Wetlands Ecol Manage
https://doi.org/10.1007/s11273-018-9633-8

ORIGINAL PAPER

Longitudinal evaluation of vegetation richness and cover at wetland compensation sites: implications for regulatory monitoring under the Clean Water Act

Morgan Robertson · Susan M. Galatowitsch · Jeffrey W. Matthews

Received: 10 January 2018 / Accepted: 5 October 2018

Abstract There has been regulatory concern over the appropriate length of time to monitor wetland sites restored or created as compensation for impacts permitted by a U.S. Clean Water Act permit. However there is very little longitudinal research on wetland compensation sites, and conclusions on compensation site development are usually drawn from the analysis of a chronosequence of sites of different ages. This approach has limitations, given the extent of changes in wetland compensation practices and performance standards over the past few decades. In this study we conducted vegetation surveys of 22 wetland compensation sites in a rapidly developing part of the Minneapolis-St. Paul metropolitan area in 1997 and 2010. We present data on changes over time in floristic richness and cover at the site level and at the level of wetland community type within each site. Our findings do not support the assumption that wetland compensation sites progress on a trajectory toward increasing diversity, floristic quality, or native cover over time. We find that, when data from all sites are considered together, emergent communities have suffered significant declines in both floristic quality and native plant cover, while wet meadow communities have gained species richness but not species diversity. There is some evidence that site richness and cover characteristics are converging toward a regional mean over time, as the species composition of wet meadows became significantly more similar over the survey period, and all community types have significant increases in woody cover. Our study suggests the importance of selecting appropriate compensation sites that avoid foreseeable hydrologic stresses, and does not support the position that 5 years of monitoring can assure the ongoing biotic integrity of wetland compensation sites.

Keywords Compensation · Restoration · Monitoring · Clean Water Act · Vegetation change

Introduction

The restoration, creation or enhancement of wetlands as compensation for wetland impacts permitted under the US Clean Water Act has been common since the late 1970s. Since the first rigorous assessment of wetland compensation sites (Race and Christie 1982; Race 1985), a large literature has sprung up around the question of whether such sites meet regulatory performance standards and other standards of

M. Robertson (✉)
University of Wisconsin – Madison, Madison, WI, USA
e-mail: mmrobertson@wisc.edu

S. M. Galatowitsch
University of Minnesota, Minneapolis, MN, USA

J. W. Matthews
University of Illinois – Urbana-Champaign, Champaign, IL, USA

ecological quality (see NRC 2001; Zedler 2004). The question of regulatory compliance has been complicated by the fact that until 2008, compensation was only loosely and briefly described in federal regulation at 40 CFR 230.75 as "habitat restoration" (Hough and Robertson 2009). In the absence of binding national-level regulations governing compensation outcomes, different performance standards and assessment methods were created in each of the 38 districts of the US Army Corps of Engineers (Corps), often in collaboration with state resource agencies and regional offices of the US Environmental Protection Agency (EPA), the US Fish and Wildlife Service, and the National Marine Fisheries Service. There was substantial communication between agency offices on best practices, but the first three decades of compensatory mitigation were characterized by an emerging patchwork of different and uncoordinated standards and practices (Doyle et al. 2013; Breaux and Serefiddin 1999; Streever 1999). In particular, practices varied concerning the appropriate length of monitoring needed to establish the compliance of a compensation site.

When the Corps and EPA issued new proposed regulations concerning compensatory wetland mitigation in 2006, some commenters were concerned with the proposed 5-year limit to monitoring, and others over the provision that monitoring could be waived "upon a determination that the compensatory mitigation site has achieved its performance standards" (Corps and EPA 2006, p. 15551). Reporting on these comments in the preamble to the final rule in 2008, the Corps and EPA noted that "There was no consensus among commenters regarding the appropriate length for monitoring periods" (Corps and EPA 2008, p. 19644). The final rule's discussion of monitoring suggests that more study is needed on the ecological changes and variability experienced by newly established compensation sites, and that diachronic data on compensation sites beyond 5 years are needed to provide a scientific groundwork for regulatory discussions concerning the monitoring of compensation sites:

> We believe that five years is an appropriate starting point for determining the required monitoring period. The final rule states that the mitigation plan must provide for a monitoring period that is sufficient to demonstrate that the compensatory mitigation project has met performance standards, but not less than five years, and a longer monitoring period must be required for aquatic resources with slow development rates (e.g., forested wetlands, bogs)…. Performance standards should be designed, to the extent practicable, to account for the ecological characteristics of early developmental stages of aquatic ecosystems, so that a determination of ecological success can be made within five years (19645).

In the final rule, the agencies created a series of extensions and exceptions to the 5-year limit that recognize, for example, difficulties in drawing conclusions from the "early developmental stages" of restoration sites, and the need to establish evidence of sustained compliance. Most significantly, the final rule allows monitoring to be waived if there are "at least two consecutive monitoring reports issued where the success criteria are met. This will help account for variability in environmental conditions, to ensure that the compensatory mitigation project is truly meeting its performance standards" (Corps and EPA 2008, p. 19645). However, the agencies offered no scientific basis for these changes because of the scarcity of scientific work documenting change over time at compensation sites. In this paper we attempt to address this scarcity by analyzing changes in species richness and cover at wetland compensation sites using surveys performed 13 years apart.

In stating that two successful reports can constitute compensation site compliance, the rule is informed by the idea that once a restoration site is on a trajectory toward success (by ecological or administrative benchmarks), that it will continue on that path. One problem with using the trajectory concept in evaluating compensation sites has been identified by Zedler and Callaway (1999), who suggest that "the time to functional equivalency may well exceed the usual monitoring periods, and long-term predictions of the time to functional equivalency may not be meaningful if they are based on short-term data from pulse-driven ecosystems". The concept of the ecological trajectory is still current, however (Clewell and Aronson 2013), and is embedded in current regulatory practice (Miller et al. 2012). If the ecological trajectory concept frames the monitoring debate, wetland compensation policies will benefit from studies on the pace and nature of ecological change in compensation sites over time. Matthews and Spyreas (2010) and Brooks et al. (2005) both suggest that the idea of convergence and divergence may be a more useful way of characterizing restoration trajectories.

The 5-year duration may have as much to do with the availability of staff time to pursue and review monitoring reports as it has to do with scientific principles. A general Corps policy placing monitoring "below the line" separating mandatory from optional staff duties (Army Corps 1999) caused controversy (NRC 2001). Research has found that in many cases, monitoring has been so lax that significant percentages of compensation sites were not even built: 33% in Florida between 1981 and 1991 (Erwin 1991), and 22% in Massachusetts between 1983 and 1994 (Brown and Veneman 2001). Those that are constructed are often unmonitored: 46% of sites in Michigan's Upper Peninsula between 2003 and 2006 (Kozich and Halvorsen 2012; see also Bernhardt et al. 2005). In contrast, Sudol and Ambrose (2002) found that the widespread failure to meet permit conditions among wetland compensation sites in Orange County, CA between 1979 and 1993 was due to inadequate mitigation plans rather than the failure of monitoring and enforcement. In a Government Accountability Office report entitled "Corps of Engineers Does Not Have an Effective Oversight Approach to Ensure That Compensatory Mitigation Is Occurring", the GAO (2005, p. 2) found that:

> The Corps required monitoring reports for 89 of the 152 permit files reviewed where the permittee was required to perform compensatory mitigation. However, only 21 of these files contained evidence that the Corps received these reports. Moreover, only 15 percent of the 152 permit files contained evidence that the Corps had conducted a compliance inspection.

While there have been many studies of wetland compensation sites, especially since the 1990s, most of these have aimed at assessing success relative to permit criteria. Fewer have engaged in ecological assessment and many of those studies have very small sample sizes (Cammen 1976a, b; Seneca et al. 1976; Kelly 2001; Morgan and Roberts 2003; Stefanik and Mitsch 2012; Gutrich and Hitzhusen 2004; Balcombe et al. 2005; Spieles 2005; Mack and Miccachion 2006; see also the literature reviewed in Kusler and Kentula 1990; NRC 2001). Of the studies that examine the ecology of compensation sites, few (e.g., Matthews and Endress 2008; Van den Bosch and Matthews 2017) have used longitudinal data from the same sites to characterize trends, rather than doing so using a chronosequence of wetland compensation sites of different ages. Given the rapid changes in compensation standards and practices between 1990 and the present, it is particularly inappropriate to draw conclusions about trends based on differences between wetland compensation sites established decades apart. Spieles et al. (2006) studied two wetland compensation sites in Ohio comparing a 10th-year survey with the first 5 years of monitoring data collected by regulators. Galatowitsch and her colleagues (Mulhouse and Galatowitsch 2003; Aronson and Galatowitsch 2008) have exampled a sample of 39 wetland restorations in the prairie pothole region over 19 years; although the sites are likely very similar to compensation sites in the region, they were not restored for compensatory purposes.

We surveyed 22 wetland compensation sites in the urban periphery of the Minneapolis-St. Paul area in 1997, and revisited these sites in 2010. The same sites were subjected to the same assessment by the same researchers, allowing us to characterize vegetative development of wetland compensation sites in the rapidly urbanizing setting where wetland compensation sites are frequently located. We believe this is the first longitudinal study to report on a large set of wetland compensation sites using the same assessment method by the same researchers. Using survey data, we measure changes in vegetative richness and cover to examine changes and trends in plant diversity within wetland community types, species presence and turnover, and changes in wetland plant guilds over 13 years. We use these findings to comment on evidence of trajectories in compensation site development.

Methods

Site selection

We conducted vegetation surveys of 22 of the 23 wetland compensation sites constructed under the permit requirements of Section 404 of the Clean Water Act or the Minnesota Wetland Conservation

Act permit program in the South Washington Watershed District between January 1992 and July 1995. The Minnesota Wetland Conservation Act (MWCA) came into effect in January 1992, and so the wetland compensation sites we surveyed were all subject to the same regulatory regime. All sites were at least 2 years old by the time of the first survey, between August 22 and September 19, 1997. To minimize the effects of seasonality, the second survey also took place late in the growing season, between August 17 and September 8, 2010. Since all sites were relatively small even by the standards of compensatory mitigation sites, between 0.07 and 0.98 ha, vegetation surveys consisted of a random walk survey to record all species present and to estimate cover for each species observed. The 22 sites represent compensation for 13 different permitted impacts, 12 of which were permitted jointly under the MWCA and the Clean Water Act through Nationwide Permit 26 and one of which was permitted only under the MWCA. All 22 sites are spatially separate from each other, but many are components of larger wetland systems (Fig. 1).

All sites were located within a large hydrologically isolated basin in the suburban cities of Woodbury and Oakdale in Washington County east of St. Paul, Minnesota, within a water management area known as the "central draw" of the South Washington Watershed District (SWWD). The compensation sites were constructed on a variety of soils: some former wetland sites, some lateral expansions of existing wetlands, some established on non-wetlands soils. Only 6 of the 22 sites were constructed on soils characterized as poorly drained by the USDA, with the rest being characterized as moderately to excessively well-drained. In assessment, each compensation site was subdivided into "upland", "wet meadow", "emergent", and "open water" communities based primarily on the location of changes in community composition and evidence of inundation such as drift lines. "Upland" communities were surveyed but are not included in this analysis, as they did not provide wetland compensation credit under the system employed in Minnesota at the time of initial survey.

Of the 22 sites surveyed, only three receive no surface water flows from the municipal storm-sewer system, and four are hemi-marsh restoration sites bordering the main municipal stormwater detention areas in the middle of the basin. Thus, 19 of 22 sites experience unnatural hydrologic fluctuations, and the landscape context consists of a matrix of developed areas dominated by introduced plant and animal species (for an analysis of landscape changes affecting these compensation sites, see Robertson and Galatowitsch, forthcoming). The SWWD's central draw basin has been under rapid urban development since the late 1980s; Woodbury was the fastest-growing municipality in Minnesota in 1996, and grew from a population of 10,297 in 1980 to 61,961 in 2010. Rapid development led to increasing stress on the city's stormwater management system. Since the SWWD's central draw is a hydrologically isolated basin, stormwater is routed through a series of lakes and wetlands and pools at a depression in the rural southern part of the city, where it is lifted through a forcemain at a pumping station to a large depression that has been used successfully to dispose of stormwater through infiltration management. Most of the major lakes and wetlands of Woodbury are connected to the city's stormwater management system, and without a typical outlet the lower part of the system experiences extreme hydrological fluctuations due to direct runoff from the growing impervious areas and from a four-mile section of US Interstate 94 running through the northern end of the basin.

Through coordination between the Corps and the State of Minnesota, all sites were held to the same performance standards regardless of permitting authority. Until 1999, Corps Nationwide Permit 26 was frequently used to permit with minimal review those impacts under 10 acres (4.05 ha)—changing to under 3 acres (1.21 ha) after 1996—which were seen to have "minimal adverse effects on the aquatic environment" in isolated wetlands and headwaters (Corps 1998, p. 36040). Due to controversy concerning its overuse, NWP 26 was discontinued in March 1999.

We do not evaluate these sites against regulatory performance standards, because appropriate general standards are difficult to identify (Streever 1999; Doyle et al. 2015) and the specific standards applied to these sites varied permit-by-permit. We analyze relative change between surveys rather than absolute change with regard to a reference site, although we did collect data from regionally-appropriate reference sites as a part of the larger research effort; forthcoming work will provide calibration using the reference sites in analyzing landscape-level floristic and hydrologic change at these sites. The permits authorizing the

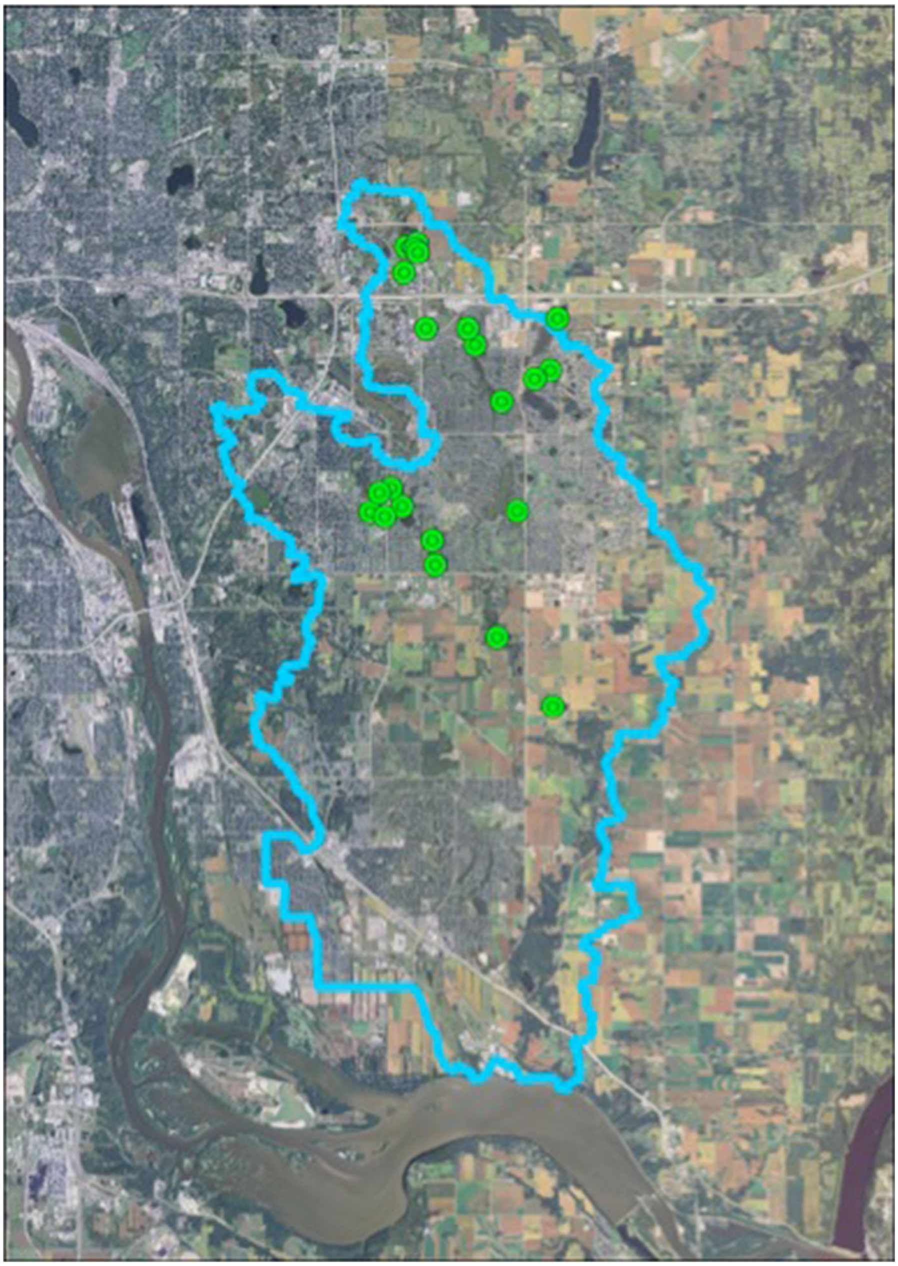

Fig. 1 The location of the 22 wetland compensation sites (green dots) in the South Washington Watershed District (in blue), on the east side of the Minneapolis-St. Paul metropolitan area. Water flow is from north to south through the watershed. (Color figure online)

wetland impacts issued by the St. Paul District of the US Army Corps of Engineers in the 1990s usually contained little guidance on what constituted a successful compensation site. One permit, issued January 18 1994 for the Marsh Creek subdivision in the SWWD, stated "The mitigation site shall be considered successful if it is constructed as shown in the mitigation plan and is dominated by hydrophytes" (Wopat 1994a). While the St. Paul District specified a recommended native seed mix, they did not require this mix to be used and only specified that the native seed mix be described by the permittee (although many permit files do not include this information). In the Marsh Creek permit, the St. Paul District Regulatory Branch Chief indicated: "We also request that you provide a description of seed mixtures used in mitigation areas" (Wopat 1994b). Many of the environmental engineers who performed the work for the permittees had their own in-house seed mix that had standing approval from the Corps. In at least one case no seed at all was used (Smyth 1994). In several other cases, the text of the project description strongly suggests that natural revegetation was expected to restore wetland habitat.

In spite of the unfavorable landscape context and the ambiguous standards frequently seen with older or "legacy" compensation sites, state and federal regulations suggest that vegetative assessment of such sites is both legitimate and important. The regulations of both the Clean Water Act and the Minnesota Wetland Conservation Act specified at the time that compensation sites should not be used primarily for stormwater detention, and must perform as functioning wetland ecosystems. Clarifying their powers of review under Section 401 of the Clean Water Act, the Minnesota Pollution Control Agency, with regard to one of the sites in this study, reminded the Corps that:

> … the creation of storm water management basins shall not be considered as wetland compensatory mitigation. If proposed creation of wetlands is primarily designed and constructed for the purpose of storm water retention, detention or sedimentation control, then the proposal cannot be considered wetland mitigation. (Holck 1995)

In light of this emphasis on habitat and vegetation in the assessment of wetland compliance, as opposed to a focus on hydrology, geomorphic characteristics, or concerns with landscape setting, our assessment focused solely on vegetation.

Data collection

At each site, the vegetation survey was limited to the area on which compensatory activities took place. A complete floristic inventory was complied, and species cover was estimated using a scale of seven cover classes (modified from Mueller-Dombois and Ellenberg 1974): one individual with insignificant cover; $<$ 1% cover; 1–5% cover; 5–25% cover; 25–50% cover; 50–75% cover, and; $>$ 75% cover. Cover was estimated within each wetland community type rather than for the site as a whole. Data on species cover were calculated by assigning each species a percentage cover value in the midpoint of the cover-class range. Summing these may produce a number under 100% if there are bare areas, or over 100% in cases where there are multiple canopy layers.

All samples were identified to the species level where possible, and where necessary assistance was obtained in species identification from faculty at the University of Minnesota and the University of Wisconsin Herbarium. Nomenclature follows the US Department of Agriculture PLANTS database. Samples identifiable only to the genus level are not included in the analysis. Out of 1528 occurrences, 27 were not identified to species in 1997, and 25 in 2010.

Data analysis

Changes in species richness, species composition, plant guild composition, and floristic quality were assessed for all sites combined as well as for individual sites. Floristic quality was assessed based on two indices: the Mean Coefficient of Conservatism (Mean C) and the Floristic Quality Index (FQI) (Swink and Wilhelm 1979, 1994). These indices are based on Coefficients of Conservatism (C-values) that range between 0 and 10, with 10 assigned only to species that are only found in non-degraded ecosystems. C-values for native species have been assigned by panels of regional experts in field botany, and we used the C-values developed by the Minnesota Pollution Control Agency (Milburn et al. 1997). Mean C is the mean of C-values for all native species found at a site, and FQI is the mean C multiplied by the square-root of native species richness. Species richness, Mean C, FQI, percent perennial species, and percent native species were compared between sample years using paired *t*-tests. Levene's test for homogeneity of variances was used to test for the convergence of plant richness and floristic quality metrics between sample years.

Plants were assigned to guilds based on origin (native or introduced), life span (annual or perennial), and growth form (herbaceous forb/vine, graminoid, or woody). Shapiro–Wilk normality tests revealed significant departure from normality for the relative cover of most guilds. Therefore, non-parametric Wilcoxon signed rank tests were used to determine if relative cover of native, annual, perennial, or woody species differed between sample years. Tests for changes in relative cover of guilds were done separately for wet meadow and emergent communities.

We used the metaMDS function in the R package vegan to perform non-metric multidimensional scaling (NMDS) and visualize trajectories of change in species composition between 1997 and 2010. To test the homogeneity of multivariate dispersions between sample years, we used the PERMDISP procedure (function betadisper in the R package vegan) to determine whether the 1997 dissimilarity values differed from the 2010 dissimilarity values in their degree of dispersion from the within-group centroid. Greater dispersion in 1997 relative to 2010 would suggest community convergence through time. NMDS and PERMDISP analyses were performed separately for species presence/absence data for the entire site, species cover data for the wet meadow zone, and species cover data for the emergent zone. NMDS and PERMDISP were based on Sørensen's index of dissimilarity for presence/absence data and Bray–Curtis dissimilarity for abundance data.

Results

Changes in species richness and floristic quality

At the level of the watershed or landscape, considering the pooled species counts of all sites, species richness increased with 219 species present in 2010 compared to 183 in 1997, a 20% gain (Table 1). In 2010, 106 new species were present (48.4% of the 2010 total, similar to Aronson and Galatowitsch's 49.3% after 9 years), while 73 had been lost (39.9% of the 1997 total, considerably higher than Aronson and Galatowitsch's 4.3% over 11 years). Of those species lost, 53% were perennial natives, almost all forbs and graminoids. Of those species gained, 67.9% were perennial natives (of which 28.3% were woody species), and only 21.7% of those gained were non-native. The guild with the greatest increase in richness was woody species, while richness of both native and introduced annuals decreased. Between 1997 and 2010, only one woody species was lost, while 43 were gained. Wet meadow zones gained richness while emergent and annual zones lost richness over the study period. Percent of native species, considering all sites together, rose only from 71 to 72% over the course of the study period, while Mean C and FQI increased for the flora as a whole (Table 8 in Appendix).

When sites are considered individually, species data showed significant increases in richness, FQI and percent perennial between surveys (Table 2). However, site averages of Mean C and percent native remained static. Mean C values across all sites showed strong convergence toward this static mean over the survey period, and percent of perennial species showed a similar convergence (Table 3). Richness, FQI and percent native showed no pattern of convergence over time.

The species recommended for planting at wetland compensation sites by the St. Paul District Corps office

Table 1 Species richness by community and guild, pooled from all sites

	Number of species					
	All years	1997	2010	% Change	New in 2010	Lost in 2010
Community						
Open water	9	6	6	0	3	3
Emergent	63	48	29	− 40	15	34
Sedge meadow	19	19	0	− 100	0	19
Wet meadow	266	156	198	27	101	68
Guild						
Introduced annual	26	21	17	− 19	4	9
Introduced perennial	31	26	19	− 27	4	12
Introduced woody	12	0	12		12	0
Native annual	38	27	24	− 11	11	14
Native perennial	139	96	102	6	30	37
Native woody	41	11	40	264	23	1
Total richness	287	181	214	18	84	73

Some species were present in more than one community. Annuals include biennials; perennials include vines; woody includes shrubs and trees

Table 2 Results of paired *t*-tests comparing mean plant richness and floristic quality variables between 1997 and 2010

	1997 Mean	2010 Mean	*T*	*p*
Richness	31.36	38.07	− 2.23	0.04
Mean C	3.26	3.24	0.17	0.86
FQI	14.15	15.86	− 1.92	0.07
% Native	71.06	71.82	− 0.31	0.76
% Perennial	76.00	83.37	− 3.67	0.001

Table 3 Results of Levene's test for homogeneity of variances between 1997 and 2010

	1997 Variance	2010 Variance	*F*	*p*
Richness	243.86	264.67	0.27	0.61
Mean C	0.50	0.11	6.93	0.01
FQI	20.77	22.40	0.45	0.51
% Native	87.69	54.00	0.85	0.36
% Perennial	76.52	28.28	5.88	0.02

A significantly smaller variance in 2010 suggests convergence of a variable through time

contributed only a small and decreasing amount (4.7% in 1997; 3.3% in 2010) to the total cover (Table 4). Once again, however, it is impossible to know if these species were actually planted at these compensation sites. Three of the 14 species on the planting list (*Calamagrostis canadensis*, *Angelica incarnata*, and *Thalictrum dasycarpum*) were not observed at any site in either survey year.

Changes in cover

Cover measurements between 1997 and 2010 indicated the development of a woody canopy layer and a corresponding loss of graminoids, the expansion of open water and a corresponding loss of emergent communities, and little change in cover of native species relative to introduced (Tables 5 and 6). Total cover in emergent communities fell by a small amount and remained below 100% indicating an expansion of bare or open areas. In spite of the considerable expansion of certain non-native species, relative cover of native species did not differ significantly between 1997 and 2010 in either wet meadow or emergent communities. Woody cover in wet meadows increased from 3 to 23% over the study period, but remained low in emergent communities (0.2% in 1997 and 0.7% in 2010). In wet meadow communities there was a significant decrease in relative cover of graminoids and a significant increase in relative cover of woody

Table 4 Frequency and cover of the species recommended by the St. Paul District of the Corps of Engineers for seeding in wet meadow areas of wetland compensation sites

	1997		2010	
	Sites present (%)	Cover	Sites present	Cover
Angelica incarnata	0	0.0	0	0.0
Andropogon gerardii	18	0.2	27	0.1
Asclepias incarnata	5	< 0.1	64	0.4
Aster simplex	59	1.5	36	1.3
Calamagrostis canadensis	0	0.0	0	0.0
Carex vulpinoidea	14	0.9	5	< 0.1
Eupatorum maculatum	9	< 0.1	9	0.1
Helenium autumnale	27	1.1	23	0.4
Panicum virgatum	14	0.3	9	0.1
Scirpus atrovirens	14	0.3	14	0.1
Spartina pectinata	14	0.3	23	0.5
Thalictrum dasycarpum	0	0.0	0	0.0
Verbena hastata	14	0.1	55	0.3
Veronicastrum virginianum	0	0.0	5	< 0.1
Average frequency	13		19	
Total cover		4.7		3.3

Table 5 Cover measures for three communities

Wetland zone and guild	1997 Cover	2010 Cover
Wet meadow (1997 $n = 22$, 2010 $n = 23$)		
Native	44	69
Introduced	53	79
Graminoid	68	68
Herbaceous forb/vine	22	46
Woody	3	35
Total cover	97	148
Emergent (1997 $n = 17$, 2010 $n = 17$)		
Native	31	21
Introduced	64	70
Graminoid	42	35
Herbaceous forb/vine	52	56
Woody	< 1	1
Total cover	95	91
Open water (1997 $n = 15$, 2010 $n = 16$)		
Native	40.	48
Introduced	0	0
Graminoid	0	0
Herbaceous forb/vine	40	48
Woody	0	0
Total cover	40.1	48.4

Two sedge meadow sites present in 1997 had transitioned to other communities by 2010

species between 1997 and 2010. In both wet meadow and emergent communities relative cover of annual species declined significantly.

Changes in species composition

In both wet meadow and emergent communities, the cover of the 10 species most frequent in site inventories declined, suggested increasing community evenness (Table 9 in Appendix). However, problematic invasive species increased in cover through time (Table 7). In wet meadows, the species that decreased the most in cover tended to be either wetland natives that are sensitive to hydrologic fluctuations (*Carex* sp.), or plants (especially introduced grasses) not typically found in saturated settings (*Elymus repens, Setaria viridis, Phleum pratense, Juncus tenuis*). Increases in cover came largely from aggressive introduced plants adapted to wetlands (*Phalaris arundinacea* and *Typha* × *glauca*), and in the woody stratum. In emergent communities, the most increases in cover as well as the most decreases in cover were found among aggressive introduced species as well as natives well-adapted to the setting. With few exceptions, however, individual species cover was fairly stable in all community types, with few species increasing or decreasing in cover by > 10%.

Non-metric multidimensional scaling of species composition, based on Sørensen's Index, revealed a clear pattern of separation between 1997 and 2010 points, indicating a common successional pattern among sites (Fig. 2). The results of the PERMDISP analysis, however, showed that there was no significant difference between the dispersion of 1997 sites from the 1997 within-group centroid and the dispersion of

Table 6 Results of Wilcoxon signed rank tests comparing mean relative cover of plant guilds between 1997 and 2010

	Mean relative cover (%)		W	p
	1997	2010		
Wet meadow ($n = 22$)				
Native	43.5	44.5	115	0.73
Annual	23.1	4.8	201	0.01
Graminoid	67.7	47.3	202	0.01
Herbaceous forb/vine	28.4	28.5	124	0.95
Woody	3.9	24.2	2	< 0.01
Emergent ($n = 14$)				
Native	36.6	21.5	72	0.24
Annual	21.0	0.6	65	< 0.01
Graminoid	28.9	41.4	40	0.46
Herbaceous forb/vine	70.9	57.8	66	0.43
Woody	0.2	0.8	1	0.20

Table 7 The ten species with the greatest increase or decrease in cover between 1997 and 2010, shown in percent of the total area of each community

Increase in cover		Decrease in cover	
Wet meadow			
<u>*Phalaris arundinacea*</u>	13.3	<u>*Phleum pratense*</u>	− 4.9
<u>*Typha*</u>× <u>*glauca*</u>	8.5	*Carex pellita*	− 4.7
<u>*Poa pratensis*</u>	8.1	<u>*Echinochloa crusgalli*</u>	− 4.3
Salix interior	6.5	*Panicum rigidulum*	− 4.2
Acer negundo	4.9	*Carex sychnocephala*	− 4.0
Agrostis stolonifera/gigantea	3.9	*Panicum dichotomiflorum*	− 3.3
Solidago canadensis	3.1	<u>*Elymus repens*</u>	− 3.0
Poa palustris	2.8	<u>*Setaria viridis*</u>	− 2.6
<u>*Lythrum salicaria*</u>	2.4	*Juncus tenuis* var. *tenuis*	− 1.4
Impatiens capensis	2.3	*Polygonum lapathifolium*	− 1.3
Emergent			
<u>*Typha*</u>× <u>*glauca*</u>	10.0	<u>*Echinochloa crusgalli*</u>	− 10.9
<u>*Phalaris arundinacea*</u>	8.7	*Alisma triviale/subcordatum*	− 7.4
Leersia oryzoides	5.2	*Polygonum lapathifolium*	− 4.0
Sparganium eurycarpum	3.7	*Polygonum amphibium* var. *emersum*	− 3.5
<u>*Phragmites australis*</u>	2.2	<u>*Melilotus officinalis*</u>	− 2.2
Scirpus cyperinus	1.3	*Cyperus lupulinus*	− 1.8
Eleocharis obtusa	0.9	*Schoenoplectus tabernaemontani*	− 1.6
Eleocharis palustris var. *palustris*	0.9	*Eleocharis acicularis*	− 1.2
Epilobium coloratum	0.9	*Polygonum pensylvanicum*	− 1.0
Salix interior	0.3	*Eleocharis engelmanii*	− 0.9
Open water			
Wolffia borealis	13.0	*Potamogeton foliosus/pusillus*	− 8.0
Ceratophyllum demersum	10.3	*Nymphaea odorata* ssp. *tuberosa*	− 1.0
Lemna minor	1.5	*Potamogeton nodosus*	− 1.0
Stuckenia pectinatus	1.2		
Elodea canadensis	0.2		

Introduced species are underlined

2010 sites from the 2010 within-group centroid ($F = 1.37, p = 0.27$), indicating a lack of convergence of species composition using presence-absence data.

The NMDS based on species cover data from wet meadow communities also suggested a successional change, indicated by the separation of 1997 and 2010 points along the second axis (Fig. 3). The data illustrate a clear shift toward woody species in addition to an increase in cover by *Phalaris arundinacea*. The PERMDISP analysis at the community-type level using Bray–Curtis distance indicates homogenization in species composition in wet meadows in 2010 relative to 1997 ($F = 6.92$, $p = 0.011$). The same convergence was not evident in PERMDISP analysis of the emergent communities ($F = 0.009$, $p = 0.923$). Instead, the NMDS suggests divergent polarizations toward emergent sites dominated by different invasive species (*Typha* × *glauca*, *Phalaris arundinacea*, or *Phragmites australis*; Fig. 4).

Discussion

The data reveal long-term trends in species richness, composition and cover at compensation sites, and suggest findings relevant to the issue of monitoring periods for compensatory mitigation sites. The overall pattern of development at the compensation sites in the SWWD shows some indications of stability, some indications of decline, and some indications of a trajectory of progress toward restoration goals between 1997 and 2010.

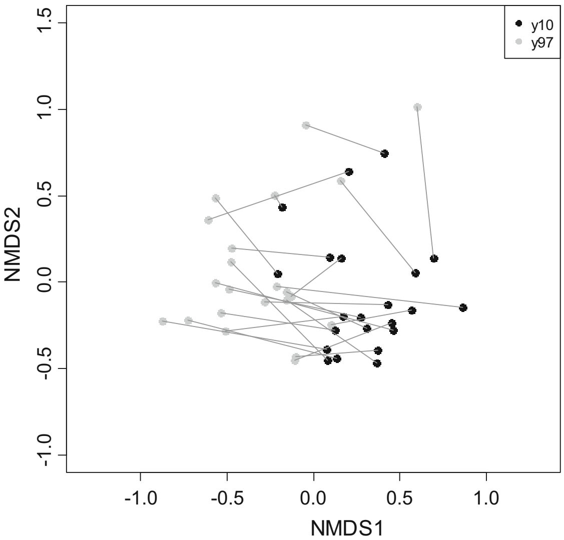

Fig. 2 NMDS graph of species composition, based on species presence-absence data and Sørensen's Index of dissimilarity, between 1997 sites (gray) and 2010 sites (black). The first two axes of a three-dimensional NMDS solution are shown (stress = 0.16)

The outcomes depend to some extent on the scale of observation. Considering all sites together as a landscape of compensation in the SWWD's central draw, Mean C and FQI were unchanged over the study period, whereas floral richness and percent perennial composition increased significantly. Introduced plants and native annual plants declined in richness, while woody and native perennial plants increased. The increase in species richness at all sites taken together is a far more modest than reported by Aronson and Galatowitsch (2008), who report a 94% increase in overall species richness at a larger population of similar restoration sites surveyed in their 3rd and 11th years. The landscape's wet meadows became significantly more woody and richer in species, but overall dominance by native cover was unchanged. The landscape's emergent communities, however, became less rich in species, and both overall and native cover declined. Thus while the landscape of compensation as a whole shows some evidence of progress toward restoration goals, data from community types show that wet meadows and emergent communities are trending in different directions, even where the two communities are both present at the same compensation sites.

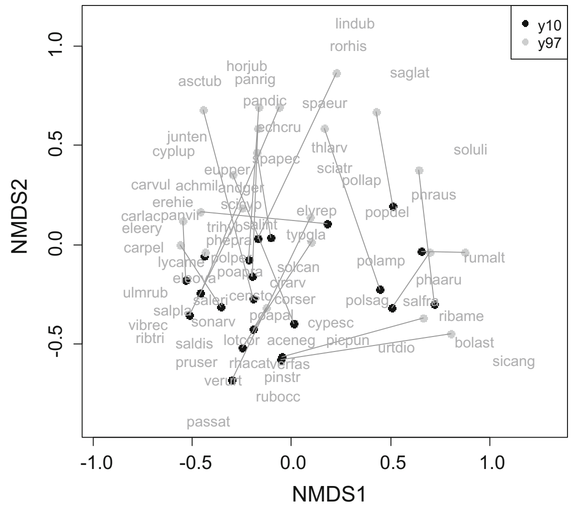

Fig. 3 NMDS graph of species composition, based on species cover data and Bray–Curtis dissimilarity, for wet meadow communities in 1997 sites (gray) and 2010 sites (black). The first two axes of a three-dimensional NMDS solution are shown (stress = 0.16). Species are indicated in gray font. If species names overlapped, priority was given to the species with the greatest summed cover. Species included: *Acer negundo* (aceneg), *Achillea millefolium* (achmil), *Andropogon gerardii* (andger), *Asclepias tuberosa* (asctub), *Boltonia asteroides* (bolast), *Carex lacustris* (carlac), *Carex pellita* (carpel), *Carex vulpinoidea* (carvul), *Centaurea stoebe* (censto), *Cirsium arvense* (cirarv), *Cornus sericea* (corser), *Cyperus esculentus* (cypesc), *Cyperus lupulinus* (cyplup), *Echinochloa crusgalli* (echcru), *Eleocharis erythropoda* (eleery), *Eleocharis ovata* (eleova), *Elymus repens* (elyrep), *Erechtites hieracifolia* (erehie), *Eupatorium perfoliatum* (eupper), *Hodeum jubatum* (horjub), *Juncus tenuis* (junten), *Lindernia dubia* (lindub), *Lotus corniculatus* (lotcor), *Lycopus americana* (lycame), *Panicum rigidulum* (panrig), *Panicum virgatum* (panvir), *Pastinaca sativa* (passat), *Phalaris arundinacea* (phaarun), *Phragmites australis* (phraus), *Picea pungens* (picpun), *Pinus strobus* (pinstr), *Pleum pratense* (phlpra), *Poa palustris* (poapal), *Poa pratensis* (poapra), *Polygonum amphibium* (polamp), *Polygonum lapathifolium* (pollap), *Polygonum persicaria* (polper), *Polygonum sagittatum* (polsag), *Populus deltoides* (popdel), *Prunus serotina* (pruser), *Rhamnus cathartica* (rhacat), *Ribes americanum* (ribame), *Ribes triste* (ribtri), *Rorippa palustris* var. *hispida* (rorhis), *Rubus occidentalis* (rubocc), *Rumex altissimus* (rumalt), *Sagittaria latifolia* (saglat), *Salix discolor* (saldis), *Salix eriocephala* (saleri), *Salix fragilis* (salfra), *Salix interior* (salint), *Salix planifolia* (salpla), *Scirpus atrovirens* (sciatr), *Scirpus cyperinus* (scicyp), *Sicyos angulatus* (sicang), *Solidago canadensis* (solcan), *Solidago uliginosa* (soluli), *Sonchus arvensis* (sonarv), *Sparganium eurycarpum* (spaeur), *Spartina pectinata* (spapec), *Thlaspi arvense* (thaarv), *Trifolium hybridum* (trihyb), *Typha* × *glauca* (typgla), *Ulmus rubra* (ulmrub), *Urtica dioica* (urtdio), *Verbena urticifolia* (verurt), *Vernonia fasciculata* (verfas), *Viburnum recognitum* (vibrec)

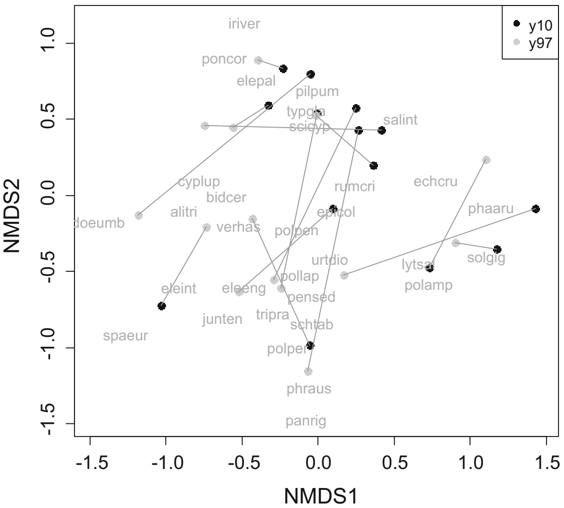

Fig. 4 NMDS graph of species composition, based on species cover data and Bray–Curtis dissimilarity, for emergent wetland communities in 1997 sites (gray) and 2010 sites (black). The first two axes of a three-dimensional NMDS solution are shown (stress = 0.14). Species are indicated in gray font. If species names overlapped, priority was given to the species with the greatest summed cover. Refer to Fig. 3 for species abbreviations; in addition, species included: *Alisma trivale* (alitri), *Bidens cernua* (bidcer), *Doellingeria umbellata* (doeumb), *Eleocharis engelmanii* (eleeng), *Eleocharis intermedia* (eleint), *Eleocharis palustris* (elepal), *Epilobium coloratum* (epicol), *Iris versicolor* (iriver), *Lythrum salicaria* (lytsal), *Penthorum sedoides* (pensed), *Pilea pumila* (pilpum), *Polygonum pensylvanicum* (polpen), *Pontedaria cordata* (poncor), *Rumex crispus* (rumcri), *Schoenoplectus tabernaemontani* (schtab), *Solidago gigantea* (solgig), *Trifolium pratense* (tripra), *Verbena hastata* (verhas)

The connection of most compensation sites to the SWWD stormwater management system means that they experience hydrologic fluctuations that are likely to be most damaging to emergent communities. Emergent communities are losing both native and overall cover, and losing native richness. Wet meadow communities, which experience less hydrologic fluctuation, are increasing in cover complexity and maintaining native cover and native richness, as well as overall cover. Hydrologic disturbance from stormwater outlets and drainage ditches has been associated with decreased plant species richness in wetlands (Kercher et al. 2004), and compensation wetlands exposed to fluctuating stormwater inputs are subject to scouring, plant stress, and loss of diversity (Mitsch and Wilson 1996).

Even though wet meadow areas of the compensation sites gained overall richness, this did not translate into cover diversity: the ten most dominant species in wet meadows constituted nearly as much of the cover in 2010 as in 1997. In emergent communities, dominant species actually constituted more of the cover in 2010. Thus, aggressive introduced species such as *Typha* × *glauca* maintained or expanded their dominance of sites even as a greater diversity of species found small footholds in wet meadow communities; this dynamism is concealed if one only looks at the percentage of natives and of native cover, which remained static. Data from both the wet meadow and emergent zones indicate that aggressive introduced species are increasing both in frequency and in cover. The pattern of changes in cover by species suggest that in wet meadows, species not suited to the hydrologic regime typical of a wet meadow are exiting, while wet meadow natives are increasing. In emergent marshes, by contrast, even natives that are tolerant of the typical hydrology of emergent communities are probably exiting under the pressure of strong hydrologic fluctuations related to the operation of the municipal stormwater system.

There was a clear increase in woody species richness, which is an expected pattern of site development in this region in isolated basin wetland restorations that are not maintained through prescribed burning or cutting. The increase in woody and perennial species richness and cover, and the decrease in annual species, reflect the general pattern of changes in plant life history strategies observed in other studies of wetland succession (Dunn and Sharitz 1987; Odland and del Moral 2002; Matthews and Endress 2010). Whether or not the development of a woody canopy is a desirable feature of wetland compensation sites is debatable: the MDOT design criteria suggest that they are meant to mimic “prairie pothole”-type isolated basins (typically dominated by herbaceous cover) even though the SWWD is in an area originally dominated by deciduous woods rather than tallgrass prairies. This is a decision that must be made by regulators, but it is a clear and predictable trend at these sites.

At the scale of individual sites’ floristic trends, the outcome differs from results at the scale of the landscape as a whole. This is important because it is the site scale—rather than the quality of the entire assemblage or of wetland community types—at which compliance with the Clean Water Act is determined. Despite considerable dynamism in species composition, especially in the case of wet meadow communities, almost all sites gained species, and gained native species, over the course of the study. This is consistent with other studies that have reported increases in species richness over the first 8–15 years following wetland restoration (Moore et al. 1999; Aronson and Galatowitsch 2008; Gutrich et al. 2009; Matthews et al. 2009). The data in Table 8 in Appendix suggest that gain in richness was concentrated in those sites that were particularly depauperate in 1997. In Aronson and Galatowitch's (2008) study, nearly all the increase in richness occurred during the first decade of site development, increasing by 94% after 11 years, but only by another 5% by 18 years.

Some researchers have observed a convergence or homogenization of compensation and restoration sites in a landscape, which suggests that higher-quality sites will not maintain a desirable trajectory when the condition of most similar sites in the landscape is lower (Brooks et al. 2005; Aronson and Galatowitsch 2008; Matthews and Spyreas 2010). At the study sites there was significant convergence in the species composition of wet meadow communities, and a significant shift toward woody species. The same convergence was not seen in emergent communities. However, although emergent communities were not converging among all sites, the NMDS plot suggested that emergent communities may have been converging on a few alternative states, each dominated by a different clonal perennial species. Taken together, these results suggest that wet meadow compensation sites in a landscape may become richer but converge on a common species composition, while emergent compensation sites that are subject to hydrologic impacts may lose richness while not universally converging in composition (Table 9).

The assumption that wetland compensation sites will develop along a trajectory that leads to increased ecological integrity is not supported by this study. Although species richness increased in most sites, it is apparent that the emergent marsh components of these compensation sites have become more degraded and the heterogeneity of dominant cover has decreased. Average site-level FQI, Mean C, and percent native species did not increase appreciably between 1997 and 2010. Furthermore, aggressive, introduced species increased in dominance in both emergent and wet meadow communities, similar to trends observed in other studies of restored wetlands (Reinartz and Warne 1993; Moore et al. 1999; Aronson and Galatowitsch 2008). It is possible that better site selection and the setting of appropriate hydrologic performance standards would have resulted in a trajectory toward ecological integrity. These regulatory decisions may be science-based, but they are also informed by the social and political context in which Clean Water Act policy is executed Future research on compensation sites should address the effect of the social and economic context on site success, and the way that these practices affect regulatory decision-making. Elements such as development patterns in the surrounding landscape, regulators' beliefs about adequate site conditions, and changes in site development and maintenance practices could potentially be included within an expanded and multi-disciplinary concept of a site's trajectory.

Structural and functional indicators measured in restored wetlands often fail to follow desired or expected trajectories or develop very slowly through time (Morgan and Short 2002; Whigham et al. 2002; Craft et al. 2003; Woodcock et al. 2011; Matthews 2015). In the SWWD landscape the decline in floristic indicators in emergent marsh communities is likely due to the continuing exposure of many sites to hydrologic stressors, and means that the study overall does not support that 5 years of monitoring can assure the ongoing biotic integrity of wetland compensation sites typical of suburban and exurban areas in the mid-continental US.

The results also suggest the importance of appropriate site selection in compensation practice: where compensation is allowed to occur in locations that are subject to known and foreseeable stormwater impacts, compensation wetlands may suffer decreases in richness, cover density and cover heterogeneity no matter how long the monitoring period is. The Corps and state regulators have considerable flexibility to approve only compensation sites likely to meet the goals of the Clean Water Act permit program, and the 2008 federal compensation rule at 33 CFR 230.93(d) requires that "the compensatory mitigation project site must be ecologically suitable for providing the desired aquatic resource functions." In making this determination, the Corps "must" take into account many factors including hydrological conditions, watershed-scale features, "compatibility with adjacent land-uses and watershed management plans," and "other relevant factors including, but not limited to, development trends

[and] anticipated land use changes". The regulatory mandate and leverage exists to avoid the problems seen in the SWWD.

Finally, the lack of comparable studies on wetland compensation sites underscores the need for additional longitudinal studies, as well as a programmatic approach to compensation performance evaluation on the part of the CWA implementing agencies that will allow for the observation and interpretation of regional and national trends in the development of wetland compensation sites. Hundreds of millions of dollars are spent every year to restore tens of thousands of acres of wetlands under the aegis of the Section 404 permit program of the CWA (ELI 2007), representing an experiment in applied wetland ecology for which federal and state agencies should develop a sampling and evaluation regime robust enough to reveal trends and guide policy.

Funding In support of this research, the first author received a small grant from the University of Kentucky Associate Dean for Research.

Appendix

See Tables 8 and 9.

Table 8 Species richness and floristic quality for 1997 and 2010 at 22 of the 23 wetland compensation sites in the SWWD established between January 1992 and July 1995

Site name	# Species			% Native			Mean C			FQI		
	1997	2010	% Change	1997	2010	% Change	1997	2010	% Change	1997	2010	% Change
Bailey Ridge	30	48	60	63	77	22	3.28	3.19	− 3	13.91	18.03	30
Company× South	36	25	− 31	53	64	21	2.35	2.93	25	9.70	11.36	17
Copper Oaks	58	39	− 33	69	74	8	3.51	3.48	− 1	21.37	17.40	− 19
Dale Rd.	29	23	− 21	72	74	2	4.26	3.56	− 16	18.58	14.25	− 23
Donnay's East	52	58	12	62	81	32	2.87	3.41	19	15.70	22.61	44
Donnay's West	33	29	− 12	70	62	− 11	2.70	3.24	20	12.07	13.34	10
Fox Run East	51	51	0	57	61	7	2.54	2.96	17	13.42	14.80	10
Fox Run West	46	64	39	78	80	2	3.09	3.63	18	17.76	25.43	43
Lake Place	16	39	144	69	72	4	3.00	2.73	− 9	9.00	12.79	42
Markgrafs Lake	7	27	286	86	56	− 35	2.40	2.58	8	5.37	8.95	67
Marsh Creek East	39	69	77	69	78	13	3.35	3.07	− 8	16.06	20.57	28
Marsh Creek West	32	39	22	72	79	11	3.90	3.69	− 5	17.44	19.87	14
Oak Run Shores 1	32	43	34	78	70	− 11	3.80	3.70	− 3	19.00	19.25	1
Oak Run Shores 2	23	20	− 13	65	70	7	3.67	3.46	− 6	14.20	12.48	− 12
Oak Run Shores 3	26	52	100	73	71	− 3	2.94	3.40	15	12.49	20.11	61
Oak Run Shores 4	31	38	23	84	82	− 3	3.13	3.33	7	15.31	18.26	19
Oak Run Shores 6	24	52	117	83	65	− 22	3.78	3.30	− 13	16.03	18.97	18
Parkside	65	50	− 23	62	76	24	3.15	3.34	6	20.15	18.92	− 6
Pendryn Hill	20	11	− 45	75	73	− 3	5.08	2.88	− 43	17.61	8.13	− 54
Pioneer Drive	13	19	46	54	68	27	2.17	2.58	19	5.31	8.95	69%
State Farm	13	23	77	85	70	− 18	4.09	3.42	− 16	13.57	11.84	− 13
Wedgewood North	12	18	50	75	83	11	2.71	3.36	24	7.18	12.56	75
Total flora	181	214	18	74	78	5	3.50	3.65	4	39.49	44.27	13
Average for sites	31.1	38.0	41	71	72	2	3.26	3.24	− 7	14.14	15.85	12
SD			0.75			0.17	0.70	0.34		4.56	4.73	

Table 9 Cover and frequency for the most frequently-encountered species based on presence at sites

1997			2010		
Species	Frequency (%)	Cover	Species	Frequency (%)	Cover
Wet meadow					
Phalaris arundinacea	82	36.8	*Cirsium arvense*	86	1.6
Rumex crispus	77	1.3	*Phalaris arundinacea*	86	41.7
Typha × *glauca*	86	33.2	*Solidago canadensis*	73	5.0
Populus deltoides ssp. *monilifera*	59	0.4	*Acer negundo*	68	5.0
Symphyotrichum lanceolatum ssp. *lanceolatum*	59	1.5	*Asclepias syriaca*	68	0.5
Solidago canadensis	55	1.8	*Poa pratensis*	68	10.3
Salix interior	59	1.5	*Salix interior*	68	8.0
Scirpus cyperinus	64	1.9	*Asclepias incarnata* ssp. *incarnata*	64	0.4
Echinochloa crusgalli	73	13.6	*Cornus sericea* ssp. *sericea*	64	1.7
Setaria viridis	50	2.9	*Rumex crispus*	64	0.4
Total		94.8	Total		74.4
Emergent					
Typha × *glauca*	86	33.2	*Typha* × *glauca*	82	49.5
Phalaris arundinacea	82	36.8	*Phalaris arundinacea*	47	19.6
Scirpus cyperinus	64	1.9	*Scirpus cyperinus*	29	2.4
Schoenoplectus tabernaemontani	50	1.8	*Schoenoplectus tabernaemontani*	29	0.3
Leersia oryzoides	36	2.4	*Leersia oryzoides*	29	7.1
Alisma triviale/subcordatum	41	6.5	*Alisma triviale/subcordatum*	24	1.0
Polygonum pensylvanicum	45	2.4	*Polygonum pensylvanicum*	18	0.2
Salix interior	59	1.5	*Penthorum sedoides*	12	0.0
Lythrum salicaria	18	0.1	*Eleocharis* spp.	12	0.2
Salix petiolaris	14	0.1	*Salix interior*	12	0.4
Total		86.6	Total		80.8
Open water					
Lemna minor	67	20.8	*Lemna minor*	94	22.3
Potamogeton spp.	50	16.9	*Wolffia borealis*	31	13.0
Nymphaea odorata ssp. *tuberosa*	20	1.4	*Elodea canadensis*	25	1.2
Elodea canadensis	7	1.0	*Stuckenia pectinatus*	25	1.2
Potamogeton nodosus	7	1.0	*Ceratophyllum demersum*	19	10.3
Utricularia macrorhiza	7	0.0	*Nymphaea odorata* ssp. *tuberosa*	13	0.4
Total		41.1	Total		48.4

References

Aronson MFJ, Galatowitsch SM (2008) Long-term vegetation development of restored prairie pothole wetlands. Wetlands 28:883–895

Balcombe CK, Anderson JT, Fortney RH, Rentsch JS, Grafton WM, Kordek WS (2005) A comparison of plant communities in mitigation and reference wetlands in the mid-Appalachians. Wetlands 25:130–142

Bernhardt ES, Palmer MA, Allan JD, Alexander G, Barnas K, Brooks S, Carr J, Clayton S, Dahm C, Follstad-Shah J, Galat D, Gloss S, Goodwin P, Hart D, Hassett B, Jenkinson R, Katz S, Kondolf GM, Lake PS, Lave R, Meyer JL, O'Donnell TK, Pagano L, Powell B, Sudduth E (2005) Synthesizing U.S. river restoration efforts. Science 308:636–637

Breaux A, Serefiddin F (1999) Validity of performance criteria and a tentative model for regulatory use in compensatory wetland mitigation permitting. Environ Manag 24:327–336

Brooks RP, Wardrop DH, Cole CA, Campbell DA (2005) Are we purveyors of wetland homogeneity? A model of degradation and restoration to improve wetland mitigation performance. Ecol Eng 24:331–340

Brown SC, Veneman PLM (2001) Effectiveness of compensatory wetland mitigation in Massachusetts, USA. Wetlands 21:508–518

Cammen LM (1976a) Macroinvertebrate colonization of *Spartina* marshes artificially established on dredge spoil. Estuar Coast Mar Sci 4:357–372

Cammen LM (1976b) Abundance and production of macroinvertebrates from naturally and artificially established salt marshes in North Carolina. Am Midl Nat 96:487–493

Clewell AF, Aronson J (2013) Ecological restoration: principles, values and structure of an emerging profession, 2nd edn. Island Press, Washington

Corps [US Army Corps of Engineers] (1998) Proposal to issue and modify nationwide permits. Fed Regist 63(126):36040–36078

Corps [US Army Corps of Engineers] (1999) Army Corps of Engineers Standard Operating Procedures for the Regulatory Program: Part II: Regulatory Program Priorities. USACE, Washington, DC

Corps [US Army Corps of Engineers], EPA [U.S. Environmental Protection Agency] (2006) Compensatory mitigation for losses of aquatic resources. Fed Regist 71:15520–15556

Corps [U.S. Army Corps of Engineers], EPA [U.S. Environmental Protection Agency] (2008) Compensatory mitigation for losses of aquatic resources. Fed Regist 73:19594–19705

Craft C, Megonigal P, Broome S, Stevenson J, Freese R, Cornell J, Zheng L, Sacco J (2003) The pace of ecosystem development of constructed *Spartina alterniflora* marshes. Ecol Appl 13:1417–1432

Doyle MW, Lave R, Robertson M (2013) River federalism. Ann Assoc Am Geogr 103:290–298

Doyle MW, Singh J, Lave R, Robertson M (2015) The morphology of streams restored for market and nonmarket purposes: insights form a mixed natural-social science approach. Water Resour Res 51:5603–5622

Dunn CP, Sharitz RR (1987) Revegetation of a *Taxodium-Nyssa* forested wetland following complete vegetation destruction. Vegetatio 72:151–157

ELI [Environmental Law Institute] (2007) Mitigation of impacts to fish and wildlife habitat: estimating costs and identifying opportunities. ELI, Washington, DC

Erwin K (1991) An evaluation of wetland mitigation in the South Florida Water Management District, vol 1. South Florida Water Management District, West Palm Beach

GAO [US Government Accountability Office] (2005) Corps of Engineers does not have an effective oversight approach to ensure that compensatory mitigation is occurring. Report GAO-05-898. GAO, Washington, DC

Gutrich JJ, Hitzhusen FJ (2004) Assessing the substitutability of mitigation wetlands for natural sites: estimating restoration lag costs of wetland mitigation. Ecol Econ 48:409–424

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. Ecol Eng 35:351–368

Holck J (1995) Letter RE: CENCS-CO-R 94-06674-IP-RJA Arcon Development and City of Oakdale–Oak Run Shores, Washington County. May 18 1995, Minnesota Pollution Control Agency. Document held in City of Oakdale Engineering Office, Oak Run Shores File

Hough P, Robertson M (2009) Mitigation under Section 404 of the Clean Water Act: where it comes from, what it means. Wetl Ecol Manag 17:15–33

Kelly NM (2001) Changes to the landscape pattern of coastal North Carolina wetlands under the Clean Water Act, 1984–1992. Landscape Ecol 16:3–16

Kercher S, Carpenter QJ, Zedler JB (2004) Interrelationships of hydrologic disturbance, reed canary grass (*Phalaris arundinacea* L.), and native plants in Wisconsin wet meadows. Nat Areas J 24:316–325

Kozich AT, Halvorsen KE (2012) Compliance with wetland mitigation standards in the Upper Peninsula of Michigan, USA. Environ Manag 50:97–105

Kusler JA, Kentula ME (1990) Wetland creation and restoration: the status of the science. Island Press, Washington, DC

Mack JJ, Miccachion M (2006) An ecological assessment of Ohio mitigation banks: vegetation, amphibians, hydrology and soils. Ohio EPA Technical Report WET/2006-1. Ohio Environmental Protection Agency, Division of Surface Water, Wetland Ecology Group, Columbus

Matthews JW (2015) Group-based modeling of ecological trajectories in restored wetlands. Ecol Appl 25:481–491

Matthews JW, Endress AG (2008) Performance criteria, compliance success, and vegetation development in compensatory mitigation wetlands. Environ Manag 41:130–141

Matthews JW, Endress AG (2010) Rate of succession in restored wetlands and the role of site context. Appl Veg Sci 13:346–355

Matthews JW, Spyreas G (2010) Convergence and divergence in plant community trajectories as a framework for monitoring wetland restoration progress. J Appl Ecol 47:1128–1136

Matthews JW, Spyreas G, Endress AG (2009) Trajectories of vegetation-based indicators used to assess wetland restoration progress. Ecol Appl 19:2093–2107

Milburn SA, Bourdaghs M, Husveth JJ (1997) Floristic quality assessment for Minnesota Wetlands. Minnesota Pollution Control Agency, St. Paul

Miller SJ, Pruitt BA, Theiling CH, Fischenich JC, Komlos SB (2012) Reference concepts in ecosystem restoration and environmental benefits analysis (EBA): Principles and Practices. ERDC Report TN-EMRRP-EBA-12. June 2012. US Army Corps of Engineers ERDC, Vicksburg

Mitsch WJ, Wilson RF (1996) Improving the success of wetland creation and restoration with know-how, time, and self-design. Ecol Appl 6:77–83

Moore HH, Niering WA, Marsicano LJ, Dowdell M (1999) Vegetation change in created emergent wetlands (1988–1996) in Connecticut (USA). Wetl Ecol Manag 7:177–191

Morgan KL, Roberts TH (2003) Characterization of wetland mitigation projects in Tennessee, USA. Wetlands 23:65–69

Morgan PA, Short FT (2002) Using functional trajectories to track constructed salt marsh development in the Great Bay Estuary, Maine/New Hampshire, USA. Restor Ecol 10:461–473

Mueller-Dumbois D, Ellenberg H (1974) Aims and methods of vegetation ecology. Wiley, New York

Mulhouse JM, Galatowitsch SM (2003) Revegetation of prairie pothole wetlands in the mid-continental US: twelve years post-reflooding. Plant Ecol 169:143–159

NRC [National Research Council] (2001) Compensating for wetland losses under the Clean Water Act. National Academy Press, Washington, DC

Odland A, del Moral R (2002) Thirteen years of wetland vegetation succession following a permanent drawdown, Myrkdalen Lake, Norway. Plant Ecol 162:185–198

Race MS (1985) Critique of present wetlands mitigation policies in the United States based on an analysis of past restoration projects in San Francisco Bay. Environ Manag 9:71–82

Race MS, Christie DR (1982) Coastal zone development: mitigation, marsh creation, and decision-making. Environ Manag 6:317–328

Reinartz JA, Warne EL (1993) Development of vegetation in small created wetlands in southeastern Wisconsin. Wetlands 13:153–164

Seneca ED, Broome SW, Woodhouse WW, Cammen LM, Lyon JT (1976) Establishing *Spartina alterniflora* marsh in North Carolina. Environ Conserv 3:185–188

Smyth J (1994) Technical Memorandum: Proposed mitigation for 0.37 acres of wetland filled by Pioneer Drive reconstruction. Our File No. 31422. Bonestron, Rosene, Anderlik and Associates, St. Paul, MN, March, 4, 1994. Held in City of Woodbury Engineering Office, Pioneer Drive File

Spieles DJ (2005) Vegetation development in created, restored, and enhanced mitigation wetland banks in 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. Environ Manag 38:837–852

Stefanik KC, Mitsch WJ (2012) Structural and functional vegetation development in created and restored wetland mitigation banks of different ages. Ecol Eng 39:104–112

Streever WJ (1999) Examples of performance standards for wetland creation and restoration in Section 404 permits and an approach to developing performance standards. WRP Technical Notes Collection. U.S. Army Engineer Research and Development Center, Vicksburg

Sudol MF, Ambrose RF (2002) The US Clean Water Act and habitat replacement: evaluation of mitigation sites in Orange County, California, USA. Environ Manag 30:727–734

Swink F, Wilhelm G (1979) Plants of the Chicago Region, revised ed. The Morton Arboretum, Lisle

Swink F, Wilhelm G (1994) Plants of the Chicago region, 4th edn. The Morton Arboretum, Lisle

Van den Bosch K, Matthews JW (2017) An assessment of long-term compliance with performance standards in compensatory mitigation wetlands. Environ Manag 59:546–556

Whigham D, Pittek M, Hofmockel KH, Jordan T, Pepin AL (2002) Biomass and nutrient dynamics in restored wetlands on the outer coastal plain of Maryland, USA. Wetlands 22:562–574

Woodcock BA, McDonald AW, Pywell RF (2011) Can long-term floodplain meadow recreation replicate species composition and functional characteristics of target grasslands? J Appl Ecol 48:1070–1078

Wopat B (1994a) Letter from Ben Wopat, US Army Corps of Engineers, to Scott Johnson, Arcon Development-Sienna Corporation Joint Venture. St. Paul, MN, January 18 1994. Held in City of Woodbury Engineering Office, Marsh Creek File

Wopat B (1994b) Letter from Ben Wopat, US Army Corps of Engineers, to Mark Putnam, Charles Cudd Co. St. Paul, MN, August 31 1994. Held in City of Woodbury Engineering Office, Marsh Creek File

Zedler JB (2004) Compensating for wetland losses in the United States. IBIS 146:92–100

Zedler JB, Callaway JC (1999) Tracking wetland restoration: do mitigation sites follow desired trajectories? Restor Ecol 7:69–73