Grazing Management Contributions to Net Global Warming Potential: A Long-term Evaluation in the Northern Great Plains

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The role of grassland ecosystems as net sinks or sources of greenhouse gases (GHGs) is limited by a paucity of information regarding management impacts on the flux of nitrous oxide (N_2O) and methane (CH_4). Furthermore, no long-term evaluation of net global warming potential (GWP) for grassland ecosystems in the northern Great Plains (NGP) of North America has been reported. Given this need, we sought to determine net GWP for three grazing management systems located within the NGP. Grazing management systems included two native vegetation pastures (moderately grazed pasture [MGP], heavily grazed pasture [HGP]) and a heavily grazed crested wheatgrass [*Agropyron desertorum* (Fisch. ex. Link) Schult.] pasture (CWP) near Mandan, ND. Factors evaluated for their contribution to GWP included (i) CO_2 emissions associated with N fertilizer production and application, (ii) literature-derived estimates of CH_4 production for enteric fermentation, (iii) change in soil organic carbon (SOC) over 44 yr using archived soil samples, and (iv) soil–atmosphere N_2O and CH_4 fluxes over 3 yr using static chamber methodology. Analysis of SOC indicated all pastures to be significant sinks for SOC, with sequestration rates ranging from 0.39 to 0.46 Mg C ha^{-1} yr^{-1}. All pastures were minor sinks for CH_4 (<2.0 kg CH_4–C ha^{-1} yr^{-1}). Greater N inputs within CWP contributed to annual N_2O emission nearly threefold greater than HGP and MGP. Due to differences in stocking rate, CH_4 production from enteric fermentation was nearly threefold less in MGP than CWP and HGP. When factors contributing to net GWP were summed, HGP and MGP were found to serve as net $CO_{2equiv.}$ sinks, while CWP was a net $CO_{2equiv.}$ source. Values for GWP and GHG intensity, however, indicated net reductions in GHG emissions can be most effectively achieved through moderate stocking rates on native vegetation in the NGP.

Contributions of grassland ecosystems to net global warming potential (GWP) are largely unknown. Much work to date has emphasized the role of grassland management to affect change in soil organic carbon (SOC) stocks (Follett et al., 2001; Derner and Schuman, 2007), which are estimated at 306 to 330 Pg C for grassland ecosystems worldwide (Batjes, 1996). Sequestration of atmospheric carbon dioxide (CO_2) as SOC under grassland ecosystems is estimated at 0.5 Pg C yr^{-1} (Schlesinger, 1997), underscoring the importance of these ecosystems to affect the global C cycle and indirectly regulate climate (Derner and Schuman, 2007; Lal and Follett, 2009). However, clear determination of grassland ecosystems as net sinks or sources of greenhouse gases (GHGs) is limited by a paucity of information regarding management impacts on the flux of nitrous oxide (N_2O) and methane (CH_4).

Within a grassland ecosystem, emission of N_2O tends to be highly variable in space and time (Mosier et al., 1991; Oenema et al., 1997), owing to the heterogeneous distribution of urine and dung patches and variability of edaphic properties that control soil water status (Mosier et al., 1998). Across grassland ecosystems, differences in N_2O emission appear to be driven by soil water status, N mineralization, and plant community dynamics (Mosier et al., 1997; Bolan et al., 2004; Liebig et al., 2005). Inherent challenges associated with upscaling N_2O emissions from grazed lands are reflected by global estimates, which range from 16 to 33% of total agricultural N_2O emissions (Clark et al., 2005).

Published information suggests grassland soils can be small CH_4 sinks (Mosier et al., 1997; Chan and Parkin, 2001) or small to moderate CH_4 sources (Yamulki et al., 1999; Flessa et al., 2002; Allard et al., 2007), depending on factors that regulate the activity of soil bacteria responsible for CH_4 production and consumption (Paul and Clark, 1996). In many grassland ecosystems, however, the magnitude of soil–atmosphere CH_4 flux is relatively small when compared with CH_4 from domestic ruminant livestock (USEPA, 2009), which currently accounts for >30% of

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Abbreviations: CWP, crested wheatgrass pasture; GHG, greenhouse gas; GHGI, greenhouse gas intensity; GWP, global warming potential; HGP, heavily grazed pasture; MGP, moderately grazed pasture; NGP, northern Great Plains; SOC, soil organic carbon; WFPS, water-filled pore space.

total CH_4 emissions worldwide (Smith et al., 2007). Ruminant CH_4 emissions vary by livestock type, stocking rate, and diet (Westberg et al., 2001), and as a result, can differ by an order of magnitude across grassland ecosystems (Clark et al., 2005).

Recent evaluations of net GWP for grassland ecosystems highlight potential GHG tradeoffs associated with management. Measurements of net ecosystem exchange across nine sites in Europe indicated managed grasslands to be a sink for atmospheric CO_2 (−240 ± 70 g C m^{-2} yr^{-1}) (Soussana et al., 2007). However, when C exports and emissions of N_2O and CH_4 were taken into account, the same grasslands exhibited a net GWP not significantly different from zero (−85 ± 77 g CO_2–$C_{equiv.}$ m^{-2} yr^{-1}). Select GHG evaluations for seminatural grasslands in Europe have shown net GWP to range from −446 to 251 g CO_2–$C_{equiv.}$ m^{-2} yr^{-1} (Flessa et al., 2002; Allard et al., 2007; Soussana et al., 2007), with increased stocking density consistently reducing GHG sink capacity.

While investigative efforts in Europe have provided valuable insight into the role of managed grasslands to affect GWP, results are very much limited to regions sharing similar climatic and edaphic attributes. To date, no long-term (≥3 yr) evaluation of net GWP for a grassland ecosystem in the northern Great Plains (NGP) of North America has been reported. Given this need, we sought to determine net GWP for three grazing management systems (two native vegetation pastures and one seeded forage pasture) located in the NGP, a region possessing a semiarid continental climate and abundant grassland resources (Padbury et al., 2002).

Materials and Methods

Site Description

Experimental sites were located within the Temperate Steppe Ecoregion of the United States (Bailey, 1995). This ecoregion possesses a semiarid continental climate, with evaporation typically exceeding precipitation in any given year. Specifically, sites were within the Missouri Plateau approximately 6 km south of Mandan, ND (46°46′12″ N, 100°55′59″ W). Average annual precipitation at the sites from 1913 to 2003 was 410 mm and long-term growing season precipitation (April–September) was 330 mm. Average annual temperature was 4°C, though daily averages ranged from 21°C in the summer to −11°C in the winter. The average frost-free period at the sites was 131 d.

The sites were on gently rolling uplands (0–3% slope) with a silty loess mantle overlying Wisconsin age till. Predominant soils were Temvik-Wilton silt loams (FAO: Calcic Siltic Chernozems; USDA: Fine-silty, mixed, superactive, frigid Typic and Pachic Haplustolls) (Soil Survey Staff, 2009). A survey of select soil properties conducted before initiating the study has been reviewed elsewhere (Liebig et al., 2006).

Description of Grazing Treatments

Grazing treatments included two native vegetation pastures and one seeded forage pasture. The two native vegetation pastures included a moderately grazed pasture (MGP) and heavily grazed pasture (HGP), both of which were established in 1916. Vegetation composition within the MGP on initiation of this study included a mixture of blue grama [*Bouteloua gracilis* (H.B.K.) Lag. ex Griffiths], needle-and-thread (*Stipa comata* Trin. and Rupr.), western wheatgrass [*Pascopyrum smithii* (Rybd) Löve], prairie junegrass [*Koeleria pyramidata* (Lam) Beauv.], Kentucky bluegrass (*Poa pratensis* L.), and sedge (*Carex filifolia* Nutt. and *Carex heliophila* Mack.). Within HGP, blue grama and sedge were the dominant plant species. A crested wheatgrass pasture (CWP) represented the seeded forage, which was planted in 1932 into plowed native range. In addition to crested wheatgrass, a small amount of blue grama was present in the CWP. The grazing treatments varied in size, with MGP, HGP, and CWP occupying 15.4, 2.8, and 2.6 ha, respectively. Per standard protocol for the establishment of experimental treatments before widespread use of statistics, none of the grazing treatments were replicated (Sarvis, 1923).

The grazing season for all three pastures extended from about mid-May to early October using yearling steers. Stocking rates for MGP and HGP were 2.6 and 0.9 ha $steer^{-1}$ (0.39 and 1.1 animals ha^{-1}), respectively. Stocking rates within CWP were 0.4 ha $steer^{-1}$ (2.3 animals ha^{-1}) in late spring–early summer and 0.9 ha $steer^{-1}$ (1.2 animals ha^{-1}) for the remainder of the grazing season. Grazing has occurred every year since pasture establishment except during times of severe drought when forage production was inadequate to support livestock grazing. To increase forage production, CWP was fertilized in the fall of each year since 1963 with NH_4NO_3 at 45 kg N ha^{-1}. Of the three grazing treatments, MGP and HGP are reflective of the predominant forms of grassland management in the northern Great Plains.

Experimental Setup

Preparations for the experiment were initiated on 6 Oct. 2003 with the placement of six polyvinyl chloride (PVC) pipe anchors (19.6-cm diam.; 15.2-cm height) in each grazing treatment. The anchors served as part of a two-piece chamber system for GHG analyses (described below). Anchors were oriented in a hexagonal pattern around the center of each grazing treatment. Distance between adjacent anchors was approximately 40 m in the MGP, and 10 m in the HGP and CWP. Treatment area covered within the six collars was 1.8×10^{-5} ha. Anchors were inserted into the soil to a depth of approximately 10 cm using a tractor-mounted Giddings hydraulic probe (Giddings Machine Co., Windsor, CO). A carpenter's level was used during collar insertion to ensure each anchor was level on north–south and east–west axes. Following insertion, headspace within each anchor was determined by lining the space within an anchor with plastic wrap and filling it with a known volume of water until the water level was flush with the upper edge of the anchor. Vegetation within each anchor was not removed during the evaluation unless it was grazed by livestock.

Soil Sample Collection and Analyses

On 7 Oct. 2003, soil samples were collected from four locations in each grazing treatment, approximately equidistant between anchors 1 and 2, 1 and 6, 3 and 4, and 4 and 5. Soil samples were collected to 100 cm in depth increments of 0 to 5, 5 to 10, 10 to 20, 20 to 30, 30 to 60, and 60 to 100 cm using a tractor-mounted Giddings hydraulic probe with an inner tip diameter of 3.5 cm. At each location, eight soil cores at 0 to 5 and 5 to 10 cm, four soil cores at 10 to 20 and 20 to

30 cm, and two soil cores at 30 to 60 and 60 to 100 cm were collected and composited by depth. Each sample was saved in a double-lined plastic bag and placed in cold storage at 5°C until processing.

Soil samples were dried at 35°C for 3 to 4 d and ground by hand to pass a 2.0-mm sieve. Identifiable plant material (>2.0 mm) was removed during sieving. Air-dry water content was determined for each sample using a 12- to 15-g subsample by measuring the difference in mass before and after drying at 105°C for 24 h. Samples were analyzed for total soil C and N by dry combustion on soil ground to pass a 0.106-mm sieve (Nelson and Sommers, 1996). Using the same fine-ground soil, inorganic C was measured on soils with a pH $\geq$7.2 by quantifying the amount of CO_2 produced using a volumetric calcimeter after application of dilute HCl stabilized with $FeCl_2$ (Loeppert and Suarez, 1996). Soil organic C was calculated as the difference between total C and inorganic C. Gravimetric data were converted to a volumetric basis for each sampling depth using field-measured soil bulk density (Blake and Hartge, 1986). All data were expressed on an oven-dry basis.

To assess grazing management effects on SOC over time, archived soil from a 1959 sampling of the grazing treatments (reviewed by Liebig et al., 2008b) was subsampled and analyzed as outlined above. While information on protocol from the 1959 sampling is limited, inspection of the sample descriptions indicated multiple locations in each of the grazing treatments were sampled to a depth of 60.9 cm in increments of 0 to 15.2, 15.2 to 30.5, and 30.5 to 60.9 cm. Specifically, nine locations were sampled in MGP and HGP (resulting in 27 total samples each), while 30 locations were sampled in CWP (resulting in 90 total samples). Given discrepancies in depth increments between the 1959 and 2003 samplings and lack of soil bulk density data from 1959, volumetric expression of SOC for the archive samples utilized soil bulk density values weighted by depth from the 2003 sampling for the respective grazing treatments (Table 1). Similarly, calculated values of SOC in 2003 were weighted to conform to depth increments used in 1959. To eliminate effects of sampling depth and soil bulk density on SOC stocks, data from the 1959 and 2003 samplings were recalculated on an equivalent mass basis assuming a soil profile mass of 7500 Mg ha^{-1} for each grazing treatment (Ellert and Bettany, 1995). The soil profile mass was selected based on the approximate soil mass in the upper 60.9-cm soil depth assuming a mean soil bulk density across grazing treatments of 1.22 Mg m^{-3}.

Greenhouse Gas Analyses

Fluxes of N_2O and CH_4 were measured in the grazing treatments from 21 Oct. 2003 to 24 Oct. 2006, employing static chamber methodology as outlined by Hutchinson and Mosier (1981). Within each treatment, gas samples were collected from two-part chambers consisting of six anchors (outlined above) each with a PVC cap (19.6-cm diam.; 10.0-cm height), vent tube, and sampling port. Gas samples from inside the chambers were collected with a 20-mL syringe at 0, 15, and 30 min after installation (approximately 1000 h each sampling day). After collection, gas samples were injected into 12-mL evacuated Exetainer glass vials sealed with butyl rubber septa (Labco Ltd., Buckinghamshire, UK).

Concentration of N_2O and CH_4 inside each vial was measured by gas chromatography 1 to 3 d after collection using a Shimadzu GC-17A gas chromatograph (Shimadzu Scientific Instruments, Kyoto, Japan) attached to an ISCO Retriever IV autosampler (Teledyne Isco, Inc., Lincoln, NE). Using this system, each sample was auto-injected and split into two sample loops, with 1 mL directed to a thermal conductivity detector (TCD) in series with a flame ionization detector (FID) using ultrapure He carrier gas. Ultrapure He and hydrocarbon-free air were used for combustion in the FID. The second sample loop directed 0.5 mL to a ^{63}Ni electron capture detector (ECD) with ultrapure N_2 as carrier gas. Before reaching each detector, samples passed through a 4-m HayeSep D column (Hayes Separations, Inc., Bandera, TX) for the TCD and FID, and 2-m Porapak Q (Waters Corp., Milford, MA) and 4-m HayeSep D columns for the ECD. The gas chromatograph was calibrated with a commercial blend of N_2O (0.100, 0.401, 1.99 μL L^{-1}) and CH_4 (1.00, 2.09, 10.1 μL L^{-1}) balanced in N_2 from Scott Specialty Gases (Scott Specialty Gases, Plumsteadville, PA). Precision analysis expressed as coefficient of variation for 18 replicate injections of 0.401 μL N_2O L^{-1} and 2.09 μL CH_4 L^{-1} standards was 1.8 and 2.0%, respectively. Standard error associated with the precision analysis for each gas was ±0.002 μL N_2O L^{-1} and ±0.008 μL CH_4 L^{-1}. Gas flux was calculated from the change in concentration in the chamber headspace over time (Hutchinson and Mosier, 1981).

Measurements of gas fluxes were made one to two times per week when near-surface soil depths were not frozen or during midwinter thawing periods. Otherwise, fluxes were measured every other week. Over the course of the evaluation gas fluxes were measured 128 times. Due to recurring problems with the FID, CH_4 flux measurements were used only from 19 Jan. 2005 to 27 Sept. 2005 and from 29 Nov. 2005 to 24 Oct. 2006, for a total of 73 measurements.

Supplementary Analyses

Precipitation, air temperature, and solar radiation were monitored daily at a North Dakota Agricultural Weather Network (NDAWN) station within 1 km of the grazing treatments. Relevant data were downloaded following each gas sampling event from the NDAWN Web site (NDAWN, 2009).

Near-surface soil water content and temperature were measured concurrently with gas flux when the soil was not frozen. Volumetric water content was measured in the surface 12 cm

Table 1. Values for soil bulk density used to calculate soil organic C for fixed depths in 1959 and 2003 within each grazing treatment.

Soil depth	Soil bulk density: Crested wheatgrass	Heavily grazed	Moderately grazed	P
cm		Mg m^{-3}		
0–15.2	1.20 (0.04) a†	1.06 (0.01) b	1.05 (0.01) b	0.01
15.2–30.5	1.26 (0.07)	1.16 (0.04)	1.15 (0.03)	0.29
30.5–60.9	1.29 (0.03)	1.33 (0.01)	1.27 (0.01)	0.16
0–60.9	1.26 (0.02)	1.22 (0.01)	1.19 (0.01)	0.08

† Values in parentheses reflect the standard error of the mean. Means compared within depth. Values in a column with unlike letters differ ($P \leq 0.05$).

of soil using a time-domain reflectometry technique with a Campbell CS620 HydroSense System (Campbell Scientific, Inc., Logan, UT). Soil temperature was measured at a 6-cm depth with an Omega HH81A handheld digital thermometer attached to a heavy-duty T-type thermocouple probe (Omega, Inc., Stamford, CT). Three measurements of soil water content and one measurement of soil temperature were made within 30 cm of each anchor during the 15-min gas sampling period. Values for volumetric water content were converted to water-filled pore space (WFPS) using field-measured soil bulk density for the surface 10 cm (Linn and Doran, 1984).

Data Analyses

Given the lack of replication of the grazing treatments, anchors and sampling locations served as pseudoreplicates for measurements of gas flux, soil properties, and aboveground biomass (Gomez, 1984). While not ideal, justification for using this approach hinges on the long-term value of the grazing treatments, the age of which makes them rare within North America. All collected data were analyzed using PROC MIXED in SAS (Littell et al., 1996), with grazing treatments and pseudoreplicates considered fixed and random effects, respectively. A significance criterion of $P \leq 0.05$ was used to document differences among means. Variation of arithmetic means was expressed using standard error (Steel and Torrie, 1980). Where appropriate, associations between measured parameters were identified using Pearson correlation analysis.

Gas flux data were tested for normality using skewness, kurtosis, and Kolmogorov–Smirnov coefficients before and after data were log-transformed. Data transformation did not improve normality of the data, so original gas flux data were used for statistical analyses. A mixed repeated-measures model was used to analyze the effects of year, quarterly period (1 December–28/29 February, 1 March–31 May, 1 June–31 August, and 1 September–30 November), and grazing treatment on N_2O and CH_4 flux. Effects of soil temperature and WFPS on CH_4 and N_2O flux were also evaluated using a repeated-measures model, but with a parsed data set including only sampling times when soil temperature and WFPS were measured. For both analyses, a time-series covariance structure was used in the repeated-measures model, where correlations decline over time (Phillips et al., 2009). Cumulative fluxes of N_2O and CH_4 were calculated by linearly interpolating data points and integrating the underlying area (Gilbert, 1987).

Net GWP was calculated for each grazing treatment as the sum of emitted CO_2 equivalents from five factors:

$$\text{Net GWP} = \text{NPA} + \text{EF} + \Delta\text{SOC} + \text{CH}_4\text{F} + \text{N}_2\text{OF} \qquad [1]$$

where NPA was N fertilizer production and application; EF was CH_4 emission from enteric fermentation; ΔSOC was SOC change; CH_4F was soil–atmosphere CH_4 flux; and N_2OF was soil–atmosphere N_2O flux. The GWP for N fertilizer production was based on the applied N rate and an emission factor of 3.14 kg CO_2 kg^{-1} N (West and Marland, 2002). Emission of CO_2 associated with fertilizer N application (via a granular spreader and small tractor) was assumed to be 45.5 kg CO_2 ha^{-1} (West and Marland, 2002). Daily CH_4 emission via enteric fermentation from each steer was estimated at 179 g CH_4 d^{-1} assuming a diet of "good pasture" (Westberg et al., 2001). Differences in SOC stocks between the 1959 and 2003 samplings were taken to represent net soil–atmosphere CO_2 exchange for each grazing treatment. Soil organic C stocks determined by the equivalent soil mass approach were used for calculation. Contributions of soil–atmosphere CH_4 and N_2O flux to GWP were based on annual flux rates. For each factor contributing to GWP, the sum of CO_2 equivalents was calculated assuming direct GWP of 1 kg CH_4 ha^{-1} = 25 kg CO_2 ha^{-1} and 1 kg N_2O ha^{-1} = 298 kg CO_2 ha^{-1} (100-yr time horizon) (Intergovernmental Panel on Climate Change, 2007). Results for net GWP were expressed as kg $CO_{2\text{equiv.}}$ ha^{-1} yr^{-1}. Global warming potential was related to animal productivity by dividing net GWP by mean annual weight gain ha^{-1} to estimate greenhouse gas intensity (GHGI) for each grazing treatment.

Results and Discussion

Experimental Conditions

Precipitation, Solar Radiation, and Air Temperature

Conditions during the study were dry and cold, coupled with an absence of continuous snow cover during the winter months. Over the 36-mo evaluation period, 189 d received precipitation (approximately 17%) (Fig. 1A). Minor precipitation events were predominant, with <5 mm event^{-1} occurring 96% of the time. Conversely, precipitation events totaling >25 mm occurred only four times. During the evaluation period, a total of 798 mm of precipitation was received near the study site, which was 432 mm below the long-term mean over the same time period (NDAWN, 2009). Growing-season precipitation (April–September) was particularly deficient in 2004 (38 mm) and 2006 (188 mm). Furthermore, precipitation received as snow during the non–growing season was limited, as continuous snow cover was present only during winter 2003–2004.

Daily sums of solar radiation averaged 14.0 ± 8.2 MJ m^{-2} d^{-1}, and exceeded 30 MJ m^{-2} d^{-1} only eight times during the evaluation period (Fig. 1B). Mean daily air temperature was 6.7 ± 12.0°C, and ranged from −29.9°C (28 Jan. 2004) to 31.0°C (30 Jul. 2006) (Fig. 1C). Over the course of the evaluation period, mean daily air temperature was <0°C on 348 d (32%). As expected, mean daily air temperature was strongly associated with daily sums of solar radiation (r = 0.68; $P \leq$ 0.01).

Water-filled Pore Space and Soil Temperature

Near-surface WFPS averaged 33.3 ± 0.4%, and ranged from 9.5 to 74.7% when the soil was not frozen (Fig. 2A). Across all measurements of WFPS during the study period, 95% of the values were 58% or less, indicating the prevalence of aerobic soil conditions. Mean WFPS was greater in HGP (35.6 ± 0.7%) than in CWP (31.9 ± 0.6%) and MGP (32.2 ± 0.6%) (P = 0.03; data not shown), possibly the result of greater available water-holding capacity from increased near-surface root biomass in the former (Lorenz and Rogler, 1967). Across grazing treatments, mean soil temperature at 6 cm when the soil was not frozen was 14.7 ± 0.2°C, ranged from 1.0 to 25.2°C, and was significantly associated with mean daily air

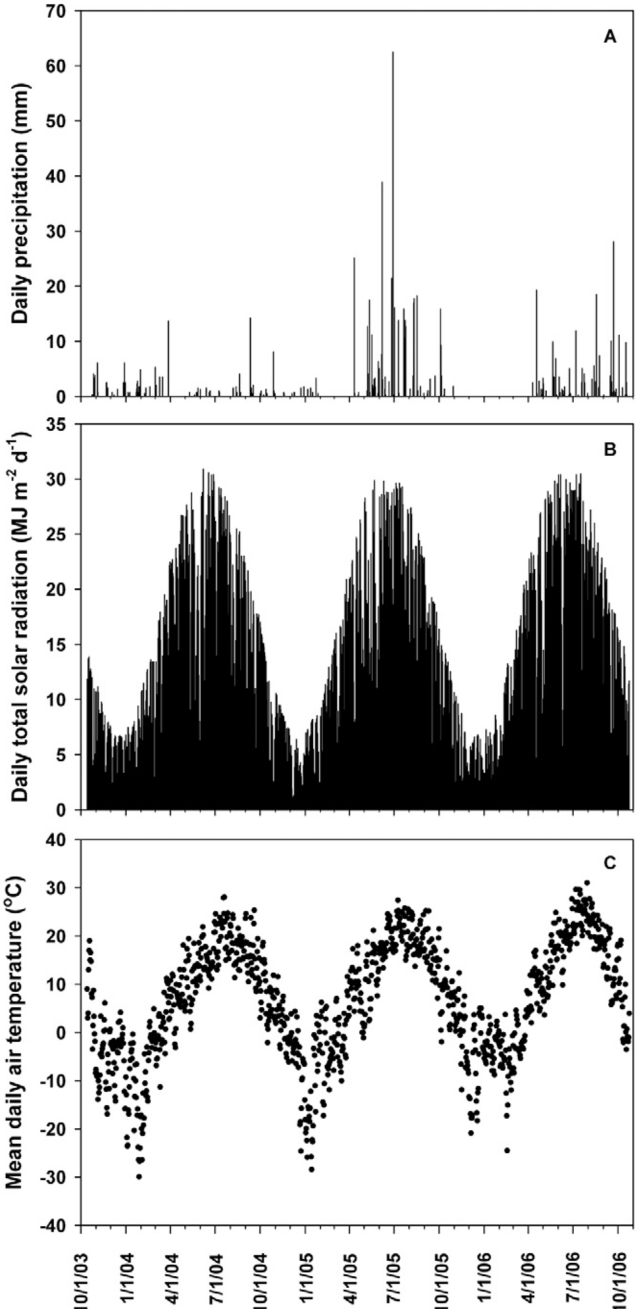

Fig. 1. (A) Daily precipitation, (B) daily total solar radiation, and (C) mean daily air temperature at the study site (Mandan, ND) from October 2003 through November 2006.

temperature (r = 0.89, $P \leq 0.01$) (Fig. 2B). Mean soil temperature did not differ among grazing treatments, with mean values of 14.5 ± 0.3°C in CWP, 14.7 ± 0.3°C in HGP, and 14.7 ± 0.3°C in MGP (P = 0.73; data not shown).

Factors Contributing to Global Warming Potential

Nitrogen Fertilization Production and Application

The contribution of N fertilization to GWP of the grazing treatments was applicable only to CWP, as synthetic N fertilizer was not applied to HGP and MGP. Using conversion factors outlined above, emission of CO_2 associated with N fertilizer production and application was 141 and 118 kg CO_2 ha^{-1} yr^{-1}, respectively.

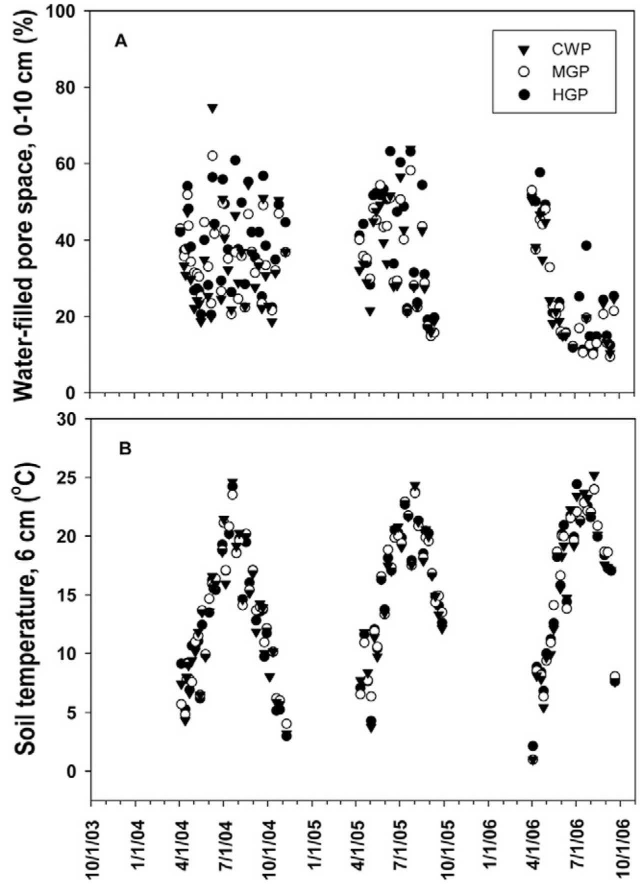

Fig. 2. (A) Water-filled pore space and (B) soil temperature within crested wheatgrass pasture (CWP), heavily grazed pasture (HGP), and moderately grazed pasture (MGP) from October 2003 through November 2006 when the soil was not frozen, Mandan, ND.

Methane Emission from Enteric Fermentation

Because consistent stocking rates were used throughout the study and estimates of CH_4 emission were derived from the literature (Westberg et al., 2001), stocking rate and grazing days dictated CH_4 emission from cattle for each grazing treatment. Accordingly, reduced grazing days in 2006 led to low CH_4 emission in all grazing treatments, with values lower in CWP and MGP (mean = 3.7 kg CH_4–C ha^{-1} yr^{-1}) than in HGP (10.2 kg CH_4–C ha^{-1} yr^{-1}) (Table 2). Estimates of CH_4 emission in 2004 and 2005 were similar among treatments, with production greatest in CWP (mean = 23.5 kg CH_4–C ha^{-1} yr^{-1}) and least in MGP (mean = 6.1 kg CH_4–C ha^{-1} yr^{-1}).

Diet quality is known to significantly affect CH_4 emission from ruminant livestock (Johnson and Johnson, 1995). Consequently, it is important to acknowledge estimates of CH_4 emission from cattle in this study may be conservative for HGP, as the diversity of available forage in this grazing treatment was less than in CWP and MGP. The magnitude of this assumption is significant, as estimates of CH_4 emission from cattle on "poor pasture" are 25% greater than on "good pasture" (Westberg et al., 2001). Furthermore, estimates of CH_4 emission from cattle are inherently variable, but not reflected in the value used in this study. Westberg et al. (2001) found standard deviations in CH_4 emission rates from beef cows on pasture to range from 12 to 23% of the mean.

Table 2. Grazing schedule, number of days grazed, livestock performance, and CH_4 production from livestock for grazing treatments, 2004–2006.

Year	Treatment†	No. of cattle	Grazing schedule		No. of days grazed	Weight gain		CH_4 production kg CH_4–C
			Begin	End		kg animal^{-1}	kg ha^{-1}	ha^{-1} yr^{-1}
2004	CWP	6; 3	25 May	2 July; 26 Aug.‡	38; 93§	40.1 (6.9) b¶	15.4 (2.7) b	20.4#
	HGP	3	25 May	10 Sept.	108	90.0 (10.9) a	32.1 (3.9) a	15.5
	MGP	6	25 May	10 Sept.	108	103.7 (5.1) a	6.7 (0.3) c	5.6
2005	CWP	6; 3	1 June	5 Aug.; 15 Sept.	65; 106	36.0 (5.5) b	13.8 (2.1) b	26.6
	HGP	3	1 June	4 Oct.	125	99.6 (4.9) a	35.6 (1.8) a	17.9
	MGP	6	1 June	4 Oct.	125	114.3 (7.6) a	7.4 (0.5) c	6.5
2006	CWP	6; 3	5 June	29 June	24	19.6 (3.2) b	7.5 (1.2) b	3.7
	HGP	3	5 June	15 Aug.	71	48.5 (6.5) a	17.3 (2.3) a	10.2
	MGP	6	5 June	15 Aug.	71	59.3 (5.2) a	3.8 (0.4) c	3.7

† CWP, crested wheatgrass pasture; HGP, heavily grazed pasture; MGP, moderately grazed pasture.

‡ Three cattle were removed from pasture on the first date. Remaining cattle were removed on the second date.

§ First value refers to number of days grazed by first group of three cattle. Second value refers to the total number of days grazed by remaining cattle.

¶ Values in parentheses reflect the standard error of the mean. Means compared within year. Values in a column with unlike letters differ ($P \leq 0.05$).

Methane production from cattle was estimated using literature-derived values for "good pasture" (Westberg et al., 2001), and was therefore dictated by the stocking rate and number of days grazed within a grazing treatment. Hence, statistical comparisons within a year were not possible.

Δ Soil Organic Carbon

Evaluation of archived soil samples provided valuable insight on stock changes of SOC within the grazing treatments. Using calculations of stock changes for fixed depths, all grazing treatments exhibited a significant increase in SOC over time within the 0- to 15.2-cm depth (Table 3). While SOC in lower depths (15.2–30.5 and 30.5–60.9 cm) did not exhibit significant change from 1959 to 2003, SOC over the entire sampling depth (0–60.9 cm) increased at $P \leq 0.05$ within all grazing treatments. Similarly, SOC stocks calculated on an equivalent mass basis significantly increased in each grazing treatment by 20.4 Mg C ha^{-1} in CWP, 18.2 Mg C ha^{-1} in HGP, and 17.0 Mg C ha^{-1} in MGP (Table 3). Soil inorganic C did not differ between sampling dates in any treatment within individual or aggregated depths (data not shown).

Table 3. Soil organic C within each grazing treatment in 1959 and 2003.

Treatment	Year		P
	1959	2003	
	Soil organic C to a fixed depth (Mg C ha^{-1})		
Crested wheatgrass			
0–15.2	45.4 (1.1)†	62.4 (2.7)	0.01
15.2–30.5	31.9 (1.1)	34.0 (1.4)	0.53
30.5–60.9	43.5 (1.5)	44.9 (1.9)	0.98
0–60.9	120.8 (2.6)	141.3 (3.1)	<0.01
Heavily grazed			
0–15.2	51.5 (1.2)	63.8 (1.2)	<0.01
15.2–30.5	28.6 (1.2)	33.1 (3.8)	0.15
30.5–60.9	36.7 (1.0)	37.9 (1.6)	0.52
0–60.9	116.8 (1.6)	134.8 (6.4)	0.03
Moderately grazed			
0–15.2	43.1 (1.0)	54.7 (2.5)	0.01
15.2–30.5	25.8 (0.9)	30.8 (2.6)	0.06
30.5–60.9	37.2 (1.8)	37.5 (2.4)	0.92
0–60.9	106.1 (3.0)	123.0 (6.2)	0.02
	Soil organic C in equivalent soil mass (Mg C ha^{-1})‡		
Crested wheatgrass	119.1 (2.5)	139.5 (3.1)	<0.01
Heavily grazed	117.2 (1.6)	135.4 (6.4)	0.03
Moderately grazed	108.5 (3.1)	125.5 (6.3)	0.03

† Values in parentheses reflect the standard error of the mean.

‡ An equivalent soil mass of 7500 Mg ha^{-1} was used for calculations following the method of Ellert and Bettany (1995). Depth to reach equivalent soil mass was 59.6, 61.3, and 62.9 cm for crested wheatgrass pasture, heavily grazed pasture, and moderately grazed pasture, respectively.

Measurements of SOC in this study were the first to utilize archived soil samples to assess stock changes within the grazing treatments. Previous evaluations (Frank et al., 1995; Wienhold et al., 2001; Liebig et al., 2006) measured SOC during specific points in time (1991, 1997, and 2003), allowing for relative comparisons of stock values. Bowen ratio/energy balance measurements of CO_2 flux within MGP in the 1990s provided opportunities to estimate SOC accrual rates for HGP and CWP using absolute differences in stock values assuming a base CO_2–C accrual rate of 0.29 Mg C ha^{-1} yr^{-1} for MGP (Frank et al., 2002; Frank, 2004). This approach yielded SOC accrual rates ranging from 0.39 to 0.60 Mg C ha^{-1} yr^{-1} for HGP and 0.38 to 0.55 Mg C ha^{-1} yr^{-1} for CWP (Frank et al., 1995; Wienhold et al., 2001; Liebig et al., 2006). Our results suggest SOC accrual in MGP over the span of the archived samples (1959–2003) was 0.39 ± 0.05 Mg C ha^{-1} yr^{-1}, approximately 0.10 Mg C ha^{-1} yr^{-1} greater than other observations in northern mixed-grass prairie (Derner and Schuman, 2007). Accordingly, SOC accrual rates for HGP and CWP were slightly greater than for MGP (0.41 ± 0.05 and 0.46 ± 0.03 Mg C ha^{-1} yr^{-1}, respectively), owing to a grazing-induced shift in vegetation composition toward greater blue grama in HGP and greater overall biomass production in CWP from N fertilizer application.

Methane Flux

Results from this study support previous observations that semiarid rangelands serve as minor sinks for atmospheric CH_4 (Mosier et al., 1997). Uptake of atmospheric CH_4 was the dominant exchange process during the measurement period, occurring >90% of the time within each of the grazing treatments (Fig. 3). Methane flux was highly variable, ranging from 6.8 to −50.9 μg CH_4–C m^{-2} h^{-1} for CWP, 40.4 to −53.4 μg CH_4–C

$m^{-2} h^{-1}$ for HGP, and 107.4 to −59.3 μg CH_4–C $m^{-2} h^{-1}$ for MGP. When the soil was not frozen (March–November), CH_4 flux was positively associated with WFPS ($r = 0.49$; $P \leq 0.01$; $n = 753$), implying greater CH_4 uptake under drier soil conditions. Soil temperature was not associated with CH_4 flux ($r = -0.18$; $P = 0.91$; $n = 753$).

Methane flux differed across quarterly periods, but not between years or grazing treatments (Table 4). Mean values of CH_4 uptake were greatest in the summer and fall (range = −33.3 to −35.5 μg CH_4–C $m^{-2} h^{-1}$), intermediate in the spring (−20.4 μg CH_4–C $m^{-2} h^{-1}$), and least in the winter (−6.6 μg CH_4–C $m^{-2} h^{-1}$). While seasonal estimates of CH_4 flux in rangeland ecosystems are lacking, summer flux rates observed in this study were comparable to previous investigations conducted in the shortgrass steppe (−26 μg CH_4–C $m^{-2} h^{-1}$) (Mosier et al., 1991) and mixed-grass prairie (−30 μg CH_4–C $m^{-2} h^{-1}$) (Liebig et al., 2008a).

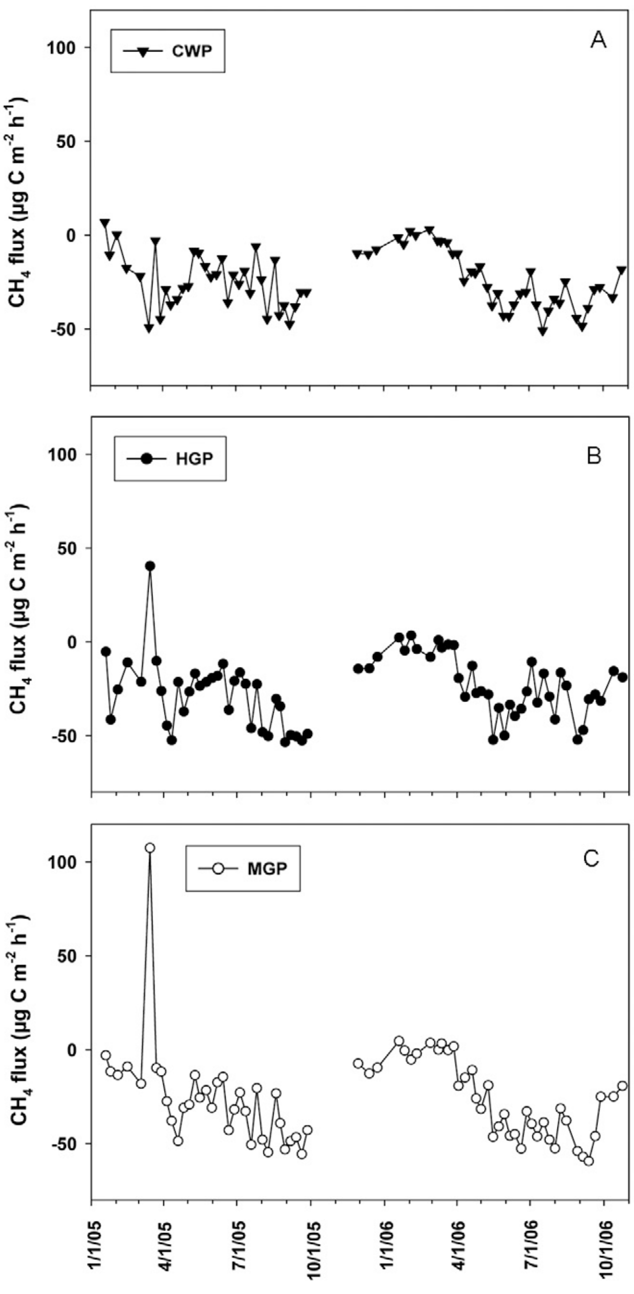

Fig. 3. Flux of CH_4 for (A) crested wheatgrass pasture (CWP), (B) heavily grazed pasture (HGP), and (C) moderately grazed pasture (MGP) from 19 Jan. 2005 through 24 Oct. 2006, Mandan, ND.

Cumulative CH_4 uptake was −3.2 ± 0.2, −3.3 ± 0.3, and −3.3 ± 0.5 kg CH_4–C ha^{-1} for CWP, HGP, and MGP, respectively (Fig. 4A). Accordingly, when calculated on an annual basis, CH_4 uptake equated to −1.8 ± 0.1, −1.9 ± 0.2, and −1.9 ± 0.3 kg CH_4–C $ha^{-1} yr^{-1}$ for CWP, HGP, and MGP. For the native vegetation treatments, CH_4 uptake rates were approximately twice that of native grassland in the shortgrass steppe (−1.0 kg CH_4–C $ha^{-1} yr^{-1}$) (Mosier et al., 1991), but less than observations for grass sod in western Nebraska (−3.2 kg CH_4–C $ha^{-1} yr^{-1}$) (Kessavalou et al., 1998). Interestingly, reduced CH_4 uptake was not observed in CWP relative to native vegetation grazing treatments. Previous studies of fertilized semiarid grasslands have observed a 35 to 40% decrease in CH_4 uptake due to compromised activity of methanogenic bacteria from excessive NH_4–N in soil solution (Mosier et al., 1991; Bronson and Mosier, 1994; Liebig et al., 2008a).

Nitrous Oxide Flux

Most observations of N_2O flux during the measurement period were positive (i.e., net emission of N_2O to the atmosphere) (Fig. 5). However, N_2O consumption was frequent enough to warrant attention, occurring on 20, 15, and 12% of the days that measurements were taken in CWP, HGP, and MGP, respectively. Such events were most prevalent from October 2005 to October 2006 when the soil was not frozen. Consumption of N_2O has been observed to occur in temperate grasslands (Chapuis-Lardy et al., 2007), though consistent relationships to abiotic factors are lacking. Unfortunately, the focus of this study did not allow for a thorough investigation of possible

Table 4. Mean hourly flux for CH_4 and N_2O partitioned by year, quarterly periods, and grazing treatments. *F* statistics and *P* values for comparisons within an effect are provided below the listed means.

Effect	CH_4 flux	N_2O flux
	μg CH_4–C $m^{-2} h^{-1}$	μg N_2O–N $m^{-2} h^{-1}$
Year†		
1	–	23.6 (1.7)
2	−26.5 (1.1)‡	26.3 (2.9)
3	−23.8 (0.8)	11.4 (1.5)
F statistic	1.7	6.5
P value	0.19	<0.01
Season		
Dec.–Feb.	−6.6 (1.2)	37.1 (5.1)
Mar.–May	−20.4 (1.3)	17.7 (1.5)
June–Aug.	−33.3 (0.9)	19.5 (1.2)
Sept.–Nov.	−35.5 (1.4)	12.1 (1.9)
F statistic	40.2	5.6
P value	<0.01	<0.01
Grazing treatment		
Crested wheatgrass	−23.8 (1.0)	34.2 (3.1)
Heavily grazed	−25.1 (1.1)	13.6 (1.2)
Moderately grazed	−26.4 (1.4)	14.5 (1.2)
F statistic	0.4	14.3
P value	0.65	<0.01

† Values for year effect determined using beginning and ending dates in mid-October.

‡ Values in parentheses reflect the standard error of the mean.

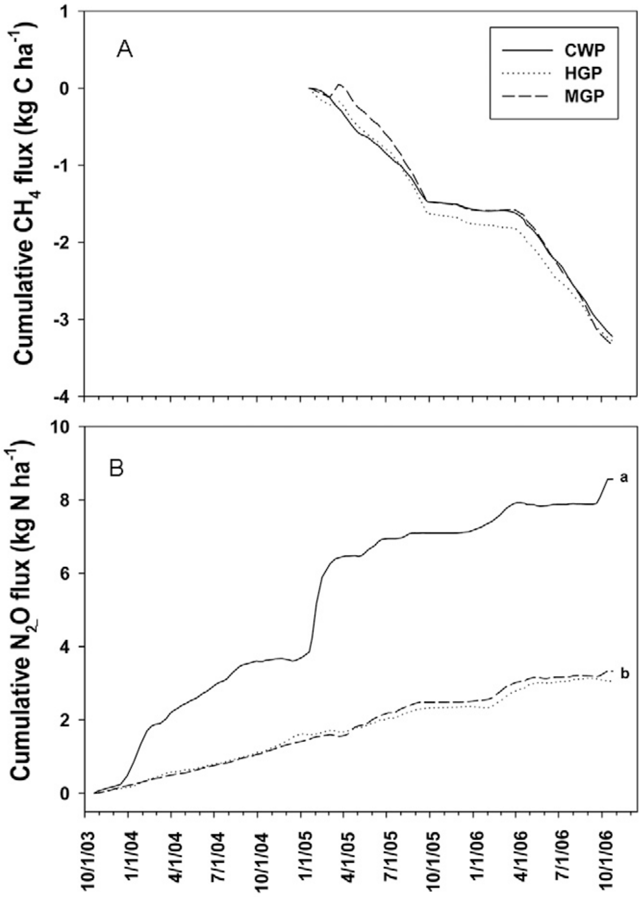

Fig. 4. (A) Cumulative flux CH_4 and (B) N_2O within crested wheatgrass pasture (CWP), heavily grazed pasture (HGP), and moderately grazed pasture (MGP) during the study period from October 2003 through November 2006, Mandan, ND. Different letters to the right of the cumulative flux values signify treatment differences at $P \leq 0.05$.

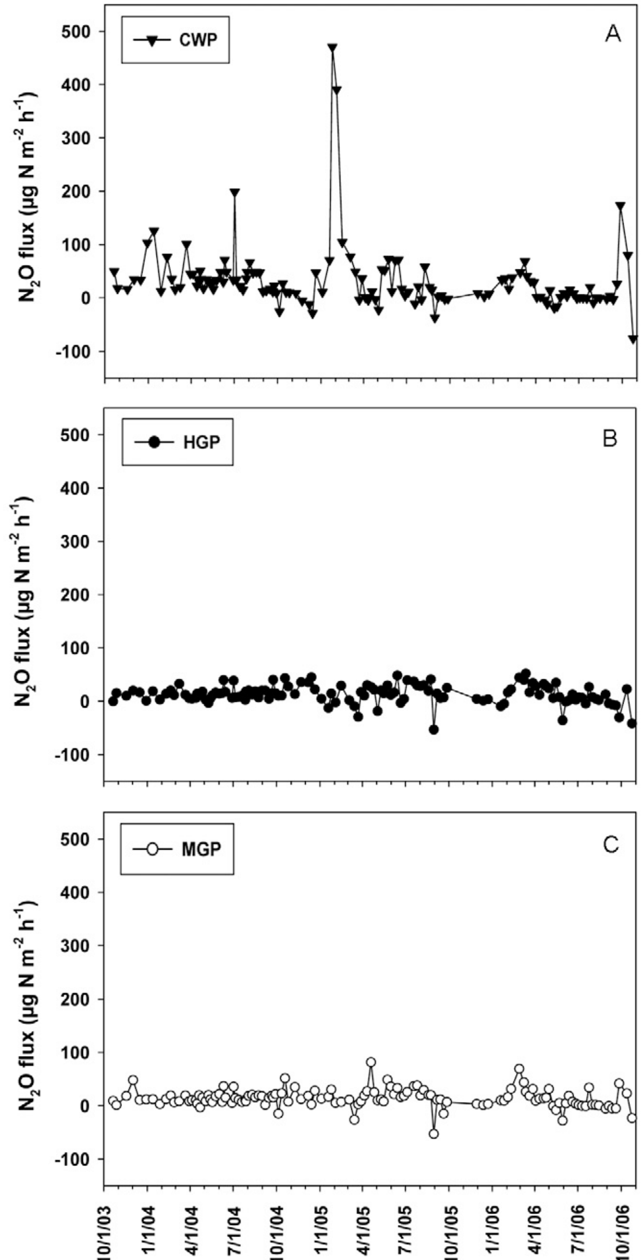

Fig. 5. Flux of N_2O for (A) crested wheatgrass pasture (CWP), (B) heavily grazed pasture (HGP), and (C) moderately grazed pasture (MGP) from 21 Oct. 2003 through 24 Oct. 2006, Mandan, ND.

mechanisms contributing to N_2O consumption. This phenomenon, however, merits further study.

Flux of N_2O was most variable in CWG, ranging from −75.8 µg N_2O–N m^{-2} h^{-1} (24 Oct. 2006) to 470.5 µg N_2O–N m^{-2} h^{-1} (25 Jan. 2005). Within the native vegetation pastures, N_2O flux ranged from −53.4 to 48.1 µg N_2O–N m^{-2} h^{-1} for HGP and −53.1 to 81.5 µg N_2O–N m^{-2} h^{-1} for MGP, with the lowest (negative) flux occurring on 30 Aug. 2005 and the highest flux occurring in spring 2005. Nitrous oxide flux was positively associated with WFPS ($r = 0.23$; $P \leq 0.01$; $n = 1343$), though (as indicated above) nearly all WFPS values in near-surface soil fell below 60% during the measurement period when the soil was not frozen. Soil temperature was weakly associated with N_2O flux over the measurement period ($r = 0.02$; $P \leq 0.01$; $n = 1343$).

Year, quarterly period, and grazing treatment affected N_2O flux (Table 4). Mean N_2O flux during Years 1 and 2 was more than twice that during Year 3, a result likely driven by limited microbial activity during the drought-stricken 2006 growing season. Among quarterly periods, N_2O flux was greatest during the winter months (37.1 µg N_2O–N m^{-2} h^{-1}), intermediate in spring and summer (range = 17.7–19.5 µg N_2O–N m^{-2} h^{-1}, and lowest in the fall (12.1 µg N_2O–N m^{-2} h^{-1}). Elevated N_2O flux during the winter was mainly the result of brief, episodic bursts within CWP during midwinter warming periods and/or spring thaw (Fig. 5). Elevated N_2O flux during the winter and early spring have been observed elsewhere in the NGP (Dusenbury et al., 2008; Lemke et al., 1998), and are thought to correspond to a burst of N_2O production from increased biological activity in near-surface soil depths (Wagner-Riddle et al., 2007).

Application of N fertilizer contributed to greater N_2O emission from CWP relative to HGP and MGP (Table 4). Hourly rates of N_2O flux from CWP were nearly 2.5 times greater than mean flux from HGP and MGP. Elevated emission of N_2O from CWP translated to greater cumulative emissions, with 8.6 ± 1.7 kg N_2O–N ha^{-1} emitted over the measurement period from CWP, and 3.1 ± 0.2 and 3.3 ± 0.5 kg N_2O–N ha^{-1} emitted from HGP and MGP, respectively ($P \leq 0.01$) (Fig. 4B). When calculated on an annual basis, N_2O emission from CWP, HGP, and MGP were 2.9 ± 0.6, 1.0 ± 0.1, and 1.1 ± 0.2 kg N_2O–N ha^{-1} yr^{-1}. Annual N_2O emission from the grazing treatments were greater than observations in the shortgrass steppe of northeast Colorado and fescue grasslands of Saskatchewan, but lower than rates observed in tallgrass prairie (Liebig et al., 2005).

Emission of N_2O from CWP as a percentage of N fertilizer applied was 4.1%, assuming the subtraction of background N_2O emissions attributable to livestock observed in HGP. Such an assumption is tenuous given the absence of a nonfertilized control for CWP. Assumptions notwithstanding, the value of the emission factor for CWP was larger than most previous evaluations within grassland ecosystems (Oenema et al., 1997). Elevated N_2O emission from CWP was likely due to a combination of factors. As a synthetic N source, NH_4NO_3 is rapidly soluble and easily metabolized by the microbial community. This fact, coupled with fall application of N, would favor N_2O production the subsequent winter and/or spring when the soil thawed. It is also important to acknowledge that N inputs from livestock were greatest in CWP among the grazing treatments due to a higher stocking rate early in the growing season. Nitrous oxide emission from pasture land has been shown to increase with increased N excretion by livestock (Bhandral et al., 2007; Flessa et al., 1996).

Net Global Warming Potential and Greenhouse Gas Intensity

Overall, factors contributing to net GWP decreased in relative impact in the order of (i) SOC change, (ii) soil–atmosphere N_2O flux, (iii) CH_4 emission from enteric fermentation, (iv) CO_2 emission associated with N fertilizer production and application, and (v) soil–atmosphere CH_4 flux (Table 5). Such a ranking among factors was anticipated, as SOC change and N_2O flux typically control net GWP in agroecosystems (Robertson et al., 2000). Individual factors differing in their contribution to net GWP across grazing treatments included N fertilizer and soil–atmosphere N_2O flux. Conversely, factors not different in their contribution to net GWP included CH_4 emission from enteric fermentation, soil–atmosphere CH_4 flux, and SOC change. Carbon dioxide emission associated with N fertilizer production was limited to CWP, as N fertilizer was not applied to HGP and MGP. Accordingly, N fertilizer application contributed to 2.7 times greater cumulative emission of N_2O from CWP compared with HGP and MGP. Consequently, N_2O emission from CWP negated a much larger percentage of CO_2 sequestered in soil (79%) relative to the native vegetation pastures (37 and 31% for MGP and HGP, respectively). Methane emission from enteric fermentation exhibited a similar effect on the negation of CO_2 sequestered by the grazing treatments. Within HGP and CWP, where higher stocking rates were employed, enteric fermentation negated approximately 33% of the CO_2 sequestered in soil, whereas only 12% of the CO_2 sequestered in MGP was negated by enteric fermentation.

Summing across factors, net GWP was negative for both HGP and MGP, implying net $CO_{2equiv.}$ uptake by both native vegetation grazing treatments (Table 5). Conversely, net GWP for CWP was positive and significantly different from zero (P = 0.03; data not shown), implying net $CO_{2equiv.}$ emission to the atmosphere. Mean net GWP differed between CWP and the native vegetation pastures by 1098 kg $CO_{2equiv.}$ ha^{-1} yr^{-1} ($P \leq 0.01$).

Table 5. Grazing management effects on net global warming potential (GWP) and greenhouse gas intensity (GHGI).

Variable	Crested wheatgrass	Heavily grazed	Moderately grazed
	——— kg $CO_{2equiv.}$ ha^{-1} yr^{-1} ———		
N fertilizer	259 (0)a†	0b	0b
Enteric fermentation	563 (227)	484 (76)	176 (28)
CH_4 flux	−61 (4)‡	−62 (6)	−63 (9)
N_2O flux	1336 (260)a	477 (39)b	520 (85)b
SOC change	−1700 (114)	−1517 (187)	−1416 (193)
Net GWP	397 (227)a	−618 (76)b	−783 (28)b
	——— kg $CO_{2equiv.}$ kg^{-1} animal gain ———		
GHGI	27 (17)b	−26 (9)b	−145 (38)a

† Values in parentheses reflect the standard error of the mean. Means in a row with unlike letters differ ($P \leq 0.05$).

‡ Negative values imply net CO_2 uptake.

Greenhouse gas intensity is similar to GWP in that negative values indicate a net sink of GHG to the soil, while positive values indicate a net source of GHG to the atmosphere (Mosier et al., 2006). Greenhouse gas intensity, however, expresses GHG results per unit of animal production, which for this study was kg $CO_{2equiv.}$ kg^{-1} animal gain. Both native vegetation pastures were found to possess a negative GHGI (MGP_{GHGI} = −145 ± 38 kg $CO_{2equiv.}$ kg^{-1} animal gain; HGP_{GHGI} = −26 ± 9 kg $CO_{2equiv.}$ kg^{-1} animal gain) (Table 5). However, GHGI of HGP was not significantly different from zero (P = 0.34; data not shown), nor did it differ from CWP, which was positive (27 ± 17 kg $CO_{2equiv.}$ kg^{-1} animal gain). Per unit of animal production, GHG benefits from HGP were reduced to a greater degree than from MGP given the significantly greater weight gain by livestock on the former (Table 2). Results for both GWP and GHGI suggest grazing management systems employing moderate stocking rates on native vegetation in the NGP are most effective at achieving net reductions in GHG emissions.

Summary and Conclusions

Within a given grassland ecosystem, climatic and management-related factors interact to influence GHG balance over a specified period of time. Assessing factors that contribute to GHG balance on annual and/or decadal timescales can help reduce uncertainties associated with identifying management strategies that minimize GWP. Given this context, we sought to estimate net GWP for three grazing management systems in the NGP using a multiyear data set of key factors that affect GHG balance.

We found all grazing treatments, representing long-term pastures of native vegetation and seeded crested wheatgrass, to be strong net sinks for SOC. Using data from archived and recently collected soil samples, SOC accrual rates for the grazing treatments were, on average, 0.10 to 0.17 Mg C ha^{-1} yr^{-1} greater than previous evaluations in the NGP. Under the dry and cold conditions of the evaluation, all grazing treatments were found be minor sinks for atmospheric CH_4 (mean

= −1.9 kg CH_4–C ha^{-1} yr^{-1}), and minor (native vegetation; 1.0 to 1.1 kg N_2O–N ha^{-1} yr^{-1}) or moderate (seeded crested wheatgrass; 2.9 kg N_2O–N ha^{-1} yr^{-1}) sources of N_2O. Relative to native vegetation, the nearly threefold greater annual N_2O emission from seeded crested wheatgrass was driven by greater N inputs from synthetic N fertilizer and excreted N from livestock. Methane uptake exhibited a strong seasonal dependence within the grazing treatments, with uptake rates greatest in the summer and fall and least in the winter. Conversely, N_2O flux in native vegetation was generally invariant to time of year, while significant N_2O emission events in January from seeded crested wheatgrass resulted in greater N_2O flux rates in winter than in other times of the year.

Summation of factors contributing to net GWP underscored the value of grazed native rangeland in the NGP to serve as a net $CO_{2equiv.}$ sink. Results from this evaluation further suggest grazing practices on native rangeland contributing to lower stocking rates confer greater GHG mitigation benefits based on calculations of GHGI. As with all temporally dependent research outcomes, continued monitoring of soil change and GHG flux will be necessary to ensure mitigation benefits are realized in the future.

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