

# Grazing depresses soil carbon storage through changing plant biomass and composition in a Tibetan alpine meadow

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## ABSTRACT

Grazing-induced variations in vegetation may either accelerate or reduce soil carbon storage through changes in litter quantity and quality. Here, a three-year field study (2005–2007) was conducted in Tibetan alpine meadow to address the responses of surface soil (0–15 cm) organic carbon (SOC) storage in the plant growing season (from May to September) to varying grazing intensity (represented by the residual aboveground biomass, with  $G_0$ ,  $G_1$ ,  $G_2$ , and  $G_3$  standing for 100%, 66%, 55%, and 30% biomass residual, respectively), and to explore whether grazing-induced vegetation changes depress or facilitate SOC storage. Our results showed that: (i) Higher grazing intensity resulted in lower biomass of grasses and sedges, lower root biomass, and in a change in plant community composition from palatable grasses and sedges to less palatable forbs. (ii) Increased grazing reduced the SOC content and storage with only  $G_3$  showing an SOC loss during the plant growing season. (iii) Soil organic carbon storage exhibited a highly positive correlation with the residual aboveground biomass and root biomass. Our results imply that a grazing-induced reduction in plant biomass productivity and changes in species composition would depress soil carbon storage, and that an increase in grazing pressure can lead to a gradual change of alpine meadow soils from being ‘carbon sinks’ to become ‘carbon sources’.

**Keywords:** ground cover; root biomass; summer pasture; soil carbon loss; grazing management

In last few years, grazing pressure has increased in most of the world’s rangeland that may represent as high as a quarter of the global terrestrial soil carbon storage potential (Scurlock and Hall 1998). Indeed, improper grazing management was often reported to lead to increased carbon emissions into the atmosphere from grassland ecosystems (Ingram et al. 2008). The effects of grazing on soil carbon were studied in various grassland ecosystems over extensive time periods (Ingram et al. 2008, Piñeiro et al. 2010), but the results are partly contradictory (Milchunas and Lauenroth 1993, Piñeiro et al. 2010). For instance, in Tibetan alpine meadows, both negative (Wu et

al. 2009, 2010) and positive effects (Gao et al. 2007, 2009) of livestock grazing on SOC were reported.

Several mechanisms, including direct and indirect mechanisms, are discussed that could explain the shift in SOC storage caused by grazing (Bardgett and Wardle 2003, De Deyn et al. 2008). Grazing directly influences soil carbon dynamics and turnover by removing biomass but also returning carbon in the form of dung and urine deposition; trampling may also have direct effects (Bardgett and Wardle 2003). Grazing can also indirectly influence soil carbon storage in several ways (Bardgett and Wardle 2003, De Deyn et al. 2008, Semmartin et al. 2010). First, herbivores regulate the quantities

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of resources that are returned to the soil through changing biomass allocation patterns (Bardgett and Wardle 2003, De Deyn et al. 2008). Second, herbivores could alter soil carbon storage through changes in plant species composition (Bardgett and Wardle et al. 2003, De Deyn et al. 2008, Semmartin et al. 2010), which was demonstrated to influence litter quality and decomposability (Bardgett and Wardle 2003). Although the mechanisms of grazing impact on soil carbon storage were extensively studied in a wide range of ecosystems worldwide (Ingram et al. 2008, Piñeiro et al. 2010), few studies examined the linkage between SOC storage and grazing-induced vegetation changes in Tibet (Wu et al. 2010).

In Tibetan alpine meadows, grazing by yaks and sheep triggers dramatic floristic changes in plant communities, such as decreasing plant productivity, increasing plant diversity (Wu et al. 2009) and altered species composition (Niu et al. 2010). However, whether these changes accelerate or reduce soil carbon storage is still unclear. Moreover, Tibetan alpine meadow soils are rich in carbon (18.2 km<sup>2</sup>, Ni 2002), and known to suffer from degradation due to improper grazing intensity (Wu et al. 2009, 2010). However, the associated effects on soil carbon storage remain poorly understood.

Therefore, we investigated the effects of grazing intensity on soil carbon over three years (2005 to 2007) to address the following questions:

How does grazing influence plant biomass and plant species composition?

What are the effects of grazing intensity on SOC content and SOC storage?

What is the relationship between SOC storage and the grazing-induced variation in plant biomass and species composition?

## MATERIALS AND METHODS

**Study site.** Field studies were conducted at the Alpine Meadow and Wetland Ecosystem Research Station of Lanzhou University, located on the eastern Tibetan Plateau in the northwestern China (33°59'N, 102°00'E) at an altitude of 3500 m. The local climate is characterized by strong solar radiation with long, cold winters, and short, cool summers. The monthly mean temperature and precipitation from 2005 to 2007 are shown in Figure 1. The sub-humid climate (mean rainfall is 620 mm over the last 35 years, with 16.6% coefficient of variation for interannual variability) supports a dense vegetation, typical for Tibetan alpine meadows, that

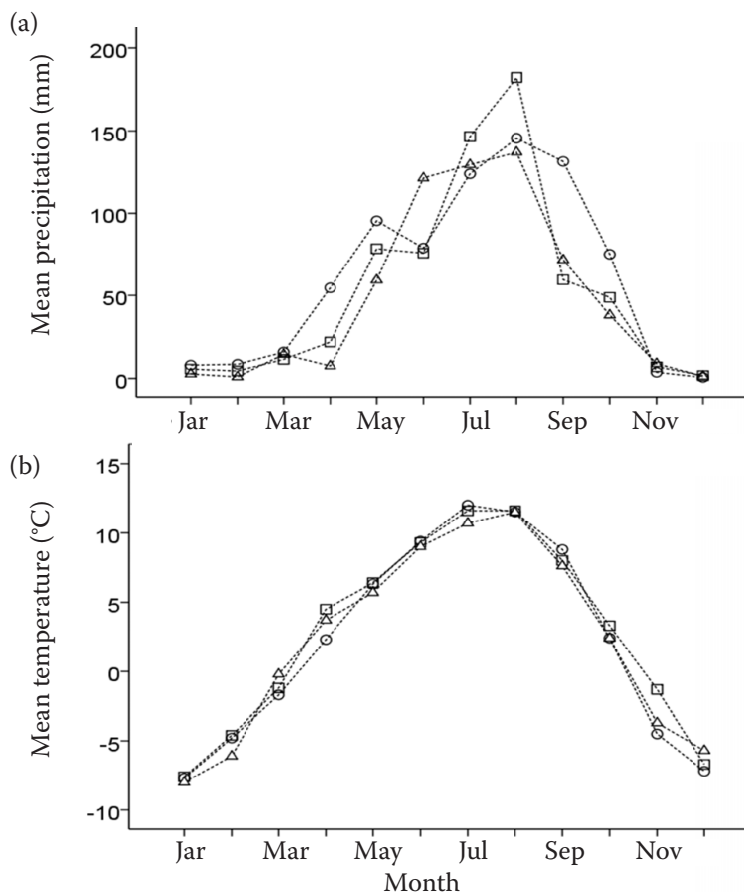


Figure 1. Monthly mean precipitation (a) and temperature (b) in 2005 (circles), 2006 (squares) and 2007 (triangles) in the study area

is dominated by sedges (e.g. *Kobresia capillifolia* and *K. humilis*), grasses (e.g. *Elymus* spp. and *Poa* spp.), compositae (e.g. *Saussurea* spp.), and other broadleaved species (e.g. *Anemone* spp.). The soil type is an alpine meadow soil (Wu et al. 2010).

**Experimental design.** This grassland has traditionally been used as summer pasture by livestock (e.g. yak and Tibetan sheep), with grazing occurring mainly from May to September (Luo et al. 2006, Ma et al. 2010). Our interviews with local people revealed that this alpine meadow has been managed by the same herders with similar grazing management for at least 15 years. The development of vegetation was monitored along a gradient of increasing grazing intensity, which is often assessed by measuring residual aboveground biomass (RAB, g/m<sup>2</sup>) at the end of grazing (Ingram et al. 2008, Ma et al. 2010). We chose 4 grassland sites – G<sub>0</sub>, G<sub>1</sub>, G<sub>2</sub>, and G<sub>3</sub> (see below for definition) – that differed significantly with respect to mean dry weight of RAB in September when the grazing terminated:

Fencing site (G<sub>0</sub>): it covers 2 ha and was fenced in October 1999 to avoid grazing of large herbivores in plant growing seasons. The mean RAB (dry biomass, same below) is 429 ± 12 g/m<sup>2</sup> (± SE, set to 100%).

Light grazing site (G<sub>1</sub>): it covers about 3 ha, with mean RAB of 285 ± 5 g/m<sup>2</sup>, corresponding to ca. 66% of aboveground biomass in G<sub>0</sub>.

Moderate grazing site (G<sub>2</sub>): it covers about 3 ha. The mean RAB is 237 ± 6 g/m<sup>2</sup>, or ca. 55% of G<sub>0</sub>.

Heavy grazing site (G<sub>3</sub>): it covers about 2 ha. The RAB is 127 ± 2 g/m<sup>2</sup>, or ca. 30% of G<sub>0</sub>. The site was exposed to long-term overgrazing and trampling by livestock, resulting in an obvious loss of vegetation cover.

These sites are 500–1000 m apart from each other, and have flat to gently rolling slopes (inclination < 1%). Both vegetation and environmental factors were homogeneous within each site and largely similar across sites. The same soil type and environmental factors assured that the differences of soil and vegetation among sites were primarily due to grazing intensity.

**Soil, vegetation sampling and analysis.** In May in 2005, i.e. at the beginning of plant growth and of summer grazing, soil samples were collected. To optimize sampling efforts, five plots (6 × 10 m) were established in each site. Then two units were randomly selected in each plot. At each unit, five soil cores (3.8 cm in diameter × 15 cm in depth) were randomly collected and pooled as one bulk sample. After removing the roots and stones, all

soil samples were taken to the laboratory and analyzed as soon as possible. In September (i.e. at the end of plant growth and summer grazing), we opted for a paired sampling in order to avoid additional spatial variability. The soil sampling lasted for three years from 2005 to 2007, during which 240 soil samples were collected for soil analysis.

In September, two quadrats (50 × 50 cm) were selected near soil sampling units to optimize pairing of plant and soil samples, but at least 0.5 m far away to avoid artificial disturbance from soil sampling in May. In each quadrat, all present plant species were recorded; the height of each species was measured thrice for three mature individuals and the ground cover of each species and of the entire community was visually estimated. In addition, ramets of clonal plants were counted as if they were separate individuals (Luo et al. 2006); ramets and shoots were counted for each species, clipped and then classified into four functional groups: grasses (Gramineous plants); sedges (Cyperaceous plants); legumes; and forbs (other annual and perennial species). At the same time, three soil samples at 3.8 cm in diameter and 15 cm in depth were taken from each unit to estimate root biomass, which was distributed mainly in the upper 15 cm of the soils (Chen et al. 2010).

The harvest for each functional group was taken to the laboratory, where it was dried at 80°C for 48 h and afterwards weighted. For the root biomass estimation, the prepared samples were rinsed in water to remove soil and debris; roots were dried at 80°C till constant weight. Soil bulk density (g/cm<sup>3</sup>) was determined using the core method and calculated as the mass of oven-dry soil (105°C) divided by its volume (Chen et al. 2010). Air-dried soil samples were passed through 0.2-mm sieve and were used to measure SOC by the dichromate oxidation method (Kalembasa and Jenkinson 1973).

**Data analysis.** Plant community RAB was calculated as a sum of all present functional groups. The SOC content (g/cm) was calculated as the amount (g) of SOC in g per square centimeter basis up to the depth of 15 cm. The SOC storage in the plant growing season (g/cm<sup>2</sup>) was calculated as the difference in SOC content between May and September. The calculation formulae are as follow:  
SOC content = SOC concentration × bulk density × height of soil column (sample depth)  
SOC storage = SOC content in September – SOC content in May

Effects of grazing intensity (G<sub>0</sub>, G<sub>1</sub>, G<sub>2</sub>, and G<sub>3</sub>) on vegetation and soil variables (Appendix 1) were evaluated with repeated measures analysis of

Appendix 1. Results of repeated measures ANOVA for variables of vegetation and soil (0–15 cm) organic carbon (SOC) over three years (2005–2007)

	Grazing intensity			Year			Year × grazing intensity		
	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>	<i>df</i>	<i>F</i>	<i>P</i>
Residual aboveground biomass	3	458.7	< 0.001	2	27.7	< 0.001	6	<b>1.8</b>	<b>0.093</b>
Ground cover	3	301.1	< 0.001	2	10.6	< 0.001	6	5.8	< 0.001
Root biomass	3	334.5	< 0.001	2	215.3	< 0.001	6	<b>3.4</b>	<b>0.006</b>
Residual grasses biomass	3	358.9	< 0.001	2	4.5	0.015	6	<b>0.4</b>	<b>0.876</b>
Residual sedges biomass	3	293.4	< 0.001	2	<b>1.2</b>	<b>0.154</b>	6	2.6	0.023
Residual legumes biomass	3	70.2	< 0.001	2	16.5	< 0.001	6	<b>1.0</b>	<b>0.430</b>
Residual forbs biomass	3	3.4	0.027	2	21.5	< 0.001	6	<b>2.4</b>	<b>0.057</b>
SOC content (May)	3	28.8	< 0.001	2	22.6	< 0.001	6	<b>0.8</b>	<b>0.561</b>
SOC content (September)	3	71.1	< 0.001	2	10.9	< 0.001	6	2.4	0.033
SOC storage	3	48.7	< 0.001	2	0.2	0.790	6	0.7	0.627

Bold numbers indicate non significant differences between treatments

variance on data collected over multiple growing seasons (2005–2007) using general linear models. One-way ANOVA was used where it was required to compare the effects of grazing intensity. Linear regression analyses were used to examine the relationship of SOC storage with vegetation biomass. Fisher's least significant difference (*LSD*) was chosen for post-hoc tests. All data analyses were conducted with the SPSS software (SPSS 16.0 for Windows, Chicago, IL, USA).

## RESULTS AND DISCUSSION

The climate data indicated that approximately 80% of precipitation and  $\geq 0^\circ\text{C}$  days were concentrated in the summer season from May to September

(Figure 1). Together, precipitation and frost-free days control the growth of plant and the activities of soil organisms (Li et al. 2004). Therefore, the soil carbon dynamics during the summer are important for understanding the local carbon budget. The climate data showed that mean annual precipitation and temperature varied among experimental years (Figure 1), which is the most straightforward explanation for the differences in vegetation and soil among years reported in earlier studies (Wu et al. 2009). However, we primarily focused on the effects of grazing intensity, thus inter-annual variation was not discussed in depth in the present study.

**Vegetation response.** Grazing intensity, represented by RAB, significantly reduced root biomass ( $F_3 = 334.5$ ,  $P < 0.001$ ; Table 1). This resulted probably from the decrease of sedge species (Table 1;

Table 1. The variations of vegetation properties ( $\text{g}/\text{m}^2$ ) along the grazing intensity gradients over three years (2005–2007)

	Sites			
	$G_0$	$G_1$	$G_2$	$G_3$
Aboveground residual biomass	429 ± 12.0	285.2 ± 4.6	237.2 ± 6.03	126.7 ± 2.1
Root biomass	1905.8 ± 22.5	1625.5 ± 21.1	1221.7 ± 22.5	629.5 ± 13.7
Residual grasses biomass	179.0 ± 0.8	69.4 ± 1.9	48.2 ± 1.5	21.4 ± 0.8
Residual sedges biomass	116.4 ± 6.4	94.5 ± 2.3	60.6 ± 2.6	1.7 ± 0.4
Residual legumes biomass	22.2 ± 1.7	19.2 ± 0.7	21.5 ± 1.0	4.1 ± 0.3
Residual forbs biomass	111.3 ± 5.5	102.1 ± 2.1	106.9 ± 2.8	99.6 ± 1.9

Values are means ( $\pm\text{SE}$ ) for 40 quadrats and over three years  $G_0$ ,  $G_1$ ,  $G_2$ , and  $G_3$  indicated 100%, ca. 66%, ca. 55%, and ca. 30% residual biomass, respectively

Table 2. Dominant species composition and their responses to grazing intensity in terms of plant ground cover (%)

Species	Functional group	Height (cm) <sup>†</sup>	Palatability*	Plant ground cover (%)			
				G <sub>0</sub>	G <sub>1</sub>	G <sub>2</sub>	G <sub>3</sub>
<i>Anemone obtusiloba</i>	forbs	14.4	unpalatable	2.1 ± 0.1 <sup>c</sup>	3.8 ± 0.2 <sup>a</sup>	3.2 ± 0.2 <sup>b</sup>	1.3 ± 0.1 <sup>d</sup>
<i>Anemone rivularis</i>	forbs	44.2	unpalatable	11.3 ± 0.4 <sup>a</sup>	7.6 ± 0.3 <sup>b</sup>	6.8 ± 0.3 <sup>c</sup>	2.2 ± 0.5 <sup>d</sup>
<i>Elymus nutans</i>	grasses	59.3	high	10.4 ± 0.3 <sup>a</sup>	5.2 ± 0.2 <sup>b</sup>	4.1 ± 0.2 <sup>c</sup>	3.0 ± 0.1 <sup>d</sup>
<i>Galium verum</i>	forbs	16.8	low	2.2 ± 0.1 <sup>c</sup>	5.0 ± 0.2 <sup>b</sup>	5.6 ± 0.2 <sup>a</sup>	–
<i>Gueldenstaedtia diversifolia</i>	legumes	12.5	high	3.9 ± 0.1 <sup>a</sup>	3.8 ± 0.1 <sup>a</sup>	4.1 ± 0.2 <sup>a</sup>	–
<i>Kobresia capillifolia</i>	sedges	20.4	high	10.3 ± 0.4 <sup>a</sup>	5.7 ± 0.2 <sup>b</sup>	4.9 ± 0.2 <sup>b</sup>	–
<i>Kobresia graminifolia</i>	sedges	43.8	high	17.5 ± 0.5 <sup>a</sup>	12.6 ± 0.5 <sup>b</sup>	10.6 ± 0.4 <sup>c</sup>	–
<i>Kobresia humilis</i>	sedges	8.4	high	2.8 ± 0.1 <sup>c</sup>	4.2 ± 0.1 <sup>b</sup>	4.6 ± 0.2 <sup>a</sup>	2.2 ± 0.1 <sup>d</sup>
<i>Leontopodium souliei</i>	forbs	4.1	low	1.1 ± 0.1 <sup>c</sup>	2.5 ± 0.2 <sup>b</sup>	3.6 ± 0.2 <sup>a</sup>	–
<i>Ligularia virgaurea</i>	forbs	40.5	unpalatable	6.3 ± 0.4 <sup>a</sup>	3.3 ± 0.3 <sup>b</sup>	3.8 ± 0.3 <sup>b</sup>	–
<i>Medicago ruthenica</i>	legumes	12.9	high	2.9 ± 0.2 <sup>b</sup>	3.2 ± 0.1 <sup>a</sup>	3.5 ± 0.2 <sup>a</sup>	1.3 ± 0.1 <sup>c</sup>
<i>Pedicularis kansuensis</i>	forbs	31.2	low	4.4 ± 0.3 <sup>a</sup>	2.1 ± 0.2 <sup>b</sup>	1.5 ± 0.2 <sup>c</sup>	1.3 ± 0.1 <sup>c</sup>
<i>Plantago depressa</i>	forbs	10.1	low	–	2.2 ± 0.2 <sup>c</sup>	3.5 ± 0.2 <sup>b</sup>	4.7 ± 0.1 <sup>a</sup>
<i>Poa chalarantha</i>	grasses	43.0	high	7.2 ± 0.5 <sup>a</sup>	3.6 ± 0.2 <sup>b</sup>	3.1 ± 0.2 <sup>b</sup>	1.6 ± 0.2 <sup>c</sup>
<i>Poa pratensis</i>	grasses	42.7	high	7.7 ± 0.4 <sup>a</sup>	4.3 ± 0.3 <sup>b</sup>	3.8 ± 0.3 <sup>b</sup>	2.2 ± 0.2 <sup>c</sup>
<i>Potentilla anserina</i>	forbs	9.8	medium	1.3 ± 0.1 <sup>c</sup>	2.4 ± 0.1 <sup>bc</sup>	3.1 ± 0.2 <sup>b</sup>	38.3 ± 0.6 <sup>a</sup>
<i>Roegneria nutans</i>	grasses	55.0	medium	4.1 ± 0.2 <sup>a</sup>	3.3 ± 0.1 <sup>b</sup>	2.8 ± 0.2 <sup>c</sup>	–
<i>Saussurea hieracioides</i>	forbs	12.5	medium	3.3 ± 0.2 <sup>b</sup>	5.6 ± 0.3 <sup>a</sup>	6.2 ± 0.4 <sup>a</sup>	1.9 ± 0.2 <sup>c</sup>
<i>Saussurea nigrescens</i>	forbs	18.2	medium	3.0 ± 0.3 <sup>b</sup>	4.7 ± 0.2 <sup>a</sup>	4.7 ± 0.3 <sup>a</sup>	1.7 ± 0.2 <sup>c</sup>
<i>Stipa aliena</i>	grasses	55.8	high	8.4 ± 0.3 <sup>a</sup>	5.3 ± 0.3 <sup>b</sup>	3.4 ± 0.2 <sup>c</sup>	–
<i>Taraxacum maurocarpum</i>	forbs	15.2	medium	1.5 ± 0.1 <sup>c</sup>	4.4 ± 0.2 <sup>b</sup>	4.8 ± 0.1 <sup>a</sup>	1.2 ± 0.1 <sup>c</sup>

Means with different letters indicate significant differences ( $P < 0.05$ ; *LSD* after significant one-way ANOVA); G<sub>0</sub>, G<sub>1</sub>, G<sub>2</sub>, and G<sub>3</sub> indicated 100%, 66%, 55%, and 30% residual biomass, respectively; <sup>†</sup>was the mean of three measured maximum values of plant individual height in all sites; \*was defined based on Zhang et al. (1990). –means species was absent in this grazing sites

Wu et al. 2009) along the grazing intensity gradient. The main Cyperaceae *Kobresia capillifolia* and *K. graminifolia* (Table 2) normally show low shoot/root ratio and dense root systems and allocate a large fraction of biomass below ground (Gao et al. 2007). Our results confirm earlier work by Cao et al. (2004) who conducted studies in a summer-grazed alpine meadow (May to September), but disagree with later studies conducted by Gao et al. (2007 and 2009) in winter-grazed (October to May) alpine meadows. Furthermore, Hejduk and Hrabě (2003) reported that underground pasture phytomass was not significantly influenced by grazing in a pasture in Rapotin, the Czech Republic. These results confirmed that response of root biomass to grazing can differ between seasons of grazing, and also between different grassland ecosystems.

Because most grasses and sedges are palatable to herbivores and tall (Table 2), the RAB of grasses ( $F_3 = 358.9$ ,  $P < 0.001$ ) and sedges ( $F_3 = 293.4$ ,  $P < 0.001$ ) were markedly reduced with increasing grazing (Table 1), which mainly accounted for a decrease in plant community RAB. Consistently, there was a shift across the grazing intensity gradient in the dominant species from palatable grasses and sedges to less-palatable forbs (Table 2), which is probably because of the preferential grazing of tall, palatable dominant grasses and sedges (Table 2). Grazing creates canopy gaps and/or bare soil patches, relaxes intra- and inter-specific competition for light (Sternberg et al. 2000, Pavlů et al. 2007, Wu et al. 2009), and ultimately favors the establishment of short-stature, less-palatable forbs (Table 2, Pavlů et al. 2007). These results

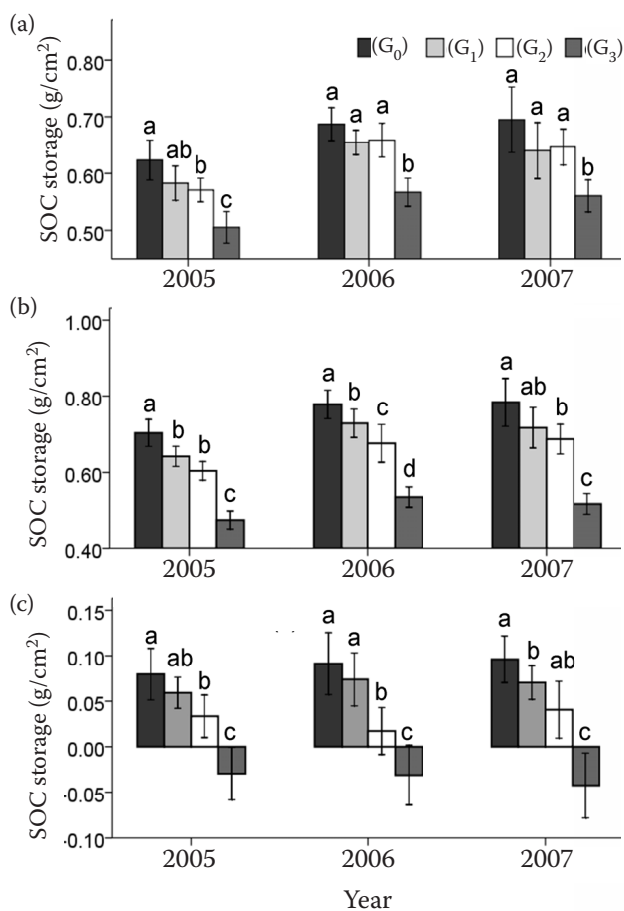


Figure 2. Effect of grazing intensity on soil organic carbon (SOC) content in May (a) and in September (b), and SOC storage (c) from May to September. Values ( $\pm$ SE) are means of ten samples for three years (2005, 2006, and 2007).  $G_0$ ,  $G_1$ ,  $G_2$ , and  $G_3$  represent mean 100%, 66%, 55%, and 30% biomass residual, respectively. Significant differences among grazing intensities are indicated by different letters at  $P < 0.05$

suggest that grazing reduces plant biomass and changes species composition mainly through the negative effects on dominant grasses and sedges.

**Responses in soil carbon stocks.** Grazing intensity had significantly negative effects on SOC content in May ( $F_3 = 28.773$ ,  $P < 0.001$ ) and September ( $F_3 = 71.127$ ,  $P < 0.001$ ) over all three years (Figures 2a and 2b). Soil organic carbon storage during the plant growing seasons was also significantly reduced by grazing intensity ( $F_3 = 48.737$ ,  $P < 0.001$ ; Figure 2c). From May to September, surface soil (0–15 cm) in  $G_0$  site sequestered the largest amount of carbon ( $0.0895 \pm 0.0084$  g/cm<sup>2</sup>), followed by  $G_1$  ( $0.0678 \pm 0.0063$  g/cm<sup>2</sup>) and  $G_2$  ( $0.0304 \pm 0.0077$  g/cm<sup>2</sup>). However,  $G_3$  led to a negative SOC storage ( $-0.0345 \pm 0.0091$  g/cm<sup>2</sup>) over three years (Figure 2c).

These results showed that grazing depresses SOC content and storage, which is in accordance with some previous studies (Snyman and Du Preez 2005, Wu et al. 2010). They are, however, in contrast to studies of Gao et al. (2007, 2009), who conducted experiments in an alpine meadow that were used as winter pasture (October to May). This inconsistent result can be attributed to the difference in seasons of grazing, which was reported to lead to various grazing response elsewhere (Milchunas and Lauenroth 1993). Furthermore, compared to  $G_0$ ,  $G_1$  and  $G_2$ , heavy grazing ( $G_3$ ) resulted in a negative soil carbon balance, which suggests that grazing at these levels caused soil carbon emission to atmosphere.

There is a range of potential indirect mechanisms through which soil carbon may be affected by increased grazing (Bardgett and Wardle 2003, De Deyn et al. 2008, Semmartin et al. 2010). Firstly, in the present study both RAB ( $P < 0.001$ ,  $n = 120$ ; Figure 3a) and root biomass ( $P < 0.001$ ,  $n = 120$ ; Figure 3b) were found to have a significantly positive correlation with SOC storage over three years, which suggests that the decrease of SOC storage with increased grazing can be attributed to grazing-induced decreases in plant biomass. Previous studies indicated that grazing alters the quantity of resources returned to the soil (Semmartin et al. 2010, Wu et al. 2010). Considering that the decrease of plant biomass was mostly caused by the removal of palatable grasses and sedges (Tables 1 and 2) and that these built especially large soil carbon pools, we conclude that grazing depresses SOC storage through reducing plant biomass, especially through the removal of grasses and sedges.

Second, grazing-induced changes in species composition could also alter SOC storage. Grime et al. (1996) suggested that palatable plant species generally produce litter of a higher quality for decomposers than do unpalatable species because palatability of foliage and decomposability of plant litter are governed by similar ecophysiological traits. Similarly, Wardle et al. (2002) also reported that litter decomposition and nutrient release are faster in palatable than in unpalatable grasses. Thus, the significant shift of dominant species composition in the present study from palatable grasses and sedges species to less palatable forbs species (Tables 1 and 2) may point to a decline in the rate of litter decomposition, and consequently reduced SOC storage in alpine meadow.

Overall, increased grazing induced a decrease in SOC storage and apparently sequestration during the growing season. This was caused by

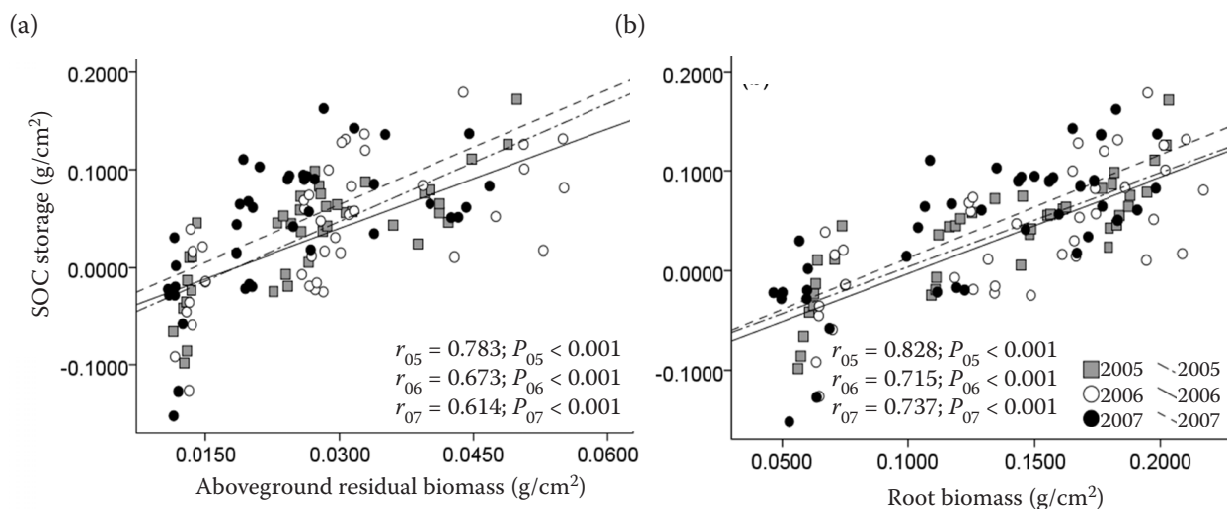


Figure 3. The relationship of soil organic carbon (SOC) storage with aboveground residual biomass (a) and root biomass (b)

reduced vegetation biomass and altered plant species composition, which both can be attributed to the negative effects of grazing on grasses and sedges. The fenced meadow ( $G_0$ ) had the highest SOC storage; light and moderate grazing also led to a positive SOC storage; and only heavy grazing showed a negative SOC storage (Figure 2c). These results imply that very heavy grazing can result in alpine meadow soils gradually changing from being 'carbon sinks' to become 'carbon sources'. Fencing is an effective restoration measure to promote SOC storage; however, when considering the loss of pastures and the reduction in biodiversity caused by fencing (Wu et al. 2009), light or moderate grazing may help to achieve a balance between sustainable livestock production and soil carbon management.

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