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Wildlife use of mitigation and reference wetlands in West Virginia

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Abstract

We evaluated avian and anuran communities in 11 mitigation and four reference wetlands throughout West Virginia, USA. Avian species richness ($P = 0.711$), diversity ($P = 0.314$), and abundance ($P = 0.856$) (expressed as mean ± S.E. per ha) were similar between mitigation (richness: 11.3 ± 0.40; diversity: 3.1 ± 0.53; abundance: 27.1 ± 2.2) and reference (richness: 11.2 ± 0.62; diversity: 2.8 ± 0.47; abundance: 28.5 ± 4.9) wetlands. Waterbird ($P = 0.013$) and waterfowl ($P = 0.013$) abundance were higher in mitigation (waterbird: 5.1 ± 1.5; waterfowl: 4.4 ± 1.4) than reference (waterbird: 0.44 ± 0.23; waterfowl: 0.24 ± 0.21) wetlands. Anuran (frogs and toads) species richness ($P = 0.023$), Wisconsin index (WI) calling values ($P < 0.001$), and abundance ($P < 0.001$) (expressed as mean ± S.E. per survey point) were higher in mitigation (richness: 2.01 ± 0.09; WI: 0.52 ± 0.03; abundance: 4.75 ± 0.66) than reference (richness: 1.47 ± 0.14; WI: 0.40 ± 0.17; abundance: 4.69 ± 1.18) wetlands. Evidence suggests that avian and anuran densities in mitigation wetlands are similar or in some cases higher than in natural (reference) wetlands.

Keywords: Anurans; Mitigation; Mitigation wetland; Habitat use; Birds; Frogs; Wetland-dependent species; Mitigation success; West Virginia

1. Introduction

Wetlands are important ecosystems that provide valuable habitat for wildlife. The destruction of wetlands across the U.S., however, has undermined the survival of some fish, shellfish, furbearing mammals, waterfowl, and amphibians that rely exclusively on these areas for survival (Mitsch and Gosselink, 2000). The Clean Water Act of 1972 was the first major legislation that protected our nation's wetland resource base, but it was not until the "no net loss" policy of the late 1980s that the government actively sought to mitigate for these losses that have impacted wetland-dependent wildlife across the country.

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Under the new policy, thousands of hectares of wetlands have been constructed to compensate for wetland destruction, but little monitoring has been conducted on the success of these newly created wetlands, particularly in West Virginia (National Research Council, 2001). Most studies that have addressed mitigation success have focused on wetland function with respect to hydrology, soils, and vegetation (Cummings, 1999; Moore et al., 1999; Zedler and Callaway, 1999; Stolt et al., 2000; Cole et al., 2001; Campbell et al., 2002). These parameters are excellent indicators of wetland function, but they yield limited insight into a wetland's direct ability to support wildlife populations. Indeed, it is assumed that adequate vegetation, hydrology, and location will precipitate wildlife colonization of newly created wetlands (Erwin, 1990; Hammer, 1992). But information regarding the ability of mitigation wetlands to replace lost wildlife habitat is lacking (National Research Council, 2001). Of particular concern is the replacement of waterbird and anuran habitat in the face of continued declines as a result of wetland destruction (Dahl, 1990; Weller, 1999; Semlitsch, 2002). For reasons listed below, these taxa are extremely important in the functioning of wetland ecosystems.

Numerous bird species require wetlands as their primary habitat. Eighty percent of breeding birds in North America, and more than 50% of the 800 protected migratory birds rely on wetlands (Wharton et al., 1982). Perhaps due to increased habitat diversity provided by the water surface (Ferguson et al., 1975; Weller, 1999), wetlands support higher avian species diversity (MacArthur, 1964; Mensing et al., 1998) and densities (Udevitz and Michael, 1982; Mensing et al., 1998) than their upland counterparts. Wetland birds are good indicators of function because, as a group, they exhibit a wide range of habitat requirements, have adapted to the variety of vegetative cover types and water regimes wetlands provide (McConnell and Samuel, 1985; Anderson et al., 1996; Melvin and Webb, 1998; Anderson and Smith, 1999; Weller, 1999; Naugle et al., 2000), and they eat a variety of foods including seeds, fruit, invertebrates, amphibians, and small mammals (Gonzalez et al., 1996; De Szalay and Resh, 1997; Davis and Smith, 1998; Anderson et al., 2000).

Like avian species, anurans are relatively easy to sample and possess unique habitat requirements. Because wetlands provide hibernation, foraging, breeding, and interspersion habitat for different life stages, anurans rely exclusively on wetlands for all or part of their life-cycle (Michael and Smith, 1985; Dodd and Cade, 1998; Lehtinen et al., 1999; Semlitsch, 2002). Hence, anuran populations can provide insight into water quality and temporal variations in hydrology (Beattie and Tyler-Jones, 1992; Anderson et al., 1999a; Semlitsch, 2002). They feed on numerous invertebrate species (Anderson et al., 1999b; Lima and Magnusson, 2000) and are an important food source for invertebrates and vertebrates alike (Bridges, 1999; Lardner, 2000). This makes them a valuable link between invertebrate populations and higher vertebrates in a complex food web (Weller, 1999). Moreover, physiological attributes such as their permeable skin, eggs without shells, gilled larvae, and ectothermic metabolism make them particularly vulnerable to habitat alterations, and thus excellent indicators of environmental health (Hall, 1980; Heyer et al., 1994; Semlitsch, 2002).

There is a need to evaluate the success of mitigation wetlands in supporting wildlife taxa that are considered good indicators of wetland health. This success can often best be determined through surveys of wildlife populations (Wilson and Mitsch, 1996; VanRees-Siewert and Dinsmore, 1996; Stevens et al., 2002). Natural wetlands are often used as standards of comparison because these areas are considered relatively stable and undisturbed (Brinson, 1993; Brinson and Rheinhardt, 1996; Wilson and Mitsch, 1996). The goal of this study was to evaluate the success of mitigation wetlands in West Virginia in supporting healthy wildlife communities. Therefore, we compared avian and anuran populations between mitigation and reference wetlands. As such, we tested the null hypotheses that anuran and avian richness, diversity, and abundance were similar between mitigation and reference wetlands.

2. Methods

2.1. Study area

We evaluated 11 constructed and partially restored mitigation wetlands (Walnut Bottom, VEPCO, Buffalo Coal, Elk Run, Leading Creek, Sugar Creek, Sand Run, Triangle, Trus Joist MacMillan, Enoch Branch, and Bear Run) and four reference wetlands (Altona Marsh, Elder Swamp, Meadowville, and Muddlety) in east-central West Virginia, USA (Fig. 1; Table 1). We

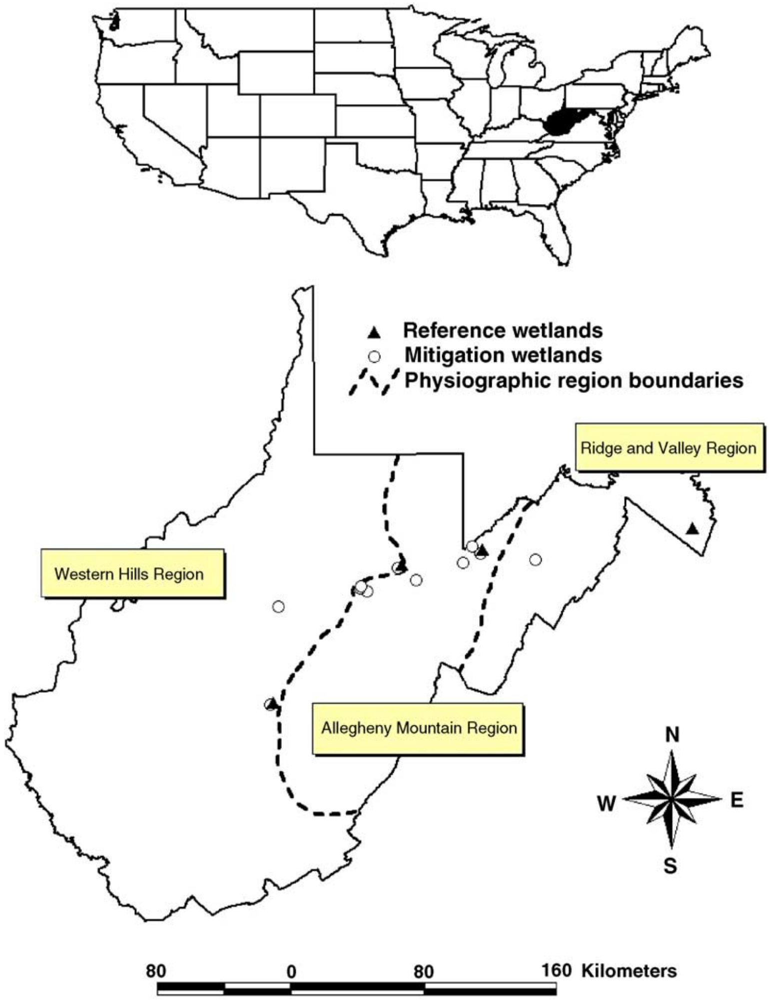

Fig. 1. Site locations of mitigation and reference wetlands in West Virginia, USA, 2001–2002.

stratified these sites into four areas representing three geomorphic settings within the state. These settings are indicated by three physiographic regions described by Fenneman (1938): Western Hills, Appalachian Plateau, and Ridge and Valley, but for statistical purposes, all mitigation wetlands were compared to all reference wetlands. We were limited to four reference wetlands based on limited disturbance of natural wetlands and their similarity in location and elevation to mitigation sites.

Mitigation study sites were created as compensation for human activities including facility construction, road construction, or mining. Almost every wetland was located near some form of human disturbance, with some lying adjacent to roads with moderate to heavy traffic (Balcombe, 2003). Many were extensively used for recreational use, adding to the level of disturbance. Mitigation sites ranged from 4 to 21 years old since the time of construction, in size from 3.0–9.5 ha, and in elevation from 265 to 1036 m (Tables 1 and 2). Average water depth per mitigation wetland ranged from 5.4 to 57.2 cm. All mitigation wetlands were classified as palustrine emergent or palustrine unconsolidated bottom wetlands (Cowardin et al., 1979). The most common dominant plant communities in mitigation wetlands were common rush (*Juncus effusus*) and

Table 1
List of 11 mitigation and four reference wetland study sites in West Virginia, USA in 2001–02 including site name, year constructed, size (ha), responsible agency or organization, universal transverse mercator (UTM) coordinates, 7.5 min quadrangle, river basin, and watershed

Site name	Year	Size (ha)	Responsibility	UTM Y	UTM X	Quad	Basin	Watershed
Altona Marsh[a]	N/A	15.2	N/A	4353000	768600	Middleway	Shenandoah River	Shenandoah River
Walnut Bottom	1997	9.5	Division of Hwys	4334210	673914	Old Fields	South branch of Potomac River	South branch of Potomac River
Elder Swamp	N/A	28.0	N/A	4340000	642200	Mt. Storm Lake	Cheat River	Blackwater River
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	North branch of Potomac River	Elk Run
Meadowville	N/A	6.5	N/A	4330920	593940	Nestorville	Tygart Valley	Laurel Creek
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	Buckhannon River
Muddlety	N/A	10.4	N/A	4248480	516790	Widen	Gauley River	Muddlety Creek
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

[a] Site names in bold indicate reference wetlands for mitigation wetland sites (listed below) in each of four areas.

reed canarygrass (*Phalaris arundinacea*) (Balcombe et al., 2005) (Table 2). Mitigation wetland cover types averaged 40.6% open water, 54.0% herbaceous vegetation, and 5.4% scrub-shrub vegetation.

Reference wetlands chosen for study were located near mitigation sites within each area, usually within the same watershed. All had established stable emergent, scrub-shrub, and forested wetland communities that were typical of undisturbed wetlands in the region. The portions of each wetland that were evaluated ranged from 6.5 to 28.0 ha in size and ranged from 170 to 1000 m in elevation. Average water depth per reference wetland ranged from 5.4 to 17.4 cm. Overall, water depth in mitigation wetlands was 2.5 times greater than water depth in reference wetlands (Table 2). All reference wetlands were classified as palustrine emergent or palustrine scrub-shrub wetlands (Cowardin et al., 1979). Detailed study site descriptions are provided

Table 2
Dominant plant species, average wetland size, and average elevation of mitigation ($n = 11$) and reference ($n = 4$) wetlands in West Virginia, USA, 2001–2002

Dominant plant species (% aerial coverage)		Mitigated		Reference	
Common name	Scientific name	$\bar{x}$	S.E.	$\bar{x}$	S.E.
Tussock Sedge	*Carex stricta*	0.42	0.22	4.23	2.19
Common Rush	*Juncus effusus var. effusus*	4.82	0.73	0.25	0.16
Rice Cutgrass	*Leersia oryzoides*	2.64	1.03	2.85	2.08
Purple Loosestrife	*Lythrum salicaria*	1.52	0.94	0.00	0.00
Reed Canarygrass	*Phalaris arundinacea*	6.99	2.71	0.00	0.00
Mild Water Pepper	*Polygonum hydropiperoides*	1.59	1.09	0.03	0.02
American Bur-reed	*Sparganium americanum*	0.26	0.24	1.25	1.25
Broad-leaved Cattail	*Typha latifolia*	1.27	0.70	6.82	3.84
Water depth (cm)		32.32	3.43	12.03	2.68
Wetland size (ha)		5.80	0.80	15.10	4.70
Wetland elevation (m)		586.00	75.90	582.00	169.50

in Balcombe (2003). The most abundant community type in reference wetlands included broad-leaved cattail (*Typha latifolia*) and tussock sedge (*Carex stricta*) (Table 2). Cover types on reference wetlands averaged 9.3% open water, 44.3% herbaceous vegetation, 41.1% scrub-shrub vegetation, and 5.3% forested.

2.2. Avian communities

We evaluated avian communities by sampling breeding bird populations using point count surveys (Ralph et al., 1995). We visited each wetland twice between 5 May and 27 June 2001–2002, 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 established a minimum of 1 (2.4 ± 0.31) 0.78 ha point count stations (50-m radius) at each wetland, which were spaced $\geq$ 250 m apart for independent bird surveys (Ralph et al., 1993). At each wetland, we established a sufficient number of sampling stations (1–5) to cover the entire wetland area.

We conducted playback surveys for some waterbirds that are generally missed with traditional bird count methodologies. Immediately following point counts, we conducted call-response surveys for Virginia rails (*Rallus limicola*), king rails (*R. elegans*), and soras (*Porzana carolina*) at the same stations used for point counts to determine rail presence/absence and relative abundance. Surveys also were conducted for American bitterns (*Botaurus lentiginosus*), least bitterns (*Ixobrychus exilis*), and pied-billed grebes (*Podilymbus podiceps*) using the same protocol. We conducted surveys according to protocol outlined by Gibbs and Melvin (1993). We played species-specific calls using a portable cassette player located 0.75 m above ground or water for 50 s per call, followed by 10 s of silence. Calls were played with a maximum sound pressure of 80 dB 1 m from the recorder. We played each species' call in the same order 1 time/station. We used the American Ornithologists Union (1998) checklist for common and scientific names of birds.

2.3. Anuran communities

We evaluated anuran communities using nocturnal call count surveys to evaluate species presence or absence and relative abundance. We visited wetlands three times (5–24 April, 7–30 May, and 5–18 June 2001–2002) to account for temporal breeding differences among species (Mossman et al., 1998; Stevens et al., 2002). These dates were selected based on recommended temperature ranges for different survey periods (i.e., period 1: >5 °C; period 2: >10 °C; period 3: >12.8 °C; Casey and Record, U.S. Fish and Wildlife Service, unpublished report). We collected data for 3 min at each sampling point following a 1–2 min settling period. We identified frogs to species and evaluated relative abundances by assigning a Wisconsin index (WI) value of intensity to each species' call (Mossman, 1994). We assigned a ranking of 1 to species with nonoverlapping calls and when an exact count of individuals could be made, a ranking of 2 to species whose calls overlapped and only estimations of numbers could be made, and a 3 to species that were calling in full chorus. If a WI value of 3 was assigned to a species, we used a standard abundance estimate of 50. We conducted surveys between 30 min after sunset and midnight. We used the species checklist from the Society for the Study of Amphibians and Reptiles (2000) for common and scientific names of frogs.

2.4. Statistical analyses

For all avian analyses, we included only those birds sampled within the 50 m radius (0.78 ha) plots. We used a split-plot analysis of variance design (ANOVA) to test for differences in avian richness (no. species/ha), abundance (no. birds/ha), and diversity (per ha) between mitigation and reference wetlands using SAS (SAS Institute, 1988). Avian diversity was calculated using the Shannon–Weiner index (Shannon and Weaver, 1949). Avian species included in the waterbird analysis were Canada geese (*Branta canadensis*), mallards (*Anas platyrhynchos*), wood ducks (*Aix sponsa*), black ducks (*Anas rubripes*), green herons (*Butorides virescen*), great blue herons (*Ardea herodius*), belted kingfishers (*Ceryle alcyon*), spotted sandpipers (*Actitis macularia*), Virginia rails, and soras. Canada geese, mallards, wood ducks, and black ducks were included in the waterfowl analysis.

We used a two-way ANOVA with a repeated measures design to compare anuran richness because three survey periods were repeated both years. For avian and anuran analyses, the independent variables tested were year, type (mitigation versus reference), and year × type interactions with the dependent variables varying depending on which taxa was being analyzed. We used individual wetlands as experimental units. Because WI and anuran abundance metrics were categorical variables, we used logistic regression to compare mitigation and reference wetlands. Abundance estimates were obtained using SAS and grouped into intervals (i.e., 2–5, 6–15, 16–25, 26–35, and 50), which allowed them to be treated as categorical variables. Only the mid-point of each interval was used for analyses. Logistic regression also was needed because of unequal variances associated with WI and abundance variables. We used an area × year × sampling period combination as a blocking factor for logistic regression. For all other avian and anuran analyses, geographic area was a blocking factor.

Assumptions of normality were tested with the univariate procedure in SAS, and Levene's test was used for homogeneity of variances. Rank and square-root transformations were used to convert dependent variables that did not meet the aforementioned assumptions (Dowdy and Wearden, 1991). Specifically, square-root transformations were incorporated in anuran WI comparisons, and rank transformations were used to analyze avian communities. We used an alpha level of 0.05 for all statistical tests.

Table 3
Richness (no. species/ha), diversity (per ha), and abundance (no. birds/ha) comparisons for avian communities between mitigation ($n = 11$) and reference ($n = 4$) wetlands in West Virginia, USA, 2001–2002

Species or group	Mitigation[a]		Reference[a]	
	$\bar{x}$	S.E.	$\bar{x}$	S.E.
Richness	11.27a	0.40	11.24a	0.62
Diversity	3.09a	0.53	2.76a	0.47
Abundance				
All birds	27.09a	2.17	28.46a	4.94
Waterbirds[b]	5.09a	1.46	0.44b	0.23
Waterfowl[c]	4.44a	1.41	0.24b	0.21
Passerines[d]	20.37a	1.51	26.73a	4.94
Top 20 species	22.21a	2.04	23.81a	5.01
Red-winged blackbird (*Agelaius phoeniceus*)	5.91a	0.94	8.00a	1.58
European starling (*Sturnus vulgaris*)	1.27a	0.50	4.05a	4.05
Song sparrow (*Melospiza melodia*)	1.44a	0.14	3.12b	0.28
Canada goose (*Branta canadensis*)	2.59a	0.03	0.21a	0.21
Common yellowthroat (*Geothlypis trichas*)	0.91a	0.13	1.36a	0.37
Tree swallow (*Tachycineta bicolor*)	2.05a	0.49	0.68a	0.27
Cedar waxwing (*Bombycilla cedrorum*)	0.64a	0.18	0.08a	0.05
American crow (*Corvus brachyrhynchos*)	0.03a	0.01	0.21a	0.14
Indigo bunting (*Passerina cyanea*)	0.64a	0.14	0.81a	0.24
Wood duck (*Aix sponsa*)	1.12a	0.46	0.00b	0.00
American goldfinch (*Carduelis tristis*)	0.69a	0.14	0.44a	0.19
Red-eyed vireo (*Vireo olivaceus*)	0.51a	0.10	0.32a	0.10
Willow flycatcher (*Empidonax traillii*)	0.46a	0.12	1.31a	0.29
Mallard (*Anas platyrhynchos*)	0.73a	0.21	0.04a	0.04
Yellow warbler (*Dendroica petechia*)	0.58a	0.13	1.21a	0.29
Gray catbird (*Dumetella carolinensis*)	0.42a	0.18	1.01a	0.26
Northern cardinal (*Cardinalis cardinalis*)	0.23a	0.06	0.41a	0.18
American robin (*Turdus migratorius*)	0.56a	0.14	0.27a	0.13
Barn swallow (*Hirundo rustica*)	0.87a	0.40	0.12a	0.09
Eastern towhee (*Pipilo erythrophthalmus*)	0.29a	0.08	0.21a	0.10
Great blue heron (*Ardea herodias*)	0.15a	0.08	0.04a	0.04

[a] The same letter following means indicates no difference between wetland types ($P > 0.05$).
[b] Includes only those birds that depend on water for all or most of their life requisites.
[c] Includes only birds in the family Anatidae.
[d] Includes only birds in the order Passeriformes.

3. Results

3.1. Avian communities

We observed 106 species of birds in mitigation and reference wetlands both inside and outside the 50 m radius plots (see Balcombe, 2003 for a complete list of species). In mitigation sites, 2074 individuals from 86 species were sampled within 50 m radius plots, and in reference sites, 771 individuals from 62 species were sampled. For all species sampled, mean species richness ($F_{1,10} = 0.15$, $P = 0.711$), diversity ($F_{1,10} = 1.1$, $P = 0.314$), and abundance ($F_{1,10} = 0.03$, $P = 0.856$) were similar between mitigation and reference wetlands (Table 3). Mean abundance for the 20 most common avian species sampled was similar ($F_{1,10} = 0.07$, $P = 0.800$) between mitigation and reference wetlands (Table 3). Out of these common species, wood ducks ($F_{1,10} = 5.80$, $P = 0.037$) and American goldfinches (*Carduelis tristis*; $F_{1,10} = 9.24$, $P = 0.013$) were more abundant in mitigation wetlands, whereas song sparrows (*Melospiza melodia*; $F_{1,10} = 5.94$, $P = 0.035$) were more abundant in reference wetlands (Table 3). Density of great blue herons were similar ($F_{1,10} = 0.28$, $P = 0.610$) between wetland types. All other common species also were similar ($F_{1,10} \leq 4.57$, $P \geq 0.058$) between wetland types. Passerine abundance (72 species combined) was similar ($F_{1,10} = 0.41$, $P = 0.537$) between mitigation and reference wetlands. However, waterbird ($F_{1,10} = 9.08$, $P = 0.013$) and waterfowl ($F_{1,10} = 9.23$, $P = 0.013$) abundance were higher in mitigation wetlands than reference wetlands.

Two rail species were detected at two mitigation wetlands. Three soras were sampled at Buffalo Coal during the first surveys of both years. Similarly, three soras were detected at Walnut Bottom during the first survey of year 2. Five and two Virginia rails were located at Buffalo Coal during the second surveys of years 1 and 2, respectively. No rail species were detected at reference wetlands, and no bitterns or pied-billed grebes were sampled at any wetland.

3.2. Anuran communities

Seven species of anurans were heard, all of which occurred in both mitigation and reference wetlands. These included northern spring peepers (*Pseudacris crucifer crucifer*), gray treefrogs (*Hyla*

Table 4
Wisconsin index value and abundance per wetland for all anuran species combined and for each of seven species heard at mitigation ($n = 11$) and reference ($n = 4$) wetlands, West Virginia, USA, 2001–2002

Common name	Scientific name	Wisconsin index						Abundance					
		Mitigation[a]		Reference[a]				Mitigation[a]		Reference[a]			
		$\bar{x}$	S.E.	$\bar{x}$	S.E.	χ^2	P	$\bar{x}$	S.E.	$\bar{x}$	S.E.	χ^2	P
All frogs		0.52a	0.03	0.40b	0.07	14.51	≤0.001	4.75a	0.66	4.69b	1.18	11.35	≤0.001
Northern Spring Peeper	*Pseudacris crucifer crucifer*	1.83a	0.16	1.86	0.27	0.61	≤0.434	22.9a	3.32	28.4a	6.95	1.46	≤0.228
Gray Treefrog	*Hyla chrysoscelis/ H. versicolor*	0.17a	0.06	0.16a	0.09	0.88	≤0.348	0.39a	0.16	0.33a	0.19	0.60	≤0.440
American Bullfrog	*Rana catesbeiana*	0.13a	0.05	0.05b	0.03	4.56	≤0.033	0.21a	0.09	0.10b	0.06	4.30	≤0.038
Wood Frog	*Rana sylvatica*	0.13a	0.06	0.08a	0.05	2.87	≤0.090	0.45a	0.29	0.21a	0.14	2.76	≤0.097
Northern Green Frog	*Rana clamitans melanota*	0.84a	0.12	0.50b	0.13	6.36	≤0.012	7.82a	2.37	3.56b	1.29	5.64	≤0.018
eastern American Toad	*Bufo americanus americanus*	0.18a	0.05	0.04a	0.03	3.66	≤0.056	0.48a	0.17	0.05a	0.04	3.58	≤0.059
Pickerel Frog	*Rana palustris*	0.35a	0.09	0.09b	0.03	8.73	≤0.003	0.82a	0.21	0.24a	0.09	8.08	≤0.005

[a] The same letter following means indicates no difference between wetland types ($P > 0.05$).

chrysoscelis/*H. versicolor*), American bullfrogs (*Rana catesbeiana*), wood frogs (*R. sylvatica*), northern green frogs (*R. clamitans*), eastern American toads (*Bufo americanus americanus*), and pickerel frogs (*R. palusris*). Mean species richness was higher in mitigation (2.01 ± 0.09 species/point) than reference (1.47 ± 0.14) wetlands ($F_{1,10} = 7.18$, $P = 0.023$). In addition, Wisconsin index (WI) values and abundance were higher in mitigation than reference wetlands (Table 4). Wisconsin Index and abundance (A) comparisons also were made for each species detected (Table 4). For these indices, American bullfrogs, northern green frogs, and pickerel frogs were higher in mitigation than reference wetlands, whereas northern spring peepers, gray treefrogs, wood frogs, and eastern American toads were similar between wetland types (Table 4).

4. Discussion

4.1. Avian communities

Almost every avian metric we measured in mitigation wetlands was equal to or greater than reference wetlands. No differences emerged in total species richness, diversity, and abundance probably because of similarities in landscape position. Both wetland types were generally located near forested stands, so wetland edge and forest-interior species had an equal chance of being sampled between wetland types. Similarly, both wetland types were either adjacent or connected to other wetlands, streams or large rivers. Although some studies have found human disturbance to negatively affect wildlife numbers (Wilson and Mitsch, 1996), the proximity of mitigation and reference sites to human disturbances (i.e., major roads) appeared to have minimal effects on avian numbers. Although it is known that wetland size affects avian richness (MacArthur and Wilson, 1967; Tyser, 1983; Delphey and Dinsmore, 1993), the fact that reference wetlands were about three times larger than mitigation wetlands had little effect on avian metrics relative to mitigation wetlands. While some studies have shown higher avian richness and diversity in natural wetlands (Delphey and Dinsmore, 1993; Melvin and Webb, 1998), others, similar to our study, have yielded similar avian indices between wetland types (Perry et al., 1996; Brown and Smith, 1998). It is likely that, given the similarities in landscape position between wetland types, the increased richness and diversity of vegetation offered in our mitigation sites was balanced by the increased percentage of emergent vegetation in reference wetlands (Balcombe et al., 2005), thus resulting in similar overall avian community structure between wetland types.

Mitigation wetlands, however, supported higher waterbird and waterfowl abundance than reference wetlands. Because mitigation sites were so young (4–20 years of age), they differed significantly in their vegetation community structure than reference sites (Balcombe et al., 2005). Not only did mitigation sites contain more open water and less emergent aquatic vegetation than reference wetlands, they contained higher plant species richness and diversity than reference wetlands (Balcombe et al., 2005). Specifically, mitigation wetlands contained 40.8% open water, whereas reference wetlands contained only 11.6% open water. This has been found to be true of other reference wetlands in the Appalachian Region (Cole and Brooks, 2000). VanRees-Siewert and Dinsmore (1996) showed that, although total bird richness increased with increasing emergent vegetation, waterfowl and shorebirds preferred younger restored wetlands with more open water and mud flats. Overall, mitigation wetlands in this study were closer to hemimarsh conditions where an equal percentage of open water to emergent vegetation exists. Hemimarsh conditions provide the best combination of food and cover for waterbirds (Kaminski and Prince, 1981; Bookhout et al., 1989; Murkin et al., 1997; Balcombe et al., in press-a). Based on these and other studies, many have concluded that "wetter is better" in terms of constructing wetlands. As a result, mitigation wetlands are often structurally dissimilar to the reference wetlands they are designed to mimic, thus indicating an inability to functionally replace those wetlands that were destroyed (Cole and Brooks, 2000). This stresses the importance of not having too much open water in mitigation wetlands.

Waterbird abundance also may be affected by higher vegetative richness and diversity indices observed in mitigation wetlands over reference wetlands (Balcombe et al., 2005). These differences may result in an increase in the type, quantity, and quality of plant foods while at the same time maximizing the distribution, density, and structure of cover available for waterbirds in mitigation wetlands (De Szalay and Resh, 1997; Brown, 1999). Differences in vegetation community structure also may have created favorable water chemistry and hydroperiod conditions in mitigation sites as well (Goslee et al., 1997; Castelli et al., 2000).

Differences in invertebrate abundance and composition also varied between wetland types. Mitigation wetlands had higher macroinvertebrate biomass from the water column for the 13 most common taxa than natural wetlands (Balcombe et al., in press-a). However, within open water habitats, total benthic invertebrate density was higher in reference wetlands than in mitigation wetlands (Balcombe et al., in press-a). Planorbidae (orb snails) density from benthic samples in emergent habitats was higher in reference than mitigated wetlands (Balcombe et al., in press-a). Benthic Oligochaeta (aquatic worms) density was higher across open water habitats in mitigation wetlands. Among the most common water column orders, Isopoda (pillbugs and sowbugs) density was higher in reference wetlands, but Physidae (physids) density was higher in mitigation wetlands (Balcombe et al., in press-a). Biomass for most taxa was similar between wetland types, although taxonomic composition and abundance will change as the wetland continues to age (Mitsch et al., 1998). Many of these taxa are important components in waterbird diets (Euliss et al., 1991; Anderson et al., 2000). These differences in macroinvertebrate populations may account for differences in waterbird abundance observed between mitigation and reference wetlands.

Other studies comparing waterbirds between mitigation and reference wetlands have shown conflicting results. Similar to our study, Havens et al. (1995) observed similar overall species diversity between mitigation and reference wetlands in Virginia, but higher wading bird abundances occurred in constructed marshes. However, Confer and Niering (1992) included waterbirds in their assessment of wildlife in constructed and natural wetlands in Connecticut, and they observed higher wildlife activity (overall species richness) in natural wetlands. They attributed low wildlife indices in constructed wetlands to their isolation and relative small size.

These data indicate that mitigation wetlands in West Virginia, despite their proximity to human disturbances, are supporting healthy avian communities, particularly waterbirds. High avian numbers in mitigation wetlands are likely due to wetland size and landscape position, as well as vegetative structure and diversity and invertebrate community structure. However, just because a diverse avian community exists in mitigation wetlands, it does not mean that birds are successfully reproducing in mitigation wetlands. Future studies should correlate changes in vegetation and invertebrate communities to avian community structure and evaluate breeding success.

4.2. Anuran communities

It is not surprising that anurans have colonized mitigation wetlands so rapidly. Northern spring peepers, American bullfrogs, eastern American toads, and gray treefrogs may colonize created wetlands $\leq$2 years after construction (Perry et al., 1996; Mierzwa, 2000; Pechmann et al., 2001). Colonization rates are generally affected by distance to other ponds, dispersal habitat, dispersal capabilities, site fidelity of a particular species, and size of source populations (Laan and Verboom, 1990). The proximity of our study sites to streams, rivers, and other wetlands along with the relatively large size of mitigation sites likely contributed to rapid dispersal and colonization (Wolfenbarger, 1949; Lacki et al., 1992; Gibbs, 1993; Stevens et al., 2002).

Mitigation wetlands in West Virginia contained higher anuran mean richness, WI, and abundance values than reference wetlands. Similar to our study, Stevens et al. (2002) observed a higher overall mean richness as well as abundance of green frogs in restored than reference wetlands. Although they observed a positive correlation between green frogs and percentage of cattail in restored wetlands, our results suggest cattail may have a relatively minimal effect on green frogs. Both wetland types in our study had low cattail abundance, although mitigation wetlands contained less cattail than reference wetlands (1.3% versus 6.8%) (Balcombe et al., 2005). However, because mitigation wetlands sustained more northern green frogs, we think open water may play a larger role in determining abundance of northern green frogs, as well as American bullfrogs and pickerel frogs. Lacki et al. (1992) also observed more green frogs in a constructed wetland in Ohio, and Pechmann et al. (2001) observed more American bullfrogs in constructed than natural wetlands in South Carolina.

Because eastern American toads, northern spring peepers, and wood frogs are less dependent on permanent water sources (Gilhen, 1984; Cook, 1984), we expected these species to be relatively more abundant than other anuran species in reference wetlands, which contained less open water. In general, the open water areas were deeper than vegetated areas. Consistent with this speculation, relative abundance of these species were similar between mitigation and reference wetlands.

Many important factors may account for anuran community differences observed between mitigation and reference wetlands. Primarily, studies have shown that open water is positively correlated with amphibian abundance (Lacki et al., 1992; Stevens et al., 2002). As previously mentioned, mitigation wetlands contain more open water than reference wetlands, thus more closely resembling hemimarsh conditions. Like avian communities, anuran communities benefit from these conditions (Stumpel and Van Der Voet, 1998; Anderson et al., 1999a). Although hydrologic data are incomplete for our study sites, existing data indicated an extended hydroperiod in some mitigation wetlands (Balcombe, 2003) that may prevent drying and subsequent tadpole mortality prior to metamorphosis. Thus, species with longer larval periods such as American bullfrogs, northern green frogs, and pickerel frogs (whose abundances and WI values were higher in mitigation wetlands) may have been excluded from reference wetlands, which contained shorter hydroperiods (Babbit and Tanner, 2000; Semlitsch, 2002). This may not necessarily be a limiting factor because pond drying is a natural process that eliminates or reduces predation on and competition among larval amphibians (Semlitsch, 2000). On the contrary, maintaining wetlands with extremely long hydroperiods may be harmful to anuran populations because it may facilitate colonization of aquatic invertebrate and fish predators (Semlitsch, 2002).

Water depth also plays an important role in amphibian colonization (Stevens et al., 2002). Deeper water prevents complete freezing, which provides winter hibernacula for anurans (Cook, 1984; Cunjak, 1986). We found that both mitigation and reference wetlands contained areas with sufficient hibernacula, but based on water depth estimations, mitigation wetlands generally contained deeper water with more potential wintering habitat (Balcombe, 2003). However, water depth in mitigation wetlands averaged 2.5 times deeper than in reference wetlands. The deep water in some areas of mitigation wetlands may have inhibited some anurans from reproducing throughout the entire wetland.

Furthermore, shorter distance to forests and higher percentage of shrub cover increases anuran richness by providing cover and dispersal corridors for post-breeding or newly metamorphosed individuals (Stevens et al., 2002). This may be of particular importance to wood frogs, which disperse long distances via forested cover types to other wetlands (Berven and Grudzien, 1990; DeMaynadier and Hunter, 1999). As well, forested perimeters may buffer wetlands from agricultural activities, which have been linked to larval death and limb deformities in amphibians (Berrill et al., 1997; Ouellet et al., 1997), although abundance may still be high (Gray et al., 2004). They also may buffer against negative impacts associated with cattle grazing. Wood frogs and chorus frogs, in particular, are known to be sensitive to this disturbance (Ambrose and Paszkowski, 1998). As mentioned in the avian discussion, mitigation and reference wetlands shared similar landscape positions adjacent to forests. Forested buffers occurred at all reference wetland sites; however, mitigation wetlands averaged only 14.5 m to the nearest forest. Thus, anurans in both wetland types likely benefit from forested perimeters.

Although reference wetlands contained a higher percentage of shrub cover than mitigation wetlands (41.1% versus 5.4%), they contained less open water, which likely limited anuran numbers. Shrub communities had successfully been established at 9 of 11 mitigation wetlands, and percent coverage should increase as these wetlands mature (Balcombe et al., 2005). This will be valuable in maintaining future diverse anuran habitat.

Similar to waterbird communities, differences in anuran communities may be attributed to differences in invertebrate and vegetation communities between mitigation and reference sites (Balcombe et al., 2005, in press-a,b). Because frogs depend on invertebrates for their diet (Anderson et al., 1999b; Lima and Magnusson, 2000), it is expected that anuran abundance and distribution could reflect higher invertebrate nektonic biomass densities across open water areas of mitigation wetlands. Similarly, higher vegetative species richness and diversity may provide more diverse microhabitats for oviposition, foraging, growth, and refuge (Stratman, 2000). However, anurans were not influenced by total percent emergent vegetation (Balcombe et al., in press-b).

Few anuran species (e.g., American bullfrogs, northern green frogs) can coexist with predatory fish species (Semlitsch, 2002), but studies offer conflicting evidence as to the effect of predatory fish on anuran populations (Hecnar and M'Closkey, 1997; Lehtinen et al., 1999; Pechmann et al., 2001; Semlitsch, 2002). Despite fish populations in 9 of 11 mitigation wetlands and three of four reference wetlands, these sites continue to support healthy anuran populations. In fact, some of the highest frog indices were obtained in wetlands that contained fish. It is important to note that some mitigation sites consisted of numerous open water cells, some of which did not contain fish. These areas serve as a refuge for breeding frogs, thus minimizing potential negative impacts caused by fish populations. Furthermore, high anuran populations in wetlands that contain fish may be attributed to an increase in the macroinvertebrate prey base, which can result indirectly from increases in predatory fish populations (Batzer et al., 2000). A more detailed study would be needed to accurately assess the impact of fish populations on anuran communities among mitigation wetlands in West Virginia. Even if data were to show a negative impact of fish populations on anurans, it would be difficult to prevent the invasion of fish into wetlands adjacent to streams or rivers.

Numerous mitigation sites were built on-site as mitigation for the construction of highways in West Virginia. However, the proximity of mitigation wetlands to major roads did not seem to adversely affect current anuran abundance. Indeed, two of the sites closest to roads scored among the highest richness and WI values of all mitigation sites. However, studies have correlated low amphibian, as well as reptile numbers to road density (Fahrig et al., 1995; Lehtinen et al., 1999; Haxton, 2000; Trombulak and Frissell, 2000). The limiting factor, however, is not necessarily the traffic, although amphibian mortality due to vehicular collisions is not uncommon (Fahrig et al., 1995). Roads, acting as barriers to dispersal, may have long-term effects on metapopulation dynamics by deteriorating the genetic integrity of localized populations (Trombulak and Frissell, 2000). In addition, roads potentially change soil density, temperature and water content, surface waters, patterns of run-off, and sedimentation, as well as adding heavy metals to roadside environments (Trombulak and Frissell, 2000), although this is not confirmed at any of our study sites. Problems associated with dispersal may not manifest themselves within anuran populations located at our study sites because of their proximity to streams, rivers, and other wetlands. Research should monitor road-related stresses to the environment and their potential effect on anuran populations within mitigation wetlands. Although wetland construction near roads can potentially have long-term negative impacts, there are numerous logistical benefits associated with on-site design and construction. As well, on-site mitigation sites can facilitate colonization by philopatric anuran species.

Recent concern over declining amphibian populations has drawn attention to the need to compensate for loss of amphibian habitat. Our data provide, both an assessment of the success of mitigation wetlands in West Virginia in supporting anuran communities, and a sound framework for future research that monitors anuran community responses to structural changes in these wetlands through time. However, similar to the limitations noted above for birds, anuran calling count surveys do not provide information on reproductive success. Future research needs to evaluate breeding and reproductive success of anurans in mitigation wetlands.

5. Conclusions

Numerous studies have written about our inability to successfully mitigate for wetland destruction (Race, 1985; Erwin, 1990; Reinartz and Warne, 1993; National Research Council, 2001), although others have not viewed the situation as bleak (Mitsch and Wilson, 1996; Mitsch et al., 1998). Although the definition of success varies depending upon project objectives, most agree that compensatory wetlands should replace functions lost during wetland destruction. These data indicate that mitigation wetlands in West Virginia currently support numerous avian and anuran species. Indeed, mitigation sites contained some higher wildlife indices than reference sites, and this could reflect actual differences in wildlife populations resulting from wetland age, design, or location within the landscape. It is likely that wildlife distribution and abundance reflect differences in vegetation and invertebrate community structure between mitigation and reference wetlands, and future monitoring should focus on monitoring the interactions between wildlife populations and these biotic factors. Monitoring the effects of beaver (*Castor canadensis*) activity on vegetation structure is of particular importance in evaluating future wildlife communities.

We should caution that it is premature to assess the full outcome of mitigation efforts within the state. First, these data represent a short-term trend resulting from only 2 years of data collection. Thus, these data do not encompass the temporal variation in avian and anuran community structure. Pechmann et al. (2001) recommended several years of census data on amphibians before meaningful comparisons between mitigation and reference sites can be made. Similarly, D'Avanzo (1990) and Zedler (1993) suggested a monitoring duration of 20 years for mitigation wetlands. Unfortunately, financial or logistical restraints often preclude long-term monitoring capabilities.

Second, created wetlands often take at least a decade before they function compatible to reference wetlands. Wilson and Mitsch (1996) recommend giving freshwater wetlands 15–20 years before judging their success, and Frenkel and Morlan (1991) recommend waiting ≥50 years for certain forested and coastal wetlands. Two wetlands included in this study were about 20 years old and an additional three sites were ≥10 years old. Although our sites do not meet the ideal age criteria for mitigation wetland development time, nearly half are ≥10 years old, and we think relatively conservative inferences can still be made regarding their success.

Finally, the variation in structure among mitigation and reference wetlands adds to the difficulty in assessing mitigation success. This is particularly important in the establishment of reference standards (Smith et al., 1995; Brinson and Rheinhardt, 1996). Natural short-term processes such as seasonal cycles of precipitation and temperature, coupled with long-term processes including population dynamics, erosion and depositional processes, succession, or drought/wet cycles can cause variation in the functional capacity of natural wetlands (Smith et al., 1995). This type of variability is common in many wetland ecosystems including coastal marshes (Oviatt et al., 1977), cypress swamps (Ewel and Odum, 1984), prairie potholes (Kantrud et al., 1989), and playa wetlands (Smith, 2003). Another factor researchers must consider in establishing reference standards concerns anthropogenic disturbance (Smith et al., 1995). Because most wetlands have been exposed to hundreds of years of continued disturbance, their functions have been fundamentally changed, so it may be difficult to construct wetlands based on undisturbed standards. Although some misuses of using natural wetlands as reference standards are possible, reference wetlands can guide mitigation, both during and after the process by making explicit the goals of mitigation and by evaluating the progress of mitigation wetlands through proper monitoring (Brinson and Rheinhardt, 1996). Similar variation in wetland structure also can occur within mitigation wetlands thus providing further evidence as to the difficulty in duplicating natural systems, especially since alternative stable states are commonly observed in ecological communities (Drake, 1990). These points illustrate the complexity in assessing mitigation success based on reference wetlands and reiterate the need to document and compare losses of wildlife habitat during wetland destruction to creation of wildlife habitat via compensatory mitigation.

Indeed, temporal variation in wildlife habitat use, wetland development time, and structural variation compound the logistics of evaluating mitigation wetland success. Nevertheless, the similarities in wildlife indices observed in this study suggest preliminary development of mitigation sites towards reference standards. We anticipate these data will help guide the creation of standardized protocols for the continued monitoring of these and other mitigation wetlands, not only in West Virginia, but also across the Appalachians.

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