Comparing carbon sequestration in temperate freshwater wetland communities

B L A N C A B E R N A L* and W I L L I A M J. M I T S C H*

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Abstract

High productivity and waterlogged conditions make many freshwater wetlands significant carbon sinks. Most wetland carbon studies focus on boreal peatlands, however, with less attention paid to other climates and to the effects of hydrogeomorphic settings and the importance of wetland vegetation communities on carbon sequestration. This study compares six temperate wetland communities in Ohio that belong to two distinct hydrogeomorphic types: an isolated depressional wetland site connected to the groundwater table, and a riverine flow-through wetland site that receives water from an agricultural watershed. Three cores were extracted in each community and analyzed for total carbon content to determine the soil carbon pool. Sequestration rates were determined by radiometric dating with ^{137}Cs and ^{210}Pb on a set of composite cores extracted in each of the six communities. Cores were also extracted in uplands adjacent to the wetlands at each site. Wetland communities had accretion rates ranging from 3.0 to 6.2 mm yr^{-1}. The depressional wetland sites had higher ($P < 0.001$) organic content (146 ± 4.2 gC kg^{-1}) and lower ($P < 0.001$) bulk density (0.55 ± 0.01 Mg m^{-3}) than the riverine ones (50.1 ± 6.9 gC kg^{-1} and 0.74 ± 0.06 Mg m^{-3}). The soil carbon was 98–99% organic in the isolated depressional wetland communities and 85–98% organic in the riverine ones. The depressional wetland communities sequestered 317 ± 93 gC m^{-2} yr^{-1}, more ($P < 0.01$) than the riverine communities that sequestered 140 ± 16 gC m^{-2} yr^{-1}. The highest sequestration rate was found in the *Quercus palustris* forested wetland community (473 gC m^{-2} yr^{-1}), while the wetland community dominated by water lotus (*Nelumbo lutea*) was the most efficient of the riverine communities, sequestering 160 gC m^{-2} yr^{-1}. These differences in sequestration suggest the importance of addressing wetland types and communities in more detail when assessing the role of wetlands as carbon sequestering systems in global carbon budgets.

Keywords: ^{137}Cs, ^{210}Pb, carbon accumulation, Gahanna Woods, *Nelumbo lutea*, Old Woman Creek, *Phragmites australis*, *Quercus palustris*, wetland community, wetland hydrgeomorphology

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Introduction

Wetlands have an important role in global carbon cycles. They are highly productive ecosystems that accumulate large amounts of organic matter in the soil, functioning as significant carbon sinks (Odum *et al.*, 1995; Chmura *et al.*, 2003; Mitra *et al.*, 2005). Despite covering 6–8% of the freshwater surface, wetlands are estimated to account for one-third of the world's organic soil carbon pool (Mitra *et al.*, 2005; Lal, 2007; Mitsch & Gosselink, 2007). Part of the carbon that the wetland produces is released to the atmosphere as methane (about 3% of the net wetland production; Schlesinger, 1997; Jokic *et al.*, 2003), a powerful green-house gas. Wetlands are estimated to be responsible for 25% of the current yearly methane emissions to the atmosphere, representing 60% of the naturally originated methane emitted each year (Bartlett & Harris, 1993; Whalen, 2005; IPCC, 2007).

The amount of carbon that a wetland stores and emits every year depends greatly on the hydrogeo-chemical characteristics of the ecosystem, which, in turn, determine the wetland vegetation communities. Therefore, to estimate with precision a wetland's carbon pool and carbon sequestration capacity, it would be more accurate to differentiate between wetland types, especially if wetlands are to be used as a carbon–sequestering systems to reduce net greenhouse gas emissions (Stern, 2007). Despite the known importance of wetlands in global carbon budgets, the lack of systematic studies and adequate models, and the limited information on their carbon turnover rates and temporal dynamics, has probably led to an underestimation of their relevance to global and

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regional levels, to the point that they are typically omitted from large-scale assessments (Trenttin & Jurgensen, 2003).

Wetlands are ecosystems defined by the presence of standing water and/or saturated soil during at least part of the year, a condition that is subsequently responsible for the development of specialized vegetation (hydrophytes) and hydric soil (Mitsch & Gosselink, 2007). These factors (hydrology, vegetation, and soil) and their interaction create the signature characteristics of wetland ecosystems and communities, and can be used to differentiate and classify wetland types. The U.S. Fish and Wildlife Service recognizes five wetland systems (marine, estuarine, riverine, lacustrine, and palustrine; Cowardin *et al.*, 1979) while the U.S. Army Corps of Engineers uses a wetland classification system developed by Brinson (1993) based on wetland hydrogeomorphology (HGM classification), that recognizes four geomorphic settings (depressional, riverine, fringe, and extensive peatlands), three water sources (precipitation, groundwater discharge, and surface inflow), and three hydrodynamics (vertical fluctuation, unidirectional flow, and bidirectional flow). Much of the rest of the world uses the 32 Ramsar wetland classes identified mainly for wildlife and biodiversity (http://www.ramsar.org). Wetland plant communities (plant associations generally described by the dominant plant species) adapt to the conditions of particular wetland zones and often reflect an environmental gradient as they change through the wetland from deep water to the upland (Boutin & Keddy, 1993; Mitsch & Gosselink, 2007). Recognition of a wetland community can be used to identify not only the plant species, but also hydrogeomorphic features that define the wetland site.

In this study, we compare the ability of wetlands to accumulate carbon in two hydrogeomorphic conditions and several vegetation communities by estimating their soil carbon pool and sediment accretion rate. Given the depositional nature of wetlands, a frequently used technique to estimate accretion rates is radiometric dating with ^{137}Cs and ^{210}Pb, two independent atmospherically deposited radionuclides of similar half-life that bind strongly to the sediments and accumulate in wetlands, functioning as a reference to date the soil (Appleby & Oldfield, 1978; Craft & Richardson, 1993; Graham *et al.*, 2005; Stark *et al.*, 2006). We determine carbon pools and sequestration rates of six different wetland communities in two different hydrogeomorphic settings of temperate humid Ohio – one flow-through riverine, inland deltaic wetland with surface inflow and directional flow (Ramsar classes L and M) and one isolated palustrine and depressional wetland, with vertical fluctuation of the water table mainly fed

by groundwater and precipitation (Ramsar classes Ts and Xf). The communities associated with the riverine system would be likely to have high total carbon accumulation because of additional external organic inputs and nutrients in the inflow water, while wetland communities on depressional and isolated sites are usually less productive than flow-through or slow-flow ones (Mitsch & Ewel, 1979; Mitsch *et al.*, 1991; Conner & Day, 1992; Watt & Golladay, 1999; Cronk & Fennessy, 2001; Wilson *et al.*, 2005). On the other hand, we might expect to find high carbon accumulation in a forested wetland community that is intermittently flooded because of the recalcitrant nature of the organic matter produced there (high in lignin and cellulose, two organic compounds that are harder to degrade by microbes due to their complexity; Dalva & Moore, 1991; Schlesinger, 1997), and because of the protection from direct wind and sun exposure that the tree canopy provides, potentially retarding plant litter decomposition rates (Kirschbaum, 1995; Fierer *et al.*, 2005; Bernal & Mitsch, 2008).

Materials and methods

Site descriptions

Our depressional forested wetland communities are located in a suburban area in central Ohio (40°0′ N, 82°50′ W). These woodland pools are mainly fed by groundwater and precipitation (Korfel *et al.*, 2010), and their surface and groundwater water tables fluctuate with precipitation events. Gamble & Mitsch (2009) and Korfel *et al.* (2010) described the seasonality of these swamp pools as depressional bodies of water that remain saturated all year-round, with permanent standing water in some of the deeper pools that fluctuates vertically, remaining frozen from December to March and drying down for the most part between June and July. The deepest pools reach up to 40 cm of water depth and saturation under driest conditions is found at 5 cm deep in the soil (Korfel *et al.*, 2010). All the pools are isolated and their water remains stagnant rather than flowing. Hydric soils (Pewamo and Carlisle muck) dominate these wetland areas (Natural Resources Conservation Service, U.S. Department of Agriculture, National Cooperative Soil Survey, 2010).

The wetland communities studied for soil carbon accumulation at this site (Table 1) include a permanent pool where cattail grows (*Typha* spp.), a semipermanently flooded forest site with pin oak (*Quercus palustris*), and an edge site of buttonbush (*Cephalanthus occidentalis*). Upland soil samples were also collected in a beech–maple (*Fagus* spp.–*Acer* spp.) forest adjacent to the wetland area.

Our flow-through wetland communities are located in the Old Woman Creek State Natural Preserve (41°22′ N, 82°31′ W), in the southwestern shores of Lake Erie. The site is a 230-ha park that connects a 69-km^2 agricultural watershed

Table 1   Description of the six wetland communities included in this study

Wetland community	Dominant vegetation	Hydrogeomorphic category	Hydrologic features	Soil type[*]	Location
Shrub	*Cephalanthus occidentalis*	Depressional, isolated	Intermittently flooded, shallow edge	Pw	Central Ohio
Forested	*Quercus palustris*	Depressional, isolated	Intermittently flooded, shallow	Pw/Cc	Central Ohio
Marsh	*Typha* spp.	Depressional, isolated	Permanently flooded, deep water	Cc	Central Ohio
Marsh	*Phragmites australis, Scirpus fluviatilis*	Riverine, flow-through	Intermittently flooded, shallow edge	FnA	Northern Ohio
Mudflat	*Leersia oryzoides*	Riverine, flow-through	Intermittently flooded, shallow	FnA/Aa	Northern Ohio
Floating bed	*Nelumbo lutea*	Riverine, flow-through	Permanently flooded, deep water	FnA	Northern Ohio

*Pw (Pewamo), Cc (Carlisle muck), FnA (Fluvaquents), Aa (Adrian muck).

(75% cropland) with the lake through a 56-ha wetland that comprises the lower 3 km of the Old Woman Creek (Mitsch & Reeder, 1991; Francko & Whyte, 1999; Herdendorf *et al.*, 2006). They receive water from the watershed (unidirectional flow) and from occasional wind-driven seiches when the outlet mouth is not blocked by the formation of a small barrier beach (bidirectional flow). The barrier is present almost half of the time due to lake wave action, and is usually broken by storm flows from the watershed, giving these wetland communities a pulsing hydrology and an ability to exchange nutrients and sediments with the watershed and the lake. Water depths range between 0.3 and 1.6 m at the mouth, with most of the wetland basin at about 0.5 ± 0.1–0.2 m deep daily (Herdendorf, 1990; Herdendorf *et al.*, 2006). The soils in the wetland are hydric (Fluvaquents and Adrian muck; Natural Resources Conservation Service, U.S. Department of Agriculture, National Cooperative Soil Survey, 2010). In the deepest areas of the wetland, floating beds of *Nelumbo lutea* (American water lotus) dominate the deep water zones that are guarded from the wind (Klarer & Millie, 1992; Whyte *et al.*, 2003; Herdendorf *et al.*, 2006). Emergent plants such as *Phragmites australis* (common reed) and *Typha angustifolia* (narrow-leaved cattail), and to a lesser extent *Schoenoplectus tabernamontani* (softstem bulrush) and *Scirpus fluviatilis* (river bulrush) grow in saturated or shallowly submerged soils, up to 50 cm of standing water (Klarer & Millie, 1992; Whyte *et al.*, 2003; Herdendorf *et al.*, 2006). The mudflats on the shallow part of the wetland (<10 cm of standing water) are characterized by *S. fluviatilis* (river bulrush) and dense stands of *Leersia oryzoides* (rice cutgrass), as described by Whyte *et al.* (2003) and Herdendorf *et al.* (2006). The upland areas surrounding the wetland are a mixed hardwood forest dominated by red oak (*Quercus rubra*), pin oak (*Q. palustris*), white ash (*Fraxinus americana*), and buttonbush (*C. occidentalis*).

Samples were collected in the three wetland communities at this riverine site: the deep water floating beds of *N. lutea*, the emergent vegetation marshes dominated by *P. australis* and *S. fluviatilis*, and the mudflats dominated by *L. oryzoides* (Table 1). The upland forest adjacent to the wetland and creek

bank of the stream that connects the wetland to the watershed were also sampled.

Soil sampling and samples preparation

Two sets of samples were collected in each of the six wetland communities (units of stratified sampling), one for radiometric analysis and the other for bulk density and carbon content determination. For radiometric analysis, a composite sample consisting of three 7-cm diameter sediment cores, up to 36 cm deep. The three cores for a composite sample were spaced within 40 cm to include variation of deposition in the sample area (Isaksson *et al.*, 2001; Stark *et al.*, 2006) and, divided *in situ* into 2-cm-thick increments. Corresponding layers were pooled together into one sample per layer and packed in sealed containers (Bernal & Mitsch, 2008). Triplicated soil samples (7 cm in diameter, 35 cm long) were extracted for the determination of bulk density and carbon content in each wetland community, divided in the field into 5 cm increments, and packed in sealed containers. One set of triplicated samples was also collected in the upland area adjacent to these wetlands. These upland samples were 10 cm diameter, 35 cm long, and were collected by the core method described by Grossman & Reinsch (2002). Upland samples were collected for carbon pool determination as a reference site for each wetland. Another extra set was collected in the creek bank feeding the wetland of Old Woman Creek, right in the inflow of the wetland basin. This set of creek cores included samples for radiometric and carbon analysis, following the procedure described above. Every soil sample taken was stored under 4 °C until analysis to minimize losses from volatilization and microbial activity, ground and passed through a 2 mm sieve after been oven-dried, and homogenized (Bernal & Mitsch, 2008). Samples collected from radiometric dating were dried at 105 °C until constant weight was reached (Craft & Richardson, 1993; Grossman & Reinsch, 2002), while the samples for carbon analysis were dried at 60 °C until constant weight to avoid potential oxidation of carbon in very rich organic soils (Grossman & Reinsch, 2002).

Estimation of sediment accretion rates

Accretion rates in the soil were determined non-destructively with ^{137}Cs and ^{210}Pb activity (pCi, 10^{-12} Ci) in each 2 cm increment soil sample by γ spectrometry for 24 h (Craft & Richardson, 1993; Smith *et al.*, 2000; Graham *et al.*, 2005) at 661.7 and 46.5 keV respectively, using a high efficiency germanium detector (Canberra, GL 2820R; Canberra Industries, Inc., Meriden, CT, USA). Radiocesium (^{137}Cs) is a man-made fallout radionuclide (30.1 years half-life) worldwide distributed as consequence of deposition from atmospheric nuclear weapon tests (Smith *et al.*, 2000; Ilus & Saxén, 2005). According to its depositional pattern, 1964 had the highest ^{137}Cs deposition on the globe. Once in the soil, ^{137}Cs binds strongly to the sediment and moves with it, remaining unaltered and making it a radionuclide widely used as tracer in dating studies, especially successful in depositional environments such as wetlands and floodplains (Yeager & Santschi, 2003; Ilus & Saxén, 2005; Stark *et al.*, 2006). The identification of the layer in the soil profile with the peak in the activity is assumed to correspond to the year 1964. Thus, the sediment accumulated in the wetland since that year can be estimated, and the accretion rate calculated assuming constant sedimentation rate, unless evidence in the profile of the opposite (Craft & Richardson, 1993; Craft & Casey, 2000; Graham *et al.*, 2005; Stark *et al.*, 2006).

In cases where ^{137}Cs profiles are not conclusive to determine accretion rates, ^{210}Pb in the core layers is often analyzed to corroborate the dating (Craft & Richardson, 1993; He & Walling, 1996; Faure & Mensing, 2005; Graham *et al.*, 2005), as they are two independent tracers that have the similar depositional pattern and behavior in the soil. ^{210}Pb (22.3 years half-life) is a naturally occurring radionuclide of the ^{238}U decay series that deposits atmospherically from the decay of ^{226}Ra (Oldfield & Appleby, 1984; Matisoff *et al.*, 2002; Faure & Mensing, 2005). The soil and the vertical accretion rate can be calculated estimating excess ^{210}Pb activity in the core and using the constant activity model, described in Eqn (1) (Appleby & Oldfield, 1978; Oldfield & Appleby, 1984; Bricker-Urso *et al.*, 1989; Pfitzner *et al.*, 2004; Faure & Mensing, 2005; Graham *et al.*, 2005):

$$A_d = A_0 e^{-(\lambda/s)} \qquad (1)$$

where A_d is the ^{210}Pb activity at depth d, A_0 is the ^{210}Pb activity at the surface, λ is the decay constant of ^{210}Pb (0.0311 yr^{-1}), d is the depth (cm), and s is the sediment accretion rate (cm yr^{-1}).

Determination of soil carbon content and carbon sequestration rate

Triplicates (50 mg) of each soil sample were analyzed for total carbon content (sum of organic and inorganic carbon) using a Total Carbon Analyzer for Soil Samples (TOC-V series, SSM-5000A; Shimadzu Corporation, Kyoto, Japan). The soil carbon concentration (C conc$_{layer}$) and pool (C pool$_{layer}$) of each soil sample increment were calculated as described by Bernal & Mitsch (2008):

$$\text{C conc}_{layer} \, (\text{gC kg}^{-1}) = 10 \times \text{TC}_{layer} \, (\%) \qquad (2)$$

$$\text{TC}_{layer} \, (g) = W_{layer} \, (g) \times \text{TC}_{layer} \, (\%) \times 10^{-2} \qquad (3)$$

$$\text{C pool}_{layer} \, (\text{kgC m}^{-2}) = 10^{-3} \times \text{TC}_{layer} \, (g)/A_{core} \, (\text{m}^2) \qquad (4)$$

where TC$_{layer}$ (%) TC$_{layer}$ (g) are the percentage of total carbon in the layer, and the total mass of carbon in the layer in grams, respectively, W_{layer} is the dry weight of soil in grams, and A_{core} is the area of the sediment sampler in m^2.

The carbon accumulation since 1964 is calculated by estimating the total soil carbon pool from the soil surface to the soil layer that, according to the radiometric profile, corresponds to the year 1964.

Statistical analysis

Statistical analyses were performed with SPSS version 19.0 (SPSS Inc., Chicago, IL, USA) for Macintosh. Student's *t*-test for independent samples at 95% significant level was used to find differences between carbon concentrations and carbon pools of the six wetland communities, between the hydrogeomorphic wetland types, and between the wetland areas and their respective non-wetland sites (upland in Gahanna Woods, upland and creek bank in Old Woman Creek). Carbon sequestration rates of each wetland were also compared using a *t*-test, at a significance level of 95% and 90%. Carbon sequestration was compared in every wetland community individually, and within each wetland hydrogeomorphic type, at these same significance levels. Significant differences indicate $P \leq 0.05$, ≤ 0.01 for 95% and 99% confidence, respectively (Fowler *et al.*, 2003).

Results

Soil carbon content

The total carbon concentration in the depressional isolated wetland sites was, on average, about three times the average concentrations of the riverine flow-through sites (146 ± 4.2 and 50.1 ± 6.9 gC kg^{-1}, respectively; Table 2, $P < 0.001$). However, the concentrations of total carbon of the depressional sites were only 1.5–2 times greater than the floating bed community (water lotus). In every site, soil carbon was predominantly organic (98–99% in the isolated depressional wetland communities, 85–98% in the riverine ones). The carbon content increased with depth of soil in the riverine deep-water community dominated by floating beds of *N. lutea* and slightly in the mudflat, as well as in the forested community and the cattail marsh of the depressional wetland (Fig. 1). Total carbon concentration in the upland site adjacent to the depressional wetlands was 16.5 ± 6.3 gC kg^{-1}, about 10% of the carbon content in the wetland sites, while the upland at the riverine sites was 7.1 ± 2.9 gC kg^{-1} (14% of what the wetland area contains; Table 2). A wetland's carbon

Table 2 Mean concentrations of total, organic and inorganic soil carbon (gC kg^{-1}) expressed as average ± standard error (n), and ratio of organic carbon over total carbon in the six wetland sites and the three non-wetland sites included in this study. All site cores are 35 cm long, except those from the shrub and the mudflat communities that are 30 cm long

| Wetland community	Hydrogeomorphic category	Total carbon (gC kg^{-1} soil)	Inorganic carbon (gC kg^{-1} soil)	Organic carbon (gC kg^{-1} soil)
Shrub	Depressional, isolated	134.4 ± 5.2 (6)	1.7 ± 0.1 (6)	132.7 ± 5.1 (6)
Forested	Depressional, isolated	143.6 ± 6.4 (7)	2.7 ± 0.6 (7)	141.0 ± 6.9 (7)
Marsh	Depressional, isolated	159.4 ± 6.8 (7)	2.7 ± 0.8 (7)	156.7 ± 7.5 (7)
All depressional wetland sites		146.4 ± 4.2 (20)	2.4 ± 0.4 (20)	144.0 ± 4.3 (20)
Upland of depressional wetland		16.5 ± 6.3 (7)	1.0 ± 0.1 (7)	15.5 ± 6.3 (7)
Floating bed	Riverine, flow-through	86.6 ± 8.4 (7)	5.4 ± 1.6 (7)	81.3 ± 9.7 (7)
Marsh	Riverine, flow-through	26.6 ± 1.8 (7)	4.3 ± 0.9 (7)	22.3 ± 1.1 (7)
Mudflat	Riverine, flow-through	35.1 ± 3.2 (6)	0.7 ± 0.3 (6)	34.9 ± 3.4 (6)
All riverine wetland sites		50.1 ± 6.9 (20)	3.6 ± 0.8 (20)	46.6 ± 6.9 (20)
Creek bank		9.1 ± 1.1 (7)	0.3 ± 0.0 (7)	8.9 ± 1.1 (7)
Upland of riverine wetland		7.1 ± 2.9 (7)	1.1 ± 0.0 (7)	5.9 ± 2.9 (7)
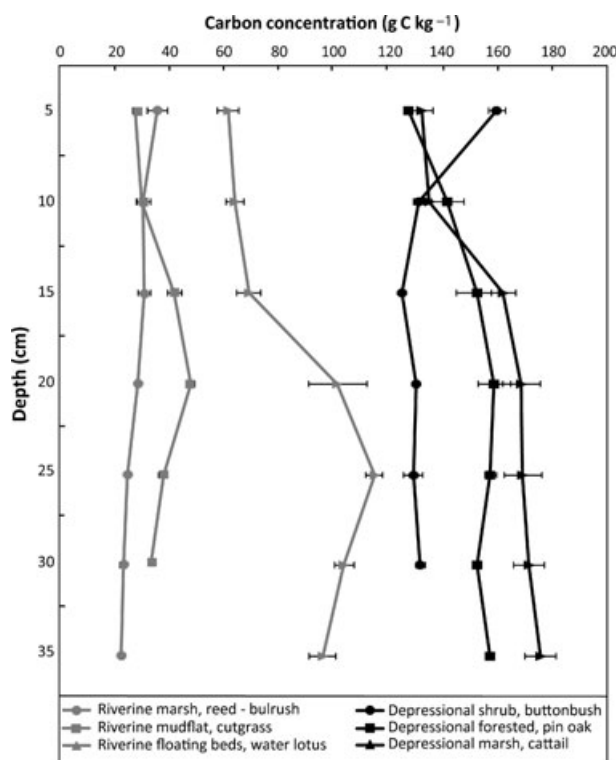
Fig. 1 Profile of carbon concentration in the soil (gC kg^{-1} soil) of the six wetland communities at the isolated depressional wetlands (in black) and the riverine flow-through site (in gray). Concentration for each depth increment is the average of that depth ($n = 3$), and the error bars represent standard error of the average.

content was significantly higher than its adjacent uplands in both cases ($P < 0.001$). The creek feeding the riverine wetland had low carbon content as well (9.1 ± 1.1 gC kg^{-1}) compared to the wetland communities, but it was 30% higher than the carbon at the

upland of the riverine site. The carbon content of the creek bank was significantly different to the wetland content ($P < 0.001$) but not to the upland.

The bulk densities (Table 3) of the depressional wetland communities were very similar, and on average (0.57 ± 0.01 Mg m^{-3}) compared to the density of the riverine ones (0.77 ± 0.06 Mg m^{-3}). Soil bulk density was consistently lower in the wetland sites compared to their adjacent uplands (upland bulk densities ranged between 1.60 and 1.68 Mg m^{-3}). The creek, despite being a non-wetland site, had a bulk density similar to the wetlands (0.84 ± 0.01 Mg m^{-3}), but much lower carbon pool (0.38 ± 0.05 kgC m^{-2}). Differences in carbon pools between wetland and non-wetland sites are evident in these sites, being the non-wetland ones one-fourth to one-third lower than in their corresponding wetland sites (1.01 ± 0.32 kgC m^{-2} vs. 4.18 ± 0.25 kgC m^{-2} in the depressional sites, and 0.64 ± 0.22 kgC m^{-2} vs. 1.50 ± 0.19 kgC m^{-2} in the riverine ones, Table 3). The average total carbon pool (up to 35 cm, Table 3) of the depressional wetland category was significantly different ($P < 0.001$) to the average pool of the riverine ones. In all the wetland communities studied, except of the *Phragmites – Scirpus* marsh, soil carbon increases with depth (Fig. 2).

Sediment accretion rates

Every wetland core analyzed for ^{137}Cs showed peaks of this radionuclide's activity (Fig. 2), corresponding to 1964. On the contrary, the creek bank had no ^{137}Cs activity detected. Old Woman Creek sites had peaks on ^{137}Cs at very similar depths (18 cm deep in both shallow reed – bulrush sites and intermittently flooded mudflat, and 16 cm deep in the water lotus floating beds), resulting in accretion estimates of 4.3 mm yr^{-1}

Table 3 Average ± standard error (number of samples) of soil carbon pool (kgC m^{-2}) and bulk density (Mg m^{-3}) in the six wetland communities (upper 35 cm in all communities except the shrub and the mudflat communities, which were 30 cm), three nonwetland communities (upper 35 cm), and overall hydrogeomorphic wetland types

Wetland community	Hydrogeomorphic category	Carbon pool (kgC m^{-2})	Bulk density (Mg m^{-3})
Shrub	Depressional, isolated	3.72 ± 0.27 (6)	0.56 ± 0.05 (6)
Forested	Depressional, isolated	3.93 ± 0.34 (7)	0.54 ± 0.03 (7)
Marsh	Depressional, isolated	4.62 ± 0.57 (7)	0.57 ± 0.05 (7)
All depressional wetland sites		4.18 ± 0.25 (20)	0.57 ± 0.01 (20)
Upland of depressional wetland		1.01 ± 0.32 (7)	1.60 ± 0.01 (7)
Marsh	Riverine, flow-through	1.10 ± 0.08 (7)	0.82 ± 0.02 (7)
Mudflat	Riverine, flow-through	1.31 ± 0.10 (6)	0.75 ± 0.02 (6)
Floating bed	Riverine, flow-through	2.73 ± 0.24 (7)	0.63 ± 0.02 (7)
All riverine wetland sites		1.50 ± 0.19 (20)	0.77 ± 0.06 (20)
Creek bank		0.38 ± 0.05 (7)	0.84 ± 0.01 (7)
Upland of riverine wetland		0.64 ± 0.22 (7)	1.68 ± 0. 03 (7)

for the former, and 3.8 mm yr^{-1} in the latter (Table 4). Based on these accretion rates, we estimated the weighted average annual sediment accumulation of the entire riverine wetland area of Old Woman Creek to be 28 tons ha^{-1} yr^{-1}. A previous study by Wilson *et al.* (2005) in this same wetland area reported a peak on ^{137}Cs activity at 17.5 cm deep (with an accumulation rate of 4.3 mm yr^{-1}), and estimated a sediment sink capacity for the entire Old Woman Creek wetland area of 47%.

In the depressional wetland sites, the ^{137}Cs activity peaked at 16 cm deep in the buttonbush community and at 18 cm deep in the *Typha* marsh, both very similar to the depths obtained in the riverine wetland sites (Fig. 2). The forested community dominated by pin oaks, however, had a much deeper peak at 28 cm. Due to the lack of consistency between these three wetland communities and because of the absence of accretion rates from the wetlands in Gahanna Woods reported in literature, we used ^{210}Pb to estimate accretion rates (Fig. 3) and support the values obtained with ^{137}Cs. The resulting accumulation of 14 cm since 1964 in the buttonbush community, 16 cm in the *Typha* marsh, and 29 cm in the forested site, are similar to the depths obtained by the ^{137}Cs method. Thus, these sites are accumulating 3.2, 3.6, and 6.5 mm yr^{-1} of soil respectively, and that this entire depressional wetland area is accumulating 23 tons ha^{-1} yr^{-1} of sediment (Table 4).

Carbon sequestration rates

On average, the depressional wetland sites accumulated almost 2.5 times more carbon per year than did the riverine sites (317 ± 93 vs. 140 ± 16 gC m^{-2} yr^{-1}, respectively; Table 4). Sequestrations are weighed averages relative to the surface area of each wetland community. The two general hydrogeomorphic types of

wetlands (depressional vs. riverine) are significantly different ($P = 0.010$). The riverine floating beds of water lotus were the most effective community sequestering carbon in the riverine type of wetland (160 gC m^{-2} yr^{-1}, about 45% higher than the other two riverine sites). The forested depressional site, however, had the highest carbon sequestration rate of all the communities studied (473 gC m^{-2} yr^{-1}), more than twice the rates estimated for the other two depressional communities (202 gC m^{-2} yr^{-1} in the shrub community and 210 gC m^{-2} yr^{-1} in the cattail marsh). The high sequestration rate of the forested community raises the average rate of the depressional sites, but in any case all of the depressional communities sequester carbon at a faster pace than did any of the riverine communities.

Discussion

Carbon profiles in wetland soils

Most of the soil carbon measured was in the organic form (85–99%). Inorganic percentages were slightly higher in the riverine sites (particularly in the marshes), which could be due to external inputs from the river or Lake Erie. In every site, high organic carbon content could be indicating that these wetlands are receiving important organic inputs despite their different placement in the landscape and their different hydrogeomorphic classification. This is usually the case for any type of wetland – these ecosystems are so productive that they have the ability to generate large amounts of organic matter (autochtonous organic source) and store it in a semidecomposed state in the soil due to the anaerobic conditions that water saturation creates (Gorham, 1998; Collins & Kuehl, 2001; Mitsch & Gosselink, 2007). In the cases
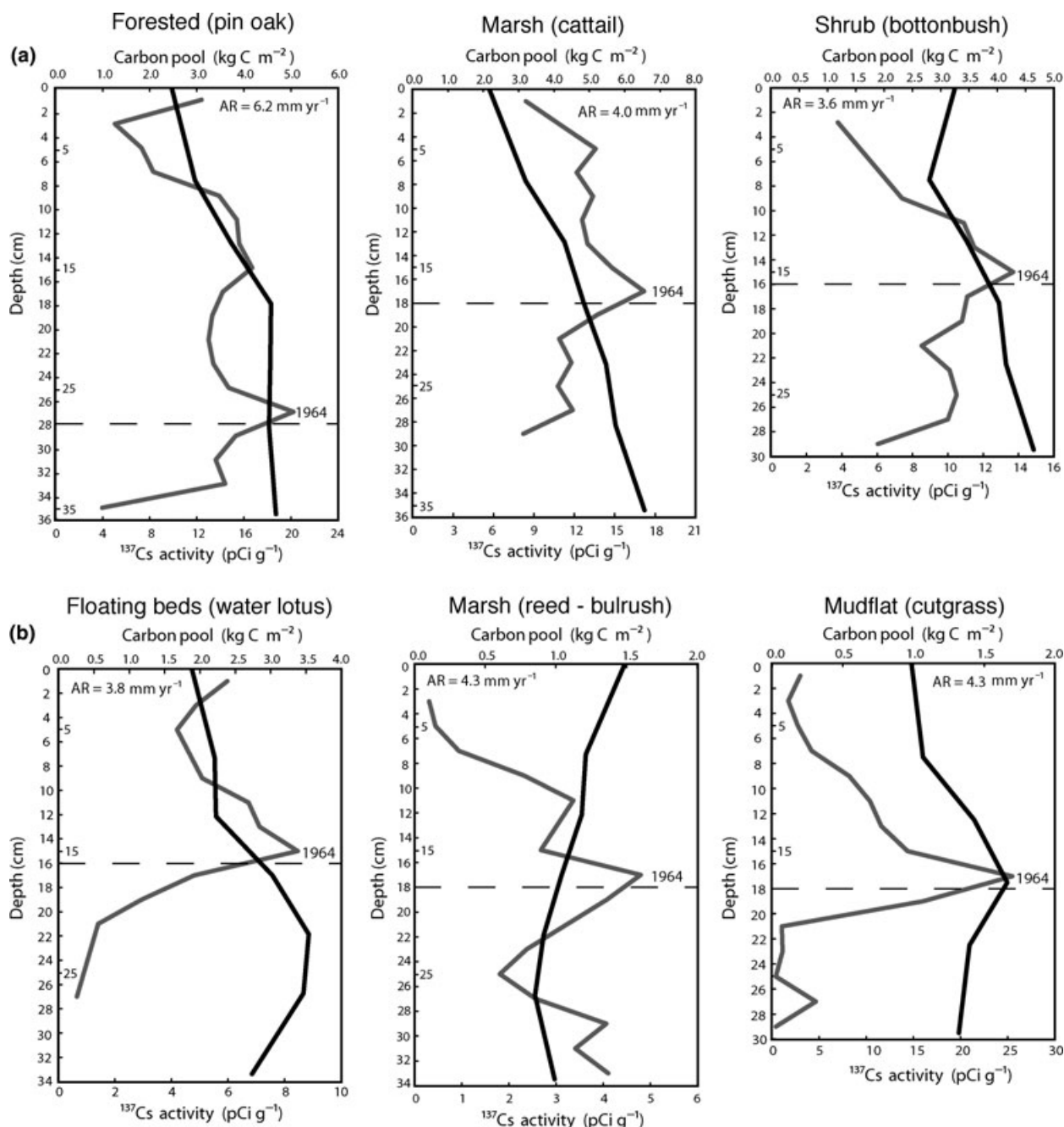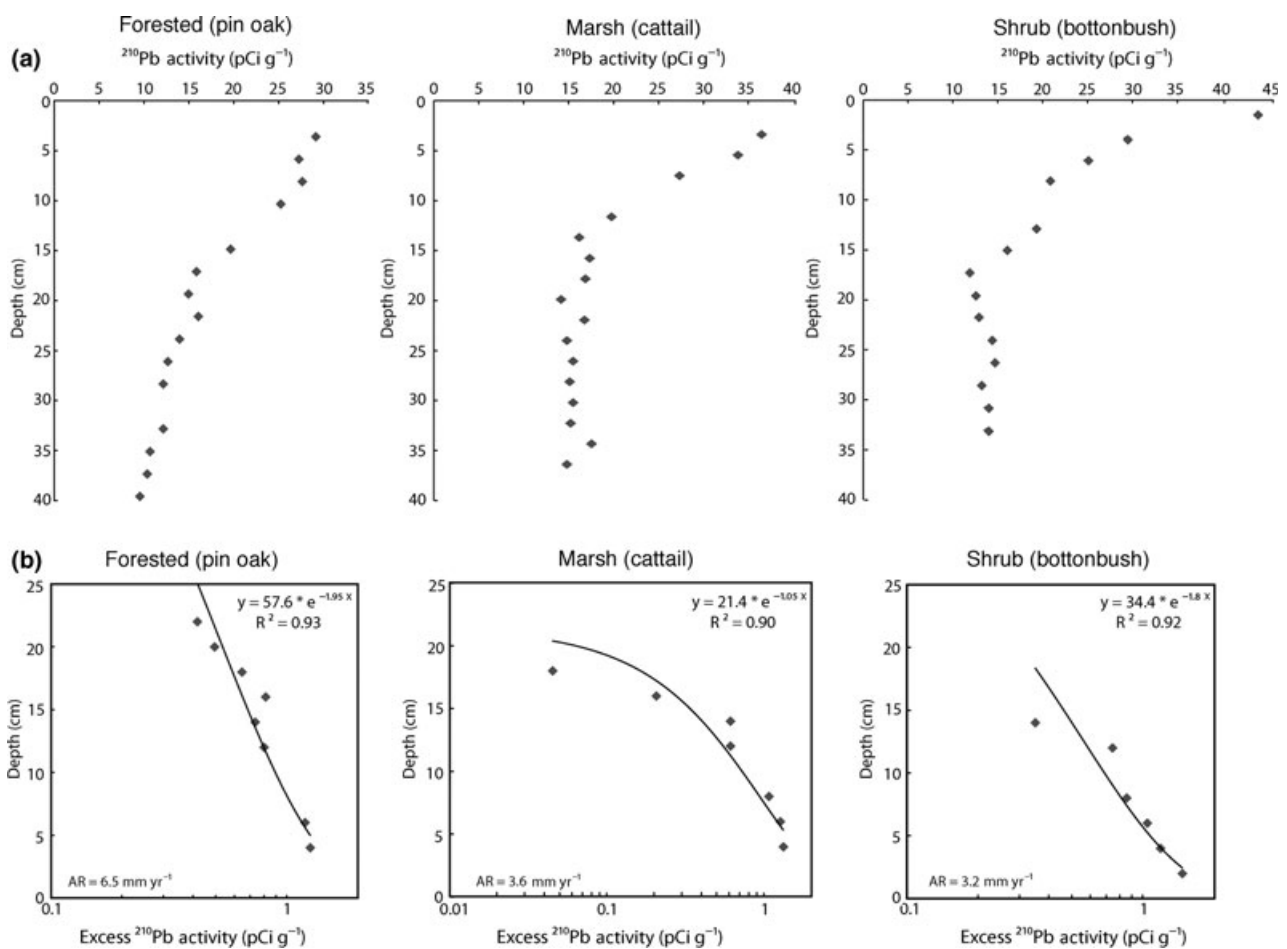
Fig. 2 ^{137}Cs activity per unit mass of soil (pCi g^{-1}) in soil cores (in gray) and carbon pool profile (in black) in six wetland communities at the (a) isolated depressional wetlands and the (b) riverine flow-through wetlands. The peak of Cs activity corresponds to the year 1964; the dotted line at that depth represents the amount of soil accumulated since that year. Sediment accretion rates (AR) are shown for each community.

where the wetlands receive external (allochtonous) organic inputs, their soil carbon content was even higher. For this reason, the carbon content of the wetland is much higher compared to its adjacent upland. The creek bank did not accumulate much carbon because of its slope and the constant fluvial processes of the creek and the occasional seiches, but it was higher than the concentration in the adjacent upland for two possible reasons: (1) being under water for most of the time could be retarding the decomposition of the organic matter contained in its soil, and (2) the water coming back and forth to the creek bank can be a source of stress (erosive agent) but also a constant subsidy of nutrients and organic matter (Odum, 1988;

Table 4 Average sedimentation and carbon accumulation in the six wetland sites and weighted average of the hydrogeomorphic wetland types

Wetland community	Hydrogeomorphic category	Annual sediment accretion (mm yr^{-1})	Annual sediment accumulation (tons ha^{-1} yr^{-1})	Carbon sequestration rate (gC m^{-2} yr^{-1})
Shrub	Depressional, isolated	3.0	14	202
Forested	Depressional, isolated	6.2	34	473
Marsh	Depressional, isolated	3.4	15	210
All depressional wetland sites		4.5	23	317
Floating bed	Riverine, flow-through	3.8	25	160
Marsh	Riverine, flow-through	4.3	35	105
Mudflat	Riverine, flow-through	4.3	31	112
All riverine wetland sites		4.1	28	140

Fig. 3 (a) Total ^{210}Pb activity per unit mass of soil (pCi g^{-1}) in soil cores from three wetland communities at the isolated depressional wetlands; and (b) excess ^{210}Pb activity profiles with estimated sediment accretion rate (AR) of each.

Odum *et al.*, 1995; Mitsch & Gosselink, 2007). All these organic rich sediments that are being eroded from the creek bed are entering the wetland, where they are deposited and temporarily stored in its soil, in a way similar to runoff from the uplands surrounding the wetlands bringing nutrient-rich sediments and organic materials into the wetland.

Soil carbon concentration increased with depth in most of our sites. Chin *et al.* (1998) measured total dissolved carbon in the pore water of the riverine wetland

at Old Woman Creek and also found that the concentration increased with depth, up to a point where it was significantly greater than the concentration in the standing water. This increase is particularly evident in the floating beds of *N. lutea*, a community that has increased in area considerably in Old Woman Creek in the last two decades (Francko & Whyte, 1999; Whyte *et al.*, 2003; Herdendorf *et al.*, 2006; Cornell & Klarer, 2008), augmenting the net primary productivity of the entire wetland area and thus the organic matter inputs into the wetland soil. The increase of carbon content with depth is frequent in sites where there is significant organic matter accumulation and the decomposition rate (carbon output) is slower than the organic matter deposition (carbon input), as described by Becker-Heidmann & Scharpenseel (1992), Gorham (1998), Schlesinger (1997), and Wolf & Wagner (2005). In the wetlands of this study, that could be due to the constant or semiconstant presence of water slowing down microbial activity (in the cattail marsh, the water lotus marsh, the reed – bulrush marsh, and the mudflat), and because of the recalcitrant nature of the organic matter entering the system (particularly in the depressional wetland sites).

The depressional isolated wetlands of Gahanna Woods are surrounded by a forested wetland community dominated by *Q. palustris*, which is able to survive because of seasonal flooding (Mitsch *et al.*, 2009). When the site is not flooded the soil is exposed and organic matter accumulated can oxidize. But overall, this site seems to be accumulating enough organic matter to compensate carbon for the loses under aerobic conditions, which could be consequence of the recalcitrant character of the organic matter coming into the soil from the trees. Trees litter reaches the wetland directly or through runoff or wind. Plant litter from woody species is rich in lignin and complex polysaccharides, difficult to degrade by microorganisms (Schlesinger, 1997; Wolf & Wagner, 2005; Berg & McClaugherty, 2008) and thus, remaining longer in the soil and accumulating deeper in the profile than rapidly degradable labile compounds (Schlesinger, 1997; Trumbore, 1997; Wolf & Wagner, 2005). Also, water from precipitation falling through the tree canopy onto the wetland is likely to contain more organic compounds than water from direct precipitation onto the wetland. Dalva & Moore (1991) studied this phenomenon and found that water richer in dissolved organic carbon after passing through the forest canopy. The profiles of these depressional isolated wetland sites are more similar among themselves than are the ones in at the riverine sites, probably because the depressional communities were closer to each other and likely received similar organic inputs from the surrounding forest. The more recalcitrant woody plant input could be a reason why the carbon soil content in the depressional communities is about one order of magnitude higher than in the riverine sites.

Despite the similar or higher carbon accretion rates on the depressional sites compared to the riverine ones, total accumulation of sediment is lower because its soil has lower density. The high annual sediment accretion in the forested community can be interpreted as evidence of the large amount of organic matter accumulating in this site from the trees litter, probably due to its high productivity and recalcitrant character.

Carbon sequestration in freshwater temperate wetlands

The studied forested isolated wetland sites had greater carbon sequestration rates (317 ± 93 gC m^{-2} yr^{-1}) than did the riverine sites (140 ± 16 gC m^{-2} yr^{-1}), but also greater variability. The high productivity of this forested site raises the average rate of all of our depressional sites. The carbon sequestration rates of the marsh and shrub depressional communities studied, however, are also higher compared to any of the riverine ones. Such a significant difference between our two wetland types is likely due to the high productivity of the forest in which they are located, cooler temperatures provided by the shade that retards litter decomposition, and the recalcitrance of the organic matter being introduced in the wetland (Fierer *et al.*, 2005). Within the riverine sites, the sites with highest carbon sequestration rate were the floating beds of water lotus (160 gC m^{-2} yr^{-1}), probably because of the combined effect of the permanent anaerobic conditions in this deepwater area and the reported high productivity of this community. It is therefore important to take into account these differences when comparing the efficiency of wetlands as carbon sequestering systems. The flooding duration and the type of vegetation growing in the wetland are factors that control soil carbon accumulation, and therefore can be managed to enhance the natural ability of a wetland to accumulate carbon, while maintaining other valuable wetland functions and ecosystem services.

To put the carbon sequestration rates obtained in this study in perspective, we compare them with other carbon sequestration rates estimated for freshwater temperate wetlands reported in literature (Table 5). Most of these wetland studies used radiometric dating with ^{137}Cs and/or ^{210}Pb. The range of carbon sequestration in temperate zone inland wetlands range over an order of magnitude from 56 to 504 gC m^{-2} yr^{-1} (Table 5). The mean of all these temperate zones estimates is 174 gC m^{-2} yr^{-1} and the median is 131 gC m^{-2} yr^{-1}. Eliminating outliers, most of the numbers range from 100 to 280 gC m^{-2} yr^{-1}. Our carbon sequestration

Table 5 Comparison of carbon sequestration rates of freshwater wetlands located in temperate regions reported in literature. Carbon sequestration rates estimated in this study are included in each appropriate wetland type. All studies except those with an asterisk used ^{137}Cs and/or ^{210}Pb to determine carbon sequestration rates

Wetland site	Carbon sequestration rate (g m^{-2} yr^{-1})	References
Isolated, depressional, and/or forested wetlands		
Dismal swamp, Virginia	105	Craft *et al.* (2008)
Cypress swamp, Florida	122	Craft *et al.* (2008)
Pocosin, North Carolina	127	Bridgham & Richardson (1993)
Okefernokee peat swamp, Georgia	25–82	Schlesinger (1978)*, Craft *et al.* (2008)
Peat meadow, Netherlands	280	Hendriks *et al.* (2007)*
Danube floodplain, Austria	180	Zehetner *et al.* (2009)
Cypress-gum floodplain, Georgia	107	Craft & Casey (2000)
Upper St. Johns floodplain, Florida	117–244	Brenner *et al.* (2001)
Pin oak swamp, Ohio	473	This study
Buttonbush swamp, Ohio	202	This study
Cattail marsh, Ohio	210	This study
Riverine and/or flow-through wetlands		
Cattail marsh, nutrient enriched, Florida	264	Reddy *et al.* (1993)
Cattail-sawgrass marsh, Florida	104–167	Reddy *et al.* (1993), Craft & Richardson (1993)
Sawgrass marsh, Florida	124	Reddy *et al.* (1993)
Grass-sedge marsh, Georgia	56	Craft & Casey (2000)
Reed-bulrush marsh, Oregon	116	Graham *et al.* (2005)
Reed marsh, Denmark	504	Brix *et al.* (2001)*
Arum arrow marsh, Virginia	97	Whiting & Chanton (2001)*
Anderson tule marsh, California	106–155	Kim (2003)
Everglades, Florida	99–190	Craft & Richardson (1993)
Reed-bulrush marsh, Ohio	105	This study
Cutgrass mudflat, Ohio	112	This study
Water lotus marsh, Ohio	160	This study

estimates (317 ± 93 and 140 ± 16 gC m^{-2} yr^{-1}; Table 4) fall within the range of previously published rates. The average values for isolated, depressional, and/or forested wetlands reported in literature (188 ± 33 gC m^{-2} yr^{-1}), however, are more similar to those of riverine, flow-through wetlands (164 ± 37 gC m^{-2} yr^{-1}) than to our forested wetland measurements. All these temperate zone mineral soil wetland estimates are, in general, considerably higher than the average rates of carbon sequestration estimated for boreal peatlands (10–61 gC m^{-2} yr^{-1}; Mitsch & Gosselink, 2007), a much less productive type of wetlands where most of the wetland carbon studies are focused. Many wetland carbon budgets are based on peatland carbon pools and sequestration rates, which may lead to a general underestimation of the role of wetland ecosystems in global carbon budgets and to the misconception that every wetland has a similar low accretion rate.

Implications of this study

Wetlands are important sinks of carbon, as evidenced in the significant differences between wetland and non-wetland sites in this study. However, we have shown that not all wetlands are equal in their ability to sequester carbon. In our study the six temperate wetland communities differed in their hydrogeomorphic type and placement in the landscape, thus allowing different vegetation communities to dominate. These differences resulted in significantly different carbon content in the soil (greater in all the depressional wetland sites than in the riverine ones, the greatest in the forested community, and greater in the riverine deeper areas than in the shallower ones). The hydrogeomorphic types of these wetland communities also had significantly different carbon sequestration rates (on average, 2.5 times higher in the depressional wetland areas than in the riverine sites). Not every wetland is equally effective in sequestering carbon; it is important to address differences in wetland types and vegetation communities when assessing the role of wetlands as carbon sinks in global carbon budgets.

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