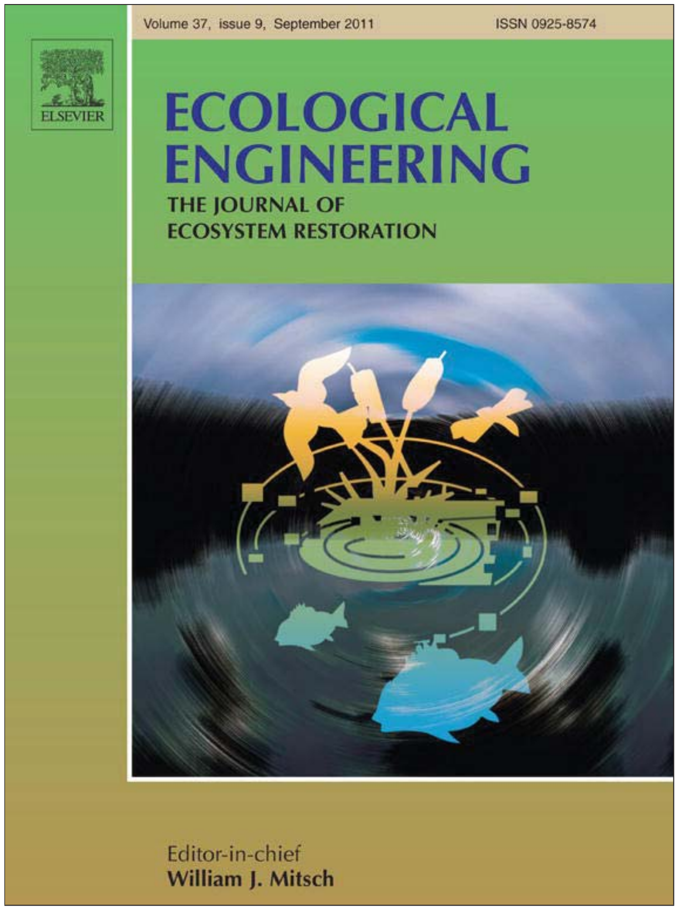
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Early development of plant community in a created mitigation wetland as affected by introduced hydrologic design elements

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ABSTRACT

The goal of this study was to investigate early plant community development in two sites (i.e., higher and lower elevation sites—LC1 and LC2, respectively) of the Loudoun County (LC) mitigation wetland created in the Virginia Piedmont. The effects of hydrologic design elements incorporated during the construction (i.e., disking-induced microtopography—MT and site level elevation difference) on vegetative and hydrologic attributes (e.g., species richness, biodiversity, plant cover, floristic quality assessment index, wetland indicator status, soil moisture content, and water table depths) were investigated. The study was conducted at the end of two growing seasons in 2008 and 2009 (i.e., second and third growing seasons). Drought conditions that persisted into the second growing season resulted in the abundance of a seeded cover grass, *Lolium multiflorum* (Italian ryegrass), intentionally planted during initial seeding for erosion control. *L. multiflorum* was subsequently phased out and replaced by facultative wet and obligate wetland species by the third growing season when above-average precipitation occurred, leading to a decrease in total percent cover. Prevalence index changed in both LC1 and LC2, dropping from overall facultative status in 2008 to obligate status in 2009 when obligate species such as *Bidens cernua*, *Carex frankii*, and *Juncus effusus* thrived and expanded with the change of hydrologic regime in 2009. Microtopographic treatment (i.e., disked or undisked) positively influenced vegetation development for LC1 site in 2008, but the positive influence was not consistent for LC2, which experienced an additional month of standing water conditions in the same year, nor for either site in 2009 when above-average precipitation occurred. The site differences in elevation, and thus hydrologic regime, seemed to overwhelm the effects on vegetation of disking-induced microtopography in each site when precipitation was at or above average range. Although shown on short spatial and temporal scales in this study, incorporation of micro- and macrotopographic design elements in creating a mitigation wetland can be beneficial to the early development of diverse vegetation communities wetland under varying climate conditions.

1. Introduction

Wetlands are created to mitigate the loss of natural wetlands under Section 404 of the Clean Water Act in the United States (Zedler and Callaway, 1999; Brown and Veneman, 2001; National Research Council, 2001). Post-creation monitoring is usually conducted to determine if mitigation wetlands are successfully supporting native wetland vegetation. Common vegetation standards are mostly structural, including targets for percent cover of hydrophytic vegetation, limits for nuisance species cover, and goals for survival of planted stock (Spieles, 2005). Structural attributes of vegetation may not fully measure ecological functions that should be in development, but have been used as a quick surrogate for biogeochemical condition and as a measure of mitigation success (Breaux and Serefiddin, 1999; Spieles, 2005).

Created wetlands are often found extremely limited in soil nutrients and organic matter, which may inhibit plant development (Campbell et al., 2002; Hoeltje and Cole, 2007; Ahn and Peralta, 2009; Moser et al., 2009). This may be due in part to methods/designs of wetland construction that usually remove and/or compact the A-horizon of soil, typically high in organic matter and rich in nutrients, with heavy machinery to form a depression. Soil compaction typical of most created wetlands might limit vegetation growth and biomass production.

Early vegetation establishment can have profound implications for the development of structure and function of created mitigation wetlands (Atkinson et al., 2005; Spieles, 2005), as the vegetation

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community influences the maturation of biogeochemical functions, such as primary productivity, soil physicochemistry, and nutrient accumulation. Wetland mitigation success thus often relies on the early establishment and development of wetland vegetation communities. However, vegetation in created wetlands often displays lower species richness, less cover, higher occurrence of non-native or invasive species, and fewer obligate wetlands species than that of natural wetlands (Galatowitsch and van der Valk, 1996; Zedler and Callaway, 1999; Campbell et al., 2002; Spieles, 2005). Thus, identifying wetland creation methods and/or design elements that enhance the development of a diverse, native wetland plant community may increase the likelihood of mitigation success.

The hydrologic regime plays a pivotal role in the growth, distribution, and abundance of wetland plants (van der Valk and Davis, 1978; Cronk and Fennessy, 2001; Ahn et al., 2004; Bruland and Richardson, 2005; Mitsch and Gosselink, 2007). Effects of temporal and spatial variability of soil moisture content and water table depth on wetland plant structure and composition have been extensively studied (Titus, 1990; Vivian-Smith, 1997; Bruland and Richardson, 2005; Dwire et al., 2006; Touchette et al., 2010; Yu and Ehrenfeld, 2010). Hydrologic conditions in wetlands are often characterized by micro- and macrotopographic variability within the site (Moser et al., 2007; Ahn and Peralta, 2009). This is truer with certain types of wetlands such as palustrine forested wetlands or alluvial swamps that have a signature hydroperiod defined as "saturated" or "temporarily flooded" where standing water is present rarely or for a short period of time (Mitsch and Gosselink, 2007). Wetlands with topographic variability include both flooded and non-flooded areas with varying moisture levels or depths to water table, supporting a variety of wetland species, and thus influencing plant community development (Moser et al., 2007).

Microtopography is important in wetlands, especially where <5 cm of variation in elevation can shift hydrologic conditions experienced by individual plant species (Vivian-Smith, 1997; Werner and Zedler, 2002; Peach and Zedler, 2006; Moser et al., 2007). At low standing water level conditions, microhighs may be exposed to aerobic conditions while microlows and flats are still under water in anaerobic conditions, creating ecological niches or a mosaic of microenvironments that vary in abiotic and biotic conditions. These microenvironments influence seed germination, seedling establishment, and seedling growth; critical phases in the life cycle of plants (Harper et al., 1965; Smith and Capelle, 1992; Schupp, 1995). Microenvironments also facilitate or inhibit plant species distribution by influencing resource heterogeneity (Baer et al., 2005; Moser et al., 2007), interspecific differences in habitat preferences (Vivian-Smith, 1997), and differential mortality and growth rates at different microtopographic positions (Jerling, 1981; Hamrick and Lee, 1987). Created mitigation wetlands frequently lack microtopography due to use of heavy machinery for grading during the construction process when compared to natural wetlands that often have a heterogeneous soil surface topography (Whittecar and Daniels, 1999; Stolt et al., 2000; Moser et al., 2007).

The goal of this study was to investigate early development of wetland plant community in a mitigation wetland created in the Virginia piedmont. The first objective of the study was to evaluate several attributes of vegetation for two early years (i.e., the second and the third years of creation), including plant cover, species richness (S), diversity (H'), wetland indicator status/prevalence index (WIS/PI), and floristic quality assessment index (FQAI). The second objective was to assess the effects of hydrologic design elements (i.e., site-level elevation difference and disking-induced microtopography) incorporated during construction on all vegetation attributes studied.

2. Methods

2.1. Site and study plot description

Field research was performed in the Loudoun County mitigation wetland (LC) (39°02.05′N, 77°36.5′W), a 0.13-km^2 wetland mitigation bank and upland buffer complex created in the northern Piedmont region of Virginia, just about 40 km southwest of Washington, DC. The study was conducted during its second and third growing seasons in 2008 and 2009 [annual precipitation-norm 83.6 cm (95.1 cm in 2008 and 100.7 cm in 2009); mean annual temperature-norm 12 °C (13 °C in both 2008 and 2009)]. LC was constructed in 2006 by Wetland Studies and Solutions, Inc. (WSSI) and is located within the floodplain of Big Branch Creek (to the east), a tributary of Goose Creek in Loudoun County. The wetland design includes 0.2 m topsoil atop a 0.3 m or thicker low permeability subsoil layer, resulting in a perched water table that fluctuates with precipitation. LC receives surface water runoff from an upland housing development and forested buffer, as well as a negligible amount of groundwater from toe-slope intercept seepage. The LC wetland was hydro-seeded in the summer of 2006 with a commercially available wetland plant seed mix of 26 herbaceous plant species (~1.5 g m^{-2}) appropriate for the region and the intended hydrology (e.g., wetland meadow as opposed to obligate wetland). LC is intended to eventually mitigate the loss of palustrine forested wetlands, but all trees planted in December 2006 were still sparsely dispersed as small saplings during the course of the study. Thus, LC was best characterized as palustrine emergent.

The LC wetland consists of two contiguous sites (i.e., LC1 and LC2) separated by a berm and connected by a drainage channel with LC1 approximately 0.4 m higher in elevation than LC2. This design causes LC1 to drain more quickly leaving it inundated for shorter periods after precipitation than LC2, while LC2 can remain under standing water (e.g., <~12 cm) for longer periods, on the order of weeks. In addition, LC2 has the ability to receive flow from an unnamed tributary of Goose Creek, through a head race attached to a cross vane structure with flow impacts regulated by a culvert and gate valve, although it is a rarity in a normal climate condition. The wetland contains at least a 0.3 m low permeability subsoil layer covered with the original topsoil from the site that was supplemented with commercially available topsoil to a depth of 0.2 m. This design creates a perched, precipitation-driven water table close to the soil surface and limits groundwater exchange in the wetland. LC contains disked areas and undisked areas with twelve 10 × 10 m plots randomly staked throughout—six in LC1 and another six in LC2. Disking to a depth of 10–20 cm using disk rollers (i.e., 30–40 cm in diameter) was conducted during the construction to create microtopographic variation on the soil surface, and half the study plots were left undisked, allowing the investigation of the effects of MT on vegetation development (Fig. 1). Each disked plot is labeled as a single alphabetical letter (i.e., A through F) and is adjacent to an undisked plot labeled as a double letter (i.e., AA through FF) to reduce variability in other environmental factors (Fig. 2).

2.2. Hydrologic regime

Hydrologic regime was evaluated through precipitation, water table depths, and plot-level soil moisture. Water table depth (WTD) was measured using five 70 cm slotted wells constructed of 3.18 cm (=1¼ in.) polyvinyl chloride (PVC) pipe, which were installed by WSSI for legal, post-construction monitoring (three wells in LC1 and two wells in LC2). Water table depth was measured weekly from March to June, then on a monthly basis throughout the rest of the year (n = 25 per year). Further, to distinguish the hydrologic signature of each plot, soil moisture content was measured monthly

Fig. 1. The picture shows (a) the disking equipment used to create microtopography and (b) how the disking was conducted, all during the construction of Loudoun County mitigation bank wetland.

throughout each growing season (i.e., March through October) for the top 10 cm of soil by taking three soil cores from each plot using a 1.8 cm-diameter soil auger. Samples were oven-dried at 105 °C for 48 h, and moisture content was determined as a percentage of mass loss between wet and dry samples (calculated as [wetmass − drymass]/drymass, expressed as a percentage). Precipitation input and drought conditions during the study seasons were evaluated using precipitation data obtained from the National Weather Service (National Weather Service, 2010) and Palmer Drought Severity Index data (Heddinghause and Sabol, 1991) obtained from the National Drought Mitigation Center (National Drought Mitigation Center, 2010).

2.3. Vegetation

Vegetation surveys were performed at the end of the growing season (i.e., September–October) for both 2008 and 2009. Percent cover was determined using a 1 m^2 grid constructed of 2.54 cm-PVC pipe and elbow joints, with holes drilled every 10 cm in each pipe. Twine was threaded through these holes, creating a 10 × 10 grid within the PVC framework. Each of these 10 cm × 10 cm squares represented 1% of the total area of the frame, enabling a means of estimating percent cover directly or through application of comparable Peet et al.'s (1998) 10-cover classes. In 2008, each individual plot was marked off in 1 m grid cells using the northwest corner of each plot as the 0–0 mark. Quadrat sampling locations were chosen by random number generation. The first two pairs of numbers generated for each quadrant (northwest, northeast, southeast, and southwest) of the plot were used for the survey, with an additional sample taken at the center of the plot for a total of nine samples per plot. In 2009, plot-level quadrants were divided into four subquadrants to reduce field labor, but remain representative, with one randomly selected for sampling for a total of four samples per plot.

For each sample, individual herbaceous and woody species were identified and percent cover per species recorded. Each plot was surveyed to identify any species that were not in the samples. Species that had less than 1% cover were listed as "present". Due to the existence of multiple canopy layers, total percent cover (TPC) can exceed 100% even when visual estimate of total cover was less than 100% (Peet et al., 1998). Plants were identified to genus and, in most cases, to species in the field. When unable to identify to the species level, physical samples were taken for further study in the lab. In addition, a representative sample of most species was

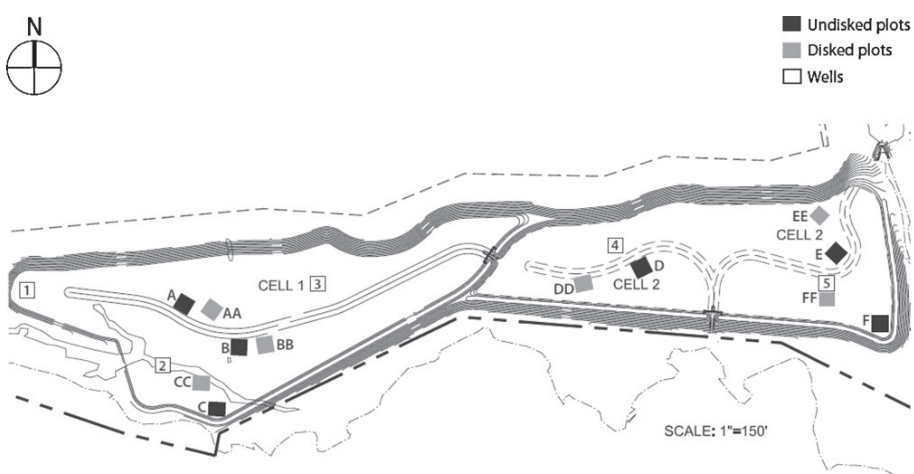

Fig. 2. Permanent plot locations for microtopography study at Loudoun County Mitigation Bank (LC). There are two cells in the wetland (i.e., LC1 and LC2) with six undisked plots labeled with single letters (A–F) and six disked plots labeled with double letters (AA–FF). The figure also shows five wells installed to study the hydrologic regime of LC. Each study plot was 10 m × 10 m.

taken for verification and documentation. Lack of cover or percent bare ground was also included in the survey. In order to assess both variation from the original wetland plant seeding plan and vegetative development, percent cover was further divided into three categories, including seeded (i.e., at creation), volunteer, and non-native species. WSSI planting lists were used for determination of seeded or volunteer status. Percent cover values were assessed at plot, microtopographic treatment (MT) (i.e., disked or undisked), and site levels for each category.

Several attributes were calculated to quantitatively assess vegetative development including richness (S), Floristic Quality Assessment Index (FQAI), Shannon–Weiner (H') index for diversity, and prevalence index (PI, see Wentworth et al., 1988). Vegetative indices were assessed at site and MT levels using individual sample values for each year. The Shannon–Weiner Biodiversity Index, $H' = -\sum p_i \log(p_i)$, was used to characterize abundance based plant diversity, and is a function of relative percent cover (p_i = species percent cover/total percent cover) for each species within a sample (Andreas et al., 2004).

Identified herbaceous species were assigned a coefficient of conservatism (C-value or c_n) based on the Virginia FQAI Advisory Committee Wetlands Plants C-value list (Davis and Harold, 2006). C-values are normally assigned by a regional panel of botanical experts and reflect tolerance to disturbance and fidelity to specific habitat integrity (Lopez and Fennessy, 2002; Davis and Harold, 2006; Nichols et al., 2006). C-values range from 0 to 10 with zero being assigned to species that are non-native to the region, low coefficients (1–3) assigned to species normally found in highly disturbed areas, average values (4–6) to species found in moderately disturbed areas, and high values (8–10) assigned to species that are sensitive, or are adapted to a narrow range of ecological conditions (Lopez and Fennessy, 2002; Davis and Harold, 2006). FQAI is a function of the C-value: $I = \Sigma c_n/(N)^{1/2}$, where c_n is the coefficient of conservatism for each species and N is the total number of native species (those with $c_n > 0$) (Wilhelm and Ladd, 1988). Mean C-values were calculated at the site and wetland level for each study season.

Plant species were assigned a regional (region 1) wetland indicator status (WIS) (e.g., obligate, facultative wet, etc.) based on the 1996 National Wetland Inventory which is instantiated in the United States Department of Agriculture Plant database (NWI, 1996; USDA, 2009). WIS values are indicative of the degree to which plant adaptations allow individual species to thrive in anaerobic conditions and range from 1 to 5 based on the regional indicators as follows: 1 = OBL (obligate), 2 = FACW (facultative wet), 3 = FAC (facultative), 4 = FACU (facultative upland), and 5 = UPL (upland) (Wentworth et al., 1988; Cronk and Fennessy, 2001). A WIS value of one (1) is given to taxa found >99% of the time in wetlands and at the opposite end of the range, five (5) is given to taxa found <1% of the time in wetlands (Wentworth, 1988; Cronk and Fennessy, 2001). The Prevalence Index (PI) is a weighted average of WIS values based on relative percent cover, calculated as $\mathrm{PI} = \Sigma(A_i W_i)/\Sigma A_i$ where A_i is the relative percent cover of species i and W_i is the WIS of species i (Wentworth et al., 1988; Cronk and Fennessy, 2001). PI values are reflective of an overall WIS for the wetland with a value less than three (WIS < 3) equivalent to an overall wetland status of 'facultative' or better (i.e., species adapted to wetlands are dominant), which is indicative of a site achieving wetland status.

2.4. Data analysis

Normality and homogeneity of variance tests were conducted on hydrologic (i.e., WTD, soil moisture content) and vegetation (TPC, FQAI, PI, H', S) variables, with transformations applied as appropriate. Percent cover variables for seeded, volunteer, and non-native species included a large number of samples with very low or zero values (severely positively skewed), so non-parametric Wilcoxon–Mann–Whitney testing was used. Mann–Whitney non-parametric tests were also conducted for mean C-value since it was comprised of a small number of plot (n = 12) level values. The effects of MT (i.e., disked vs. undisked plots), site hydrologic regime (i.e., LC1 vs. LC2), and year (i.e., 2008 vs. 2009) were analyzed for soil moisture content and vegetation variables using multi-way (3-way) analysis of variance (ANOVA) through application of the General Linear Model (GLM). ANOVA analyses were conducted using Type III sum-of-squares and an alpha level of 0.05. Bi-variate Pearson's correlation of WTD, soil moisture content, and precipitation was conducted to determine the significance of the relationship between hydrologic variables. Bi-variate correlations between vegetative attributes were also assessed to determine whether significant relationships existed between percent covers and the other variables. All statistical analyses were performed using SPSS version 18 (SPSS, 2010).

3. Results and discussion

3.1. Precipitation, water table depth (WTD) and soil moisture regime

For the first growing season of LC (i.e., 2007), precipitation data obtained from the National Weather Service for Dulles International Airport (Loudoun County, VA) during the period of March through November was 34.3 cm below normal, being defined as "extremely dry", while it was 11.4 and 17 cm above normal for the same periods in 2008 and 2009, respectively (National Weather Service, 2010).

WTD is assessed in created and/or restored wetlands as the primary hydrologic attribute for determining the success of restoration projects (Mitsch and Gosselink, 2007; Moser et al., 2007). Water budgets developed for the purpose of wetland mitigation in Virginia must target the regulatory hydrology requirement of a free water table within the upper 30 cm of the soil profile for at least 12.5% of the growing season (~30 days), as a single saturation event (Federal Interagency Committee for Wetland Delineation, 1989). The WTD readings for 2008 and 2009 met the legal criteria for jurisdictional hydrology for a mitigation wetland in Virginia (Fig. 3). The mean [±SE] WTDs for LC1 and LC2 during the 2008 growing season were significantly different ($p < 0.05$), [LC1 −6.09 ± 3.2 cm; LC2 6.37 ± 3.3 cm], while there was not a significant difference in site WTD means ($p = 0.718$) for the 2009 growing season, [LC1 −2.44 ± 3.3 cm; LC2 2.31 ± 3.8 cm] (Fig. 3). Overall, no significant difference was found in mean wetland WTDs based on study year ($p = 0.953$). During the early 2008 growing season, mean LC2 WTD levels were in the standing water range until late-June, experiencing longer standing water conditions than LC1, where WTD fluctuated below the soil surface twice in the same period prior to completely receding in late-May (Fig. 3). In contrast, both sites experienced standing water conditions through the early months of the 2009 growing season before receding below-ground in late-June. WTD variation at the well level was indicative of topographical variation within each site.

Soil moisture content varies with soil properties such as texture and organic matter content, which affect the water-holding capacity of the soils, in addition to the balance between inputs and outputs to the water budget (Bruland and Richardson, 2006; Sleutel et al., 2008). Thus the moisture content of wetland soils responds to changes in wetland hydrologic regime (Sleutel et al., 2008; Yu and Ehrenfeld, 2010). Mean plot level soil moisture content (%) ranged between 30 and 40% during the two growing seasons of our study in both sites, with no significant difference between years ($p = 0.482$).

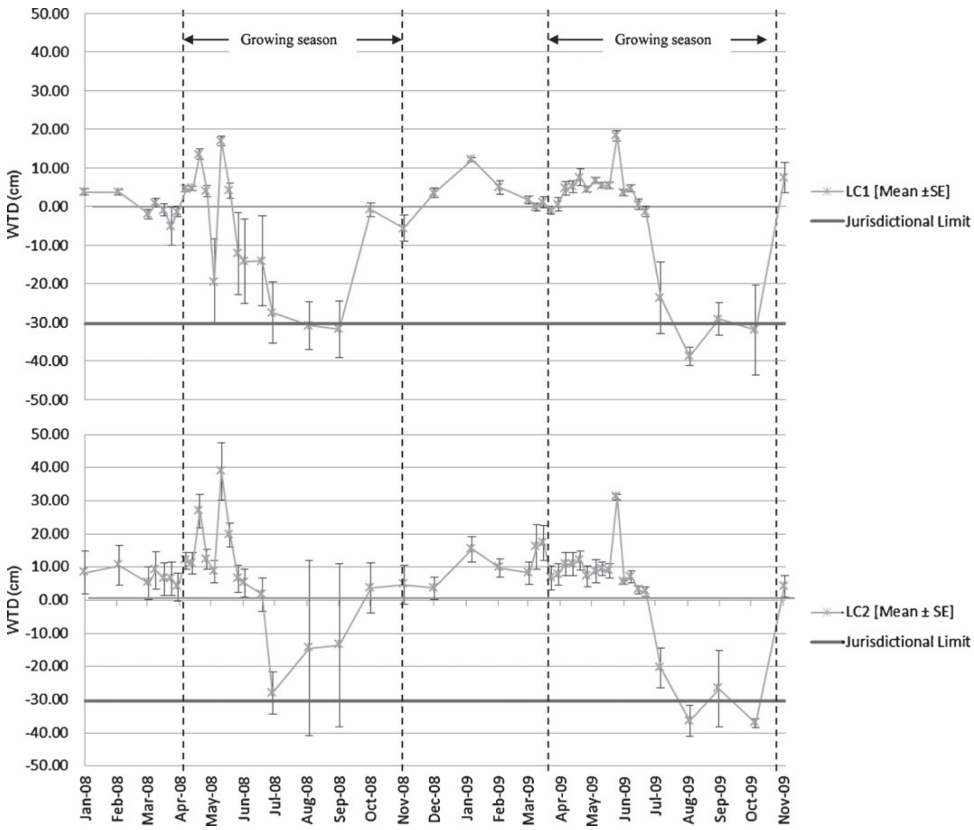

Fig. 3. Water table depths [mean ± SE] for LC1 (3 wells) and LC2 (2 wells) for two study years, 2008 and 2009.

Monthly mean soil moisture content was highly correlated with mean WTD ($r = 0.62$, $p < 0.01$) for the LC wetland over both years, suggesting that soil moisture can be a good indicator of proper hydrologic regime for vegetation development when soil is saturated even with little standing water. The hydrologic design (the difference between LC1 and LC2), though, which supported significantly higher mean WTD in LC2 in 2008, did not appear to positively correlate with higher soil moisture content as would have been expected. There was no significant difference found in soil moisture between disked and undisked plots ($p = 0.588$) across site and year. Previous studies suggested that microtopography induced by disking creates a prevalence of inundated/saturated conditions that would better support native wetland plant communities (Tweedy et al., 2001; Moser et al., 2007), which was not evident in this study (Table 2).

3.2. Vegetation

3.2.1. Species richness and percent cover

A total of 48 plant taxa were observed in the newly created wetland during the second growing season with a reduction to 22 in the third growing season (Table 1). The wetland experienced a marked reduction in seeded species between 2008 and 2009 from 13 to 5. Three of the seeded species absent in 2009 were graminoids that had FAC or higher wetland indicator status (i.e., FACU, UPL). For example, *Lolium multiflorum* (Italian ryegrass), planted intentionally for erosion control during construction, was a seeded species that was still widespread in LC1 (present in 90% of samples with mean sample percent cover of 72 ± 5[mean ± SE]) in 2008, on the heels of the prior drought conditions that encouraged FAC and up species germination. However, it disappeared completely in 2009 when the hydrologic conditions supported extended standing water durations (Fig. 3), indicating a positive transition of the site to a wetland. Another three species that were observed in 2008 at extremely low abundance in only a few samples (i.e., percent cover <5%), but absent in 2009 were perennial forbs or grasses (i.e., *Ascelpia incarnata*, *Verbena hastata*, *Gyceria striata*). By the 2009 growing season, only 35% of the originally seeded FAC or up herbaceous taxa were found in the sites (Table 1). Species richness was significantly reduced in 2009 compared to 2008 ($p < 0.005$), attributable to the loss of FAC and up herbaceous taxa and the expansion of *Juncus effusus* (in LC2), with a significant interaction between year and site ($p < 0.05$) (Tables 2 and 3).

There were 35 volunteer species, originally not included in the seed mix, present in 2008, yet only 17 were found in 2009 (Table 1). Three of those 17 species were new arrivals since 2008 (Table 1). Eight of the 17 volunteers which were FAC or up disappeared in 2009, largely in LC1. Drought conditions persisted with drought severity (e.g., "abnormally dry" to "severely dry", National Drought Mitigation Center, 2010) for northern Virginia into the early part of the 2008 growing season until May, which may have facilitated drought tolerant, volunteer plants to become established in 2008, but as expected, they were replaced by more wetland species (i.e., ≤FACW) when precipitation exceeded normal levels in 2009. Overall, the number of volunteers in both years indicates a high rate of natural seed dispersal, germination, and survival within the study sites.

Total percent cover (TPC) of vegetation is the most common attribute observed in a newly created wetland for mitigation success (Spieles, 2005), and has significant impacts on soil organic

Table 1
Plant species found in Loudoun County mitigation wetland (LC) during the growing seasons of 2008 and 2009.

Scientific name	Common name	WIS[1]	c_n[2]	2008	2009	Annual	Perennial
Seeded taxa							
Asclepias incarnata L.	Swamp milkweed	OBL	5	X	X		
Bidens aristosa Michx.	Bearded beggarticks	FACW	2	X	X	X	X
Carex lurida Wahlenb.	Sallow sedge	OBL	4	X	X	X	
Carex vulpinoidea Michx.	Fox sedge	OBL	3	X	X	X	
Elymus virginicus L.	Virginia wild rye	FACW−	4	X	X		
Glyceria striata Lam.	Fowl mannagrass	OBL	5	X	X		
Juncus effusus L.	Common rush	FACW+	3	X	X	X	
Lolium multiflorum Lam.	Italian ryegrass	FACU−	0	X	X		
Panicum virgatum L.	Switchgrass	FAC	4	X	X		
Polygonum pensylvanicum L.	Pennsylvania smartweed	FACW	2	X	X		
Scirpus atrovirens Willd.	Green bulrush	OBL	5	X	X		
Setaria italica L.	Foxtail bristlegrass	FACU	0	X	X		
Symphyotrichum novi-belgii L.	New York Aster	FACW+	6	X	X		
Verbena hastata L.	Swamp verbena	FACW+	4	X			X
				13	5	3	12
Volunteer taxa							
Alisma subcordatum Raf.	American water plantain	OBL	6	X	X	X	
Bidens cernua L.	Nodding beggarticks	OBL	4	X	X	X	
Boehmeria cylindrical L.	Smallspike false nettle	FACW	4	X	X		
Carex frankii Kunth.	Frank' Sedge	OBl	4	X	X	X	
Carex tribuloides Wahlenb.	Blunt broom sedge	FACW+	3	X	X	X	
Cyperus polystachyos Rottb.	Flat sedge	FACW	5	X	X	X	
Cyperus strigosus L.	Strawcolored flatsedge	FACW	3	X	X	X	
Digitaria sanguinalis L.	Hairy crabgrass	FACU	0	X	X		
Echinochloa crus-galli L.	Barnyard grass	FACW	0	X	X	X	
Eclipta prostata	Yerba de Tajo	FAC	2	X	X	X	
Eleocharis obtusa Willd.	Blunt spikerush	OBL	2	X	X	X	X
Erechtites hieraciifolia L.	American burnweed	FACU	2	X	X		
Juncus tenuis Willd.	Poverty rush	FAC−	2	X	X	X	
Lactuca canadensis L.	Canada lettuce	FACU−	2	X	X		
Leersia oryzoides L.	Rice cutgrass	OBL	4	X	X		
Lespedeza virginica L.	Slender lespedeza	UPL	3	X	X		
Ludwigia palustris L.	Marsh seedbox	OBL	2	X	X	X	
Lysimachia nummilaria L.[a]	Creeping jenny	OBL	0	X	X		
Parthenosissus quinquefolia L.	Virginia creeper	FACU	4	X	X		
Phleum pretense[a]	Timothy grass	FACU	0	X	X		
Polygonum cespitosum Bl.[b]	Oriental ladysthumb	FACU−0	X	X	X		
Polygonum hydropiper L.	Marshpepper knotweed	OBL	4	X	X	X	
Polygonum hydropiperoides Michx.	Mild water pepper	OBL	4	X	X		
Polygonum lapathifolium L.	Curlytop knotweed	FACW+	4	X	X		
Polygonum persicaria L.	Spotted ladysthumb	FACW	0	X	X	X	X
Polygonum punctatum Elliot	Dotted smartweed	OBL	4	X	X	X	
Polygonum sagittatum L.	Arrowleaf tearthumb	OBL	5	X	X	X	
Rorippa palustris L.	Bog yellowcress	OBL	3	X	X	X	
Rotala ramosior L.	Lowland rotala	OBL	4	X	X		
Rumex crispus L.[a]	Curly dock	FACU	0	X	X		
Setaria faberi Herrm.[a]	Japanese bristlegrass	UPL	0	X	X		
Scirpus cyperinus L.	Woolgrass	FACW	3	X	X	X	
Solanum carolinense L.	Carolina horsenettle	UPL	2	X	X		
Solidago altisima Mill.	Wrinkleaf goldenrod	FAC	3	X	X		
Trifolium pretense L.	Red clover	FACU−	0	X	X		
Trifolium repens L.	White clover	FACU−	0	X	X		
Typha latifolia L.	Broadleaf cattail	OBL	2	X	X		
Verbesina alternifolia L.	Wingstem	FAC	3	X			X
				35	17	19	25

Invasiveness was determined by using the September 2009 Invasive Alien Plant Species of Virginia list prepared by the Virginia Department of Conservation and Recreation and the Virginia Native Plant Society.

[1] Wetland indicator status for the Northeast Region (Region 1) (USDA Plants): OBL (obligate wetland species); FACW (facultative wetland species); FAC (facultative species); FACU (facultative upland); UPL (obligate upland); NI, no wetland indicator status assigned for Region 1.

[2] Coefficient of conservation scores were obtained from Virginia wetlands plants C-value list prepared by FQAI Advisory Committee (2006) of the Department of Conservation and Recreation, Virginia.

[a] *Note*: Moderately invasive species.

[b] *Note*: Highly invasive species.

matter accumulation that is fundamental for biogeochemical processes to be developed (Giese et al., 1999; Bruland and Richardson, 2006). Percent cover variables included those for TPC, seeded, volunteer, and non-native (i.e., $c_n = 0$) species cover (Table 2). TPC values ranged from 92 to 254% in 2008 and from 80 to 136% in 2009 (Table 2). TPC values were significantly higher in disked plots ($p < 0.05$), in LC1 ($p < 0.005$), and in 2008 ($p < 0.005$), with significant interaction effects among MT, site and year (Table 3). LC1 values for TPC in 2008 were dramatically different in all respects and strongly influenced the results. Disked plots experienced significantly greater TPC values than undisked plots in 2008 for both LC1 and LC2, clearly showing the positive effects of disking-induced

Table 2
Vegetation attributes and soil moisture content both by site (LC1 vs. LC2) and by disking-induced microtopography (disked vs. undisked) in 2008 and 2009 in the LC wetland. The attributes include %cover (total-TPC, seeded, volunteer, non-native); richness (*S*), Shannon biodiversity index (*H*′), prevalence index (P.I.), wetland indicator status (WIS), floristic Quality Assessment Index (FQAI).

	LC1				LC2				LC overall			
	Disked		Undisked		Disked		Undisked		LC1		LC2	
	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009
%TPC (total)[a]	203 ± 9	113 ± 4	164 ± 8	120 ± 9	111 ± 7	107 ± 6	90 ± 7	101 ± 6	183 ± 7	116 ± 5	100 ± 5	104 ± 4
%Seeded	60 ± 4	11 ± 2	52 ± 5	20 ± 12	53 ± 10	89 ± 23	53 ± 3	77 ± 20	62 ± 8	16 ± 6	53 ± 5	83 ± 14
%Volunteers	41 ± 4	90 ± 2	49 ± 5	82 ± 14	55 ± 10	19 ± 19	56 ± 7	27 ± 12	45 ± 7	86 ± 7	55 ± 6	23 ± 10
%Non-native	60 ± 3	6 ± 2	60 ± 3	18 ± 10	18 ± 1	3 ± 3	15 ± 8	5 ± 5	60 ± 3	12 ± 5	17 ± 4	4 ± 3
S (richness)	7.6 ± 0.3	5.0 ± 0.1	5.7 ± 0.4	4.3 ± 0.4	8.0 ± 1.4	3.6 ± 1.6	7.0 ± 0.6	3.5 ± 0.7	6.7 ± 0.6	4.6 ± 0.2	7.5 ± 0.7	3.5 ± 0.8
H′[b]	0.6 ± 0.0	0.5 ± 0.0	0.5 ± 0.0	0.5 ± 0.1	0.6 ± 0.1	0.3 ± 0.2	0.6 ± 0.0	0.3 ± 0.1	0.5 ± 0.1	0.5 ± 0.0	0.6 ± 0.1	0.3 ± 0.1
P.I.[c]	2.9 ± 0.1	1.4 ± 0.1	2.7 ± 0.1	1.6 ± 0.3	1.8 ± 0.1	1.6 ± 0.1	1.8 ± 0.0	1.5 ± 0.0	2.8 ± 0.1	1.5 ± 0.1	1.8 ± 0.0	1.5 ± 0.0
WIS	FAC	OBL	FAC	FACW+	FACW+	FACW+	FACW+	OBL	FAC	OBL	FACW	+ OBL
FQAI[d]	4.8 ± 0.2	7.1 ± 0.5	5.1 ± 0.4	6.4 ± 0.5	6.6 ± 0.4	5.3 ± 0.6	6.8 ± 0.3	5.4 ± 0.5	4.9 ± 0.2	6.8 ± 0.3	6.7 ± 0.2	5.3 ± 0.4
%Soil moisture	36 ± 1.7	34 ± 1.8	35 ± 1.6	32 ± 1.6	32 ± 1.8	32 ± 2.2	32 ± 2.0	31 ± 2.2	36 ± 1.3	33 ± 1.9	32 ± 1.3	31 ± 1.5

[a] Due to multiple herbaceous canopy layers, the total cover estimates could exceed 100%, even when visual estimate was <100%.
[b] $H' = -\Sigma p_i \log p_i$ where p_i is the relative percent cover fraction for each species in the sample (species percent cover/total percent cover).
[c] $PI = (\Sigma W_i A_i)/(\Sigma A_i)$ where W_i is the species wetland indicator status for Region 1 and A_i is the fraction of the percent cover.
[d] $FQAI = (\Sigma C_n)/N_{native}$ where C_n is the coefficient of conservatism assigned to species by the Virginia Department of Environmental Quality (VDEQ), and N is the number of native species bases on a C-value ≥ 1. Values calculated using sample-level means.

MT on vegetation cover. The positive effects were more pronounced in LC1 than in LC2, whereas, in 2009, TPC was still significantly higher in LC1 than in LC2, but with no MT effect for either site (Tables 2 and 3), suggesting that the residual benefits of initially induced MT can vary over the change of climate condition (i.e., drought- to above-normal precipitation condition). Seeded covers drastically changed between 2008 and 2009, with different directionality for LC1 and LC2, decreasing precipitously in LC1, but increasing dramatically in LC2. *L. multiflorum* had a mean TPC in LC1 of 72% in 2008, driving the overall seeded percentage, but completely disappeared in 2009, contributing to a marked reduction in both seeded cover and TPC (Table 2). Other species that experienced significant changes in percent cover in LC1 from 2008 to 2009 were *Bidens aristosa* (decreased), *Carex frankii* (increased), and *Bidens cernua* (increased), transitioning LC1 toward more of an obligate status. All the seeded species observed in 2009 were wetland species (i.e., FACW and obligate). In 2009, we could see a major expansion of the seeded *J. effusus* in LC2 (i.e., increased from 24 to 62%), which contributed greatly to an increase in the seeded cover in the site (Table 2). *J. effusus* is considered invasive in some regions (USDA, 2009) though not in Virginia, and thrives in standing water conditions up to a meter with high variability (Magee and Kentula, 2005). Attention should be paid to the significant expansion of *J. effusus* in LC2 since its spread is indicative of a potentially unhealthy progression toward a monotypic system in LC2, attributable to extended standing water conditions in 2009 (Fig. 3). Overall, seeded percent cover was significantly different by site ($p < 0.005$) due to reasons explained above, but not significantly different between years ($p = 0.238$) because the trends reversed direction in the two sites (Tables 2 and 3). Seeded percent cover was not different between disked and undisked plots in both sites ($p = 0.245$), but was significantly higher in LC2 in 2009 ($p < 0.005$) (Tables 2 and 3).

Percent volunteer cover patterns displayed a significant difference between the two sites ($p < 0.05$). It increased in LC1 and decreased in LC2, both significantly between 2008 and 2009, showing the opposite trend to the pattern of seeded cover percentage (Table 2). There were almost equal amount of seeded and volunteer species colonized in each site in 2008 (Table 2), but the ratio between the two categories showed a significant difference in 2009 when volunteer species drastically increased in LC1 and decreased in LC2. It may be that the drier condition in LC1 in 2008 facilitated natural colonization of volunteer species that are either less flood-tolerant or non-wetland species (e.g., FAC and upland species), but, in 2009, up to 80% of volunteers in LC1 were wetland species (e.g., FACW and obligate), contributing to the drastic increase in volunteer wetland species cover in LC1. The opposite was true for LC2 where volunteer species cover dropped, and seeded species significantly increased as *J. effusus* expanded. The spread of *J. effusus* may have limited the establishment of volunteer native wetland species in LC2. Because the volunteer cover pattern reversed between sites, it was not affected by year ($p = 0.506$) nor by MT ($p = 0.170$) (Table 3).

Table 3
General linear model/MANOVA (SPSS v18.0) results of vegetative attributes as affected by design elements such as microtopography (MT: disked or undisked), hydrologic elevation difference (i.e., site), and by year (2008 or 2009) including interaction terms. Mann–Whitney non-parametric tests were used for seeded, volunteer, and non-native cover.

Adjusted R^2	*S* (richness) 0.306		PI 0.624		FQAI 0.172		*H*′ 0.209		Total cover 0.553		Seeded cover NA		Volunteer cover NA		Non-native cover NA	
	F	Sig	*F*	Sig	*F*	Sig	*F*	Sig	*F*	Sig	*U*	Sig	*U*	Sig	*U*	Sig
MT	4.7	*	0.02	NS	0.8	NS	1.7	NS	6.0	*	3370	NS	2655	NS	2959	NS
Year	58.1	**	118.2	**	0.2	NS	25.8	**	26.7	**	2285	NS	2765	NS	1210	**
Site	0.4	NS	37.7	**	0.5	NS	0.8	NS	60.6	**	3884	**	2409	*	884	**
MT × year	1.4	NS	1.7	NS	2.7	NS	1.2	NS	5.9	*	–	–	–	–	–	–
MT × site	1.4	NS	0.05	NS	1.1	NS	1.3	NS	0.03	NS	–	–	–	–	–	–
Year × site	5.4	*	49.5	**	9.3	**	15.0	**	33.2	**	–	–	–	–	–	–
MT × year × site	0.2	NS	2.2	NS	0.01	NS	0.2	NS	1.6	NS	–	–	–	–	–	–

Note: NS—not significant.
* $P < 0.05$.
** $P < 0.005$.

All volunteer species with WIS of FACU or up in 2008 disappeared in 2009 (Tables 1 and 2). The increase in volunteer cover in LC1 from 2008 to 2009 was attributed to an increased colonization of wetland species (i.e., OBL), including *B. cernua* (0–20%), *C. frankii* (12–37%), *Polygonum hydropiperoides* (0–11%) over non-wetland species (i.e., FAC and up).

Non-native percent cover was significantly greater in LC1 than in LC2, with a significant decrease from 2008 to 2009 in both sites ($p < 0.005$) (Tables 2 and 3). The C-values for the 52 total plant species identified in our study over 2008 and 2009 ranged from 0 to 6 (Table 1), with 12 species having C-values of zero (0), being non-native to Virginia (Davis and Harold, 2006). Of these 12, two were seeded cover graminoids, and all but two were absent in sampled plots by 2009, including the seeded *L. multiflorum*. The dominant non-native species in 2009 was *Echinochloa crus-galli* and it was present in 21 of the 48 samples, 17 of them in LC1, at a mean 20% cover in those plots. *E. crus-galli* is not considered invasive in Virginia, and has a "high" wildlife habitat value as a terrestrial and water fowl food source (VDCR, 2009; USDA, 2009). The relatively low mean C-values found in this study suggest that LC is dominated by early colonizers or generalist species that are capable of growth and survival under wide variety of ecological and environmental conditions (Cronk and Fennessy, 2001). It seems that non-native species dramatically decreased as the wetland stabilized through the third growing season in 2009. It is not unusual to observe higher non-native species dominance at early stages of created wetlands due to the disturbance (i.e., construction) that opened up a large space for opportunistic species to colonize (Moser et al., 2007).

3.2.2. Invasive species

There were five species in 2008 samples that were classified as invasive to Virginia, of which only one was still found in 2009 (VDCR, 2009; Table 1). The invasive species were present in very few of the 12 study plots at low mean percent cover (<1–3%) were as follows: *Lysimachia nummularia* (in 2008), *Phleum pretense* (in 2008), *Polygonum cespitosum* (in both 2008 and 2009), *Rumex crispus* (in 2008), and *Setaria faberi* (in 2008). It has been shown that invasive species can change or control the structure and function of wetlands (Galatowitsch et al., 1999; Kercher and Zedler, 2004), thus careful monitoring of invasive species is important, especially during early plant community development in created wetlands. The footprint of invasive species appears to be trending downward as LC matures.

3.2.3. Shannon–Weiner biodiversity index (H')

Biodiversity values (H') declined by 30% overall in 2009 compared to 2008 (0.6 ± 0.04[2008]/0.4 ± 0.05[2009]: $p < 0.005$), which was largely attributable to the spread of *J. effusus* in LC2, indicating the interaction between site and year. In addition, this may have been due to the overall decline of species richness in 2009 compared to 2008, but remains to be further observed since the wetland was still in the early stages of maturation. There was no MT effects on H' ($p = 0.190$) in either year (Tables 2 and 3). Moser et al. (2007) reported a positive effect of disking on species richness and H', but the positive effects of initial disking during the construction on vegetation might have faded over time, as observed in the changes from 2008 to 2009 in this study.

3.2.4. Wetland prevalence index (PI)

Mean PI values for most study plots in both years indicate that wetland plants ($\leq$FACW) were fairly successful in their early stage of development. Mean PI values were significantly lower in 2009 compared to 2008, especially in LC1 as the cover of obligate species (i.e., *B. cernua* and *C. frankii*) increased and the FACU-species (i.e., *L. multiflorum*) decreased, showing a strong interaction between site and year ($p < 0.005$; Tables 2 and 3). In 2009, all the seeded species observed were FACW or lower. The relatively lower PI values in LC2 in 2008 were due to the extended duration standing water conditions in the site, which may have better supported wetland vegetation (i.e., facultative wet and obligate species) than the drier LC1 did. No disking-induced MT effect on PI was found ($p = 0.186$) (Table 3) in either year. The PI values consistently dropped in each site from 2008 to 2009 (e.g., FAC down to OBL), showing a trend toward maturing into a wetland over time. The result reveals that site hydrologic conditions were critical in supporting the development of wetland vegetation in a newly created wetland.

3.2.5. Floristic Quality Assessment Index (FQAI)

FQAI values were not significantly affected by any of the factors (i.e., MT, site, and year). There was a significant interaction between site and year (Table 3) with FQAI significantly increased in LC1, and decreased in LC2 from 2008 to 2009. FQAI values less than 5 usually indicate that the area is extremely weedy or in an early successional stage (Swink and Wilhelm, 1979). Most (19 out of 24) of our plots for both years had FQAI values higher than 5, and just one plot was below 4 due to the dominance of *J. effusus* that has a relatively low C-value of 3 (i.e., plot DD, see Fig. 1). FQAI values approaching, but less than 20 indicate a habitat being between 'disturbed' and starting to show some 'native character' (Swink and Wilhelm, 1979; Lopez and Fennessy, 2002; Miller and Wardrop, 2006). Therefore, it seems that LC has not reached that level yet. Given that LC is still in the early stages of its development, floristic quality may continue to change in response to climate and as the wetland stabilizes, being more relevant as an indicator of fully developed vegetation community later on. Floristic Quality Assessment (FQA) has been recognized as an effective technique to accurately assess the overall health of wetland sites over time (Wilhelm and Ladd, 1988; Lopez and Fennessy, 2002). Lopez and Fennessy (2002) interpreted the index as a measure of environmental factors that maintain and control vegetation communities, and recommended as being useful for tracking wetland restoration projects. They also found that FQAI positively correlated with soil organic carbon, phosphorous, and calcium (Lopez and Fennessy, 2002). We plan to monitor LC for an extended period of time (e.g., 10 years) to continue assessing the change of FQAI, relating it to other functional development being concurrently studied.

3.3. Hydrologic design elements and early development of vegetation community

In 2008 when water table depth (WTD) was significantly higher in LC2 (Fig. 2) than in LC1, while still under the lasting influence of the drought of the previous year, disking-induced MT resulted in significant ($p < 0.05$) differences for four of eight vegetative variables including S, H', TPC, and non-native cover, while elevation difference affected four variables significantly including PI, FQAI, H', and TPC. However, in 2009 when there was no difference in WTD between LC1 and LC2 (Fig. 3), the effects of disking-induced MT on vegetative variables became insignificant, while site hydrology still drove the differences for six of eight variables including S, FQAI, H', TPC, volunteer, and seeded cover. This may suggest that that artificially induced microtopographic heterogeneity (i.e., micro-hydrologic variation) and its influence on vegetative diversity may be applicable only when dry conditions predominate (i.e., when precipitation is below average levels), but not a factor otherwise since hydrologic variation (e.g., WTD) by MT would be outdone by regional hydrology driven by precipitation and standing water condition. The impact of the transition out of extended drought conditions on six variables (i.e., S, PI, H', TPC, seeded and volunteer cover) was clear, in addition to strong interactions between year

Table 4
Pearson bi-variate correlation coefficients for vegetative indices and percent cover variables for 2008–2009.

	Total cover	Seeded cover	Volunteer cover	Non-native cover
S	**0.207**	−**0.210**	**0.217**	**0.243**
H′	0.134	−**0.371**	**0.325**	0.111
FQAI	−**0.204**	−**0.229**	**0.233**	−**0.298**
PI	**0.565**	0.139	−0.153	**0.704**

Values in boldface indicate correlation is significant at the 0.01 level and those underlined are significant at the 0.05 level (1-tailed).

and site for all vegetative variables (Table 3). The 3-factor model explained more than 50% of the variation for PI (adj $R^2 = 0.624$) and TPC (adj $R^2 = 0.553$), both most importantly being used to determine the success of mitigation (Table 3).

Table 4 shows correlation coefficients between percent cover variables and all the other vegetative attributes, showing significant correlations in most cases. All qualitative attributes of vegetation community in development turned out significantly correlated with all percent cover variables (Table 4). Both *S* and *H′* correlate negatively with seeded, and positively with volunteer cover, respectively (Table 4). Considering the negative impact of seeded species, mainly in the second year of this study, one might question the necessity of seeding or planting generalist species. It was a common practice and entirely appropriate to plant the cover grasses though since they provided a good erosion control in the first year (personal communication with WSSI), and disappeared as intended in the second year. We would rather caution the planting of species like *J. effusus* that can spread and dominate fairly quickly in slightly extended standing water conditions, as observed in this study.

FQAI correlated very significantly ($p < 0.01$) with all percent cover variables, and with negative directionality to seeded and non-native cover indicating that FQAI decreased as cover increased, which illustrated that seeded and non-native species tended to be of lower quality during the study years. *S* also correlated significantly with all percent cover variables, and negatively with seeded cover indicating a tendency for seeded species to have invasive tendencies and thus reduce the number of species. PI correlated significantly and positively with total, seeded, and non-native cover, and negatively with volunteer cover indicating that most volunteer species were FAC or lower. More studies are needed to further investigate the net effects of seeding/planting on overall structural development of wetland vegetation. The positive relationships between volunteer cover and *S*, *H′*, and FQAI indicated, though, that naturally colonized species assisted LC to become a wetland habitat over varying hydrologic conditions associated with both design elements (e.g., MT and site) and climate conditions (e.g., year) (Table 3). Although the effects of disking-induced MT and site design on quality development of vegetation community (*H′* and FQAI, Davis and Harold, 2006) were not consistent across both years, they were all related to the percent cover attributes most influenced by the hydrologic design elements incorporated. The differences in *H′* and FQAI may be more relevant later on when this newly created wetland has stabilized and matured further.

The effects of MT may not be conclusively definitive, since it has been observed in other studies that initially induced MT also co-evolved with autogenic sources of MT that develop as the vegetation communities mature in created wetlands, especially by tussock-forming plants and/or sedimentation (Werner and Zedler, 2002; Wolf et al., in press). Wolf et al. (in press) showed that autogenic sources of MT (e.g., tussock-forming vegetation) in concert with variable hydrology and sedimentation maintained and even increased MT over time in the same types of created wetlands as LC, suggesting a dynamic equilibrium of MT forming and -eroding processes at play in created mitigation wetlands. MT may be an important element of self-design for created and restored wetlands (Mitsch and Wilson, 1996). The positive effects of MT induced by disking during the construction seem still valid and recommendable for the early establishment of vegetation community.

4. Conclusion

This study examined the effects of two hydrologic design elements (i.e., MT and site elevation difference) incorporated during the construction on the early vegetation development in a newly created mitigation wetland over two growing seasons. The lack of MT in most created wetlands due to use of heavy machinery for grading during the construction process often leads to a vegetatively monotypic system with few species. This study found that disking-induced MT positively influenced the early vegetation community in the higher elevation site (i.e. LC1) during the second growing season when precipitation was below average levels, increasing total percent cover and species richness. The other hydrologic design element (i.e., the elevation difference between LC1 and LC2) positively influenced *S*, *H′*, and FQAI, enhancing natural colonization of wetland plants. It overwhelmed the positive effects of disking when precipitation was at or above average range.

The positive influence of induced MT on vegetation seemed likely to be influential on short spatial and temporal scales, either fading away over time or becoming insignificant under the effects of macro-hydrologic elements/conditions (e.g., basin elevation difference and precipitation pattern). Disking during the construction of a mitigation wetland is not costly (i.e., $100–200/acre, personal communication with WSSI), but is currently not mandatory. The positive effects shown by disking-induced MT still encourage us to recommend disking-induced MT to enhance early vegetation development under varying climate conditions. Wetland engineers and managers are advised to diversify hydrologic designs that would assist in early establishment and development of diverse vegetation community when creating mitigation wetlands.

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References

Andreas, B.K., Mack, J.J., McCormac, J.S., 2004. Floristic Quality Assessment Index (FQAI) for Vascular Plants and Mosses for the State of Ohio. Ohio Environmental Protection Agency, Division of Surface Water, Wetland Ecology Group, Columbus, OH, 219 pp.
Ahn, C., White, D.C., Sparks, R.E., 2004. Moist-soil plants as ecohydrologic indicators for recovering the flood pulse in the Illinois River. Restor. Ecol. 12, 207–213.
Ahn, C., Peralta, R.M., 2009. Soil bacterial community structure and physicochemical properties in mitigation wetlands created in the Piedmont region of Virginia (USA). Ecol. Eng. 35, 1036–1042.
Atkinson, R.B., Perry, J.E., Cairn Jr., J., 2005. Vegetation communities of 20-year-old, created depressional wetlands. Wetl. Ecol. Manage. 13, 469–478.
Baer, S.G., Collins, S.L., Blair, J.M., Knapp, A.K., Fiedler, A.K., 2005. Soil heterogeneity effects on tallgrass prairie community heterogeneity: an application of ecological theory to restoration ecology. Restor. Ecol. 13, 413–424.
Breaux, A., Serefiddin, F., 1999. Validity of performance criteria and a tentative model for regulatory use in compensatory wetland mitigation permitting. Environ. Manage. 24, 327–336.
Brown, S.C., Veneman, P.L.M., 2001. Effectiveness of compensatory wetland mitigation in Massachusetts, USA. Wetlands 21 (4), 508–518.
Bruland, G.L., Richardson, C.J., 2005. Hydrologic, edaphic, and vegetative responses to microtopographic reestablishment in a restored wetland. Restor. Ecol. 13, 515–523.

Bruland, G., Richardson, C., 2006. Comparison of soil organic matter in created, restored and paired natural wetlands in North Carolina. Wetl. Ecol. Manage. 14, 245–251.

Campbell, D.A., Cole, C.A., Brooks, R.P., 2002. A comparison of created and natural wetlands in Pennsylvania, USA. Wetl. Ecol. Manage. 10, 41–49.

Cronk, J.K., Fennessy, M.S., 2001. Wetland Plants: Biology and Ecology. CRC Press, Boca Raton, FL.

Davis, D.L., Harold, C.M., 2006. Determining Coefficient of Conservatism Values (C-values) for Vascular Plants Frequently Encountered in Tidal and Nontidal Wetlands in Virginia. Virginia Department of Environmental Quality (VDEQ) Office of Wetlands and Water Quality.

Dwire, K.A., Kauffman, J.B., Baham, J.E., 2006. Plant species distribution in relation to water-table depth and soil redox potential in montane riparian meadows. Wetlands 26, 131–146.

Federal Interagency Committee for Wetland Delineation, 1989. Federal Manual for Identifying, and Delineating Jurisdictional Wetlands. US Army Corps of Engineers, US Environmental. Protection Agency, US Fish and Wildlife Service, USDA Soil Conservation Service, Washington, DC, USA.

Galatowitsch, S.M., van der Valk, A.G., 1996. Vegetation and environmental conditions in recently restored wetlands in the prairie pothole region of the USA. Vegetatio 126, 89–99.

Galatowitsch, S.M., Anderson, N.O., Ascher, P.D., 1999. Invasiveness in wetland plants in temperate North America. Wetlands 19, 733–755.

Giese, L.A., Aust, W.M., Trettin, C., Kolka, R., 1999. Spatial and temporal patterns of carbon storage and species richness in three South Carolina coastal plain riparian forests. Ecol. Eng. 15, 157–170.

Hamrick, J.L., Lee, J.M., 1987. Effect of soil surface topography and litter cover on the germination, survival, and growth of Musk Thistle (*Carduus nutans*). Am. J. Bot. 74, 451–457.

Harper, J.L., Williams, J.T., Sagar, G.R., 1965. The heterogeneity of soil surfaces and its role in determining the establishment of plants from seed. J. Ecol. 53, 273–286.

Heddinghause, T.R., Sabol, P., 1991. A review of the palmer drought severity index and where do we go from here? In: Proceeding of the 7th Conference on Appl. Climatol. ,. American Meteorological Society, Boston, MA.

Hoeltje, S.M., Cole, C.A., 2007. Losing function through wetland mitigation in central Pennsylvania, USA. Environ. Manage. 39 (3), 385–402.

Jerling, L., 1981. Effects of microtopography on the summer survival of *Plantago maritima* seedlings. Holarct. Ecol. 4, 120–126.

Kercher, S.M., Zedler, J.B., 2004. Multiple disturbances accelerate invasion of reed canary grass (*Phalaris arundinacea* L.) in a mesocosm study. Oecologia 138, 455–464.

Lopez, R.D., Fennessy, S.M., 2002. Testing the floristic quality assessment index as an indicator of wetland condition. Ecol. Appl. 12, 487–497.

Magee, T.K., Kentula, M.E., 2005. Response of wetland plant species to hydrologic conditions. Wetl. Ecol. Manage. 13, 163–181.

Miller, S.J., Wardrop, D.H., 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecol. Indic. 6, 313–326.

Mitsch, W.J., Gosselink, J.G., 2007. Wetlands. Wiley, New Jersey, pp. 107–110.

Mitsch, W.J., Wilson, R.F., 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. Ecol. Appl. 6, 77–83.

Moser, K.F., Ahn, C., Noe, G.B., 2007. Characterization of microtopography and its influence on vegetation patterns in created wetlands. Wetlands 27, 1081–1097.

Moser, K.F., Ahn, C., Noe, G.B., 2009. The influence of microtopography on soil nutrients in created mitigation wetlands. Restor. Ecol. 17 (5), 641–651.

National Drought Mitigation Center, 2010. U.S Drought Monitor., http://drought.unl.edu/dm/archive.html (accessed February 2010).

National Research Council, 2001. Compensating for Wetland Losses Under the Clean. Water Act. Natl. Acad. Press, Washington, DC, USA.

National Weather Service, 2010. Dulles International Airport Monthly Total Precipitation Records., http://www.erh.noaa.gov/lwx/climate/iad/iadprecip.txt (accessed February 2010).

National Wetlands Inventory, 1996. National List of Vascular Plant Species that Occur in Wetlands: 1996 National Summary. U.S. Fish and Wildlife Service, www.fws.gov/wetlands/ (accessed February 2010).

Nichols, J.D., Perry, J.E., DeBerry, D.A., 2006. Using a Floristic Quality Assessment technique to evaluate plant community integrity of forested wetlands in Southeastern Virginia. Nat. Areas J. 26, 360–369.

Peach, M., Zedler, J.B., 2006. How tussocks structure sedge meadow vegetation. Wetlands 26, 322–335.

Peet, R.K., Wentworth, T.R., White, P.S., 1998. A flexible, multipurpose method for recording vegetation composition and structure. Castanea 63 (3), 262–274.

Schupp, E.W., 1995. Seed-seedling conflicts, habitat choice, and patterns of plant recruitment. Am. J. Bot. 82, 399–409.

Sleutel, S., Moeskops, B., Huybrechts, W., Vandenbossche, A., Salomez, J., De Bolle, S., Buchan, D., De Nev, S., 2008. Modeling soil moisture effects on net nitrogen mineralization in loamy wetland soils. Wetlands 28 (3), 724–734.

Smith, M., Capelle, J., 1992. Effects of soil surface microtopography and litter cover on germination, growth and biomass production of Chicory (*Cichorium intybus* L.). Am. Midl. Nat. 128, 246–253.

Spieles, D.J., 2005. Vegetation development in created, restored, and enhanced mitigation wetland banks of the United States. Wetlands 25, 51–63.

SPSS, 2010. IBM SPSS/PASW Statistics Version 18. SPSS Inc., Chicago, IL.

Stolt, M.H., Genthner, M.H., Daniels, W.L., Groover, V.A., Nagle, S., Haering, K.C., 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. Wetlands 20, 671–683.

Swink, F., Wilhelm, G., 1979. Plants of the Chicago Region: A Checklist of the Vascular Flora of the Chicago Region, with Keys, Notes on Local Distribution, Ecology, and Taxonomy, and a System for Evaluation of Plant Communities. Morton Arboretum, Lisle, IL, USA.

Titus, J.H., 1990. Microtopography and woody plant regeneration in a hardwood floodplain swamp in Florida. Bull. Torrey Bot. Club. 117, 429–437.

Touchette, B.W., Iannacone, L.R., Turner, G., Frank, A., 2010. Ecophysiological responses of five emergent-wetland plants to diminished water supply: an experimental microcosm study. Aquat. Ecol. 44, 101–112.

Tweedy, K.L., Scherrer, E., Evans, R.O., Shear, T.H., 2001. Influence of microtopography on restored hydrology and other wetland functions. In: Am. Soc. Agric. Eng. Annual Int. Meet. ,. ASAE, St. Joseph, MI.

USDA, 2009. Plants Database. United States Department of Agriculture Natural Resources Conservation Service, http://plants.usda.gov/ (accessed February 2010).

van der Valk, A.G., Davis, C.B., 1978. The role of seed banks in the vegetation dynamics of prairie glacial marshes. Ecology 59, 322–335.

Virginia Department of Conservation and Recreation, 2009. Invasive Alien Plant Species of Virginia., www.dcr.virginia.gov (accessed May 2009).

Vivian-Smith, G., 1997. Microtopographic heterogeneity and floristic diversity in experimental wetland communities. J. Ecol. 85, 71–82.

Wentworth, T.R., Johnson, G.P., Kologiski, R.L., 1988. Designation of wetlands by weighted averages of vegetation data—a preliminary evaluation. Water Resour. Bull. 24, 389–396.

Werner, K.J., Zedler, J.B., 2002. How sedge meadow soils, microtopography, and vegetation respond to sedimentation. Wetlands 22, 451–466.

Whittecar, G.R., Daniels, W.L., 1999. Use of hydrogeomorphic concepts to design created wetlands in southeastern Virginia. Geomorphology 31, 355–371.

Wilhelm, G., Ladd, D., 1988. Natural area assessment in the Chicago region. In: Trans. 53rd North Am. Wildl. Nat. Resour. Conf. , Chicago, IL, pp. 361–365.

Wolf, K.L., Ahn, C., Noe, G.B., 2011. Microtopography enhances nitrogen cycling and removal in created wetlands. Ecol. Eng. 37, 1398–1406.

Yu, S., Ehrenfeld, J.G., 2010. Relationships among plants, soils and microbial communities along a hydrological gradient in the New Jersey Pinelands, USA. Ann. Bot. 105, 185–196.

Zedler, J.B., Callaway, J.C., 1999. Tracking wetland restoration: do mitigation sites follow desired trajectories? Restor. Ecol. 7, 69–73.