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LONG-TERM GRAZING ALTERS SPECIES COMPOSITION AND BIOMASS OF A SHRUB MEADOW ON THE QINGHAI -TIBET PLATEAU

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Abstract

Livestock grazing has long been the most widespread land use on the Qinghai–Tibet Plateau, one of the world's highest ecosystems. However, there has been increasing concern during recent decades because of the rapid increase in livestock numbers. To assess the possible influences of grazing on the vast grassland, a long-term grazing experiment in a shrub meadow on the northern Qinghai–Tibet Plateau was carried out. The experiment included five treatments with different stocking rates and one non-grazing (N) treatment. After 17 years of grazing, treatment differences were clear. The species composition differed markedly between grazing intensities, with a decrease in palatable grass species and an increase in unpalatable forbs at higher grazing intensities. The species richness and species diversity, however, were not significantly different between treatments. Vegetation height decreased significantly at higher grazing intensities. Total above ground biomass declined considerably and the biomass of forbs increased significantly under the higher grazing intensities. The amount of litter was significantly lower under the higher grazing intensities. The results suggest that long-term grazing alters the species composition, vegetation height and biomass production of the alpine grassland ecosystem without significantly changing species richness.

Introduction

Livestock grazing is one of the most important disturbance agents in grassland ecosystems. Its ecological and environmental impacts have been documented such as effects on plant species richness, biodiversity and productivity (Milchunas *et al.*, 1988; Fleischner, 1994; Jones, 2000). Grazing can affect plant species richness and grassland biodiversity (White & Cosgrove, 1990; Killeen, 1991; O'Connor, 1994; Bertiller, 1996; Kazmaier *et al.*, 2001; Critchley *et al.*, 2004; Gillen & Sims, 2004). These effects can be positive or negative, depending mainly on grazing intensity and ecosystem properties. In ecosystems with moderate to high productivity, grazing often increases plant species richness, but it can reduce the species richness in ecosystems with low productivity (Olff & Ritchie, 1998; Proulx & Mazumder, 1998; Osem *et al.*, 2002; Bakker *et al.*, 2003). At landscape scale, however, grazing probably has little effect on native species richness (Stohlgren *et al.*, 1999). A mesic meadow ecosystem may have higher resistance to grazing pressure than an arid grassland ecosystem because the former is more productive (Hayes & Holl, 2003). On the other hand, grazing intensity seems an important determinant of plant species diversity. Low to moderate degrees of grazing can often increase plant species diversity (Collins, 1987; Milchunas *et al.*, 1992; Collins *et al.*, 1998), whereas, heavy grazing can reduce diversity (Biondini *et al.*, 1998). Moreover, grazing reduces the abundance, biomass, and number of palatable species, and increases the unpalatable and grazing-resistant species (Adler & Morales, 1999; Hickman & Hartnett, 2002).

The effect of livestock grazing on an ecosystem can be brief or long-term. Some long-term studies have focused on the effects of grazing in temperate and tropical grassland ecosystems (Lauenroth & Sala, 1992; Jones, 2000; Stohlgren *et al.*, 2000; Weber *et al.*, 2000; Harris *et al.*, 2003). However, few long-term studies have evaluated the effects of grazing on tundra, Arctic and alpine vegetations (Sundriyal, 1992).

The Qinghai–Tibet Plateau has one of the world's most extensive and highest grazing ecosystems. More than 60% of the plateau, or about 1.5 million km^2, is covered by alpine grassland, which supports some 70 million livestock (Li & Zhou, 1998). With the rapid increase of population and the rapid economic growth in China, the grazing pressure on the grassland has been consistently increasing during recent decades (Zheng *et al.*, 2000; Du *et al.*, 2004). The high grazing intensity may be contributing to the recent degradation and desertification of the grassland, which could influence regional and even global climate change on account of the geographic features of the plateau. Conservation and sustainable use of the vast alpine ecosystem have therefore become of increasing concern because of the ecosystem's potential importance in terms of the ecological and environmental roles it plays on both regional and global scales. However, little information is available on how the current grazing regime influences this unique alpine ecosystem.

The alpine meadow ecosystem occupies more than 40% of the grassland on the plateau. With the high irradiation and adequate temperature and precipitation during the growing season from May to September, the meadow ecosystem seems highly productive (Zheng *et al.*, 2000; Gu *et al.*, 2003; Cui *et al.*, 2004). We expected that light to moderate livestock grazing may thus increase the species richness, biodiversity and productivity of the alpine meadow. An alternative hypothesis was that the species composition would be altered but that species richness would not change significantly if the numbers of plant species invading and disappearing from the ecosystem were similar. To test the hypotheses, our major objective was to determine the influence of long-term grazing on vegetation structure, species composition and diversity and grassland productivity.

Study site and methods

Study site: The study was conducted in an alpine meadow dominated by *Potentilla fruticosa* shrubs. The meadow ecosystem (37°29′–37°45′N, 101°12′–101°23′E) is located in a large valley oriented NW–SE and surrounded on all sides by the Qilian Mountains at the northern edge of the Qinghai–Tibet Plateau. The average elevations are about 4000 m above sea level for the mountains and 3200 m for most of the valley. The shrub meadow is located on a small hill in the valley at an elevation about 3250 m.

The climate of the meadow ecosystem is dominated by the southeast monsoon and by high-pressure systems centered on Siberia. A continental monsoon-type climate prevails, with a long cold winter and a short cool summer. During the last 30 years, the annual air temperature has averaged –1.7°C, with extremes of 27.6 and –37.1°C (Li & Zhou, 1998; Li, 1998). During winter, the mean temperature can drop to –20°C. During the warmest month (July), the daytime temperature ranges from 14 to 22°C in the valleys and from 4 to 10°C in the mountains. Mean annual precipitation ranges widely (from 426 to 860 mm), and 80% falls in the short summer growing season from May to September (Zhao & Zhou, 2000).

The alpine meadow is dominated by Mollic-Gleyic Cambisols and Mat Cryo-Sod soils, both of which are rich in nitrogen, phosphorus and potassium. The soils are also characterized by high organic matter content (Zhao & Zhou, 2000). Two major vegetation types are found in the alpine meadow. The *Kobresia humilis* meadow occurs widely in the valley floor, and the shrub meadow dominated by *P. fruticosa* often occupies northern or northwestern slopes. The *K. humilis* meadow has long been used as cold-season pasture in spring and winter and the *P. fruticosa* shrub meadow as warm-season pasture in summer and autumn (Li & Zhou, 1998).

Experimental design: The grazing experiment was started in 1985 on a north-west facing slope covered by an extensive *Potentilla fruticosa* shrubby meadow (Zhao & Zhou, 2000). An area of about 8 ha (402 m × 200 m) was chosen and fenced with barbed wire and divided into five plots of different sizes (Zhang, 1990). According to the report from Wang *et al.*, (1991), the 8-ha site was coved by highly homogenous vegetation and was thus determined for the grazing experiment. Before the experiment, regular livestock grazing had long been conducted in the slope area. An investigation in 1985 showed a total number of 45 high plant species dominated by *Potentilla fruticosa* with 41 herbaceous species and 4 shrubby species within the site. The aboveground net primary productivity was estimated as 267g dry matter m^{-2} per year (Wang *et al*., 1991). Since 1985, no significant disturbance has been observed in terms of vegetation, soil or any other environmental factors within the study site. It is thus reasonable to assume that the differences observed between the grazing conditions in the current study would be due to the experimental treatments. Therefore, we may be able to not consider here the pseudo-replication nature as a major limitation for the study started in 1985, when the pseudo-replication became a topic in ecological studies.

The six treatments were high density (H), moderately high density(Mh), moderate density l, moderately low density (Ml), low density (L), and non-grazing (N), not grazed in the summer after 1985 and it was enclosed for all the year after 1988. The initial (May 1985 to December 1987; higher) and subsequent (1988 to 2002; lower) stocking densities of livestock are shown in Table 1. The livestock were all healthy, 2-year-old, male Tibetan sheep. New sheep of similar weights were stocked every May or June so that the grazing intensities remained comparable. The grazing ceased at the end of September every year.

Table 1. Area, number of sheep, and stocking density in the long-term grazing experiment.

Treatment	H	Mh	M	Ml	L	N
Plot area (ha)	0.92	1.12	1.39	1.85	2.75	1.00
Number of sheep 1985–1987	10.0	10.0	10.0	10.0	10.0	0.0
Stocking rate (sheep/ha)	10.87	8.93	7.19	5.41	3.64	0.0
Number of sheep 1988–2002	5.0	5.0	6.0	6.0	7.0	0.0
Stocking rate (sheep/ha)	5.43	4.46	4.32	3.24	2.55	0.0

[*]All grazing plots were fenced in 1985. The non-grazing plot was fenced in 1988, but it was ungrazed from 1985 to 1988. All grazing treatments continued from 1985 to 2002, except for a fallow year in 1996.

There were 6 grazing intensities: high-density (H), moderately high-density (Mh), moderate-density (M), moderately low-density (Ml), low-density (L), and no grazing (N).

Measurements and data analysis: To examine the long-term effects of grazing on community structure, plant species diversity, and biomass, we collected samples in quadrats (Walker, 1996) at the beginning of October 2002. A transect paralleling to the slope was set across almost the center of each plot. Along both sides of the transect, we established a total number of 10 quadrats, each 5 m from the transect line and 5 m from its neighbours. Quadrats were at least 2 m from the nearest fence. The size of each quadrat was 50 x 50 cm^2. In each quadrat, vegetation coverage, canopy height, aboveground biomass of fresh shoots and foliage, biomass of dead standing material and litter on the ground surface was measured.

The point-intercept method to assess the percentage vegetation coverage by species was used according to Walker (1996). A 50-cm × 50-cm frame with 100 squares defined by nylon strings was laid 5 cm apart on each side of the frame. The frame was placed over the vegetation. A short, thin, metal rod was inserted from the canopy top down to the ground in each square and each species hit by the rod was recorded. Multiple species could be hit in a single insertion. Relative coverage was the ratio of total hits of each species to 100 squares. At the same time, the height of one individual plant was measured for each species in each square. The number of individuals of each monocot species were determined by counting tillers and each tiller was looked as a separated plant (Zhang *et al.*, 1990).

The aboveground biomass was harvested from each quadrat after these measurements (Chapin *et al.*, 1995). The samples were separated into living and dead material species. All samples were oven-dried at 60 °C until they reached constant weight and were then weighed. The biomass of *P. fruticosa* was measured according to Wang *et al.*, (1991). Aboveground biomass was clipped from six 50cm x 50cm plots for each grazing treatment. Biomass was weighed after drying to constant weight at 80°C.

To identify any change in the relative importance of species under grazing disturbance, we calculated the combined importance value of coverage and aboveground biomass (Kiviniemi & Eriksson, 2002) for each species *i* in a quadrat (Jiang, 1986): *IV* = (relative coverage + relative aboveground biomass)/2 (1)

The relative coverage was calculated by dividing the total hitting numbers of the species *i* by the total square number (100) in each plot. The relative aboveground biomass is the ratio of the total biomass of species *i* to the total biomass of the plot.

Species diversity was assessed from species richness (*S* = total number per quadrat of 0.25 m^2), the Shannon–Wiener index (*H*) (Zar, 1996), and (E_1) Pielou's index ((Pielou, 1966):

$$H = -\sum_{i=1}^{S}(IV_i \cdot \ln IV_i), \quad (2)$$

$$E_1 = H / \ln(S) \quad (3)$$

The index of combined similarity of coverage and aboveground biomass among treatments, *R* (degree of overlap between samples *x* and *y*), was developed following (Horn, 1966):

$$R = \frac{\sum_{i}^{n}(x_i + y_i)\cdot\ln(x_i + y_i) - \sum_{i}^{n} x_i \cdot \ln x_i - \sum_{i}^{n} y_i \cdot \ln y_i}{(X+Y)\cdot\ln(X+Y) - \ln X - \ln Y} \quad (4)$$

where x_i and y_i are relative IV_i of species *i* in samples *x* and *y*; *X* and *Y* are the sum of relative *IV* of all species in samples *x* and *y*; and $0 \le R \le 1$.

The forage quality under the different grazing intensities was also assessed. Plant species were classified as excellent (3), good (2), adequate (1), poor (0), or poisonous (–1) according to their palatability to livestock (Zhang *et al.* 1990). An index of forage palatability (Q_{fl}) was estimated as follows:

$$Q_{fl} = \sum_{-1}^{3} (i \cdot S_i), \quad (5)$$

where i is the value of palatability (from –1 to 3) defined above and S_i is the coverage of different species in all quadrats.

Duncan's multiple comparison tests (N=10) was used for all the comparisons between grazing treatments. No data transformation was employed in the analysis. All calculations were performed using the Statistical Program for Social Sciences (SPSS, version 10.0).

Results

Species richness and community composition: There was no significant change in the total species richness under different grazing intensities, although it tended to decrease as grazing intensity increased (Table 2). Species diversity indicated by the Shannon–Wiener index showed no statistically significant difference under the different grazing treatments, although it tended to be highest under the moderate grazing intensity (Table 2). The Pielou index of diversity, however, was significantly higher in the Mh and M plots.

On the other hand, the species composition showed marked variation under different grazing conditions (Tables 3, 4). The most dominant species was *P. fruticosa* in the M, Ml, L, and N plots. The dominant high shrub species were replaced by the low, unpalatable forb *Leontopodium nanum* in the H and Mh plots. Species with an *IV* > 10 at low grazing densities changed dramatically under higher grazing intensities (Table 3): *Kobresia capillifolia* and *Stipa aliena* had high *IV*s in the Ml, L, and N plots, but decreasing *IV*s in the direction of M to H. As grazing intensity increased, the proportion of palatable species in the Gramineae and the Cyperaceae decreased, while the proportion of non-palatable forbs increased significantly in comparison with the N and L plots (Table 3).

Species replacement indicated by the local disappearance or invasion of species was different under different grazing pressures (Table 4). Compared with the non-grazing plot, plots at higher grazing pressures showed larger total numbers of both disappearing and invading species. In the H plot, 9 species disappeared, but another 12 species invaded.

The change of species composition between plots caused the similarity index to vary between plots (Table 5). The similarity indices were higher between plots with more similar grazing treatments. The smallest similarity was found between the H and N plots.

Canopy height: There is a clear, two-layer structure in the canopy of the alpine *P. fruticosa* meadow. The top layer consists of graminoids, high forbs and shrubs. The bottom layer consists mainly of low forbs and sedges. The canopy height of both layers increased as the grazing intensity decreased (Fig. 1). The average height of the top layer in the N plot was 37 cm about 3.7 times that in the H plot. The average height of the bottom layer in the N plot was only 14 cm, but was still 3.5 times that in the H plot.

Table 2. Means ± standard deviations of total species richness (mean species number per sampling quadrat), species richness of different functional groups, the Shannon–Wiener index, and the Pielou index under different livestock grazing intensities.

Grazing intensity	H	Mh	M	Ml	L	N
Total species richness	20.83 ± 3.97^{a}	21.00 ± 1.79^{a}	22.83 ± 3.06^{a}	23.67 ± 1.03^{a}	23.17 ± 2.23^{a}	23.50 ± 3.39^{a}
Gramineae	2.17 ± 0.41^{a}	2.00 ± 0.63^{a}	2.34 ± 0.52^{a}	3.33 ± 1.03^{b}	4.00 ± 1.10^{b}	4.00 ± 0.89^{b}
Cyperaceae	1.66 ± 1.21^{a}	2.17 ± 0.75^{ab}	2.83 ± 0.75^{bc}	3.17 ± 0.75^{bc}	2.83 ± 0.75^{bc}	3.33 ± 0.52^{c}
Fabaceae (forb)	1.50 ± 0.55^{a}	1.00 ± 0.00^{ab}	1.50 ± 0.55^{a}	0.50 ± 0.55^{b}	0.50 ± 0.55^{b}	1.00 ± 0.89^{ab}
Non-Fabaceae forb	14.67 ± 3.56^{a}	15.00 ± 1.41^{a}	15.33 ± 3.08^{a}	15.67 ± 1.21^{a}	14.67 ± 1.21^{a}	13.83 ± 2.32^{a}
Shrub	0.83 ± 0.41^{a}	0.83 ± 0.41^{a}	0.83 ± 0.41^{a}	1.00 ± 0.00^{a}	1.17 ± 0.41^{a}	1.33 ± 0.52^{a}
Shannon–Wiener index	2.47 ± 0.38^{a}	2.64 ± 0.23^{a}	2.67 ± 0.25^{a}	2.59 ± 0.12^{a}	2.51 ± 0.11^{a}	2.36 ± 0.11^{a}
Pielou index	0.81 ± 0.09^{ab}	0.87 ± 0.08^{b}	0.86 ± 0.10^{b}	0.82 ± 0.05^{ab}	0.80 ± 0.02^{ab}	0.75 ± 0.04^{a}

Values in a horizontal row followed by the same letter are not significantly different between grazing intensities (Duncan's multiple comparison tests; n=10; $P > 0.05$).

Table 3. Total numbers of species and families, and species composition under different grazing intensities.

Grazing intensity	Species number	Dominant species (***Bold underlined***), major species (***Underlined***; ***IV*** **> 10), and companion species (*IV* ≤ 10)**
High	39 (14) (4: 5: 29: 1)	***Leontopodium nanum*** Hand.-Mazz; *Potentilla anserina* L.; *Gueldenstaedtia diversifolia* Maxim.; *P. fruticosa* Rydb.; *Glaux maritima* L.; *P. bifurca* L.; *Saussurea katochaete* Maxim.; *S. superba* Anth.; *Aster flaccidus* Bunge; *Thalictrum alpinum* L.; *Anaphalis lacteal* Maxim.; *Taraxacum mongolicum* Hand.-Mazz.; *Festuca ovina* L.; and *Elymus nutans* Griseb.
Moderately high	38 (14) (4: 5: 28: 1)	***Leontopodium nanum***; *Gueldenstaedtia diversifolia*; *Potentilla fruticosa*; *Festuca ovina*; *Stipa aliena* Keng; *P. anserina*; *Taraxacum mongolicum*; *Lancea tibetica* Hook. f. et Thoms; *P. nivea* L.; *Kobresia humilis* Serg.; *Saussurea superba*; and *Polygonum viviparum* L.
Moderate	37 (13) (5: 5: 26: 1)	***Potentilla fruticosa***; *Leontopodium nanum*; *Gueldenstaedtia diversifolia*; *Stipa aliena*; *Festuca ovina*; *Saussurea katochaete*; *Lancea tibetica*; *Anaphalis lacteal*; *Kobresia humilis*; *K. capillifolia* Clarke; *S. superba*; *Gentiana farreri* Balf.; and *Stellera chamaejasme* L.
Moderately low	37 (13) (6: 4: 26: 1)	***Potentilla fruticosa***; *Kobresia capillifolia*; *Stipa aliena*; *Festuca ovina*; *Elymus nutans*; *Helictotrichon tibeticum* Keng; *Lancea tibetica*; *P. nivea*; *Leontopodium nanum*; *Aster flaccidus*; and *Taraxacum mongolicum*
Low	37 (13) (7: 4: 25: 1)	***Potentilla fruticosa***; *Kobresia capillifolia*; *Stipa aliena*; *Festuca ovina*; *Elymus nutans*; *Helictotrichon tibeticum*; *P. nivea*; *Leontopodium nanum*; *Polygonum viviparum* and *Saussurea katochaete*
None	36 (12) (6: 4: 25: 1)	***Potentilla fruticosa***; *Stipa aliena*; *Helictotrichon tibeticum*; *Elymus nutans*; *Kobresia capillifolia*; *Festuca ovina*; *Saussurea katochaete*; *S. superba*; *Lancea tibetica*; and *Ligularia virgaurea* Mattf.

Species number" indicates the numbers of species, (families), and species in the functional groups of (grass: sedge: forb: shrub).

Aboveground biomass: The total aboveground biomass decreased considerably with increasing grazing intensity (Fig. 2). The total aboveground biomass in the H plot was only half of that in the L or N plots. The biomass of litter was significantly lower in the H and Mh plots than in the other plots. The N, L and Ml plots had the highest litter biomass (Fig. 2). The biomass variation in response to changes in the grazing intensity differed markedly among the functional groups (Fig. 3). The biomasses of shrubs, graminoids and sedges all decreased with the increase of grazing intensity. Forbs, however, increased in biomass in the higher-grazing-intensity plots.

The most evident biomass change was the decrease of palatable species under the higher grazing intensities (Fig. 4). The aboveground biomass of palatable species was highest in the L plot and lowest in the H plot. The ratio of palatable biomass to total aboveground biomass showed a similar pattern. The palatability of forage was significantly lower in the H, Mh, and M plots than in the other three plots (Fig. 5).

Table 4. Change in species occurrence after 17 years' livestock grazing in an alpin shrub meadow relative to the non-grazing plot.

Plot	Locally disappeared species	Invaded species
H	*Bupleurum longicaule* Wall.; *Galium verum* L.; *Helictotrichon tibeticum* Keng; *Ligularia sagitta* Mattf.; *L. virgaurea* Mattf.; *Notopterygium forbesii* Boiss.; *Ptilagrostis concinna* Roshev.; *P. dichotoma* Keng.; *Thermopsis lanceolata* R.Br.	*Ajania tenuifolia* Tzvel.; *Euphrasia tatarica* Fisch.; *Glaux maritima* L.; *Kobresia pygmaea* C. B. Clarke; *Potentilla anserina* L.; *P. bifurca* L.; *Pedicularis kansuensis* Maxim.; *Poa alpigena* Lindm.; *Polygonum sibiricum* Laxm.; *Ranunculus pulchellus* C. A. Mey.; *Trigonella ruthenica* L.; *Veronica ciliata* Fisch.
Mh	*Bupleurum longicaule*; *Gentiana straminea* Maxim.; *Helictotrichon tibeticum*; *Ligularia virgaurea*; *Oxytropis kansuensis* Bunge; *Ptilagrostis dichotoma*; *P. concinna* Roshev.; *Thermopsis lanceolata*	*Astragalus przewalskii* Bunge; *Morina chinensis* Batal.; *Kobresia pygmaea*; *Poa alpigena*; *Polygonum sibiricum*; *P. sphaerostachyum* Meissn.; *Potentilla anserina*; *P. bifurca*; *Ranunculus pulchellus*; *Trigonella ruthenica*
M	*Bupleurum longicaule*; *Geranium pylzowianum* Maxim.; *Ligularia virgaurea*; *Ptilagrostis concinna*; *Thermopsis lanceolata*	*Euphrasia tatarica*; *Kobresia pygmaea*; *Poa alpigena*; *Potentilla bifurca*; *Ranunculus pulchellus*.; *Stellera chamaejasme* L.; *Veronica ciliata*
Ml	*Bupleurum longicaule*; *Galium bungei* Steud.; *Ligularia virgaurea*; *Ptilagrostis concinna*; *Thermopsis lanceolata*	*Euphrasia tatarica*; *Halerpestes tricuspis* Hand.-Mazz.; *Koeleria cristata*; *Potentilla bifurca*; *Saussurea kokonorensis* Ling.; *Veronica ciliata*
L	*Galium bungei*; *Ligularia virgaurea*	*Koeleria cristata* Pers.; *Lonicera tibetica* Bur. et Franch.; *Polygonum sphaerostachyum*

Table 5. Similarity indices of plant composition in different plots with different grazing intensities.

	H	Mh	M	Ml	L	N
H	1.000					
Mh	0.855	1.000				
M	0.819	0.859	1.000			
Ml	0.709	0.786	0.833	1.000		
L	0.602	0.734	0.782	0.919	1.000	
N	0.598	0.720	0.756	0.899	0.921	1.000

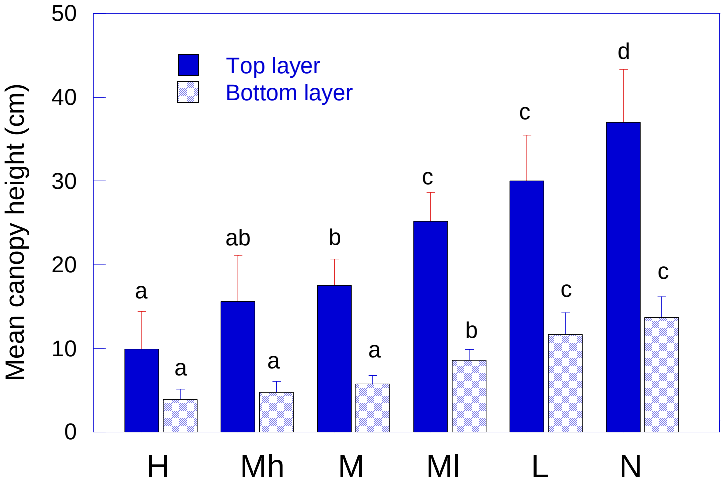

Fig. 1. Mean and SD (bars) of canopy height of *Potentilla fruticosa* meadow in plots with different grazing intensities: high (H), moderately high (Mh), moderate (M), moderately low (Ml), low (L), and non-grazed (N). Different letters indicate significant difference of the mean between plots (Duncan's multiple comparison tests; n=10; $P < 0.05$).

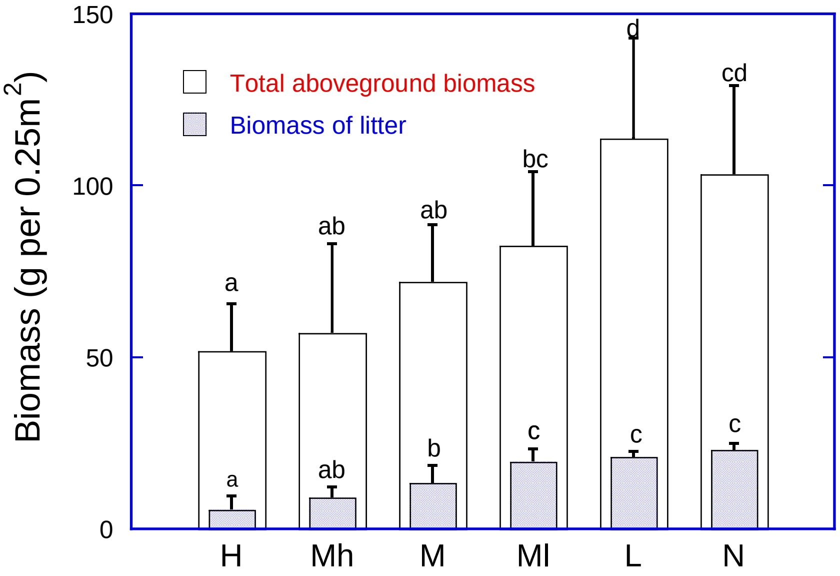

Fig. 2. Mean and SD (bars) of total aboveground biomass and the biomass of litter under different grazing intensities (see abbreviations in Figure 1). Different letters indicate significant difference of the mean between plots (Duncan's multiple comparison tests; n=10; P < 0.05).

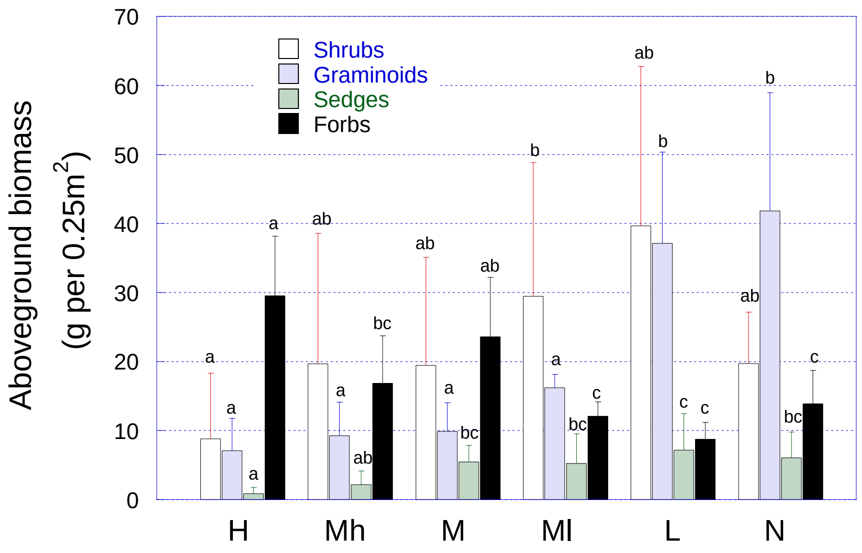

Fig. 3. Mean and SD (bars) of aboveground biomass in different functional groups under different grazing intensities (see abbreviations in Figure 1). Different letters indicate significant difference of the mean between plots (Duncan's multiple comparison tests; n=10; P < 0.05).

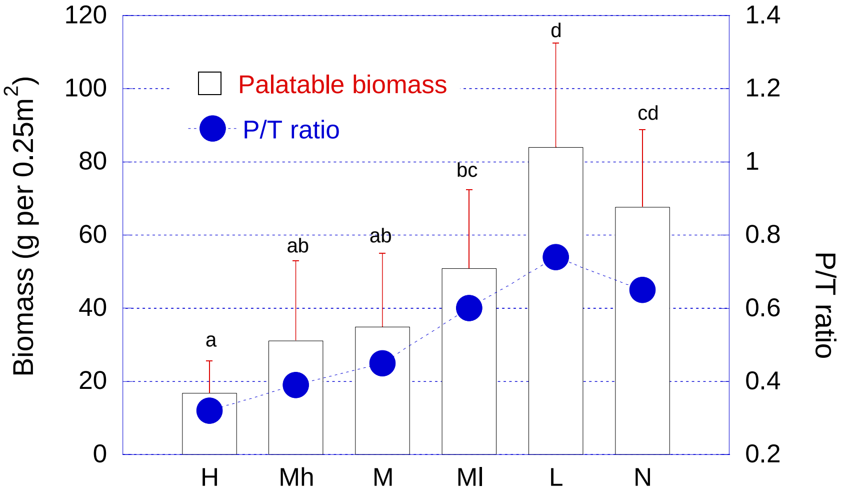

Fig. 4. Mean and SD (bars) of aboveground biomass of palatable species and the biomass ratio of palatable species to total aboveground biomass (P/T) (see abbreviations in Figure 1). Different letters indicate significant difference of the mean of palatable biomass between plots (Duncan's multiple comparison tests; n=10; $P < 0.05$).

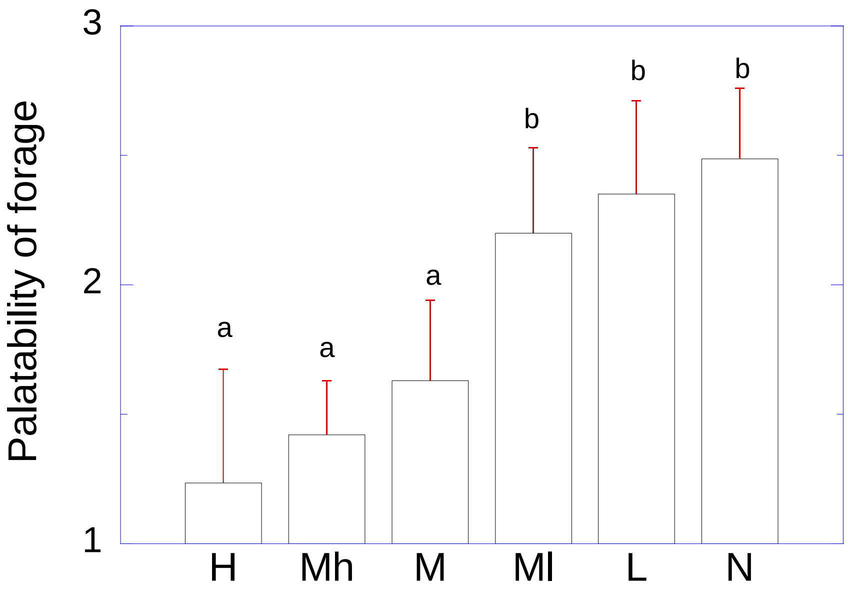

Fig. 5. Mean and SD (bars) of forage quality under different grazing intensities (see abbreviations in Fig. 1). Different letters indicate significant difference of the mean between plots (Duncan's multiple comparison tests; n=10; $P < 0.05$).

Discussion

Species composition change under different grazing intensities: The effects of grazing on species composition have been frequently documented (Fleischner, 1994; Smith *et al.*, 1996; Adler & Morales, 1999; Eccard *et al.*, 2000; Bullock *et al.*, 2001; Rodriguez *et al.*, 2003). Environmental variables was 22% of the variation in species composition and grazing-related variables was 24% (Adler & Morales, 1999). Species composition changed with grazing, in particular during the early years of a grazing experiment (Rodriguez *et al.*, 2003). Species composition seems also be affected by grazing periods and timing (Smith *et al.*, 1996). In the present study, the change of species composition in relation to grazing intensity (Table 3) can be summarized as follows:

1. The number of species moving in and out of a plot increased with increasing grazing intensity.
2. Taller, more palatable species disappeared first as grazing intensity increased.
3. More xeric species occurred more frequently under the higher grazing intensities.

Various mechanisms could be involved in the change of species composition. Selective grazing by livestock is the first (Adler & Morales, 1999; Hickman & Hartnett, 2002). Chronic overgrazing in the central Andes altered the species composition of vegetation and unpalatable or toxic species and very low-growing species were significantly more abundant on heavily grazed sites than on relatively protected sites (Adler & Morales, 1999). In our study, we found that selective grazing by sheep influenced plant composition by increasing the dominance of unpalatable forbs such as *L. nanum*, *G. diversifolia*, and *Potentilla anserina* and decreasing the abundance of palatable, perennial sedges and grasses such as *P. fruticosa*, *K. capillifolia*, and *S. aliena.* The most palatable and nutritious plants were the first to be removed under heavy grazing. Many of the invading species under the high grazing intensity were weeds that greatly reduce the fodder value of pasture. *Kobresia humilis* is a typical sedge species that is tolerant to grazing and trampling (Li & Zhou, 1998). It was also found that some forbs such as *Anaphalis lacteal*, *S. superba* and *Aster flaccidus* tolerated long-term grazing disturbance. They were almost always present in all 6 plots (Tables 3, 4). The biomass of these species also showed less change between different plots.

Grazing affects life-form composition and heavy grazing is often associated with small plant size (small leaves, low meristems etc.) and prostrate habit (Hadar *et al.*, 1999; Stohlgren *et al.*, 1999). In our study, many grazing-tolerant forbs found in high grazing plots were often dwarf like e.g., *Gueldenstaedtia diversifolia* Maxim, or prostrate e.g., *Saussurea superba* Anth. Large-sized forbs e.g., *Ligularia virgaurea* Mattf were only found in non-grazing or low grazing plots.

Secondly, grazing decreases vegetation height and litter cover (Figs. 1, 2). Similar results have been reported in other studies. For example, long-term grazing kept shrub vegetation short (Rosenstock, 1996) and prevented litter accumulation (Virtanen, 2000). The decrease in vegetation height and litter cover leads to the increase of light availability in grassland, increasing the invasion of “sun species” and the disappearance of shade-tolerant species.

Thirdly, the change of vegetation and litter cover changes the soil environment. The most prominent change could be the decrease of soil water availability with the increase of grazing intensity, which has been reported in different grasslands (Dormaar & Willms, 1998; Krzic *et al.*, 2000; Sarmiento *et al.*, 2004). The increase of xeric species observed in the present study indicates that soil water availability decreased under the high grazing intensity. The decline of soil water under high grazing intensity could also be partly due to the removal of litter cover.

Species richness and diversity: The effects of grazing on plant species richness and diversity have been frequently documented and debated (Milchunas *et al.*, 1998; Wang *et al.*, 2002; Hickman *et al.*, 2004; Sarmiento *et al.*, 2004). We still have difficulty in drawing a general conclusion on how species diversity and richness change under different grazing intensities, in part because of the wide range of grazing herbivores and forage plants and the corresponding wide range of possible interactions in these studies. However, species diversity and richness decrease under conditions with very high grazing intensity or no grazing (Hobbs & Huenneke, 1992).

Despite the marked change in species composition under different grazing intensities, we could not detect striking differences in plant species diversity or richness. The numbers of species that moved into and out of each experiment plot were very close, although the number of species movements increased with the increase of grazing intensity (Table 4). This perhaps explains why there were changes in species composition, but not in species richness. Species richness may decrease if the heavily grazed plots were isolated from species source.

The small changes of species diversity and richness under different grazing conditions in the current study may also have been due to the productive environment and rich species resource in the study area. We noticed that during the growing season, the alpine shrub meadow ecosystem is favored by good water availability and temperature conditions. About 80% of the annual precipitation is concentrated within the period from May to September. In addition, the high species richness in the alpine meadow ecosystem provides sufficient species resources for the habitats created by various grazing intensities.

Moreover, both the Shannon–Wiener index and Pielou's index indicated that species diversity tended to be highest under moderate grazing intensity (Table 2). This result seems to provide evidence for the moderate grazing hypothesis (Tilman, 1997). Further evidence should be accumulated to explain how different species acclimatize or adapt to microclimates under different grazing conditions.

Grassland degradation and long-term heavy grazing: A large livestock population will often cause grassland degradation through overgrazing, trampling, the accumulation and concentration of organic matter and chemicals from excreta, and other associated consequences (Brown & McDonald, 1995). Long-term heavy grazing was reported to cause rangeland degradation in Inner Mongolia (Kawanabe *et al.*, 1998) and the Heilongjiang Steppe of China (Tsutsumi *et al.*, 2003). Seventeen years' grazing produced some typical degradation characteristics, such as the lowest community height, lowest cover, low biomass, and low palatable herbage percentage (Figs. 1-4) in the H and Mh plots. The occurrence and abundance of *L. nanum*, *G. diversifolia*, *Lancea tibetica*, and *P. anserina* in these plots have been suggested to indicate alpine rangeland degradation (Zhao & Zhou, 1999).

However, degradation was not apparent in the low-grazing-intensity plots. This may help grassland management that aims to attain sustainable use, more palatable forage and higher species diversity (McNaughton, 1993). Moderate grazing intensity can increase not only primary productivity and species diversity, but also the proportion of palatable herbage (Pucheta *et al.*, 1998; Knapp *et al.*, 1999). The moderate grazing plot in this study lost 45% of annual aboveground biomass (Zhao & Zhou, 1999), but retained high species diversity. These facts seem to indicate that a moderate level of grazing may be desirable in the alpine shrub pastures of the Qinghai–Tibet Plateau. Our results suggest that the general guidelines of "take half, leave half" in terms of biomass may be a good measure for conservative management in the alpine meadow ecosystem. A model study also indicated that the optimum grazing intensities in warm-season and cold-season pastures are 4.11 and 3.80 Tibetan sheep per hectare (Li & Zhou, 1998). Further studies should pay attention to proper grazing management for the world's highest grassland.

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