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The role of hydropedologic vegetation zones in greenhouse gas emissions for agricultural wetland landscapes

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Abstract

Net greenhouse gas (GHG) source strength for agricultural wetland ecosystems in the Prairie Pothole Region (PPR) is currently unknown. In particular, information is lacking to constrain spatial variability associated with GHG emissions (CH_4, CO_2, and N_2O). GHG fluxes typically vary with edaphic, hydrologic, biologic, and climatic factors. In the PPR, characteristic wetland plant communities integrate hydropedologic factors and may explain some variability associated with trace gas fluxes at ecosystem and landscape scales. We addressed this question for replicate wetland basins located in central North Dakota stratified by hydropedologic vegetation zone on Jul 12 and Aug 3, 2003. Data were collected at the soil-atmosphere interface for six plant zones: deep marsh, shallow marsh, wet meadow, low prairie, pasture, and cropland. Controlling for soil moisture and temperature, CH_4 fluxes varied significantly with zone ($p<0.05$). Highest CH_4 emissions were found near the water in the deep marsh (277,800 μg m^{-2} d^{-1} CH_4), which declined with distance from water to −730 μg m^{-2} d^{-1} CH_4 in the pasture. Carbon dioxide fluxes also varied significantly with zone. Nitrous oxide variability was greater within zones than between zones, with no significant effects of zone, moisture, or temperature. Data were extrapolated for a 205.6 km^2 landscape using a previously developed synoptic classification for PPR plant communities. For this landscape, we found croplands contributed the greatest proportion to the net GHG source strength on Jul 12 (45,700 kg d^{-1} GHG-C equivalents) while deep marsh zones contributed the greatest proportion on Aug 3 (26,145 kg d^{-1} GHG-C equivalents). This was driven by a 30-fold reduction in cropland N_2O–N emissions between dates. The overall landscape average for each date, weighted by zone, was 462.4 kg km^{-2} d^{-1} GHG-C equivalents on Jul 12 and 314.3 kg km^{-2} d^{-1} GHG-C equivalents on Aug 3. Results suggest GHG fluxes vary with hydropedologic soil zone, particularly for CH_4, and provide initial estimates of net GHG emissions for heterogeneous agricultural wetland landscapes.

Keywords: Methane; Carbon dioxide; Nitrous oxide; Prairie Pothole Region

1. Introduction

The Missouri Coteau extends through the Dakotas and into central Canada, comprising over 750,000 km^2 of area within the Prairie Pothole Region (PPR). Approximately 12–15 wetlands populate each km^2 (Beeri and Phillips, 2007), for an estimated 9 million glacially-formed depressional wetland basins. Each wetland surrounds shallow water bodies, which play a major role in wetland pedogenesis through frequent seasonal expansion and contraction. Surface waters vacillate according to above-ground precipitation and evapotranspiration and by subsurface discharge and recharge, with the long-term areal extent of surface waters following patterns associated with regional drought and deluge (Winter, 2000; Winter, 2003; Beeri and Phillips, 2007). Hydroperiod drives a pedogenic gradient characteristic of areas surrounding surface waters in glacial basins. This gradient, in conjunction with hydroperiod, drives soil redox, chemical transport, and plant community distribution. Plant communities follow the gradient with specific canopy architecture and life-form (Stewart and Kantrud, 1971) along with soil pH, clay content, and water holding capacity (Phillips et al., 2005). The plant communities inhabiting these zones serve to integrate soil physical and biological properties and provide indicators of ecosystem physiography and eco-physiology (Stewart and Kantrud, 1972; Kantrud et al., 1989a,b).

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Both pedogenic and hydrologic factors exert proximal control on greenhouse gas (GHG) flux at the soil surface (Schimel et al., 1993; Conrad, 1989), especially for wetland ecosystems, which are primary sources of global atmospheric methane (CH_4) (Matthews and Fung, 1987). The naturally-occurring trace gases associated with global warming also include carbon dioxide (CO_2) and nitrous oxide (N_2O). Together, they comprise the net GHG flux term commonly referred to in the literature, which is calculated using the global warming potential (GWP) associated with each gas species on a mass basis. Methane fluxes at the soil-atmosphere interface vary seasonally and inter-annually for wetlands in Nova Scotia (Dalva and Moore, 2001), peatlands in Minnesota (Crill et al., 1988), Alaskan tundra (Whalen and Reeburgh, 1992), and for fens in northeastern Greenland (Friborg et al., 2000). Moreover, CH_4 flux spatial variability reported for these studies is high, often depending upon latitude (Fung et al., 1991), water table depth (Whalen and Reeburgh, 1992), soil moisture and temperature (Dalva and Moore, 2001), and thaw depth (Friborg et al., 2000). However, environmental factors alone (such as soil temperature) do not always adequately predict soil CH_4 fluxes for spatially diverse ecosystems, and may be explained more directly by an integrating factor (Whalen and Reeburgh, 1992) indicative of hydropedology. For Missouri Coteau prairie glacial basins, we suggest that plant community zones bordering surface waters represent a potential integrator of soil-plant-hydrologic relationships keystone to trace gas fluxes. Based on this premise, a primary goal of this research was to determine if GHG flux varies according to these hydropedologic vegetation zones. We aimed to determine sources of spatial variability in net GHG flux (and GHG-carbon) for the purpose of constraining uncertainties for agricultural wetland ecosystems first at the basin ecosystem scale.

Quantification of ecosystem trace gas exchange at the soil surface for Missouri Coteau prairie glacial basins is lacking, largely due to the paucity of data for prairie wetlands and by insufficient tools for meaningfully "scaling-up" trace gas exchange data across a diverse and dynamic ecoregion. Hydropedologic gradients in soil moisture, texture, temperature, and substrate supply within wetland basins impose a high degree of spatial variability, thereby eluding simple estimates for net GHG flux. In addition, the enormous number of vacillating surface waters over the region cannot reasonably be measured through use of ground surveys alone. Synoptic data, encompassing hundreds of km^2, are needed to constrain the GHG budget and support global carbon (C) models by identifying spatial variability in ecosystem structure potentially linked to relevant processes. These synoptic data must indicate key properties known to influence the net exchange of gaseous C to the atmosphere (calculated as global warming potential) on a wetland by wetland basis, since hydrologic variability at the basin scale exceeds variability at landscape scales (Beeri and Phillips, 2006). Explicit analyses of hydropedologic controls on GHG fluxes within prairie glacial basins or for Missouri Coteau landscapes has not been reported to date. We address the question of net GHG-C spatial variability first by determining how GHG fluxes vary among hydropedologic vegetation zones in a replicated, basin-scale experiment. Then, we employ recently developed remote sensing-based tools for mapping hydropedologic zones using vegetation spectral signatures to estimate GHG source strength for a Missouri Coteau landscape.

Explicit analysis of spatial heterogeneity associated with GHG-C fluxes is essential when extrapolating local flux data to regional scales (Bridgham et al., 1995). The areal extent of wetlands and factors associated with wetland soil variability must be known to extrapolate from field experiments to landscape or regional scales. We respond to the call for basic information regarding C fluxes in prairie wetlands and for the development of geospatial information tools (Roulet, 2000) with a geospatial model for estimating GHG emissions using the hydropedological gradient common to glacial basin ecosystems. The remote sensing component allows explicit treatment of anthropogenic activities in GHG emission calculations, a factor often absent from studies based on data from "natural" systems only. Additionally, Landsat-based observations of water level changes, a proximal control for CH_4 flux, can be fused with time-series data to approximate effects of water-level drawdown (Beeri and Phillips, 2007) on landscape GHG fluxes both in real-time and under future climate scenarios.

2. Methods

2.1. Research site

Three neighboring semi-permanent wetland basins of similar size, water permanence class (semi-permanent), and vegetation were selected (Fig. 1) near Max, ND, on the eastern edge of the Missouri Coteau in Ward County (47°53′N, 101°02′W). The terrain is characterized by glacial moraine, with hilly and irregular topography and deep glacial deposits (Bluemle, 1991), and wetlands surrounded by agricultural land-use. Natural upland vegetation is northern mixed grass prairie, or wheatgrass-needlegrass association (Barker and Whitman, 1988). Hydric soils in the wetland basins are classified as fine, smectitic, frigid Vertic Argiaquolls. Upland soils are classified as fine-loamy, superactive, frigid Typic Haplustolls mixed with Typic Argiustolls (USDA, 1974). Average precipitation is 38 cm, most of which occurs during the growing season (April–Sept). Temperatures from April through September range from −3 to 38 °C (North Dakota State Agricultural Weather Network, 2000). The site has been consistently owned and managed by the same individual for over 40 years, where uplands are planted to small grains and pastures are used for cattle grazing.

Concentric bands of specific plant communities, which vary predictably with distance from open water, comprise the hydropedologic vegetation zones found within these gently sloping depressional wetland basins (Stewart and Kantrud, 1971). They are referred to as deep marsh, shallow marsh, wet meadow, and low prairie plant communities, and are bordered by croplands or pastures. The deep marsh zone is nearest to open water, where soils are often saturated. Moving away from the open water past the deep marsh is the shallow marsh zone, where soils are normally saturated from spring to early-summer. Further from the open water is the wet meadow, where water loss from bottom seepage is fairly rapid and soils are saturated only a few weeks after

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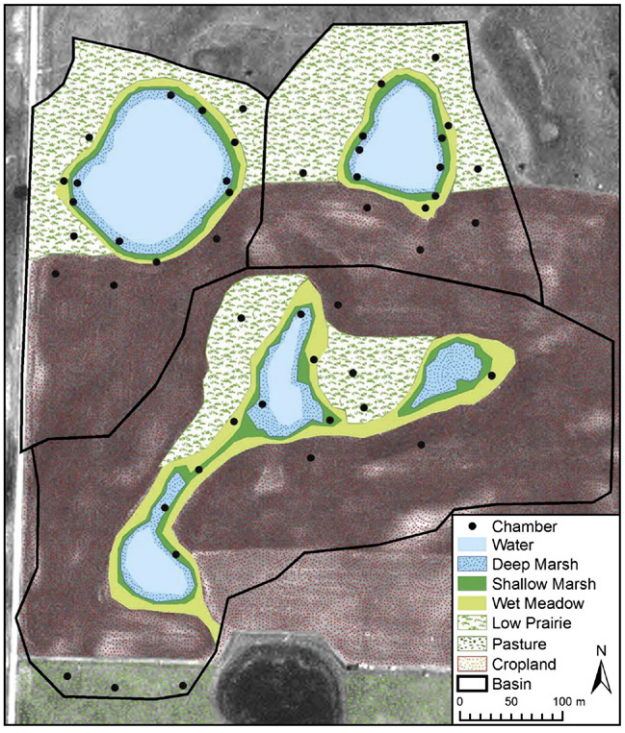

Fig. 1. Research site, located near Max, North Dakota. Three depressional wetland basins were surveyed at random points (black dots) stratified by hydropedologic zone.

snowmelt. Upland from the wet meadow is the low prairie zone, where soils are no longer hydric. Crop fields bordering the low prairie were planted to wheat and fertilized with urea (70 kg ha^{-1}) in May 2003 using no-till agricultural practices. Within one basin is a pasture grazed by cattle every summer. Dominant plant species found for deep marsh, shallow marsh, wet meadow, and low prairie zones were similar for all wetlands surveyed (Phillips et al., 2005).

Deep marsh, shallow marsh, wet meadow, and low prairie zones were mapped according to plant community distribution using a corrected, real-time differential Trimble Geo XT GPS Beacon receiver with an external antenna (Trimble Navigation, Sunnyvale, CA) to achieve sub-meter locational accuracy. The borders of each gently sloping depressional basin (represented by the elevation high points surrounding each wetland) were mapped similarly in addition to the area of cropland and pastureland within each basin.

2.2. Experimental design

Gas fluxes were measured using the static chamber method (Whalen and Reeburgh, 1992) Jul 12 and Aug 3 2003, when air and surface soil temperatures (10 cm depth) were near the annual maximum, recorded on Jul 12 for air temperature (38 °C) and on Jul 24 for soil temperature (31 °C) (NDAWN). The wheat crop flowering was complete on Jul 12 and was in the soft dough stage on Aug 3. To test the effect of hydropedologic zone on GHG fluxes, three points were selected within each zone in each of the three wetland basins using ERDAS Imagine 8.7 (Leica Geosystems GIS & Mapping LLC, Norcross, GA.) randomization procedure (Fig. 1). On Jul 2 2003, polyvinyl chloride collars (20-cm diameter × 11-cm height) were deployed in the soil ~7 cm deep at each random point. Collars were placed between rows of wheat and between the tall, emergent vegetation in the deep marsh. Plant heights did not exceed chamber heights for the remaining zones, so these collar footprints included plants respiring CO_2. Soils in the deep marsh chamber sites were nearly saturated but did not hold standing water. Initial flux measurements were collected 10 d later on Jul 12 and then again on Aug 3 2003. Measurements commenced immediately after placing polyvinyl chloride covers fitted with butyl O-rings onto the soil collars. Covers included a capillary bleed to equalize pressure and an O-seal fitting and septa for syringe sampling. Samples of the headspace gas were withdrawn from each chamber at 0.25-h intervals during a 0.75-h time course beginning at 1300 h. All zones were sampled on both dates, with the exception of the wet meadow, which was sampled only on 3 Aug due to equipment failure.

Sample aliquots were immediately injected into 15-ml exetainers (Labco Unlimited, Buckinghamshire, UK) and analyzed for CH_4, CO_2, and N_2O 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 either 1) a flame-ionization detector with ultra-pure He carrier gas or 2) a thermal conductivity detector with ultra-pure He carrier, or 3) a ^{63}Ni electron-capture detector with ultra-pure 95% Argon/5% CH_4 carrier gas. The gas chromatograph was calibrated with a commercial blend of CO_2, CH_4, and N_2O balanced in N_2 (Scott Specialty Gases) following verification of stated concentrations with standards from the National Institute of Standards and Technology. The precision of analysis expressed as a coefficient of variation for 10 replicate injections was less than 2%. The minimum detectable concentration change was ±5 μL L^{-1} for CH_4, ±8 μL L^{-1} for CO_2, and ±4 nL L^{-1} for N_2O. Gas samples were stored <12 h prior to analysis, and tests showed no change in CH_4, CO_2, and N_2O concentration during storage.

Air temperature was measured near the soil surface during flux measurements, and soil temperature was measured at 3-cm depth increments with a multithermistor probe to 15 cm and average temperature for the 0–15 cm depth interval reported. One soil core (2-cm diameter × 15-cm depth) was extracted from each of the 45 points for determination of soil moisture and C, and one soil core (5-cm diameter × 15 cm depth) was extracted from each basin and zone for determination of soil bulk density. Cores were kept in cold storage (−4 °C) prior to laboratory analysis. Soil moisture was measured gravimetrically (oven dried at 105 °C). Bulk density was computed as the quotient of oven dried mass divided by field volume. Cores were analyzed for total-C by dry combustion (Carlo Erba NA 1500 Elemental Analyzer). Ground soil was used to measure inorganic C on soils after application of dilute hydrochloric acid stabilized with $FeCl_2$ by measuring the amount of CO_2 produced by gas chromatography (Loeppert and Suarez, 1996). Soil organic C was calculated as the difference between total C and inorganic C. Soil organic and inorganic C were normalized by soil weight, due to differences in bulk density.

2.3. Data analysis

We tested for the effect of zone on fluxes of CO_2, CH_4, N_2O, and the net GHG flux [Net $GHG_{flux} = CO_2 + 23(CH_{4flux}) + 296$

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Table 1
Area of interest (AOI) geographical classification and contribution of GHG-C for each zone, based on average GHG-C emissions measured for each zone

Class	Area (km^2) Jul	Area (km^2) Aug	Vegetated landscape prop. (%) Jul	Vegetated landscape prop. (%) Aug	Flux GHG-C equiv. (kg km^{-2} d^{-1}) Jul	Flux GHG-C equiv. (kg km^{-2} d^{-1}) Aug	Total GHG-C equiv. (kg d^{-1}) Jul	Total GHG-C equiv. (kg d^{-1}) Aug
Deep Marsh	10.1	13.9	4.9	6.7	1778.4	1886.8	17,900	26,145
Shallow Marsh	3.6	3.6	1.8	1.8	571.9	735.8	2,076	2,671
Wet Meadow	5.6	5.6	2.7	2.7	–	325.4	–	1,804
Low Prairie	14.0	14.0	6.8	6.8	389.2	186.6	5,432	2,604
Water	14.3	10.5	7.0	5.1	–	–	–	–
Pasture	60.1	60.1	29.2	29.2	398.9	390.6	23,966	23,467
Crop	98.0	98.0	47.7	47.7	466.1	80.9	45,700	7,934
Total			100	100				
Vegetated landscape total	205.6	205.6					95,075[a]	64,625

[a] Not including wet meadow.

(N_2O_{flux})], as defined by the Intergovernmental Panel on Climate Change (Houghton et al., 1997). We used a mixed, repeated measures analyses of variance (Littell et al., 1996) in SAS (SAS Institute, Cary, NC) to test for the effect of zone for each gas separately. Included in the model were random effects of wetland basin, with soil temperature, soil moisture, and interactions (temperature * zone, moisture * zone, temperature * moisture, temperature * zone * moisture) as covariates. Only significant interactions were retained in the model. Data for CH_4, CO_2, N_2O and net GHG fluxes were natural log transformed to achieve normality prior to analysis. We used a first-order autoregressive structure to properly model trace gas flux variances that decline exponentially with distance. Data for each trace gas and net GHG flux [CO_2 equivalents (equiv.)] are reported as fluxes by sample date (g m^{-2} d^{-1}). Each gas is also reported in units of CO_2–C equiv. using the GWP coefficients referenced above in the GHG_{flux} equation. Soil organic C and inorganic C were also analyzed for differences among zones with a mixed analysis of variance, which included the random effects of wetland basin.

Large-scale delineation of hydropedologic zones were performed using cloud-free images acquired by the SPOT satellite sensor on 7 Aug 2003 and by the Landsat 5 sensor on Jul 13 and Aug 14 2003. The 13.30 × 16.07 km scene was centered at 47°53′ N, 101°07′ W and entirely within the Missouri Coteau ecoregion. We used ERDAS Imagine to geo-reference and calibrate each image to UTM, Zone 14 projection and WGS 84 Datum with ground-control points. We converted the Landsat 5 pixel digital numbers to ground reflectance by using MODTRAN-4 radiative transfer code inside the ERDAS Imagine ATCOR 2 package and the specific calibration files for each band (Chandler and Markham, 2003). The SPOT digital numbers were converted to ground reflectance using the empirical line method (Moran et al., 2001).

Our 213.7 km^2 landscape area-of-interest (AOI) was classified according to agricultural land-use and wetland plant community zones using a Missouri Coteau vegetation-based spectral model (Phillips et al., 2005) with the SPOT data. Surface waters were classified on Jul 13 and Aug 14 using the surface water detection model developed for the Prairie Pothole Region (Beeri and Phillips, 2007) and Landsat 5 data. Change detection between dates indicated the areal extent of water drawdown. Hydropedologic vegetation zone areas were calculated to report proportion of each across the landscape AOI. Net GHG flux for the landscape was calculated by multiplying average net GHG flux measured for each zone by the unit area for the entire image on each date.

3. Results

Wetland water levels were in the drawdown stage between Jul and Aug, where surface water area declined 3.8 km^2 or 2% of our

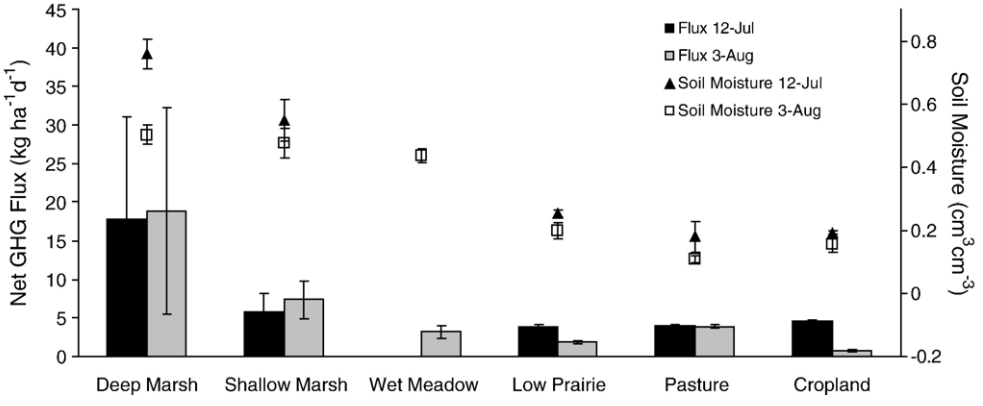

Fig. 2. Average (±standard error of the mean; number of samples = 9) net flux GHG-C and soil moisture content (0–15 cm depth) by zone for each date. Actual means are reported in this figure; data were natural log transformed to achieve normality prior to statistical analyses.

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Table 2
Average (±standard error of the mean) flux by zone for CH_4, CO_2, and N_2O–N on July 12 and on August 3

Zone	Flux CH_4 ($\mu g\ m^{-2}\ d^{-1}$)		Flux CO_2 ($mg\ m^{-2}\ d^{-1}$)		Flux N_2O–N ($\mu g\ m^{-2}\ d^{-1}$)	
	Mean	s.e.m.	Mean	s.e.m.	Mean	s.e.m.
12 Jul 2003						
Deep Marsh	277760	245390	131	46	10	20
Shallow Marsh	43960	12300	1077	179	20	20
Wet Meadow	–	–	–	–	–	–
Low Prairie	200	670	1313	65	230	70
Pasture	−730	530	1447	64	70	40
Cropland	−110	80	722	84	2080	1090
3 Aug 2003						
Deep Marsh	241060	209060	760	271	1290	790
Shallow Marsh	48550	31610	1248	223	700	480
Wet Meadow	160	200	856	65	700	550
Low Prairie	−420	90	676	102	40	30
Pasture	−2110	500	1413	369	140	10
Cropland	30	150	261	49	70	10

Data reported here for each gas were natural log transformed prior to statistical analysis.

Table 3
Statistical results for individual analyses of variance for CH_4, CO_2, N_2O–N fluxes

Fixed effect	Num	Den	Flux CH_4		Flux CO_2		Flux N_2O–N	
	DF	*DF*	*F*	Pr>*F*	*F*	Pr>*F*	*F*	Pr>*F*
Zone	1	74	3.12	**0.01**	11.01	**0.01**	1.09	0.37
Soil moisture	5	74	2.84	0.10	7.55	**0.01**	0.47	0.49
Soil temp	1	74	0.12	0.73	4.58	**0.05**	0.26	0.61

Data for each gas were natural log transformed to achieve normality prior to analysis.

landscape AOI (Table 1). Soil moisture (0–15 cm depth) for the three basins declined between Jul 12 and Aug 3 (Fig. 2), especially for the deep marsh zone, where the soil moisture average fell from 0.76 to 0.50 $cm^3\ cm^{-3}$. Air temperature during flux collection was 31 °C and 30 °C, respectively, on Jul 12 and Aug 3. Average soil temperature was 20 °C on Jul 12 and 23 °C on Aug 3. Potential evapotranspiration recorded at the Karlsruhe, ND weather station was 0.91 and 0.71 cm d^{-1}, respectively, for Jul 12 and 3 Aug 3 (North Dakota State Agricultural Weather Network, 2000).

Methane, CO_2, and N_2O–N fluxes were highly variable (Table 2), but spatial patterns associated with hydropedologic zones did emerge from the dataset (Fig. 3; Table 3). Both CH_4 and CO_2 emissions varied significantly with hydropedologic vegetation zone ($p<0.01$), with some variability in CO_2 likely contributed by plant respiration in the shallow marsh, wet meadow, low prairie, and pasture zones. In addition, soil moisture and temperature were significant covariates for CO_2 ($p<0.05$), and these effects were consistent among zones (Table 3; no interactions with zone). For N_2O–N, we found no evidence that zone, moisture, or temperature alters N_2O–N flux for these systems. Although average [±standard error of the mean (s.e.m.)] cropland N_2O–N emissions ranked highest Jul 12 (2080 $\mu g\ m^{-2}\ d^{-1}$ N_2O–N± 1080), N_2O–N emissions in the deep marsh were highest on Aug 3 (1290 $\mu g\ m^{-2}\ d^{-1}$ N_2O–N±790). Low N_2O–N fluxes for remaining zones suggest soil N may have been limiting. We did not assay soil N, but we found dissolved N in surface waters to be very low (nitrate was below the detection limit and ammonium was les than 0.1 $\mu L\ L^{-1}$), suggesting N availability may not be high at this site (unpublished data). The effect of zone, moisture, and temperature on the combined gaseous CO_2 emission, expressed as net GHG flux (CO_2 equivalents), was similar to N_2O–N. We found no significant effect of zone, moisture, or temperature (Table 3), although net GHG flux appeared to increase along the hydric soil gradient, from low prairie to deep marsh (Fig. 2). This gradient in net GHG flux was largely due to CH_4 emissions nearest to the open water, where the average (±s.e.m.) contribution for both dates of CH_4 to net GHG flux was 16.3 (±9.8) kg $ha^{-1}\ d^{-1}$ CH_4–C equivalents (Fig. 3). The contribution of CH_4 to net GHG flux dropped considerably in the shallow marsh and wet meadow zones, where overall averages (both date combined) were 2.9 (±1.0) and 0.01 (±0.01) kg $ha^{-1}\ d^{-1}$ CH_4–C equiv., respectively. Low prairie, cropland, and pastureland soils consumed CH_4, where overall average fluxes were −0.01 (±0.02), −0.01 (±0.01), and −0.12 (±0.05) kg $ha^{-1}\ d^{-1}$ CH_4–C equiv., respectively (Fig. 3).

Net GHG-C flux for these ecosystems was dominated by the deep marsh (Fig. 2). Combined shallow marsh, wet meadow, low

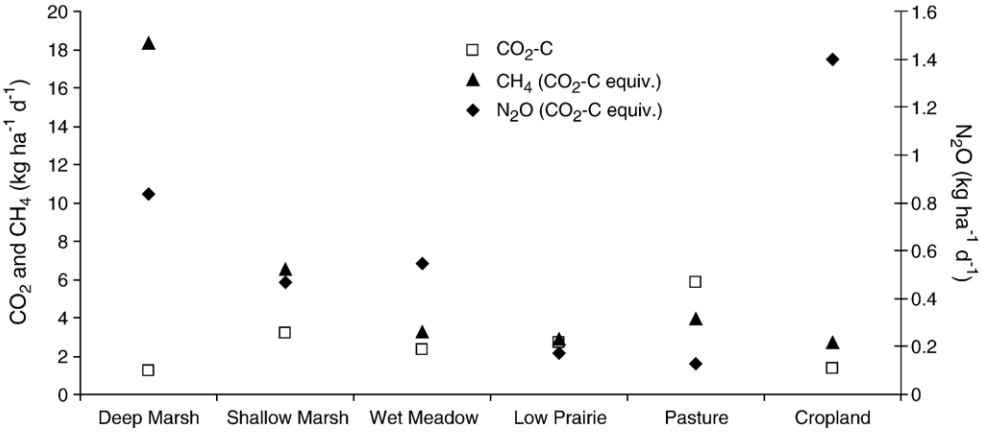

Fig. 3. Contribution of CO_2, CH_4, and N_2O to net GHG flux, including the global warming factor, average by zone for both dates combined.

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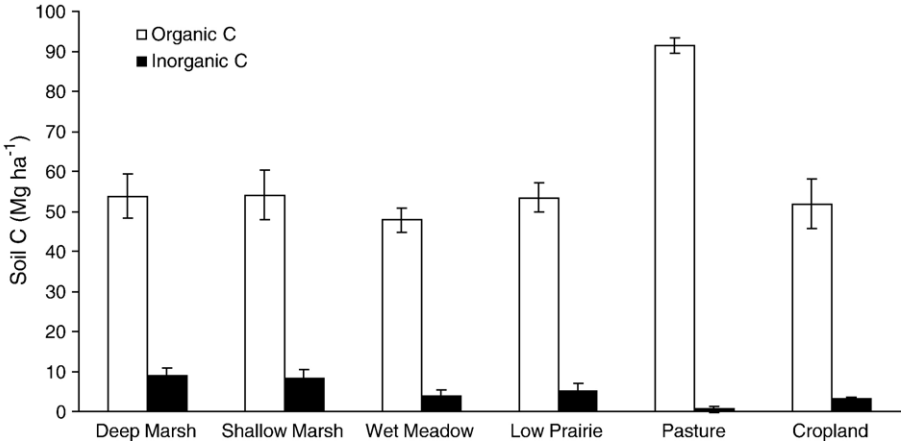

Fig. 4. Average (±standard error of the mean; number of samples=9) soil organic C and soil inorganic C (0–15 cm depth) for both dates and all zones.

prairie, cropland and pastureland fluxes contributed 52% of the net GHG flux, compared with 48% for deep marsh. The overall average contribution of CO_2 to net GHG-C flux by zone ranged from 1.2 (±0.4) kg ha^{-1} d^{-1} CO_2–C equiv. in the deep marsh to 5.9 (±0.5) kg ha^{-1} d^{-1} CO_2–C equiv. in the pasture (Fig. 3). The overall average contribution of N_2O to net GHG flux ranged from 0.1 (±0.03) kg ha^{-1} d^{-1} CO_2–C equiv. in the pasture to 1.4 (±1.1) kg ha^{-1} d^{-1} CO_2–C equiv. in the cropland (Fig. 3). All measured trace gases contributed to GHG flux but was dominated by CH_4.

While average soil organic C (0–15 cm depth) for deep marsh, shallow marsh, low prairie, and annual cropland zones ranged between 47.8 (±2.2) and 54.1 (±6.1) Mg ha^{-1}, organic C for the pasture averaged 91.5 (±7.6) Mg ha^{-1} (Fig. 4). Pasture soils contributed most strongly to significant differences found among zones for organic C ($F_{5,\ 84}=5.6$; $p<0.01$). Inorganic soil C was lower than organic C and ranged from 8.9 (±1.9) Mg ha^{-1} in the deep marsh soils to 0.6 (±0.4) Mg ha^{-1} in the pasture. Inorganic C contributed less than 10% to total soil C, with clear differences between marsh and upland soils ($F_{5,\ 84}=2.9$; $p<0.05$).

Our landscape AOI is primarily used for crop production (46%) and livestock grazing (28%). Fig. 5 illustrates a subset of the AOI. Anthropogenic materials (roads, gravel mines, houses) comprise less than 4% of the area and are not included in calculations for the vegetated landscape. The remaining 22% is comprised of wetlands and water bodies. Surface water area is included in the total landscape evaluation, since the area covered by marsh vegetation expands as the area of open water contracts between Jul and Aug (Table 1).

Extrapolated across the vegetated landscape, total net GHG-C flux was 95,075 kg d^{-1} CO_2–C equiv on Jul 12 and 64,625 kg d^{-1} CO_2–C equiv. on Aug 3 (Table 1), although Jul 12 net GHG-C flux does not include 3.6 km^2 of wet meadow vegetation. Differences between dates for each class were largely driven by cropland declines in N_2O–N emissions (Table 2, Fig. 2). N-fertilization is known to enhance N_2O emissions (Mosier et al., 1998), so the effect of fertilization likely diminished by 3 Aug, indicated by N_2O–N flux dropping from 2080 to 70 µg m^{-2} d^{-1} (Table 2). Alternatively, the encroachment of

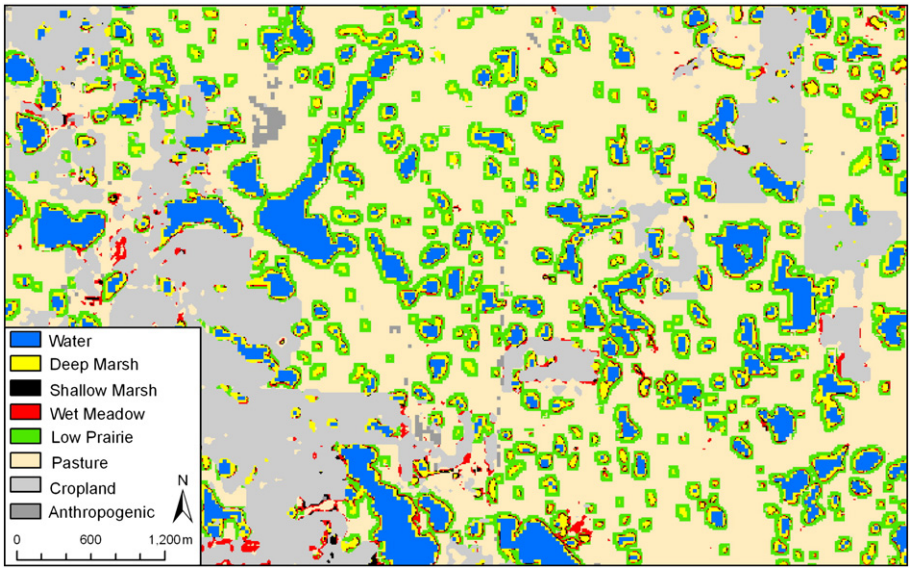

Fig. 5. Subset of the 205.6 km^2 landscape classification, where all zones were remotely delineated using SPOT V and Landsat 5 sensors in Jul and Aug 2003.

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deep marsh vegetation into receding waters between Jul 12 and Aug 3 increased the landscape area of deep marsh, where CH_4 emissions are greatest. While the contribution by the deep marsh to net GHG-C flux ranks highest on m^{-2} basis (Fig. 2), the large areal extent of croplands and pastures (Fig. 5) contributed the most gaseous C to the atmosphere for the entire landscape scale, especially on Jul 12 (Table 1).

Net GHG-C flux for this 205.6 km^{-2} vegetated landscape represents an average weighted by zone. When divided by this total area, the Jul 12 average is 462.4 kg km^{-2} d^{-1} GHG-C and the Aug 3 average is 314.3 kg km^{-2} d^{-1} GHG-C. Without taking multiple zones and land uses into account with the landscape classification tool, the average of all zones together would overestimate GHG flux for this agricultural wetland landscape. For example, taking an average for data collected for a given date and extrapolating this average across the landscape (effectively ignoring land area differences in the distribution hydropedologic zones) GHG-C emissions would have been overestimated by 50%. Under this scenario, landscape GHG-C would total 152,968 kg d^{-1} GHG-C equiv. on Jul 12 and 124,414 kg GHG-C equiv on Aug 3, compared to our reported 95,075 kg d^{-1} on Jul 12 and 64,625 kg d^{-1} on Aug 3 (Table 1).

4. Discussion

On a basin scale, CH_4 emissions were greatest in the deep marsh and contributed 48% to the net GHG flux (Fig. 3). Methane and GHG fluxes among zones tended to parallel soil water content differences (Fig. 2), following the environmental gradient common to prairie glacial basins (Fig. 1). High CH_4 fluxes have been observed near other shallow, open, water bodies (Roulet, 2000), where warm sediment temperatures, labile organic matter, and anaerobic conditions enhance CH_4 production. Methane emissions are often correlated with net primary production (NPP) (Whiting and Chanton, 1993), and biomass tends to be greater where water tables are higher (Dwire et al., 2004), as found in the deep marsh zone. Not only do CH_4 emissions tend to follow productivity gradients, which tend to co-vary with distance from water, but they also follow gradients in soil redox (Conrad, 1989). High moisture in the deep marsh would reduce soil redox potential and enhance methanogenesis, compared to less hydric soils. The water level drawdown between Jul 12 and Aug 3 did not strongly affect CH_4 fluxes for the marsh zones, suggesting soil redox remained low enough to sustain CH_4 production. Despite anaerobic conditions in marsh soils, we did not find evidence of greater SOC, suggesting long-term rates of C sequestration may be similar among zones, despite variable soil redox.

Net CH_4 flux includes both CH_4 production and CH_4 consumption (Conrad, 1989; Schimel et al., 1993), and we do not differentiate between the two processes with our net flux measurements collected at the soil-atmosphere interface. However, negative CH_4 fluxes occurred in the more aerated, upland soil zones (Table 2). In the drier pasture and low prairie zones, where soil moisture is <0.2 cm^3 cm^{-3}, atmospheric CH_4 diffused into soil pore spaces and was oxidized by methanotrophic organisms (Mosier et al., 1991). While methanogenesis may have also contributed to CH_4 fluxes, net CH_4 consumption at the soil surface suggests rates of consumption exceeded rates of production (Phillips et al., 2000). Conversely for the more saturated soil zones, methanotrophy may also occur; however, positive CH_4 fluxes suggest methanogenesis is predominate for the hydric soil zones.

Zonation patterns for SOC were not evident in this study and SOC was not correlated with CH_4 flux. Data for SOC in the PPR are scarce, but reports for grazed rangelands in ND indicate values from 50–60 Mg SOC ha^{-1} (Liebig et al., 2006), which are similar to values reported here (Fig. 4). Soil organic C was considerably higher for pasture soils than expected. However, areas with greater SOC correlated with higher CO_2 emissions (Fig. 3), in agreement with studies pointing to higher soil C fueling higher rates of microbial and soil respiration (Zak et al., 2000; Phillips et al., 2002). In the deep marsh, low CO_2 emissions also suggest soil redox was low and conditions were anaerobic. The gradient in CO_2 flux is sharp and distinctive on Jul 12, when differences in soil moisture between deep marsh and shallow marsh were nearly 10 cm^3 cm^{-3}. The drawdown in August soil water was accompanied by greater CO_2 emissions for the deep marsh zone, which is also evident by the significant effect of moisture on CO_2 flux (Table 3). The significance of soil moisture and temperature on CO_2 emissions is well-established; however, hydropedologic vegetation zone also contributed to basin-scale spatial variability for CO_2.

Temporal variability associated with N_2O production in soils often a function of soil N availability and soil redox status. Under highly anaerobic conditions, N_2O is often reduced completely to N_2 (Davidson, 1991). Deep marsh emissions on Jul 12 increased on Aug 3 by two orders of magnitude, likely because lower soil moisture increased the redox potential of the soil, thereby inhibiting the complete reduction of N_2O to N_2. For croplands, N_2O emissions commonly decline with time since fertilization, usually within the first 6–8 weeks (Wagner-Riddle et al., 1997; Venterea et al., 2005). Consequently, the thirty-fold reduction in N_2O–N flux 10 weeks after fertilization was expected. Although soil hydric status and available soil N are key factors, other elements, such as soil C and clay content, also contribute to N_2O variability. These data suggest croplands are a major source of N_2O–N and that future work should focus more intensively on temporal variability associated with cropland emissions, as noted for other regions (Wagner-Riddle et al., 1997; Mosier et al., 1998; Venterea et al., 2005).

These results must be interpreted based as an analysis of spatial (rather than temporal) variability and at a basin scale (rather than pedon scales). In other studies, predictive models based on soil temperature, thaw depth, and the ratio of CH_4 to net primary production (NPP) effectively estimated CH_4 fluxes with a high degree of certainty for ecosystem exchange at the canopy level (Friborg et al., 2000). In a study more similar to ours, plant production and plant community type was highly predictive of CH_4 spatial variability (Christensen et al., 2000). Fluxes of over 300 mg m^{-2} d^{-1} CH_4 were reported for plant communities inhabiting the hollows of northeast Greenland, compared to less than 100 mg CH_4 m^{-2} d^{-1} in the hummocks. Whalen and Reeburgh (1992) found that tussock and *Carex* sites emitted more

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CH_4 than mossy areas in the Alaskan tundra. While environmentally-based models for CH_4 flux are useful for predicting temporal variability, Christensen et al., Whalen and Reeburgh, and this study suggest integrative factors influencing spatial variability should be included to properly weight landscape-level CH_4 fluxes.

5. Conclusion

Using plant community spectral signatures as indicators of the hydropedologic gradient, we determined spatial variability in GHG source strength for a glacial basin agricultural wetland landscape. The data suggest gradients of vegetation surrounding surface waters parallel gradients in CH_4 emissions. Hydric soils were CH_4 sources and upland soils were CH_4 sinks. Using remote sensing-based tools, total net GHG source strength for the landscape (205.6 km^2) was estimated on Jul 12 (95,075 kg d^{-1}) and on Aug 3 (64,625 kg d^{-1}) for a landscape-weighted average of 462.4 mg m^{-2} d^{-1} CO_2–C on Jul 12 and 314.3 mg m^{-2} d^{-1} CO_2–C on Aug 3. Additional data for basins under different management scenarios are needed to appropriately "scale up" to the Missouri Coteau ecoregion. Here, we demonstrate the importance of hydropedologic vegetation zone to CH_4 fluxes, which ultimately influences net GHG flux and soil C sequestration. Moreover, we demonstrate the combination of bottom-up GHG research and top-down satellite tool application to address questions of ecosystem function as it relates to hydropedology at landscape scales. Future research should include productivity measurements, time series data, and a detailed investigation of hydropedologic processes contributing to GHG fluxes within depressional basin zones.

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