

Fertilizer Application Timing Influences Greenhouse Gas Fluxes Over a Growing Season

Rebecca L. Phillips,* Donald L. Tanaka, David W. Archer, and Jon D. Hanson USDA-ARS

Microbial production and consumption of greenhouse gases (GHG) is influenced by temperature and nutrients, especially during the first few weeks after agricultural fertilization. The effect of fertilization on GHG fluxes should occur during and shortly after application, yet data indicating how application timing affects both GHG fluxes and crop yields during a growing season are lacking. We designed a replicated ($n = 5$) field experiment to test for the short-term effect of fertilizer application timing on fluxes of methane (CH_4), carbon dioxide (CO_2), and nitrous oxide (N_2O) over a growing season in the northern Great Plains. Each 0.30-ha plot was planted to maize (*Zea mays* L.) and treated similarly with the exception of fertilizer timing: five plots were fertilized with urea in early spring (1 April) and five plots were fertilized with urea in late spring (13 May). We hypothesized time-integrated fluxes over a growing season would be greater for the late-spring treatment, resulting in a greater net GHG flux, as compared to the early-spring treatment. Data collected on 59 dates and integrated over a 5-mo time course indicated CO_2 fluxes were greater ($P < 0.0001$) and CH_4 fluxes were lower ($P < 0.05$) for soils fertilized in late spring. Net GHG flux was also significantly affected by treatment, with 0.84 ± 0.11 kg CO_2 equivalents m^{-2} for early spring and 1.04 ± 0.13 kg CO_2 equivalents m^{-2} for late spring. Nitrous oxide fluxes, however, were similar for both treatments. Results indicate fertilizer application timing influences net GHG emissions in dryland cropping systems.

AGRICULTURAL practices may contribute significantly to the global increase in atmospheric concentrations of CH_4 , CO_2 , and N_2O . Land under cultivation comprises the single largest total land area worldwide (Dutaur and Verchot, 2007), and roughly a third of CH_4 emissions and two-thirds of N_2O emissions come from soils (Prather et al., 2001). Knowledge of how agricultural management influences sources and sinks of CH_4 , CO_2 , and N_2O relative to crop yields is important to understanding potential anthropogenic impacts on climate forcing (Mosier et al., 2006). Agriculturally-related CO_2 emissions include direct sources (e.g., diesel fuel), indirect sources (e.g., fertilizer production), and land use changes (West and Marland, 2002). Many estimates for GHG exchanges at the soil surface are poorly constrained, which limits the potential for agriculture to minimize or manage GHG emissions (Mosier et al., 2006; Dutaur and Verchot, 2007). One of the key factors influencing production and consumption of CH_4 , CO_2 , and N_2O is fertilization of arable soils (Breitenbeck and Bremmer, 1986; Mosier et al., 1996; Chantigny et al., 1998; Bouwman et al., 2002). The effects of fertilization on GHG fluxes at the soil surface tend to occur within the initial 8 to 10 wk following N application (Breitenbeck and Bremmer, 1986; Chantigny et al., 1998; Bouwman et al., 2002; Phillips, 2007), but the magnitude of these effects likely vary with temperature at the time of application. Since microbial respiration is strongly influenced by temperature, microbial response in the northern Great Plains is likely greater for soils fertilized in late spring than in early spring.

While rate of fertilizer application reportedly affects soil emissions of CH_4 , CO_2 , and N_2O in cropping systems (Robertson et al., 2000; Amos et al., 2005; Mosier et al., 2006; Sainju et al., 2008), it is unclear how timing of fertilizer application influences emissions. Since N_2O and CO_2 emissions tend to increase and CH_4 uptake tends to decline during the first few weeks following fertilization, it is generally accepted that GHG emissions tend to increase with additions of N (Chantigny et al., 1998; Smith et al., 2000; Bouwman et al., 2002; Mosier et al., 2006; Sainju et al., 2008). Effects of fertilizer-N addition on fluxes of GHGs, however, are not consistent across studies. Sometimes fertilization of arable soil does not affect the strength of soil as a source of N_2O (Amos et al., 2005) and CO_2 (Tate and Striegl, 1993; Amos et al., 2005) or the strength of the soil as a sink for CH_4 (Tate and Striegl, 1993; Delgado and

Copyright © 2009 by the American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America. All rights reserved. No part of this periodical may be reproduced or transmitted in any form or by any means, electronic or mechanical, including photocopying, recording, or any information storage and retrieval system, without permission in writing from the publisher.

Published in *J. Environ. Qual.* 38:1569–1579 (2009).

doi:10.2134/jeq2008.0483

Received 14 Nov. 2008.

*Corresponding author (Rebecca.phillips@ars.usda.gov).

© ASA, CSSA, SSSA

677 S. Segoe Rd., Madison, WI 53711 USA

USDA-ARS, Northern Great Plains Research Lab., 1701 10th Ave SW, Mandan, ND 58554. The U.S. Department of Agriculture, Agricultural Research Service is an equal opportunity/affirmative action employer and all agency services are available without discrimination. Mention of commercial products and organizations in this manuscript is solely to provide specific information. It does not constitute endorsement by USDA-ARS over other products and organizations not mentioned.

Abbreviations: GHG, greenhouse gases; WFPS, water-filled pore space.

Mosier, 1996; Koga, 2004). These conflicting results compelled us to question what other factors might influence the response of soil microbial consumption and production of GHGs to fertilization, particularly in the northern Great Plains, where rainfall limits crop production. Here, we investigated how timing of fertilizer application affects fluxes of CH₄, CO₂, and N₂O at the soil surface for maize produced in central North Dakota under dryland farming conditions.

During the short growing season in the northern Great Plains, producers often apply fertilizer in early spring to avoid over-scheduling activities near planting and/or to achieve commercial fertilizer price discounts. Soil respiration varies strongly with temperature (Wildung et al., 1975; Wagai et al., 1998; Fang and Moncrieff, 2001; Parkin and Kaspar, 2003), so GHG emissions associated with application of fertilizer during the cold early spring (air temperatures <10°C) should be lower than later in the season. Fertilizing earlier in the spring, however, could affect crop yields if nutrients are less available following prolonged periods in the soil without plant growth. Currently, it is not clear if fertilizer application timing affects net GHG flux for arable soils cropped under dryland conditions in the northern Great Plains and if early fertilization would affect yield. With the rising cost of synthetic fertilizers and public concern about anthropogenic GHG emissions, there is a need for field studies that quantify effects of fertilizer application timing on GHG emissions in a crop-production setting.

We initiated a study to examine how timing of fertilization application (early spring vs. late spring) influences fluxes of CH₄, CO₂, N₂O, and net GHG for soils planted to maize in a North Dakota dryland cropping system, where net GHG flux was calculated according to the radiative forcing potential of CH₄ and N₂O relative to CO₂ (Ramaswamy et al., 2001; IPCC, 2007). We hypothesized net GHG flux for soils fertilized in early spring would be lower than for soils fertilized in late spring, since cool temperatures would limit soil microbial activity (Wagai et al., 1998; Fang and Moncrieff, 2001; Parkin and Kaspar, 2003). We also hypothesized that grain yield would be greater for soils fertilized just before planting in late spring. Our primary objective was to determine if time-integrated net GHG flux for soils fertilized in early spring were significantly different from those soils fertilized in late spring for a dryland cropping system.

Materials and Methods

Study Area

A fertilizer timing experiment was initiated on 25 Mar. 2008 at the USDA-ARS Northern Great Plains Research Laboratory (NGPRL) in south-central North Dakota near the city of Mandan (46°46' N, 100°55' W). Climate is semiarid with mean annual temperature of 10°C and mean annual precipitation of 41.2 cm (mean from 1913–2006). Mean minimum and maximum temperatures from April to September are 8.4 and 22.5°C, respectively (Mandan Experiment Station, ND). During the flux measurement period, between 1 April and 26 Aug. 2008, the area received 26.0 cm of rainfall (Fig. 1). Maize, wheat (*Triticum aestivum* L.), and barley (*Hordeum vulgare*

L.) are major crops in local agriculture. The soil is classified as a Temvik-Wilton silt loam [FAO: Calic Siltic Chernozems; USDA: Fine-silty, mixed, superactive, frigid Typic and Pachic Haplustolls (Soil Survey Staff, 2008)].

Experimental Design and Crop Management

A completely randomized experiment with five replicates was established with two fertilizer application dates in 2008 (early spring, 1 April; and late spring, 12 May) for soils cropped annually and under no-till management since 1992 (Fig. 2). Each 0.30-ha plot was fertilized with a dry mixture of urea, phosphorous pentoxide (P₂O₅), and potash (K₂O) at a rate of 70–40–12 kg ha⁻¹ N-P-K with a metered-feed fertilizer spreader (Barber Engineering Co., Spokane WA). The 3-m broadcast spreader delivered fertilizer in three passes to each plot (9.1 m wide by 330 m long). Care was taken to ensure fertilizer was spread within the treatment plot only, and a 10-m buffer was established around the experiment border so as to avoid chemical inputs from surrounding agricultural fields (Fig. 2). Both treatments received 0.25 cm of precipitation within 4 d of urea application (Fig. 1). A neighboring grassland site (uncultivated and unfertilized) was selected as background reference. This reference site is densely covered by introduced grasses [*Bromus tectorum* (L.) and *Poa pretensis* (L.)], and has not been hayed or grazed since 2005.

Agronomic inputs were based on agronomic product applications recommended for production of 4300 kg ha⁻¹ of maize in a dryland cropping system. Besides fertilizer, herbicide was applied twice (pre- and post-planting) for weed control, resulting in plots being essentially weed-free. Maize hybrid Legend Seed Brand 9584 was planted with a 76-cm row spacing on 15 May 2008. Planting rate was approximately 84,000 seeds ha⁻¹. Both early and late-spring fertilized plots were seeded at the same time and at the same rate with a John Deere 7340 Maxemerge 2 Vacumeter, for a total of 12 rows per plot. Before initiation of the experiment in March 2008, fertilizer had not been applied for 1 yr. In 2007, the area was planted to sunflower (*Helianthus annuus* L.) and fertilized in early spring with 67 kg N ha⁻¹ as urea. In 2006, the area was planted to spring wheat (*Triticum aestivum* L.), and fertilized in early spring with 78 kg N ha⁻¹ as urea and 6–29–0 kg ha⁻¹ of N-P-K as monoammonium phosphate. In 2005, the area was planted to soybean (*Glycine max* L.) and fertilized in early spring with 6–29–0 kg ha⁻¹ of N-P-K as monoammonium phosphate.

A planting map (Fig. 2) was created using a submeter, real-time differential Trimble Geo XT Global Positioning System (GPS) Beacon receiver (Trimble Navigation, Sunnyvale, CA) and a 2004 ortho-rectified, high resolution (2.8 m²) satellite image (Digital Globe, Inc., Longmont, CO) to ensure plot areas (0.30 ha) were equal and to identify row locations. Further, one smaller subplot (0.05 ha in area) was also mapped within each 0.30-ha plot. While fertilizer treatment covered the entire plot, soil and gas fluxes were sampled only from the subplot areas to ensure that a large number of samples could be collected during a 0.75 h time-course. Three points were randomly selected for soil and gas flux sampling within each subplot using the randomization procedure in ERDAS Imagine 9.1 software

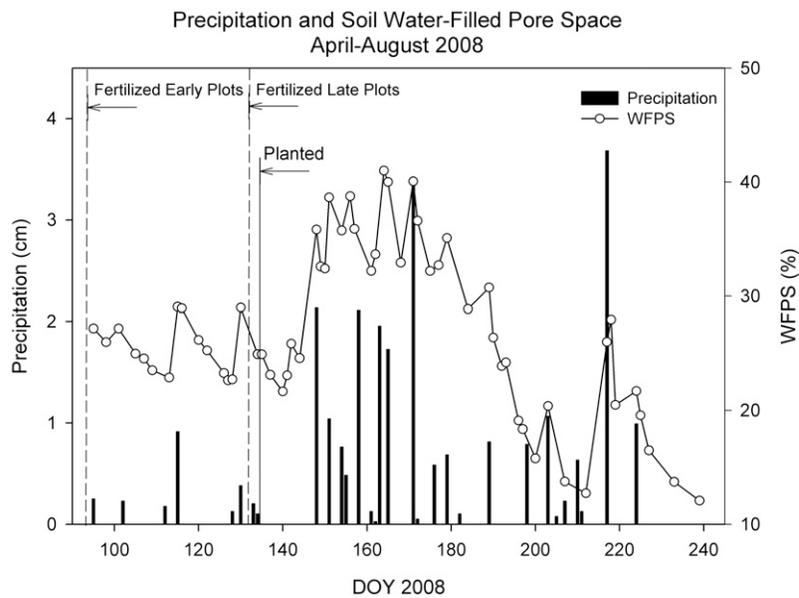


Fig. 1. Precipitation received during the study period and average ($n = 30$) soil water-filled pore space (%) determined each sample date at the 0- to 15-cm soil depth interval.

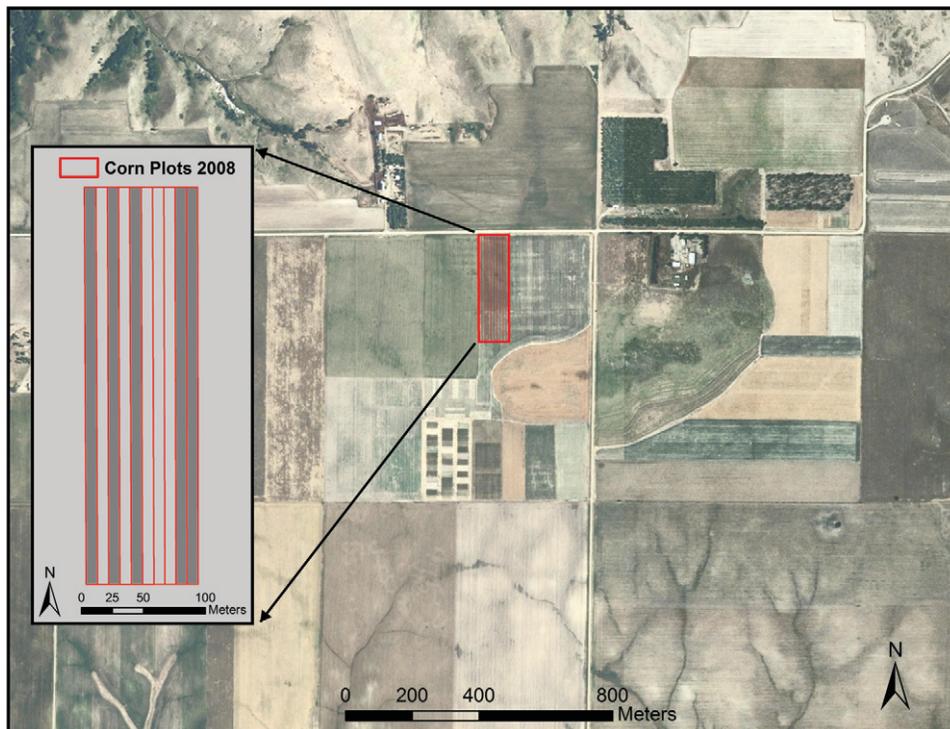


Fig. 2. Map of the study area and experimental plots (expanded for clarity), where treatments ($n = 5$) were randomly assigned. Early-spring plots are filled in a dark gray color and late-spring plots are filled in with a light gray color.

(Leica Geosystems GIS and Mapping LLC, Norcross, GA). Polyvinyl chloride collars (20-cm diam. \times 10-cm height) were oriented between maize rows according to the planting map in areas that were not trafficked by farm equipment. Collars were deployed as soon as soil conditions allowed insertion into the soil, which was 7 d before the first flux measurement. Collars remained in the soil for the duration of the experiment (fertilization, planting, harvesting). Collars were also deployed at five

random locations within a 0.05-ha reference site where plants inside the collars were clipped to ground level.

Aboveground maize production in shoot dry biomass was determined by clipping whole maize plants to ground level from a 7.5-m² area within each plot before harvest (Mosier et al., 2006). The plants were separated into cobs and shoots for mass determinations. We also determined grain yields by plot with a combine and weigh-wagon for each 0.30-ha plot on 30

October. Final grain yield was expressed at 155 g kg⁻¹ water content (Mosier et al., 2006). Emissions associated with energy used by farm equipment was calculated based on estimated diesel fuel use for each field operation (Lazarus, 2008) with published coefficients for energy content and emissions from diesel fuel use, manufacture, and transport (EIA, 1999; West and Marland, 2002). Emissions associated with fertilizer and herbicide use, were based on the quantities applied and using published coefficients for emissions from manufacture, packaging, and transportation (West and Marland, 2002; Lal, 2004).

Soil Measurements

Soil pH, electrical conductivity (EC), texture, bulk density, total C and N, and water holding capacity were measured using soil cores collected for each plot just before applying treatments with a hydraulic soil corer (5 by 15 cm) equipped with a clear plastic sleeve, to ensure the entire core volume would be retained in the sample (Table 1). Soil pH and EC were measured potentiometrically on 1:2 soil/deionized water slurries. Soil texture was determined hydrometrically and bulk density computed as the quotient of oven-dried mass divided by core volume. Total soil C and N were measured by dry combustion using a Carlo Erba NA 1500 Elemental Analyzer (Haake Buchler Instruments, Inc., Saddle Brook, NJ). Water holding capacity was calculated as the mass of water retained after 1 hr on a Whatman no. 1 filter per mass of oven-dried soil.

At each gas flux sampling date, air temperature near the soil surface and soil temperature at 10-cm depth were measured with a Type K temperature probe. Soil cores were also collected with a hand probe on each sampling date (3-cm diam. × 15-cm depth) within a 1-m radius of each chamber. Each individual core was analyzed for mineral N (ammonium and nitrate) by Cd reduction using a continuous flow analyzer (Lachat Quick-Chem FIA 8000 Series, Lachat Instruments, Loveland, CO) after extraction with 1 mol L⁻¹ KCl (soil to solution ratio = 1.5) on selected dates. Mineral N was analyzed 1-wk before and 1-, 4-, and 9-wks after fertilization for each treatment. Soil moisture content of the each collected core was determined gravimetrically (oven-dried at 105°C). Percentage water-filled pore space (%WFPS) was then calculated as the ratio of volumetric soil water content to total soil porosity. To determine if %WFPS varied between early and late-spring fertilized soils, we tested for effect of treatment on %WFPS with the mixed, repeated measures model described above (Phillips et al., 2001).

Methane, Carbon Dioxide, and Nitrous Oxide Flux Measurements

Measurements of the soil-atmosphere exchange of CH₄, CO₂, and N₂O began 4 April, 3 d following fertilization of the early-spring plots. Maize plot measurements were made one to four times per week from 4 April to 26 August between 1000 and 1200 h, which is the time period most representative of daily fluxes for these soils (Frank et al., 2002; M. Liebig, unpublished data, 2004). Reference plot measurements were collected similarly from 3 June to 13 August. Polyvinyl chloride covers fitted with butyl O-rings

were placed onto the soil collars during each gas flux measurement. Covers included a capillary bleed to equalize pressure and an O-seal fitting and septa for syringe sampling. Samples of the head-space gas were withdrawn from each chamber at 0.25-h intervals during a 0.75-h time course. Sample aliquots were immediately injected into 12-mL exetainers (Labco Unlimited, Buckinghamshire, UK) and analyzed for CH₄, CO₂, and N₂O with a Varian Model 3800 Gas Chromatograph and Combi-Pal auto-sampler. In this system, sample is auto-injected into a 1-mL sample loop, then loaded onto columns and routed through three detectors: a ⁶³Ni electroncapture detector (ultra-pure 95% Argon/5% CH₄ carrier gas), a thermal conductivity detector (ultra-pure He carrier gas), and a flame ionization detector (ultra-pure He carrier gas). The gas chromatograph was calibrated with commercial blends of CH₄, CO₂, and N₂O balanced in N₂ (Scott Specialty Gases, Trenton, NJ) following verification of stated concentrations with standards from the National Institute of Standards and Technology. The precision of GC analysis, expressed as a coefficient of variation for 10 replicate injections of a low concentration standard (2.0 μL L⁻¹ for CH₄, 369.7 μL L⁻¹ for CO₂, and 363.7 nL L⁻¹ N₂O) and a high concentration standard (10.0 μL L⁻¹ for CH₄, 1748.2 μL L⁻¹ for CO₂, and 1682.1 nL L⁻¹ N₂O) was consistently <2% for all three gases. All field measurements were within the linear ranges for each detector. The minimum detectable concentration change was ± 50 nL L⁻¹ for CH₄, 10 μL L⁻¹ for CO₂, and 7 nL L⁻¹ for N₂O. Typical concentration changes over the time course were 400 nL L⁻¹ for CH₄, 500 μL L⁻¹ for CO₂, and 30 nL L⁻¹ for N₂O. Fluxes were calculated from chamber geometry and linear changes in concentration with time (Whalen and Reeburgh, 1992). Slopes were not linear when CH₄ and N₂O concentration changes vacillated near their respective detection limits; these fluxes were lower than our ability to detect and considered to be no different from zero (Whalen and Reeburgh, 1992; Phillips et al., 2001; Phillips, 2007). Slopes for CO₂ were consistently linear. Approximately 3% of the data collected were missing due to instrument malfunction or sample handling/transfer errors. Gas samples were stored <8 h before analysis and tests showed no change in gas concentration during storage. A positive flux was defined as net emission to the atmosphere (source) and a negative flux was defined as consumption (sink) by the soil microbial community.

Methane and N₂O fluxes were multiplied by 23 and 296, respectively, to express these in units of CO₂ equivalents (based on radiative forcing potentials over a 100-yr time horizon) and to calculate net GHG fluxes (IPCC, 2007). Seasonal fluxes from each of the 30 field sites were calculated by time integration of the daily fluxes (CH₄, CO₂, N₂O, and net GHG) from each chamber. Time-integrated flux was numerically approximated with the midpoint rule (Berkey and Blanchard, 1992) using the set of flux observations over time by chamber (Phillips et al., 2001). We also calculated cumulative flux for each treatment by taking the average flux ($n = 15$) on the first observation day and adding this to the mean for the next observation day. We tested for significant differences in time-integrated flux between treatments for each gas with a mixed analysis of variance (Littell et al., 1996). A nested hierarchical model was used with chambers nested inside plots and treatments (Phillips et al., 2001). Further,

Table 1. Soil properties measured at 0 to 15 cm (mean ± SD) in March 2008 before initial spring fertilization (n = 15).

Treatment	Soil bulk density	Total soil C	Total soil N	WHC†	EC‡	Sand	Silt	Clay	pH
	Mg m ⁻³	g kg ⁻¹			dS m ⁻¹	g kg ⁻¹			
Early-spring fertilization	1.26 (0.07)§	24.8 (2.9)§	2.3 (0.2)§	643 (50)§	0.87 (0.41)§	364 (19)§	420 (19)§	216 (14)§	5.69 (0.18)§
Late-spring fertilization	1.25 (0.06)§	23.4 (3.8)§	2.2 (0.3)§	626 (30)§	0.85 (0.52)§	371 (27)§	406 (27)§	223 (21)§	5.74 (0.16)§
Grassland reference	1.20 (0.03)§	42.2 (7.8)§	3.6 (0.6)§	669 (56)§	0.78 (0.04)§	374 (49)§	497 (49)§	129 (39)§	6.22 (0.11)§

† Water holding capacity.

‡ Electrical conductivity.

§ Values in parentheses represent standard deviation.

we used a mixed, repeated measures model to analyze which variables (air temperature, soil temperature, %WFPS, treatment, or time) influenced fluxes of CH₄, CO₂, N₂O, and total net GHG. We included a time series covariance structure in the repeated measures model, where correlations decline over time (Phillips et al., 2001). Nitrous oxide fluxes were not normally distributed, so we used the general linear mixed model to fit a log-normally distributed response for N₂O. The diagnostics of the residuals indicated normality for N₂O was achieved with a log-normal fit distribution. Level of significance was set at $\alpha = 0.05$.

Results

Crop Yields and Farm Operations

Mean (± SD) grain yields were 2528 ± 334 kg ha⁻¹ and 2656 ± 550 kg ha⁻¹ for the early and late-spring treatments, respectively. Mean phytomass was 4890 ± 621 kg ha⁻¹ for the early spring and 4880 ± 547 kg ha⁻¹ for the late-spring treatments, or 1956 ± 248 kg C ha⁻¹ and 1952 ± 219 kg C ha⁻¹, respectively. The coefficients of variation for aboveground phytomass values were 13% and 11% for early and late-spring treatments, respectively. Similar values for grain yield and aboveground phytomass suggest these variables were not affected by date of urea application. Carbon emitted from farm operations and fertilizer production was estimated at 148 kg C ha⁻¹.

Soil Measurements

Soils cores collected before the experiment for both early and late-spring fertilized plots were similar with respect to texture, WHC, C, N, bulk density, pH, and EC (Table 1). There was no significant difference in %WFPS ($F_{1,28} = 0.21$; $P = 0.58$) between treatments during the time-course of the experiment (April–August), with an overall mean for all soils of 26.4 ± 8.1. Percentage WFPS did not exceed 30% until 27 May (Day 148), where it remained above 30% through 27 June (Day 179). After 27 June, soil WFPS remained below 30% (Fig. 2). Soil WFPS exceeded 40% on 3 d (12 June, 13 June and 19 June). Soil temperature ranged from 1°C on 7 April to 26°C on 18 July (Fig. 3). Soil temperature was at or below 15°C from 4 April (Day 95) until 16 June (Day 168). From 19 June (Day 171) to 26 August (Day 233) soil temperature was >15°C. Mean soil NO₃⁻-N and NH₄⁺-N levels were similar for both treatments 1 wk before fertilization and 9 wk following fertilization (Table 2). Soil NO₃⁻-N and NH₄⁺-N were <12 µg g⁻¹ and 39 µg g⁻¹, respectively, for all sampling dates. Highest mean soil NH₄⁺-N was found 1 wk following fertilization for both early and late-spring plots. Soil NH₄⁺-N

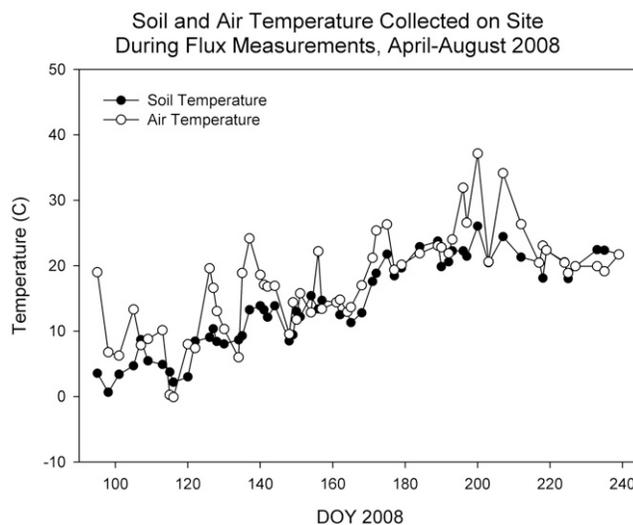


Fig. 3. Mean soil and air temperature data collected at the maize field site on each flux sample date. Sampling was performed between 1000 and 1200 h two to three times per week from April through August 2008. Temperatures were recorded at the beginning and end of the flux measurement time course.

reached 22.4 µg g⁻¹ 1 wk following early-spring fertilization and 38.7 µg g⁻¹ 1-wk following late-spring fertilization. Spatial variability of soil NO₃⁻-N and NH₄⁺-N 1 wk following fertilization was greater than at any other time point, with highest standard deviations for both treatments (Table 2).

Gas Flux Measurements

Of the three gases contributing to the net GHG flux between 4 April and 26 August, CH₄ and CO₂ were significantly affected by fertilization timing. Time-integrated CH₄ flux was significantly greater ($F = 7.01$, $P < 0.05$) for soils fertilized in late spring (-116.58 ± 24.81 mg CH₄ m⁻²) than for soils fertilized in early-spring (-91.65 ± 26.73 mg CH₄ m⁻²). Time-integrated CO₂ flux was significantly greater ($F = 22.96$, $P < 0.0001$) for soil fertilized in late spring (1.03 ± 0.12 kg CO₂ m⁻²) than for soils fertilized in early spring (0.83 ± 0.11 kg CO₂ m⁻²). Time-integrated N₂O flux, on the other hand, was not significantly affected by fertilization timing ($F = 2.09$, $P = 0.15$). Mean flux was 55.10 ± 30.89 mg N₂O m⁻² for early spring and 69.22 ± 21.74 mg N₂O m⁻² for late spring. The time-integrated net GHG flux for soil fertilized in late spring was significantly greater ($F = 20.00$, $P < 0.0001$) than the time-integrated net GHG flux for soil fertilized in early spring. Mean time-integrated net GHG flux for the 15 sites fertilized in early spring was 0.84 ± 0.11 kg CO₂ equivalents m⁻². Mean time-integrated

Table 2. Mean (\pm SD) soil nitrate and ammonium concentrations at 0 to 15 cm during pre- and postfertilization ($n = 15$).

Treatment (DOY)	Prefertilization		1 wk Postfertilization		4 wks Postfertilization		9 wks Postfertilization	
	Early spring (85)	Late spring (128)	Early spring (101)	Late spring (144)	Early spring (128)	Late spring (168)	Early spring (168)	Late spring (203)
NO ₃ -N, $\mu\text{g g}^{-1}$	5.7 (2.5)†	7.7 (4.8)†	4.9 (1.7)†	6.1 (3.3)†	7.0 (2.4)†	12.2 (6.6)†	11.8 (4.8)†	11.8 (1.8)†
NH ₄ -N, $\mu\text{g g}^{-1}$	15.0 (2.0)†	13.3 (2.2)†	22.4 (14.0)†	38.1 (22.7)†	20.8 (9.6)†	12.8 (3.7)†	15.5 2.8†	13.2 (1.4)†

† Values in parentheses represent standard deviation.

net GHG flux for the 15 sites fertilized in late spring was 1.04 ± 0.13 kg CO₂ equivalents m⁻². The coefficients of variation associated with integrating net GHG flux for 15 chambers per treatment over 59 time points were 14% and 13% for early and late spring treatments. The percentage of N emitted as N₂O relative to fertilizer input (70 kg ha⁻¹) during the 20-wk measurement period was 0.63%.

Differences between early and late-spring fertilized soils for cumulative CH₄, CO₂, and N₂O were evident following the late-spring treatment (Fig. 4–6). During the 6-wks following early-spring fertilization, there were no differences between soil fertilized 1 April and soils fertilized the previous year with respect to CH₄ (Fig. 4), CO₂ (Fig. 5), and N₂O fluxes (Fig. 6). Within a week following late-spring application, however, fluxes of CO₂ were more positive (Fig. 5) and fluxes of CH₄ (Fig. 4) were more negative. Differences between early and late-spring treatments were also more pronounced during the time period when %WFPS exceeded 30%, from Day 148 to 179 (Fig. 2). Divergence between treatments only occurred for a few days for N₂O (Fig. 6), but persisted for nearly 4 wk for CH₄ (Fig. 4) and CO₂ (Fig. 5). Cumulative net GHG fluxes plotted over the growing season tracked cumulative CO₂ fluxes (Fig. 7). Carbon dioxide fluxes were at least three orders of magnitude greater than CH₄ or N₂O (Table 3).

The test for effects of %WFPS, air temperature, soil temperature, fertilization application date, and time during the growing

season on repeated measurements of CH₄ flux indicated CH₄ was significantly influenced by %WFPS ($F = 129.42, P < 0.0001$), air temperature ($F = 19.02, P < 0.0001$), fertilization date ($F = 5.32, P < 0.05$), and time ($F = 18.13, P < 0.0001$). Higher values for soil %WFPS, air temperature, and time all resulted in more positive CH₄ fluxes (less uptake). Late-spring fertilization resulted in more negative CH₄ fluxes (greater uptake). For CO₂, higher values for soil %WFPS ($F = 76.08, P < 0.0001$), air temperature ($F = 25.37, P < 0.0001$), and time ($F = 29.14, P < 0.0001$) resulted in more positive CO₂ fluxes, as well as late-spring fertilization ($F = 22.52, P < 0.0001$). Nitrous oxide was not affected by late-spring fertilization ($F = 2.84, P = 0.103$); however, higher values for soil %WFPS ($F = 473.38, P < 0.0001$) and air temperature ($F = 11.75, P < 0.001$) resulted in more positive N₂O fluxes. A comparison of N₂O flux and %WFPS graphs (Fig. 2 and 6) suggests %WFPS contributed to the sharp rise in N₂O emissions between Days 148 and 179. For net GHG flux, higher values for soil %WFPS ($F = 94.81, P < 0.0001$), air temperature ($F = 27.05, P < 0.0001$), and time ($F = 29.92, P < 0.0001$) resulted in more positive net GHG fluxes, in addition to late-spring fertilization ($F = 20.19, P < 0.0001$).

Reference site data were collected to provide background information using unfertilized, undisturbed prairie (Table 1). There was a tendency toward greater CO₂ fluxes for prairie reference, where grass roots and crowns were actively respiring, compared to arable bare soil. Mean daily values for %WFPS

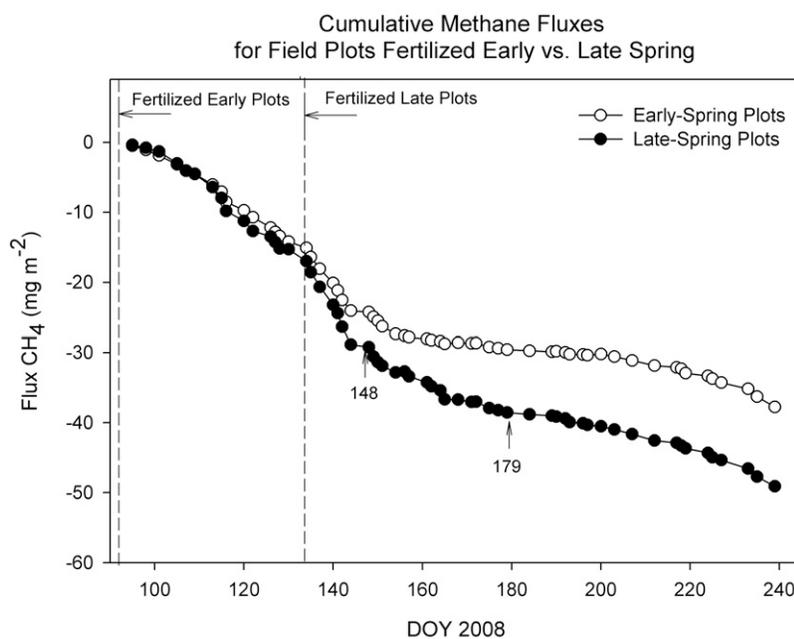


Fig. 4. Cumulative CH₄ flux by treatment between 4 Apr. and 26 Aug. 2008. Mean flux on 4 April was added to subsequent mean flux collected on each sample date by treatment.

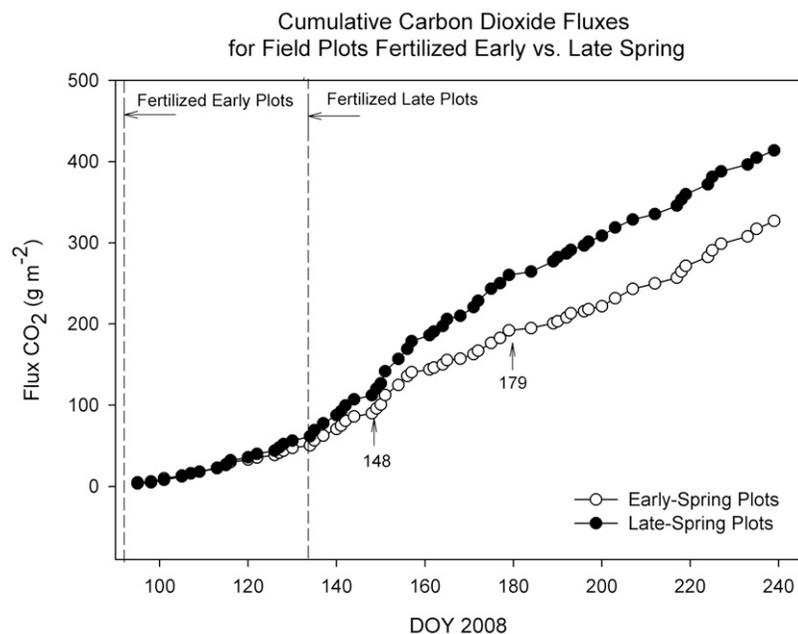


Fig. 5. Cumulative CO₂ flux by treatment between 4 Apr. and 26 Aug. 2008. Mean flux on 4 April was added to subsequent mean flux collected on each sample date by treatment.

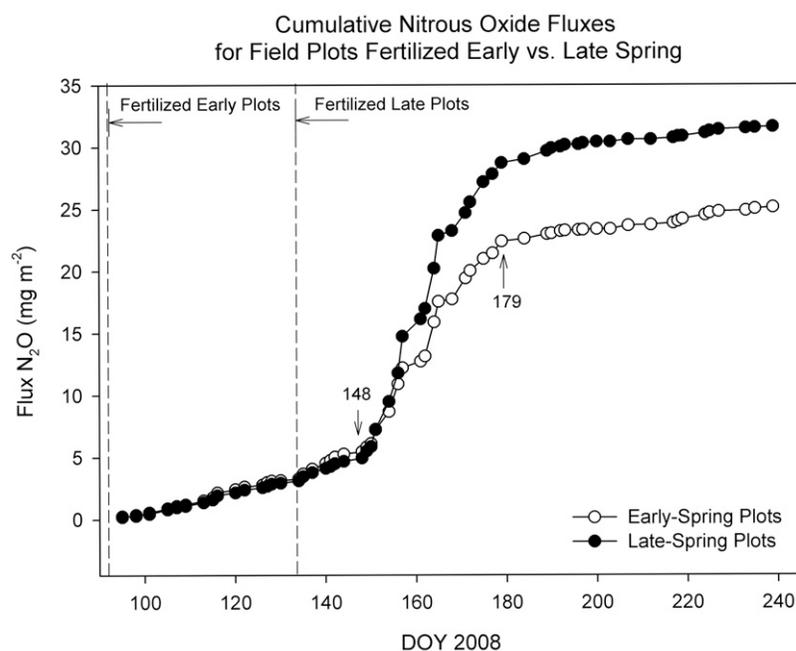


Fig. 6. Cumulative N₂O flux by treatment between 4 Apr. and 26 Aug. 2008. Mean flux on 4 April was added to subsequent mean flux collected on each sample date by treatment.

and soil temperature from 3 June to 13 August was 29.6 ± 1.0 and 18°C , respectively. During the same time period, mean [\pm standard error of the mean (SEM)] CH₄ flux for reference soil was $-23.63 \pm 3.00 \mu\text{g CH}_4 \text{ m}^{-2} \text{ h}^{-1}$, while mean flux for arable soil was $-17.39 \pm 1.57 \mu\text{g CH}_4 \text{ m}^{-2} \text{ h}^{-1}$. Mean CO₂ flux for reference was $348.23 \pm 16.95 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$, while mean flux for arable soil was $298.65 \pm 12.30 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$. Mean N₂O flux for reference was $23.23 \pm 5.64 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$, while mean flux for arable soil was $25.02 \pm 3.39 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$. Mean net GHG flux for reference was $354.56 \pm 17.42 \text{ mg CO}_2$

equivalents $\text{m}^{-2} \text{ h}^{-1}$, while mean flux for arable soil was $296.07 \pm 12.74 \text{ mg CO}_2 \text{ equivalents m}^{-2} \text{ h}^{-1}$ (Fig. 8).

Discussion

The mean time-integrated net GHG flux for soils fertilized in early spring was $548.43 \text{ kg C ha}^{-1}$ less than soils fertilized in late spring. More than 98% of net GHG flux was due to soil CO₂ emissions, with minor consumption and production of CH₄ and N₂O for soil fertilized in late spring. Carbon dioxide

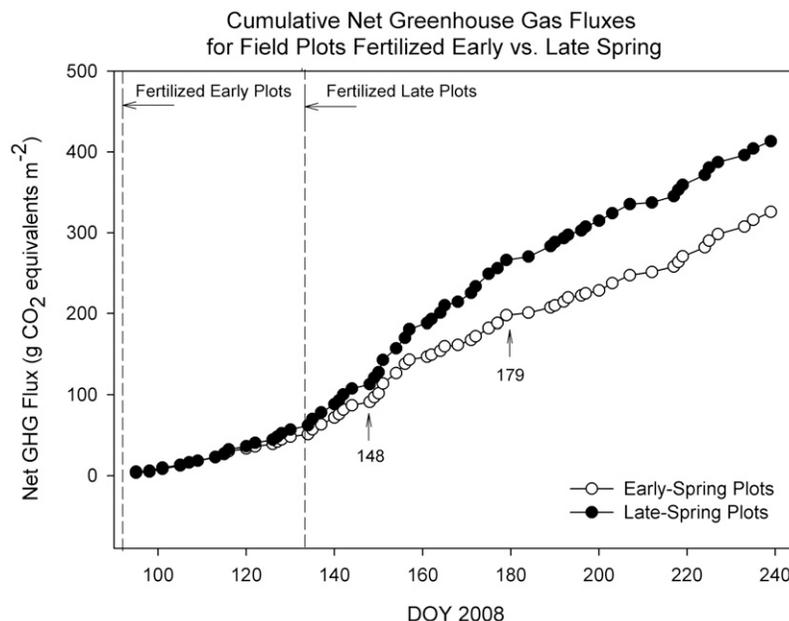


Fig. 7. Cumulative greenhouse gas (GHG) flux by treatment between 4 Apr. and 26 Aug. 2008. Mean flux on 4 April was added to subsequent mean flux collected on each sample date by treatment.

Table 3. Mean seasonal fluxes of CH_4 , CO_2 , N_2O , and net greenhouse gas (GHG) observed from 1 April to 28 August for arable soil planted to maize. Lowest data flux and highest daily flux observed for each gas highlight differences between each gas.

Greenhouse gas	Lowest daily mean	Date observed	Highest daily mean	Date observed	Seasonal mean	
					Early spring	Late spring
Methane, $\text{mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$	-2.50	2 July	-0.14	19 May	-0.64	-0.83
Carbon dioxide, $\text{g CO}_2 \text{ m}^{-2} \text{ d}^{-1}$	1.10	7 April	15.16	2 July	5.53	7.02
Nitrous oxide, $\text{mg N}_2\text{O} \text{ m}^{-2} \text{ d}^{-1}$	0.03	6 August	2.96	5 July	0.42	0.54
Net GHG flux, $\text{g CO}_2 \text{ equivalent m}^{-2} \text{ d}^{-1}$	1.10	7 April	14.14	2 July	5.52	7.00

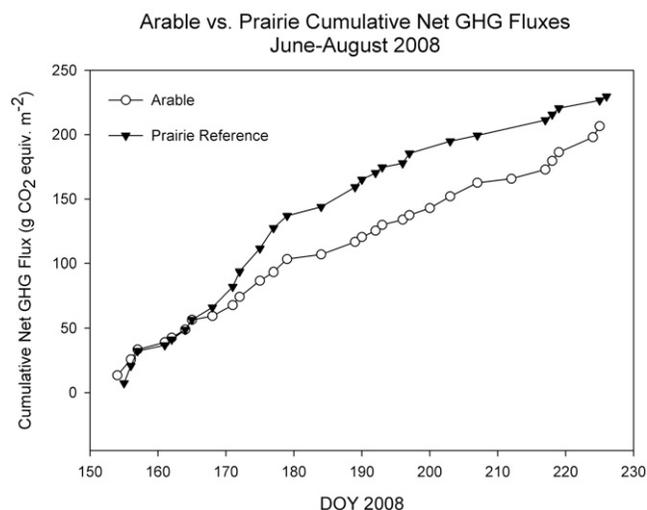


Fig. 8. Cumulative greenhouse gas (GHG) flux for all arable chamber sites and for all prairie references chamber sites between 3 June and 13 Aug. 2008. Mean flux 3 June was added to subsequent mean flux on each sample date for arable versus prairie soils.

emission from soils to the atmosphere is the primary mechanism of soil C loss (Raich and Schlesinger, 1992; Parkin and Kaspar, 2003). Fertilization application in late spring, when mean air temperature rose from $<10^\circ\text{C}$ in the early spring to

over 15°C in the late spring, likely stimulated soil respiration (Fig. 5). While temperature tends to be the strongest predictor of CO_2 flux (Wildung et al., 1975; Wagai et al., 1998; Fang and Moncrieff, 2001; Parkin and Kaspar, 2003), higher %WFPS following late-spring fertilization and greater inorganic N availability (Table 2) likely contributed to greater net GHG flux for the late-spring treatment. One week following urea application, inorganic N for early spring was lower than for late spring (Table 2), suggesting soil N retention was greater following urea application in the late spring. Both enhanced inorganic N availability and higher %WFPS following late-spring fertilization likely contributed to greater soil organic mineralization and higher rates of microbial respiration (Fig. 5). We found no evidence of treatment differences in aboveground phytomass or grain yield, so it is not clear if maize planted only a week earlier was influenced by greater N availability.

Net GHG flux during the growing season was dominated by CO_2 emissions, but this proportion changes with rate of N fertilization (Amos et al., 2005; Mosier et al., 2006) and with season (Flessa et al., 1995; Phillips 2007). Nitrous oxide contributes more to net GHG flux in arable soils at higher rates of N fertilization (Chantigny et al., 1998) and during the nonvegetated period (Flessa et al., 1995). The net loss of N_2O to the atmosphere as a percentage of fertilizer N, however, tends to be $<1\%$ at high, medium, and low rates (Mosier et al., 2006).

Treatments diverged substantively between Days 148 and 179, when %WFPS was >30% (Fig. 4–7). To assess how greater moisture in the early spring may have influenced the treatment effect, we simulated higher %WFPS in the early spring to match %WFPS in the late spring using parameter estimates in the statistical model. If mean %WFPS during April and May was 35% instead of 25% (Fig. 2), the difference between treatments for time-integrated net GHG flux would have narrowed from 548.43 to 447.94 kg C ha⁻¹. This simulation indicated the effect of soil moisture on microbial respiration was greater in the late spring, when soil temperatures were >10°C (as compared to early spring) which corroborated findings by Fang and Moncrieff (2001) and Wildung et al. (1975).

In most cases (93% of the time) negative CH₄ fluxes were observed, indicating net CH₄ uptake at the soil surface. Methane tends to be a minor source or a minor sink in agricultural ecosystems, contributing <1% to net GHG fluxes in the northern Great Plains (Mosier et al., 2006). The overall strength, however, of the soil CH₄ sink was on the high end for soils under cultivation (Dutaur and Verchot, 2007). Dutaur and Verchot (2007) estimated mean CH₄ flux for cropped soils was 1.23 kg ha⁻¹ yr⁻¹ while mean CH₄ flux for grassland soils was 2.32 kg ha yr⁻¹. Our rates of CH₄ uptake more closely approximate grasslands than crops, as also reported by Koga (2004). Rates of CH₄ uptake for soils under no-till management were twice as high as those under conventional tillage, totaling 2.4 kg ha⁻¹ yr⁻¹ (Koga, 2004). Similarly, Tate and Striegl (1993) found the most negative CH₄ fluxes for soils planted to wheat (–3.0 mg m⁻² d⁻¹), compared to prairie (–1.5 mg m⁻² d⁻¹). Like Tate and Striegl (1993), native prairie soil CH₄ uptake at our site was not greater than cropped soil. Seasonal mean CH₄ uptake rates (Table 3; –0.64 to –0.83 mg m⁻² d⁻¹) were comparable to mean values observed by others. Tate and Striegl (1993) reported rates of –0.65 to –1.05 mg m⁻² d⁻¹ for wheat, Kammann et al. (2001) reported rates of –0.60 to –0.84 mg m⁻² d⁻¹ for grasslands, Mosier et al. (1996) reported rates of –0.36 to –0.62 mg m⁻² d⁻¹ for prairie, and Liebig et al. (2006) reported rates of –0.29 to –0.72 mg m⁻² d⁻¹ for prairie.

The small but significant stimulation of CH₄ uptake (Fig. 4) by soils fertilized in late spring is consistent with concurrent stimulation of CO₂ emissions for late-spring plots (Fig. 5). Both suggest application of urea at higher soil and air temperatures generally stimulated microbial activity. The divergence in mean CH₄ fluxes between treatments (Fig. 4) was also closely coupled to greater mean N₂O fluxes (Fig. 6). Near the end of May, greater N₂O emissions and higher soil ammonium concentrations (Table 2) were likely due to greater rates of nitrification. In late spring, soil temperature and %WFPS were nearly optimal for nitrification (Avrahami et al., 2002) and the potential co-oxidation of CH₄ by nitrifying organisms (Jones and Morita, 1983). Alternatively, methane oxidation could have been N-limited, so N addition could have stimulated CH₄ oxidation (Bodelier and Laanbroek, 2004). While N limitation for high-affinity CH₄ oxidizers in well-drained soils has not been reported, this is a potential contributing factor. Since N addition in our case stimulated CH₄ consumption only when conditions were optimum

for nitrification, we suggest N may have limited nitrification and therefore opportunistic co-oxidation of CH₄. While mineral N is not a reliable predictor of changes in either N₂O or CH₄ flux (Mosier et al., 1996), soil ammonium levels observed here suggest nitrification as a potential driver to treatment differences in CH₄ flux for these very similar, well-aerated soils (Table 1).

Our time-integrated CO₂ fluxes for a 5-mo time period fell within the annual estimate for temperate agroecosystems of 477 to 2875 g CO₂ m⁻² (Raich and Schlesinger, 1992). Growing season mean CO₂ fluxes reported here were higher than between-row fluxes soil reported by Mosier et al. (2006) for maize in Colorado (220 mg CO₂ m⁻² h⁻¹) and lower than between-row soil fluxes reported by Amos et al. (2005) for maize in Nebraska (700 mg CO₂ m⁻² h⁻¹). Carbon dioxide flux is controlled by microenvironment, microbial activity, vegetative cover, and land use. Lower CO₂ emissions under dryland conditions at higher latitudes in the northern Great Plains were expected, given the strong temperature control on soil respiration (Wagai et al., 1998; Fang and Moncrieff, 2001; Parkin and Kaspar, 2003). The abundance of fine root biomass at the prairie reference site (Wagai et al., 1998) resulted in greater CO₂ flux than the arable soil. Reductions in tillage may conserve soil carbon and minimize CO₂ emissions (Wagai et al., 1998; Koga, 2004). Our data suggest application of fertilizer in early spring, when temperatures are <10°C, might also reduce growing-season emissions (or GWP_{seasonal}) for dryland cropping systems in the northern Great Plains.

Nitrous oxide fluxes were not significantly affected by timing of fertilizer application, and time-integrated fluxes were lower than those reported for irrigated maize in Colorado (Mosier et al., 2006) and in Nebraska (Amos et al., 2005). Data are lacking for N₂O fluxes in the northern Great Plains, but models estimate North Dakota soils emit 0.8 to 2.3 kg N ha⁻¹ yr (Li et al., 1996). We report emissions in central ND of <0.1 kg N₂O-N ha⁻¹ over a 5-mo time course. While dormant season fluxes were not measured, they would have to be eight times greater than those measured during the growing season to meet the minimum estimated by Li et al. (1996). Modeled estimates for ND may more closely represent fluxes on the eastern side of the state, where time-integrated N₂O fluxes over a 5-mo time course ranged from 0.64 to 1.63 kg N₂O-N ha⁻¹ (Phillips, 2007). We suspect the source strength of N₂O-N in dryland cropping systems is lower than expected because yields are lower than in irrigated or more mesic regions. As such, yield goals are lower and management inputs are less intensive than in irrigated or more highly productive areas.

Urea application just before planting did not result in greater aboveground phytomass or greater grain yield, compared to urea application 6-wk earlier. The GHG treatment effect observed here (548 kg C ha⁻¹) was 3.7 times greater than GHG emissions associated with farm equipment and chemical use (148 kg C ha⁻¹) and represented 25% of net aboveground C in plant phytomass (1950 kg C ha⁻¹). However, without data representative of plant C fixed belowground and net GHG fluxes within crop rows (Mielnick, 1996; Amos et al., 2005), the effect of treatment relative to net primary production cannot be determined. We can conclude that application of urea early, when soil temperatures during the day were just above 0°C, resulted

in 19% lower soil GHG emissions over a growing season. These results suggest a closer investigation of the N budget and GHG emissions in dryland cropping systems are warranted.

Summary and Conclusions

Timing of fertilizer application affected fluxes of CH₄, CO₂, and net GHG for soils planted to maize in a production environment, with greater emissions for soils fertilized in late spring. Results suggest that when mean air and soil temperatures warmed above 10°C in late spring, microbial activity was stimulated by the recent addition of urea. Differences between treatments in late spring were most evident when %WFPS rose above 30%. Emissions of CO₂ accounted for over 98% of net GHG flux. Despite greater CH₄ uptake, the net GHG flux calculated for the growing season was 19% lower for soils fertilized in early spring as compared to late spring. Our results suggest that fertilizer application timing should be considered when evaluating effects of agricultural practices on GHG emissions. Additional studies are warranted to (i) test how the effect of fertilizer application on GHG emissions is influenced by air temperature, (ii) test how the effect of fertilizer on GHG emissions is influenced by soil characteristics, and (iii) test how the residence time of urea influences GHG emissions in dryland cropping systems. Further, investigation is needed to determine how timing of urea application can be used as a management tool to reduce GHG emissions without sacrificing yield within dryland cropping systems.

Acknowledgments

Thanks to Scott Bylin, April Blackbird, and Frances Podrebarac for mapping and data collection assistance. Special thanks to Mark Wutzke, Varian Inc., for outstanding technical assistance. We appreciate comments by Mark Liebig and Scott Kronberg and the anonymous reviewers. This work was supported by USDA ARS Soil Resource Management Program (Project 5445-11120-001-00D) and Agricultural System Competitiveness and Sustainability Program (Project 5445-21660-002-00D). This publication contributes to the USDA ARS GRACenet Project.

References

Amos, B., T.J. Arkebauer, and J.W. Doran. 2005. Soil surface fluxes of greenhouse gases in an irrigated maize-based agroecosystem. *Soil Sci. Soc. Am. J.* 69:387–395.

Avrahami, S., R. Conrad, and G. Braker. 2002. Effect of soil ammonium concentration on N₂O release and on the community structure of ammonia oxidizers and denitrifiers. *Appl. Environ. Microbiol.* 68:5685–5692.

Berkey, D.D., and P. Blanchard. 1992. *Calculus*. Saunders College, Fort Worth, TX.

Bodelier, P.L.E., and H.J. Laanbroek. 2004. Nitrogen as a regulatory factor of methane oxidation in soils and sediments. *FEMS Microbiol. Ecol.* 47:265–277.

Bouwman, A.F., L.J.M. Boumans, and N.H. Batjes. 2002. Emissions of N₂O and NO from fertilized fields: Summary of available measurement data. *Global Biogeochem. Cycles* 16:1058–1071.

Breitenbeck, G.A., and J.M. Bremner. 1986. Effects of rate and depth of fertilizer application on emission of nitrous oxide from soil fertilized with anhydrous ammonia. *Biol. Fertil. Soils* 2:201–204.

Chantigny, M.H., D. Prévost, D.A. Angers, R.R. Simard, and F.P. Chalifour. 1998. Nitrous oxide production in soils cropped to corn with varying N fertilization. *Can. J. Soil Sci.* 78:589–596.

Delgado, J.A., and A.R. Mosier. 1996. Mitigation alternatives to decrease nitrous oxide emissions and urea-nitrogen loss and their effect on

methane flux. *J. Environ. Qual.* 25:1105–1111.

Dutaur, L., and L.V. Verchot. 2007. A global inventory of the soil CH₄ sink. *Global Biogeochem. Cycles* 21, GB4013, doi:10.1029/2006GB002734.

EIA. 1999. Annual energy outlook, 2000. DOE/EIA-0383(2000). U.S. Dep. of Energy, Energy Information Admin., Washington, DC.

Fang, C., and J.B. Moncrieff. 2001. The dependence of soil CO₂ efflux on temperature. *Soil Biol. Biochem.* 33:155–165.

Flessa, H., P. Dörch, and F. Beese. 1995. Seasonal variation of N₂O and CH₄ fluxes in differently managed arable soils in southern Germany. *J. Geophys. Res.* 100:23,115–23,142.

Frank, A.B., M.A. Liebig, and J.D. Hanson. 2002. Soil carbon dioxide fluxes in northern semi-arid grasslands. *Soil Biol. Biochem.* 34:1235–1241.

IPCC. 2007. *Climate change 2007. The Physical Science Basis*, Cambridge Univ. Press, Cambridge, UK.

Jones, R.D., and R.Y. Morita. 1983. Methane oxidation by *Nitrosococcus oceanus* and *Nitrosomonas europaea*. *Appl. Environ. Microbiol.* 45:401–410.

Kammann, C., L. Grünhage, H.J. Jäger, and G. Wachinger. 2001. Methane fluxes from differentially managed grassland study plots: The important role of CH₄ oxidation in grassland with a high potential for CH₄ production. *Environ. Pollut.* 115:261–273.

Koga, N. 2004. N₂O emission and CH₄ uptake in arable fields managed under conventional and reduced tillage cropping systems in Northern Japan. *Global Biogeochem. Cycles* 18:1–11.

Lal, R. 2004. Carbon emission from farm operations. *Environ. Int.* 30:981–990.

Lazarus, W.F. 2008. Machinery cost estimates. Available at <http://www.apec.umn.edu/faculty/wlazarus/documents/mf2008.pdf> (accessed 7 Nov. 2008; verified 24 Apr. 2009). MF2008. Univ. of Minnesota Ext. Serv. St. Paul.

Li, C., V. Narayanan, and R.C. Harriss. 1996. Model estimates of nitrous oxide emissions from agricultural lands in the United States. *Global Biogeochem. Cycles* 10:297–306.

Liebig, M.A., J.R. Gross, S.L. Kronberg, J.D. Hanson, A.B. Frank, and R.L. Phillips. 2006. Soil response to long-term grazing in the northern Great Plains of North America. *Agric. Ecosyst. Environ.* 115:270–276.

Littell, R.C., G.A. Milliken, W.W. Stroup, and R.D. Wolfinger. 1996. SAS system for mixed models. SAS Inst., Cary, NC.

Mielnick, P.C. 1996. Soil carbon dioxide flux in agricultural ecosystems. Ph.D. diss. Univ. of Nebraska, Lincoln (Diss. Abstr. AAT9712521).

Mosier, A.R., A.D. Halvorson, C.A. Reule, and X.J. Liu. 2006. Net global warming potential and greenhouse gas intensity in irrigated cropping systems in northeastern Colorado. *J. Environ. Qual.* 35:1584–1598.

Mosier, A.R., W.J. Parton, D.W. Valentine, D.S. Ojima, D.S. Schimel, and J.A. Delgado. 1996. CH₄ and N₂O fluxes in the Colorado shortgrass steppe: I. Impact of landscape and nitrogen addition. *Global Biogeochem. Cycles* 10:387–399.

Parkin, T.B., and T.C. Kaspar. 2003. Temperature controls on diurnal carbon dioxide flux: Implications for estimating soil carbon loss. *Soil Sci. Soc. Am. J.* 67:1763–1772.

Phillips, R.L. 2007. Organic agriculture and nitrous oxide emissions at sub-zero soil temperatures. *J. Environ. Qual.* 36:23–30.

Phillips, R.L., S.C. Whalen, and W.H. Schlesinger. 2001. Influence of atmospheric CO₂ enrichment on nitrous oxide flux in a temperate forest ecosystem. *Global Biogeochem. Cycles* 15:741–752.

Prather, M., D. Ehhalt, F. Dentener, J. Derwent, E. Dlugokencky, E.A. Holland, I. Isaksen, J. Katima, P. Kirchhoff, P.A. Matson, P. Midgley, and M. Wang. 2001. Atmospheric chemistry and greenhouse gases. p. 240–287. *In* J.T. Houghton et al. (ed.) *Climate change 2001: The scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge Univ. Press, Cambridge, UK.

Raich, J.W., and W.A. Schlesinger. 1992. The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. *Tellus* 44:81–99.

Ramaswamy, V., O. Boucher, J. Haigh, D. Haughustaine, J. Haywood, and G. Myhre. 2001. Radiative forcing of climate change. p. 349–416. *In* J.T. Houghton et al. (ed.) *Climate change 2001: The scientific basis*. Cambridge Univ. Press, Cambridge.

Robertson, G.P., E.A. Paul, and R.R. Harwood. 2000. Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. *Science (Washington, DC)* 289:1922–1925.

Sainju, U.M., J.D. Jabro, and W.B. Stevens. 2008. Soil carbon dioxide emission and carbon content as affected by irrigation, tillage, cropping system, and nitrogen fertilization. *J. Environ. Qual.* 37:98–106.

- Smith, K.A., K.E. Dobbie, B.C. Ball, L.R. Bakken, B.K. Sitaula, S. Hansen, R. Brumme, W. Borken, S. Christensen, A. Priemé, D. Fowler, J.A. Macdonald, U. Skiba, L. Klemetsson, A. Kasimir-Klemetsson, A. Degórska, and P. Orlanski. 2000. Oxidation of atmospheric methane in Northern European soils, comparison with other ecosystems, and uncertainties in the global terrestrial sink. *Global Change Biol.* 6:791–803.
- Soil Survey Staff. 2008. Official soil series description. USDA Washington, DC Available at <http://soils.usda.gov/technical/classification/osd/index.html> (verified 24 Apr. 2009).
- Tate, K.R., and R.G. Striegl. 1993. Methane consumption and carbon dioxide emission in tallgrass prairie: Effects of biomass burning and conversion to agriculture. *Global Biogeochem. Cycles* 7:735–748.
- Wagai, R., K.R. Brye, T.G. Stith, J.M. Norman, and L.G. Bundy. 1998. Land use and environmental factors influencing soil surface CO₂ flux and microbial biomass in natural and managed ecosystems in southern Wisconsin. *Soil Biol. Biochem.* 30:1501–1509.
- West, T.O., and G. Marland. 2002. A synthesis of carbon sequestration, carbon, emissions, and net carbon flux in agriculture: Comparing tillage practices in the United States. *Agric. Ecosyst. Environ.* 91:217–232.
- Whalen, S.C., and W.S. Reeburgh. 1992. Interannual variations in tundra methane emissions: A four-year time series at fixed sites. *Global Biogeochem. Cycles* 6:139–159.
- Wildung, R.E., T.R. Garland, and R.L. Buschbom. 1975. The interdependent effect of soil temperature and water content on soil respiration rate and plant root decomposition in arid grassland soils. *Soil Biol. Biochem.* 7:373–378.