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Small mammal communities of restored and natural wetlands in West Virginia

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ABSTRACT

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Wetland restoration is commonly practiced as part of conservation programs or wetland mitigation, which attempts to offset human-created losses of natural wetlands. However, because of the intrinsic and human-derived value of wetlands, it is critical to determine whether these wetlands truly act similarly to natural wetlands. One role of wetlands is to provide habitat for a diverse array of wildlife species. Small mammals are often overlooked taxa in wetland restoration efforts. However, they are essential to the wetland system because they influence vegetation and are prey for higher trophic level wildlife. I discuss considerations of restored wetlands, wildlife responses to these wetlands, and the role of small mammals in wetlands in Chapter 1.

In Chapter 2, I devise a study to determine whether small mammal communities are similar in restored and natural wetlands. I assess apparent abundance, occupancy, relative density, mass, diversity, richness, evenness, and community composition of small mammal communities from 14 restored wetlands and 12 natural wetlands in West Virginia, USA, sampled from June–August of 2020 and 2021. Over 10,060 trap nights, I captured deer mice (*Peromyscus maniculatus*), white-footed mice (*Peromyscus leucopus*), meadow voles (*Microtus pennsylvanicus*), northern short-tailed shrews (*Blarina brevicauda*), meadow jumping mice (*Zapus hudsonius*), and eastern chipmunks (*Tamias striatus*) at both wetland types, and woodland jumping mice (*Napaeozapus insignis*), masked shrews (*Sorex cinereus*), and one southern flying squirrel (*Glaucomys volans*) at exclusively natural wetlands. I found all aspects to be similar between wetland types, apart from apparent abundance of deer mice, which was higher in natural wetlands ($P<0.01$), and total small mammal apparent abundance was again higher in natural wetlands ($P<0.01$). My results suggest that restored wetlands are similar to natural wetlands for small mammal communities in most aspects.

In chapter 3, I determine the features of restored wetlands that most affect small mammal communities. Specifically, I examined the effects of age and environmental variables in 14 restored wetlands spanning the three ecoregions in West Virginia. I determined the apparent abundance of deer mice ($P = 0.01$), white-footed mice (*Peromyscus leucopus*) ($P<0.01$), and meadow voles ($P<0.01$) decreased with wetland age. Furthermore, occupancy probabilities of meadow voles decreased with wetland age ($P = 0.03$). Diversity and richness increased with wetland age ($P<0.01$), but apparent species richness was not affected by wetland age ($P = 0.895$). Of 18 environmental variables, including wetland age, model selection showed apparent abundance of white-footed mice, meadow jumping mice, and total small mammals to be most influenced by wetland size, meadow vole apparent abundance to decline with average tree and shrub canopy cover, and deer mice apparent abundance to decrease with vegetation community similarity. Although both white-footed mice and meadow jumping were affected by wetland size, apparent abundance of white-footed mice decreased with wetland size. In contrast, apparent abundance of meadow jumping mice increased with wetland size. There was model selection uncertainty for the apparent abundance of northern short-tailed shrews and eastern chipmunk, occupancy probability of individual species, diversity, richness, or evenness in created and restored wetlands. Therefore, when designing wetlands, managers should consider how age impacts different aspects of the community at various stages of wetland restoration and should manage accordingly. Moreover, wetland managers should consider the impact of wetland size on different species to better host small mammal communities.

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CHAPTER 1

Considerations of wetland restoration and wildlife: A literature review

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Introduction

The general importance of wetlands

Wetlands are of great importance to the landscape. Deemed “kidneys” of the earth due to their ability to purify water (Mitsch and Gosselink 1986), wetlands provide many ecosystem services. These ecosystem services include provisioning services such as supplying food and water, regulating services such as flood mitigation and water quality, supporting services such as habitat provisioning and nutrient cycling, and cultural services such as educational or recreational use (Millennium Ecosystem Assessment 2005). Moreover, wetlands are responsible for an estimated 40% of global ecosystem services, despite covering only a tiny fraction of the earth’s surface (Zedler and Kercher 2005).

While wetlands have a high contribution to ecosystem services, they are in decline and continuously face threats to their existence (Hu et al. 2017). It is estimated that > 50% of wetlands have been lost globally; however, wetland loss may be as high as 87% since the 1700s (Davidson 2014). The United States (US) has lost > 50% of its wetlands (Dahl and Allord 1996), and West Virginia has potentially lost > 80% of its wetlands (West Virginia Department of Environmental Protection 2021). Although wetland loss has decreased in recent years (Dahl and Allord 1996), wetlands are still in decline. Currently, wetlands face many threats, including but not limited to climate change (Burkett and Kusler 2000; Erwin 2009; Mitsch and Hernandez 2013), invasive species (Houlahan and Findlay 2004), and human activity, such as development (Keddy 1983; Hu et al. 2017).

Brief legal history

While current awareness of wetland function is well known today, wetlands were maligned in early American history; they were regarded as centers for diseases and viewed as barriers to travel and food production (Dahl and Allord 1996). Economic interests combined with these negative assessments of wetlands encouraged mass filling and drainage of wetlands. Early legislation further promoted wetland loss with acts such as The Swamp Lands Acts in 1849–1860, which relinquished federal ownership of swamplands to state governments with the expectation of developing the land (Olsen et al. 2016). The Sugar Act of 1934 promoted sugarcane production in Florida's wetlands, and the Watershed Protection and Flood Prevention Act of 1954, which resulted in wetland drainage and channelization (Erickson et al. 1979), also contributed to wetland decline.

However, other pieces of legislation worked to protect wetlands at similar points in history. For instance, the Migratory Bird Hunting and Conservation Stamp Act (also known as the "Duck Stamp Act"), enacted for wetland restoration, was passed in the same year as the Sugar Act of 1934. The Rivers and Harbors Act of 1899 was a start to wetland protection, but it only protected interstate navigable waterways. By the 1970s, there was a mass shift to a more positive view of wetlands, resulting in the Federal Water Pollution Control Act of 1972. This Act was later changed to the Clean Water Act of 1977, which is the leading piece of legislation in wetland protection, specifically under section 404, which protects some wetlands (EPA 2021). During this period, the United States also took part in the Ramsar Convention and signed an intergovernmental treaty to protect exceptionally biodiverse and unique wetlands. Other wetland protections arose from the Swampbuster provision in the Food Security Act of 1985, the Emergency Wetlands Resources Act of 1986, and the no net loss policy in 1989 by the George

H. W. Bush administration. The no net loss policy emphasized the ecosystem value of a wetland rather than acreage. Adopting this policy meant that if a wetland were to be destroyed or harmfully altered, it must be replaced with another wetland similar in ecosystem value to the original. Between Section 404 of the Clean Water Act and the “no net loss” policy, wetland mitigation quickly became the recommended process by the United States Army Corps of Engineers (USACE) and Environmental Protection Agency (EPA) to prevent wetland loss. Mitigated wetlands have been restored, created, enhanced, or preserved to replace natural wetlands that have been lost due to human development (Federal Register 1995). Many mitigated wetlands are restored or created, and if avoidance is not possible, restoration is the recommended approach, as the necessary hydrology is already present at the site (Federal Register 1995). Restored and created wetlands are the most controversial wetland mitigation strategies, as success may be difficult to gauge (Mitsch et al. 1998; Gutrich and Hitzhusen 2004). In theory, mitigation techniques may seem sufficient to offset losses to natural wetlands; however, criticism of mitigation includes potential global loss of ecosystem function (Moreno-Mateos et al. 2012) and yet unknown efficacy (Levrel et al. 2017).

While mitigation is the lawfully imposed solution to planned wetland loss, voluntary programs to restore wetlands exist. Voluntary programs such as the Agricultural Conservation Easement Program (formerly the Wetland Reserve Program) run through the Natural Resources Conservation Service, and the Partners for Fish and Wildlife Program run through the U.S. Fish and Wildlife Service work with private landowners to restore and conserve wetlands on their property (Benson et al. 2018). Although the Partners for Fish and Wildlife Program is not strictly dedicated to wetlands, their work includes riparian and wetland habitat restoration. The National Fish and Wildlife Foundation partner with governmental agencies to give grants designated for restoring wetlands, such as the Five Star and Urban Waters Restoration Grant Program in collaboration with the EPA. Voluntary restoration work is also done by non-governmental organizations, including Ducks Unlimited, which claims to have provided benefits to >900 species through its wetland restoration efforts (Tori et al. 2002).

Considerations of wetland restoration: in mitigation and voluntary projects

Wetland mitigation ratios are the proportion of mitigation wetland required for every unit area of natural wetland impacted (Bendor 2009). While ratios are not standardized in wetland mitigation and can differ from state to state, Brown and Lant (1999) conducted a nationwide review of 68 mitigation banks. They found most use a 1:1 ratio of natural wetland destroyed to wetland replaced. In West Virginia, wetland replacement ratios are 1:1 for open water wetlands, 2:1 for emergent wetlands, and 3:1 for scrub-shrub or forested wetlands (State of West Virginia 2014). While wetland replacement acreage is higher for some wetland types, the rules do not specify that the replacement wetland must be the same type of wetland lost.

Moreover, Robb (2002) estimated that ratios should be higher than this for wetland types, with 3.5:1 for forested wetland and 7.6:1 for wet meadows. In the neighboring state of Pennsylvania, the law is slightly different; the natural wetland function must be considered and replaced with a 1:1 ratio for specific functions (Commonwealth of Pennsylvania 1991). Although variation in laws exists, King and Price (2004) put forth a 5-step method for determining these ratios. These steps involve 1) considering natural wetland functionality before mitigation, 2) the resulting level of wetland functionality if the mitigation is entirely successful, 3) the length of time before the mitigation is considered successful, 4) the risk mitigation will not succeed, and 5) differences in location between the lost wetland and the mitigated wetland that may affect functionality (King and Price 2004).

Another consideration is the lag time in ecosystem services between when natural wetlands are lost and when new restored or created wetlands for mitigation are functional. One study of eight Ohio mitigated freshwater marshes estimated that the total average economic loss from restoration to the time these wetlands were fully functional was $41,600 USD per hectare (Gutrich and Hitzhusen 2004). However, these costs can vary depending on the system as high-elevation wetlands of the same type result in an average lag-time cost of $67,687 USD per hectare (Gutrich and Hitzhusen 2004). In a model presented by Bendor (2009), poor wetland restorations and delays in mitigation could potentially amount to upwards of $130 million USD, thus providing evidence that lag time in wetland mitigation projects can result in severe economic impacts from loss of ecosystem services. Additionally, there are long-term considerations of mitigation. In a global study of over 600 wetlands, researchers found that within a century, restored wetlands had not acquired the vegetative structure they were expected to, nor had they acquired the same capacity for biogeochemical functioning (such as carbon storage in soils) (Moreno-Mateos et al. 2012). Although these ecological timescales are too large for mitigators to assess, it has been proposed that mitigated wetlands may never recover in terms of soil (Zedler and Callaway 1999). While functional replacement may be achieved in 10 years for low-stress systems, species-rich systems or systems with special water quality needs may take up to 200 years to develop (Zedler and Callaway 1999). However, King and Bohlen (1994) suggested the issue is not restoration science but rather a failure of the agencies to enforce mitigation requirements.

Vegetation at mitigated wetlands also has a time consideration. Balcombe et al. (2005a) found as mitigated wetlands age, they become similar in composition to that of natural wetlands. Vegetation restoration at mitigated wetlands may vary by location. One study assessing mitigation banks in Ohio found native hydrophytes could become the predominant vegetation at mitigated wetlands after 5 years, and vegetation species richness was similar between mitigated and reference wetlands by the tenth year (Spieles et al. 2006). However, a Massachusetts study found those plant communities at compensatory wetlands would not be like natural wetlands for at least 12–15 years, if ever (Brown and Veneman 2001).

The surrounding abiotic environment must be considered, such as differences in hydrologic regimes, nutrient levels, and climate. Hoeltje and Cole (2007) found a loss of wetland function in created wetlands compared to natural wetlands in a similar geomorphic setting because of unnatural hydrologic regimes in created wetlands. Therefore, mitigation project design should consider hydrology to improve wetland functionality (Morgan and Roberts 2003). Nutrient levels could also reduce functionality in mitigated wetlands, as they perform more poorly than natural wetlands at nitrate removal and carbon sequestration (Hossler et al. 2011). In a created riverine wetland, phosphorus retention declined 10 years after creation (Mitsch et al. 2005). In another study, created wetlands ranging in age from 1 to 8 years old were assessed in terms of soil and found to have higher pH, bulk density, and matrix chroma and lower total nitrogen compared to reference wetlands (Bishel-Machung et al. 1996). Climate may also affect the success of wetland restoration; wetlands in warmer climates recovered faster than their colder climate wetland counterparts (Moreno-Mateos et al. 2012).

Along with the abiotic environment, the biotic environment of restored wetlands should also be considered for similarities or differences to natural wetlands. Decomposers are essential in wetland ecosystems because they can influence many aspects of wetland function, such as nutrient cycling (Allison and Vitousek 2004). Gingerich and Anderson (2011) found decomposition rate and percent remaining litter varied among litter types but were similar between mitigated and natural wetlands. Moreover, taxa responsible for litter decomposition, fungi and invertebrates, were comparable in biomass between mitigation and natural wetlands (Gingerich et al. 2015). Although biomass of the two groups was similar between natural and mitigated wetlands overall, there were differences in the biomass of individual groups, such as that of Oligochaetes and collector/gatherers being higher in number in mitigated wetlands than in natural wetlands. Macroinvertebrates are an important group because they are often used to determine water quality (Armitage et al. 1983; Lazorchak et al. 2015). Balcombe et al. (2005b) found no difference in macroinvertebrate abundance, diversity, and productivity between natural and mitigated wetland sites in West Virginia. Similar results were found by Strain et al. (2014), who studied the macroinvertebrate diet of red spotted newts (*Notophthalmus viridescens viridescens*) between created and natural wetlands and found they were similar, suggest the presence of macroinvertebrate prey species was also similar.

Wildlife responses to wetland restoration

Wildlife responses to restored or created wetlands vary by taxa. Birds have been researched in wetland mitigation; Balcombe et al. (2005c) studied avian communities in mitigated and natural wetlands and found them similar in species richness, diversity, and abundance between mitigated and natural wetlands. However, when comparing only waterbirds and waterfowl between mitigated and natural wetlands, these birds had a higher abundance in mitigated wetlands than in natural wetlands (Balcombe et al. 2005c). These results suggest mitigated wetlands are not only succeeding at providing quality habitat but may be exceeding that of natural wetlands for some species. This difference may be apparent with waterbirds and waterfowl because mitigated wetlands in Appalachia tend to contain more open water than natural wetlands (Cole and Brooks 2000). Lewis et al. (2019) examined how Agricultural Conservation Easement Program (ACEP) wetlands, which are restored wetlands, compared to natural wetlands by measuring avian species richness and occupancy probability of four Passerellidae species: song sparrow (*Melospiza melodia*), dark-eyed junco (*Junco hyemalis*), swamp sparrow (*Melospiza georgiana*), and white-throated sparrow (*Zonotrichia albicollis*). The findings show both avian species richness and occupancy probability of these four Passerellidae species were not different between ACEP and natural wetlands, meaning restored wetlands can provide adequate habitat (Lewis et al. 2019). However, the wetland system type may be essential in determining whether mitigated wetlands will succeed for birds. Desrochers et al. (2008) researched the avian community in a salt marsh ecosystem and found that the created salt marshes had lower avian abundance and richness than natural wetlands during the breeding season.

Amphibians have also been researched in restored and created wetlands. Strain et al. (2017a) measured the occupancy of anurans at both human-created wetlands and beaver (*Castor canadensis*)-created natural wetlands in West Virginia to determine if mitigated wetlands were providing similar chorusing habitat. Overall, the results of this study match the growing body of research that mitigated wetlands are functionally like natural wetlands, as the authors found occupancy was not affected (Strain et al. 2017a). Mitigated wetlands provide chorusing habitat, but they also provide breeding habitat for two species of amphibians (Strain et al. 2017b). When Strain et al. (2017b) researched amphibian metamorphs, they found a similar abundance of spring peeper (*Pseudacris crucifer*) and green frog (*Lithobates clamitans*) metamorphs between human-created and beaver-created (natural) wetlands. In green frogs, Strain et al. (2017b) also found survival, growth curves, and mass to be similar between larvae raised in mitigated wetlands and larvae raised in natural wetlands, indicating mitigated wetlands can provide habitat for metamorphs, as well as breeding habitat for these two species of amphibians. Moreover, Balcombe et al. (2005c) found that anuran species richness and abundance were higher in mitigated wetlands than in natural wetlands.

Researchers have also studied reptiles in a created wetland environment. Hartwig and Kiviat (2007) discovered Blanding's turtles (*Emydoidea blandingii*) used created wetlands. However, as these wetlands dried up, they moved to natural wetlands. This finding suggests that created wetlands may not fully support Blanding's turtles' life history as they moved back to natural wetlands. The created wetland was only three years old at the time of research which may have impacted their findings. Snapping turtles (*Chelydra serpentina*) and painted turtles (*Chrysemys picta*) were also compared between restored wetlands and reference wetlands in West Virginia (Gulette 2018). There was no significant difference in the abundance of snapping turtles between the two wetland types; however, painted turtle abundance was higher in restored wetlands (Gulette 2018). These studies illustrate that individual species may respond differently to created or restored wetlands than natural wetlands.

Small mammals

Mammals may be useful to assess the functionality of restored or created wetlands. Taxa such as bats can be excellent bioindicators of wetland use (Kalcounis-Rueppell et al. 2007; Maslonek 2010; Parker et al. 2019). However, nonvolant small mammals may be a useful taxon to compare restored or created and natural wetlands. Generally, small mammals have been used to study landscape ecology, particularly to examine habitat fragmentation (Bayne and Hobson 1998; Pardini 2004; Presley et al. 2019). Not only can these communities be useful indicators because of their sensitivity to changes in habitat (Pearce and Venier 2005; Leis et al. 2008; Myers et al. 2009; Levykh and Panin 2019), but small mammals are also considered a model taxon for research in habitat systems (Barrett and Peles 1999) and landscape ecology (Bowers and Barrett 1999). They are considered model taxa for several reasons. These include having small home range sizes (<0.20 ha) and dispersal distances of <200 m; in fact, they do not typically move >100 m (Diffendorfer et al. 1999). Additionally, they have short generation times, making it easier for them to be studied short-term (Bowers and Barrett 1999).

Although small mammals are largely overlooked (Frey 2018), previous research has compared small to mid-sized mammals in restored and natural wetlands; no significant differences in abundance, richness, or species composition in these two wetland types in Ohio, USA, were discovered (Kurz et al. 2013). All species captured were considered habitat generalists, such as northern short-tailed shrews (*Blarina brevicauda*), meadow voles (*Microtus pennsylvanicus*), and white-footed mice (*Peromyscus leucopus*). These findings were attributed to the "Field of Dreams" hypothesis, which suggests if the conditions are right, wildlife species will move into the area (Palmer et al. 1997). Whitsitt and Tappe (2009) also sampled a small mammal community at a restored wetland to study the influence of water level and duration of water on the population. Their findings show different species had different peaks in relative abundance at other times; marsh rice rats (*Oryzomys palustris*) were associated with higher water levels, most likely because they are semi-aquatic and more efficient at using areas with standing water. Other changes in species composition throughout sampling were more likely due to species-specific interactions (for instance, the hispid cotton rat (*Sigmodon hispidus*) is aggressive towards other small mammals) and the seasons in which they were trapping, rather than water level or duration (Whitsitt and Tappe 2009). In another study, small mammals were trapped at a wetland site to determine whether small mammal community differences could be observed between wetlands that had natural regeneration after disturbance and wetlands that experienced plantings as part of a remediation effort (Wike et al. 2000). Although researchers failed to find differences between the two categories of wetlands, they concluded that this finding may be attributed to a small study area and the likely overlap of the same small mammal communities at their sites (Wike et al. 2000). While research put forth by these studies provides insight of small mammal communities at restored wetlands, these studies were limited in terms of wetland age (all were $\leq$ 7 years) and study sites were few in each study ($\leq$ 6).

Objectives and Hypotheses

The goal of this research was to determine if small mammal communities at restored (n = 14) and natural wetlands (n = 12) are similar, thus supporting the hypothesis that these wetlands can act similarly to natural wetlands for small mammal communities. Because mitigated wetlands are typically restored wetlands, and wetland mitigation assumes that they entirely replace natural wetlands, it is essential to compare the two wetland types. My objectives were to:

1. Determine apparent abundance and occupancy of each species at each site, then compare between restored and natural wetlands.
2. Compare the relative density of *Peromyscus* spp. and the mass of each captured species between wetland types.

3. Compare site community composition, diversity, evenness, and apparent richness between restored and natural wetlands.
4. Analyze effects of wetland age and environmental variables on small mammals 1) apparent abundance, 2) occupancy, 3) diversity, 4) apparent richness, and 5) evenness across a range of restored wetland ages (1–29 years).

I hypothesized that restored wetlands may have higher scores for all aspects of the small mammal community than natural wetlands, as mitigated wetlands have higher vegetation diversity, richness, and evenness (Balcombe et al. 2005a), and small mammals are reliant upon vegetation diversity (Wywialowski 1987). Therefore, I expected to find significant differences in small mammal community metrics between restored and natural wetlands. Again, due to the dependence of small mammals on vegetation diversity (Wywialowski 1987), I hypothesized that small mammal communities at restored wetlands would decrease in all metrics with wetland age because vegetation species richness is higher at younger versus older created wetlands (Stefanik and Mitsch 2012), displaying a temporary response to succession. Moreover, I predicted that small mammal communities would be influenced most by vegetation-related environmental variables, such as vegetation diversity, canopy cover, and dominant vegetation type according to Cowardin (emergent, scrub-shrub, and forested) at restored wetlands. The following null hypotheses were tested:

1. Apparent abundance and occupancy of each species were similar between restored and natural wetlands.
2. Relative density of *Peromyscus* spp. and mass of each captured species were the same between wetland types.

3. Site community composition, diversity, evenness, and apparent species richness were similar between restored and natural wetlands.
4. Wetland age and environmental variables do not affect small mammals 1) apparent abundance, 2) occupancy, 3) diversity, 4) apparent richness, and 5) evenness across a range of restored wetland ages (1–29 years).

Study Areas

The three primary ecoregions of the state were represented: Central Appalachians (23.1%), Western Allegheny Plateau (38.4%), and Ridge and Valley (38.4%). I sampled 26 wetlands in total; 14 were restored/created wetlands (i.e., wetlands were restored, but small areas that historically were not wetlands were likely created [hereafter, restored]), and 12 were natural. All wetlands were classified as palustrine emergent (7 restored, 3 natural), scrub-shrub (6 restored, 4 natural), or forested (1 restored, 5 natural), although most had elements of all three classification types. Wetland size varied by site, from 2 to 28.7 ha (mean ± SE ha = 8.1 ± 1.9 ha) for restored wetlands and 1.5 to 45 ha (mean ± SE ha = 12.6 ± 3.6 ha) for natural wetlands. I excluded high elevation wetlands (>730 m) from sampling, as they are unique from other wetlands in the state because they resemble bogs and fens of a boreal climate and can provide habitat to species that are otherwise uncommon in West Virginia (Francl et al. 2004; Byers et al. 2007). Elevation ranged from 146 to 695 m (mean ± SE m = 429.5 ± 36.3 m). Restored wetland age ranged from 1 to 29 years old at the time of sampling (mean ± SE years = 14.2 ± 2.8 years).

Restored wetland sites

Buckhannon Triangle

Buckhannon Triangle is a 2.8-ha restored wetland in Buckhannon, in Upshur County, WV (4317185 N 568494 E; Figure 1). It was built in 1992 as a mitigation project for the building of Corridor H (Balcombe 2003). It is owned by the West Virginia Division of Highways (WVDOH) and under jurisdiction of District 7. It sits at an elevation of 435 m and is adjacent to the Buckhannon River and Corridor H. Buckhannon Triangle has loamy soils and an underlying geology of alluvium. Vegetation was mainly scrub-shrub; however, there were emergent and forested areas at the time of sampling in 2021.

Glade Farms

Glade farms is a 11.8-ha mitigated wetland (Figure 2). It is located close to the Pennsylvania border in Preston County, WV (4397093 N 626684 E) and was restored in 2019. Historically, this site was used as pasture and several ditches exist within the wetland, indicating the site was previously a wetland before being converted to pasture. There is ongoing monitoring and maintenance of the wetland by Decota Consulting Company, including spraying for reed canary grass (*Phalaris arundinacea*). An unnamed tributary of Fike Run flows throughout the wetland. Glade farms is at an elevation of 630 m and has silty clay loam soil types. The underlying geology of Glade farms is sandstone. Vegetation was emergent with scrub-shrub areas at the time of sampling in 2021.

Hazelton

Hazelton is a 2-ha wetland located in Hazelton, in Preston County, WV (4390974 N 625712 E; Figure 3). It is owned by the WVDOH and is under the jurisdiction of District 4. It was mitigated in 2007 in response to the Mon-Fayette Expressway system project (Gingerich

2010). It sits alongside Interstate 68 at an elevation of 660 m. Primary water sources include Little Sandy Creek and roadside runoff (Gingerich 2010). Steep slopes lead into the wetland from surrounding roads. There is a ponded area within the wetland that was crafted into the shape of WV. Hazelton was predominantly composed of scrub-shrub vegetation, although there were patches of emergent vegetation at the time of sampling in 2020. Soil types at the site consist of silt loams and there is an underlying geology of sandstone.

Hillcrest 1

The mitigation project at the Hillcrest Wildlife Management Area (WMA) was implemented in three different places within the large WMA. Hillcrest 1 is also known as North Fork of Tomlinson Run and was mitigated in 2016 (Figure 4). It is 6 ha in area and mitigation was led by the West Virginia Department of Environmental Protection (WVDEP) from in-lieu-fee (ILF) funds. It is located near Fairhaven in Hancock County, WV (4490638 N 538011 E). Natural hydrology flowing into the wetland is from the North Fork of Tomlinson Run. Runoff from Route 8 may also flow into the outer edges of the wetland. Active vegetation and water monitoring co-occurred during sampling in 2021. Vegetation was primarily emergent with some patches of scrub shrub. Hillcrest 1 sits at an elevation of 307 m and is characterized by silt loams with an underlying sandstone geology.

Hillcrest 2

Hillcrest 2 is also known as the Middle Fork of Tomlinson Run and was mitigated as part of the same project as Hillcrest 1 above (Figure 5). It is geographically separated from Hillcrest 1 by mountains (~2 km); therefore, the two wetlands were treated as separate sites for small mammal sampling purposes. Like Hillcrest 1, it was mitigated in 2016 and was led by the WVDEP from ILF funds. It includes 9.2 ha of wetland and is located near Fairhaven in Hancock

County (4488993 N 539026 E). Hunting occurs in the WMA. Natural hydrology of the site is from the Middle Fork of Tomlinson Run. A few areas in the northern section of the wetland were dammed due to beaver activity at the time of sampling in 2021. Vegetation at the site was shrub-scrub with emergent areas. With mountains on either side of the wetland, the elevation of the Hillcrest 2 is 330 m. Silt loam is the main soil type at the site, and it is underlain by shale geology.

Hoeft Marsh

Hoeft Marsh is the given name of the mitigated wetland in the Green Bottom WMA, although some sources refer to it simply as Green Bottom (Figure 6). It is located near Glenwood in Cabell County, WV (4271673 N 389982 E). It was mitigated by the WVDEP from ILF funds and is a local area for recreational activities, such as hunting and fishing. This freshwater marsh was created in 2020 and is composed of 4.88 ha of wetland. The wetland is adjacent to a buttonbush (*Cephalanthus occidentalis*) swamp and the WMA itself is adjacent to the Ohio River; the wetland is in its floodplain. Vegetation is mostly emergent, with some shrub-scrub areas. It is underlain by silt loam soils and alluvium geology and is situated at 164 m in elevation. Hoeft Marsh was sampled in 2021, a year after its completion.

McClintic

McClintic is a 4-ha mitigated wetland located within the McClintic WMA and is near Point Pleasant in Mason County, WV (4309633 N 406220 E; Figure 7). Mill Run, a small stream, flows through the wetland. When sampling occurred in 2021, water levels within the embankment at the site were low. Previous usage of the site and surrounding land was as a storage area for explosives during World War II, although ongoing remediation of harmful waste was occurring before the completion of the wetland in 2019 by the WVDEP from ILF funds.

This young wetland is composed of mainly emergent vegetation with some shrubs and has primary silt loam for soil type. The site is underlain by alluvium and sits at an elevation of 187 m.

Montrose

The Montrose mitigated wetland, also known as the Leading Creek wetland (Balcombe 2003; Gingerich 2010; Strain 2014), was also sampled in 2020 (Figure 8). It is located near Montrose in Randolph County, WV (4321684 N 602659 E). Owned by the WVDOH under the jurisdiction of District 8, they mitigated the site in 1996 due to the construction of the Appalachian Corridor H highway project (Gingerich 2010). Past land use of the wetland was likely agricultural. It contains 3.2 ha of wetland, with a successful mitigated black willow swamp and mitigated ponds, one of which partially failed due to failure of ditch plugs. Vegetation is mainly forested wetland, although some areas are emergent and scrub-shrub. It has an elevation of 593 m, silt loam soils, and alluvium bedrock.

Nicholas

The Nicholas wetland, sampled in 2020, is a mitigated wetland made in 1997 to offset construction of U.S. Route 19 (Strain 2014). Owned by the WVDOH under jurisdiction of District 9, it is located near Summersville, in Nicholas County, WV (4247775 N 514458 E; Figure 9). It is composed of 15.8 ha of wetland and has flooded areas within the wetland as well. Enoch Branch is the natural stream that flows through the wetland. This site contains mainly scrub-shrub with some emergent wetland areas. It is underlain by drained silt loam soils and sandstone bedrock. Elevation of the site is 575 m.

Queens

The Queens wetland is in Tucker County within the Monongahela National Forest (4320054 N 611562 E; Figure 10). Formerly a ditched old field, it was restored in 2009 by Tom Biebighauser. It became a functioning wetland in just a few years after restoration. It sits in the floodplain of Shaver's Fork at an elevation of 541 m. It encompasses 2.7 ha of predominantly scrub-shrub wetland, with some forested patches. The site is also characterized by silt loam soil and shale bedrock. The Queens wetland was sampled in 2020.

Stauffer's Marsh

Stauffer's Marsh, sampled in 2021, is a public recreation area in Shanghai, Berkeley County, WV (4368482 N 746415 E; Figure 11). It was completed in 1992 by the Natural Resource Conservation Service (NRCS) as part of its Wetlands Reserve Program (WRP) and is currently owned and preserved by the Potomac Valley Audubon Society (PVAS). It is thought that this was historically a natural wetland that was drained for agriculture in the early 1900s (Strain 2014). Currently it is composed of 11.7 ha of wetland, with some of its area covered by open pools of water. The wetland has emergent vegetation with some scrub-shrub. Soil type at the site is silt loam and is underlain by alluvium geology. The elevation of Stauffer's Marsh is 146 m.

Sugar Creek

The Sugar Creek site consists of 28.7 ha of mitigated wetland (Figure 12). Located near Philippi, in Barbour County WV (4329112 N 591473 E), it was made in 1995 as mitigation for Corridor H (Balcombe 2003). It is owned by the WVDOH, under jurisdiction of District 7. Natural hydrology is supplied by Sugar Creek, which the wetland site is named after, and unnamed tributaries. The site is composed of frequently flooded silt loam soil and has shale bedrock. Elevation is 484 m and the dominant vegetation type is both emergent and scrub-shrub. Sugar Creek was sampled in 2020.

Tygart Mitigation Bank

The Tygart Valley Mitigation bank (5.2 ha) was made in 2011 and is privately owned. The site sits at 585 m in elevation and is a floodplain of the Tygart Valley River near Elkins, Randolph County, WV (4307018 N 597273 E; Figure 13). Historic land use was agricultural. Soil type at the site is characterized by silt loam and has an underlying geology of alluvium. Vegetation was scrub-shrub with many emergent areas at the time of sampling in 2020.

Walnut Bottom

Like other DOH projects, the Walnut Bottom mitigated wetland was made in 1997 to mitigate for construction of Appalachian Highway Corridor H (Balcombe 2003). It is of 6.6 ha and is located near Moorefield in Hardy County, WV (4334561 N 674160 E; Figure 14). It is currently owned by the WVDOH and is under jurisdiction of District 5. An unnamed tributary of Anderson Run flows through the wetland and it is likely highway runoff also drains into the wetland, given its proximity to Corridor H and steep slopes leading into the wetland from the road. The main soil types at the site are loam, silty clay loam, and silt loam, with an underlying geology composed of shale. Elevation of the site is 335 m. Vegetation is predominantly emergent, with few patches of scrub-shrub vegetation. This site was sampled in 2021.

Natural wetland sites

Beaver Pond

Beaver Pond is named after the run that feeds into the wetland and is located within Randolph County, WV (4314929 N 607738 E; Figure 15). It is within the Monongahela National

Forest and is adjacent to Shaver’s Fork. At an elevation of 606 m, it is underlain by both fluvaquent soils and a bedrock of shale. At the time of sampling in 2020, vegetation was predominantly scrub-shrub with some forested areas.

Bruceton Mills

The Bruceton Mills wetland is the remnant of an old beaver dam located on private property (Gingerich 2010; Strain 2014). There are nearby houses, but 1.5 ha of wetland remains untouched by development. As its name suggests, it is located near Bruceton Mills in Preston County, WV (4393355 N 615515 E; Figure 16). Natural hydrology into the wetland is from an unnamed tributary of Glade Run (Gingerich 2010). The elevation is 512 m, the dominant soil type is silt loam, and the soil is underlain by sandstone and shale. At the time of sampling in 2020, vegetation was emergent with scrub-shrub areas.

Burches Run

The Burches Run wetland, 5.9 ha, is owned by the WVDNR and is part of the Burches Run WMA (Figure 17). This WMA formerly went by the name of Burches Run Lake WMA but was aptly changed after a dam removal in the early 2000s. Located near Wheeling, in Marshall County, WV (4424751 N 531973 E), it has natural hydrology flowing into the wetland in the form of Burch Run. The wetland has silt loam soils and sandstone for bedrock. It is located at an elevation of 251 m. Vegetation in the wetland was mainly forested at sampling in 2021.

Cross Creek

The Cross Creek wetland is within the Cross Creek WMA, formerly strip-mining lands, is owned by the WVDNR (Figure 18). It is a natural wetland near Wellsburg, in Brooke County, WV (4464433 N 540518 E). The Cross Creek wetland, 12 ha, is surrounded by steep slopes that drain into the wetland. Water sources into the wetland include run-off from the surrounding slopes and Parmar Run, which flows through the wetland. Beaver-created open water pools were plentiful at the site at the time of sampling in 2021. Vegetation at the wetland is primarily emergent with some scrub-shrub and forested areas southwards into the wetland. Silt loams are the main soil types present. Cross Creek is located at an elevation of 312 m and has a bedrock of sandstone.

Fairfax

This wetland resides in the Fairfax Pond-Rehe WMA and is a mix of scrub-shrub and forested wetland that is adjacent to a pond complex where people often recreate (Figure 19). Located between Arthurdale and Reedsville in Preston County, WV, the wetland is owned by the WVDNR and was acquired by them in 2014 (4372126 N 603022 E). The wetland has an area of 14.5 ha, an elevation of 529 m, and is underlain by silt loam soil and shale bedrock. This site was sampled in 2021.

Green Bottom

The natural wetland at Green Bottom is owned by the WVDNR and located within the WMA of the same name near Glenwood in Cabell County, WV (4271542 N 393756 E; Figure 20). This wetland has an area of 15 ha and is in the floodplain of the Ohio River which is directly adjacent to it. Vegetation at the wetland is scrub-shrub. It sits at an elevation of 167 m and is characterized by silt loam soils with alluvium bedrock. Green Bottom was sampled in 2021.

Little Indian Creek

Little Indian Creek, sampled in 2021, is a mostly forested wetland, 3.7 ha, that is owned by the WVDNR and is located near Georgetown, in Monongalia County (4382126 N 579231 E; Figure 21). The land was purchased by the state as a WMA in 2006 and was formerly a mining site. It sits at an elevation of 294 m and is underlain by silt loam soils and sandstone bedrock.

Lower Glady

The Lower Glady wetland, sampled in 2020, is in Randolph County, WV (4315293 N 622151 E; Figure 22). It is within the boundaries of the Monongahela National Forest and is 4 ha in area. The Cheat River is directly adjacent to the wetland. The wetland is forested primarily but has emergent and scrub-shrub areas. It sits at an elevation of 693 m and is underlain by fluvaquent soils and shale bedrock.

Old Town Creek

Sampled in 2021, the wetland at Old Town Creek is located near Point Pleasant in Mason County, WV (4306199 N 406683 E; Figure 23). It is located within the boundaries of the McClintic WMA, and a portion of the 30 ha-wetland underlies powerlines. Adjacent to the wetland is a large pond complex with beaver activity. Vegetation at this wetland is scrub-shrub with some forested vegetation eastwards. It has an elevation of 176 m and is underlain by silt loam soil with alluvium bedrock.

Short Mountain

The Short Mountain wetland, 45 ha, is located within the Short Mountain WMA, and is owned by the WVDNR near Baker in Hampshire County, WV (4341796 N 701821 E; Figure 24). Meadow Run flows through the wetland providing natural hydrology. The wetland is occupied by emergent vegetation with scrub-shrub areas. Soil types at the wetland include both silt loam and loam. It sits at an elevation of 628 m and is underlain by sandstone bedrock. Short Mountain wetland was sampled in 2021.

Sleepy Creek

Named after the WMA it resides in, Sleepy Creek is owned and actively managed by the WVDNR (Figure 25). It is in Berkeley County, WV (4376125 N 743791 E). The wetland is adjacent to Sleepy Creek Lake, which also resides in the WMA. The 7 ha-wetland is also surrounded by recreational activities occurring within the WMA, such as hiking and camping. It is situated at 332 m in elevation and is primarily forested wetland with some scrub shrub areas. This wetland is underlain by silt loam and loam soils, as well as by shale bedrock. Sleepy Creek was sampled in 2021.

Three Springs

The Three Springs wetland, sampled in 2020, is located near Alpena in Randolph County, WV (4314270 N 620954 E; Figure 26). It is within the boundaries of the Monongahela National Forest and is adjacent to a dispersed camping area. Although not directly beside the wetland, the Cheat River flows near to it. Vegetation at the site is scrub-shrub. The wetland is at an elevation of 695 m and has an area of 6.8 ha. Fluvaquent soils and shale bedrock characterize it.

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Figures

Figure 1. Photograph of Buckhannon Triangle, a West Virginia Division of Highways owned mitigated wetland in Upshur County, West Virginia, 2021.

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Figure 20. Photograph of Green Bottom Wildlife Management Area, a natural wetland in Cabell County, West Virginia, 2021.

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Figure 25. Photograph of Sleepy Creek Wildlife Management Area, a natural wetland in Berkeley County, West Virginia, 2021.

Figure 26. Photograph of the Three Springs natural wetland in Randolph County, West Virginia, 2020.

CHAPTER 2

A comparison of small mammal communities of restored and natural wetlands in West Virginia, USA

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This chapter is written in the style of *Land (MDPI).*

Abstract

Wetland restoration is a common practice, and in many cases, it is for mitigation to offset losses to natural wetlands due to human interference. Researchers commonly compare bird, amphibian, and reptile communities between these wetlands and natural wetlands but overlook small mammals. However, they are essential to consider as they serve a fundamental role in the ecosystem as seed dispersers and prey for larger wildlife. We conducted small mammal trapping on 26 wetlands (n = 14 restored, n = 12 natural) in West Virginia, USA, in the summers of 2020 and 2021 to obtain community metrics and compare these metrics between wetland types. We found most aspects of small mammal communities in restored and natural wetlands were similar, including species mass and occupancy, the relative density of *Peromyscus* spp., diversity, richness, evenness, and community composition. However, the apparent abundance of deer mice (*Peromyscus maniculatus*) was higher in natural wetlands ($P<0.001$), as was the total small mammal apparent abundance ($P<0.001$). Because the three rarest species were captured exclusively in natural wetlands, the ability of restored wetlands to provide adequate habitat for rare or wetland-obligate species may be biologically significant. Restored wetlands mainly offer sufficient habitat for small mammal communities, but apparent abundance at restored wetlands may differ from natural wetlands depending on species.

Keywords: created wetlands, deer mice, mammals, meadow voles, *Microtus pennsylvanicus, Peromyscus leucopus, Peromyscus maniculatus*, restored wetlands, white-footed mice, wildlife

Introduction

Wetlands are relied upon for many functions, such as flood mitigation and improving water quality (Millennium Ecosystem Assessment 2005). Due to the diverse functionality wetlands provide, many programs to restore wetlands exist. Many United States-based restoration programs are voluntary and run through the government, such as the Agricultural Conservation Easement Program (Natural Resources Conservation Service 2022) or the Partners for Fish and Wildlife Program (U.S. Fish and Wildlife Service 2022). There are also voluntary, non-governmental organizations that restore wetlands, such as Ducks Unlimited. While many voluntary programs exist, wetland mitigation is compulsory; it is the lawfully imposed solution to planned human-induced wetland losses in the United States (Hough and Robertson 2009). Wetland mitigation offsets losses to natural wetlands due to human activities by creating, restoring, enhancing, or preserving wetlands (Brown and Lant 1999). Wetland creation involves the establishment of wetlands where they have not historically existed. Degraded wetlands restored to their original ecosystem functioning capacity are known as restored wetlands. Restoration is often the first approach to wetland mitigation because necessary hydrologic conditions are already present (Federal Register 1995). Researchers often focus on restored or created wetlands because the success of these wetland types is controversial (Mitsch et al. 1998; Gutrich and Hitzhusen 2004). Researchers need to assess restored wetland functionality because of the widespread practice of wetland restoration and the assumption that these wetlands can truly replace natural wetlands.

One function of wetlands is to provide habitat for an array of wildlife. Wildlife responses to wetland conservation vary depending on taxa. Birds, for instance, may be indifferent to restored wetlands, as demonstrated by passerines on restored conservation easement wetlands in

West Virginia, USA; they did not differ in occupancy probability or species richness between restored and natural wetlands (Lewis et al. 2019). Similarly, when the avian community was compared between mitigated and natural wetlands, no significant differences were found between the two types of wetlands (Balcombe et al. 2005a). Researchers have also studied amphibian responses to wetland creation and restoration. Occupancy of some amphibian species is similar between natural and mitigated wetlands because restored wetlands provide chorusing (Strain et al. 2017a) and breeding (Strain et al. 2017b) habitat. However, these wetlands may not be a viable option for some amphibian species because some created wetlands dry up prematurely, leading to high larvae mortality and little recruitment (Swartz et al. 2020). The problem of created wetlands potentially drying up is not only a problem for amphibians but also reptiles. Hartwig and Kiviat (2007) described one species of turtle that would move from created wetlands back to natural wetlands when this occurred. In other species of turtles, the abundance between restored and natural wetlands was not different (Gulette 2018). Additionally, bats were studied in restored and created wetlands; no differences in bat activity were observed between these wetlands and natural wetlands, except in eastern pipistrelles (*Perimyotis subflavus*), which were positively associated with natural wetlands (Maslonek et al. 2010). Furthermore, macroinvertebrate assemblages, a prey base for some wildlife species, may be similar between natural and mitigated wetlands (Balcombe et al. 2005b; Strain et al. 2014).

One largely understudied taxon in wetland restoration is small mammals. Lack of data is problematic because small mammals play an essential role in the ecosystem. Small rodents are significant seed dispersers, influencing the landscape (Brewer and Rejmanek 1999). Additionally, differing individual tendencies of scatter-hoarding species like deer mice (*Peromyscus maniculatus*), southern red-backed voles (*Myodes gapperi*), and northern short-tailed shrews (*Blarina brevicauda*) can affect how far seeds travel (Brehm et al. 2019). Aside from their importance as seed dispersers, they are important prey species for birds (Korpimaki and Norrdahl 1991), bobcats (*Lynx rufus*) (Hass 2009), and coyotes (*Canis latrans*) (Miller et al. 2012). Small mammals can also influence plant biomass, richness, and diversity (Root-Bernstein and Ebensperger 2013). Apart from their roles as seed dispersers, prey, and vegetation influencers, small mammals can also be valuable bioindicators because of their sensitivity to changes in habitat (Pearce and Venier 2005; Leis et al. 2008). The vegetative community impacts small mammal diversity and abundance (Birney et al. 1976; Wywialowski 1987).

Researchers have not completely overlooked small mammals in restored wetlands. Small mammal community composition changed due to variations in hydroperiod and water depth in a restored wetland in Arkansas, USA (Whitsitt and Tappe 2009). In South Carolina, USA, researchers showed that the small mammal community was similar between restored wetland areas that received plantings and those revegetated naturally (Wike et al. 2000). Kurz et al. (2013) found small to mid-sized mammal abundance, richness, and species composition was similar between three reference and three restored wetlands in Ohio, USA. Therefore, existing research suggests small mammal communities respond to wetland restoration favorably. Age of restoration in these studies may have impacted the findings, as restored wetlands were no older than seven years at the time of research for Kurz et al. (2013) and even less for Whitsitt and Tappe (2009) and Wike et al. (2000). Researchers should sample a broad range of ages because restored wetlands may become more similar to natural wetlands with age (Balcombe et al. 2005c).

The objective of this study was to examine small mammal community characteristics in natural versus restored wetlands to determine if these wetlands can provide for small mammal communities. Wetland mitigation is designed to offset losses of destroyed natural wetlands. If restored wetlands, which account for many mitigated wetlands, cannot support similar community characteristics of vital wildlife taxa, then the wetland has not fulfilled its purpose in its entirety. Specifically, we examined 1) abundance, 2) occupancy, 3) relative density of *Peromyscus* spp., 4) mass, 5) species diversity, 6) species richness, 7) species evenness, and 8) community composition of small mammals between restored and natural wetlands. Given the finding by Balcombe et al. (2005c) that mitigation wetlands had higher vegetation species richness, diversity, and evenness, coupled with the result that small mammals are reliant on vegetation diversity (Wywialowski 1987), we hypothesized that small mammal community metrics would be higher in restored wetlands than in natural wetlands.

Methods

Study area

We sampled 26 wetlands (14 restored/created (i.e., wetlands were primarily restored but likely included created/enhanced areas and hereafter are referred to as restored) and 12 natural wetlands) across West Virginia, USA, and its three main ecoregions: Ridge and Valley (5 restored and 5 natural), Central Appalachians (4 restored and 2 natural), and Western Allegheny Plateau (5 restored and 5 natural) (EPA 2022; Figure 1). Broad characterizations of these ecoregions are that the Ridge and Valley have forested ridges with agricultural valleys, the Central Appalachians have a higher elevation and increased rainfall, and the Western Allegheny Plateau has many hills and agriculture (Woods et al. 1999). Mean annual precipitation for the Ridge and Valley, Central Appalachians, and Western Allegheny Plateau ecoregions are 1,138 mm, 1,180 mm, and 1,063 mm, respectively (Wiken et al. 2011). Elevation reaches its highest in the Central Appalachians (1,402 m), with elevation in Ridge and Valley ranging from 152–1,311 m, and lower elevation in the Western Allegheny Plateau (<610 m) (Woods et al. 1999). Although elevation varies by each ecoregion within the state, the average elevation is higher in West Virginia than in any other state east of the Mississippi River (Brack et al. 2002).

West Virginia is primarily forested (79%; Widmann 2014). Wetlands cover <1% of West Virginia's land surface, which totals about 40,468 ha (WVDEP and WVDNR 2021). Sampled wetlands ranged in size from 2 to 28.7 ha (mean ± SE ha = 8.1 ± 1.9 ha) for restored wetlands and 1.5 to 45 ha (mean ± SE ha = 12.6 ± 3.6 ha) for natural wetlands (Appendix 1). Most sampled wetlands were owned by a governmental agency, such as the West Virginia Division of Natural Resources (4 restored, 8 natural), West Virginia Division of Highways (6 restored, 0 natural), and the U.S. Forest Service (1 restored, 3 natural); however, some wetlands were privately owned (2 restored, 1 natural) and one restored wetland was owned by the Potomac Valley Audubon Society and is a public recreation area.

We classified wetlands as palustrine emergent (7 restored, 3 natural), scrub-shrub (6 restored, 4 natural), or forested (1 restored, 5 natural) (Cowardin et al. 1979), although many wetlands had patches of emergent, scrub-shrub, and forested areas. Soil at sampled wetlands were primarily silt loam (Soil Survey Staff 2021), and all sites had underlying sedimentary geology of shale, sandstone, or alluvium (WVDEP 1998) (Appendix 1). Elevation ranged from 146 to 660 m (mean ± SE m = 426.5 ± 48.2 m) for restored wetlands and from 167 to 695 m (mean ± SE m = 432.9 ± 57.3 m) for natural wetlands (Appendix 1). Restored wetland sites were of differing ages (mean ± SE years = 14.2 ± 2.8 years), with older sites being made in 1992 (29 years old), to newer sites having recently been made in 2020 (1 year old). Five wetlands were between 1–6 years old (mean ± SE years = 3 ± 0.83 years), two wetlands were between 7–12 years old (mean ± SE years = 10 ± 1 years), one wetland was between 13–18 years old (13 years old), three wetlands were between 19–24 years old (mean ± SE years = 22.67 ± 1.33 years), and three wetlands were between 25–30 years old (mean ± SE years = 27.67 ± 1.33 years) (Appendix 1). All wetlands were sampled in 2020 (6 restored, 4 natural) or 2021 (8 restored, 8 natural). Between wetlands, we cleaned equipment to avoid the spread of invasive species and diseases (Bryzek et al. 2022).

Small mammal trapping

We used a transect trapping design to capture small mammals because they have a higher rate of captures, are more efficient at sampling communities (Pearson and Ruggeiro 2003) and are more likely to capture rare species than a traditional grid array (Harkins et al. 2019). Our transect design consisted of 240 m long transects, with traps spaced 10 m apart (Read et al. 1988); therefore, there were 25 traps per transect. Larger wetlands (>240 m in length) received more transects for full sampling coverage of the wetland. Transects were placed on both edge and interior locations of the wetland to sample all wetland areas. We placed transects ≥50 m apart and established 2–6 (mean = 3.3; SE = 0.2) transects per wetland. We determined wetland boundaries before trapping by referring to the National Wetlands Inventory (NWI) GIS layer (USFWS 2021), as well as by on-site evaluation of wetland boundaries (using wetland plant and hydrology indicators).

We used 5.08 cm × 6.35 cm × 16.51 cm folding Sherman Live Traps (H.B. Sherman Traps, Inc, Tallahassee, Florida, USA) that were baited with a mixture of peanut butter and oats wrapped in wax paper; bait was replaced as needed throughout the trapping session (Edalgo and Anderson 2007). To encourage small mammal survival, we added cotton to each trap (Szebor and Strubel 2013). We checked traps ≤ 24 hours in the morning during the trapping session. A trapping session consisted of five consecutive nights of trapping. Trapping was restricted to June through August of both sampling years because seasonal, temporal variation among sites can lead to incorrectly drawn conclusions about the small mammal community (Asher and Thomas 1985). To limit the effects of weather and precipitation variation between wetland types, we trapped restored and natural wetlands simultaneously in pairs; therefore, keeping the influence of weather consistent between a restored and its natural wetland counterpart.

We marked most small mammals with a #1005-1 Monel ear tag (National Band and Tag Company, Newport, Kentucky, USA) in their left ear. We did not ear tag shrews due to the tendency of the tag to quickly fall off and damage their ears (Craig 1995). Instead, we marked shrews with hair dye in a unique pattern of dots to distinguish among captured individuals (Craig 1995; Stromgren 2008). If a small mammal occupied a trap, we recorded species, mass, body and tail length, sex, and reproductive condition (Bruseo et al. 1999; Glennon et al. 2002). Sex was recorded as male or female, and reproductive condition was recorded as adult or juvenile. Additionally, we recorded if females were pregnant or lactating. We differentiated white-footed mice (*Peromyscus leucopus*) and deer mice (*Peromyscus maniculatus*), two similar species, from each other by body and tail lengths and tail pelage (Kays and Wilson 2009).

Vegetation Assessment

We assessed vegetation using 1 × 1 m quadrats along transects; traps were centered within each quadrat in each transect. We identified plant species and assessed their cover within the plot using the Daubenmire scale with percent ranges of 1 = 1–5%, 2 = 6–25%, 3 = 26–50%, 4 = 51–75%, 5 = 76–95%, and 6 = 96–100% (Daubenmire 1959). We determined percent coverage by counting the number of times that a particular plant species occurred in each cover class, multiplying this number by the mid-point (2.5, 15, 37.5, 62.5, 85.5, 97.5 for each cover class, respectively), then summing the products of all the cover classes to obtain a total, and finally dividing this sum by the number of quadrats that were completed at the site (U.S. Department of Agriculture and U.S. Department of the Interior 1996). If leaf litter was present, we measured depth to the nearest cm and recorded at its deepest point within the quadrat. We used a spherical densiometer to estimate canopy cover to the nearest percent at each quadrat; we took a reading at each of the four corners of the quadrat and averaged the values to obtain a reading for each quadrat (Lemmon 1956).

Statistical Analysis

Apparent Abundance

We calculated apparent abundance using count data of unique individuals for each observed species and a total count of small mammals across all species. We call this apparent abundance because it is not a direct estimate of abundance but instead assumes the number of captures (counts) of the species is proportional to true abundance. We estimated apparent abundance instead of abundance using our capture-mark-recapture data because there were not enough captures made to estimate abundance for each species. Apparent abundance was assumed using a generalized linear model with a Poisson distribution based on count data, our response variable. Within this model, we specified wetland type (restored or natural) as the predictor variable and included an offset for trap nights to standardize trapping effort across sites. To calculate trapping effort (number of trap nights) at each site, we multiplied the number of traps in a transect (25) times the number of transects in a wetland (two to six depending on the site) and multiplied by the number of nights in a trapping session (five). Therefore, each wetland met a minimum of 250 planned trap nights. However, we subtracted a half-trap night from the trap night total when a trap had been falsely snapped and was empty (Nelson and Clark 1973; Hodo et al. 2020). This half-trap night subtraction assumes the trap was open for at least half the night before it was snapped, leaving the opportunity for a small mammal to enter the trap earlier (Nelson and Clark 1973; Hodo et al. 2020). Using apparent abundance models for each species (n = 9), we determined expected apparent abundance per wetland type (restored or natural) for species that were found in both wetland types (n = 6). We also found expected apparent abundance by wetland type for a total count across all small mammal species. Our hypothesis test included a type 1 error rate of 0.05.

Relative Density

To determine the relative density of *Peromyscus* at each site, we first found abundance using capture-mark-recapture data and the M_0 model described by Otis et al. (1978). Deer mice and white-footed mice were grouped in the abundance estimation because they occur in similar microhabitats, use similar nesting sites, and have a similar summer diet (Wolff et al. 1985). Additionally, there were not enough captures for either species of *Peromyscus* to perform analyses for each separately. We defined relative density as *Peromyscus* abundance divided by the number of traps at each site. Because the distance small mammals traveled to reach our traps is unknown, and we do not know the exact trap coverage area, we cannot calculate density. However, we assume this relative value is proportional to density. The relative density model assumes a closed population, meaning no births, deaths, immigration, or emigration throughout the sampling period. This is a reasonable assumption for a five-night trapping period, as we did not capture any lactating females or mothers with babies during our trapping sessions. We implemented this model in MARK using the full likelihood p and c model (White 2020) and specified that capture probability did not change among trap nights. For each site, we divided abundance estimates from MARK by the total number of traps deployed at the site. The resulting value is the relative density of *Peromyscus* at each site. We compared relative density between wetland type using a general linear model (an ANOVA analysis) with a type 1 error rate of 0.05.

Mass

To determine if the mass of each species differed between wetland types, we used mass data collected on individuals (initial capture only) and constructed a general linear model with a type 1 error rate. Our response variable was mass and our predicator variables were wetland type, sex, reproductive condition, an interaction between wetland type and sex, an interaction between wetland type and reproductive condition. Interaction terms were included in order to be able to conclude whether adult, juvenile, female, or male mass differed by wetland type for each species. This model was implemented for three species that were captured in both wetland types (deer mice, white-footed mice, and meadow voles). A variation of this model excluding the reproductive condition variable and the interaction term between reproductive condition and wetland type was implemented for northern short-tailed shrews and eastern chipmunk because only adults were captured for those species. A simple linear model was run for meadow jumping mice with mass as the response variable and wetland type as the predictor variable since only adult males were captured. We used contrasts to account for the influence of sex (male and female) and reproductive condition (juvenile and adult) on mass for both wetland types. Contrasts were made using the package 'multcomp' (Hothorn et al. 2008).

Occupancy

We ran single season occupancy models described by Mackenzie et al. (2002) because each site was only visited once for a 5-night trapping session. Our response variable was the detection of a species; we collapsed detection of a species within a transect as a 1 if detected, and as a 0 if not detected. Wetland type, restored or natural, was used as our site-level covariate and our survey level covariate (detection covariate) was held constant. These models were implemented using unmarked (Fiske and Chandler 2011), using a maximum likelihood estimation approach (MacKenzie et al. 2018). We fit occupancy models for all observed species except for deer mice; they were detected at each site, therefore leading to poor estimates of occupancy. We did not fit an occupancy model for total small mammals observed for the same reason.

Richness, Diversity, and Evenness

Apparent richness, diversity, and evenness were calculated for small mammal communities at each wetland site. We determined apparent species richness as the number of species observed at each site. Because there is potential for species to have been missed due to not being captured, we created a species accumulation curve to assess how likely we were to capture all species present given our sampling effort for all sites, and for restored and natural sites. To calculate apparent diversity for each site, we used previously calculated apparent abundance values and the Shannon-Wiener diversity index (Shannon 1948):

$$H_j' = -\sum_{i=s_j}^{s} p_i \ln{(p_i)}$$

Where H_j' is estimated diversity for a site, 's_j' is the number of species at a site, and 'p_i' is the proportion of each abundance relative to other species present at a site (Shannon 1948). This equation was then implemented using vegan in R specifying the Shannon diversity index (Oksanen et al. 2020). Additionally, apparent evenness was calculated for each site using Pielou's evenness index (Pielou 1966): $J = H' / \ln(S)$ where H' represents the previously calculated Shannon diversity and S is total species richness.

To compare apparent species richness between wetland types, we used a generalized linear model; we treated each wetland as an independent replicate and compared counts of species richness between wetland types using a Poisson generalized linear model with a type 1 error rate of 0.05. Because sites had various trapping effort, we included an offset of trapping effort in the model and predicted species richness expected counts per 100 trap nights by wetland type. Additionally, we constructed species accumulation curves using the specaccum function in vegan (Oksanen et al. 2020). To compare diversity and evenness between restored and natural wetlands, we created linear models (an ANOVA analysis) with diversity and evenness as our response variable and wetland type as the predictor variable with a type 1 error rate of 0.05.

Community Composition

To assess differences in small mammal community composition, we used non-metric multidimensional scaling (NMDS). This technique uses rank orders, making it more flexible and able to accommodate more kinds of data than other ordination methods (Minchin 1987). We specified Bray-Curtis dissimilarity to obtain a distance matrix which we then plotted for a visual assessment of community composition between the two wetland types. To numerically assess differences in community composition between wetland types, we used an analysis of similarity

($\alpha = 0.05$), as it is used to complement the NMDS plot (Clark 1993; Santos-Filho et al. 2015), and again specified Bray distance. Community data for this analysis did not use previously calculated estimates of apparent abundance; the previous apparent abundance analysis used an offset within the model to standardize trapping effort and produced abundance estimates that took wetland type into account within one cohesive model. Because of the nature of this analysis, we needed abundance estimates that were not already influenced by wetland type so that we could visually assess community overlap, or lack thereof. Instead, community data for the analysis was our raw count data for each species for each site, with each value being standardized by catch-per-unit effort (CPUE); we calculated this as the percentage of captured individuals divided by the trapping effort at a given site (Nicolas and Colyn 2006), which reads as the number of captures per 100 trap nights. To implement NMDS, we used the function metaMDS in vegan (Oksanen et al. 2020) and specified two reduced dimensions with 100 iterations until a solution was reached. We then used the function anisom in vegan (Oksanen et al. 2020) to analyze small mammal similarity.

Vegetation

Vegetation diversity was calculated using percent canopy cover as a proxy for abundance in the Shannon-Weiner diversity estimation. We calculated Shannon-Weiner diversity with the diversity function in the vegan package in R (Oksanen et al. 2020). We compared vegetation diversity between wetland type using linear models with diversity as the response variable and wetland type as the predictor variable. We assumed a type 1 error rate of 0.05. Additionally, we assessed vegetation community composition between restored and natural wetlands using NMDS. Vegetation data was based on percent coverage of a species at each wetland site. Again, this was implemented using metaMDS in vegan (Oksanen et al. 2020) with 2 reduced dimensions and 100 iterations.

Results

After adjusting for snapped traps, we had a total trapping effort of 10,060 trap nights across our 26 wetland sites over 2 summers. Trapping effort at restored wetlands ranged from 204.5 to 719 (mean ± SE = 412.8 ± 40.7), and 214 to 584 (mean ± SE = 356.6 ± 42.57) at natural wetlands. There were 6.36 captures per 100 trap nights (640 total captures), and 4.25 captures per 100 trap nights were unique individuals (428 unique individuals). Recaptured individuals made up 2.10 captures per 100 trap nights (212 recaptures) and were primarily *Peromyscus* spp. Nine total species were represented. We captured deer mice (N = 187), white-footed mice (N = 111), meadow voles (N = 73), meadow jumping mice (N = 16), northern short-tailed shrews (N = 23), and eastern chipmunks (*Tamias striatus*) (N = 11) at both wetland types, and woodland jumping mice (*Napaeozapus insignis*) (N = 2), masked shrews (*Sorex cinereus*) (N =4), and a southern flying squirrel (*Glaucomys volans*) (N =1) exclusively at natural wetlands.

Apparent abundance

Peromyscus was the most captured genus, with deer mice accounting for 43.69% of total unique captures, and white-footed mice 25.93%. Apparent abundance of deer mice was greater for natural wetlands ($Z = 4.84$, $P < 0.001$) (Table 1; Appendix 3), but similar between wetland types for white-footed mice ($Z = 1.488$, $P = 0.137$). Meadow voles were the second most captured taxa, accounting for 17.05% of unique individuals captured. Apparent abundance of meadow voles was statistically similar between wetland types by a narrow margin ($Z = -1.89$, P

= 0.06). We experienced more captured meadow voles at restored wetlands. We captured northern short-tailed shrews, meadow jumping mice, and eastern chipmunks less frequently (5.37%, 3.73%, and 2.57% of unique captures, respectively), and we did not see a difference in apparent abundance between wetland types (P = 0.246, 0.684, and 0.170, respectively). Woodland jumping mice, masked shrews, and the southern flying squirrel were captured only in natural wetlands and were our three least common captures (0.46%, 0.93%, 0.23%, respectively). Furthermore, total apparent abundance was higher in natural wetlands (Z = 3.494, P < 0.001) (Table 1).

Relative density

Abundance estimates for *Peromyscus* were obtained for 24 sites (Appendix 4). Two restored sites did not have any captures of *Peromyscus*, so density at those sites was considered zero. We found no difference in *Peromyscus* relative density between restored (mean ± SE = 0.143/trap ± 0.038) and natural (mean ± SE = 0.190/trap ± 0.055) wetlands ($F_{1, 24}$ = 2.406, P = 0.134; Appendix 4).

Mass

We found no differences in the mass of deer mice between restored and natural wetlands (P ≥ 0.92; Figure 2). While no differences were significant, adult males in restored wetlands weighed less than adult males in natural wetlands; however, adult females in restored wetlands weighed more than adult females in natural wetlands. Juvenile males and females weighed more in restored wetlands than in natural wetlands (Appendix 5). Similar results were obtained for white-footed mice; there was no difference in mass between wetland types (P ≥ 0.40; Figure 2).

Adults of both sexes and juvenile females on average weighed more in restored wetlands, although juvenile males weighed more on average in natural wetlands (Appendix 5). Similarly, meadow vole mass was similar between wetland types ($P \geq 0.79$; Figure 2), although on average weighed more in natural wetlands (Appendix 5). Because northern short-tailed shrew juveniles were not captured, we only compared adult males and females between wetland types and found no difference in mass for either sex between wetland types ($P \geq 0.86$; Figure 2). On average, both sexes weighed more in natural wetlands. Eastern chipmunks followed the same trend as the northern short-tailed shrews; mass was not different between wetland types ($P \geq 0.84$; Figure 2), and on average both sexes weighed more in natural wetlands. Lastly, average mass of adult male meadow jumping mice were similar between wetland types ($P = 0.45$; Figure 2).

Occupancy

Occupancy probability was similar between wetland types for white-footed mice ($Z = 0.315$, $P = 0.753$), meadow voles ($Z = -0.287$, $P = 0.774$), northern short-tailed shrews ($Z = 0.119$, $P = 0.905$), meadow jumping mice ($Z = -0.018$, $P = 0.985$), and eastern chipmunks ($Z = 0.216$, $P = 0.828$) (Table 2).

Richness, Diversity, and Evenness

We did not find a difference between small mammal apparent species richness by wetland type (Wald Test $P = 0.117$; Figure 3). A species accumulation curve indicated that enough sites (including both restored and natural wetland sites) were sampled to discover present species (Figure 4). The species accumulation curve for restored wetland sites indicated enough restored sites were sampled to discover present species (Figure 5). However, the species accumulation curve for natural wetland sites indicated that we may have observed more species with greater sampling effort at natural wetlands (Figure 5). Additionally, we found no difference in small mammal community diversity ($F_{1, 24}$ = 0.128, P = 0.724) or evenness ($F_{1, 24}$ = 0.128, P = 0.724) between wetland types (Table 3).

Community Composition

NMDS analysis showed there is great overlap in small mammal community composition between restored and natural wetlands (Figure 6; Appendices 6–7). The stress value associated with the distance matrix used to make the plot for small mammal community was 0.19. This value is regarded as fair, but still useful, because it signifies potential for the analysis to be misleading; traditionally, stress values are most reliable when they are <0.05 (Dexter et al. 2018). However, increasing the number of dimensions to reduce stress did not significantly alter the results. The analysis of similarity indicated that small mammal community composition was similar between wetland types (R = 0.032, P = 0.226).

Vegetation

We found no difference in vegetation diversity between wetland type ($F_{1, 24}$ = 0.9694, P = 0.335; Appendix 8). Moreover, vegetation community composition was similar between wetland type (Figure 7). The stress value associated with the distance matrix was 0.18.

Discussion

Restored wetlands were largely similar to natural wetlands, though not equivalent. Two main discrepancies between wetland type were discovered, both suggesting natural wetlands were more favorable to the small mammal community. One difference was that natural wetlands supported greater apparent abundance of deer mice. Another difference was that only six species were found in restored wetlands, as opposed to nine species found in natural wetlands, suggesting natural wetlands may have a greater ability to host a wider variety of small mammals. In terms of small mammal community, restored wetlands do appear to be mostly successful in providing adequate habitat for small mammals, though are certainly not equivalent to natural wetlands.

Apparent Abundance, Relative Density, and Occupancy

We found apparent abundance of deer mice was higher in natural wetlands, and that average expected count for deer mice per 100 trap nights for natural wetlands was more than double the estimate for that of restored wetlands. This trend was also observed for small mammal total apparent abundance, as natural wetlands supported a higher average apparent abundance per 100 trap nights. However, total small mammal apparent abundance was likely driven by deer mice, as they were our most encountered species and represented 43% of unique individuals captured. Species that were captured at natural wetlands but not restored wetlands (woodland jumping mice, masked shrews, and southern flying squirrel) also contributed, in part, to this result. This result suggests that restored wetlands cannot support the abundance in numbers as natural wetlands can for the most prevalent species (deer mice), nor for the species that were captured exclusively in natural wetlands. It is possible we observed a higher apparent abundance of deer mice at natural wetlands because of competition with other species at restored wetlands, as their distribution of an area can be driven by inter-specific competition (Hallet et al. 1983); although not statistically significant, average apparent abundances for meadow voles, northern short-tailed shrews, and meadow jumping mice were higher in restored wetlands than in natural wetlands suggesting potential support for this hypothesis. Kurz et al. (2013) found no difference in relative abundance for any of their captured species between restored and reference wetlands, which is not consistent with our finding for deer mice. However, Kurz et al. (2013) did not capture deer mice in their study. Despite our observed difference in apparent abundance for deer mice, we did not determine a difference in relative density of *Peromyscus* between wetland type, possibly because understory vegetation is more important in mice density (Anderson and Meikle 2006). Apparent abundance considers deer mice and white-footed mice separately, while the relative density estimate considers both species together; although we found deer mice to be significantly more abundant in natural wetlands in the apparent abundance estimation, it is important to address that the relative density calculation uses MARK abundance estimates which take detection probability into account, creating a more meaningful estimate of abundance. Therefore, the relative density estimation could be more meaningful than the apparent abundance calculation for deer mice.

Meadow voles were bordering a significant difference in apparent abundance between wetland types; therefore, this result may be of biological significance. Meadow vole apparent abundance was 61% higher in restored than natural wetlands. Although meadow voles are a common non-obligate wetland species (Francl 2003) and have been found to be the most abundant small mammal species in another West Virginia wetland (Becker et al. 2022), our nearly significant finding may be related to wetland age. Meadow voles have a higher apparent abundance at younger wetlands (Noe et al., Chapter 3), presumably because younger wetland sites were typically palustrine emergent which better corresponds to their habitat affinity for grassland, as opposed to woody vegetation (Grant 1971; Yahner 1982). Specifically, younger wetlands (<5-yrs-old), such as McClintic and Glade Farms, had more grasses and less woody vegetation, while older sites (>10-yrs-old), such as Hazelton or Montrose, were characterized by more woody vegetation because they had more time to become established. Moreover, meadow voles had a similar occupancy probability between the two wetland types; however, their estimate of occupancy probability was higher in restored wetlands, signifying a trend that they may be more likely to occur in restored wetlands given larger sample size.

Unlike deer mice, we could not determine a difference in apparent abundance between wetland types for white-footed mice, the other *Peromyscus* species captured. Given both species are similar in that they share the same food resources and exist in similar microhabitats (Wolff et al. 1985), it was unexpected that they did not have a higher average apparent abundance in natural wetlands, like that of deer mice. This may be explained by variations in food production and climatic conditions from year to year, which could change the competitive advantage of one species over the other (Wolff 1996); this potentially leads to one species being more common than the other, and, therefore why we encountered more deer mice than white-footed mice. We may have also observed this result because of smaller sample size of white-footed mice than deer mice, as we captured more deer mice in this study. The occupancy probability estimate of white-footed mice was higher in natural wetlands; although this result was not significant it shows that white-footed mice may follow the same trend as deer mice as expected.

Northern short-tailed shrews are prevalent in this region and are not specific to wetlands or other habitat types (Webster et al. 1985; Francl 2003). Therefore, it is not surprising that we did not find a difference between wetland types. Meadow jumping mice are more habitat-specific in that they prefer moist areas (Getz 1961; Zwank et al. 1997), with one of its subspecies, *Zapus hudsonius preblei*, even considered a wetland-obligate (Trainor et al. 2012). However, no differences in apparent abundance or occupancy probability were evident between wetland type. Meadow jumping mice also exhibit an association with jewelweed (*Impatiens capensis*) (Urban and Swihart 2009), which we frequently observed in both wetland types. Although occupancy was similar between wetland types for northern short-tailed shrews and meadow jumping mice, detection probability was <0.3, creating more uncertainty in the results for these species.

Mass is commonly used to evaluate the environment of small mammals (Avenant 2011; Ofori et al. 2016). We expect to find differences in small mammal mass if food abundance, such as insect community or mast production, was different between wetland types. Because we found mass to be similar between wetland types, we can assume the environment of restored wetlands provides for small mammal communities similarly to natural wetlands. Therefore, no difference in mass between wetland types suggests that the ecological condition of wetland types is similar. Live trapping can cause the mass of small mammals to decline (Kaufman and Kaufman 1994; Pearson et al. 2003), especially for consecutive nights of trapping (Suazo et al. 2005). However, we assume this effect would be consistent in both wetland types. We evaluated and compared mass at initial capture only (not recaptures); we assumed this comparison would be a better representation of mass at wetland sites, as recaptured individuals may have experienced fluctuations in weight because of being trapped multiple times. Seasonality may also affect the mass of small mammals, specifically shrews (Merritt 1986; Taylor et al. 2013). However, we again assume we have made a fair comparison between wetland types as we trapped during the summer in both years and generally trapped at both wetland types simultaneously. Other variables such as patch size or pollution did not affect small mammal mass in previous research (Harper et al. 1993; Boonstra and Bowman 2003).

Richness, Diversity, Evenness

Despite one-third of our encountered species occurring exclusively in natural wetlands, we did not detect statistical differences in species richness, diversity, or evenness between wetland types. However, these species may not have significantly impacted the statistical analysis of richness, diversity, or evenness because their encounter histories were minute (4 masked shrews, 2 woodland jumping mice, and 1 southern flying squirrel encountered throughout the entire study). These results match that of Kurz et al. (2013), who found diversity and richness of small mammals in restored wetlands were like reference wetlands. Additionally, Juni and Berry (2001) found mammals as a taxon, not just small mammals, were similar in species richness between compensatory wetlands and natural wetlands. However, the capture of three additional species in exclusively natural wetlands suggests that restored wetlands may not be able to support a more comprehensive small mammal assemblage. Most species we captured were habitat generalists, similar to the findings of Francl et al. (2004) in high-elevation wetlands. We caught the habitat specialist, woodland jumping mice (Brannon et al. 2005). Because we only captured them in natural wetlands, it is still unclear as to whether restored wetlands can support habitat specialists. However, woodland jumping mice are not wetland-obligate species or wetland-specialists and are a specialist of another habitat type. Similarly, Francl et al. (2004) found captured habitat specialists were habitat specialists from a surrounding habitat type in high elevation wetlands. More focus and future research efforts should be given to determine if restored wetlands can support wetland-obligate species.

Most of our captured species are considered secure in their populations within the state and globally, according to NatureServe (NatureServe 2022). All species except for meadow jumping mice and woodland jumping mice are in the secure category (ranked as S5; NatureServe

2022). Meadow jumping mice, whose populations are considered vulnerable in the state (ranked as S3 by NatureServe), were found in both wetland types and were similar in apparent abundance and occupancy between wetland types. Woodland jumping mice, whose populations are uncommon but not rare in the state (ranked S4 by NatureServe), were only captured twice in the study, and both were in natural wetlands. While meadow jumping mice (an S3 species) can be supported by restored wetlands similarly to natural wetlands, woodland jumping mice (an S4 species) appear to only be supported by natural wetlands.

It may also be important to note the species we did not catch. Francl (2003) found star-nosed moles (*Condylura cristata*) and southern bog lemmings (*Synaptomys cooperi*) in addition to many of the species we captured. We expected to catch more shrews, as the Central Appalachians are known for high shrew diversity (Ford and Rodrigue 2001). Moreover, we expected to capture southern water shrews (*Sorex palustris punctulatus*). While this could indicate the ability of restored wetlands to provide for these species, we did not capture them in our natural wetlands either. We may not have seen star-nosed moles, southern bog lemmings, or southern water shrews due to their infrequency and difficulty to trap (Francl 2003).

Additionally, there may be competitive exclusion between meadow voles and southern bog lemmings (Kruppa and Haskins 1996), leading us to only observe the former. Francl (2003) captured southern bog lemmings (8 of 20 sites), although researchers acknowledged that more individuals were likely present than they could confirm by capture. The elusive southern water shrew was not captured by Francl (2003). Francl (2003) captured star-nosed moles only at two of 20 high-elevation wetland sites. While research by Francl (2003) occurred at high elevations, star-nosed moles have also been observed in elevations as low as 243 m (Simpson 1923), as well as in low elevation regions of South Carolina: the Outer and Inner Coastal Plains and Sandhill

Ecoregions (Bunch et al. 2005). Star-nosed moles were also previously observed at one of our restored sites, Glade Farms, in 2019. For rare species, such as star-nosed moles or southern water shrews, Sherman traps are not as conducive for capture as pitfall traps, which are better at capturing rare species (Umetsu et al. 2006). However, we did not use pitfall traps because they increase mortality and are challenging to use in wetlands due to high water tables; pitfall traps are seldom used in wetland surveys (Enge 2001). Although rare species were likely missed in our sampling, our species accumulation curve for species richness based on all sites suggests that we would not have observed additional species if we added more sampling sites using the same methodology. However, when we separated restored and natural wetland sites and calculated a species accumulation curve for each wetland type, we found that enough sites were sampled for restored wetlands, but it is possible that we would have observed more species at natural wetlands with additional natural wetland sites.

We ran a post-hoc analysis to assess lost trapping effort. Many falsely sprung traps had evidence of raccoon (*Procyon lotor*) presence or in some instances, black bear (*Ursus americanus*) presence which triggered traps to snap. We compared the lost trapping effort between restored and natural wetlands to assess trap disturbance between wetland types and as a proxy for predator presence using a general linear model. Lost trapping effort was similar between restored and natural wetlands, we can surmise predator presence in restored wetlands may be similar to that of natural wetlands.

There was little mortality as a result of trapping. One benefit of this is that the scent of death would not have deterred animals from entering our traps, as small mammals can be drawn to or deterred from traps by scents (Mazdzer et al. 1976; Beckmann et al. 2022). Although disinfectant may carry a scent, and traps were disinfected between sites to limit the spread hantavirus and invasives (Mills et al. 1995; Bryzek et al. 2022), we do not believe this deterred small mammals from entering our traps, as their ability to trapped does not change with disinfected traps (Yunger and Randa 1999; Van Horn and Douglass 2000; Wilson and Mabry 2011).

<u>Community Composition</u>

We found small mammal community composition was similar between wetland types. Policies like no net loss tend to overlook species composition changes because they are only concerned with totals (Xu et al. 2019). Our findings suggest this lack of oversight has not resulted in major consequences for the small mammal community, which is encouraging. Like our findings, Kurz et al. (2013) also did not discover differences in community composition when researching small mammal communities between restored and natural wetlands. Sundell et al. (2021) found differences in small mammal community composition between beaver-modified habitats and control wetlands; two species were only captured in control wetlands, wood lemmings (*Myopus schisticolor*) and yellow-necked mice (*Apodemus flavicollis*). While they did not quantify these differences in community composition using statistics, observations such as this should be addressed.

While there was community composition overlap between the wetland types, we also observed variations of species. Three of the nine species we caught were found only in natural wetlands: masked shrews, woodland jumping mice, and southern flying squirrel. It is also noteworthy that these were our three rarest species in terms of capture. One reason could be that masked shrews and woodland jumping mice may be sensitive to habitat modification. Racey and Euler (1982) found both species were sensitive to development. Potentially, both species left the wetland area when it was a degraded wetland in the years before restoration and they simply have yet to re-colonize the patch. In addition to being a sensitive species (Racey and Euler 1982), the woodland jumping mouse has strong associations with a high volume of decomposed logs (Brannon 2005). These decomposed logs are also where fungi, such as *Endogone* spp., *Melanogaster* spp., and *Hymenogaster* spp. grow, an important food source for the woodland jumping mouse (Whitaker 1962; Orrock et al. 2003; Brannon 2005). Conditions at most wetlands may not have been adequate for this fungus to grow, including many natural wetlands. One southern flying squirrel was captured once, and it was at a natural wetland. Perhaps the reason why it was captured only at a natural wetland is because the species prefers mature forests (Taulman and Smith 2004). The oldest restored wetland sampled is approaching 30 years old, which may be too young still to have mature forest.

<u>Vegetation</u>

We found vegetative diversity and community composition was similar between wetland types. This outcome was expected since we found small mammal community metrics were similar, and we know they rely heavily on their surrounding vegetation (Birney et al. 1976; Wywialowski 1987). We included assessments of both diversity and community composition because, while vegetation diversity is important to small mammals, it does not capture changes in the vegetation community over time; vegetation community composition can be entirely different but still be similarly diverse.

Our findings resonate with those of Stefanik and Mitsch (2012), who found vegetation diversity at mitigation banks did not differ between the mitigation banks and reference wetlands. Contrastingly, Campbell et al. (2002) found higher species richness of vegetation at natural wetlands than at mitigated wetlands, and Tillman et al. (2022) found plant communities at mitigation banks are higher in species richness and floristic quality than at lower quality natural wetlands; however, this does not apply to high-quality wetlands. While we did not determine a difference in vegetation diversity or community composition, it may be due to sampling a variety of lower and higher quality natural wetlands, and different ages of wetlands.

Balcombe et al. (2005c) observed a difference in vegetation communities of both mitigated and natural wetlands, with mitigated wetlands having species that are considered pioneer species, or species of low conservation value. Despite our study being conducted at some of the same wetlands, we did not find a difference in vegetation community between the two wetland types; this is likely because our studies were conducted 19–20 years apart. Additionally, as mitigated wetlands age, their vegetation community becomes like natural wetlands (Balcombe et al. 2005c). Due to the amount of time between studies, it is possible wetland age (average wetland age ± SE = 14.2 ± 2.8 years) has played a role in the similarity of the vegetative community between the two types.

Conclusion

We conclude that restored wetlands can be an adequate treatment for replacing natural wetlands in terms of small mammal communities but are not equivalent to natural wetlands. While we did not discover differences in mass, occupancy, diversity, richness, evenness, and community composition between wetland types, we did find that natural wetlands supported a higher apparent abundance of deer mice and the presence of three species that were not found at restored wetlands; this suggests species may respond to, and be supported by, restored wetlands differently. The presence and abundance of small mammals in restored wetlands is beneficial for biodiversity and serves as an important prey base for larger mammals and avian predators. Our results are encouraging because small mammals are not traditionally considered in wetland restoration efforts; however, we found that despite mostly being overlooked, they use restored wetlands similarly. Because we found our three most rarely captured species in exclusively natural wetlands and did not find wetland-obligate species, future research should aim to capture wetland-obligates and rare species in restored wetlands to discern whether they can provide quality habitat for these species. Although apparent abundance between wetland types appears to depend on the species, we conclude restored wetlands can mostly provide for small mammal communities like natural wetlands.

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Figures

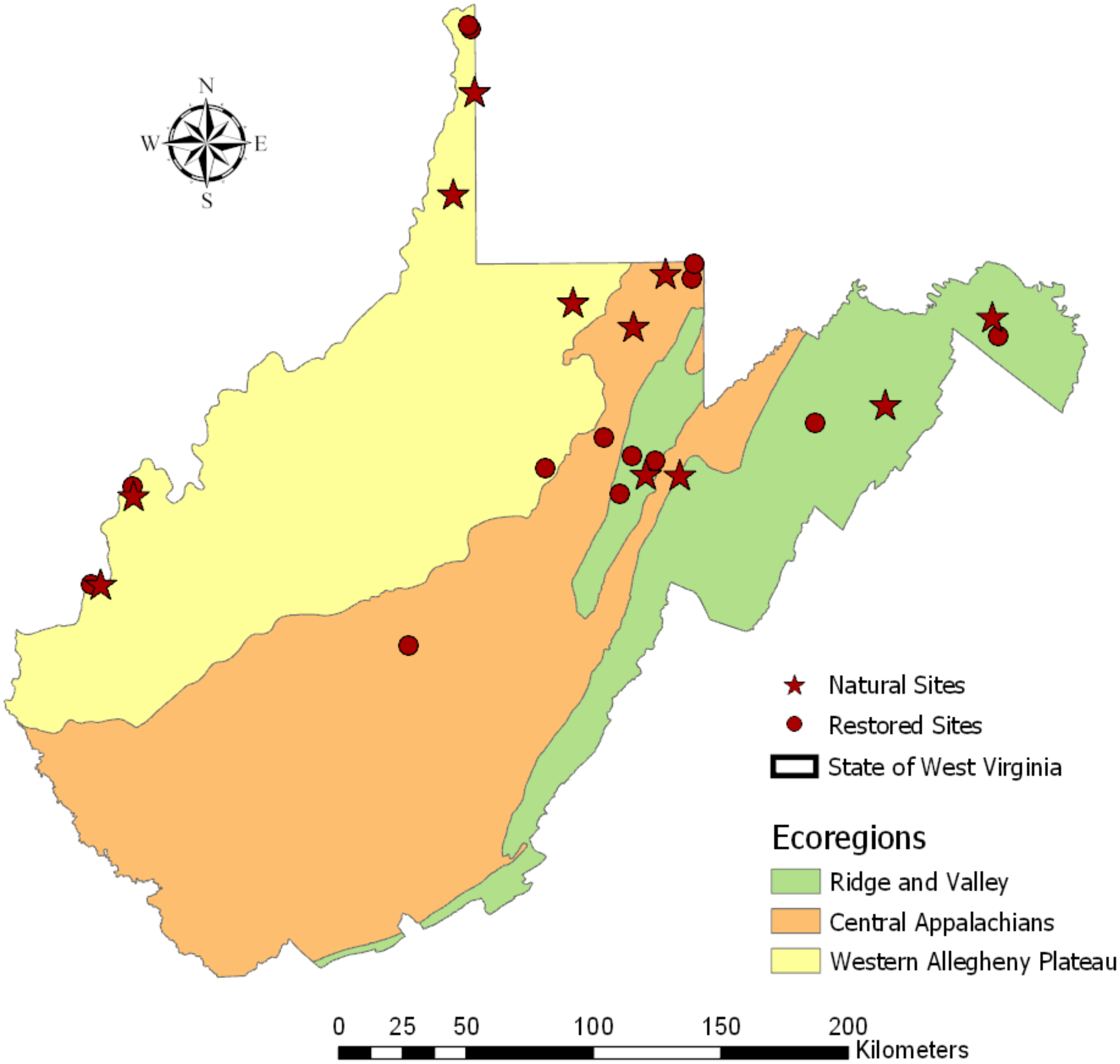

Figure 1. Twenty-six wetland sites (14 restored and 12 natural) were sampled in 2020 and 2021 for small mammal and vegetation communities throughout West Virginia, USA. Sites are plotted against the state's three ecoregions: Ridge and Valley, Central Appalachians, and the Western Allegheny Plateau.

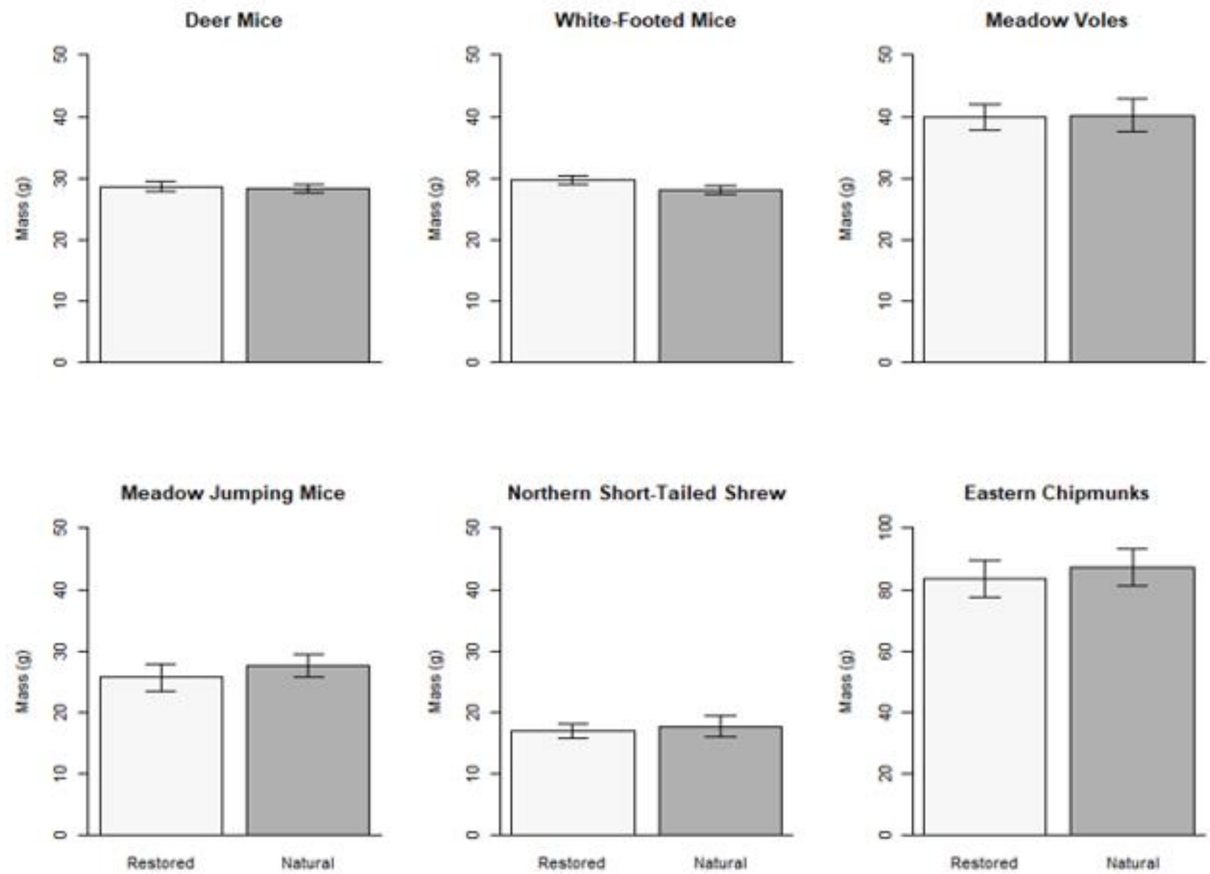

Figure 2. Average mass by species captured between both restored and natural wetlands in West Virginia, USA, in the summers of 2020 and 2021. Six species are represented: deer mice, white-footed mice, meadow voles, northern short-tailed shrews, meadow jumping mice, and eastern chipmunk. No significant difference in mass between wetland types was observed for any species. For each species, mass by sex and age is further delineated in Appendix 5.

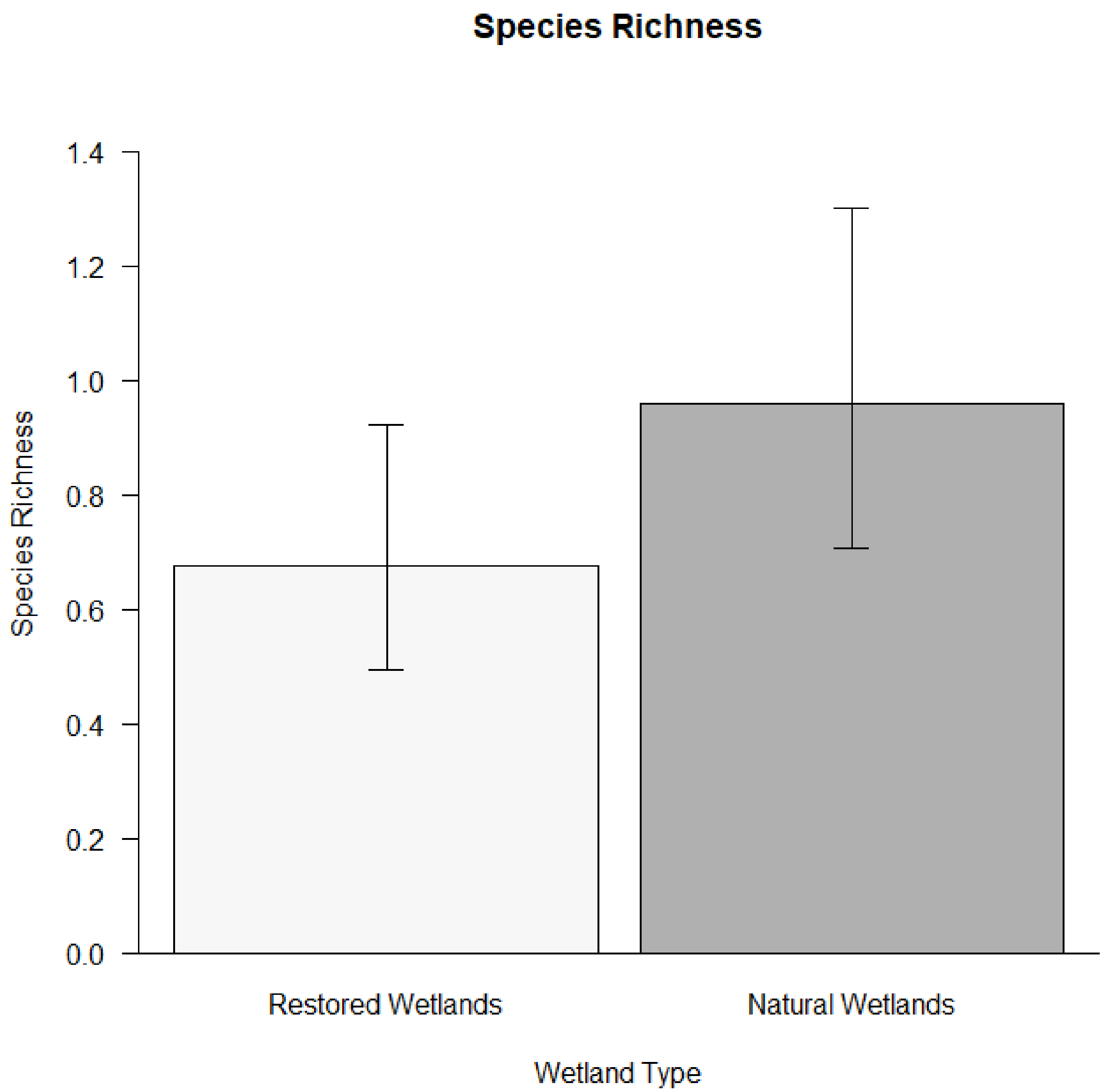

Figure 3. Small mammal species richness expected counts per 100 trap nights from generalized linear model predictions for restored wetlands (n = 14, expected count: 0.675) and natural wetlands (n = 12, expected count: 0.958) sampled in West Virginia, USA, from 2020–2021. The 95% confidence intervals for restored wetlands (0.492, 0.923) and natural wetlands (0.705, 1.300) overlapped. P-value (Wald Test P = 0.117) indicates no significant difference in expected count per 100 trap nights of species richness between wetland types.

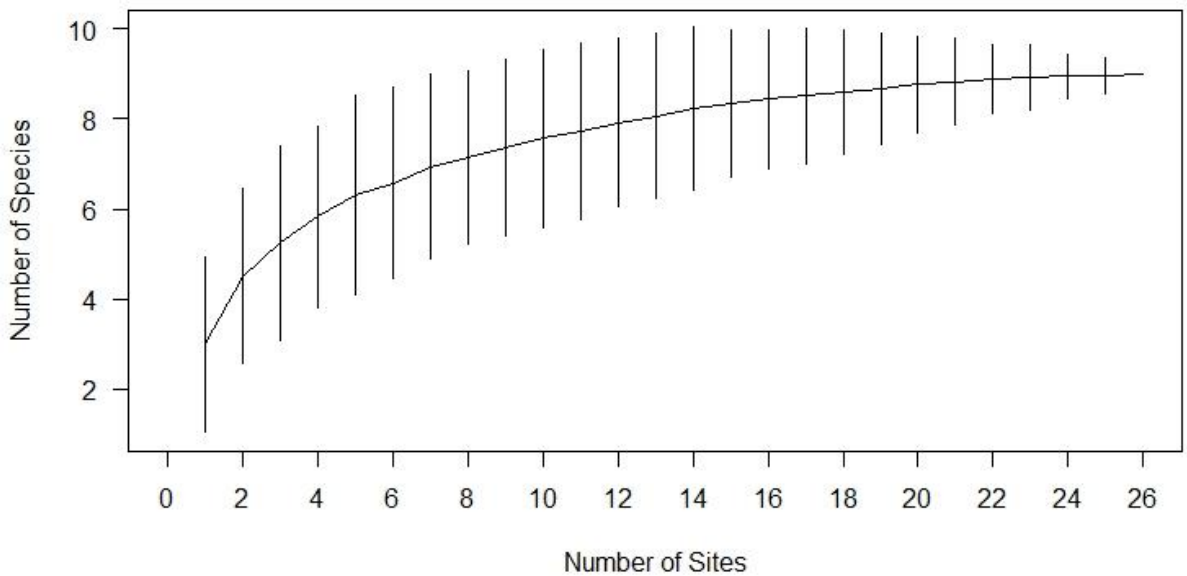

Figure 4. Species accumulation curve calculated from small mammal apparent species richness data for 26 wetland sites in West Virginia, USA, from 2020–2021. Over 26 wetland sites, apparent species richness was 9 species. The accumulation curve indicates enough sites sampled to attain apparent species richness data; sampling more sites would not have resulted in observing additional species.

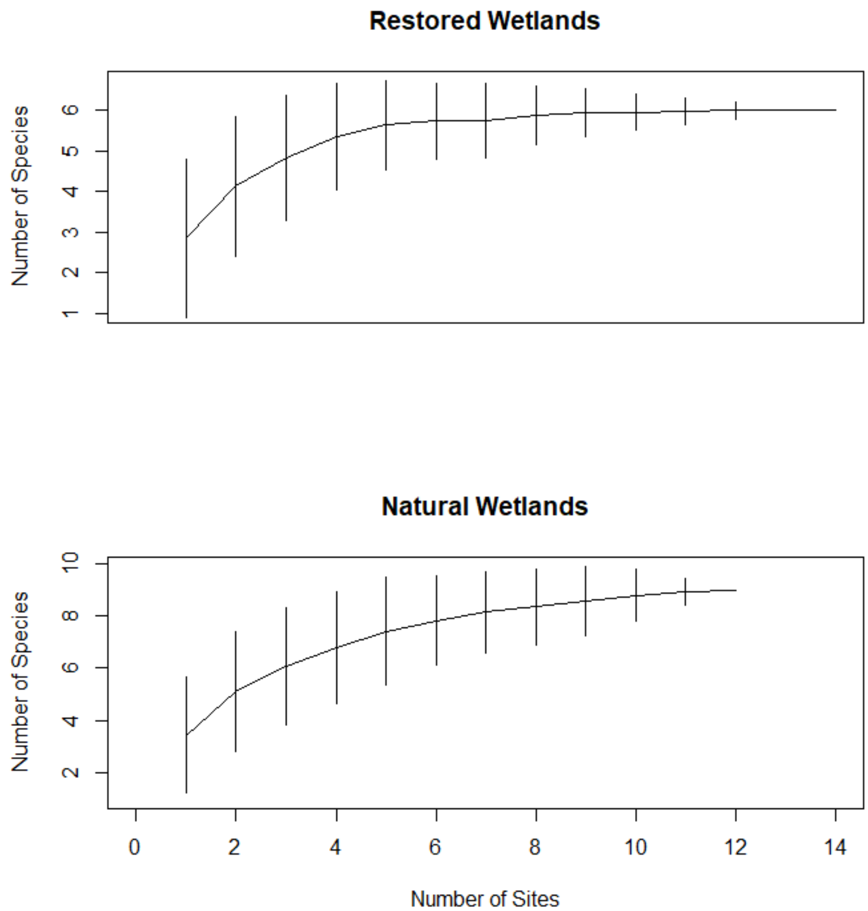

Figure 5. Species accumulation curves based on small mammal apparent species richness data for 14 restored wetlands (top) and 12 natural wetlands (bottom) in West Virginia, USA, from 2020–2021. At restored wetlands, apparent species richness was 6 species, and at natural wetlands apparent species richness was 9 species. The accumulation curve for restored wetlands indicates enough sites were sampled to attain apparent species richness data; sampling more restored sites would not have resulted in observing additional species. However, the accumulation curve for natural wetlands indicates that more sampling effort may result in more species being observed.

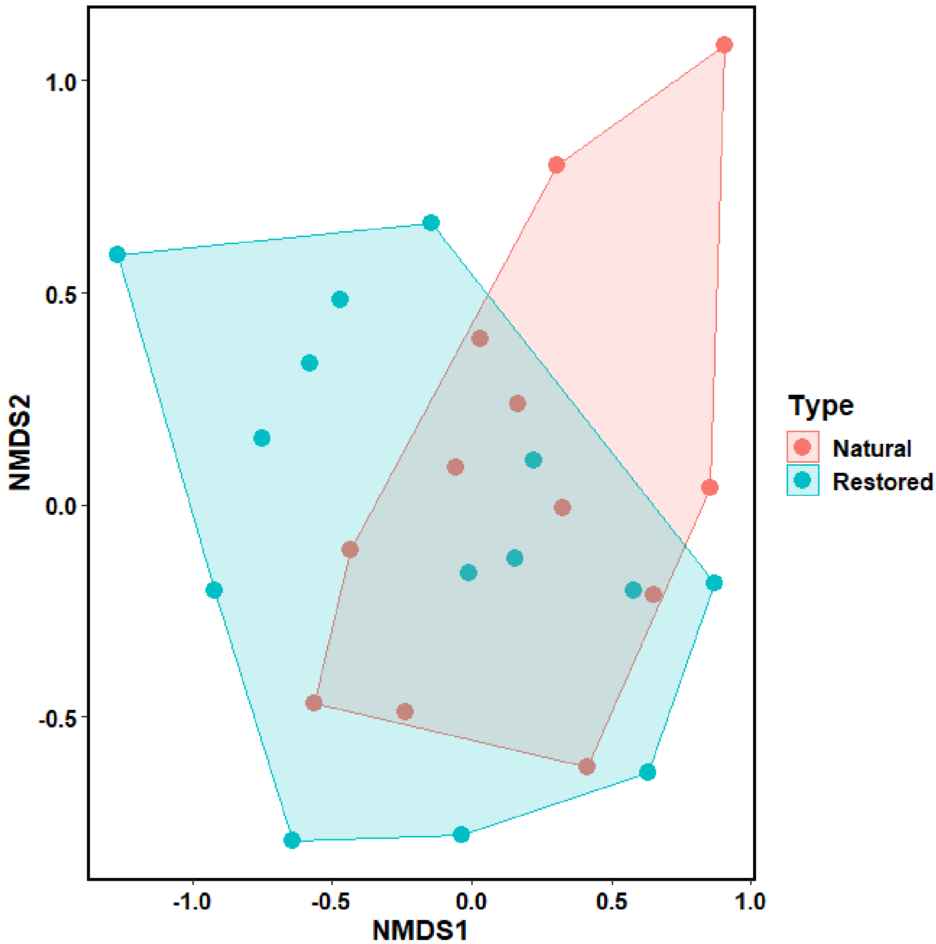

Figure 6. Non-metric multidimensional scaling (NMDS) scores for small mammal community composition from restored (n = 14) and natural (n = 12) wetland sites in West Virginia, USA, sampled from 2020–2021. Convex hulls surround restored wetland communities (blue) and natural wetland communities (pink). We specified 2 dimensions with 100 iterations. The distance matrix stress value was 0.19. Analysis of similarity indicates no significant difference in small mammal communities by wetland type (R = 0.032, P = 0.226).

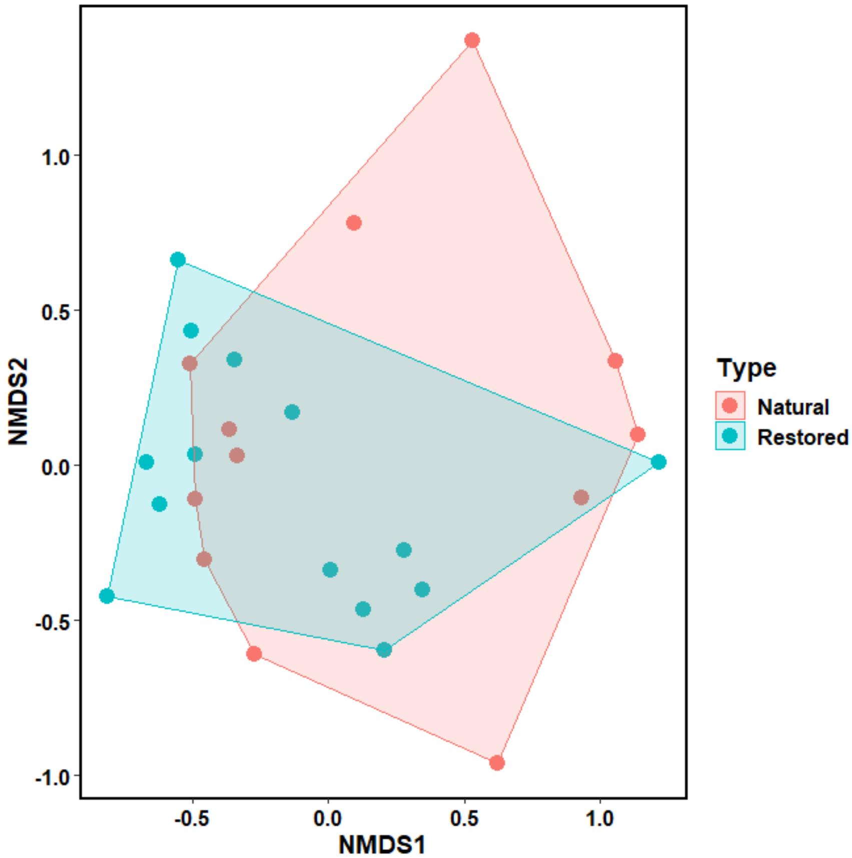

Figure 7. Non-metric multidimensional scaling (NMDS) scores for vegetation community composition from restored (n = 14) and natural (n = 12) wetland sites in West Virginia, USA, sampled from 2020–2021. Convex hulls surround restored communities (blue) and natural communities (pink). We specified 2 dimensions with 100 iterations. The distance matrix stress value was 0.18. Overlap in the graph is indicative of a similar community composition between wetland types.

Tables

Table 1. Mean and standard error of the apparent abundance of unique small mammals per 100 trap nights by wetland type sampled in the summers of 2020 and 2021 in West Virginia, USA. Over both years, 3 species (woodland jumping mouse, masked shrew, and southern flying squirrel) were only observed in natural wetlands and were excluded from the table.

	Restored (n = 14)		Natural (n = 12)	
Species	Mean	SE	Mean	SE
Peromyscus maniculatus	1.280	1.123	2.640	1.098
Peromyscus leucopus	0.968	1.142	1.285	1.144
Microtus pennsylvanicus	0.865	1.151	0.537	1.231
Blarina brevicauda	0.276	1.284	0.163	1.459
Zapus hudsonius	0.173	1.371	0.140	0.150
Tamias striatus	0.069	1.648	0.163	1.459
Total	3.633	1.071	5.093	1.070

Table 2. Occupancy probabilities by wetland type and detection probabilities for five species captured in restored (n = 14) and natural (n = 12) wetlands throughout West Virginia, USA, from 2020–2021. *Peromyscus maniculatus* was omitted from occupancy modeling because it was discovered at each site, leading to poor occupancy estimations. Species that were only captured at one wetland type were also omitted from occupancy comparison (i.e., *Napaeozapus insignis*, *Sorex cinereus,* and *Glaucomys volans* were only captured at natural wetlands).

	Occupancy Probability				Detection Probability	
	Restored		Natural			
Species	Estimate	SE	Estimate	SE	Estimate	SE
Peromyscus leucopus	0.651	0.154	0.724	0.180	0.463	0.081
Microtus pennsylvanicus	0.568	0.154	0.503	0.176	0.501	0.087
Blarina brevicauda	0.689	0.235	0.998	0.851	0.230	0.062
Zapus hudsonius	0.633	0.393	0.625	0.430	0.162	0.097
Tamias striatus	0.164	0.111	0.200	0.135	0.458	0.220

Table 3. Shannon diversity and Pielou's evenness estimates from 14 restored and 12 natural wetland sites in West Virginia, USA, using R package vegan (Oksanen et al. 2020). Diversity estimates were informed by previously calculated apparent abundance for each species at each site from data collected in the summers of 2020 and 2021 (Appendix 3).

Site	Type	Diversity	Evenness
Beaver Pond	Natural	1.292	0.588
Bruceton Mills	Natural	1.313	0.597
Buckhannon Triangle	Restored	1.384	0.630
Burches Run	Natural	1.192	0.542
Cross Creek	Natural	1.008	0.459
Fairfax	Natural	1.317	0.599
Glade Farms	Restored	1.108	0.504
Green Bottom	Natural	1.101	0.501
Hazelton	Restored	1.347	0.613
Hillcrest 1	Restored	1.350	0.614
Hillcrest 2	Restored	1.134	0.516
Hoeft Marsh	Restored	1.142	0.519
Little Indian Creek	Natural	1.310	0.596
Lower Glady	Natural	1.295	0.589
McClintic	Restored	1.132	0.515
Montrose	Restored	1.215	0.553
Nicholas	Restored	0.948	0.431
Old Town Creek	Natural	1.009	0.459
Queens	Restored	1.215	0.553
Short Mountain	Natural	1.076	0.489
Sleepy Creek	Natural	1.110	0.505
Stauffers Marsh	Restored	1.240	0.564
Sugar Creek	Restored	0.928	0.422
Three Springs	Natural	1.304	0.593
Tygart Mitigation Bank	Restored	1.222	0.556
Walnut Bottom	Restored	1.089	0.495

Appendices

Appendix 1. Physical site characteristics of 26 wetlands sampled for small mammal and vegetation communities, including both restored (n = 14) and natural (n = 12) wetlands in West Virginia, USA. Sites were sampled from June–August of 2020 (n = 10) and 2021 (n = 16). Restored wetlands ranged in age from 1 to 29 years old (mean ± SE years = 10.2 ± 2.0 years), size of all wetlands ranged from 2 to 45 hectares (mean ± SE ha = 13.4 ± 3.7 ha), and elevation of all wetlands ranged from 146 to 695 m (mean ± SE m = 429.5 ± 36.3 m). The most prevalent Cowardin et al. (1979) wetland type was listed for each site, although elements of other types were typically present at sites as well; types identified as palustrine emergent (PEM), palustrine scrub shrub (PSS), or palustrine forested (PFO). Most wetlands had silt loam soils, and all were composed of sedimentary bedrock.

Site	Type	Year Sampled	Year Made	Size (ha)	Elevation (m)	Cowardin Type	UTM Coordinates (Zone 17 N)		Geology (sedimentary)	Soil Type
							Northing	Easting		
Beaver Pond	Natural	2020	N/A	6.2	606	PFO	4314929	607738	Shale	Fluvaquents/Udifluvents complex (Fu); BB; BF; LP
Bruceton Mills	Natural	2020	N/A	1.5	512	PEM	4393355	615515	Sandstone and Shale	Atkins silt loam (At)
Buckhannon Triangle	Restored	2021	1995	2.8	435	PSS	4317185	568494	Alluvium	Udorthents, loamy (Ua)
Burches Run	Natural	2021	N/A	5.9	251	PFO	4424751	531973	Sandstone	Sensabaugh silt loam (SeA)
Cross Creek	Natural	2021	N/A	12	312	PEM	4464433	540518	Sandstone	Strip mines (Sm); Lindside silt loam (Ld); (WeD); Atkins silt loam (At)
Fairfax	Natural	2021	N/A	14.5	529	PSS	4372126	603022	Shale	(BFE) - base flood elevation (silt loam)
Glade Farms	Restored	2021	2019	11.8	630	PEM	4397093	626684	Sandstone	Atkins silty clay loam (Av)

Green Bottom	Natural	2021	N/A	15	167	PSS	4271542	393756	Alluvium	Melvin silt loam (Me); Lindside silt loam (Lm)
Hazelton	Restored	2020	2007	2	660	PSS	4390974	625712	Sandstone	Atkins silt loam (At); Tyler silt loam (TyA)
Hillcrest 1	Restored	2021	2016	6	307	PEM	4490638	538011	Sandstone	Philo silt loam (Ph); Ernest silt loam (ErC); Ernest silt loam (ErB); (BeD)
Hillcrest 2	Restored	2021	2016	9.2	330	PSS	4488993	539026	Shale	Atkins silt loam (At); (BeD); (GID)
Hoeft Marsh	Restored	2021	2020	4.88	164	PEM	4271673	389982	Alluvium	Melvin silt loam (Me); Lindside silt loam (Lm); Ashton silt loam (Asa)
Little Indian Creek	Natural	2021	N/A	3.7	294	PFO	4382126	579231	Sandstone	Lobdell silt loam (Lb)
Lower Glady	Natural	2020	N/A	4	693	PFO	4315293	622151	Shale	Fluvaquents/Udifluvents complex (Fu); (CaF)
McClintic	Restored	2021	2019	4	187	PEM	4309633	406220	Alluvium	Duncannon silt loam (DuC)
Montrose	Restored	2020	pre-97	3.2	593	PFO	4321684	602659	Alluvium	Philo loam (Ph); Tygart silt loam (Tg); Atkins silt loam (At); (Pm)
Nicholas	Restored	2020	2000	15.78	575	PSS	4247775	514458	Sandstone	Elkins silt loam drained (Ed)
Old Town Creek	Natural	2021	N/A	30	176	PEM	4306199	406683	Alluvium	Melvin silt loam (MdA)
Queens	Restored	2020	2009	2.7	541	PSS	4320054	611562	Shale	(BB); Ernest silt loam (EnC); Atkins silt loam (At)
Short Mountain	Natural	2021	N/A	45	628	PSS	4341796	701821	Sandstone	Atkins silt loam (At); Buchanan channery loam (BvC); Dekalb and Lehew (DIC)
Sleepy Creek	Natural	2021	N/A	7	332	PFO	4376125	743791	Shale	Atkins silt loam (At) / Buchanan loam - Atkins (BxC)
Stauffers Marsh	Restored	2021	1992	11.73	146	PEM	4368482	746415	Alluvium	Philo Silt loam - Atkins (Ph)
Sugar Creek	Restored	2020	1995	28.73	484	PEM	4329112	591473	Shale	Atkins silt loam, 0 to 3 percent slopes, frequently flooded (At)
Three Springs	Natural	2020	N/A	6.8	695	PSS	4314270	620954	Shale	Fluvaquents/Udifluvents complex (Fu); (MkC)
Tygart Mitigation Bank	Restored	2020	2011	5.2	585	PSS	4307018	597273	Alluvium	Atkins silt loam (At); (BkF); (Pm)
Walnut Bottom	Restored	2021	1997	6.6	335	PEM	4334561	674160	Shale	Massanetta loam (Ma); Dunning silty clay loam (Du); Tygart Silt Loam (TgA)

Appendix 2. Small mammal species captured (n = 9) in the summers of 2020 and 2021 in 26 wetlands in West Virginia, USA with scientific name, common name, and 4-letter species code used in Appendices 3, 6, and 7.

Scientific name	Common name	Code
Peromyscus maniculatus	Deer mouse	PMAN
Peromyscus leucopus	White-footed deer mouse	PLEU
Microtus pennsylvanicus	Meadow vole	MPEN
Zapus hudsonius	Meadow jumping mouse	ZHUD
Napaeozapus insignis	Woodland jumping mouse	NINS
Blarina brevicauda	Northern short-tailed shrew	BBRE
Sorex cinereus	Masked shrew	SCIN
Tamias striatus	Eastern chipmunk	TSTR
Glaucomys volans	Southern flying squirrel	GVOL

Appendix 3. Estimates of apparent abundance per 100 trap nights from a generalized linear model specifying wetland type as the predictor variable and assuming a Poisson random variable; data input was raw counts of small mammals sampled at 14 restored and 12 natural wetlands in West Virginia, USA, in the summers of 2020 and 2021. Four-digit species codes are found in Appendix 2.

	Species								
Site	PMAN	PLEU	MPEN	BBRE	ZHUD	TSTR	SCIN	NINS	GVOL
Beaver Pond	2.640	3.116	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Bruceton Mills	2.640	2.820	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Buckhannon Triangle	1.280	1.981	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Burches Run	2.640	4.510	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Cross Creek	2.640	7.504	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Fairfax	2.640	2.750	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Glade Farms	1.280	4.582	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Green Bottom	2.640	5.885	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Hazelton	1.280	2.301	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Hillcrest 1	1.280	2.276	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Hillcrest 2	1.280	4.292	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Hoeft Marsh	1.280	4.214	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Little Indian Creek	2.640	2.859	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Lower Glady	2.640	3.071	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
McClintic	1.280	4.316	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Montrose	1.280	3.478	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Nicholas	1.280	6.656	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Old Town Creek	2.640	7.485	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Queens	1.280	3.483	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Short Mountain	2.640	6.296	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Sleepy Creek	2.640	5.750	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Stauffers Marsh	1.280	3.245	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Sugar Creek	1.280	6.966	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Three Springs	2.640	2.949	0.537	0.163	0.140	0.163	9.3e-02	4.6e-02	2.3e-02
Tygart Mitigation Bank	1.280	3.415	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10
Walnut Bottom	1.280	4.791	0.865	0.276	0.173	0.069	3.9e-10	3.9e-10	1.4e-10

Appendix 4. Relative density of *Peromyscus* (including both *P. maniculatus* and *P. leucopus*). Abundance estimates were created using small mammal data collected at 26 wetlands (16 restored, 12 natural) in West Virginia, USA, from 2020–2021 and obtained from MARK. Relative density estimates were creating using MARK abundance estimates and the number of traps at each wetland site; estimates are defined as relative density since the sampling area of traps are unknown. Some sites did not have enough *Peromyscus* captures for MARK model.

Site	Type	Abundance	Traps	Relative Density
Beaver Pond	Natural	34.24	50	0.68
Bruceton Mills	Natural	8.73	50	0.17
Buckhannon Triangle	Restored	8.73	50	0.17
Burches Run	Natural	12.21	75	0.16
Cross Creek	Natural	5.24	125	0.04
Fairfax	Natural	4.07	50	0.08
Glade Farms	Restored	2	100	0.02
Green Bottom	Natural	18.01	100	0.18
Hazelton	Restored	13.37	50	0.268
Hillcrest 1	Restored	0	50	0
Hillcrest 2	Restored	0	100	0
Hoeft Marsh	Restored	18.02	100	0.18
Little Indian Creek	Natural	1	50	0.02
Lower Glady	Natural	22.66	50	0.45
McClintic	Restored	67.85	100	0.68
Montrose	Restored	9.90	75	0.13
Nicholas	Restored	1	150	0.01
Old Town Creek	Natural	13.38	125	0.11
Queens	Restored	13.38	75	0.17
Short Mountain	Natural	12.22	100	0.12
Sleepy Creek	Natural	6.40	100	0.06
Stauffers Marsh	Restored	3	75	0.04
Sugar Creek	Restored	35.41	150	0.24
Three Springs	Natural	9.90	50	0.20
Tygart Mitigation Bank	Restored	3	75	0.04
Walnut Bottom	Restored	5.24	100	0.05

Appendix 5. Average mass, standard error, and the number of individuals of six species of small mammals captured in restored and natural wetlands in the summers of 2020 and 2021 in West Virginia, USA. Categories were divided by sex and reproductive stage of individuals, as these variables influence mass.

		Deer Mice			White-footed mice			Meadow Voles			Northern Short-tailed Shrews			Meadow Jumping Mice			Eastern Chipmunks		
		n	Mean	SE	n	Mean	SE	n	Mean	SE	n	Mean	SE	n	Mean	SE	n	Mean	SE
Restored	Male	54	30.40	0.66	47	30.87	0.68	39	42.94	1.98	5	15.80	1.31	4	30.00	1.68	1	90.00	-
	Female	18	29.56	1.03	8	32.25	0.94	11	43.27	3.07	8	17.75	1.27	6	24.00	2.95	3	79.00	6.80
	Adult	67	30.67	0.55	50	31.78	0.56	40	47.27	1.38	13	17.00	0.94	7	29.57	1.52	4	81.75	5.54
	Juvenile	5	23.80	1.01	5	24.00	0.70	10	26.00	1.84	0	-	-	3	19.00	2.88	0	-	-
Natural	Male	86	30.39	0.43	44	30.68	0.58	15	38.93	2.79	5	16.60	2.42	5	27.80	2.08	1	92.00	-
	Female	29	29.55	0.78	12	28.00	1.04	8	45.50	4.33	5	18.80	1.46	0	-	-	6	84.33	6.37
	Adult	107	30.74	0.34	53	30.49	0.51	15	48.06	1.81	10	17.70	1.38	6	27.33	1.76	7	85.42	5.49
	Juvenile	8	22.62	1.33	3	23.33	0.66	8	28.37	1.88	0	-	-	0	-	-	0	-	-

Appendix 6. The number of unique individuals captured per species in restored (n = 14) and natural (n = 12) wetlands in West Virginia, USA, in the summers of 2020 and 2021. Nine species were observed in total (Appendix 2). A total of 428 individuals were captured; we had 212 total recaptures. Four-digit species codes are found in Appendix 2.

		Unique Individuals Captured								
Site	Type	PMAN	PLEU	MPEN	ZHUD	NINS	BBRE	SCIN	TSTR	GVOL
Beaver Pond	Natural	12	23	0	0	0	0	0	4	0
Bruceton Mills	Natural	6	2	1	1	0	0	1	0	0
Buckhannon Triangle	Restored	3	1	0	0	0	3	0	0	0
Burches Run	Natural	8	0	0	0	0	0	1	0	0
Cross Creek	Natural	11	0	1	0	0	1	0	0	0
Fairfax	Natural	4	1	0	0	0	0	0	0	0
Glade Farms	Restored	2	0	9	1	0	2	0	0	0
Green Bottom	Natural	15	1	16	1	0	0	0	0	0
Hazelton	Restored	3	10	0	1	0	0	0	3	0
Hillcrest 1	Restored	0	0	6	0	0	0	0	0	0
Hillcrest 2	Restored	0	0	2	3	0	5	0	0	0
Hoeft Marsh	Restored	13	3	3	0	0	0	0	0	0
Little Indian Creek	Natural	1	0	0	0	1	0	0	0	0
Lower Glady	Natural	22	0	4	0	0	2	2	3	0
McClintic	Restored	18	29	12	0	0	0	0	0	0
Montrose	Restored	6	4	0	0	0	1	0	0	0
Nicholas	Restored	1	0	0	0	0	0	0	0	0
Old Town Creek	Natural	6	6	1	4	0	1	0	0	0
Queens	Restored	10	6	0	0	0	0	0	0	0
Short Mountain	Natural	2	8	0	0	0	1	0	0	0
Sleepy Creek	Natural	17	14	0	0	0	1	0	0	0
Stauffers Marsh	Restored	3	0	0	0	0	0	0	1	0
Sugar Creek	Restored	8	1	0	5	0	0	0	0	0
Three Springs	Natural	9	0	0	0	1	1	0	0	1
Tygart Mitigation Bank	Restored	4	0	7	0	0	4	0	0	0
Walnut Bottom	Restored	3	2	11	0	0	1	0	0	0

Appendix 7. Catch-per-unit effort (CPUE) for 26 wetlands (16 restored, 12 natural) across West Virginia, USA, in 2020 (n = 10) and 2021 (n = 16) used to standardize sampling effort for a comparison of community composition between wetland type. Values are calculated as a percentage of unique individuals captured (Appendix 6), divided by the number of trap nights at a site and read as the number of individuals captured per 100 trap nights. The trap nights at each site spanned from 204.5 to 719 (mean ± SE = 386.92 ± 29.41). Four-digit species codes are found in Appendix 2.

		CPUE								
Site	Trap nights	PMAN	PLEU	MPEN	ZHUD	NINS	BBRE	SCIN	TSTR	GVOL
Beaver Pond	242.5	4.948	9.485	0.000	0.000	0.000	0.000	0.000	1.649	0.000
Bruceton Mills	219.5	0.834	0.278	0.139	0.139	0.000	0.000	0.139	0.000	0.000
Buckhannon Triangle	204.5	1.467	0.489	0.000	0.000	0.000	1.467	0.000	0.000	0.000
Burches Run	351	2.279	0.000	0.000	0.000	0.000	0.000	0.285	0.000	0.000
Cross Creek	584	1.884	0.000	0.171	0.000	0.000	0.171	0.000	0.000	0.000
Fairfax	214	1.869	0.467	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Glade Farms	473	0.423	0.000	1.903	0.211	0.000	0.423	0.000	0.000	0.000
Green Bottom	458	3.275	0.218	3.493	0.218	0.000	0.000	0.000	0.000	0.000
Hazelton	237.5	1.263	4.211	0.000	0.421	0.000	0.000	0.000	1.263	0.000
Hillcrest 1	235	0.000	0.000	2.553	0.000	0.000	0.000	0.000	0.000	0.000
Hillcrest 2	443	0.000	0.000	0.451	0.677	0.000	1.129	0.000	0.000	0.000
Hoeft Marsh	435	2.989	0.690	0.690	0.000	0.000	0.000	0.000	0.000	0.000
Little Indian Creek	222.5	0.449	0.000	0.000	0.000	0.449	0.000	0.000	0.000	0.000
Lower Glady	239	9.205	0.000	1.674	0.000	0.000	0.837	0.837	1.255	0.000
McClintic	445.5	4.040	6.510	2.694	0.000	0.000	0.000	0.000	0.000	0.000
Montrose	359	1.671	1.114	0.000	0.000	0.000	0.279	0.000	0.000	0.000
Nicholas	687	0.146	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Old Town Creek	582.5	1.030	1.030	0.172	0.687	0.000	0.172	0.000	0.000	0.000
Queens	359.5	2.782	1.669	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Short Mountain	490	0.408	1.633	0.000	0.000	0.000	0.204	0.000	0.000	0.000
Sleepy Creek	447.5	3.799	3.128	0.000	0.000	0.000	0.223	0.000	0.000	0.000

Stauffers Marsh	335	0.896	0.000	0.000	0.000	0.000	0.000	0.000	0.299	0.000
Sugar Creek	719	3.645	0.456	0.000	2.278	0.000	0.000	0.000	0.000	0.000
Three Springs	229.5	3.922	0.000	0.000	0.000	0.436	0.436	0.000	0.000	0.436
Tygart Mitigation Bank	352.5	1.135	0.000	1.986	0.000	0.000	1.135	0.000	0.000	0.000
Walnut Bottom	494.5	0.607	0.404	2.224	0.000	0.000	0.202	0.000	0.000	0.000

Appendix 8**.** Vegetation diversity estimate based on percent cover of each species in ≥ 50 1 x 1 m^2 quadrats per site at restored (n = 14) and natural (n = 12) wetlands in West Virginia, USA in 2020 and 2021. Percent cover of each species was calculated by summing the occurrences of each species within a Daubenmire cover class (1 = 0–5%, 2 = 6–25%, 3 = 26–50%, 4 = 51–75%, 5 = 76–95%, and 6 = 96–100%; Daubenmire 1959), multiplying this number by a mid-point (2.5, 15, 37.5, 62.5, 85.5, 97.5) for each cover class, summing products of all cover classes to obtain a total, then dividing the sum by the number of quadrats present at the site.

Wetland	Type	Vegetation Diversity
Beaver Pond	Natural	2.163
Bruceton Mills	Natural	1.419
Buckhannon Triangle	Restored	3.481
Burches Run	Natural	2.904
Cross Creek	Natural	3.311
Fairfax	Natural	3.150
Glade Farms	Restored	2.997
Green Bottom	Natural	3.344
Hazelton	Restored	2.736
Hillcrest 1	Restored	3.361
Hillcrest 2	Restored	3.392
Hoeft Marsh	Restored	3.261
Little Indian Creek	Natural	3.427
Lower Glady	Natural	2.129
McClintic	Restored	3.035
Montrose	Restored	2.790
Nicholas	Restored	2.234
Old Town Creek	Natural	3.331
Queens	Restored	2.059
Short Mountain	Natural	2.154
Sleepy Creek	Natural	3.246
Stauffers Marsh	Restored	3.332
Sugar Creek	Restored	2.888
Three Springs	Natural	1.726
Tygart Mitigation Bank	Restored	2.502
Walnut Bottom	Restored	2.799

CHAPTER 3

Restored wetland size and age influence small mammal communities in West Virginia, USA

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This chapter is written in the style of *Wetlands*.

Abstract

Small mammals are important, albeit often overlooked, fauna in wetland restoration projects. However, it is essential to evaluate factors that may determine small mammal community metrics in restored wetlands to maximize wetland restoration effectiveness. Previous studies found vegetation to differ as restored wetlands age, and that wetland age may play a role in the presence of amphibians and birds. Therefore, we assessed whether small mammals were influenced by wetland age. Because small mammals are vital to wetland systems, we also evaluated 17 environmental factors in restored wetlands that could influence small mammal communities at these wetlands. To evaluate the age and environmental factors on the small mammal community, we assessed 14 restored wetlands in West Virginia, USA, in the summers of 2020 and 2021 for small mammal community metrics; specifically, species apparent abundance, occupancy, diversity, richness, and evenness. We found apparent abundance of deer mice (*Peromyscus maniculatus*), white-footed mice (*Peromyscus leucopus*), and meadow voles (*Microtus pennsylvanicus*) decreased with wetland age. However, both species diversity and evenness increased with wetland age. We found wetland size influenced the apparent abundance of white-footed mice, meadow jumping mice (*Zapus hudsonius*), and all small mammals combined. Although, white-footed mice and total small mammal apparent abundance decreased with wetland size, while the apparent abundance of meadow jumping mice increased with wetland size. There was model selection uncertainty for the apparent abundance of northern short-tailed shrews (*Blarina brevicauda*) and eastern chipmunks (*Tamias striatus*), occupancy of any species, and diversity, richness, and evenness. Therefore, wetland managers need to consider wetland age and size when designing wetlands to facilitate small mammal communities.

Keywords: Appalachia, deer mice, meadow voles, *Microtus pennsylvanicus, Peromyscus leucopus, Peromyscus maniculatus,* restoration age, wetland age, wetland creation, wetland mitigation, wetland restoration, wetland size, white-footed mice

Despite the numerous ecosystem services wetlands provide (Millennium Ecosystem Assessment 2005), wetland loss occurred throughout much of U.S. history and still occurs today, albeit at a slower rate (Dahl and Allord 1996; Hu et al. 2017). Many programs exist to ameliorate the problem of wetland loss. The U.S. Department of Agriculture's Natural Resource Conservation Service's Wetland Reserve Program, now known as the Agricultural Conservation Easement Program, functions to restore and conserve wetlands by working with private landowners (USDA 2022). Similarly, the Partners for Fish and Wildlife Program of the U.S. Fish and Wildlife Service works with private landowners to restore habitats for fish and wildlife, including wetlands (Benson et al. 2018). Furthermore, the U.S. Environmental Protection Agency recommends voluntary wetland restorations and provides grants to aid this endeavor, such as the Five Star/Urban Waters Restoration Program grant (EPA 2022). Wetland restoration is also performed by non-governmental organizations such as Ducks Unlimited (Tori et al. 2002). Apart from voluntary wetland restoration efforts, legislation to compensate for wetland losses, such as the "no net loss policy" of 1989 and Clean Water Act of 1977, has prompted the widespread use of wetland mitigation to replace lost wetland function (Turner et al. 2001).

There are many considerations for wetland restoration, including re-establishing native vegetation (Kettenring and Tarsa 2020), wetland location on the landscape (Van Lonkhuyzen et al. 2004), capacity for biogeochemical functioning (Moreno-Mateos et al. 2012), and providing sufficient wildlife habitat (Kurz et al. 2013). Considerations for wetland mitigation include functional and economic loss from lag time (Zedler and Callaway 1999; Gutrich and Hitzhusen

2004; Bendor 2009) and proper replacement of vegetation (Brown and Veneman 2001; Spieles et al. 2006).

Small mammals are an especially important taxa to assess in wetlands because they are key seed dispersers (Brewer and Rejmanek 1999; Brehm et al. 2019), useful bioindicators (Pearce and Venier 2005; Leis et al. 2008), vegetation influencers (Root-Bernstein and Ebensperger 2013), and impact presence of other species due to their role as prey for higher-trophic level wildlife species (Korpimaki and Norrdahl 1991; Haas 2009). Because of their critical role in ecosystems, it is vital to know variables affecting small mammal communities in restored wetlands. These variables include environmental factors within the wetland, and external landscape variables.

Wetland age may influence small mammal communities in restored wetlands. Age influences invertebrate communities (Fairchild et al. 2000; Swartz et al. 2019), soil properties and nutrients (Stolt et al. 2000; Campbell et al. 2002; Wolf et al. 2011; Ahn and Jones 2013), and microtopography (Wolf et al. 2011), all of which can, in turn, affect small mammal communities (Sieg 1987; Whittsitt and Tappe 2009). Small mammals have an especially close association with vegetation complexity and protective cover (Birney et al. 1976), which also can change with wetland age (Campbell et al. 2002; Stefanik and Mitsch 2012). The effect of vegetation may be species-specific, as habitat generalists like deer mice (*Peromyscus maniculatus*) may be able to thrive in less diverse vegetation, while habitat specialists like southern red-backed voles (*Myodes gapperi*) have a greater reliance on vegetative structural diversity (Wywialowski 1987). In Appalachia, high elevation wetlands (>730 m) are composed mainly of habitat generalists (Francl et al. 2004); the same seems to be true of lower elevation wetlands (<730 m), as Becker et al. (2022) found a herbaceous riparian wetland dominated by meadow voles (*Microtus* *pennsylvanicus*), a common habitat generalist. Kurz et al. (2013) also captured more meadow voles than any other species at restored and reference wetlands in Ohio. Moreover, canopy openness can positively affect the capture probability of deer mice (Weldy et al. 2019) and occupancy of shrew species and meadow voles (Cassel et al. 2020), while canopy cover positively influences occupancy of eastern chipmunks (*Tamias striatus*), gray squirrels (*Sciurus carolinensis*) and white-footed mice (*Peromyscus leucopus*) (Cassel et al. 2020). These species-specific interactions with vegetation may determine small mammal presence in restored wetlands by age because there is a significant difference in vegetation in young versus old wetlands (Campbell et al. 2002), with younger wetlands having higher vegetation species richness than older wetlands (Stefanik and Mitsch 2012). Although, as wetlands age, they become similar in vegetation community composition to that of reference wetlands (Balcombe et al. 2005). Moreover, commonly used wetland vegetation-based indicators show positive results for the first 5–10 years following restoration, the traditional monitoring period for mitigation (Zedler and Callaway 1999), but then declined after 10 years (Matthews et al. 2009); thus, reinforcing the notion that age plays a role in wetland restoration.

The association between wetland age and other wildlife has previously been investigated. While some researchers have failed to find a correlation of wetland age and anuran and amphibian communities (Porej and Hetherington 2005; Denton and Richter 2013), other researchers have determined the influence of wetland age may affect amphibian species occupancy differently (Birx-Raybuck et al. 2010; Oja et al. 2011). For birds, wetland age may not affect the average number of all bird species, waterfowl species, and breeding waterfowl species, but older restored wetlands had a significantly higher average number of breeding birds than younger wetlands (VanRees-Siewert and Dinsmore 1996). Moreover, wetlands made 12 years apart could provide habitat for winter waterbirds similarly, regardless of age; however, species composition differed between the two wetlands (Clipp et al. 2017). Although not a wetland study, Mulligan et al. (2013) found small mammal community structure to become more similar with age following a grassland restoration. While studies on wildlife have considered wetland age for some species and grassland age for small mammal communities, this age-related concern has not yet been extended to small mammals in wetlands.

External landscape features may also influence small mammal communities in restored wetlands, including roads, recreational trails, railways, and powerlines. While roads may function as barriers in dispersal (Oxley et al. 1974; Richardson et al. 1997; Clark et al. 2001; McGregor et al. 2008), they may also have positive effects on small mammals, potentially because they have negative impacts on several predators of small mammals (Rytwinski and Fahrig 2007). The effect of roads on small mammals may be species-specific, as meadow voles, northern short-tailed shrews (*Blarina brevicauda*), and masked shrews (*Sorex cinereus*) were associated with higher road density (Francl et al. 2004). Although, the effect of roads may depend on the season (Rytwinski and Fahrig 2007). Roads may be a crucial factor to consider, as many mitigated wetlands in West Virginia have been built adjacent to major roads and highways, potentially influencing small mammal communities at these wetlands. Another landscape variable that affects small mammal communities is railways, which may deter predators away from the area due to the noise from trains, thus influencing small mammal communities (Cerboncini et al. 2016).

It is important to know the variables that most affect small mammal communities at wetland sites before and after wetland restoration occurs. Although not all wetland restoration projects are for mitigation, many mitigated wetlands are restored wetlands; since the goal of mitigation is to replace the functionality of naturally occurring wetlands, including their important role in habitat provisioning restored wetlands must be evaluated. This research aimed to determine features of restored wetlands that may affect small mammal communities. Our objectives were to analyze the effects of restored wetland age and environmental variables (vegetation, landscape, and wetland characteristics) on small mammal 1) abundance, 2) occupancy, 3) diversity, 4) richness, and 5) evenness across a range of wetland ages (1–29 years). We hypothesized small mammal communities would decrease in all metrics as wetlands age because they are reliant on vegetation (Wywialowski 1987), and vegetation richness at younger wetlands is higher (Stefanik and Mitsch 2012). Likewise, due to the reliance of small mammals on vegetation (Wywialowski 1987), we hypothesized that vegetative variables would have the greatest effect on small mammal metrics in restored wetlands.

Methods

Study area

We sampled 14 restored wetlands of differing ages in West Virginia, USA, across three ecoregions: Ridge and Valley (n = 5), Central Appalachians (n = 4), and Western Alleghany Plateau (n = 5) (Figure 1). Landscape throughout the state includes agricultural valleys, hills, forested ridges, and high elevation areas (Woods et al. 1999). Mean annual precipitation ranges from its highest at 1,180 mm in the Central Appalachians to its lowest at 1,063 mm in the Western Alleghany Plateau (Wilken et al. 2011). Sampled wetlands were palustrine emergent (n = 7), scrub-shrub (n = 6), and forested (n = 1), but typically exhibited traits of all three dominant vegetation types (Cowardin et al. 1979). Many palustrine emergent wetlands were younger, although this habitat type was not exclusive to young wetlands; four of five wetlands ≤5 years old were classified as palustrine emergent. Wetlands were primarily restored although some small patches were potentially created, and 12 of the 14 were mitigated wetlands specifically restored to offset natural wetland losses. Sampled wetlands are owned by the West Virginia Division of Natural Resources (WVDNR) ($n = 4$), West Virginia Division of Highways (WVDOH) ($n = 6$), the U.S. Forest Service ($n = 1$), the Potomac Valley Audubon Society (PVAS) ($n = 1$), and privately owned ($n = 2$). Wetlands were between 2 and 28.7 ha in area (mean ± SE ha = 8.1 ± 1.9 ha). Mean elevation at wetlands was 426.5 m (± SE = 48.2) and ranged from 146 to 660 m. Wetland age ranged from 1 year to 29 years old at the time of sampling (mean ± SE = 14.2 ± 2.8 years); the oldest sites were established in 1992, while the youngest site was established in 2020.

Small mammal trapping

We conducted small mammal trapping along 240 m-long transects using 5.08 cm × 6.35 cm × 16.51 cm folding Sherman Live Traps (H.B. Sherman Traps, Inc, Tallahassee, FL, USA) placed 10 m apart on each transect. We placed transects 50 m apart from each other. Each wetland had a minimum of 2 transects, although to sample larger wetlands, we included up to six transects (mean = 3.57; SE = 0.34) depending on size. We checked traps each morning (≤ 24 hours), during trapping sessions consisting of 5 consecutive nights.

We baited traps with peanut butter and oats wrapped in wax paper and replaced them throughout the trapping session as needed (Edalgo and Anderson 2007). We added cotton to traps to enhance survival (Szebor and Strubel 2013). When small mammals were captured in traps, we first checked for pre-existing tags or marks from previous trap nights; if there were none, we marked all (except for shrews) with #1005-1 Monel ear tag (National Band and Tag

Company, Newport, Kentucky, USA) on the left ear and recorded species, mass, length of body and tail, sex, and reproductive condition (Bruseo et al. 1999; Glennon et al. 2002). Shrews received a unique pattern of dots made with hair dye to identify individuals (Craig 1995; Stromgren 2008), due to the tendency of an ear tag to damage their ears and quickly get lost (Craig 1995). We distinguished between deer mice and white-footed mice in the field by evaluating proportions of tail to body length (deer mice have a longer tail in proportion to their bodies), tail hair density (white-footed mice have a sparsely haired tail in comparison to deer mice), and distinct bicoloration on their tail (for deer mice) (Kays and Wilson 2009). Between sites, we cleaned equipment to avoid the potential spread of invasive species and diseases among wetlands (Bryzek et al. 2022).

Gathering environmental variables

To collect vegetation data at each site, we used 1 × 1 m quadrats along our transects and positioned them, so each trap was in the center of the quadrat (25 quadrats per transect). In each quadrat, we identified herbaceous vegetation to species and estimated its cover using Daubenmire (1959) cover classes (1–5%, 6–25%, 26–50%, 51–75%, 76–95%, and 96–100%), determined average tree and shrub canopy cover in the quadrat to the nearest percent by using a spherical densiometer at each of the four corners, and measured water and leaf litter depth to the nearest cm at the deepest point within the quadrat. Using the percent coverage of each plant species at each wetland, we calculated Shannon diversity using the package vegan in R (Oksanen et al. 2020).

Additionally, we used a combination of public shapefiles from environmental datasets and West Virginia Wetland Rapid Assessment [WVWRAM] scores to determine other environmental variables that may affect small mammal communities in restored wetlands. Scores obtained for the WVWRAM were created using both remote sensing data and data from site visits (West Virginia Department of Environmental Protection 2020; Table 1). WVWRAM scores included metrics of road and rail, wetland condition, habitat function, habitat function without biodiversity rank, habitat condition, habitat potential, floristic quality, vertical vegetation structure, woody vegetation, and wetland breeding bird occupancy. On average, WVWRAM site visits to inform scores occurred 1.36 years (± 0.36 years) before our site visits (Appendix 1). On four occasions, WVWRAM field assessments came after our field sampling, although these were always ≤ 2 months later. We also used the National Wetlands Inventory (NWI) GIS layer created by the U.S. Fish and Wildlife Service to assess wetland size and type of wetlands (Cowardin et al. 1979; U. S. Fish and Wildlife Service 2014). Because the NWI layer is not always reflective of actual wetland presence or type (Matthews et al. 2016), GIS data were corroborated by site visits. We obtained the landscape integrity metric using GIS data created using distances that were weighted by different anthropogenic landscape features (Dougherty and Byers 2008). Additionally, we used WVWRAM scores collected, calculated, and provided by the West Virginia Department of Environmental Protection for our restored wetlands; scores included assessments of wetland condition, function, and vegetation (West Virginia Department of Environmental Protection 2020; Table 1).

Statistical analysis

<u>Age</u>

Apparent abundance

We estimated the apparent abundance of small mammal species at each site for all captured species (n = 6) and total small mammals using count data from unique individuals. We calculated apparent abundance with a generalized linear model specifying count data as our response variable and wetland age as our predictor variable. We assumed a Poisson random variable and included trapping effort as an offset in our model to account for different trapping efforts across sites. To implement species apparent abundance models, we used the glm function in R statistical software (R Core Team 2022). Our hypothesis test was interpreted with a type 1 error rate of 0.05.

Occupancy

We fit single-season occupancy models described by Mackenzie et al. (2002) for five species; occupancy models for the total number of small mammals and deer mice were excluded due to their detection at each site, resulting in poor occupancy model estimates. Detection was collapsed to 1 if a species was detected at a transect, and 0 if it was not. We treated wetland age as our site-level covariate and detection covariate was held constant. To implement these models, we used 'unmarked' (Fiske and Chandler 2011) in R statistical software (R Core Team 2022).

Richness, diversity, and evenness

We first determined species richness as the number of species observed at each site. Species richness was then used to estimate Shannon-Weiner diversity:

$$H_j' = -\sum_{i=s_j}^{s} p_i \ln{(p_i)}$$

Where Hj' is estimated diversity for a site, 'sj' is the number of species at a site, and 'pi' is the proportion of each abundance relative to other species present at a site (Shannon 1948). We implemented this equation using ‘vegan’ and specifying the Shannon diversity index (Oksanen et al. 2020). Additionally, we calculated Pielou’s evenness index (J) for each site using our previously estimated diversity from each site (H') and total species richness (S) (Pielou 1966): $J = H' / \ln(S)$.

Apparent species richness was assessed using a generalized linear model assuming a Poisson random variable to determine if wetland age, our predicator, had an effect. We included an offset within the model to consider different trapping efforts among sites. Our hypothesis test had a type 1 error rate of 0.05. To determine the potential effect of age on diversity and evenness, we created general linear models (ANOVA analysis) with wetland age as the predictor variable and diversity and evenness as response variables. Again, these models had a type 1 error rate of 0.05.

Model selection of environmental variables

We first created generalized linear models for the apparent abundance of each species. For each species, 18 models were created with apparent abundance as our response variable; each of the 18 models had a different environmental predictor variable. We summarized these environmental predictor variables at the wetland scale. These environmental variables include wetland age, size, wetland type, assessment of vegetation community, average canopy cover, landscape integrity, ecoregion, and WVWRAM scores: road and rail, condition, habitat function, habitat function without biodiversity rank, habitat condition, habitat potential, floristic quality, vertical vegetation structure, woody vegetation, and wetland breeding bird occupancy (Table 1). To estimate the similarity of vegetation community among sites, we used a principal component analysis (PCA). We used the first score provided by the PCA analysis, as this is the score that explains most of the variation in the data. We used the “prcomp” function in R to obtain this score. We also included an intercept-only model for later model comparison. We assumed a Poisson random variable for apparent abundance models and standardized different site trapping efforts by including an offset within the model.

Next, we constructed occupancy models for 5 species: white-footed mice, meadow voles, meadow jumping mice, northern short-tailed shrews, and eastern chipmunk. As with apparent abundance, we created 18 occupancy models for each species; occupancy probability was the response variable in each model. However, the predictor variable would be one of the 18 environmental variables previously described (Table 1) for each model. Occupancy models were constructed as previously described, with detection collapsed as 1 if the species was detected at a transect and 0 if it was not and implemented using ‘unmarked’ (Fiske and Chandler 2011).

We then created generalized linear models for apparent species richness, with apparent species richness per site as our response variable for all models, and each of the 18 different environmental variables as our predictor variables, totaling 18 models for species richness. We assumed a Poisson random variable and included an offset of trapping effort in each model to account for different trapping efforts among sites.

To fairly evaluate diversity and evenness across all environmental variables, we first determined catch-per-unit effort (CPUE) for each species at each site. This CPUE had to be calculated differently from apparent abundance models, where we used an offset to standardize trapping effort across sites and obtained an abundance estimate that was influenced by environmental variables all in one cohesive model; here we needed to have a CPUE that was not influenced by any factor to fairly calculate diversity and evenness, but still fairly adjusted to accommodate for different trapping effort across sites. CPUE was defined as the count of unique individuals per 100 trap nights (Nicolas and Colyn 2006). Shannon-Weiner diversity was calculated for each site using CPUE with 'vegan' (Oksanen et al. 2020) in R statistical software (R Core Team 2022). Pielou's evenness (*J*) for each site was calculated from site diversity estimates and the equation: $J = H' / \ln(S)$, where *H'* is diversity and *S* is total species richness (Pielou 1966).

To assess environmental influence on diversity, we created 18 general linear models, all with site diversity as the response variable and a different environmental variable (predictor variable) for each of the 18 models. We created models for evenness similarly, but with site evenness as the response variable in all evenness models.

We then used the Akaike information criterion (AIC) to compare models and perform model selection for the top environmental variable (Akaike 1973), specifically AICc, a version of AIC that is corrected for small sample sizes (Brewer et al. 2016). We were limited to one predictor variable per model because of the 1:10 rule (1 predictor variable: 10 samples) (Steyerberg et al. 2000). We defined top models as models with confidence intervals that did not overlap zero and had model selection certainty ($\Delta_i < 2$). We selected models using 'AICcmodavg' (Mazerolle 2020) with R statistical software (R Core Team 2022).

Results

During 5,780 trap nights at 14 restored wetlands from June to August 2020 (n = 6) and 2021 (n = 8), we captured 210 unique individuals. We captured six species of small mammals: deer mice, white-footed mice, meadow voles, meadow jumping mice (*Zapus hudsonius*), northern short-tailed shrews, and eastern chipmunks; deer mice were the most captured species (35% of all unique captures).

Age

Apparent abundance

Apparent abundance decreased with wetland age for deer mice ($Z = -2.543$, $P = 0.01$), white-footed mice ($Z = -4.415$, $P < 0.01$), meadow voles ($Z = -4.108$, $P < 0.01$), and total small mammals ($Z = -6.01$, $P < 0.01$) (Appendix 2). Apparent abundance of meadow jumping mice, northern short-tailed shrews, and eastern chipmunks were unaffected by wetland age ($P > 0.05$).

Occupancy

Wetland age affected the occupancy probability of meadow voles ($Z = -2.15$, $P = 0.03$), which had lower occupancy probabilities as wetland age increased (Figure 2; Appendix 3). Wetland age did not affect the occupancy probabilities of white-footed mice ($Z = 0.683$, $P = 0.494$), meadow jumping mice ($Z = -0.622$, $P = 0.534$), northern short-tailed shrews ($Z = 0.2613$, $P = 0.794$), or chipmunks ($Z = 0.959$, $P = 0.337$) (Figure 2; Appendix 3).

Diversity, richness, and evenness

Diversity (Figure 3A) and evenness (Figure 3B) increased with wetland age ($F_{1, 12} = 1259$, $P < 0.01$). Mean expected richness of wetlands per 100 trap nights was 0.67 (± SE = 1.22) and was not influenced by wetland age ($Z = -0.299$, $P = 0.765$; Figure 4).

Environmental variables and models selected

Apparent abundance

The top model for white-footed mice suggested greater apparent abundance in smaller wetlands, with no competing models (Appendix 5A). Likewise, total small mammal apparent abundance was greater in smaller wetlands, with no competing models (Appendix 5B). The top model for meadow jumping mice suggested greater apparent abundance at larger wetlands, again with no competing models (Appendix 5C). Meadow vole apparent abundance was best explained by decreasing canopy over with no competing models (Appendix 5D). The apparent abundance of deer mice decreased with vegetation community similarity, with no competing models (Appendix 5E). The vegetation community was also the top model for the apparent abundance of northern short-tailed shrews, although this model was in competition with wetland type. Unlike deer mice, the apparent abundance of northern short-tailed shrews increased with vegetation community similarity (Appendix 5F). The top model for the apparent abundance of eastern chipmunk was wetland size but had confidence intervals that overlapped 0, suggesting no substantial effect of the variable. Models within 2ΔAICc also had confidence intervals that overlapped 0; the only model within 2ΔAICc without confidence intervals that overlapped 0 was the intercept-only model (Appendix 5G), indicating that the tested environmental variables do not predict eastern chipmunk apparent abundance in restored wetlands.

Occupancy

We found competing models within 2ΔAICc for the apparent abundance of white-footed mice, meadow voles, northern short-tailed shrews, meadow jumping mice, and eastern chipmunk. No model had a strong effect on white-footed mice occupancy at restored wetlands as the top model, and models within 2ΔAIC of the top model had confidence intervals that overlapped zero (Appendix 6A). Meadow vole occupancy probability may decrease with wetland age, as this was the only model within 2ΔAICc of the top model which had confidence intervals that did not overlap 0, suggesting that age may be the best predictor, albeit weak, of meadow voles in restored wetlands (Appendix 6B). For occupancy of both northern short-tailed shrews and meadow jumping mice, confidence intervals overlapped 0 for top models and those within 2ΔAICc of the top model (Appendix 6C; Appendix 6D). For eastern chipmunks, most tested models came within 2ΔAICc, suggesting the tested environmental variables do not predict eastern chipmunk occupancy in restored wetlands well (Appendix 6E).

Diversity, richness, and evenness

We found competing models for diversity, evenness, and richness in restored wetlands. For diversity, 3 models fell within 2ΔAICc of the top model, with the top model being the intercept-only model (Appendix 7), and the two remaining models having confidence intervals that overlapped 0, suggesting none of our tested environmental variables influence small mammal diversity at restored wetlands. Likewise, 2 models fell within 2ΔAICc of the top model in predicting small mammal evenness (Appendix 8); the first model had confidence intervals which overlapped 0, and the second model was the intercept-only model, again suggesting there is no strong environmental predictor of small mammal evenness in restored wetlands. Richness in restored wetlands was best explained by the intercept-only model (Appendix 9), again indicating that the tested environmental variables do not predict small mammal apparent species richness in restored wetlands well.

Discussion

Many of our tested small mammal community metrics varied with wetland age; therefore, it is likely that small mammal communities may be dependent on the stage of wetland succession post-restoration, which may be tied to a variety of variables including vegetation community or other physical wetland characteristics. When we did isolate variables in model selection for the small mammal community, we found wetland size is an important variable that matters to the small mammal community, ultimately influencing species that may be present at a restored wetland given its size.

Peromyscus

We found deer mice had higher apparent abundances in younger wetlands than in older wetlands. Similarly, previous research has shown deer mice are more frequent post-disturbance years, regardless of habitat type. For instance, Smith (1940) and Hansen and Warnock (1978) found that deer mice are generally more abundant in earlier stages of succession following ridgetop strip-mining. In an old field habitat, Schweiger et al. (2000) found deer mouse density to be highest in the earlier stage of succession of the old field, then decreased in the later successional stage, although the severity of this decline was dependent on patch size. Additionally, deer mice were captured more frequently by Doyle et al. (1991) in immediate years following forest fire disturbance than in mature forests. Therefore, research suggests that deer mice can be a pioneer species of small mammals for recently disturbed or developing areas, regardless of habitat type, and, given our findings, is likely also true for restored wetlands. The top model for deer mice apparent abundance was vegetation community, with no competing models. Specifically, apparent abundance decreases as vegetation community similarity increases. This may be because they are considered habitat generalists (Wywialowski 1987) and can thrive in various plant communities, thus, dissimilar vegetation communities among restored wetlands may be advantageous to them. In model selection, we interpreted models within ΔAICc of ≥2 as being selected; while some researchers interpret models within ΔAICc of ≥4 as being selected, most of our 18 models were between ΔAICc of 2 and 4, and it, therefore, made more sense to use a model selection of ≥2.

Like deer mice, we found white-footed mouse apparent abundance decreased with wetland age. However, we did not determine the occupancy probability of white-footed mice to differ by wetland age. Although wetland age influences the apparent abundance of white-footed mice, we found through model selection, that wetland size was a better predictor of white-footed mice apparent abundance than wetland age. Specifically, we found as wetland size increased, white-footed mice apparent abundance decreased. This finding complements other research that has determined white-footed mice have higher density in fragmented patches (Nupp and Swihart 1998; Krohne and Hock 1999; Anderson et al. 2003), possibly because structural complexity was higher in smaller patches (Anderson et al. 2003), or because of increased mast availability in smaller fragments (Nupp and Swihart 1998). In our wetlands, on average, smaller wetlands tended to be adjacent to roads and major highways. However, on average, larger wetlands were typically located in more remote areas. We found white-footed mice to have a higher apparent abundance in smaller wetlands, potentially due to the proximity of these wetlands to rights-of-way, which white-footed mice prefer (Adams and Geis 1983).

Moreover, fragmentation has a negligible effect on the genetic structure of white-footed mice (Mossman and Waser 2001), again suggesting that the species can persist in smaller fragmented patches. Our results may also reflect the trapping season, as small, fragmented areas provide good habitat for white-footed mice seasonally, and populations could experience decreases in the winter (Wilder et al. 2005). The environmental variables evaluated for the occupancy probability of white-footed mice in restored wetlands showed no strong effect in model selection, which is likely because they are habitat generalists and can persist in many habitat types (Alder and Wilson 1987; Francl et al. 2004).

Meadow voles, northern short-tailed shrews, and meadow jumping mice

We found wetland age influenced apparent abundance of meadow voles. Meadow voles had higher apparent abundance in younger wetlands as opposed to older wetlands. This may be due to their preference for grasses over woody vegetation (Grant 1971; Yahner 1982), as many younger sites were classified as palustrine emergent, while older sites were usually classified as palustrine scrub-shrub or forested and had more woody vegetation. Our youngest sites (≤5 yrs old) were mostly palustrine emergent (n= 4), while older sites (≥5 yrs old) were mostly palustrine scrub-shrub (n = 6) or palustrine forested (n = 1), although three older sites were palustrine emergent. We observed a preference for younger restored sites because woody vegetation takes longer to develop (Niswander and Mitsch 1995; Balcombe et al. 2005). In support of this explanation, we found the most important environmental variable in predicting meadow vole apparent abundance was average canopy cover, with meadow vole apparent abundance decreasing as average canopy cover increases. This is consistent with findings by Cassel et al. (2020) that canopy openness was positively associated with occupancy of meadow voles. Both canopy cover and woody vegetation are related and may be correlated with wetland age. Additionally, we found occupancy probability of white-footed mice decreased with wetland age. However, model selection for occupancy probability of white-footed mice later showed wetland age as having a model weight of only 22%, in addition to having competing models. Thus, wetland age may be a weak predictor of meadow vole occupancy in restored wetlands. In summary, wetland age influenced both meadow vole apparent abundance and occupancy probability. However, it is likely due to the lack of woody vegetation and consequently canopy cover at younger wetlands.

Although we expected to find an effect of age on the apparent abundance and occupancy of northern short-tailed shrews because they are insectivores and invertebrates may differ by wetland age (Fairchild et al. 2008; Swartz et al. 2019), we did not detect an effect. However, our detection probability of the species was low (0.36), as insectivores like *Blarina*, are challenging to capture in live traps (Rose et al. 1990; Rossell and Rossell 1999). Meadow jumping mice also had a low detection probability (0.24). Potentially our trapping technique was not conducive to capturing these species. As was found for amphibians (Birx-Raybuck et al. 2010; Oja et al. 2011), small mammal response to wetland age seems to be species-specific. We found total apparent abundance to decline with wetland age. However, this result is driven by deer and white-footed mice, which account for 61.9% of all unique individuals captured.

Model selection showed vegetation community and wetland type were competing models in the apparent abundance of northern short-tailed shrews. Wetland classification (Cowardin et al. 1979) affected northern short-tailed shrew apparent abundance; northern short-tailed shrews had higher apparent abundance at palustrine emergent wetlands than at scrub-shrub wetlands, but abundance was not different between palustrine emergent wetlands and forested wetlands. Shrews prefer grass-sedge marsh and willow-alder fen (Wrigley et al. 1979), and specifically, northern short-tailed shrews are associated with herbaceous cover and coarse woody debris in Appalachia (Laerm et al. 2007), which resonates with our findings. Vegetation community similarity positively affected northern short-tailed shrew apparent abundance, potentially because of a specific insect community that a particular vegetation community may harbor. Because macroinvertebrate abundance and community composition are associated with vegetation

(Stewart and Downing 2008; Swartz et al. 2019), and northern short-tailed shrews are insectivores, they may be drawn to areas with higher prey abundance. Sites that had a higher vegetation community similarity scores were composed of a plant community consisting of a large amount of alder (*Alnus spp*.) in the shrub stratum, and large amounts of reed canary grass (*Phalaris arundinacea*), sensitive fern (*Onoclea sensibilis*), soft rush (*Juncus effusus*), *Carex spp*, and goldenrods (*Solidago spp*.) in the herbaceous stratum. Although these models may weakly predict northern short-tailed shrew apparent abundance, we could not determine any environmental variables that influenced northern short-tailed shrew occupancy probability from the model selection. Our tested environment variables did not include a measure of insect community or soil attributes; because arthropod availability and well-developed soil profile are important variables to northern short-tailed shrews (Pruitt 1953; O'neill and Robel 1985; Laerm et al. 2007), as they are insectivores and semi-fossorial, these variables may be more important to northern short-tailed shrews than the environmental variables tested in this study.

Like white-footed mice, the apparent abundance of meadow jumping mice was best predicted by wetland size; unlike in the case of white-footed mice, the apparent abundance of meadow jumping mice increased with wetland size. According to Bowers and Dooley (1993), species found in larger patch sizes (1 ha), had larger home ranges and were seemingly more territorial than species found in smaller patches (0.062 ha). Although home range estimations for meadow jumping mice are variable, estimates have been as large as 1.1 ha (Quimby 1951; Whitaker 1972). In contrast, the average home range of white-footed mice is smaller, averaging 0.1 ha (Lackey et al. 1985). Therefore, this result may have stemmed from meadow jumping mice having a larger and more variable home range size. Although wetland size was the best predictor of apparent abundance for meadow jumping mice, we did not find a model to best predict meadow jumping mice occupancy probability. Urban and Swihart (2009) came to a similar conclusion: patch size did not affect species' occupancy.

We found apparent abundance for all species combined was best predicted by wetland size. Like for white-footed mice, this trend showed a decrease in apparent abundance as wetland size increased. Because *Peromyscus* accounted for most of our captures, we believe total apparent abundance was influenced chiefly by deer mice and white-footed mice, who also exhibited this trend with wetland size.

Beaver activity is one variable present at some sites that may have indirect effects on the results of wetland age. Beavers can alter hydrology leading to changes in vegetation community and wetland type (Bonner et al. 2009). This may be problematic because beaver activity is monitored and addressed by wetland managers in younger restored wetland sites, as these younger sites are still within their monitoring period. However, older restored wetlands are likely not monitored or managed, yet beaver activity at these restored wetlands is still occurring, creating a possibility for younger sites to have different attributes than older sites because of a difference in actions taken towards beavers in these different wetland age groups. Another variable that was not directly assessed but is closely tied with small mammals is microtopography. However, it is known that microtopography changes with age (Wolff et al. 2011), and therefore differences observed in the small mammal community by age (decreasing apparent abundance of certain species and increasing small mammal diversity with age) may also be attributed to changes in wetland microtopography as wetlands age.

Richness, diversity, and evenness

Although we determined that diversity and evenness increased with wetland age when looking at the variable isolated, model selection showed a high AICc score and low AICc model weight for age for both diversity and evenness. Therefore, wetland age affects small mammal diversity and evenness but may be a weak predictor. We did not determine a difference in wetland age on apparent species richness; although not significant, apparent richness did show a decreasing trend with wetland age. Similarly, birds did not exhibit a difference in richness in younger versus older wetlands (VanRees-Siewert and Dinsmore 1996). Specifically, vegetation had a greater effect on total birds and breeding bird richnesss, and wetland area had a greater effect on species richness for waterfowl and breeding waterfowl (VanRees-Siewert and Dinsmore 1996).

Porej and Heatherington (2005) found amphibian richness depends on other habitat features rather than age. Furthermore, when determining significant differences in invertebrate density between age classes of wetlands, researchers found invertebrate taxa richness to be consistent across all age classes (Hart and Davis 2011). However, Thiere et al. (2009) found local richness of aquatic macroinvertebrates increases with wetland age. Overall, our data matches most of the research for other wildlife taxa and macroinvertebrates that species richness does not differ by wetland age. Model selection for diversity, richness, and evenness showed no strong effect, suggesting other factors such as community interspecific competition or predation may have a greater effect on these metrics rather than the environmental variables chosen for this study.

Conclusion

In summary, we found wetland age affected some aspects of small mammal communities. While age may negatively affect the apparent abundance of individual species, it positively affected small mammal diversity and evenness. While managers cannot change wetland age, knowing general small mammal community trends will facilitate better wetland management due to a more complete understanding of their wetland system through time. Managers should consider that wetland size will influence species differently and should therefore strive for a diversity of wetland sizes when designing restoration projects to accommodate species that require larger wetlands (such as meadow jumping mice) and species that thrive in smaller wetlands (such as white-footed mice). Environmental variables could not predict diversity, richness, or evenness in restored wetlands, suggesting other factors such as interspecific competition, predation, or other untested environmental variables are impactful. From this study, we learn that wetland restoration success, in terms of the small mammal community, depends on wetland age and size.

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Figures

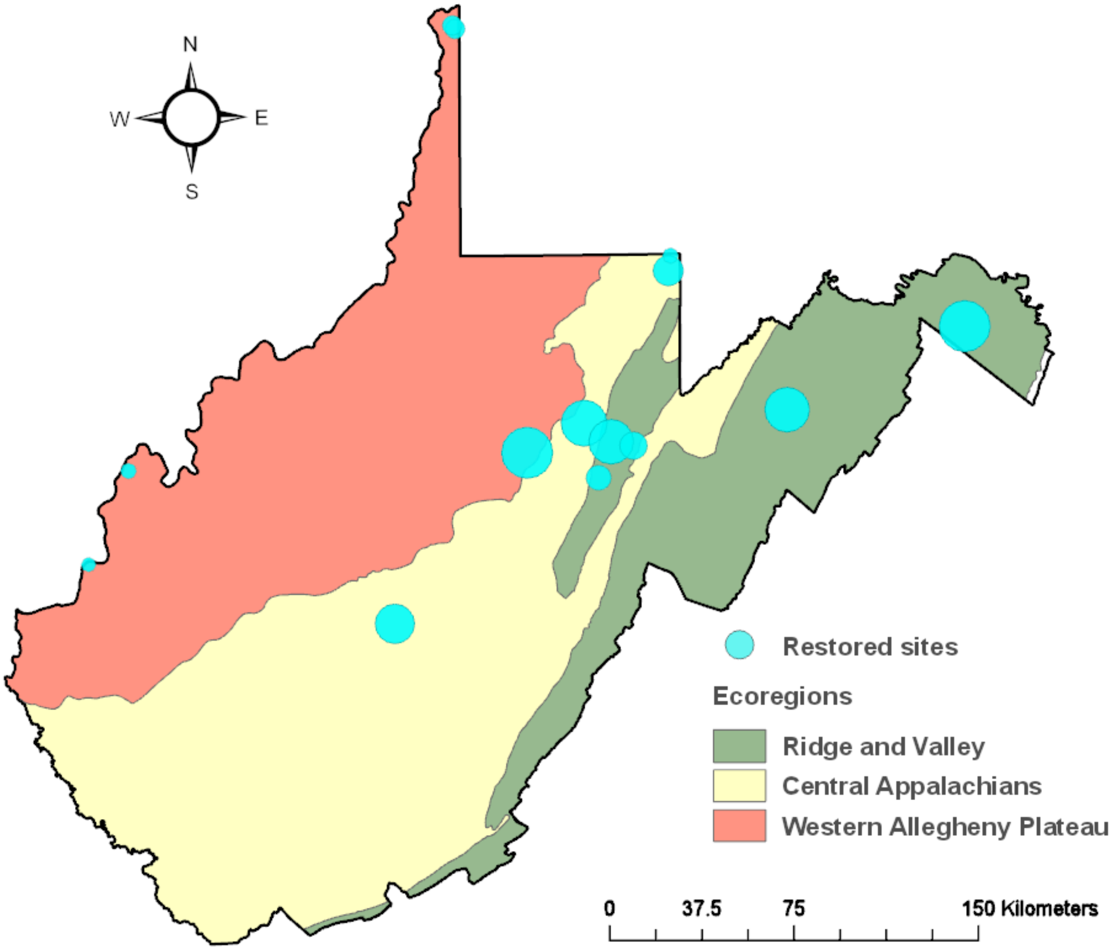

Figure 1. Restored wetlands (n = 14) were sampled in 3 ecoregions of West Virginia, USA, from June to August of 2020 (n = 6) and 2021 (n = 8). The age of wetlands ranged from 1 to 29 years (mean ± SE years = 14.2 ± 2.8 years); the size of the data point on the map represents age, with circle size increasing with wetland age.

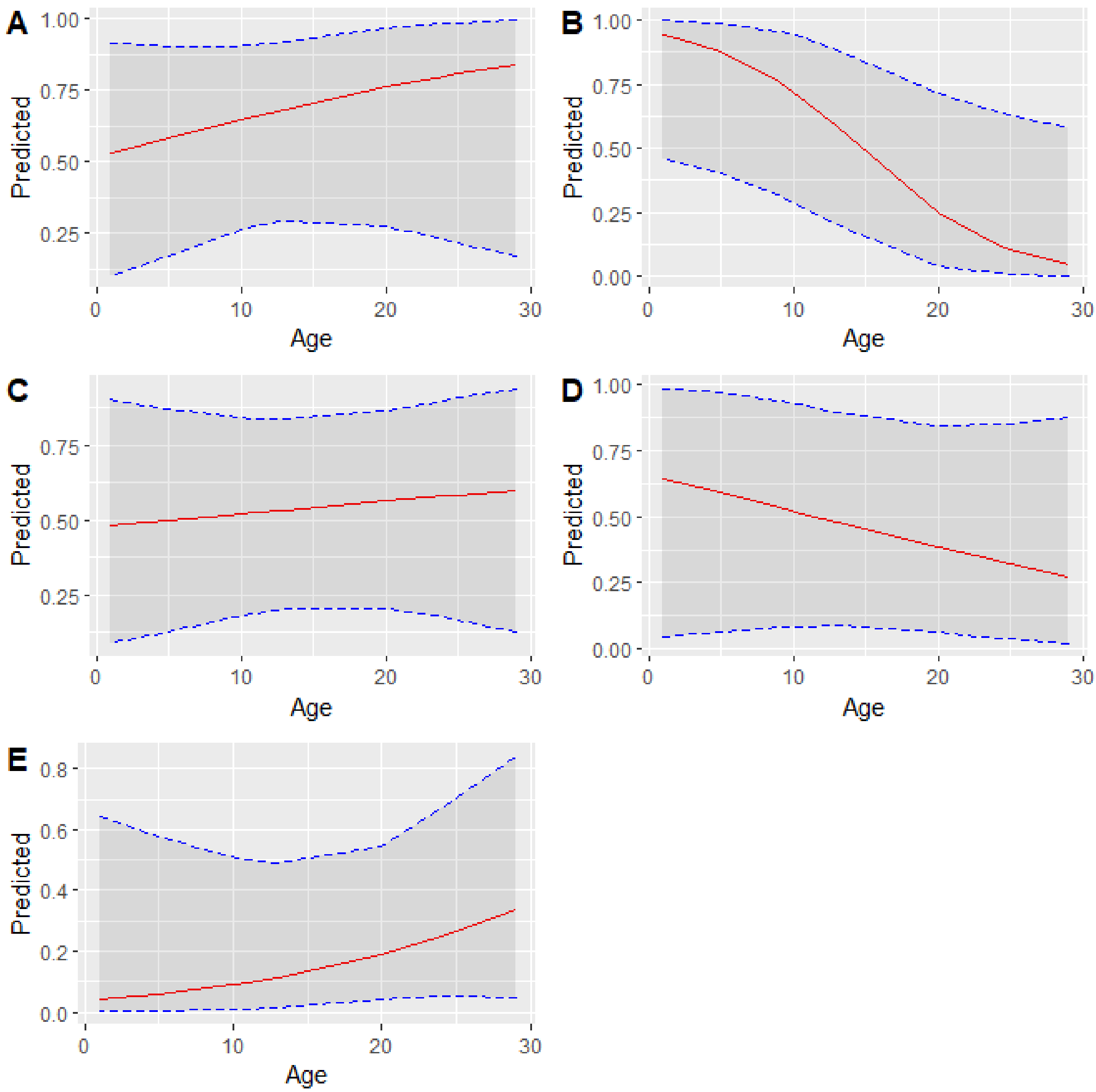

Figure 2. Predicted occupancy probabilities of white-footed mice [A], meadow voles [B], northern short-tailed shrews [C], meadow jumping mice [D], and eastern chipmunk [E] as wetlands age in West Virginia, USA. Restored wetlands (n = 14) ranged in age from 1 to 29 years old at the time of sampling in 2020 and 2021.

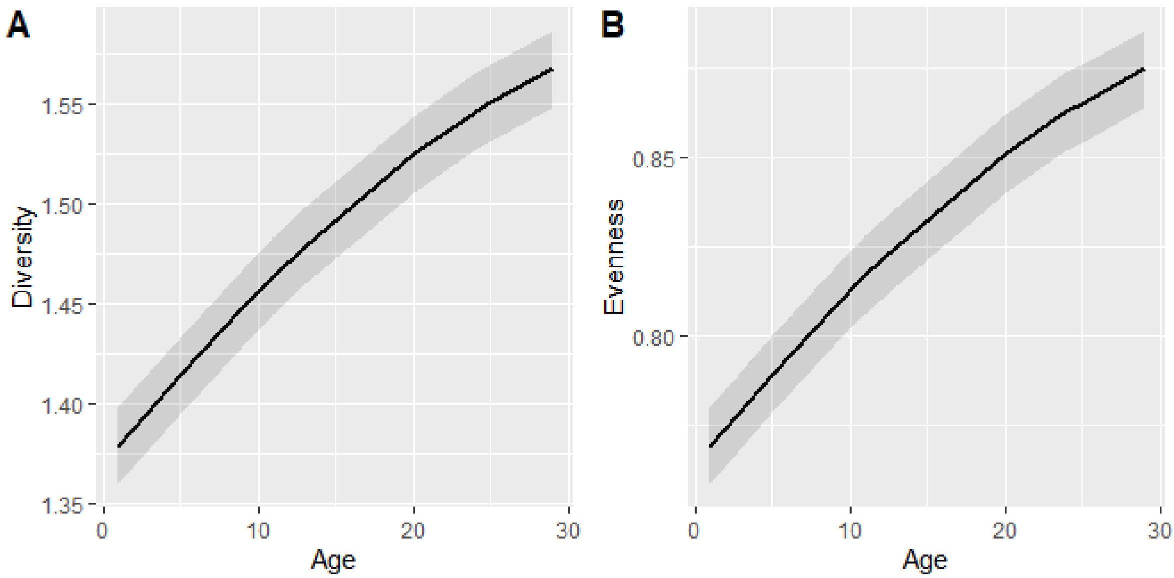

Figure 3. Small mammal community metrics at sites ranging from 1 to 29 years old, including: A) Shannon diversity and B) Pielou's evenness. Data were obtained from 14 restored wetland sites in 2020 and 2021 in West Virginia, USA.

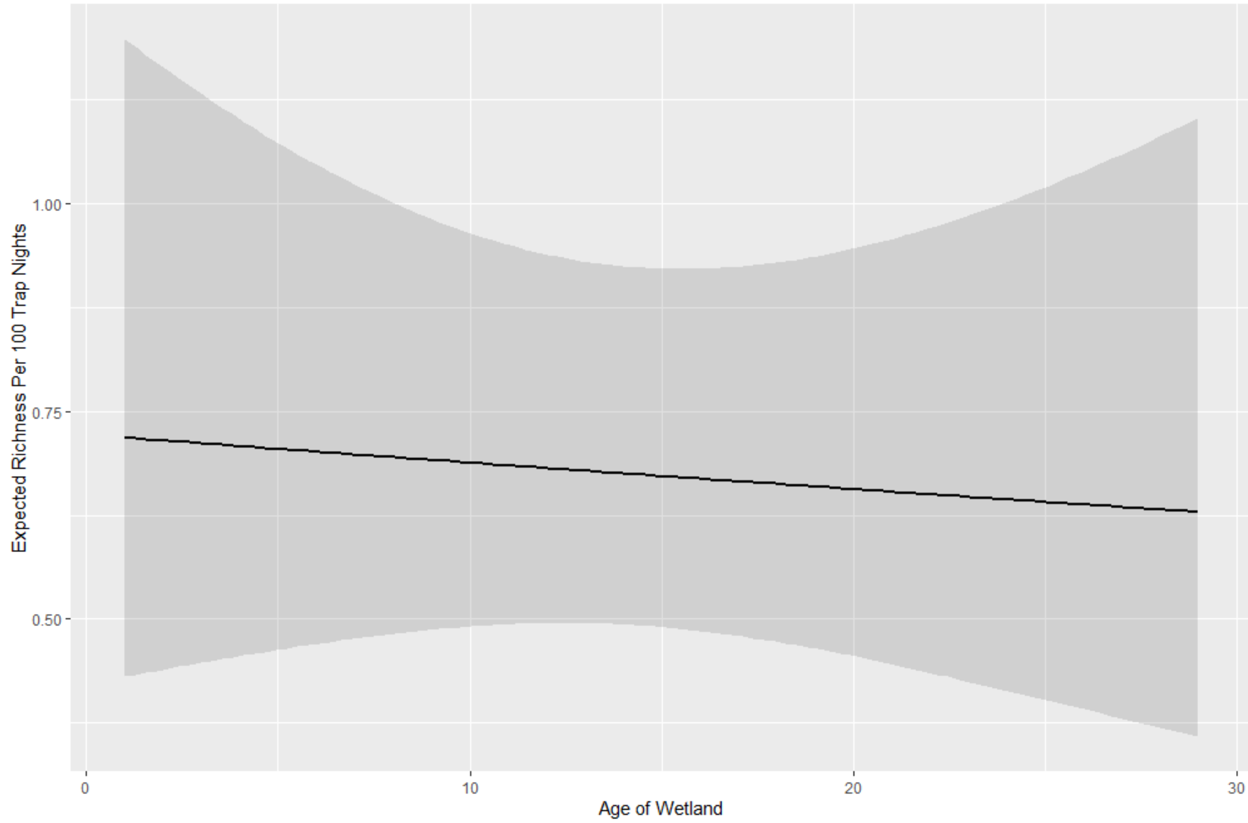

Figure 4. Small mammal expected species richness per 100 trap nights at 26 wetland sites ranging from 1 to 29 years old in 2020 and 2021 in West Virginia, USA.

Tables

Table 1. Description of environmental variables used in models to determine which environment variables most influence aspects of small mammal communities (such as species apparent abundance, occupancy, and site diversity, richness, and evenness). Many variables were obtained from West Virginia Wetland Rapid Assessment Method (WVWRAM) scores at each site and are briefly described according to the WVWRAM reference manual (West Virginia Department of Environmental Protection 2020). WVWRAM were obtained using both remote sensing data and data from field visits. Both age and intercept only models were assessed along with the other models in the AIC model comparison.

Variable	Description
Size	Wetland size based on national wetlands inventory (NWI) GIS layer (U. S. Fish and Wildlife Service 2014) in corroboration with site visits.
Wetland Type	Palustrine emergent, scrub-shrub, and forested according to Cowardin et al. (1979).
Vegetation Community	Vegetation community was estimated using a principle component analysis (PCA). We used the first score provided by the PCA in model selection, as this score explains most of the variation in the data.
Average Canopy	Average canopy was estimated using methodology described in methods section.
Landscape Integrity	This metric was calculated using the WVDNR layer for landscape integrity (Dougherty and Byers 2008); the layer was made using distance from weighted landscape features.
Ecoregion	Level 3 ecoregions comprising West Virginia: Western Alleghany Plateau, Central Appalachians, and Ridge and Valley ecoregions.
Rail/Road	Wetlands receive a score of 0–2 points, depending on proximity to railway: 2 points wetland if within 5 m (16 ft) of a road or railroad track, 1 point if wetland is 5–50 m (16–164 ft) from a road or railroad track, 0 points if wetland is > 50 m (164 ft) from a road or railroad track (West Virginia Department of Environmental Protection 2020)
Habitat & Ecological Integrity Intrinsic Potential	This score was calculated as a combination of all previously calculated WVWRAM vegetation, soil, and hydrology scores (West Virginia Department of Environmental Protection 2020).

Habitat & Ecological Integrity Function Without Biodiversity rank	Calculated using previously determined WVWRAM scores: habitat & ecological integrity intrinsic potential, habitat & ecological integrity landscape opportunity, and habitat & ecological integrity value to society (West Virginia Department of Environmental Protection 2020).
Habitat & Ecological Integrity Function	Calculated using same variables as habitat & ecological integrity function without biodiversity rank, plus a biodiversity rank; additional points added to final score based on the biodiversity rank of the site (1–6) (West Virginia Department of Environmental Protection 2020).
Habitat & Ecological Integrity Condition	Calculated using scores of habitat & ecological integrity intrinsic potential, habitat & ecological integrity landscape opportunity, and site biodiversity rank (1–6) (West Virginia Department of Environmental Protection 2020).
Condition	The WVWRAM condition score was developed using previously calculated WVWRAM scores: intrinsic potential of wetland water quality and flood attenuation, and habitat/ecological integrity condition (West Virginia Department of Environmental Protection 2020).
Floristic Quality Assessment	Floristic quality assessment was calculated using abundance-weighted mean coefficient of conservatism (wmC) and field collected vegetation data (West Virginia Department of Environmental Protection 2020).
Vegetation Vertical Structure	This score is representative of the number of vertical vegetative strata (overstory, understory, herbaceous vegetation) present for each site. Score calculated using both a GIS score using NWI layer and a field-assessed score evaluating present strata (West Virginia Department of Environmental Protection 2020).
Woody Vegetation	This metric is based upon a combination of different GIS layers to assess the amount of woody vegetation present at the site (West Virginia Department of Environmental Protection 2020).
Wetland Breeding Bird Occupancy	This score was developed from a West Virginia wetland breeding bird database from breeding bird atlas data. A score of 0–3 points was given to each site depending on the ranking of atlas blocks. This metric was strictly landscape-assessed and not evaluated at site visits (West Virginia Department of Environmental Protection 2020).

Appendices

Appendix 1. Year in which site visits were conducted in the creation of West Virginia Wetland Rapid Assessment Method (WVWRAM) scores for restored wetlands in West Virginia, USA. Months between site visits describe how many months were between the WVWRAM site visit and our site visits for small mammal trapping. Most WVWRAM site visits occurred before our sampling (n = 12), although on four occasions was obtained after our site visit. On average, 16.42 (± 4.39) months passed from WVWRAM site visit to our site visit (1.36 ± 0.36 years).

Site	WVWRAM Year Sampled	Months After Visit
Buckhannon Triangle	2017	48
Glade Farms	2019	23
Hazelton	2018	23
Hillcrest 1	2021	-2
Hillcrest 2	2021	-2
Hoeft Marsh	2020	11
McClintic	2020	11
Montrose	2017	37
Nicholas	2017	36
Queens	2018	22
Stauffers Marsh	2021	-1
Sugar Creek	2020	2
Tygart Mitigation Bank	2018	23
Walnut Bottom	2021	-1

Appendix 2. Apparent abundance of deer mice [PMAN], white-footed mice [PLEU], meadow voles [MPEN], northern short-tailed shrews [BBRE], meadow jumping mice [ZHUD], eastern chipmunks [TSTR], and total small mammals per 100 trap nights at each wetland, based on a generalized linear model using age as a predictor variable and count data collected 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

		Species						
Site	Age	PMAN	PLEU	MPEN	BBRE	ZHUD	TSTR	Total
Buckhannon Triangle	29	0.788	0.278	0.256	0.204	0.198	0.096	1.737
Glade Farms	2	1.783	1.830	1.618	0.346	0.151	0.048	5.719
Hazelton	13	1.278	0.849	0.764	0.279	0.169	0.064	3.519
Hillcrest 1	5	1.628	1.484	1.319	0.326	0.156	0.052	5.009
Hillcrest 2	5	1.628	1.484	1.319	0.326	0.156	0.052	5.009
Hoeft Marsh	1	1.838	1.962	1.732	0.353	0.150	0.047	5.977
McClintic	2	1.783	1.830	1.618	0.346	0.151	0.048	5.719
Montrose	24	0.916	0.394	0.361	0.225	0.189	0.085	2.166
Nicholas	20	1.034	0.521	0.474	0.243	0.181	0.077	2.584
Queens	11	1.358	0.976	0.876	0.290	0.166	0.061	3.844
Stauffers Marsh	29	0.788	0.278	0.256	0.204	0.198	0.096	1.737
Sugar Creek	25	0.889	0.367	0.337	0.220	0.191	0.087	2.072
Tygart Mitigation Bank	9	1.442	1.123	1.004	0.302	0.162	0.058	4.199
Walnut Bottom	24	0.916	0.394	0.361	0.225	0.189	0.085	2.166

Appendix 3. Occupancy and detection probability estimates of white-footed mice [PLEU], meadow voles [MPEN], northern short-tailed shrews [BBRE], meadow jumping mice [ZHUD], and eastern chipmunk [TSTR] captured in 14 restored wetlands of varying ages (1–29) in 2020 and 2021 in West Virginia, USA.

	White-footed mice		Meadow voles		Northern short-tailed shrews		Meadow jumping mice		Eastern chipmunks	
	Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
Buckhannon Triangle	0.838	0.222	0.048	0.077	0.599	0.282	0.272	0.300	0.338	0.265
Glade Farms	0.542	0.274	0.931	0.092	0.487	0.277	0.629	0.409	0.047	0.080
Hazelton	0.683	0.181	0.582	0.210	0.533	0.188	0.478	0.288	0.113	0.103
Hillcrest 1	0.582	0.234	0.879	0.129	0.500	0.244	0.589	0.375	0.060	0.088
Hillcrest 2	0.582	0.234	0.879	0.129	0.500	0.244	0.589	0.375	0.060	0.088
Hoeft Marsh	0.529	0.289	0.943	0.081	0.483	0.289	0.642	0.419	0.043	0.077
McClintic	0.542	0.274	0.931	0.092	0.487	0.277	0.629	0.409	0.047	0.080
Montrose	0.797	0.212	0.125	0.141	0.579	0.231	0.331	0.275	0.249	0.172
Nicholas	0.760	0.197	0.247	0.192	0.562	0.201	0.382	0.263	0.190	0.128
Queens	0.659	0.185	0.678	0.197	0.525	0.196	0.506	0.306	0.097	0.100
Stauffers Marsh	0.838	0.222	0.048	0.077	0.599	0.282	0.272	0.300	0.338	0.265
Sugar Creek	0.806	0.215	0.104	0.127	0.583	0.241	0.319	0.280	0.265	0.187
Tygart Mitigation Bank	0.634	0.196	0.760	0.178	0.517	0.209	0.534	0.328	0.083	0.097
Walnut Bottom	0.797	0.212	0.125	0.141	0.579	0.231	0.331	0.275	0.249	0.172
Detection Probability	0.392	0.109	0.623	0.102	0.363	0.133	0.249	0.142	0.547	0.260

Appendix 4. Shannon diversity and Pielou's evenness index for small mammal communities sampled in 2020 and 2021 in West Virginia, USA, at each site (n = 14), organized by wetland age (1–29). As wetlands age, Shannon diversity and Pielou's evenness increase.

Site	Age	Diversity	Evenness
Hoeft Marsh	1	1.379	0.769
Glade Farms	2	1.388	0.774
McClintic	2	1.388	0.774
Hillcrest 1	5	1.414	0.789
Hillcrest 2	5	1.414	0.789
Tygart Mitigation Bank	9	1.448	0.808
Queens	11	1.464	0.817
Hazelton	13	1.479	0.825
Nicholas	20	1.525	0.851
Montrose	24	1.546	0.863
Walnut Bottom	24	1.546	0.863
Sugar Creek	25	1.551	0.865
Buckhannon Triangle	29	1.568	0.875
Stauffers Marsh	29	1.568	0.875

Appendix 5A. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence the apparent abundance of white-footed mice (*Peromyscus leucopus*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Size	2	104.44	0.00	1.00	-49.67	1.00
Age	2	144.33	39.89	0.00	-69.62	1.00
Wetland Breeding Bird Occupancy	2	149.61	45.17	0.00	-72.26	1.00
Ecoregion	4	152.35	47.91	0.00	-69.95	1.00
Landscape Integrity	2	153.56	49.12	0.00	-74.23	1.00
Vegetation Vertical Structure	2	155.35	50.92	0.00	-75.13	1.00
Habitat Function (without Biodiversity Rank)	2	159.86	55.42	0.00	-77.38	1.00
Woody Vegetation	2	160.47	56.03	0.00	-77.69	1.00
Vegetation Community	2	162.09	57.66	0.00	-78.50	1.00
Habitat Potential	2	163.02	58.58	0.00	-78.96	1.00
Habitat Function	2	164.71	60.28	0.00	-79.81	1.00
Intercept-Only	1	164.96	60.52	0.00	-81.31	1.00
Canopy Cover	2	166.12	61.69	0.00	-80.52	1.00
Habitat Condition	3	166.53	62.10	0.00	-80.72	1.00
Road & Rail	2	166.89	62.46	0.00	-80.90	1.00
Floristic Quality	2	167.11	62.68	0.00	-81.01	1.00
Condition	2	167.14	62.71	0.00	-81.03	1.00
Cowardin Type	3	168.99	64.56	0.00	-80.30	1.00

Appendix 5B. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence the apparent abundance of total small mammals captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Size	2	151.66	0.00	0.99	-73.28	0.99
Age	2	160.68	9.03	0.01	-77.80	1.00
Ecoregion	4	171.33	19.67	0.00	-79.44	1.00
Woody Vegetation	2	181.70	30.04	0.00	-88.30	1.00
Landscape Integrity	2	183.11	31.45	0.00	-89.01	1.00
Habitat Potential	2	183.43	31.77	0.00	-89.17	1.00
Wetland Breeding Bird Occupancy	2	185.12	33.46	0.00	-90.01	1.00
Vegetation Vertical Structure	2	192.03	40.38	0.00	-93.47	1.00
Habitat Function	2	193.91	42.26	0.00	-94.41	1.00
Floristic Quality	2	195.76	44.10	0.00	-95.33	1.00
Cowardin Type	3	195.83	44.17	0.00	-93.71	1.00
Intercept-Only	1	196.85	45.19	0.00	-97.26	1.00
Habitat Condition	2	197.56	45.91	0.00	-96.24	1.00
Habitat Function (without Biodiversity Rank)	2	198.05	46.39	0.00	-96.48	1.00
Vegetation Community	2	198.08	46.43	0.00	-96.50	1.00
Road & Rail	2	199.13	47.48	0.00	-97.02	1.00
Canopy Cover	2	199.35	47.70	0.00	-97.13	1.00
Condition	2	199.38	47.73	0.00	-97.15	1.00

Appendix 5C. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence the apparent abundance of meadow jumping mice (*Zapus hudsonius*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Size	2	31.75	0.00	0.45	-13.33	0.45
Wetland Breeding Bird Occupancy	2	33.96	2.22	0.15	-14.44	0.60
Vegetation Community	2	35.86	4.11	0.06	-15.38	0.66
Vegetation Vertical Structure	2	36.11	4.36	0.05	-15.51	0.71
Woody Vegetation	2	36.23	4.49	0.05	-15.57	0.76
Habitat Condition	2	37.01	5.26	0.03	-15.96	0.79
Habitat Function	2	37.06	5.31	0.03	-15.98	0.83
Intercept-Only	1	37.08	5.34	0.03	-17.38	0.86
Ecoregion	4	37.48	5.74	0.03	-12.52	0.88
Road & Rail	2	37.62	5.87	0.02	-16.26	0.91
Landscape Integrity	2	37.83	6.09	0.02	-16.37	0.93
Floristic Quality	2	38.45	6.71	0.02	-16.68	0.95
Canopy Cover	2	38.79	7.04	0.01	-16.85	0.96
Condition	2	39.07	7.32	0.01	-16.99	0.97
Habitat Potential	2	39.43	7.68	0.01	-17.17	0.98
Age	2	39.74	8.00	0.01	-17.33	0.99
Habitat Function (without Biodiversity Rank)	2	39.83	8.08	0.01	-17.37	1.00
Cowardin Type	3	41.85	10.10	0.00	-16.73	1.00

Appendix 5D. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence the apparent abundance of meadow voles (*Microtus pennsylvanicus*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Canopy Cover	2	58.63	0.00	1.00	-26.77	1.00
Woody Vegetation	2	80.30	21.67	0.00	-37.61	1.00
Age	2	93.60	34.97	0.00	-44.25	1.00
Cowardin Type	3	97.31	38.68	0.00	-44.46	1.00
Habitat Potential	2	100.64	42.01	0.00	-47.78	1.00
Size	2	101.78	43.15	0.00	-48.34	1.00
Floristic Quality	2	104.38	45.75	0.00	-49.65	1.00
Ecoregion	4	105.99	47.36	0.00	-46.77	1.00
Condition	2	109.59	50.96	0.00	-52.25	1.00
Vegetation Vertical Structure	2	109.70	51.07	0.00	-52.30	1.00
Landscape Integrity	2	110.67	52.03	0.00	-52.79	1.00
Intercept-Only	1	110.93	52.30	0.00	-54.30	1.00
Habitat Function (without Biodiversity Rank)	2	111.72	53.09	0.00	-54.31	1.00
Habitat Condition	2	113.31	54.68	0.00	-54.11	1.00
Road & Rail	2	113.49	54.86	0.00	-54.20	1.00
Vegetation Community	2	113.53	54.90	0.00	-54.22	1.00
Habitat Function	2	113.68	55.04	0.00	-54.29	1.00
Wetland Breeding Bird Occupancy	2	113.68	55.05	0.00	-54.29	1.00

Appendix 5E. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence the apparent abundance of deer mice (*Peromyscus maniculatus*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Vegetation Community	2	96.04	0.00	0.54	-45.47	0.54
Habitat Function	2	99.00	2.96	0.12	-46.95	0.66
Size	2	99.22	3.18	0.11	-47.07	0.77
Landscape Integrity	2	98.98	3.94	0.07	-47.44	0.84
Condition	2	100.01	3.97	0.07	-47.46	0.91
Habitat Condition	2	100.88	4.84	0.05	-47.89	0.96
Age	2	103.40	7.36	0.01	-49.15	0.98
Ecoregion	4	104.36	8.32	0.01	-45.96	0.98
Road & Rail	2	106.02	9.98	0.00	-50.46	0.99
Habitat Function (without Biodiversity Rank)	2	106.31	10.27	0.00	-50.61	0.99
Intercept-Only	1	107.34	11.30	0.00	-52.50	0.99
Habitat Potential	2	107.36	11.32	0.00	-51.13	0.99
Canopy Cover	2	107.47	11.43	0.00	-51.19	1.00
Wetland Breeding Bird Occupancy	2	107.70	11.67	0.00	-51.31	1.00
Vegetation Vertical Structure	2	109.30	13.26	0.00	-52.10	1.00
Cowardin Type	3	109.33	13.29	0.00	-50.47	1.00
Woody Vegetation	2	109.82	13.79	0.00	-52.37	1.00
Floristic Quality	2	110.07	14.03	0.00	-52.49	1.00

Appendix 5F. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence the apparent abundance of northern short-tailed shrews (*Blarina brevicauda*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Vegetation Community	2	50.30	0.00	0.37	-22.60	0.37
Cowardin Type	3	52.16	1.87	0.15	-21.88	0.52
Condition	2	52.73	2.44	0.11	-23.82	0.63
Habitat Function (without Biodiversity Rank)	2	54.02	3.72	0.06	-24.46	0.68
Size	2	54.41	4.12	0.05	-24.66	0.73
Landscape Integrity	2	54.57	4.27	0.04	-24.74	0.77
Woody Vegetation	2	54.61	4.32	0.04	-24.76	0.82
Intercept-Only	1	55.11	4.84	0.03	-26.39	0.85
Road & Rail	2	55.67	5.37	0.02	-25.29	0.88
Habitat Condition	2	55.95	5.65	0.02	-25.43	0.90
Habitat Potential	2	56.19	5.90	0.02	-25.55	0.92
Ecoregion	4	56.46	6.17	0.01	-25.01	0.93
Habitat Function	2	56.79	6.49	0.01	-25.85	0.95
Age	2	57.25	6.95	0.01	-26.08	0.96
Canopy Cover	2	57.34	7.05	0.01	-26.13	0.97
Vegetation Vertical Structure	2	57.50	7.20	0.01	-26.20	0.98
Floristic Quality	2	57.52	7.22	0.01	-26.21	0.99
Wetland Breeding Bird Occupancy	2	57.56	7.26	0.01	-26.23	1.00

Appendix 5G. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence the apparent abundance of eastern chipmunk (*Tamias striatus*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Size	2	26.88	0.00	0.15	-10.90	0.15
Landscape Integrity	2	27.28	0.39	0.13	-11.09	0.28
Canopy Cover	2	27.51	0.63	0.11	-11.21	0.39
Vegetation Vertical Structure	2	27.54	0.65	0.11	-11.22	0.50
Intercept-Only	1	27.67	0.79	0.10	-12.67	0.60
Vegetation Community	2	28.11	1.23	0.08	-11.51	0.69
Habitat Function (without Biodiversity Rank)	2	29.61	2.73	0.04	-12.26	0.73
Condition	2	29.88	3.00	0.03	-12.39	0.76
Wetland Breeding Bird Occupancy	2	30.05	3.17	0.03	-12.48	0.79
Age	2	30.18	3.30	0.03	-12.55	0.82
Habitat Condition	2	30.25	3.37	0.03	-12.58	0.85
Habitat Function	2	30.36	3.48	0.03	-12.63	0.88
Road & Rail	2	30.40	3.52	0.03	-12.65	0.90
Habitat Potential	2	30.42	3.54	0.03	-12.66	0.93
Floristic Quality	2	30.43	3.55	0.03	-12.67	0.95
Woody Vegetation	2	30.43	3.55	0.03	-12.67	0.98
Cowardin Type	3	31.45	4.57	0.02	-11.52	1.00
Ecoregion	4	34.03	7.15	0.00	-10.79	1.00

Appendix 6A. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence occupancy probability of white-footed mice (*Peromyscus leucopus*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Vegetation Community	3	57.91	0.00	0.31	-24.76	0.31
Canopy Cover	3	58.82	0.90	0.20	-25.21	0.51
Intercept-Only	2	60.26	2.34	0.10	-27.58	0.61
Landscape Integrity	3	61.26	3.35	0.06	-26.43	0.67
Habitat Potential	3	61.86	3.95	0.04	-26.73	0.71
Wetland Breeding Bird Occupancy	3	61.95	4.03	0.04	-26.77	0.76
Road & Rail	3	62.47	4.56	0.03	-27.04	0.79
Woody Vegetation	3	62.88	4.97	0.03	-27.24	0.81
Size	3	62.92	5.00	0.03	27.26	0.84
Vegetation Vertical Structure	3	62.99	5.08	0.02	-27.30	0.86
Floristic Quality	3	63.04	5.13	0.02	-27.32	0.89
Age	3	63.04	5.13	0.02	-27.32	0.91
Condition	3	63.12	5.21	0.02	-27.36	0.94
Habitat Function	3	63.37	5.46	0.02	-27.49	0.96
Habitat Function (without Biodiversity Rank)	3	63.41	5.50	0.02	-27.51	0.98
Habitat Condition	3	63.48	5.57	0.02	-27.54	1.00
Cowardin Type	4	66.58	8.67	0.00	-27.07	1.00
Ecoregion	5	71.56	13.65	0.00	-27.03	1.00

Appendix 6B. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence occupancy probability of meadow voles (*Microtus pennsylvanicus*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Woody Vegetation	3	50.30	0.00	0.38	-20.95	0.38
Canopy Cover	3	51.02	0.72	0.27	-21.31	0.65
Age	3	51.37	1.07	0.22	-21.49	0.88
Vegetation Community	3	55.29	4.99	0.03	-23.45	0.91
Intercept-Only	2	56.28	5.98	0.02	-25.59	0.93
Habitat Potential	3	56.59	6.29	0.02	-24.10	0.94
Vegetation Vertical Structure	3	58.15	7.85	0.01	-24.88	0.95
Floristic Quality	3	58.18	7.88	0.01	-24.89	0.96
Landscape Integrity	3	58.52	8.22	0.01	-25.06	0.96
Size	3	58.61	8.31	0.01	-25.11	0.97
Condition	3	58.65	8.35	0.01	-25.13	0.98
Wetland Breeding Bird Occupancy	3	59.02	8.72	0.00	-25.31	0.98
Habitat Function (without Biodiversity Rank)	3	59.37	9.07	0.00	-25.48	0.98
Habitat Function	3	59.49	9.19	0.00	-25.54	0.99
Road & Rail	3	59.56	9.26	0.00	-25.58	0.99
Habitat Condition	3	59.58	9.28	0.00	-25.59	1.00
Cowardin Type	4	60.53	10.23	0.00	-24.04	1.00
Ecoregion	5	61.11	10.81	0.00	-21.80	1.00

Appendix 6C. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence occupancy probability of northern short-tailed shrews (*Blarina brevicauda*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Habitat Function (without Biodiversity Rank)	3	47.43	0.00	0.28	-19.51	0.28
Intercept-Only	2	49.44	2.01	0.10	-22.18	0.38
Size	3	50.03	2.60	0.08	-20.81	0.46
Condition	3	50.06	2.63	0.08	-20.83	0.54
Habitat Function	3	50.37	2.94	0.06	-20.99	0.60
Habitat Condition	3	50.51	3.08	0.06	-21.06	0.66
Road & Rail	3	50.68	3.25	0.06	-21.14	0.72
Habitat Potential	3	51.11	3.68	0.04	-21.36	0.76
Vegetation Community	3	51.12	3.69	0.04	-21.36	0.81
Vegetation Vertical Structure	3	51.17	3.74	0.04	-21.39	0.85
Floristic Quality	3	51.43	4.00	0.04	-21.52	0.89
Wetland Breeding Bird Occupancy	3	52.43	5.00	0.04	-22.02	0.91
Landscape Integrity	3	52.61	5.18	0.02	-22.10	0.93
Woody Vegetation	3	52.66	5.23	0.02	-22.13	0.95
Age	3	52.68	5.26	0.02	-22.14	0.97
Canopy Cover	3	52.75	5.32	0.02	-22.18	0.99
Cowardin Type	4	54.77	7.34	0.01	-21.16	1.00
Ecoregion	5	58.66	11.23	0.00	-20.58	1.00

Appendix 6D. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence occupancy probability of meadow jumping mice (*Zapus hudsonius*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Habitat Function	3	34.66	0.00	0.34	-14.33	0.34
Habitat Condition	3	35.03	0.37	0.28	-14.51	0.63
Vegetation Community	3	38.55	3.89	0.05	-16.28	0.67
Ecoregion	5	39.06	4.40	0.04	-14.53	0.71
Intercept-Only	2	39.19	4.53	0.04	-17.59	0.75
Vegetation Vertical Structure	3	39.43	4.77	0.03	-16.72	0.78
Size	3	39.64	4.98	0.03	-16.82	0.81
Canopy Cover	3	39.86	5.19	0.03	-16.93	0.83
Floristic Quality	3	40.11	5.45	0.02	-17.05	0.86
Landscape Integrity	3	40.20	5.54	0.02	-17.10	0.88
Woody Vegetation	3	40.22	5.56	0.02	-17.11	0.90
Condition	3	40.31	5.65	0.02	-17.16	0.92
Age	3	40.73	6.07	0.02	-17.37	0.93
Wetland Breeding Bird Occupancy	3	40.75	6.09	0.02	-17.38	0.95
Habitat Potential	3	40.95	6.29	0.01	-17.47	0.97
Habitat Function (without Biodiversity Rank)	3	40.99	6.33	0.01	-17.50	0.98
Road & Rail	3	41.19	6.53	0.01	-17.59	0.99
Cowardin Type	4	42.50	7.84	0.01	-17.25	1.00

Appendix 6E. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence occupancy probability of eastern chipmunk (*Tamias striatus*) captured in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Vegetation Community	3	21.38	0.00	0.15	-7.69	0.15
Intercept-Only	2	21.95	0.58	0.11	-8.98	0.26
Vegetation Vertical Structure	3	22.50	1.12	0.08	-8.25	0.34
Age	3	22.85	1.47	0.07	-8.42	0.41
Habitat Function (without Biodiversity Rank)	3	23.20	1.82	0.06	-8.60	0.47
Condition	3	23.47	2.09	0.05	-8.73	0.52
Habitat Condition	3	23.53	2.15	0.05	-8.76	0.57
Habitat Function	3	23.56	2.18	0.05	-8.78	0.62
Landscape Integrity	3	23.60	2.23	0.05	-8.80	0.67
Floristic Quality	3	23.62	2.25	0.05	-8.81	0.72
Canopy Cover	3	23.65	2.28	0.05	-8.83	0.76
Wetland Breeding Bird Occupancy	3	23.68	2.31	0.05	-8.84	0.81
Road & Rail	3	23.71	2.33	0.05	-8.85	0.86
Size	3	23.82	2.44	0.04	-8.91	0.90
Habitat Potential	3	23.85	2.47	0.04	-8.93	0.94
Woody Vegetation	3	23.95	2.57	0.04	-8.97	0.98
Cowardin Type	4	25.62	4.24	0.02	-8.81	1.00
Ecoregion	5	51.16	29.78	0.00	-20.58	1.00

Appendix 7. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence small mammal diversity in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Intercept-Only	2	15.89	0.00	0.18	-5.40	0.18
Landscape Integrity	3	16.45	0.55	0.14	-4.02	0.31
Road & Rail	3	16.92	1.02	0.11	-4.26	0.42
Habitat Potential	3	17.89	2.00	0.07	-4.75	0.49
Vegetation Vertical Structure	3	17.99	2.09	0.06	-4.79	0.55
Habitat Function (without Biodiversity Rank)	3	18.05	2.16	0.06	-4.83	0.61
Size	3	18.53	2.63	0.05	-5.06	0.66
Floristic Quality	3	18.67	2.78	0.04	-5.14	0.70
Canopy Cover	3	18.68	2.79	0.04	-5.14	0.75
Wetland Breeding Bird Occupancy	3	19.02	3.13	0.04	-5.31	0.79
Vegetation Community	3	19.10	3.21	0.04	-5.35	0.82
Age	3	19.14	3.25	0.04	-5.37	0.86
Condition	3	19.17	3.27	0.03	-5.38	0.89
Woody Vegetation	3	19.18	3.28	0.03	-5.39	0.93
Habitat Function	3	19.18	3.29	0.03	-5.39	0.96
Habitat Condition	3	19.20	3.30	0.03	-5.40	0.99
Cowardin Type	4	23.07	7.18	0.00	-5.31	1.00
Ecoregion	5	27.64	11.75	0.00	-5.07	1.00

Appendix 8. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence small mammal evenness in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Wetland Breeding Bird Occupancy	3	-19.17	0.00	0.23	14.08	0.23
Intercept-Only	2	-18.76	0.40	0.19	12.05	0.41
Condition	3	-16.48	2.69	0.06	12.74	0.47
Age	3	-16.44	2.73	0.06	12.72	0.53
Habitat Function	3	-16.30	2.87	0.05	12.65	0.59
Vegetation Community	3	-16.19	3.98	0.05	12.59	0.64
Habitat Condition	3	-16.15	3.02	0.05	12.57	0.69
Woody Vegetation	3	-15.66	3.51	0.04	12.33	0.73
Landscape Integrity	3	-15.53	3.64	0.04	12.26	0.76
Size	3	-15.51	3.66	0.04	12.25	0.80
Canopy Cover	3	-15.48	3.68	0.04	12.24	0.84
Vegetation Vertical Structure	3	-15.31	3.85	0.03	12.16	0.87
Road & Rail	3	-15.29	3.87	0.03	12.15	0.90
Habitat Potential	3	-15.24	3.92	0.03	12.12	0.93
Floristic Quality	3	-15.19	3.98	0.03	12.09	0.97
Habitat Function (without Biodiversity Rank)	3	-15.12	4.05	0.03	12.06	1.00
Cowardin Type	4	-11.06	8.11	0.00	12.39	1.00
Ecoregion	5	-6.91	12.26	0.00	13.45	1.00

Appendix 9. Model selection of 17 environmental variables, plus an intercept-only model, using AICc to predict variables that influence small mammal richness in 14 restored wetlands in 2020 and 2021 in West Virginia, USA.

Model	K	AICc	Delta AICc	AICc Weight	Log Likelihood	Cumulative Weight
Size	2	51.39	0.00	0.16	-23.15	0.16
Landscape Integrity	2	51.73	0.34	0.14	-23.32	0.30
Habitat Function (without Biodiversity Rank)	2	52.19	0.80	0.11	-23.55	0.40
Intercept-Only	1	52.22	0.83	0.11	-24.94	0.51
Road & Rail	2	52.42	1.02	0.10	-23.66	0.61
Habitat Potential	2	53.13	1.74	0.07	-24.02	0.67
Floristic Quality	2	54.13	2.73	0.04	-24.52	0.72
Canopy Cover	2	54.38	2.99	0.04	-24.65	0.75
Habitat Condition	2	54.59	3.20	0.03	-24.75	0.78
Condition	2	54.60	3.21	0.03	-24.77	0.82
Habitat Function	2	54.63	3.24	0.03	-24.77	0.85
Woody Vegetation	2	54.63	3.24	0.03	-24.86	0.88
Vegetation Vertical Structure	2	54.82	3.42	0.03	-24.89	0.91
Wetland Breeding Bird Occupancy	2	54.88	3.48	0.03	-24.90	0.94
Age	2	54.89	3.49	0.03	-24.94	0.97
Vegetation Community	2	54.97	3.58	0.03	-24.83	0.99
Cowardin Type	3	58.07	6.68	0.01	-24.83	1.00
Ecoregion	4	60.39	9.00	0.00	-23.97	1.00