Bee Communities of Emergent Wetlands Under Restoration in the Lower Mississippi Alluvial Valley of Arkansas

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Abstract - Native bee communities in wetlands are poorly described and the recognized loss of wetlands in the United States adds to the need to better understand these communities. In particular, wetlands in the lower Mississippi Alluvial Valley (LMAV) have declined >50% during the past 70 years. Loss of wetlands and intensification of the region's agriculture are thought to be putting native bee communities at risk. We sampled palustrine emergent wetlands being restored through either active or passive management in the Arkansas portion of the LMAV to assess how those management practices related to bee species richness and diversity. During the 2015 and 2016 growing seasons, we captured 17,860 individual bees representing 5 families, 83 species, and 31 genera; 17 species of which were singletons. Thirteen species were unique to actively managed stands, and 15 species were unique to passively managed stands. Neither species richness nor diversity differed between actively and passively managed wetland sites. Although management type did not have a strong impact on bee communities, we maintain that these restored wetlands have created attractive patches of habitat for native bees.

Introduction

Native bee populations may be in decline in part due to the loss of diverse native plant communities resulting from the expansion of agriculture (Brown and Paxton 2009). Of those habitat types cleared for agricultural practices, wetlands have suffered the most (Dahl 2011). Bottomland hardwood forests, emergent wetlands, and prairies once dominated the lower Mississippi Alluvial Valley (LMAV; King et al. 2006) of Arkansas. These lands were extensively cleared and converted to agriculture between the 1950s and 1970s after commodity prices reached an all-time high (King et al. 2006). Land conversions, as well as agricultural intensification, have rendered native bee communities in critical need of conservation. Models suggesting that compared to other regions, the LMAV's wild bee communities have relatively low abundances while at the same time the region's need for their pollination services is the highest (Koh et al. 2016).

As land-use patterns changed over time, federal conservation programs were established to counteract habitat loss. The Agricultural Conservation Easement Program, previously known and hereafter referred to as the Wetland Reserve

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Program (WRP), was established in 1990 under the United States Farm Bill to offer landowners the opportunity to voluntarily protect, restore, and enhance wetlands or previous wetlands on their property. Since the WRP was established, >279,235 ha of wetlands have been restored/reestablished in the LMAV, of which 91,886 ha are in Arkansas (NRCS 2017a, Twedt and Uihlein 2005). Though the intent of the WRP is to provide flood protection, reduce soil erosion, improve water quality, and provide wildlife habitat, its potential role in providing nesting and food resources for bees and other pollinators has not been documented (Brown and Paxton 2009, NRCS 2017b).

Bees require nesting sites (bare-ground, pithy stems, cavities), nest materials, and food resources (pollen and nectar) to survive and reproduce (Gathmann and Tscharntke 2002, Steffan-Dewenter 2003). Not only do bees require pollen and nectar for food resources, but Fowler and Droege (2018) found that about 30% of ~450 species of bees native to the mid-Atlantic and northeastern United States are pollen specialists restricted to native plants at the family level or lower; many of which are wetland associated. Wetlands surrounded by a matrix of agricultural landscapes may provide unique floral resources for specialist bees.

Floral resources must also be within an appropriate foraging range of species-specific nesting habitat. Body length of bees has been correlated with flight distance, with small-bodied (6–13 mm) and large-bodied (21–25 mm) bees traveling up to 300 m and 1200 m, respectively, from their nest to forage (Gathmann and Tscharntke 2002, Zurbuchen et al. 2009). These relatively short foraging distances suggest the need for proper juxtaposition of suitable habitats to sustain a variety of bee species. Foraging distance is not the same as maximum flight distance, but limited information exists for a majority of solitary bees as tracking options have not evolved for such small organisms.

Of the many types of wetlands that occur in the LMAV (Nelms 2007), palustrine emergent (Cowardin et al. 1979) is targeted for management by wetland managers to meet food and cover requirements for waterfowl (Frederickson and Taylor 1982, Manley et al. 2004, Reinecke et al. 1989, Smith et al. 1989). Palustrine emergent wetlands are defined by Cowardin et al. (1979) as areas <8 ha in size, lacking active wave-formed or bedrock shoreline features, with water depth in the deepest part of the basin <2.5 m at low water, and salinity due to ocean-derived salts less than 0.5 ppt. These palustrine emergent wetlands are managed in the LMAV either actively during the first ~7 years to promote an annual plant community or passively to allow natural succession to promote a perennial plant community (Manley et al. 2004, Reinecke et al. 1989). We categorized these 2 management types into active and passive management treatments. Emergent wetlands are actively managed by manipulating vegetation, seed banks, hydrology, and soils through disking, herbicide applications, or mowing generally every 1–3 years (Frederickson and Taylor 1982, Nelms 2007). These actively managed wetlands, referred to as moist-soil units, are typically moist in the spring, dry in the summer, and moist or inundated again in the fall. Actively managed emergent wetlands convert to passive emergent wetlands within ~4 years in the LMAV once active managment measures cease and if major disturbance does not occur (Manley et al. 2004, Reinecke et al. 1989, Strader and Stinson 2005). Passively managed emergent wetlands are generally disturbed every 4–7 years, but a perennial emergent wetland will convert to the next successional stage (scrub-shrub) due to the lack of disturbance beginning at about 8 years. These passive emergent wetlands can be impounded or naturally occurring and are managed to produce perennial plants. More often than not, these passive emergent wetlands result from not actively managing emergent wetlands. These passively managed wetlands are typically moist or inundated from autumn to late spring and fluctuate with natural evaporation and rain events throughout the summer.

Given our limited knowledge of native bees in the LMAV and the unknown effects of management of palustrine emergent wetlands, we sampled palustrine emergent wetlands managed actively for annual plant communities or passively for perennial plant communities to assess how those management options related to bee species richness and diversity. We anticipate that our results will help managers better understand how these 2 wetland management options relate to native bee communities in the LMAV.

Study Area

We collected bees on palustrine emergent wetlands in the LMAV of Arkansas (Fig. 1, Table 1). Because our study sites included restored palustrine emergent wetlands (see below), we included palustrine emergent wetlands that exceeded the 8 ha maximum defined in Cowardin et al. (1979).

Historically the LMAV of Arkansas included vast wetlands in the floodplains and wet prairies between the floodplains (Branner 1908, Foti 2001). The elevation of the LMAV varies by 46 m throughout the entire 402 km length of the LMAV in Arkansas (Crow 1974). The region is now dominated by agriculture (soybean, rice, corn, sorghum, and cotton; ~61%) with fragments of remnant emergent wetlands (1%) and bottomland hardwood forest (17%) (King et al. 2006, USDA 2016). All sites we surveyed had been in row crops from 5 to 20 years previously and have since been impounded. Our sites were either being managed as moist-soil units, reestablished to emergent wetlands through the WRP, or were naturally succeeding back to emergent wetlands.

Methods

We selected 2 groups of emergent wetland sites based on their previous management histories and successional stages. We defined sites as actively managed wetlands if >10% of the unit had been disked, sprayed, or mowed that year and/or if >75% of the unit was disked or sprayed in the previous 2 years. We defined sites as passively managed wetlands if <10% of the unit had been disked, sprayed, or mowed that year and if <75% of the unit was disked or sprayed in the previous 2 years. Actively managed wetlands were usually drained by late May and disked in early July to reset succession and produce seeding grasses (e.g., *Leptochloa* spp. [sprangletop], *Echinochloa crus-galli* (L.) P. Beauv. [Barnyardgrass], *Dichanthelium* spp. [panicgrass]) for migratory birds. Passively managed wetlands evaporated naturally throughout the growing season, thereby retaining soil moisture well into the growing season. Passively managed sites had a longer bloom period for hydrophytic plants such as *Hydrolea uniflora* Raf. (Oneflower False

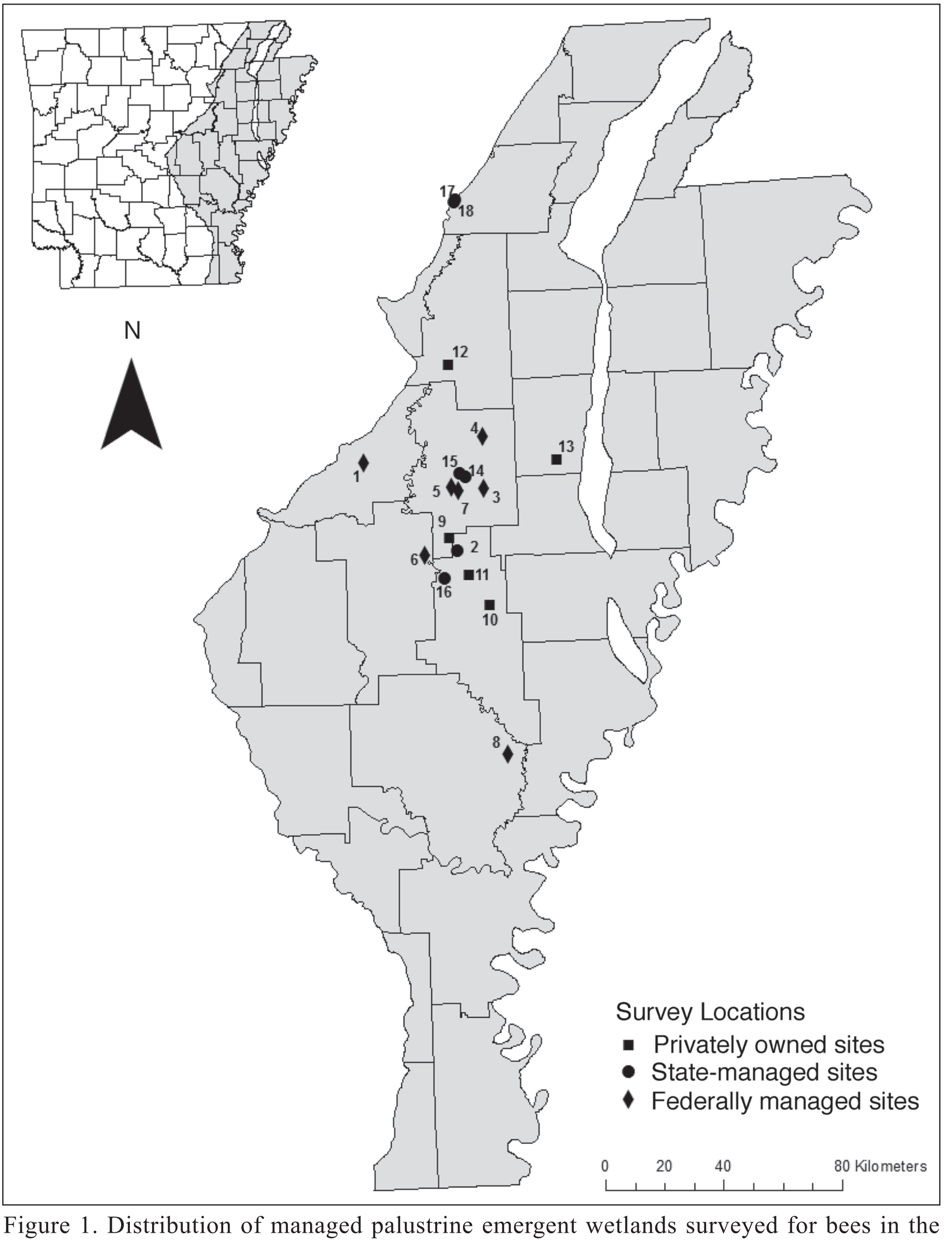

Figure 1. Distribution of managed palustrine emergent wetlands surveyed for bees in the lower Mississippi Alluvial Valley of Eastern Arkansas in 2015 and 2016. See Table 1 for site names.

Fiddleleaf) and *Ludwigia peploides* (Kunth) P.H. Raven ssp. *glabrescens* (Kuntze) P.H. Raven (Floating Primrose-willow).

Bee survey

We sampled bees across 18 sites each at least 600 m apart, with most sites (89%) more than 2 km apart to account for documented foraging distances in solitary bees (Gathmann and Tscharntke 2002). The 2015 field season was a pilot season during which 11 sites were sampled 4–8 times each depending on time available, for a total of 65 trapping days. In 2016, we sampled 17 wetland sites 8 times each for 136 total trapping days. We sampled bees during 19 May–18 September 2015 and 22 May–9 September 2016 at each site, every other week once initiated. We grouped sampling

Table 1. Site number, site name, ownership, pre-management status, method of wetland reestablishment, and times surveyed during 2015 and 2016 in the lower Mississippi Alluvial Valley of eastern Arkansas. NWR = National Wildlife Refuge, WMA = Wildlife Management Area. Management type: P = passive, A = active. Site ownership: USFWS = US Fish and Wildlife Service, ANHC = Arkansas Natural Heritage Commission, Private = private land, and AGFC = Arkansas Game and Fish Commission. Reesatblishment method: Natural = natural succession, WRP = wetland reserve program, and MSM = moist-soil management

Site #	Study site	Management type	Site ownership	Pre-management	Reestablish. method	Number of times surveyed 2015	2016
1	Bald Knob NWR	P	USFWS	Agriculture	Natural	5	8
2	Benson Creek Natural Area	P	ANHC	Agriculture	WRP	6	8
3	Cache River NWR Cabin	A	USFWS	Agriculture	WRP	-	8
4	Cache River NWR Hwy 64	P	USFWS	Agriculture	WRP	-	8
5	Cache River NWR Lower Howell Unit	P	USFWS	Agriculture	WRP	8	8
6	Cache River NWR Plunkett Farm Unit	A	USFWS	Agriculture	MSM	5	-
7	Cache River NWR Upper Howell Unit	A	USFWS	Agriculture	WRP	8	8
8	White River NWR Farm Pond #2	A	USFWS	Agriculture	MSM	-	8
9	Gin Road	P	Private	Agriculture	Natural	-	8
10	Gumbo	P	Private	Aquaculture	Natural	6	8
11	Hallum Cemetery Road	P	Private	Agriculture	Natural	-	8
12	Jackson County Hwy 224	P	Private	Aquaculture	Natural	-	8
13	Oldham Duck Club	P	Private	Aquaculture	Natural	-	8
14	Black Swamp WMA Wiville East	P	AGFC	Agriculture	MSM	6	8
15	Black Swamp WMA Wiville West	A	AGFC	Agriculture	WRP	6	8
16	Dagmar WMA Conway George C	A	AGFC	Agriculture	MSM	7	8
17	Shirey Bay Rainey Brake WMA North	A	AGFC	Agriculture	MSM	4	8
18	Shirey Bay Rainey Brake WMA South	A	AGFC	Agriculture	MSM	4	8

events into 4 collection periods: late spring (19 May–20 June), early summer (21 June–13 July), mid-summer (18 July–12 August), and late summer (15 August–18 September).

We captured bees by placing 10 pan-trap stations ~20 m apart throughout managed emergent wetlands along a permanent transect following an opportunistic path avoiding semi-permanent open water. Pan-trap station platforms held three 266-mL plastic Solo™ cups (Lake Forest, IL) that were painted either fluorescent blue, fluorescent yellow, or white (Droege et al. 2009, Kirk 1984, Leong and Thorp 1999). Each pan-trap station platform held all 3 colored cups. We filled these cups ¾ full with soapy water (Dawn Ultra Dish Soap, Cincinnati, OH). We adjusted pan-trap station platforms to the average vegetation height at every collection point. We placed traps out at all sites during 0700–0900 hrs and collected them the same day during 1800–2000 hrs. We strained pan-trapped bees using a 180-µm sieve in the field and transferred them to Whirl-Pak® bags (Fort Atkinson, WI) with 70% ethanol. We used 1 blue-vane trap (1.89-L jar) per field site suspended from a shepherd's hook pole, with the bottom of the trap ~1 m above the ground, and filled with 475 ml of soapy water (Kimoto et al. 2012, Stephen and Rao 2005). We placed and collected blue-vane traps on the same schedule as the pan traps and similarly extracted samples. We used sweep netting to sample for bees that were not attracted to either pan or blue-vane traps. We conducted 5 random-direction transects of 50 sweeps apiece within each wetland per collection period to capture bees. Sweeps were conducted during 1100–1345 hrs (Stephen and Rao 2007) in 2015 and during 0900–1000 hrs (Roulston et al. 2007) in 2016. We altered the sweep-net collection period in 2016 because we observed bees were more active during 0730–1000 hrs in 2015. All sweep net samples were placed in 3.8-L Ziploc® bags (Racine, WI) and stored in a freezer until processed.

We acknowledge it may appear counterproductive to use lethal collecting methods for a project studying pollinator conservation, but Gezon et al. (2015) found that sampling the same sites every other week during the flight season did not affect bee catch rate or diversity over 3 consecutive years. Solitary bee species can be seasonal in their emergences, thus limiting oversampling of particular species as turnover between collection efforts occur. Solitary bees also provision brood cells as they forage and offspring do not require ongoing parental care (Gezon et al. 2015).

We identified bees to species when possible or to genus (Stephenson et al. 2018) using identification guides such as Michener's (2007) *The Bees of the World* or DiscoverLife.org (DiscoverLife 2017). We confirmed identifications with taxonomic bee experts: H.W. Ikerd (Insect Ecologist, USDA-Agricultural Research Services, Logan, UT), T.L. Griswold (Research Entomologist, USDA-Agricultural Research Services, Logan, UT), M.S. Arduser (Insect Ecologist, Missouri Department of Conservation – retired, Webster Groves, MO), K.A. Parys (Research Entomologist, USDA-Agricultural Research Services, Stoneville, MS), and S. Droege (Wildlife Biologist, US Geological Survey, Laurel, MD). We deposited a subset of voucher specimens at the University of Arkansas Arthropod Museum (~250), Fayetteville, AR; the US National Pollinating Insect Collection (~50), Logan, UT; and M.S. Arduser's reference collection (~17,560), Webster Groves, MO.

Plant surveys

To assess the relative abundance of flowers available to bees at each site, we assigned a floral score (1–3) to each wetland site during each sampling period: 1 (low) if <30% of the site was covered in desirable flowering plants, 2 (medium) if 30–60% of the unit was covered in desirable flowering plants, and 3 (high) if 60–100% of the unit was covered in desirable flowering plants. Desirable plants were any flowering plant that we observed being visited by a bee during sampling. Representative specimens of all desirable flowering plants were mounted and identified to species (Gentry et al. 2013) and catalogued in the University of Arkansas Herbarium, Fayetteville, AR. Plant identifications were confirmed by K.L. Willard (University of Arkansas Herbarium, Fayetteville, AR).

Analysis

We estimated bee species richness for active and passive management types using program SPECRICH (Burnham and Overton 1979). This program computes "species richness" or the total number of species from empirical species abundance distribution data that is based on the idea that capture probabilities among species exhibit heterogeneity across time and sites (Burnham and Overton 1979, Chao and Lee 1992, Chao et al. 1992, Lee and Chao 1994). This program assumes that not all species have an equal probability of detection and so a capture–recapture context allows detectability of a given species to be explicitly incorporated in the estimates of richness. This assumption is important and needs to be addressed in the estimation procedure as some approaches, e.g., rarefaction (Sanders 1968, Simberloff 1972) are overly restrictive and make no such assumptions (Cam et al. 2002). Another assumption that we had to account for was that we knew that some bee species would be present at certain sampling times and not at others because we sampled across most of the growing season (see below). Hence, we assumed that the bee community was not closed across the entire growing season. Additionally, we used removal sampling, which possibly impacted the probability of capture for rarer species if those capture rates were high, thus possibly violating the model assumption of population closure. Given these constraints, we estimated species richness and 95% confidence intervals (CI) for each of the 4 collection periods in both years rather than across an entire sampling period. Note that we considered bees only identified to genus as a single species.

We compared bee species richness between management types by examining the overlap in 95% confidence intervals for: (1) the pattern of species richness by management type across the consecutive 4 collection periods within each year, and (2) comparing species richness between management types by collection period and by year. Next, we were interested whether there was a pattern in the counts of bee species detected in each management type within and among collection periods. To make this examination, we tallied the number of bee species detected for each management type in each collection period and among all possible combination of adjacent periods. For example, we tallied the number of bees in active management stands that were detected in only a single collection period as compared to the number of bee species in active management stands that were detected in 2 adjacent periods and so on until we tallied the number of bees in active management stands that were detected in all 4 capture periods. We also tallied the number of bees that were detected in either 1, 2, 3, or 4 periods, in any combination. We compared these raw species counts between management types to determine whether the management practices resulted in different patterns. These counts also allow an assessment of whether the bee communities were closed (see above).

To assess the bee community structure for each management type, we calculated Shannon diversity index (Shannon 1948), which is a measure taking into account both species richness and relative abundance across species, and evenness (Elliott 1990), which is a measure of how close in numbers each species is among sites. We tested for community structure differences by management type using a one-way ANOVA. The reader should be aware that these 2 metrics do not formally account for detectability of bee species and for this reason, the strength of inference from these metrics is uncertain (Williams et al. 2002).

We implemented non-metric multidimensional scaling (NMDS) to visualize potential bee community composition differences among actively and passively managed sites, incorporating the Bray–Curtis dissimilarity index within the vegan package (Oksanen et al. 2019) in R version 3.6.1 (R Core Team 2013). Stress values from models with 2 to 6 dimensions were compared to determine the most appropriate summary of the relationship between management actions and community composition.

Finally, to relate our species richness estimates to other studies of native bees in wetlands and other basic habitat types in North America, we conducted a literature review. We thought this exercise was important because wetland bee communities are poorly studied.

Results

Natural history

During 2015, we captured 3740 individual bees comprising 23 genera and 64 species across 11 sites (see Supplementary Table A, available online along with all other Supplemental Tables at https://www.eaglehill.us/SENAonline/suppl-files/s19-3-S2599-Stephenson-s1, and for Bioone subscribers, at https://dx.doi.org/10.1656/S2599.s1); 18 (28%) species were represented by only a single capture ("singletons"). The 5 most commonly detected species in both management types were similar: *Augochlorella aurata* (Smith) (28%), *Ptilothrix bombiformis* (Cresson) (Hibiscus Bee; 11%), *Melissodes* (*Melissodes*) *communis* (Cresson) (Common Long-horned Bee; 8%), *Florilegus* (*Florilegus*) *condignus* (Cresson) (8%), and *Lasioglossum* (*Dialictus*) *creberrimum* (Smith) (7%) in actively managed stands, and *A. aurata* (43%), *Lasioglossum* (*Hemihalictus*) *nelumbonis* (Robertson) (12%), *P. bombiformis* (7%), *Lasioglossum* (*Dialictus*) *hartii* (Robertson) (7%), and *L. creberrimum* (4%) in passively managed stands. The number of individuals captured per species varied from 1 to 292 with a median count of 4 across actively managed stands, and from 1 to 1081 with a median count of 6 in passively managed stands. The interquartile distance of abundance for active stands between the first (n = 1.5) and third quartiles (n = 29.5) was 28 individuals, while in the passive stands the interquartile distance between the first (n = 2) and third quartiles (n = 47) was 45 individuals. In actively managed stands, 36 of 49 species had 20 or fewer individuals captured, while in passively managed stands, 37 of 51 species had 20 or fewer individuals captured. Across both stand types, the average number of individuals captured per species was 58 (SE = 22.78). The large difference between the median and mean resulted from the large numbers of *A. aurata* captured in each management type.

During 2016, we captured 14,120 individual bees comprising 29 genera and 73 species across 17 sites (see Supplementary Table B); 14 (18%) were singletons. The most commonly detected genera and species in both managed wetland types overlapped: *A. aurata* (35%), *P. bombiformis* (11%) *Melissodes* (*Melissodes*) *comptoides* (Robertson) (Brown-winged Long-horned Bee; 7%), *L. creberrimum* (6%), and *Melissodes* (*Melissodes*) *bimaculata* (Lepeletier) (Two-spotted Longhorn; 5%) in actively managed stands, and *A. aurata* (55%), *P. bombiformis* (7%), *L. creberrimum* (7%), *L. nelumbonis* (5%), and *L. hartii* (4%) in passively managed stands. The number of individuals captured per species varied from 1 to 1478 with a median count of 6.5 across actively managed stands and from 1 to 5484 with a median count of 8 across passively managed stands. The interquartile distance of abundance for actively managed stands between the first (n = 3) and third quartiles (n = 46) was 43 individuals, while in the passively managed stands the interquartile distance between the first (n = 2) and third quartiles (n = 61) was 59 individuals. In actively managed stands, 42 of 60 species had 20 or fewer individuals captured, whereas in passively managed stands, 43 of 64 species had 20 or fewer individuals captured. The average number of individuals captured in both active and passive management types was some 3 times higher in 2016 (mean = 193, SE = 97.4) than in 2015. Note that the large standard error was a result of the large numbers of *A. aurata* captured (n = 6962), almost 6 times more individuals than the next most abundant species captured (*P. bombiformis*; n = 1192). Although the average number of individuals captured increased in 2016 compared to 2015, the number of species with 20 or fewer captures per species increased in 2016. Thus, the increase in average number of individuals captured in 2016 was a result of the abundant species being captured more often. Increased sampling effort as well as the changes in sampling protocol in 2016 were likely the main factors contributing to the increase in total captures in 2016 versus 2015.

Across both 2015 and 2016, we captured 17,860 individual bees, representing 5 families, 83 species, and 31 genera; 17 species (20%) were singletons. Of the 49 species collected in actively managed stands across both years, 13 were not captured at passively managed stands, whereas 15 of 51 species we captured in passively managed stands across both years were not captured at actively managed stands (see Supplementary Tables A, B). *Augochlorella aurata* accounted for 47% (8335 individuals) of the total number of bees captured during both years. *Apis*

(*Apis*) *mellifera* (L.) (Western Honey Bee) was detected at all sites throughout the study but was poorly represented (n = 244) in our collections. While estimating floral abundance, we observed Honey Bees visiting wetland plants in large numbers (10–30 per m^2) during peak bloom.

Plant surveys

Hibiscus lasiocarpos Cav. (Rosemallow), Floating Primrose-willow, *Persicaria* spp. (smartweed), and *Sesbania herbacea* (Mill.) McVaugh (Bigpod Sesbania) were found in both stand types. *Coreopsis tinctoria* Nutt. (Golden Tickseed), *Croton capitatus* Michx. (Woolly Croton or Hogwort), annual smartweed spp., and grasses (e.g., sprangletop spp., barnyardgrass spp.) were more frequently found in actively managed stands, whereas *Asclepias perennis* Walter (Aquatic Milkweed), *Cephalanthus occidentalis* L. (Common Buttonbush), *Echinodorus cordifolius* (L.) Griseb. (Creeping Burhead), *Heliotropium indicum* L. (Indian Heliotrope), Oneflower False Fiddleleaf, *Nelumbo lutea* Willd. (American Lotus), *Sagittaria brevirostra* Mack. & Bush (Shortbeak Arrowhead), perennial smartweed spp., and *Salix nigra* Marshall (Black Willow) were more frequently found in passively managed stands. Note that no woody species occurred in actively managed stands while 2 woody plants (Common Buttonbush and Black Willow) were found in passively managed stands. Average floral score remained constant at ~1.5 (~35% cover) in actively managed stands throughout the sampling period in 2015 and 2016. Actively managed stands did not have many flowering plants between mid-June and early July until smartweed spp. began to bloom. Average floral score steadily increased from ~1.5 to 2.5 (~35–50% cover) in passively managed stands over the collection period in both 2015 and 2016. Overall, passively managed stands provided floral resources continuously throughout the growing season. Reduced soil disturbance sustained floral availability in passively managed stands providing a longer bloom period for hydrophytic plants such as Oneflower False Fiddleleaf and Floating Primrose-willow.

Species comparisons between management stands

Examining the patterns of bee species richness among collection periods by management type by year revealed only a single difference (Fig. 2): between periods 3 and 4 in 2016, the species richness of both management types declined significantly. The apparent coincident increase in species richness for both management types between collection periods 1 through 3 was not significant as the 95% CIs overlapped. Comparing species richness by management type between years, we found 3 differences: in both collection periods 1 and 2, passive species richness was greater in 2016 than 2015, while only in collection period 2 was the species richness in active sites greater in 2016 than in 2015. Note that the only point estimate that was lower in 2016 than in 2015 was for passive species richness in collection period 4. Most importantly, we found no evidence of bee species richness being different between management types by collection period in either 2015 or 2016.

When we compared the tallies of bee species captured in various collection periods by year and management type, few patterns were evident (Table 2). Based on these raw counts, there were only 4 apparent differences: (1) twice as many species were captured in period 4 in 2015 as in 2016; (2) twice as many species were captured in both periods 2 and 3 in 2016 as in 2015; (3) triple the number of species were captured in periods 1, 2, and 3 in 2016 as in 2015; and (4) ~1.5 times as many species were captured across all 4 periods in 2016 as in 2015. The only noticeable difference between management types in the number of species captured was in 2015 during period 3 when 4 times as many species were captured in actively managed stands as in passively managed stands. Overall, in 2015, more species were captured in 1 or 2 collection periods (any combination of periods, ~30 species) compared to those captured in 3 or 4 collection periods (any combination of periods, ~20 species); no such differences were present in 2016.

Community metrics

In 2015, we estimated the average Shannon index was 2.38 (95% CI = 2.133–2.633) in the actively management stands and 2.08 (95% CI = 1.466–2.693) in passively management stands. We found no difference in diversity between management types ($F_{1,10}$ = 0.926, P = 0.31). Species evenness averaged 0.786 (95% CI = 0.710–0.863) for the active management type and 0.648 (95% CI = 0.521–0.775) for the passive management type. Evenness was not significantly different ($F_{1,10}$ = 3.614, P = 0.09) between management types.

In 2016, the average Shannon index was 2.32 (95% CI = 2.065–2.564) in actively managed stands and 1.95 (95% CI = 1.583–2.313) in passively managed stands. There was no difference in diversity between stand types ($F_{1,16}$ = 2.19, P = 0.16). Species evenness averaged 0.684 (95% CI = 0.601–0.768) for actively

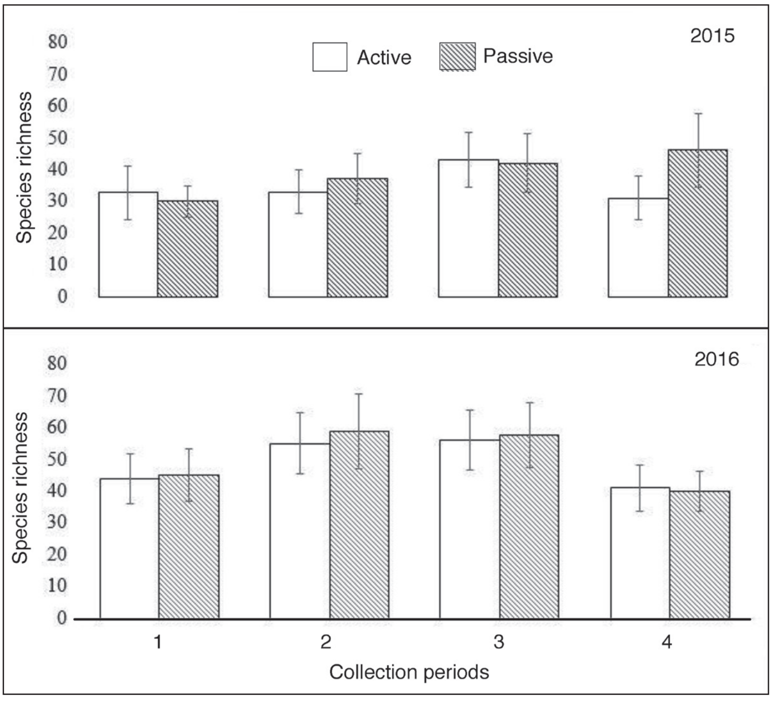

Figure. 2. Native bee species richness estimates (±95% CI) for palustrine emergent wetlands managed under either an active or passive treatment prescription in the lower Mississippi Alluvial Valley of Eastern Arkansas in 2015 and 2016.

managed stands and 0.566 (95% CI = 0.462–0.67) for passively managed stands. We determined that stand type evenness was not significantly different ($F_{1,16}$ = 2.64, $P = 0.12$).

NMDS indicated overlap in bee community composition between actively and passively managed sites (Fig. 3) across years and all collection periods. The overall stress of the two-dimensional NMDS was 0.226, indicating poor confidence in the ordination distances.

Discussion

Wetland management of palustrine emergent wetlands in the LMAV resulted in apparent differences in the plant communities between active and passive management prescriptions—the most important differences being that actively managed stands had no woody plants present and had lower floral scores throughout the growing season compared to passively managed stands. We think that moisture loss and disking were contributing factors to floral score differences between actively and passively managed stands across the collection periods. Observing these differences in the floral communities before we started our study, we originally anticipated that the native bee species richness and community dynamics would have been different between management types. However, after extensive

Table 2. Number of species captured in each of the 4 collection periods and in various combinations of periods by stand type and year in the lower Mississippi Alluvial Valley of Arkansas. For example, 4 species of bees were captured in collection periods 2, 3, and 4 in passive stands in 2015. The bottom 2 rows represent the number of bee species captured in either 1 or 2 collection periods (any combination) or in 3 or 4 collection periods (any combination). Collection periods: 1 (19 May–20 June), 2 (21 June–3 July), 3 (18 July–12 August), and 4 (15 August–18 September).

Collection period(s)	2015		2016	
captured in	Active	Passive	Active	Passive
1	4	6	5	8
2	3	4	4	4
3	11	3	3	4
4	4	6	2	3
1_2	2	1	2	0
1_3	1	1	1	2
1_4	1	2	1	2
2_3	2	3	4	6
2_4	1	0	1	3
3_4	2	4	4	3
1_2_3	2	2	6	5
1_2_4	1	1	1	0
1_3_4	0	0	1	0
2_3_4	3	4	5	5
1_2_3_4	14	14	19	20
Total number of collection periods captured in				
1 or 2	31	30	27	35
3 or 4	20	21	32	30

sampling of the bee communities across many sites and over 2 years, we found no major differences in either bee species richness or community dynamics between wetlands managed with either the active or the passive management treatments.

This project represents the first study to document emergent wetland bee communities in the LMAV. In both years, the sweat bee *Augochlorella aurata* was the most commonly captured bee in both management treatments. More than 40% of the bee species captured in this study were reported as new Arkansas state records in Stephenson et al. (2018) and provide evidence there are more bees in these systems than previously realized. This study also emphasizes the importance of emergent wetlands to specialist and generalist bee species. Seven species collected (*F. condignus*, *Hylaeus* (*Prosopis*) *nelumbonis* (Robertson) [Lotus Masked Bee], *Hylaeus* (*Prosopis*) *ornatus* (Mitchell) [Ornate Masked Bee], *L. hartii*, *L. nelumbonis*, *P. bombiformis*, and *Svastra* (*Epimelissodes*) *atripes* (Cresson) are wetland-dependent species, and over half of the remaining species collected are known to be common in bottomland areas (M.S. Arduser, pers. comm.). While there were some bee species unique to a stand type for reasons not determined in this study, most species were detected in both stand types. Across palustrine emergent wetlands sampled in the LMAV, we collected 83 species and morphospecies. Putting our bee species richness into context, we conducted a literature review of bee species richness across a variety of habitat types, focusing on wetlands. We were surprised to locate only a few studies on bee species richness in wetlands (Table 3). In particular, palustrine emergent wetlands are poorly studied in North America. The number of bee species we documented was comparable to the number of bee species documented in emergent wetlands in Florida (Pascarella et al. 1999) and Nebraska (Park et al. 2017), and to the number of bee species documented in palustrine forested wetlands in Louisiana (Bartholomew and Prowell 2006) (Table 3). Note that had

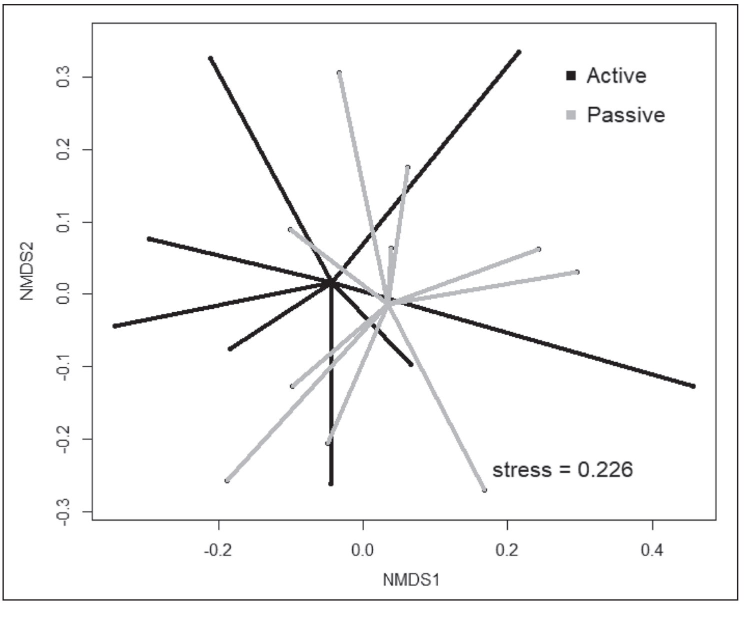

Figure 3. Two-dimensional, non-metric multidimensional scaling (NMDS) ordination plot indicating overlap in bee community composition when separated by treatment (active and passive) from both years and all collection periods. Each line correlates to a specific study site.

we sampled during the early spring (March/April) when species of Andrenidae are most active (DiscoverLife 2017; S. Droege, pers. comm.), we anticipate that our species richness values would have been higher. Compared to other habitats sampled for bee communities in North America, emergent wetland bee communities are not as species rich (Table 3), but emergent wetlands do harbor bee species unique to these systems (Stephenson et al. 2018).

We collected fewer than 20 individuals over the entire study for the majority of the species we encountered, which conforms to models of low abundance of native bees in this region (Koh et al. 2016). Although under-sampling can be a problem when examining arthropod biodiversity, we believe the bee species that were collected in this study area represent those species that have either survived a variety of perturbations including land-use changes, pesticide use, and Honey Bee competition or are in the process of recolonizing these emergent wetland habitats. Land conversion not only decreases the amount of usable habitat for bees, but also inhibits foraging visits from other isolated patches because of known flight

Table 3. Literature review of North American bee community species richness, habitat description, and locations.

No. specimens	No. species	No. genera	Habitat type	Location	Reference
17,860	83	31	Emergent wetland	Eastern Arkansas	This study
-	104	34	Emergent wetland	Everglades National Park, FL	Pascarella et al. 1999
86,500+	77	47	Emergent wetland/ upland edge	South-central Nebraska	Park et al. 2017
1225	81	-	Wet flatwood*	SE Louisiana	Bartholomew and Prowell 2006
494	36	9	Cranberry and natural bogs	SE Massachusetts	MacKenzie and Averill 1995
7095	119	26	Riparian forest	Georgia Piedmont	Hanula and Horn 2011
12,637	166	30	Highbush blueberry	SW Michigan	Tuell et al. 2009
584	57	22	Sand/dune/beach	Assateague Island National Seashore, MD	Orr 2010
6138	118	36	Black Belt Prairie	Northern Mississippi	Smith et al. 2012
10,437	150	27	Old field	Southern Ontario	Grixti and Packer 2006
2551	130	-	Pine–oak heath	New Jersey Pine Barrens	Winfree et al. 2007
3407	165	41	Longleaf Pine savanna	Louisiana and Mississippi	Bartholomew et al. 2006
6542	144	26	Upland hardwood forest	Eastern New York	Giles and Ascher 2006

*Abita Creek site only.

distance limitations of solitary bees (MacKenzie and Averill 1995). Pesticides are used in surrounding agricultural fields to control pest insects harmful to desirable crops. Aerial applications of these pesticides are common in the LMAV creating the chance for drift, accidental spray, or deliberate applications to sensitive invertebrate communities (Tome et al. 1991). Palustrine emergent wetlands in the LMAV often do not have adequate buffers or protection to mitigate the use of insecticides near their edges (Park et al. 2015). Honey Bees, managed and feral, also compete with native bees for nectar and pollen in these emergent wetlands (Cane and Tepedino 2017). Although Honey Bees are not considered a direct threat to the survival of native bees, they have been known to exploit patches of resources until moving to the next location (Aslan et al. 2016).

We found that the active and passive management practices promoted different plant communities through natural successional development, but overall still harbored components of one another as shown in our NMDS output of bee community overlap (Fig. 3). Wetland managers often view particular wetland-associated plants as undesirable (Strader and Stinson 2005, Nelms 2007), but we believe that some plants mentioned in our study should be recognized as important because we observed consistent use of those plants by native bees (e.g., Floating Primrose-willow, Golden Tickseed, Oneflower False Fiddleleaf). We offer that managing these less desirable plants at <20% coverage can be done without adverse consequences for waterfowl (Strader and Stinson 2005). Once plants that are considered "undesirable" for waterfowl exceed coverage of ~20%, then active management is required to set succession back (Nelms 2007). We also found that mid- to late summer could be the most limited time of the year for flowering plants in actively managed stands in part because of the lack of available standing water and disturbance events (disking/planting). Managers should take into account that some desirable plant species for native bees (e.g., Rosemallow, Aquatic Milkweed, American Lotus) require multiple years to mature and produce flowers, thus requiring some portions of managed emergent wetlands to be undisturbed for multiple seasons for successful establishment. Additionally, wetland managers should realize that other characteristics of emergent wetlands are valuable to bee communities. For example, standing water and moist soil are important to the construction of nesting sites and serve a crucial role in the survival of specialized species (Michener 2007). Future work should investigate the difference between bee communities in managed wetlands that undergo various levels of soil disturbance, not only from the perspective of the resultant plant communities, but also from the perspective of nesting habitat availability in the impoundment and along the levees.

Managing palustrine emergent wetlands under either active or passive management prescriptions attract many native bees. Farm bill programs such as the WRP have the capacity to create wetland habitats in a mosaic of agriculture/wetland interfaces (Otto et al. 2017) that native bees find attractive. These restored emergent wetlands serve as a refuge for sensitive invertebrate communities by producing habitats with suitable host plants (Marlin and LaBerge 2001) for a wide variety of native bees, including potentially rare species (Stephenson et al. 2018).

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