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Vegetation, invertebrate, and wildlife community rankings and habitat analysis of mitigation wetlands in West Virginia

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Abstract

Numerous efforts have been made in West Virginia to construct and restore compensatory wetlands as mitigation for natural wetlands destroyed through highway development, timbering, mining, and other human activities. Because such little effort has been made to evaluate these wetlands, there is a need to evaluate the success of these systems. The objective of this study was to determine if mitigation wetlands in West Virginia were adequately supporting ecological communities relative to naturally occurring reference wetlands and to attribute specific characteristics in wetland habitat with trends in wildlife abundance across wetlands. Specifically, avian and anuran communities, as well as habitat quality for eight wetland-dependent wildlife species were evaluated. To supplement this evaluation, vegetation and invertebrate communities also were assessed. Wetland ranks were assigned based on several parameters including richness, abundance, diversity, density, and biomass, depending on which taxa was being analyzed. Mitigation wetlands consistently scored better ranks than reference wetlands across all communities analyzed. Canonical correspondence analysis revealed no correlations between environmental variables and community data. However, trends relating wetland habitat characteristics to community structure were observed. These data stress the need to maintain specific habitat characteristics in mitigated wetlands that are compatible with wildlife colonization and proliferation.

Introduction

An enormous array of wildlife depends on wetlands for all or part of their lives. Dwindling populations of wetland-dependent wildlife populations have resulted from years of losses in the wetland resource base across the U.S. (Mitsch and Gosselink 2000). In an attempt to mitigate for losses in wetland habitat, current legislation has mandated the construction of thousands of hectares of wetlands. To evaluate these wetlands, researchers have attempted to describe and quantify wetland functions (e.g., their role in providing ecosystem services) relative to naturally occurring reference wetlands. Such functions commonly evaluated include soil (Stolt et al. 2000) and hydrology (Ashworth 1997) processes, vegetation communities (Campbell et al. 2002), wildlife habitats (Delphey and Dinsmore 1993), or combinations of these (Brinson 1993;

Brinson and Rheinardt 1996; Wilson and Mitsch 1996). Although these functions have been evaluated exclusively in numerous studies, few studies have engaged in a comprehensive evaluation of multiple wetland functions to assess mitigation success. The term 'success' in itself, is quite variable, and often varies by project objectives (National Research Council 2001). This study addressed mitigation success in terms of a wetland's ability to support diverse native vegetation, avian, anuran, and invertebrate communities at a similar level to natural reference wetlands.

Vegetation communities were evaluated not only because they directly determine the distribution and abundance of wildlife populations by providing essential food and cover (MacArthur and MacArthur 1961; Evans and Wilson 1982; Anderson et al. 1999a; King et al. 2000; Naugle et al. 2000), but also because they indirectly affect wildlife by contributing to a variety of other wetland attributes including quantity and type of substrate for invertebrates (Murkin et al. 1992; Anderson and Smith 1998, 1999, 2000; King et al. 2000) and water chemistry (Goslee et al. 1997; Castelli et al. 2000). For a variety of reasons, invertebrates are extremely important in the functioning of wetlands as well and thus, similar to vegetation communities, can be viewed as surrogates to wetland health. They are particularly sensitive to long-term hydrologic cycles, water quality, and habitat type (Wiggens et al. 1980; Doupe and Horwitz 1995; Brooks 2000; Anderson and Smith 2004), which is often associated with vegetative structure and composition. In turn, invertebrates contribute to other wetland functions by assisting in litter decomposition, nutrient cycling (Cummins 1973; Merritt et al. 1984) and plant community regulation (Weller 1994). Thus, invertebrates aid in the transfer of nutrients from the sediments, detritus, and water column to higher-level organisms. They also have direct impacts on wildlife species that depend on them for food. In particular, waterfowl and other waterbirds (De Szalay and Resh 1996; Davis and Smith 1998; Anderson and Smith 1999; Anderson et al. 2000), as well as anurans (Anderson et al. 1999a; Lima and Magnusson 2000), depend on invertebrates for food. It is clear that invertebrates play a vital role in wetland function and are integral in analyzing the health of these ecosystems.

Considering more than 50% of the 800 protected migratory birds rely on wetlands (Wharton et al. 1982), it is clear that an avian component is necessary in the evaluation of mitigation wetlands. Avian communities are good indicators of wetland function because, as a group, they exhibit a wide range of habitat requirements, and have adapted to the variety of vegetative cover types and water regimes wetlands provide (Anderson et al. 1996; Davis and Smith 1998; Melvin and Webb 1998; Anderson and Smith 1999; Weller 1999; Naugle et al. 2000). As well, they have diverse diets with many being herbivorous or omnivorous, preferring such foods as seeds, fruit, invertebrates, amphibians, and small mammals (Gonzalez et al. 1996; Anderson et al. 1996; De Szalay and Resh 1997; Davis and Smith 1998; Anderson and Smith 1999; Weller 1999). A greater diversity of birds should indicate suitable and diverse habitats and an adequate food supply.

Anurans rely exclusively on wetlands (Michael and Smith 1985; Dodd and Cade 1998; Lehtinen et al. 1999; Semlitsch 2002), specifically for hibernation, foraging, breeding, and interspersion habitat for different life stages. In turn, anuran populations provide insight into water quality and temporal variations in hydrology (Beattie and Tyler-Jones 1992; Anderson et al. 1999a; Semlitsch 2002). While anurans often feed on numerous invertebrate species (Anderson et al. 1999b; Lima and Magnusson 2000), they are an important food source for numerous other invertebrates and vertebrates alike (Bridges 1999; Lardner 2000), thus making them a valuable link in a complex food web (Weller 1999).

The development of wildlife habitat models is important because researchers must often assign relative values to habitat to support objectives for mitigation. Some models that have been created include the Wetland Evaluation Technique (Adamus 1983; Adamus and Stockwell 1983), Habitat Assessment Technique (Cable et al. 1989), and the Avian Richness Evaluation Model (Adamus, 1993). Species-specific models often used today are the Habitat suitability index (HSI) models developed by the U.S. Fish and Wildlife Service (1981). Based on natural history requirements for a particular species, these models use habitat parameters considered pertinent to a species survival to calculate an index ranging from 0 to 1 (1 represents optimal habitat). Depending on the HSI model, the habitat parameters evaluated may have significant implications for other wildlife taxa as well, which can provide further insight into overall habitat quality for wildlife for a given area. Although the validity of HSI models are sometimes questioned (Bender et al. 1996), the models still provide insight into the habitat structure of wetlands and are often used by resource agencies to determine mitigation requirements (Morrison et al. 1992).

Only three studies have evaluated the success of mitigation wetlands in West Virginia, and two of them (McConnell and Samuel 1985; R.H. Fortney, West Virginia University unpublished report) excluded an evaluation of invertebrates while the other exclusively evaluated production of only one invertebrate taxon in one constructed wetland (Johnson et al. 2000). Moreover, two papers (McConnell and Samuel 1985; Johnson et al. 2000) did not evaluate vegetation. As such, there is a need to evaluate the ability of mitigation wetlands in the mid-Appalachians, and in particular West Virginia, to support diverse vegetation and wildlife communities. Likewise, to maintain the significant role wildlife plays in the development of wetland ecosystems across this region, there is a need to identify wetland habitat characteristics that are associated with wildlife distribution and abundance. Therefore, researchers can develop adequate monitoring protocols and construct future wetlands that are compatible with wildlife proliferation. The objective of this study was to rank order the mitigated wetlands to determine which mitigation wetlands were best and to determine why wildlife were distributed in which wetlands. In doing so, we sought to attribute specific characteristics in wetland habitat with trends in wildlife abundance across wetlands. Specifically, avian and anuran communities, as well as habitat quality for eight wetland-dependent wildlife species were evaluated. To complement this evaluation, vegetation and invertebrate communities also were assessed. This study was designed to assist in the creation of future monitoring protocols for mitigation wetlands in West Virginia and to guide in the development of future mitigation projects by identifying individual wetlands that have abundant and diverse vegetative, wildlife, and invertebrate communities.

Methods

Study sites

West Virginia can be classified into three regions (Fenneman 1938; Figure 1). The unglaciated Western Hill section is the largest province in West Virginia, and includes the Appalachian Plateau between the Ohio River and the mountainous area to the east. Most of the hills in the northern and western portions of the state are $\leq$450 m in elevation. Southern sections of this region, however, reach elevations $\geq$900 m and can exceed 1000 m. The Allegheny Mountain section includes the high mountains that lie in the Cheat River system and in the headwaters of the North Branch of the Potomac River. This section contains the highest elevations in West Virginia with many ridges reaching between 1200 and 1375 m in elevation. This area contains the Allegheny Mountains that extend northward from West Virginia into western Maryland and central Pennsylvania. The Ridge and Valley region is located east of the Allegheny Front, and is drained primarily by the Potomac River. This region is a lowland area that, as its name implies, contains numerous interspersed ridges that form a narrow belt along the eastern margin of the state. The elevation of valley floors ranges from 300 to 400 m with ridges reaching $\geq$1219 m in elevation.

Eleven mitigation wetlands were evaluated in this study: Walnut Bottom, VEPCO, Buffalo Coal, Elk Run, Leading Creek, Sugar Creek, Sand Run, Triangle, Trus Joist MacMillan, Enoch Branch, and Bear Run (Table 1 and Figure 1). These wetlands were created or restored as compensation for wetland losses sustained for different human activities including highway development, facility construction, and mining. Almost every mitigation site was located near some form of human disturbance. In fact, many were located adjacent to roads with moderate to heavy traffic. Wetlands ranged in age from 4 to 21 years ($\bar{x} = 9.0$, SE = 1.7; Table 1) and ranged in elevation from 265 to 1036 m ($\bar{x} = 586$, SE = 75.9). Size ranged from 3.0 to 9.5 ha ($\bar{x} = 5.8$, SE = 0.80). All mitigation wetlands were classified as palustrine emergent or unconsolidated bottom (Cowardin et al. 1979).

Four naturally occurring reference wetlands were selected for comparisons with mitigation

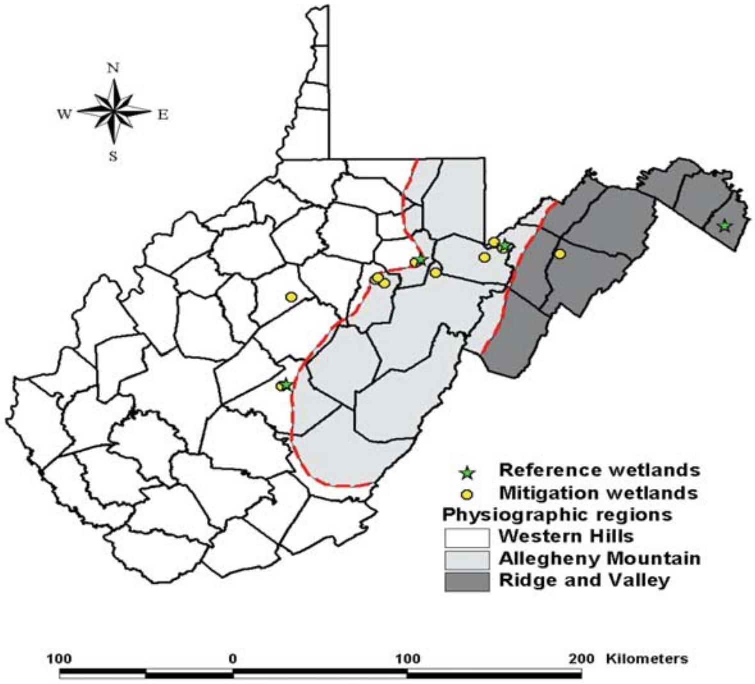

Figure 1. Study site locations for 11 mitigation and 4 reference wetlands in West Virginia, 2001–2002.

Table 1. List of 11 mitigation and 4 reference wetland study sites in West Virginia, including site name, year constructed, size (ha), source builder, Universal Transverse Mercator (UTM) coordinates, 7.5 minute quadrangle, basin, and watershed, 2001–2002.

Site name	Year	Size (ha)	Source	UTM Y	UTM X	Quad	Basin	Watershed
Mitigation sites								
Walnut Bottom	1997	9.5	Division of Hwys	4334210	673914	Old Fields	S. Branch of Potomac R.	S. Branch of Potomac R.
VEPCO	1995	7.0	VA Electric Power	4337900	641300	Mt. Storm	Cheat River	Blackwater River
Buffalo Coal	1981	9.0	Davis Trucking Co.	4332100	630900	Davis	Cheat River	Blackwater River
Elk Run	1981	3.8	Island Crk Coal Co.	4342000	636250	Davis	N. Branch of Potomac R.	Elk Run
Leading Creek	1995	8.6	Division of Hwys	4321563	602550	Montrose	Tygart Valley	Leading Creek
Sugar Creek	1995	6.8	Division of Hwys	4328850	591470	Belington	Tygart Valley	Laurel Creek
Sand Run	1992	3.0	Division of Hwys	4315060	573140	Buckhannon	Tygart Valley	Sand Run
Triangle	1992	3.1	Division of Hwys	4316950	568500	Buckhannon	Tygart Valley	Buckhannon River
Trus Joist Macmillan	1994	3.2	TJM Timber Co.	4318340	569560	Century	Tygart Valley	Buckhannoan River
Enoch Branch	1997	3.4	Division of Hwys	4247300	514550	Widen	Gauley River	Muddlety Creek
Bear Run	1993	6.2	WV Dept Env. Prot.	4305780	519750	Glenville	Little Kanawha	Little Kanawha
Reference sites								
Altona Marsh	N/A	15.2	N/A	4353000	768600	Middleway	Shenandoah River	Shenandoah River
Elder Swamp	N/A	28.0	N/A	4340000	642200	Mt. Storm Lake	Cheat River	Blackwater River
Meadowville	N/A	6.5	N/A	4330920	593940	Nestorville	Tygart Valley	Laurel Creek
Muddlety	N/A	10.4	N/A	4248480	516790	Widen	Gauley River	Muddlety Creek

wetlands: Altona Marsh, Elder Swamp, Meadowville, and Muddlety (Table 1, Figure 1). Each reference wetland represented a geomorphic setting (as described above) within the state and was selected relative to mitigation wetlands within that setting. One reference wetland was chosen for each area based on its similarity in location and elevation to mitigation sites. All reference wetlands were undisturbed (i.e., lacked evidence of logging or grazing) and were typical of the undisturbed palustrine scrub–shrub and emergent wetlands with scrub–shrub borders that occurred in the region. Because some reference wetlands were relatively larger than mitigation sites, only portions of reference sites were selected for study. Reference sites ranged in elevation from 170 to 1000 m ($\bar{x} = 582$, SE $= 169.5$; Table 1) and ranged in size from 6.5 to 28.0 ha ($\bar{x} = 15.1$, SE $= 4.7$. All reference wetlands were classified as palustrine emergent or scrub–shrub wetlands (Cowardin et al. 1979). Detailed descriptions of mitigation and reference wetlands are provided in Balcombe (2003).

Vegetation community sampling

We conducted vegetation sampling in June and July of 2001, and in July of 2002. Sampling was conducted according to techniques incorporated by Stephenson and Adams (1986). Plant communities were stratified based on distinct communities present, and representative communities were sampled using permanently marked 0.05 ha quadrats (25×20 m). At each wetland, at least one quadrat was used to sample each distinct plant community. Within each quadrat all live stems of trees (≥10 cm diameter at breast height, DBH) and small trees (2.5–9.9 cm DBH) were measured at DBH and counted to species. In addition, saplings (individuals <2.5 cm DBH but ≥1.0 m tall) were counted. Within each 0.05 ha quadrat, two 5.0×5.0 m plots were placed evenly along the center line of the transect. Within these plots, seedlings (individuals >10 cm but less than <1.0 m tall) and shrubs (including woody vines) were counted to species. Five 1.0×1.0 m plots were placed along the same center line. Within these plots, small seedlings (individuals ≤ 10 cm tall) were counted to species. In addition, percent cover of herbaceous plants, exposed substrate, woody debris, and bryophytes were recorded within 1.0×1.0 m plots. Detailed descriptions of vegetation community sampling are provided in Balcombe (2003).

Invertebrate sampling

We conducted invertebrate sampling according to Anderson and Smith (1996, 2000) during the summers of 2001 and 2002. Specifically, we collected 620 samples in July and September of 2001 and another 620 samples were collected in April and June of 2002. Samples were taken at different times both years to maximize representative taxa. Wetlands were stratified based on wetland classification (Cowardin et al. 1979), and specimens were collected at each of 10 random points within open water and emergent from all wetlands, and from scrub–shrub areas in Elder Swamp, because scrub–shrub did not exist in the other wetlands. At each point, we used a 5-cm diameter benthic core (15-cm deep) and a 7.5-cm diameter water-column sampler (Swanson 1983) to collect benthic and water column specimens, respectively. Water column samples were sieved in the field using a 500-μm sieve (Huener and Kadlec 1992) and preserved in 70% ethanol. Benthic samples were placed in bags, refrigerated, and processed within 10 days of collection (Anderson and Smith 2000). Biomass was obtained by oven-drying samples at 55° for ≥48 h to a constant mass and using an analytical scale. Details of invertebrate sampling methodologies are provided in Balcombe (2003).

Avian and anuran communities

We evaluated avian communities by sampling breeding bird populations using point count (0.78 ha plots) surveys (Ralph et al. 1995). We visited wetlands twice between late May and late June, 2001 and 2002, when breeding birds were most active. We conducted 10-min point counts that occurred between 30 min before sunrise and 1000 h, under acceptable weather conditions (Ralph et al. 1995).

We evaluated anuran communities using nocturnal call count surveys that followed standardized protocols developed by the U.S. Fish and Wildlife Service (Casey and Record 1999). To account for temporal breeding differences between species, each wetland was visited monthly in March, April, and May of 2001 and 2002. We collected data for 3 min at each sampling point following a 1–2 min settling period. Frogs were identified to species and relative abundances were recorded by assigning a Wisconsin index value of intensity to each species' call (Mossman 1994). Detailed descriptions of avian and anuran sampling schemes are included in Balcombe (2003).

Habitat quality

Habitat quality was assessed using species-specific habitat suitability index (HSI) models. The models chosen had broad taxonomic coverage and included one reptile (snapping turtle, *Chelydra serpentina*, Graves and Anderson 1987), one amphibian (red-spotted newt, *Notophthalmus virdescens*, Sousa 1985), three mammals (beaver, *Castor canadensis*, Allen 1983; muskrat, *Ondatra zibethicus*, Allen and Hoffman 1984; mink *Mustela vison*, Allen 1984), and three bird species (one wading bird: great blue heron, *Ardea herodias*, Short and Cooper 1985; one waterfowl species: wood duck, *Aix sponsa*, Sousa and Farmer 1983; one passerine: red-winged blackbird, *Agelaius phoeniceus*; Short 1985). All evaluated species had wide distributions throughout West Virginia, and possessed life-history components (i.e., foraging, reproduction, and interspersion) that were compatible with palustrine emergent and palustrine scrub-shrub wetlands. Numerous methodologies were incorporated in quantifying the 38 variables encompassing the eight models (Balcombe, 2003).

Statistical analyses

Vegetation metrics (species richness, diversity, and evenness were calculated using PC-ORD software (McCune and Mefford 1999) for each of 45 and 15 quadrats within mitigation and reference wetlands, respectively. Metrics were calculated for all species and for native species only. Species diversity was calculated using the Shannon index (Shannon and Weaver 1949). Average cover was calculated for each species and totaled to get a total coverage for each plot. These values were averaged to obtain mean total coverage for each wetland. Each herbaceous species was assigned a wetland indicator status value (WIS): obligate = 1, facultative wetland = 2, facultative = 3, facultative upland = 4, and upland = 5 (U.S. Fish and Wildlife Service 1996). From coverage and WIS values, mean weighted averages (Carter et al. 1988; Wentworth et al. 1988; Atkinson et al. 1993) were calculated based on the following formula:

$$\mathrm{WA} = (y_1u_1 + y_2u_2 + \cdots + y_mu_m)/100$$

where y_1y_2 = relative basal area (trees and small trees) or relative cover estimates (herbaceous plants) for each species, and u_1u_2 = the WIS for each species (Atkinson et al. 1993).

Species richness was calculated for birds and anurans by averaging the total number of species observed in each sampling plot per wetland (for birds, this was expressed as the no. of species/50 m plot). Invertebrate richness was calculated in a similar manner, but taxa were classified only to family. Avian abundance was calculated by averaging the total number of individuals observed in each sampling plot per wetland. Similar to vegetation diversity, avian and invertebrate diversity were calculated using the Shannon index. Anuran abundance was calculated separately both by Wisconsin index (WI) calling intensity values of particular species (Mossman 1994), and by actual estimations of calling individuals. The WI value was based on the following ranking system: we assigned a ranking of one to species with nonoverlapping calls and when an exact count of individuals could be made, a ranking of two for species whose calls overlapped and only estimations of numbers could be made, and a three for species that were calling in full chorus. Details involving abundance estimations are provided in Balcombe (2003). Invertebrate density and biomass were calculated separately for both the core sampler and the water-column sampler.

The purpose of this study was to rank order the mitigated and reference wetlands to determine which had the overall best communities based on the metrics we measured and not to compare between mitigation and reference wetlands. A detailed account of statistical mean comparisons between mitigated and reference wetlands for vegetation, invertebrate, and wildlife communities are provided in Balcombe (2003). Instead, we used a rank transformation (Conover and Iman 1981; Potvin and Roff 1993) of the metric means for each wetland (ranked on a scale of 1–15) based on observed means of each wetland relative to other mitigation and reference wetland means. A rank of 1 was given to wetlands that scored the best or highest value for a particular metric, whereas 15 was given to wetlands that scored the worst or lowest value relative to other evaluated wetlands. Wetlands with similar means were averaged and ranked the same number, so in some instances, scales may not extend all the way to 15. Separate ranks were calculated individually for vegetation, invertebrate, avian, and anuran communities, as well as for habitat quality to gauge the relative success of individual wetlands in supporting a particular community. Furthermore, an overall rank representing means across all metrics were assigned to each wetland. Ranks are an appropriate way to deal with normality issues of numerous dependent variables simultaneously to provide each factor equal weight and when trying to show differences between numerous factors simultaneously Potvin and Roff 1993, Anderson et al. 1999c, 2000; Anderson and Tacha 2002).

Vegetation ranking was based on combining ranks for species richness, evenness, diversity, and weighted averages. Overall avian ranks were calculated by averaging total species richness, diversity, and abundance ranks, as well as abundance ranks for waterbirds, waterfowl, and passerines. Overall anuran ranks were based on mean rankings of total species richness, Wisconsin index (WI) value, and abundance, as well as individual WI and abundances for the seven frog species sampled. These species included spring peeper (*Pseudacris crucifer*), gray treefrog (*Hyla chrysoscelis*), American bullfrog (*Rana catesbeiana*), wood frog (*Rana sylvatica*), green frog (*Rana clamitans*), American toad (*Bufo americanus*), and pickerel frog (*Rana palustris*). Overall invertebrate ranks represented combined rankings of familial richness, diversity, density, and biomass for core and water-column samples. Habitat suitability index (HSI) ranks were based on mean ranks for all eight species evaluated.

A completely randomized analysis of variance (ANOVA) model on rank transformed data (Conover and Iman 1981; Potvin and Roff 1993; Anderson et al. 2000) followed by the Tukey's honestly significantly difference (HSD) test was used to test for differences among individual wetlands. All differences were considered significant at an alpha level of 0.05.

We used canonical correspondence analysis (CCA; ter Braak 1986), using PC-ORD software to correlate environmental variables to avian, anuran, and invertebrate abundance. Canonical correspondence analysis is a multivariate direct ordination method that performs a least-squares linear regression of environmental variables on site (wetland) scores determined through correspondence analysis (Gauch 1982). Species are ordered on axes constrained by linear combinations of environmental factors. The eigenvalues associated with each axis indicate the relative ability of the axis to order or separate species distributions. Intraset correlation coefficients represent the strength of environmental variables in structuring the ordination.

We ordinated avian, waterbird, anuran, and invertebrate abundances to eight environmental factors to determine which variables influenced abundance of each taxa (Table 2). Only species present in ≥10% of wetlands were used in this analysis due to the potential negative effect of outliers (Gauch 1982). We used eigenvalues, percentage of variation explained in species data, and intraset correlations of environmental variables to each axis to assess the relative importance of environmental variables in structuring species composition. A Monte Carlo simulation (McCune and Mefford 1999) with 1000 permutations was used to test the null hypothesis that there was no relationship between species and environmental matrices (p = 0.05). Although p values were reported for all three axes, the significance of correlations between matrices was determined only by axis one p values because this axis accounted for the most variation in all analyses (B. McCune, Oregon State University, personal communication).

Results

Wetland rankings

Total mean ranks combining vegetation, anuran, avian, invertebrate, and habitat rankings were similar between all wetlands ($F_{14,60}$ = 1.26, p = 0.260; Table 3). Nonetheless, Leading Creek,

Table 2. Variables that were used for ordination of abundance of organisms for all 15 wetlands and 11 mitigated wetlands, West Virginia 2001–2002.

Wetland variables	All wetlands				Mitigated wetlands			
	Avian	Waterbirds	Anurans	Invertebrates	Avian	Waterbirds	Anurans	Invertebrates
Age					X	X	X	X
Benthic invertebrate diversity	X	X	X		X	X	X	
Nektonic invertebrate diversity	X	X	X		X	X	X	
Percent emergent vegetation	X	X	X	X	X	X	X	X
Percent open water	X	X	X	X	X	X	X	X
Percent submergent vegetation				X				X
Size					X	X	X	X
Vegetation diversity	X	X	X	X	X	X	X	X

Table 3. Vegetation, invertebrate, avian, anuran, and habitat suitability index (HSI) ranks, as well as total mean and scaled ranks[b] for 11 mitigation and 4 reference wetlands in West Virginia, 2001–2002.

	Mitigation wetlands															
	Walnut Bottom		VEPCO		Buffalo Coal		Elk Run		Leading Creek		Sugar Creek		Sand Run		Triangle	
	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Vegetation rank[a]	10.4abcd	0.3	2.8e	1.1	7.1abcde	0.8	4.1de	0.6	6.3bcde	0.9	13.1a	1.4	5.6cde	0.9	7.2abcde	1.6
Invertebrate rank[a]	3.9c	1.5	12.0ab	0.7	8.4abc	1.6	5.0c	0.7	8.3abc	1.1	7.8abc	1.4	9.8abc	1.4	4.6c	0.8
Avian rank[a]	4.3b	1.7	11.8a	1.4	6.7ab	1.8	4.2b	0.7	6.7ab	1.8	8.9ab	1.4	8.3ab	0.9	7.9ab	1.4
Anuran rank[a]	6.2bcd	0.9	9.8abc	1.0	6.5bcd	1.2	9.8bcd	1.0	6.1cd	0.4	8.2abc	0.6	7.4abcd	0.9	6.8bcd	1.1
HSI rank[a]	7.3ab	1.7	5.8ab	1.3	8.7ab	1.8	9.4ab	2.1	3.2b	1.1	6.1ab	1.8	12.3a	0.6	6.1ab	1.3
Total mean rank[a]	6.4a	1.2	8.4a	1.8	7.5a	0.4	6.5a	1.3	6.1a	0.8	8.8a	1.2	8.7a	1.1	6.5a	0.6
Scaled rank[a]	2.0		11.0		6.0		3.5		1.0		13.0		12.0		3.5	
	Mitigation Wetlands cont.						Reference Wetlands									
	Trus Joist MacMillan		Enoch Branch		Bear Run		Altona Marsh		Elder Swamp		Meadow-ville		Muddlety			
	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE		
Vegetation rank[a]	5.3de	1.8	5.7cde	0.9	6.2bcde	1.7	12.0abc	0.3	10.3abcd	1.7	11.7abc	0.9	12.3ab	2.3		
Invertebrate rank[a]	7.3abc	1.9	12.9a	0.5	6.1bc	0.8	4.9c	1.7	12.7a	1.1	9.7abc	1.4	6.6bc	1.0		
Avian rank[a]	3.7b	0.8	9.1ab	1.6	8.6ab	1.6	7.0ab	0.9	13.1ab	0.4	8.8ab	1.4	8.8ab	2		
Anuran rank[a]	8.8abc	1.2	3.7d	0.7	9.0abc	1.1	11.8a	0.4	8.4abc	0.7	10.6ab	1.0	6.6bcd	1		
HSI rank[a]	7.9ab	1.8	7.8ab	1.7	9.6ab	1.6	5.4ab	2.2	6.4ab	1.7	8.8ab	1.9	5.8ab	1.5		
Total mean rank[a]	6.6a	0.9	7.8a	1.6	7.9a	0.7	8.2a	1.5	10.2a	1.3	9.9a	0.6	8.0a	1.2		
Scaled rank[a]	5.0		7.0		8.0		10.0		15.0		14.0		9.0			

[a] Different letters following means indicate a significant difference at $p = 0.05$.
[b] Wetlands with similar mean ranks were assigned similar scaled ranks.

Trus Joist MacMillan, Triangle, Walnut Bottom, and Elk Run scored the lowest five overall ranks of all wetlands (Table 3). Triangle and Trus Joist MacMillan scored similar ranks, as did Walnut Bottom and Elk Run. The lowest overall rank was assigned to Leading Creek whereas the fourth and fifth lowest were Walnut Bottom and Elk Run. On the contrary, Elder Swamp, Meadowville, Sugar Creek, Altona Marsh and Sand Run scored the five highest ranks, with Elder Swamp scoring the highest rank of all wetlands. Altona Marsh and Sand Run scored similar scores. The other three wetlands scored ranks in the middle.

Vegetation rankings were significantly different among the lowest and highest ranked wetlands ($F_{14,75} = 6.66$, $p < 0.001$; Table 3). VEPCO, Elk Run, and Trus Joist MacMillan scored lower vegetation ranks than Sugar Creek, Muddlety, Altona Marsh, and Meadowville, which scored the highest vegetation ranks.

Enoch Branch and Sand Run also scored relatively low ranks, but these were only statistically lower than Muddlety and Sugar Creek.

Invertebrate rankings also were different among the lowest and highest ranked wetlands ($F_{14,105} = 5.15$, $p < 0.001$; Table 3). Walnut Bottom, Triangle, Elk Run, and Altona Marsh ranks were significantly lower than Elder Swamp, Enoch Branch, and VEPCO. Bear Run also scored a low rank, which was significant only to Elder Swamp and Enoch Branch.

Avian rankings were different among the lowest and highest ranked wetlands as well ($F_{14,75} = 3.03$, $p = 0.001$; Table 3). Trus Joist MacMillan, Elk Run, and Walnut Bottom scored ranks significantly lower than the highest ranked wetlands, Elder Swamp and VEPCO. Although not significant, Buffalo Coal and Leading Creek scored fourth and fifth lowest ranks while Enoch Branch and Sugar Creek scored the third and fourth highest ranks.

In addition, anuran ranks were different among the lowest and highest ranked wetlands ($F_{14,240} = 5.13, p < 0.001$; Table 3). The wetlands with the 2 lowest ranks, Enoch Branch and Leading Creek, were statistically lower than the wetlands with the 2 highest ranks, Altona Marsh and Meadowville. Although results were not significant, Walnut Bottom, Buffalo Coal, and Muddlety scored the next lowest anuran ranks of all wetlands. Similarly, next to Altona Marsh and Meadowville, VEPCO, Elk Run, and Bear Run scored the next highest anuran ranks of all wetlands.

Habitat suitability index ranks were similar among all wetlands $F_{14,105} = 1.76$, $p = 0.055$; Table 3). Nonetheless, Leading Creek, Altona Marsh, Muddlety, VEPCO, Sugar Creek, and Triangle scored the lowest ranks and Sand Run, Bear Run, Elk Run, Meadowville, and Buffalo Coal scored the highest habitat ranks.

Canonical correspondence analysis

The Monte Carlo simulation of all three axes indicated that environmental variables predicted species all taxa abundance no better than sets of scores randomly assigned to samples, both for all wetlands and for mitigation sites only. The probabilities of achieving the relationships by chance for all avians, waterbirds, anurans, benthic invertebrates, and nektonic invertebrates are provided in Table 4.

Discussion

Wetland rankings

Reference wetlands generally scored higher (worse) overall ranks than mitigation wetlands, with most reference wetlands scoring at or near the bottom. These ranks largely reflected differences in wetland size, heterogeneity, and disturbance. For instance, larger, more heterogeneous wetlands with fewer disturbances were more successful than smaller, more monotypic wetlands (i.e., too much open water or emergent vegetation, or lacking diverse hydrologic gradients). This trend distinguished mitigation wetlands not only from reference wetlands, but from other mitigation wetlands as well. Within this trend emerged the relative importance of which characteristics appeared to drive wetland success. It appeared, for instance, that heterogeneity was more important than size, which was more important than disturbance. These specifications were mostly accurate in predicting the positions of individual wetlands along the rank spectrum (i.e., larger, more heterogeneous wetlands scored better overall ranks).

In addition, specific habitat characteristics tended to affect the position of wetland rankings for each metric evaluated. For instance, wetlands that scored good vegetation ranks tended to be younger and more disturbed while wetlands that scored good anuran ranks were generally larger and more heterogeneous, contained relatively equal ratios of emergent vegetation to open water, and lacked predatory fish. Wetlands that scored good avian ranks also tended to be large and heterogeneous with relatively equal ratios of emergent vegetation to open water; but the existence of natural and artificial perching structures as well as the proximity to forest cover also were important. The habitat characteristic that appeared to position wetlands along the invertebrate rank spectrum was submerged aquatic vegetation (SAV), with increased SAV leading to better ranks. Finally, the relative abundance of tree and shrub cover, both within and around wetlands, appeared to be the most significant characteristics resulting in good wetland ranks among HSI models.

Table 4. Summary of parameters including general structure and intraset correlation coefficients for all environmental variables in the canonical correspondence analysis of all avian species, waterbirds, anurans, and benthic and nektonic invertebrates abundance within all wetlands ($n = 15$) and for mitigation wetlands only ($n = 11$) in West Virginia, 2001–2002.

General structure	Axis 1		Axis 2		Axis 3	
	All wetlands	Mitigated only	All wetlands	Mitigated only	All wetlands	Mitigated only
Total bird abundance						
Eigenvalue	0.191	0.279	0.110	0.211	0.097	0.137
Monte Carlo Test result (p value[a])	0.841	0.764	0.907	0.391	0.415	0.255
% Variance explained	12.1	21.4	6.9	16.2	6.1	10.6
Predictor variables (correlation coefficients)						
Size		0.800		0.651		0.303
Age		0.163		−0.317		−0.706
% Emergent vegetation	−0.503	−0.091	−0.526	−0.219	−0.053	0.61
Vegetation diversity	0.367	−0.325	−0.627	−0.720	0.209	0.173
% Open water	0.689	−0.031	0.312	0.439	0.442	−0.477
Benthic invertebrate diversity	−0.736	−0.149	0.032	−0.176	−0.053	−0.175
Nektonic invertebrate diversity	0.395	−0.211	−0.152	0.668	−0.413	0.074
Waterbird abundance						
Eigenvalue	0.265		0.082		0.057	
Monte Carlo Test result (p value[a])	0.570		0.622		0.170	
% Variance explained	23.5		30.8		35.9	
Predictor variables (correlation coefficients)						
% Emergent vegetation	0.629		−0.030		0.419	
Vegetation diversity	0.728		0.402		−0.281	
% Open water	−0.307		−0.186		−0.340	
Benthic invertebrate diversity	0.018		−0.178		−0.636	
Nektonic invertebrate diversity	0.393		−0.888		0.147	
Anuran abundance						
Eigenvalue	0.040	0.088	0.025	0.048	0.016	0.030
Monte Carlo Test result (p value[a])	0.843	0.819	0.307	0.390	0.057	0.036
% Variance explained	15.0	32.2	9.5	17.5	6.2	11.0
Predictor variables (correlation coefficients)						
Size		0.234		−0.167		0.339
Age		−0.327		−0.374		−0.571
% Emergent vegetation	−0.273	−0.003	−0.244	−0.365	0.115	0.382
Vegetation diversity	0.092	0.154	−0.564	−0.597	0.66	−0.071
% Open water	0.564	0.048	0.295	0.621	0.303	−0.229
Benthic invertebrate diversity	0.211	−0.71	−0.478	−0.282	−0.677	−0.219
Nektonic invertebrate diversity	0.814	−0.433	0.106	−0.514	−0.230	−0.253
Benthic invertebrates						
Eigenvalue	0.273	0.329	0.099	0.172	0.046	0.053
Monte Carlo Test result (p value[a])	0.316	0.116	0.031	0.035	0.011	0.547
% Variance explained	25.3	36.2	9.1	18.9	4.3	5.9
Predictor variables (correlation coefficients)						
Size		0.636		0.343		−0.303
Age		−0.278		−0.831		−0.402
% Emergent vegetation	−0.742	−0.546	−0.664	0.318	−0.091	0.410
Vegetation diversity	0.556	−0.408	0.088	−0.346	0.827	−0.071
% Open water	0.537	0.758	0.830	−0.260	0.148	−0.212
Nektonic invertebrates						
Eigenvalue	0.331	0.434	0.167	0.247	0.069	0.179
Monte Carlo Test result (p value[a])	0.069	0.180	0.154	0.320	0.607	0.106
% Variance explained	17.1	25.4	8.7	14.5	3.6	10.5
Predictor variables (correlation coefficients)						
Size		0.387		0.640		−0.628
Age		−0.706		−0.165		−0.406
% Emergent vegetation	0.000	−0.043	−0.706	0.122	−0.708	−0.276
Vegetation diversity	−0.289	−0.511	0.845	−0.014	−0.450	0.518
% Open water	0.440	0.429	0.767	−0.278	0.467	0.426

[a] P = proportion of randomized runs with eigenvalue greater than or equal to the observed eigenvalue [i.e., $P = 1\ (1 + \text{no. permutations} \geq \text{observed})/(1 + \text{no. permutations})$].

Environmental data

Canonical correspondence analysis yielded weak correlations between species and environmental data throughout all metrics analyzed. Some factors that may account for such weak correlations include species dominance overriding environmental factors, factor interactions, unmeasured variables, and chance (Kazmierczak et al. 1995). We believe that a larger sample size would have revealed the importance of these variables. Although statistical significance did not emerge regarding wetland habitat characteristics, it is clear that these attributes play a large role in structuring wildlife communities (Balcombe 2003). Indeed these data indicate that size, as well as percent emergent vegetation, submerged aquatic vegetation, open water, and shrub cover, in addition to the presence of snags and artificial nesting and perching structures, may play important roles in determining the habitat quality for a variety of communities in the wetlands we evaluated. Wetland size is known to affect overall avian richness (MacArthur and Wilson 1967; Tyser 1983; Delphey and Dinsmore 1993). Moreover, percent emergent vegetation is known to affect waterbird and waterfowl distribution (Kaminski and Prince 1981; Bookhout et al. 1989; Murkin et al. 1997), as well as anuran abundance (Stumpel and Van Der Voet 1998). Other studies have linked invertebrate community structure to quality and quantity of aquatic vegetation (Brown et al. 1988; Wilcox 1992; Streever et al. 1995; Zimmer et al. 2000), including submerged aquatic vegetation (Carpenter and Lodge 1986). Our study, however, found no such links between wildlife distribution and abundance and environmental factors.

Furthermore, although no statistical significance emerged correlating emergent vegetation to anuran abundance, a trend did appear to exist in correlating these two variables. For example, with the exception of VEPCO, the wetlands with the poorest rankings fell at the extreme ends of the spectrum with regards to percent emergent vegetation (i.e., ≤ 22.3 or ≥ 81.0; Balcombe 2003). One variable that was not quantified was the amount of snags present among wetlands. Indeed, the two wetlands that scored the best avian ranks (Trus Joist MacMillan and Elk Run) were the only two wetlands that contained abundant snags. These data, combined with those data presented in Balcombe (2003) show trends in habitat characteristics that contribute to wildlife colonization and proliferation.

Conclusions

An underlying assumption with respect to evaluating the success of mitigation wetlands based on vegetation and wildlife rankings, is that first, all ranks are weighted equally. Of course, some components may be more important than others and should therefore be weighted more heavily. For instance, overall avian and anuran species abundances were weighted similar to individual or guild species abundances. Although total species abundance could be weighted more heavily since it represents a combination of all species observed, it would be difficult to calibrate the value of metrics accurately, and thus, would lead to spurious results. Hence, metrics were weighted equally, and despite the implicit error in this assumption, comparisons can still be made as to the relative success of individual wetlands in supporting wildlife communities.

Another assumption of these analyses is that wetlands that scored lower ranks were 'better' or 'more successful' than wetlands that scored higher ranks. An important aspect to consider is development time. These data provided insight into the community dynamics of these mitigation wetlands at only one point in time. Based on results obtained in Balcombe (2003), it was clear that development time affects vegetation community structure and composition, and these results are reflected in the ranks assigned to individual wetlands. Specifically, three of four reference wetlands scored among the highest vegetation ranks of all 15 wetlands, which reflects the lower species richness and diversity values observed in reference wetlands (Balcombe 2003). Two (Elder Swamp and Meadowville) of the four wetlands (scored the highest rankings for all metrics combined). An evaluation of these wetlands in 10 or 20 years may yield entirely different rankings all together as autogenic and allogenic factors influence vegetative structure and composition, and hence, wildlife distribution and abundance. Thus, we do not think that poor rankings of reference wetlands reflects inadequate selection of reference wetlands. Nonetheless, these data provide researchers with a current index of the success of mitigation wetlands in West Virginia in supporting wildlife communities.

The strength of the overall index rankings lies in the comprehensive nature of the ranks themselves. By combining vegetation, invertebrates, avians, and anurans, researchers are provided with a comprehensive view of the ecological functions of these wetlands. This allows researchers to document trends in wetland structure that improve general habitat quality for wildlife, or to assess more specific trends that contribute to improving habitat quality for one particular taxa (i.e., anurans). This provides more latitude in creating management objectives for mitigated wetlands. Specifically, if future mitigation efforts focus on replacing anuran habitat, as opposed to general wildlife habitat, one could look for correlations between anuran distribution and abundance and wetland structure. The anuran rankings provided in this study could be applicable in such a scenario.

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