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ORIGINAL ARTICLE

Grazing intensity and soil depth effects on soil properties in alpine meadow pastures of Qilian Mountain in northwest China

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In the mountainous rangeland of inland arid regions of Eurasia, seasonal grazing has been important to local communities for production of food, fiber, and for income, for the past thousand years. Recent population increases and other changes have put traditional grazing systems under pressure. However, empirical data describing soil properties or the impact of traditional grazing practice on the thresholds at which increase in animal stocking rate (SR) may result in degradation are lacking. Here, we provide, for alpine “typical steppe” at Qilian Mountain on the Tibetan plateau in China, a description of variation in some soil properties with soil depth, and with variation in grazing intensity. The soils studied have a humus-rich epipedon typically exceeding our sampling depth of 40 cm. As commonly reported, increased grazing intensity has correlated with depletion in soil organic carbon (SOC). Regression of our SOC data on SR indicated no “safe” threshold for grazing intensity below which SOC depletion would not occur. Other soil changes linked to increased grazing intensity in our study included a lowering of the carbon:nitrogen ratio (indicating possibly increased risk of nitrogen loss from farming systems to the wider environment), an increase in soil bulk density, a decrease in soil moisture content, and transfer of phosphorus from less intensively grazed areas toward animal night pens. Our study site is experiencing a climate warming trend which may be contributing to loss of SOC.

Keywords: alpine typical steppe; soil nitrogen; soil organic carbon; soil phosphorus; soil salt; soil water content; grazing

Introduction

Vast areas of land worldwide are unsuitable for intensive crop production because of slope, altitude, or low rainfall factors but where the vegetation cover is artificial or native grassland or shrub-land, such land is able to yield perhaps 0.5–3.0 tons herbage dry matter (DM) per ha annually and in this way support livestock grazing and provide a livelihood for rural communities and meat and milk products for neighboring urban populations. Often such landscapes are also rich in aesthetic beauty which deserves to be preserved, and provide a range of ecosystem services. Historically, management practices have been determined by the land occupiers. In the twenty-first century, there is an international trend toward regulation by government agencies seeking to ensure

both that grasslands are used in a multifunctional way to provide ecosystem services to the neighboring urban populations and that the grazing practices employed are sustainable in the longer term and not causing environmental degradation. Such land use planning requires a sound platform of scientific knowledge to provide a basis for informed discussion between graziers and regulatory authorities to evolve sound future land use practices.

Our study area comprised alpine meadow pastures (Ren et al. 2008) located on the edge of the Tibetan plateau on Qilian Mountain, in Gansu province of northwestern China. Livestock farming in this area has a history of more than a thousand years (Hou et al. 2008), which in common with many other alpine rangelands worldwide, employs a grazing system based on seasonal rotation between higher

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altitude pastures utilised in summer and lower altitude pastures utilised in winter (Watson et al. 1990; Hou et al. 2008). Specifically, at this site pastures at approximately 3000 m a.s.l. are grazed in summer, and pastures at around 2400 m a.s.l. are grazed in winter pasture (WP). In spring and in autumn, pastures at intermediate altitude of around 2700 m a.s.l. spring-autumn pasture (SAP) are used (Figure 1). The SAP tends to be utilized by graziers as a convenient hub for livestock to move to higher elevation in summer or lower altitude in winter (Helle & Aspi 1983; Larson et al. 2008). The pastures studied are part of a total area of 5900 ha of pasture managed by a single commercial entity engaged in farming wapiti deer.

Our research on the Qilian Mountain commenced in 1997 and data collection has been wide ranging, including soil and vegetation characteristics and ecology of the rangeland and economic aspects of farming operations. One of the key pieces of information required to facilitate sound decision making in longer term land use planning is knowledge of the impact of stocking rate (SR) or grazing intensity on soil properties. There are few published data on the impact of grazing intensity on soil properties, and documentation of these effects was our primary aim, though we do also present some data on the botanically dominant species at each soil sampling site. At the study site, animals are penned at night and taken by herders to daytime grazing and left to wander. Land located nearer to the pens is more often grazed than paddocks at greater distance. We hypothesized that since these farm practices have been in place for many years, it might be possible by sampling soil from sampling sites at different distances from the pens, to gain information of the longer-term impact of grazing intensity on soil

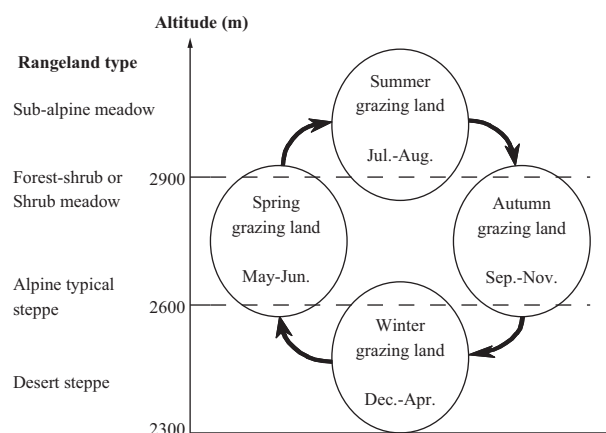


Figure 1. Schematic diagram of the seasonal grazing rotation in use at the experiment site. Animals are moved to higher altitude pastures in summer and lower altitude pastures in winter with SAP at intermediate altitude.

properties at this site. It is also likely that results for the Qinghai-Tibetan plateau would be relevant to alpine pastures in other continents such as Europe and the Americas.

Materials and methods

Study site

This study was conducted in Sunan Yugur Autonomous County in Gansu Province, China (38.8°N, 99.6°E), on rangeland classified by Wu et al. (1980) as “Alpine Typical Steppe.” The elevation ranges from 2600 to 3000 m and the study areas are located on wide, flat “outwash” plains with slope <3°. The data presented here were collected from a 275 ha block of SAP at 2700–2900 m grazed from May to June and August to November and from a 160 ha block of WP at 2600–2750 m altitude grazed from November from April by Wapiti deer (Supplemental data, Figure 1), used in a seasonal rotation as discussed above (Figure 1). The annual mean temperature (1957–2012) is 3.6°C and the annual accumulated temperature $\geq 0^{\circ}\text{C}$ is 2336°C with 127 frost-free days per year and an average of about 60% of the daily photoperiod as full sunshine. The mean annual rainfall (1957–2012) is 260 mm (Figure 2). Annual raised pan evaporation is 1785 mm. Generally, spring growth and greening of herbage commences in late April and winter yellowing occurs from late August. These alpine soils are referred to by several authors (see e.g. Dong et al. 2013) as “Black soils” and the authors are not aware of any published classification using the USDA (United States Department of Agriculture) Soil Taxonomy System but we believe they would key out as Cryolls, a suborder of Mollisols. Other Mollisols in the region have been discussed by Kravchenko et al. (2011). Given that there is a very large area of this class of soil in the high-altitude regions of Gansu and

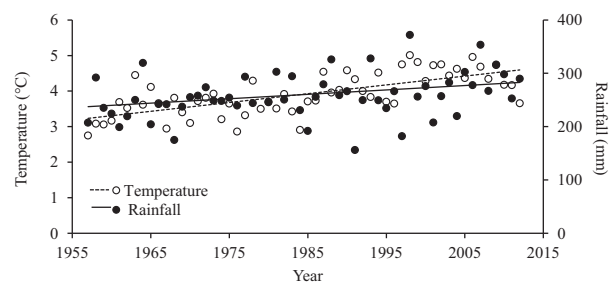


Figure 2. Rainfall and mean annual temperature from 1957 to 2012 in the vicinity of Sunan Yugur Autonomous County, 20 km from the experiment site. Data are from the Gansu Meteorological Bureau. Regression equations are: temperature ($^{\circ}\text{C}$) = $3.2078 + 0.0248^{\circ}\text{C yr}^{-1}$, $R^2 = 0.465$, $p < 0.0001$; rainfall (mm) = $237 + 0.822 \text{ mm yr}^{-1}$, $R^2 = 0.082$, $p = 0.033$.

Qinghai provinces and in the Tibetan autonomous region of China, and in neighbouring countries, there is a need for further study of the soil properties.

SR gradient for sampling

In these grazing systems, animals are penned at night and led out to surrounding rangeland pastures by herders during the day, with the result pastures close to the pen are subject to the heaviest grazing intensity, with gradually diminished herbage utilization with greater distance away from the pen (Hou et al. 2003). After consulting with herdsmen about their grazing practices which they advised had been unchanged over a number of years, sampling areas located 0 m, 300 m, 600 m, 900 m, 1200 m, and 1500 m from the pens in both WP and SAP, and expected to display a grazing intensity gradient, were selected and marked with stones to facilitate revisiting for soil and herbage sampling (Hou et al. 2003). In 1999, animal grazing behavior observations were carried out on 15 consecutive days for both SAP and WP to ascertain the level of grazing activity at various distances from the night pens. Fifteen Wapiti deer around 5–6 years old were marked and observed by telescope from a distance so as not to disturb their normal grazing behavior. The locations of the marked animals were logged over a 15-minute interval within each 2-hour period of each observation day. These grazing observation data were then integrated over time to convert the observed grazing intensity to the equivalent expressed as a SR for a continuously grazed pasture (Wang et al. 2013). Briefly, the calculation was as follows:

$$SR_i = SR_M \times \frac{\sum Hd_i}{\mu_{i=1:n} \sum Hd_i},$$

where SR_i = equivalent SR under continuous grazing of the grazing intensity at the i th sampling area, SR_M = mean SR for this pasture block calculated as number of animals/total area, $\sum Hd_i$ = cumulative head count of animals observed grazing at the i th sampling area during the observation periods, and $\mu_{i=1:n} \sum Hd_i$ = average cumulative head count for all six sampling areas. Animal SR data were converted to “animal units” used in agricultural extension practice in China (1 animal unit = one 50 kg female sheep rearing a lamb to weaning). The herders advised in 2012 that grazing practices had not changed since 1999, so for data analysis purposes the grazing intensity distribution was assumed to be the same in 2012 as in 1999. A modest SR increase arising from the inclusion of a small number of yaks grazing with the deer was factored into the 2012 grazing intensity calculations, however.

Soil sampling and analysis

Soil samples were analyzed in State Key Laboratory of Grassland Agro-ecosystems, College of Pastoral Agriculture Science and Technology, Lanzhou University, Lanzhou, Gansu 730020, China. Soil sampling at each of the six identified sampling areas along the SR gradient was carried out as described below in July 1999, and again in July 2012. To collect soil samples, at each of the six sampling areas, three plots (100 m × 100 m) were marked, three parallel 100 m transects 30–40 m apart were randomly placed within each plot, and 3–4 quadrats (1 m × 1 m) were set at 30-m intervals along each 100 m sampling transect. Using a soil auger and sampling to 40 cm depth in 10 cm steps, soil samples for four soil depths (0–10, 10–20, 20–30, and 30–40 cm) were collected in every quadrat. In this way, soil cores were collected from 10 points and mixed to obtain a composite sample, and three replicate samples from each of four soil depths were collected from the different 100 m × 100 m plots in each sampling area. Soil was sub-sampled from the evenly mixed composite sample by quartering, then brought to the laboratory, air-dried at room temperature and ground and passed through 1-mm sieves for chemical analyses (Jastrow et al. 1998; Wang et al. 2002; Li et al. 2005). Soil bulk density (BD) was determined by the core method (Blake & Hartge 1986). Cutting rings used to obtain samples for soil BD determination were 100 cm³ in volume (5 cm diameter and 5 cm height) and again samples were collected from four soil depths as above and were aggregated from 10 samples in different quadrats in plots within sampling areas. Wet soil cores were weighed, dried in a fan-assisted oven set at 105°C for 48 h, and weighed again to determine the dry soil weight for BD calculation, with soil water content determined by difference between wet and dry weights (Ferraro & Ghera 2007).

Soil organic carbon (SOC) was analyzed using standard procedures for determination of SOC content (Ding et al. 2004; Long et al. 2010). Total nitrogen (TN) was measured by the Kjeldahl method (Hirota et al. 2005), available nitrogen (AN) was extracted using the micro-diffusion method (Hirota et al. 2005). Soil total phosphorus (TP) was determined using a flame photometer (Dick & Tabatabai 1977; Borie et al. 1989), while available phosphorus (AP) was extracted using the Olsen method (Olsen & Dean 1965). To determine soil soluble salts, a 20 g sample of soil and 100 ml water de-gassed by boiling were placed in a 250 ml closed flask and shaken for 3 minutes. The extract was then filtered with a standard filter paper, and a 25 ml aliquot evaporated to dryness in a pre-weighed

crucible, then wet with 15% H₂O₂ until the residue whitened. The crucible was then re-dried and re-weighed and, hydrogen peroxide treatment repeated for several cycles until no further weight loss occurred on H₂O₂ treatment, and the soluble salts taken as the residue determined by the difference between final and initial crucible weights.

Botanical data

Aboveground vegetation biomass was sampled at the same time as the soil sampling described above. Herbage was cut to ground level from nine 0.25 m² quadrats in each of the three plots within each of the six sampling areas in July of 1999 and 2012. Harvested herbage was spread out on a table and herbage of three four visually dominant species was sorted into individual piles which were dried at 65°C for 48 h and weighed.

Statistical analysis

Trends over time in mean annual temperature and rainfall were evaluated by ordinary least squares (OLS) linear regression in SAS Institute (2011). To gain an overview of change in soil properties with soil depth, soil test data for each of the four soil depths were averaged over 1999 and 2012 measurements for WP and SAP pasture types, and significance of the soil depth effect also analyzed by OLS linear regression, as was the relationship between grazing intensity and distance from night pens along the soil testing transects. For a more comprehensive analysis of soil properties, multiple regression analyses were performed separately for data of each pasture type (WP and SAP) in each year (1999 and 2012), for each soil characteristic. In these regression analyses, the effects of grazing intensity and soil depth and their interaction on the soil characteristics were assessed using the General Linear Model (GLM) procedure in SAS, and a rejection level of $p = 0.05$. Type III sums of squares and a command to output solutions for regression coefficients were specified in the model.

Results

Meteorological data

A warming trend of 0.248°C per decade was detected from regression of the temperature data for the period 1967–2012 on time (intercept 3.23°C mean annual temperature, $R^2 = 0.465$; $p < 0.0001$). There was a less pronounced indication of increasing rainfall, of 8.22 mm per decade (intercept 238 mm total annual rainfall, $R = 0.286$; $p = 0.033$) (Figure 2).

Grazing intensity variation with distance from animal pens

The hypothesized gradient in grazing intensity was confirmed although grazing intensities were somewhat higher on WP than on SAP as a result of the smaller area of WP. In WP, effective SR reduced in a near linear relationship with distance from 7.10 AU ha⁻¹ at sampling areas adjacent to the pens where animals shelter at night to 1.05 AU ha⁻¹ at 1500 m distance. In SAP, the corresponding reduction was from 3.48 AU ha⁻¹ in sampling areas adjacent to pens to 0.20 AU ha⁻¹ at 1500 m distance (Supplemental data, Figure 2).

Change in soil properties with soil depth

In general, the humus-rich mollic epipedon extended below the 40 cm sampling depth and most of the soil properties measured varied comparatively little with depth. For the averaged data, OC, TN, AN, and TP showed a modest but statistically significant decrease with soil depth and BD increased with soil depth, while AP and salt concentration showed more substantive soil depth effects (Figure 3). Soil AP was some 3A higher in the top 20 cm than in the 20–40 cm layer. Soil salt concentration was markedly higher in the subsoil than in the topsoil, especially in the 30–40 cm soil depth, and exceeded 5 kg m⁻² for the 30–40 cm soil depth in WP (Figure 3). For specific years and pasture types, there was some variability but data for SOC, TN, TP, and BD in particular largely reflected patterns in the averaged data (Table 1).

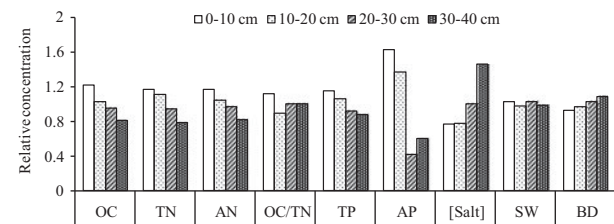


Figure 3. Relative variation of measured soil properties with soil depth. For each soil depth, each measurement is expressed as a proportion of the mean for 0–40 cm soil depth. Data are averaged across SAP and WP types and across 1999 and 2012 sampling dates. Values of means and their units are shown in Supplemental data, Figure S3. SOC, soil organic carbon; TN, soil total nitrogen; C/N, carbon nitrogen ratio; STP, soil total phosphorus; SAN, soil available nitrogen; AP, soil available phosphorus; [Salt], soil salt concentration; WC, soil water content; BD, soil bulk density.

Table 1. Change in soil properties with grazing intensity (GI) and soil depth (SD) in SAP and WP in 1999 and 2012 measurements.

Soil characteristics	Year	Pasture	Regression parameters ^a				<i>R</i> ²	<i>p</i> ^b
			Intercept (SE)	GI	SD	GI × SD		
OC (%)	1999	SAP	13.3948 (1.2490)	-1.8002	-0.0626	-0.0263	0.7410	<0.0001
		WP	7.3268 (0.4645)	-0.2795	-0.1298	0.0096	0.8215	<0.0001
	2012	SAP	13.5725 (0.9499)	-2.2654	-0.1607	0.0265	0.8716	<0.0001
		WP	4.8991 (0.2377)	-0.0170	-0.0367	-0.0032	0.8549	<0.0001
TN (g kg ⁻¹)	1999	SAP	16.0283 (0.7775)	-2.4645	-0.1218	0.0070	0.9128	<0.0001
		WP	11.4705 (0.6743)	-0.7417	-0.2043	0.0188	0.8475	<0.0001
	2012	SAP	19.2916 (1.0374)	-2.4058	-0.2267	0.0264	0.8683	<0.0001
		WP	13.9279 (1.1295)	-0.5667	-0.1207	0.0072	0.5920	0.0006
AN (mg kg ⁻¹)	1999	SAP	80.9819 (9.5206)	2.0204	-0.5242	-0.2211	0.5138	0.0020
		WP	101.7670 (7.1778)	-2.5138	-0.5557	-0.0075	0.4691	0.0047
	2012	SAP	14.2152 (1.2806)	-0.0927	-0.1728	-0.0184	0.7409	0.0001
		WP	35.0461 (2.3615)	-2.8152	-0.4196	0.0261	0.8424	<0.0001
OC/TN	1999	SAP	8.0197 (1.5934)	0.4380	0.0417	-0.0099	0.0456	0.8117
		WP	6.1326 (0.4089)	0.2980	0.0079	0.0006	0.6559	<0.0001
	2012	SAP	7.0707 (0.4795)	-0.5063	0.0042	0.0008	0.6481	<0.0001
		WP	10.7141 (0.2612)	-1.1328	-0.2468	0.0411	0.2088	0.1874
TP (g kg ⁻¹)	1999	SAP	0.0463 (0.0036)	-0.0387	-0.0006	0.0002	0.7794	<0.0001
		WP	0.0675 (0.0069)	0.0014	-0.0004	0.0001	0.3647	<0.0257
	2012	SAP	0.0927 (0.0054)	-0.0035	-0.0016	0.0001	0.8764	<0.0001
		WP	0.1403 (0.0167)	0.0056	-0.0017	0.0003	0.6652	<0.0001
AP (mg kg ⁻¹)	1999	SAP	38.3502 (3.8672)	-4.8419	0.7227	-0.3870	0.9380	0.0048
		WP	42.3572 (2.0004)	-3.9632	-1.8753	0.2249	0.9659	<0.0001
	2012	SAP	3.2961 (0.4086)	-0.3641	0.1121	-0.0008	0.8926	<0.0001
		WP	-0.7598 (0.9024)	0.1996	0.1589	-0.0152	0.6005	0.0003
[Salt] (kg·m ⁻²)	1999	SAP	2.3025 (0.4374)	-0.2347	0.0552	-0.0181	0.6909	<0.0001
		WP	0.5076 (0.9798)	0.1156	0.1550	-0.0187	0.5529	0.0009
	2012	SAP	1.5142 (0.3512)	-0.0241	0.0156	-0.0044	0.1492	0.3462
		WP	0.1969 (0.8656)	0.1627	0.1261	-0.0100	0.5331	0.0014
SW (%)	1999	SAP	9.3947 (0.5353)	-1.0605	0.1807	-0.0323	0.9383	<0.0001
		WP	4.0061 (0.2936)	-0.0971	0.1833	-0.0091	0.9675	<0.0001
	2012	SAP	29.3556 (1.6884)	-1.5091	-0.0845	-0.0339	0.8029	<0.0001
		WP	20.9320 (1.1579)	-0.0580	-0.0864	-0.0258	0.7746	<0.0001
BD (Mg m ⁻³)	1999	SAP	0.6979 (0.0434)	0.0960	0.0043	0.0009	0.8957	<0.0001
		WP	0.5424 (0.0452)	0.0563	0.0149	-0.0006	0.9170	<0.0001
	2012	SAP	0.5939 (0.0584)	0.1123	0.0068	-0.0019	0.6918	<0.0001
		WP	1.0564 (0.0138)	-0.0027	-0.0019	0.0001	0.0847	0.6122

^aAll intercepts were significant ($p < 0.05$) except for [Salt] in WP in both years; other significant parameters are highlighted in bold.

^bThe R^2 and p values listed are for the multiple regression models in each case.

SOC, soil organic carbon; TN, soil total nitrogen; C/N, carbon nitrogen ratio; STP, soil total phosphorus; SAN, soil available nitrogen; AP, soil available phosphorus; [Salt], soil salt concentration; WC, soil water content; BD, soil bulk density.

Change in soil properties with distance on the grazing intensity gradient and with time

For SAP, SOC was strongly reduced with increasing grazing intensity (Table 1), or with proximity to the night pens (Figure 4). However, in WP, the equivalent pattern was only weakly seen in 1999, and was not statistically discernible in 2012 (Table 1, Figure 4). For WP, SOC as indicated by the intercept of the regression analysis (Table 1), fell from 7.3 (± 0.47) to 4.9 (± 0.24)% between 1999 and 2012. While the regression intercept for soil OC% in SAP appeared stable from 1999 to 2012, the mean OC% as assessed by analysis of variance and indicated in Figure 4, fell from 8.5% to 6.8% (s.e. + 0.27%). Soil TN, like SOC, decreased with increasing grazing intensity, and more strongly so on SAP than WP. However unlike SOC, soil TN increased in both SAP and WP between 1999 and 2012, as indicated by regression intercepts (Table 1).

For SAP, despite a fall in soil OC and rise in soil TN between measurements in 1999 and 2012 the data separated based on grazing intensity (Table 1), but the resulting C:N slopes did not differ significantly with time and had a near 1:1 stoichiometry (Figure 5a). Since the C:N ratio at low grazing intensity on SAP in 1999 was approximately 10:1, the near 1:1 stoichiometry between C loss and N loss along the grazing intensity gradient indicates a narrowing of the C:N ratio in soil, associated with and more or less linearly proportional to grazing intensity (Figure 5a). For WP, the situation was more complex, with the grazing intensity effect on soil TN and OC much weaker and the narrowing of the C:N ratio to below 4:1 at some sampling sites in WP in 2012 (Figure 5a). When the soil TN and OC data were plotted in three dimensions against grazing intensity for the respective soil sampling locations, it is evident that depletion of soil OC and reduction in the soil C:N ratio is strongly associated with increased grazing intensity (Figure 5b).

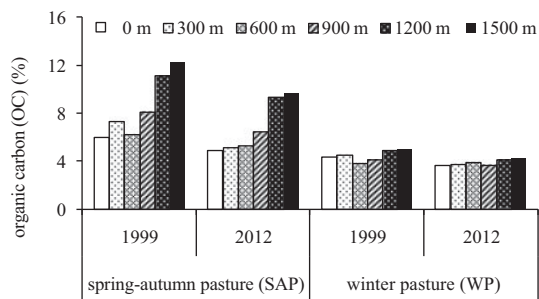
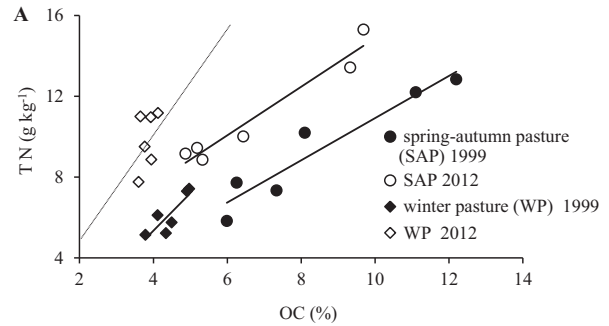


Figure 4. Variation in SOC% with distance from animal night pens. Grazing activity reduces with increasing distance from the pens; hence, the lower values at the left of each series indicate the effect of increased grazing intensity in reducing soil organic carbon (SOC).



SAP 1999: $TN = 1.045 (\pm 0.1668) OC + 0.476$, $R^2 = 0.908$, $F_{1,4} = 39.28$, $P = 0.0033$;
 SAP 2012: $TN = 1.203 (\pm 0.1419) OC + 2.843$, $R^2 = 0.947$, $F_{1,4} = 71.91$, $P = 0.0011$;
 WP 1999: $TN = 1.857 (\pm 0.5576) OC - 2.080$, $R^2 = 0.735$, $F_{1,4} = 11.08$, $P = 0.0291$;
 WP 2012: $TN = 3.453 (\pm 3.0507) OC - 3.371$, $R^2 = 0.243$, $F_{1,4} = 1.28$, $P = 0.3210$.

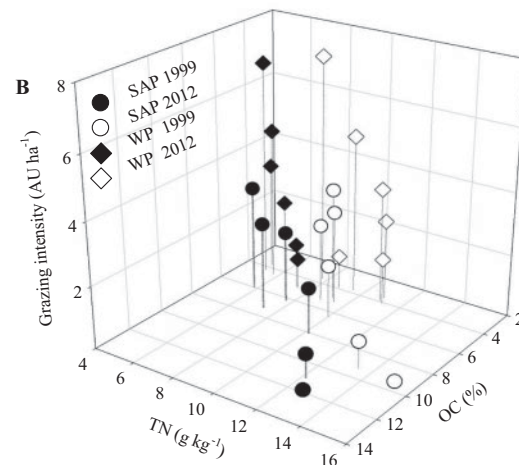


Figure 5. (A) Relationship between soil organic carbon (SOC) and total nitrogen (TN). The dashed line at left indicates a C:N ratio of 4:1 and it is seen that C:N ratio of the soil reduces systematically with SOC decline from 1999 to 2012. The stoichiometry of the relationships is given in the equations below. (B) Visualization of the link between increased intensity and reduced SOC. The Figure plots organic carbon (OC) and TN data with grazing intensity [animal units (AU) ha^{-1}] as a third dimension.

Soil water contents were higher at sampling in 2012 than in 1999. Unexpectedly, there was a significant decrease in soil moisture content with increased grazing intensity both in 1999 and 2012 (Table 1), and further investigation indicated that this was linked to the decline in SOC at increased grazing intensity (Figure 6).

Soil BD was significantly increased with increased grazing intensity, for SAP both in 1999 and 2012. Although the “compaction coefficient” (representing the grazing intensity effect on soil BD from regression analysis) superficially increased from 0.0960 $Mg m^{-3}$ per animal unit per ha in 1999 to 0.1123 in 2012 (Table 1) the associated standard error of the compaction coefficients was ± 0.023 , so compaction increase over time was not statistically significant. For WP in the 2012 measurement, however, the soil

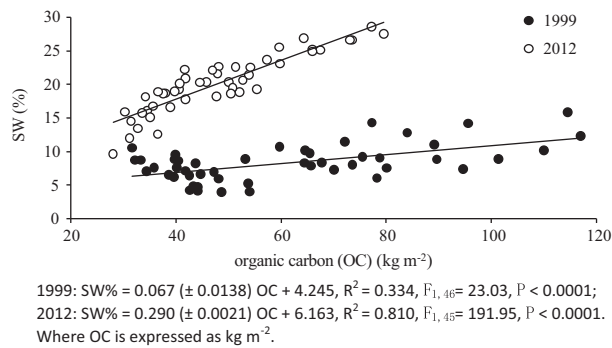


Figure 6. Relationship between soil water (SW)% and soil organic carbon (SOC) at sampling in 1999 and 2012.

BD profile was not displaying soil depth or grazing intensity effects but had increased to a uniform value of just over $1.0\ Mg\ m^{-3}$ (Table 1; Supplemental data, Figure 3).

Botanical composition

Botanical composition data are presented in Supplemental data, Table 1. In SAP, at low grazing intensity, pasture communities were dominated by *Stipa purpurea* and *Potentilla anserine* in 1999, with a shift to increased abundance of *Anaphalis lacteal* and *Potentilla multifida* in 2012. Dominant species of the SAP pasture community at medium grazing intensity were *S. purpurea* and *Kobresia capillifolia* in 1999, and *S. Purpurea* and *Potentilla bifurca* in 2012. At high grazing intensity, dominant species in 1999 were *Achnatherum inebrians*, which is poisonous and *Convolvulus ammannii* which is undesirable, with a marked shift toward *S. purpurea*, *Carex atrofusca*, and *K. capillifolia* in 2012.

In WP, at low grazing intensity, the pasture community was dominated by *S. purpurea*, *K. capillifolia*, and *Agropyron cristatum* in 1999, with a shift to *Stipa breviflora*, *Leymus secalinus*, and *Heteropappus altaicus* in 2012. At medium grazing intensity, the pasture community was dominated by *A. cristatum*, *S. breviflora*, and *S. purpurea* in 1999, while 13 years later the dominant species were *S. breviflora*, *A. cristatum*, and *L. secalinus*. At high grazing intensity, the dominant species were *A. inebrians* and *C. ammannii*, with a shift to *S. breviflora*, *A. cristatum*, and *H. altaicus* 13 years later.

Discussion

Climate trends

The finding of a rising temperature trend at this site agrees well with other analyses of weather data for the region that have emerged in recent years. For example, a comprehensive analysis of 81 weather

stations on the northern Tibetan plateau by Cuo et al. (2013) reported a general increase per decade in maximum and minimum temperatures of $0.4^\circ C$ and $0.3^\circ C$, respectively. Also, Dong et al. (2013) reported a $1^\circ C$ rise in temperature since the 1950s at two different sites in neighboring Qinghai Province. Reported precipitation change is not only less pronounced from a statistical perspective but also regionally variable and can have complex ramifications such as increased incidence of flooding if increased rainfall arises from an increased intensity of rainfall events rather than an increased number of rain days (Qian & Lin 2005). Hence, it is difficult to make general predictions about the likely present and future impact of the observed meteorological trends for temperature and total rainfall.

However, increased temperatures could be expected to exacerbate summer moisture deficit from an agricultural perspective, and while the authors are not aware of a similar analysis for the Tibetan plateau, Liu et al. (2009) have reported reference evapotranspiration to be rising at $4.8\ mm\ yr^{-1}$ since 1994 in northwest China as a result of rising temperature and reduced average wind speed. Rising temperature would also tip the soil C sequestration/oxidation balance more in favor of oxidation, and this point will be discussed further below.

Pattern in vegetation data

Significant rangeland degradation occurred in China over the period 1950–1990, but more recently the government has implemented policies to reverse degradation (Wang et al. 2010). This degradation was linked both to government policies of that time which resulted in increased SRs on much of China's rangeland (Wang et al. 2010), and population increase. Zhang et al. (2004) report a fourfold increase in human and livestock populations from 1954 to 2000 in one part of the Tibetan Plateau for which they gathered records. Hence, one question for the present study was whether or not there would be evidence of degradation reversal between 1999 and 2012 measurements. For SAP, this seems to be the case. In particular, in high grazing intensity pasture, the dominant species had changed from *A. inebrians* and *C. ammannii* (indicator species for pasture degradation) in the 1999 record, to *S. purpurea*, *C. atrofusca*, and *K. capillifolia* in 2012. The latter species are considered more appropriate for grazing. With lower and medium grazing intensity in SAP, the communities were not showing signs of degradation in 1999, and remained in good condition in 2012.

Grazing intensity of WP was higher than SAP (Supplemental data, Figure S2), and as noted above

lower energy levels of WP herbage result in a correspondingly larger herbage demand of animals. Hence, the botanical data for 1999 indicated serious degradation at high grazing intensity with a dominance of *A. inebrians* and *C. ammannii*. After 13 years of winter grazing, the pasture had improved to some extent, but some *C. ammannii* remained. For medium and low grazing intensity WP, there was no obvious change in species dominance after 13-year grazing.

Previous studies of rangeland degradation on the Tibetan Plateau

In seeking to place the present results in context with findings of other researchers, it emerges that several studies in the last decade have addressed the question of land degradation across China and on the Tibetan Plateau in particular, but that approaches and thinking on the issue are as yet rather uncoordinated. For example, Han et al. (2008) define rangeland degradation as “a decrease in plant species diversity, plant height, vegetation cover and plant productivity” but they note that others extend the term to include a much wider range of effects, including loss of ecosystem function and decline in soil carbon status. Wu et al. (2014) used a number of criteria to define four stages of degradation from light to extreme. The soil OC criteria for these ranged from >20% for original vegetation to <5% for extreme degradation, while biomass for sites so-classified ranged from 1520 kg DM/ha in the first case to 160 kg DM/ha in the latter case. On these criteria, the Qilian Mountain pastures studied would be considered moderately to severely degraded based on soil carbon status, but would be considered less degraded on herbage biomass criteria. Li et al. (2014) considered an extensive range of factors including among others: % bare ground, small mammal burrow density, average proportion of Cyperaceae, and average proportion of Poaceae species present. These authors concluded from multivariate analysis of data spanning this wider range of criteria that degradation categories determined in earlier studies are not discrete states, but rather overlap along a continuous gradient. This is consistent with the continuous change in soil OC along a gradient of increasing grazing intensity (Figure 5b).

Grazing intensity effects on SOC

Our primary hypothesis was that sampling along grazing intensity gradients would reveal change in soil properties attributable to grazing activity and so help in determination of SRs sustainable in the

longer term on these soils. The hypothesis was largely confirmed, in that soil OC and TN showed highly significant increase with decreasing SR in every case except for OC in WP in 2012 (Table 1; Figure 4 and Figure 5a). One possible explanation for the very “flat response” of soil OC to grazing intensity in 2012 (and 1999) might be that actual grazing intensity in WP was more uniform with distance from night pens when integrated over the entire season than it was during the particular time period in 1999 when observations of distribution of grazing intensity along the distance transect were made. Grazing of WP at this site is more intensive than grazing of SAP, both because SR is higher as a result of the smaller area and longer grazing duration (Supplemental data, Figure S2) and because frosted winter herbage has a lower nutritive value and metabolizable energy level than fresh spring herbage and so animals require higher intakes. Following methodology widely used in New Zealand and Australia (C. Matthew, personal communication) and applied in China by Chen et al. (2010), we performed energy demand calculations based on animal numbers and body weights recorded by Hou et al. (2003) and on updated SR data for 2012. Even after allowing for a winter feed saving in the form of a 20% weight loss of animals in winter, regained while grazing SAP and summer pastures, the estimated annual herbage consumption for WP was 1.03 and 1.07 t DM ha⁻¹ yr⁻¹ in 1999 and 2012, respectively, with the corresponding values for SAP being 0.51 and 0.53 t DM ha⁻¹ yr⁻¹ (details available from the authors on request). This higher annual feed requirement on WP could have led graziers to vary their grazing strategy as the winter progressed and utilize pastures further from the night pens to a greater extent than on SAP after the collection of grazing distribution data.

Despite these reservations, it is clear that Figure 5a and 5b describes a soil OC decline linked to grazing that is approximately proportional to grazing intensity. In this sense, Figure 5a defines a general trajectory from high right to low left which shows a progressive, continuous shift in soil status characterized by loss of soil OC. The further decline in soil OC between 1999 and 2012 (Figures 4 and 5a) has at least two possible interpretations. This decline may represent the cumulative effect of grazing intensity over time, or it may be an independent effect arising from the climate warming indicated in Figure 2. Information held by the authors from a major experiment at another site on the Tibetan plateau (in preparation for publication) indicates that annualized soil respiration for similar Tibetan Plateau pastures at Maqu in Gansu province was 28.0 t CO₂ ha⁻¹ yr⁻¹ while ecosystem photosynthesis was

25.3 t CO₂ ha⁻¹ yr⁻¹, or 270 g m⁻² yr⁻¹. Allowing for a 3.67 conversion factor for the difference in molecular weight of elemental C and CO₂ and a mean soil BD of 0.97 Mg m⁻³, the net soil OC loss over time averaged across the Qilian Mountain data from this study is 306 g m⁻² yr⁻¹, remarkably similar to the annual soil CO₂ loss determined at Maqu. However, it should be kept in mind that the present experiment did not have replicates at the whole farm level to confirm that the results are general on a regional basis, so this is a point for further research. An important conclusion from our study is there is no lower threshold of grazing intensity below which soil OC loss does not occur. This appears to be a new finding, at least for China, as previous studies we are aware of have not examined soil properties like OC% along a structured grazing intensity gradient.

Stoichiometry of soil TN and SOC depletion under grazing

Parsons et al. (2013) in a theoretical discussion of ecosystem feedback mechanisms between N and C, note in their Table 1 that lactating herbivore animals return 30% of ingested herbage C and 85% of ingested herbage N to the soil via excreta, with a similar proportion of C and a somewhat higher proportion of N in animals reared for meat. This then indicates a mechanism for “concentration” of N in soils of these grazing systems that would operate in proportion to animal grazing intensity. This effect likely explains the shift between 1999 and 2012 in soil OC:TN ratio, and concentration of N in this way would be further exacerbated by the practice of recovering dung from night pens for use as domestic fuel, since a large part of animal N excretion is associated with urine rather than dung. It is almost certain that disassociation of C and N previously tightly linked in soil organic matter, in this way, would increase the risk of N loss to ground water and river systems. This point is also little recognized in previous studies of rangeland degradation on the Tibetan plateau or elsewhere in China, and indicates an important issue needing to be explored in ongoing research aimed at identifying sustainable food production practices and environment protection guidelines.

Other soil properties

The soils of the Tibetan plateau have been inventoried at a regional level in terms of questions like their carbon storage (Ni 2002; Wang et al. 2002), but there appear to be few if any published data describing their intrinsic properties in more detail.

Mollisols are also found in northeast China, where the colder winter temperatures required for their formation occur at lower altitudes, and where they are extensively cropped and there is some published data on their properties (e.g. Kravchenko et al. 2011). Data from the present study (Figure 4) highlight the depth of the mollic epipedon with properties relatively unchanged (compared to lowland mineral soils) from the surface to beyond the 40 cm sampling depth employed in this study, and its high soil OC content with associated N and P. By contrast with the present results, typical SOC, TN, and total P levels for a loessial soil in the same region at lower altitude are 2.64 kg C m⁻² (approximately 2.4% C) (Liu et al. 2011), 0.33 kg N m⁻², and 0.32 kg P m⁻² (Liu et al. 2013), respectively. In the case of loessial soils with poorly formed topsoil SOC can at times be < 1.0 kg C m⁻² (Huang et al. 2007).

Soil AN is known to vary widely within a few weeks, depending on background conditions. For example, Ball and Field (1982) show soil AN increasing by almost an order of magnitude after an autumn rainfall event and attribute the effect to heightened microbial activity triggered by the rainfall. Therefore, the point-of-time values for AN (Table 1) from this study are hard to interpret, except to note that they do indicate that similar temporal variation in soil AN occurs on the Tibetan plateau.

A large proportion of soil TP is typically unavailable to plants, but soil AP is generally more stable than soil AN is used in some countries as a general measure of soil nutrient status. In this study, the major feature in the soil AP data is an increase along the grazing intensity gradient toward the night pens (Table 1) which can be attributed to nutrient transfer in animal excreta.

In studies on salinity, salt level is often measured in units of electrical conductivity, and only a minority of the available reports quantify salt concentration. Hence, it is difficult to benchmark the present results against other studies. However, Li et al. (2011) in surveying soils of alluvial plains in the lower reaches of the Yellow River define a soil salt content of 4–6 g kg⁻¹ (1.3–1.7 dS m⁻¹) as highly saline and >6 g kg⁻¹ (>1.7 dS m⁻¹) as very highly saline. At the higher end of the spectrum, Glenn and Brown (1998) considered a salt concentration of 85 mol m⁻³ (approximately 6 g kg⁻¹) as optimal for the halophyte species *Atriplex canescens* and a concentration of 520 mol m⁻³ (approximately 30 g kg⁻¹) as supra-optimal for that species. In comparison to these benchmarks, the average value for soluble salts obtained in the upper 10 cm of soil in WP and SAP was approximately 10 g kg⁻¹ and up to 35 g kg⁻¹ was recorded for the 30–40 cm soil depth in WP.

This high salt level needs to be verified by repeat measurement in a future study, but it does seem clear that the salt load of these soils is significant and an implication is that locally adapted species of grasses and legumes might be screened as a source of genetic material conferring salt tolerance in plant breeding programs, not only in China but also worldwide. In terms of land use planning, anecdotal observation suggests that salt at depth is likely to be brought to the surface or move laterally in ground water when water tables rise during snow melt in spring, and this issue clearly needs to be considered in developing management plans for these grasslands. We have not noted previous reports of a relationship between depleted soil OC and soil moisture, perhaps because it is less common to sample and report soil moisture as a component of soil spatial variability. However, a direct correlation between reduced soil OC and decreased soil water content is clearly seen in Figure 6 and from the slope coefficients of those data, reduction in soil moisture storage in land near the night pens compared with land 1.5 km distant is about 15×10^4 liters ha^{-1} . It could be expected then that loss of soil OC will impact negatively on the resilience of the grazing lands in the case of summer drought, and this issue may become more significant in future if summer evapotranspiration is increased by climate warming.

While soil BD was apparently increased by increased grazing intensity, except for WP in 2012 (Table 1) care is needed in assigning a cause of this effect. Soil BD was similarly correlated with animal grazing intensity and soil OC ($R = 0.526$ and -0.582 , respectively). Hence, increase in soil BD might relate either to compaction effect from impact of trampling or to loss of soil OM where the OM fraction had a lower specific gravity than the mineral fraction.

To conclude, our data allow quantification of some major livestock grazing effects on soil properties for these vulnerable soils, including herbage consumption, excreta return, and treading effects. The data indicate that the livestock grazing effects are linearly correlated with SR. Herbage consumption intercepts C that would normally be returned to the soil and discharges it instead to the atmosphere (Parsons et al. 2013), resulting in a decline in soil C in proportion to grazing intensity. There is no lower threshold below which the defoliation effect is negligible, and the effect is also cumulative with time. In parallel with soil C loss through grazing activity, soil N is “concentrated” to the point that soil C:N ratios in the soils studied narrowed to less than 4:1 in those sampling points with 13 years continuous exposure to high grazing intensity. Further research is needed to assess if defoliation will lead to equilibration of the

soil–plant–animal system to lower soil OC levels after time and the implications of soil N concentration for eutrophication of waterways and other water bodies.

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Supplemental data

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