12-2009

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

Dong-Gill Kim
Iowa State University

Thomas M. Isenhart
Iowa State University, isenhart@iastate.edu

Timothy B. Parkin
United States Department of Agriculture

Richard C. Schultz
Iowa State University, rschultz@iastate.edu

Thomas E. Loynachan
Iowa State University, teloynac@iastate.edu

Methane Flux in Cropland and Adjacent Riparian Buff ers with Different Vegetation Covers

Abstract
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 (CH(4)), these systems have the potential to serve as sinks of CH(4) or may provide favorable conditions for CH(4) production. This study quantifies CH(4) 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 CH(4) 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 CH(4) flux and soil moisture or soil temperature among sites within the range of conditions observed. Cumulative CH(4) flux was -0.80 kg CH(4)-C ha(-1) yr(-1) in the cropped field, -0.46 kg CH(4)-C ha(-1) yr(-1) within the forest buffers, and 0.04 kg CH(4)-C ha(-1) yr(-1) within grass filters, but difference among vegetation covers was not significant. Results suggest that CH(4) flux was not changed after establishment of perennial vegetation on cropped soils, despite significant changes in soil properties.

Keywords
rivers, methane, pollutants, crops, riparian buffer

Disciplines
Agronomy and Crop Sciences | Hydrology | Natural Resources Management and Policy

Comments
This article is from *Journal of Environmental Quality* 39 (2009): 97, doi:10.2134/jeq2008.0408.

Published January, 2010

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

Dong-Gill Kim and Thomas M. Isenhart* Iowa State University
Timothy B. Parkin USDA-ARS
Richard C. Schultz and Thomas E. Loynachan Iowa State University

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 (CH_4), these systems have the potential to serve as sinks of CH_4 or may provide favorable conditions for CH_4 production. This study quantifies CH_4 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 CH_4 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 CH_4 flux and soil moisture or soil temperature among sites within the range of conditions observed. Cumulative CH_4 flux was –0.80 kg CH_4–C ha^{-1} yr^{-1} in the cropped field, –0.46 kg CH_4–C ha^{-1} yr^{-1} within the forest buffers, and 0.04 kg CH_4–C ha^{-1} yr^{-1} within grass filters, but difference among vegetation covers was not significant. Results suggest that CH_4 flux was not changed after establishment of perennial vegetation on cropped soils, despite significant changes in soil properties.

Published in J. Environ. Qual. 39:97–105 (2010).
doi:10.2134/jeq2008.0408
Published online 22 Oct. 2009.
Received 12 Sept. 2008..
*Corresponding author (isenhart@iastate.edu).

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

The global atmospheric concentration of CH_4 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 CH_4 (Topp and Pattey, 1997; Le Mer and Roger, 2001). In a recent review, Dutaur and Verchot (2007) summarized net CH_4 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 CH_4 by soil. It is well known that forest soils are the most active sink of CH_4, followed by grass lands and cultivated soils, and that the CH_4 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; Dutaur and Verchot, 2007). It has been reported that land-use change can also influence CH_4 uptake rates. For instance, higher rates of CH_4 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 CH_4 uptake resulting from land-use change are attributed to changes in soil porosity, moisture content, and methanotroph population (Priemé 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

D.-G. Kim, T.M. Isenhart, and R.C. Schultz, Dep. of Natural Resource Ecology and Management, Iowa State Univ., Ames, IA 50011-3221. T.B. Parkin, USDA-ARS, National Soil Tilth Lab., Ames, IA 50011. T.E. Loynachan, Dep. of Agronomy, Iowa State Univ., Ames, Iowa 50011-1010.

Abbreviations: IPCC, Intergovernmental Panel on Climate Change; NPS, nonpoint source.

riparian buffers re-established on cropped soils may produce less and consume more CH_4 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 CH_4 production. For example, Ambus and Christensen (1995) reported that CH_4 was produced in temporarily flooded riparian areas at rates of 78.8 kg CH_4–C ha^{-1} yr^{-1}. Methane was also produced from riparian areas of ponded depressions in northern Germany at rates of 0.33 to 330.3 kg CH_4–C ha^{-1} yr^{-1} (Merbach et al., 1996). In contrast, rates of CH_4 consumption in temperate regions have been estimated as 1.29 ± 0.16 kg CH_4 ha^{-1} yr^{-1} in crop fields (n = 48), 5.75 ± 0.59 kg CH_4 ha^{-1} yr^{-1} in grasslands (n = 24), and 2.40 ± 0.40 kg CH_4 ha^{-1} yr^{-1} in forests (n = 91) (data extracted from Dutaur and Verchot (2007)). These results suggest that reestablished riparian buffers may produce more CH_4 than crop fields and natural lands, at least when they are flooded, 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 many 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 found conflicting results regarding the effect of vegetation type on biogeochemical process regulation. For example, there are uncertainties about the effect of vegetation type on groundwater NO_3^- removal or denitrification in riparian buffers (e.g., Groffman et al., 1991; Schnabel et al., 1996; Hefting et al., 2003). With respect to CH_4 flux, several studies have compared rates among vegetation types, (Topp and Pattey, 1997; Le Mer and Roger, 2001; Chan and Parkin, 2001a, 2001b; Dutaur and Verchot, 2007). However, few studies of CH_4 flux have 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 CH_4 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.

Materials and Methods

Study Site

The study area consisted of three forest buffers, three warm-season grass filters, one cool-season grass filter, 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 area is now rain-fed agriculture cultivated with soybean [*Glycine max* (L.) Merr.] and corn (*Zea mays* L.), which are usually grown in rotation. Re-established forest buffers, and warm-season and cool-season grass filters were previously under row-crop cultivation or livestock grazing. The forest buffers and grass filters ranged in age from 7 to 17 yr since re-establishment.

Forest buffers included the following tree and shrub species: silver maple (*Acer saccharinum* L.), green ash (*Fraxinus pennsylvanica* Marsh.), black walnut (*Juglans nigra* L.), willow (*Salix* spp.), cottonwood hybrids (*Populus* spp.), red oak (*Quercus rubra* L.), and bur oak (*Q. bicolor* Willd). Shrub species included chokecherry (*Prunus virginiana* L.), Nanking cherry (*P. tomentosa* Thunb), wild plum (*P. americana* Marsh), red osier dogwood (*Cornus stolonifera* Michx.), and ninebark (*Physocarpus opulifolius* Maxim.). Warm-season grass filters included native species such as switchgrass (*Panicum virgatum* L.), Indian grass (*Sorghastrum nutans* Nash), and big bluestem (*Andropogon gerardii* Vitman). Numerous forb species were present, including purple prairie clover (*Petalostemum purpureum* Vent.), black-eyed susan (*Rudbeckia hirta* L.), yellow coneflower (*Ratibida pinnata* Vent.), stiff goldenrod (*Solidago rigida* L.), prairie blazing star (*Liatris pycnostachya* Michx.), and others. The cool-season grass buffers were dominated by nonnative forage grasses (*Bromus inermis* Leysser, *Phleum pratense* L., and *Poa pratensis* L). Details of the riparian buffer design, placement, and plant species are given in Schultz et al. (1995). The crop fields adjacent to the riparian buffers served as a control, representing conditions before buffer establishment. The crop fields were planted to soybean in 2007. Pelletized urea (134 kg N ha^{-1}) was applied during corn rotation years and cultivation consists of fall chisel plowing (15–20 cm depth). All areas used in this study were located on Coland soil (fine-loamy,mixed, superactive, mesic Cumulic Endoaquoll) which is well- to poorly drained and formed from till or local alluvium and colluvium derived from till (DeWitt, 1984).

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 (NO_3^-) and ammonium (NH_4^+) contents were analyzed by colorimetric method (Mulvaney, 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:

$$\text{Minimum detectable } CH_4 \text{ flux}(\mu L\ CH_4\text{-C}\ m^{-2}\ h^{-1}) = \frac{2 \times \text{average of standard deviation}(\mu L\ L^{-1}) \times \text{chamber volume (L)}}{\text{chamber footprint } (m^2) \times \text{chamber enclosure time (h)}} \quad [1]$$

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}. 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 Scientifc, 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, Oneset 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_2H_2 and three were retained as controls (no C_2H_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_2H_2 (Chan and Parkin, 2000, 2001b). Net CH_4 flux was determined from CH_4 flux in soil incubations without C_2H_2.

$$\text{Aerobic } CH_4 \text{ production} = CH_4 \text{ flux under 10 Pa } C_2H_2 \quad [2]$$

$$\text{Net } CH_4 \text{ flux} = CH_4 \text{ flux under no } C_2H_2 \quad [3]$$

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

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)].

Site	Soil texture†	Bulk density	pH	TC	TN	NH_4–N	NO_3–N
		Mg m^{-3}		g kg^{-1} soil		mg N kg^{-1}soil	
Crop field	Loam	1.67 ± 0.02a‡	5.9 ± 0.1c	22.8 ± 1.0c	1.9 ± 0.1c	1.7 ± 0.2b	1.2 ± 0.5a
Forest buffer	Loam, Sandy loam	1.10 ± 0.03c	7.3 ± 0.1a	42.9 ± 3.2a	3.8 ± 0.3a	4.1 ± 0.6a	0.7 ± 0.2a
Warm-season grass filter	Loam	1.29 ± 0.05b	6.7 ± 0.2b	29.1 ± 2.7bc	2.6 ± 0.2bc	3.9 ± 0.5a	0.2 ± 0.1a
Cool-season grass filter	Loam	1.19 ± 0.04bc	6.9 ± 0.1ab	32.4 ± 1.6bc	2.9 ± 0.1b	4.3 ± 0.4a	0.9 ± 0.3a

† Marquez et al. (2004).

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

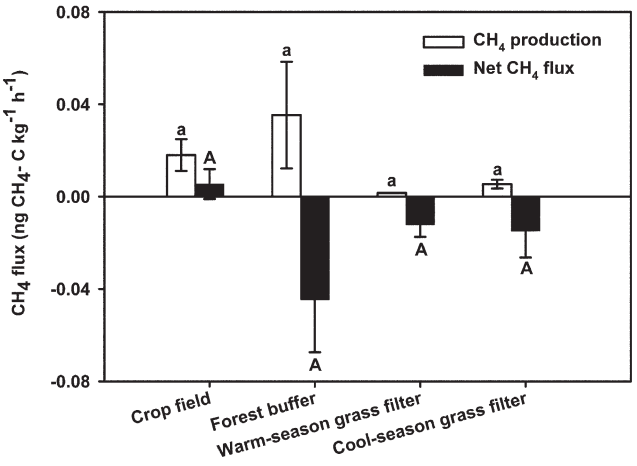

Fig. 1. Aerobic CH_4 production and net CH_4 flux from incubated soil core under controlled laboratory conditions. Soil cores were obtained from crop field, forest buffer, warm-season grass filter and cool-season grass filter soils. Each mean represents three observations (one observation for CH_4 production in warm-season grass filter soil) and bars are the standard error of the mean.

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 CH_4 flux, correlation analysis using the GLM procedure was applied. Statistical analyses were conducted using SAS ver 8.1 (SAS Institute, 1999).

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 NH_4^+ ($P < 0.01$) than crop field soils. TC ($P < 0.01$) and TN ($P < 0.01$) within the pooled riparian buffer vegetation soils were significantly higher that crop field soils. However, this difference is apparently driven by the forest buffer soils, as indicated in pairwise comparisons. Soil NO_3^- was not significantly different among sites (Table 1). Within the same sites, soil carbon content (0–15 cm soil depth) was 30.4 ± 1.6 g C kg $soil^{-1}$ ($n = 6$) in the forest buffer, 24.4 ± 1.0 g kg^{-1} ($n = 6$) in the warm-season grass filter, and 31.0 ± 1.8 g kg^{-1} ($n = 6$) in the cool-season grass filter in 1998 and 1999 (J. Raich, unpublished data, 1999). Comparing these data with those of this study (Table 1), soil C in the forest buffer (42.9 ± 3.2 g kg^{-1}, $n = 6$) in this study was significantly higher than those in 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 significantly 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 significantly 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 significant (one-way ANOVA $P < 0.01$). Aerobic CH_4 production in the incubated forest buffer soils ($3.5 \times 10^{-2} \pm 2.3 \times 10^{-2}$ ng CH_4–C kg soil $^{-1}$ h^{-1}, $n = 3$), warm-season grass filter soils (1.6×10^{-5} ng CH_4–C kg soil $^{-1}$ h^{-1}, $n = 1$), and cool-season grass filter soils ($5.4 \times 10^{-3} \pm 1.9 \times 10^{-3}$ ng CH_4–C kg soil $^{-1}$ h^{-1}, $n = 3$) and crop field soils ($1.8 \times 10^{-2} \pm 6.9 \times 10^{-3}$ ng CH_4–C kg $soil^{-1}$ h^{-1}, $n = 3$) were not significantly different (Fig. 1). Similarly, net CH_4 fluxes in the incubated forest buffer soils ($-4.5 \times 10^{-2} \pm 2.3 \times 10^{-2}$ ng CH_4–C kg $soil^{-1}$ h^{-1}, $n = 3$), warm-season grass filter soils ($-1.2 \times 10^{-2} \pm 5.3 \times 10^{-3}$ ng CH_4–C kg $soil^{-1}$ h^{-1}, $n = 3$), and cool-season grass filter soils ($-1.5 \times 10^{-2} \pm 1.2 \times 10^{-2}$ ng CH_4–C kg $soil^{-1}$ h^{-1}, $n = 3$) and crop field soils ($5.4 \times 10^{-3} \pm 6.5 \times 10^{-3}$ ng CH_4–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 CH_4 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 CH_4 flux among vegetation types (Fig. 2A). The average CH_4 flux ($n = 8$) was 6.9 ± 12.2 μg CH_4–C m^{-2} h^{-1} in crop field,-23.6 ± 8.1 μg CH_4–C m^{-2} h^{-1} within the forest buffer, –19.1 ± 10.6 μg CH_4–C m^{-2} h^{-1} within the warm-season grass filter and –25.5 ± 10.2 μg CH_4– C m^{-2} h^{-1} within the cool-season grass filter. The variation of CH_4 flux in the crop field, forest buffer, warm-season grass filter and cool-season grass filter was not correlated with soil temperature. Cumulative CH_4 flux over 24 h was 165.7 μg CH_4–C m^{-2} d^{-1} in the crop field,-567.3 μg CH_4–C m^{-2} d^{-1} in the forest buffer, –459.1 μg CH_4–C m^{-2} d^{-1} in the warm-season

grass filter, and-612.7 μg CH_4–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 CH_4 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 CH_4 flux was 2.4 mg CH_4–C m^{-2} d^{-1} in crop fields (7 August), 2.4 mg CH_4–C m^{-2} d^{-1} in forest buffers (1 May), and 1.6 mg CH_4–C m^{-2} d^{-1} and grass filters (11 September) (Fig. 3). Observed maximum negative daily CH_4 flux was –2.2 mg CH_4–C m^{-2} d^{-1} in crop fields (16 January), –2.6 mg CH_4–C m^{-2} d^{-1} in forest buffers (20 September), and –3.7 mg CH_4–C m^{-2} d^{-1} (20 September) grass filters (Fig. 3). Mean daily CH_4 flux was –0.2 ± 0.1 mg CH_4–C m^{-2} d^{-1} (n = 40) in the crop field, –0.5 to 0.9 mg CH_4–C m^{-2} d^{-1} (n = 45–50) in forest buffers, and –0.2 to 0.1 mg CH_4–C m^{-2} d^{-1} (n = 41–49) in grass filters, with no significant differences (Fig. 4). There were no significant relationships between CH_4 flux and soil moisture (P > 0.05) or soil temperature (P > 0.05) in any of the sites. Cumulative CH_4 flux from January to December 2007 was –0.80 kg CH_4–C ha^{-1} yr^{-1} (n = 1) in the crop field, –0.46 ± 0.48 kg CH_4–C ha^{-1} yr^{-1} (n = 3) in forest buffers, and 0.04 ± 0.2 kg CH_4–C ha^{-1} yr^{-1} (n = 4) in grass filters (Fig. 5). Cumulative CH_4 flux in forest buffers (95% confidence interval (CI): –2.54 to 1.61 kg CH_4–C ha^{-1} yr^{-1}) and grass filters (95% CI: –0.51 to 0.61 kg CH_4–C ha^{-1} yr^{-1}) were not significantly different from zero. The cumulative CH_4 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).

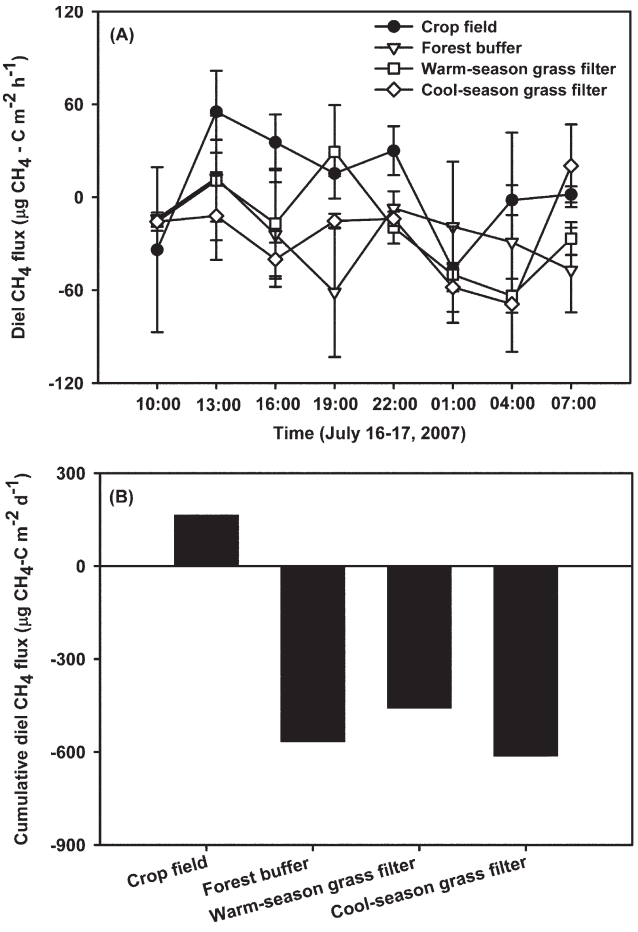

Fig. 2. (A) Diel variation of CH_4 flux in crop field, forest buffer, warm-season grass filter and cool-season grass filter soils on 16–17 July 2007 and (B) cumulative diel CH_4 flux. Each mean represents three observations and bars are the standard error of the mean.

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 NH_4^+ 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 CH_4–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 CH_4 sink averaged 26.1 ± 6.3 μg CH_4 m^{-2} h^{-1} in the semiarid riparian soils of southeastern Arizona. In a riparian alder stand in southern Estonia, Teiter and Mander (2005) observed an average CH_4 flux of 0.1 to 265 μg CH_4–C m^{-2} h^{-1}. However, the CH_4 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

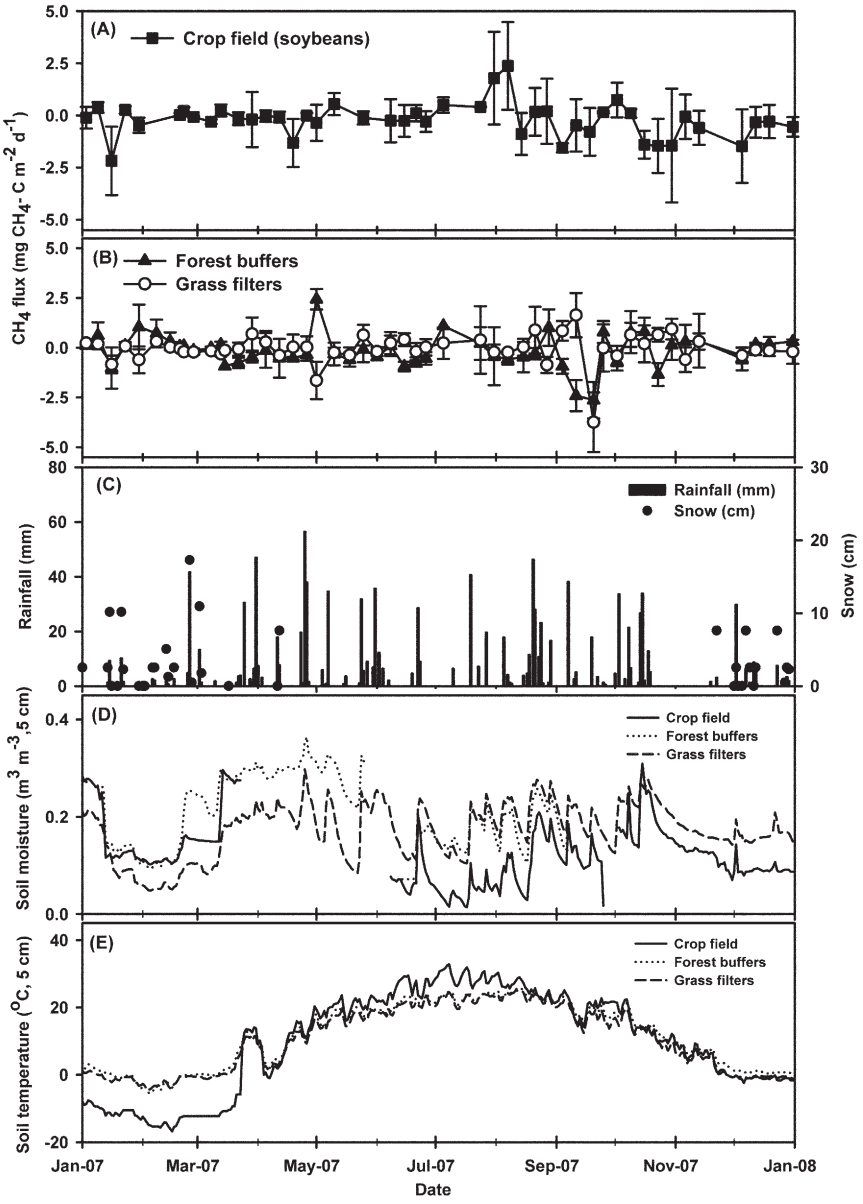

Fig. 3. (A,B) Methane flux, (C) daily precipitation, (D) daily soil moisture, and (E) soil temperature in crop fields (n = 1), forest buffers (n = 3), and grass filters (n = 4) in 2007. Each mean represents observations and bars are the standard error of the mean. Gaps in the soil moisture data (D) were caused by either data loggers malfunction or removing a data logger during planting and harvesting periods in the crop field.

(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 CH_4 flux cannot be assessed for these sites. However, a main objective of this study was to compare CH_4 flux among crop fields and adjacent riparian buffers 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 CH_4 flux measurements indicate that 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 CH_4 consumers, with cumulative CH_4 fluxes ranging from –0.27 to –0.07 g CH_4 m^{-2} (–2.7 to –0.7 kg CH_4 ha^{-1}) over the 258-d sampling season, while agricultural sites were net CH_4 producers, with cumulative CH_4 fluxes ranging from –0.02 to 3.19 g CH_4 m^{-2} (–0.2 to 31.9 kg 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 CH_4 (Chan and Parkin, 2001b). Within temperate regions globally, reported CH_4 consumption rates include 1.29 ± 0.16 kg CH_4 ha^{-1} yr^{-1} in crop fields (n = 48), 5.75 ± 0.59 kg CH_4 ha^{-1} yr^{-1} within grasslands (n = 24), and 2.40 ± 0.40 kg CH_4 ha^{-1} yr^{-1} within forests (n = 91) [data extracted from Dutaur and Verchot (2007)]. These reports indicate that CH_4 consumption within re-established riparian forest buffer and grass filter soils examined in this study were much lower than other reported CH_4 consumption rates within grasslands and forests in Iowa and the temperate regions. Such a contrast suggests that 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 significantly. It is well known that 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 CH_4 oxidation as following: (i) an increase in NH_4^+ content of soil by fertilizer application inhibits CH_4 oxidation because NH_4^+ produces competition at the level of methane-mono-oxygenase, a transfer of the 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 CH_4 oxidizers reduce CH_4 oxidation (Hütsch et al., 1994; Sitaula et al., 2000). Slow recovery of CH_4 oxidation after land use change has been reported. In a range of successional sites on former arable land in Denmark and Scotland, 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 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 CH_4 flux to be manifested.

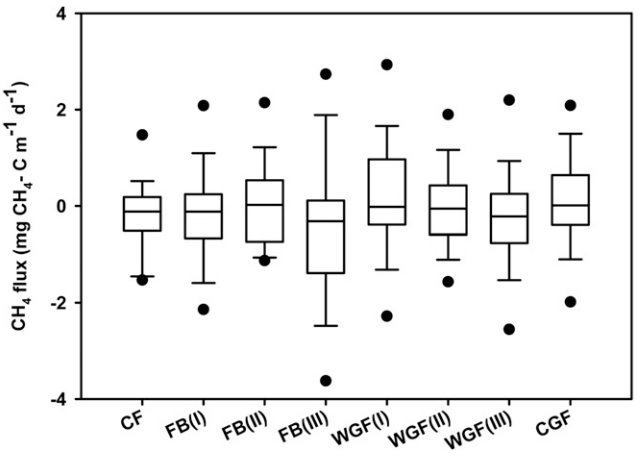

Fig. 4. CH_4 flux in crop fields (CF), forest buffers (FB), warm-season grass filters (WGF), and cool-season grass filter (CGF) soils in 2007 (n = 40–49). I, II, and III indicate replicates. The boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles. Solid circles indicate outliers.

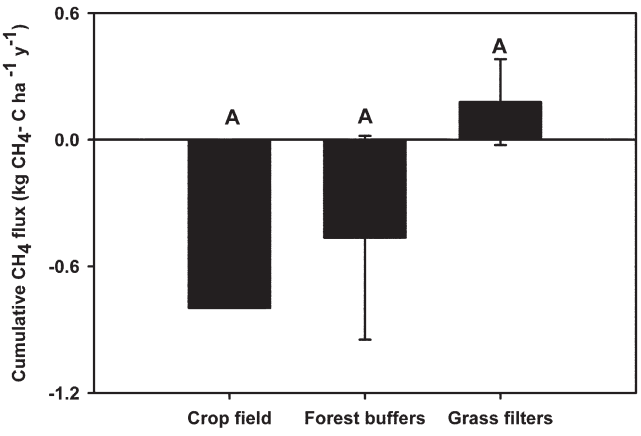

Fig. 5. Annual CH_4 flux in crop fields (n = 1), forest buffers (n = 3), and grass filters (n = 4) sites in 2007. Each mean represents observations and bars are the standard error of the mean.

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 crop field soils. Soil incubation experiments provide some indication that CH_4 consumption was higher than CH_4 production in forest buffers

and grass filters soils, while crop field soils showed the opposite response. However, none of the CH_4 fluxes from incubation experiments were significantly different, nor were diel and seasonal variation of CH_4 fluxes in forest buffers, grass filters, and adjacent crop fields. The cumulative CH_4 flux –0.80 kg CH_4–C ha^{-1} yr^{-1} in the crop field, –0.46 kg CH_4–C ha^{-1} yr^{-1} in forest buffers, and 0.04 kg CH_4–C ha^{-1} yr^{-1} in grass were also not significantly different. The CH_4 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 sources of CH_4. The CH_4 flux rates reported here are also greater than those reported for forests and grasslands, which are known to be sinks of CH_4. 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 CH_4 as has been found in other riparian areas or systems with more frequent saturation. However, any potential benefit as increased sinks of CH_4, as has been found for other perennial plant systems within the region, 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.

Acknowledgments

The authors wish to thank Leigh Ann Long, Otis Smith, and numerous students for technical assistance in the field and lab. We are also grateful to Dr. James Raich for his valuable comments on earlier versions of the manuscript. Financial support was provided US EPA, USDA NRI, USDA SARE Graduate Student Grant Program, Leopold Center for Sustainable Agriculture, University of Iowa Center for Global and Regional Environmental Research, and University of Missouri Center for Agroforestry.

References

Altor, A.E., and W.J. Mitsch. 2006. Methane flux from created riparian marshes: Relationship to intermittent versus continuous inundation and emergent macrophytes. Ecol. Eng. 28:224–234.

Ambus, P., and S. Christensen. 1995. Spatial and seasonal nitrous oxide and methane fluxes in danish forest-ecosystems, grassland-ecosystems, and agroecosystems. J. Environ. Qual. 24:993–1001.

Arnold, S.L., T.B. Parkin, J.W. Doran, B. Eghball, and A.R. Mosier. 2001. Automated gas sampling system for laboratory analysis of CH_4 and N_2O. Commun. Soil Sci. Plant Anal. 32:2795–2807.

Ball, B.C., I.P. McTaggart, and C.A. Watson. 2002. Influence of organic ley-arable management and afforestation in sandy loam to clay loam soils on fluxes of N_2O and CH_4 in Scotland. Agric. Ecosyst. Environ. 90:305–317.

Bharati, L., K.-H. Lee, T.M. Isenhart, and R.C. Schultz. 2002. Riparian zone soil-water infiltration under crops, pasture and established buffers. Agrofor. Syst. 56:249–257.

Castro, M.S., W.T. Peterjohn, J.M. Melillo, P.A. Steudler, H.L. Gholz, and D. Lewis. 1994. Effects of nitrogen-fertilization on the fluxes of N_2O, CH_4, and CO_2 from soils in a Florida slash pine plantation. Can. J. For. Res. 24:9–13.

Chan, A.S.K., and T.B. Parkin. 2000. Evaluation of potential inhibitors of methanogenesis and methane oxidation in a landfill cover soil. Soil Biol. Biochem. 32:1581–1590.

Chan, A.S.K., and T.B. Parkin. 2001a. Effect of land use on methane flux from soil. J. Environ. Qual. 30:786–797.

Chan, A.S.K., and T.B. Parkin. 2001b. Methane oxidation and production activity in soils from natural and agricultural ecosystems. J. Environ. Qual. 30:1896–1903.

DeWitt, T.A. 1984. Soil survey of Story County, Iowa. USDA Soil Conserv. Serv., Washington, DC.

Duda, A.M. 1993. Addressing nonpoint sources of water pollution must become an international priority. Water Sci. Technol. 28:1–11.

Dutaur, L., and L.V. Verchot. 2007. A global inventory of the soil CH_4 sink. Global Biogeochem. Cycles 21:GB4013 doi:10.1029/2006GB002734.

Gebhart, D.L., H.B. Johnson, H.S. Mayeux, and H.W. Polley. 1994. The CRP increases soil organic carbon. J. Soil Water Conserv. 49:488–492.

Gilbert, R.O. 1987. Statistical methods for environmental pollution monitoring. Van Nostrand Reinhold Company, New York.

Groffman, P.M., E.A. Axelrod, J.L. Lemunyon, and W.M. Sullivan. 1991. Denitrification in grass and forest vegetated filter strips. J. Environ. Qual. 20:671–674.

Grossman, R.B., and T.G. Reinsch. 2002. Bulk density and linear extensibility. p. 201–228. *In* J.H. Dane and G.C. Topp (ed.) Methods of soil analysis, Part 4: Physical methods. SSSA Book Ser. no. 5. ASA and SSSA, Madison, WI.

Guo, L.B., and R.M. Gifford. 2002. Soil carbon stocks and land use change: A meta analysis. Glob. Change Biol. 8:345–360.

Hefting, M.M., R. Bobbink, and H. de Caluwe. 2003. Nitrous oxide emission and denitrification in chronically nitrate-loaded riparian buffer zones. J. Environ. Qual. 32:1194–1203.

Holland, E.A., G.P. Robertson, J. Greenburg, P. Groffman, R. Boone, and J. Gosz. 1999. Soil CO_2, N_2O, and CH_4 exchange. p. 185–201. *In* G.P. Robertson et al. (ed.) Standard soil methods for long-term ecological research. Oxford Univ. Press, New York.

Hubbard, R.K., J.M. Sheridan, R. Lowrance, D.D. Bosch, and G. Vellidis. 2004. Fate of nitrogen from agriculture in the southeastern Coastal Plain. J. Soil Water Conserv. 59:72–86.

Hütsch, B.W., C.P. Webster, and D.S. Powlson. 1994. Methane oxidation in soil as affected by land-use, soil-pH and N-fertilization. Soil Biol. Biochem. 26:1613–1622.

Intergovernmental Panel on Climate Change. 2007. Summary for policy makers. p. 1–18. *In* S.D. Solomon et al. (ed.) Climate change 2007: The physical science basis. Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge Univ. Press, New York.

Johnson, J.M.F., D.C. Reicosky, R.R. Allmaras, T.J. Sauer, R.T. Venterea, and C.J. Dell. 2005. Greenhouse gas contributions and mitigation potential of agriculture in the central USA. Soil Tillage Res. 83:73–94.

Knops, J.M.H., and D. Tilman. 2000. Dynamics of soil nitrogen and carbon accumulation for 61 years after agricultural abandonment. Ecology 81:88–98.

Le Mer, J., and P. Roger. 2001. Production, oxidation, emission and consumption of methane by soils: A review. Eur. J. Soil Biol. 37:25–50.

Marquez, C.O., V.J. Garcia, C.A. Cambardella, R.C. Schultz, and T.M. Isenhart. 2004. Aggregate-size stability distribution and soil stability. Soil Sci. Soc. Am. J. 68:725–735.

Mayer, P.M., S.K. Reynolds, M.D. McCutchen, and T.J. Canfield. 2007. Meta-analysis of nitrogen removal in riparian buffers. J. Environ. Qual. 36:1172–1180.

McLain, J.E.T., and D.A. Martens. 2006. Moisture controls on trace gas fluxes in semiarid riparian soils. Soil Sci. Soc. Am. J. 70:367–377.

McLauchlan, K.K., S.E. Hobbie, and W.M. Post. 2006. Conversion from agriculture to grassland builds soil organic matter on decadal timescales. Ecol. Appl. 16:143–153.

Merbach, W., J. Augustin, T. Kalettka, and H.J. Jacob. 1996. Nitrous oxide and methane emissions from riparian areas of ponded depressions of northeast Germany. J. Appl. Bot. 70:134–136.

Merino, A., P. Perez-Batallon, and F. Macias. 2004. Responses of soil organic matter and greenhouse gas fluxes to soil management and land use changes in a humid temperate region of southern Europe. Soil Biol. Biochem. 36:917–925.

Mulvaney, R.L. 1996. Nitrogen–Inorganic forms. p. 1123–1184. *In* D.L. Sparks et al (ed.) Methods of soil analysis. Part 3. SSSA Book Ser. no. 5. ASA and SSSA, Madison, WI.

Nesbit, S.P., and G.A. Breitenbeck. 1992. A laboratory study of factors influencing methane uptake by soils. Agric. Ecosyst. Environ. 41:39–54.

Post, W.M., and K.C. Kwon. 2000. Soil carbon sequestration and land-use change: Processes and potential. Glob. Change Biol. 6:317–327.

Priemé, A., S. Christensen, K.E. Dobbie, and K.A. Smith. 1997. Slow increase in rate of methane oxidation in soils with time following land use change

from arable agriculture to woodland. Soil Biol. Biochem. 29:1269–1273.

Sabater, S., A. Butturini, J.C. Clement, T. Burt, D. Dowrick, M. Hefting, V. Matre, G. Pinay, C. Postolache, and M. Rzepecki. 2003. Nitrogen removal by riparian buffers along a European climatic gradient: Patterns and factors of variation. Ecosystems 6:20–30.

SAS Institute. 1999. SAS/STAT user's guide. Version 8. SAS Inst., Cary, NC.

Schnabel, R.R., L.F. Cornish, W.L. Stout, and J.A. Shaffer. 1996. Denitrification in a grassed and a wooded, valley and ridge, riparian ecotone. J. Environ. Qual. 25:1230–1235.

Schultz, R.C., J.P. Colletti, T.M. Isenhart, C.O. Marquez, W.W. Simpkins, and C.J. Ball. 2000. Riparian forest buffer practices. p. 189–281. *In* H.E. Garrett et al. (ed.) North American agroforestry: An integrated science and practice. ASA, Madison, WI.

Schultz, R.C., J.P. Colletti, T.M. Isenhart, W.W. Simpkins, C.W. Mize, and M.L. Thompson. 1995. Design and placement of a multispecies riparian buffer strip system. Agrofor. Syst. 29:201–226.

Schultz, R.C., T.M. Isenhart, W.W. Simpkins, and J.P. Colletti. 2004. Riparian forest buffers in agroecosystems-lessons learned from the Bear Creek Watershed, central Iowa, USA. Agrofor. Syst. 61:35–50.

Singh, B.K., K.R. Tate, G. Kolipaka, C.B. Hedley, C.A. Macdonald, P. Millard, and J.C. Murrell. 2007. Effect of afforestation and reforestation of pastures on the activity and population dynamics of methanotrophic bacteria. Appl. Environ. Microbiol. 73:5153–5161.

Sitaula, B.K., S. Hansen, J.I.B. Sitaula, and L.R. Bakken. 2000. Methane oxidation potentials and fluxes in agricultural soil: Effects of fertilisation and soil compaction. Biogeochemistry 48:323–339.

Suwanwaree, P., and G. Robertson. 2005. Methane oxidation in forest, successional, and no-till agricultural ecosystems: Effects of nitrogen and soil disturbance. Soil Sci. Soc. Am. J. 69:1722–1729.

Tate, K.R., D.J. Ross, S. Saggar, C.B. Hedley, J. Dando, B.K. Singh, and S.M. Lambie. 2007. Methane uptake in soils from *Pinus radiata* plantations, a reverting shrubland and adjacent pastures: Effects of land-use change, and soil texture, water and mineral nitrogen. Soil Biol. Biochem. 39:1437–1449.

Teiter, S., and U. Mander. 2005. Emission of N_2O, N_2, CH_4, and CO_2 from constructed wetlands for wastewater treatment and from riparian buffer zones. Ecol. Eng. 25:528–541.

Tonderski, A. 1996. Landuse-based nonpoint source pollution. A threat to water resources in developing countries. Water Sci. Technol. 33:53–61.

Topp, E., and E. Pattey. 1997. Soils as sources and sinks for atmospheric methane. Can. J. Soil Sci. 77:167–178.

Tufekcioglu, A., J.W. Raich, T.M. Isenhart, and R.C. Schultz. 1999. Root biomass, soil respiration, and root distribution in crop fields and riparian buffer zones. Agrofor. Syst. 44:163–174.

United States Department of Commerce–National Oceanic and Atmospheric Administration. 2009. Available at http://www.ncdc.noaa.gov/oa/ncdc.html (verified 27 Aug. 2009). Natl. Climatic Data Ctr., Asheville, NC.

Uri, N.D. 2000. Conservation practices in US agriculture and their implication for global climate change. Sci. Total Environ. 256:23–38.

Van Miegroet, H. 1995. Inorganic nitrogen determined by laboratory and field extractions of two forest soils. Soil Sci. Soc. Am. J. 59:549–553.