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RESEARCH ARTICLE

Effect of mowing on N_2O and CH_4 fluxes emissions from the meadow-steppe grasslands of Inner Mongolia

Zedong LU, Rui DU (✉), Pengrui DU, Ziming LI, Zongmin LIANG, Yaling WANG, Saisai QIN, Lei ZHONG

College of Resources and Environment, University of Chinese Academy of Sciences, Beijing 100049, China

Abstract To assess the impacts of mowing on N_2O and CH_4 fluxes emissions from the meadow-steppe grasslands of Inner Mongolia, China, two regimes were investigated: unmown since 2005 (UM), and mown once every three years since 2009 (M3). On-site measurements were conducted continuously during a year-round period (August 2011 to August 2012). During the observation period, three diurnal cycles were also measured. In addition, a targeted laboratory experiment was conducted to make up for the few measurements in winter. A large pulse of N_2O emissions related to freeze-thaw cycles was observed at M3 during the spring thaw. Results showed that the meadow-steppes played a role as a sink for CH_4 and a source for N_2O. Significantly lower mean CH_4 uptake at UM (40.3 $\mu g\ C\cdot m^{-2}\cdot h^{-1}$) as compared to M3 (70.5 $\mu g\ C\ \cdot m^{-2}\cdot h^{-1}$) ($p < 0.01$), and significantly higher mean N_2O efflux at UM (6.3 $\mu g\ N\cdot m^{-2}\cdot h^{-1}$) as compared to M3 (4.3 $\mu g\ N\cdot m^{-2}\cdot h^{-1}$) ($p < 0.05$) were found. The laboratory experiment results revealed that mowing changed the soil conditions that favor the activity of denitrifiers during thawing periods. The CH_4 and N_2O fluxes were significantly correlated with soil temperature ($p < 0.05$). Mowing affected CH_4 uptake and N_2O emission mainly through its effect on vegetation types and some soil properties, such as soil inorganic N content, soil temperature, and soil moisture content, while soil inorganic N and moisture were not leading factors. Our results also suggested that mowing could mitigate the potential global warming in terms of CH_4 uptake and N_2O emissions.

Keywords mowing, meadow-steppe, methane, nitrous oxide, freeze-thaw cycles, global warming

1 Introduction

Methane (CH_4) and nitrous oxide (N_2O) are the most important greenhouse gases in the atmosphere, the radiative forcing of which are, on a molar basis, about 20–30 and 220–290 times greater than that of CO_2, respectively (Rodhe,1990; Lloyd, 1995). The atmospheric concentrations of CH_4 and N_2O have increased rapidly due to anthropogenic activities, such as land use/cover changes, agriculture, and fossil fuel use (Ojima et al., 1993; Houghton et al., 1999; Dlugokencky et al., 2003; IPCC, 2007; Zhou et al., 2007a; Zhang et al., 2014a). Because of these human activities, the global atmospheric concentrations of CH_4 and N_2O have increased from a pre-industrial values of approximately 715 ppb and 270 ppb to current values of 1,774 ppb and 319 ppb, respectively (IPCC, 2007). Furthermore, N_2O emissions contribute to the catalytic depletion of the ozone layer via reactions with singled oxygen atoms (Crutzen, 1970; Ravishankara et al., 2009). Therefore, more attention should be paid to the increment of CH_4 and N_2O concentrations in the atmosphere.

Grassland covers roughly 25% of the earth's land surfaces, and is one of the most important global terrestrial ecosystems. The natural grassland of China covers an area of 3.9×10^8 ha, and the *Leymus chinensis* steppe, which is the dominant grassland type of the semi-arid grassland area in Inner Mongolia, covers an area of approximately 90,000 km^2 (Wang, 2004). In the semi-arid grasslands of Hulunber, Inner Mongolia, grazing and mowing are important management practices. Wolf et al. (2010) carried out a year-round study with high and low temporal resolution at ten steppe grassland sites in Inner Mongolia of China, and concluded that grazing decreased rather than increased N_2O emissions. Smith et al. (2000) reported that the conversion of native grasslands into pastures decreased the sink strength of soils for atmospheric CH_4. These land use and management changes will directly affect CH_4

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E-mail: ruidu@ucas.ac.cn

uptake and N_2O emission. Therefore, variation, even if minor, in the exchange of these trace gases (N_2O and CH_4) between grassland ecosystems and the atmosphere can be of significance for the global atmospheric budgets.

Mowing, an important human practice in the Eurasian steppe management, is increasingly being used as a method to collect forage to feed livestock, replacing traditional practices (Foster et al., 2009). Huston (1979) and Collins et al. (1998) found that mowed plots had more than double the species richness of plots that were not mown, and mowing in anthropogenically stressed grasslands enhanced biodiversity. Meanwhile, mowing also has various other effects on grassland ecosystems, e.g., changes in soil temperature and moisture, soil microbial growth, and biomass (Parr and Way, 1988; Wan et al., 2002; Bahn et al., 2006; Robson et al., 2007; Zhou et al., 2007b; Ilmarinen and Mikola, 2009; Luo et al., 2009; Gavrichkova et al., 2010). Soil–atmosphere exchange of CH_4 and N_2O relies on a suite of complex processes: production, consumption, and diffusion processes in the sequential biochemical reactions. In general, the magnitude of their uptake or emission depends on soil physical properties, environmental conditions, and soil biologic characteristics such as soil temperature and moisture content, soil mineral N content, microbial activity and population size, and vegetation type (Willison et al., 1997; Smith et al., 2000; Groffman et al., 2006; Maljanen et al., 2007; Chen et al., 2010). Some of the changes induced by mowing have been shown to influence CH_4 and N_2O fluxes in grassland (Wang et al., 2005; Du et al., 2006; Blagodatsky and Smith, 2012; Cai, 2012). Thus, in the present study we hypothesized that mowing would affect CH_4 and N_2O fluxes by: (i) affecting the soil inorganic N content and species composition, (ii) changing the soil moisture content, and (iii) the soil temperature.

Only a few studies have been conducted on the effect of mowing on greenhouse gas fluxes from the meadow-steppe grasslands of Inner Mongolia, and the effects on CH_4 and N_2O fluxes in particular are not well understood (Guo et al., 2010, 2011a, b; Zhang et al., 2014b; Zhong et al., 2014). In these previous studies, however, the measurements of grassland greenhouse gas fluxes were often performed over short time periods, and mostly during the growing season (from May to September). Thus, the estimations of emissions feature high levels of uncertainty due to the absence of measurements during the non-growing seasons in which the seasonal and repeated freeze-thaw events occur. Freeze-thaw cycles affect C and N dynamics in soils, and thus also have a considerable impact on the emissions of GHGs (Zhu et al., 2009). Mosier et al. (1996) reported that winter fluxes contributed 20%–40% of the annual N_2O emissions and 15%–30% of the CH_4 consumption in North American prairie systems. Röver et al. (1998) found that N_2O-emission-related freeze-thaw cycles could contribute up to 70% of the annual N_2O emissions. In our field measurements, CH_4 uptake peaks at both sites were not found, and a large pulse of N_2O emissions was observed at mown sites, but not at unmown sites, during the spring thaw. To better understand this phenomenon of the freeze-thaw effect on the soil N_2O flux at mown meadow-steppe sites, a laboratory incubation experiment under controlled conditions was carried out.

The hope in conducting these field and laboratory measurements of CH_4 and N_2O fluxes in response to mowing on the meadow-steppe grassland of the Hulunber high plain in Inner Mongolia was that we can contribute to an improved understanding and more accurate evaluation of the global carbon budget and climate change. In this context, the aim of the study was threefold: (1) to study the effect of mowing on environmental factors; (2) to examine the effect of mowing on CH_4 and N_2O fluxes from the meadow-steppe grassland; and (3) to calculate the net effect on potential global warming in terms of net CH_4 and N_2O emissions.

2 Materials and methods

2.1 Site descriptions and field treatments

The experimental site was located in a meadow-steppe grassland adjacent to the Hulunber Grassland Ecosystem Research Station of the Chinese Academy of Agricultural Sciences in Inner Mongolia, China (49°19′N, 120°03′E; 628 m a.s.l.). The climate in this region is semi-arid, with a frost-free growing season of 95–110 days from early May to early September. Annual mean precipitation is 400 mm (150–550 mm), with a large inter-annual variation, falling primarily between June and August, which coincides with the growing season. Annual mean air temperature is −2°C to 1°C, with a minimum monthly mean of −26°C in January, and a maximum of 21°C in July. The region has a long winter period of nearly six months, from October to April of the following year, a short autumn (September; Julian days 241–270), spring (May, June; Julian days 120–180), and summer (July, August; Julian days 181–240) (Du et al., 2006). The soil in this region is frequently exposed to the episodes of intensive freeze-thaw cycles, starting to freeze in late October, and thawing in April. The unmown steppes (UM) have been fenced since 2005, while for the mown steppes a machine has been used since 2009 to mow the grasses in late August once every three years (M3). Grasses are mowed at a height of 5 cm, and a control of unmown grass (height of approximately 25 cm) is maintained. The study site's soil is chernozem with an average thickness of 30–40 cm. The major soil characteristics are provided in Table 1. The meadow-steppe grassland is dominated by *Leymus Chinensis*.

2.2 In situ measurements of CH_4 and N_2O fluxes

The CH_4 and N_2O flux measurements were carried out from August 2011 to August 2012. Four replicate plots were simultaneously observed for each field treatment. Gas samples were collected using the static chamber method. The static chamber was made of stainless steel and equipped with a fan, 10 cm in diameter, installed on the top wall of each chamber to ensure complete gas mixing. The chamber consisted of two parts: a square box (without a top and bottom; dimensions: 0.5 m × 0.5 m × 0.1 m (length × width × height)) and a box with a removable cover (without a bottom; dimensions: 0.5 m × 0.5 m × 0.5 m (length × width × height)). The square box was inserted directly into the soil at about 10 cm below the soil surface, and the cover was placed on top during sampling times and removed afterwards. A white adiabatic cover was added outside of the stainless steel cover to reduce the impact of direct radiative heating during sampling. The chamber was closed for almost an hour and five gas samples were collected every 10 min using plastic syringes fitted with three-way stop-cocks. The collected gas samples were analyzed with a gas chromatograph (Agilent 7890 GC USA) within 24 hours. The gas chromatograph was equipped with a flame ionization detector (FID) for CH_4 analysis, and an electron capture detector (ECD) for N_2O analysis. The gas chromatography configurations for analyzing concentrations of CH_4 and N_2O, and methods for calculating the fluxes of CH_4 uptake and N_2O emission, were the same as those reported by Wang and Wang (2003). The flux measurements were taken once or twice per week during the growing season, and once or twice per month during the non-growing season. Intensive flux measurements at two-hour intervals to investigate the diurnal variations of the meadow-steppe were taken on 10 August 2011, 18 June 2012, and 10 August 2012, respectively. Annual GHG emission or uptake was estimated by calculating cumulative fluxes over an experimental period. Based on the equation of global warming potential (GWP), an estimate of the degree of contribution to global warming, the CO_2 equivalent [$CO_{2(e)}$] of the net emission of GHGs (only CH_4 and N_2O were taken into account) was determined using a transformation coefficient of 298 for the N_2O emission flux and 26 for the CH_4 uptake flux (Li et al., 2004; IPCC, 2007). The $CO_{2(e)}$ was calculated using the following equation:

$$\mathrm{CO}_{2(\mathrm{e})} = \mathrm{N_2O} \div 28 \times 12 \times 298 + \mathrm{CH_4} \div 44 \times 16 \times 26,$$

where $CO_{2(e)}$ ($\mathrm{kg\ C \cdot ha^{-1} \cdot yr^{-1}}$) is the CO_2 equivalent of net emissions of GHGs, and N_2O and CH_4 are the N_2O flux ($\mathrm{kg\ N \cdot ha^{-1} \cdot yr^{-1}}$) and the CH_4 flux ($\mathrm{kg\ C \cdot ha^{-1} \cdot yr^{-1}}$), respectively.

2.3 Laboratory experiment

The incubation experiment was performed in a 360 mL flask. Tap water (directly extracted local groundwater) was used to just flood the top of the soil to simulate field conditions. The flask still had sufficient headspace left at the top of the intact soil for gas sampling. Each flask was then sealed with an airtight lid and placed in a freezer to simulate freeze-thaw cycles. To begin with, the soil was frozen for 1 week at −15°C, and then we raised the temperature up to 5°C for 1 week. In subsequent cycles, the temperature was changed from −10°C to 5°C, from −5°C to 10°C, and from −5°C to 5°C. After a temperature change, the soil cores were allowed to equilibrate to the new temperature at least 12 hours before the start of flux measurements.

Gas samples from the headspace of the flasks were taken with 60 mL plastic syringes fitted with three-way stop-cocks at 12 hours, 24 hours, 3 days, and 7 days. The gas samples were immediately analyzed using gas chromatography (Agilent 7890 GC USA). The gas chromatography was equipped with an electron capture detector for N_2O analysis. N_2O fluxes were calculated using equation (Du et al., 2006)

$$f = \frac{\Delta m}{M \Delta t} = \frac{\rho v \Delta c}{M \Delta t},$$

where ρ (ppm) is the density of the gas, $\Delta c / \Delta t$ is the variation ration of the gas concentration, M (g) is the mass of the soil, and v (mL) is the volume of the headspace in the flask.

2.4 Measurement of environmental factors

During each gas sampling process, the internal temperature of the chamber and the soil surface temperature were simultaneously recorded, using a portable digital thermometer (JM624, Tianjin, China). Soil temperature and moisture at depths of 10 and 20 cm were continuously monitored by an automated measuring system using a Hobo Micro Station Data Logger (H21-002, USA) every 10 seconds, and were stored every 1 minute during the growing season, and every 1 minute and every 5 minutes during the non-growing season. Climatic data (daily precipitation, air temperature) were obtained from a local meteorological station.

2.5 Soil sampling and vegetation description

Soil cores (0–20 cm layer) were collected nine times using a steel soil corer (7 cm in diameter): in spring (15 May, 31 May, 15 June, 30 June), summer (13 July, 1 August, 16 August, 27 August), and in autumn (20 September). Five soil cores were randomly taken for each site at each time. Intact soil was separated into five depth intervals (0–3 cm, 3–6 cm, 6–9 cm, 9–12 cm, 12–15 cm) for laboratory incubation in a 360 mL flask. For soil analysis, each soil sample was separated into three depth intervals (0–5 cm, 5–10 cm, 10–20 cm), and then five soil samples were mixed evenly by interval for determination of soil organic matter (SOM), soil NH_4^+-N and NO_3^--N concentrations. Soil NH_4^+-N and NO_3^--N concentrations were determined photometrically after extracting the soil with 1 $mol \cdot L^{-1}$ KCl solution (1:2 w:w) for 1 h. Organic matter content was determined by the soil bath-$K_2Cr_2O_7$ titration method.

The grassland vegetation was observed and measured once a month (from May to September) using six quadrats of 1 m^2 in each site. Plant standing biomass was harvested, oven-dried at 65°C for 48 hours, and weighted for each species (Zhong et al., 2014).

2.6 Statistical analysis

The statistical software packages SPSS 17.0 (SPSS Inc., Chicago, USA) and Origin 8.0 (Origin Laboratory Corporation, USA) were used. Differences in CH_4 and N_2O fluxes, aboveground biomass, average soil temperature, soil moisture, SOM, soil NH_4^+-N and NO_3^--N among treatments were determined using ANOVA. Pearson correlation was used to determine correlations between CH_4 and N_2O fluxes and the environmental factors. All statistical tests were performed at a significance level of level of $\alpha = 0.05$.

3 Results

3.1 Environmental conditions and soil conditions

The daily means of air temperature, precipitation, and soil temperature and moisture at 10 and 20 cm depth at the two sites are given in Fig. 1. The average air temperature was −1.7°C, ranging from −37.8°C (winter) to 25.5°C (summer). Annual precipitation (247.6 mm) was lower than the long-term average (400 mm), and seasonally precipitation in spring and summer was higher. The relatively high and low daily precipitation generated comparatively wet and dry conditions, which altered soil moisture and soil temperature. In winter, heavy snow generated soils with saturated conditions, and the freeze period was from roughly the first 5 days of October (Julian days 305) to the middle 15 days of April the following year (Julian days 105) (Fig. 1(a)). The surface snow and ice then began to thaw with great fluctuation, and totally thawed in

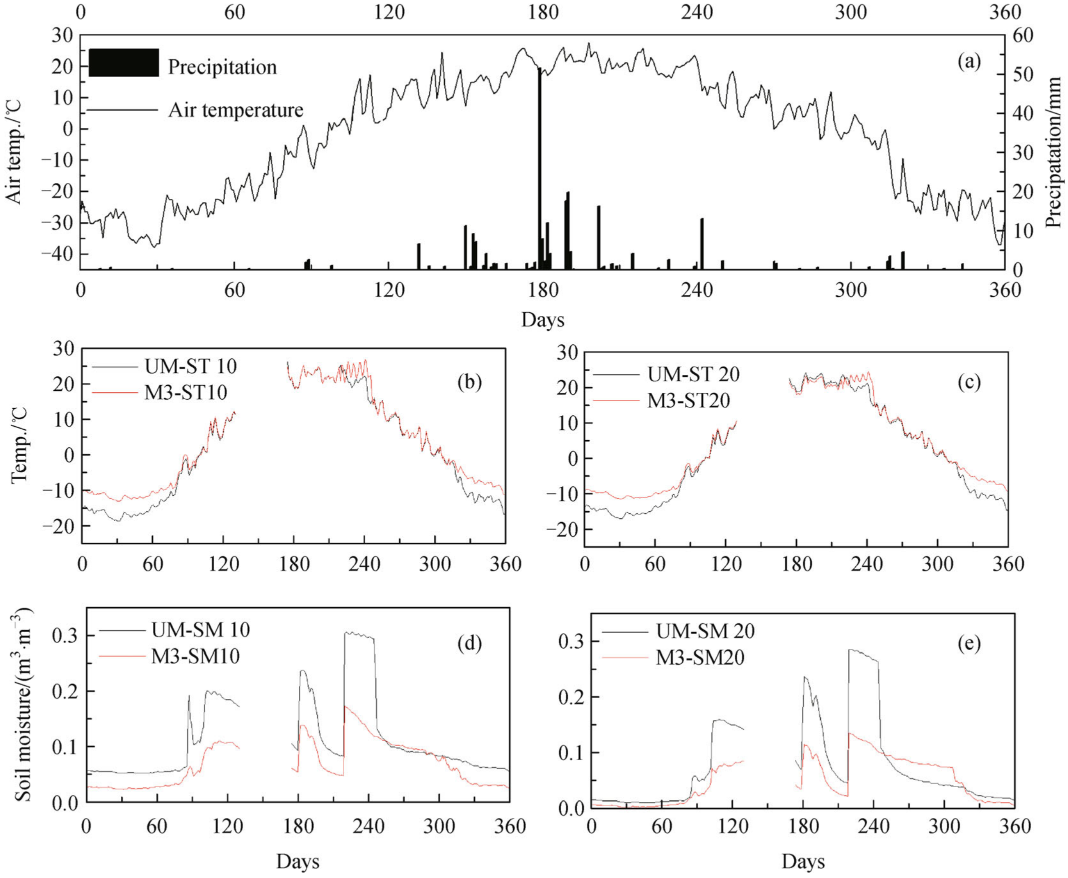

Fig. 1 Daily mean (a) air temperature and precipitation, soil temperature at (b) 10 cm and (c) 20 cm depth, and soil moisture at (d) 10 cm and (e) 20 cm soil depth.

the first 5 days of May (Julian days 120) as temperature increased.

Mowing affected soil temperature at 10 cm depth in summer and winter, soil temperature at 20 cm depth in winter significantly ($p < 0.01$), but no significant effects were found on soil temperature in other seasons. However, there was a significant effect on soil moisture in all seasons except at 20 cm in autumn (Table 2).

3.2 Soil analysis and vegetation conditions

Both NH_4^+-N, NO_3^--N and SOM showed the same distribution, decreasing with increasing depth, i.e., the NH_4^+-N of soil at depths of 0–5, 5–10 and 10–20 cm was 5.07, 4.46, and 4.44 $mg \cdot kg^{-1}$, respectively (Table 1). There was no significant difference between M3 and UM in terms of NH_4^+-N, NO_3^--N, and SOM, indicating that inorganic nitrogen (NH_4^+-N and NO_3^--N) and SOM restored gradually after mowing, but their concentrations were higher in UM than M3 fields.

Aboveground biomass showed strong seasonal fluctuations, with peak values in July (Fig. 2). Mowing had no effect on aboveground biomass ($p > 0.05$); however, a marginally significant effect of mowing on species composition was found ($p < 0.01$) (data not shown). The mown meadow-grassland was found to be dominated by *Leymus Chinensis*, whereas the dominant species of unmown grassland-meadow was *Carex tristachya*, indicating moderate mowing can prevent grassland degradation in meadow-steppe grassland.

3.3 Diurnal variation in soil CH_4 and N_2O fluxes

The diurnal cycle in three different growth stages of grass was analyzed and was found to vary greatly with no stable pattern (Fig. 3). During the panicle initiation stage, diurnal variation of CH_4 uptake was slight. The daily average CH_4 uptake flux was 63.1 $\mu g\ C \cdot m^{-2} \cdot h^{-1}$ in UM, which was rather lower than in the M3 steppe, and mean uptake flux was 75.1 $\mu g\ C \cdot m^{-2} \cdot h^{-1}$, reaching a maximum at 16:00 and

Table 1 Main soil characteristics of the different experimental sites

	Soil depth	M3						UM					
		May	Jun	Jul	Aug	Sep	Mean	May	Jun	Jul	Aug	Sep	Mean
NH_4^+-N /($mg \cdot kg^{-1}$)	0–5 cm	5.24	5.27	5.76	4.08	5.00	5.07	8.65	3.44	5.14	9.14	5.04	6.28
	5–10 cm	5.55	4.39	4.27	3.39	4.70	4.46	6.24	4.99	5.22	8.63	4.75	5.97
	10–20 cm	4.20	4.76	4.47	3.87	4.89	4.44	6.28	5.63	5.20	4.10	4.84	5.21
NO_3^--N /($mg \cdot kg^{-1}$)	0–5 cm	21.38	20.79	4.20	9.06	23.44	15.77	18.63	23.80	6.99	19.30	15.99	16.94
	5–10 cm	7.03	10.57	4.18	5.63	18.51	9.18	7.02	19.31	4.88	15.97	8.89	11.22
	10–20 cm	4.80	8.72	5.87	3.67	3.19	5.25	3.68	7.71	4.42	5.60	4.10	5.10
SOM /($g \cdot kg^{-1}$)	0–5 cm	80.80	67.71	71.55	62.21	62.94	69.04	73.96	80.67	57.37	68.43	63.66	68.82
	5–10 cm	56.16	55.59	50.41	56.35	44.47	52.60	54.03	66.66	60.96	59.33	48.51	57.90
	10–20 cm	58.56	39.41	43.66	37.54	37.64	43.36	51.72	48.94	55.78	41.35	41.68	47.89

No significant differences were found between M3 and UM.

Table 2 Seasonal means of soil temperature and soil moisture at the M3 and UM steppe sites

Sites	Period	Soil temperature/°C		Soil moisture/($m^3 \cdot m^{-3}$)	
		10 cm	20 cm	10 cm	20 cm
M3	Spring	14.51	13.02	0.09**	0.07**
	Summer	22.87**	21.54	0.11**	0.08**
	Autumn	13.89	13.96	0.11*	0.09
	Winter	−4.90**	−4.11**	0.05**	0.03*
	Mean	3.14	3.33	0.07**	0.05**
UM	Spring	14.44	13.18	0.15**	0.12**
	Summer	22.02**	21.54	0.20**	0.18**
	Autumn	12.72	13.00	0.14*	0.11
	Winter	−7.61**	−6.77**	0.08**	0.04*
	Mean	1.05	1.48	0.11**	0.08**

* Significant difference at the 0.05 level between UM and M3; ** Significant difference at the 0.01 level between UM and M3. Only significant differences are shown.

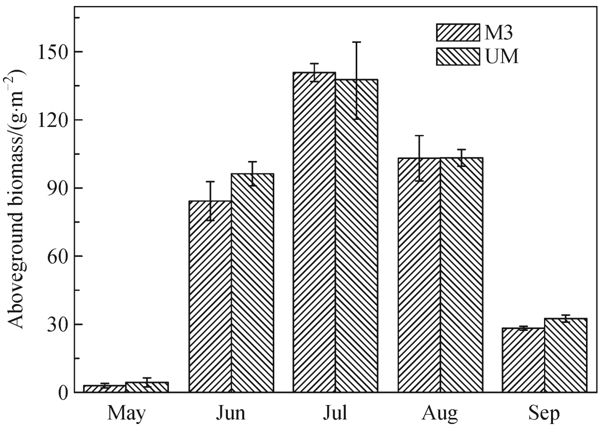

Fig. 2 Aboveground biomass in the M3 and UM steppes.

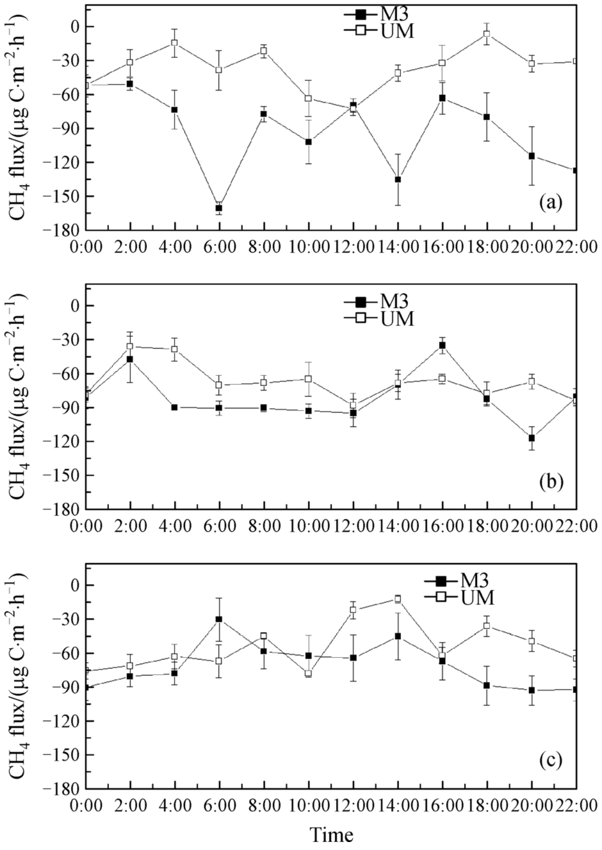

Fig. 3 Diurnal distribution of CH_4 uptake fluxes from the M3 and UM steppes in different growth stages: (a) fruiting stage (7–8 August 2011); (b) panicle initiation stage (18–19 June 2012); (c) fruiting stage (10–11 August 2012).

falling to a minimum at 20:00. Figures 3(a) and 3(c) suggest that there was a great variation in CH_4 uptake, even when observed in the same growing stage. Figure 3 also shows that the timing of the peak uptake varied in different stages, but on a diurnal flux basis, average CH_4 uptake over the time period of 08:00–14:00 would be representative for a diurnal period.

The average values of CH_4 uptakes during the whole day, daytime and nighttime for UM and M3, are shown in Table 3. The maximum CH_4 uptake was observed with a daily average of up to 63.1 μg $C \cdot m^{-2} \cdot h^{-1}$ for UM and up to 87.5 μg $C \cdot m^{-2} \cdot h^{-1}$ for M3. In general, the daily mean CH_4 uptake flux from the M3 meadow-steppe tended to be roughly 1.58-fold greater than that of the UM field. Meanwhile, mowing changed the proportion of CH_4 flux during daytime and nighttime, i.e., 64% and 36% for UM and 54% and 46% for M3 on 7 and 8 August, respectively. As shown in Table 4, mowing enhanced CH_4 uptake mainly through increasing its flux during nighttime, e.g., in the panicle initiation stage, the average CH_4 uptake flux at night was 54.1 μg $C \cdot m^{-2} \cdot h^{-1}$ for UM and 72.6 μg $C \cdot m^{-2} \cdot h^{-1}$ for M3.

Similar to the CH_4 uptake, no stable pattern of diurnal variation in N_2O emission by the UM and M3 steppes was revealed, with mean emissions of 3.7 and 0.5 μg $N \cdot m^{-2} \cdot h^{-1}$ respectively in the panicle initiation stage (Fig. 4, Table 3). Figures 4(a) and 4(c) suggest that there was a great inter-annual variation in N_2O uptake (3.5 μg $N \cdot m^{-2} \cdot h^{-1}$, and 7.6 μg $N \cdot m^{-2} \cdot h^{-1}$ for UM in 2011 and 2012, respectively). These variations all implied that any low-frequency diurnal observation may greatly under- or over-estimate the total N_2O emission, due to the enormous variability.

On a diurnal flux basis, the average N_2O emission over the time period of 08:00–10:00 would be representative of a diurnal period. The data from diurnal-cycle measurements conducted in various growth stages (Table 3) indicated that the daily average N_2O emission fluxes from the UM and M3 steppes varied greatly from 0.5 to 7.6 μg $N \cdot m^{-2} \cdot h^{-1}$, and the daily mean N_2O emission by the UM steppe tended to be roughly 2.7-fold greater than that of the M3 field (means of 4.9 μg $N \cdot m^{-2} \cdot h^{-1}$ and 1.9 μg $N \cdot m^{-2} \cdot h^{-1}$, respectively). In contrast to the influence of mowing on CH_4, mowing decreased N_2O emissions by changing the proportion of its flux during daytime and nighttime, i.e., 55% and 45% for UM, and 60% and 40% for M3 on 18 and 19 June, respectively.

3.4 Seasonal variation in soil CH_4 and N_2O fluxes

Figure 5 shows that the year-round measured CH_4 uptake fluxes were usually negative. This suggests that the meadow-steppe of the Inner Mongolia played a role as a sink for the atmospheric CH_4. At the same time, the CH_4 uptake of either site showed a large seasonal variability (Fig. 6). The maximum CH_4 uptake was observed in summer or autumn, with means reaching −102.2 μg $C \cdot m^{-2} \cdot h^{-1}$ for UM and −220.7 μg $C \cdot m^{-2} \cdot h^{-1}$ for M3. During winter, CH_4 uptake at the two sites was still observed with a range of −2.23 – 42.8 μg $C \cdot m^{-2} \cdot h^{-1}$ (Fig. 5). As shown in Table 5, the CH_4 uptake at the M3 steppe was 1.75 times higher than that at the UM site, with the

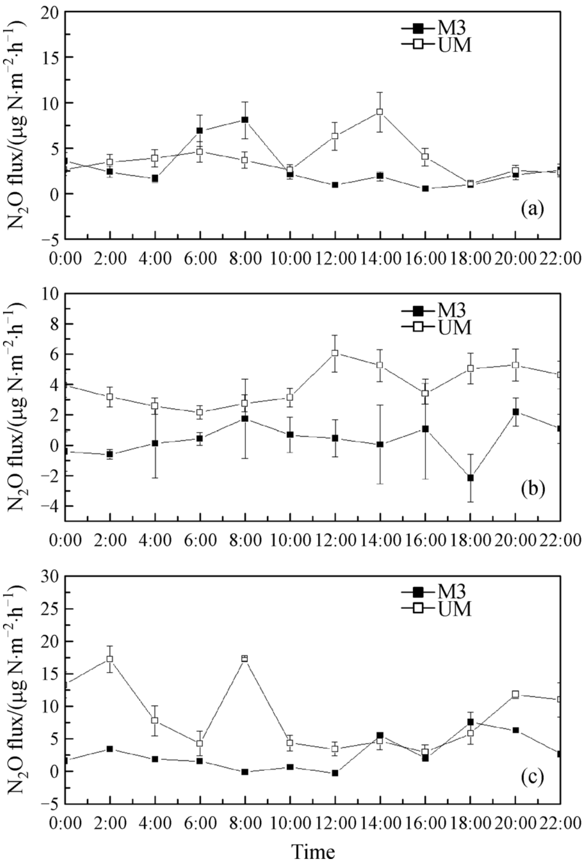

Fig. 4 Diurnal distribution of N_2O emission fluxes from the M3 and UM steppes in different growth stages: (a) fruiting stage (7–8 August 2011); (b) panicle initiation stage (18–19 June 2012); (c) fruiting stage (10–11 August 2012).

annual average reaching 70.5 and 40.3 μg C·m^{-2}·h^{-1}, respectively. The CH_4 uptake fluxes at M3 averaged −52.8 μg C·m^{-2}·h^{-1}, −116.2 μg C·m^{-2}·h^{-1}, −118.9 μg C·m^{-2}·h^{-1}, and −55.6 μg C·m^{-2}·h^{-1} during the seasons of spring, summer, autumn and winter, respectively. At UM, the average flux during each season was −33.0 μg C·m^{-2}·h^{-1}, −54.9 μg C·m^{-2}·h^{-1}, −44.6 μg C·m^{-2}·h^{-1}, and −37.6 μg C·m^{-2}·h^{-1}, respectively. Accumulatively, the former was higher than the latter by 4% in spring, 14% in summer, 9% in autumn, and 15% in winter. This indicated that mowing drastically enhanced CH_4 uptake in summer, autumn and winter.

As Fig. 5 shows, the year-round measured N_2O emission fluxes from the M3 and UM meadow-steppes were usually positive, with some sporadic N_2O uptake events, especially during soil freeze-thaw periods. This suggests that the meadow-steppe of the Inner Mongolia played a role as a source of the atmospheric N_2O, with an annual emission flux of 4.2 and 6.2 μg N·m^{-2}·h^{-1} for the M3 and UM steppe, respectively (Table 4). There was also an obvious difference in N_2O emission intensity between different seasons (Fig. 6). The N_2O emission at M3 averaged 2.2 μg N·m^{-2}·h^{-1}, 3.9 μg N·m^{-2}·h^{-1}, 11.0 μg N·m^{-2}·h^{-1}, and 4.0 μg N·m^{-2}·h^{-1} during the seasons of spring, summer, autumn, and winter, respectively. On average, the N_2O emission at M3 was lower than that at UM by 2.4 μg N·m^{-2}·h^{-1} in spring, 2.6 μg N·m^{-2}·h^{-1} in summer, 6.2 μg N·m^{-2}·h^{-1} in autumn, and 1.1 μg N·m^{-2}·h^{-1} in winter. Accumulatively, the latter were 2.1, 1.7, 1.6, and 1.3 times higher than the former in spring, summer, autumn, and winter, respectively. This indicates that mowing decreased N_2O emissions by mainly having an effect in spring, summer, and autumn.

3.5 Laboratory experiment

Measurements of N_2O in the gas profile showed that the soil N_2O concentration never reached zero and was generally small in frozen soils, and the highest soil N_2O concentration was generally observed during soil thawing periods (Fig. 7). In UM, N_2O emissions increased (from 0.001 μg·g^{-1}·h^{-1} to 0.03 μg·g^{-1}·h^{-1}) at the start of soil thawing only, and declined strongly and remained constant during the subsequent continuation of thawing. In M3, the N_2O emission was characterized by a fast increase from 0.002 μg·g^{-1}·h^{-1} to 0.05 μg·g^{-1}·h^{-1} followed by a rapid decline to 0.02 μg·g^{-1}·h^{-1} 2 days later from −15°C to 5°C, and an increase from 0.004 μg·g^{-1}·h^{-1} to 0.04 μg·g^{-1}·h^{-1} followed by a rapid decline to 0.02 μg·g^{-1}·h^{-1} 3 days later from −10°C to 5°C during the thawing period, before finally remaining constant. The spring-thaw N_2O flux in M3 was larger than in UM, which was in agreement with our field findings. Measurements of N_2O in the gas profile

Table 3 Average fluxes during the whole day, daytime, and nighttime at the two sites

Sites	Date	CH_4 average flux /(μg C·m^{-2}·h^{-1})			Cumulative ratio of CH_4		N_2O average flux /(μg N·m^{-2}·h^{-1})			Cumulative ratio of N_2O	
		Diurnal	Daytime	Nighttime	$R_{day./diu.}$	$R_{nig./diu.}$	Diurnal	Daytime	Nighttime	$R_{day./diu.}$	$R_{nig./diu.}$
M3	Aug. 7–8	−87.6	−94.3	−80.9	54	46	2.6	3.0	2.2	57	43
	Jun. 18–19	−75.2	−77.7	−72.6	52	48	0.5	0.6	0.4	60	40
	Aug. 10–11	−63.4	−59.1	−67.7	47	53	2.5	2.0	3.0	41	59
UM	Aug. 7–8	−32.7	−41.9	−23.5	64	36	3.5	4.6	2.3	66	34
	Jun. 18–19	−63.1	−72.1	−54.1	57	43	3.7	4.1	3.3	55	45
	Aug. 10–11	−47.3	−44.7	−50.0	47	53	7.6	6.5	8.7	43	57

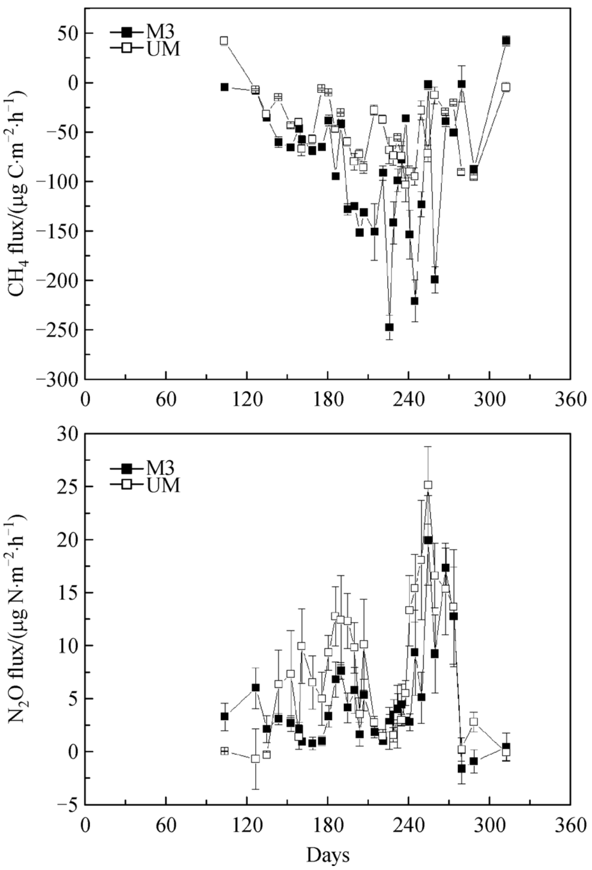

Fig. 5 Daily fluxes of CH_4 uptake and N_2O emission in the M3 and UM fields.

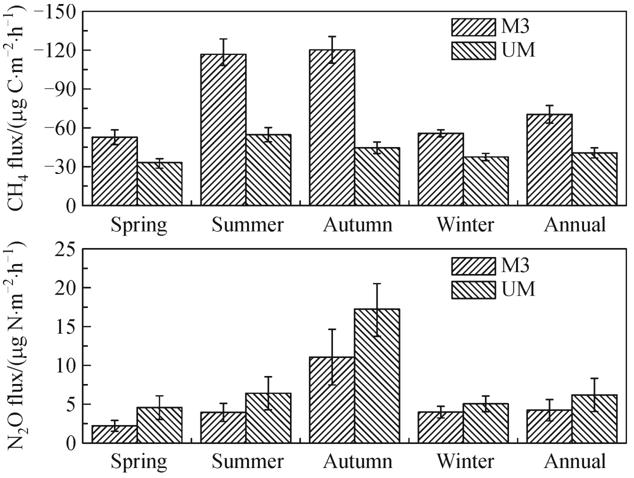

Fig. 6 Seasonal distributions of CH_4 uptake and N_2O emission in the M3 and UM field.

also revealed that the observed peak N_2O emission during spring thaw occurred in the topmost layer (0–6 cm).

The soil NO_3^- concentration decreased during the experiment (Table 4). The nitrate concentration after the freeze-thaw cycle experiment was 6.90 $mg \cdot kg^{-1}$ in UM and 9.14 $mg \cdot kg^{-1}$ in M3 (8.32 $mg \cdot kg^{-1}$ and 11.15 $mg \cdot kg^{-1}$ at the beginning, respectively). The soil NH_4^+ concentration increased from 4.45 $mg \cdot kg^{-1}$ to 12.12 $mg \cdot kg^{-1}$ in UM, and from 4.34 $mg \cdot kg^{-1}$ to 12.86 $mg \cdot kg^{-1}$ in M3 during the whole experiment. The total N concentration and SOM showed little change.

3.6 Net effect on potential global warming in terms of net N_2O and CH_4 emissions

The annual net effect of CH_4 uptake and N_2O emission was determined by the sum of $CO_{2(e)}$, which was calculated by multiplying the global warming potential with the annual total net emissions of N_2O and CH_4 (i.e., 298 and 26 for the unit mass of N_2O and CH_4, respectively, at the 100-yr time scale (IPCC, 2007)). The calculated results listed in Table 5 showed that the $CO_{2(e)}$ of the annual net emissions of CH_4 and N_2O from the UM and the M3 steppe sites was 36.1 and – 10.3 $kg\ C \cdot ha^{-1} \cdot yr^{-1}$, respectively. This indicates that the mowing of the meadow-steppe mitigated the potential global warming; however, the role of mowing on the meadow-steppe in the terms of the atmospheric greenhouse effect has rarely been considered.

4 Discussion

4.1 N_2O emissions from frozen soil and during thawing

In winter months, the grasslands in this study are usually subject to heavy snowfall and experience several days or weeks of subzero temperature. The grasslands are exposed to freeze-thaw cycles for a period of time. Müller et al. (2002) reported that N_2O emissions occurred even when the soil was frozen. Our results also demonstrated that N_2O emissions could be measured for several days at both sites during the period of soil freezing, indicating that soil microorganisms were still active in frozen soils. The N_2O emission peaks were observed after soil thawing, which lasted only for a few days, and the emission of N_2O dropped off very quickly at the onset of the freezing cycle in M3 (Figs. 5 and 7). Meanwhile, the emissions decreased in the subsequent thawing periods (Fig. 7). There was no matching peak of N_2O emissions at the UM site in the field measurements. Contrary to expectations, the accumulation of N_2O in winter accounted for 70% of the annual accumulation for the UM site and 55% of the annual accumulation for the M3 site (Table 5). The results do not contradict our laboratory findings, as in our field measurements N_2O emissions were lower at M3 than UM in frozen periods, and only higher in a short-lived thawing periods. Therefore, mowing had changed the soil and environmental conditions that determine emissions of N_2O during thawing periods. The NH_4^+ concentration increased by approximately 7.67 $mg \cdot kg^{-1}$ at UM and by 8.52 $mg \cdot kg^{-1}$

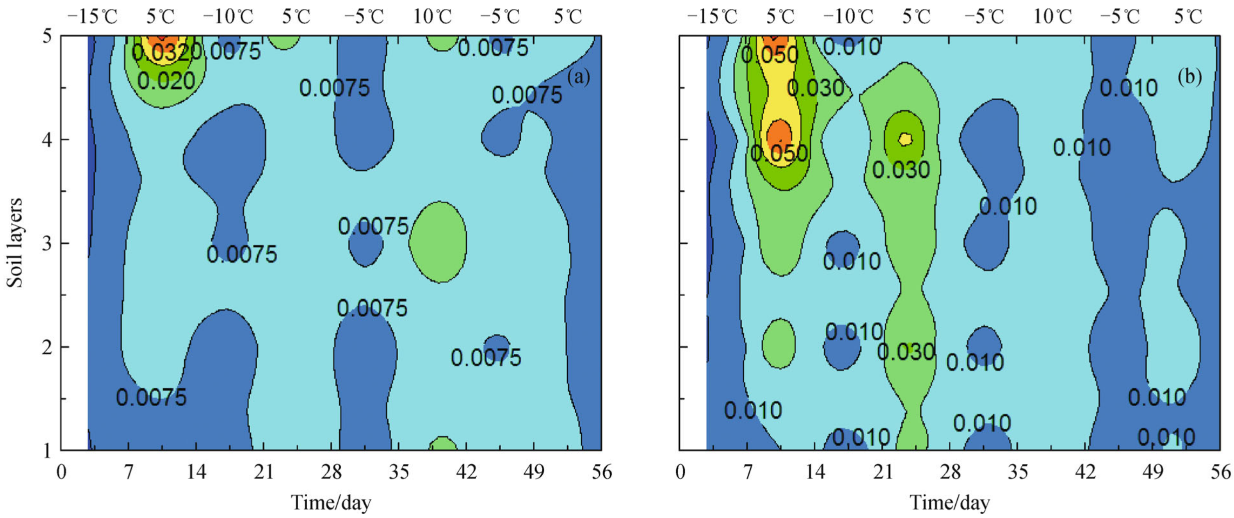

Fig. 7 N_2O fluxes ($\mu g \cdot g^{-1} \cdot h^{-1}$) along the soil profile (0–15 cm) of the undisturbed soil cores from (a) UM and (b) M3 under different soil temperatures during the entire incubation period; soil layer 1, 12–15 cm; soil layer 2, 9–12 cm; soil layer 3, 6–9 cm; soil layer 4, 3–6 cm; soil layer 5, 0–3 cm.

Table 4 Main soil characteristics in the laboratory experiment.

Sites	Total nitrogen /($g \cdot kg^{-1}$)		NO_3^--N /($mg \cdot kg^{-1}$)		NH_4^+-N /($mg \cdot kg^{-1}$)		SOM /($g \cdot kg^{-1}$)	
	Start	End	Start	End	Start	End	Start	End
UM	2.71	2.69	8.32	6.90	4.45	12.12	55.13	53.40
M3	1.86	2.20	11.15	9.14	4.34	12.86	44.87	39.75

at M3. Meanwhile, the NO_3^- concentration decreased by approximately 1.42 $mg \cdot kg^{-1}$ at UM and 2.01 $mg \cdot kg^{-1}$ at M3, respectively (Table 4). N_2O emissions peaks occurred as a consequence of favorable denitrification conditions due to the death and lysis of microbial cells and/or the destruction of soil aggregates in saturated soil as a result of snowmelt (Groffman and Tiedje, 1989; Christensen and Christensen, 1991; Herrmann and Witter, 2002; Müller et al., 2002).

4.2 Effects of mowing on soil CH_4 and N_2O fluxes

Mowing reduced soil moisture, which acted as a diffusion barrier for methanotrophic microorganisms and determined the proportion of anaerobic/aerobic conditions in the soil. As Table 2 shows, there was higher soil moisture at UM (mean of 9.5%, v/v) than at M3 (mean of 6%, v/v), especially in spring and summer ($p < 0.01$). Although positive correlations between CH_4 uptake and soil moisture have also been reported in several other studies (Smith et al., 2000; Jones et al., 2005; Livesley et al., 2009), our results showed no significant correlation between soil moisture and CH_4 uptake during the year-round measurements ($p > 0.05$). This suggested mowing enhanced CH_4 uptake (70.5 versus 40.3 $\mu g\ C \cdot m^{-2} \cdot h^{-1}$) by affecting other more important factors, not primarily

Table 5 Fluxes of N_2O emissions and CH_4 uptake by the meadow-steppe grasslands

Gases		Annual		Spring		Summer		Autumn		Winter	
		Flux[a)]	Total[b)]	Accumulation[b)]	Ratio[c)]	Accumulation[b)]	Ratio	Accumulation[b)]	Ratio	Accumulation[b)]	Ratio
N_2O	UM	6.3*	0.5	0.07	18	0.09	25	0.12	33	0.26	70
	M3	4.3*	0.4	0.03	9	0.06	15	0.08	21	0.20	55
CH_4	UM	−40.3**	−3.5	−0.48	8	−0.79	13	−0.32	5	−1.9	31
	M3	−70.5**	−6.1	−0.76	12	−1.67	27	−0.86	14	−2.80	46
CO_2	UM	–	36.1								
	M3	–	−10.3								

* Significant difference at the 0.05 level between UM and M3; ** Significant difference at the 0.01 level between UM and M3. a) Units: $\mu g\ N \cdot m^{-2} \cdot h^{-1}$, $\mu g\ C \cdot m^{-2} \cdot h^{-1}$; b) Units: $kg\ N \cdot ha^{-1} \cdot yr^{-1}$, $kg\ C \cdot ha^{-1} \cdot yr^{-1}$; c) Units: contribution to the annual total (%).

through the effect of soil moisture. However, because mowing reduced soil vegetation coverage and litter accumulation, soil moisture and nutrient transported to the soil decreased. This would influence root growth, root respiration, soil microbial activity, and community composition. Secondly, mowing increased soil temperature [nearly 3°C higher than that at M3 in winter ($p < 0.01$)]. Simultaneously, the CH_4 uptake decreased dramatically since CH_4 oxidation was likely to be limited due to low microbial activity with reduced soil temperature (Liu et al., 2007; Zhang et al., 2012). In our study, we further found significant positive correlation between soil temperature and CH_4 uptake ($p < 0.01$), which was consistent with that reported by Groffman et al. (2006). The R^2 values of the fitted regression curve indicate that the variation in soil temperature may explain 32%–34% ($p < 0.01$) of the variation in CH_4 uptake (Figs. 8(a) and 8(b)). Other previous studies have also reported that soil temperature in the topmost layer might be the primary driving factor for CH_4 uptake (Wang et al., 2005). Thirdly, we found soil NH_4^+-N and NO_3^--N content at UM (means of 5.82 and 11.09 $mg \cdot kg^{-1}$, respectively) were greater than those at M3 (means of 4.66 and 10.07 $mg \cdot kg^{-1}$, respectively) (Table 3). The increase in CH_4 consumption may be attributable to the decrease in soil mineral N contents, since both NH_4^+-N (Steinkamp et al., 2001) and NO_3^--N (Kähkönen et al., 2002) could inhibit CH_4 oxidation. Whiting and Chanton (1993) also reported that reduction in inorganic N by mowing resulted in an increase of CH_4 oxidation. However, no significant difference ($p > 0.05$) was found with respect to the soil NH_4^+-N and NO_3^--N content because of restoration in three years, suggesting soil inorganic nitrogen content was not the leading factor in terms of the differences in CH_4 flux. There are some other factors that may affect the CH_4 uptake, such as the variation in root/shoot ratios (Nitschke et al., 2010) and root exudation, favoring the microbial activity (Lipson and Schmidt, 2004). The oxidation of atmospheric CH_4 by soils is mainly controlled by the activity of methanotrophic bacteria. Singh and Tate (2007) observed atmospheric CH_4 consumption in soil cores in New Zealand, for which type II methanotrophs may be chiefly responsible. We presumed that mowing increased CH_4 uptake perhaps due to changes in methanotrophic community structure and activity.

In this study, there was no correlation found between

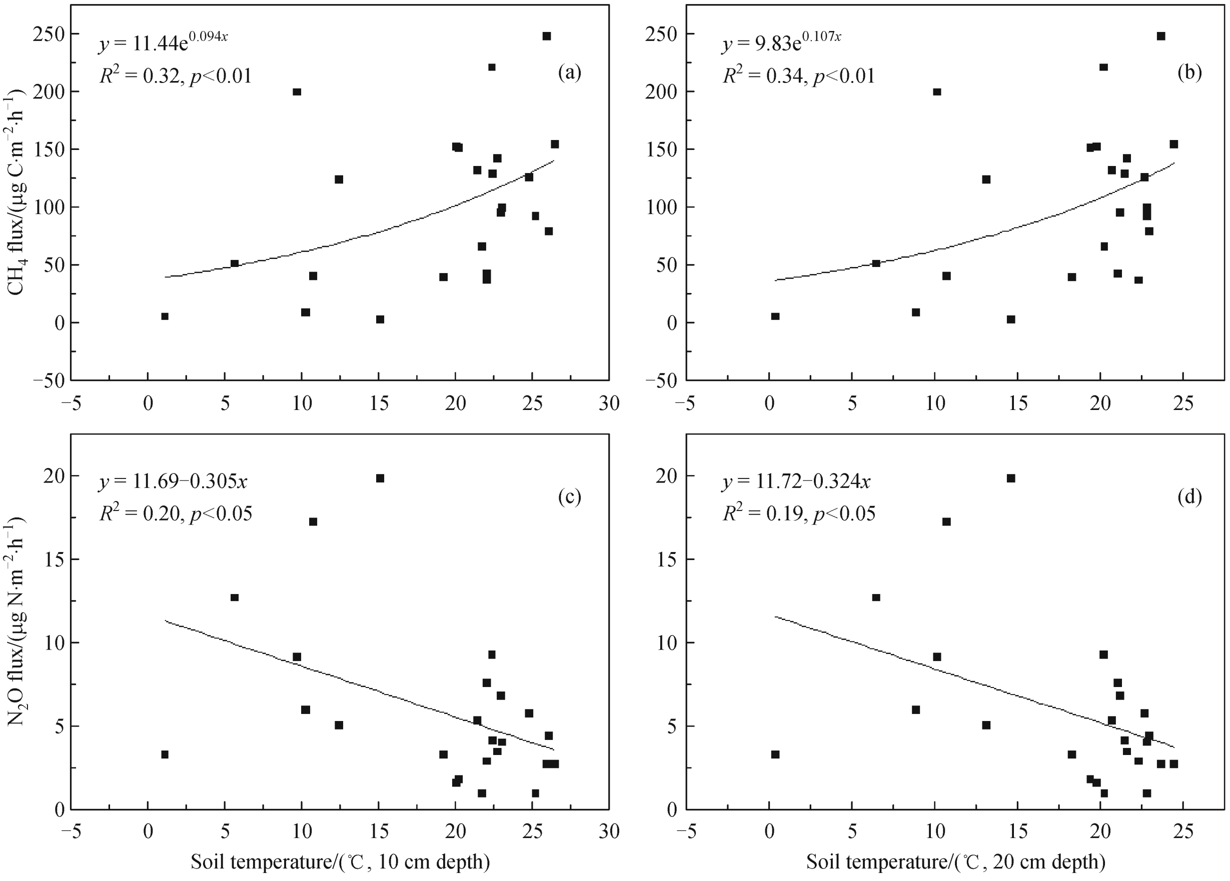

Fig. 8 Effects of soil temperature at 10 cm and 20 cm depth on N_2O emission and CH_4 uptake fluxes at the mown site: (a, b) effects of soil temperature on CH_4 uptake flux; (c, d) effects of soil temperature on N_2O emission flux.

daily N_2O emission fluxes and simultaneously measured soil temperature and moisture in the UM steppe over the whole year ($p > 0.05$), but significant correlation was found between daily N_2O emission flux and soil temperature in the M3 steppe ($p < 0.05$). The R^2 values of the fitted regression curve indicate that the variation in soil temperature may explain 19%–20% ($p < 0.05$) of the variation in N_2O emission (Figs. 8(c) and 8(d)). The annual range of temperature was wide (from −13°C to 26.5°C) in the mowing region, while the optimum temperature ranges for nitrification and denitrification were 15°C–35°C and 5°C–75°C, respectively (Liu et al., 2010). Therefore, soil microbial activity was limited by temperature. We can thus conclude that soil moisture was not the key factor, but that soil temperature was the dominant factor accounting for the difference in N_2O emission between the UM (6.3 μg N $\cdot m^{-2}\cdot h^{-1}$) and M3 sites (4.3 μg N$\cdot m^{-2}\cdot h^{-1}$). Secondly, soil NH_4^+-N (mean of 5.82 versus 4.66 mg$\cdot$kg^{-1}) and NO_3^--N (11.09 versus 10.07 mg$\cdot$kg^{-1}) concentrations were higher in the UM than M3 fields. This may be explained by a faster relative mineral-N turnover in the mown fields. Li (1999) revealed that N mineralization by roots was accelerated as a result of increased NH_4^+-N and NO_3^--N concentrations in the soil, especially during the maximum growth stage of grassland vegetation. In addition, differences in population density of individual plant communities was also found (*Leymus chinense* and *Pulsatilla turczaninovii* as the dominant species in M3 and UM, respectively) (data not shown), and so the change in availability of inorganic N also reflects a shift in plant community structure to some extent. Thirdly, N_2O is predominantly produced by microbial nitrification, which is affected by alterations of the soil environment in most semi-arid grasslands (Verchot et al., 2002; Xu et al., 2008). Newton et al. (2014) reported that a change of the dominant species would affect the composition of soil microorganisms. We cannot exclude the effects of differences in SOM quality or competition processes between plants and microorganisms for N on nutrient availability between both sites, because such differences have been shown to also feed back on N_2O emissions (Groffman et al. 2006), which is inversely related to soil inorganic nitrogen contents (NH_4^+-N and NO_3^--N) and methane uptake. The alteration of soil environment would affect the process of N_2O production and the related microbial functional groups. Therefore, further investigations are necessary.

Generally, the practice of mowing may have promoted CH_4 uptake and inhibited N_2O emissions through three pathways. First, mowing could reduce soil moisture content and increase soil temperature. Secondly, it could reduce soil inorganic N (NH_4^+-N and NO_3^--N) and SOM. And finally, a shift in plant species richness and soil microbial community composition could be affected by mowing.

4.3 Comparison with another grassland management practice

In the semiarid grasslands of Hulunber, Inner Mongolia, grazing is another important management practice. Up to now, published results about the effect of grazing on N_2O production are rather controversial. Some studies have found grazing could suppress N_2O production (Wang et al., 2005; Xu et al., 2008; Wolf et al., 2010). Liu et al. (2010) presented results of 2 years (from January 2005 to December 2006) of measurements of N_2O fluxes from the native and grazed *Leymus chinensis* (LC) steppes in Inner Mongolia, and found the annual net emission of N_2O was 0.30 and 0.26 kg N_2O-N ha^{-1} from the native and grazed LC steppes, respectively. Others, meanwhile, have reported that grazing could enhance N_2O production (Frank et al., 2000; Saggar et al., 2004). Previous studies have reported that grazing tends to reduce CH_4 uptake in some grassland ecosystems. Liu et al. (2007) conducted CH_4 flux measurements during the periods of June to September 2004, May to September 2005, and March to June 2006, and concluded that the annual rate of CH_4 uptake was 3.58 and 1.91 kg C$\cdot$ha^{-1} at ungrazed and winter-grazed sites, respectively. Chen et al. (2010) noted that the annual CH_4 uptake was 3.7 and 2.1 kg C$\cdot$ha^{-1} during the year-round CH_4 flux measurements at an ungrazed site and winter-grazed site from June 2007 to October 2008. In contrast to grazing, mowing also affected the fluxes of CH_4 and N_2O in the meadow-steppe. Moreover, it had a larger potential to increase the CH_4 uptake and reduce N_2O emissions. As shown in Fig. 6 and Table 5, the CH_4 uptake was significantly higher at M3 (70.5 μg C$\cdot m^{-2}\cdot h^{-1}$) than at UM (40.3 μg C$\cdot m^{-2}\cdot h^{-1}$) ($p < 0.01$); whereas, the N_2O emission was lower than that at UM (4.3 versus 6.3 μg N$\cdot m^{-2}\cdot h^{-1}$) ($p < 0.05$) during the observation periods.

The area of natural grassland in China is 3.9×10^8 ha. If all natural grassland behaved in the same way as the mown site in this study, the CO_2 equivalent of the annual net emission of CH_4 and N_2O from all natural grassland in China would be -40.3×10^2 Tg C$\cdot$yr^{-1} (Table 5). An estimation by neglecting the effect of mowing yields a figure of approximately 140.6×10^2 Tg C$\cdot$yr^{-1}. If we have overestimated, as described above, at global and regional/national scales, then the sink and source are obviously reversed. Therefore, this is an important sink that cannot be neglected.

5 Conclusions

Our study has demonstrated that mowing could enhance CH_4 uptake and inhibit N_2O emissions in the semi-arid steppe ecosystem, mainly due to its effect on vegetation types and soil factors, especially on soil temperature. Soil moisture and inorganic nitrogen were not found to be the primary and key dominant factors. Therefore, further investigation is needed. Laboratory experiment results revealed that mowing changed the soil and environmental conditions that determine emissions of N_2O during thawing periods. Our results suggest that the practice of mowing could mitigate potential global warming in terms of CH_4 uptake and N_2O emissions, and mown steppes may be a neglected sink for GHGs.

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References

Bahn M, Knapp M, Garajova Z, Pfahringer N, Cernusca A (2006). Root respiration in temperate mountain grasslands differing in land use. Glob Change Biol, 12(6): 995–1006

Blagodatsky S, Smith P (2012). Soil physics meets soil biology: towards better mechanistic prediction of greenhouse gas emissions from soil. Soil Biol Biochem, 47: 78–92

Cai Z (2012). Greenhouse gas budget for terrestrial ecosystems in China. Science China Earth Science, 55(2): 173–182

Chen W W, Wolf B, Yao Z S, Bruggemann N, Butterbach-Bahl K, Liu C Y, Han S H, Han S G, Zheng X H (2010). Annual methane uptake by typical semiarid steppe in Inner Mongolia. J Geophys Res, 115(D15): D15108

Christensen S, Christensen B T (1991). Organic-matter available for denitrification in different soil fractions: effect of freeze/thaw cycles and straw disposal. J Soil Sci, 42(4): 637–647

Collins S L, Knapp A K, Briggs J M, Blair J M, Steinauer E M (1998). Modulation of diversity by grazing and mowing in native tallgrass prairie. Science, 280(5364): 745–747

Crutzen P J (1970). The influence of nitrogen oxide on the atmospheric ozone content. Q J R Meteorol Soc, 96(408): 320–325

Dlugokencky E J, Houweling S, Bruhwiler L, Masarie K A, Lang P M, Miller J B, Tans P P (2003). Atmospheric methane levels off: temporary pause or a new steady-state? Geophys Res Lett, 30(19),

Du R, Lu D R, Wang G C (2006). Diurnal, seasonal, and inter-annual variations of N_2O fluxes from native semi-arid grassland soils of Inner Mongolia. Soil Biol Biochem, 38(12): 3474–3482

Foster B L, Kindscher K, Houseman G R, Murphy C A (2009). Effects of hay management and native species sowing on grassland community structure, biomass, and restoration. Ecol Appl, 19(7): 1884–1896

Frank D A, Groffman P M, Evans R D, Tracy B F (2000). Ungulate stimulation of nitrogen cycling and retention in Yellowstone Park grasslands. Oecologia, 123(1): 116–121

Gavrichkova O, Moscatelli M C, Kuzyakov Y, Grego S, Valentini R (2010). Influence of defoliation on CO_2 efflux from soil and microbial activity in a Mediterranean grassland. Agric Ecosyst Environ, 136(1–2): 87–96

Groffman P M, Hardy J P, Driscoll C T, Fahey T J (2006). Snow depth, soil freezing, and fluxes of carbon dioxide, nitrous oxide and methane in a northern hardwood forest. Glob Change Biol, 12(9): 1748–1760

Groffman P M, Tiedje J M (1989). Denitrification in north temperate forest soils: spatial and temporal patterns at the landscape and seasonal scales. Soil Biol Biochem, 21(5): 613–620

Guo M Y, Wei Z J, Xu L J, Yang G X, Liu H M, Wu Y L, Xin X P (2011a). Soil respiration of different mowing types of meadows. ACTA AGRESTIA SINICA, 19(1): 51–57 (in Chinese)

Guo M Y, Wei Z J, Yun X J, Wu Y L, Liu H M, Li Y, Xin X P (2011b). Effect of grazing on grassland soil respiration. Pratacultural Science, 28(5): 729–736 (in Chinese)

Guo M Y, Xu L J, Yang G X, Liu R, Liu H M, Wu Y L, Xin X P (2010). Effect of cutting treatment on soil respiration rate of Leymus chinensis grassland. Grassland and turf, 30(6): 10–14 (in Chinese)

Herrmann A, Witter E (2002). Sources of C and N contributing to the flush in mineralization upon freeze-thaw cycles in soils. Soil Biol Biochem, 34(10): 1495–1505

Houghton R A, Hackler J L, Lawrence K T (1999). The U.S. carbon budget: contributions from land-use change. Science, 285(5427): 574–578

Huston M (1979). A general hypothesis of species diversity. Am Nat, 113(1): 81–101

Ilmarinen K, Mikola J (2009). Soil feedback does not explain mowing effects on vegetation structure in a semi-natural grassland. Acta Oecol, 35(6): 838–848

IPCC (2007). Summary for policymakers. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt K B, Tignor M, Miller H, eds. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press

Jones S K, Rees R M, Skiba U M, Ball B C (2005). Greenhouse gas emissions from a managed grassland. Global Planet Change, 47(2–4): 201–211

Kähkönen M A, Wittmann C, Ilvesniemi H, Westman C J, Salkinoja-Salonen M S (2002). Mineralization of detritus and oxidation of methane in acid boreal coniferous forest soils: seasonal and vertical distribution and effects of clear-cut. Soil Biol Biochem, 34(8): 1191–1200

Li C S, Mosier A, Wassmann R, Cai Z C, Zheng X H, Huang Y, Tsuruta H, Boonjawat J, Lantin R (2004). Modeling greenhouse gas emissions from rice-based production systems: sensitivity and upscaling. Global Biogeochem Cycles, 18(1): GB1043

Li X Z (1999). Effects of grazing on characteristics of carbon, nitrogen, phosphorus pools in soil-plant system of typical steppe. Dissertation for Ph.D degree. Beijing: Institute of Botany, Chinese Academy of Science, pp 74 (In Chinese)

Lipson D A, Schmidt S K (2004). Seasonal changes in an alpine bacterial community in the Colorado Rocky Mountains. Appl Environ Microbiol, 70(5): 2867–2879

Liu C Y, Holst J, Bruggemann N, Butterbach-Bahl K, Yao Z S, Yue J, Han S H, Han X, Krummelbein J, Horn R, Zheng X (2007). Winter-grazing reduces methane uptake by soils of a typical semi-arid steppe in Inner Mongolia, China. Atmos Environ, 41(28): 5948–5958

Liu X R, Dong Y S, Qi Y C, Li S G (2010). N_2O fluxes from the native and grazed semi-arid steppes and their driving factors in Inner Mongolia, China. Nutr Cycl Agroecosyst, 86(2): 231–240

Livesley S J, Kiese R, Miehle P, Weston C J, Butterbach-bahl K, Arndt S K (2009). Soil-atmosphere exchange of greenhouse gases in a Eucalyptus marginata woodland, a clover-grass pasture, and Pinus radiata and Eucalyptus globulus plantations. Glob Change Biol, 15 (2): 425–440

Lloyd D (1995). Microbial processes and the cycling of atmospheric trace gases. Trends Ecol Evol, 10(12): 476–478

Luo Y, Sherry R, Zhou X, Wan S (2009). Terrestrial carbon-cycle feedback to climate warming: experimental evidence on plant regulation and impacts of biofuel feedstock harvest. Glob Change Biol, 1(1): 62–74

Maljanen M, Martikkala M, Koponen H T, Virkajarvi P, Martikainen P J (2007). Fluxes of nitrous oxide and nitric oxide from experimental excreta patches in boreal agricultural soil. Soil Biol Biochem, 39(4): 914–920

Mosier A R, Parton W J, Valentine D W, Ojima D S, Schimel D S, Delgado J A (1996). CH_4 and N_2O fluxes in the Colorado shortgrass steppe: 1. Impact of landscape and nitrogen addition. Global Biogeochem Cycles, 10(3): 387–399

Müller C, Martin M, Stevens R J, Laughlin R J, Kammann C, Ottow J C G, Jäger H J (2002). Processes leading to N_2O emissions in grassland soil during freezing and thawing. Soil Biol Biochem, 34(9): 1325–1331

Newton P C D, Lieffering M, Parsons A J, Brock S C, Theobald P W, Hunt C L, Luo D, Hovenden M J (2014). Selective grazing modifies previously anticipated responses of plant community composition to elevated CO_2 in temperate grassland. Glob Change Biol, 20(1): 158–169

Nitschke N, Ebeling A, Rottstock T, Scherber C, Middelhoff C, Creutzburg S, Weigelt A, Tscharntke T, Fischer M, Weisser W W (2010). Time course of plant diversity effects on Centaurea jacea establishment and the role of competition and herbivory. Journal of Plant Ecology, 3(2): 109–121

Ojima D S, Valentine D W, Mosier A R, Parton W J, Schimel D S (1993). Effect of land use change on methane oxidation in temperate forest and grassland soils. Chemosphere, 26(1–4): 675–685

Parr T W, Way J M (1988). Management of roadside vegetation: the long-term effects of cutting. J Appl Ecol, 25(3): 1073–1087

Ravishankara A, Daniel J S, Portmann R W (2009). Nitrous oxide (N_2O): the dominant ozone-depleting substance emitted in the 21st century. Science, 326(5949): 123–125

Robson T, Lavorel S, Clement J, Roux X (2007). Neglect of mowing and manuring leads to slower nitrogen cycling in subalpine grasslands. Soil Biol Biochem, 39(4): 930–941

Rodhe H (1990). A comparison of the contribution of various gases to the greenhouse effect. Science, 248(4960): 1217–1219

Röver M, Heinemeyer O, Kaiser E A (1998). Microbial induced nitrous oxide emissions from an arable soil during winter. Soil Biol Biochem, 30(14): 1859–1865

Saggar S, Bolan N S, Bhandral R, Hedley C B, Luo J (2004). A review of emissions of methane, ammonia, and nitrous oxide from animal excreta deposition and farm effluent application in grazed pastures. N Z J Agric Res, 47(4): 513–544

Singh B K, Tate K (2007). Biochemical and molecular characterization of methanotrophs in soil from a paristine New Zealand beech forest. FEMS Microbiol Lett, 275(1): 89–97

Smith K A, Dobbie K E, Ball B C, Bakken L R, Sitaula B K, Hansen S, Brumme R, Borken W, Christensen S, Prieme A, Fowler D, Macdonald J A, Skiba U, Klemedtsson L, Kasimir-Klemedtsson A, Degórska A, Orlanski P (2000). Oxidation of atmospheric methane in Northern European soils, comparison with other ecosystems, and uncertainties in the global terrestrial sink. Glob Change Biol, 6(7): 791–803

Steinkamp R, Butterbach-Bahl K, Papen H (2001). Methane oxidation by soils of an N-limited and N-fertilized spruce forest in the Black Forest, Germany. Soil Biol Biochem, 33(2): 145–153

Verchot L V, Groffman P M, Frank D A (2002). Landscape versus ungulate control of gross mineralization and gross nitrification in semi-arid grasslands of Yellowstone National Park. Soil Biol Biochem, 34(11): 1691–1699

Wan S, Luo Y, Wallace L L (2002). Changes in microclimate induced by experimental warming and clipping in tallgrass prairie. Glob Change Biol, 8(8): 754–768

Wang R (2004). Photosynthetic pathways and life form types for native plant species from Hulunbeier Rangelands, Inner Mongolia, North China. Photosynthetica, 42(2): 219–227

Wang Y S, Wang Y H (2003). Quick measurement of CH_4, CO_2 and N_2O emission from a short-plant ecosystem. Adv Atmos Sci, 20(5): 842–844

Wang Y S, Xue M, Zheng X H, Ji B M, Du R, Wang Y F (2005). Effects of environmental factors on N_2O emission from and CH4 uptake by the typical grasslands in the Inner Mongolia. Chemosphere, 58(2): 205–215

Whiting G J, Chanton J P (1993). Primary production control of methane emission from wetlands. Nature, 364(6440): 794–795

Willison T W, O'Flaherty M S, Tlustos P, Goulding K W T, Powlson D S (1997). Variations in microbial populations in soils with different methane uptake rates. Nutr Cycl Agroecosyst, 49(1–3): 85–90

Wolf B, Zheng X H, Brüggemann N, Chen W, Dannenmann M, Han X G, Sutton M A, Wu H H, Yao Z S, Butterbach-Bahl K (2010). Grazing-induced reduction of natural nitrous oxide release from continental steppe. Nature, 464(7290): 881–884

Xu Y Q, Wan S Q, Cheng W X, Li L H (2008). Impacts of grazing intensity on denitrification and N_2O production in a semiarid grassland ecosystem. Biogeochemistry, 88(2): 103–115

Zhang B, Chen G Q, Li J S, Tao L (2014a). Methane emissions of energy activities in China 1980–2007. Renew Sustain Energy Rev, 29: 11–21

Zhang L H, Guo D F, Niu S L, Wang C H, Shao C L, Li L H (2012). Effects of mowing on methane uptake in a semiarid grassland in Northern China. PLoS ONE, 7(4): e35952

Zhang P D, Wang X, Chen B R, Xin X P (2014b). CO_2 release charateristics from Stipa baicalensis meadow steppe in the Hulunbeir region, Inner Mogolia, China. Chinese Journal of Applied Ecology, 25(2): 387–393 (in Chinese)

Zhong L, Du R, Ding K, Kang X, Li F Y, Bowatte S, Hoogendoorn C J, Wang Y, Rui Y, Jiang L,Wang S (2014). Effects of grazing on N_2O

production potential and abundance of nitrifying and denitrifying microbial communities in meadow-steppe grassland in northern China. Soil Biol Biochem, 69: 1–10

Zhou J B, Jiang M M, Chen G Q (2007a). Estimation of methane and nitrous oxide emission from livestock and poultry in China during 1949–2003. Energy Policy, 35(7): 3759–3767

Zhou X H, Wan S Q, Luo Y Q (2007b). Source components and interannual variability of soil CO_2 efflux under experimental warming and clipping in a grassland ecosystem. Glob Change Biol, 13(4): 761–775

Zhu R B, Liu Y S, Ma E D, Sun J J, Xu H, Sun L G (2009). Greenhouse gas emissions from penguin guanos and ornithogenic soils in coastal Antarctica: effects of freezing–thawing cycles. Atmos Environ, 43 (14): 2336–2347