Methane Flux in Cropland and Adjacent Riparian Buffers with Different Vegetation Covers

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While water quality functions of conservation buffers established adjacent to cropped fields have been widely documented, the relative contribution of these re-established perennial plant systems to greenhouse gases has not been completely documented. In the case of methane (CH₄), these systems have the potential to serve as sinks of CH₄ or may provide favorable conditions for CH₄ production. This study quantifies CH₄ flux from soils of riparian buffer systems comprised of three vegetation types and compares these fluxes with those of adjacent crop fields. We measured soil properties and diel and seasonal variations of CH₄ flux in 7 to 17 yr-old re-established riparian forest buffers, warm-season and cool-season grass filters, and an adjacent crop field located in the Bear Creek watershed in central Iowa. Forest buffer and grass filter soils had significantly lower bulk density (P < 0.01); and higher pH (P < 0.01), total carbon (TC) (P < 0.01), and total nitrogen (TN) (P < 0.01) than crop field soils. There was no significant relationship between CH₄ flux and soil moisture or soil temperature among sites within the range of conditions observed. Cumulative CH₄ flux was –0.80 kg CH₄-C ha⁻¹ yr⁻¹ in the cropped field, –0.46 kg CH₄-C ha⁻¹ yr⁻¹ within the forest buffers, and 0.04 kg CH₄-C ha⁻¹ yr⁻¹ within grass filters, but difference among vegetation covers was not significant. Results suggest that CH₄ flux was not changed after establishment of perennial vegetation on cropped soils, despite significant changes in soil properties.

The global atmospheric concentration of CH₄ has increased from a preindustrial value of about 715 to 1774 μL L⁻¹ in 2005, likely a result of anthropogenic activities such as agricultural production and fossil fuel use (IPCC, 2007). Soils have been shown to both produce and consume CH₄ (Topp and Pattey, 1997; Le Mer and Roger, 2001). In a recent review, Dutar and Verchot (2007) summarized net CH₄ flux as the result of the balance between the two offsetting processes of methanogenesis (microbial production under anaerobic conditions) and methanotrophy (microbial consumption). These authors identified methanotrophy as the dominant process in upland soils, where oxidation generally exceeds production with a resulting net uptake of atmospheric CH₄ by soil. It is well known that forest soils are the most active sink of CH₄, followed by grass lands and cultivated soils, and that the CH₄ uptake potential of many upland soils is reduced by cultivation and application of ammonium N fertilizer (Topp and Pattey, 1997; Le Mer and Roger, 2001; Dutar and Verchot, 2007). It has been reported that land-use change can also influence CH₄ uptake rates. For instance, higher rates of CH₄ oxidation have been observed in soils afforested from croplands or pastures (e.g., Ball et al., 2002; Merino et al., 2004; Tate et al., 2007). Observed increases in CH₄ uptake resulting from land-use change are attributed to changes in soil porosity, moisture content, and methanotroph population (Prieß et al., 1997).

Nonpoint source (NPS) pollutants such as sediment, N, P, and pesticides are major causes of water quality problems around the world (Duda, 1993; Tonderski, 1996; Sabater et al., 2003). Riparian buffers have been recommended as one of the most effective tools for mitigating NPS pollution (Hubbard et al., 2004; Mayer et al., 2007). Some of the important functions of riparian buffers related to NPS pollution control are filtering and retaining sediment and immobilizing, storing, and transforming chemical inputs from uplands (Schultz et al., 2000). Generally, riparian buffers re-established on cultivated crop fields consist of combinations of grasses, forbs, shrubs, and trees (Schultz et al., 2004). While these systems have been well documented for their water quality functions, little is known about other ecosystem processes such as their relative greenhouse gas flux. If these systems perform similar to perennial plant systems in upland positions, it would suggest that...
Chronic and cumulative CH4 emissions from forests and agricultural areas are important contributors to global climate change. Forests can produce CH4 through biogeochemical processes, while agricultural areas can consume CH4 through denitrification or crop production. However, the impact of riparian areas on CH4 emissions and consumption remains uncertain.

### Materials and Methods

#### Study Site

The study area consisted of three forest buffers, three warm-season grass filters, and one adjacent crop field located in the Bear Creek watershed, Story County and Hamilton County, Iowa (42° 11’ N, 93° 30’ W). The Bear Creek watershed (6810 ha) is a predominantly agricultural watershed typical of north central Iowa, with a mean annual air temperature of 8.7°C and mean annual precipitation of 810 mm over the period of record (United States Department of Commerce–National Oceanic and Atmospheric Administration, 2009). Most of the area was originally covered with prairie and wetland vegetation except for riparian forests along higher order streams. Most of the riparian buffers re-established on cropped soils may produce less and consume more CH4 than crop fields. However, riparian buffers are often flooded and also sustain relatively high soil moisture conditions caused by high water tables, long residence time and slow discharge (Schultz et al., 2000). These conditions may be favorable for CH4 production. For example, Ambus and Christensen (1995) reported that CH4 was produced in temporarily flooded riparian areas at rates of 78.8 kg CH4-C ha–1 yr–1 in crop fields (Christensen, 1995; Marquez et al., 2004). However, few studies of CH4 flux have focused on re-established riparian buffers, and the benefits of reduced nonpoint-source pollution from riparian buffers may be offset by increased greenhouse gas emissions.

Numerous studies have emphasized the role of vegetation in soil biogeochemical processes within natural or re-established riparian buffers, with studies demonstrating an improvement in soil quality indicators (e.g., Tufekcioglu et al., 1999; Bharati et al., 2002; Marquez et al., 2004). However, most studies have often focused on riparian soils, particularly those re-established to perennial vegetation, or on the relationship between observed changes in soil quality on conditions regulating methane flux. Specific objectives of this study were to compare CH4 flux from riparian buffer systems comprised of forest, warm-season grasses, and cool-season grasses and an adjacent crop field, and to relate these fluxes to changes in soil properties after re-establishment of perennial plants.

### Soil Sampling and Analysis

Six intact soil cores (5.3-cm diam.) were collected to a depth of 15 cm around each of three gas sampling points in a forest buffer, a warm-season grass filter, a cool-season grass filter, and an adjacent crop field in October 2006 and September 2007. A plastic sleeve liner was placed inside the metal core tube and the liner and intact soil core pulled from the tube and capped for transport to the laboratory. Soil samples were transported back to lab in a cooler and stored at 4°C until analysis. Soil pH was determined by using 1:1 diluted soil solution. Gravimetric moisture content was determined by oven drying a subsample at 105°C for 24 h. Bulk density was estimated using the core method (Grossman and Reinsch, 2002). For C and N analysis, soils were air dried at room temperature, and sieved (2 mm). Total C and TN were measured using a Flash EA 2000 (ThermoFinnigan, Italy) elemental analyzer. Soil inorganic N was extracted with 2 mol L–1 potassium chloride (KCl) within 4 h of sampling and stored at 4°C until filtration (Van Miegroet, 1995). Filtrates were frozen and stored until analysis. Nitrate (NO3\(^{-}\)) and ammonium (NH4\(^{+}\)) content were analyzed by colorimetric method (Mulaney, 1996) with an auto analyzer (Quikchem 8000 FIA+, Lachat Instruments, Milwaukee, WI).
Field Gas Sampling, Methane Gas Analysis, and Flux Calculation

Soil CH$_4$ flux from riparian forest buffers, warm- and cool-season grass filters and one crop field was measured from January through December 2007. To assess the temporal variation of flux, five locations were randomly selected in each of three forest buffers, three warm-season grass filters, one cool-season grass filter, and one crop field with the distance between gas sampling points ranging from 5 to 10 m. A polyvinyl chloride (PVC) ring (30 cm diam. by 15 cm height) served as base for gas chambers and was installed to a depth of approximately 10 cm. In the crop field, rings were placed either between plants within the row or between rows. These rings were left in place between sampling periods, but were removed for fertilization, planting, and tillage events in the crop field. Vegetation inside the rings was cut before gas sampling in the forest buffers and grass filters during growing seasons. Gas samples were collected within static vented chambers (PVC, 30 cm diam. by 15 cm height with a vent) weekly or biweekly during the mid-morning. Chambers were equipped with a thermometer to measure air temperature within the chambers at the time of sampling. Ten milliliters of air was sampled from the chamber with a polypropylene syringe at 15-min intervals for 45 min (four samples 0, 15, 30, and 45 min) and the gas stored in pre-evacuated glass vials (6 mL, fitted with butyl rubber stoppers) until analysis. Glass vials were prepared by alternately evacuating the vial headspace and flushing with helium to remove air. In addition to the regular measurements, diel variation of CH$_4$ flux was measured on 16–17 July 2007. For this assessment, three gas sampling points were randomly selected in a forest buffer, a warm-season grass filter, a cool-season grass filter, and an adjacent crop field and gas samples were collected every 3 h for 24 h. Methane concentrations were determined with a gas chromatograph (GC) (Model GC17A; Shimadzu, Kyoto, Japan) equipped with a flame ionization detector (FID) and a stainless steel column (0.3175 cm diam. × 74.54 cm long) with Porapak Q (80–100 mesh). Samples were introduced into the gas chromatograph using an autosampler described by Arnold et al. (2001). Three different CH$_4$ standards (0, 2.0, and 10 μL L$^{-1}$) were used to perform calibration curves and field ambient samples and CH$_4$ standards were analyzed every 20 gas samples to verify accuracy in GC results. Methane fluxes were obtained by applying linear regression to the CH$_4$ concentration vs. time data (Holland et al., 1999). Linearity with $R^2 > 0.8$ was accepted as a valid flux rate, which resulted in the inclusion of 90% of the flux rates in this study. Where removing a sample corrected a poor linearity ($R^2 < 0.8$) to $R^2 > 0.9$, the sample was eliminated from the calculation of flux rate (Altor and Mitsch, 2006). Therefore, flux rate was determined using a minimum of three gas samples.

The minimum detectable CH$_4$ flux was calculated using an average of standard deviations of CH$_4$ concentrations of lab ambient air samples and CH$_4$ standards ($n = 500$) analyzed with collected gas samples, chamber volume, chamber footprint, and chamber enclosure time in the fields as following:

\[
\text{Minimum detectable CH}_4 \text{ flux (μL CH}_4\text{C m}^{-2} \text{h}^{-1}) = 2 \times \frac{\text{average of standard deviation(μL L}^{-1}) \times \text{chamber volume (L)} \times \text{chamber footprint (m}^2\text{)} \times \text{chamber enclosure time (h)}}{\text{}}
\]

The calculated minimum detectable CH$_4$ flux (μL CH$_4$–C m$^{-2}$ h$^{-1}$) converted to mass unit (μg CH$_4$–C m$^{-2}$ h$^{-1}$) through application of the universal gas law (Holland et al., 1999). Our estimated minimum detectable flux was 33.2 μg CH$_4$–C m$^{-2}$ h$^{-1}$.$^{99}$ Some of the fluxes measured from the individual chambers were smaller than our detection limit. In these situations, we followed the recommendation of Gilbert (1987) and Chan and Parkin (2001a) and included the measured values of these “nondetects” in computing mean fluxes. Cumulative CH$_4$ fluxes from each site over the 1-d study period (16–17 July 2007) and the 1-yr study period (January–December 2007) were calculated by linear interpolation and numerical integration between sampling times.

Soil temperature (ST) and soil water content (SWC) were measured simultaneously with CH$_4$ gas collection around the chamber at a 5 cm depth using a digital thermocouple (ThermoWorks, Orem, UT) and a digital soil moisture meter (HydroSense, Campbell Scientific, Inc., Logan, UT). Air temperature was measured simultaneously with CH$_4$ gas collection inside and outside the gas chamber. A soil temperature and soil moisture data logger (HOBO Micro station data logger with sensors, Onset Computer Corporation, Bourne, MA) was installed at 5 cm soil depths around a chamber at each site to measure hourly ST and SWC at each site. Daily rainfall and snow data were provided by a nearby meteorology station (Colo, IA, 42° 1’ N, 93° 19’ W).

Soil Incubation with Control and 10 Pa Acetylene ($C_2H_2$)

Aerobic CH$_4$ production and net CH$_4$ flux were estimated using the intact soil cores (0–15 cm depth) collected in September 2007. Soil samples were transported in a cooler and stored at 4°C until experiments, and incubation experiments with the intact soil cores were conducted within 6 h of sampling. All aboveground vegetation in the soil cores was cut off before the experiments. Six intact soil cores (5.3 cm diam. by 8 cm long) collected at each site were placed into 350-mL glass jars with gas-tight lids containing a gas-sampling port and all jars were sealed. Gravimetric moisture content of each soil was determined by oven drying a subsample at 105°C for 24 h. Three soil cores from each site were treated with 10 Pa C$_2$H$_2$, and three were retained as controls (no C$_2$H$_2$). Soil cores were incubated at 22°C, the on-site soil temperature. Ten milliliters of air was sampled from the jars with a polypropylene syringe at 3, 9, and 16 h, and stored until analysis. Storage, gas analysis, and flux calculations were as described above. Aerobic CH$_4$ production was estimated from soil incubations in which CH$_4$ oxidation was inhibited by 10 Pa C$_2$H$_2$ (Chan and Parkin, 2000, 2001b). Net CH$_4$ flux was determined from CH$_4$ flux in soil incubations without C$_2$H$_2$.

Aerobic CH$_4$ production = CH$_4$ flux under 10 Pa C$_2$H$_2$ [2]

Net CH$_4$ flux = CH$_4$ flux under no C$_2$H$_2$ [3]

Statistical Analyses

The Shapiro-Wilk normality test was used to assess normality of data. A two-sample t test was used to evaluate differences in soil C measured in 1998–1999 and 2006–2007 in the same sites. One-way ANOVA was used to evaluate the differences in soil properties, and diel and seasonal CH$_4$ flux by site. When the
higher than crop field soils. However, this difference is apparently

soils were significantly

the relationship between soil properties and CH4 flux, correlation

higher than those in forest buffers and grass filters (21.8°C, n = 93) (P < 0.01).

Soil Incubation Experiments

Soil gravimetric moisture at the time of soil incubations was

14.8 ± 0.3% in the crop field soils (n = 18), 22.8 ± 0.8% within

forest buffers soils (n = 18), 19.0 ± 0.7% within warm-season

grass filters soils (n = 18), and 26.1 ± 0.3% within cool-season

grass filters soils (n = 18) with the observed differences sig-
nificant (one-way ANOVA P < 0.01). Aerobic CH4 production

in the incubated forest buffer soils (3.5 × 10–2 ± 2.3 × 10–2 ng CH4–C kg soil–1 h–1, n = 3), warm-season grass filter soils

(1.6 × 10–3 ng CH4–C kg soil–1 h–1, n = 1), and cool-

season grass filter soils (5.4 × 10–2 ± 1.9 × 10–3 ng CH4–C kg

soil–1 h–1, n = 3) and crop field soils (1.8 × 10–2 ± 6.9 × 10–3 ng

CH4–C kg soil–1 h–1, n = 3) were not significantly different (Fig.

1). Similarly, net CH4 fluxes in the incubated forest buffer soils

(–4.9 × 10–4 ± 2.3 × 10–3 ng CH4–C kg soil–1 h–1, n = 3), warm-

season grass filter soils (–1.2 × 10–2 ± 5.3 × 10–3 ng CH4–C kg

soil–1 h–1, n = 3), and cool-season grass filter soils (–1.5 × 10–2

± 1.2 × 10–2 ng CH4–C kg soil–1 h–1, n = 3) were not significantly different (Fig. 1).

Diel Variation of Methane Flux and Controlling Factors

The diel variation of CH4 flux for sampling conducted during

16–17 July 2007 did not differ among any of the vegetation

types. Similarly, there was no significant difference in CH4 flux among vegetation types (Fig. 2A). The average CH4 flux

(n = 8) was 6.9 ± 12.2 μg CH4–C m–2 d–1 in crop field, 23.6 ±

8.1 μg CH4–C m–2 h–1 within the forest buffer, –19.1 ± 10.6 μg

CH4–C m–2 h–1 within the warm-season grass filter and –25.5 ±

10.2 μg CH4–C m–2 h–1 within the cool-season grass filter. The variation of CH4 flux in the crop field, forest buffer, warm-season

grass filter and cool-season grass filter was not correlated with

soil temperature. Cumulative CH4 flux over 24 h was 165.7 μg

CH4–C m–2 d–1 in the crop field, –567.3 μg CH4–C m–2 d–1 in

the forest buffer, –459.1 μg CH4–C m–2 d–1 in the warm-season

Fig. 1. Aerobic CH4 production and net CH4 flux from incubated soil core under controlled laboratory conditions. Soil cores were obtained from crop field, forest buffer, warm-season grass filter and cool-season grass filter soils. Each mean represents three observations (one observation for CH4 production in warm-season grass filter soil) and bars are the standard error of the mean.

Table 1. Soil properties (mean ± standard error) within a re-established riparian forest buffer, a warm-season grass filter, a cool-season grass filter, and an adjacent crop field in October 2006 and September 2007 (depth 0–15 cm, n = 6–9 except bulk density (n = 27)).

<table>
<thead>
<tr>
<th>Site</th>
<th>Soil texture</th>
<th>Bulk density (Mg m–1)</th>
<th>pH</th>
<th>TC (g kg–1 soil)</th>
<th>TN (g kg–1 soil)</th>
<th>NH4–N (mg N kg–1 soil)</th>
<th>NO3–N (mg N kg–1 soil)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Crop field</td>
<td>Loam</td>
<td>1.67 ± 0.02a</td>
<td>5.9 ± 0.1c</td>
<td>22.8 ± 1.0c</td>
<td>1.9 ± 0.1c</td>
<td>1.7 ± 0.2b</td>
<td>1.2 ± 0.5a</td>
</tr>
<tr>
<td>Forest buffer</td>
<td>Loam, Sandy loam</td>
<td>1.10 ± 0.03c</td>
<td>7.3 ± 0.1a</td>
<td>42.9 ± 3.2a</td>
<td>3.8 ± 0.3a</td>
<td>4.1 ± 0.6a</td>
<td>0.7 ± 0.2a</td>
</tr>
<tr>
<td>Warm-season grass filter</td>
<td>Loam</td>
<td>1.29 ± 0.05b</td>
<td>6.7 ± 0.2b</td>
<td>29.1 ± 2.7bc</td>
<td>2.6 ± 0.2bc</td>
<td>3.9 ± 0.5a</td>
<td>0.2 ± 0.1a</td>
</tr>
<tr>
<td>Cool-season grass filter</td>
<td>Loam</td>
<td>1.19 ± 0.04bc</td>
<td>6.9 ± 0.1ab</td>
<td>32.4 ± 1.6bc</td>
<td>2.9 ± 0.1b</td>
<td>4.3 ± 0.4a</td>
<td>0.9 ± 0.3a</td>
</tr>
</tbody>
</table>

† Values in the same column followed by a different letter are significantly different (P < 0.05).

‡ Marquez et al. (2004).

Results

Soil Properties

The texture of all treatment site soils (Coland) was loam

(Marquez et al., 2004) (Table 1). Soils within all riparian buffer

vegetation types had significantly lower bulk density (one-way

ANOVA P < 0.01); higher pH (P < 0.01), and NH4+ (P < 0.01) than crop field soils. TC (P < 0.01) and TN (P < 0.01) within

the pooled riparian buffer vegetation soils were significantly

higher than that of 1998 and 1999 (two sample t test P = 0.006, 95% CI for difference of means: 4.5 – 20.5 g kg–1).

From 15 June to 15 August 2007 (93 d), average daily soil

moisture in the crop field (8.7 ± 0.2%, n = 93), was signifi-
cantly lower (one-way ANOVA P < 0.01) than soils within either

forest buffer (16.9 ± 0.2%, n = 93), or grass filter (19.0 ± 0.2%,
n = 93). During the same period, average daily soil temperature

in the crop fields (22.8 ± 0.3°C, n = 93) was signifi-
cantly higher than those in forest buffers and grass filters (21.8°C, n = 93) (P < 0.01).

standard assumptions of normality were violated, nonparametric

Kruskal–Wallis one-way ANOVA on ranks was used. Differences

were considered significant at the P < 0.05 level. To determine

the relationship between soil properties and CH4 flux, correlation

analysis using the GLM procedure was applied. Statistical analyses

were conducted using SAS ver 8.1 (SAS Institute, 1999).
grass filter, and -612.7 μg CH₄–C m⁻² d⁻¹ in the cool-season grass filter (Fig. 2B).

**Seasonal Variation of Methane Flux and Annual Methane Emission**

Since there was no significant variation in CH₄ flux through time within a day (Fig. 2A), daily fluxes were calculated by multiplying measured hourly fluxes (mid-morning) by 24 h. Observed maximum positive daily CH₄ flux was 2.4 mg CH₄–C m⁻² d⁻¹ in crop fields (7 August), 2.4 mg CH₄–C m⁻² d⁻¹ in forest buffers (1 May), and 1.6 mg CH₄–C m⁻² d⁻¹ and grass filters (11 September) (Fig. 3). Observed maximum negative daily CH₄ flux was −2.2 mg CH₄–C m⁻² d⁻¹ in crop fields (16 January), −2.6 mg CH₄–C m⁻² d⁻¹ in forest buffers (20 September), and −3.7 mg CH₄–C m⁻² d⁻¹ (20 September) grass filters (Fig. 3). Mean daily CH₄ flux was −0.2 ± 0.1 mg CH₄–C m⁻² d⁻¹ (n = 40) in the crop field, −0.5 to 0.9 mg CH₄–C m⁻² d⁻¹ (n = 45–50) in forest buffers, and −0.2 to 0.1 mg CH₄–C m⁻² d⁻¹ (n = 41–49) in grass filters, with no significant differences (Fig. 4). There were no significant relationships between CH₄ flux and soil moisture (P > 0.05) or soil temperature (P > 0.05) in any of the sites. Cumulative CH₄ flux from January to December 2007 was −0.80 kg CH₄–C ha⁻¹ yr⁻¹ (n = 1) in the crop field, −0.46 ± 0.48 kg CH₄–C ha⁻¹ yr⁻¹ (n = 3) in forest buffers, and 0.04 ± 0.2 kg CH₄–C ha⁻¹ yr⁻¹ (n = 4) in grass filters (Fig. 5). Cumulative CH₄ flux in forest buffers (95% confidence interval (CI): −2.54 to 1.61 kg CH₄–C ha⁻¹ yr⁻¹) and grass filters (95% CI: −0.51 to 0.61 kg CH₄–C ha⁻¹ yr⁻¹) were not significantly different from zero. The cumulative CH₄ flux in the crop field, forest buffers and grass filters were not significantly different from one another (one-way ANOVA P = 0.40) (Fig. 5).

**Discussion**

**Change of Soil Properties after Re-establishment of Riparian Buffers**

Soils within forest buffers and grass filters had significantly lower bulk density, higher pH, TC, TN, and NH₄⁺ than those in adjacent crop fields. This suggests that the re-establishment of the perennially vegetated buffers changed these properties in soils that were previously under row-crop cultivation. This conclusion is corroborated when comparing data collected from the same sites in 1998 and 1999 and indicate a 29% increase in soil C in the forest buffer over the last 9 yr. Decomposition of above and below-ground litter, root exudates, and microbial C accumulation may contribute to the observed C increase. Increased soil C resulting from conservation practices such as conversion from crop lands to grasslands or forest has been reported in other studies (Gebhart et al., 1994; Knops and Tilman, 2000; Uri, 2000; Post and Kwon, 2000; Guo and Gifford, 2002; McLauchlan et al., 2006). Johnson et al. (2005) reported that conversion of previous cropland to grass increased soil organic C by 4.2 ± 4.5 Mg C ha⁻¹ yr⁻¹ after 6–8 yr since establishment in the central United States.

We observed significantly higher soil moisture and lower soil temperature in the soils of riparian buffers compared to those of the crop field. This may be the result of the vegetation within riparian buffers providing more shade to prevent high temperatures in the summer and the lower soil bulk density and high organic matter of riparian buffers holding more soil moisture. In contrast, soils in conventionally cultivated crop fields are more exposed to direct sunlight, have higher bulk density and lower soil organic matter and tend to hold less soil moisture compared with riparian buffers soils.

**Methane Flux in Riparian Buffers**

Methane flux observed within the forest buffers and grass filter soils (−0.5 to 0.9 mg CH₄–C m⁻² d⁻¹, n = 45–50) is similar to results of studies conducted in other riparian systems with infrequent saturation. McLain and Martens (2006) found the CH₄ sink averaged 26.1 ± 6.3 μg CH₄ m⁻² h⁻¹ in the semiarid riparian soils of southeastern Arizona. In a riparian alder stand in southeastern Estonia, Teiter and Mander (2005) observed an average CH₄ flux of 0.1 to 265 μg CH₄–C m⁻² h⁻¹. However, the CH₄ flux in the forest buffers and grass filters soils was lower than those reported in other studies conducted in temporarily submerged areas such as rice (Oryza sativa L.) fields, wetlands, or riparian areas with frequent saturation. For example, Ambus and Christensen
(1995) found CH$_4$ produced at rates of 7877 mg CH$_4$–C m$^{-2}$ yr$^{-1}$ (78.8 kg CH$_4$–C ha$^{-1}$ yr$^{-1}$) in a temporarily flooded riparian area in Denmark. Methane was produced from riparian areas in northern Germany at rates of 33 to 33,030 mg CH$_4$–C m$^{-2}$ yr$^{-1}$ (0.33–330.3 kg CH$_4$–C ha$^{-1}$ yr$^{-1}$) (Merbach et al., 1996). Altor and Mitsch (2006) reported that annual CH$_4$ flux from intermittently flooded zones was 13 g CH$_4$–C m$^{-2}$ yr$^{-1}$ (130 kg CH$_4$–C ha$^{-1}$ yr$^{-1}$) in the midwestern United States. Le Mer and Roger’s (2001) review of the literature found that the median of CH$_4$ emissions were 0.72 kg CH$_4$ ha$^{-1}$ d$^{-1}$ (3 mg CH$_4$ m$^{-2}$ h$^{-1}$) in swamps, 0.43 kg CH$_4$ ha$^{-1}$ d$^{-1}$ (1.8 mg CH$_4$ m$^{-2}$ h$^{-1}$) in peat lands and 1.0 kg CH$_4$ ha$^{-1}$ d$^{-1}$ (4.2 mg CH$_4$ m$^{-2}$ h$^{-1}$) in rice fields. These results suggest that riparian zones soils under certain conditions are not major sources of CH$_4$ compared to wetlands, rice fields, or riparian zones with more frequent saturation. In the case of riparian zones in many areas of the midwestern United States, changes in landscape hydrology resulting from the conversion to agriculture have resulted in incised stream channels and lowered riparian water tables, likely altering conditions favorable to CH$_4$ production. At our sites, some riparian buffers were easily affected by flooding caused by snow melting (14 March) and heavy rainfall (26 April) and we conducted gas sampling when we were able to access the sites after flooding subsided. However, it is likely that because these conditions were so ephemeral, that observed CH$_4$ mass flux reflect the hydrologic characteristics of riparian buffers within this landform region.

Interannual variation of greenhouse gas flux within this region can be significant, as demonstrated by Chan and Parkin (2001a).
Because this study was conducted for only 1 yr (January 2007–December 2007), the interannual variation of \( \text{CH}_4 \) flux cannot be assessed for these sites. However, a main objective of this study was to compare \( \text{CH}_4 \) flux among crop fields and adjacent riparian buffers re-established for water quality. Since all sites were in close proximity and experienced similar conditions, it can be assumed that annual (or interannual) climate variability did not affect study conclusions. Since climatic conditions in 2007 (mean air temperature 9.4°C; annual precipitation 1097 mm) were within the standard deviation of conditions over the last 37 yr of record (mean air temperature 8.7 ± 0.8°C; mean annual precipitation 914 ± 210 mm) results from this study could be considered representative of flux rates measured over multiple years.

Results of soil incubation experiments, and diel and seasonal \( \text{CH}_4 \) flux measurements indicate that \( \text{CH}_4 \) flux in the crop field, forest buffers, and grass filters were not significantly different from one another. In contrast, in the same region (central Iowa), Chan and Parkin (2001a) found that forest and prairie soils were net \( \text{CH}_4 \) consumers, with cumulative \( \text{CH}_4 \) fluxes ranging from –0.27 to −0.07 g \( \text{CH}_4 \) m\(^{-2}\) (–2.7 to –0.7 kg \( \text{CH}_4 \) ha\(^{-1}\)) over the 258-d sampling season, while agricultural sites were net \( \text{CH}_4 \) producers, with cumulative \( \text{CH}_4 \) fluxes ranging from –0.02 to 3.19 g \( \text{CH}_4 \) m\(^{-2}\) (–0.2 to 31.9 kg \( \text{CH}_4 \) ha\(^{-1}\)) over the same season. The prairie and forest soils were found to have the greatest potential to oxidize atmospheric concentrations of \( \text{CH}_4 \) (Chan and Parkin, 2001b). Within temperate regions globally, reported \( \text{CH}_4 \) consumption rates include 1.29 ± 0.16 kg \( \text{CH}_4 \) ha\(^{-1}\) yr\(^{-1}\) in crop fields \((n = 48)\), 5.75 ± 0.59 kg \( \text{CH}_4 \) ha\(^{-1}\) yr\(^{-1}\) within grasslands \((n = 24)\), and 2.40 ± 0.40 kg \( \text{CH}_4 \) ha\(^{-1}\) yr\(^{-1}\) within forests \((n = 91)\) [data extracted from Dutaur and Verchot (2007)]. These reports indicate that \( \text{CH}_4 \) consumption within re-established riparian forest buffer and grass filter soils examined in this study were much lower than other reported \( \text{CH}_4 \) consumption rates within grasslands and forests in Iowa and the temperate regions. Such a contrast suggests that \( \text{CH}_4 \) soil oxidation capacity has not been improved during the 7 to 17 yr following re-establishment of perennial vegetation (forest buffers and grass filters) on conventional crop fields, even when soil properties such as soil bulk density pH, TC, and soil moisture have changed. It is well known that \( \text{CH}_4 \) oxidation potential of upland soils is reduced by cultivation and ammonium N-fertilizer application (e.g., Topp and Pattey, 1997; Le Mer and Roger, 2001; Dutaur and Verchot, 2007). Le Mer and Roger (2001) summarized the effects of cultural practices on \( \text{CH}_4 \) oxidation as following: (i) an increase in \( \text{NH}_4^+ \) content of soil by fertilizer application inhibits \( \text{CH}_4 \) oxidation because \( \text{NH}_4^+ \) produces competition at the level of methane-mono-oxygenase, a transfer of the \( \text{CH}_4 \) oxidizing activity toward nitrification (Castro et al., 1994; Nesbit and Breitenbeck, 1992), and (ii) cultural practices that destroy micro-aerophilic niches suitable for \( \text{CH}_4 \) oxidizers reduce \( \text{CH}_4 \) oxidation (Huntsch et al., 1994; Sitaula et al., 2000). Slow recovery of \( \text{CH}_4 \) oxidation after land use change has been reported. In a range of successional sites on former arable land in Denmark and Scotland, \( \text{CH}_4 \) oxidation rates took more than 100 yr to reach precultivation levels (Priemé et al., 1997). Similarly, Suwanwaree and Robertson (2005) observed that rates of \( \text{CH}_4 \) oxidation in soils of 40 to 60 yr-old successional fields were between those of the no-till and deciduous forest sites in southwest United States. Singh et al. (2007) reported that afforestation and reforestation of pastures (30–50 yr later) resulted in changes in methane oxidation by altering the community structure of methanotrophic bacteria in these soils. In the case of the re-established riparian buffers investigated in this study, it appears that, while soil properties have been altered, additional time is needed for changes in \( \text{CH}_4 \) flux to be manifested.

Conclusions

Soil properties such as soil bulk density, pH, TC, and soil moisture in riparian forest buffer and grass filter soils were significantly different from those in adjacent crop fields, suggesting that soil properties have changed since re-establishment of perennial vegetation on previously cultivated field soils. Soil incubation experiments provide some indication that \( \text{CH}_4 \) consumption was higher than \( \text{CH}_4 \) production in forest buffers.
and grass filters soils, while crop field soils showed the opposite response. However, none of the CH₄ fluxes from incubation experiments were significantly different, nor were diel and seasonal variation of CH₄ fluxes in forest buffers, grass filters, and adjacent crop fields. The cumulative CH₄ flux of −0.80 kg CH₄-C ha⁻¹ yr⁻¹ in the crop field, −0.46 kg CH₄-C ha⁻¹ yr⁻¹ in forest buffers, and 0.04 kg CH₄-C ha⁻¹ yr⁻¹ in grass were also not significantly different. The CH₄ flux in forest buffers and grass filter soils was less than that reported for wetlands, rice fields, or riparian areas with more frequent saturation, which are known to be sinks of CH₄. These results suggest that these re-established riparian forest buffers and grass filters, possibly due to altered hydrology, cannot be considered as major sources of CH₄ as has been found in other riparian areas or systems with more frequent saturation. However, any potential benefit as increased sinks of CH₄ has not yet been achieved after 7 to 17 yr since re-establishment. These results have important management implications given the significant effort to promote such systems for water quality improvement and other ecosystem services.

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