

Creating Wetlands: Primary Succession, Water Quality Changes, and Self-Design over 15 Years

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The succession of vegetation, soil development, water quality changes, and carbon and nitrogen dynamics are summarized in this article for a pair of 1-hectare flow-through-created riverine wetlands for their first 15 years. Wetland plant richness increased from 13 originally planted species to 116 species overall after 15 years, with most of the increase occurring in the first 5 years. The planted wetland had a higher plant community diversity index for 15 years, whereas the unplanted wetland was more productive. Wetland soils turned hydric within a few years; soil organic carbon doubled in 10 years and almost tripled in 15 years. Nutrient removal was similar in the two wetlands in most years, with a trend of decreased removal over 15 years for phosphorus. Denitrification accounted for a small percentage of the nitrogen reduction in the wetlands. The wetlands were effective carbon sinks with retention rates of 1800–2700 kilograms of carbon per hectare per year, higher than in comparable reference wetlands and more commonly studied boreal peatlands. Methane emission rates are low enough to create little concern that the wetlands are net sources of climate change radiative forcing. Planting appears to have influenced carbon accumulation, methane emissions, and macrophyte community diversity.

Keywords: ecosystem experiment, long-term ecosystem research, freshwater marshes, riparian wetlands, Olentangy River Wetland Research Park

Over a decade ago, Mitsch and colleagues (1998) described in *BioScience* the first three years of a wetland planting and self-design experiment at the Olentangy River Wetland Research Park (ORWRP; figure 1) on Ohio State University's campus in Columbus. In that article, it was hypothesized after the three years of comparison of the two created riverine 1-hectare (ha) wetland basins that

The introduction of plant species, although often necessary to meet legal requirements of early plant cover in created wetlands, may not always be necessary to start wetlands on a trajectory toward becoming a functional ecosystem.... A hydrologically open created wetland can develop, through self-design, a diverse assemblage of species even where no propagules existed before.... Nature's functions and subsequent services are often best provided when natural processes are mostly in control and are given room to operate.

p. 1027–1028

In that article, it was further suggested that “a long-term experimental ecosystem approach appears to be a viable

method for investigating general questions such as ‘Can we create and restore wetlands?’ and ‘How much human intervention is required?’” (p. 1027).

In the present article, we compare those two experimental wetlands at the ORWRP more than a decade after that article (Mitsch et al. 1998) was published and for 15 years after the wetlands were created. For the purposes of this study, we compared these wetlands both with each other and with a reference (control) natural flow-through wetland in Ohio. The experimental wetlands were not designed exclusively for water quality improvement, nor were they designed for any specific biological population. Rather, they were designed to be sustainable, multiple-ecosystem-service, flow-through freshwater wetland ecosystems in a riverine setting for use in research. The inflow rate is almost 10 centimeters (cm) per day, two to four times that of traditional wastewater treatment wetlands and more similar to riverine floodplain sloughs. The source of the water for the wetlands—the Olentangy River—has nutrient and sediment concentrations sufficient for the investigation of long-term trends in wetland retention without excessive overload or saturation within a few years.

Wetland ecosystem services and the mitigation of lost functions

Wetlands are among the most important yet vulnerable ecosystems on the planet (Mitsch and Gosselink 2007). They are keenly tuned to their climate, their watersheds and landscape geomorphology, and, in some cases, their coastline processes. Wetlands provide many ecosystem services, including biodiversity support, flood and storm mitigation, and water quality improvement, particularly through nutrient reduction, which has led some to suggest that wetlands provide more services to human society than does almost any other ecosystem (Costanza et al. 1997, Mitsch and Gosselink 2007). More recently, there have been many suggestions that wetlands provide buffers to climate change through sequestration and the permanent burial of carbon (Mittra et al. 2005, Lenhart 2009, Mitsch et al. 2010). Yet wetlands are now described as among the largest natural sources of the greenhouse gases methane (CH_4) and nitrous oxide (N_2O) (Whalen 2005, Bergamaschi et al. 2007, Bloom et al. 2010) in climate change discussions. Created and restored wetlands need to be evaluated in terms of their balance between carbon sequestration and greenhouse gas emissions.

There have been many studies in which the effectiveness and functioning of specific created

and restored wetlands have been evaluated—particularly in the United States, in reference to compliance for the mitigation of wetland loss required by the amended section 404 requirements of the US Clean Water Act of 1972 (Mitsch and Wilson 1996, Wilson and Mitsch 1996, Zedler 2000, Robb 2002, Spieles 2005, Spieles et al. 2006, Fennessy et al. 2008, Matthews and Endress 2008, Gutrich et al. 2009)—but most of these have been relatively short-term studies or studies in which wetlands were evaluated synoptically, with infrequent measurements, and were primarily of wetland structure, not function. This wetland-loss mitigation policy was examined by a National Research Council (2001) panel

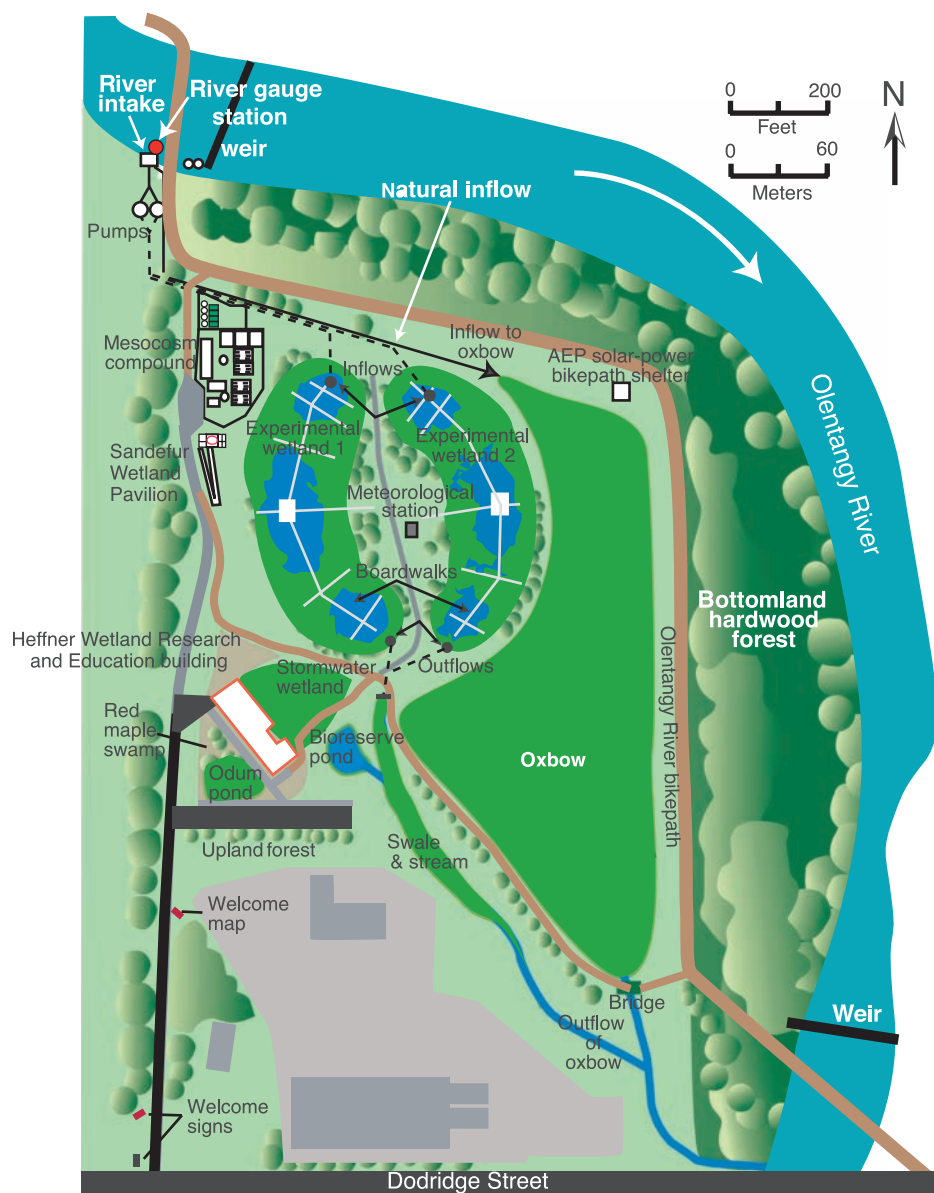


Figure 1. The Wilma H. Schiermeier Olentangy River Wetland Research Park at The Ohio State University, in Columbus. The two 1-hectare experimental wetlands received water from the Olentangy River over the 15-year study period. They received pumped water from a river shown in the upper left corner of the map and returned the flow to the river via a swale in the lower right of the map.

Identical hydrologic inflows, proportional to the adjacent river flow, were maintained in the two wetlands for that entire 15 years. We extended many of the same functional characteristics used in the 1998 article and investigated several new biogeochemical processes. Overall, in this study, we provide insight into the strategy of managing wetland ecosystems in primary succession; the long-term trends in ecosystem services such as water quality improvement; the importance of hydrologic pulsing; and, with more recent data, the relative magnitude of greenhouse gas emissions, carbon sequestration, and denitrification in newly created wetlands.

that concluded that the United States had not yet achieved the goal of no net loss of wetland function, although considerable progress had been made since the policy started in the 1980s. A recent reassessment by several of the members of that National Research Council panel concluded that although significant progress has continued in the regulation and enforcement of the Clean Water Act with regards to mitigating wetland loss, several improvements could be implemented, including extramural reviews of wetland mitigation projects and greater emphasis on adaptive management and the avoidance or minimization of wetland loss (Gardner et al. 2009).

Despite all of the attention given to the legal aspects of creating and restoring wetlands, there is generally poor scientific understanding of exactly what functions and ecosystem services to expect from created and restored wetlands, how these compare with those of “natural” wetlands, over what time frames these functions and services manifest themselves, and how we can collectively measure and compare these functions and services. Federal policy generally requires only five years of simplistic monitoring of wetland structure for wetlands created or restored for mitigating wetland loss. Long-term, controlled, whole-ecosystem research of the functions of created and restored wetlands is rare. Although the experimental wetlands described here have river water pumped through them, the pumping rate was always maintained as a function of the adjacent river flow, so in many ways, these wetlands may represent reference conditions of what to expect in riverine freshwater wetlands. The trends and amplitudes over 15 years for nutrient retention, greenhouse gas emissions, carbon sequestration, and plant succession are reasonable conditions to expect in other created and restored riverine wetlands over a similar period.

The experimental wetlands

Two 1-ha experimental wetlands (figure 1) were created over 1993 and 1994 at the 20-ha Wilma H. Schiermeier ORWRP, a complex of created and natural freshwater riverine wetlands located on the campus of The Ohio State University in Columbus. The two wetlands have been extensively studied since water was first added in 1994 (year 1), with research results regarding water quality, plant community structure and function, soil development, sedimentation, and gas exchange (see Mitsch et al. 1998, 2005a, 2005b, and a complete list of publications at <http://swamp.osu.edu/Research/index.html>). The urban-located ORWRP is in the easternmost edge of the Central Plains portion of Eastern Temperate Forest ecological region (biome) of North America and is close to the intersection of three level II ecoregions in that biome: Mixed Wood Plains, Central Plains, and Appalachian Forests. The original alluvial soil type at the site is the Ross series—a deep, dark, and well-drained silt loam, silty clay loam, or loam that forms on floodplains, commonly on the high part and in alluvium and outwash. Soils in the experimental wetland basins were

nonhydryc before the wetlands were created. The western basin was planted with 13 native species of macrophytes in May 1994 (see supplemental material A, available online at <http://dx.doi.org/10.1525/bio.2012.62.3.6>) and is therefore labeled the *planted wetland*, whereas the eastern basin was allowed to be colonized naturally (Mitsch et al. 1998) and so will be called the *unplanted wetland*.

Water from the adjacent Olentangy River was first added by a pumping system to the wetland basins on 4 March 1994; pumping has continued uninterrupted except for brief pump and inlet repairs and electric outages for over 15 years. Water is pumped continuously from the Olentangy River, a third-order stream in the agriculture-dominated Scioto River Watershed of central Ohio, according to a formula relating pumping rates to river stages. Therefore, the water in the wetlands is deep when there is river flooding and shallow when the river is at base flow conditions.

Hydrology and hydroperiods. For the 15-year period described here, the annual inflow rates into the wetlands ranged from on average from 626 to 1552 cubic meters (m^3) per day. Over the 15 years of this study, the inflows for the planted and unplanted wetlands averaged $990 \pm 44 m^3$ per day and $986 \pm 45 m^3$ per day, respectively, and were identical for each year and for the overall 15 years ($p = .569$). These flow rates are equivalent to hydraulic loading rates of 9.9 cm per day, or 36 m per year for the 1-ha wetland basins. Because pumped inflow is programmed to be a function of river flow, the wetlands' hydroperiods are variable (figure 2), with high levels generally in the wet season (December–June) and low-water levels in the dry season (July–November).

Soil development. A 10×10 m grid was established in the two wetlands in 1993, prior to water's being flooded into the wetlands (Anderson et al. 2005). Experimental wetland soils were then sampled in 1993 (before flooding), 1995 (after 1.5 years after flooding), 2004 (after 10 years after flooding), and 2009 (after 15 years after flooding) to estimate physical and chemical changes. The measurements in 2004 and 2009 were both used to estimate the annual accumulation of carbon. Little change in carbon and nutrient content has occurred in antecedent soils, and most accumulation occurred from the accretion of mineral and organic sediment on top of antecedent soils (Anderson et al. 2005). Sediment depths and density were estimated by extracting soil cores at the 10-m grid points throughout the two wetlands using a 10-cm-diameter steel corer to a maximum of 16 cm or until the antecedent soil layer was reached (Anderson et al. 2005). A total of 46 of the 2004 soil cores and 36 of the 2008 soil cores (distributed throughout the wetlands) were analyzed for their total carbon content, which was used with sediment depths and bulk densities to estimate the total carbon accumulation in open water and in the emergent sections of the wetlands.

Carbon in the upper soil layers in the experimental marshes increased by an average of 14% per year and

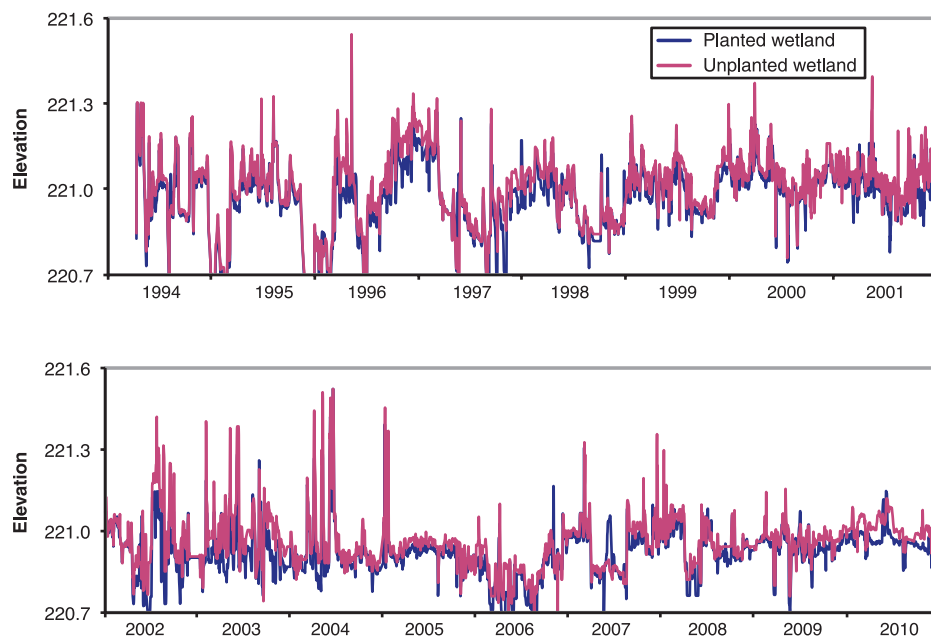


Figure 2. Water stage in the experimental wetlands measured each January from 1994 through 2008 in meters above sea level. The depth in the wetlands generally averaged between 15 and 60 centimeters.

more than doubled in the first 10 years after the wetlands' creation, from 16 to 38 grams (g) per kilogram (kg) (table 1). Soil carbon increased by another 8% over the last five years (2004–2009), to an average of 41 g per kg. By 1995, 17 months after the wetlands were created, 78% of the soil samples in the top 8 cm and 24% of the soil samples in depths of 8–16 cm had Munsell-code chroma values less than or equal to 2, a standard indicator of hydric soils (Anderson et al. 2005). By 2004 (10 years after the wetlands were created), all soil samples from the upper 16 cm in the experimental wetlands were hydric.

Macrophyte cover, community diversity, and productivity. The macrophyte-dominant community cover was estimated each year in middle to late August from color aerial photography and ground-truth verifications. The maps for each year were normalized to a basin map of a standard size, using geographic information system software. A 10 × 10 m grid system marked with permanent, numbered white poles was used to identify the locations of plant communities in each wetland. A macrophyte community diversity index (CDI), described by Mitsch and colleagues (2005a) and in supplemental material B, was used to quantify diversity in the wetland basins. Overall, 22 different vegetation communities, including open water and algal mats, were identified during the 15-year study and were generally named for the dominant taxa in the community.

Aboveground biomass was measured each August as an estimate of aboveground net primary productivity (ANPP), beginning in 1997, by the direct aboveground harvesting of 16 1-m² plots in each wetland along sampling boardwalks.

For the 1994–1996 data, ANPP was estimated from fewer plots. The aboveground biomass in each plot was clipped at the soil surface and weighed immediately. Subsamples were taken to the lab and dried to a constant weight to estimate dry:wet ratios.

Vegetation patterns in the experimental wetlands changed dramatically over the 15 years, with dramatic fluctuations in macrophyte communities and a gradual development of wetland forested communities on the periphery of the basins (figure 3). The distribution of macrophyte communities—and particularly those of the clonal-dominant macrophyte *Typha* spp.—were variable from year to year (figure 4a). The planted wetland, in its 15th growing season (2008) was dominated by the following five emergent

communities: *Sparganium eurycarpum* Engelm, *Typha* spp. (*T. angustifolia* L./*T. latifolia* L./*Typha* X *glauca*), *Scirpus fluviatilis* (Torr.) M.T. Strong, *Schoenoplectus tabernaemontani* (C.C. Gmel.) Palla, and *Leersia oryzoides* (L.) Sw. The macrophyte communities found from time to time in the deepwater areas (more than 30 cm deep) were *N. lutea* and *Potamogeton pectinatus*. Of the seven dominant emergent plant taxa listed above, five were planted in the planted wetland in 1994. In contrast, the unplanted wetland, in its 15th growing season, had three dominant plant communities: *Typha* spp., *S. tabernaemontani*, and *L. oryzoides*. *Phragmites australis* (Cav.) Trin. ex Steud. recently invaded the western edge of this wetland but still covered only a small fraction of it. There was always more area covered by *Typha* spp. in the naturally colonized (unplanted) wetland than in the planted wetland, except during year three, when 4% of the planted wetland and only 2% of the unplanted wetland were covered by *Typha*. In 2008, *Typha* cover, which included *T. angustifolia*, *T. latifolia*, and the hybrid *T. X glauca*, covered 26% of the planted wetland and 32% of the unplanted wetland. That represents the highest percentage of cover by this clonal-dominant species in the planted wetland but is well below the 56% cover of *Typha* seen in the naturally colonized (unplanted) wetland in 1999 (see figure 4a).

Thirteen macrophyte species were introduced to the planted wetland in May 1994, and the unplanted wetland was left to be naturally colonized (Mitsch et al. 1998). Plant community diversity, estimated by our CDI measure, was consistently higher ($p = 0$) in the planted wetland after 1995 (year 2) and generally increased in both wetlands for the first 10 years after they were created (figure 5a). The CDI dropped

Table 1. Changes in the upper 8–10 centimeters of soil in the experimental wetlands at Olentangy River Wetland Research Park.

Year	Wetland age (years)	Bulk density				Percentage of soil samples with chroma values less than or equal to 2				Soil carbon				
		Planted wetland		Unplanted wetland		Planted wetland		Unplanted wetland		Planted wetland		Unplanted wetland		
		Mean (g per cm ³)	Standard error (g per cm ³)	Number of samples	Standard error (g per cm ³)	Mean (g per cm ³)	Standard error (g per cm ³)	Number of samples	Standard error (g per cm ³)	Mean (g per kg of soil)	Standard error (g per kg of soil)	Number of samples	Standard error (g per kg of soil)	
1993 ^a	-1	1.30	0.01	19	1.29	0.01	21	0	16	0.1	19	16	0.2	21
1995 ^a	1	1.00	0.01	19	0.73	0.01	21	78	20	0.3	19	20	0.3	21
2004 ^b	10	0.53	0.02	33	0.49	0.03	36	100	39	1.0	22	38	2.0	24
2009	15	0.60	0.02	17	0.71	0.04	19	100	36	1.6	17	46	2.8	19

cm³, cubic centimeters; g, grams; kg, kilograms.

^aData from Nairn (1996).

^bData from Anderson and colleagues (2005) and Anderson and Mitsch (2006).

in both wetlands in years six and seven (1999–2000) because both wetlands became dominated by *Typha* spp. That brief domination was cut short by an equally dramatic muskrat eat-out over the winter of 2000–2001, which resulted in both wetlands' becoming mostly open water, with their lowest CDI since year two (figure 5a). After that brief decline, the CDI recovered dramatically, mostly from soft-stemmed bulrush (*S. tabernaemontani*), which naturally revegetated from an accumulated seed bank. The CDI more or less leveled at 1.6 in the planted wetland and 1.3 in the unplanted wetland for the last 4 years of the study (years 12–15, 2005–2008).

Plant richness data over time (table 2; see also supplemental material C) reflect the rate at which plants other than the 13 planted species came into both wetlands. By 1996, three growing seasons after planting, there were 67 plant species in the planted wetland and 56 in the unplanted wetland (a difference of 11 species). The number of species increased to 96 in the planted wetland and 87 in the unplanted wetland by year five (1998), a difference in richness of 9 species between the two wetlands. Of these species found after five years, 9 of the 13 originally planted species remained in the planted wetland, whereas only 2 of those species were found in the unplanted wetland (table 2). In 2008, the wetlands' 15th year, there were 101 plant species in the planted wetland and 97 species in the unplanted wetland. The difference in the wetlands had decreased further to four species. The same number of originally planted species as that from 10 years earlier remained in the two wetlands—nine planted species in the planted wetland and only two in the unplanted wetland.

To normalize the comparison for wetland vegetation, we counted all of the plants that would normally be considered “wetland plants” (obligate and facultative wetland species) in wetland delineations (Reed 1988). There were 55 wetland plants in the planted wetland and 52 wetland plants in the unplanted wetland after the 15th growing season (year 15). Among the wetland species listed for the unplanted wetland and not in the planted wetland are the invasives *P. australis* and *Lythrum salicaria*, which are listed as *common* and *present*, respectively, in that wetland (see supplemental material C). There were seven and nine invasive plants, following the definitions on Ohio invasive species lists (see Web citation in table 2), in the planted and unplanted wetlands, respectively, after 15 years.

In years five through eight (1998–2001), ANPP was higher ($p \leq .05$) in the unplanted wetland than in the planted wetland (figure 4b). Productivity in both wetlands decreased somewhat between year six (1999) and year eight (2001) because of a muskrat eat-out, stimulated in all probability by the rapid spread of *Typha* monocultures in the early years. From year 10 through year 13, productivity was higher ($p \leq .05$) in the planted wetland than in the unplanted wetland in three of the four years. Cumulative productivity consistently remained higher in the unplanted wetland since year 5 (figure 5b).



Figure 3. Aerial photographs of the two experimental wetlands in 1995 (year 2, left panel) and 2008 (year 15, right panel). The planted wetland is on the left, and the unplanted wetland is on the right in each panel. Note the fringe of planted macrophytes around the planted wetland's perimeter in year 2 and the fringe of wetland forest around both wetland perimeters in year 15.

Water quality changes. Water temperature, dissolved oxygen, pH, and conductivity were measured manually twice per day (at dawn and dusk) for 15 years, with water quality sondes at the inflow of the wetlands and at the outflows of both wetland basins. In addition to the twice-daily manual sampling, weekly water samples were taken at the inflow, at the middle, and at the outflows of the wetlands for nutrient concentrations (total phosphorus [TP], soluble reactive phosphorus [SRP], and nitrate + nitrite nitrogen [NO₃-N]) as was determined by standard methods (USEPA 1983, APHA 1989). Details of the field and laboratory analyses are in Mitsch and colleagues (2005b).

Water quality changes through the wetlands over the years were normalized not by describing concentrations in

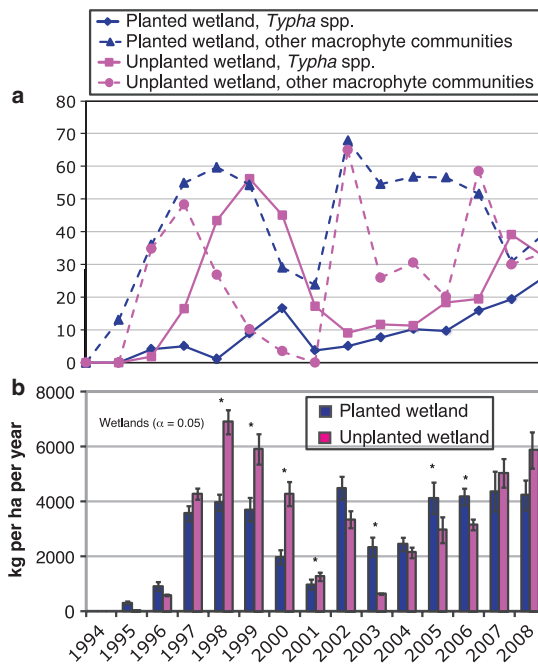


Figure 4. Comparison of (a) dominant wetland communities and (b) aboveground net primary productivity (ANPP, in kilograms [kg, dry weight] per hectare [ha] per year) for the planted wetland and the unplanted wetland between 1994 and 2008. *Typha* spp. and other macrophyte cover are shown in panel (a) for the two experimental wetlands. A vegetation eat-out in the winter of 2000–2001 by muskrats, particularly of *Typha*, led to the resurgence of other macrophyte cover in 2001. The asterisks (*) indicate a significant difference between the ANPP of the two wetlands at $\alpha = .05$. During the 14 years of measured productivity, the differences between the wetlands were significant seven years, with the unplanted wetland ANPP greater on four of those occasions.

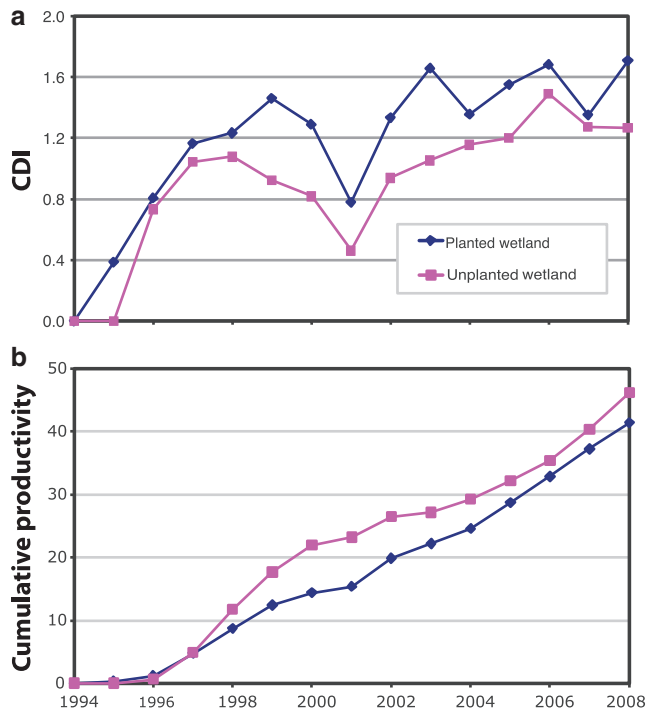


Figure 5. (a) Community diversity index (CDI) and (b) cumulative net primary productivity (in megagrams [Mg]) from macrophytes in the two experimental wetlands, 1994–2008. CDI values are statistically different between the two experimental wetlands ($p = 0$) over the 15 years of the present study. Overall, the unplanted wetland had 8 Mg more organic matter production at the end of the 15-year study.

Table 2. Summary of vegetation species richness in the two 1-hectare experimental wetlands at the Olentangy River Wetland Research Park during 1996 (3 years after the wetlands' creation), 1998 (5 years after creation) and 2008 (15 years after creation).

Year	Number of species			Number of wetland species			Number of planted wetland species ^a		Number of woody species		Number of invasive species ^b	
	Planted wetland	Unplanted wetland	Total	Planted wetland	Unplanted wetland	Total	Planted wetland	Unplanted wetland	Planted wetland	Unplanted wetland	Planted wetland	Unplanted wetland
1996	67	56	72	43	31	44	9	1	5	7	1	1
1998	96	87	99	56	46	57	9	2	15	15	4	4
2008	101	97	116	55	52	61	9	2	18	21	7	9

Note: Wetland species includes both obligate and facultative wetland species, following Reed (1988). The vegetation species names are in supplemental material C.

^aFrom 13 species planted in the planted wetland in May 1994 (see Mitsch et al. 1998).

^bFrom the Ohio list www.ohiodnr.com/Portals/3/invasive/pdf/OHIO%20INVASIVE%20PLANTS.pdf.

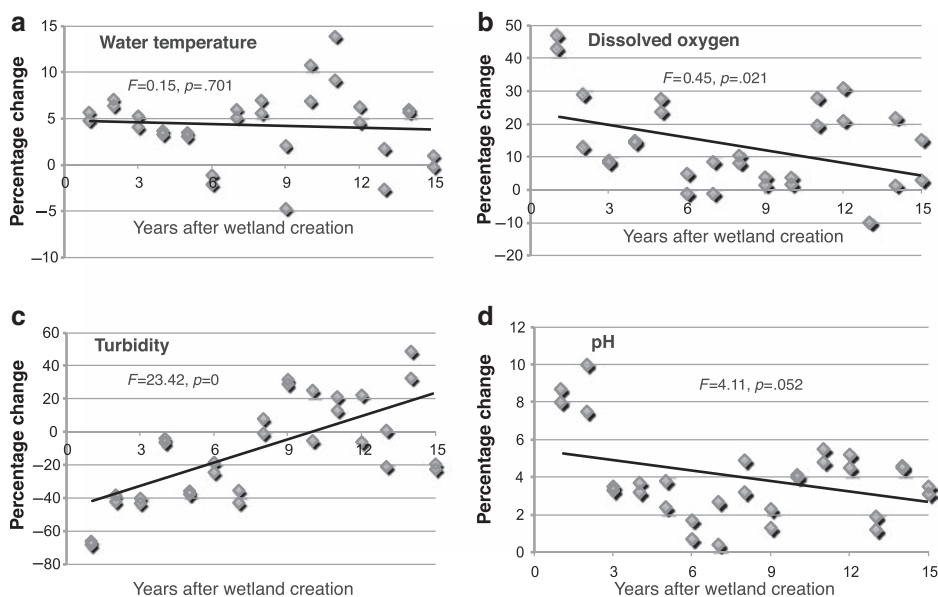


Figure 6. Fifteen-year trends of water quality changes through the experimental wetlands for (a) water temperature, (b) dissolved oxygen, (c) turbidity (as an indicator of suspended solids), and (d) pH. Each data point is the summary difference of several hundred readings taken annually at the inflow and outflows of the wetlands. Degrees of freedom for all F functions are (1,29).

the outflows but as the percentage change in concentrations between the inflow and that measured at the outflow. The 15-year trends of basic water quality changes for both of the wetlands lumped together (figure 6) show that (a) there was little significant trend in temperature change ($F(1,29) = 0.15, p = .701$); (b) dissolved oxygen consistently increased through the wetlands, most probably because of primary productivity in the water column, but there was a significant trend of that increase slowing by about 1.2% per year ($F(1,29) = 0.45, p = .021$); (c) the wetlands changed from being a sink for suspended sediments (estimated by turbidity) to being a source of those sediments at about year 10 ($F(1,29) = 23.42, p < .001$); and (d) pH increased in the wetlands each year, probably because of aquatic primary

productivity, with no significant trend in that increase ($F(1,29) = 4.11, p = .052$).

Two measures of phosphorus (TP and SRP) and the one measure of nitrogen ($\text{NO}_3\text{-N}$) consistently decreased through the wetlands, with only two exceptions (TP increased in both wetlands in 2003) out of 82 annual measurements. There were only four times (once each for TP and $\text{NO}_3\text{-N}$ and twice for SRP) out of 41 paired annual numbers that the two experimental wetlands differed significantly ($p = .05$) with regards to nutrient retention. In three of those four situations, the planted wetland showed more nutrient retention than did the unplanted wetland.

The trends of annual reductions of these three nutrient species over 15 years (figure 7) show that TP retention decreased

from 60% to 10% ($F(1,27) = 32.26, p = 0$), SRP retention decreased from 80% to 30% ($F(1,25) = 39.21, p < .001$), and $\text{NO}_3\text{-N}$ reductions decreased from 35% to 25% ($F(1,27) = 6.22, p = .019$), although recent data suggest that $\text{NO}_3\text{-N}$ reduction may be steady state or even improving.

Denitrification and a nitrogen budget. Denitrification was measured monthly in 2004, 2005, and 2008, with an *in situ* acetylene-inhibition technique in 4-cm-diameter PVC (polyvinylchloride) chambers. Details of the field and laboratory methods are in supplemental material D. Overall, the annual denitrification rates of the experimental wetlands were estimated to range from 2.1 to 3.7 g of nitrogen per m^2 per year (table 3), with no obvious trend over the 5 years of

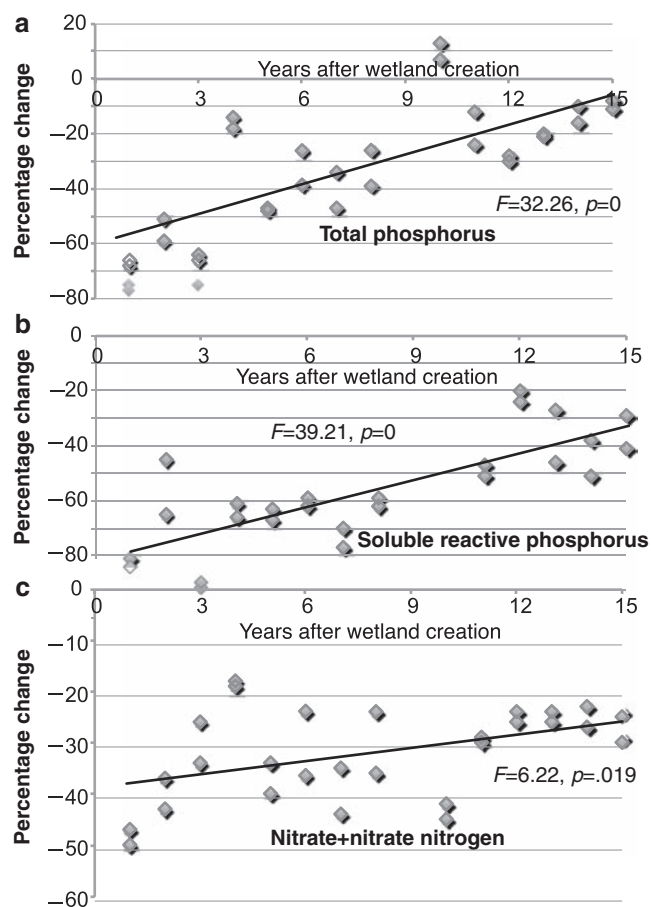


Figure 7. Fifteen-year trends of nutrient concentration changes through the experimental wetlands for (a) total phosphorus, (b) soluble reactive phosphorus, and (c) nitrate + nitrite nitrogen. Each data point is the percentage change of weekly samples at the inflow and similar samples at the outflows of the two wetlands. See the article text for degrees of freedom.

measurement (years 10–15, 2004–2008). The hydrologic pulsing imposed on both experimental wetlands in 2004 appears to have increased the overall rate of denitrification, at least relative to the artificial steady-flow conditions maintained the following year (2005; table 3). This suggestion is supported by findings that reported higher ($p \leq .05$) denitrification in late spring during the pulsing year than during the steady-flow year in the high marsh (*edge* or *high marshes* refers to those that flood less frequently) (Hernandez and Mitsch 2007).

Denitrification accounts for only about 2%–10% of the nitrogen reduction in these wetlands (table 3). Nitrogen accumulation in the soil, estimated to be 16.2–16.6 g of nitrogen per m^2 per year in the planted and the unplanted wetland, respectively over the first 10 years of wetland development (Anderson and Mitsch 2006), is an order of magnitude higher as a nitrogen sink in these flow-through wetlands than is denitrification. On a mass basis, the accumulation of nitrogen in soil accounts for 42%–61% of the nitrogen retained in these wetlands (table 3).

Greenhouse gas emissions. Methane emissions were measured over the five years between 2004 and 2008, and N_2O emissions were measured between 2004 and 2005, with non-steady-state gas emission chambers. Details of the field and laboratory methods are in supplemental material D. Although the overall average rates of methane emissions (table 4) were steady at about 30 g carbon (CH_4 -C) per m^2 per year between 2004 and 2008 (years 11–15), there was a consistent pattern of more methane emitted from the unplanted wetland (Nahlik and Mitsch 2010). The mean annual methane emissions from our reference wetland was 57 g of CH_4 -C per m^2 per year between 2006 and 2008, about double the rate measured at these created wetlands.

The overall rate of N_2O emissions was about 0.13 g of N_2O -N m^2 per year, a small fraction of the denitrification rates (table 3). The low (wetter) marsh zone showed N_2O/N_2 emission ratio percentages from 0.15% to 4.5%. Overall, N_2O emissions were highest in the edge (intermittently wet) marsh plots (0.191 ± 0.02 g of N_2O -N per m^2 per year) and lowest in the low (permanently wet) marsh plots (0.061 ± 0.004 g of N_2O -N per m^2 per year) (Hernandez and Mitsch 2006). The N_2O emissions were affected by pulsing hydrology. Spring N_2O fluxes from the high marsh plots were significantly ($p = .03$) higher under steady-flow conditions ($0.20 \pm .01$ g of N_2O -N per m^2 per year) than under pulsing conditions (0.09 ± 0.04 g of N_2O -N per m^2 per year).

Carbon sequestration. The cumulative ANPP (figure 5b) in the unplanted wetland began to increase at a more rapid rate in year 4 than in the planted wetland and, although the gap narrowed in the last 8 years of the study, there was still an estimated 5 megagrams (Mg; dry weight plant material, or 500 g per m^2), or 12%, more accumulated productivity in the unplanted wetland than in the planted wetland after 15 years.

The sequestration of carbon after 10 years in the experimental wetlands was 181 and 193 g of carbon per m^2 per year in the planted and unplanted wetlands, respectively (table 4), and a significant portion of that carbon accumulation (about 10%) was inorganic. The measurements of carbon sequestration after 15 years (2008) were higher, at rates of 219 and 266 g of carbon per m^2 per year over 15 years in the planted and unplanted wetlands, respectively (table 4). All of these rates are considerably higher than the carbon sequestration rates of 105–160 g per m^2 per year estimated at our reference wetland.

Ordination model results. For the vegetation ordination (figure 8a, 8b), there was a positive relationship between age and the three dependent variables (percentage of vegetation cover, CDI, and ANPP) for both the planted and the unplanted wetland. In both cases, CDI appeared to be more strongly correlated with age than were the other two variables. For water quality change ordinations (figure 8c, 8d), NO_3 -N, SRP, TP, and turbidity changes were positively

Table 3. Nitrous oxide emissions and estimates of overall denitrification in the experimental wetlands and its contribution to overall nitrogen retention by the wetlands.

	Hydrologic conditions				Normal river pulse conditions for both wetlands (2008) ^b
	Artificial spring pulses (2004) ^a		Flood pulses suppressed (2005) ^a		
	Planted wetland	Unplanted wetland	Planted wetland	Unplanted wetland	
Nitrous oxide emissions (g-N per m ² per year)					
Low marsh	0.019	0.102	0.027	0.022	—
High marsh	0.264	0.164	0.110	0.094	—
Edge	0.071	0.063	0.094	0.110	—
Denitrification (g-N per m ² per year)					
Open water	3.9	3.1	2.2	2.5	2.1
Marsh	1.72	2.14	1.42	1.83	1.7
Proportion open water	0.33	0.58	0.34	0.62	0.34
Proportion vegetated	0.67	0.42	0.66	0.38	0.66
Overall	2.5	2.7	1.7	2.3	1.8
Nitrogen accumulation in soil (g-N per m ² per year)					
	16	17			
Nitrogen surface inflow (g-N per m ² per year)					
	107	108	98	92	139
Nitrogen surface outflow from wetland (g-N per m ² per year)					
	69	80	44	37	56
Nitrogen removal in wetland (g-N per m ² per year)					
	38	28	54	55	83
Percentage nitrogen removal					
	35.5	25.9	55.1	59.8	59.7
Percentage of nitrogen removed due to denitrification					
	6.6	9.6	3.1	4.2	2.2
Percentage of nitrogen removed in soil sequestration					
	42	61	—	—	—

g-N, grams of nitrogen; m², square meters.
^aData from Hernandez and Mitsch (2007).
^bData from Song and colleagues (2012)

correlated with age, whereas conductivity, pH, dissolved oxygen, and temperature changes were negatively correlated with age in both wetlands. The correlation between temperature and age in the unplanted wetland was weak. For the planted wetland, TP change was more strongly correlated with age than was any other variable. Turbidity change was more strongly correlated with age than was any other variable in the unplanted wetland.

Methane emissions and denitrification were estimated three years out of the last five years in this study. For the methane and denitrification ordination (figure 8e, 8f), all variables were negatively correlated with age in the planted wetland, with denitrification seeming to have little correlation with the age of the wetland. In the unplanted wetland, methane emissions were positively correlated with age over these years, whereas denitrification was negatively correlated with age.

Primary succession of the created wetlands

With some variability, there appeared to be a trend of increasing community diversity in both the planted and the unplanted wetland for the first 10 years after the wetlands were created, with a leveling off over the subsequent 5 years.

The two experimental wetlands in this study achieved 90%–95% of their final species richness within 5 years of development. In these riverine flow-through wetlands at least, species introduction was rapid.

Planting had an effect on the structure (e.g., community diversity) of our primary-succession freshwater wetland for at least 15 years. We reported that after three years of wetland development, 9 of the 13 plant species introduced to the planted wetland were still in the wetland (Mitsch et al. 1998). The same nine species were still present after 15 years, although some were relatively rare. In year 15, the communities dominated by these introduced species covered 39% of the planted wetland and only 3% of the unplanted wetland. Several of the planted species, particularly *S. tabernaemontani* and *S. eurycarpum*, appear to have provided stiff competition in the planted wetland for the more productive *Typha* spp., which was illustrated in mesocosm experiments with different nutrient treatments at the same site (Svengsouk and Mitsch 2001).

Nutrient removal

Although both planted and unplanted wetlands generally continued to be nutrient sinks throughout the 15-year

Table 4. Methane emissions and carbon sequestration in the experimental wetlands at the Olentangy River Wetland Research Park (ORWRP) between 2004 and 2008 as well as those in other wetlands.

Wetland	Hydrologic conditions	Methane emissions ^a	Carbon sequestration ^a	Carbon dioxide sequestration to methane emissions ratio ^b	References
ORWRP experimental wetlands					
Planted wetland	Pulsing hydrology (2004)	16	181	37:1	Altor and Mitsch 2006, 2008, Anderson and Mitsch 2006
	Steady flow (2005)	16			Altor and Mitsch 2008
	Normal river pulse conditions (2006–2008)	13	219 (SE = 15)	46:1	Nahlik and Mitsch 2010
Unplanted wetland	Pulsing hydrology (2004)	32	193	17:1	Altor and Mitsch 2006, 2008, Anderson and Mitsch 2006
	Steady flow (2005)	31			Altor and Mitsch 2008
	Normal river pulse conditions (2006–2008)	47	266 (SE = 17)	16:1	Nahlik and Mitsch 2010
Reference wetland		57	140 (range, 105–160; SE = 16)	7:1	Nahlik and Mitsch 2010, Bernal and Mitsch 2012
General range for wetlands		range: 1 to 1000	range, 20–40		Mitra et al. 2005
General range for boreal peatlands		range: 2 to 7	range, 15–26	12:1 (range, 6:1 to 36:1)	Turunen et al. 2002

SE, standard error.
^aIn grams carbon per square meter per year.
^bRatio of carbon dioxide sequestered (CO₂) per methane (CH₄) emitted.

period of our study, the trends, especially for total phosphorus, suggest a pattern of decreased retention. The phosphorus trend may be a result of the wetlands' saturation of storages in the soil, detritus, and plant biomass. More sediment, soil, or organic matter in the outflow may also contribute to the decrease in phosphorus-removal effectiveness. NO₃-N retention appeared to stabilize in years 10–15 and should be effective and maybe even more effective in subsequent years.

The planted and unplanted wetlands were rarely different in terms of overall nutrient removal; there were only a few annual statistical differences for nitrates, SRP, or TP between the wetlands over the 15-year study period. This is enlightening, given the great attention often paid to the selection of plants when wetlands are created for water quality enhancement. In the end, plant selection may not be one of the important design questions for wetlands involved in nutrient removal, except perhaps when very low concentrations are of interest, such as in the Florida Everglades (Chimney and Goforth 2006). As was described elsewhere in this article, planting may be more important for other functions.

The wetlands were sinks for 26%–60% of the surface water flow of nitrogen by mass (and 22%–29% by concentration of NO₃-N) for the last 5 years of the 15-year study period. Denitrification was not an important factor after 15 years of the wetlands as sinks of nitrogen,

because it accounted for only 2%–10% of the nitrogen removed by these wetlands. Nitrogen sequestration into the soil accounted for an order of magnitude more. It could be that these created wetlands need many more years to develop denitrification rates comparable to those of similar natural wetlands.

Soil development and carbon fluxes

The opportunity to show the development of wetland soil conditions in formerly nonwetland soils over this amount of time is uncommon. In other studies, created-wetland soils examined over a shorter time period have not often shown—as they have in our study—a relation between organic matter accumulation and wetland age (Bishel-Machung et al. 1996, Shaffer and Ernest 1999, Anderson and Cowell 2004). The accumulation of organic carbon in the soil, along with the standing water itself, caused the nonhydric soils to begin to develop characteristics of hydric soils within two years. After 1.4 years of flooding, over three-quarters of the wetlands would have been characterized as hydric, with chroma values less than 2; after 10 years, all of the soil samples taken throughout the basins were hydric. Hydric characteristics first developed at the soil surface and became deeper with succeeding years. Gleying of iron-depleted soils to a depth of 16 cm or more was commonly observed in these wetlands near the end of our study.

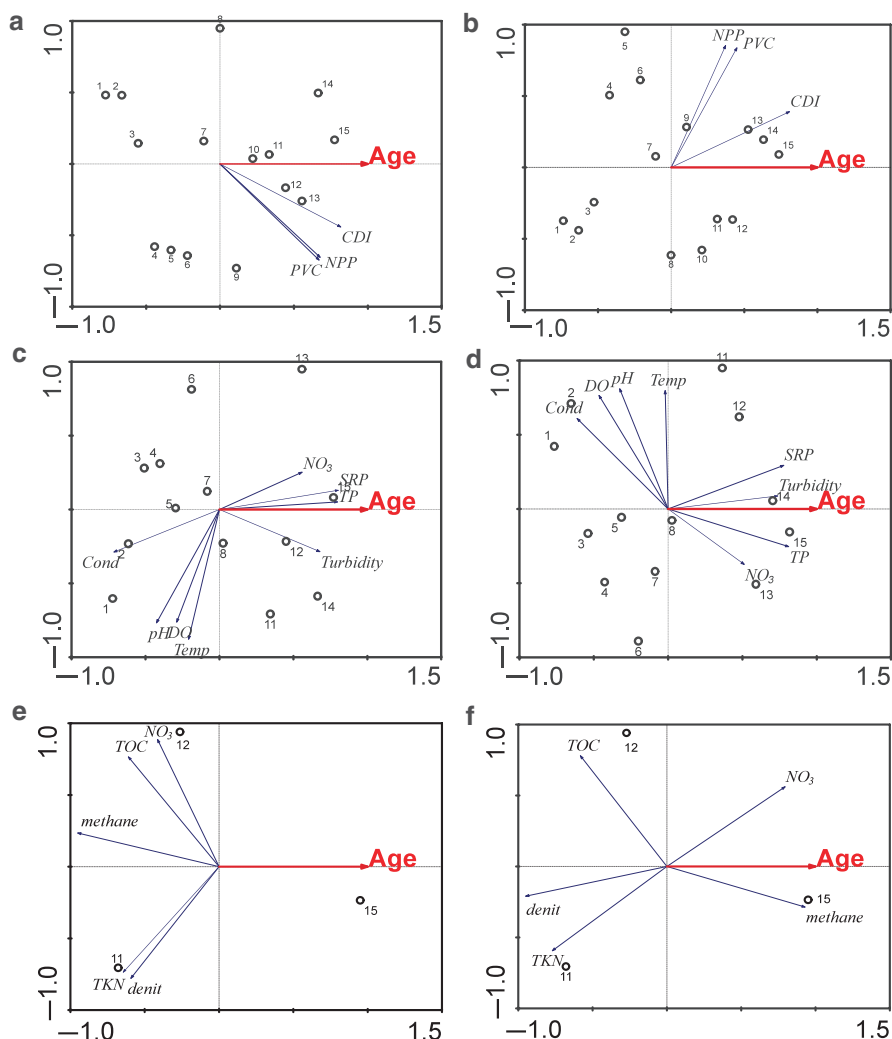


Figure 8. Ordination redundancy analysis triplots for the influence of wetland age on vegetation in the planted wetland (a) and unplanted wetland (b), water quality changes in the planted wetland (c) and unplanted wetland (d), and carbon and nitrogen in the planted wetland (e) and unplanted wetland (f). Dependent variables are depicted by thin blue arrows, the independent variable is depicted as a thick red arrow, and the sample year is depicted by open circles. Abbreviations: CDI, community diversity index; Cond, conductivity; denit, denitrification; DO, dissolved oxygen; NO₃, nitrate-nitrogen; NPP, net primary production; PVC, percent vegetation cover; SRP, soluble reactive phosphorus; Temp, temperature; TKN, total Kjeldahl nitrogen; TP, total phosphorus.

Surface soil carbon (primarily as organic carbon) increased dramatically in these created wetlands over the 15-year study. The carbon content almost tripled over that period, and organic matter concentrations in the upper 8 cm of the soil increased from 5.3% immediately before the wetlands were created to 6.1% 17 months after flooding then to 9.4% after 10 years (see Anderson et al. 2005). Soil organic matter distribution varied throughout the wetlands, and there was a slightly broader range in the unplanted wetland, particularly near the inflow, where the accretion rates were highest (Anderson and Mitsch 2006).

The measured soil carbon sequestration rates in the experimental wetlands of 181–266 g of carbon per m² per year were considerably higher than similar measurements in our reference flow-through wetland (105–160 g of carbon per m² per year) and the general range of carbon sequestration in wetlands reported by Mitra and colleagues (2005) of 20–140 g of carbon per m² per year (table 4). Euliss and colleagues (2006) reported higher carbon sequestration in restored prairie pothole marshes (305 g of carbon per m² per year) than in reference wetland sites (83 g of carbon per m² per year), so the high carbon sequestration rate that we saw may not continue indefinitely. The carbon sequestration figures for these created and restored wetlands, including those for the sites in the present study, are considerably higher than those for much more frequently cited boreal-zone peatlands, where general rates of 10–25.5 g of carbon per m² per year have been reported (Gorham 1991, Turunen et al. 2002).

Part of the reason that newly created wetlands have high carbon sequestration could be the algal-influenced inorganic carbon accumulation in these wetlands in the early years. Inorganic carbon rates were estimated at 23 and 26 g of carbon per m² per year between 1994 and 2004 for the planted and unplanted wetlands, respectively, a figure that is likely tied to calcite precipitation, particularly in the open-water portions of the wetland, where algae production was particularly high and emergent macrophyte productivity was low (Wu and Mitsch 1998, Liptak 2000, Anderson and Mitsch 2006). The rate of calcite deposition that we measured in these wetlands (1.2 g of carbon per m² per day) is similar to the rates reported by Stabel (1989) and Otsuki and Wetzel (1974) for eutrophic hardwater lakes (1.24 and 1.22 g of carbon per m² per day, respectively). Calcite precipitation may be more widespread as a carbon-retention process in wetlands than was previously thought; it may also play a significant role in phosphorus immobilization through coprecipitation of calcium phosphates.

The mean annual methane emissions were consistently higher for our five years of measurements (2004–2008) in the unplanted wetland than in the planted wetland. This difference could logically be due to the higher accumulation of organic carbon in the unplanted wetland. Over the 15 years of study, there was an accumulation of 500 g (dry weight) per m² more organic matter in the unplanted wetland than in the planted wetland, which was estimated by the differences in ANPP. Nevertheless, the emission rates in both wetlands appear to have been lower than those of similar naturally occurring wetlands in the region, including our reference wetland. Pulsing hydrology, which is common in these riverine wetlands, was shown to slow down the methane emissions in certain parts of these wetlands as well in a comparison in 2004 and 2005 (Altor and Mitsch 2008). Higher methane emissions under steady-flow hydrology have also been reported in several rice paddy studies (Mishra et al. 1997, Rath et al. 1999, Wang et al. 1999).

The net effect of wetlands on climate radiative forcing

The overall impact of a wetland on climate change is a balance between the wetland's ability to sequester and retain carbon dioxide (CO₂) from the atmosphere into semi-permanent storage in the soil and its emissions of greenhouse gases. We show here the ratio of CO₂ sequestered to CH₄ emitted (expressed as molecular weight) as ranging from 31:1 to 37:1 for the planted wetland and 15:1 to 17:1 for the unplanted wetland. Comparing these numbers with the global warming potential of CH₄ relative to that of CO₂ of 25:1, we could conclude that the planted wetland is a sink of climate radiative forcing and that the unplanted wetland is a source of radiative forcing. Our reference flow-through wetland had a ratio of only 7:1 and so is even more of a source of radiative forcing than was our unplanted wetland. We found, in an analysis of 18 wetlands around the world, that the average molecular ratio of CO₂ sequestered to CH₄ emitted is about 20:1. Planting may have resulted in more favorable conditions related to carbon sequestration relative to methane emissions, but in terms of kilograms of carbon sequestered, the unplanted wetland was more effective.

Trends, cycles, and projections

The planted and unplanted wetlands appear to be developing along similar paths as far as vegetation and water quality are concerned. For both of the wetlands, primary productivity, community diversity, vegetation cover, and nutrient (nitrogen and phosphorus) and turbidity export increased with age, whereas conductivity, dissolved oxygen, and pH seem to have decreased with age. Temperature for both wetlands seems to be the variable least associated with age. Dissolved oxygen and pH may be influenced slightly by age, but the water quality of the river itself may be more predictive of temperature, pH, and dissolved oxygen changes within the wetlands than processes within the wetlands themselves are.

Within the planted wetland, methane emissions seem to have decreased over time. The opposite was seen in the unplanted wetland. As age increased in that wetland, methane emissions increased. Denitrification seems to have little to do with age in the planted wetland, but it decreased with age in the unplanted wetland.

The dominance of *Typha* spp. appears to have a 10-year cycle intermixed with muskrat herbivory. We expect that pattern to continue and for the *Typha* patterns and overall productivity of macrophytes to converge but become less important as woody plants on the edges develop and gradually enter much of the wetland other than for the deepwater areas. We expect the wetlands to be neither sinks nor sources of phosphorus over the next 20–30 years, assuming that a similar hydrology to that of the first 15 years is continued. NO₃-N retention appears to have been steady or to have even increased slightly over the last 5 years of the study. As denitrification rates, which were quite low at the end of the study relative to natural wetlands, continue to increase as organic carbon accumulates in these wetlands, we expect the nitrogen retention of these riverine flow-through wetlands to increase.

Ecosystem experimentation versus experimental ecology

This study was a 15-year whole-ecosystem study, not a replicated reductionistic study. Miao and Carstenn (2006), following up on the work of Carpenter (1990), discussed the applicability of ecosystem science in the restoration of the Florida Everglades. They pointed out—accurately, in our view—that

ecosystem ecology and restoration are developing rapidly to meet the demand for urgent action to restore degraded habitats. This has revealed the inadequacy of traditional statistical methods in addressing large-scale, unreplicated research and unplanned events, because these methods rely on a statistical paradigm of replicates, homogeneity, randomness, normal distributions, and controlled experiments. However, the subjects of ecosystem studies are often complex, nonrandom, nonnormal, not replicable, and generally violate traditional statistical assumptions.

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Our ordination analyses helped somewhat to show processes that sync or go in different directions, and other tests allowed us to compare paired and unpaired data in the two experimental wetlands. Ultimately, a multiparameter dynamic ecosystem model may be the optimum methodology to tie all of these dependent variables into a single system.

A strategy of wetland ecosystem development

Natural ecosystems do not develop according to human wishes or desires. We closely followed the primary succession of two freshwater flow-through wetlands over 15 years

on a floodplain that, when we started our experiment, was a cleared landscape otherwise devoid of anything except aerobic microbes and some upland plant seed banks. In this primary succession, the wetlands showed both allogenic and autogenic succession. There are signs of biotic control, such as muskrat eat-outs, but there are also wetland functions that appear to be determined directly by forcing functions (now sometimes called *drivers*), particularly the hydrologic pulsing of river water and the resulting hydroperiods.

It appears to us that the unplanted wetland basin maximized power (i.e., it was more productive) in the Odum (1971) concept of maximum power, whereas the planted wetland had more community diversity. In the midwestern United States, where almost all of the landscape and many of the rivers and streams have excessive nutrients, highly productive *Typha* marshes are the norm when wetlands are created and restored. We have shown that planting does help maintain consistently more plant community diversity over even a 15-year time frame. In a strange way, the planting appeared to be only a slight bump in the road to *Typha* marsh dominance and hence to the planted wetland system's maximization of power. Planting led to more diversity of plant communities and perhaps to better wetland aesthetics (the introduced plants, well shorter than 2 m, provided a better vista across the wetland for humans). The natural colonization of plants in the unplanted wetland led to better ecosystem services, such as carbon sequestration, but also led to higher emissions of the greenhouse gas methane.

The extended time frame of the present study gave us an opportunity to see how time works as a partner with self-design. We were left at the end of 15 years wondering whether that is even enough time for a wetland to become stable. Perhaps it is the wrong question, since these wetlands (and wetlands in general) are a function of dynamic landscapes, such as flooding rivers of different amplitudes and frequencies in this case. These wetlands are relatively stable systems amid dramatic seasonal changes and flooding instability. Perhaps the concept suggested by Ives and Carpenter (2007) that diversity enhances an environment's ability to resist invasive species and its temporal stability is more appropriate for these developing wetlands. That is the point of sound management of natural ecosystems—to provide resilient biological havens in an uncertain environment through stability. But this stability comes in its own time, not in ours.

Acknowledgments

We appreciate the many student and other researchers who helped during the 15 years of research for this study, especially Changwoo Ahn, Anne E. Altor, Virginie Bouchard, Michael A. Liptak, Douglas Spiles, Naiming Wang, and Xinyuan Wu, who were helpful at critical points in the project. Funding for this project came from US Department of Agriculture (National Research Initiative Cooperative State Research, Education and Extension Service Award no. 2003-35102-13518) from the US Environmental

Protection Agency (US EPA Agreements no. EM83329801-0 from Cincinnati, Ohio, and no. MX95413108-0 from the Gulf of Mexico Program, Stennis Space Center, Mississippi), and from the Wilma H. Schiermeier Olentangy River Wetland Research Park, the School of Environment and Natural Resources, the Environmental Science Graduate Program, and the Ohio Agricultural Research and Development Center at The Ohio State University. The present article is Olentangy River Wetland Research Park publication no. 2012-002.

References cited

- Altor AE, Mitsch WJ. 2008. Methane emissions and carbon dioxide fluxes in created wetland mesocosms: Effects of hydrologic regime and hydric soils. *Ecological Applications* 18: 1307–1320.
- Anderson CJ, Cowell BC. 2004. Mulching effects on the seasonally flooded zone of west-central Florida, USA wetlands. *Wetlands* 24: 811–819.
- Anderson CJ, Mitsch WJ. 2006. Sediment, carbon, and nutrient accumulation at two 10-year-old created riverine marshes. *Wetlands* 26: 779–792.
- Anderson CJ, Mitsch WJ, Nairn RN. 2005. Temporal and spatial development of surface soil conditions in two created riverine marshes. *Journal of Environmental Quality* 34: 2072–2081.
- [APHA] American Public Health Association. 1989. *Standard Methods for the Analysis of Wastewater*, 17th ed. APHA.
- Bergamaschi P, et al. 2007. Satellite cartography of atmospheric methane from SCIAMACHY on board ENVISAT: 2. Evaluation based on inverse model simulations. *Journal of Geophysical Research* 112: D02304. doi:10.1029/2006JD007268
- Bernal B, Mitsch WJ. 2012. Comparing carbon sequestration in temperate freshwater wetland communities. *Global Change Biology*. Forthcoming. doi:10.1111/j.1365-2486.2011.02619.x
- Bishel-Machung L, Brooks RP, Yates SS, Hoover KL. 1996. Soil properties of reference wetlands and wetland creation projects in Pennsylvania. *Wetlands* 16: 532–541.
- Bloom AA, Palmer PI, Fraser A, Reay DS, Frankenberg C. 2010. Large-scale controls of methanogenesis inferred from methane and gravity spaceborne data. *Science* 327: 322–325.
- Carpenter SR. 1990. Large-scale perturbations: Opportunities for innovation. *Ecology* 71: 2038–2043.
- Chimney MJ, Goforth G. 2006. History and description of the Everglades Nutrient Removal Project, a subtropical constructed wetland in south Florida (USA). *Ecological Engineering* 27: 268–278.
- Costanza R, et al. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253–260.
- Euliss NH Jr, Gleason RA, Olness A, McDougal RL, Murkin HR, Robarts RD, Bourbonniere RA, Warner BG. 2006. North American prairie wetlands are important nonforested land-based carbon storage sites. *Science of the Total Environment* 361: 179–188.
- Fennessy MS, Rokosch A, Mack JJ. 2008. Patterns of plant decomposition and nutrient cycles in natural and created wetlands. *Wetlands* 28: 300–310.
- Gardner RC, Zedler J, Redmond A, Turner RE, Johnston CA, Alvarez VA, Simenstad CA, Prestegard KL, Mitsch WJ. 2009. Compensating for wetland losses under the Clean Water Act (redux): Evaluating the federal compensatory mitigation regulation. *Stetson Law Review* 38: 213–249.
- Gorham E. 1991. Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecological Applications* 1: 182–195.
- Gutrich JJ, Taylor KJ, Fennessy MS. 2009. Restoration of vegetation communities of created depressional marshes in Ohio and Colorado (USA): The importance of initial effort for mitigation success. *Ecological Engineering* 35: 351–368.
- Hernandez ME, Mitsch WJ. 2006. Influence of hydrologic pulses, flooding frequency, and vegetation on nitrous oxide emissions from created riparian marshes. *Wetlands* 26: 862–877.

- . 2007. Denitrification in created riverine wetlands: Influence of hydrology and season. *Ecological Engineering* 30: 78–88.
- Ives AR, Carpenter SR. 2007. Stability and diversity of ecosystems. *Science* 317: 58–62.
- Lenhart M. 2009. An unseen carbon sink. *Nature Reports Climate Change*. doi:10.1038/climate.2009.125
- Liptak MA. 2000. Water Column Productivity, Calcite Precipitation, and Phosphorus Dynamics in Freshwater Marshes. PhD dissertation, The Ohio State University, Columbus.
- Matthews JW, Endress AG. 2008. Performance criteria, compliance success, and vegetation development in compensatory mitigation wetlands. *Environmental Management* 41: 130–141.
- Miao S, Carstenn S. 2006. A new direction for large-scale experimental design and analysis. *Frontiers in Ecology and the Environment* 4: 227.
- Mishra S, Rath AK, Adhya TK, Rao VR, Sethunathan N. 1997. Effect of continuous and alternate water regimes on methane efflux from rice under greenhouse conditions. *Biology and Fertility of Soils* 24: 399–405.
- Mitra S, Wassmann R, Vlek PLG. 2005. An appraisal of global wetland area and its organic carbon stock. *Current Science* 88: 25–35.
- Mitsch WJ, Gosselink JG. 2007. *Wetlands*, 4th ed. Wiley.
- Mitsch WJ, Wilson RF. 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. *Ecological Applications* 6: 77–83.
- Mitsch WJ, Wu X, Nairn RW, Weihe PE, Wang N, Deal R, Boucher CE. 1998. Creating and restoring wetlands: A whole-ecosystem experiment in self-design. *BioScience* 48: 1019–1030.
- Mitsch WJ, Wang N, Zhang L, Deal R, Wu X, Zuwerink A. 2005a. Using ecological indicators in a whole-ecosystem wetland experiment. Pages 211–235 in Jørgensen SE, Xu F-L, Costanza R, eds. *Handbook of Ecological Indicators for Assessment of Ecosystem Health*. CRC Press.
- Mitsch WJ, Zhang L, Anderson CJ, Altor AE, Hernández ME. 2005b. Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. *Ecological Engineering* 25: 510–527.
- Mitsch WJ, Nahlik A[M], Wolski P, Bernal B, Zhang L, Ramberg L. 2010. Tropical wetlands: Seasonal hydrologic pulsing, carbon sequestration, and methane emissions. *Wetlands Ecology and Management* 18: 573–586.
- Nahlik AM, Mitsch WJ. 2010. Methane emissions from created riverine wetlands. *Wetlands* 30: 783–793.
- Nairn RW. 1996. Biogeochemistry of newly created riparian wetlands: Evaluation of water quality changes and soil development. PhD dissertation, The Ohio State University, Columbus.
- National Research Council. 2001. *Compensating for Wetland Losses under the Clean Water Act*. National Academy Press.
- Odum HT. 1971. *Environment, Power, and Society*. Wiley.
- Otsuki A, Wetzel RG. 1974. Calcium and total alkalinity budgets and calcium carbonate precipitation of a small hard-water lake. *Archiv für Hydrobiologie* 73: 14–30.
- Rath AK, Swain B, Ramakrishnan B, Panda D, Adhya TK, Rao VR, Sethunathan N. 1999. Influence of fertilizer management and water regime on methane emission from rice fields. *Agriculture, Ecosystems and Environment* 76: 99–107.
- Reed PB Jr. 1988. National list of plant species that occur in wetlands: Northeast (Region 1). US Fish and Wildlife Service. Report no. 88-26.1.
- Robb JT. 2002. Assessing wetland compensatory mitigation sites to aid in establishing mitigation ratios. *Wetlands* 22: 435–440.
- Shaffer PW, Ernest TL. 1999. Distribution of soil organic matter in freshwater emergent/open water wetlands in the Portland, Oregon metropolitan area. *Wetlands* 19: 505–516.
- Song K, Kang H, Zhang L, Mitsch WJ. 2012. Seasonal and spatial variations of denitrification and denitrifying bacterial community structure in created riverine wetlands. *Ecological Engineering* 38: 130–134.
- Spieles DJ. 2005. Vegetation development in created, restored, and enhanced mitigation wetland banks of the United States. *Wetlands* 25: 51–63.
- Spieles DJ, Coneybeer M, Horn J. 2006. Community structure and quality after 10 years in two central Ohio mitigation bank wetlands. *Environmental Management* 38: 837–852.
- Stabel HH. 1989. Coupling of strontium and calcium cycles in Lake Constance. *Hydrobiologia* 176–177: 323–329.
- Svengsouk LM, Mitsch WJ. 2001. Dynamics of mixtures of *Typha latifolia* and *Schoenoplectus tabernaemontani* in nutrient-enrichment wetland experiments. *American Midland Naturalist* 145: 309–324.
- Turunen J, Tomppo E, Tolonen K, Reinkainen A. 2002. Estimating carbon accumulation rates of undrained mires in Finland: Application to boreal and subarctic regions. *The Holocene* 12: 69–80.
- [USEPA] US Environmental Protection Agency. 1983. *Handbook for Methods in Water and Wastewater Analysis*. USEPA.
- Van den Wollenberg AL. 1977. Redundancy analysis: An alternative for canonical correlation analysis. *Psychometrika* 42: 207–219.
- Wang B, Xu Y, Wang Z, Li Z, Guo Y, Shao K, Chen Z. 1999. Methane emissions from ricefields as affected by organic amendment, water regime, crop establishment, and rice cultivar. *Environmental Monitoring and Assessment* 57: 213–228.
- Whalen SC. 2005. Biogeochemistry of methane exchange between natural wetlands and the atmosphere. *Environmental Engineering Science* 22: 73–94.
- Wilson RF, Mitsch WJ. 1996. Functional assessment of five wetlands constructed to mitigate wetland loss in Ohio, USA. *Wetlands* 16: 436–451.
- Wu X, Mitsch WJ. 1998. Spatial and temporal patterns of algae in newly constructed freshwater wetlands. *Wetlands* 18: 9–20.
- Zedler JB. 2000. Progress in wetland restoration ecology. *Trends in Ecology and Evolution* 15: 402–407.

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