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Effects of increasing precipitation and nitrogen deposition on CH_4 and N_2O fluxes and ecosystem respiration in a degraded steppe in Inner Mongolia, China

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ABSTRACT

Most rangelands in temperate semiarid steppes have degraded due to over-grazing. However, the exchanges of greenhouse gases (GHG) between the degraded steppes have been poorly studied. In this study we investigated the fluxes of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) as ecosystem respiration during the growing season and their responses to simulated increases in water availability and nitrogen supply at a degraded steppe in Inner Mongolia, China. Temporal variation of ecosystem respiration (i.e., CO_2 flux) was dominated by the interaction of soil temperature and moisture, whereas N_2O emissions were mainly dependent on soil moisture. The ambient degraded steppe (i.e., not receiving additional water and nitrogen supplies) was a sink of CH_4 (-1.41 ± 0.04 kg C ha^{-1}) and a source of N_2O (0.17 ± 0.09 kg N ha^{-1}) during the growing season, respectively. Increases in water and nitrogen supplies significantly stimulated N_2O emissions by 65–94% ($p<0.05$) and promoted ecosystem respiration by 47–70% ($p<0.01$), but did not significantly change CH_4 uptake during the growing season in degraded plots. This result indicates that soil source of N_2O and ecosystem respiration in degraded semiarid steppe may be strengthened with increasing precipitation and atmospheric nitrogen deposition. However, this conclusion should be examined at the annual scale in future studies.

1. Introduction

Grasslands, covering approximately 40% of global ice-free land areas, are important sources or sinks of greenhouse gases (GHG) (Sutie et al., 2005; IPCC, 2007). The microbial production and consumption processes of the carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) in grasslands is intimately linked to climatic variations and human disturbances (Ojima et al., 1993; Mosier et al., 1996; Xu et al., 2004; Hao et al., 2010). At present, most grassland suffers from degradation due to overgrazing and poor management, especially arid and semiarid grassland, where 73% of the rangeland is degraded (Steinfeld et al., 2006). The clarification of GHG exchanges in these degraded grasslands is necessary to better understand the contribution of grassland ecosystems to the global budgets and fluxes of C and N trace gases.

Inner Mongolian steppes (approximately 87 million hectare) cover more than 20% of the total grassland area in China. These steppes are characterized by a typical temperate semiarid climate (Wang et al., 2005). In the last few decades, natural steppes have been overgrazed to meet the increasing demand for livestock products (Tong et al., 2004). Overgrazing has resulted in widespread rangeland degradation associated with changes in soil properties (Kang et al., 2007), plant diversity and productivity (Gao et al., 2011) and the activities of soil microorganisms (Su et al., 2005). Recently it was shown that the potential of steppe soils to act as sinks or sources of atmospheric CH_4 and N_2O may be reduced by heavy grazing (Wolf et al., 2010; Chen et al., 2011a). This level of grazing intensity indicates that the magnitude of the sink or source of GHGs in steppe ecosystems may have changed significantly with the expansion of degraded grasslands. The CO_2, CH_4 and N_2O fluxes in grasslands are also sensitive to precipitation (Du et al., 2006; Mariko et al., 2007; Liu et al., 2008; Hao et al., 2010) and soil nitrogen content (Mosier et al., 1991; Willison et al., 1995). In Inner Mongolian steppes, a significant inter-annual variability in precipitation with a trend of increasing annual precipitation has been observed since 1950 (You et al., 2002). In addition, atmospheric nitrogen deposition originating from surrounding industrial nitrogen fixation, crop mediated nitrogen fixation and fossil fuel burning is gradually increasing (Galloway et al., 2003). Although there have been no reports about atmospheric N deposition in steppe regions, however, the atmospheric N deposition has been reported to be 15–50 kg N a^{-1} in Northern China Plain (Xie et al., 2010). Therefore, changes of precipitation

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and atmospheric nitrogen deposition as well as the effect of degradation will increase uncertainties in the magnitudes of the GHG exchanges of steppe ecosystems.

This study presents the hypothesis that the soil sinks or sources of CH_4 and N_2O and ecosystem respiration are significantly affected after the long-term increase of precipitation and nitrogen deposition. Using chamber-based systems, we measured CH_4 and N_2O fluxes and ecosystem respiration during the growing season at a severely degraded steppe site. The responses of GHG fluxes to simulated increases in water and nitrogen addition (by irrigation and fertilization) were analyzed in this study.

2. Materials and methods

2.1. Site description

Our study site was in a severely degraded steppe (43°35′ N, 116°41′ E; 1218 m a.s.l) (100 ha) located in the Xilin River catchment near the Inner Mongolia Grassland Ecosystem Research Station (IMGERS), Chinese Ecosystem Research Network (43°38′ N, 116°42′ E; 1100 m a.s.l.). The local climate is characterized as temperate continental monsoon climate with a frost-free period of 90–110 days. The mean annual air temperature recorded for the period 1982–2007 at the meteorological station of IMGERS was 0.7 °C. The maximum monthly mean air temperature was 19 °C in July and the minimum was −21 °C in January. The annual mean precipitation was 330 mm (range: 166–507 mm yr^{-1}), of which approximately 80% of the precipitation was concentrated in the growing season from May to September. The trend of precipitation increases based on the meteorological data in the last 30 years in this region. The region is dominated by grasses especially *Stipa grandis* and *Leymus chinensis* (88% of the species abundance). The *Stipa grandis* and *Leymus chinensis* steppes account for 65% of the land area of the Xilin River catchment. The soil types of the *Stipa grandis* and *Leymus chinensis* steppes are *kastanozem* and *chernozem*, respectively.

More than 30 years of overgrazing (with at least 2 sheep ha^{-1} yr^{-1}) have changed the soil (0–4 cm) properties and vegetation composition at the degraded steppe (Steffens et al., 2008; Chen et al., 2011b). Compared with adjacent ungrazed steppe, the bulk density increased significantly ($p<0.01$), whereas organic C, total N and total S concentrations decreased significantly ($p<0.01$). The dominant plant species at the degraded steppe investigated in 2005 were *Carex duriuscula*, *Artemisia scoparia*, *A. frigida* and *Potentilla tanacetifolia*.

2.2. Experimental design

Part of the degraded steppe was fenced (0.2 ha) in 2005, and plot experiments were established to study the combined and individual effects of increased precipitation (by water supply) and nitrogen deposition (by supply of nitrogen fertilizer) on GHG fluxes. The experiment was designed as a 2-factorial split-plot with 4 replicates. Two water supply levels (rainfed, W_0; irrigated, W_1) were combined with three levels of nitrogen fertilizer supplies (unfertilized ambient, N_0; urea applications of 25 and 50 kg N ha^{-1}, N_1 and N_2) resulting in 24 subplots (length × width: 5 m × 8 m for each subplot). Pathways of 0.8 m and 3 m in width separated the subplots.

Precipitation data from 1982 to 2003 as obtained by IMGERS were used to calculate the natural water supply to the plots, with irrigation being realized by a pump-line injector. Using the annual precipitation sums of all years, the 22 years were classified into 3 levels, i.e., dry (6 years), moderate (10 years) and wet (6 years), with average growing season (May 1–September 30) rainfall of 236 mm, 280 mm, and 375 mm, respectively. To incorporate seasonal rainfall distribution patterns during the growing season, the average rainfall of a 10-day interval was used to decide if and how much irrigation water had to be supplied additionally to the W_1 plots. The W_1 plots were irrigated at the end of 10-day intervals with the amount of water matching the precipitation occurring during wet years. If the rainfall amount during the 10-day interval was higher than the intended water supply, irrigation in the following 10-day interval was reduced accordingly. In our measuring year of 2008 the growing season precipitation amounted to 295 mm. To match the total growing season precipitation of a wet year three irrigation events (by underground water) with 95 mm water in total were performed on May 17 (+10 mm), July 19 (+55 mm) and July 30 (+30 mm), respectively (Fig. 1). The N_1 and N_2 plots were fertilized annually once, starting from 2005, at the beginning of May by the surface application of urea mixed with fine-sieved, air-dried soil to simulate nitrogen input (i.e., urea) by wind sediment deposition.

Three treatments (i.e., W_0N_0, W_1N_0 and W_1N_2) were chosen to measure the GHG fluxes at this degraded steppe during the growing season of 2008. The W_0N_0 treatment reflected an ambient environment with natural rainfall and nitrogen deposition; the W_1N_0 treatment simulated the increasing rainfall (95 mm) without increasing the nitrogen deposition, while the W_1N_2 treatment represented both the increasing rainfall and nitrogen deposition (50 kg N ha^{-1}). According to Chen et al. (2011b), the forbidding of grazing for 3 years increased the aboveground plant biomass but did not change the plant species (i.e., *Carex duriuscula*, *Artemisia scoparia*, *A. frigida* and *Potentilla tanacetifolia*). They also found that water and nitrogen fertilization significantly increased plant productivity, and the plant species shifted to *Leymus chinensis*, which is widely distributed in natural steppes. Moreover, the content of soil total N and C also significantly ($p<0.01$) increased in water and nitrogen treatment.

2.3. GHG flux measurements

Fluxes of CO_2, CH_4 and N_2O were measured using static opaque chamber and gas chromatography techniques (Zheng et al., 2008; Wang et al., 2010). The static chambers were made of stainless steel and

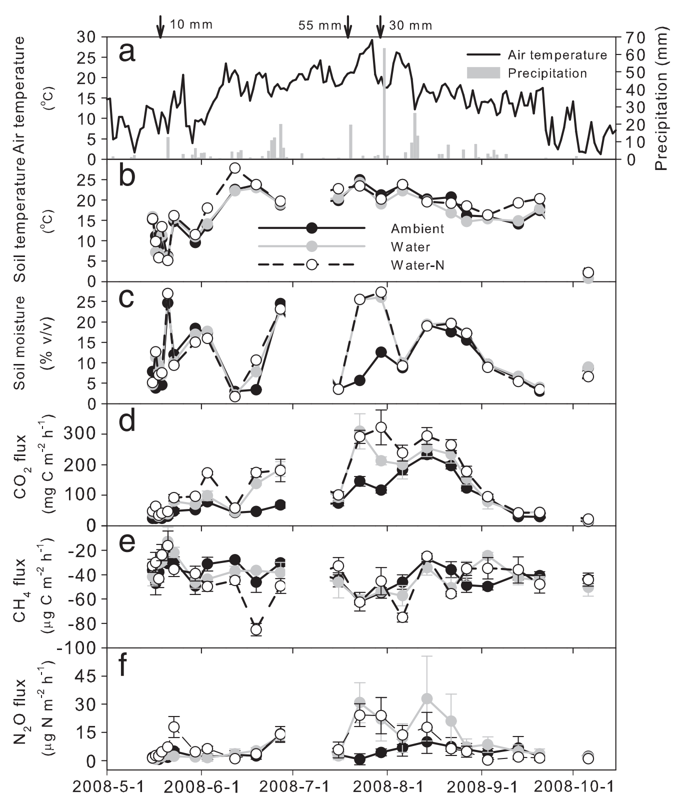

Fig. 1. Temporal variations in air temperature, precipitation (a), soil (5 cm depth) temperature (b), soil (0–5 cm depth) moisture (c) and fluxes of CO_2 (derived from ecosystem respiration) (d), CH_4 (e) and N_2O (f) subject to different treatments during the observation period. Each point of gas flux represents the mean value and standard error.

consisted of two parts: a square base frame (length × width × height = 0.4 m × 0.4 m × 0.1 m) and a removable top (length × width × height = 0.4 m × 0.4 m × 0.4 m). The frames were inserted directly into the soil to a depth of 10 cm and remained fixed during the entire observation period. The chamber tops were mounted onto the base frames during gas sampling and were immediately removed after collection of the air samples. A fan (10 cm in diameter) was installed inside at the top of each chamber to generate turbulence in the chamber closure during sampling. A styrofoam coating at the outside of the chambers prevented an increase in the headspace air temperature due to heating by direct solar radiation.

Gas samples were collected daily during the first week following the nitrogen applications and weekly thereafter during the other investigation periods. Sampling time usually started at approximately 9:00 a.m. local time because the fluxes of CO_2, CH_4 and N_2O during this period are suggested to be representative of the daily average values (Dong et al., 2000). The chambers were closed for an hour, and gas samples (60 ml) were collected every 15 min using plastic syringes. The concentrations of CO_2, CH_4 and N_2O in the gas samples were analyzed by gas chromatography. The gas chromatograph (Agilent 6820, Shanghai, China) was equipped with an electron capture detector (ECD) for N_2O analysis and a flame ionization detector for CH_4 and CO_2 analysis. The CO_2 was converted to CH_4 in a methanizer prior to analysis. The analytical conditions for analyzing the concentrations of CO_2, CH_4 and N_2O and the methods for calculating the fluxes of each gas have been described by Zheng et al. (2008) and Wang et al. (2010). We used a purge gas (CO_2:N_2 = 10:90) for the ECD at a rate of 1–3 mL min^{-1} to avoid cross-interference between CO_2 and N_2O in the air samples (Zheng et al., 2008). The gas fluxes were calculated from changes in the initial rate of concentration change within the chambers' headspace; this was calculated from the initial slope of a non-linear regression of concentration against time (Kroon et al., 2008). Negative flux values indicate gas uptake from the atmosphere, and positive flux values indicate gas emissions to the atmosphere. The detection limits for our approach were 3 μg N m^{-2} h^{-1} for N_2O, 8 μg C m^{-2} h^{-1} for CH_4 and 2 mg C m^{-2} h^{-1} for CO_2 based on the chamber dimensions, the sampling time, and the reproducibility of repeated measurements for standard gases (0.7% for CH_4, 0.8% for CO_2 and 0.6% for N_2O). The daily mean fluxes for each treatment were calculated by averaging the four replicates for each sampling day; these were further integrated into the seasonal cumulative sums using a simple linear approach.

2.4. Ancillary measurements

Aboveground plant biomass at the three treatments was determined on July 15, 2008. Soil temperature at a 5 cm depth was measured for all three treatments using portable digital thermometers (JM 624, JinMing Instrument Co., Ltd., Tianjin, China). The volumetric soil (0–5 cm depth) moisture (%) of the three treatments was measured using a portable TDR probe (ThetaKit, Delta-T Devices, Cambridge, UK). Data of daily precipitation, air pressure and temperature were obtained from the meteorological station at IMGERS.

2.5. Statistical analysis

The significance of differences in mean GHG flux and net cumulative GHG emission among the treatments was investigated using a one-way ANOVA with Tukey's HSD test (SPSS 11.5, SPSS Inc., Chicago, USA). The combined effects of soil temperature and moisture on GHG fluxes and multivariate non-linear regression analysis were evaluated using SigmaPlot 10.0 (SPSS Inc., Chicago, USA).

3. Results

3.1. Environmental variables

The mean air temperature and precipitation during the growing season of 2008 was 0.5 °C warmer and 12 mm higher, respectively, than the long-term average values for the same period (15.1 °C and 283 mm in 1982 to 2007) at IMGERS (Fig. 1a). The most intense rain with 63 mm in one day was recorded on July 31, 2008. During the observation period, there were no significant differences in soil temperatures among the three treatments (Fig. 1b). However, increases in the soil moisture of the W_1N_0 and W_1N_2 treatments occurred during the last 10 days of July following two irrigation events (Fig. 1c). For the entire period, the average soil moisture (% v/v) of the irrigated plots (i.e., W_1N_0 and W_1N_2) amounted to 13.8%, which was 3.2% higher ($p<0.01$) than for the W_0N_0 treatment.

3.2. Ecosystem respiration

The ecosystem respiration of this degraded steppe ranged from 14 to 322 mg C m^{-2} h^{-1} during the sampling period (Fig. 1d). The seasonal pattern of CO_2 fluxes from the W_0N_0 was unimodal, with the maximum flux of 232 mg C m^{-2} h^{-1} on August 14, 2008, and the minimum of CO_2 fluxes for all the treatments recorded in October, i.e., the soil temperatures were lowest. The mean CO_2 fluxes exhibit significant ($p<0.05$) differences among the treatments (Table 1). Compared with the W_0N_0 treatment (2957 ± 141 kg C ha^{-1}), the cumulative CO_2 fluxes from the W_1N_0 and W_1N_2 treatments during the growing season increased significantly ($p<0.01$) by 47% and 70%, respectively (Table 2).

3.3. Methane fluxes

The CH_4 fluxes at the investigated steppe varied in the range of − 85 to − 13 μg C m^{-2} h^{-1} (Fig. 1e), showing that the soil acted as a net sink for atmospheric CH_4. There was a limited seasonality of CH_4 uptake during the growing season, resulting in relatively low temporal CVs for the three treatments (25–38%). The maximum of CH_4 uptake was recorded for the W_1N_2 treatment on June 19 when soil moisture was moderate, while the minimum CH_4 uptake rate was observed for the W_1N_0 treatment on May 21 following a heavy rain event. The CH_4 fluxes did not vary between treatments and were cumulated to a seasonal uptake of 1.4–1.6 kg C ha^{-1} (Tables 1 and 2).

Table 1
Greenhouse gas (ecosystem-respiratory CO_2, CH_4 and N_2O) fluxes, soil (5 cm depth) temperature (ST), soil (0–5 cm depth) moisture (SM), and aboveground plant biomass (AB) in the three treatments.

Treatment	CO_2 flux (mg C m^{-2} h^{-1})	CH_4 flux (μg C m^{-2} h^{-1})	N_2O flux (μg N m^{-2} h^{-1})	ST (°C)	SM (%v/v)	AB (g m^{-2})
W_0N_0	77 ± 4^c	− 40 ± 1^a	4 ± 2^b	16.0	10.6^b	90 ± 8^c
	(14–232)	(− 62 to − 25)	(1–14)	(2.0–24.9)	(3.0–24.6)	(74–98)
W_1N_0	111 ± 10^b	− 40 ± 2^a	8 ± 3^a	15.4	13.9^a	192 ± 13^b
	(17–309)	(− 61 to − 13)	(1–33)	(0.7–24.3)	(2.1–26.9)	(172–232)
W_1N_2	132 ± 4^a	− 43 ± 2^a	8 ± 1^a	16.9	13.7^a	307 ± 17^a
	(22–322)	(− 85 to − 16)	(1–24)	(2.2–27.9)	(1.7–27.3)	(294–330)

The figures outside the parentheses are the means of the four replicates and the standard error, and those inside the parentheses are their ranges. Superscripts as lowercase letters indicate the significant ($p<0.05$) differences among the treatments.

Table 2
Cumulative CO_2 (derived from ecosystem respiration), CH_4 and N_2O fluxes (mean ± standard deviation) during the growing season in each treatment, and changes in cumulative gas fluxes in the W_1N_0 and W_1N_2 treatments compared with the W_0N_0 treatment and between the watered treatments (W_1N_0, W_1N_2).

Gas	Cumulative fluxes (kg C/N ha^{-1})			Relative change (%)		
	W_0N_0	W_1N_0	W_1N_2	$\frac{W1N0}{W0N0}-1$	$\frac{W1N2}{W0N0}-1$	$\frac{W1N2}{W1N0}-1$
CO_2	2957 ± 141	4338 ± 359	5030 ± 149	47^{**}	70^{**}	16^{*}
CH_4	−1.41 ± 0.04	−1.46 ± 0.09	−1.58 ± 0.08	4^{ns}	12^{ns}	8^{ns}
N_2O	0.17 ± 0.09	0.34 ± 0.15	0.28 ± 0.06	100^{*}	65^{*}	-18^{ns}

The symbols of * and ** indicate significant differences in relative changes among three treatments at $p<0.05$ and $p<0.01$, respectively, whereas the ^{ns} represents no significance ($p \geq 0.05$).

3.4. Nitrous oxide fluxes

The range of N_2O fluxes from the degraded steppe was between 1 and 33 μg N m^{-2} h^{-1} (Fig. 1f). Significant N_2O emissions occurred in the W_1N_0 and W_1N_2 treatments following intense rainfall or irrigation events (e.g. June 27, July 20–August 14), with emission rates being significantly higher ($p<0.05$) and the duration of peak emission periods being longer compared with those observed for the W_0N_0 treatment. In addition, a N_2O emission event (18 μg N m^{-2} h^{-1}) at the W_1N_2 treatment was detected on the eighth day after nitrogen addition. The seasonal mean N_2O fluxes were doubled by the addition of water and N, following the order$W_1N_0 = W_1N_2 > W_0N_0$ (Table 1). The growing season cumulative N_2O fluxes showed that the W_0N_0 treatment emitted 0.17 kg N ha^{-1}, whereas in the W_1N_0 and W_1N_2 treatment the N_2O emissions were significantly increased by 94% and 65%, respectively (Table 2).

3.5. Relationships of GHG fluxes and soil temperature and moisture

The soil temperature at a 5 cm depth explained 22–43% of the CO_2 flux variations based on an exponential regression model (Table 3). The CO_2 fluxes were also positively correlated to changes in the soil moisture except W_0N_0 (0–5 cm). There was no significant correlation ($p>0.05$) between CH_4 uptake and soil temperature or moisture in all treatments. The N_2O emissions from the three treatments were significantly correlated to soil moisture (Table 3). Combined changes in soil temperature and moisture significantly ($p<0.01$) explained 73%, 30% and 53% of the observed changes in the CO_2, CH_4 and N_2O fluxes, respectively, in each treatment (Table 3 and Fig. 2).

4. Discussion

4.1. Ecosystem respiration

Atmospheric CO_2 assimilated by photosynthesis evolves by ecosystem respiration from metabolic activity of plant and soil microbes. Changes in ecosystem respiration under global change may affect the functions of steppe ecosystem as sinks or sources of atmospheric CO_2. Our study showed that ecosystem respiration rates at the degraded steppe were closely related to both soil temperature and soil moisture (Table 3). The changes in soil temperature and moisture during the growing season could explain a considerable fraction (76%) of the CO_2 flux variation. A previous study has suggested that soil temperature is the dominating factor of the ecosystem respiration at < 15 °C, while soil moisture will override temperature effects during the other periods at water-limited semiarid steppes (Jia et al., 2006). In this study, ecosystem respiration was also high (123–210 mg C m^{-2} h^{-1}) during the period of daily average soil temperatures > 15 °C. Significantly lower respiration rates (39–66 mg C m^{-2} h^{-1}) were observed during the periods of low soil temperature (< 15 °C) with less living biomass (at the beginning and the end of the growing season). Furthermore, extreme drought (June 12 and July 16 with soil moisture values < 4%) also inhibited ecosystem respiration at the steppe (Fig. 1d).

Ecosystem respiration in this degraded steppe increased markedly with simulated increases in water and nitrogen supplies (Table 2). The highest respiration occurred in the treatment with both additional water and nitrogen supplies (Table 2). These supplies generally promote autotrophic plant respiration including both above- and belowground parts (Chen et al., 2011b) as well as rhizosphere respiration by microbes due to the accelerated decomposition of soil organic matter (Nakano et al., 2008). This explanation is well supported by the significant linear dependence of aboveground biomass ($y = 2247 + 9x$, $n = 3$, $r^2 = 0.95$, $p<0.05$). In addition, during the drought (i.e., mid-July), the additional water supply (by irrigation) induced strong CO_2 emission (from mid-July to the beginning of August) in the water treatments (W_1N_0, W_1N_2) compared with the W_0N_0 treatment (Fig. 1d).

4.2. Methane

The CH_4 uptake rates (13–85 μg C m^{-2}) presented here fall within the range (2–105 μg C m^{-2} h^{-1}) of that observed for other arid or semiarid ecosystems such as deserts (Striegl et al., 1992), prairies (Mosier et al., 1996), croplands (Galbally et al., 2008), and steppes (Wang et al., 2005; Chen et al., 2011a). However, the magnitude of cumulative CH_4 uptake (1.4 kg C ha^{-1}) during the growing season in this degraded steppe was only 52% of that of the surrounding natural ungrazed steppes

Table 3
Dependency of CO_2 (as ecosystem respiration), CH_4 and N_2O fluxes (F) on soil (5 cm depth) temperature (T, °C) and soil (0–5 cm depth) moisture (M, % v/v).

Gas	Factor	Treatment	Regression equation	r^2	p
CO_2 (mg C m^{-2} h^{-1})	T	W_0N_0	$F = 18.3 \times e^{0.084 \times T}$	0.36	<0.01
		W_1N_0	$F = 22.8 \times e^{0.094 \times T}$	0.43	<0.01
		W_1N_2	$F = 54.3 \times e^{0.050 \times T}$	0.22	<0.05
	M	W_1N_0	$F = 22.3 + 6.8 \times M$	0.38	<0.01
		W_1N_2	$F = 27.5 + 8.2 \times M$	0.45	<0.01
	$T \times M$	Overall	$F = 1.1 \times T + 18.1 \times M + 0.3 \times T^2 - 0.4 \times M^2 - 138$	0.76	<0.01
CH_4 (μg C m^{-2} h^{-1})	$T \times M$	Overall	$F = 2.3 \times T - 2.0 \times M - 0.1 \times T^2 + 0.1 \times M^2 - 35$	0.30	<0.01
N_2O (μg N m^{-2} h^{-1})	T	W_0N_0	–	–	–
		W_1N_0	$F = 1.11 \times e^{0.117 \times T}$	0.29	<0.01
		W_1N_2	–	–	–
	M	W_0N_0	$F = 2.1 \times e^{0.06 \times M}$	0.28	<0.05
		W_1N_0	$F = 2.7 \times e^{0.08 \times M}$	0.35	<0.01
		W_1N_2	$F = 2.4 \times e^{0.07 \times M}$	0.47	<0.01
	$T \times M$	Overall	$F = 0.3 \times T + 0.4 \times M - 0.005 \times T^2 + 0.007 \times M^2 - 7$	0.53	<0.01

during the growing season (2.7 ± 0.1 kg C ha^{-1}), even though the prohibition of grazing had already started in 2005. This reduction is most likely due to decreased soil gas permeability as was shown by Liu et al. (2007) and Chen et al. (2011a) for adjacent sites. The 30 years of overgrazing has led to an increase in bulk density at the 0–4 cm soil depth (1.28 g cm^{-3}) compared with surrounding ungrazed sites (0.97–1.07 g cm^{-3}), indicating a lower soil gas permeability (Liu et al., 2007; Chen et al., 2011a).

Four years of increased water and nitrogen supply did not change the soil CH_4 sink significantly in the degraded steppe of the present study (Table 2). For soils in arid or semiarid regions, wetting after rainfall or irrigation may even stimulate CH_4 uptake, as the activity of

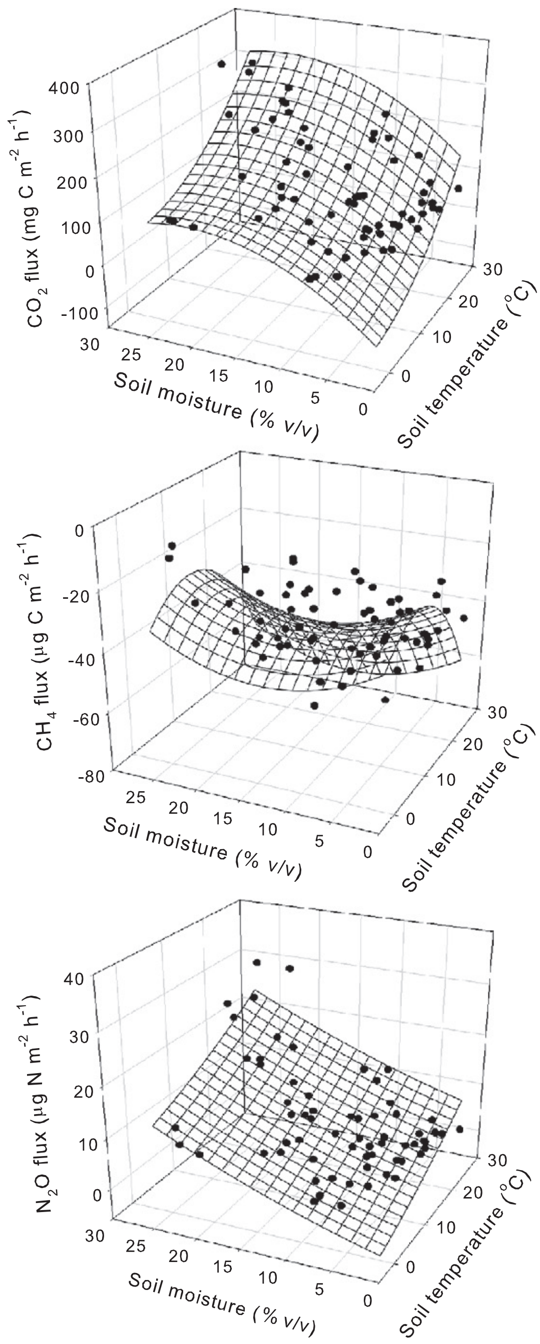

Fig. 2. The CO_2, CH_4 and N_2O fluxes as a function of soil (5 cm depth) temperature (°C) and soil (0–5 cm depth) moisture (% v/v) in the degraded steppe. The data of gas fluxes and soil temperature and moisture are collected from the three treatments and the parabolic mesh graphs are plotted according to the multiple nonlinear regression equations shown in Table 3.

the soil microbial community can become water-limited (Steenwerth et al., 2005; Liu et al., 2008). However, the higher water content could also reduce soil aeration, thus slowing down the diffusion of atmospheric CH_4 to the sites of CH_4 oxidation. The low sensitivity of CH_4 uptake to soil moisture may partially account for the negligible effect of irrigation on the CH_4 uptake. Previous studies have shown that CH_4 oxidation can be suppressed by inputs of nitrogen to the soils of temperate grassland (Mosier et al., 1996) but can also be promoted if sites are found to be strongly N-limited (Bodelier and Laanbroek, 2004). At our degraded steppe site, the water and nitrogen additions in the W_1N_0 and W_1N_2 treatment did not significantly increase total nitrogen contents, which indicates the amount of nitrogen addition was not enough to change the CH_4 uptake by the steppe soil because most of the supplied nitrogen was used for plant growth.

4.3. Nitrous oxide

Temperate semiarid grasslands have been identified as sources of N_2O with an annual mean emission of approximately 0.15 kg N ha^{-1} (ranges: 0–0.28 kg N ha^{-1}) (Mosier et al., 1996; Wolf et al., 2010). In the present study, the N_2O emission during the growing season in the degraded steppe cumulated to 0.17–0.34 kg N ha^{-1}, which was already at the high-end of the annual N_2O emission reported above. However, it should be noted that the pulse of N_2O emissions during the non-growing season are expected to contribute significantly to the annual N_2O budget, indicating that the intensity of the N_2O source of the investigated degraded steppe may be even greater on an annual scale. Compared with N_2O emission from adjacent steppes observed during the growing season (Wolf et al., 2010), the N_2O emission from our degraded steppe without water and nitrogen supply (i.e., the W_0N_0) was more than four times higher. Long-term animal trampling can severely compact topsoil (Steffens et al., 2008). This compaction on the one hand reduces the diffusion of gas but one the other hand forms more anaerobic areas for N_2O production by denitrification after rainfall (Wang et al., 2005) which was supported by the significant relationship between N_2O flux and soil moisture (Table 3). The soil capacity of N_2O production rather than the diffusion rate was most likely the dominant driver of the N_2O flux in the compacted steppe soil.

The water and nitrogen increases significantly stimulated N_2O emission in our degraded steppe (Tables 2 and 3). Increases in soil moisture after water input provide anaerobic conditions for N_2O production and promote the decomposition of residual organic matter enhancing the supply of nitrogen and carbon substrates for denitrification. However, we did not observe significant N_2O emission following the rainfall event on May 21 2008, even if soil moisture exceeded 25% (Fig. 1f). The absence of strong N_2O emission during this period may be seen as the consequence of low soil temperatures (average: 5.5 °C). In this study, the effect of nitrogen input on N_2O emission was also obvious and was reflected in the increased N_2O emissions (18 µg N m^{-2} h^{-1}, 23 May) in the W_1N_2 treatment after nitrogen addition. At the same time N_2O emissions remained low in the W_1N_0 treatment (Fig. 1f). Although the W_1N_2 soils emitted more N_2O than the W_0N_0 treatment, the importance of low nitrogen addition seems to be masked by the role of water addition (e.g., Steudler et al., 1989; Mosier et al., 1991, 1996), with no significant difference in N_2O emission between the W_1N_0 and W_1N_2 treatments. It is possible that most of the nitrogen was consumed by plants, with less nitrogen remaining in the soil as a substrate for N_2O production.

5. Conclusion

Increased precipitation (<95 mm yr^{-1}) and atmospheric nitrogen deposition (<50 kg N ha^{-1} yr^{-1}) had no effect on the soil CH_4 sink during the growing season, whereas N_2O emissions doubled and ecosystem respiration increased 1.5 and 1.7-fold. This result indicates that soil N_2O source functions and ecosystem respiration in degraded semiarid steppe may be further strengthened with increasing precipitation and atmospheric nitrogen deposition. However, these conclusions should be required to be elucidated at the annual scale, which could not be performed here due to infrastructural constraints.

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