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Long-term large N and immediate small N addition effects on trace gas fluxes in the Colorado shortgrass steppe

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Abstract Land use changes in semiarid grasslands have long-lasting effects. Reversion to near-original conditions with respect to plant populations and productivity requires more than 50 years following plowing. The impact of more subtle management changes like small, annual applications of N fertilizer or changing cattle stocking rates, which alters N redistribution caused by grazing and cattle urine deposition, is not known. To investigate the long-term effects of N addition to the Colorado shortgrass steppe we made weekly, year-round measurements of N₂O and CH₄ from the spring of 1990 through June 1996. Fluxes of NO_x (NO plus NO₂) were measured from October 1995 through June 1996. These measurements illustrated that large N applications, either in a single dose (45 g N m⁻²), simulating cattle urine deposition, or in small annual applications over a 15-year period (30 g N m⁻²) continued to stimulate N₂O emissions from both sandy loam and clay loam soils 6–15 years after N application. In sandy loam soils last fertilized 6 years earlier, average NO_x emissions were 60% greater than those from a comparable, unfertilized site. The long-term impact of these N additions on CH₄ uptake was soil-dependent, with CH₄ uptake decreased by N addition only in the coarser textured soils. The short-term impact of small N additions (0.5–2 g N m⁻²) on N₂O, NO_x emissions and CH₄ uptake was observed in field studies made during the summer of 1996. There was little short-term effect of N addition on CH₄ uptake in either sandy loam or clay loam soils. Small N additions did not result in an imme-

diately increase in N₂O emissions from the sandy loam soil, but did significantly increase N₂O flux from the clay loam soil. The reverse soil type, N addition interaction occurred for NO_x emissions where N addition increased NO_x emissions in the coarser textured soil 10–20 times those of N₂O.

Key words Nitrous oxide · Methane consumption · Nitrification · Oxides of nitrogen

Introduction

The shortgrass steppe is within the Great Plains of central North America. The Great Plains cover about 2.23×10^{12} m² of land area, and represents approximately 19% of the world's temperate grassland (Bouwman 1990; Lauenroth et al. 1994). Land use changes in semiarid grasslands have been shown to have long-lasting ecosystem effects. Cropped areas within the shortgrass steppe that have been abandoned and reverted to grasslands appear to require >50 years to recover their typical vegetation types and distribution (Hyder et al. 1975; Briske and Wilson 1980; Reichardt 1982). Recovery of microbial populations which regulate soil consumption of atmospheric CH₄ and production of N₂O requires more than 8 years, but less than 50 years (Mosier et al. 1997). Recent studies within the Colorado shortgrass steppe (SGS) demonstrate that grassland fluxes of N₂O and CH₄ represent an important part of the global atmospheric budget (Mosier et al. 1996, 1997).

The soil-atmosphere exchange of N₂O and CH₄ influences the radiative properties of the troposphere, and CH₄ is involved in the balance of oxidants as well (IPCC 1996). Temperate grasslands comprise about 8% (about 11.5×10^{12} m²) of the global land surface area (Bouwman 1990). Assuming a spatial distribution in all temperate grasslands of 70% upland and 30% lowland (Schimel et al. 1986), average SGS N₂O emissions and CH₄ uptake rates are $1.6 \mu\text{g N m}^{-2} \text{ h}^{-1}$ and $31.2 \mu\text{g C}$

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$\text{m}^{-2} \text{h}^{-1}$, respectively, from long-term observations (1990–1994) presented in Mosier et al. (1991, 1996, 1997). Assuming that these values are typical of the rest of the world, N_2O emissions average $0.16 \text{ Tg N year}^{-1}$, and CH_4 uptake averages $3.2 \text{ Tg CH}_4\text{-C year}^{-1}$ for temperate grasslands globally.

Within the Great Plains $0.69 \times 10^{12} \text{ m}^2$ is cropped, mostly to winter wheat (Laurenroth et al. 1994). Assuming the same gas flux, annual means and spatial distribution across the Great Plains as noted above, we can estimate the impact of converting part of the Great Plains grassland into wheat-fallow cropping. Approximately $1.54 \times 10^{12} \text{ m}^2$ of land remains as grassland. If we assume that wheat-fallow cropping decreases the annual CH_4 uptake to $17 \mu\text{g CH}_4\text{-C m}^{-2} \text{h}^{-1}$ or $149 \text{ mg C m}^{-2} \text{year}^{-1}$ and increases N_2O emissions to $3.6 \mu\text{g N m}^{-2} \text{h}^{-1}$ or $31.5 \text{ mg N m}^{-2} \text{year}^{-1}$, as we observed in our studies, then across the Great Plains, CH_4 uptake would be decreased by a total of 14% and N_2O emissions increased by 35% annually (Mosier et al. 1997).

The impact of more subtle management changes like small annual applications of N fertilizer or changing cattle stocking rates, which alters N redistribution caused by grazing and cattle urine deposition, is not known. Additionally, studies (Bronson and Mosier 1993) indicated that additions of small amounts of $\text{NH}_4^+\text{-N}$ to soils from our grassland study sites caused an immediate decrease in CH_4 oxidation and increased N_2O emissions (Parton et al. 1988). We had not tested the effect of small N additions, similar to the approximately 0.5 g N m^{-2} total annual mineral N deposition at the research site, under field conditions. The study reported herein was conducted to compare the observed long-term effects of $30\text{--}45 \text{ g N m}^{-2}$ added to sites within the SGS 6–13 years earlier to the immediate effect of adding $0.5\text{--}2 \text{ g N m}^{-2}$ of different forms of mineral N on CH_4 uptake, and N_2O and NO emissions from sites within the SGS.

Material and methods

Research sites

Long-term N effects

In this paper we present data from two sets of sites that we established in the SGS within the United States Department of Agriculture-Agricultural Research Service Central Plains Experimental Range (CPER), which is located about 60 km northeast of Fort Collins, Colorado ($40^\circ 48' 23'' \text{N}$, $104^\circ 45' 15'' \text{W}$). At these sites annual precipitation averages about 350 mm year^{-1} with about 80% occurring between May and September. Typically, the vegetation is dominated by blue grama (*Boutelous gracilis* Lag.) Fringed sagebrush (*Artemisia frigida* Willd.) and plains prickly pear (*Opuntia polyacantha* Haw.) are secondary dominants (Laurenroth and Milchunas 1991).

The first set of unfertilized sites (catena) was established along a toposequence on a midslope (MN) sandy loam soil classified as an Ustollic Haplargid and swale (SN) sandy clay loam (Aridic Argiustoll). In July 1981 and May 1982, 5-m^2 plots were fertilized on the midslope (MF) and swale (SF), respectively, with a

solution containing 45 g m^{-2} of $\text{CO}(\text{NH}_2)_2\text{-N}$ to simulate the $\text{CO}(\text{NH}_2)_2\text{-N}$ concentration from cattle urine patches (Mosier and Parton 1985).

The second site (pasture) was established in April, 1990 within a paired fertilized (PF) and native unfertilized (PN) pasture. The PF site had been fertilized with $2.2 \text{ g N m}^{-2} \text{year}^{-1}$ of NH_4NO_3 from 1976 until October 1989. Both pastures were grazed from May through October of each year by cattle (20 head/130 ha). The terrain of the two pastures is essentially flat, and the soil, an Ascalon sandy loam (Fine-loamy, mixed, mesic Aridic Argiustolls), is similar to the catena midslope. The data reported herein cover the measurement period of January 1995 through June 1996. These data are from a continuation of the long-term studies reported in Mosier et al. (1991, 1996, 1997), which report data collected during 1990 through 1994.

Immediate N addition effects

During the week of 30 June 1996, micro plots were established within two additional study sites by installing gas flux chamber anchors (see gas flux measurements below). Four replicate micro plots were established for each fertilizer treatment at each site. Site 1, called "sand" from hereon, was established on a sandy loam soil of the same soil series as the above PN site. Site 2, called "clay" from hereon, was established on a clay loam soil [Nunn clay loam (fine montmorillonitic, mesic Aridic Argiustoll)] site that is located about 1 km from the sand site (Table 1). On 8 July 1996 the following treatments were imposed within the micro plots at both sites: Control (CONT; no N addition, 0.5 cm of distilled water only sprayed onto the soil surface to cover the micro-plot as uniformly as possible); KNO_3 (KN; 0.5 g N m^{-2} in 0.5 cm of water); NH_4NO_3 (AN; 0.5 g N m^{-2} in 0.5 cm of water); $(\text{NH}_4)_2\text{SO}_4$ (AS.5; 0.5 g N m^{-2} in 0.5 cm of water) and $(\text{NH}_4)_2\text{SO}_4$ (AS2; 2 g N m^{-2} in 0.5 cm of water). Gas fluxes were measured daily for 3 days following application, then weekly until the end of August 1996.

Gas flux measurements

N_2O and CH_4 measurements

We established four flux measurement locations within previously fertilized and unfertilized areas at both positions in the catena and six locations in the PF and PN sites. Chamber anchors for each gas sampling location were installed randomly, about 5 m apart, within each site. They were installed in the sand catena in March, 1990, in pasture A in April 1990 and in the clay catena in September 1991. The permanently placed anchors were made from 20.3-cm inside diameter PVC pipe which was driven 8 cm into the soil. We measured the soil-atmosphere exchange of CH_4 and N_2O within each location by fitting a 7.5-cm high, closed, vented chamber (Hutchinson and Mosier 1981) onto the anchor. Gas samples from inside the chambers were removed with 60-ml polypropylene syringes fitted with nylon stopcocks, at 0, 15, and 30 min after the chambers were installed. We usually analyzed the samples within 6 h by gas chromatography. Flux measurements were made mid-morning of each sampling day, generally weekly at each site. Details of gas sampling and analyses are noted in Mosier et al. (1991, 1993, 1996). The unpublished N_2O and CH_4 gas fluxes reported in this paper are for the period of 1 January 1995 through June 1996.

NO_x measurements

At the catena and PN and PF sites NO_x (NO plus NO_2) flux measurements were made weekly from October 1995 through June 1996 and only during the months of July and August 1996 at the sand and clay sites. Fluxes were measured by flowing outside air at the rate of 1 l min^{-1} through a 6-l chamber for 6 min after

Table 1 Soil properties (0–15 cm) of the different sites

Site	Sand (%)	Clay (%)	Bulk density g cm ⁻³	Total N (%)	Total C (%)	pH		Gravimetric soil water content at water-filled pore space		
								0.2 (kg kg ⁻¹ × 10 ²)	0.4 (kg kg ⁻¹ × 10 ²)	0.6 (kg kg ⁻¹ × 10 ²)
Long-term sites										
Catena										
Midslope (native)	76	13	1.42	0.141	1.25	6.5	6.5	13.1	19.6	
Midslope (fertilized)	76	13	1.42	0.138	1.22	5.6	6.5	13.1	19.6	
Swale (native)	58	24	1.34	0.189	1.84	6.0	7.4	14.8	22.2	
Swale (fertilized)	58	24	1.34	0.223	2.22	5.6	7.4	14.8	22.2	
Pasture										
Pasture (native)	74	13	1.41	0.12	1.07	6.2	6.6	13.3	19.9	
Pasture (fertilized)	74	15	1.34	0.14	1.18	5.6	7.4	14.8	22.2	
Short-term sites										
Sandy loam	74	11	1.36	0.127	1.207	7.1	7.2	14.4	21.6	
Clay loam	21	39	0.91	0.298	3.087	7.3	14.4	28.9	43.2	

the chamber was placed on the microplot anchor, either 30 min before or after other chambers were used for measuring N₂O and CH₄ fluxes. The change in concentration of the NO and NO₂ in the air flowing through the chamber was measured with a Scintrex LNC-3 converter and LMA-3 analyzer by measuring a chemiluminescent reaction with Luminol (Scintrex, Concord, Ontario, Canada; Martin 1996) at the clay site. At the sand site the same chamber system was used with a Thermo Environmental Instruments model 42C chemiluminescence NO-NO₂-NO_x analyzer that is housed in a field mobile laboratory. Emissions from the soil are typically >90% NO, so fluxes will generally be discussed in terms of NO only (Martin 1996).

Ancillary measurements

Soils (0–15 cm) were collected from each sampling site at the time of gas flux measurement. The soils were analyzed gravimetrically for soil water content and extracted, field moist, with 2 M KCl, and the extracts analyzed colorimetrically for NH₄⁺ and NO₃⁻ plus NO₂⁻ content using a flow-injection analyzer. We analyzed soils from each site for total C and N content with a combustion C/N analyzer (Table 1). Soil temperature at 5 cm depth was measured at each sampling time at each site using a hand-held digital thermometer that was calibrated to within 0.1 °C over the temperature range observed. Air temperature before and after each sampling period was measured with the same device. The mean of initial and final air temperature was used in gas flux calculations. Weather variables were measured continuously at a meteorological station located about 200 m from the sand catena.

Statistical analysis

The probability that mean flux rates, during a specified period for a given site, differed from those of another site over the same time period was tested using a paired *t*-test. Within each site, fluxes were measured at four to six measurement locations. Over the time period for which differences were being tested, the mean flux rate at each measurement location was calculated, and each *t*-test was then conducted with these measurement site means to compare sites. For the field data, sites were considered to be significantly different if *P* ≤ 0.1 (SAS 1991).

Results and discussion

Long-term impact of N additions to SGS soil on N₂O, NO_x and CH₄ fluxes

N₂O

Studies of N₂O emissions in the SGS during the year following N application, used to simulate cattle urine deposition (Mosier and Parton 1985; Parton et al. 1988), showed that N losses were small (0.5–1% of N added). Both field and laboratory studies showed that when only N₂O and N₂ are considered, N₂O is generally the primary nitrification-denitrification product, with nitrification accounting for 60–80% of the total annual flux of N (Parton et al. 1988). Annual N₂O emissions from SGS soils that have only natural N addition through N deposition averaged about 100 g N ha⁻¹ year⁻¹, i.e. about 2% of annual N input estimated from wet and dry deposition (Parton et al. 1988). We did not know if there was longer term enhancement of N₂O emissions following N addition to SGS soils. In our more recent studies we measured N₂O emissions from two different pastures that had been fertilized 1–9 years before starting the studies.

Catena N₂O

The MF site was fertilized with 45 g CO(NH₂)₂-N m⁻² in July 1981 and the SF site was fertilized with the same amount of N in April, 1982. For the 1990–1994 measurement period the MN site N₂O flux averaged 1.5 μg N m⁻² h⁻¹ compared to 1.9 μg N m⁻² h⁻¹ for the MF site. N₂O fluxes were also typically higher (*P* < 0.05) in the SF than in the SN site (Fig. 1). The 4-year mean flux for the SN site was 1.9 μg N m⁻² h⁻¹ compared to

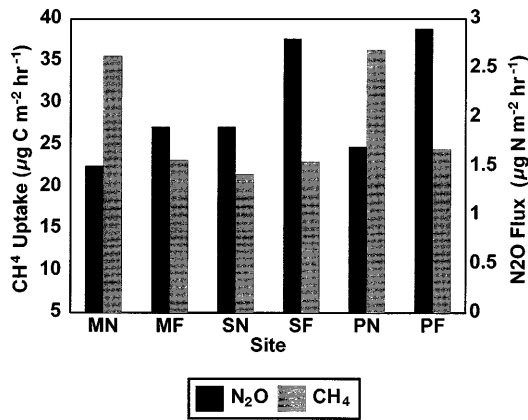


Fig. 1 Mean CH₄ uptake and N₂O emission rates from weekly flux measurements from 1990 through 1994 at six sites within the Colorado shortgrass steppe. The unfertilized site, *MN*, is from the midslope of the catena. Site *MF* on the catena midslope was fertilized with 45 g N m⁻² in 1981. The unfertilized site, *SN*, is from the swale of the catena, while site *SF* on the swale was fertilized with 45 g N m⁻² in 1982. The unfertilized site, *PN*, is a native pasture, and site *PF* was fertilized with 2.2 g N m⁻² year⁻¹ from 1976 until 1989

2.8 µg N m⁻² h⁻¹ for the *SF* site. In an analysis of soils from micro plots at these sites that were fertilized with ¹⁵N-labeled CO(NH₂)₂, Delgado et al. (1996) found that 29% and 85% of the N applied on the midslope and swale in 1981 and 1982, respectively, were recovered in 1992 in plant tissue and soil organic matter. Although soil mineral N values were not elevated in the sites that were fertilized in 1981 and 1982, a residual effect of fertilization on N₂O emissions remained evident 12 and 13 years later.

Pasture N₂O

Application of 22 kg N ha⁻¹ year⁻¹ from 1976 through 1989 increased ($P < 0.05$) N₂O emissions by an average of 73% during 1990–1994 (Fig. 1). The mean flux from the *PN* compared to the *PF* site over the measurement period was 1.7 µg N m⁻² h⁻¹ and 3.0 µg N m⁻² h⁻¹, respectively (Fig. 1). Emissions were lower ($P < 0.05$) in 1992 from the *PF* and *PN* sites compared to the other 3 years. It is interesting to note that even though the soil mineral N content was much higher (an average of 10 µg N g⁻¹ of both NO₃⁻ and NH₄⁺ in 1991 compared to an average of 1 µg N g⁻¹ of both in 1994) in 1991 in the *PF* site than in later years, that N₂O fluxes were not lower in 1994. This observation suggests that it is the N turnover rate that is controlling N₂O emissions coupled with precipitation patterns, rather than the bulk soil mineral N content. In the *PN* site N₂O emissions were higher ($P < 0.05$) in 1993 than in 1991 and 1992 (Fig. 1). The bulk soil NH₄⁺ concentration may not regulate nitrification rates (Davidson and Hackler 1994) since the process may be limited by NH₄⁺ supply on the microsite scale (Davidson et al. 1990). In the years immediately

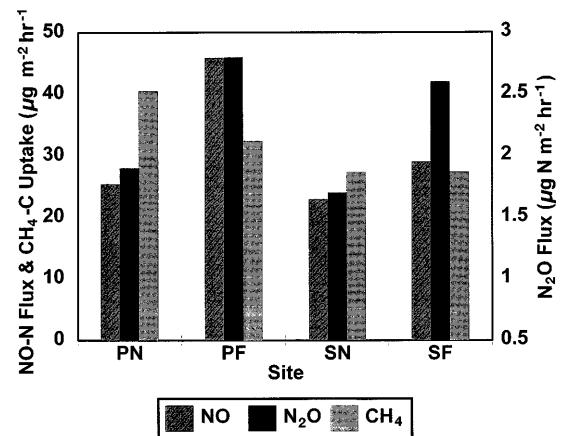


Fig. 2 Mean N₂O emissions and CH₄ uptake from weekly flux measurements taken during January 1995 through June 1996. Mean NO_x fluxes were calculated from weekly fluxes measured over the time period October 1995 through June 1996. See Fig. 1 for site descriptions

following fertilization, when bulk soil NH₄⁺ and NO₃⁻ concentrations were high the NH₄⁺ supply may have exceeded nitrifier demand. Later, when the bulk soil NH₄⁺ content was similar to nonfertilized levels, bulk soil measurements may not have reflected the relative amounts of N turnover between the *PF* and *PN* sites (Schimel et al. 1989; Parton et al. 1996). Annual, in situ N mineralization rates in 1993 were 8.5 g N m⁻² year⁻¹ and 4.3 g N m⁻² year⁻¹ in the top 10 cm of soil at sites *PF* and *PN*, respectively.

To demonstrate the continued long-term effect of N additions to soils within the SGS we used data collected weekly from the *SN*, *SF*, *PN* and *PF* sites throughout 1995 and until the end of June 1996 (Fig. 2). N₂O fluxes averaged higher in the fertilized sites, 1.8 µg N m⁻² h⁻¹ and 2.6 µg N m⁻² h⁻¹ from *SN* and *SF*, respectively, and 1.9 µg N m⁻² h⁻¹ and 2.8 µg N m⁻² h⁻¹ from the *PN* and *PF* sites ($P = 0.05$). These 18-month flux means were similar to the mean fluxes during the previous 54-month period when fluxes averaged 1.9, 2.8, 1.7 and 3.0 µg N m⁻² h⁻¹ from sites *SN*, *SF*, *PN* and *PF*, respectively. These data show, as did the earlier data (Mosier et al. 1991, 1996) that N₂O emissions are significantly enhanced ($P = 0.05$) 14 and 7 years after N addition at these sites.

NO_x

The NO_x flux was measured periodically from June 1994 through September 1995 at the *PN* and *PF* sites to assess the effect of N fertilization about 5 years before measurements began (Martin 1996). Over this 16-month period the mean annual fluxes of NO (calculated by averaging across seasons for each observation location) were 24 µg NO-N m⁻² h⁻¹ and 11.5 µg NO-N m⁻² h⁻¹ for *PF* and *PN* sites, respectively. On an annual basis about 0.21 g N m⁻² year⁻¹ and 0.10 g N

$\text{m}^{-2} \text{ year}^{-1}$ was emitted from these sites. Generally fluxes at both sites followed the same temporal patterns with emissions from the previously fertilized site exceeding those from the native site (Martin 1996).

During the October 1995–June 1996 measurement period (Fig. 2) emissions of NO_x were 10–20 times greater during the 8-month observation period than N_2O emissions. These NO_x fluxes averaged 23, 29, 25 and $46 \mu\text{g N m}^{-2} \text{ h}^{-1}$ from SN, SF, PN and PF sites, respectively. The long-term N addition gave rise to increases in NO_x emissions which were significantly different between the PN and the PF sites ($P=0.07$) but this was not the case when the SN and SF sites were compared ($P=0.30$). These former emissions were about double those observed during the previous year in the PN and PF sites. Timing of winter precipitation in 1995–1996 and the very dry spring of 1996 may explain the larger fluxes. In another study, periodic measurements of NO_x emissions from one site within the SGS were conducted March–July 1988 by Stocker et al. (1993) using micrometeorological eddy correlation, and by Williams and Fehsenfeld (1991) using chambers. Using the information presented in Stocker et al. (1993) we estimated an annual NO emission of $0.11 \text{ g N m}^{-2} \text{ year}^{-1}$ for this unfertilized site. This estimate was made by using their estimates of midsummer flux ($26 \mu\text{g N m}^{-2} \text{ h}^{-1}$ for 30 days), summer flux ($21 \mu\text{g N m}^{-2} \text{ h}^{-1}$ for 90 days), and a flux of $7.9 \mu\text{g N m}^{-2} \text{ h}^{-1}$ for 265 days, and summing the three periods to obtain an annual flux estimate.

CH₄

Stuedler et al. (1989) showed that mineral N additions to forest soils in the northeast U.S. significantly decreased soil consumption of atmospheric CH_4 . Studies with SGS soils showed that CH_4 consumption was decreased by 80–90% with the addition of $25 \mu\text{g NH}_4\text{-N g}^{-1}$ soil and about 30% with the same amount of NO_3^- (Bronson and Mosier 1994). When we returned to the sites that had been fertilized in 1981 we were surprised to find that CH_4 consumption rates were much lower than in unfertilized locations (Mosier et al. 1991).

Catena CH₄

CH_4 uptake was decreased ($P<0.05$) by an average of 35% by the 1981 N fertilization in the MF site but not from the 1982 N fertilization in the SF site (Fig. 1). During 1990–1994, CH_4 uptake averaged $36 \mu\text{g C m}^{-2} \text{ h}^{-1}$ and $23 \mu\text{g C m}^{-2} \text{ h}^{-1}$ in MN and SN sites, respectively.

Pasture CH₄

Fertilization of the PF site 2–6 years before CH_4 uptake measurements were begun decreased CH_4 ($P<0.05$) by

26–37% compared to the adjoining PN site. The 4-year uptake rate average was lower ($P<0.05$), i.e. $24.6 \mu\text{g C m}^{-2} \text{ h}^{-1}$ in the PF compared to $36.2 \mu\text{g C m}^{-2} \text{ h}^{-1}$ in the PN site (Fig. 1). As in the MN site, N application decreased CH_4 uptake in the sandy soil. Also as in the MN and MF sites, the variation between fertilized and nonfertilized site CH_4 uptake rates follow parallel patterns from year to year (Mosier et al. 1996).

Monthly CH_4 uptake rates for PF and PN also followed parallel patterns. Uptake rates in the PN site were always higher than in the PF site. We assumed that this lower CH_4 consumption in the PF site was a result of higher N turnover rates which are shown by the generally higher N_2O flux from the PF site. Since the CPER soils are typically wettest during the winter months, the winter CH_4 uptake rates are proportionately lower than in the spring and summer (Mosier et al. 1996).

The long-term decrease in CH_4 uptake rates due to N addition more than 5 years earlier was observed in the comparison between the PN and PF sites ($P=0.1$). The CH_4 uptake rates averaged $41 \mu\text{g C m}^{-2} \text{ h}^{-1}$ and $32 \mu\text{g C m}^{-2} \text{ h}^{-1}$ in PN and PF sites, respectively (Fig. 2). The $\text{CO}(\text{NH}_2)_2\text{-N}$ added to the SF site in 1982 did not decrease CH_4 uptake when compared to that of the SN site. Mean uptake rates were $27 \mu\text{g C m}^{-2} \text{ h}^{-1}$ at both sites. These uptake rates were compared to 21, 22, 36 and $25 \mu\text{g C m}^{-2} \text{ h}^{-1}$ in SN, SF, PN and PF sites during the 1990 through 1994 measurement period. The lack of an added N effect on CH_4 uptake in the SF site compared to the PF site and the MF site of the catena (Mosier et al. 1991, 1996) is attributed to total N turnover being greater in the SN site compared to the PN site or MN site of the catena, where N added in 1981 inhibited CH_4 uptake by about 35%, 10–15 years later (Mosier et al. 1991, 1996).

Immediate effect of N addition on trace gas exchange

Addition of mineral N to any of the grassland soils was expected to result in an “immediate” increase in N_2O and NO emissions and a decrease in CH_4 uptake. In a 2-day and 12-day laboratory incubation study, addition of $25 \mu\text{g NH}_4\text{Cl g}^{-1}$ dry soil (soils from the MN and SN sites were used), inhibited CH_4 uptake by 88% and 79%, respectively (Bronson and Mosier 1994). Another set of laboratory studies showed that N additions also increased N_2O production in soils from these sites (Parton et al. 1988). As discussed earlier, the initial addition of $\text{CO}(\text{NH}_2)_2$ to the MF and SF sites in 1981 and 1982 led to large increases in the normally small N_2O emissions from the sites during the following 18 months (Mosier and Parton 1985).

Because of the immediate laboratory response of soils from the SGS to $\text{NH}_4^+\text{-N}$ addition in terms of CH_4 oxidation and N gas production, and the long-term effect on trace gas exchange, we expected to see similar responses to N additions in the field.

N_2O

In the sand site mean N_2O emission rates were not increased by addition of 0.5 g N m^{-2} (approximately $9 \mu\text{g N/g}$ of the top 5 cm of soil) of KN, AN or AS (Fig. 3a). In the clay site, soil N_2O emissions were significantly increased ($P=0.1$) in the KN and both AS amended plots (Fig. 3a), but not when AN was added. N_2O emissions were higher in the clay site with emissions ranging from $3.0 \mu\text{g N m}^{-2} \text{ h}^{-1}$ in CONT to $8.7 \mu\text{g N m}^{-2} \text{ h}^{-1}$ in the AS (2) treatment.

 NO_x

In the sand site NO_x emissions averaged $46 \mu\text{g N m}^{-2} \text{ h}^{-1}$ in the CONT and $73 \mu\text{g N m}^{-2} \text{ h}^{-1}$ and $144 \mu\text{g N m}^{-2} \text{ h}^{-1}$ in the AS-amended soils. Although mean NO_x emissions increased to $60 \mu\text{g N m}^{-2} \text{ h}^{-1}$ and $65 \mu\text{g N m}^{-2} \text{ h}^{-1}$ (Fig. 3b) in AN and KN treated plots, these increases were not statistically significant ($P=0.17$ and 0.26 , respectively). In the clay site NO_x emissions were lower than from the sand site. The NO_x emissions from the clay CONT plots averaged $20 \mu\text{g N m}^{-2} \text{ h}^{-1}$ compared to 26 , 22 , 34 and $46 \mu\text{g N m}^{-2} \text{ h}^{-1}$ in KN, AN, AS (0.5) and AS (2) plots, respectively. These NO_x emissions were significantly higher ($P=0.1$) in the KN, AS (0.5) and AS (2) treatments compared to the CONT.

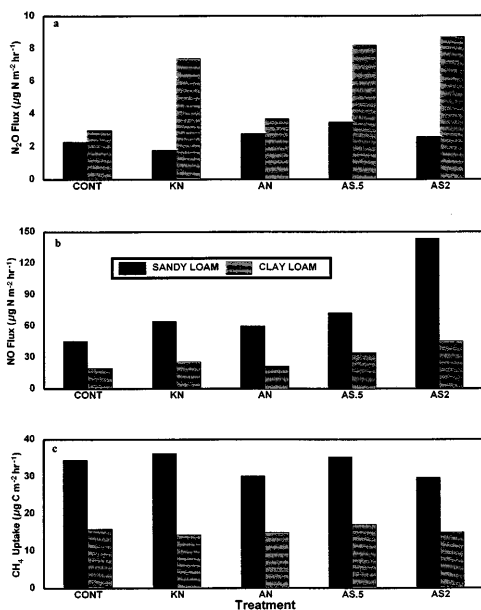


Fig. 3a-c Mean N_2O emission rates measured 11 times during July and August 1996 (a); mean NO_x emission rates from the N treated plots (b); mean CH_4 uptake rates in the N treated plots (c). Treatments include a nonfertilized control (CONT), plots fertilized with 0.5 g N m^{-2} of KNO_3 (KN), plots fertilized with 0.5 g N m^{-2} of NH_4NO_3 (AN), plots fertilized with 0.5 g N m^{-2} of $(NH_4)_2SO_4$ (AS.5) and plots fertilized with 2 g N m^{-2} of $(NH_4)_2SO_4$ (AS2)

 CH_4

As we observed in all of the SGS field studies, CH_4 oxidation rates were much higher in the coarse-textured soils. Uptake rates in the CONT plots averaged $35 \mu\text{g C m}^{-2} \text{ h}^{-1}$ and $16 \mu\text{g C m}^{-2} \text{ h}^{-1}$ in the sand and clay sites, respectively (Fig. 3c). The immediate effect of N additions on CH_4 uptake was smaller than expected. In the sand site, the lowest mean uptake rates were in the AS (2) plots which decreased ($P=0.1$) only 14% compared to the control (Fig. 3c). In the clay site the CH_4 uptake ranged between $14.4 \mu\text{g C m}^{-2} \text{ h}^{-1}$ and $17.1 \mu\text{g C m}^{-2} \text{ h}^{-1}$, with the highest uptake rates observed in the AS (0.5) plots.

The following gives a summary of the results.

1. *Long-term effects of fertilization on trace gas exchange:* weekly soil-atmosphere exchange measurement of N_2O and CH_4 were conducted from spring 1990 through June 1996 at sites within the SGS. The 1990–1994 (Fig. 1) results were reported in Mosier et al. (1991, 1996). The 1995–1996 data (Fig. 2) confirm that large N applications, either in a single dose, simulating cattle urine deposition (45 g N m^{-2}) or in small annual applications over a 15-year period (30 g N m^{-2}) continued to stimulate N_2O emissions from both sandy loam and clay loam soils. These data demonstrate that N fertilization and N redistribution through cattle grazing continued to increase N_2O emission for decades within this semiarid grassland. Even though these N_2O emissions are only $<5\%$ of the annual N deposition they do make an important contribution to grassland N_2O emissions and the global budget of N_2O (Mosier et al. 1996).

The emissions of NO_x from both fertilized and natural SGS soils constitute an important N loss from the system. In natural sites we found that NO_x emissions can contribute to the loss of about 40% of the N deposited annually through wet and dry deposition (Martin 1996). In sandy loam soils last fertilized 6 years earlier, NO_x emissions averaged 60% greater than from the unfertilized site. NO_x emissions from the clay loam site fertilized 13 years earlier averaged about 30% higher than from unfertilized soils. These data clearly show that the enhancement of NO_x emissions following fertilization continues for more than a decade (Fig. 2).

The long-term impact of N addition on CH_4 uptake is soil-dependent (Figs. 1, 2). Fertilization of a sandy loam soil continued to decrease CH_4 uptake 6 and 14 years later. In a clay loam soil no inhibition due to fertilization was observed.

2. *Immediate effects of small N additions on trace gas exchange:* a short-term study was conducted during the summer of 1996 to determine the immediate effect of small N additions on N_2O , NO_x and CH_4 fluxes. Averaged over 11 observation periods, N_2O emissions in the sand site were not significantly increased by addition of 0.5 g N m^{-2} or 2 g N m^{-2} of three different forms of

mineral N. In the clay site NO_2O fluxes were small but significantly higher in the KN, AS (0.5) and AS (2) treatments than in the CONT. These results are similar to those found in the Parton et al. (1988) field study where N_2O emissions were higher in the clay loam soil than in the sandy loam.

Conversely, NO_x fluxes were 10 to 20 times higher than those of N_2O and were much higher in the coarse-textured soil (Fig. 3b). Addition of both 0.5 g N m^{-2} and 2 g N m^{-2} of AS increased the NO_x flux in the sand site. Addition of 0.5 g N m^{-2} of KN or AN increased the average NO_x emissions, but only by a small, not statistically significant, amount. In the clay site soil NO_x emissions were significantly increased by N addition in all but the AN-amended soils.

N fertilization had relatively little short-term effect upon CH_4 uptake in either soil (Fig. 3c). Only in the coarser-textured soil did AS (2) addition decrease CH_4 uptake ($P=0.1$); in this case by an average of only 14%. This is much lower than the 35–40% long-term decrease observed in all of the sandy loam soil sites throughout the study (Mosier et al. 1991, 1996).

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