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Influence of Created Wetland Size on Herpetofaunal Diversity and Richness in Northeastern Texas, USA

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Abstract.—**Habitat degradation is a main contributor to biodiversity loss, and wetland degradation in North America has been pronounced since colonial settlement. Conservation efforts have created wetlands, but creation and maintenance costs can surge with increasing wetland size. To examine herpetofauna responses to the size of created wetlands, we tested the effects of wetland size on herpetofaunal species richness, diversity, and captures per trap night at two created wetlands in northeastern Texas, USA. Study sites exhibited similar ground cover, hydrology, locality, proximity to flowing water, and age but one wetland was 22 times larger in size than the other. Across 3,866 trap nights, we captured 2,745 individual herpetofauna comprising 29 species. Amphibians, squamates, and testudines had the greatest to least captures per trap night at both wetlands. Species richness at the two wetlands never differed between amphibians, squamates, testudines, or across all herpetofauna. Shannon Diversity estimates for amphibians and across all herpetofauna were greater at the larger wetland, but squamate diversity was higher at the smaller wetland. We encourage future conservation efforts to consider opportunities for creating and protecting smaller wetlands, as small wetlands can be effective habitats that support similarly rich communities of herpetofauna.**

Key Words.—amphibian; assemblage; biodiversity; community; habitat loss; reptile; Shannon Diversity Index

Introduction

Biodiversity loss is one of the most pertinent concerns of wildlife conservation agencies, as current extinction rates exceed historic rates (Ceballos et al. 2015). Although many direct and indirect causes exist for the worldwide extinction rate, habitat loss is one of the largest contributors (Wood et al. 2013). Many habitat types have experienced degradation throughout the 20th Century, but declines in wetland habitats have been precipitous worldwide due to wetland development or conversion into agricultural lands or deep-water areas (Hefner and Brown 1984; Davidson 2014).

Although recent wetland loss is low in North America (Davidson 2014), the current area of wetlands within the U.S. is far less than in pre-colonial times. During colonial settlement, the contiguous U.S. contained 89.4 million ha of wetland (Dahl 1990); however, at least 53% have been drained, dredged, filled, leveled, or flooded, leaving just 42.7 million ha of wetland remaining (Dahl 2000). Such declines in wetlands are important for both conservation of biodiversity and society at large, as wetlands are vital contributors to many economic, recreational, and ecological processes (Bergstrom et al. 1990; Woodward and Wui 2001; Keddy 2010). A diverse assemblage of flora and fauna are obligately dependent on wetlands during their natural history, including mammals (McDonald and Fuller 2005; Wattles 2015), avifauna (Haukos and Smith 1994; Webb et al. 2010), and herpetofauna (Gibbons 2003; Russell et al. 2005).

Herpetofauna have specifically suffered extremely high extinction rates within the last century, perhaps because of acute dependence on particular habitat types (including wetlands), frequent aquatic dependence, and permeable skin (Collins and Storfer 2003; Alroy 2015). Over 50% of all endangered species within the Atlantic Coast Flatwoods, Eastern

Gulf Coast Flatwoods, and northern California mountains and valleys use wetlands (Flather et al. 1998). For these reasons, remaining wetlands are crucial for biodiversity persistence.

In response to the historic widespread loss of wetlands, government agencies have taken steps to protect wetlands against degradation or destruction. In 1987, the U.S. Environmental Protection Agency refocused regulations that strove to have no net loss of remaining wetlands in the U.S. (The Conservation Foundation 1988). Federal and state policies (e.g., section 404 of the Clean Water Act and Massachusetts General Laws Chapter 131, Section 40) of no net loss of wetlands have since required permits for conducting activities potentially destructing wetlands and ensured that deleterious impacts on wetlands are offset through either creation or restoration of wetlands in other locations by permittees (Turner et al. 2001; Bendor 2009). In addition to ensuring no net loss, other governmental programs have created incentives for landowners to restore and protect wetlands on their properties. For instance, the Natural Resource Conservation Service initiated the Agricultural Conservation Easement Program, which has incentives for landowners and assists them with protecting, restoring, and enhancing wetlands through Wetland Reserve Easements and Wetland Reserve Enhancement Partnerships. Other non-governmental organizations also strive to create and restore wetlands for recreational purposes or to ensure healthy ecosystems (e.g., Ducks Unlimited, The Nature Conservancy).

Like their natural counterparts, created wetlands help support biodiversity (VanRees-Siewert and Dinsmore 1996; Wike et al. 2000; Palis 2007; Brown et al. 2012). Unfortunately, land with the potential to be converted into wetland may be scarce and expensive, especially larger land parcels (Rossiter 1995; Thorsnes and McMillen 1998). Transforming a smaller parcel of land into a created wetland may be more financially feasible, but the size of created wetlands may affect herpetofaunal communities that inhabit created wetlands (Lehtinen and Galatowitsch 2001; Houlahan and Findlay 2003). Therefore, to better understand the effects of the size of created wetlands on herpetofauna, we compared the herpetofaunal community at two created wetlands that were similar in terms of age, ground-cover characteristics, hydrology, proximity to flowing water, and location; however, one wetland was 22 times larger in size than the other.

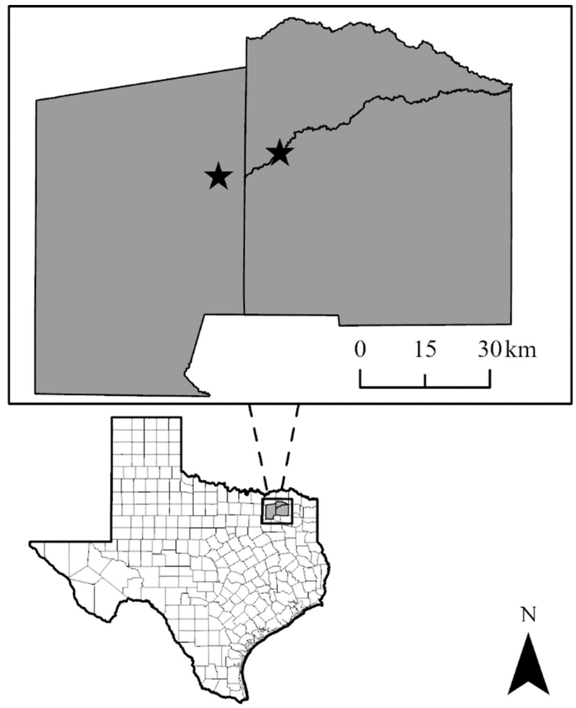

Figure 1. Location of two created wetlands in northeastern Texas, USA: (Left star) The created wetland owned by Texas A&M University-Commerce in Commerce; (Right star) The Cooper Lake Wildlife Management Area in Cooper.

Materials and Methods

Study site.—We sampled herpetofauna at two created wetlands within 14.18 km of each other (Fig. 1). The smaller of the two sites that we surveyed was a created wetland owned by Texas A&M University-Commerce (henceforth SW) in Commerce, Texas, USA. The SW contained 9.09 ha of wetland and was immediately west of Texas State Highway 24. The SW comprised several separated small basins (< 30 m^2) of various shapes that remained wet year-round. The property was previously used as a hayfield in part of a larger university farm until 2007, when it was converted into a created wetland and prairie restoration site. The SW was created using excavators that dug out basins that naturally filled with water. The SW was located south of bottomland mesic forest and pasture periodically browsed by cattle. Several species of wetland thriving herbaceous plants, such as Winged Loosestrife (*Lythrum alatum*), sunflowers (*Helianthus* spp.), buttonweed (*Diodia* spp.), sedges (*Carex* spp.), and rushes (*Juncus* spp.), thrived in between the basins, and small stands of Green Ash (*Fraxinus pennsylvanica*) saplings were dispersed

Table 1. Classification, drainage, and geomorphological attributes of two created wetlands including the Texas A&M University-Commerce wetland (SW) and the Cooper Wildlife Management Area (LW), Texas, USA.

	Attribute	SW	LW
Classification	System	Palustrine	Palustrine
	Class	Emergent wetland	Emergent wetland
	Subclass	Palustrine persistent emergent wetland	Palustrine persistent emergent wetland
	Water regime	Intermittently exposed	Intermittently exposed
	Water chemistry	Fresh–circumneutral	Fresh–circumneutral
	Soil	Kaufman	Kaufman
	Special modifier	Excavated	Excavated and impounded
Drainage	River basin	Sulphur River	Sulphur River
	River sub basin	Sulphur Headwaters	Sulphur Headwaters
	Watershed	Spring Creek-South Sulphur River	Spring Creek-South Sulphur River
	Sub watershed	Rays Creek-South Sulphur River	City of Commerce-South Sulphur River
Geomorphology	Rock unit	Alluvium	Fluviatile terrace deposits
	Sheet	Texarkana	Texarkana
	Epoch	Holocene	Pleistocene

throughout the property. The SW was 520 m from the South Sulphur River, and overflow drainage from this river periodically flooded the SW.

The larger of the two sites was the Cooper Lake Wildlife Management Area (henceforth LW) in Cooper, Texas, USA. In 2005, basins were excavated, and levees were created for impoundment followed by implementation of an active management plan at the LW. The LW contained 200.76 ha of wetland, and much of the property was surrounded by bottomland mesic forest. A periodically flooded lowland savanna forest bisected the northern boundary, and the southwest boundary of the property bordered a cattle pasture. The LW was periodically flooded by overflow drainage from the South Sulphur River, which was 178 m away. The wet basins of the property were dominated by a mixture of herbaceous plants (Giant Ragweed, *Ambrosia trifida*, and sunflowers) and graminoids (sedges, rushes, Giant Cutgrass, *Zizaniopsis miliacea*).

Classification (Cowardin et al. 1979), water regime, water chemistry, soil type, drainage, and geomorphological characteristics varied only slightly for SW and LW (Table 1). Both wetlands are in the Northern Blackland Prairies ecoregion. The SW and LW were separated by only 14.18 km and thus experienced very similar weather. Historical average temperatures throughout the year varied from 2.22° C to 35° C (https://www.ncdc.noaa.gov/cdo-web/). The hottest months spanned from June to mid-September, and the coolest from December to February. Precipitation was highest in spring from mid-March through June. May experienced the most rainfall (annual average = 11.43 cm) and August the least (annual average = 4.57 cm) throughout the year. Snowfall occasionally occurred from mid-December through January, with an average accumulation of 2.79 cm in January. Dew point was generally higher from May through September and peaked in July. Wind speeds increased from November through May, and peaked in March at 18.02 kph. In addition to weather, we found ground cover characteristics to be similar within each of the two created wetlands (Fig. 2). Using the National Land Cover Database 2019 land cover raster file (https://www.mrlc.gov/data/nlcd-2019-land-cover-conus), we found landcover within a 500-m buffer around each wetland to exhibit some fractional differences. Specifically, while the fraction of landcover consisting of development and open water were similar between buffers around the SW and LW, grass and woody vegetation were more common in buffers around the SW and LW, respectively (Fig. 2B).

Data collection.—To survey the herpetofaunal community at both study sites, we used drift fencing combined with funnel traps and pitfall traps. We deployed nine funnel traps and six pitfall traps per study site. Both pitfall and funnel traps were used because a diversity of trapping methods can improve capture rates for some species and result in a more representative assessment of the herpetofaunal community (Jenkins et al. 2003). We constructed funnel traps using 0.32-cm mesh hardware cloth and

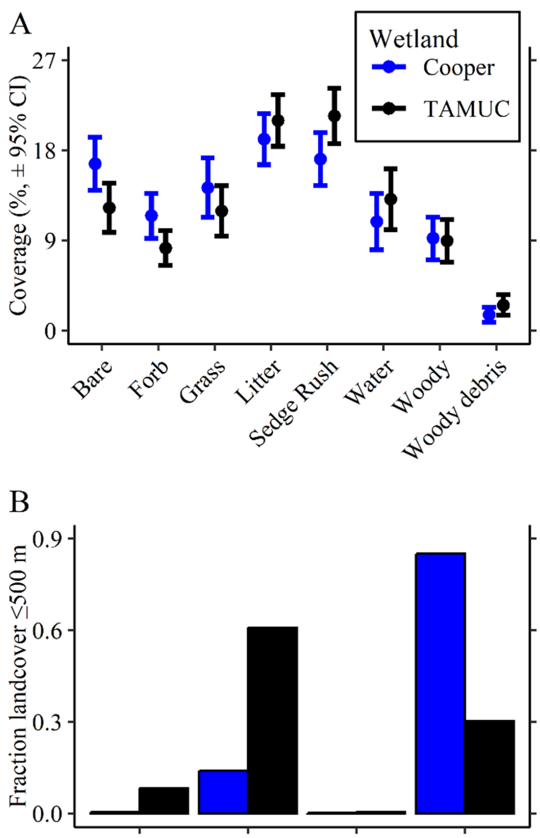

Figure 2. (A) Characteristics of ground coverage (± 95% confidence intervals, CI) inside 713 randomly located 1-m2 plots within the Cooper Wildlife Management Area (Cooper; larger wetland) and Texas A&M University-Commerce wetland (TAMUC; smaller wetland), northeastern Texas, USA. Ground characteristics include bare ground (bare), forbs, grasses, litter, sedge or rush, standing water, woody vegetation (woody), and coarse woody debris. (B) Fraction of the total landcover within 500 m of each wetland consisting of development, grass, open water, or woody vegetation.

built funnel traps to 0.9 × 0.32 × 0.32 m (Length × Width × Height). We used 0.46-m aluminum flashing for drift fencing and 18.9 L buckets for pitfall traps. In both study sites, we constructed three linear drift fence arrays each containing three funnel traps and two pitfall traps separated by 15 m of drift fencing. We placed the two pitfall traps on the ends of each linear array, both of which had 15 m of drift fencing on each side (i.e., in between the pitfall and funnel trap, and at the end of the drift fence). Hence, each drift fence array totaled about 90 m in length. All traps contained shelter boards for shade.

We strove to set traps for seven consecutive days (henceforth trapping bout) followed by 14 d of no trapping (henceforth trapping break). We repeated this trapping schedule from 24 April to 13 October 2017, 19 March to 9 September 2018, and 18 March to 21 April 2019. We checked traps daily in the morning from 0700 to 1000. The temporal span of trapping in the first 2 y was based on apparent herpetofaunal activity, and the third year concluded funding. In a few instances, flooding in the wetlands caused delays and we altered the lengths of trapping bouts and trapping breaks accordingly. Such flooding caused the total trap nights to differ slightly between SW and LW (SW = 2,060 trap nights, LW = 1,806 trap nights). We placed drift fence arrays randomly within the wetland using ArcMap 10.2, contingent upon being placed within land that generally remained dry when water was at typical levels. We sampled the exact same locations with the three linear drift-fence arrays in each study area during each trapping bout. During daily checks of traps, we tallied and released each captured animal 10 m away from the drift fence on the opposing side. We only enumerated captured species for each trap night, and neither marked nor measured captured animals.

Data analysis.—We used several metrics to compare herpetofaunal communities between SW and LW. These included the total number of amphibians, squamates, testudines, and all herpetofauna captures per trap night (i.e., the total number of captures for each group divided by the total number of trap nights), species richness, and estimates of diversity using the Shannon Diversity Index. To estimate richness, we used extrapolated rarefaction estimates (Colwell et al. 2012). We used extrapolated measures because the trap nights slightly differed between our two study sites. Similarly, we used the extrapolated diversity estimator from Chao et al. (2013) to estimate the Shannon Diversity Index. We approximated 95% confidence intervals with 1,000 bootstraps and considered richness and estimates of diversity between sites to differ significantly when confidence intervals did not overlap (Manly 2017).

We estimated extrapolated richness and diversity using the iNEXT package in the R programming language (Hsieh et al. 2016; R Core Team 2022). Additionally, we calculated the relative dominance of each *i*th species (i.e., what fraction of the total number of captures belonged to each species) in each created wetland by:

$$d_i = \frac{n_i}{N}$$

where d_i = the dominance of species i, n_i = the number of individual captures of species i, and N = the total number of captures across all species.

Results

We trapped for a total of 3,866 trap nights across 41 trapping bouts and captured 2,745 herpetofauna (LW = 1,162, SW = 1,583) comprising 29 species (LW = 23, SW = 22; Fig. 3). The average trapping bout was 6.90 ± (standard error) 0.53 d and the average break between trapping was 14.08 ± 0.61 d. We captured 2,441 anurans (LW = 1,071, SW = 1,370), 211 squamates (LW = 83, SW = 128), and 93 testudines (LW = 8, SW = 85). The Southern Leopard Frog (*Lithobates sphenocephalus*) was the most dominant species at SW and LW (Table 4). The next two most dominant species at SW were the Northern Cottonmouth (*Agkistrodon piscivorus*) and Eastern Musk Turtle (*Sternotherus odoratus*), and the next two most dominant species at LW were the Blanchard's Cricket Frog (*Acris blanchardi*) and Eastern Narrow-mouthed Toad (*Gastrophryne carolinensis*).

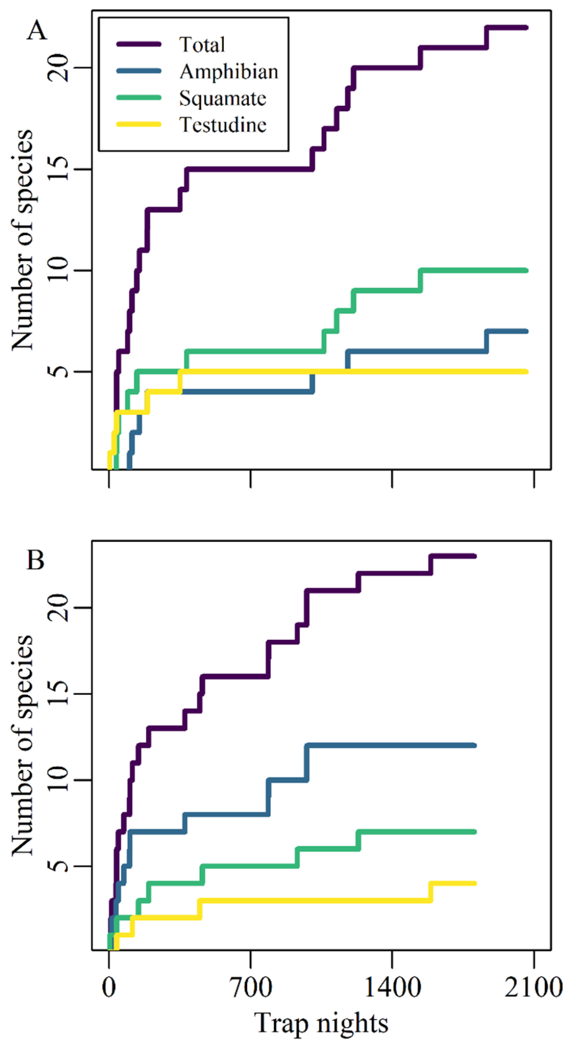

Figure 3. Accumulation of herpetofaunal species captured across trap nights at (A) small and (B) large created wetlands in northeastern Texas, USA. Herpetofauna were captured using pitfall and funnel traps in conjunction with drift fence arrays that sampled from spring 2017 to spring 2019.

Table 2. Captures per trap night for all captured herpetofauna (Cumulative), amphibians, squamates, and testudines within a small and large wetland in northeastern Texas, USA. Captures per trap night are shown for pitfall traps, funnel traps, and both pitfall and funnel traps (Cumulative) that were deployed on drift fence arrays, and across both the small and large wetlands (Cumulative). Drift fence arrays sampled from spring 2017 to spring 2019.

Group	Trap type	Small wetland	Large wetland	Cumulative
Cumulative	Funnel	0.818	0.860	0.837
	Pitfall	0.690	0.319	0.514
	Cumulative	0.768	0.643	0.710
Amphibian	Funnel	0.715	0.791	0.750
	Pitfall	0.586	0.296	0.449
	Cumulative	0.665	0.593	0.631
Squamate	Funnel	0.073	0.065	0.070
	Pitfall	0.045	0.017	0.032
	Cumulative	0.062	0.046	0.055
Testudine	Funnel	0.030	0.004	0.018
	Pitfall	0.059	0.006	0.034
	Cumulative	0.041	0.004	0.024

Across all herpetofauna, captures per trap night were 0.77 and 0.64 at SW and LW, respectively (Table 2). Amphibians, squamates, and testudines had the greatest to least captures per trap night across both trap types at SW and LW. At both wetlands, captures per trap night were greater in funnel traps for all herpetofauna, amphibians, and squamates but higher in pitfall traps for testudines.

Species richness between the two wetlands never differed regarding all herpetofauna, amphibians, testudines, or squamates (Table 3). Shannon Diversity estimates between the two sites did not differ for testudines (Table 3). We documented significantly larger estimates of diversity at the LW for both amphibians and all herpetofauna. Squamates had significantly larger Shannon Diversity estimates at the SW.

Discussion

We found species richness for amphibians, testudines, and squamates, as well as all herpetofauna, to be similar between the two wetlands even though the size of these study sites dramatically differed.

Table 3. Extrapolated richness and Shannon Diversity estimates and associated 95% confidence intervals (CI) for herpetofauna and subclasses of herpetofauna captured at a small (SW) and large (LW) created wetland in northeastern Texas, USA. Herpetofauna were captured using pitfall and funnel traps in conjunction with drift fence arrays that sampled from spring 2017 to spring 2019. Richness and diversity metrics were estimated using the iNEXT package in the R programming language.

Wetland	Species	Estimate	Lower 95% CI	Upper 95% CI
Richness				
SW	Herpetofauna	26.5	22.5	62.89
LW	Herpetofauna	31.16	24.57	65.39
SW	Amphibians	7.5	7.03	15.44
LW	Amphibians	17.99	12.95	49.99
SW	Testudines	5	5	6.58
LW	Testudines	4.22	4.01	8.3
SW	Squamates	10	10	11.87
LW	Squamates	8.98	7.18	28.89
Diversity				
SW	Herpetofauna	0.9	0.89	0.98
LW	Herpetofauna	1.52	1.51	1.6
SW	Amphibians	0.23	0.23	0.28
LW	Amphibians	1.22	1.21	1.29
SW	Testudines	1.28	1.26	1.4
LW	Testudines	1.55	1.32	2.24
SW	Squamates	1.77	1.74	1.94
LW	Squamates	1.22	1.16	1.46

Small wetlands can hence be adequate habitats for similarly rich communities of herpetofauna when compared to larger, yet otherwise similar, wetlands. Unlike species richness, we documented differences in the species diversity of amphibians, squamates, and all herpetofauna at the two wetlands. Specifically, we found higher herpetofaunal diversity at the LW. Larger wetlands may cause greater spatial dispersion of resources that herpetofauna require (Delisle et al. 2019), which could decrease competition between herpetofaunal species (Petren and Case 1998). Such decreases in competition may enable more even occurrence of species, as evidenced by our findings of no differences in species richness but differences in diversity, which suggests unequal species evenness. Indeed, a single species (Southern Leopard Frog) accounted for over 80% of all captures at the SW, while captures were more evenly distributed across species at the LW.

Amphibian diversity was higher at the LW, but squamate diversity, and amphibian and squamate captures per trap night were higher at the SW. This pattern of captures per trap night and diversity may reflect the ecology of our target assemblages. Many of the squamate species we captured consume amphibians as a regular part of their diet (Hibbitts and Hibbitts 2015; Dixon et al. 2020). Additionally, the higher amphibian captures per trap night at SW could indicate greater amphibian abundance (Slade and Blair 2000). Such high prey abundance may decrease interspecific competition for amphibian prey between squamate species (Luiselli 2006), resulting in a more even, diverse, and abundant squamate assemblage at SW (Zipkin et al. 2020). Subsequently, this increased predation pressure may restrict the available niche space of amphibian prey (Chase et al. 2002), which may be reflected in the lesser amphibian diversity found at SW. Another possibility is that other assemblages of amphibian predators (e.g., fish), which we did not sample, were dissimilar between the two wetlands and may have caused a similar response in the amphibian communities (Henrikson 1990; Hecnar and M'Closkey 1997). Ultimately, we encourage more examinations of the roles that amphibian predators have on amphibian communities.

Our study was conducted at only two sites. Thus, a lack of replicates prevented modeling of species richness, diversity, or captures per trap night as a function of wetland size, which would potentially yield more robust statistical evidence and confidence in our conclusions. Our goal of comparing created wetlands that were close in age, spatial proximity, ground-cover characteristics, and hydrology, however, limited our ability to find suitable wetland replicates that also facilitated field sampling. Secondly, we compared only herpetofaunal assemblages, but many other groups besides herpetofauna could be compared. We encourage future research to address these two limitations. Moreover, because past research has inconsistently found relationships between wetland size and various indices of herpetofaunal communities (Richter and Azous 1995; Lehtinen et al. 1999; Lehtinen and Galatowitsch 2001; Houlahan and Findlay 2003; Porej and Hetherington 2005), we generally encourage more examinations of the effects of the size of created wetlands and other wetland attributes on wildlife communities.

We used pitfall and funnel traps in conjunction with drift fencing to survey our two study sites, but other trapping or sampling techniques may be more appropriate for capturing specific types of herpetofauna (Ali et al. 2018). For instance, we did not use aquatic hoop traps, which are better suited for capturing aquatic testudines (Mali et al. 2012), and

Table 4. Relative dominance of herpetofauna captured at a small and large created wetland located in northeastern Texas, USA. See text for method of calculation. Herpetofauna were captured using pitfall and funnel traps in conjunction with drift fence arrays that sampled from spring 2017 to spring 2019. The top three most dominant species for both wetlands are bolded.

Common name	Scientific name	Small wetland	Large wetland
American Bullfrog	*Lithobates catesbeianus*	0.009	0.011
Cajun Chorus Frog	*Pseudacris fouquettei*	0.000	0.004
Common Snapping Turtle	*Chelydra serpentina*	0.002	0.002
Northern Cottonmouth	*Agkistrodon piscivorus*	**0.033**	0.045
Diamond-backed Watersnake	*Nerodia rhombifer*	0.016	0.003
Dekay's Brownsnake	*Storeria dekayi*	0.003	0.001
Eastern Mud Turtle	*Kinosternon subrubrum*	0.018	0.002
Eastern Narrow-mouthed Toad	*Gastrophryne carolinensis*	0.006	**0.050**
Eastern Yellow-bellied Racer	*Coluber constrictor*	0.002	0.000
Common Five-lined Skink	*Plestiodon fasciatus*	0.003	0.001
Graham's Crawfish Snake	*Regina grahamii*	0.002	0.000
Green Frog	*Lithobates clamitans*	0.004	0.035
Little Brown Skink	*Scincella lateralis*	0.009	0.000
Green Treefrog	*Hyla cinerea*	0.000	0.014
Blanchard's Cricket Frog	*Acris blanchardi*	0.015	**0.256**
Ornate Box Turtle	*Terrapene ornata*	0.001	0.000
Red-eared Slider	*Trachemys scripta*	0.014	0.003
Spotted Chorus Frog	*Pseudacris clarkii*	0.000	0.001
Speckled Kingsnake	*Lampropeltis holbrooki*	0.001	0.000
Southern Leopard Frog	*Lithobates sphenocephalus*	**0.830**	**0.526**
Eastern Musk Turtle	*Sternotherus odoratus*	**0.019**	0.001
Small-mouthed Salamander	*Ambystoma texanum*	0.001	0.000
Texas Toad	*Anaxyrus speciosus*	0.000	0.001
Western Ratsnake	*Pantherophis obsoletus*	0.000	0.002
Upland Chorus Frog	*Pseudacris feriarum*	0.000	0.001
Woodhouse's Toad	*Anaxyrus woodhousii*	0.001	0.022
Western Narrow-mouthed Toad	*Gastrophryne olivacea*	0.000	0.001
Western Ribbonsnake	*Thamnophis proximus*	0.010	0.014
Plain-bellied Watersnake	*Nerodia erythrogaster*	0.002	0.006

auditory call surveys could yield a broader sample of breeding amphibians (Pierce and Gutzwiller 2004). Unfortunately, financial restraints precluded our use of other sampling techniques. Therefore, we implemented a trapping strategy that could potentially sample the widest array of herpetofauna (e.g., testudines, squamates, amphibians) rather than multiple strategies that each target specific types of herpetofauna.

Our findings suggest that small wetlands can support similarly rich communities of herpetofauna when compared to larger counterparts. But small wetlands are still more likely to be degraded than larger wetlands (Asselen et al. 2013). Moreover, a recent ruling by the U.S. Supreme Court in Sackett v. EPA (Sackett v. EPA. 2023. 598 U.S. 651, 143 S. Ct. 1322, 215 L. Ed. 2d 579) restricts the way in which the U.S. Environmental Protection Agency can define wetlands and hence regulate wetland degradation. Therefore, we encourage future conservation efforts to consider opportunities for creating smaller wetlands and legislation that continues to protect these important ecosystems.

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Literature Cited

Ali, W., A. Javid, S.M. Bhukhari, A. Hussain, S.M. Hussain, H. Rafique. 2018. Comparison of different trapping techniques used in herpetofaunal monitoring: a review. Punjab University Journal of Zoology 33:57–68.

Alroy, J. 2015. Current extinction rates of reptiles and amphibians. Proceedings of the National Academy of Sciences 112:13003–13008.

Asselen, S.V., P.H. Verburg, J.E. Vermaat, and J.H. Janse. 2013. Drivers of wetland conversion: a global meta-analysis. PLoS ONE 8: e81292. https://doi.org/10.1371/journal.pone.0081292.

Bendor, T. 2009. A dynamic analysis of the wetland mitigation process and its effects on no net loss policy. Landscape and Urban Planning 89:17–27.

Bergstrom, J.C., J.R. Stoll, J.P. Titre, and V.L. Wright. 1990. Economic value of wetlands-based recreation. Ecological Economics 2:129–147. https://doi.org/10.1016/0921-8009(90)90004-E

Brown, D.J., G.M. Street, R.W. Nairn, and M.R. Forstner. 2012. A place to call home: amphibian use of created and restored wetlands. International Journal of Ecology 2012:989872.

Ceballos, G., P.R. Ehrlich, A.D. Barnosky, A. García, R.M. Pringle, and T.M. Palmer. 2015. Accelerated modern human–induced species losses: Entering the sixth mass extinction. Science Advances 1:e1400253. https://doi.org/10.1126/sciadv.1400253.

Chao, A., Y.T. Wang, and L. Jost. 2013. Entropy and the species accumulation curve: a novel entropy estimator via discovery rates of new species. Methods in Ecology and Evolution 4:1091–1100.

Chase, J.M., P.A. Abrams, J.P. Grover, S. Diehl, P. Chesson, R.D. Holt, S.A. Richards, R.M. Nisbet, and T.J. Case. 2002. The interaction between predation and competition: A review and synthesis. Ecology Letters 5:302–315.

Collins, J.P. and A. Storfer. 2003. Global amphibian declines: Sorting the hypotheses. Diversity and Distributions 9:89–98.

Colwell, R.K., A. Chao, N.J. Gotelli, S.Y. Lin, C.X. Mao, R.L. Chazdon, and J.T. Longino. 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. Journal of Plant Ecology 5:3–21.

Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service, Washington, D.C., USA. 142 p.

Dahl, T.E. 1990. Wetland losses in the United States 1780's to 1980's. U.S. Fish and Wildlife Service, Washington, D.C., USA. 11 p.

Dahl, T.E. 2000. Status and trends of wetlands in the conterminous United States 1986 to 1997. U.S. Fish and Wildlife Service, Washington, D.C., USA. 84 p.

Davidson, N.C. 2014. How much wetland has the world lost? Long-term and recent trends in global wetland area. Marine and Freshwater Research 65:934–941.

Delisle, Z.J., D. Ransom, Jr., W.I. Lutterschmidt, and J. Delgado-Acevedo. 2019. Site-specific differences in the spatial ecology of Northern Cottonmouths. Ecosphere 10:e02557. https://doi.org/10.1002/ecs2.2557.

Dixon, J.R., J.E. Werler, and M. Forstner. 2020. Texas Snakes: A Field Guide. University of Texas Press, Austin, Texas, USA.

Flather, C.H., M.S. Knowles, and I.A. Kendall. 1998. Threatened and endangered species geography. BioScience 48:365–376.

Gibbons, J.W. 2003. Terrestrial habitat: A vital component for herpetofauna of isolated wetlands. Wetlands 23:630–635.

Haukos, D.A., and L.M. Smith. 1994. The importance of playa wetlands to biodiversity of the Southern High Plains. Landscape and Urban Planning 28:83–98.

Hecnar, S.J., and R.T. M'Closkey. 1997. The effects of predatory fish on amphibian species richness and distribution. Biological Conservation 79:123–131.

Hefner, J.M., and J.D. Brown. 1984. Wetland trends in the southeastern United States. Wetlands 4:1–11.

Henrikson, B.I. 1990. Predation on amphibian eggs and tadpoles by common predators in acidified lakes. Ecography 13:201–206.

Hibbitts, T.D., and T.J. Hibbitts. 2015. Texas Lizards: A Field Guide. University of Texas Press, Austin, Texas, USA.

Houlahan, J.E., and C.S. Findlay. 2003. The effects of adjacent land use on wetland amphibian species richness and community composition. Canadian Journal of Fisheries and Aquatic Sciences 60:1078–1094.

Hsieh, T.C., K.H. Ma, and A. Chao. 2016. iNEXT: an R package for rarefaction and extrapolation of species diversity (Hill numbers). Methods in Ecology and Evolution 7:1451–1456.

Jenkins, C.L., K. McGarigal, and L.R. Gamble. 2003. Comparative effectiveness of two trapping techniques for surveying the abundance and diversity of reptiles and amphibians along drift fence arrays. Herpetological Review 34:39–42.

Keddy, P.A. 2010. Wetland Ecology: Principles and Conservation. Cambridge University Press, Cambridge, UK.

Lehtinen, R.M., S.M. Galatowitsch, and J.R. Tester. 1999. Consequences of habitat loss and fragmentation for wetland amphibian assemblages. Wetlands 19:1–12.

Lehtinen, R.M., and S.M. Galatowitsch. 2001. Colonization of restored wetlands by amphibians in Minnesota. American Midland Naturalist 145:388–396.

Luiselli, L. 2006. Resource partitioning and interspecific competition in snakes: The search for general geographical and guild patterns. Oikos 114:193–211.

Mali, I., D.J. Brown, M.C. Jones, and M.R. Forstner. 2012. Switching bait as a method to improve freshwater turtle capture and recapture success with hoop net traps. Southeastern Naturalist 11:311–318.

Manly, B.F.J. 2017. The Bootstrap. Pp. 41–79 *In* Randomization, Bootstrap and Monte Carlo Methods in Biology. Manly, B.F.J. (Ed.). Chapman and Hall/CRC, New York, New York, USA.

McDonald, J.E. and T.K. Fuller. 2005. Effects of spring acorn availability on black bear diet, milk composition, and cub survival. Journal of Mammalogy 86:1022–1028.

Palis, J.G. 2007. If you build it, they will come: herpetofaunal colonization of constructed wetlands and adjacent terrestrial habitat in the Cache River drainage of southern Illinois. Transactions of the Illinois State Academy of Science 100:177–189.

Pierce, B.A., and K.J. Gutzwiller. 2004. Auditory sampling of frogs: Detection efficiency in relation to survey duration. Journal of Herpetology 38:495–500.

Petren, K., and T.J. Case. 1998. Habitat structure determines competition intensity and invasion success in gecko lizards. Proceedings of the National Academy of Sciences 95:11739–11744.

Porej, D., and T.E. Hetherington. 2005. Designing wetlands for amphibians: The importance of predatory fish and shallow littoral zones in structuring of amphibian communities. Wetlands Ecology and Management 13:445–455.

R Core Team. 2022. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org.

Richter, K.O., and A.L. Azous. 1995. Amphibian occurrence and wetland characteristics in the Puget Sound Basin. Wetlands 15:305–312.

Rossiter, D.G. 1995. Economic land evaluation: why and how. Soil Use and Management 11:132–140.

Russell, A.P., A.M. Bauer, and M.K. Johnson. 2005. Migration in amphibians and reptiles: an overview of patterns and orientation mechanisms in relation to life history strategies. Pp. 151–203 *In* Migration of Organisms. Elewa, A.M. (Ed.). Springer, Berlin, Germany.

Slade, N.A., and S.M. Blair. 2000. An empirical test of using counts of individuals captured as indices of population size. Journal of Mammalogy 81:1035–1045.

The Conservation Foundation. 1988. Protecting America's wetlands: an action agenda, the final report of the National Wetlands Policy Forum. The Conservation Foundation, Washington, D.C., USA. 79 p.

Thorsnes, P., and D.P. McMillen. 1998. Land value and parcel size: a semiparametric analysis. Journal of Real Estate Finance and Economics 17:233–244.

Turner, R.E., A.M. Redmond, and J.B. Zedler. 2001. Count it by acre or function - mitigation adds up to net loss of wetlands. National Wetlands Newsletter 23:5–6.

VanRees-Siewert, K.L., and J.J. Dinsmore. 1996. Influence of wetland age on bird use of restored wetlands in Iowa. Wetlands 16:577–582.

Wattles, D.W. 2015. The effect of thermoregulation and roads on the movements and habitat selection of moose in Massachusetts. Ph.D. Dissertation, University of Massachusetts, Amherst, Massachusetts, USA. 173 p.

Webb, E.B., L.M. Smith, M.P. Vrtiska, and T.G. Lagrange. 2010. Effects of local and landscape variables on wetland bird habitat use during migration through the Rainwater Basin. Journal of

Wildlife Management 74:109–119.
Wike, L.D., F.D. Martin, H.G. Hanlin, and L.S. Paddock. 2000. Small mammal populations in a restored stream corridor. Ecological Engineering 15:S121–S129.
Wood, A., P. Stedman-Edwards, and J. Mang. 2013. The Root Causes of Biodiversity Loss. Routledge, London, UK.
Woodward, R.T., and Y.S. Wui. 2001. The economic value of wetland services: A meta-analysis. Ecological Economics 37:257–270.
Zipkin, E.F., G.V. DiRenzo, J.M. Ray, S. Rossman, and K.R. Lips. 2020. Tropical snake diversity collapses after widespread amphibian loss. Science 367:814–816.

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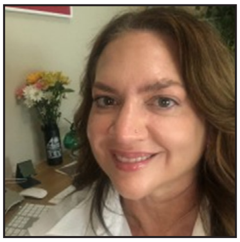

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