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Running Title: Vegetation Diversity in Southeast Arkansas

Abstract

The loss of forested wetlands in the Lower Mississippi River Alluvial Valley in Arkansas has altered regional vegetation communities. Multiple restoration projects have been established in this region to restore wetlands and the services they provide. In order to return these functions to the environment, microtopographic features were constructed in 2001 at Bob White Memorial Wetlands Research and Teaching Station (Bob White). Vegetation diversity was examined at Cut-Off Creek Wildlife Management Area (Cutoff), a naturally forested wetland, and Bob White, an area formally converted to cropland that is now undergoing forest wetland restoration. Vegetation diversity is one way to determine if restoration efforts are effectively restoring ecosystem structure and functions to natural wetland conditions. Vegetation diversity and composition were examined across three topographical features: hummocks/ridges, swales, and flats. Vegetation diversity was examined in the spring, summer, and fall. Indices were used for determining composition similarities between Bob White and Cutoff. Bob White had a species richness of 33 and Cutoff's species richness was 47. Beta diversity between the two sites was 76 species, this value is high and suggests there is low similarity between the two sites. Sorensen-Dice Similarity Index value was calculated as 0.05, where on a scale of zero to one a low value indicates low similarity in composition. The low similarity between the two sites suggests that vegetation composition at Bob White has not been fully restored to conditions comparable to a natural setting. An explanation for this is the presence of *Baccharis halimifolia* (Eastern baccharis). *B. halimifolia* inhibits other species from colonizing. Another factor for the difference between the vegetation at Bob White and Cutoff is that Cutoff is an older forest. Hydrophyte communities in a forested wetland take 50 years after restoration begins for them to resemble a natural forested wetland. The results from this study provide mixed evidence that restoration at Bob White is succeeding; there is a high percentage of wetlands species, while vegetation lacks similarity. This study improves our understanding of the influence that anthropogenic changes have on wetland functions as agricultural lands are restored to their previous land cover. Ecosystem functions should continue to be monitored to determine time frames as these functions are restored to Bob White.

Introduction

Wetlands provide important environmental services to human and biological communities, such as flood attenuation, wetland-dependent wildlife habitat, water quality enhancement, sediment filtration, and pollution control through denitrification, wood products, and food production (Walbridge 1993, Gilliam 1994, Kleiss 1996, Mitsch and Gosselink 2000, Zedler 2003). In 1972 the Clean Water Act Section 404 was created to mitigate wetland loss (Hough and Robertson 2009), but this Act was largely unsuccessful. An estimated 25,212 hectare of wetlands were lost in the US between 2004 and 2009 (Dahl 2011). Wetland loss has decreased in recent years due to not only the Clean Water Act Section 404 but also successful education programs and fewer economic incentives available to drain wetlands (EPA 2011).

The physical properties of a soil determine its hydraulic character (Mitsch and Gosslink 2000). Depositional events in the Lower Mississippi Alluvial Valley (LMAV) are variable over time and result in differential microtopography and hydrological gradients. The Lower Mississippi River, its tributaries, and the landscape within these watersheds have undergone immense change in the past (Dahl and Johnson 1991), resulting in the loss of hydrologic characteristics found in natural wetlands (Mitsch and Gosselink 1993). Since the 1970s, the most extensive losses of wetlands have occurred in Arkansas, Mississippi, and Louisiana (Dahl and Johnson 1991, Kress et al. 1996). The change in land cover type from wetland to agriculture likely change the capacity of the land to lessen flood events (Hopkinson and Day 1980a, Hopkinson and Day 1980b), and improve downstream water quality (Hupp and Morris 1990, Hupp and Bazemore 1993).

Mature forested wetlands are known to exhibit varied microtopography due to erosion and sedimentation processes (Barry et al. 1996). Microtopography refers to any surface roughness in a forest stand or wetland area, usually at most ± one m of average elevation. Microtopographic lows hold water seasonally and have been suggested to be beneficial for maintaining hydric soil properties and hydrophytes during restoration (Bruland and Richardson 2005, Moser et al. 2009, Simmons et al. 2011). Microtopographic manipulations (Swales, Flats, and Hummocks) benefit a wetland's hydrologic regime (Tweedy and Evans 2001). Restoration with roughened microtopography ± 0.5 m was found to retain surface water more frequently throughout a year compared to a planar restoration site of similar age in coastal North Carolina (Tweedy and Evans 2001). Soil survey hydrologic classifications, poorly drained, somewhat poorly drained, and somewhat excessively drained can be predictors of the duration water is ponded on a soil.

Swale and hummock microtopography reestablishment during wetland restoration is expected to provide hydrological and edaphic benefits on otherwise planar sites. In less than a year saturated soils tend to develop characteristic redox potentials as well as denitrification processes (Megonigal et al. 1993). Following these processes prolonged anoxic conditions, slow soil microorganism activity, and promote the accumulation of soil organic matter (SOM), a key soil response to wetlands restoration (Bruland and Richardson 2006; Ballantine and Schneider 2009). SOM refers to any undecayed animal and plant matter as well as other humic substances, mainly composed of four elements: C (52-58%), O (34-39%), H (3.3%-4.8%), and N (3.7%-4.1%) (Sparks 2002). No differences in soil total carbon was found though total nitrogen was greater in the swale features five years after swale and hummock establishment in east Texas. (Simmons et al. 2011). It was also noted that hydrophytes colonized an initially bare site and swales held surface water as expected (Simmons et al. 2011).

Topographic modifications were evaluated for hydrologic, edaphic, and vegetative responses at a three year old wetland restoration site in coastal North Carolina (Bruland and Richardson 2005). Restored features included, hummocks rising one meter above average elevation, swales, and unaltered flats. Soil bulk density and soil organic matter did not differ among recreated swales, hummocks, and associated flats. Soil nitrogen, nitrate-nitrite-ammonium, significantly differed among the features. The authors suggest that microtopography heterogeneity provides heterogeneous aerobic and anaerobic zones within a wetland, benefiting nitrogen transformation and retention during restoration. Greater plant species richness and diversity has been found at intermediate flat areas (Simmons et al. 2011, Bruland and Richardson 2005). This is attributed to intermediate duration hydroperiod, whereas swales and hummocks experience moisture extremes.

The Wetland Reserve Program (WRP) developed by USDA Natural Resources Conservation Service (NRCS) was created to provide incentives for private land owners to conserve their property. These incentives have been the subject of public attention for their lack of effectiveness in conserving land. Specific research has focused on the assessment of the effectiveness of riparian buffers and wetland restoration methods (Gilliam 1994, Hill 1996, Hughes et al. 2005). Conservation incentives have been provided for the establishment of riparian buffers and wetlands in agricultural watersheds totaling 51,757 hectares in the LMAV (Faulkner et al. 2011). The titles of these conservation programs are varied: Riparian Forest Buffer, Wetland Wildlife Habitat Improvement, and Wetland Reserve Enhancement. The objective of these programs is to return marginally productive, poorly drained farmlands and pasture to forested wetlands. It has been suggested that ecological restoration strategies must address the challenge of structural complexities that vary greatly by ecoregion and the criticism over the lack of standard restoration monitoring practices (Suding 2011). Limited progress of wetland restoration toward reference features is due to failure to establish hydrophytes and planted hardwood trees at desired locations (Stanturf et al. 2001, Patterson and Adams 2003). In wetland restoration, progress toward reference conditions can refer to an abundance of hydrophytes, as well as appropriate water levels, and soil organic matter accumulation (Ballantine and Schneider 2009).

Evaluating wetland function is difficult because there is no standard to use for comparison when studying a specific function. Therefore, numerous studies have used reference wetlands to represent optimal habitat conditions (Brinson 1993, Brinson and Rheinhardt 1996, Wilson and Mitsch 1996, Ashworth 1997, Brown and Smith 1998, Stolt et al. 2000). Studying vegetation composition is one of the most commons metrics used to compare restored wetlands to reference wetlands. Studying vegetation alone is a poor measure of a wetland's function but is a quick and effective method for studying the biogeochemical condition of a wetland and is commonly used as a measure of success (Breaux and Serefiddin 1999). There are many advantages to using vegetation as a biological indicator: they are present in most wetlands, relatively easy to identify, sampling methods are well established, and their low mobility creates a direct link between onsite environmental conditions and plant community characteristics (Cronk and Fennessy 2001). Balcombe et al. (2005) found that species composition indicate wetland quality. Atkinson et al. (2005) found that there are very few studies that examined vegetation of wetlands more than ten years after construction. The long-term development of wetlands is poorly understood (Zedler 2000).

It has been accepted that changes in the vegetation diversity are associated with the different stages of succession (Hill and Jones 1978, Sykes et al. 1989, Gilliam et al. 1995). It is expected that more mature sites have a higher plant species richness than restored sites, because the vegetation on the mature sites has been undisturbed for a longer time than restored sites. Hydrophytic vegetation generally returns to wetlands three to five years after restoration begins (Brown 1999).

Materials and Methods

Site Description

Two study sites were used in this study. Their relation to each other is shown in Figure 1. The two sites are 32.2 kilometers apart. One area, Cutoff is a natural forested wetland, and Bob White is a former agricultural land undergoing wetland restoration. Cutoff is located in Drew County in southeast Arkansas. The 3,488 hectare wildlife management area consists mainly of bottomland hardwood forest. The majority of the property was acquired in a purchase in 1955 through the Wildlife Restoration Program Grant Fund. Before the purchase, Cutoff was high-graded for timber, only the highest quality of trees were harvested for timber resulting in relatively poor quality residual stands. The average forest age ranges from 50-100 years. The area is located within the Bayou Bartholomew Ecobasin, which is located within the LMAV ecoregion. Elevations range from 38 - 46 meters above mean sea level. The majority of hydrology in Cutoff is influenced by two drainage points: one just north of the management area, and the Cut-off Creek-Bayou Bartholomew drainage. The Perry Clay soil series (very-fine, smectitic, thermic Chromic epiaquerts) is the primary soil mapped for the site. This hydric soil is very poorly drained, with zero to three percent slopes and composed of Arkansas River sediments (Cloutier and Finger 1967). Cutoff had a gradient in microtopography features: swales (low laying areas), hummocks/ridges (mounds), and flats (the intermediate between the other two features).

Bob White comprises 146 hectares and is located in the LMAV in Chicot County, Arkansas. Elevations range from 32-34 meters above sea level. Similar to Cutoff, the soils on the Bob White tract are mapped as Perry Clay series. Bob White was bottomland hardwood forest (BLHF), until the 1960's when it was converted into row crop production. Once Bob White was enrolled in the WRP in 2001 Swale excavation and hardwood planting of bald cypress (*Taxodium distichum*), water oak (*Quercus nigra*), overcup oak (*Quercus lyrata*), green ash (*Fraxinus pennsylvanica*), and willow oak (*Quercus phellos*) occurred. The swale excavations created low laying features called swales, the excavated material was used to create mounds or hummocks, high laying areas, and lastly the areas not manipulated created flats, the intermediate between the two other features. The region where the microtopographic manipulations were done and where the plots for this study were placed has an average planted tree survival zero per ha (Smith 2006).

Study Design

Five plots were randomly placed within each of three microtopographic features at each study site. The gradient of microtopographic features included swales (low laying areas), hummocks/ridges (mounds), and flats (the intermediate between the other two features). The vegetation plots were one m^2 plots. These plots were surveyed May 6th – May 11th, August 28th - August 31st, and October 22nd – October 29th in 2015. Plants were identified using Schummer et al. (2011). Species richness was determined for each plot. A plant was considered in the plot if it was rooted within the plot. Vines were counted if they were present in the plot. Plants that could not be identified were collected from the surrounding area and identified in a lab. Some of the species were only found in the plots, so pictures

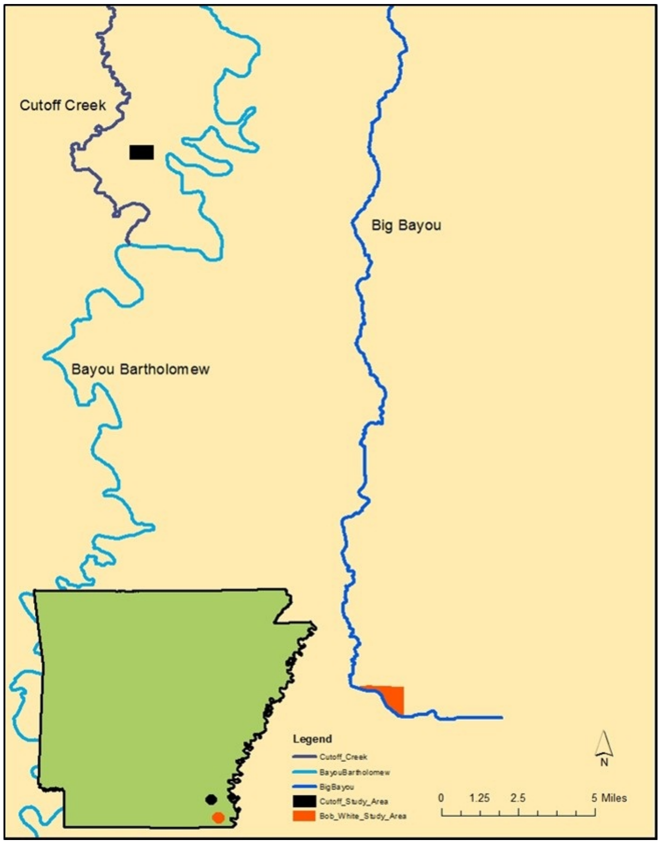

Figure 1. The location of the study sites coordinates for the vegetation plots at both study sites.

were taken and used for identification. Beta diversity was defined as the spatial variation in species composition and abundance between sampling units (Whittaker 1972). Sorensen-Dice Similarity Index and Jaccard/Tanimoto Coefficient were calculated across the two sites and within the sites and topographic features. The index ranges from zero to one where zero is no similarity between sites and one is exact similarity in species composition (Murguía and Villaseňor 2003, Tan et al. 2005).

The wetland indicator status of the plants that were identified to species were determined for the Atlantic and Gulf Coastal Plain region using the National Wetland Plant List by Lichvar et al. (2014). The wetland indicator status categories are defined as: Obligate (OBL)- almost always occurring under natural wetland conditions, Facultative Wetland (FACW)- usually occurring in wetlands but occasionally found in non-wetlands, Facultative (FAC)- equally likely to occur in wetlands and non-wetlands, Facultative Upland (FACU)- usually occurs in non-wetlands but occasionally found in wetlands, and Upland (UPL)- occurs in wetlands in another region, but almost always under natural conditions in non-wetlands in the region specified (Lichvar et al. 2012).

Beta diversity was calculated as

$$\beta = (A - C) + (B - C)$$

Where A is the total number of species found at a site, B is the total number of species found at another site, and C is the total number of species shared between the sites. Sorensen-Dice Similarity Index was calculated as

$$S = (2 * C)/(A + B)$$

Where A, B, and C are the same as above. Jaccard/Tanimoto Coefficient was calculated as

$$J = C/(A + B - C)$$

Where A, B, and C are the same as above.

Linear mixed-effects regression models were used to assess species richness. All models were fit using PROC MIXED in SAS 9.4 for Windows (SAS Institute, Cary, NC, U.S.A.). Species richness, models included study site, topographic feature, and topographic feature within study site. The random statements used included study site, topographic feature within study site, and survey period within topographic feature. All parameters were evaluated at an alpha level of 0.05.

Results

Thirty-three different plant species were identified at Bob White compared with 47 different species at Cutoff with two shared species between these two sites. There was no significant difference in the species richness between the two sites or the three topographic features. Species richness among the topographic features within site however was significantly different (Table 1). Within a study site only Bob White ridges and Bob White swales were not significantly different. Across study site only Bob White swales and Cutoff swales were not significantly different, shown in Table 1.

There were only two species found at both study sites. The shared species were *Brunnichia ovata*, and *Juncus effusus*. A list of the most common and rare species at both sites and topographic features within site is shown in Table 2.

When comparing the two study areas, the Beta diversity was calculated as 76, with a Sorensen-Dice Similarity Index of 0.05, and a Jaccard/Tanimoto Coefficient of 0.03. Cutoff ridges had the greatest overall diversity, whereas flats had the highest diversity at Bob White as shown in Table 3.

Table 1. The p-values for species richness at each topographic feature within each site.

Site and Topographic Feature	Site and Topographic Feature	P-value
Bob White Swale	Bob White Flat	<0.01
Bob White Swale	Bob White Ridge	>0.05
Bob White Flat	Bob White Ridge	<0.01
Cutoff Swale	Cutoff Flat	<0.01
Cutoff Swale	Cutoff Ridge	<0.01
Cutoff Flat	Cutoff Ridge	<0.01
Bob White Swale	Cutoff Swale	>0.05
Bob White Flat	Cutoff Flat	<0.01
Bob White Ridge	Cutoff Ridge	<0.01

Table 2. List of the most common and rare species at each site and at each topographic feature within site.

Site and Feature	Common Species	Rare Species
Bob White	*Campsis radicans*	*Ammannia auriculata*
Bob White Swales	*Typha latifolia*	*Salix nigra*
Bob White Flats	*Juncus marginatus*	*Ammannia auriculata*
Bob White Hummocks	*Baccharis halimifolia*	*Geranium carolinianum*
Cutoff	*Berchemia scandens*	Arisema dracontium
Cutoff Swales	*Gleditsia triacanthos*	*Quercus lyrata*
Cutoff Flats	*Berchemia scandens*	*Arisema dracontium*
Cutoff Ridges	*Toxicodendron radicans*	*Ostrya virginiana*

Table 3. The shared species, Beta diversity, Sorensen-Dice Similarity Index, Jaccard/Tanimoto Coefficient between each topographic feature and study site.

Topographic feature vs. Topographic feature	Shared Species	Beta Diversity	Sorensen-Dice Similarity Index	Jaccard Tanimoto Coefficient
Bob White Flat vs. Bob White Hummock	5	22	0.31	0.19
Bob White Flat vs. Bob White Swale	7	20	0.44	0.29
Bob White Hummock vs. Bob White Swale	2	18	0.18	0.1
Bob White Flat vs Cutoff Flat	1	41	0.05	0.02
Bob White Hummock vs. Cutoff Ridge	0	36	0	0
Bob White Swale vs. Cutoff Swale	0	25	0	0
Cutoff Flat vs. Cutoff Ridge	6	35	0.26	0.15
Cutoff Flat vs. Cutoff Swale	5	22	0.31	0.19
Cutoff Ridge vs. Cutoff Swale	3	33	0.15	0.08

Table 4. The species richness, percent OBL species, % of FACW species, % of FAC species, % of FACU species, % of UPL species, found at each topographic feature at each study site.

Topographic feature	Species Richness	% of OBL Species	% of FACW Species	% of FAC Species	% of FACU Species	% of UPL Species
Bob White Flat	23	38%	31%	19%	13%	0%
Bob White Hummock	9	0%	33%	33%	17%	17%
Bob White Swale	13	67%	17%	17%	0%	0%
Cutoff Flat	20	9%	36%	36%	18%	0%
Cutoff Ridge	27	0%	18%	24%	41%	18%
Cutoff Swale	12	30%	20%	20%	30%	0%

Of the plants identified to species, swales at both sites had the most plants and highest percentage of OBL species shown in Table 4. Two-thirds of plants identified to species were OBL or FACW species at Bob White. Only 34% of the species found at Cutoff were OBL or FACW species. Bob White flats had the highest number of individual species with a wetland indicator status of OBL or FACW. At Cutoff the topographic feature with the most OBL and FACW species were the swales.

Ipomoea wrightii (Wright's morning glory) and *Rumex crispus* (Curly dock) *Triadica sebifera* (Chinese tallow), and *Cardiospermum halicacabum* (Balloon vine) are introduced species found at Bob White. *T. sebifera* and *C. halicacabum* were not in any of the survey plots but was seen at the site. *Vicia sativa* (Common vetch) was the only introduced species at Cutoff.

Discussion

The results demonstrated that Bob White did not have a significantly different species richness from Cutoff even though species composition was different. Vegetation diversity differences are, however, common between restored wetlands and natural wetlands (Galatowitsch and van der Valk 1996, Ashworth 1997, Fennessy and Roehrs 1997). There were differences in species richness between the topographic features across site except for the swales. There should be a difference in plant diversity between the topographic features at both sites because it is accepted that hydrologic regime impacts plant diversity (Brinson et al. 1981, Keddy 2000). Flats had the highest species richness at Bob White which is supported by Simmons et al. (2011) and Bruland and Richardson (2005) who both reported greater plant species richness and diversity at intermediate elevation, flat areas. This is attributed to intermediate duration of hydroperiod, whereas swales and hummocks experience moisture extremes. These extremes in moisture may explain why there was not a significant difference in species richness between the swales and ridges at Bob White although the composition of the diversity was different. The flats shared the most species in common with swales and ridges/hummocks within study sites. The amount of shared species is likely because flats are a transitional landscape between the two topographic features and could have varying hydrologic regimes that do not favor swale or ridge/hummock species over the other. Strangely the shared species that were found were found at Bob White's flats and hummocks and Cutoff's flat and swales. This was unusual because the gradient of topographic features should have different hydrologic regimes, which should influence species composition. Bob White hummocks had the lowest species richness, while the ridges at Cutoff had the highest species richness. The presence of *B. halimifolia* may be the reason for the low richness at Bob White hummocks.

There were not many tree species at Bob White. Only *Salix nigra* was found in the survey plots but planted *T. distichum*, *Q. nigra*, *Q. lyrata*, *F. pennsylvanica*, and *Q. phellos* can be found at Bob White. *T. sebifera* has naturally invaded Bob White. Smith (2006) found a tree survival of zero trees per ha in the area where the topographic manipulations were done. In New Jersey a tree layer was not found at old-field sites in the coastal plain region until 25 years after abandonment (Hanks 1971). In addition Noon (1996) found no tree recruitment for the first 11 years following wetland creation on mineral soils of reclaimed mine lands. Battaglia et al. (2002) reported on a restoration site in the LMAV where ten species were planted. Fifteen years after planting there was a total of 16 tree and shrub species found at the site. After 16 years Battaglia et al. (2002) found a tree layer emerging from beneath the ground layer of vegetation. This shift in community is indicative of the transition phase between an old-field and young forest community (Bonck and Penfound 1945, Hopkins and Wilson 1974). A successful bottomland restoration project should include more than 15 woody species, such as oak, and hickory species (Allen 1990, 1997, Allen et al. 1998). Our study suggests that Bob White has not yet transitioned to a young forest community. One factor could be the presence of *B. halimifolia* (Groundsel bush). *B. halimifolia* was found within all topographic features of Bob White but was absent from Cutoff. Battaglia et al. (2008) found that *B. halimifolia* inhibits the regeneration of some plant species. This is consistent with many other studies that have shown that shrub species negatively interact with colonizing plants (Callaway 1992, Holl et al. 2000, Posada et al. 2000, Slocum 2001, Gomez-Aparicio et al. 2004, Zanini and Ganade 2005, Battaglia et al. 2008). Management of *T. sebifera* and *C. halicacabum* is suggested so they like *B. halimifolia* will not inhibit other plants from colonizing.

Woody vegetation generally takes longer to establish in created wetlands (Niswander and Mitsch 1995). De Steven et al. (2010) found that the cover of woody species in restored wetlands average 40% after five years and that restored sites had 53% of the same species in forested reference sites. Yepsan (2014) however found no correlation between time since restoration and percent of woody species. Seed banks and seed dispersal are the major sources of propagules in restored wetlands. However, the seed banks of farm fields are dominated by herbaceous species and often do not contain woody species (De Steven et al. 2006, Middleton 2003). Restored wetlands including Bob White are often surrounded by agricultural fields rather than forests, limiting dispersal of the propagules of woody species may explain their lower abundance in restored wetland (Herault and Thoen 2009, Kettenring and Galatowitsch 2011). Swales and flats are exceptionally isolated because they rarely receive overland flow of water from other wetlands, which leaves wind transport as the major source of woody propagules (Greene and Johnson 1996). Diversity in early succession is positively influenced by external dispersal of seeds (Pacala and Rees 1998). The previous land use history is important in determining the species composition of secondary forests (Grau et al. 1997, Chinea and Helmer 2003, Ito et al. 2004). Both sites in this study are surrounded by agricultural fields, and the dispersal rate should be low. Bob White does have a narrow riparian zone of trees that consists of animal or wind dispersed seed species surrounding the dike system that supplies the adjacent agricultural fields. This riparian area may be a source of seeds for Bob White but shortest distance from the surrounding patch of trees to one of the excavated swales is 230 m. The majority of preferred bottomland species density decreases at 60-80 m from a remnant forest but this issue decreases as the forests ages (Allen 1997, Brunet et al. 2000) as animal dispersion increases. Natural invasion should not be the only method of dispersion for plant species other than trees. Mid-story and shrub species should be planted; they are an important part of the natural forest structure.

At both sites and at all three topographic features there was a decrease in species richness over the study period. During the first survey in May water was still present at both sites which may have inhibited some plants from growing in the swales. The presence of water could explain why the inundated swales had an increase in diversity between the spring and summer surveys, as the presence of water may have inhibited some plants from growing early in the spring. Besides the changing of the seasons and a plant's natural life cycle causing a decrease in species richness over time there was also a dry spell from August 28th – October 19th, where only 1.8 cm of rain fell and 0.43 cm of that rain fell on October 9th (NADP 2016). This dry spell may have also been a factor in the loss of species richness over the time period of this study.

One indication that wetlands species are becoming reestablished at Bob White is that the swales had a significant proportion of OBL vegetation. Hydrophytic vegetation typically returns to a wetland 3-5 years after restoration begins (Erwin and Best 1985, Confer and Niering 1992, Reinartz and Warne 1993, Mitsch et al. 1998, Brown 1999). It had been 14 years after restoration at Bob White therefore the proportion of hydrophytes is to be expected. The U.S. Army Corps of Engineers (1987) stated that in order for a site to be considered to have a hydrophytic vegetation community the total number of species that are OBL, FACW and FAC+ must equal at least 50%. Examining just OBL and FACW species, Bob White Swales and Flats and Cutoff Swales met this criterion. The hummocks at Bob White also do not have any OBL species but have UPL species suggesting that swale excavation was successful at forming a landscape with diverse hydrology and therefore a wide range of plant species. OBL species are indicative of wetlands and ecosystem services (Naeem et al. 1999). Bob White may be providing ecosystem services since ecosystem

The time frame for a restored bottomland hardwood forest to resemble a natural bottomland hardwood forest can take decades (Kusler 1986, Mitsch and Gosselink 1993, King et al. 2006). Cutoff is in the range of 50-100 years old on average. Bob White's restoration began in 2001; it is therefore unreasonable to expect Bob White to resemble a natural bottomland forest at this point in time. According to Frenkel and Morlan (1991) success of forested wetlands should not be judged until 15-50 years after restoration. It is too early to tell if Bob White has been a success because the factor one uses to determine success will determine whether or not restoration has been successful. To determine if Bob White will be successfully restored more studies like this should be conducted.

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Literature Cited

Allen JA. 1990. Establishment of bottomland oak plantations on the Yazoo National Wildlife Refuge Complex. Southern Journal of Applied Forestry 14:206–210.

Allen JA. 1997. Reforestation of Bottomland Hardwoods and the Issue of Woody Species Diversity. Restoration Ecology 5(2):125-134. http://www.fs.fed.us/psw/publications/allen/psw_1997_allen001.pdf

Allen JA, J McCoy and **BD Keeland**. 1998. Natural establishment of woody species on abandoned agricultural fields in the lower Mississippi Valley: first- and second-year results. p. 263–268. *In* T. A. Waldrop (ed.) Proceedings of the ninth biennial southern silvicultural research conference. USDA Forest Service, Southern Research Station, Asheville, NC, USA.

Ashworth SM. 1997. Comparison between restored and reference sedge meadows in south-central Wisconsin. Wetlands 17:518– 527.

Atkinson RB, JE Perry and **J Cairns Jr**. 2005. Vegetation communities of 20-year-old created depressional wetlands. Wetlands Ecology and Management 13(4):469-78.

Ballantine K and **R Schneider**. 2009. Fifty-five years of soil development in restored freshwater depressional wetlands. Ecological Applications 19(6):1467-1480.

Balcombe CK, JT Anderson, RH Fortney, JS Rentch, WN Grafton and **WS Kordek**. 2005. Wetlands. 25(1):130-142.

Barry WJ, AS Garlo and **CA Wood**. 1996. Duplicating the mound-and-pool microtopography of forested wetlands. Ecological Restoration. 14(1):15-21.

Battaglia LL, PR Minchin and **DW Pritchett**. 2002. Sixteen years of old-field succession and establishment of a bottomland hardwood forest in the Lower Mississippi Alluvial Valley. Wetlands 22:1–17.

Battaglia LL, DW Pritchett and **PR Minchin**. 2008. Evaluating Dispersal Limitation in Passive Bottomland Forest Restoration. Restoration Ecology 16(2): 417-424

Bonck J and **WT Penfound**. 1945. Plant succession on abandoned farmland in the vicinity of New Orleans, Louisiana. American Midland Naturalist 33:520–529.

Breaux A and **F Serefiddin**. 1999. Validity of performance criteria and a tentative model of regulatory use in compensatory wetland mitigation permitting. Environmental Management 24:327–336.

Brinson MM, AE Lugo and **S Brown**. 1981. Primary productivity, decomposition, and consumer activity in freshwater wetlands. Annual Review of Ecology and Systematics 12:123–161.

Brinson MM and **RD Rheinhardt**. 1996. The role of reference wetlands in functional assessment and mitigation. Ecological Applications 6:69–76

Brown SC and **CR Smith**. 1998. Breeding season bird use of recently restored versus natural wetlands in New York. Journal of Wildlife Management 62:1480–1491.

Brown SC. 1999. Vegetation similarity and avifaunal food value of restored and natural marshes in northern New York. Restoration Ecology 7:56–68.

Bruland GL and **CJ Richardson**. 2005. Hydrologic, edaphic, and vegetative responses to microtopographic reestablishment in a restored wetland. Restoration Ecology 13(3):515-523.

Bruland GL and **CJ Richardson**. 2006. Comparison of soil organic matter in created, restored and paired natural wetlands in North Carolina. Wetlands Ecology and Management 14:245-251.

Brunet J, G von Oheimb and **M Diekmann**. 2000. Factors influencing vegetation gradients across ancient-recent woodland borderlines in southern Sweden. Journal of Vegetation Science 11:515–524.

Callaway RM. 1992. Effects of shrubs on recruitment of *Quercus douglasii* and *Quercus lobata* in California. Ecology 73:2118–2128.

Chinea JD and **EH Helmer**. 2003. Diversity and composition of tropical secondary forests recovering from large-scale clearing: results from the 1990 inventory in Puerto Rico. Forest Ecology and Management 180:227–240.

Cloutier H and **CJ Finger**. 1967. Soil survey of Chicot County, Arkansas. US Department of Agriculture, Soil Conservation Service. 58 p.

Confer SR and **WA Niering**. 1992. Comparison of created and natural freshwater emergent wetlands in Connecticut, USA. Wetlands Ecology and Management 2:143–156.

Cronk JK and **MS Fennessy**. 2001. Wetland Plants: Biology and Ecology. Lewis Publishers, Boca Raton, FL.

Dahl TE and **CE Johnson**. 1991. Status and trends of wetlands in the conterminous United States, Mid-1970s to Mid-1980s. U.S. Department of Interior, Fish and Wildlife Service.

Dahl TE. 2011. Status and Trends of Wetlands in the Conterminous United States 2004 to 2009. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. 108 pp. http://www.fws.gov/wetlands/Status-And-Trends-2009/index.html.

De Steven D, RR Sharitz, JH Singer and **CD Barton**. 2006. Testing a passive revegetation approach for restoring Coastal Plain depression wetlands. Restoration Ecology 14:452–460.

De Steven D, RR Sharitz and **CD Barton**. 2010. Ecological outcomes and evaluation of success in passively restored southeastern depressional wetlands. Wetlands 30:1129–1140.

Erwin KL and **GR Best**. 1985. Marsh and forested wetland reclamation of a central Florida phosphate mine. Wetlands 4:87–104

Faulkner S, W Barrow, B Keeland, S Walls and **D Telesco**. 2011. Effects of conservation practices on wetland ecosystem services in the Mississippi Alluvial Valley. Ecological Applications 21:S31-S48.

Fennessy S and **J Roehrs**. 1997. A functional assessment of mitigation wetlands in Ohio: comparisons with natural systems. Ohio Environmental Protection Agency Division of Surface Waters, Columbus, OH, USA.

Frenkel RE and **JC Morlan**. 1991. Can we restore our salt marshes? Lessons from the Salmon River, Oregon. Northwest Environmental Journal 7:119–135.

Galatowitsch SM and **AG van der Valk**. 1996. The vegetation of restored and natural prairie wetlands. Ecological Applications 6:102–112.

Gilliam JW. 1994. Riparian wetlands and water quality. Journal of Environmental Quality. 23(5):896-900.

Gilliam FS, NL Turrill and **MB Adams**. 1995. Herb-layer and overstory species in clear-cut and mature central Appalachian hardwood forests. Ecological Applications 5:947–955.

Gomez-Aparicio L, R Zamora, JM Gomez, JA Hodar, J Castro and **E Baraza**. 2004. Applying plant facilitation to forest restoration: a meta-analysis of the use of shrubs as nurse plants. Ecological Applications 14:1128–1138.

Grau HR, MF Arturi, AD Brown and **PG Acenolaza**. 1997. Floristic and structural patterns along a chronosequence of secondary forest succession in Argentinean subtropical montane forests. Forest Ecology and Management 95:161–171.

Greene DF and **EA Johnson**. 1996. Wind dispersal of seeds from a forest into a clearing. Ecology. 77 (2): 595–609.

Hanks JP. 1971. Secondary succession and soils on the inner coastal plain of New Jersey. Bulletin of the Torrey Botanical Society 98(6):315–321.

Herault B and **D Thoen**. 2009. How habitat area, local and regional factors shape plant assemblages in isolated closed depressions. Acta Oecologica. 35:385–392.

Hill AR. 1996. Nitrate removal in stream riparian areas. Journal of Environmental Quality. 25:743-755.

Hill MO and **EW Jones**. 1978. Vegetation changes resulting from afforestation of rough grazing in Caeo forest, south Wales. Journal of Ecology 66:433-456.

Holl KD, ME Loik, EHV Lin and **IA Samuels**. 2000. Tropical montane forest restoration in Costa Rica: overcoming barriers to dispersal and establishment. Restoration Ecology 8:339–349.

Hopkins WE and **RE Wilson**. 1974. Early oldfield succession on bottomlands of southeastern Indiana. Castanea 39:57–71.

Hough P and **M Robertson**. 2009. Mitigation under Section 404 of the Clean Water Act: where it comes from, what it means. Wetlands Ecology and Management 17:15-33.

Hughes FM, A Colston and **JO Mountford**. 2005. Restoring riparian ecosystems: the challenge of accommodating variability and designing restoration trajectories. Ecology and Society 10(1):1-12.

Hupp CR and **EE Morris**. 1990. A dendrogeomorphic approach to measurement of sedimentation in a forested wetland, Black Swamp, Arkansas. Wetlands. 10:107-124.

Hupp CR and **DE Bazemore**. 1993. Temporal and spatial patterns of wetland sedimentation, West Tennessee. Journal of Hydrology. 141:179-196.

Ito S, R Nakayama and **GP Buckley**. 2004. Effects of previous landuse on plant species diversity in semi-natural and plantation forests in a warm-temperate region in southeastern Kyushu, Japan. Forest Ecology and Management 196:213–225.

Keddy PA. 2000. Wetland Ecology: Principles and Conservation. Cambridge Studies in Ecology, Cambridge University Press, Cambridge, UK.

Kettenring KM and **SM Galatowitsch**. 2011. Carex seedling emergence in restored and natural prairie wetlands. Wetlands 31, 273–281.

King SL, DJ Twedt and **RR Wilson**. 2006. The role of the wetland reserve program in conservation efforts in the Mississippi river alluvial valley. Wildlife Society Bulletin 34(4):914-920.

Kleiss BA. 1996. Sediment retention in a bottomland hardwood wetland in eastern Arkansas. Wetlands. 16:321-333

Kress MR, MR Graves and **SG Bourne**. 1996. Loss of bottomland hardwood forests and forested wetlands in the Cache River Basin, Arkansas. Wetlands 16:258-263.

Kusler JA. 1986. Wetland creation/restoration: a summary of science views and perspectives. Pages 440–446 in J. A. Kusler, M. L. Quammen, and G. Brooks (eds.), Proceedings of the national wetland symposium: Mitigation of impacts and losses. ASWM Technical Report 3. Association of State Wetland Managers, Berne, New York.

Lichvar RW, NC Melvin, ML Butterwick and **WN Kirchner**. 2012. National Wetlands Plant List Indicator Rating Definitions. Wetland Regulatory Assistance Program, US Army Corps of Engineers.

Lichvar RW, M Butterwick, NC Melvin and **WN Kirchner**. 2014. The national wetland plant list: 2014 update of wetland rating. US Army Corps of Engineers Phytoneuron 2014-41: 1-42.

Naeem S, FS Chair, RC Chapin III, PR Ehrlich, FB Golley, DU Hooper, JH Lawton, et al. 1999. Biodiversity and Ecosystem Functioning: Maintaining Natural Life Support Processes. Ecological Society of America. 4: 1-14.

Megonigal JP, WH Patrick and **SP Faulkner**. 1993. Wetland identification in seasonally flooded forest soils: soil morphology and redox dynamics. Soil Science Society of America Journal. 57:140-149

Middleton BA. 2003. Soil seed banks and the potential restoration of forested wetlands after farming. Journal of Applied Ecology. 40: 1025–1034.

Mitsch WJ and **JG Gosselink**. 1993. Wetlands. Van Nostrand Reinhold, New York, 722 pp.

Mitsch WJ and **JG Gosselink**. 2000. Wetlands, third. John Wiley and Sons, Canada

Mitsch WJ, X Wu, RW Nairn, PE Weithe, N Wang, R Deal and **CE Boucher**. 1998. Creating and restoring wetlands: a whole ecosystem experiment in self-design. BioScience 48:1019–1030.

Moser K, C Ahn and **GB Noe**. 2009. The influence of Microtopography on soil nutrients in created mitigation wetlands. Restoration Ecology 17(5):641-651

Murguía M and **JL Villaseňor**. 2003. Estimating the effect of similarity coefficient and the cluster algorithm on biogeographic classifications. Finnish Zoological and Botanical Publishing Board. 40:415-421.

National Atmospheric Deposition Program (NADP). 2016. Precipitation data. Retrieved February 26th, 2016. http://nadp.sws.uiuc.edu/siteOps/ppt/default.aspx

Niswander SF and **WJ Mitsch**. 1995. Functional analysis of a two-year-old created in-stream wetland: hydrology, phosphorus retention, and vegetation survival and growth. Wetlands 15:212–225.

Noon KF. 1996. A model of created wetland primary succession. Landscape and Urban planning 34:97–123.

Pacala SW and **M Rees**. 1998. Models suggesting field experiments to test two hypotheses explaining successional diversity. The American Naturalist 152:729–737.

Patterson WB and **JC Adams**. 2003. Soil, hydroperiod and bedding effects on restoring bottomland hardwoods on flood-prone agricultural lands North Louisiana, USA. Forestry 76(2):181-188

Posada JM, TM Aide and **J Cavelier**. 2000. Cattle and weedy shrubs as restoration tools of tropical montane rainforest. Restoration Ecology 8:370–379.

Reinartz JA and **Warne EL**. 1993. Development of vegetation in small created wetlands in southeast Wisconsin. Wetlands 13: 153–164.

Ruiz-Jaén MC and **TM Aide**. 2005. Vegetation structure, species diversity, and ecosystem processes as measures of restoration succession. Forest Ecology and Management 218:159–173

SAS Institute. 2013. The SAS system for Windows. Release 9.4. SAS Inst.., Cary, NC

Schummer ML, HM Hagy, KS Fleming, JC Cheshier and **JT Callicutt**. 2011. A Guide to Moist Soil Plants of the Mississippi Alluvial Valley. University Press of Mississippi.

Simmons ME, X Ben Wu and **SG Whisenant**. 2011. Plant and soil responses to created microtopography and soil treatments in bottomland hardwood forest restoration. Restoration Ecology 19(1):136-146

Slocum MG. 2001. How tree species differ as recruitment foci in a tropical pasture. Ecology 82:2547–2559.

Smith J, S Holli and **L Vickers.** 2006. Bob White Memorial Wetlands Research and Teaching Station management plan. University of Arkansas at Monticello. School of Forest Resources, Monticello AR.

Sparks D. 2002. Environmental Soil Chemistry. Academic Press, London

Stanturf JA, SH Schoenholtz, CJ Schweitzer and **JP Shepard**. 2001. Achieving Restoration Success: Myths in Bottomland Hardwood Forests. Restoration Ecology 9(2):189-200.

Stolt MH, MH Genthner, WL Daniels, VA Groover, S Nagle and **KC Haering**. 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. Wetlands 20:671–683.

Suding KN. 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. Annual Review of Ecology, Evolution and Systematics 42(1):465-487.

Sykes JM, VPW Lowe and **DR Briggs**. 1989. Some effects of afforestation on the flora and fauna of an upland Sheepwalk during 12 years after planting. Journal of Applied Ecology 26:299–320

Tan PN, M Steinbach and **V Kumar**. 2005. Introduction to Data Mining.

Tweedy KL and **RO Evans**. 2001. Hydrologic characterization of two prior converted wetland restoration sites in eastern North Carolina. Transaction of the American Society of Agricultural Engineers 44:1135-1142.

United States Department of Agriculture, **NRCS**. 2016. The PLANTS Database (http://plants.usda.gov, 25 February 2016). National Plant Data Team, Greensboro, NC 27401-4901 USA.

US Environmental Protection Agency (EPA). 2011. Wetlands-status and trends. EPA, Washington. Retrieved November 4, 2014. http://water.epa.gov/type/wetlands/vital_status.cfm

Walbridge MR. 1993 Functions and values of forested wetlands in the southern United States. Journal of Forestry 91:15-19.

Whittaker RH. 1972. Evolution and measurement of species diversity. Taxonomy 21:213-251

Wilson RF and **WJ Mitsch**. 1996. Functional assessment of five wetlands constructed to mitigate wetland loss in Ohio, USA. Wetlands 16:436–451.

Yepsan M, AH Baldwin, DF Whigham, E McFarland, M LaForgia and **M Lang**. 2014. Agricultural wetland restorations on the USA Atlantic Coastal Plain achieve native wetland plant communities but differ from natural wetlands. Agriculture, Ecosystems and Environment 197:11-20.

Zanini L and **G Ganade**. 2005. Restoration of Araucaria forest: the role of perches, pioneer vegetation, and soil fertility. Restoration Ecology 13:507–514.

Zedler JB. 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecology and the Environment 1(2):65-72.