Soil Properties and Vegetative Development in Four Restored Freshwater Depressional Wetlands

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The creation and restoration of wetlands is widely seen as a critical tool for replacing ecosystem functions lost by historic wetland destruction. However, studies have shown that these wetlands often take hundreds of years to achieve the functions for which they are restored. We used controlled field-scale manipulations in four recently restored depressional freshwater wetlands in western New York to investigate the impact of organic amendments of differing lability on the soil and vegetative development during the first 3 yr. Results showed that the addition of soil amendments to wetland plots stimulates development of key soil properties that are critical for wetland functioning. In particular, initial increases in soil C and decreases in bulk density in topsoil and biochar amended plots were still present 3 yr after restoration. Plant biomass recovered quickly and had reached levels of comparable natural wetlands within 2 yr, irrespective of amendments. Amendments did not influence plant diversity. Site differences, however, did influence plant diversity and different sites hosted different numbers and types of species. Two years after restoration, both desirable native wetland species and undesirable weedy species had colonized each site. Results of this research reveal that organic amendments can improve key soil properties critical for wetland functioning. The strength of treatment effects and the development of the plant community, however, are highly influenced by initial site conditions. These results confirm the importance of focusing on both hastening soil development via amendments and careful site selection in restoration design.

Abbreviations: BD, bulk density; CEC, cation exchange capacity; SOM, soil organic matter.

Although they cover less than 2% of earth's surface, wetlands perform more ecosystem services (e.g., water purification, aquifer recharge, climate regulation, long-term C storage, flood abatement, and habitat provision) per hectare than any other ecosystem type (Aselmann and Crutzen, 1989; Costanza et al., 1997; Mitsch and Gosselink, 2000). More than 50% of the earth's wetlands have been lost to agriculture and development, however, with some U.S. states having destroyed more than 90% of their wetlands between 1780 and 1980 (Dahl, 1990). In response to both historic losses and the continuing threat of wetland destruction, numerous federal, state, and private agencies in the United States have initiated wetland restoration programs. Current federal policy for mitigating damage to wetlands commonly assumes that a restored ecosystem will replace losses in wetland structure and function within 5 to 10 yr. However, research has shown that some soil properties essential for wetland functions, such as water quality improvement, do not approach natural wetland levels for centuries (Ballantine and Schneider, 2009). These findings have seri-

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ous implications for the ability of restored wetlands to perform their intended functions.

Wetland functions are predominantly dependent on extensive interactions between water and wetland soils. Therefore, the condition of the soil may be one of the most critical components in restoration of wetlands. Soil organic matter (SOM) in particular is a key property of soils that directly influences ecosystem functions, but this critical property of wetlands has proven especially challenging to restore (Gwin and Kentula, 1990; Morgan and Short, 2002; Bruland et al., 2003). The SOM contributes to soil structure, promoting aeration, microbial habitat, root penetration, and water-holding capacity (Brady and Weil, 2002). The SOM controls hydrologic properties, such as bulk density (BD) and porosity, both of which influence water infiltration and flow rates. The SOM is also important to plants, holding a large proportion of nutrients, cations, and trace elements critical for their growth. Finally, SOM buffers soil from strong changes in pH and has also been shown to control properties that remove contaminants from water, such as trace metal adsorption, nutrient sequestration, and denitrification, an important biogeochemical process responsible for nitrate reduction in groundwater (Ponnamperuma, 1972; Craft et al., 1988; Hogan et al., 2004; Anderson et al., 2005). For all of these reasons, SOM is widely acknowledged as an indicator of wetland health.

Despite the importance of soil in providing the substrate for many of the biological and chemical functions that wetlands perform, soil conditions are often the least considered component of wetland systems (Bruland et al., 2003). Draining wetlands for agriculture or construction creates an aerobic soil environment in which SOM is oxidized and soil C is lost (Sutton-Grier et al., 2009). Many depressional wetland restorations involve excavations that intersect the groundwater level, leaving subsoils exposed and soils severely compacted from the weight of wide-tracked wetland bulldozers and other heavy equipment. This compaction increases the BD of the soil making it more difficult for soil organisms and plant roots to penetrate soils. Re-grading may also involve the complete removal of the topsoil layers that tend to be richest in SOM. Thus, wetland disturbance and restoration often create conditions that decrease the soil quality in newly restored wetlands. Once the wetland restoration is complete, soil development is a relatively slow process that only appears to accelerate later in the successional recovery sequence (Ballantine and Schneider, 2009). Because soil processes are critical to overall wetland development and to achieving desired ecosystem services, the development of soil parameters should be incorporated into initial restoration goals, project design, and site construction. Research investigating restoration practices that hasten soil development have been recommended to improve the likelihood of functional success of restored wetlands by maximizing the potential for soil development (Ballantine and Schneider, 2009).

In particular, the use of soil amendments could be a promising strategy to stimulate functions of restored wetlands. Organic matter additions in the form of compost or salvaged marsh soil have been shown to improve soil by stimulating nutrient cycling and microbial community development, increasing soil moisture as well as C and N pools and P sorption, and decreasing BD in both coastal and inland restored and created wetlands (Duncan and Groffman, 1994; Stauffer and Brooks, 1997; Bruland and Richardson, 2004; Bailey et al., 2007; Bruland et al., 2009; Sutton-Grier et al., 2009). In particular, initial addition of topsoil in nontidal freshwater wetland soils has been shown to be an effective strategy for increasing plant biomass, cation exchange capacity (CEC), soil moisture, water-holding capacity, P sorption, and denitrification (Brown and Bedford, 1997; Burke, 1997; Burchell et al., 2007; Jacinthe and Lal, 2007).

Unfortunately, specific recommendations for incorporating amendments into wetland restoration plans are rare. Furthermore, recommendations that have been published are often conflicting because some studies report no response effects, implying that the time and money invested into incorporating amendments are not worthwhile. Vegetation parameters in particular have yielded mixed results. While some studies report increased plant biomass or diversity (Erwin and Best, 1985; Stauffer and Brooks, 1997), others have reported no difference in plant growth in the first few years (Bailey et al., 2007; Sutton-Grier et al., 2009). This is significant as plant growth and diversity are commonly the variables used to determine success of wetland mitigation projects approved by the Army Corps of Engineers (Hoeltje and Cole, 2007).

Conflicting results may in large part be due to differences in initial soil conditions unique to specific restoration sites. For example, while organic amendments have increased soil C and N in some restoration sites, high decomposition rates and sandy soils resulted in no increase in C or N pools after amendments were added to a created salt marsh in southern California (Gibson et al., 1994). It is difficult to assess the applicability of reported recommendations to other potential restoration sites if the initial conditions of the soils in that site are unknown. This is problematic because even sites in close proximity can have vastly different hydrology and baseline soil conditions. Clearly, more information is needed concerning amendment effects at different site types, the effects of different types of amendments in the same site, and long-term benefits if we are to determine whether the initial costs of soil amendments are worthwhile (Bendfeldt et al., 2001b).

To address these gaps in the literature, we examined the effect of amendments of different organic matter on the development of restored wetland soils in each of four newly restored wetlands. Our objectives were to: (i) determine if the soil amendments influenced key soil variables or vegetation parameters, and (ii) determine if the effect of amendments on soil or plant variables was influenced by individual site characteristics.

MATERIALS AND METHODS

Experimental Design

The field experiment was conducted in four newly restored wetlands (Sites 1, 2, 3, and 4), each within 120 km of Ithaca,

Table 1. Site soil and amendment chemical properties based on 2007 pre-restoration conditions. Soils were sampled to 0.1-m depth. Phosphorus, K, Mg, Ca, Fe, Al, Mn, Zn, Cu, and NO_3 extracted using the Morgan method (Morgan 1941).

Treatment	C	N	P	K	Mg	Ca	Fe	Al	Mn	Zn	Cu	pH	NO_3
	– g/kg –						– mg/kg –						mg/kg
Straw	441.7	4.4											
Biochar	614.7	6.6	34.40	6028.00	274.00	2346.00	70.40	0.40	48.00	3.42		7.18	0.00
Topsoil (Site 1)	45.9		4.18	55.20	413.34	3658.40	3.54	8.00	6.92	0.21	0.72	6.68	0.00
Topsoil (Site 2)	198.6		3.20	31.00	485.40	7067.00	495.20	140.30	17.70	7.90	1.90	5.21	27.02
Topsoil (Site 3)	39.3		2.84	30.60	689.04	5699.40	6.40	15.24	19.20	0.45	1.70	7.11	1.20
Topsoil (Site 4)	25.8		1.34	49.20	101.08	664.00	37.12	161.94	39.18	1.20	0.30	5.38	0.00
Subsoil (Site 1)	21.3	1.1	0.80	38.67	1077.57	14427.67	29.80	35.93	62.43	0.18	19.20	7.90	0.00
Subsoil (Site 2)	30.2	1.2	0.96	24.80	820.46	6491.20	70.14	43.94	27.60	1.64	1.75	6.98	0.00
Subsoil (Site 3)	16.6	0.6	0.96	31.60	1074.92	13182.60	3.78	51.82	30.42	0.17	16.16	7.88	1.10
Subsoil (Site 4)	06.2	1.1	0.66	23.40	47.88	370.20	30.56	120.42	17.28	0.43	0.42	5.13	0.00

NY. Two sites, 1 and 2, were located relatively close to each other, separated by approximately 400 m and a hedgerow. Each wetland was restored in July 2007 on retired agricultural fields by removing topsoil and using that soil to build a flood control berm. To ensure minimal elevation variation between plots, the bottom topography was leveled with bulldozers. Immediately after restoration, before flooding occurred at each of the four sites, we established 25 2 by 2 m experimental plots to measure soil parameters (five replicates of each of four treatments plus one control) and 15 2 by 2 m experimental plots to measure vegetation parameters (three replicates of each of four treatments plus one control). Each plot was separated from its nearest neighbors by 2 m.

The treatments (straw, topsoil, a 50:50 mix of straw and biochar, biochar, and the control) were assigned to plots in a randomized block design. Carbon content was applied at the same rate across all treatments, with 8 kg of organic C added to each plot. This represented an increase of 66% to more than 350% above the amount of pretreatment C levels, depending on the site. The control plots received no organic addition, but like all other plots, were roto-tilled to 0.1-m depth. The straw treatment was composed of dry stalks of organically grown wheat, *Triticum aestivum* subsp. *spelta,* obtained from Oescher Farm in Newfield, NY. The biochar was made from a mixture of hardwoods by fast pyrolysis at 450°C with a retention time of <5 s (Dynamotive, Vancouver, Canada). The topsoil amendment of each site was taken from homogenized topsoil of that same site (Table 1).

Representative 0.1-m deep soil cores were taken using a chrome molybdenum corer (0.019 m diam.) pushed gently into the soil. Eight randomly distributed cores per site were collected before restoration and again of the subsoil postrestoration. One core per treatment plot was taken immediately after the plots were established (2007) and in July 2008 and 2010, 1 and 3 yr after the wetlands were restored. To avoid interactions between plants and microbes that would confound the results, plants were removed from the soil plots by hand or with an oscillating hoe throughout every growing season. Separate plots in which vegetation was allowed to grow were established in three of the four sites (1, 2, and 4), and were used to measure plant biomass and diversity. Vegetation plots were not set up in Site 3 due to insufficient space.

Study Sites

The restored wetlands are all palustrine emergent depressional wetlands (Cowardin et al., 1979). Although they are all similar in topography, size, and history, they differ in soil type and hydrology (Table 2). Sites 1 and 2 were restored on the property of Jim Carter by Marshland Excavating and were permitted by the Seneca County Soil & Water Conservation District as a part of the USDA Natural Resources Conservation Service Wetland Reserve Program. Site 3 was restored on the property of the Cornell University Biological Field Station, also as a part of the USDA Natural Resources Conservation Service Wetland Reserve Program. Site 4 was restored by the Upper Susquehanna

Table 2. Site characteristics of the four restored wetlands examined in this study.

Site	Location	Landscape position	Soil type	Soil saturation	Area
					ha
1	42°55′39″ N 76°51′31″ W	Depression	Canandaigua: very deep, poorly drained, fine-silty, nonacid, mesic Mollic Endoaquepts	Consistent	1.2
2	42°55′37″ N 76°51′22″ W	Depression	Alden: deep, poorly drained, fine-loamy, nonacid, mesic Mollic Endoaquepts	Consistent	0.8
3	42°23′11″ N 76°18′17″ W	Depression	Canandaigua: very deep, poorly drained, fine-silty, nonacid, mesic Mollic Endoaquepts	Intermittent	0.8
4	43°10′11″ N 75°56′04″ W	Depression	Middlebury: very deep, moderately well drained, coarse-laomy, mesic Fluvaquentic Eutrudepts	Intermittent	2.4

Coalition as a mitigation wetland and is located in the Goetchius Wetland Preserve, now property of the Finger Lakes Land Trust. Each site was surveyed and the water level was measured with a series 12 0.6-m deep PVC wells distributed evenly throughout each site. Elevation of the water table was measured in wells once monthly during the growing season in 2008 and 2010. Water table depths relative to the soil surface were averaged to create a single overall index of soil flood condition across each site.

Laboratory Analysis

Each soil core was analyzed for total soil C, total soil N, BD, and soil moisture. Each sample was dried to constant weight at 65°C, weighed, and passed through a 22 mm diam. mesh sieve. The sieved coarse material was weighed again and stored in the dark at 44°C until processing.

Bulk density was calculated using the air-dried weight of the soil after correcting for the moisture content (Blake and Hartge, 1986). Soil moisture of each sample was measured gravimetrically by drying each sample at 105°C for 24 h. Total C and N of the amendments and the soil samples were analyzed using the combustion gas analyzer method combined with a gas chromatographic separation and thermal conductivity detection by the Stable Isotope Facility, University of California, Davis.

Standing biomass samples were taken in September 2008 from a random 0.25 m^2 quadrant of each plant plot and oven dried at 65°C to constant weight and weighed to the nearest 0.1 g. Vegetative diversity was measured in plant plots by identifying every plant in the plot to the species level where possible.

Statistical Analysis

A mixed-model MANOVA (fixed effects = treatment, site, year, treatment × site, treatment × year, site × year, treatment × site × year; random effect = plot ID) was performed to assess significant effects across all soil variables measured in this study (Statistical package R). Next, univariate mixed-model ANOVAs were performed using the same model design as the MANOVA to assess significant effects for individual response variables (JMP version 9, SAS Institute, Inc.). In cases where significant fixed effects were detected, pair-wise comparisons among groups were made with Tukey's test of Honestly Significant Difference (HSD). All variables were tested for normality and homoscedascity and were transformed to meet these criteria where necessary.

RESULTS

Properties of the subsoil differed among the newly restored sites (Table 1), as did site hydrology. In particular, pretreatment soil C concentration differed among sites (p = 0.0251), and was highest in Site 2, followed by Site 1, Site 3, and, finally by Site 4. Sites 1 and 2 were consistently flooded for much of the growing seasons of 2008 and 2010, with water levels dropping below the soil surface in August of 2008 in Site 1, and August of both 2008 and 2010 in Site 2. In contrast, Site 4 was drier, with intermittent inundation throughout the growing season. Site 3 was not submerged in 2008, but flooded for much of 2010 (Fig. 1).

The mixed model MANOVA identified significant effects of treatment, site, year, site × year, and treatment × site across all response variables (Wilks' Lambda p = <0.0001 for all). For soil C, there was a significant effect of treatment and site (p = <0.0001 for both) based on the mixed model ANOVA. There was no effect of year. Nonetheless, a qualitative assessment of C across years revealed several consistent noteworthy trends. Soil C among treatments did not differ immediately after amendment addition. In 2008, 12 mo after amendment addition, C in Biochar plots appeared greater than C in Control plots (Fig. 2). That year,

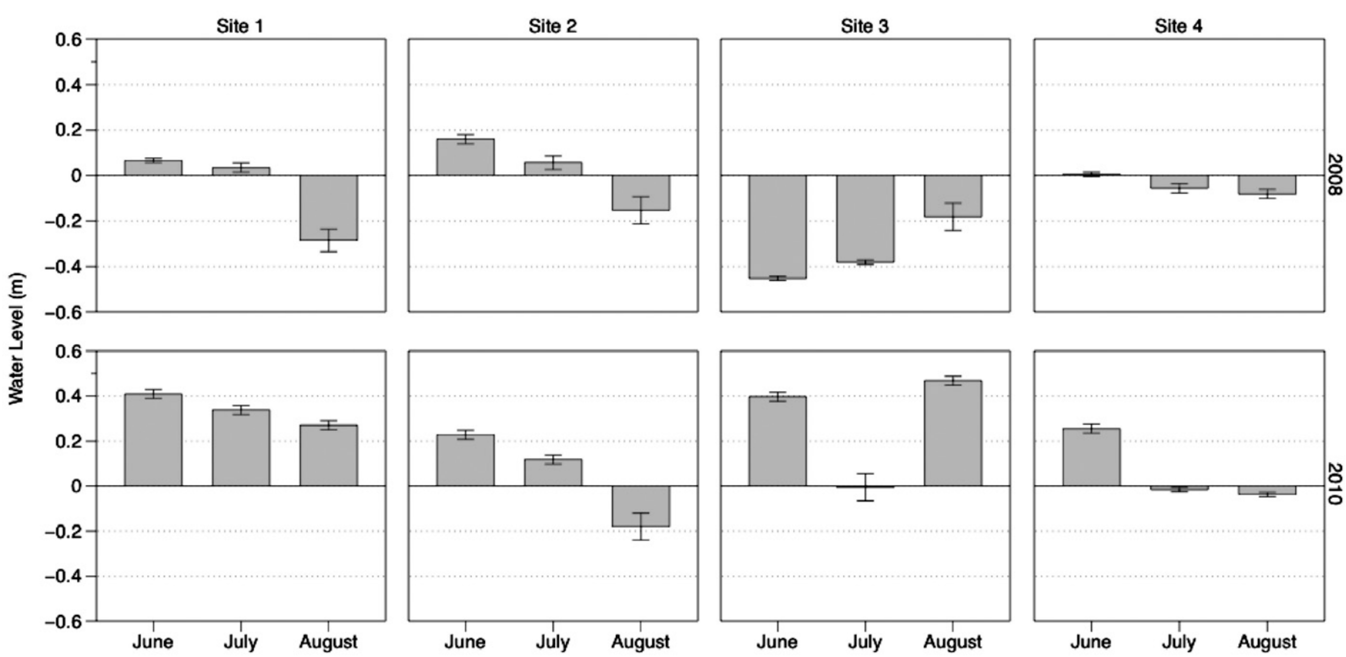

Fig. 1. Water level (m) above or below soil surface (zero level) as an average of 12 well measurements across each site on each date (mean + standard error).

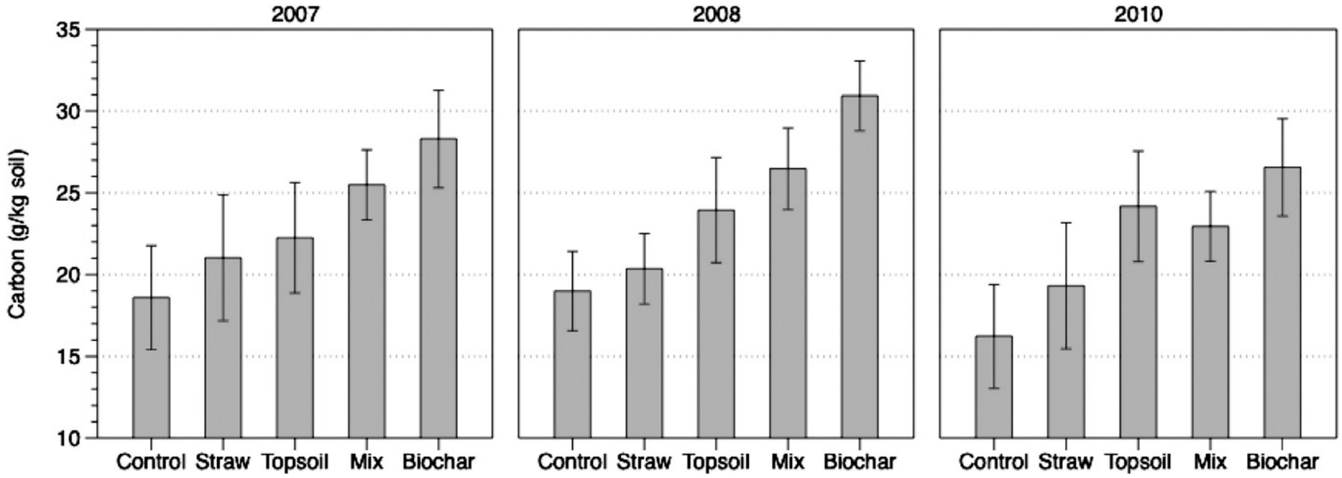

Fig. 2. Soil carbon by treatment and year, averaged across all sites (mean + standard error).

Biochar plots had the highest C, followed by Mix, Topsoil, Straw, and finally Control plots. In 2010, 3 yr after amendment addition, the pattern across treatments was the same, except Topsoil had slightly more C than Mix plots. Treatment differences in 2010 appeared greater, with Biochar still having the highest C, followed by Topsoil, Mix, Straw, and finally Control plots. In 2010, mean C of Biochar, Mix, and Topsoil plots was 145 to 165% greater than Control plots. The overall pattern of increasing C across treatments (Control < Straw < Topsoil < Mix < Biochar) was consistent across sites, though overall C was higher in Sites 1 and 2, than in Site 3, which was significantly higher than Site 4 (Fig. 3).

Soil C was positively correlated with soil moisture at the time of sampling (p = 0.0057). The mixed model ANOVA of soil moisture found a significant effect of treatment and site (p = 0.0012, <0.0001, respectively). Topsoil and Straw plots had the highest soil moisture, followed by Mix, Biochar, and finally Control plots. Specifically, Topsoil plots had significantly higher soil moisture than Biochar and Control plots, while Straw and Mix plots had significantly higher levels than Control plots alone. Soil moisture was significantly higher in Site 1 than all other sites. Site B had significantly higher levels than Site 3.

Amendment additions decreased BD in all sites. The mixed model ANOVA of BD found a significant effect of treatment, site, year, and site × year (p = 0.0019, <0.0001, <0.0001, <0.0001, respectively). Control plots had higher BD than all the other plots, though levels were not significantly different than Biochar plots. Sites 1, 2, and 3 had significantly higher BD than Site 4 (Fig. 4). Bulk density decreased from 2007 to 2010 in all Sites except Site 4, where it significantly increased.

The mixed model ANOVA of soil N found a significant effect of treatment, site, and site × treatment (p = <0.0001, 0.0152, 0.0121, respectively). Levels of N were highest in Topsoil plots in each site ($p \leq 0.05$), though the order of other treatments was variable ($p \leq 0.05$) (Fig. 5).

Plant biomass recovered quickly in the first year of development, increasing from zero immediately after restoration to a mean of 724.8 g/m^2. There were no significant differences among treatments or sites (Table 3). Plant diversity did not differ significantly by treatment, but Site 2 had significantly more

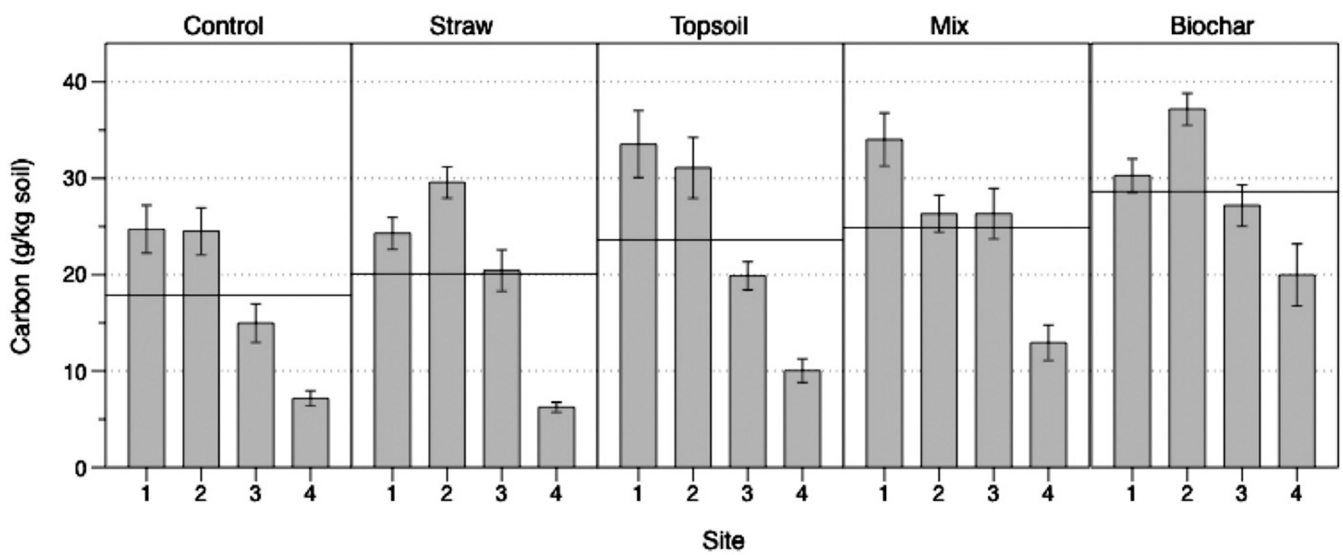

Fig. 3. Soil carbon by treatment and site (1–4), averaged across all years (mean + standard error). Dark horizontal lines signify mean soil C averaged across all sites and years for each treatment.

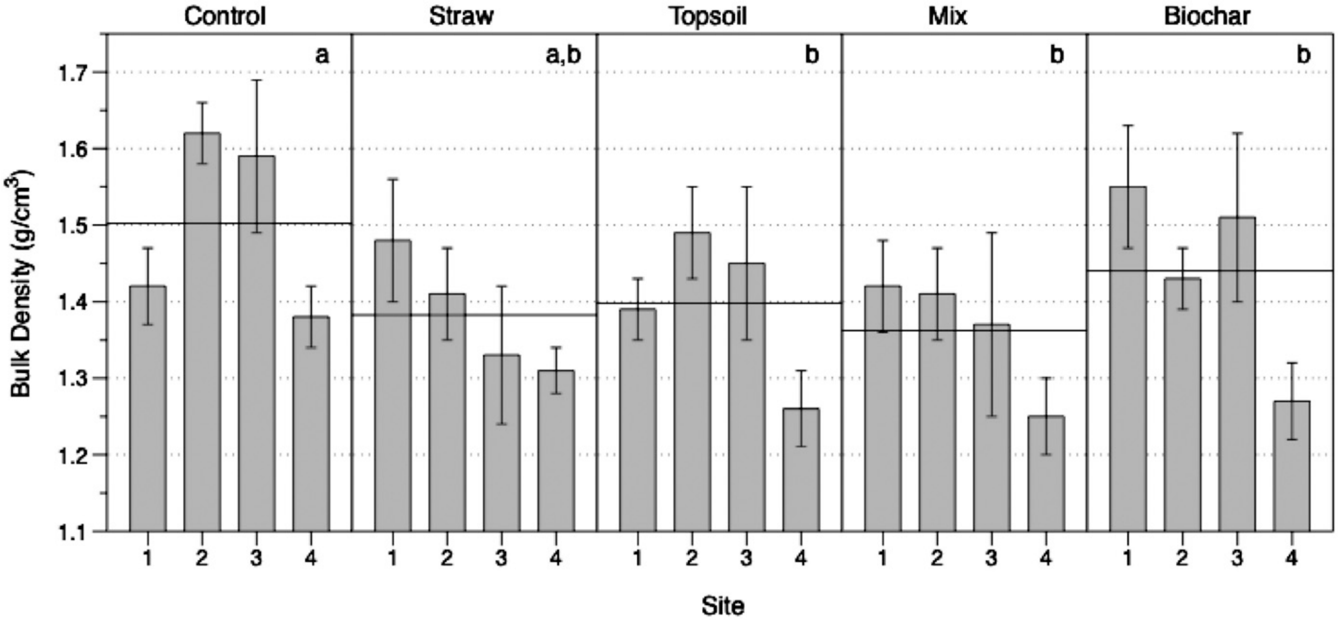

Fig. 4. Bulk density by treatment and site (1–4), averaged across all years (mean + standard error). Dark horizontal lines signify mean BD over all sites and years for each treatment. Letters in the top right corner of each segment summarize the results of post hoc comparisons among treatments. Treatments not linked by a common letter are significantly different.

plant species than Sites 4 and 1 ($p = 0.0069$) (Table 4). Across all sites and treatments, there was a mixture of wetland and upland plants, about 50% of which are considered potential undesirable plants, and 18% of which are considered endangered or threatened (http://plants.usda.gov/wetland.html).

DISCUSSION

Research has revealed that restored and created wetlands may take decades to hundreds of years to develop the important soil attributes of their natural counterparts (Bishel-Machung et al., 1996; Shaffer and Ernst, 1999; Bruland and Richardson, 2006; Ballantine and Schneider, 2009; Hossler and Bouchard, 2010). This indicates that ecosystem function in these wetland sites may be severely limited for much longer than previously anticipated (Shaffer and Ernst, 1999). Therefore, there is a need to establish methods that stimulate the development of these important soil parameters if we are to optimize ecosystem functions of restored and created wetlands.

Soil Properties

Our results show that the addition of biochar and topsoil to wetland soils as a part of the restoration process will help achieve ecosystem function goals within 3 yr of restoration. Plots amended with straw will also likely increase C and N and decrease BD

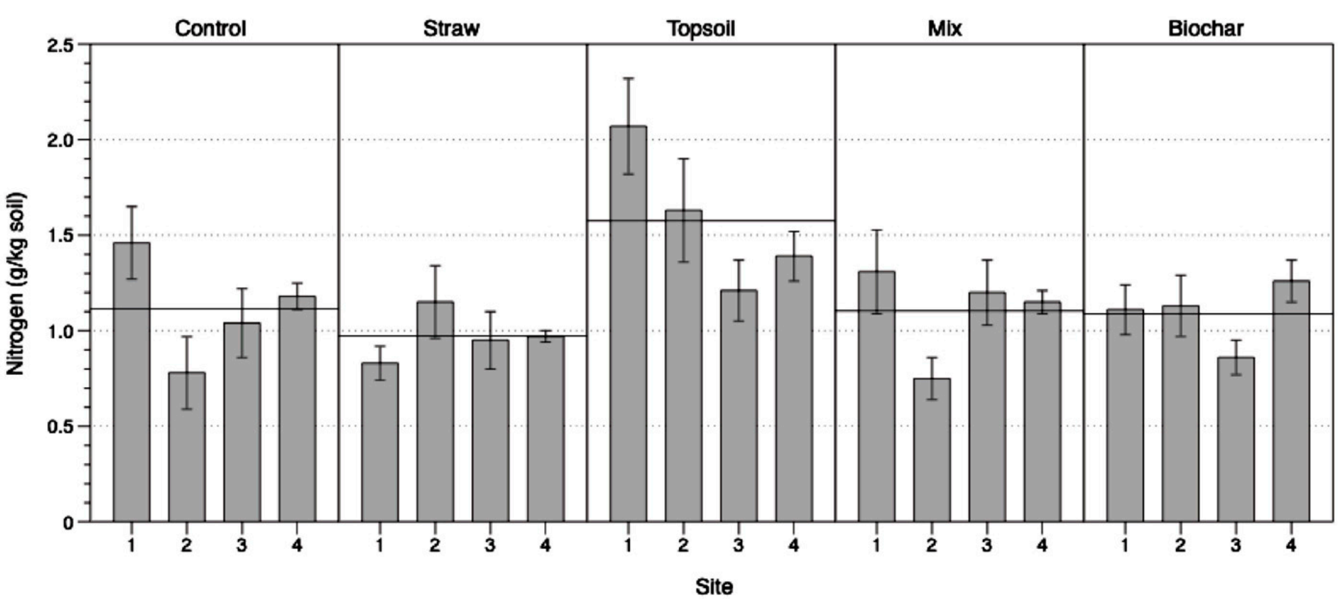

Fig. 5. Soil nitrogen by treatment and site (1–4), averaged across all years (mean + standard error). Dark horizontal lines signify mean soil N over all sites and years for each treatment.

relative to Control plots after more time has passed, and we plan to monitor the development of these properties as straw decomposes and is incorporated into the soil. Of particular importance, biochar and topsoil additions significantly increased soil C. Just 3 yr after restoration, Biochar, Mix, and Topsoil plots had 145 to 165% greater C than Control plots. Topsoil came to have higher C than Mix plots over time, likely as a result of the decomposition of the straw in the Mix amendment. Biochar, however, consistently had the highest C. This may be explained by the recalcitrance of biochar, the least labile soil amendment. The structure of biochar is dominated by a core of aromatic rings, making it highly stable and more resistant to decomposition than straw (Lehmann and Rondon, 2006). Percent C differed by site as well, though these differences are likely due primarily to initial variation in soil C among sites. Specifically, Sites 1 and 2 had higher starting levels of C than Sites 3 and 4, and this trend persisted after the addition of amendments and throughout the study.

The influence of initial site differences on C is reflected by the range of soil C values reported in the wetland restoration and creation literature (Table 5). The mean C of our wetlands across treatments and sites was <60% of the levels found in 44 similar wetland types created in Pennsylvania (Bishel-Machung et al., 1996), but about four times higher than a different mitigation wetland created in Pennsylvania (Stauffer and Brooks, 1997). In this study, Sites 1 and 2 were located only 400 m apart, yet C on their initial topsoil differed by approximately 15%. Determining the influence of amendment treatments on soil C based on comparisons among studies is complicated by differences in site history, restoration methodology, sampling depth, and underlying site characteristics. Nonetheless, it is clear that while amendments can significantly increase soil C, restored and created wetlands still typically have far lower levels than their natural counterparts. In this study, for example, amendment additions significantly increased C, but these increases fell far below the range expected for nearby comparable natural wetlands (~15–25%) (Ballantine and Schneider, 2009). Numerous other investigators report lower C in restored and created wetlands than in natural reference wetlands (Lindau and Hossner, 1981; Craft et al., 1991; Langis et al., 1991; Bishel-Machung et al., 1996; Galatowitsch and van der Valk, 1996; Shaffer and Ernst, 1999; Stolt et al., 2000; Nair et al., 2001; Campbell et al., 2002; Bruland and Richardson, 2005; Ballantine and Schneider, 2009; Hossler and Bouchard, 2010, and references in Table 5). Some authors predict increases in C over time due to accumulation of organic matter, though we did not observe any significant change from 2007 to 2010. The SOM increases were likely inhibited by weeding, which precluded the accumulation of dead plant matter. However, given the relatively slow rate of litter accumulation observed in similar wetlands (Ballantine and Schneider, 2009), it is unlikely that C input from plants would have been substantial over the course of this study.

While it is known that low soil C levels can limit plant establishment and growth (Zedler and Langis, 1991; Stauffer and Brooks, 1997; van der Valk et al., 1999) as well as nutrient cycling and other key soil processes (Groffman et al., 1996), it is unknown what minimum amount of soil C is necessary to achieve equivalent functions of natural reference wetlands or to sufficiently stimulate wetland processes so that functional goals may be met in an acceptable time frame. Because it may be practically infeasible or cost prohibitive to add amendments sufficient to achieve equivalent C levels of natural reference soils within the first few years after restoration, attention should be focused on determining what amount of amendment is necessary to stimulate processes and achieve functional equivalency within a given time.

Table 3. 2008 plant biomass (g/m^2) averaged across all plots per treatment. Mean (Standard Error) are denoted.

Site	Control	Straw	Topsoil	Mix	Biochar
Site 1	857.60 (345.54)	431.46 (99.78)	514.67 (129.01)	505.60 (137.21)	796.80 (194.12)
Site 2	691.73 (258.10)	821.33 (311.75)	904.53 (370.03)	552.53 (128.91)	605.87 (158.25)
Site 4	997.87 (304.19)	426.13 (259.05)	1002.13 (159.45)	788.80 (248.90)	974.93 (143.18)

In addition to jumpstarting soil processes that lead to functional equivalency with natural reference wetlands, adding amendments to wetlands could work as a strategy to sequester C. This is because organic material, such as C-rich topsoil, is less likely to be oxidized to CO_2 if preserved as submerged wetland soil. Anaerobic processes proceed at slower rates than decomposition under aerobic conditions, causing organic matter to accumulate (Ponnamperuma, 1972). Therefore, a wetland's ability to store C is dependent on the submerged status of the soil. Likewise, wetlands that experience prolonged dry periods will lose C by oxidation by aerobic microorganisms. This indicates that in addition to lower levels of C at the time of restoration, Sites 3 and 4 may also have had lower C than Sites 1 and 2 because they had periods where the soil was not submerged. While wetlands are favored as long-term C stores, warming and drying from global climate change put this function at risk. In consistently submerged restored wetlands such as Sites 1 and 2, where the anaerobic soil environment already depresses microbial decomposition rates of organic matter, addition of C through soil amendments may serve as a significant C sink.

Related to site hydrology is soil moisture, which was also a strong predictor of C contents in our study. Topsoil had the highest soil moisture, likely due to its relatively low BD. Low BD indicates the soil has a large amount of pore space, most of which may be filled with water (Reddy and DeLaune, 2008). In contrast, Control plots had a relatively high BD, reflecting that the soil was very dense and that there was minimal porosity (Fig. 4). Correspondingly, Control plots had the lowest soil moisture. In addition to reduced water holding capacity, highly compacted soil can limit mixing and establishment of soil fauna, thereby reducing microbial community development and, ultimately, decomposition and nutrient cycling (Ruiz-Jaen and Aide, 2005). High BD can also reduce root penetration, in turn limiting plant establishment or favoring more aggressive plants with stronger

Table 4. Species list by site. Undesirable and threatened plants are identified, and wetland indicator status (WIS) is noted. Amended plots are indicated with letters as follows: C = Control, S = Straw, T = Topsoil, M = 50:50 mix for straw and biochar, B = Biochar.

Species name	Undesirable or desirable	WIS†	Site 1					Site 2					Site 4				
			C	S	T	M	B	C	S	T	M	B	C	S	T	M	B
Abutilon theophrasti Medik.	Noxious weed in CO, IA, OR, WA	UPL								X							
Acer rubrum L.	Potentially weedy	FAC	X	X			X	X	X	X	X	X			X		
Alisma triviale Pursh	Endangered in NJ and PA	OBL	X	X	X	X	X		X		X	X					
Alnus incana L.	Endangered in IL	NI	X	X		X	X	X	X	X	X						
Ambrosia artemisifolia L.	Noxious weed in IL, MI, OR	FACU	X	X	X	X	X	X		X	X	X	X	X	X	X	X
Asclepias incarnata L.	Potentially weedy	OBL							X	X	X						
Symphyotrichum lanceolatum (Wild.) G.L. Nesom		FACW							X								
Bidens frondosa L.	Weedy	FACW	X	X	X	X	X	X	X	X	X	X					
Calamagrostis canadensis (Michx.) P. Beauv	Endangered in KT	FACW+		X	X	X	X			X			X				
Carex spp.		OBL FAC FAC	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Convolvulus sepium L.	Noxious weed in AR, TX Endangered in NJ	FAC-											X	X	X	X	
Daucus carota L.	Noxious weed in IA, MI, OH, WA	NI												X	X	X	X
Eleocharis obtusa (Wild.) Schult.	Endangered in PA	OBL	X	X	X	X	X	X	X	X	X	X					
Epilobium leptophylium Raf.	Threatened in TN	OBL										X			X		
Equisetum spp.		OBL FACW FAC								X							
Fraxinus pennsylvanica Marsh.	Weedy	FACW														X	
Galium palustre L.	Endangered in OH special concern in TN	OBL			X												

Species name	Weedy or threatened	WIS	Site 1					Site 2					Site 3				
			C	S	T	M	B	C	S	T	M	B	C	S	T	M	B
Gramineae		OBL FACW FAC FACU UPL		X	X	X	X	X	X	X	X	X	X	X	X	X	X
Hypericum mutilum L.		FACW													X		
Hypericum kalmianum L.	Endangered in IL Threatened in OH	FAC											X				
Juncus spp.		OBL FACW FAC					X										
Lobelia inflata L.		FACU											X		X		
Lonicera spp.	Potentially weedy	OBLFACWFAC								X							
Ludwigia palustris (L.) Elliott		OBL	X	X	X	X	X	X	X	X	X	X					
Lysimachia spp.	Potentially weedy	OBL FACW FAC FACU UPL												X			
Lythrum salicaria L.	Widespread noxious weed	FACW+	X	X	X	X	X	X	X	X	X	X					
Onoclea sensibilis L.		FACW													X		
Oxalis stricta L.	Weedy	UPL		X				X				X	X	X	X	X	X
Panicum virgatum L.	Potentially weedy	FAC						X	X		X	X	X	X	X	X	
Phalaris arundinacea L.	Noxious weed in CT, MA, WA	FACW+						X	X	X	X	X	X	X	X	X	X
Plantago lanceolata L.	Weedy	UPL											X	X	X	X	
Plantago major L.	Weedy	FACU						X	X	X	X	X	X	X	X	X	X
Polygonum amphibium L.	Weedy	OBL	X	X	X	X	X	X	X	X	X	X					

Continued next page

Table 4 (continued).

Species name	Weedy or threatened	WIS	Site 1					Site 2					Site 3				
			C	S	T	M	B	C	S	T	M	B	C	S	T	M	B
Polygonum pensylvanicum L.	Weedy	FACW		X	X								X	X	X	X	X
Potentilla simplex Michx.	Weedy	FACU											X	X	X	X	X
Ranunculus spp.		OBL FACW FAC FACU UPL						X	X	X	X	X			X		X
Rubus allegheniensis Porter		FACU-														X	
Rumex orbiculatus A. Gray	Weedy	OBL						X	X	X	X	X		X		X	
Saxifraga spp.		OBL FACW FAC													X		X
Scirpus spp.		OBL FACW+	X	X	X	X	X	X	X			X	X		X		
Sium suave Walter		OBL			X												
Solidago spp.	Potentially weedy	OBL FACW FAC FACU UPL	X	X	X	X	X						X	X	X	X	X
Taraxacum officinale F.H. Wigg.	Weedy	FACU-						X	X	X	X		X	X	X	X	
Trifolium hybridum L.		FACU-		X									X	X	X	X	X
Trifolium pratense L.		FACU-											X	X	X	X	X
Trifolium procumbens L.	Potentially weedy	NA											X				
Typha angustifolia L.		OBL	X	X	X	X	X		X	X	X	X					
Typha latifolia L.	Weedy	OBL		X	X	X	X	X	X	X	X	X					
Typha × *glauca* Godr. (pro sp.)	Potentially weedy	OBL		X													
Veronica americana Schwein. ex Benth.	Endangered in IL Extirpated in IN Historical in KT Special Concern in TN	OBL								X	X	X		X	X		

† Wetland Indicator Status for Region 1 (WIS) found at: http://plants.usda.gov/wetland.html. OBL = Obligate Wetland-occurs almost always (estimated probability 99%) under natural conditions in wetlands. FACW = Facultative Wetland-usually occurs in wetlands (estimated probability 67–99%), but occasionally found in non-wetlands. FAC = Facultative- Equally likely to occur in wetlands or non-wetlands (estimated probability 34–66%). FACU = Facultative Upland-usually occurs in non-wetlands (estimated probability 67–99%), but occasionally found on wetlands (estimated probability 1–33%). UPL = Obligate Upland-occurs in wetlands in another region, but occurs almost always (estimated probability 99%) under natural conditions in non-wetlands in the regions specified. NI = No indicator-Insufficient information was available to determine an indicator status.

root systems. In our study, BD was decreased by the addition of soil amendments. All plots were tilled equally, so the immediate decrease in BD of the amended plots was due to the mixing in of lighter material (topsoil, straw, and/or biochar). Over time, the lower density of the amended plots may allow for further mixing by soil fauna, creating a positive feedback mechanism.

As was the case for soil C, mean levels of BD improved with the use of amendments, but were still far from those of natural wetlands. The mean BD in our sites over all treatments was 1.42 g/cm^3, compared to 0.2 to 0.3 g/cm^3 in comparable natural wetlands (Mitsch and Gosselink, 2000; Ballantine and Schneider, 2009). The mean BD was, however, similar to other recently restored and created wetlands (Table 5). Studies of amendments in various soil types and land-uses have shown a reduction in compaction with an increase in amendment level (Bendfeldt et al., 2001a; Cogger, 2005; Bruland et al., 2009). It is unlikely, however, that the resources of a given project will be sufficient to add enough organic matter to achieve natural reference levels of BD. Therefore, it would be useful to know what levels are necessary to meet functioning criteria. Over time, SOM accumulation in submerged soil will prevent aerobic decomposition of litter and surface SOM. As decomposed material is incorporated into the soil, BD will gradually decrease. These changes are slow, however, and rarely detectible over short timeframes.

Soil N may also be slow to recover in restored wetlands. Unlike a previous study of a restored riparian wetland in North Carolina by Sutton-Grier et al. (2009), N did not decrease over time either, but remained constant. It is possible that the N in our topsoil was in a more recalcitrant form than the compost mix of topsoil, wood chips, and pathogen-free wastewater biosolids used by Sutton-Grier and colleagues. This would make it less available for microbial and plant use and therefore soil levels would not decrease as quickly over time.

Topsoil was the only amendment to significantly increase soil N. This is unsurprising because topsoil contained more N than straw or biochar. The higher N in Topsoil plots could ben-

Table 5. Literature comparison of soil properties among restored/created and natural wetlands, including the cited study, wetland classification (WL Type), location of study sites (Location), years since restoration/creation vs. natural wetlands (Age), amendments used as part of restoration methodology (Amendments), number of wetlands examined in the study (No. WL), depth of the soil samples (Depth), and response variables. When necessary, percent carbon was calculated from percent soil organic matter (SOM) using the formula SOM = 2.0 × soil organic carbon (Mitsch and Gosselink 2000). All values are means.

Study	WL Type	Location	Age	Amendments	No. WL	Depth	Organic C	Total N	Bulk density
						m	—— g kg^{-1} ——		g cm^{-3}
Ahn and Peralta, 2009	palustrine shrub/scrub	Virginia	1,5,8	0.2 m topsoil	3	0.1	15.8	1.4	1.01
Ballantine and Schneider, 2009	palustrine emergent	New York	3–5	no			30.3		1.10
Bishel-Machung et al., 1996	slope, riverine, depression, fringe	Pennsylvania	1–8	sometimes w/ 0.1–0.3 m topsoil	44	0.05	31.0	1.1	1.15
Bruland and Richardson, 2006	headwater riverine, mainstem riverine, non-riverine mineral soil flat, non-riverine organic soil flat	North Carolina	3–9	not specified	11		59.0		
Bruland et al., 2009	non-tidal forested	Virginia	5	pre-amendment	1	0–0.2	11.4	0.8	1.25
Campbell et al., 2002	palustrine emergent	Pennsylvania	2–10	not specified	12	0.05	24.0		1.20
Card et al., 2010	prairie potholes	Saskatchewaan Alberta	1–3	no	28	0.06	47.60		
Craft et al., 2002	created brackish-water marsh	North Carolina	15	no	1	0.3	1866 (kmol/ha)	1.65	1.21
Craft et al., 2002	natural brackish-water marsh	North Carolina	natural	natural	1	0.3	10270 (kmol/ha)	5.42	0.13
Fennessy et al., 2008	created, emergent marshes	Ohio	1–9	not specified	10	0.1	24.0	2.6	1.75
Fennessy et al.,2008	comparable natural wetlands	Ohio	natural	natural	9	0.1	75.5	11.3	0.72
Galatowitsch and van der Valk, 1996	emergent wet meadow	Minnesota Iowa	3	not specified	10		38.3		0.90
Hogan et al., 2004	palustrine emergent	Maryland	5,5,12	not specified	3	0.13	12.0		1.10
Hogan et al., 2004	palustrine forest	Maryland	natural	natural	3	0.13	57.0		0.90
Hossler and Bouchard, 2010	palustrine emergent	Ohio	3–8	1 w/wetland soil	5	0–0.05	0.2–0.45		
Hossler and Bouchard, 2010	palustrine emergent	Ohio	natural	natural	4	0–0.05	0.5–2.4		
Langis et al., 1991	constructed salt marsh	California	4	fertilzed w/urea	1	0.08		10.99	
Langis et al., 1991	natural salt marsh	California	natural	natural	1	0.08		0.13	
Lindau and Hossner, 1981	intertidal salt marsh	Texas	2	dredge substrate	1	0.3	1.5	0.95	
Lindau and Hossner, 1981	natural salt marsh	Texas	natural	natural	3	0.3	2–7	2.27–5.88	
Mitsch and Gosselink, 2000	organic wetland soil		natural	natural			120–200		0.2–0.3
Nair et al., 2001	phosphate-reclaimed wetlands	Florida	1–16	not specified	5	0.1	0.56	0.04	0.70
Nair et al., 2001	adjacent native wetlands	Florida	natural	natural	5	0.1	1.93	0.09	0.40
Shaffer and Ernst, 1999	freshwater palustrine	Oregon	5	not specified	50	0.05	29.2		1.1–1.6
Stauffer and Brooks, 1997	not specified, mitigation site	Pennsylvania	1	unamended control plots	1	0.15	4.5	0.5	
Stauffer and Brooks, 1997	not specified, mitigation site	Pennsylvania	1	salvaged marsh surface soil	1	0.15	27.5	2	
Stolt et al., 2000	palustrine forested shrub/scrub	Virginia	4–7	not specified	3	0.05–0.15	0.093		
Taylor and Middleton, 2004	coal slurry pond	Illinois	5	no	1	0.05–0.15	6.0		
This study	palustrine emergent	New York	3	see text	4	0.1	23.02	1.2	1.42

efit both plant and microbial growth, especially during community establishment at a newly restored or created site. Our results are similar to other studies reporting that soil N increased with organic matter additions (O'Brien and Zedler, 2006; Bailey et al., 2007; Sutton-Grier et al., 2009), but as was the case for C and BD, these levels were not equivalent to those found in comparable natural wetlands (O'Brien and Zedler, 2006; Bailey et al., 2007; Fennessy et al., 2008; Sutton-Grier et al., 2009). We expect N levels in the soil will gradually increase as plants grow and decompose and organic matter accumulates in the soil. The rate of increase will depend on how quickly the bacteria and processes of the N cycle become established. How long this process will take is unknown. In the future, we plan to investigate what levels of key chemical and physical soil variables are necessary to jumpstart desirable wetland processes. This will help to establish achievable target levels of C, BD, and N in similar restored and created wetlands. This knowledge for different types of wetlands would provide useful guidance for practitioners eager to improve wetland projects and stimulate wetland functions.

Plant Properties

In contrast to soil properties, plant communities in our sites were quick to recover to natural levels. This finding is consistent with other studies showing that plant communities return to desired reference levels faster than other wetland parameters (Ballantine and Schneider, 2009). Plants in all four sites established themselves rapidly, and by 2008, most of the plots were covered in dense growth. The mean biomass of 724.8 g/m^2 was within the range of aboveground biomass reported for similar wetlands sites, both natural and restored/created. Mitsch and Gosselink (2000) stated that aboveground biomass in natural inland freshwater marshes is typically 500 to 5500 g/m^2. In created marshes similar to ours, Cole et al. (2001) reported aboveground biomass ranging from 676 to 1694 g/m^2.

While plant biomass did not differ among treatments or sites 3 yr after restoration, species diversity was higher in Site 2 than in Sites 1 and 4. This is likely due to Site 2's closer proximity to older restored wetlands and relatively diversely populated fallow fields (personal observation). In particular, the fields neighboring the wetland sites appeared to supply volunteer plants, as evidenced by the large proportion of upland species. Site 2 had more desirable endangered plants than Sites 1 and 4, but it also had more undesirable species that are known to outcompete neighboring plants and dominate the system. Although rapid plant establishment is desired, it appears that most colonization in our plots was by undesirable vegetation including some invasive plant species. Invasive plants are those that rapidly and aggressively spread by expanding into native plant communities (Rejmanek and Richardson, 1996; Galatowitsch et al., 1999; Richardson et al., 2000). The rapid colonization by undesirable and invasive species was reported in other restored and created wetlands as well (Cole et al., 2001; Zedler and Kercher, 2004; Spieles, 2005; Matthews and Endress, 2008).

Topsoil, straw, and/or biochar amendments did not affect plant biomass or diversity in this study. We were particularly surprised that Topsoil plots did not have higher plant diversity than other treatments. We expected topsoil would act as a seed bank, providing seeds and propogules that may not have otherwise become quickly established. It appears, however, that the proximity of nearby fields to the plant plots enabled windborne volunteers to easily colonize. These results again reflect the importance of site selection in wetland restoration and creation. Sites that are located in close proximity to seed sources, be they aggressive invasive plants or sensitive wetland natives, are likely to quickly become colonized by those plants.

We also expected biochar additions to cause lower abundance of non-native species than native species due to effects on heterotrophic, symbiotic, and pathogenic soil organisms, as well as a potential ability to sequester allelochemicals. This hypothesis was based on a study showing that activated C additions in ex-arable fields dominated by non-native plants in Washington have been shown to increase native plant dominance by decreasing non-native abundance 6 yr after addition (Kulmatiski, 2011). Like activated C, biochar has high microporocity and indiscriminately binds organic molecules through physical adsorption and ionic bonding (Lehmann and Joseph, 2009), but 3 yr after addition, there was no different in native or non-native plant abundance from other plot types.

We also hypothesized that plots amended with biochar would have higher plant biomass because, in agricultural systems, biochar has been shown to revive depleted soils and substantially increase crop growth (Glaser et al., 2002; Lehmann et al., 2003; Oguntunde et al., 2004; Rondon et al., 2007; Steiner et al., 2007). However, there were no significant effects of biochar on plant biomass observed in this study. This could be because background soil conditions were sufficient to support plant growth or the particular biochar used did not address soil constraints. It is also possible that the beneficial results of biochar application are impeded in wetlands due to the anoxic nature of submerged soils. For example, biochar may stimulate crop growth in part because it has been shown to increase soil pH by up to 1.0 unit and decrease available aluminum. In wetland soils, however, pH is generally close to neutral and aluminum toxicity is rarely a problem.

The recovery of vegetation is traditionally used as the only measure to assess the success of restored and created wetlands (Wilson and Mitsch, 1996; Laidig and Zampella, 1999; Perry and Hershner, 1999; Young, 2000; Cole et al., 2001; Matthews and Endress, 2008; Ahn and Peralta, 2009). The use of plant biomass and diversity as a surrogate for wetland function, however, is misleading (Breaux and Serefiddin, 1999; Ruiz-Jaen and Aide, 2005; Spieles, 2005; Ahn and Peralta, 2009). Numerous studies demonstrate that if the goal of a wetland restoration or creation project is to establish a wetland that is self-supporting and resilient to perturbation (SER, 2004), ecosystem processes essential for long-term persistence must be assessed. Just as there are vegetative success criteria for restored and created wetlands, our results support the mounting evidence that there should be success criteria based on soil properties (Ballantine and Schneider, 2009; Bruland et al., 2009). Not only can poor soil conditions lead to low plant survival or invasion by exotic species (Zedler and Kercher, 2004), measures of soil development serve as indicators for overall wetland health and function. Unfortunately, at present, very few wetland restoration, creation, or mitigation projects require a soil restoration component in the project design. With the development of success criteria that include soil measures such as C, BD, and N, projects will be encouraged to incorporate techniques that have been shown to improve soil properties at similar sites. The next step forward is to improve our knowledge of the effects of soil amendments on wetland development. Strategic field experiments that examine the impacts of alternative amendment uses (amendment type, amount, site influence, incorporation techniques) on key ecosystem functions will improve our understanding of both how ecosystems function and how to improve the effectiveness of future restoration projects (Zedler, 2000, 2003; Sutton-Grier et al., 2009).

CONCLUSIONS

While none of the amendments used in this study affected plant properties, amendments did influence the key soil properties of C, N, and BD. If the success of this project were determined solely from plant biomass data, the evaluation would deem the restorations successful and the use of amendments unnecessary. This conclusion would miss the beneficial effect of amendments on soil quality. Furthermore, both plant and soil responses were significantly influenced by differences among sites. This emphasizes the importance of evaluating site parameters such as background soil conditions and proximity to desirable and undesirable volunteer plants in choosing where restoration should take place. Finally, while topsoil and biochar amendments in particular improved soil conditions, key soil properties were still substantially lower than comparable natural wetlands. We plan to examine how this influences long-term development and function in our sites over the coming years.

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