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ORIGINAL RESEARCH

The Effects of Organic Matter Amendments on Greenhouse Gas Emissions from a Mitigation Wetland in Virginia's Coastal Plain

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Abstract There is concern that widespread restoration and/or creation of freshwater wetlands may present a radiative forcing hazard because of the potential for methane (CH_4) emissions. Yet data on greenhouse gas (GHG) emissions from restored wetlands are sparse and there has been little investigation into the GHG effects of amending wetland soils with organic matter (OM), a practice used to improve function of mitigation wetlands in the Eastern United States. In this study we measure GHG emissions across an OM gradient at the Charles City Wetland (CCW) in Charles City County, Virginia. We found soils heavily loaded with OM emit significantly more CO_2 than those that have received little or no OM. CH_4 emissions from CCW are low compared to reference wetlands and show no relationship with the loading rate of added OM or total soil carbon. We conclude that adding moderate amounts ($< \sim 150$ kg m^{-2}) of OM to the CCW does not greatly increase GHG emissions, while the addition of high amounts of OM produces additional CO_2, but not CH_4. CH_4 flux is highest under flooded conditions during warmer months but it still a modest contribution to global warming potential compared to soil CO_2 flux.

Keywords Methane · Greenhouse gas · Restoration · Soil organic matter

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Introduction

Despite making up only five to eight percent of world land cover (Mitsch and Gosselink 2007), wetland ecosystems play an important role in regulating the Earth's climate. Wetland soils contain 16 to 33 % of the earth's soil carbon (C) pool of 2,500 Pg (Lal 2005; Bridgham et al. 2006) and emit 20 to 40 % of methane (CH_4) (Bloom et al. 2010), an important greenhouse gas (GHG) (Myhre et al. 2013).

A review of North American wetland C exchange found that because of CH_4 emissions, most wetlands emit more GHG than they sequester on century timescales and therefore: "…that creating and restoring wetlands may increase net radiative forcing…" (Bridgham et al. 2006). Others have claimed that because wetlands are sustainable ecosystems and persistent as C sinks, the widely-used 100-year time horizon is too short, and that: "…wetlands can be created and restored to provide C sequestration and other ecosystem services without great concern of creating net radiative sources on the climate due to methane emissions" (Mitsch et al. 2013). But errors in both the math and reasoning underpinning this latter view have been exposed (Bridgham et al. 2014; Neubauer 2014), which reaffirms the potential century-scale impact of restored and created wetland CH_4 emissions on regional climate budgets.

While this controversy over the C balance of wetland restoration and creation is partly a disagreement about the appropriate use and calculation of global warming potential, versus sustained flux models, which account for annual pulses of GHGs (i.e., Frolking et al. 2006; Neubauer 2014), it also reflects the great uncertainty (100 %) around wetland GHG flux estimates (Bridgham et al. 2006). It thus may be particularly difficult to make long-term assumptions regarding restored and created wetland GHG fluxes given their complex histories of human disturbance and intervention and that they routinely

fail to achieve the same ecological function of reference ecosystems over short timescales (Zedler and Callaway 1999). An important remaining question is whether created or restored freshwater wetlands with mineral soils are in fact a sink or source of GHG over policy-relevant timescales?

In the eastern United States large areas of freshwater wetlands are created or restored as part of compensatory mitigation mandated by section 404 of the Clean Water Act, and they commonly suffer from an initial deficiency of soil organic matter (OM) (Stauffer and Brooks 1997; Whittecar and Daniels 1999) compared to natural wetlands (Bailey et al. 2007). Many studies have advocated for the amendment of created wetlands with OM in the form of salvaged topsoil or compost to help them achieve reference functionality (Stauffer and Brooks 1997; Whittecar and Daniels 1999; Bruland and Richardson 2004). Indeed, studies have found that moderate loading of compost OM into a created wetland increase woody plant development (Bailey et al. 2007) and soil functions, such as microbial decomposition and increased denitrification enzyme activity (Bruland and Richardson 2009; Sutton-Grier et al. 2009).

Few studies have measured GHG emissions from created or restored wetlands and fewer still have done so at sites amended with OM. It is unclear whether or not the practice of adding OM to created wetlands will have an effect on their radiative impact, though one study at a freshwater tidal wetland in Virginia may provide a clue. In this case wetland soil cores, when amended with leaf litter OM, tended to produce more CH_4 and CO_2 over unamended controls, while soils amended with compost OM produced more CO_2, but no additional CH_4 (Morrissey et al. 2013). These findings are compelling, but need to be replicated at an independent site and confirmed in situ.

The purpose of this study is to investigate how a gradient of added compost OM affects GHG emissions from a created mitigation wetland. Included in our analysis is an estimate of how long it would take for our restored wetland to change from a GHG source to a sink, calculated as the radiative forcing switchover time following Frolking et al. (2006).

Methods

Site Description

The study took place within the 20.8-ha Charles City Wetland Mitigation Site (CCW), which is located in Charles City County, Virginia, USA (37°20'37"N, 76°55'33"W), and owned by the Virginia Department of Transportation (VDOT) as part of its compensatory mitigation program (Bailey et al. 2007). Precipitation is the dominant hydrologic input and the CCW may hold up to 0.5 m of standing water during cooler months (Bailey et al. 2007). Site history is described in detail by Bergschneider (2005) and Bailey et al. (2007), but briefly summarized here. Prior to restoration the site was covered by upland mixed hardwood forest that had been partially converted to agricultural field. The soil was mapped as a complex of Chickahominy (fine, mixed, semiactive, thermic Typic Endoaquults) and Newflat (fine, mixed, subactive, thermic Aeric Endoaquults) series (Bergschneider 2005). Mitigation efforts attempted to convert field and remnant forest to wetland status during the winter of 1997–1998 by excavating into the subsoil (E or Btg horizon) to the depth of the presumed seasonal high water table. After revegetation, many parts of the site were found to be covered in facultative or upland plant species with much less hydrophytic cover than desired for mitigation purposes, a result attributed to restoration activities in which topsoil was lost, leaving compacted, low organic matter (OM) subsoil at the surface. The addition of an OM source had been proposed as a method for improving function of mitigation wetlands (Stauffer and Brooks 1997), but no data existed regarding the quantity of added OM required to achieve sufficiently improved wetland function in this setting. With a goal of determining optimal OM amendment loads for the wetland, a research group from Virginia Polytechnic Institute and State University implemented a gradient experiment in 2001 with 4 replicate plots of 4 OM loading rates (plus control) in an experimental block. Municipal wood and yard waste compost was rototilled into the topsoil of 4.6 by 3.1 m plots at loading rates of 56, 112, 224 and 336 kg m^{-2} (dry weight) in July, 2002. Control plots received only rototilling. Each plot was planted with five Pin Oak (*Quercus palustris*) and River Birch (*Betula nigra*) saplings, but otherwise the site was allowed to revegetate naturally from seed bank. In January, 2013 we found a mean count of 3.4 *Q. palustris* and 4.6 *B. nigra* survived in each 14.3 m^2 plot with some volunteer tree species, such as Red Maple (*Acer rubrum*) and Black Willow (*Salix nigra*), established sporadically.

Site Characterization

We measured the relative elevation of each plot near the gas collars used for measuring GHGs using a Topcon RL-H3A laser level and collected soil cores in each plot in September, 2011 using a 10-cm diameter soil-corer. Cores were split into 0 to 5 and 5 to 10 cm depth sections in the field. In the lab each core section was weighed wet and a subsample was weighed, oven-dried and re-weighed to estimate wet:dry ratios and calculate bulk density. Subsamples were analyzed for total carbon (C) and total (N) using a CE Instruments Flash Elemental Analyzer. We sampled soils again in September, 2012 using a punch tube and separated depth sections of 0 to 2 cm, 4 to 6, 9 to 11 and 19 to 21 cm in the field, and then composited corresponding depths from three replicate punches. These soils were analyzed for total C, total N (following the same method as above), digested following a nitric-perchloric acid method followed by colorimetric analysis of total phosphorus (P)

using a Beckman DU-64 spectrophotometer, Meilich-3-extractable P, KCl-extractable nitrate/nitrite (NO_x) and ammonia/ammonium (NH_x) using a Lachat Quickchem 8000 autoanalyzer. We installed litter fall traps (approximately 1000 cm^2) in each plot in September, 2012 and litter was collected during subsequent site visits.

Greenhouse Gas Sampling

We installed one 20-cm diameter PVC collar 10 to 15 cm into the soil in each plot for static chamber GHG gas sampling (Livingston and Hutchinson 1995; Weishampel and Kolka 2008) in fall 2011. After observing that the close approach necessary for setup and sampling of PVC collars was generating unreliable CH_4 data due to soil disturbance, we redeployed 30-cm diameter static chamber collars affixed with a water-fillable gutter in the spring of 2012. To avoid soil disturbance and improve CH_4 data quality, we used a remote rod sampling system, which allows chambers to be set up and sampled from a distance of 2 m (Winton and Richardson, in review) for subsequent trace gas sampling every 2 months from May, 2012 until January, 2013. We excluded from analysis unreliable CH_4 data generated by the original PVC collars, but include complete, unimpacted CO_2 data collected from these collars on October 2011 and February 2012. Sampling errors on September 12, 2012 impacted a large portion of CH_4 flux data, so we repeated measurements on September 26 for CH_4 analysis. CO_2 results were not affected and we averaged results from both September, 2012 dates for subsequent analysis.

Static chambers were opaque and total extracted gas volume was never greater than 5 % of chamber headspace. We left any plants growing within chamber footprints intact unless they were long enough to interfere with chamber setup, in which case we clipped them. On each sampling date we collected headspace gas four times over the course of half-hour incubations from collars in each of the 20 plots. Following placement of the static chamber top on the collar we immediately extracted a 50-ml headspace sample via a plastic syringe and deposited it into a mylar gas-tight sample bag. We recorded ambient air temperature (T), internal chamber T, soil T at 5 cm depth for initial and subsequent samples taken approximately 5, 15 and 30 min following chamber setup. We transported gas bags to the Duke University Wetland Center laboratory and analyzed within 1 week of collection on a Varian 450 Gas Chromatograph (GC) equipped with a flame ionization detector and methanizer to analyze CH_4 and CO_2 concentrations synchronously. All samples were run in duplicate with the mean value used for gas flux calculations unless duplicate values differed by >10 %, in which case the obviously outlying value was assumed to stem from analytical error and discarded. Flux was estimated by linear regression of sample concentrations as a function of time elapsed. If a threshold r-squared value of 0.90 was not met, we removed one outlying point if it improved fit to>0.90 (approximately 5 % of incubations), otherwise such estimates were treated as failed incubations and discarded. We estimate the minimum detectable flux for CO_2 was 52 mg m^{-2} h^{-1} and for CH_4 was 0.037 mg CH_4 m^{-2} h^{-1}. We analyzed all samples for nitrous oxide, but found flux to be below minimum detection thresholds for approximately 90 % of incubations, so we focus our results and discussion on CH_4 and CO_2.

Hydrologic Data

We measured soil volumetric water content (SVWC) in the top 5 cm using a Fieldscout 100 time domain reflectometry probe (Spectrum Technologies) starting in May, 2012. We recorded the mean of five measurements taken adjacent to each chamber collar. We installed five Odyssey loggers (Dataflow Systems, Christchurch, New Zealand), which monitored water level hourly starting in February 2012. We collected daily rainfall and air T data from nearby meteorological stations in Williamsburg and Norfolk, Virginia, which are publicly accessible via the National Climatic Data Center (Lawrimore et al. 2011).

Statistical Analyses

We used ANOVA and Tukey's honest significant difference test to test for differences in gas flux between groups of plots with different OM treatments and linear regression to look for trends in gas flux across the OM gradient. We evaluated all data for normality by generating box-and-whisker, histogram and quantile-quantile plots and log-transformed data when necessary. We explored relationships between gas flux and potential explanatory variables using the Ecodist package (Goslee and Urban 2007) and by building generalized linear models (GLM). We used these models to estimate daily emissions of CO_2 and CH_4 and generate an annual flux budget from March 1, 2012 to February 28, 2013. We used JMP Pro 11 (SAS Institute Inc 2013) to plot GLM outputs. All other statistics were computed using the R programming language (R Core Team 2014) and in Microsoft Excel 2010.

Carbon Balance

We compare the relative radiative impacts of soil CH_4 and CO_2 fluxes by multiplying CH_4 by its 100-year sustained global warming potential of 45 (Neubauer and Megonigal 2015). We estimate radiative forcing switchover time, determined by the ratio of CO_2 stored : CH_4 emitted, (Frolking et al. 2006) for the CCW by assuming annual net ecosystem exchange (NEE) estimates (Bailey 2006) and annual CH_4 flux estimates generated in this study will be sustained over centuries. Bailey (2006) found NEE to be negative (net emission of CO_2) for most of the CCW plots because of rapid oxidation of

added OM early in the experiment, therefore we only used his positive mean NEE values (net uptake of CO_2) from the lowest loading rates (141.1 and 29.9 g CO_2-C m^{-2} y^{-1}) to generate a range of potential radiative forcing switchover times.

Results

GHG Fluxes

CO_2 Flux

We observed the highest CO_2 fluxes (>400 mg m^{-2} h^{-1}) during warmer, drier months and these contrast with fluxes approaching minimum analytical detection limits during cold, wet months (Fig. 1). CO_2 emissions from soil directly responded to increases in soil T (Fig. 2) and in general, the higher CO_2 emissions are associated with higher OM loading rates; linear regression of log-transformed CO_2 flux as a function of OM treatment shows significant positive relationships across all sampling months except September (Table 1). The relationship between OM and CO_2 emission is strongest during peak flux in July which is one of only 2 months (the other being January) where significant differences in CO_2 flux between OM treatments occur.

A GLM with three parameters: soil T (coefficient=0.058; $p<0.001$), soil volumetric water content (SVWC; coefficient=−0.025; $p<0.001$), and surface soil total C (coefficient=0.031; $p<0.001$), explains much of the variability ($r^2=0.75$) in log-transformed CO_2 flux across all sampling dates (Fig. 3a). Soil T and SVWC are the two most important terms in our GLM explaining log-transformed CO_2 flux variability across seasons, with r-squared values of 0.50 and 0.49 respectively. Soil T and SVWC are correlated with each other (r-squared of 0.40), but this relationship is driven by one sampling date in July when the site was both very warm and very dry. Including both soil T and SVWC improves model r-squared to 0.71. The third model parameter, total surface soil C partly reflects the amount of OM available to be decomposed. The effects of soil C on CO_2 flux become obvious when the site is sufficiently dry (i.e., July), but during wetter periods the importance of surface soil C is obscured. So while soil C is very weakly correlated with log-transformed CO_2 flux across all sampling dates (r-squared of 0.05), including it in the GLM helps improve fit (r-squared of 0.75) and reduces the Akaike information criterion.

With T and soil moisture held relatively constant across the site during a given sampling date (relative to seasonal changes), we found surface soil C to be the most important parameter (coefficient=25; $p=0.001$) explaining CO_2 flux in July (r-squared of 0.52). The inclusion of total soil N at 20 cm depth (coefficient=3900; $p=0.05$) improved our GLM r-squared to 0.63 (see Fig. 3b) and it was not highly correlated with surface soil C (r-squared of 0.24).

Since we do not have high frequency SVWC and soil T data, we used highly correlated water level ($r^2=0.93$) and mean air T from the preceding 7 days ($r^2=0.98$; as measured in Norfolk, Virginia; National Climatic Data Center), as respective substitutes to model daily CO_2 flux from March 1, 2012 through February 28, 2013. We assume surface soil total C would remain constant over the year. From this model we estimate an annual CO_2 flux ranging from 0.44 to 0.64 kg CO_2-C m^{-2} $year^{-1}$ from the low to high end of the OM gradient.

CH_4 Flux

We find CH_4 fluxes consistently exceed minimum analytical detection only when soil T was at least 15 °C and some ponded water was present at the CCW (see Table 2). We identify a threshold of 50 % SVWC, below which CH_4 was never greater than 0.13 mg CH_4 m^{-2} h^{-1} (Fig. 4). When conditions at the CCW are favorable for methanogenesis fluxes are highly variable across plots. We observed maximum CH_4 fluxes of approximately 3 to 5 mg m^{-2} h^{-1} during sampling in

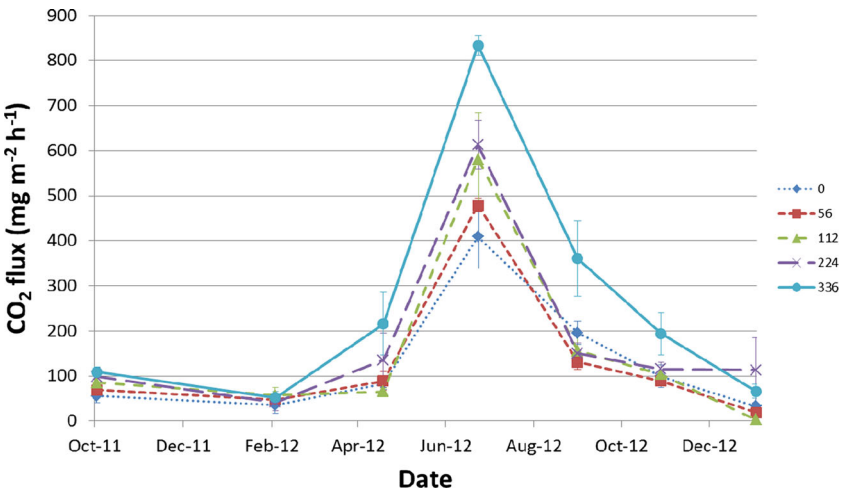

Fig. 1 Mean (±SE) carbon dioxide flux from the organic matter experimental plots at the Charles City Wetland in Charles City County, Virginia across nine sampling dates from September, 2011 to January, 2013. Different *dash patterns* represent loading rates of organic matter in kg m^{-2}

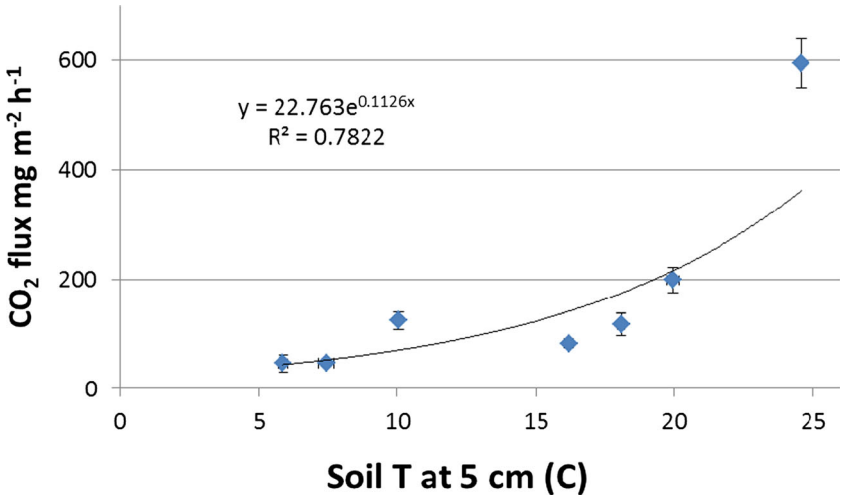

Fig. 2 Mean carbon dioxide flux from a range of organic matter treatments as a function of soil temperature at 5 cm depth from experimental plots at the Charles City Wetland in Charles City County, Virginia across eight sampling dates from November, 2011 to January, 2013. *Error bars* represent standard errors of the mean

May and September 2012. But even on these high flux dates we are unable to detect any statistically significant patterns in CH_4 flux related to OM loading rate using linear regression (May, 2012: p=0.16; and Sept. 2012: p=0.21) and Tukey's honest significant difference test (May, 2012: p=0.92; and Sept. 2012: p=0.60) when comparing the highest loading rate to control). Furthermore the non-significant linear regression coefficients were negative, with lower CH_4 flux associated with greater OM amendment, the opposite of what would be expected if added OM is fueling CH_4 production and emission.

To model annual CH_4 flux we assume CH_4 will respond in a binary fashion to soil T and SVWC based on the thresholds we identified. We imputed mean measured CH_4 flux from CH_4-favorable sampling dates (1.1 mg CH_4 m^{-2} h^{-1}) for each day during which conditions met the threshold we identified (soil T>15 °C and SVWC>50 %). If the soil T and SVWC thresholds were not both met, we assumed CH_4 flux would be equal to mean measured CH_4 flux from CH_4-unfavorable sampling dates (0.012 mg CH_4 m^{-2} h^{-1}). By running this model from March 1, 2012 through February 28, 2013 we estimate an annual flux of 3.1 g CH_4-C m^{-2} y^{-1} from the CCW.

Table 1 Summary of linear regression and ANOVA tests for differences and trends in log-transformed carbon dioxide (CO_2) emissions between and across gradient of plots treated with different levels of organic matter (OM) at the Charles City Wetland in Charles City County, Virginia

Month	Linear regression		ANOVA
	p-value	r-squared	p-value
Oct., '11	**0.021**	0.22	0.231
Feb., '12	0.93	−0.05	0.813
May, '12	**0.028**	0.24	0.165
July, '12	**<0.001**	0.55	**0.018**
Sept., '12	0.133	0.12	0.116
Nov., '12	**0.009**	0.40	0.126
Jan., '13	**0.043**	0.21	**0.003**

Values that meet p<0.05 are *bolded*

Carbon Balance

During the sampling dates when CH_4 flux was large enough to be detectable (May and September), its contribution to radiative forcing was relatively minor on average (less than one-third) when compared to soil CO_2 flux using a newly suggested 100-year sustained global warming potential of 45 for CH_4 (Neubauer and Megonigal 2015) (Fig. 5). If we make the same comparison using our modelled annual CO_2 and CH_4 flux estimates we find CH_4 contributes approximately 8 to 12 % of the radiative forcing budget. The CO_2^-sequestration:CH_4-flux ratio of CCW ranges from 125 to 26, corresponding to a radiative forcing switchover time range of less than 300 years following Neubauer's (2014) model.

Hydrology and Soil Elevations

Water level data suggest that the hydrology of CCW is controlled by precipitation inputs with storm events and dry spells driving periodic fluctuations of more than 1 m in the water table (Fig. 6). Ponded water was present at the site 59 % of the time from 22 February, 2012 to 21 January, 2013 and reached a maximum depth of 14 cm above the mean elevation of unamended plots. The distribution of plot elevations is approximately normally distributed with a standard deviation of 4 cm and two outliers: a 12 cm "hummock" and a −9 cm "hollow." Pairwise comparison (ANOVA) of plots grouped by OM loading rate shows no significant differences in mean elevation, though there is a weak (r^2=0.18), but significant (p<0.05) positive linear trend in elevation across the OM gradient.

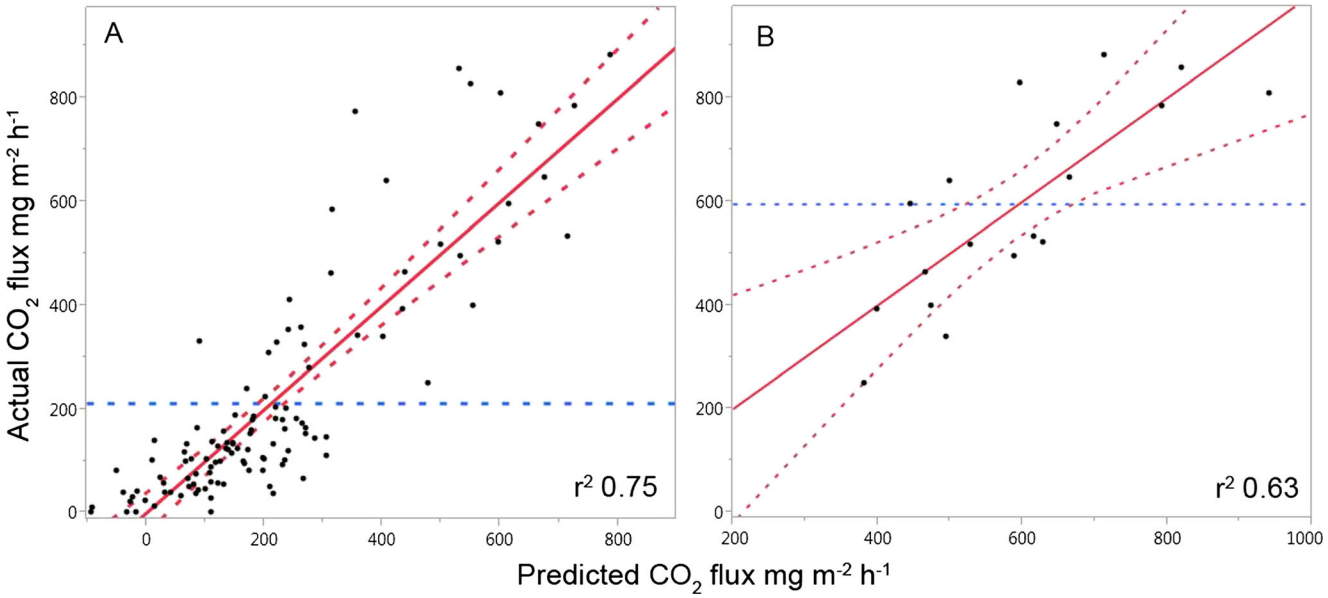

Fig. 3 Actual carbon dioxide flux compared to linear model predictions at the Charles City County Wetland in Charles City County, Virginia for: A) data across five sampling dates from May 2012 to January 2013 and multiple regression predictions based on soil temperature (5 cm depth), soil volumetric water content, and total soil carbon (top 5 cm); and B) data from 22 July 2012 and linear predictions based on total soil carbon (top 5 cm) and total soil nitrogen at 20 cm depth. *Dashed curves* represent 95 % confidence intervals for the regression line. *Dashed horizontal line* indicates mean carbon dioxide flux value

Soil Nutrients

Total soil C data show that as much as 50 % of the added OM has been lost since 2005 (Bailey et al. 2007), with the biggest loss observed in plots loaded with 112 and 224 kg m^{-2} OM. Nevertheless, the gradient, as originally established, persists (see Fig. 7), with total C in the top 10 cm of soil ranging from approximately 2 to 13 %. Mean and standard error (SE) litter fall across the plots during the fall of 2012 was 0.37± 0.045 kg m^{-2}, which assuming litter is 50 % C by weight (Bocock 1964), represents an input of 0.19±0.023 kg C m^{-2} y^{-1} to surface soils. Total soil C, N and P are generally higher in plots that received higher loading rates of OM, but decrease with depth such that differences between loading rates are negligible at 10 and 20 cm depth. KCl-extractable NH_x and NO_x and Mehlich-3-extractable P follow roughly similar patterns.

Discussion

Controls of Greenhouse Gas Emissions

Wetland GHG flux at the CCW is moderated by soil T because of the temperature dependence of soil respiration (Lloyd and Taylor 1994), as well as hydrologic dynamics. Hydrology is important because saturation inhibits aerobic decomposition and creates conditions favorable for CH_4 emission (Whalen

Table 2 Summary of monthly averages (±SE) soil temperature (at 5 cm depth), hydrology and soil carbon emissions from the Charles City Wetland in Charles City County, Virginia. Methane (CH_4) values reported below our minimum detection threshold (0.037 mg m^{-2} h^{-1}) are the results of averaging many zero fluxes with a few low values. All data collected in 2012 except for October, 2011 and January, 2013

Month	Soil temp. °C	Water level cm	Soil volumetric water content %	CH_4 emissions mg·m^{-2}·h^{-1}	CO_2 emissions
Oct.	16.2±0.2	3.7±0.8[a]	NA[b]	NA[c]	77±8
Feb.	7.3±0.3	5.9±0.9	NA[b]	NA[b]	47±5
May	18.1±0.1	5.3±0.6	65.6±1.7	0.82±0.20	117±21
July	24.6±0.08	−39.9±6.4	29.9±2.6	0.02±0.01	595±43
Sept.	20.0±0.2	6.0±0.5	60.2±1.0	1.29±0.27	188±30
Nov.	10.1±0.09	3.0±0.5	55.5±1.5	0.02±0.01	124±14
Jan.	5.9±0.2	9.2±0.6	60.5±1.4	0	32±6

[a] water depth measured by hand within plots rather than in wells; may not be comparable to other data

[b] not measured

[c] methodological issue led to unusable CH_4 data, therefore we omit from analysis. See Methods section

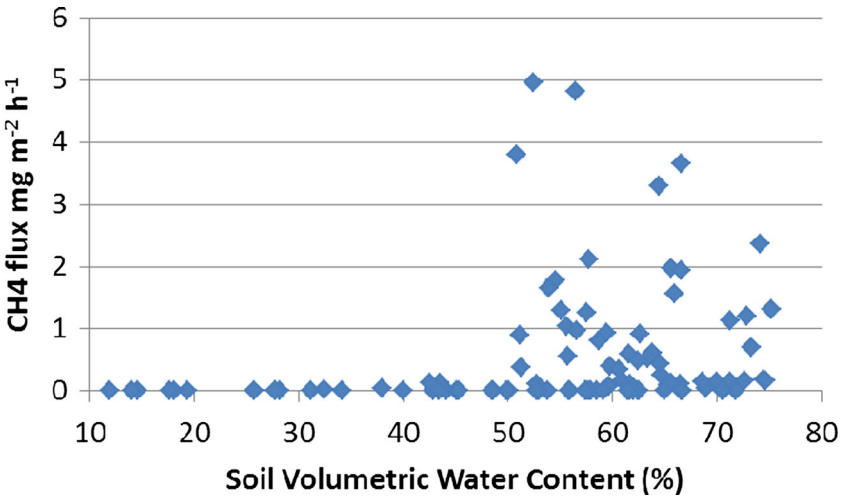

Fig. 4 Methane flux (CH_4) as a function of soil volumetric water content measured from the organic matter experimental plots at the Charles City Wetland in Charles City County across five sampling dates from May 2012 to January 2013

2005), as is illustrated by the soil volumetric water content threshold we identified (Fig. 4). In addition to facilitating CH_4 flux, high water levels are also associated with lower rates of CO_2 emission because soil respiration is typically oxygen-limited in a wetland setting. Our hydrologic data are consistent with previous work indicating that the CCW is a groundwater recharge system with hydrologic inputs dominated by precipitation (Despres 2004).

Related to hydrology is the elevation of surface soil. While the site has very little relief, microtopographic features (<10 cm) are related to OM loading rate and can be significant relative to water depth at the CCW. During OM addition to the CCW in 2001 there was difficulty in completely incorporating the highest OM loading rates into plots, which led to mounding (Daniels et al. 2005). Therefore we suspected elevation might be a confounding factor in predicting GHG flux since it correlates with OM loading rate, but when we included it as a predictor of GHG fluxes in our GLM tests it did not emerge as significant. Thus we conclude that differences in elevation across the OM gradient are not driving seasonal GHG patterns.

In addition to soil T and hydrology, surface soil C also correlates with CO_2 flux, which suggests that the addition of OM causes higher rates of aerobic decomposition and/or root respiration. We assume that soil N at depth correlates with CO_2 flux because a greater N pool in the rooting zone should stimulate higher rates of autotrophic and heterotrophic respiration related to N mineralization (Schlesinger 1997).

While we were able to find some relationships between soil C/N and CO_2 flux, CH_4 did not correlate with any of the soil chemical properties we measured. Hydrology and T both control rates of CH_4 production by dictating oxygen availability and demand (Whalen 2005), which explains why we found CH_4 flux to be very low during cold and/or dry periods. CH_4 flux variability is consistent with results from other forested wetlands of the Southeastern US but our annual CH_4 flux estimate was on the low end of the range of published estimates for analogous systems (Table 3).

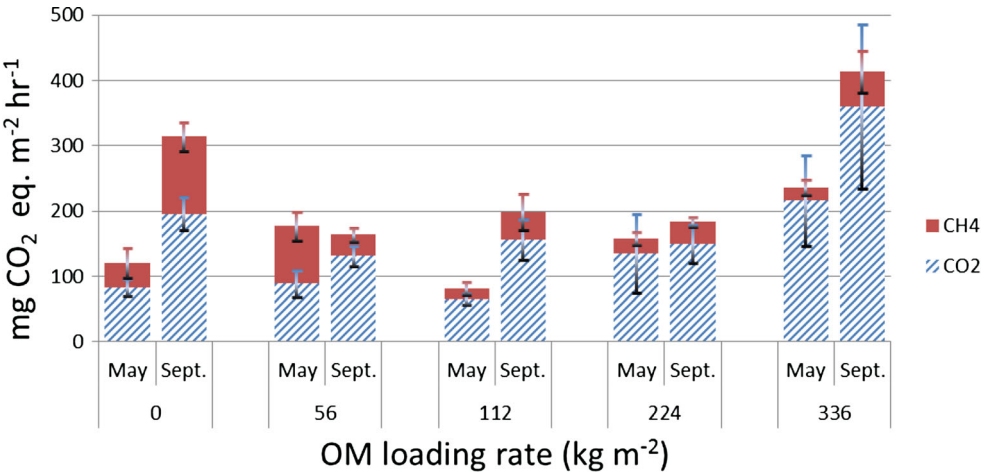

Fig. 5 Carbon dioxide (CO_2) and methane (CH_4) flux from soil across five levels of organic matter loading rates estimated from sampling on 7 May and 26 September, 2012 at the Charles City Wetland in Charles City County, Virginia, USA. Note: CH_4 was converted to CO_2-equivalents by multiplying by 45—its 100-year sustained global warming potential following Neubauer and Megonigal (2015). *Error bars* represent standard errors of the mean

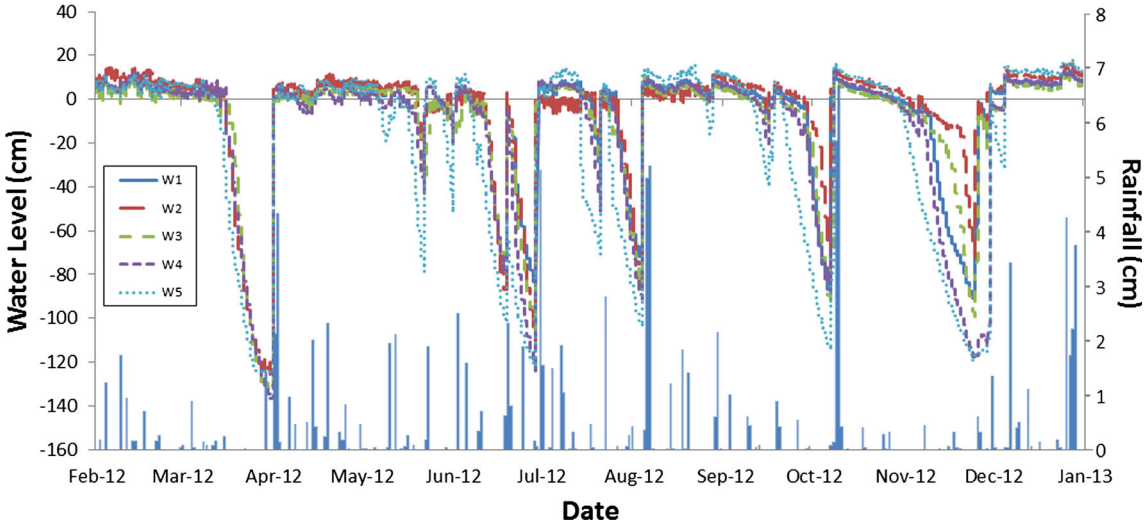

Fig. 6 Water level as recorded by five 1.5 m loggers placed in 1.5 m wells (W1 through W5) at the Charles City Wetland in Charles City County, Virginia, USA from 22 February, 2012 to 21 January, 2013. Positive values indicate standing water. Overlaid precipitation data are from a station in nearby James City County, Virginia (National Climate Data Center)

CCW CH_4 flux shows no significant relationship with OM loading rate, suggesting that if excess nutrients and enhanced primary productivity are enhancing methane production, then the increase is being cancelled out by concomitant CH_4 oxidation. This result contrasts with findings from another study in which addition of OM to intact wetland soil cores led to higher rates of potential net methane emissions compared to controls (Ballantine et al. 2014). Ballantine et al. 2014 found that adding OM created conditions more favorable for methanogenesis through an indirect mechanism—by increasing water retention. At the CCW, OM additions may have had the opposite effect on soil moisture because of a slight mounding effect as described earlier. Our data from the relatively drier months of May and July show weak (r-squared of 0.16 and 0.14, respectively), marginally significant ($p<0.09$ and $p<0.11$, respectively) relationships between SVWC and OM loading rate. Increasing soil C by adding OM does not necessarily provide additional C substrate for methanogens, but it may alter methane production and/or oxidation because of indirect hydrologic effects. Heavy OM addition may elevate the soil surface allowing for more oxic conditions (our study), or conversely, it may enhance water holding capacity facilitating anoxia (Ballantine et al. 2014).

The OM gradient at the CCW does not incorporate a test of the effects of different OM types and results of other studies conflict as to whether or not OM quality matters. Ballentine et al. (2015) found that several OM sources (straw, topsoil, straw+biochar, biochar) led to statistically indistinguishable increases in potential net CH_4 emissions over control. In contrast Morrissey et al. (2013) found added leaf litter led to increased CH_4 flux, but added compost did not. It is noteworthy that our results are consistent with one of the conclusions reached by Morrissey et al. (2013): that adding compost increases CO_2 flux, but not CH_4 flux.

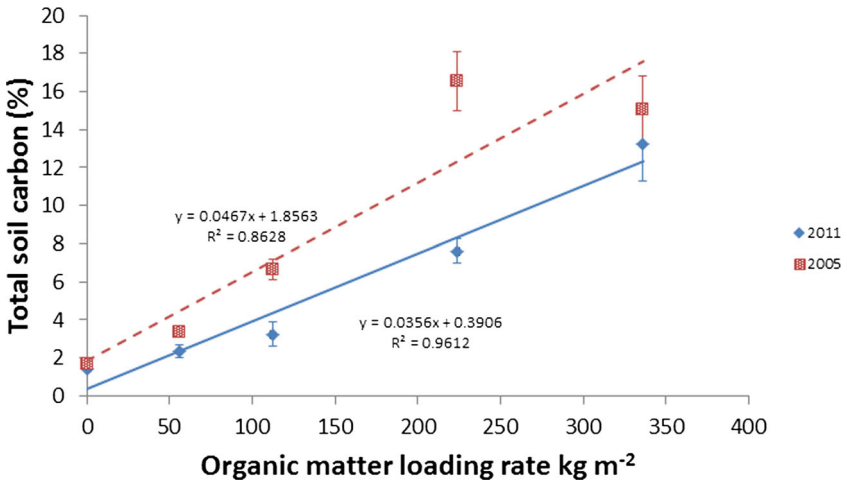

Fig. 7 Linear regressions of mean (±SE) total carbon in top 10 cm of soil in 2011 across organic matter amendment plots at the Charles City Wetland in Charles City County, Virginia, USA. 2005 data from Bailey et al. (2007)

Table 3 Review of methane (CH_4) emissions rates in kg CH_4-C ha^{-1} y^{-1} from natural and restored forested wetlands of the Southeastern United States

CH_4 flux	Location	Type	Reference
554	Newport News Swamp, Va.	Natural	(Wilson et al. 1989)
427	Newport News Swamp, Va.	Natural	(Wilson et al. 1989)
311	Ogeechee River, Ga. (west)	Natural	(Pulliam 1993)
297	Okeefenokee Swamp, Ga.	Natural	(Flebbe 1982)
262	Creeping Swamp, NC	Natural	(Mulholland 1981)
107	Timberlake Restoration Preserve, NC	Restored	(Morse et al. 2012)
92	Ogeechee River, Ga. (east)	Natural	(Pulliam 1993)
72	Palmetto Peartree Preserve, NC	Natural	(Morse et al. 2012)
31	Charles City Wetland, Va.	Restored	This study
14	Timberlake Restoration Preserve, NC	Restored	(Morse et al. 2012)
0.5	Timberlake Restoration Preserve, NC	Restored	(Morse et al. 2012)

Climate Impacts of Created/Restored Wetlands

The radiative forcing switchover time (Frolking et al. 2006) for CCW is uncertain because of high variability in NEE (Bailey 2006) and CH_4 flux data (this study). Furthermore, for this analysis we must assume that CH_4 emissions and NEE will remain constant over many decades. In reality NEE is likely to be dynamic over at least several decades of succession (Odum 1969). The temporal mismatch in our input data—NEE data from 2006 and CH_4 data from 2012—is a potential source of error. If the CCW ecosystem has become a more efficient C sink over the 6-year period as trees have matured, then we may be overestimating radiative forcing switchover time. Therefore it would take a long-term monitoring approach to improve the certainty of radiative forcing switchover time for the CCW. But despite the shortcomings mentioned above, we may conclude that CCW has a relatively short radiative forcing switchover time due to its low CH_4 flux. Importantly, CCW will likely become a net GHG sink more quickly than at least six out of eight wetlands analyzed by Neubauer (2014).

While we may be tempted to generalize the results from CCW to all other created/restored wetlands, we caution that wetlands are diverse, idiosyncratic ecosystems, especially those which have undergone intense transformations, as the CCW has. Many factors determine how much CH_4 is produced and emitted, including the quality of any added OM, type of plant cover, productivity and hydrology. Since all data were collected from a single site amended with a single OM type (compost), our conclusions may or may not apply to other created/restored wetlands but our analytical approach for assessing radiative forcing switchover time should prove useful.

Temporal Changes at Charles City Wetland

We found micro-elevational differences between plots, described above, to be less pronounced in 2012 compared to conditions in 2005 reported by Bailey et al. (2007). The relationship between OM loading rate and elevation was far weaker in 2012 (see Table 4), which could be the result of settling or subsidence due to more rapid OM oxidation in elevated, high-OM plots. The higher rates of soil respiration that we and Bailey et al. (2007) detected coming from higher OM plots are consistent with an oxidation-subsidence explanation for the loss of elevation, as is the difference in total soil C between 2005 and 2012 we observed (see Fig. 7).

Our annual soil respiration budget does not fully account for the soil C losses in the 112 and 224 kg m^{-2} plots we observe between 2005 and 2012, which were approximately 3.4 and 8.9 % C, respectively (Fig. 7), corresponding to respective losses of 0.8 and 1.6 kg C m^{-2} y^{-1} over 7 years from the top 10 cm of soil. These loss rates are two to four times greater than our estimated annual soil respiration loss from these plots: 0.42 and 0.49±0.032 kg CO_2-C m^{-2} y^{-1} respectively, but bracket estimates of approximately 1.3 and 1.4 kg CO_2-C m^{-2} y^{-1} respectively made by Bailey (2006). The relatively wet conditions in 2012 relative to 2005 (Table 4) could explain why we found soil respiration to be lower than Bailey

Table 4 Comparison of microtopographic and growing season hydrologic conditions at the Charles City Wetland in Charles City County, Virginia between 2005 (Bailey et al.) and 2012 (this study)

Year	Rainfall (Apr. - Oct.; cm)[b]		Elevation across OM loading rates		
	Total	Depart. from normal	Range (cm)	Lin. reg. r-squared	Lin. reg. p-value
2005	69	−7.5	11	0.55	<0.001
2012	82	+5.7	6	0.17	<.05

(2006) since saturation inhibits soil respiration. The CCW was relatively wet during the 2012 growing season when it received 82 cm of rain (7.5 % above mean; National Climatic Data Center; Lawrimore et al. 2011) and held ponded water 52 % of the time. This contrasts with conditions during the 2005 growing season when the CCW received 10 % less rainfall than average (National Climatic Data Center; Lawrimore et al. 2011) and water was ponded just 25 % of the time (Bailey et al. 2007). We also suspect that the rate of C loss has decreased as soil C content decreased leaving more recalcitrant materials over the past 7 years. Some C loss may also be due to leaching of dissolved OM into lower soil layers and/or transport of particulate OM during floods.

The overall seasonal pattern in soil CO_2 flux we observe is similar to what Bailey (2006) reported from the CCW for 2005/2006 with peak respiration of greater than 400 mg m^{-2} h^{-1} during summer dry periods and low CO_2 flux of less than 100 mg m^{-2} h^{-1} during wet winter months. The positive relationship between CO_2 flux and soil OM loading rate is also consistent with Bailey's (2006) results.

Conclusions

We found little evidence to suggest that added composted yard waste increases CH_4 emissions from CCW a decade after restoration. CH_4 emissions are only significant when soils are warm and water levels and soil moisture are high. Even when CH_4 flux is at its greatest magnitude, it still represents a relatively modest contribution to radiative forcing compared to soil CO_2 flux.

Yet even if CCW were to produce no CH_4, it would still be a net CO_2 source at high OM loading rates because of negative NEE (Bailey 2006), at least until the excess OM is respired. Therefore the addition of high levels of OM (>112 kg m^{-2}) to created wetlands may be detrimental to greenhouse gas budgets due to enhanced soil respiration. At the CCW, heavy additions of OM decompose over time while adding little in the way of tangible productivity increases (Bailey et al. 2007) or soil geochemistry improvement (Bruland and Richardson 2009), not to mention incurring greater material transport and associated construction costs.

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