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Response of soil N_2O emissions to precipitation pulses under different nitrogen availabilities in a semiarid temperate steppe of Inner Mongolia, China

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Abstract: Short-term nitrous oxide (N_2O) pulse emissions caused by precipitation account for a considerable portion of the annual N_2O emissions and are greatly influenced by soil nitrogen (N) dynamics. However, in Chinese semiarid temperate steppes, the response of N_2O emissions to the coupling changes of precipitation and soil N availability is not yet fully understood. In this study, we conducted two 7-day field experiments in a semiarid temperate typical steppe of Inner Mongolia, China, to investigate the N_2O emission pulses resulting from artificial precipitation events (approximately equivalent to 10.0 mm rainfall) under four N addition levels (0, 5, 10 and 20 g N/(m^2·a)) using the static opaque chamber technique. The results show that the simulated rainfall during the dry period in 2010 caused greater short-term emission bursts than that during the relatively rainy observation period in 2011 (P<0.05). No significant increase was observed for either the N_2O peak effluxes or the weekly cumulative emissions (P>0.05) with single water addition. The peak values of N_2O efflux increased with the increasing N input. Only the treatments with water and medium (WN10) or high N addition (WN20) significantly increased the cumulative N_2O emissions (P<0.01) in both experimental periods. Under drought condition, the variations in soil N_2O effluxes were positively correlated with the soil NH_4-N concentrations in the three N input treatments (WN5, WN10, and WN20). Besides, the soil moisture and temperature also greatly influenced the N_2O pulse emissions, particularly the N_2O pulse under the relatively rainy soil condition or in the treatments without N addition (ZN and ZWN). The responses of the plant metabolism to the varying precipitation distribution and the length of drought period prior to rainfall could greatly affect the soil N dynamics and N_2O emission pulses in semiarid grasslands.

Keywords: temperate semiarid steppe; nitrous oxide; nitrogen availability; precipitation

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Nitrous Oxide (N_2O) is one of the most important global greenhouse gases (Wuebbles, 2009). In contrast to other greenhouse gases, N_2O is often released instantaneously and circumstantially from the soils, particularly after precipitation during drought season (Priemé and Christensen, 2001; Freibauer and Kaltschmitt, 2003; Yao et al., 2010; Harrison-Kirk et al., 2013). By experiments Birch et al. (1964) found that the soil drying and wetting cycles caused by precipitation stimulated the mineralization of soil organic matter, resulting in rapid soil carbon losses; such phenomenon was called the "Birch effect". By following Birch's experiment, several studies (Priemé and Christensen, 2001; Haren et al., 2005; Muhr et al.,

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2008; Kim et al., 2010; Trost et al., 2013) have proven that precipitation also triggers a considerable N_2O emission pulse. Haren et al. (2005) found that the first rainfall after a drought period triggered an N_2O pulse for several hours in Brazilian tropical forests. This pulse accounted for approximately 25% of the drought-associated reduced N_2O efflux and for 1.3%±0.4% of the annual N_2O emissions. Meanwhile, as the intermediate product or by-product of soil nitrification and denitrification, soil mineral N concentrations greatly influenced the amplitude of N_2O emissions from the soil (Weier et al., 1993; Doobie et al., 2003; Peng et al., 2011; Rafique et al., 2011). Burger et al. (2005) indicated that the greatest N_2O effluxes occurred immediately after the application of N fertilizer during the drying and wetting cycles in the agricultural soils in the California Central Valley. In addition, plants and microbes have different sensitivities to water pulses. The variations in precipitation distribution affect the biological activity of microbes and plants, thereby affecting the soil N dynamics and the corresponding soil N_2O effluxes (Austin et al., 2004; Barton et al., 2008; Austin 2011; Dijkstra et al., 2012; Song et al., 2012). Furthermore, the N_2O emission pulses induced by water pulses substantially vary among different ecosystems and under different soil N conditions (Rudaz et al., 1991; Davidson et al., 1992a, b, 1993; Priemé and Christensen, 2001; Huxman et al., 2004a; Burger et al., 2005; Barton et al., 2008; Yao et al., 2010; Li et al., 2011; Xu et al., 2012; Harrison-Kirk et al., 2013).

Grassland soil N_2O emission is an important part of the global N_2O budget and accounts for approximately 14% of the global annual anthropogenic N_2O emissions (Zhang et al., 2010). In China, grasslands comprise the largest terrestrial ecosystems, covering approximately 3.93×10^8 hm^2 or 41.7% of the national land area (Fan et al., 2008). The reported value of N_2O emissions from grasslands in China is 76.5±12.8 Gg N/a (Mummey et al., 2000; Zhang et al., 2010). Most of the grassland ecosystems in China are distributed in arid and semiarid regions. The precipitation in arid and semiarid areas causes rapid transitions between the drought and wet conditions in soil, resulting in marked changes in the soil N cycle (Austin et al., 2004; Saetre and Stark, 2005; Borken and Matzner, 2009; Dijkstra et al., 2012; Zhang et al., 2013). Several previous studies (e.g. Du et al., 2006; Liu et al., 2010; Peng et al., 2011) have explained the relationship between N_2O effluxes and soil water availability. The response of soil N_2O effluxes to the available soil N concentrations in Chinese semiarid grasslands has also been explored (Peng et al., 2011). However, few studies have focused on the short-term pulse dynamic of soil N_2O after precipitation in temperate semiarid grasslands in China (Yao et al., 2010). Moreover, the differences in the N_2O emission pulses caused by precipitation among different soil N availabilities have seldom been discussed previously. Thus, we investigated the effects of simulated rainfall on the N_2O pulse emissions under four different N addition levels, respectively during drought and rainy periods by performing two 7-day field experiments in a typical semiarid temperate steppe in Inner Mongolia. The objectives of this study were (1) to quantify the N_2O emission pulses triggered by precipitation in temperate grassland soils, (2) to explore the responses of major environmental factors to different water and N addition levels, as well as their effects on N_2O emission pulses, and (3) to evaluate the effects of different soil available N concentrations on short-term N_2O losses during rainfall events.

1 Materials and methods

1.1 Study area

The experimental plots were built at a *Leymus chinensis* steppe in the Baiinxile Pasture, Xilin River Basin, Inner Mongolia, China, which is located at 43°26′N to 44°39′N, 115°32′E to 117°12′E, 1,265 m asl. The annual mean temperature in this area is –0.4°C, whereas the mean month temperature ranges from –21.41°C (January) to 18.53°C (July). The mean annual precipitation approximately ranges from 350 to 450 mm, 70% of which falls between June and September (Peng et al., 2011). The soil is classified as chestnut soil under Chinese soil classification or as calcic-orthic Aridisol under soil taxonomic classification. The soil texture consists of 60% sand, 21% clay and 19% silt. The soil depth changes between 100 and 150 cm and horizon A is approximately 20 to 30 cm, and soil organic carbon, total nitrogen, pH and soil buck density were shown in Table 1. The community is dominated by *Leymus chinensis*, *Stipa grandis*, *Agropyron michnoi* and *Cleistogenes squarrosa*. Distinctive grass layer differentiation is observed in the community, with an average height of 40 to 70 cm from June to August and a community canopy density of 60% to 80%. The site was used for grazing prior to enclosure, with a grazing intensity of approximately 2.25 sheep per hectare.

Table 1 Soil physical-chemical properties of the sampling sites

Soil depth (cm)	Organic C (g/kg)	Total N (g/kg)	C/N	pH	Soil bulk density (g/cm^3)
0–10	16.520±1.049	1.426±0.272	11.585	7.03	1.284±0.012
10–20	14.047±1.412	1.354±0.277	10.374	7.09	1.336±0.033
20–30	13.831±1.652	1.206±0.289	11.468	7.23	1.347±0.030

1.2 Experimental design

A field experiment was conducted *in situ* in a *Leymus chinense* steppe during a drought period (from 24 June 2010 to 1 July 2010) and during a rainy period (from 3 August 2011 to 10 August 2011). Fifteen 8 m×8 m experimental plots, which were separated by 1 m buffers, were established in May 2010. These 15 plots were randomly designed with five treatments and three replicates to simulate rainfall events under different soil available N conditions. The treatments were as follows: (1) control, without N or water addition (ZN); (2) with water addition and without N (ZWN); (3) with N addition at a rate of 5 g N/(m^2·a) and with water (WN5); (4) with N addition at a rate of 10 g N/(m^2·a) and with water (WN10); and (5) with N addition at a rate of 20 g N/(m^2·a) and with water (WN20). Approximately 640 L of water, which is equivalent to 10.0 mm rainfall, was manually applied to the experimental plots on June 24 and August 3 by using backpack sprayers (Sponseller, 2007; Jin et al., 2009). A simulated 10 mm rainfall was chosen because it can effectively increase nitrification and affect the average seasonal N_2O effluxes in the semiarid grasslands in Inner Mongolia (Du et al., 2006; Liu et al., 2010). In addition, recent studies have indicated that a 10 mm rainfall surpasses the water threshold for activating microbial N cycling and plant N uptake in semiarid grasslands (Schwinning and Sala, 2004; Dijkstra et al., 2012). NH_4NO_3 was ground into fine powder, weighted, and evenly distributed on the fertilised treatment plots simultaneously with the water addition.

1.3 Sample collection

Static closed opaque chambers with inner dimensions of 50 cm×40 cm×30 cm, were used in this study to collect gas samples. The chambers were made of 8 mm thick black acrylic material, and the chamber surface was covered with reflecting tin foil to overcome the rapid increase in chamber temperature (Dong et al., 2000; Qi et al., 2007; Peng et al., 2011). Sampling was respectively conducted before water and N addition (0 h) and at 2, 5, 24, 48 and 168 h (1 week) after adding water and N. The sampling chambers were placed in the groove of a stainless steel frame, which was inserted at a depth of 5 cm in the soil, then carefully sealed with distilled water. The aboveground parts of the plants were cut off 1 day before each sampling to avoid any plant disturbance on the soil N_2O emission. Gas samples were collected within 30 min to avoid disturbing the natural conditions of the sampling sites. After closing the chamber, the gas samples were collected at 0, 7, 14 and 21 min (Dong et al., 2000; Qi et al., 2007). For each sample, approximately 200 mL of gas was collected through a PVC tube and a silica gel pipe that were connected to the chamber through a three-way stopcock. The collected gas was stored in polyethylene-coated aluminum bags, and the N_2O gas concentrations of the samples were analyzed using a gas chromatograph (Hewlett-Packard 6890, Agilent Technologies, Palo Alto, USA) equipped with an electron capture detector (ECD).

The air, soil, and internal chamber temperatures were recorded during gas sampling. The air and the soil temperatures at depths of 0, 5 and 10 cm were measured using DHM2 mechanical ventilated thermometers and SN2202 digital thermo detectors (Sinan Instruments Plant of Beijing Normal University), respectively. Soil water content was determined using oven-drying method and was then converted to water-filled pore space (WFPS) by using the bulk density and porosity values. During gas sampling, soil samples were collected at the depth of 0 to 10 cm and were analyzed for soil NH_4-N and NO_3-N contents. Soil NH_4-N and NO_3-N were extracted with 2 M KCl and analyzed using an automated flow injection analyzer (Braun and Lübbe, Norderstedt, Germany).

1.4 Data analysis

The N_2O efflux was calculated using the following method by Dong et al. (2000) and Qi et al. (2006):

$$F_{N_2O} = \rho \times h \times \frac{\Delta c}{\Delta T} \times \frac{273}{273+T} . \quad (1)$$

Where F_{N_2O} refers to the N_2O flux (μg/(m^2·h)), ρ is the N_2O gas density (μg/m^3), and h is the chamber height (m). $\Delta C/\Delta T$ denotes the change in gas concentrations inside the chamber during the sampling period (m^3/(m^3·h)), and T represents the average chamber temperature during the sampling period (°C).

The cumulative N_2O fluxes were calculated by interpolating the N_2O fluxes that were measured during the sampling periods (Dong et al., 2000; Peng et al., 2011). The statistical analyses, i.e. nested repeat ANOVA (LSD Test was used) and Pearson's correlation were conducted using SPSS 17.0. The graphs were prepared using Origin 8.0 and the cumulative N_2O emissions were calculated using Matlab 7.0.

2 Results

2.1 Response of short-term N_2O emission to water addition under different N input levels

The response of the N_2O effluxes to water addition was rapid, and the N_2O emission pulse lasted for a short time in both sampling periods (Fig. 1). For the first observation period in late June 2010, water addition and N input significantly affected the temporal variation of the N_2O flux (Table 2, P<0.01). The maximum N_2O emissions occurred within 2 to 5 h and varied between 12.52±0.80 μg/(m^2·h) (ZN) and 442.75±15.39 μg/(m^2·h) (ZWN). Relative to the peak efflux values of the ZN treatment, the maximum effluxes of the ZWN treatment increased by 219.20%. The peak values of the N_2O efflux increased with increasing N input levels. The peak effluxes of N_2O in the WN5, WN10, and WN20 treatments increased by

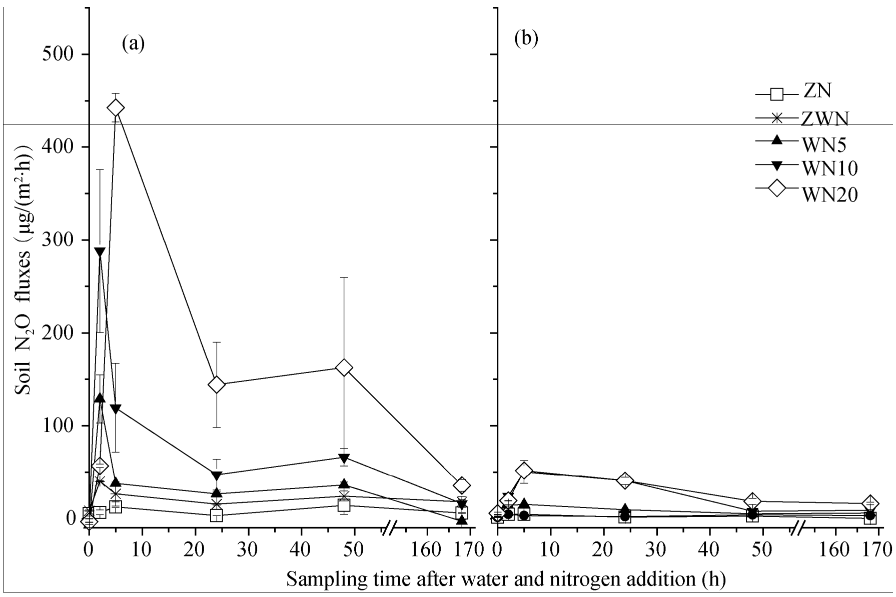

Fig. 1 Short-term dynamics of the soil N_2O effluxes under different water and N treatments, including (a) water and N addition on 24 June 2010 and (b) water and N addition on 3 August 2011. ZN, without N or water addition; ZWN, water addition and without N; WN5, N addition at a rate of 5 g N/(m^2·a) and with water; WN10, N addition at a rate of 10 g N/(m^2·a) and with water; and WN20, N addition at a rate of 20 g N/(m^2·a) and with water.

Table 2 Results of the nested repeated ANOVA on the effects of the sampling time, water addition, N addition and their interaction on the soil N_2O effluxes in both sampling periods

	June 2010				August 2011			
	df	Mean square	*F*	*P*	*df*	Mean square	*F*	*P*
Sampling time	1	614.260	76.684	0.000	1	397.322	12.058	0.006
Water addition	1	896.164	29.932	0.000	1	2,277.698	25.193	0.000
N addition	3	833.142	26.343	0.000	3	2,250.370	27.287	0.000
Sampling time×Water addition	1	645.961	37.505	0.000	1	58.028	1.761	0.214
Sampling time×N addition	3	103.948	26.143	0.000	3	163.410	4.959	0.023

222.29%, 621.11% and 1007.92%, relative to that in the ZWN treatments ($P<0.01$). However, after 48 h, water addition did not show any significant effect on the N_2O effluxes for N treatment levels ($P>0.05$).

For the second observation period in early August 2011, the temporal variation of the N_2O effluxes was insignificantly affected by single water addition (Table 2), and no significant difference in the N_2O effluxes was observed between the WZN and ZN treatments ($P>0.05$). The N_2O peak effluxes in this period were significantly smaller than those in late June for all treatments ($P<0.01$) and ranged between 4.63±0.45 μg/(m^2·h) (ZWN) and 51.89±0.08 μg/(m^2·h) (WN20). Similar to the results obtained in late June 2010, the peak effluxes in the second observation period were significantly greater in the treatments with greater N applied levels within the first 48 h ($P<0.01$).

The cumulative N_2O effluxes for one week after adding water and N were calculated (Fig. 2). The weekly cumulative N_2O emissions of the different treatments ranged from 0.016±0.007 (ZN) to 0.216±0.078 kg N_2O/hm^2 (WN20) for the first sampling period and from 0.004±0.001 (ZN) to 0.042±0.013 kg N_2O/hm^2 (WN20) for the second sampling period. Similar to the difference in the peak values of N_2O effluxes, the weekly cumulative N_2O effluxes in the wet sampling period were significantly less than those in the dry sampling period of late June, 2010 ($P<0.05$). Nested ANOVA showed that only the treatments with water and medium N addition (WN10) or high N addition (WN20) in both

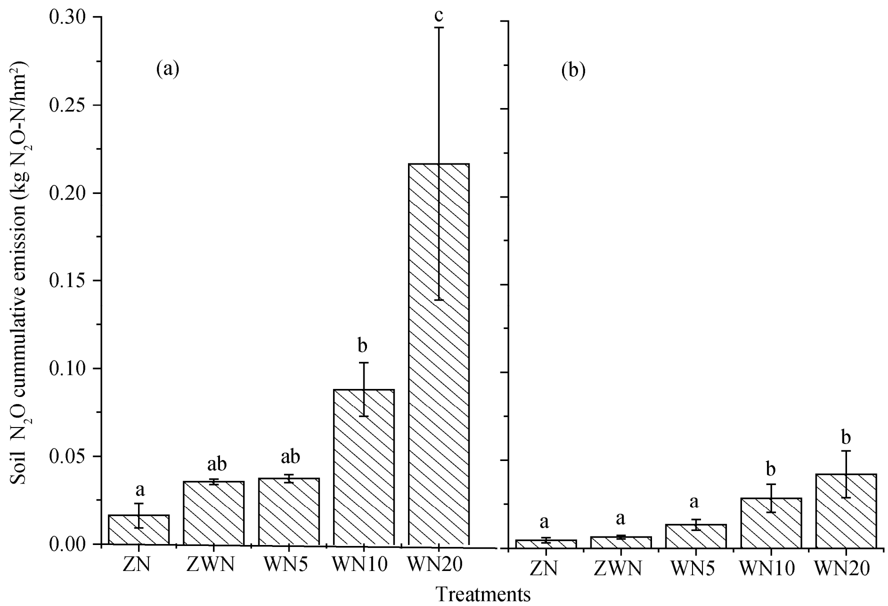

Fig. 2 Cumulative N_2O emissions of the different water and N treatments within 168 h after water and N addition on 24 June 2010 (a) and on 3 August 2011 (b). Bars with different letters denote the treatments that are significantly different at $P<0.05$.

Table 3 Results of the nested ANOVA on the effects of water and N addition on the soil N_2O cumulative effluxes in both sampling periods

Statistical period	Water addition		N addition		R^2
	F	*P*	*F*	*P*	
June 2010	11.617	0.007	17.171	0.000	0.863
August 2011	15.015	0.003	14.892	0.001	0.857

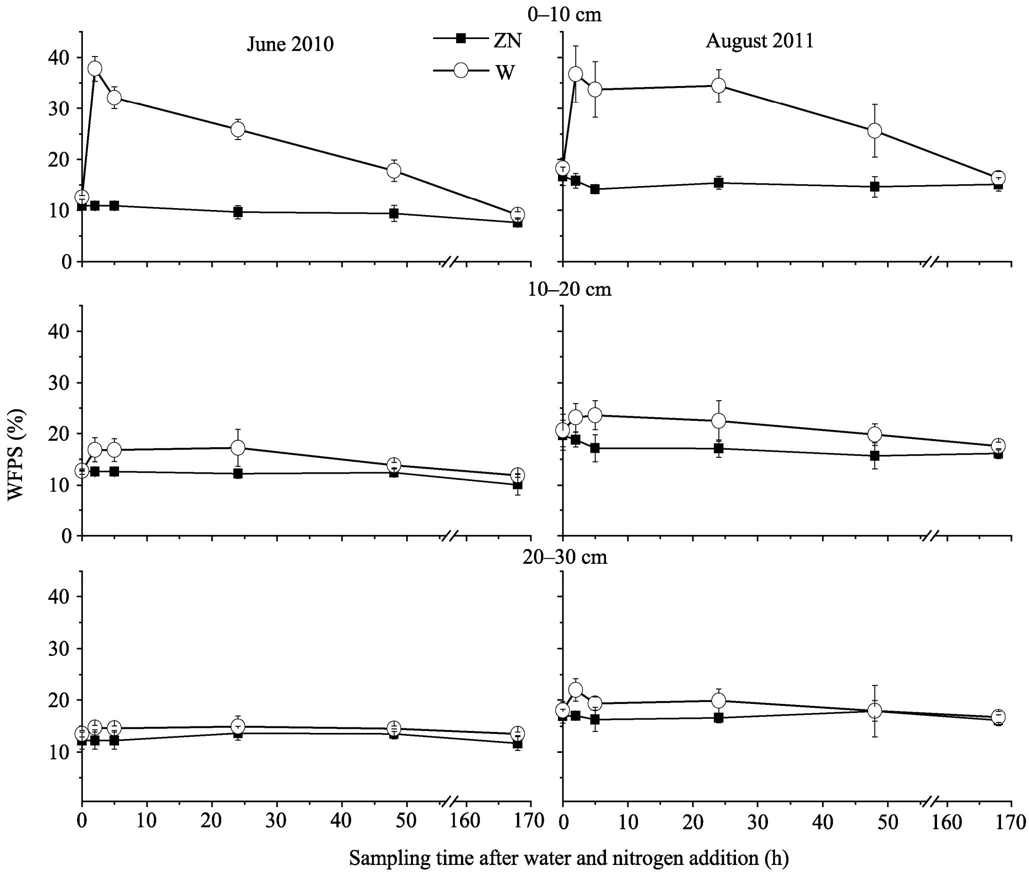

Fig. 3 Dynamics of water-filled pore spaces (WFPSs) in three surface soil layers (0–10 cm, 10–20 cm and 20–30 cm) for the two observation periods. W refers to the mean of the WFPS in the ZWN, WN5, WN10 and WN20 treatments.

sampling periods significantly increased the weekly cumulative N_2O emissions (P<0.01, Table 3).

2.2 Variations in soil WFPS, temperature and mineral N content after water and N addition

The WFPS in three soil layers (0–10 cm, 10–20 cm and 20–30 cm) for the two observation periods are shown in Fig. 3. WFPS remained low and exhibited small fluctuations in the ZN treatments for both sampling periods. However, the WFPS values of the ZN treatment in early August 2011 were significantly greater than those of the same layer in late June 2010 (P<0.01) because of the higher natural precipitation prior to the sampling. The simulated precipitation significantly increased the WFPS of treatments with water addition (ZWN, WN5, WN10 and WN20) in the 0 to 10 cm soil layer for both periods (P<0.01). However, water addition had limited influence on the soil WFPS for the 10 to 20 cm and 20 to 30 cm soil layers.

For both observation periods, air temperature ranged from 22.4°C to 33.1°C (Fig. 4). Within 24 h after water addition, the soil temperatures at the depth of 0 and 5 cm in the treatments added with water were significantly lower than those temperatures in the ZN treatment (P<0.05). The soil temperatures of the treatments added with water at a soil depth of 10 cm

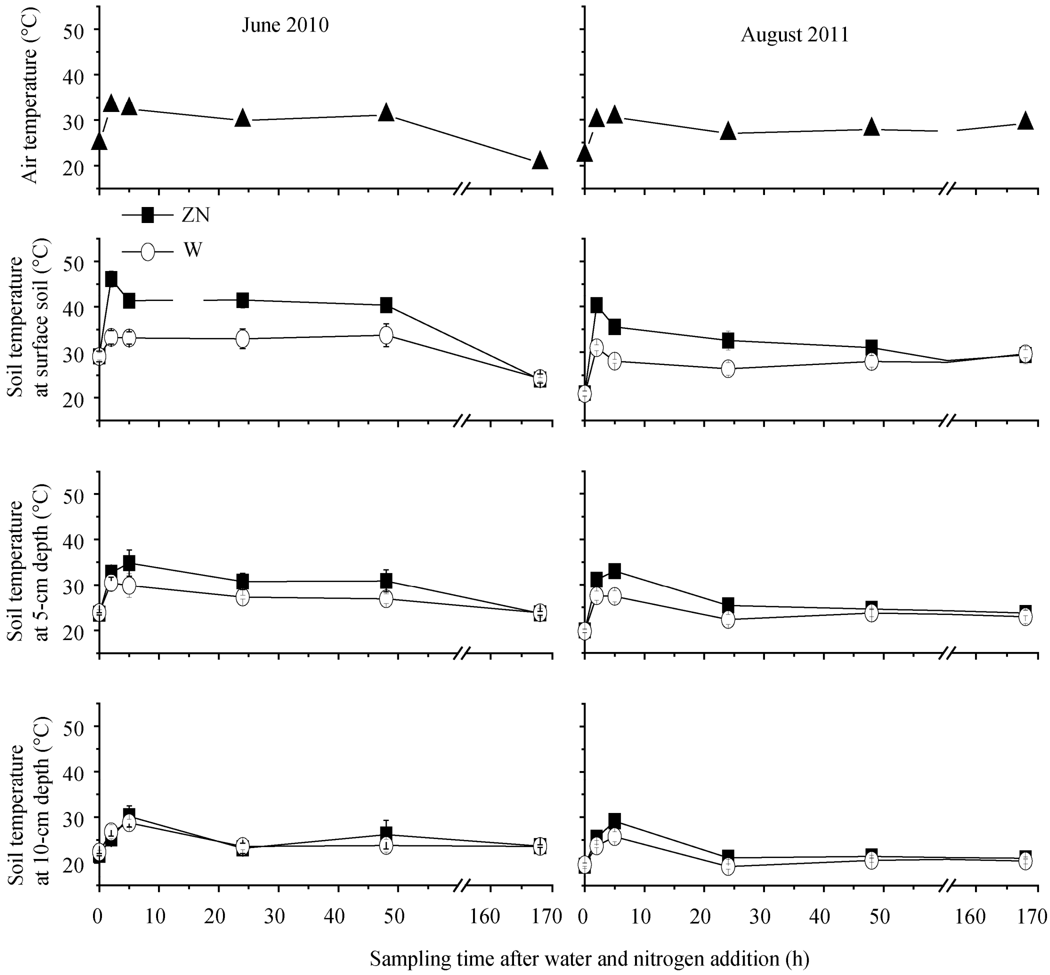

Fig. 4 Air and soil temperatures at three soil depths (0, 5, and 10 cm) for the two observed periods. W refers to the mean of the temperature in the ZWN, WN5, WN10 and WN20 treatments.

were also lower than those of ZN treatment; however, the difference was insignificant (P>0.05).

The soil NH_4-N and NO_3-N contents in the surface soil layer of 0 to 10 cm significantly increased with increasing N fertilizer application (Fig. 5, P<0.05). Meanwhile, the soil NH_4-N and NO_3-N contents in the ZWN treatment significantly increased in the first 2 h after water addition (P<0.01). The soil NH_4-N and NO_3-N concentrations in the WN5, WN10, and WN20 treatments in early August 2011 were significantly lower than the corresponding concentrations in late June 2010 in the first 5 h (P<0.01).

2.3 Correlations between the N_2O effluxes and the environmental factors

The Pearson correlations between the N_2O effluxes and the main environmental factors during the first 24 h were determined for the different treatments because significant variations in the environmental factors and N_2O effluxes were found in this duration (Table 4). Regression equations were established using multiple stepwise regression method to determine the quantitative contributions of the different environmental factor changes to the variations in the N_2O efflux (Table 5).

For the first sampling period, the N_2O effluxes were positively correlated with the soil temperatures in the ZN and ZWN treatments (P<0.1). Besides, the correlations between the soil NH_4-N, WFPS and N_2O effluxes in the ZN treatment were weaker than those in the ZWN treatment. In the treatments with both water and N additions (WN5, WN10 and WN20), the N_2O effluxes were significantly and positively correlated with the soil NH_4-N concentrations, particularly in the

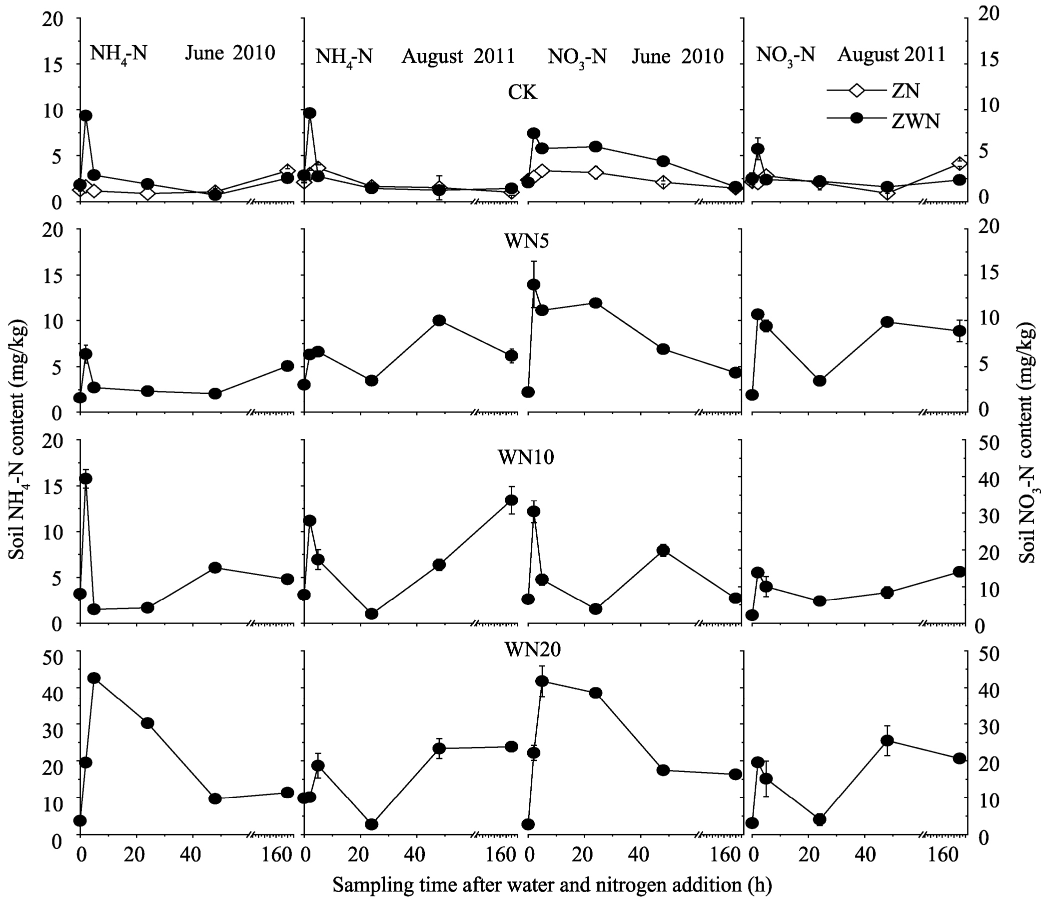

Fig. 5 Variations in mean NH_4-N and NO_3-N contents (n=3) for the different treatments at a depth of 0 to 10 cm during both observation periods

Table 4 Correlations between the N_2O effluxes and the environmental factors

Treatments	WFPS			T^a	T^b			NH_4-N^c	NO_3-N^c
	0–10	10–20 (cm)	20–30		0	5 (cm)	10		
Late June 2010									
ZN	0.619	0.519	0.006	0.474	0.202	0.576	0.927*	0.192	0.472
WZN	0.938*	0.630	0.572	0.887	0.730	0.917*	0.902*	0.781	0.844
WN5	0.868	0.576	0.557	0.787	0.659	0.808	0.558	0.994***	0.776
WN10	0.887	0.555	0.514	0.815	0.654	0.846	0.660	0.964**	0.876
WN20	0.421	0.520	0.413	0.542	0.541	0.536	0.764	0.912*	0.795
Early August 2011									
ZN	−0.583	−0.196	−0.217	0.865	0.807	0.925*	0.922*	0.919*	0.816
WZN	0.791	0.791	0.916*	0.848	0.952**	0.919*	0.727	0.823	0.825
WN5	0.893	0.989**	0.703	0.999***	0.930*	0.970**	0.840	0.920*	0.918*
WN10	0.759	0.848	0.263	0.786	0.584	0.602	0.509	0.039	0.452
WN20	0.563	0.768	0.115	0.698	0.458	0.506	0.460	0.285	0.221

Note: *, ** and *** indicate the correlation was significant at P<0.1, 0.05 and 0.01 level, respectively. a, air temperature (°C); b, soil temperature (°C); c, quality contents in dry soil (mg/kg).

Table 5 Stepwise regression equations between the N_2O effluxes and the environmental factors

Statistical period	Treatments	Regression equation	F	α	R^2
24 June to 1 July 2010	ZN	Y=1.017T_{10}–18.661	12.180	0.073	0.859
	WZN	Y=1.188W_0–9.544	14.619	0.062	0.880
	WN5	Y=27.107N_a–41.561	166.165	0.006	0.988
	WN10	Y=10.210N_a–21.205	26.046	0.036	0.893
	WN20	Y=10.962N_a–102.894	9.930	0.088	0.832
3 August to 10 August 2011	ZN	Y=0.288T_5–4.891	11.884	0.075	0.856
	WZN	Y=0.337T_5–6.225	19.196	0.048	0.906
	WN5	Y=1.466T_a–29.621	929.298	0.001	0.997
	WN10	Y=13.423W_{10}–271.677	5.105	0.152	0.578
	WN20	Y=12.275W_{10}–246.068	2.882	0.232	0.385

Note: T_a means air temperature; T_5, soil temperature at 5 cm depth; T_{10}, soil temperature at 10 cm depth; N_a, NH_4-N contents in the 0 to 10 cm soil layer; W_0, WFPS in the 0 to 10 cm soil layer; W_{10}, WFPS in the 10 to 20 cm soil layer.

WN5 treatment (P<0.01). Multiple stepwise regression results indicated that the changes of soil temperatures at a depth of 10 cm could explain 85.9% of the variation in the N_2O efflux in the ZN treatment. With regard to the ZWN treatment, the changes in the WFPS at 0 to 10 cm were mainly responsible for the increased N_2O efflux and explained 88.0% of the N_2O efflux variation. The increasing soil NH_4-N concentrations in the three treatments with N addition accounted for 77.6% (WN5), 87.6% (WN10) and 79.5% (WN20) of the N_2O efflux variation. The correlations between the N_2O effluxes and the NH_4-N concentrations in early August 2011 were weaker than those in late June 2010 for all the treatments added with water. In the ZN treatment, the N_2O effluxes were positively correlated with the soil temperature and the NH_4-N concentrations (P<0.1). Multiple stepwise regression analysis showed that the temperature and the soil WFPS were the major factors that influenced the N_2O effluxes in early August 2011.

3 Discussion

3.1 Magnitude of the N_2O emission pulses from a single precipitation event

Previous studies have indicated that a single wetting event could affect the annual N_2O efflux by 2% to 50% in different soil ecosystems (Nobre et al., 2001; Barton et al., 2008; Goldberg et al., 2010; Kim et al., 2012). In this study, within one week of water and N addition during drought period, the weekly accumulated emissions of N_2O accounted for 24.12% (ZWN), 21.94% (WN5), 16.69% (WN10), and 25.23% (WN20) of the annual total emissions in 2010 (unpublished data). Meanwhile, during the rainy observation period in 2011, the pulse effects of water and N addition on the N_2O emissions were weak, and the weekly emissions only accounted for 4.92% (ZWN), 6.15% (WN5), 9.46% (WN10) and 7.78% (WN20) of the annual total emissions in 2011 (unpublished data). These results indicate that the precipitation distribution and the length of dry period prior to rainfall significantly affected the N_2O pulse emissions and its contribution to the annual N_2O emissions. Similar results have been observed in different ecosystems. Goldberg et al. (2010) studied German fen soil and found that the soil rewetting resulted in greater N_2O emissions for several days during long drought periods (6 to 8 weeks), and that the N_2O pulse emissions accounted for approximately 20% to 40% of the total annual N_2O emissions. Kim et al. (2010) also observed greater N_2O emissions following the rainfall events after a long dry period in fertilized temperate grassland relative to the N_2O emissions under normal precipitation conditions.

3.2 Soil N dynamics after simulated precipitation and their influence on N_2O emissions

For the relative drought observation period in late June 2010, the soil NH_4-N and NO_3-N concentrations in the ZWN treatment rapidly increased within the first 2 h after water application. The increases in soil

NH_4-N and NO_3-N potentially aroused the N_2O efflux peak at 2 h; such phenomenon can be explained by the following mechanisms: (1) the dead microbial biomass and the disrupted soil macroaggregate were accumulated in the soil during the drought period prior to rainfall, (2) less rainfall during the dry periods reduces the N loss by leaching, and (3) the consumption of soil N is limited because of the lower microbial activity and plant uptake during the dry period. Therefore, more organic substrates and mineral N accumulated in dry soil. When the soils were rewetted, the accumulated organic matter and mineral N were rapidly released, providing a sufficient substrate for the N_2O generation (Davidson et al., 1993; Fierer and Schimel, 2002; Austin et al., 2004; Barton et al., 2008; Yao et al., 2010; Kim et al., 2010; Xu et al., 2012).

In addition, the soil NH_4-N and NO_3-N concentrations also increased during the first 2 h after the water application on August 3, 2011 in the ZWN treatment. However, no obvious increase in the N_2O efflux was observed. What's more, the N_2O emission pulse was even smaller for the N addition treatments (WN5, WN10 and WN20) in this period than those in the late June 2010 ($P<0.01$). We inferred that the differences in the N_2O effluxes between the two sampling periods are due to the different responses of plants and microbes to the water pulses. In semiarid ecosystems, the net N mineralization by microbes and the N uptake by plants can be asynchronous during the drying and wetting cycles. Huxman et al. (2004b) found that microbial populations are sensitive to small precipitation events. By contrast, plants are only responsive to relatively larger precipitation or to multiple small rainfall events. Collins et al. (2008) also found that plants in arid ecosystems are highly sensitive to drought stress because plant metabolism is more easily restrained during drought conditions compared with microbe metabolism. In the present study, no natural precipitation occurred during the two weeks preceding the sampling period in late June 2010, except for a 1.1 mm rainfall on 17 June 2010, and the plant N uptake was limited by the drought weather condition (Kim et al., 2010; Dijkstra et al., 2012). Thus, the soil available N concentration increased during the drought periods, thus providing more substrates for microbes and resulting in greater N_2O emissions when the soil is rewetted. Meanwhile, two heavy rainfalls occurred during the week proceeding the sampling period in early August 2011. The N uptake by plants increased because of the previous rainfall and water addition; thus, the soil mineral N accumulation in this period was lower than that in late June, 2010. Consequently, the N_2O emissions were restrained by the N threshold, because plants were better competitors for soil N than the N_2O producing microbes. In addition, previous studies have confirmed that N_2O production would be suppressed until the plant N demands are fully satisfied in an N-limited ecosystem (McSwiney and Robertson, 2005; Peng et al., 2011; Kim et al., 2012).

3.3 Main influencing factors on the short-term N_2O effluxes under the different water and N conditions

For the ZN treatment in both observation periods, the soil WFPS remained low and did not show obvious variation, thus, the N_2O effluxes were insignificantly correlated with the soil WFPS (Weier et al., 1993; Weitz et al., 2001; Dobbie and Smith, 2003). Previous studies in semiarid climate zones have confirmed the significant influence of temperature on the N_2O emissions during drought soil conditions (Hernandez-Ramirez et al., 2007; Yao et al., 2010; Cantarel et al., 2011). In the present study, the N_2O efflux was positively correlated with temperature for both observation periods. However, the significance of correlation was weak ($P<0.1$) potentially because of the relatively appropriate temperature conditions, which varied between 22.4°C and 33.1°C. The temperatures in this range are generally suitable for nitrification and denitrification and are not the restricting factor of N_2O fluxes during the growing season, thereby resulting in the weak correlation between temperature and N_2O effluxes (Liu et al., 2010). But for the ZWN treatment, the WFPS at depths of 0 to 10 cm significantly increased from less than 20% to more than 30% within a few hours; this marked increase in WFPS would significantly increase soil nitrification (Davidson, 1992). Meanwhile, soil rewetting stimulated C and net N mineralization and increased the soil mineral N concentration. Overall, N_2O generation greatly benefited from increasing WFPS, and accordingly the N_2O effluxes in the ZWN treatment were positively correlated with the soil WFPS values in June 2010.

In this study, N addition greatly increased the soil NH_4-N and NO_3-N concentrations in all the N input treatments. Significant correlations were observed between the N_2O effluxes and the soil NH_4-N concentrations in these treatments. Previous studies have indicated that nitrification is the dominant mechanism for the N_2O production in the semiarid grassland ecosystems (Dong et al., 2000; Du et al., 2006; Yao et al., 2010; Peng et al., 2011). Xu et al. (2003) found that approximately 76% of the N_2O fluxes were produced by nitrification in the grasslands in Inner Mongolia. As the important substrate for nitrification process, changes in the NH_4-N concentration would inevitably influence the soil N_2O effluxes in this region. In addition, the correlation between the N_2O emissions and the soil NH_4-N concentration in the WN5 treatment was stronger than those in the WN10 and WN20 treatments, although the low N input level (WN5) did not significantly increase the weekly N_2O emission pulse. The reason for this perhaps was that the low N input did not fully satisfy the N demand of the ecosystem, and the N_2O production was restrained by the availability of NH_4-N to a certain degree. Thus, the variations in the N_2O efflux were significantly correlated with those in the NH_4-N concentration in the WN5 treatment. With the increase in N input in the WN10 and WN20 treatments, the improvement in the availability of NH_4-N reduced the N restriction for N_2O generation, as well as the correlations between them. During the second observation period, the correlations between the N_2O emissions and the soil NH_4-N concentrations were insignificant (at a significance level of 0.05) in the WN5, WN10 and WN20 treatments. The reason we inferred was that the N uptake by microbes may be limited by more plant N uptake during the rainy observation period in 2011. However, this hypothesis lacks sufficient data support and should be validated with further research.

4 Conclusions

The peak N_2O emissions in all treatments occurred shortly after water and N addition, and no significant difference in N_2O emissions was observed among the different treatments after 48 h. These results suggest that the N_2O emission pulse caused by water addition only lasted for a short period. Weekly or monthly measurements with a manual chamber were not conducive for capturing this N_2O pulse. The improved sampling frequency should be used in future studies to reduce the uncertainties in estimating the annual soil N_2O effluxes in this region, particularly in the area with high available N concentrations. The soil N_2O fluxes for the dry and the rainy periods responded differently to the N additions. The weekly emission pulse accounted for approximately 16.69% to 25.23% of the total annual N_2O emissions during the dry observation period in 2010. During the rainy period, smaller N_2O pulse emissions were observed in the plots treated with single water addition or with both water and N addition. The contributions of the weekly accumulated emissions to the total annual emissions, which varied between 4.92% and 9.46%, were much smaller during this period. The amounts of soil available N accumulated during the dry period or during soil rewetting greatly affected the response of the N_2O emissions to the precipitation pulses and the soil available N contents are largely decided by the different responses of plant and microbial N uptakes to the changes in precipitation in semiarid ecosystems. However, this speculation must be validated with additional research.

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References

Austin A T, Yahdjian L, Stark J, et al. 2004. Water pulses and biogeochemical cycles in arid and semiarid ecosystems. Oecologia, 141(2): 221–235.

Austin A T. 2011. Has water limited our imagination for aridland biogeochemistry? Trends in Ecology & Evolution, 26(5): 229–235.

Barton L, Kiese R, Gatter D, et al. 2008. Nitrous oxide emissions from a cropped soil in a semi-arid climate. Global Change Biology, 14(1): 177–192.

Birch H F. 1964. Mineralisation of plant nitrogen following alternate wet and dry conditions. Plant and Soil, 20(1): 43–49.

Borken W, Matzner E. 2009. Reappraisal of drying and wetting effects on C and N mineralization and fluxes in soils. Global Change Biology, 15(4): 808–824.

Burger M, Jackson L E, Lundquist E J, et al. 2005. Microbial responses and nitrous oxide emissions during wetting and drying of organically and conventionally managed soil under tomatoes. Biology and Fertility of Soils, 42(2): 109–118.

Cantarel A, Bloor J, Deltroy N, et al. 2011. Effects of climate change drivers on nitrous oxide fluxes in an upland temperate grassland. Ecosystems, 14(2): 223–233.

Collins S L, Sinsabaugh R L, Crenshaw C, et al. 2008. Pulse dynamics and microbial processes in aridland ecosystems. Journal of Ecology, 96(3): 413–420.

Davidson E A. 1992a. Sources of nitric oxide and nitrous oxide following wetting of dry soil. Science Society of America Journal, 56(1): 95–102.

Davidson E A. 1992b. Pulses of nitric oxide and nitrous oxide flux following wetting of dry soil: an assessment of probable sources and importance relative to annual fluxes. Ecological Bulletins, 42: 149–155.

Davidson E A, Matson P A, Vitousek P M, et al. 1993. Processes regulating soil emissions of NO and N_2O in a seasonally dry tropical forest. Ecology, 74(1): 130–139.

Dijkstra F A, Augustine D J, Brewer P, et al. 2012. Nitrogen cycling and water pulses in semiarid grasslands: are microbial and plant processes temporally asynchronous? Oecologia, 170(3): 799–808.

Dobbie K E, Smith K A. 2003. Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water–filled pore space and other controlling variables. Global Change Biology, 9(2): 204–218.

Dong Y S, Zhang S, Qi Y C, et al. 2000. Fluxes of CO_2, N_2O and CH_4 from a typical temperate grassland in Inner Mongolia and its daily variation. Chinese Science Bulletin, 45(17): 1590–1594.

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 Biology and Biochemistry, 38(12): 3474–3482.

Fan J W, Zhong H P, Harris W, et al. 2008. Carbon storage in the grasslands of China based on field measurements of above- and below-ground biomass. Climatic Change, 86(3–4): 375–396.

Fierer N, Schimel J P. 2002. Effects of drying–rewetting frequency on soil carbon and nitrogen transformations. Soil Biology and Biochemistry, 34(6): 777–787.

Freibauer A, Kaltschmitt M. 2003. Controls and models for estimating direct nitrous oxide emissions from temperate and sub-boreal agricultural mineral soils in Europe. Biogeochemistry, 63(1): 93–115.

Goldberg S D, Knorr K H, Blodau C, et al. 2010. Impact of altering the water table height of an acidic fen on N_2O and NO fluxes and soil concentrations. Global Change Biology, 16(1): 220–233.

Harrison-Kirk T, Beare M H, Meenken E D, et al. 2013. Soil organic matter and texture affect responses to dry/wet cycles: Effects on carbon dioxide and nitrous oxide emissions. Soil Biology and Biochemistry, 57: 43–55.

Hernandez-Ramirez G, Brouder S M, Smith D R, et al. 2009. Greenhouse gas fluxes in an eastern corn belt soil: weather, nitrogen source, and rotation. Journal of Environmental Quallity, 38(3): 841–854.

Huxman T E, Snyder K A, Tissue D, et al. 2004a. Precipitation pulses and carbon fluxes in semiarid and arid ecosystems. Oecologia, 141(2): 254–268.

Huxman T E, Smith M D, Fay P A, et al. 2004b. Convergence across biomes to a common rain-use efficiency. Nature, 429(10): 651–654.

Jin Z, Dong Y S, Qi Y C, et al. 2009. Precipitation pulses and soil CO_2 emission in desert shrubland of *Artemisia ordosica* on the Ordos Plateau of Inner Mongolia, China. Pedosphere, 19(6): 799–807.

Kim D G, Mishurov M, Kiely G. 2010. Effect of increased N use and dry periods on N_2O emission from a fertilized grassland. Nutrient Cycling in Agroecosystems, 88(3): 397–410.

Kim D G, Vargas R, Bond-Lamberty B, et al. 2012. Effects of soil rewetting and thawing on soil gas fluxes: a review of current literature and suggestions for future research. Biogeosciences, 9(7): 2459–2482.

Li Y, Barton L, Chen D L. 2011. Simulating response of N_2O emissions to fertiliser N application and climatic variability from a rain-fed and wheatcropped soil in Western Australia. Journal of the Science of Food and Agriculture, 92(5): 1130–1143.

Liu X R, Dong Y S, Qi Y C, et al. 2010. N_2O fluxes from the native and grazed semi-arid steppes and their driving factors in Inner Mongolia, China. Nutrient Cycling in Agroecosystems, 86(2): 231–240.

McSwiney C P, Robertson G P. 2005. Nonlinear response of N_2O flux to incremental fertilizer addition in a continuous maize (*Zea mays* L.) cropping system. Global Change Biology, 11(10): 1712–1719.

Muhr J, Goldberg S D, Borken W, et al. 2008. Repeated drying–rewetting cycles and their effects on the emission of CO_2, N_2O, NO, and CH_4 in a forest soil. Journal of Plant Nutrition and Soil Science, 171(5): 719–728.

Mummey D L, Smith J L, Bluhm G. 2000. Estimation of nitrous oxide emissions from US grasslands. Environmental Management, 25(2): 169–175.

Nobre A, Keller M, Crill P, et al. 2001. Short–term nitrous oxide profile dynamics and emissions response to water, nitrogen and carbon additions in two tropical soils. Biology and Fertility of Soils, 34(5): 363–373.

Peng Q, Qi Y C, Dong Y S, et al. 2011. Soil nitrous oxide emissions from a typical semiarid temperate steppe in inner Mongolia: effects of mineral nitrogen fertilizer levels and forms. Plant and Soil, 342(1): 345–357.

Priemé A, Christensen S. 2001. Natural perturbations, drying–wetting and freezing–thawing cycles, and the emission of nitrous oxide, carbon dioxide and methane from farmed organic soils. Soil Biology and Biochemistry, 33(15): 2083–2091.

Qi Y C, Dong Y S, Domroes M, et al. 2006. Comparison of CO_2 effluxes and their driving factors between two temperate steppes in Inner Mongolia, China. Advances in Atmospheric Sciences, 23(5): 726–736.

Qi Y C, Dong Y S, Liu J Y, et al. 2007. Effect of the conversion of grassland to spring wheat field on the CO_2 emission characteristics in Inner Mongolia, China. Soil and Tillage Research, 94(2): 310–320.

Rafique R, Hennessy D, Kiely G. 2011. Nitrous oxide emission from grazed grassland under different management systems. Ecosystems, 14(4): 563–582.

Rudaz A O, Davidson E A, Firestone M K. 1991. Sources of nitrous oxide production following wetting of dry soil. FEMS Microbiology Letters, 85(2): 117–124.

Saetre P, Stark J M. 2005. Microbial dynamics and carbon and nitrogen cycling following re-wetting of soils beneath two semi–arid plant species. Oecologia, 142(2): 247–260.

Schwinning S, Sala O E. 2004. Hierarchy of responses to resource pulses in arid and semi-arid ecosystems. Oecologia, 141(2): 211–220.

Song L, Bao X, Liu X, et al. 2012. Impact of nitrogen addition on plant community in a semi-arid temperate steppe in China. Journal of Arid Land, 4(1): 3–10.

Sponseller R A. 2007. Precipitation pulses and soil CO_2 flux in a Sonoran Desert ecosystem. Global Change Biology, 13(2): 426–436.

Trost B, Prochnow A, Drastig K, et al. 2013. Irrigation, soil organic carbon and N_2O emissions. A review. Agronomy for Sustainable Development, 33(4): 733–749.

Van Haren J L M, Handley L L, Biel K Y, et al. 2005. Drought-induced nitrous oxide flux dynamics in an enclosed tropical forest. Global Change Biology, 11(8): 1247–1257.

Weier K L, Doran J W, Power J F, et al. 1993. Denitrification and the dinitrogen/nitrous oxide ratio as affected by soil water, available carbon, and nitrate. Science Society of America Journal, 57(1): 66–72.

Weitz A M, Linder E, Frolking S, et al. 2001. N_2O emissions from humid tropical agricultural soils: effects of soil moisture, texture and nitrogen availability. Soil Biology and Biochemistry, 33(7–8): 1077–1093.

Wuebbles D J. 2009. Nitrous oxide: no laughing matter. Science, 326(2): 56–57.

Xu R, Wang Y S, Zheng X H, et al. 2003. A comparison between measured and modeled N_2O emissions from Inner Mongolian semi-arid grassland. Plant and Soil, 255(2): 513–528.

Xu Z W, Wan S Q, Ren H Y, et al. 2012. Effects of water and nitrogen addition on species turnover in temperate grasslands in Northern China. PLoS One, 7(6): e39762. doi: 10.1371/journal. pone.0039762.

Yao Z, Wu X, Wolf B, et al. 2010. Soil-atmosphere exchange potential of NO and N_2O in different land use types of Inner Mongolia as affected by soil temperature, soil moisture, freeze-thaw, and drying-wetting events. Journal of Geophysical Research: Atmospheres, 115(D17): D17116. doi: 10.1029/2009JD013528.

Zhang F, Qi J, Li F M, et al. 2010. Quantifying nitrous oxide emissions from Chinese grasslands with a process-based model. Biogeosciences, 7(6): 2039–2050.

Zhang Y, He N, Zhang G, et al. 2013. Ammonia emissions from soil under sheep grazing in inner mongolian grasslands of China. Journal of Arid Land, 5(2): 155–165.