WSSI Consortium Projects Progress Report

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WSSI Grant #	**25000.01E7**
Grant Years:	**2010-2013**
Progress for the period:	**Year: 2010-13 (Final Report)**
University	**Duke University**
Project Title	**A comparison of wetland functions and services on restored wetlands of the Piedmont: carbon storage and GHG release estimates.**
Prepared by:	
Date: 10/31/2014	

NARRATIVE: *Summarize activities accomplished during this reporting period using only space provided below (10 pt. min).*

Restoration sites in Virginia were studied to determine changes in soil carbon flux in response to the experimental carbon additions. Analysis of bulk density, total carbon and total nitrogen were consistent with earlier studies at the site, although lower amounts of soil organic matter (OM) were found. However, plots that received greater OM amendment loads have higher total carbon and nitrogen and lower bulk density. There is concern that widespread restoration and/or creation of wetlands may present a radiative forcing hazard because of the potential for high rates of methane (CH_4) emissions. Yet data on greenhouse gas (GHG) emissions from restored wetlands remains relatively sparse and there has been little investigation into the GHG effects of amending wetlands with soil organic matter (OM), a practice used to improve function in mitigation wetlands in the Eastern United States. In this study we evaluate the effect of added OM on GHG across an organic matter gradient at the Charles City Wetland (CCW) in Charles City County, Virginia, ten years post original OM additions. Our data suggest that soils heavily loaded with OM are emitting significantly more CO_2 than those that have received little or no OM amendment. Emissions of CH_4 are low compared to those of other forested wetlands in the region and show no relationship with the loading rate of added OM or total soil carbon. We conclude that adding moderate amounts of OM to the CCW does not greatly increase GHG emissions, while the addition of high OM loading rates produces additional CO_2, but not CH_4.

COMMENTS: *Note any delays, problems, or special circumstances affecting progress and how you intend to address them.*

See the attached final report for a complete analysis of the study. A new method for reducing variability and errors in measuring GHG fluxes from static chambers was developed for this study and is presented in the report as well as annual GHG fluxes

Project Benchmarks and Deliverables	% Completion	Anticip. Completion Date
Collars installed for gas sampling	100	Complete
Wells and soil moisture probes installed and tested	100	March, 2012
A full year of monthly greenhouse gas flux data collected	100	April 2013
Manuscript submitted for publication	1	November 2014

WSSI Project Final Report

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October 29, 2014

Title

The effects of organic matter amendment on greenhouse gas emissions from a mitigation wetland in Virginia's coastal plain

Abstract

There is concern that widespread restoration and/or creation of wetlands may present a radiative forcing hazard because of the potential for high rates of methane (CH_4) emissions. Yet data on greenhouse gas (GHG) emissions from restored wetlands remains relatively sparse and there has been little investigation into the GHG effects of amending wetlands with soil organic matter (OM), a practice used to improve function in mitigation wetlands in the Eastern United States. In this study we evaluate the effect of added OM on GHG across an organic matter gradient at the Charles City Wetland (CCW) in Charles City County, Virginia. Our data suggest that soils heavily loaded with OM are emitting significantly more CO_2 than those that have received little or no OM amendment. Emissions of CH_4 are low compared to those of other forested wetlands in the region and show no relationship with the loading rate of added OM or total soil carbon. We conclude that adding moderate amounts of OM to the CCW does not greatly increase GHG emissions, while the addition of high OM loading rates produces additional CO_2, but not CH_4.

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 percent of the earth's soil Carbon (C) pool of 2,500 Pg (Lal 2005; Bridgham et al 2006) and emit 20 to 40 percent 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 are net emitters of GHG on century timescales and therefore: "…large CH_4 emissions from conterminous US wetlands suggest 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, which reaffirms the potential century-scale impact of restored and created wetland CH_4 emissions on regional climate budgets (Neubauer 2014; Bridgham et al 2014).

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 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 wetlands are created 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 mulch 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 OM into a created wetland increase woody plant development (Bailey et al 2007) and soil functions, such as microbial biomass and 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.

Increased C substrate and/or productivity due to nutrient content of added OM could enhance CH_4 flux given the relationship between OM loading rate and primary productivity (Bailey 2006), which across wetland systems has been correlated with CH_4 flux rate (Whiting and Chanton 1993). Alternatively, added OM could reduce CH_4 emissions by altering the physical structure of soil. The addition of OM increases soil elevation (Bailey et al 2007) and reduces bulk density (Bruland and Richardson 2009), which could allow surface soil to remain more oxic, facilitating methane oxidation as well as aerobic, rather than anaerobic methanogenic, respiration.

The purpose of this study is to investigate how a gradient of added OM affects GHG emissions from a created mitigation wetland on mineral soils. 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-hectare Charles City Wetland Mitigation Site (CCW), which is located in Charles City County, Virginia, USA, and owned by the Virginia Department of Transportation (VDOT) as part of its compensatory mitigation program (Bailey et al., 2007; see Fig. 1AB). 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 Type Endoaquults) and Newflat (fine, mixed, subactive, thermic Aeric Endoaquults) (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 wet and dry experimental blocks (see Fig. 1C). 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} 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 (1112 series) 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 30 cm^2) in each plot in September, 2012 and litter was collected during subsequent site visits.

Greenhouse Gas Sampling

In late summer 2011 we imbedded 20 cm diameter PVC collars 10 to 15 cm into the soil in each plot of the wet block for static chamber GHG gas sampling (Livingston and Hutchinson, 1995). During chamber setup we placed a PVC cap with a rubber gasket over collars, but after sampling in September and October, 2011 and February, 2012, we found that this chamber design and/or sampling technique was producing CH_4 data that frequently failed to follow a linear pattern of accumulation within chamber headspace. CO_2 concentrations accumulated in a linear fashion within headspace as expected, but extraordinarily high initial CH_4 concentrations (up to 1500 ppm; roughly 1000 time ambient concentration) within the headspace indicated that capping the collar and/or standing near the collar during sampling was purging CH_4 stored within soil pores. To mitigate this problem we redesigned our chambers and collars in spring of 2012 to minimize collar disturbance during chamber setup. We accomplished this by building new permanent collars with gutters that could be filled with water, capped and sampled from a distance of 2 m (see Fig. 2). An internal computer fan powered by a 9-volt battery circulates chamber head space from which air samples are extracted using a 2 m tube, 1 mm inner diameter plastic tube. Chamber caps were also equipped with a thermocouple allowing for internal chamber temperature (T) to be recorded during each sample extraction and we coated them with reflective aluminum foil to minimize solar warming as recommended by the US Department of Agriculture (Parkin and Venterea 2010). We installed these new collars in April, 2012 and sampled for estimation of trace gas flux every two months from May, 2012 until January, 2013.

On each sampling date we collected headspace gas four times over the course of a half-hour incubation from collars in each of the 20 plots. Following placement of the 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 T, internal chamber T, soil T at 5 cm depth for initial and subsequent samples taken approximately 5, 15 and 30 minutes following chamber setup. Gas bags were transported to the Duke University Wetland Center laboratory and analyzed within one week of sampling on a Varian 450 Gas Chromatograph (GC) equipped with a flame ionization detector, methanizer, and electron capture detector to analyze CH_4, CO_2 and nitrous oxide (N_2O) concentrations synchronously. All samples were run in duplicate and when duplicate values differed by <10% the mean was used for gas flux calculations. Flux rate 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, one outlying point was occasionally (approximately 5% of incubations) removed to improve fit.

Supplementary Data

On each sampling date after finishing headspace incubations we measured soil moisture in the top 5 cm using a Fieldscout 100 time domain reflectometry probe (Spectrum Technologies). We recorded the mean of five measurements taken adjacent to each chamber collar.

In September 2012 we installed pore water wells in each plot and starting in November, 2012 began collecting pore water samples for subsequent analysis of total dissolved C and dissolved organic C using a Shimadzu TOC-5000 A, total phosphorus (P) following persulfate I digestion method (Wetzel and Likens 1979), nitrate/nitrite (NO_x) and ammonia/ammonium (NH_x) using a Lachat Quickchem 8000 autoanalyzer (EPA method 350.1).

Statistical analyses

We used ANOVA 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 if 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 JMP Pro 11 (SAS Sintitute Inc.) to plot GLM outputs. All other statistics were computed using the R programming language (R Core Team 2013) and in Microsoft Excel 2010. We estimated annual emissions of CO_2 and CH_4 by extrapolating hourly flux from each sampling day across the nearest adjacent unsampled days.

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 38 (Neubauer 2014) and estimate radiative forcing switchover time (Frolking et al 2006) for the CCW using net ecosystem exchange (NEE) data (Bailey 2006) and CH_4 flux data generated in this study. Bailey (2006) found NEE to be negative for most of the CCW plots because of rapid oxidation of added OM, therefore we used his positive mean NEE values from the lowest loading rates (141.1 and 29.9 g CO_2-C m^{-2} yr^{-1}) to generate a range of radiative potential radiative forcing switchover times.

Results

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 (see Fig. 3). Ponded water was present at the site 59 percent of the time from 22 February, 2012 to 21 January, 2013 and reached a maximum depth of 14 cm above the mean elevation of unammended 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 ($p < .05$) positive linear trend in elevation across the OM gradient.

Soil Nutrients

Total soil C data shows that while some of the added OM may have been lost since 2005 (Bailey et al 2007), particularly from plots loaded with 112 and 224 mg ha^{-1} OM, the gradient, as originally established, persists (see Fig. 4), with total C in the top 10 cm of soil ranging from approximately 2 to 13 percent. Mean 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} yr^{-1} to surface soils.

Total soil C, N and P are generally higher in plots that received higher loading rates of OM, but decreases with depth such that differences between loading rates are negligible at 10 and 20 cm depth (see Fig. 5A-5C). KCl-extractable NH_x and NO_x and Mehlich-3-extractable P follow roughly similar patterns, with some exceptions (see Fig. 5D-5F). KCl-extractable NH_x shows no clear pattern related to loading rate and KCl-extractable NO_x is uniformly low with as much variability with depths and loading rates as across them.

We observe a strong linear correlation (r^2=.96) between total soil C and N had throughout the top 20 cm of soil (see Fig. 6A). The relationship between total N and KCl-extractable NH_x is much weaker (r^2=.56; see Fig. 6C). Total soil C and P show a logistic correlation (r^2=.67; Fig. 6B), and total P and Mehlich-3-extractable P show a relatively weaker but significant (r^2=.58) quadratic correlation (see Fig. 6D).

GHG Fluxes

We analyze the three most-important GHGs (CO_2, CH_4 and N_2O). Because of the high spatial and temporal heterogeneity in N_2O flux (Firestone 1982; Groffman et al 2009), and the fact we that found N_2O flux to be below minimum detection thresholds for approximately 90 percent of incubations we focused our results and discussion on CH_4 and CO_2 flux.

CO_2 Flux

The highest CO_2 fluxes (>400 mg m^{-2} hr^{-1}) were observed during warmer, drier months and contrast with fluxes approaching minimum analytical detection limits during cold, wet months (Fig. 7). CO_2 emissions from soil directly responded to increases in soil T (Fig.8) 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 two months (the other being January) where significant differences in CO_2 flux between OM treatments occur. From summed monthly data we estimate an annual soil CO_2 flux ranging from 0.33 ± 0.019 to 0.71 ± .11 kg CO_2-C from the respective low to high end of the OM gradient.

A GLM with three parameters: soil T, soil volumetric water content (SVWC), and soil total C (top 5 cm), explains much of the variability (r^2 = 0.75) in CO_2 flux across all sampling dates (Fig. 9A). During any given sampling date soil T and soil moisture are essentially constant across plots (relative to seasonal changes) and cannot explain differences in soil respiration. For example, variability in July CO_2 flux could only be partially explained ($r^2 = 0.61$) by a GLM incorporating soil total C (top 5 cm) and soil total N (20 cm depth; Fig. 9B).

CH_4 Flux

We find CH_4 flux rates above minimum analytical detection thresholds only when soil T was at least 18 ˚C and some ponded water was present at the CCW (see Table 2). We identify a threshold of 50 percent SVWC, below which CH_4 was never greater than 0.13 mg CH_4 m^{-2} hr^{-1} (Fig. 10). When conditions at the CCW are favorable for methanogenesis (soil T > 15 ˚C and ponded water), flux rates are highly variable across plots. Maximum observed CH_4 flux rates are approximately 3 to 5 mg m^{-2} hr^{-1}. We estimate an annual efflux of 40.5 kg CH_4-C ha^{-1} yr^{-1} from our bi-monthly measurements (see Table 4). We are unable to detect any statistically significant patterns in CH_4 flux related to soil C or OM loading rate.

Carbon Balance

During the sampling dates when CH_4 flux was large enough to be detectable, its contribution to radiative forcing was relatively minor compared to soil CO_2 flux based on a 100-year sustained global warming potential of 38 for CH_4 (Neubauer 2014) (Fig. 11).

Discussion

Hydrology

Wetland GHG flux is moderated by hydrologic dynamics because saturation inhibits decomposition and creates conditions favorable for CH_4 emission (Whalen 2005). Thus it is important to consider site hydrology as we discuss gas flux. 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). The CCW was relatively wet during the 2012 growing season when it received 82 cm of rain (7.5 percent above mean; National Climatic Data Center; Lawrimore et al 2011) and held ponded water 52 percent of the time. This contrasts with conditions during the 2005 growing season when the CCW received 10 percent 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).

Elevation and OM incorporation

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). We found micro-elevational differences between plots 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 3), 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 discrepancy in total soil C between 2005 and 2012 we observed (see Fig. 4).

CO_2 flux

Our annual soil respiration budget cannot account for the soil C losses in the 112 and 224 Mg ha^{-1} plots we observe between 2005 and 2012, which were approximately 1 and 5 percent, respectively, corresponding to respective losses of 1.5 and 5.5 kg C m^{-2} yr^{-1} over seven years. This rate is an order of magnitude greater than our estimated annual soil respiration loss from these plots: 0.42 and 0.49 ± 0.032 kg CO_2-C m^{-2} yr^{-1}respectively. Therefore we suspect that some of the C loss may be due to leaching of dissolved OM and/or transport of particulate OM during floods. In calculating the annual budget we assume that CO_2 flux will be similar on average across a 2-month window to what we measure during our relatively short period of observations, which means that our calculations are susceptible to bias from idiosyncrasies of weather preceding each sampling date. Such effects could be especially pronounced during the fall and spring, which experience extreme within-season and inter-annual climate variability.

However, 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} hr^{-1} during summer dry spells and low CO_2 flux of less than 100 mg m^{-2} hr^{-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.

Soil respiration rate is typically limited by T and oxygen availability, so it is not surprising that soil T and SVWC are the two most important terms in our generalized linear model explaining log-transformed CO_2 flux variability across seasons, with r-squared values of .50 and .49 respectively. Soil T and SVWC are slightly correlated with each other (r-squared of -.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 .71. The third model parameter, total surficial soil C simply reflects the amount of OM available to be decomposed. The effects of OM 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 .05), including it in the GLM helps improve fit (r-squared of .75) and reduces the Akaike information criterion (AIC).

With T and soil moisture held relatively constant across the site during a given sampling date, we found surface soil C to be the most important parameter explaining CO_2 flux in July (r-squared of .52). The inclusion of total soil N at 20 cm depth improved our model r-squared to .63 and it was not highly correlated with surface soil C (r-squared of .24). 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).

CH_4 flux

Hydrology and T both control rates of methane 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 (See table 4).

CH_4 flux shows no significant relationship with OM loading rate, suggesting that if excess nutrients and enhanced primary productivity are increasing methane production, then the increase is being cancelled out by increased oxidation. This result contrasts with that of Ballantine et al. (in press) which shows that addition of OM led to higher rates of potential net methane emissions from intact soil cores compared to controls (Ballantine et al., in press). Higher soil moisture in amended plots correlate with Ballentine et al.'s observed differences in CH_4 production and they suggest that OM amendments increased water retention creating conditions more favorable for methanogenesis. At CCW, OM additions appeared to have the opposite effect on soil moisture because of the slight mounding effect described above. Our data from the relatively drier months of May and July show weak (r-squared of 0.16 and .014, respectively), marginally significant ($p < 0.09$ and $p < 0.11$, respectively) relationship between SVWC and OM loading rate.

The higher CH_4 production in response to added OM found by Ballantine et al. (in press) appears to be an indirect effect caused by increased soil moisture as there was no relationship between C quality and CH_4 flux among different types of added OM. Therefore 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, or conversely, increased OM may enhance water holding capacity facilitating anoxia (Ballantine et al., in press).

Carbon Balance

The radiative forcing switchover time (Frolking et al 2006) for CCW is highly uncertain because of high variability in NEE (Bailey 2006) and CH_4 flux data (this study). Furthermore in 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). Therefore it would take a long-term monitoring approach to improve certainty of radiative forcing switchover time for the CCW. Despite these shortcomings we may conclude that CCW has a relatively short radiative forcing switchover time due to its low CH_4 flux. The CO_2^{-}sequestration:CH_4-flux ratio of CCW ranges from to 96 to 20, corresponding to a radiative forcing switchover time range of 0 to approximately 200 years following Neubauer's (2014) model. CCW will likely become a net GHG sink more quickly than at least six out of eight wetlands analyzed by Neubauer (2014).

Conclusions

We found little evidence to suggest that added composted yard waste increases CH_4 or N_2O 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 global warming potential 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 we recommend that only moderate levels of OM need to be added to created wetlands. Adding more than ~160 Mg ha^{-1} does not improve soil geochemistry (Bruland and Richardson 2009) and excess OM simply decomposes while adding little in the way of tangible productivity increases, not to mention incurring greater material transport and associated construction costs.

Acknowledgments

We appreciate the help of Jonathan Bills, who provided critical assistance with site instrumentation and field sampling as well as sample processing and analysis. Wes Willis also contributed greatly to the water and soil processing effort. Ashley Helton and Emily Bernhardt gave valuable advice and training in static chamber design and gas sampling and Todd Smith assisted with chamber construction. Jim Perry, Lee Daniels, David Bailey and Leo Snead provided valuable insight into the site history and scope of previous investigations. Darmawan Prasadojo assisted with programming an Excel Visual Basic macro designed to wrangle data output from the gas chromatograph. We thank Ann and Jay Kinney for graciously allowing us to use of their driveway to access the site. This work was supported by a grant from the Peterson Family Foundation and the Duke Wetland Center Endowment.

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Tables

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. Values that meet $p < 0.05$ are bolded

Month	linear regression		ANOVA
	p-value	r-squared	p-value
May	**0.028**	0.24	0.165
July	**<0.001**	0.55	**0.018**
Sept.	0.133	0.12	0.116
Nov.	**0.009**	0.40	0.126
Jan.	**0.043**	0.21	**0.003**

Table 2. Summary of monthly averages (±SD) soil temperature (at 5 cm depth), hydrology and soil carbon emissions from the Charles City Wetland in Charles City County, Virginia. All data collected in 2012 except for January, 2013

Month	Soil Temp.	Water Level	Soil volumetric water content	CH_4 emissions	CO_2 emissions
	˚C	cm	%	$mg \cdot m^{-2} \cdot hr^{-1}$	
May	18.1±0.7	5.3±2.8	65.6±7.5	0.82±0.88	117±94
July	24.6±0.3	-39.9±28.6	29.9±11.5	0.02±0.04	595±191
Sept.	20.0±1.5	6.0±2.3	60.2±6.4	1.29±1.41	188±130
Nov.	10.1±0.4	3.0±2.3	55.5±6.7	0.02±0.04	124±63
Jan.	5.9±0.8	9.2±2.8	60.5±6.2	0	32±27

Table 3. Review of methane (CH_4) emissions rates in kg CH_4-C ha^{-1} yr^{-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)
41	**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)

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. 2007) and 2012 (this study)

Year	Rainfall (Apr. - Oct.; cm)*		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

*Source: National Climatic Data Center (Lawrimore et al. 2011)

Figure captions

Fig. 1 Location of Charles City Wetland in Charles City County, Virginia, USA, with A) showing the geographic location of Charles City County, Virginia, B) showing the siting of the experimental block within the wetland, and C) indicating the arrangement of plots, treatments and wells (labeled a through e) within the block, and site dimensions

Fig. 2 Illustrations of the reduced-disturbance static chamber design: A) Photograph of chamber being deployed in the Charles City Wetland in Charles City County, Virginia; B) schematic of chamber disassembled to reveal water fillable gutter on rim of collar that creates an air tight seal, internal fan to mix headspace air and thermocouple to monitor internal chamber temperature; C) schematic of chamber assembled

Fig. 3 Water level as recorded by five 1.5 meter Odyssey loggers (Dataflow Systems, Christchurch, New Zealand) placed in water level 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 is from a station in nearby James City County, Virginia (National Climate Data Center)

Fig. 4 Linear regressions of mean (±SE) total carbon in top 10 cm of soil across organic matter amendment plots at the Charles City Wetland in Charles City County, Virginia, USA. 2005 data from Bailey *et al* (2007.)

Fig. 5 Depth profiles of mean (±SE): A) percent soil carbon by mass, B) percent soil nitrogen by mass, C) total soil phosphorus by mass in µg g^{-1}, D) extractable ammonia/ammonium in µg NH_x-N g^{-1} dry soil, E) extractable nitrate/nitrite in µg NO_x-N g^{-1} dry soil, F) extractable phosphorus in µg NO_x-N g^{-1} dry soil. Different dash patterns represent loading rates of organic matter in Mg ha^{-1} added to the Charles City Wetland in Charles City County, Virginia, USA

Fig. 6 Relationships between total soil elemental content and extractable nutrients by depth, as indicated by shapes, at the Charles City Wetland in Charles City County, Virginia, USA. A) shows a linear relationship between percent total soil carbon by mass and percent total soil nitrogen by mass, B) a logarithmic relationship between percent total soil carbon by mass and total soil phosphorus by mass in ppm, C) a linear relationship between percent total nitrogen of soil by mass and extractable ammonia/ammonium in µg NH_x-N g^{-1} dry soil, D) a quadratic relationship between total soil phosphorus by mass in µg.g^{-1} and extractable phosphorus in µg g^{-1} dry soil

Fig. 7 Mean carbon dioxide flux as a function of soil temperature at 5 cm depth from the organic matter 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 ± 1 standard deviation

Fig. 8 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 Mg ha^{-1}

Fig. 9 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 percent confidence intervals for the regression line. Dashed horizontal line indicates mean carbon dioxide flux value

Fig. 10 Methane flux (CH_4) rates 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

Fig. 11 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 38—its 100-year sustained global warming potential following Neubauer (2014). Error bars represent standard errors of the mean

Figures (color) for Peterson Report

Fig. 1 (174 mm)

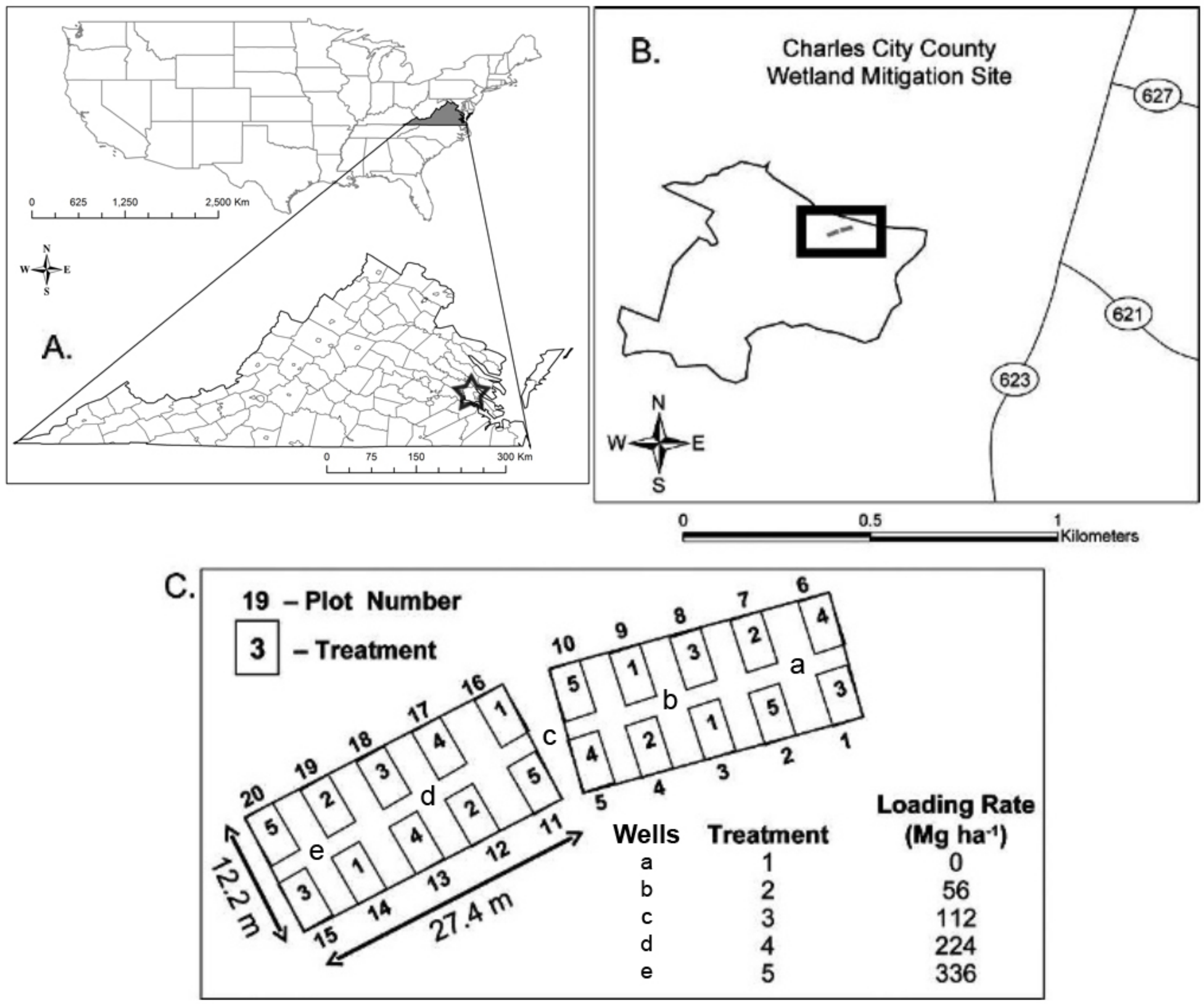
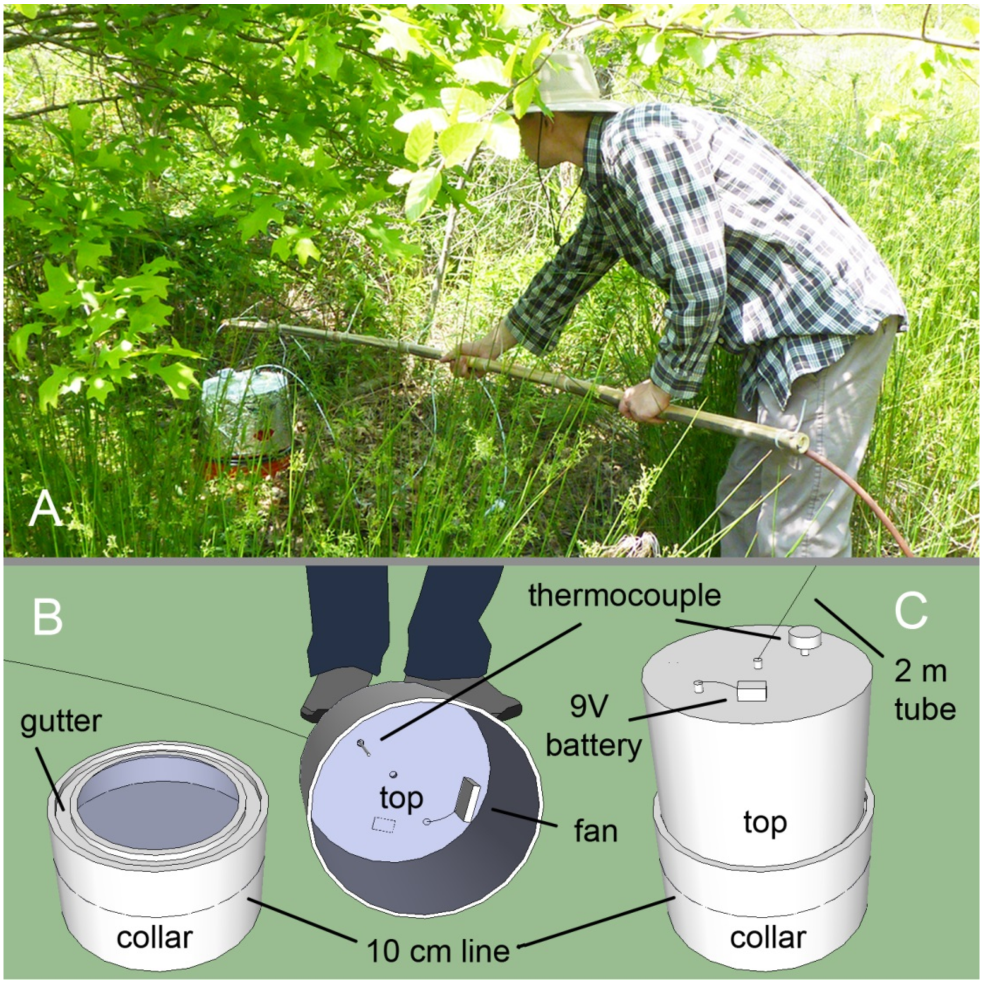


Fig. 2 (129 mm wide)



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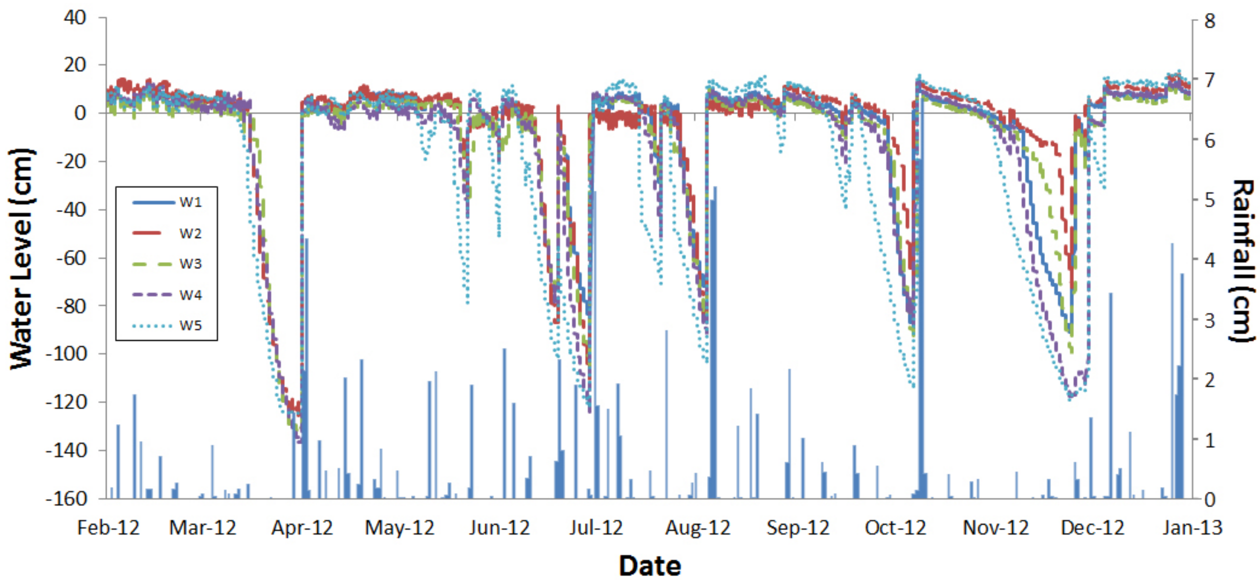

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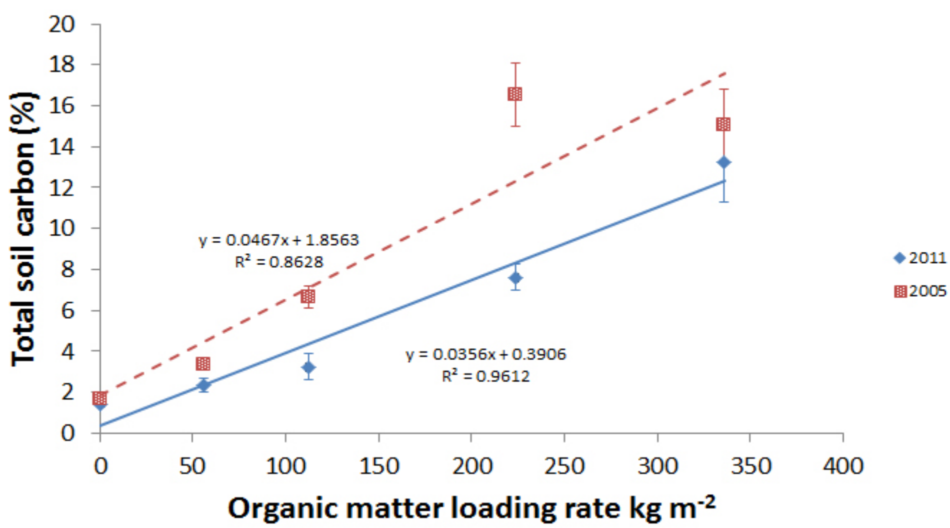

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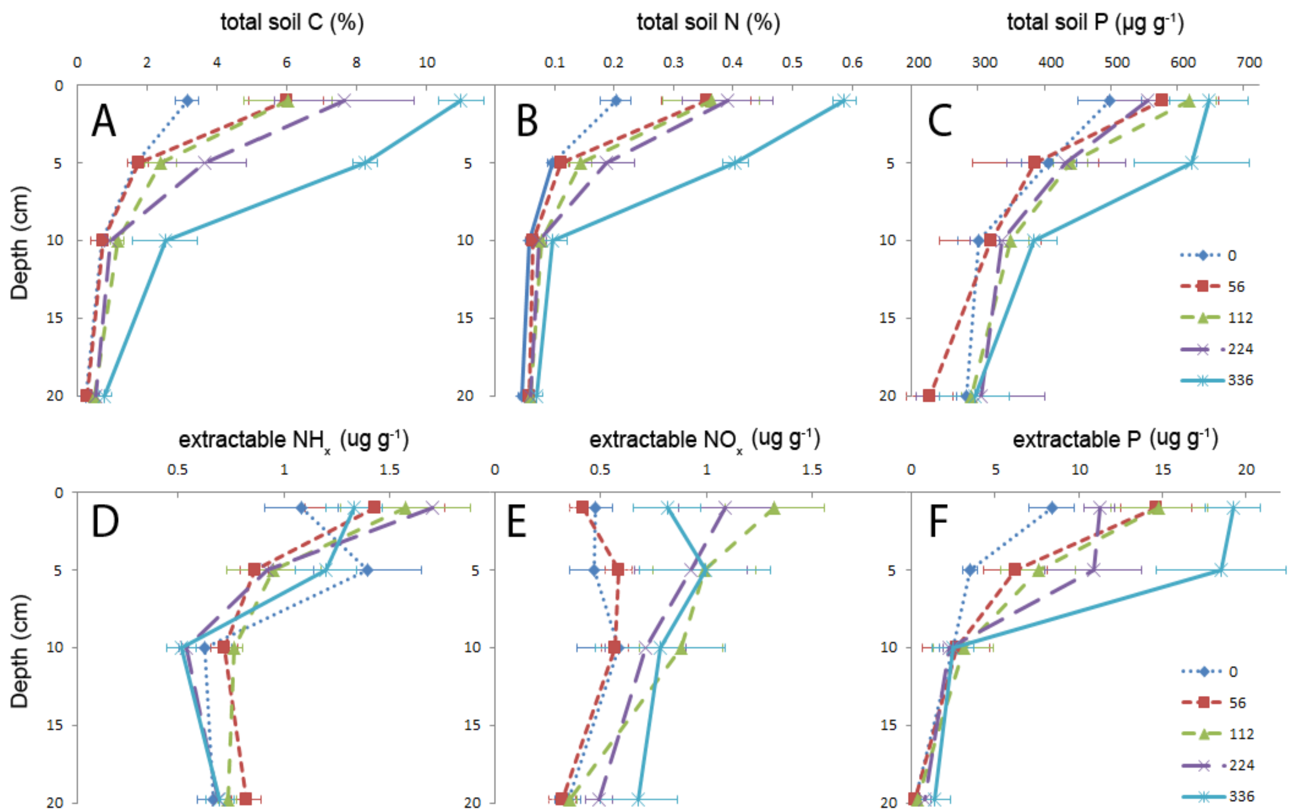
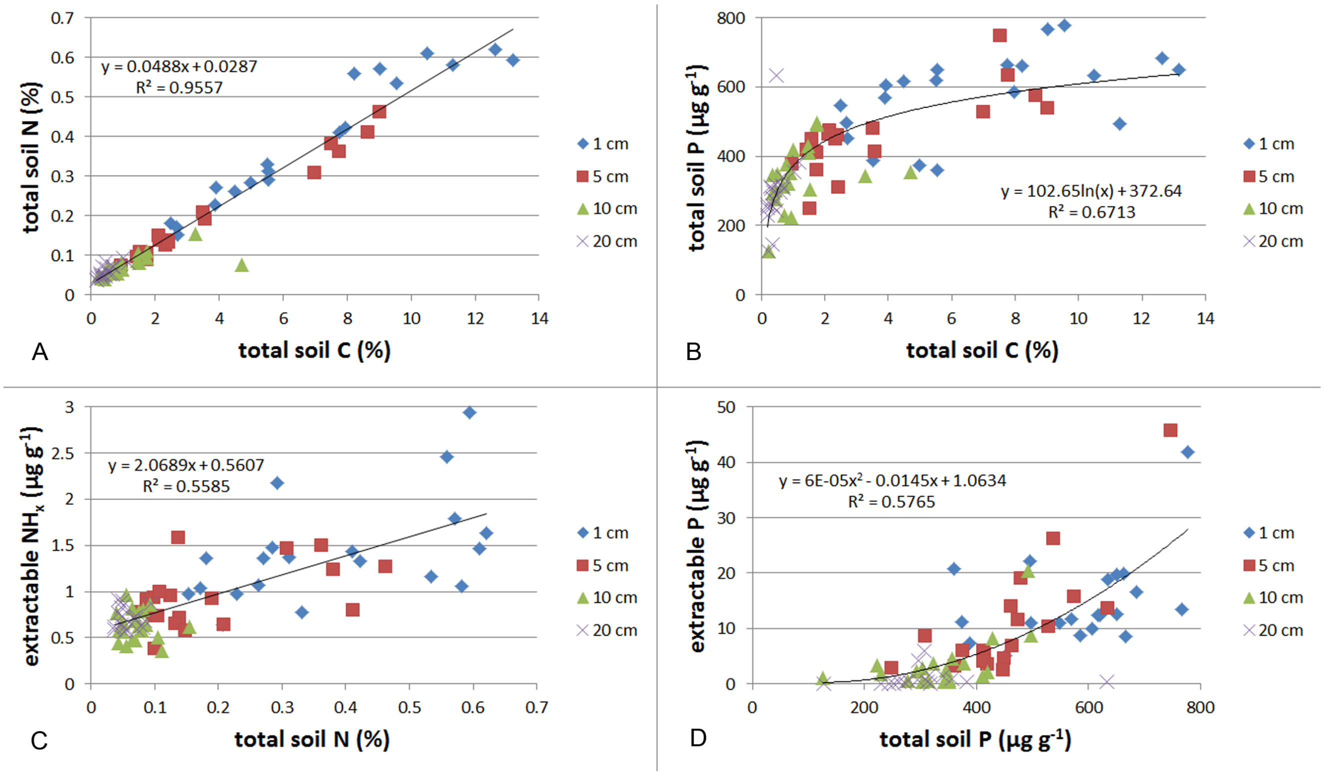


total soil C (%)
total soil N (%)
total soil P (µg g-1)
extractable NHx (ug g-1)
extractable NOx (ug g-1)
extractable P (ug g-1)
Depth (cm)
A
B
C
D
E
F
0
56
112
224
336

Fig. 6 (174 mm)



y = 0.0488x + 0.0287
R² = 0.9557
total soil N (%)
total soil C (%)
A
y = 102.65ln(x) + 372.64
R² = 0.6713
total soil P (µg g-1)
total soil C (%)
B
y = 2.0689x + 0.5607
R² = 0.5585
extractable NHx (µg g-1)
total soil N (%)
C
y = 6E-05x² - 0.0145x + 1.0634
R² = 0.5765
extractable P (µg g-1)
total soil P (µg g-1)
D
1 cm
5 cm
10 cm
20 cm

Fig. 7 (129 mm)

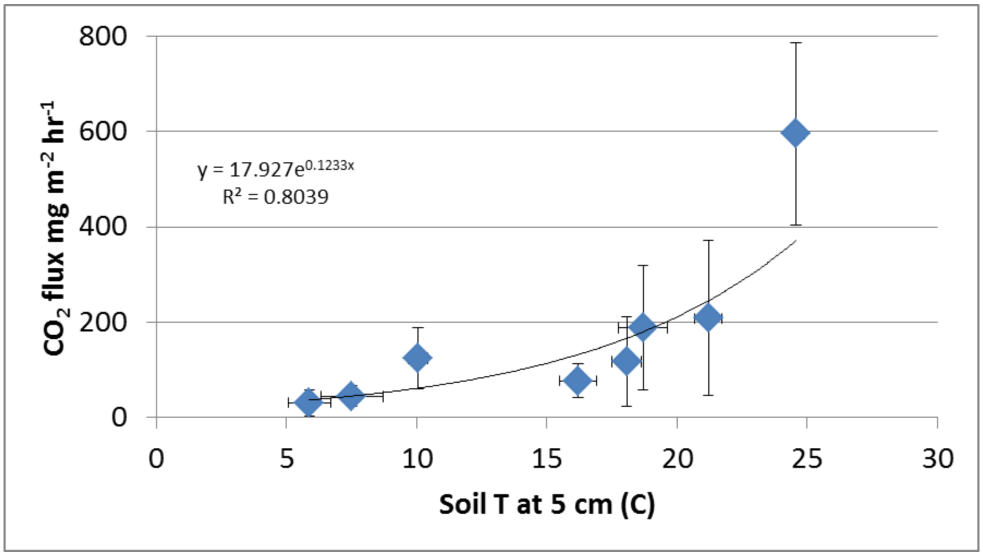


Fig. 8 (129 mm)

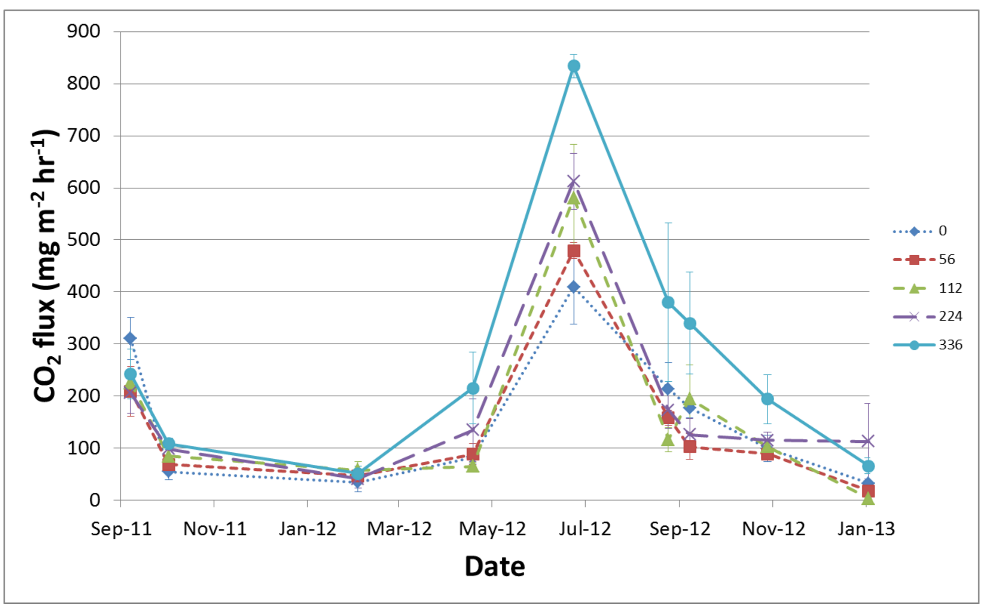


Fig. 9 (174 mm)

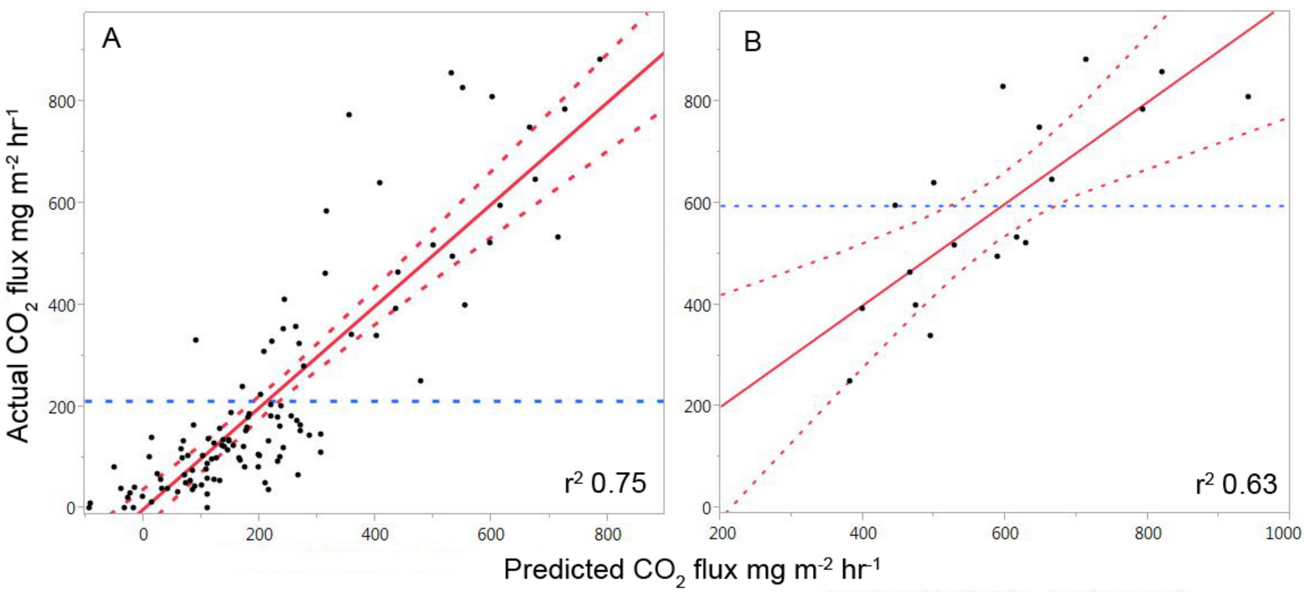

A
B
r² 0.75
r² 0.63
Actual CO₂ flux mg m⁻² hr⁻¹
Predicted CO₂ flux mg m⁻² hr⁻¹
0
200
400
600
800
1000

Fig. 10 (129 mm)

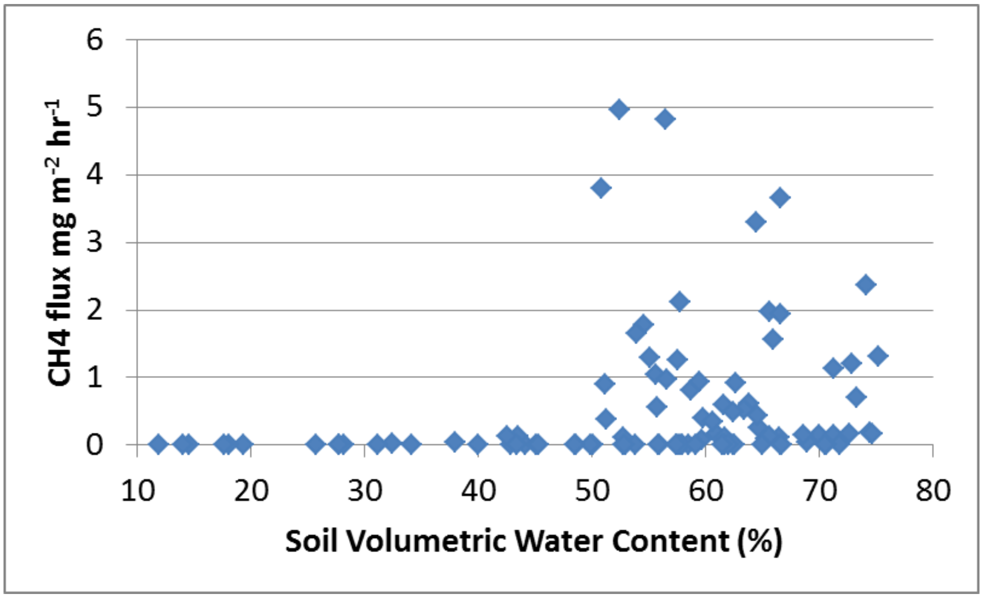


Fig. 11 (174 mm)