Chapter 2
Greenhouse Gas Fluxes from Restored Agricultural Wetlands and Natural Wetlands, Northwestern Indiana

Brianna Richards and Christopher B. Craft

Abstract We measured gas fluxes and production efficiency rates of CO_2, CH_4, and N_2O from natural and restored freshwater marshes in northwestern Indiana to evaluate the contribution of restored wetlands to regional greenhouse gas fluxes. Anaerobic soil incubations were used to determine production efficiencies, and static flux chamber measurements were used to measure fluxes during the growing season. Restored wetlands contained less soil organic carbon (1.5 % versus 6.3 %) than natural wetlands yet emitted comparable greenhouse gases in anaerobic incubations. Production efficiency rates, though, were significantly higher in restored wetlands. Mean growing season fluxes from static flux chamber measurements were 10.1 kg CO_2-C ha^{-1} day^{-1}, −0.2 g CH_4-C ha^{-1} day^{-1}, and 0.6 g N_2O-N ha^{-1} day^{-1} from natural wetlands and 3.8 kg CO_2-C ha^{-1} day^{-1}, 0.1 g CH_4-C ha^{-1} day^{-1}, and 0.4 g N_2O-N ha^{-1} day^{-1} from restored wetlands and did not differ among the two wetland types. We conclude that the ecological benefits gained from restoring wetlands in the glaciated interior plains outweigh the negative impact of their greenhouse gas contribution.

Keywords Gas flux • Methane • Restored wetlands • Natural marsh

2.1 Introduction

Freshwater wetlands are known sources of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) (Bartlett and Harriss 1993), but their individual and cumulative contribution to global warming potential is poorly quantified (Bridgham et al. 2006). High variability exists in GHG fluxes between and within created and restored freshwater wetlands (Gleason et al. 2009; Sha et al. 2011), leading to difficulty in modeling landscape-scale fluxes. This difficulty is compounded by a

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scarcity of data on GHG emissions from restored wetlands (Brinson and Eckles 2011).

We measured GHG (CO_2, CH_4, and N_2O) emissions from four restored wetlands and four natural wetlands using anaerobic incubations and static flux chambers to better understand their relative contribution to local and regional fluxes. We hypothesized that because restored wetlands often contain less soil organic matter (SOM) than natural wetlands (Bruland and Richardson 2005), they would exhibit lower fluxes of GHG. Furthermore, many restored wetlands have shorter hydroperiods and thus more aerobic soil conditions than natural sites (Knutsen and Euliss 2001) that may lead to lower CH_4 fluxes from restored sites.

2.2 Methods

2.2.1 Site Description

Four natural wetlands and four restored wetlands located in Newton County, northwestern Indiana, were selected for sampling (Fig. 2.1). The natural wetlands were located in Willow Slough Fish and Wildlife Area, a publicly owned area maintained by the Indiana Department of Natural Resources (DNR). Willow Slough, which encompasses 4,030 ha of land that was mostly used for agriculture in the past, was purchased by the state of Indiana between 1949 and 1951 (IN DNR 2011).

The restored wetlands were embedded in a restored prairie and oak-savanna landscape and were located in the 3,160 ha Kankakee Sands Efroymson Family Prairie Restoration owned by The Nature Conservancy (TNC). Prior to restoration, the land was a lake bed that was drained in the late 1800s for row crop agriculture (USFWS 1999). In 1999 and 2001, TNC filled in drainage ditches with sediment and regraded topography to restore wetlands as part of the Wetlands Reserve Program (C. O'Leary pers. comm.). The restored wetlands have been actively managed using a variety of methods including prescribed burns, woody and invasive species removal, and reintroduction of native species. Sites are burned approximately every three years. Undesirable plant species, such as *Equisetum hyemale* (L.), *Populus deltoides*, and *Salix nigra*, are controlled by mechanical removal and through the application of herbicides through backpack spraying, boom spraying, and basal bark treatments. Native grasses and forbs are introduced through annual seeding and plantings.

The natural wetlands are dominated by *Calamagrostis canadensis* (Michx.), *Boehmeria cylindrica* (L.), and *Scirpus cyperinus* (L.) (Hopple and Craft 2013). Dominant species in the restored wetlands include *Schoenoplectus pungens* (Vahl), *Scirpus cyperinus* (L.), *Leersia oryzoides* (L.), and *Solidago altissima* (L.). The natural wetland soils are predominantly in the Newton series (sandy, mixed, mesic typic humaquept) (NRCS 2011). Soils of the restored wetlands are in the Conrad

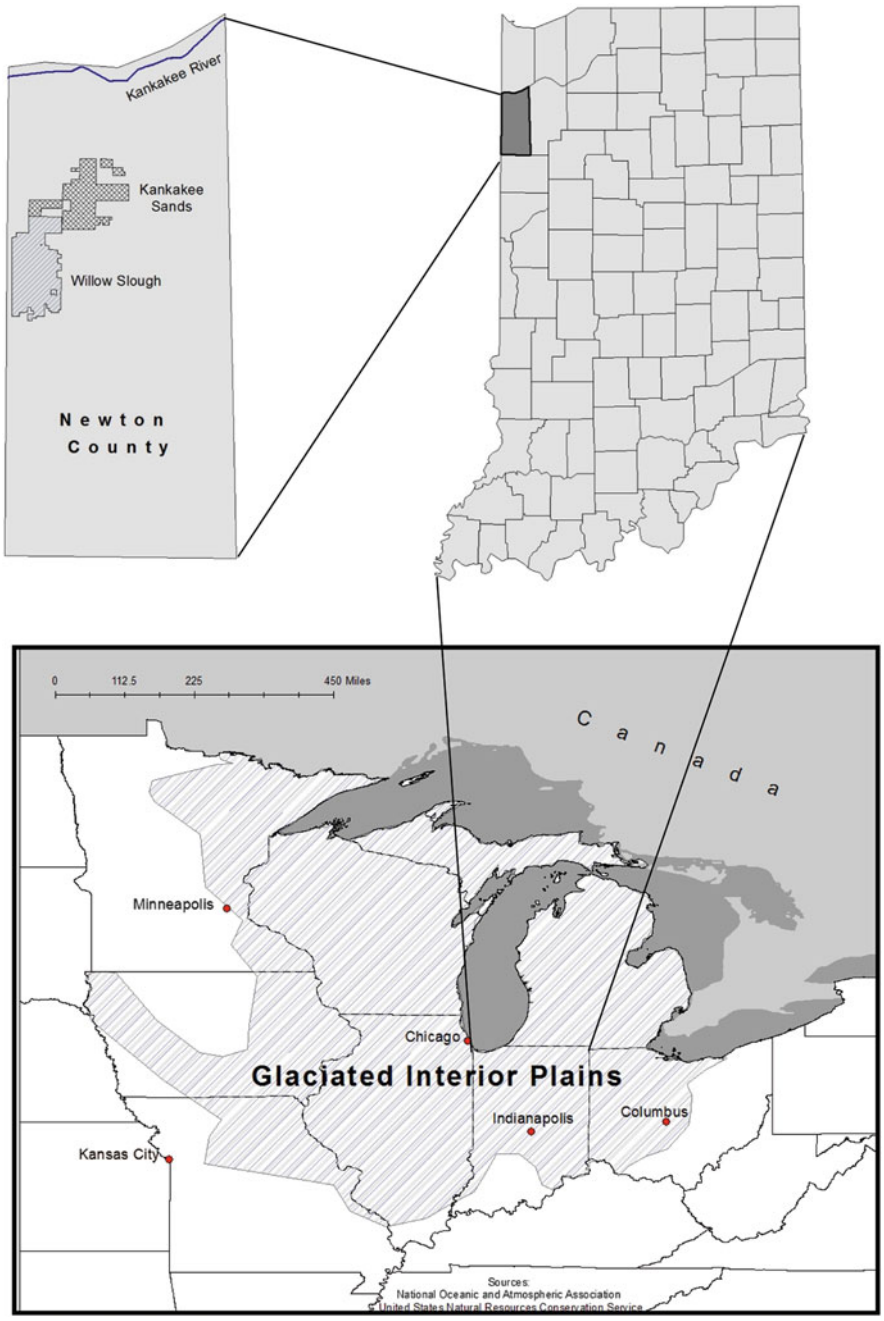

Fig. 2.1 Sampling locations of the restored (Kankakee Sands) and natural (Willow Slough) wetlands. The sites are located in Newton County, IN, and are within the glaciated interior plains region

series (mixed, mesic typic psammaquent), which contains much less soil organic matter than the Newton series (NRCS 2011). All natural and restored wetlands sampled in this study are classified as depressional wetlands according to the HGM classification (Brinson 1993) receiving most of their water and nutrients from precipitation and are approximately 0.5 ha in size.

2.2.2 Greenhouse Gas Production Efficiency: Anaerobic Incubations

Five soil cores (8.5 cm diameter by 10 cm deep) were collected monthly from each wetland during the growing season from May 2011 through September 2011. Soils were transported to the lab on ice where they were homogenized and stored at 4 °C. For incubations, two replicates were prepared from each core for a total of 80 incubations per month. Incubations were conducted in glass bottles equipped with gray butyl septa. Forty mL of deoxygenated, deionized water were added to 25 g of field-moist soil at room temperature to create anaerobic conditions. Samples were then flushed with N_2. Five mL gas samples were withdrawn at intervals of 0.25, 1, 2, 3, and 4 h and injected into evacuated 5 mL glass vials with gray butyl septa. Five mL of N_2 was injected into the incubation bottles to replace the headspace. Gas samples were stored in the light at room temperature until they were analyzed.

2.2.3 Greenhouse Gas Measurements: Static Flux Chamber

A static flux chamber was used to measure fluxes of CO_2, CH_4, and N_2O from permanent plots at two restored wetlands and two natural wetlands. Four aluminum chamber bases (50.8 cm × 50.8 cm × 3.8 cm in size) were installed in each wetland prior to the first sampling event and were held in place by 20 cm long spikes driven into the ground. Chamber bases were placed at randomly selected locations. The chamber (51 cm × 51 cm × 121 cm) was made from ¼″ thick clear plexiglass and was large enough to include vegetation. During sampling, the chamber was shaded with an opaque tarp to regulate the internal chamber temperature. A small, battery-operated fan was attached to the inside of the chamber to ensure mixing. Septa were installed in the plexiglass at heights of 0.3 m and 0.9 m, and gas samples taken from each height were compared to confirm that the air in the chamber was well mixed.

Gases were sampled monthly from June 2011 to September 2011 between 8 am and 5 pm, and the order in which wetlands were sampled was varied each month. At 0, 5, 10, 20, and 30 min intervals, 5 mL of gas was withdrawn from the chamber and injected into evacuated 5 mL glass vials with gray butyl septa. Vials were stored in the light at room temperature and were analyzed as described below.

2.2.4 Laboratory Analyses

Gas samples were analyzed using a SRI 8610C gas chromatograph (GC) (SRI Instruments, Menlo Park, CA). Methane concentrations were determined using a flame ionization detector, and CO_2 concentrations were determined using a flame ionization detector coupled with a methanizer. Nitrous oxide concentrations were determined using an electron capture detector. 500 μL of each sample was injected, and N_2 was used as a carrier gas. The GC was calibrated using four standard concentrations, each for CO_2, CH_4, and N_2O (Scotty Analyzed Gases balanced with N_2, Plumsteadville, PA). A blank and check standard were run after every ten samples.

2.2.5 Supporting Data

Surface water levels were measured monthly at the chamber base and soil core locations. Soil cores were collected from the top 10 cm. Soil moisture content and percent soil moisture of incubation soils (0–10 cm) were determined gravimetrically by weighing 3–5 g of soil at field-moist conditions and then reweighing after drying to a constant weight at 70 °C. Bulk density was calculated by dividing the weight of the dried core by the volume of the core (Blake and Hartge 1986). Porosity was calculated by the formula (Flint and Flint 2002):

$$1.\ (\text{Bulk Density}/\text{Particle Density})$$

The particle density was assumed to be 2.65 (Singer and Munns 2006). Water-filled pore space (WFPS) was calculated using soil moisture and porosity data. Total nitrogen and organic carbon were measured using a Perkin-Elmer CHN Analyzer (Perkin-Elmer, Waltham, MA). An in-house soil standard (5.85 % C, 0.35 % N) had average recovery rates for organic carbon and total nitrogen of 96 % and 95 %.

2.2.6 Statistical Analyses

The data were analyzed using a three-way analysis of variance with wetland type, wetland number, and sampling date as the main effects variables (SAS 1996). The Kolmogorov-Smirnov and Shapiro-Wilk tests were used to test for normality. Chamber fluxes were determined from the change in concentration over time and were analyzed in SAS using repeated measure analysis of variance with wetland type and sampling date as the main effects variables (SAS 1996). Fluxes were natural log transformed prior to statistical analysis to achieve normality. Organic carbon, total nitrogen, bulk density, and WFPS were analyzed with a *t*-test to test

for differences between natural and restored wetlands. Pearson correlation test was used to explore associations between supporting data, incubation emissions, and chamber fluxes. All tests of significance were performed at $\alpha = 0.05$

2.3 Results

2.3.1 Greenhouse Gas Measurements: Anaerobic Incubations

There were no differences in mean GHG emissions of anaerobic incubations from natural and restored wetland soils, and though differences occurred in some months, there were no consistent trends (Table 2.1). In June and September, mean CO_2 emissions from natural wetland soils (0.5 and 0.4 mg C kg^{-1} h^{-1}) were significantly higher than from restored wetland soils (0.2 and 0.1 mg C kg^{-1} h^{-1}). Conversely, mean CO_2 emissions in May and July were significantly higher from restored wetland soils (1.3 and 0.4 mg C kg^{-1} h^{-1}) than from natural wetland soils (0.8 and 0.1 mg C kg^{-1} h^{-1}). Nitrous oxide emissions also varied among months. Emissions in May and September were greater from natural wetland soils (0.7 and 0.2 μg N kg^{-1} h^{-1}) than from restored wetland soils (0.1 and −0.02 μg N kg^{-1} h^{-1}), whereas in July and August, N_2O emissions from restored wetlands (1.0 and 0.2 μg N kg^{-1} h^{-1}) were greater than from natural wetlands (0.1 and 0.1 μg N kg^{-1} h^{-1}). Methane emissions did not vary between months, and natural and restored emissions did not differ in any month.

Restored wetlands had significantly higher GHG production efficiency rates than natural wetlands (Table 2.2). Carbon dioxide production efficiency rates were higher in restored wetlands in May, July, August, and September and were not significantly different in June. Methane production efficiency rates were higher in restored wetlands in June and August and were not significantly different in May, July, and September. Nitrous oxide production efficiencies were higher in restored wetlands in July and August and were not significantly different in May, June, and September.

2.3.2 Greenhouse Gas Measurements: Static Flux Chamber

Similar to the incubations of greenhouse gas production, wetland fluxes measured with the chamber did not differ between natural and restored wetlands (Fig. 2.2). No differences in CO_2 fluxes were observed between natural and restored wetlands in any month. Differences in CH_4 and N_2O fluxes were observed in a few months, but there were no consistent trends. All fluxes had very high spatial and temporal variability. Mean CO_2 fluxes from natural and restored wetlands were 10.1 kg CO_2-C ha^{-1} day^{-1} and 3.6 kg CO_2-C ha^{-1} day^{-1}, respectively. CH_4 flux means

Table 2.1 Mean emissions (± standard errors) during the 2011 growing season of N_2O, CO_2, and CH_4 from restored and natural wetland soil using an anaerobic incubation laboratory method

		N_2O emission	CO_2 emission	CH_4 emission
Month	Type	(μg N kg^{-1} h^{-1})	(mg C kg^{-1} h^{-1})	(μg C kg^{-1} h^{-1})
May	Restored	$0.13 \pm 0.03^*$	$1.27 \pm 0.25^*$	0.17 ± 0.09
	Natural	$0.73 \pm 0.26^*$	$0.79 \pm 0.10^*$	0.09 ± 0.05
June	Restored	0.04 ± 0.03	$0.18 \pm 0.09^*$	0.31 ± 0.18
	Natural	0.09 ± 0.06	$0.47 \pm 0.11^*$	0.40 ± 0.15
July	Restored	$1.03 \pm 0.84^*$	$0.40 \pm 0.05^*$	0.07 ± 0.23
	Natural	$0.08 \pm 0.03^*$	$0.11 \pm 0.08^*$	0.09 ± 0.17
Aug	Restored	$0.23 \pm 0.03^*$	0.33 ± 0.07	0.28 ± 0.08
	Natural	$0.09 \pm 0.01^*$	0.25 ± 0.12	0.21 ± 0.09
Sept	Restored	$-0.02 \pm 0.01^*$	$0.13 \pm 0.01^*$	0.00 ± 0.00
	Natural	$0.19 \pm 0.01^*$	$0.41 \pm 0.03^*$	-0.06 ± 0.09
Mean	**Restored**	$\mathbf{0.28 \pm 0.19}$	$\mathbf{0.37 \pm 0.29}$	$\mathbf{0.17 \pm 0.07}$
	Natural	$\mathbf{0.23 \pm 0.13}$	$\mathbf{0.39 \pm 0.15}$	$\mathbf{0.15 \pm 0.08}$
	GA FW forests[a]	38–400	0.40–1.1	120–190
	NC salt marshes[b]			
	Created	–	0.01–0.03	<0.001
	Natural	–	0.01–0.04	<0.001

Emissions are reported per kilogram of soil on a dry weight basis. Asterisks represent significant differences ($p < 0.05$) between wetland types according to ANOVA analysis

[a]Marton et al. (2012)

[b]Cornell et al. (2007)

Table 2.2 Mean production efficiency rates (±standard errors) during the 2011 growing season of N_2O, CO_2, and CH_4 from restored and natural wetland soil using an anaerobic incubation laboratory method.

		N_2O emission	CO_2 emission	CH_4 emission
Month	Type	(μg N kg^{-1} h^{-1})	(mg C kg^{-1} h^{-1})	(μg C kg^{-1} h^{-1})
May	Restored	0.13 ± 0.05	$1.01 \pm 0.24^{*}$	0.08 ± 0.07
	Natural	0.18 ± 0.07	$0.14 \pm 0.03^{*}$	0.02 ± 0.03
June	Restored	0.02 ± 0.01	0.18 ± 0.05	$0.40 \pm 0.16^{*}$
	Natural	0.01 ± 0.01	0.07 ± 0.03	$0.06 \pm 0.04^{*}$
July	Restored	$0.79 \pm 0.28^{*}$	$0.38 \pm 0.08^{*}$	0.10 ± 0.17
	Natural	$0.01 \pm 0.01^{*}$	$0.02 \pm 0.01^{*}$	-0.03 ± 0.07
Aug	Restored	$0.14 \pm 0.05^{*}$	$0.24 \pm 0.04^{*}$	$0.27 \pm 0.11^{*}$
	Natural	$0.03 \pm 0.01^{*}$	$0.04 \pm 0.01^{*}$	$0.05 \pm 0.01^{*}$
Sept	Restored	-0.01 ± 0.06	$0.11 \pm 0.01^{*}$	0.00 ± 0.00
	Natural	0.04 ± 0.01	$0.08 \pm 0.01^{*}$	-0.02 ± 0.02
Mean	**Restored**	$\mathbf{0.23 \pm 0.09^{*}}$	$\mathbf{0.39 \pm 0.10^{*}}$	$\mathbf{0.17 \pm 0.11^{*}}$
	Natural	$\mathbf{0.06 \pm 0.02^{*}}$	$\mathbf{0.07 \pm 0.02^{*}}$	$\mathbf{0.02 \pm 0.03^{*}}$
	GA tidal	0.53–1.48	0.001–0.002	392–4,076
	FW forests[a]			
	NC salt marshes[b]			
	Natural	–	0.70−1.11	<0.001
	Created	–	0.81−3.89	<0.001

Rates are reported in carbon and nitrogen per kilogram of soil organic carbon basis. Asterisks represent significant differences ($p < 0.05$) between wetland types according to ANOVA analysis

[a]Marton et al. (2012)

[b]Cornell et al. (2007)

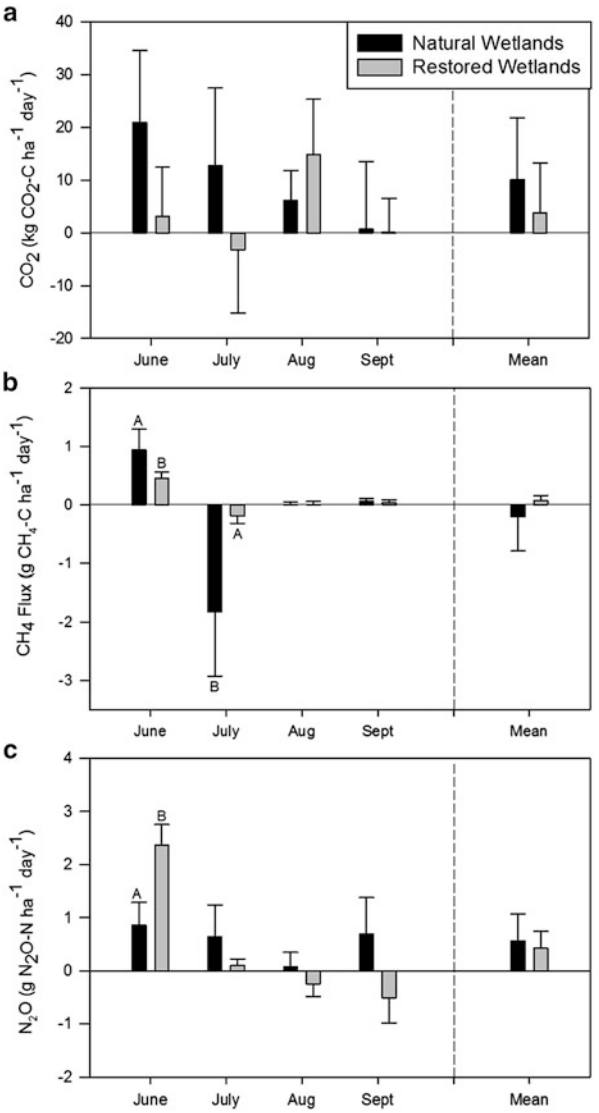

Fig. 2.2 Monthly net ecosystem fluxes of (**a**) CO_2 (g-CO_2-C ha^{-2} day^{-1}), (**b**) CH_4 (g-CH_4-C ha^{-2} day^{-1}), and (**c**) N_2O (g-N_2O-N ha^{-2} day^{-1}) from natural and restored wetlands sampled from June to September 2011 using a static flux chamber. Different letters represent significant differences between wetland types according to ANOVA analysis. Fluxes were natural log transformed prior to statistical analysis. Error bars represent the untransformed standard error of the mean

from natural and restored wetlands were very low, −0.2 and 0.1 g CH_4-C ha^{-1} day^{-1}, respectively. Negative fluxes of CH_4 were observed in natural and restored wetlands in July. Growing season N_2O fluxes from natural and restored wetlands were 0.6 g N_2O-N ha^{-1} day^{-1} and 0.4 g N_2O-N ha^{-1} day^{-1}, respectively. Restored wetlands had a greater flux than natural wetlands in June.

Carbon dioxide was the largest contributor to global warming potential (in CO_2 equivalents) in restored (97.9 %) and natural (98.3 %) wetlands, and CH_4 had the smallest contribution, 0.1 % and 0 % in restored and natural wetlands, respectively

Table 2.3 Growing season means (±standard errors) of water-filled pore space (WFPS), bulk density, percent organic carbon, and percent total nitrogen (equivalent mass to 15 cm) for restored and natural wetland soils

	Bulk density	Organic C		Total N		Moisture	Porosity	WFPS
	(g cm^{-3})	Mg ha^{-1}	(%)	Mg ha^{-1}	(%)	(%)	(%)	(%)
Restored	0.97 ± 0.02*	21.2	1.5 ± 0.6*	1.1	0.12 ± 0.02*	24.7 ± 0.7*	63.2 ± 0.9*	24.7 ± 1.0*
Natural	0.44 ± 0.02*	41.9	6.3 ± 2.5*	2.6	0.58 ± 0.02*	41.2 ± 1.4*	83.3 ± 0.9*	18.1 ± 1.0*
ND restored wetlands[a]	….	49.8	….	2.9	….	….	….	20–80
ND natural wetlands[b]								
Deep marsh	….	54	….	….	….	65	….	….
Shallow marsh	….	54	….	….	….	55	….	….
Wet meadow	….	49	….	….	….	45	….	….
OH created wetlands[c]	….	….	7.2	….	0.3	….	….	….
Prairie pothole restored								
Wetlands[d]	….	50–100	….	….	….	….	….	….

Asterisks represent significant differences ($p < 0.05$) between wetland types according to ANOVA analysis. Values of mean organic carbon, mean total nitrogen, and mean water-filled pore space (WFPS) of natural and restored wetlands in northwestern Indiana are compared with other freshwater mineral soil wetlands

[a]Gleason et al. (2009) *Values were used from the grassland wetlands only

[b]Phillips and Beeri (2008)

[c]Hernandez and Mitsch (2006) *Values were used from the 2005 high marsh study only

[d]Badiou et al. (2011)

(Table 2.3). Nitrous oxide contributed 2.0 % of the global warming potential in restored wetlands and 1.7 % in natural wetlands.

2.3.3 *Supporting Data*

Although we observed no differences in GHG emissions between natural and restored wetlands, soil properties were dramatically different (Table 2.3). Bulk density in natural wetlands (0.44 g cm^{-3}) was half of that measured in restored wetlands (0.97 g cm^{-3}). Soil organic C and total N were four times greater in natural wetlands (6.33 % C, 0.58 % N) than in restored wetlands (1.45 % C, 0.12 % N). Soil organic C concentrations in the top 10 cm of soil were 21.8 Mg ha^{-1} in restored wetlands and 41.6 Mg ha^{-1} in natural wetlands. Porosity and percent moisture were also greater in natural wetlands (41 %, 83 %) than in restored wetlands (25 %, 63 %). Water-filled pore space (WFPS) was significantly higher in restored wetland soils (25 %) than in natural wetland soils (18 %).

2.4 Discussion

Greenhouse gas emissions and fluxes did not differ between natural and restored wetlands. Fluxes were comparable with prairie pothole wetlands in North Dakota, which share similar characteristics such as undergoing periodic burns and existing in a prairie mosaic. Natural (10.1 kg CO_2-C ha^{-1} day^{-1}) and restored (3.8 kg CO_2-C ha^{-1} day^{-1}) flux chamber measurements of CO_2 were comparable to fluxes from deep marsh (4.5 kg CO_2-C ha^{-1} day^{-1}), shallow marsh (11.6 kg CO_2-C ha^{-1} day^{-1}), and wet meadow (8.6 kg CO_2-C ha^{-1} day^{-1}) vegetative zones in North Dakota prairie pothole wetlands (Phillips and Beeri 2008).

Methane fluxes also did not differ between natural and restored wetland soils, but unlike CO_2 emissions, levels were low compared with other studies. Methane fluxes for natural (−0.2 g CH_4-C ha^{-1} day^{-1}) and restored (0.1 g CH_4-C ha^{-1} day^{-1}) wetlands were five orders of magnitude lower than in North Dakota deep marshes (18,000 g CH_4-C ha^{-1} day^{-1}), shallow marshes (7,000 g CH_4-C ha^{-1} day^{-1}), and wet meadows (3,000 g CH_4-C ha^{-1} day^{-1}) (Phillips and Beeri 2008). We attribute the low rates of CH_4 emissions in our natural and restored wetland to the predominantly aerobic soil conditions during sampling. Our wetlands dried down in late June and maintained a mean water-filled pore space (WFPS) of between 15 and 30 % throughout the entire growing season (Fig. 2.3). Gleason et al. (2009) observed net CH_4 consumption in restored grassland wetlands when WFPS dropped below 40 %. These findings suggest that, during the growing season, our wetlands did not maintain the anaerobic conditions required for net CH_4 production. Furthermore, the prolonged aerobic conditions may have suppressed

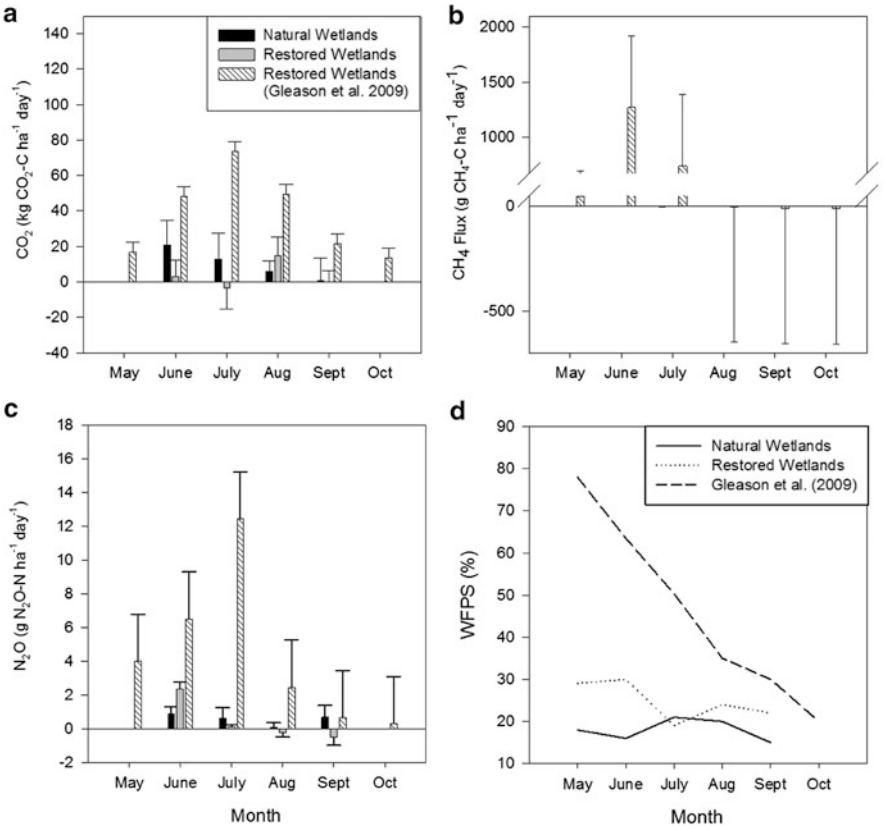

Fig. 2.3 Monthly net ecosystem fluxes of (**a**) CO_2 (g-CO_2-C ha^{-2} day^{-1}), (**b**) CH_4 (g-CH_4-C ha^{-2} day^{-1}), and (**c**) N_2O (g-N_2O-N ha^{-2} day^{-1}) and (**d**) monthly water-filled pore space from the natural and restored wetlands in this study and the restored grassland wetlands in Gleason et al. (2009). Flux measurements were made during the 2011 growing season using a static flux chamber. No measurements for natural and restored wetlands were made in May and October. *Error bars* represent the untransformed standard error of the mean

the populations of methanogens in the soil (Shannon and White 1994), leading to low CH_4 production rates observed in our anaerobic laboratory incubations.

Similar to CH_4, N_2O fluxes did not differ between natural and restored wetlands and were low compared to values reported in the literature. Nitrous oxide fluxes from our natural and restored wetlands measured using static chambers were one order of magnitude lower than natural and restored wetlands in North Dakota (Fig. 2.3c). However, once the WFPS of the restored grassland wetlands measured by Gleason et al (2009) dropped below 40 %, their mean flux (0.6 g N_2O-N ha^{-1} day^{-1}) was comparable to growing season fluxes of our natural wetlands (0.6 g-N_2O-N ha^{-1} day^{-1}) and restored wetlands (0.4 g-N_2O-N ha^{-1} day^{-1}) (Fig. 2.3c, d), suggesting that the aerobic soil conditions limited net N_2O production.

Table 2.4 The relative global warming potential contributions of CO_2, CH_4, and N_2O from natural and restored wetlands compared with other freshwater mineral soil wetlands

	CO_2 (%)	CH_4 (%)	N_2O (%)
Restored	97.9	0.1	2.0
Natural	98.3	0.0	1.7
ND restored wetlands[a]	90	9	1
ND natural wetlands[b]			
Deep marsh	6.0	92.0	2.0
Shallow marsh	49.0	48.0	3.0
Wet meadow	83.6	0.4	16.0
OH created wetlands[c]	99.8	0.2	<0.1

[a]Gleason et al. (2009). Values were used from the grassland wetlands only
[b]Phillips and Beeri (2008)
[c]Altor and Mitsch (2008) (CO_2 and CH_4 values), Hernandez and Mitsch (2006) (N_2O values). Values were used from 2005 only

We observed no correlations between greenhouse gas fluxes and measured soil properties (organic carbon, total nitrogen, bulk density, and WFPS). This is likely because hydrology was the main limiting factor. Because the wetlands dried down so early in the growing season, the soil did not maintain anaerobic conditions ideal for greenhouse gas production. This led to a low WFPS. Other studies have observed that maximum N_2O emissions occur when WFPS is between 40 % and 60 % (Davidson et al. 2000; Gleason et al. 2009), and maximum CH_4 emissions occur when WFPS approaches 100 % (Gleason et al. 2009). The WFPS of wetland soils in this study only exceeded 40 % in 3.5 % of the soil samples and never exceeded 60 %. This suggests that soil saturation is a key limiting factor in N_2O and CH_4 production of our restored and natural wetlands.

Our study is among the few that compared greenhouse gas emission (CO_2, CH_4, and N_2O) between natural and restored freshwater mineral soil wetlands. Our chamber measurements found no differences in greenhouse gas fluxes between natural and restored wetlands. This finding is in conflict with Badiou et al. (2011) and Nahlik and Mitsch (2010). Badiou et al. (2011) found that restored wetlands in the prairie pothole region of Canada contributed more CH_4 and N_2O than natural wetlands. Nahlik and Mitsch (2010) found the opposite trend and observed higher CH_4 fluxes in a natural Ohio wetland than in two nearby created wetlands. All three studies exhibited high spatial and temporal variability in greenhouse gas emissions, suggesting that more frequent measurements are needed to constrain the contribution of restored wetlands on a landscape scale.

Of the three GHGs, carbon dioxide contributed the most (98 %) to cumulative GHG emissions in our natural and restored wetlands (Table 2.4). This trend was also seen in created wetlands in Ohio (99.8 %) (Nahlik and Mitsch 2010), restored wetlands in North Dakota (90 %) (Gleason et al 2009), and, to a lesser extent, shallow marshes (49 %) and wet meadows (84 %) in North Dakota (Phillips and Beeri 2008). Methane was the lowest contributor to global warming potential in our natural (0 %) and restored (0.1 %) wetlands and was also the lowest contributor in the North Dakota wet meadows (0.4 %). Nitrous oxide was the lowest contributor in

created marshes in Ohio (<0.1 %), restored wetlands in North Dakota (1 %), and deep (2 %) and shallow (3 %) marshes in North Dakota.

Using our flux data, we estimate that the restored wetlands contributed 0.66 Mg CO_2 equivalents ha^{-1} during the growing season, and natural wetlands contributed 1.9 Mg CO_2 equivalents ha^{-1}. It should be noted that these values are likely lower than the true emissions as sampling began after late April, thus missing the beginning of the growing season.

Previous studies have demonstrated that the restored wetlands used in this study provide water quality and biodiversity benefits (Hopple and Craft 2013, Marton et al. 2014), and from our measurements, we conclude that the ecological benefits of the restored wetlands outweigh potential negative impacts due to GHG emissions. Because of the high spatial and temporal variability of fluxes observed, additional GHG measurements are needed to constrain greenhouse emissions of restored wetlands in the glaciated interior plains and elsewhere.

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References

Altor, A., & Mitsch, W. (2008). Pulsing hydrology, methane emissions and carbon dioxide fluxes in created marshes: a 2-year ecosystem study. *Wetlands, 28*(2), 423–438.

Badiou, P., McDougal, R., Pennock, D., & Clark, B. (2011). Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian prairie pothole region. *Wetlands Ecology and Management, 19*, 237–256.

Bartlett, K. B., & Harriss, R. C. (1993). Review and assessment of methane emissions from wetlands. *Chemosphere, 26*, 261–320.

Blake, G. R., & Hartge, K. H. (1986). Bulk density. In A. Klute (Ed.), *Methods of soil analysis, Part I. Physical and mineralogical methods* (Agronomy monograph no. 9 2nd ed., pp. 363–375). Madison: American Society of Agronomy.

Brander, L. M., Florax, R., & Vermaat, J. E. (2006). The empirics of wetland valuation: A comprehensive summary and a meta-analysis of the literature. *Environmental and Resource Economics, 33*(2), 223.

Bridgham, S. D., Megonigal, J. P., Keller, J. K., Bliss, N. B., & Trettin, C. (2006). The carbon balance of North American wetlands. *Wetlands, 26*(4), 889–916.

Brinson, M. (1993). *A hydrogeomorphic classification for wetlands*. Wetlands Research Program TR-WRP-DE-4. US Army Waterways Exp. Station, Vicksburg, MS.

Brinson, M., & Eckles, S. (2011). U.S. Department of Agriculture conservation program and practice effects on wetland ecosystem services: A synthesis. *Ecological Applications, 21*(3), S116–S127.

Bruland, G., & Richardson, C. (2005). Spatial variability of soil properties in created, restored, and paired natural wetlands. *Soil Science Society of America Journal, 69*, 273–284.

Cornell, J., Craft, C., & Megonigal, P. (2007). Ecosystem gas exchange across a created salt marsh chronosequence. *Wetlands, 27*(2), 240–250.

Craft, C. B., & Casey, W. P. (2000). Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. *Wetlands, 20*(2), 323–332.

Craft, C. B., Broome, S., & Campbell, C. (2002). Fifteen years of vegetation and soil development after brackish-water marsh creation. *Restoration Ecology, 10*(2), 248–258.

Craft, C., Krull, K., & Graham, S. (2007). Ecological indicators of nutrient enrichment, freshwater wetlands, Midwestern United States (U.S.). *Ecological Indicators, 7*, 733–750.

Dahl, T. E. (1990). *Wetland losses in the United States: 1780's to 1980's*. Washington, DC: U.S. Fish and Wildlife Service.

Davidson, E. A., Keller, M., Erickson, H. E., Verchot, L. V., & Veldkamp, E. (2000). Testing a conceptual model of soil emissions of nitrous and nitric oxides. *BioScience, 50*, 667–680.

Euliss, N. H., Gleason, R. A., Olness, A., McDougal, R. L., Murkin, H. R., Robarts, R. D., Bourbonniere, R. A., & Warner, B. G. (2006). North American prairie wetlands are important nonforested land-based carbon storage sites. *Science of the Total Environment, 361*, 179–188.

Fennessy, S., & Craft, C. (2011). Agricultural conservation practices increase wetland ecosystem services in the Glaciated Interior Plains. *Ecological Applications, 21*(3), S49–S64.

Flint, L. E., & Flint, A. L. (2002). Porosity. In A. Klute (Ed.), *Methods of soil analysis, Part I. Physical and mineralogical methods* (Agronomy monograph no. 9 2nd ed., pp. 241–253). Madison: American Society of Agronomy.

Gleason, R. A., Tangen, B. A., Browne, B. A., & Euliss, N. H. (2009). Greenhouse gas flux from cropland and restored wetlands in the Prairie Pothole Region. *Soil Biology and Biochemistry, 41*, 2501–2507.

Hernandez, M. E., & Mitsch, W. J. (2006). Influence of hydrologic pulses, flooding frequency, and vegetation on nitrous oxide emissions from created riparian marshes. *Wetlands, 26*, 862–77.

Hopple, A., & Craft, C. (2013). Managed disturbance enhances biodiversity of restored wetlands in the agricultural Midwest. *Ecological Engineering, 61B*, 505–510.

Indiana Department of Natural Resources (IN DNR). Willow Slough. http://www.in.gov/dnr/fishwild/3080.htm. Accessed 15 Nov 2011.

Keller, M., Mitre, M. E., & Stallard, R. F. (1990). Consumption of atmospheric methane in soils of central Panama: Effects of agricultural development. *Global Biogeochemical Cycles, 4*, 21–27.

Knutsen, G. A., & Euliss, N. H., Jr. (2001). *Wetland restoration in the Prairie Pothole Region of North America: A literature review* (Biological science report 2001–0006). Reston: U.S. Geological Survey.

Marton, J., Herbert, E., & Craft, C. (2012). Effects of salinity on denitrification and greenhouse gas production from laboratory-incubated tidal forest soils. *Wetlands, 32*, 347–357.

Marton, J. M., Fennessy, M. S., & Craft, C. B. (2014). Functional differences between natural and restored wetlands in the Glaciated Interior Plains. *Journal of Environmental Quality, 43*, 409–417.

Mitsch, W. J., Day, J. W. Jr., Gilliam, J. W., Groffman, P. M., Hey, D. L., Randall, G. W., & Wang, N. (1999). *Reducing nutrient loads, especially nitrate–nitrogen, to surface water, ground water, and the Gulf of Mexico: Topic 5 report for the integrated assessment on hypoxia in the Gulf of Mexico*. NOAA Coastal Ocean Program Decision Analysis Series No. 19. NOAA Coastal Ocean Program, Silver Spring, MD, 111 pp.

Nahlik, A. M., & Mitsch, W. J. (2010). Methane emissions from created riverine wetlands. *Wetlands, 30*, 783–793.

Natural Resources Conservation Service. (2011). Web soil survey, United States Department of Agriculture. http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm. Accessed Nov 2011.

Nelson, D. W., & Sommers, L. E. (1996). Total carbon, organic carbon, and organic matter. In D. L. Sparks (Ed.), *Methods of soil analysis, Part 3: Chemical methods* (Soil Science Society of America, book series, Vol. 5, pp. 961–1010). Madison: Soil Science of Society of America.

Phillips, R., & Beeri, O. (2008). The role of hydropedologic vegetation zones in greenhouse gas emissions for agricultural wetland landscapes. *Catena, 72*, 386–394.

Poffenbarger, H. J., Needleman, B. A., & Megonigal, J. P. (2011). Salinity influence on methane emissions from tidal marshes. *Wetlands, 31*, 831–842.

Power, J. F., Koerner, P. T., Doran, J. W., & Wilhelm, W. W. (1998). Residual effects of crop residues on grain production and selected soil properties. *Soil Science Society of America Journal, 62*, 1393–1397.

SAS (Statistical Analysis Systems). (1996). *SAS user's guide*. Cary: SAS Institute.

Segers, R. (1998). Methane production and methane consumption: A review of processes underlying wetland methane fluxes. *Biogeochemistry, 41*, 23–51.

Sha, C., Mitsch, W., Mander, U., Lu, J., Batson, J., Zhang, L., & He, W. (2011). Methane emissions from freshwater riverine wetlands. *Ecological Engineering, 37*, 16–24.

Shannon, R., & White, J. (1994). A three-year study of controls on methane emissions from two Michigan peatlands. *Biogeochemistry, 27*(1), 35–60.

Singer, M. J., & Munns, D. N. (2006). *Soils: An introduction* (6th ed.). Upper Saddle River: Pearsons Prentice Hall.

Soil Survey Staff. (2010). Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey, Newton County, Indiana. Available online at: http://websoilsurvey.nrcs.usda.gov/. Accessed 18 Oct 2011.

U. S. FWS. (1999). Chronology of Important Events on the Kankakee River. In *Grand Kankakee Marsh National Wildlife Refuge environmental assessment*. Fort Snelling, MN. Available online at: http://www.fws.gov/midwest/planning/GrandKankakee/. Accessed 4 Dec 2011

United States Department of Agriculture. (2008). *At a glance: Wetlands reserve program*. Natural Resource Conservation Service.

Zedler, J. (2003). Wetlands at your service: Reducing impacts of agriculture at the watershed scale. *Frontiers in Ecology and the Environment, 1*(2), 65–72.