Carbon Sequestration in Two Created Riverine Wetlands in the Midwestern United States

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Wetlands have the ability to accumulate significant amounts of carbon (C) and thus could provide an effective approach to mitigate greenhouse gas accumulation in the atmosphere. Wetland hydrology, age, and management can affect primary productivity, decomposition, and ultimately C sequestration in riverine wetlands, but these aspects of wetland biogeochemistry have not been adequately investigated, especially in created wetlands. In this study we investigate the ability of created freshwater wetlands to sequester C by determining the sediment accretion and soil C accumulation of two 15-yr-old created wetlands in central Ohio—one planted and one naturally colonized. We measured the amount of sediment and soil C accumulated over the parent material and found that these created wetlands accumulated an average of 242 g C m^{-2} yr^{-1}, 70% more than a similar natural wetland in the region and 26% more than the rate estimated for these same wetlands 5 yr before this study. The C sequestration of the naturally colonized wetland was 22% higher than that of the planted wetland (267 ± 17 vs. 219 ± 15 g C m^{-2} yr^{-1}, respectively). Soil C accrual accounted for 66% of the aboveground net primary productivity on average. Open water communities had the highest C accumulation rates in both wetlands. This study shows that created wetlands can be natural, cost-effective tools to sequester C to mitigate the effect of greenhouse gas emissions.

The increased rate of wetland loss in the last century and the recognition of wetland values have led to a "no net loss" policy in the United States (Dahl, 2000; NRC, 2001; Mitsch and Gosselink, 2007), whereby wetlands are often created to replace those lost or damaged. There are concerns regarding the ability of created wetlands to provide the same functions as natural wetlands due to the time it can take the wetland to develop (e.g., hydric soil and vegetation cover) and for ecosystem services to become established (e.g., water quality improvement and provision of wildlife haven). Wetland creation and restoration have been thoroughly evaluated over the years to assess the success of the new wetlands in their ability to replace functional natural wetlands and to assess the accomplishment of the no net wetland loss policy (Mitsch and Wilson, 1996; Zedler and Calloway, 1999; Kentula, 2000; NRC, 2001; Campbell et al., 2002; Gutrich et al., 2009). Through the failure of some of these mitigation projects (Erwin, 1991; Spieles, 2005; Matthews and Endress, 2008), we have learned that the typical 5-yr jurisdictional monitoring period might not be long enough for a wetland to achieve adequate structure and function. This is particularly true for wetland functions such as soil organic matter accumulation, which depends greatly in the successful establishment of vegetation and hydrology (Campbell et al., 2002; Bruland and Richardson, 2005; Fennessy et al., 2008; Hossler and Bouchard, 2010). The development of hydric soils, waterlogged conditions, and high productivity allow created wetlands to sequester carbon (C) as effectively as natural wetlands.

The use of natural systems to accumulate C is one of the most cost-effective tools to reduce the net effect of greenhouse gas emissions and abate climate change (Hanley and Spash, 2003; IPCC, 2005; Bedard-Haughn et al., 2006; Stern, 2007; Lal, 2008). Wetlands are known to be significant C sinks; their high productivity introduces large amounts of organic matter into the soil, but the semipermanent presence of water slows its decomposition (Collins and Kuehl, 2001; Mitsch and Gosselink, 2007). Even though anaerobic decomposition under the presence of water produces methane (CH_4) (making wetlands accountable for about 25% of the yearly emissions, up to 85% of which is estimated to come from rice paddies and tropical wetlands) (Bartlett and Harris, 1993; Neue, 1993; Schrope et al., 1999; Melack et al., 2004; Whalen, 2005; IPCC, 2007), wetlands' soil C stock represents one third to one half of the organic terrestrial

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C pool (Mitra et al., 2005; Bridgham et al., 2006; Mitsch and Gosselink, 2007; Lal, 2008). These characteristics allow wetlands to sequester C, and thus policymakers have considered creating wetlands for C capture and sequestration to counteract the increasing levels of greenhouse gases in the atmosphere (IPCC, 2007; Schrag, 2007; Stern, 2007; Badiou et al., 2011). This would entail that wetlands could be created to serve a specific purpose (in this case, to sequester C) rather than to solely compensate for the loss of a similar ecosystem. To get to this point, much research is needed for a full understanding of the wetland conditions that enhance C sequestration while keeping CH_4 emissions low.

Factors that Can Favor Carbon Accumulation in Wetland Soils

Although any wetland ecosystem can accumulate significant amounts of C in its soil, C sequestration in wetlands can be optimized given the appropriate hydrology and vegetative community. In created wetlands, this state is reached at different ages depending on the initial wetland design. Once it is achieved, the two ways to enhance its soil C pool are by (i) increasing C inputs and (ii) decreasing the C outputs. Enhancing wetland productivity can potentially increase the amount of organic matter introduced into the soil and thus enhance C sequestration. Freshwater wetlands with mineral soil are significantly more productive than most peatlands (Mitsch and Gosselink, 2007), potentially introducing more C into the soil and accumulating it at a faster rate than peat-forming wetlands (Badiou et al., 2011). Flow-through and pulsing wetlands are typically more productive because they receive additional nutrient and organic matter inputs from a connected water body (Mitsch, 1988; Mitsch and Reeder, 1991; Odum et al., 1995; Cronk and Fennessy, 2001). They retain and transform these inputs as they flow through, removing them from the water column and improving water quality.

Wetland hydrogeochemistry is also a key factor affecting C accumulation in the soil. There is usually a gradient of inundation frequency, from the deepest and typically permanently flooded area of the wetland to the shallower and semipermanently flooded edges (Mitsch and Gosselink, 2007). This gradient determines the vegetation communities being established (open water, algae mats, and floating plants typically dominate the deepest areas of the wetland), whereas emergent macrophytes develop in the intermediate and shallower zones (Boutin and Keddy, 1993; Cronk and Fennessy, 2001; Mitsch and Gosselink, 2007). Edge vegetation communities tend to be more productive and densely vegetated because they are more affected by hydrological pulses (Odum, 1969; Odum et al., 1995) and less limited by the permanent presence of water. Thus, these areas are potentially introducing larger amounts of organic matter into the soil than open water and floating communities. On the other hand, deeper sites have lower organic matter decomposition rates due to the permanent anaerobic conditions of the soil, whereas edge areas experience strong respiration pulses when the soil is dried and rewetted (Stevenson and Cole, 1999; Jassal et al., 2005; Miller et al., 2005), potentially decreasing the C stock in the soil. It is therefore unclear which communities have greater soil C accumulation. Bernal and Mitsch (2012) found that, in a temperate riverine wetland, C sequestration rates were higher in the floating communities of the deep water sites than in the emergent and edge communities. However, C accumulation studies in wetlands typically do not differentiate between wetland communities because the differences are usually not significant (Bruland and Richardson, 2005; Gutrich et al., 2009; Hossler and Bouchard, 2010), and traditionally the concern has been the effect of nutrient gradients on soil properties rather than vegetation gradients (Reddy et al., 1993; Craft and Richardson, 1993; Mack et al., 2004; Stern et al., 2007).

The goals of this study were (i) to determine the soil C sequestration rate of two created riverine wetlands and (ii) to investigate the role that factors such as wetland age, aboveground productivity, and wetland vegetation community may have in the C accumulation capacity of these wetlands. Anderson et al. (2005) and Anderson and Mitsch (2006) measured C accumulation rates in these wetlands when they were 10 yr old. We predict that, 15 yr after wetland creation, these wetlands are increasing their soil C pool at a rate greater than that reported 5 yr earlier in these wetlands due to an increase in hydric soil thickness and standing live and dead biomass cover. In addition, we predict higher sequestration rates in the wetland basin that was naturally colonized than in the planted wetland basin because the naturally colonized basin has been more productive over these 15 yr than the planted basin (Mitsch et al., 2012). We predict higher sequestration rates in the open water communities of both of these wetlands, as was found in a similar natural riverine wetland in the same region by Bernal and Mitsch (2012).

Materials and Methods

Site Description

This research was conducted at the Olentangy River Wetland Research Park (40°01′ N, 82°01′ W), where two symmetrical 1-ha wetlands were created in 1994 adjacent to the Olentangy River (Fig. 1). Water from the river has been pumped through these wetlands since their creation in 1994, with pumping following the water level of the river to mimic the hydrologic pulses that a natural riverine wetland in that location would have experienced (Mitsch et al., 1998, 2005a, 2005b, 2012). These wetlands receive the same amount of water, experience the same pulsing hydrology, and have similar water retention times. Hydric wetland soils developed over the nonhydric parent material (Ross silt loams; NRCS [2010]) within a few years of flooding (Mitsch et al., 2005a, 2012), and sediment accretion in the two wetland basins has been occurring ever since (Harter and Mitsch, 2003; Anderson et al., 2005; Nahlik and Mitsch, 2008). These wetlands are diversely vegetated with wetland plants (*OBL* and *FACW* categories according to Reed [1988]). The western basin was planted in 1994 with 13 native species of macrophytes, whereas the eastern basin was left unplanted to be colonized naturally (Mitsch et al., 1998, 2005a, 2005b, 2012). This is the only design difference between the two wetlands. Over time the unplanted wetland, which was rapidly colonized by *Typha* spp., has had cumulatively higher aboveground biomass and net primary productivity but lower vegetative diversity compared with the planted wetland (Mitsch et al., 2012). Despite the initial planting, both wetlands have essentially converged in species richness after 15 yr (101 species in the planted wetland and 97 species in the naturally colonized wetland), with a total of 22 vegetation communities (including open water and algal mats) inventoried over that time period (Mitsch et al., 2012). Water quality has also been measured in these wetlands for 15 yr in the inflow and outflow of the basins (Mitsch et al., 1998, 2005a, 2005b, 2012). These analyses, coupled with detailed hydrology budgets, reveal that both wetlands functioned as surface water sinks for nutrients such as nitrogen and phosphorus (i.e., inflow concentrations and mass) were, on an annual basis during these 15 yr, almost always greater than outflow concentrations and mass. On the other hand, dissolved C concentration remained somewhat similar between wetland inflow and outflow, but because outflow discharge was generally less than inflow in these created wetlands (see Batson et al., 2012), they are probably retaining dissolved C as well.

Soil Sampling and Samples Preparation

A total of 44 soil cores (22 per wetland) were extracted in May 2009, 15 yr after the wetlands were created, following a 10-m grid spatial pattern that covers both wetlands entirely (Fig. 1). This same sampling design (10 m × 10 m grid) was used in 1993 (before the wetlands were flooded), 1 yr after flooding (1995), and 10 yr after flooding (2004) (Anderson et al., 2005; Anderson and Mitsch, 2006). By maintaining a consistent sampling, it is possible to compare results from this study with earlier assessments (Anderson et al., 2005; Anderson and Mitsch, 2006) of C stocks at the wetlands and to compare our 15-yr rates with their 10-yr rates. The cores were 7 cm in diameter, and their length varied depending on the depth of the sediment accumulated over the underlying nonhydric soil (10–35 cm). Extracted cores were immediately divided in the field into 5-cm increments and packed in sealed containers that were stored at 4°C until analysis. Sampling points in both wetlands were evenly distributed in the inflow, middle, and outflow sections (seven, eight, and seven cores, respectively). In every one of these sections, there are shallow areas that are not flooded when the water level drops and deeper areas that are permanently flooded. Permanently flooded areas have zones with and without vegetation. As a result, there are three distinct generic wetland communities in each wetland: (i) open water (deeper area of the wetland where water level is higher, soils are permanently flooded, and no emergent macrophytes grow), (ii) emergent macrophyte community (area where the soil is permanently flooded yet shallow enough for emergent wetland plants to grow), and (iii) edge community (where wetland macrophytes grow but flooding is intermittent). Soil sampling points were evenly distributed within these three communities throughout both wetlands. As a reference nonwetland site, two extra cores (15 cm deep and 10 cm in diameter) were collected in a forested upland area between the created wetland basins following the core method of Grossman and Reinsch (2002) and Tan (2005). All soil samples were oven dried until constant weight was reached (60°C for the wetland soils and 105°C for the upland soils, according to the standard methods of soil analysis (Grossman and Reinsch, 2002; Bernal and Mitsch, 2012), weighed to determine bulk density (Mg m^{-3}), ground to a 2-mm particle size, and homogenized.

Soil Carbon Analysis

Total C content of the soil samples was determined by combusting triplicate soil samples (150 mg each) in a Total Carbon Analyzer with solid sample module (TOC-V series, SSM-5000A, Shimadzu Corp.) at 900°C. To determine inorganic C content triplicate soil samples (150 mg each) were digested by 10 mol L^{-1} H_3PO_4 at 200°C, also in the Total Carbon Analyzer instrument. Organic C was determined as the difference between the two. The soil C concentration (g C kg^{-1}) and pool (kg C m^{-2}) of each core were calculated following the equations described in Bernal and Mitsch (2008, 2012).

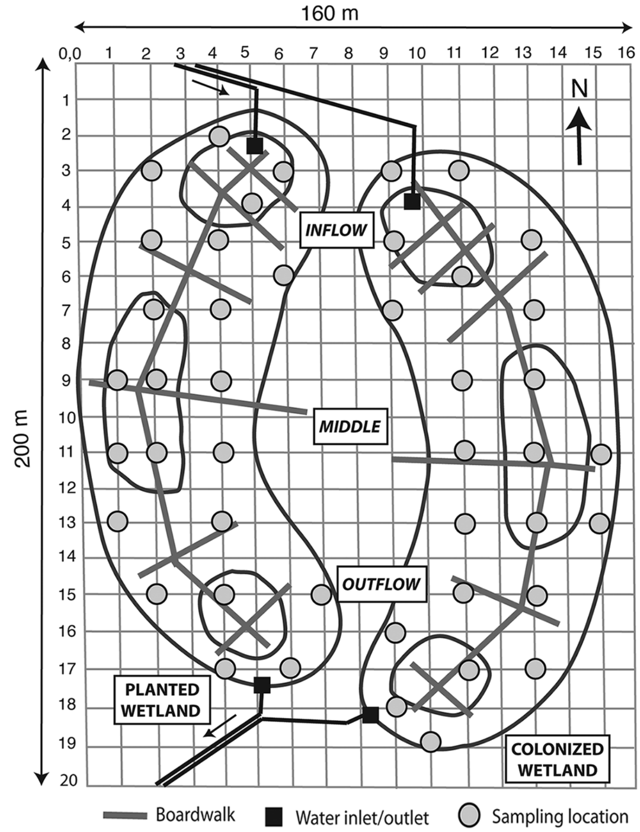

Fig. 1. Schematic of the two created wetlands at the Olentangy River Wetland Research Park showing the soil sampling locations within the 10-m grid, water inflows and outflows, direction of flow, and boardwalks. The wetland sections (inflow, middle, and outflow) and the three major open water zones (enclosed areas within each wetland) are also indicated.

The C and sediment accumulation since wetland creation in 1994 was calculated by estimating the total soil C and sediment pool of the hydric layer (i.e., from the soil surface to the underlying nonwetland soil over which the wetland soil was developed) in each point of the grid. The accumulation rates were determined by dividing the pools by the age of the wetland at the time of sampling (15 yr). Similarly, sedimentation rates were computed by measuring the amount of hydric soil accumulated over the nonhydric parent material and then divided by the 15 yr since the wetlands were created. Given the age of these wetlands, other common methods to estimate sequestration rates in natural wetlands, such as ^{137}Cs (Craft and Richardson, 1993; Bernal and Mitsch, 2012), could not be used here. Rates were estimated for each community, section, and wetland; total wetland averages were weighted based on the surface area of each community. Between 2004 and 2009 (10–15 yr after creation), the planted wetland averaged 35% open water and 65% vegetated area (emergent and edge communities), whereas the naturally colonized wetland had 30% open water and 70% vegetated area. To estimate the C sequestration rates in the upland area, we compared the upland soil C pool before the wetlands were created (in 1993, from Anderson et al. [2005]) with our measurements 15 yr after wetland creation (2009).

Aboveground Net Primary Productivity

Details on procedures for water quality measurements and aboveground net primary productivity (ANPP) determinations in these two wetlands since they were created can be found in Mitsch et al. (2005a, 2005b, 2012). Fifteen years of ANPP data were used for comparisons with our soil physicochemical analysis.

Statistical Analysis

Statistical analyses were performed with IBM SPSS Statistics version 19.0 for Macintosh (SPSS Inc.). For every data set, we conducted an exploratory data analysis and normality checks (with *Q-Q* plot, Kolmogorov–Smirnov test, and Shapiro–Wilk test) to ensure that they fit the normal distribution and tested the homogeneity of variances using the Levene Statistic. A statistical ANOVA with Tukey HSD was used to test the effect of independent variables (section and community) on the C pools as well as the C sequestration rates (dependent variables). This test was also used to determine significant differences in C sequestration between the two wetlands. Soil C sequestration and net primary productivity were examined with Pearson correlations to explore the relationship between aboveground biomass and soil C sequestration (Fowler et al., 2003).

Results

Wetland Soil Carbon Content

Average total soil C content (g C kg^{-1}) was significantly higher by 30% ($P < 0.01$) in the naturally colonized wetland compared with the planted wetland (Table 1). The naturally colonizing wetland had a 25% greater soil C pool than the planted wetland basin (4.0 ± 0.3 and 3.2 ± 0.2 kg C m^{-2}, respectively; $P = 0.02$) (Table 1). The lowest C pool was found in emergent communities (3.2 ± 0.3 kg C m^{-2}), and the highest was found in open water sites (4.0 ± 0.3 kg C m^{-2}; $P \leq 0.05$) (Table 1).

Total soil C was not significantly different when comparing the three wetland communities or the vegetated and nonvegetated areas. The highest soil C content was found in the edge community of the naturally colonizing wetland (46.5 ± 5.6 g C kg^{-1}); soil C in the emergent and open water communities was about 85% of that amount (Table 1).

Most of the C accumulated in the soil was organic (90% organic, on average, in the planted basin and 93% in the naturally colonized basin), but the highest inorganic C content was found in the open water areas (19% in both wetlands) and the lowest in the vegetated areas of the naturally colonized wetland basin (<1%) (Table 1).

Sedimentation Rates

With similar bulk densities and hydric layer depth, both wetlands yielded almost the same estimated sedimentation rates (6.0 ± 0.4 kg m^{-2} yr^{-1} in the planted wetland; 5.9 ± 0.4 kg m^{-2} yr^{-1} in the naturally colonized wetland) (Table 1). The open water sites had greater sedimentation rate than the vegetated areas (6.8 ± 0.5 vs. 5.5 ± 0.3 kg m^{-2} yr^{-1}, respectively) even though their soil was the least dense (0.59 ± 0.04 vs. 0.70 ± 0.03 Mg m^{-3} in the open water sites and the vegetated areas, respectively). The edge communities had the highest bulk density and the least accretion depth of the three wetland communities (Table 1).

Carbon Sequestration

There were significant differences in C sequestration rates between the two wetlands (267 ± 17 g C m^{-2} yr^{-1} in the naturally colonized wetland and 219 ± 15 g C m^{-2} yr^{-1} in the planted wetland; $P = 0.02$) (Table 1). However, C sequestration in the inflow of the planted wetland (201 ± 29 g C m^{-2} yr^{-1}) was 31% lower ($P = 0.04$) than C sequestration in the naturally colonizing wetland (291 ± 26 g C m^{-2} yr^{-1}).

When comparing wetland communities, C sequestration was consistently higher in both wetlands in open deepwater communities than in the shallower emergent vegetation communities (267 ± 21 vs. 212 ± 20 g C m^{-2} yr^{-1}, respectively). This difference was significant only at the 85% confidence level ($P = 0.12$). No differences were found between edge

Table 1. Physiochemical conditions, pools, and rates of the planted and unplanted (naturally colonized) wetlands at the Olentangy River wetland Research Park 15 yr after creation and of their three wetland communities.

	Planted† wetland ($n = 17$)	Unplanted† wetland ($n = 19$)	Wetland community		
			Open water ($n = 12$)	Emergent ($n = 15$)	Edge ($n = 9$)
Physiochemistry					
Bulk density, Mg m^{-3}	0.63 ± 0.02‡	0.71 ± 0.04	0.59 ± 0.04	0.67 ± 0.03	0.75 ± 0.06
Hydric soil depth, cm	14.6 ± 1.3	13.2 ± 1.2	17.9 ± 1.7	12.0 ± 0.8	11.7 ± 0.8
Total C content, g C kg^{-1}	36.1 ± 1.6*	46.5 ± 2.8*	39.6 ± 1.6	39.9 ± 2.8	46.5 ± 5.6
Inorganic C content, g C kg^{-1}	3.6 ± 1.2	3.1 ± 1.1	7.7 ± 1.6*	1.0 ± 0.5	1.6 ± 1.0
Organic C:Total C ratio	0.90 ± 0.03	0.93 ± 0.01	0.81 ± 0.04	0.97 ± 0.02	0.97 ± 0.01
Rates and pools					
Soil C pool, kg C m^{-2}§	3.2 ± 0*	4.0 ± 0.3*	4.0 ± 0.3*	3.2 ± 0.3*	3.8 ± 0.3*
Sedimentation rate, kg m^{-2} yr^{-1}	6.0 ± 0.4	5.9 ± 0.4	6.8 ± 0.5	5.3 ± 0.3	5.9 ± 0.6
C accumulation rate, g m^{-2} yr^{-1}	219 ± 15*	267 ± 17*	267 ± 21a¶	212 ± 20b	255 ± 19ab

* Significant at the 0.05 probability level.

† Weighted averages based on corresponding surface area of open water and vegetated communities (i.e., emergent and edge).

‡ Values are average ± SE.

§ Soil C pools calculated for the average hydric soil depth in each wetland.

¶ Values followed by different lowercase letters are significant at $P < 0.15$.

Table 2. Comparison of physiochemical conditions at 10 and 15 yr after the wetlands were created (average of both wetlands), including the percentage change in the conditions between both periods, the values for the reference natural wetland, and the values for the reference upland adjacent to the created wetlands.

Created wetlands	Bulk density	Soil accretion	Total carbon content	Carbon pool†	Carbon accumulation rate	Reference
	Mg m⁻³	cm	g C kg⁻¹	kg C m⁻²	g C m⁻² yr⁻¹	
10 yr since creation (1994–2004)	0.5	9	38.5	2.3	190	Anderson and Mitsch, 2006
15 yr since creation (1994–2009)	0.7	14	41.3	3.7	242	this study
% Change (2004–2009)	+37%	+55%	+7%	+76%	+26%	this study
Reference natural wetland	0.8	–	50.1	1.5	140	Bernal and Mitsch, 2012
Reference upland	1.2	–	24.7	1.5‡	99	this study

† Carbon pool to average hydric soil depth in created wetlands, to 35 cm in reference wetland, and to 15 cm in the reference upland.

‡ Increase in upland soil C pool in 15 yr (1994– 2009).

communities (255 ± 19 g C m^{-2} yr^{-1}) and these other two communities (Table 1).

No hydric conditions developed on the upland soil adjacent to the wetlands, and its bulk density remained high (1.22 ± 0.08 Mg m^{-3}) (Table 2), almost twice that of the average density in both wetlands (0.67 ± 0.02 Mg m^{-3}). Carbon content was 67% higher in the new wetland soils compared with the new upland soils (41.3 ± 2.2 vs. 24.7 ± 4.1 g C kg^{-1}, respectively). In 15 yr the upland sites have increased their total soil C pool by 1.5 kg C m^{-2}, yielding a C sequestration rate of 99 g C m^{-2} yr^{-1} (Table 2). The average wetland C sequestration rate of 243 g C m^{-2} yr^{-1} is 2.5 times this upland soil C sequestration rate; in other words, creating wetlands on this floodplain led to a net increase of C sequestration of 144 g C m^{-2} yr^{-1}, a 145% increase in the C sequestration over what would have happened if no wetlands had been created.

Primary Productivity and Carbon Sequestration

The ANPP of the naturally colonized wetland overall has been higher than in the planted basin (394 ± 29 and 342 ± 22 g C m^{-2} yr^{-1}, respectively), with a significant increase in ANPP from the inflow to the outflow in the planted wetland basin ($P = 0.11$; $R^2 = 0.89$) and a significant decrease from inflow to outflow in the colonized basin ($P = 0.01$; $R^2 = 0.99$). Carbon sequestration follows the same trends, with $P = 0.07$ and $R^2 = 0.95$ in the planted wetland and with $P = 0.09$ and $R^2 = 0.92$ in the wetland that was not planted (Fig. 2).

Discussion

Soil Carbon Sequestration and the Effects of Planting

In one sense, we considered these two experimental wetlands as replicates because they were created identically and have the same forcing functions (i.e., similar hydrology, nutrient loads, and climate). In another sense, they represent a paired experiment, with the experimental basin being the planted wetland (western basin) while the control wetland (eastern basin) was left to colonize naturally (Mitsch et al., 1998). Over 15 yr, the planting led to differences in vegetation structure and function between these wetlands. The naturally colonized or "control" wetland is dominated by rapidly growing and highly productive plant species (e.g., *Typha*), whereas the "experimental" planted basin has consistently had lower productivity despite higher plant community diversity (Mitsch et al., 2012). In the naturally colonized wetland, the *Typha* patches are not evenly distributed; they are very dense in the inflow, whereas the outflow remains mostly open water, probably due to extensive herbivory of the *Typha* spp. in the years 2000 through 2003 (Mitsch et al., 2012) by muskrats (*Ondatra zibethicus*) and Canada Geese (*Branta canadensis*). This is likely due to the lower diversity in the colonized wetland, which makes it less resilient and more susceptible to herbivore eat-outs. In the planted wetland, *Typha* was historically more of an edge community, with the exception of the last few years of this study when it began to dominate (Mitsch et al., 2012).

Carbon accumulates in the soil due to the environmental conditions that determine production and decomposition of soil organic matter. The correlations between macrophyte productivity (ANPP used as an indicator) and C sequestration in these wetlands suggest that wetland productivity can be a good indicator of C accumulation in the wetland soil. We found that these wetlands stored 64 to 68% of what is coming into the system as aboveground biomass (Fig. 3). The remaining C is likely leaving the wetland through emissions of CH_4 and CO_2

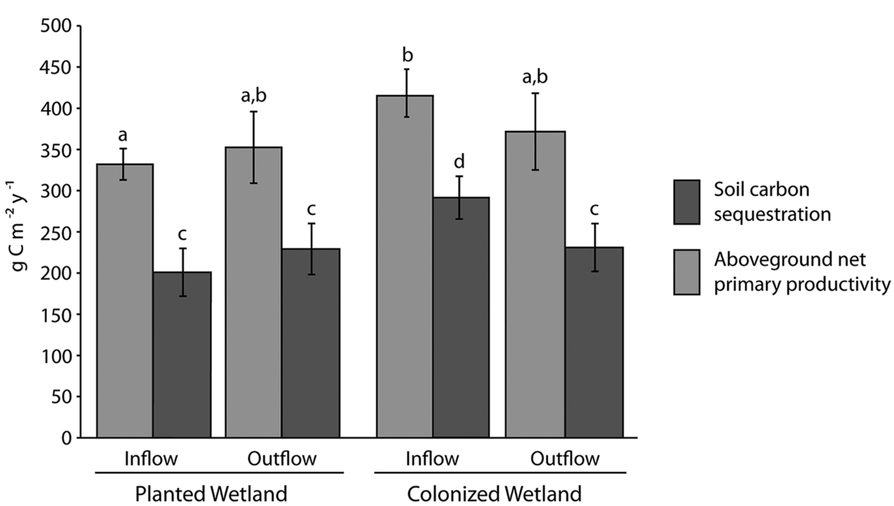

Fig. 2. Comparison of soil C sequestration (dark grey) and aboveground net primary productivity (light grey) rates (g C m^{-2} yr^{-1}) in the inflow and outflow sections of both created wetlands at the Olentangy River Wetland Research Park. Bars represent SE, and different letters indicate significant differences at $\alpha < 0.15$.

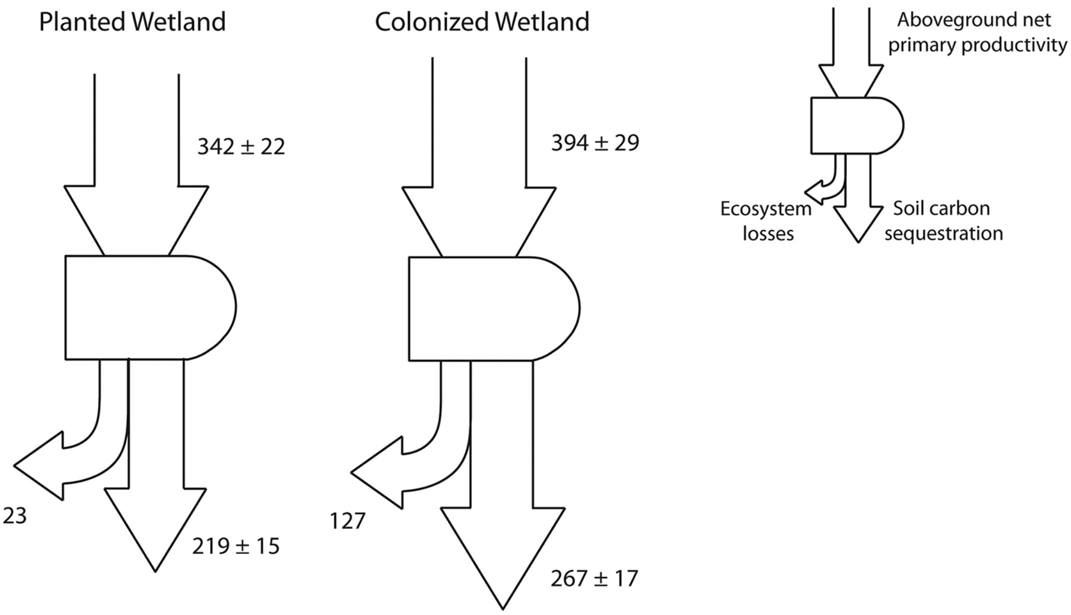

Fig. 3. Diagram summarizing wetland C inputs (aboveground net primary productivity) and outputs (soil C sequestration and ecosystem C losses). Rates for each flow are indicated next to the arrows (in g C m^{-2} yr^{-1}), whose width is proportional to their rate. (Illustration after Twilley et al. [1986] for mangrove swamps litter dynamics.)

or as C suspended or dissolved in the water and transferred laterally by animals or winds (Chapin et al., 2006). In ecosystems such as these two wetlands where organic matter and biomass accumulate significantly, ecosystem respiration is lower than productivity (i.e., $P/R < 1$), a quality often attributed to "immature" ecosystems in early stages of ecological succession (Odum, 1969; Odum and Barrett, 2004). However, Mitsch and Gosselink (2007) suggest that wetlands in general do not easily fit the Odum succession model but rather have characteristics of mature and immature ecosystems. In this study, we do not address belowground C input to the soil, which can have greater impact on soil C pool accretion than aboveground biomass (Schlesinger, 1997; Frolking et al., 2001). It would be expected, however, that the areas of the wetland with high ANPP would have high belowground biomass as well, and thus the correlation with soil C sequestration rates would be similar. If future studies address the different C pools of a wetland ecosystem, they will have to address the belowground biomass C pool along with the C pool in the aboveground biomass, the soil, and the water column.

Wetland Communities and Carbon Sequestration

We explored the effect of wetland community on soil C sequestration. Rates were higher in the open water community than in the vegetated community, a difference that is due to areas with emergent vegetation having lower C sequestration rates. These differences, however, are only significant at the 85% confidence interval, a significance level commonly not considered very impressive in ecosystem sciences but very frequently used in other fields because of its appropriateness in experiments with few replications (Christensen, 1998). Our intention in pointing out this difference is to remark on the effect that permanent anaerobic conditions have on C accumulation in these wetlands. Open water communities are not expected to be as productive as those with vegetation (emergent and edge communities in this study), although floating plants and algae mats can introduce important amounts of C into the soil (Wu and Mitsch, 1998; Bernal and Mitsch, 2012). Given the small size of these wetlands and the proximity between the various communities, the open water areas are also likely receiving large organic inputs from the plant debris produced within the wetlands.

We studied the possible interaction effect between the wetlands (planted and naturally colonized) and the communities (open water, emergent, and edge) on C sequestration to see if there was a difference in C accumulation in the communities depending on whether the wetland was planted or naturally colonized. Sequestration rates of each community type were not significantly different in both wetlands, and there was no interaction effect between community and wetland type ($P = 0.94$; $F = 0.66$). Therefore, the sequestration rates of each community do not depend on the wetland where they are located, and vice versa.

Carbon Sequestration over Time

Anderson and Mitsch (2006) measured the C content, pool, and sequestration rate of these two wetlands 10 yr after they were created. From year 10 to year 15 (Table 2), the thickness of the hydric layer increased, on average, from 9 to 14 cm. This increase, along with an increase in bulk density of 0.2 Mg m^{-3}, increased the soil C pool to 76% relative to that reported by Anderson and Mitsch (2006). However, the total C content did not increase as dramatically (from 38.5 g C kg^{-1} after 10 yr to 41.3 g C kg^{-1} after 15 yr). These two wetlands have experienced an exponential increase in their soil C pool (Fig. 4), being especially fast in the initial years after wetland creation (starting from zero because before the wetland was created there was no wetland C pool per se). If we look at the C pool of the two wetlands individually, we see that both wetlands had about the same C content shortly after creation and at the age of 10 yr (Fig. 4). However, in 2009 (15 yr after they were created), the two wetlands appeared to be diverging in their soil C pools. Even though aboveground biomass increased over time in both wetlands, the naturally colonized wetland is increasing its soil C pool at a faster rate than the planted wetland.

An increase in the C pool shows an increase in the C sequestration rate of 26%, from 190 g C m^{-2} yr^{-1} when these wetlands were 10 yr old (Anderson et al., 2005; Anderson and Mitsch, 2006) to 242 g C m^{-2} yr^{-1} when they were 15 yr old (Tables 1 and 2). This results in an average sequestration rate of 242 g C m^{-2} yr^{-1} over 15 yr; the actual C sequestration rate in the last 5 yr (from 2004 to 2009) was therefore 346 g C m^{-2} yr^{-1}. In the long term, and as long as these wetlands are not disturbed, they will probably reach "maturity" and their soil C pool will likely increase at a slower rate (Odum, 1969; Mitsch et al., 2012), approaching the rate of similar natural riverine wetlands of the temperate region (Badiou et al., 2011; Bernal and Mitsch, 2012). Therefore, it is important to address the age of a created wetland when assessing its C sequestration capacity. We compared our two created wetlands with a similar flow-through wetland in Ohio (our reference natural wetland is described in Table 2 and in detail by Bernal and Mitsch [2012]). This reference natural wetland had hydrology, nutrient loading, vegetation, and climate similar to our two created wetlands, and it sequestered 140 g C m^{-2} yr^{-1} on average, which is 43% less than what our two created wetlands are currently sequestering.

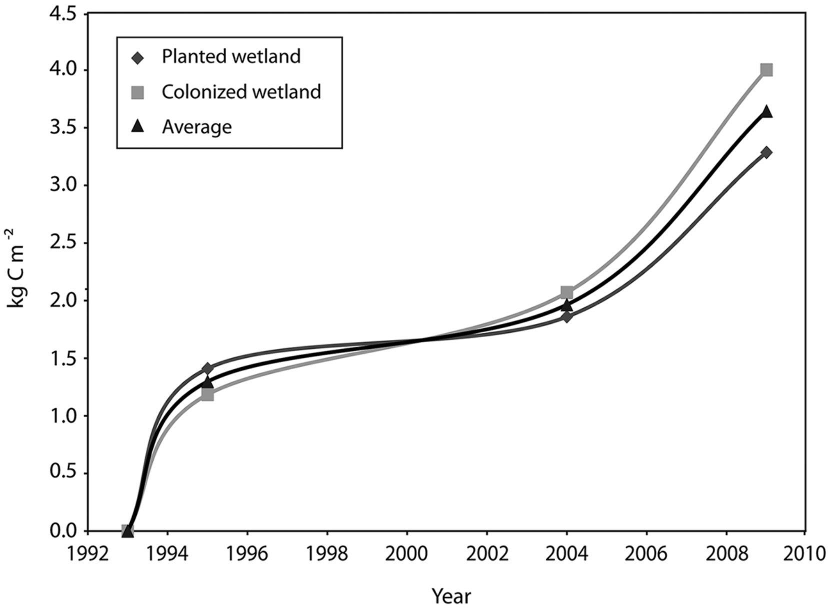

Fig. 4. Trend of total soil C (kg C m^{-2}) accumulating in the created wetlands at the Olentangy River Wetland Research Park over 15 yr (since they were created in 1994 to 2009). Carbon data of years 1995 and 2004 are from Anderson et al. (2005) and Anderson and Mitsch (2006); wetland soil C pool in 1993 (before the wetlands were created) is zero. Individual C pools of the planted and the unplanted wetlands are represented in dark and light grey, respectively; back line represents the average of both wetlands.

Creating Wetlands to Sequester Carbon

This is one of the few studies on C sequestration in created wetlands; most of the studies on soil C sequestration focus on agricultural soils, and some focus on natural wetlands. Euliss et al. (2006) estimated soil C sequestration rates of several restored prairie potholes more than a decade after restoration and found that they were accumulating 305 g C m^{-2} yr^{-1}, 3.7 times faster than their reference natural marshes in the region. Badiou et al. (2011) studied C accumulation in the soil of newly restored and long-term restored prairie pothole wetlands and estimated that, after accounting for the greenhouse gases emitted from these wetlands, their net soil C sequestration was 90 g C m^{-2} yr^{-1}. Most of the previous soil C studies in created or constructed wetlands focus on soil C pool buildup (Campbell et al., 2002; Bruland and Richardson, 2005; Fennessy et al., 2008; Gutrich et al., 2009; Hossler and Bouchard, 2010), not on C sequestration rates. Although these studies found in their created or restored sites that soil C was lower than in their natural reference sites, they all concur that the soil organic matter content in their created wetland sites increases with time (i.e., with wetland age) and that the successful soil C accretion requires successful establishment of hydrology, vegetation, and microbial communities.

There are concerns about creating wetlands to sequester C because the same anaerobic process that favors the build-up of C in the soil also favors the production of CH_4 (Mitsch et al., 2013). Therefore, for a full assessment of the net effect of a created or restored wetland as a sink or source of C to the atmosphere, one would have to take into account not only the soil C sequestration rate but also the emission of CH_4 through aerobic and anaerobic processes. Soil C sequestration rates serve as an indication of how efficient an ecosystem is in functioning as a C sink. However, the current debate on climate change has raised much interest on greenhouse gas emissions (mostly CH_4 in wetlands and, to a lesser extent, nitrous oxide) and their relation to the sequestration rate. Gleason et al. (2009) and Badiou et al. (2011) studied the balance in restored freshwater wetlands in the prairie pothole region and found that greenhouse gas emissions from these wetlands do not offset the sequestered soil C. Mitsch et al. (2013) compared C sequestration rates of 21 freshwater wetlands from around the world to their rate of CH_4 emission and found that, even after accounting for the high global warming potential of CH_4, all of these wetlands become a net sink of C and radiative forcing, given that CH_4 oxidizes to CO_2 over time. Methane is a powerful greenhouse gas, but the amount emitted from a wetland is estimated to be 1 to 3% of the wetland biomass productivity (Whiting and Chanton, 1993; Schlesinger, 1997; Melack et al., 2004).

Previous studies in these two created wetlands from Altor and Mitsch (2008), Nahlik and Mitsch (2010), and Sha et al. (2011) indicate that although CH_4 emissions are difficult to predict, open water sites had consistently higher emission rates than the other sections of the wetlands. Overall CH_4 emissions from these wetlands were higher in the open water communities under steady flow than under a pulsing hydrology (Altor and Mitsch, 2008). Additionally, greater wetland productivity leads to more available C in the soil and thus to greater CH_4 emissions (Nahlik and Mitsch, 2010).

The results from this study and others (e.g., Euliss et al., 2006; Anderson and Mitsch, 2006; Downing et al., 2008; Gleason et al., 2009; Badiou et al., 2011) indicate that created and restored wetlands can effectively sequester C. Macrophyte productivity appears to be one of the main factors enhancing C accumulation in the wetland soil. In these created wetlands, aboveground net primary productivity was high, and was higher in the unplanted, naturally colonized wetland (Mitsch et al., 2012). Nutrient-rich waters, typical of agricultural watersheds such as the one in this study, favor vegetation growth and, thus, enhancement of soil C sequestration. What seems clear from our analysis is that deeper open water sites have high C sequestration rates similar to those in the vegetated edge communities; the former is likely to have slow organic matter decomposition rates, whereas the latter is likely to be more productive.

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