Soil methane and carbon dioxide fluxes from cropland and riparian buffers in different hydrogeomorphic settings

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ABSTRACT

Riparian buffers contribute to the mitigation of nutrient pollution in agricultural landscapes, but there is concern regarding their potential to be hot spots of greenhouse gas production. This study compared soil carbon dioxide (CO_2) and methane (CH_4) fluxes in adjacent crop fields and riparian buffers (a flood-prone forest and a flood-protected grassland along an incised channel), and examined the impact of water table depth (WTD) and flood events on the variability of gas fluxes in riparian zones. Results showed significantly ($P < 0.001$) higher CO_2 emission in riparian areas than in adjoining croplands (6.8±0.6 vs 3.6±0.5 Mg CO_2-C ha^{-1} y^{-1}; mean± standard error). Daily flux of CO_2 and soil temperature were significant related ($P < 0.002$), with Q_{10} values ranging between 1.75 and 2.53. Significant relationships ($P < 0.05$) were found between CH_4 daily flux and WTD. Flood events resulted in enhanced CH_4 emission (up to +44.5 mg CH_4-C m^{-2} d^{-1} in a swale) under warm soil conditions (> 22 ^oC), but the effect of flooding was less pronounced in early spring (emission < 1.06 mg CH_4-C m^{-2} d^{-1}) probably due to low soil temperature. Although CH_4 flux direction alternated at all sites, overall the croplands and the flood-affected riparian forest were CH_4 sources with annual emission averaging +0.04±0.17 and +0.92±1.6 kg CH_4-C ha^{-1}, respectively. In the riparian forest, a topographic depression (<8 % of the total area) accounted for 78% of the annual CH_4 emission, underscoring the significance of landscape heterogeneity on CH_4 dynamics in riparian buffers. The non-flooded riparian grassland, however, was a net CH_4 sink (-1.08±0.22 kg CH_4-C ha^{-1} y^{-1}), probably due to the presence of subsurface tile drains, and a dredged/incised channel at that study site. While these hydrological alterations may have contributed to improvement in the CH_4 sink strength of the riparian grassland, this must be weighed against the water quality maintenance functions and other ecological services provided by riparian buffers.

1. INTRODUCTION

Riparian buffers have been promoted as a best management practice to protect water quality in agricultural landscapes. Located at the interface between upland and streams, riparian buffers act as natural filters by retaining sediments and nutrients that would otherwise be transported to adjacent surface water bodies. The nutrient removal capacity of riparian soils has been the focus of numerous studies during the last several decades (Hill, 1990; Jacinthe et al., 2003; Vidon and Hill, 2006). Collectively, that research has documented links between biogeochemical functions of riparian zones and their hydrological connectivity with underlying groundwater and adjacent surface water systems. Jacinthe et al. (2012) noted that flood frequency has both short-term and long-term impact on the nitrous oxide-producing capacity of riparian soils, but information remains lacking with regard to the impact of flood regime on the exchange of carbon dioxide (CO_2) and methane (CH_4) between riparian soils and the atmosphere. This information is important given the link between these gases and the accelerated greenhouse effect, and the steady increase in their atmospheric concentration (CO_2: 280 in 1850's to 380 μL L^{-1} in 2005; CH_4: 0.715 to 1.77 μL L^{-1}) during the last 150 years (IPCC 2007).

Riparian ecosystems are often characterized by seasonally-variable water tables that, through periodic contact with the upper soil layers, may affect biological activity and CO_2 production. Further, water table drawdown may also facilitate the transport of metabolizable organic substrates and nutrients to the subsurface, where the combination of high microbial activity and restricted O_2 diffusion can lead to the development of anoxic conditions conducive to methanogenesis (Glatzel et al., 2004; Ballantyne et al., 2014). Thus, water table dynamics can be a key driver of CO_2 and CH_4 production in riparian buffers. Depending on landscape geomorphology, riparian zones can also be variably affected by flooding. Flood events result in

sediment deposition, soil organic carbon (SOC) accumulation and variable redistribution of nutrients and organic debris across riparian landscapes (Blazejewski et al., 2009). In addition to their immediate impact on riparian soil processes, such hydrological events indirectly affect riparian zone biogeochemistry through creation of micro-topographic features such as ridges and low-gradient swales in which wet soil conditions can persist and CH_4 production can accelerate. McNamara et al. (2008) found that, although occupying <10 % of the total land area, gullies accounted for 96 % and 22 %, respectively of the CH_4 and CO_2 fluxes in a UK ombrotrophic peatland. Dinsmore et al. (2009) also reported that a riparian zone, covering 0.5% of a catchment area, accounted for 12% of the total catchment CH_4 emission. Thus, flood history can exacerbate the temporal (hot moments) and spatial (hot spot) variability of trace gas production in riparian buffers.

Locally, mesocosm- and plot-scale experiments have identified key drivers of C cycling in riparian soils. Data from Jacinthe et al. (2003) suggested a link between soil drainage characteristics and CO_2 production potential of riparian soils, with significantly higher CO_2 production observed in poorly-drained than in well-drained soils. Pacific et al. (2011) reported higher CO_2 efflux from riparian areas (downslope) than from forested hillslopes in the Tenderfoot Creek Experimental Forest in Montana (USA). Samaritani et al. (2011) noted that CO_2 emission was more intense and more spatially heterogeneous in riparian buffers along flood-affected sections of the Thur River (Switzerland) compared to riparian areas near channelized sections of the river. Several studies (Dinsmore et al., 2009; Salm et al., 2012; Ballantyne et al., 2014) have reported negative relationships between water table depth (WTD) and CH_4 fluxes, with increased emission generally observed when the water table intersects the topsoil layer (0-30 cm). Groundwater level was the best predictor of CH_4 emission in grassland

on peat soils (Van den Pol-van Dasselaar et al., 1998). Results from Itoh et al. (2008) suggested that CH_4 formation pathways in a forested riparian wetland are determined by temperature and water table position; specifically, a predominance of the CO_2 reduction pathway relative to acetate fermentation under high water table and warm soil conditions. It is important to note that previous investigations linking water table and CH_4 dynamics were conducted almost exclusively in peatlands and permanently-flooded riparian areas which may not necessarily be representative of riparian areas typical of agricultural landscapes of the US Midwest.

Additionally, past studies of CO_2 emission in agricultural watersheds have mostly focused on the effect of land management practice (tillage, fertilizer) and crop type, but few studies have compared CO_2 and CH_4 fluxes between intensively-managed agricultural fields and riparian areas. Irrespective of vegetation cover, Tufekcioglu et al. (2001) reported consistently higher (1.5-fold) rate of CO_2 efflux in riparian buffers than in adjacent crop fields. This trend was strongly linked to the higher soil organic matter content in the riparian areas than adjacent croplands. However, the results of Pacific et al. (2011) suggested that, more than organic matter and related soil properties, CO_2 flux in riparian zones was driven primarily by flood regime. Given differences in vegetation phenology (length of the growing season, amount of biomass produced) and land management (tillage, fertilizer application), marked seasonal differences in CO_2 flux would be expected between cropped fields and adjacent riparian areas. Further, biological consumption of CH_4 is a soil function known to be very sensitive to disturbance. It is widely reported that cropland soils exhibit lower CH_4 oxidation capacity compared to undisturbed soils (grassland and forestland), and this has primarily been ascribed to tillage disturbance and application of NH_4^+-containing fertilizer (Le Mer and Roger, 2001). Since many riparian habitats are not similarly disturbed, it is reasonable to assume higher CH_4 uptake in

riparian buffers than in croplands. This expectation should be tempered, however, in light of the results of Kim et al. (2010), who reported higher rates of CH_4 uptake in cropped fields than in adjacent riparian buffers in Iowa. At the present, it is unclear if these results are specific to the riparian area investigated or representative of other common types of riparian buffers in the US Midwest agricultural region. For example, through integration of floodplain geometry, channel gradient and discharge data (to derive flood duration and height), Panunto (2012) identified five major types of riparian buffers in the White River watershed in Central Indiana. Although past studies (Jacinthe et al., 1998; Vidon and Hill, 2006) have shown that hydrology and geologic settings (till, outwash, alluvial) are strong predictors of the nutrient-filtering capacity of riparian buffers, linkages between these factors and trace gases production in riparian areas have not been elucidated. Thus, with the general hypothesis that hydro-geomorphic settings primarily determine the intensity and variability of gas fluxes in riparian buffers, the present study was undertaken to compare CO_2 and CH_4 fluxes from riparian zones and adjacent agricultural fields at the soil-atmosphere interface, and examine relationships between gas fluxes, soil properties, water table, and landscape attributes (outwash plain vs. till plain sites).

2. MATERIALS AND METHODS

2.1 Description of study sites

This study was conducted between December 2009 and May 2011 at two locations in distinct geomorphic regions (till plains versus glacial outwash) of the White River watershed, central Indiana (USA). These sites are representative of the two most common types of riparian buffers in the glaciated landscapes of the US Midwest. Each location included an actively-managed agricultural field and a riparian buffer located down-slope from the field. The

agricultural fields were planted to corn (*Zea mays*, L.), and received on average 121.5 kg N ha^{-1} y^{-1} as urea ammonium-nitrate.

The two study sites (Fig. 1) present contrasting hydro-geomorphic features. The first site (hereafter referred to as White River, WR) is a riparian forest located 40 km south of Indianapolis (39^0 29' 39" N, 86^0 25' 2" W; Morgan County) along a 4^{th} order segment of the White River. The site comprises a cultivated field to the north and a broad riparian area (~150 m wide) along the river channel to the south. Located along the inner bank of a meander, the site is prone to flooding (bankfull discharge: ~ 380 m^3 s^{-1}) with flood events typically occurring after snowmelt and major rainstorms during the spring and early summer. Most common tree species at this riparian forest include silver maple (*Acer saccharinum*), American beech (*Fagus sylvatica* L.), American sycamore (*Platanus occidentalis*), white oak (*Quercus alba*), bur oak (*Quercus macrocarpa*) and red ash (*Fraxinus pennsylvanica*). Soils are well-drained Genesee silt loam (mesic fluventic Eutrudepts) and Stonelick sandy loam (mesic Typic Udifluvents) overlying fluvio-glacial outwash and alluvium deposits. Field observations and regional surficial geology maps indicate that this site is underlain by about 2 m of silt loam atop a 50 cm layer of compacted gravel.

The second site (39^0 51' 20" N, 85^0 50' 24" W; Hancock County, 30 km east of Indianapolis) is a grass-dominated riparian buffer along the Leary Weber Ditch (hereafter referred to LWD) in the Tipton Till Plain physiographic region. Vegetation is dominated by barnyard grass (*Echinochloa crusgalli*), smooth bromegrass (*Bromus inermis*), virginia wild rye (*Elymus virginicus* L), and foxtail (*Setaria* spp.). The riparian area is uniformly flat (slope < 2%), and extends approximately 25 m on each side of the agricultural ditch flowing west to east (Fig. 1). Over the years, the ditch has periodically been dredged and deepened to prevent flooding of

adjacent crop fields. As a result of this channelization, flood-induced sediment deposition is not active in this riparian zone. The site is also characterized by poorly-drained Brookston (fine-loamy mesic typic Argiaquolls) soils that develop in dense glacial till. Given the landscape topography and natural soil drainage characteristics, agricultural fields are equipped with a network of subsurface tile drains (on average 1.5 m deep) that flow underneath the crop field and riparian area, and discharge into the Leary Weber Ditch.

The regional climate is humid continental with a long-term mean annual precipitation of 1040 mm. For the period of study, air temperature and rainfall data were obtained from the Indiana State Climate Office (www.iclimate.org) for weather stations in Martinsville (id: 125407) and Greenfield (id: 123527), both located about 10 km south of the WR and LWD study sites, respectively. In 2010, mean air temperature was 10.5 ^{o}C in Martinsville and 11.5 ^{o}C in Greenfield. Total rainfall (992 mm) across the region was near normal in 2010, but seasonal distribution was uneven with a wet spring (360 mm in May and June) and dry autumn (91 mm between August and November). White River discharge data were downloaded from a gauging station (USGS 335400; N 39° 29', W 86° 24') about 1 km north of the WR site.

2.2 Study sites instrumentation and trace gases monitoring

At the WR site, CO_2 and CH_4 fluxes and soil properties were measured along three transects, with two transects in the riparian zone and one transect in the adjacent crop field. In the riparian area, transects were delineated so as to include high (ridge) and low (swale) topography, and extended from the field edge to the river channel edge (Fig. 1). Along each transect, five static chambers were installed. The chambers installed in the crop field were removed during harvest and fertilizer application. Given the flat and uniform topography at

LWD, chambers were installed along a predetermined grid from field to ditch edge (Fig. 1). Chambers installed in the crop fields were removed as needed to accommodate farming operations. The adjacent crop fields at LWD were under corn-soybean (*Glycine max*, L.) rotation but, in any given year, there was a corn field on either the north or south side of the ditch. To maintain consistency with regard to crop type, only the field planted to corn was monitored; therefore, chambers (#15 to 18; Fig. 1) were relocated each year to the corn field on either side of the ditch. Soil temperature (20 cm) and moisture (20 cm and 60 cm depth) were monitored with TMB-M006 and SMA-M005 sensors, respectively (Onset Corp., MA) deployed in the riparian area at each site. Observation wells and piezometers were installed in each riparian area to monitor water table depth and groundwater chemistry. Water table depth (expressed in cm below ground surface at the well location, bgs) was continuously recorded every 15-20 min using a level logger (Solinst, Inc.) in observation well 1 at LWD and in well 4 at WR (Fig. 1). Water level in the piezometers was measured manually approximately once a month during the study. Details regarding groundwater flow, sources and fate of chemical elements in groundwater can be found in Vidon et al. (2013) and Liu et al. (2014).

Gas flux was monitored by the static chamber technique from December 2009 to May 2011 on a monthly to bimonthly basis depending on weather events and accessibility of the study sites (e.g. high flood water). Chambers consisted of PVC cylinders (30 cm diam., 12-15 cm above ground) securely inserted 8-10 cm into the ground. The bottom edge of the chamber was beveled to facilitate ground insertion. During sampling, chambers were covered with PVC lids secured to the base with bungee cords. The lid was fitted with a gasket at its underside edge to make an air-tight seal, and butyl rubber septa at its center to form a sampling port. Chamber headspace was sampled 0, 20, 40 and 60 min after closing with the lid, and air samples (~20 ml)

were stored in 10 ml evacuated glass vials fitted with butyl rubber septa. Soil air samples were generally collected at mid-day (between 11 AM and 2 PM local time). Air samples were analyzed using a Varian CP-3800 gas chromatograph (Palo Alto, CA) interfaced with a Combipal headspace auto-sampler (CTC Analytics, Zurich, Switzerland), and equipped with a thermal conductivity detector (at 300 ˚C, for CO_2 detection) in series with a flame ionization detector (at 300 ˚C, for CH_4 detection). The stationary phase consisted of a pre-column (L: 0.3 m; id: 2 mm) and an analytical column (L: 1.8 m; id: 2 mm) filled with Porapak Q (80-100 mesh). The carrier gas was UHP He (60 ml min^{-1}), and oven temperature was 90 ^oC. The gas chromatograph (detection limit: 20 µL CO_2 L^{-1} and 0.12 µL CH_4 L^{-1}) was calibrated with CO_2 and CH_4 standards obtained from Alltech (Deerfield, IL). Daily flux of CO_2 or CH_4 was computed as:

$$F = \frac{dC}{dt}\frac{V}{A}\ k$$

dC/dt: concentration change in chamber headspace (mg CO_2/CH_4-C m^{-3} min^{-1}) determined by linear regression; V: chamber volume (m^3); A: area of soil circumscribed by chamber (m^2); k: time conversion factor (1440 min d^{-1}). Positive values for gas flux indicate net emission while negative values correspond to net soil uptake. Annual flux of CO_2 and CH_4 for the year 2010 was computed for each sampling point by plotting daily fluxes against time, interpolating linearly between sampling occasions, and integrating the area under the curve using SigmaPlot 11.0 (Systat, San Jose, CA) software (Dobbie and Smith, 2003; Fisher et al., 2014). We acknowledge that this method of aggregation might introduce some scaling error in the annual estimates of gas fluxes at sites exhibiting high spatial variability.

2.3 Soil samples collection and analysis

Soil samples (0-20 cm) were collected on November 17-19, 2010 at the study sites. Samples were extracted next to each static chamber for determination of general soil properties including soil bulk density, texture (hydrometer method), pH (1:2 soil-to-water suspension with an Accumet pH/ion meter), mineral N (NO_3^- and NH_4^+ extracted with 2 M KCl and analyzed with an EST Aquakem analyzer, Fairfield, OH), net nitrogen mineralization (net amount of NH_4^+ + NO_3^- produced during aerobic incubation of field-moist soil for 15 days), total C and N (dry combustion at 960 ^oC using a Vario-Cube analyzer, Elementar Americas, NJ), water extractable organic carbon, and microbial biomass carbon (MBC; Anderson and Domsch, 1978). Gravimetric soil moisture constant was determined (drying at 105 $^\circ C$ for > 48 h), and all results were reported on a dry soil mass basis.

2.4 Data analysis

The distribution of the gas flux data was first examined for normality using the Univariate procedure in SAS and the Shapiro-Wilk test to determine whether transformation was needed prior to analysis of variance (ANOVA). This analysis was carried out to assess the effect of site, land-use and sampling date on CO_2 and CH_4 fluxes. In this analysis, the class variables were site geomorphology (glacial outwash at WR and till plain at LWD) and land-use (riparian buffer and crop field) while sampling date was used as the repeated measure factor. Analysis was also conducted for each site separately to compare gas fluxes between land-uses. Regression analysis was employed to explore relationships between environmental variables (soil temperature, moisture, and water table depth), soil properties and gas fluxes. All data were analyzed using procedures available in SAS (SAS, 2002). For all tests, statistical significance was determined at the 95% confidence level.

3. RESULTS

3.1 Study sites hydrology and rainfall

During the study period, no evidence of flooding was detected at the LWD riparian area although it is possible that short-duration (< 2 days) ponding may have occurred after snowmelt and rainstorms. However, the WR riparian site was flooded 3-4 times during the study period with the most extensive flood events occurring in May/June 2010 and spring 2011 (Fig. 2b). A strong relationship ($y = 0.006\ Q\ -1.01$, r^2: 0.94, $P < 0.001$) was found between river discharge (Q; $m^3\ s^{-1}$) and water table depth (y; m bgs) measured in the observation well at the WR riparian forest (well# 4; Fig. 1). At both study sites, the water table responded to rainfall events (Figs. 2-3), but temporal variation was much more pronounced at WR (mean depth at well 4: 60 cm) than at LWD (mean depth at well 1: 128 cm). The water table never came closer than 13 cm bgs (below ground surface) at LWD, but at WR floodwater as high as 3.2 m above ground surface at the well location was recorded. At both sites, the water table reached its lowest depth (98 cm bgs at WR; 164 cm bgs at LWD) between August and November, 2010, a period characterized by low precipitation (53 mm at WR and 129 mm at LWD as compared to normal of 238 mm) and low surface soil moisture (mean ± SE; 0.07 ± 0.01 $m^3\ m^{-3}$ at WR; 0.20 ± 0.01 $m^3\ m^{-3}$ at LWD). During the dormant season (November - March), soil temperature was warmer at the WR riparian forest (7.1 ^oC) than at LWD (5.2 ^oC), whereas the opposite was observed during the growing season (April - October) with warmer soil temperature at LWD (19.3 ^oC) than at WR (16.5 ^oC).

3.2 Variability of CO_2 emission

During the study period, CO_2 fluxes ranged from 0.02 to 11.67 g CO_2-C m^{-2} d^{-1}, and temporally followed soil temperature (Figs. 2 and 3). Irrespective of location and land-use, soil respiration increased between April and August, reached its peak in late June/early July, and then decreased as soil temperature declined in late autumn and winter (Figs. 2d, 3d). At each study site, significant difference in CO_2 flux was observed with respect to land-use ($P < 0.001$) and sampling date ($P < 0.001$) but not their interactions. Although the effect of land-use was not significant on several sampling dates, overall significantly greater rate of CO_2 emission was measured in the riparian buffer than in the crop field, regardless of study site (Table 2). When the data for all types of land-use was combined, CO_2 emission was significantly ($P < 0.001$) higher at LWD than at WR. Annual (April 2010 - April 2011) CO_2 emission averaged (mean ± SE) 4.95±0.68 and 2.56±0.19 Mg CO_2-C ha^{-1} at the WR riparian buffer and crop field, respectively (Table 2). At LWD, annual emission was 8.63±0.58 and 4.54±0.21 Mg CO_2-C ha^{-1} for these respective types of land-use (Table 2). Thus, irrespective of land-use, soil respiration was 1.75 times higher at LWD than at WR.

3.3 Variability of CH_4 fluxes

During the study period, instances of CH_4 emission and consumption were detected at all sites, but net emission cases were more frequent (45 % of the time) in the crop fields than in the riparian areas (18 % of the time). Methane flux in the WR riparian area exhibited a high degree of spatial variability (range: -2.54 to +44.53 mg CH_4-C m^{-2} d^{-1}), and most notably so after major hydro-meteorological events (Figs. 2-3). The highest rate of CH_4 uptake (-2.54 mg CH_4-C m^{-2} d^{-1}) was measured on September 12, 2010 at sampling points located on the ridges near the river

margin (Fig. 4). At LWD, the highest rate of CH_4 consumption (-1.78 mg CH_4-C m^{-2} d^{-1}) was also recorded during that same period (Aug. 6, 2010). Conversely, the highest rate of CH_4 emission (+44.53 mg CH_4-C m^{-2} d^{-1}; Fig. 4) was measured on July 1, 2010 at the WR riparian forest in a depression following almost 10 days of flooding (June 16-26; Fig. 2b-c). Soil moisture and temperature in the depression averaged 0.46 m^3 m^{-3} and 22.9 ºC, respectively.

Overall, ANOVA showed a significant effect ($P < 0.001$) of land-use with respect to CH_4 fluxes at both study sites. The LWD riparian area was consistently a CH_4 sink, the WR riparian forest alternated between source and sink, and the crop fields were generally net CH_4 emitters (Fig. 2e, Fig 3e). Annual CH_4 emission averaged +0.05 and +0.04 kg CH_4-C ha^{-1} at the WR and LWD crop fields, respectively. The WR riparian forest was a net source, emitting +0.92 ± 1.6 kg CH_4-C ha^{-1} annually. However, the riparian buffer at LWD was a net sink with an annual rate of CH_4 uptake averaging -1.08 ± 0.22 kg CH_4-C ha^{-1} (Table 2).

Inspection of the soil temperature data showed that spring-thaw was underway when gas flux measurements were made on March 4-5, 2010 and on February 16-18, 2011 (Fig. 2c, 3c). At the WR location (Fig. 2e), the riparian buffer consistently shifted from a small CH_4 sink (-0.16 mg CH_4-C m^{-2} d^{-1}) to a small source (+0.02 mg CH_4-C m^{-2} d^{-1}) at spring thaw. At LWD, a similar shift was observed, from a moderate (-0.41 mg CH_4-C m^{-2} d^{-1}) to a weak sink (-0.09 mg CH_4-C m^{-2} d^{-1}) (Fig. 3e).

3.4 Relationships between soil properties and gas fluxes

Consistent with the temporal trend in CO_2 emission (Fig. 2d, Fig. 3d), linear relationships (r^2: 0.65, $P < 0.001$ at WR; r^2: 0.51, $P < 0.002$ at LWD) were found between mean daily flux of CO_2 (y) and soil temperature (x) in the riparian areas (Table 3). Relationships between these

variables were also described using exponential functions ($y = 0.58 \exp^{0.056x}$, r^2: 0.31, $P < 0.05$ at WR; $y = 0.56 \exp^{0.093x}$, r^2: 0.53, $P < 0.001$ at LWD). To express the temperature sensitivity of soil respiration, Q_{10} (proportional increase in CO_2 flux per 10 ^oC increase in temperature) was derived ($y = a \exp^{\beta x}$ and $Q_{10} = \exp^{10\beta}$). For the WR and LWD riparian areas, Q_{10} values averaged 1.75±0.28 and 2.53±0.83, respectively. No relationship was found between soil moisture and water table depth with daily rate of CO_2 emission at any of the sites.

Of all the environmental, landscape, and soil factors considered in the study, water table depth (measured in the observation wells) was the only variable that significantly ($P < 0.04$) correlated with CH_4 flux in the riparian area at both WR and LWD (Table 3; Fig. 5). When daily flux of CH_4 for each chamber was plotted against WTD measured in a nearby piezometer, similar trends were observed, but the strength of the relationship varied with sampling date, being generally higher in the summer than in the winter (Fig. 5c-f).

Regression analysis was conducted using the soil properties measured on November 17-19, 2010 to examine relationships with gas flux measurements made on these same dates (Table 3). None of the soil properties correlated with CO_2 flux at LWD, but at the WR site CO_2 flux was significant ($P < 0.04$) related to MBC, C/N ratio of soil organic matter and NH_4^+ concentration (Table 3). At both sites, significant relationships were found between CH_4 flux and soil bulk density (ρ_b). At WR, additional relationships were found with sand and clay content, soil moisture, NO_3^- and N mineralization (Table 3).

4. DISCUSSION

4.1 Soil respiration in relation to land cover in riparian buffers

One of the primary objectives of this study was to compare soil respiration between

adjacent crop fields and riparian buffers, and to assess the importance of biotic and abiotic drivers. In the context of this study, as was found in many studies conducted in temperate humid regions (Wagai et al., 1998; Jacinthe and Lal, 2004), CO_2 emission was not limited by soil moisture. Difference in CO_2 production among the study sites was primarily a reflection of soil temperature and land-use, with the latter determining factor the amount and distribution of plant residue returned to soils.

Significantly higher rates of CO_2 production were consistently measured in the riparian buffers than in the crop fields (Table 2). Similar results were reported in past studies comparing soil respiration between cropland and perennial systems (Wagai et al., 1998; Tufekcioglu et al., 2001), and can be linked to difference in biomass input between systems. While annual CO_2 emission from the crop fields were in the range expected for corn fields in the region (Tufekcioglu et al., 2001; Lehman and Osborne, 2013), emission was significantly greater (1.8 times) at the LWD than at the WR corn field (Table 2). Given the average corn yield (9.5 Mg grain ha^{-1} y^{-1}; Fisher et al., 2014) during the study period at the crop fields, using a harvest index of 0.46 and computational approach described in Johnson et al. (2006), an estimated 9.94 Mg dm ha^{-1} of crop residue (1.52 Mg as root tissues, assuming a root/shoot ratio of 0.18) is returned to the crop fields annually. Assuming an average C content of 420 g C kg^{-1} for corn (Johnson et al., 2006), this translates into an annual input of 4.17 Mg C ha^{-1} as crop residue to the agricultural fields. This amount is nearly equal to the annual C emission as CO_2 from the LWD crop field (4.54 ± 0.21 Mg CO_2-C ha^{-1}, Table 2). In the WR crop field, however, annual CO_2 emission (2.56 ± 0.19 Mg CO_2-C ha^{-1}, Table 2) only accounts for 62 % of C input as crop residue. These results suggest the existence of additional pathways of C loss in this crop field. Given the permeability of the sandy subsurface in that alluvial floodplain, C loss via leaching could be

significant. Further, since the WR crop field is located within a flood-prone meander of the White River, it is conceivable that overland flow could remove large amounts of crop residue when floodwater runs through that crop field. Over time, these flood-induced residue removals would contribute to the lower SOC content and CO_2 emission measured at the WR crop field during our study (Tables 1 and 2).

The studied riparian areas are relatively unmanaged natural systems supporting woody (WR) and grass (LWD) vegetation. Although biomass production in the riparian areas was not measured, available data suggest that biomass input must be much greater in these systems than in the croplands (Jackson et al., 1996; Ussiri et al., 2006; Bills et al., 2010). In addition to the total mass, the spatial distribution of that input (shoot versus root) is equally important, considering the strong relationships reported in several studies between root biomass production, and soil respiration, C cycling and storage (Tufekcioglu et al., 2001; Ussiri et al., 2006). In a compilation of root biomass studies, Jackson et al. (1996) noted that 83 % of root biomass resides in the top 20 cm soil layer in temperate grassland as opposed to 65 % in forest. Data from US Midwest locations showed even a greater stratification with 84-90 % of root biomass in the top 20 cm soil layer in grasslands as compared to 37 % in forests (Ussiri et al., 2006; Bills et al., 2010). In light of these considerations and with the knowledge that < 50 % of CO_2 emission could originate from root respiration (Hanson et al., 2000), it is not surprising that soil respiration was highest in the grass-dominated LWD buffer than in any of our other study sites (Table 2).

When either the crop fields or the riparian areas were compared, CO_2 emission was consistently higher at LWD than at WR. This trend is consistent with the significant site effect ($P < 0.001$) shown by ANOVA and also suggests that, besides land-use and plant biomass, there could be other site-specific drivers of soil respiration that need to be considered. Since most of

the soil C quality parameters were not markedly different between the sites (Table 1) and regression analysis failed to demonstrate strong links (Table 3), it is difficult to ascribe difference in CO_2 efflux directly to soil characteristics. However, it is likely that the greater CO_2 emission at LWD was the result of higher soil temperature (19.3 ^oC at LWD during the 2010 growing season as opposed to16.5 ^oC at WR; Figs. 2c and 3c) and greater temperature sensitivity of soil respiration at LWD as suggested by the Q_{10} values (2.53 at LWD vs 1.75 at WR). The LWD riparian area is an open field site (absence of tall trees) and receives an abundant amount of solar radiation. These factors contribute to warmer soil temperature and thus greater rate of microbial decomposition of organic matter (including root exudates released by the grass/shrub vegetation community). Conversely, less intense soil respiration at the WR riparian forest likely reflects the moderating effect of tree canopy on soil temperature during the growing season.

4.2 Flooding history and soil-atmosphere CH_4 exchange in riparian buffers

Among terrestrial ecosystems, grasslands and forests generally act as strong CH_4 sinks, whereas croplands act as CH_4 sources or weak sinks depending on management (Castro et al., 1994; Chan and Parkin, 2001; Le Mer and Roger, 2001). The limited CH_4 consumption in croplands has been ascribed to physical disturbance of soils by plowing, and to application of N fertilizers through generation of NH_4^+ ions that are inhibitory to methanotrophs (Le Mer and Roger, 2001). Consistent with that general understanding, the relationships between CH_4 fluxes and soil bulk density (Table 3) suggest that gas transport processes and related soil physical conditions control CH_4 uptake at the study sites. In contrast to the results of Kim et al. (2010) who reported net consumption in crop fields, the present study found that the crop fields were

small CH_4 sources. The flood-affected riparian forest was also found to be a moderate CH_4 source (Table 2). Flood events and seasonal fluctuations in water table position may have contributed to the difference in soil-atmosphere CH_4 exchange observed at the two riparian sites. Elevated soil moisture affects CH_4 dynamics through stimulation of CH_4 formation, and restriction on gas transport (CH_4, O_2), which ultimately limits methanotrophic activity.

In contrast to the flood-affected riparian forest, the grass-dominated LWD riparian area was a CH_4 sink with annual consumption (-1.08 ± 0.22 kg CH_4-C ha^{-1} y^{-1}) comparable to upland forests of the region (Chan and Parkin, 2001; Jacinthe and Lal, 2004). This moderate sink is probably the product of several factors including absence of mechanical disturbance, limited interaction with agrochemicals-laden runoff from adjacent crop fields, facilitated soil drainage due to subsurface tile drains, and an incised channel that protects the grass buffer from flooding. While these hydrological alterations of riparian areas, highly common in US Midwest agricultural landscapes, have been blamed for inefficient nutrient retention in riparian buffers and water quality problems, they apparently contribute to the strength of riparian areas as CH_4 sinks. Additional studies at other sites in similar hydrogeo-morphologic settings are needed to confirm this observation.

4.3 Hot spots and hot moments of CH_4 fluxes

The flux of greenhouse gases (GHG) in terrestrial system is notoriously variable both in space and time. Results of the present study are no exception. While generally limited at LWD and in the crop fields, GHG fluxes at the WR riparian forest often exhibited a high degree of spatial variability (Fig. 3). Variability tended to be highest during period of intense soil

respiration (Figs. 2d, 2e, 3d, 3e). The most extreme case of spatial variation was recorded after the late June 2010 flood at the WR site. Methane emission (44.5 mg CH_4-C m^{-2} d^{-1}, Fig. 4) measured in a depression (< 8 % of total study area) was so intense that inclusion of that single measurement (out of ~ 170 flux measurements made at the riparian forest during the study) changed the average CH_4 flux at the site from a sink (-0.163 mg CH_4-C m^{-2} d^{-1}) to a source (+0.103 mg CH_4-C m^{-2} d^{-1}). This powerfully illustrates the effect of landscape heterogeneity on CH_4 flux in riparian areas, and corroborates several past studies that have documented the disproportionate contribution of small hotspots to CO_2 and CH_4 emission from peatlands (Dinsmore et al., 2009; McNamara et al., 2008). Enhanced biological activity in these depressions could conceivably lead to biased estimates of annual gas fluxes in the flood-affected riparian forest. However, the uncertainty arising from landscape heterogeneity can be minimized through appropriate sampling design and distribution of chambers in proportion to the area represented by each landform type within a riparian buffer (Levy et al., 2011).

Besides the June 2010 flood, the WR riparian forest was flooded on other occasions (March 15-16, 2010, March 1-9 and April 20-23, 2011), but these events did not result in elevated CH_4 emission, even in the depression (max: 1.06 mg CH_4-C m^{-2} d^{-1}). Methane formation in saturated soil involves several phases, including carbon mineralization, depletion of higher energy-yielding electron acceptors, and acetate formation. All these phases are temperature-sensitive, and steady production of CH_4 by methanogens generally occurs in the 20-30 ^{o}C temperature range (van Hulzen et al., 1999). Therefore, methanogenesis may have been limited by low soil temperature (range: 6.8-11.9 ^{o}C) during the early spring floods, whereas temperature (21.4 - 23.9 ^{o}C) was within the optimum range during the summer 2010 flood.

Therefore, soil temperature must be considered when assessing the effect of flooding on CH_4 emission in riparian zones.

The temporal variation in CH_4 flux was marked by two additional weather-related factors. The first trend is the constant shift in the direction of CH_4 flux (from sink to source) observed at spring-thaw; this shift coincided with snowmelt and occurred in early spring when both water table and soil moisture began to rise (between 30-70 cm bgs). Similar shifts during spring thaw have been reported previously in in forestland (Jacinthe and Lal, 2004) and cropland (Ussiri et al., 2009) of the Midwest. This trend has been associated to spring-thaw release of gases accumulated below frozen soil layers during the winter (Burton and Beauchamp, 1994). The second trend corresponds to the low precipitation period in late summer/autumn 2010 (Fig. 2a, 3a). While the direction of CH_4 flux was poorly defined in the agricultural fields (near zero emission at LWD and small source at WR) during that stretch of dry weather, CH_4 uptake was strongest in the forested (WR) and grass-dominated riparian areas (LWD). Liu et al. (2014) reported groundwater oxidation-reduction potential (ORP) averaging -11 mv at WR and -52 mv at LWD. Methane formation likely occurred in the subsurface but did not translate into increased CH_4 emission. As suggested by others (Lessard et al., 1997; Jacinthe and Lal, 2004), it is likely that CH_4 produced in the subsurface was oxidized by methanotrophs during upward migration. Methanotrophs are sensitive to soil moisture stress and exhibit optimum CH_4 consumption at about 20 % soil water content (Castro et al., 1994). Although surface (20 cm) soil was dry, moisture condition at 60 cm was near optimum for CH_4 oxidation at the riparian sites during that period.

4.4 Effect of water table position on CH_4 fluxes

Numerous studies (Van den Pol-van Dasselaar et al., 1998; Itoh et al., 2008; Dinsmore et al., 2009; Salm et al., 2012; Ballantyne et al., 2014) have documented linkages between WTD and CH_4 flux in natural ecosystems, but almost all of these investigations were conducted in frequently-saturated systems such as wetlands and peatlands. Results of the present study showed, perhaps for the first time, that WTD is also an important controlling factor of CH_4 dynamics in non-wetland riparian ecotones with variable flooding history. Water table depth was the only factor that significantly related to CH_4 flux at both our study sites. We also found a strong relationship (r^2: 0.93) between river discharge and WTD at the WR site (outwash); if similar relationships were to be obtained at other sites with similar geomorphic characteristics, river gauge data could be used as a proxy for WTD in regional studies of CH_4 budget in well drained riparian zones where stream stage often correlated to WTD (e.g. Jung et al., 2004). However, since WTD explained less than 35% of the variability in the data, other factors likely contributed to the dynamics of CH_4 at the study sites. Examination of the study results and regression analysis suggests that soil temperature, site micro-topography and physical soil characteristics (bulk density, texture) are additional factors that could improve the predictive power of CH_4 flux models. In this study, soil properties were measured only in the surface (0-20 cm); this sampling procedure was likely inadequate to capture variations in physical properties throughout the soil profile. This is particularly relevant to topographically-diverse riparian sites, and with regard to CH_4 – a gas that is produced, transported and consumed at varying rates in different soil layers.

5. CONCLUSIONS

The protection and installation of riparian buffers have been promoted as an approach to mitigate the export of nutrients and other agrochemicals from croplands to surface water bodies, but the contribution of these buffers to regional greenhouse gases budget is not well established. In this study, we monitored CO_2 and CH_4 fluxes, and assessed soil properties in crop fields and adjacent riparian buffers in two hydrogeomorphic settings (flood-affected and flood-protected) in the White River watershed. Significantly higher SOC, MBC and soil respiration were measured in riparian buffers than in crop fields, underscoring the high level of biological activity in riparian soils and indirectly their pollutant-removal capacity. Soil temperature was the main driver of soil respiration, with plant biomass being a possible secondary factor. Water table depth was a good predictor (r^2: 0.24-0.35, $P < 0.05$) of the average CH_4 flux measured at the riparian sites during the study. Concurrent measurements of soil properties and gas fluxes also showed strong relationships (r^2: 0.42-0.61, $P < 0.01$) between CH_4 fluxes and bulk density. Flood events had variable impact on CH_4 flux: resulted in enhanced emission after summer floods (soil temperature > 22 ^oC) but not after spring floods probably due to low soil (< 11 ^oC) temperature. Although CH_4 flux direction alternated at all sites, results showed that the crop fields and the flood-affected riparian forest were CH_4 sources. A small depression (< 8 % of the area) contributed to the bulk (78 %) of the CH_4 emitted from the riparian forest. The non-flooded riparian buffer acted as CH_4 sink, with an annual consumption of -1.08 kg CH_4-C ha^{-1} y^{-1}. Hydrological alterations (subsurface tile, channel incision through dredging) may have improved the CH_4 sink strength of this riparian buffer through flood protection and maintenance of a stable water table. However, the benefits of that improvement must be weighed against its impact on water quality and other ecological services provided by riparian buffers.

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Figure captions

Fig.1**.** Schematic layouts of the White river (WR) and Leary Weber Ditch (LWD) study sites in Central Indiana. Numbered circles (filled) represent the location of the static chambers deployed at each site. The small triangle denotes the location of the soil moisture and temperature sensors, and the observation well for continuous measurement of water table depth. Mini-piezometers were also installed near the chambers for periodic measurements of water table. The closed polygons at the WR site represent topographic depressions. Arrow indicates the general water flow direction in the adjacent channel.

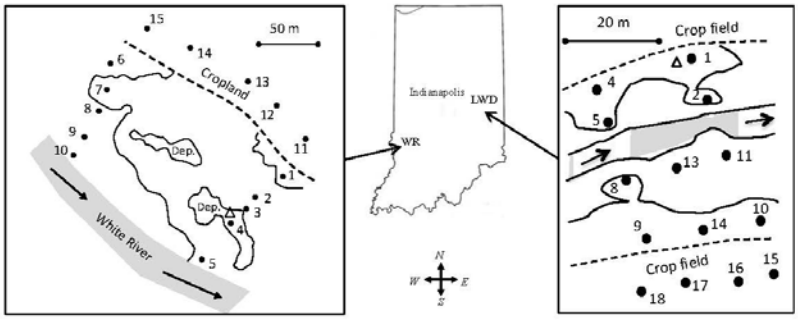

Fig. 2. Rainfall (A), river discharge and water table depth (B), soil moisture and temperature (C), CO_2 flux (D) and CH_4 flux (E) at the White river (WR) riparian forest and adjacent corn field. Discharge data for the White River were obtained from USGS station 03354000 located 1 km north of the WR site. The gap in soil moisture and temperature data is due to soil probe and logger malfunction due to stagnant flood water. The filled triangles and open circles respectively represent discrete measurements of soil moisture and temperature made during that period. Gas flux data are reported as the average of 10 and 5 observations in the riparian zone and crop field, respectively. Error bar represents standard deviation of the mean. Significant difference between land-use is indicated by placing different letters next to the average fluxes on a given sampling date.

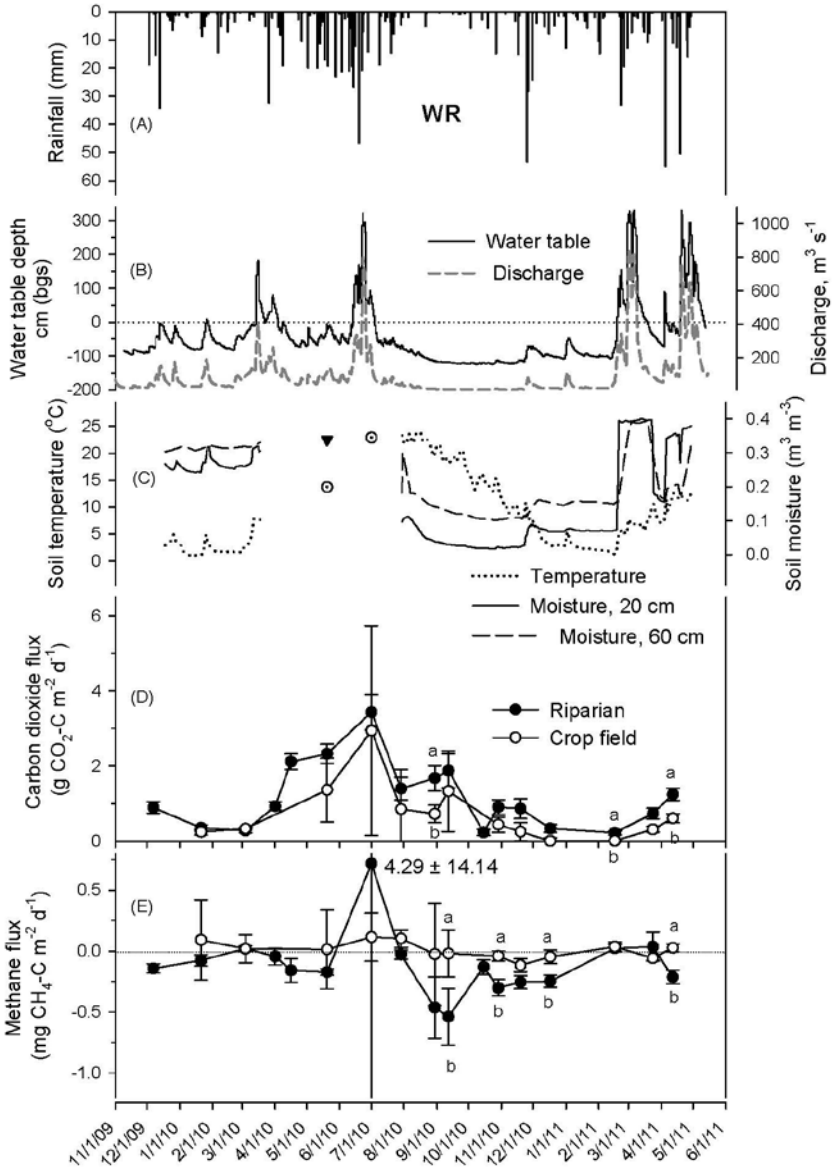

Fig. 3. Rainfall (A), water table depth (B), soil moisture and temperature (C), CO_2 flux (D) and CH_4 flux (E) at the Leary Weber Ditch (LWD) riparian buffer and adjacent corn field. The gap in soil moisture and temperature data is due to rodent damage to soil probe and loggers. Gas flux data are reported as the average of 10 and 4 observations in the riparian zone and crop field, respectively. Error bar represents standard deviation of the mean. Significant difference between land-use is indicated by placing different letters next to the average fluxes on a given sampling date.

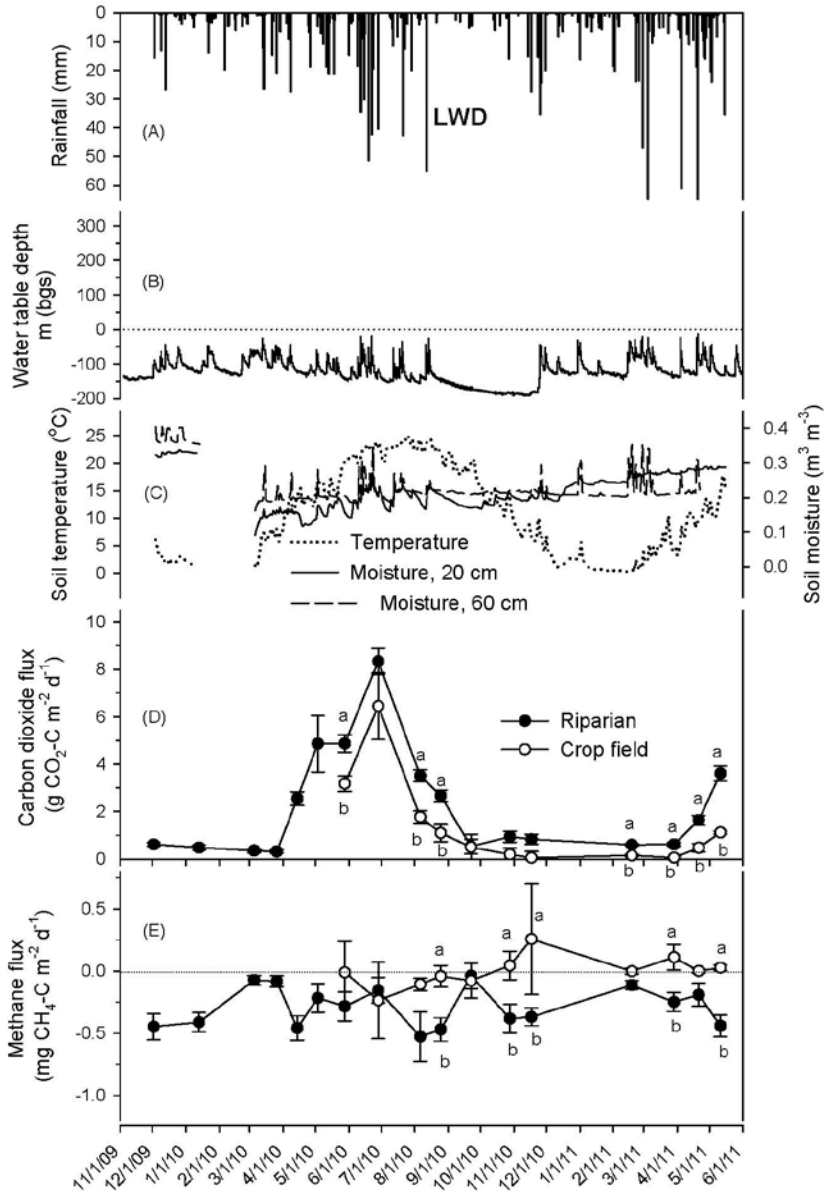

Fig. 4. Relative ground surface elevation and CH_4 flux across the White River (WR) riparian buffer after the flood of June/July 2010 (*left*), and during a dry period in autumn 2010 (*right*). Filled circles indicate the location of the static chambers. Chamber number is listed next to each circle. Vertical exaggeration is 0.0435. Note the difference in scale among the graphs.

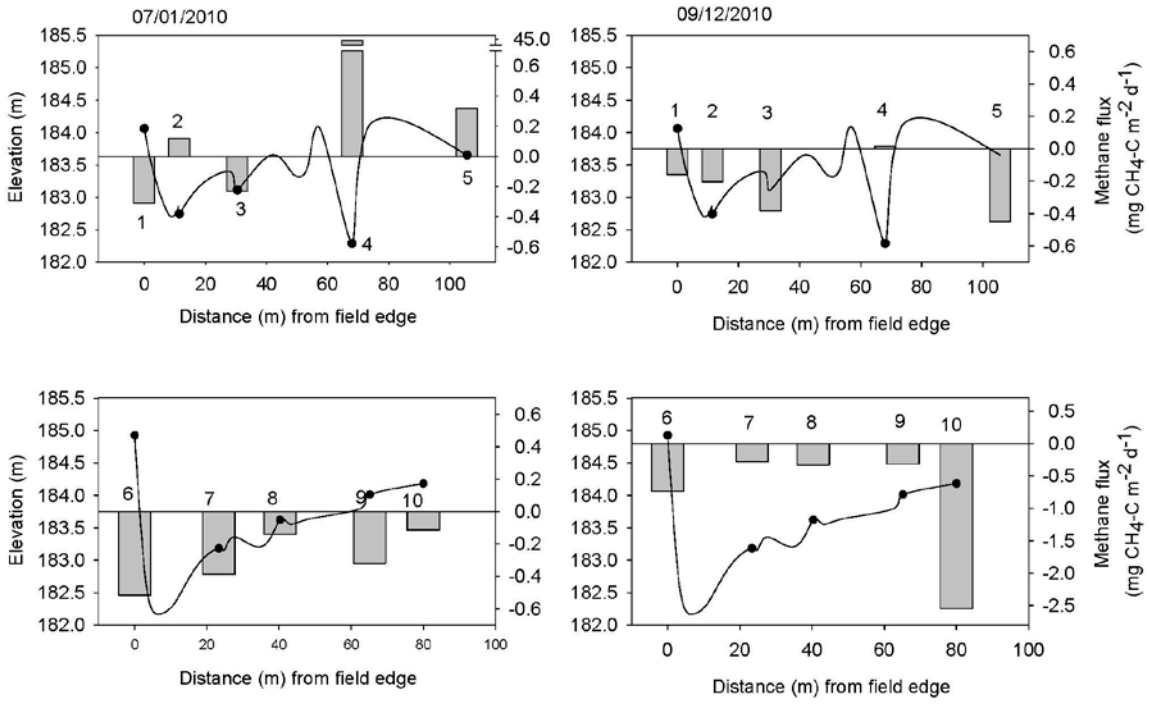

Fig. 5. Relationships between water table depth (WTD) and methane flux at the White River (WR, *left*) and Leary Weber Ditch (LWD, *right*) riparian buffers. The data shown in graphs a-b depict daily flux of CH_4 and daily average WTD measured at an observaton well with a level logger. Relationships between CH_4 flux and WTD measured in a piezometer next to each static chamber are reported in graphs c-d for the summer/fall season, and in graphs e-f for the winter season of 2010. The difference in the number of sampling points shown in Figs. a-f are due to WTD in the piezometer being deeper than 250 cm, the maximum depth of the piezometer (that occurred 45 and 8 % of the time at WR and LWD, respectively). Water table depth is reported in unit of cm below ground surface (cm bgs). A negative value for WTD (graphs a and e) indicates standing water above ground surface. Note the difference in scale among the graphs.

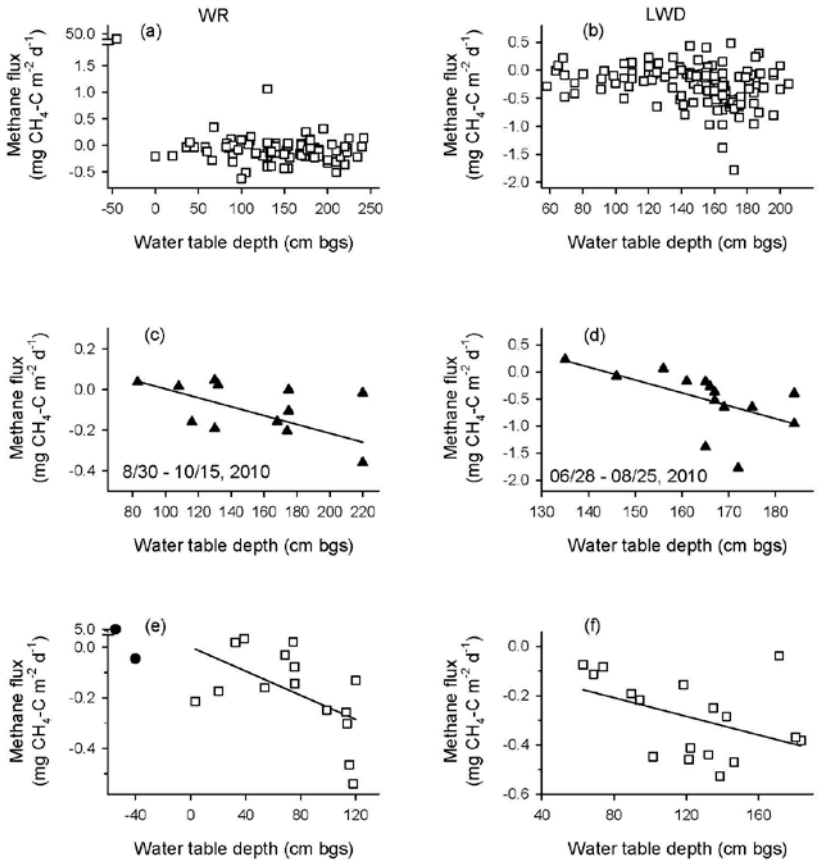

Table 1. Properties of soils (0 - 20 cm) at the study sites in November 2010. Values are mean ± standard error; n=10 for the riparian areas, and 4-5 for the crop fields.

Soil property	WR[†]		LWD	
	Riparian zone	Crop field	Riparian zone	Crop field
pH	7.3 ± 0.03	7.4 ± 0.03	6.9 ± 0.16	6.6 ± 0.08
Bulk density, g cm^{-3}	1.17 ± 0.04b[‡]	1.43 ± 0.04a	1.1 ± 0.05b	1.34 ± 0.02a
Sand, %	39.1 ± 2.6	47.2 ± 2.7	41.9 ± 1.1	42.1 ± 0.8
Clay, %	32.6 ± 1.9	29.5 ± 1.9	34.1 ± 1.2	32.1 ± 1.6
Soil moisture, g water g^{-1} soil	0.16 ± 0.02	0.12 ± 0.01	0.19 ± 0.01a	0.14 ± 0.01b
Soil organic carbon, g C kg^{-1}soil	36.6 ± 0.77a	28.7 ± 1.01b	41.2 ± 1.73a	17.6 ± 1.86b
C:N ratio	16.42 ± 0.41a	14.8 ± 0.63b	15.13 ± 0.67a	8.98 ± 0.96b
Water extractable C, mg C kg^{-1}soil	15.8 ± 0.89	18.4 ± 1.89	25.7 ± 1.9a	13.4 ± 1.19b
Microbial biomass C, mg C kg^{-1}soil	415 ± 27	325 ± 61	723 ± 109a	239 ± 86b
NO_3^--N, mg N kg^{-1}soil	10.2 ± 1.29b	18.9 ± 2.71a	2.56 ± 0.4b	4.79 ± 0.37a
NH_4^+-N, mg N kg^{-1}soil	1.57 ± 0.31	1.96 ± 0.27	0.57 ± 0.06a	0.35 ± 0.05b
N mineralization, mg N kg^{-1}soil d^{-1}	0.49 ± 0.09	0.49 ± 0.16	0.34 ± 0.08	0.10 ± 0.04

[†] Abbreviations: WR= White River; LWD = Leary Weber ditch.

[‡] In adjacent columns, values followed by different letters are significantly different at $P<0.05$.

Table 2. Average daily flux and annual emission of carbon dioxide and methane at the White River (WR) and Leary Weber Ditch (LWD) sites. Mean daily flux was computed for the entire period of the study. Annual emission was computed for the period April 2010 - April 2011 at WR, and May 2010-May 2011 at LWD. Values are means with standard error in parentheses.

	WR		LWD	
	Riparian	Crop field	Riparian	Crop field
		Carbon dioxide		
Daily flux, g CO_2-C m^{-2} d^{-1}	1.17 (0.09) a[†]	0.69 (0.12) b	2.19 (0.19) a	1.36 (0.35) b
Annual flux, Mg CO_2-C ha^{-1}	4.95 (0.68) A	2.56 (0.19) B	8.63 (0.58) A	4.54 (0.21) B
		Methane		
Daily flux, mg CH_4-C m^{-2} d^{-1}	+0.123 (0.03) a	-0.004 (0.03) b	-0.29 (0.03) b	+0.004 (0.04) a
Annual flux, kg CH_4-C ha^{-1}	+0.92 (1.61) A	+0.05 (0.24) B	-1.08 (0.22) B	+0.04(0.12) A

[†] For a given site, values followed by different letters are significantly different at $P < 0.05$.

Table 3. Regression coefficients (r^2) for the relationships between soil properties and gas (CO_2 and CH_4) fluxes measured at the study sites in mid-November 2010. Asterisks denote level of significance: *, $P < 0.05$; **, $P < 0.01$; ns = not significant.

Variable	Carbon dioxide flux		Methane flux	
	WR	LWD	WR	LWD
_____ mean daily fluxes during the entire study _____				
Water table depth[†]	ns	ns	0.35*	0.24*
Soil moisture	ns	ns	ns	ns
Soil temperature	0.66**	0.46**	ns	ns
_____ daily fluxes measured on November 17 and 19, 2010[‡] _____				
Bulk density	ns	ns	0.61**	0.42*
Clay	ns	ns	0.54**	ns
Sand	ns	ns	0.63** (-)[§]	ns
Moisture	ns	ns	0.45*	ns
SOC	ns	ns	ns	ns
C:N	0.32*	ns	ns	ns
DOC	ns	ns	ns	ns
MBC	0.41*	ns	ns	ns
NO_3^-	ns	ns	0.37* (-)	ns
NH_4^+	0.39*	ns	ns	ns
N mineralization	ns	ns	0.59**	ns

[†] For these relationships, the average value measured with sensors located near well 4 (WR) and well 1 (LWD) was used.

[‡] At the time of soil sampling and gas fluxes measurement, water table depth was 1.13 and 1.8 m below ground surface at WR (well 4) and LWD (well 1), respectively.

[§] Indicate a negative relationship.