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Research Paper

Trade-offs relating to grassland and forest mine reclamation approaches in the central Appalachian region and implications for the songbird community

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ABSTRACT. Surface mining in the Appalachian region (USA) converts large areas of mature forest to early successional habitat. This shift in landscape structure has the potential to reduce habitat availability and suitability for forest-dwelling songbirds by reducing and fragmenting mature forest, but also to increase habitat availability for grassland- and shrubland-associated songbirds. We examined the influence of mountaintop mining/valley fill (MTMVF) reclamation habitats (grassland, shrubland, and remnant forest) on songbird community composition and abundance at three former MTMVF mines in southwestern West Virginia, relative to intact forest. We quantified the songbird community in 1999 and 2000 using point counts arranged throughout the mine complexes to assess landscape composition of the songbird community. Community analysis showed songbirds had strong associations with their respective guild based on species habitat preferences. Although remnant and intact forest treatments had similar species compositions, the forest interior guild had greater richness in intact rather than remnant forest. Total species richness was greatest in the reclaimed shrubland treatment. Focal species analysis followed similar trends as community assessments, because most species abundances within treatment types were strongly associated with species habitat preferences. Our results indicate that reclamation habitat decisions, i.e., grasslands versus forests, can have large effects on avian community composition. Determining appropriate mine restoration actions depends on the suite of species desired for long-term occupancy and their conservation priority.

Compromis relatifs aux approches de remise en état de mines en prairie ou forêt dans la région centrale des Appalaches et répercussions sur la communauté de passereaux

RÉSUMÉ. L'exploitation de mines à ciel ouvert dans la région des Appalaches (É.-U.) a entrainé la conversion de vastes secteurs de forêts matures en milieux de début de succession. Ce changement dans la structure du paysage peut engendrer une réduction de la disponibilité et de la qualité des milieux pour les passereaux forestiers en réduisant et fragmentant les forêts matures, mais peut aussi entrainer une augmentation de la disponibilité de milieux pour les passereaux de prairies et d'arbustaies. Nous avons examiné l'effet de la remise en état (en prairie, arbustaie ou forêt résiduelle) d'anciens sites miniers situés au sommet de montagnes ou dans des vallées remblayées (SMVR) sur la composition et l'abondance des communautés d'oiseaux à trois anciennes mines SMVR dans le sud-ouest de la Virginie-Occidentale, comparativement à des forêts intactes. Nous avons déterminé la composition des communautés d'oiseaux à l'échelle du paysage au moyen de dénombrements par points d'écoute répartis dans l'ensemble des complexes miniers en 1999 et 2000. L'analyse des communautés a révélé que les passereaux étaient fortement associés selon leur guilde respective fondée sur leurs préférences d'habitat. Même si les forêts résiduelles et intactes avaient une composition spécifique similaire, la richesse de la guilde d'oiseaux d'intérieur de forêt a été plus élevée dans les forêts intactes que les forêts résiduelles. Le nombre d'espèces le plus élevé a été trouvé dans les arbustaies restaurées. L'analyse par espèce a montré les mêmes tendances que celles des communautés, parce que l'abondance de la plupart des espèces pour un même type de traitement était fortement liées aux préférences d'habitat des espèces. Nos résultats indiquent que les décisions relatives à la remise en état de sites miniers, c.-à-d. en prairie ou forêt, peuvent avoir des répercussions importantes sur la composition des communautés aviaires. La détermination des actions appropriées pour la restauration de mines dépend des espèces désirées à long terme et de leur priorité de conservation.

Key Words: *central Appalachians; mine reclamation; songbird community; surface mining*

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INTRODUCTION

The continued modification of forested landscapes from surface mining and their subsequent revegetation represents a potential crossroads for mine reclamation practices. Because of the degree of land modification associated with surfacing mining, reclamation decisions can influence long-term occupancy of wildlife communities. In the central Appalachian region of the United States, reclamation can generally be split into the traditional grassland reclamation approach and the newer forest reclamation approach. The grassland approach plants fast-growing grasses (often nonnative) to reduce soil erosion and increase vegetative cover, in accordance with the Surface Mining Control and Reclamation Act of 1977 (SMCRA). The forest approach is a newer approach that was developed by the Appalachian Regional Reforestation Initiative to encourage the restoration of forests on reclaimed surface mines and help establish forests through natural succession while still meeting the requirements of SMCRA (Angel et al. 2005). These two approaches have diverging long-term outcomes that affect plant composition, which in turn affect wildlife communities.

Early successional plant communities, primarily grasslands, are often predominant following mine closure (Chaney et al. 1995, Zipper et al. 2011). The outcome of revegetating primarily with grasses is the creation of extensive grasslands on mines, which benefits grassland birds (DeVault et al. 2002, Galligan et al. 2006, Wood and Ammer 2015). In North America, grassland bird populations have substantially declined over the last several decades (Ribic et al. 2009, Sauer et al. 2014) because of reduced areas in grasslands from land conversion, e.g., agriculture, or woody encroachment (Coppedge et al. 2001, Grant et al. 2004). Grassland plant communities can remain on mines for an extended period of time (>20 years; Ingold and Dooley 2013, Wood and Ammer 2015) because of poor soil quality, which inhibits establishment and growth of natural woody vegetation (Chaney et al. 1995) and creates an arrested state of succession (Wickham et al. 2013). Despite the benefits of grassland reclamation for grassland songbirds, grassland communities are novel in the central Appalachians (Hall 1983) and may not contribute functional ecosystem services when compared with forests, e.g., carbon storage and watershed and water quality protection (Zipper et al. 2011).

The forest approach attempts to expedite forest succession on mine lands and return landscapes to their premining land cover by limiting soil compaction and planting native trees (Angel et al. 2005). Within the central Appalachians (eastern Kentucky, northeastern Tennessee, southwestern Virginia, and southern West Virginia), the primary land cover is forest (89%; Sayler 2008), and forest loss and fragmentation are largely driven by natural resource extraction, e.g., mining (Sayler 2008, Townsend et al. 2009, Drummond and Loveland 2010, Palmer et al. 2010). Additionally, the central Appalachian region provides core forest, i.e., areas of forest >100 m from any forest edge, for forest-dwelling birds that are experiencing substantial population declines throughout their range, e.g., Cerulean Warbler (*Setophaga cerulea*) and Wood Thrush (*Hylocichla mustelina*; Sauer et al. 2014). Because of forest birds' reliance on core forest habitat (Becker et al. 2015, Farwell et al. 2016), the Appalachian region (Bird Conservation Region 28 [BCR 28]; see Fig. 1) is important to the conservation of North American forest bird populations. However, few studies have reported mining's negative effects on forest bird species (Weakland and Wood 2005, Wood et al. 2006, Becker et al. 2015) or whether these can be reduced or offset. Despite the limited data relating to the forest approach and the trade-offs for forest birds, the forest reclamation approach nonetheless provides a framework that focuses on reforestation and the return of forest-based ecosystems (Zipper et al. 2011, McDermott et al. 2013).

Fig. 1. Location of mountaintop mining complexes in southwestern West Virginia, USA. Map on the right represents mine complexes (shaded black) used to assess mountaintop mining/valley fill effects on the songbird community. Top left map shows location of West Virginia (shaded gray) and Bird Conservation Region 28 (crosshatched region). Bottom left map shows county locations (shaded gray) where study was conducted within West Virginia.

The decision to reclaim mined lands with either a grassland or forest approach can be based on the suite of species desired for long-term occupancy, but the return of premining bird communities is important to factor into reclamation objectives as well. Our purpose was to evaluate potential trade-offs between the grassland and forest reclamation approaches by quantifying the songbird community on mountaintop mining/valley fill (MTMVF) complexes and adjacent forest patches in southern West Virginia. A large portion of the Appalachian region (BCR 28; Fig. 1) is available for surface mining activities (~40%; Fig. 2); thus it is important to evaluate the potential trade-offs between grassland reclamation versus forest reclamation and their effects on songbird conservation for future application in the Appalachian region. Because of songbird habitat associations, we expect that songbirds will be closely related with their a priori habitat associations on MTMVF complexes and adjacent forest patches, although this has not been explicitly tested relative to mine reclamation. Our results highlight gains and losses in the avian community that will help reclamation practitioners develop guidelines when managing mine landscapes in the Appalachian region.

METHODS

Study areas included 3 MTMVF complexes and adjacent unmined mature forest areas in Boone, Logan, Kanawha, and Fayette counties in southwestern West Virginia (Fig. 1). We categorized four treatments represented within the study area: reclaimed grassland, reclaimed shrubland, remnant forest, and intact forest. Reclaimed mine landscapes on our MTMVF sites included areas primarily reclaimed to grassland (1672, 1819, and

2003 ha on each of the 3 sites), reclaimed shrubland areas (0, 428, and 508 ha), and remnant forest patches (hereafter termed remnant forest; 155, 214, and 339 ha; Fig. 3). Grassland areas were planted during the reclamation process primarily with nonnative grasses and legumes such as tall fescue (*Festuca arundinacea*), sericea lespedeza (*Lespedeza cuneate*), orchard grass (*Dactylis glomerata*), and perennial rye grass (*Lolium perenne*). Grassland areas were 5-19 years postreclamation at the time of the study. Shrubland areas consisted primarily of shrubs and pole-sized trees (0-8 cm diameter-at-breast height [DBH]) and were approximately 13-27 years postreclamation. Vegetation in shrubland habitats included species seeded or planted during reclamation such as tall fescue, sericea lespedeza, autumn olive (*Elaegnus umbelleta*), black locust (*Robinia pseudoacacia*), scotch pine (*Pinus sylvestris*), and species that regenerated naturally including goldenrod (*Solidago* spp.), red maple (*Acer rubum*), American sycamore (*Platanus occidentalis*), tuliptree (*Liriodendron tulipifera*), multiflora rose (*Rosa multiflora*), and blackberry/raspberry (*Rubus* spp.). Shrublands on mine complexes comprised little of the total reclaimed area at our study areas (936 of 6430 ha, ~14% of all reclaimed area), indicating the limited use of shrubland reclamation techniques at the time (ca. 1980s). Mine ages were the estimated year that sites were reclaimed based on information provided by the mining companies. We defined treatments based on vegetation characteristics, so lack of succession on older grassland sites resulted in an age overlap between grasslands and shrublands. Forest areas were primarily mixed mesophytic hardwood forest that were ~60-80 years old and were composed mostly of red, white, and black oak (*Quercus rubra*, *Q. alba*, and *Q. velutina*); pignut, bitternut, and shagbark hickory (*Carya glabra*, *C. cordiformis*, and *C. ovata*); red and sugar maple (*A. saccharum*); and tuliptree. Remnant forests were patches of mature forest surrounded by reclaimed mine land on at least 3 sides. Intact forest sites were forested areas undisturbed by mining activities and were in close proximity to MTMVF areas either within the same watershed as a mining site or in an adjacent watershed, but sample points were ≥135 m from mine edge and shared no more than one edge with MTMVF areas.

Fig. 2. Geographic extent of bituminous coal bed (dark gray) and Bird Conservation Region 28 (BCR 28; light gray). Coal bed region and BCR 28 overlap approximately 40% within the Appalachian region of the eastern United States.

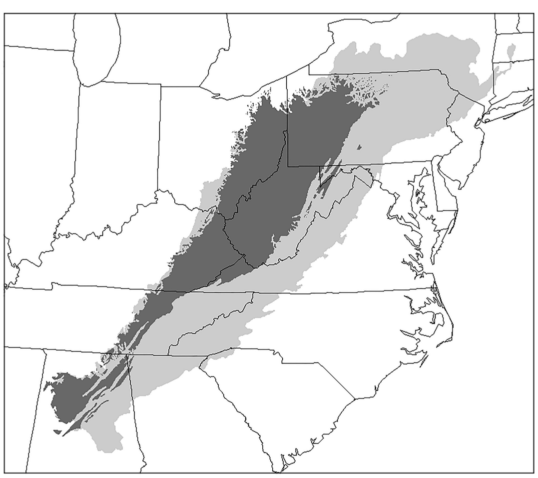
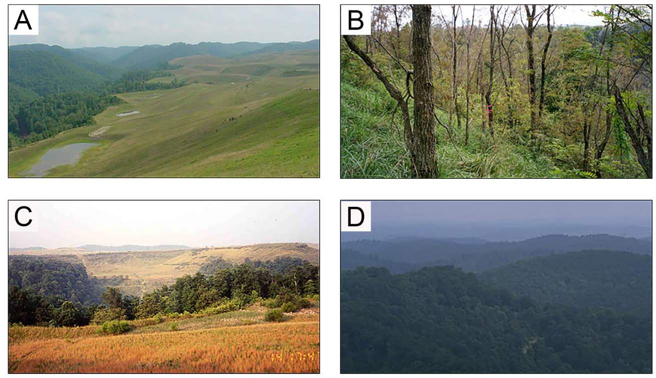


Fig. 3. Treatments used to assess songbird community response to mountaintop mining/valley fill in the central Appalachian region. Photos of treatments show typical habitat at the study site: reclaimed grassland (A), reclaimed shrubland (B), remnant forest (C), and intact forest (D).



Sampling points were distributed systematically throughout each treatment type to sample available aspects, elevations, and slope positions. Points were placed ≥75 m from the edge of any other treatment and ≥250 m apart. In 1999, 96 points across the 4 treatments were sampled: 30 in reclaimed grassland, 6 in reclaimed shrubland, 24 in remnant forest, and 36 in intact forest. In 2000, 156 points across the 4 treatments were sampled, i.e., 40 in reclaimed grassland, 33 in reclaimed shrubland, 36 in remnant forest, and 47 in intact forest, with some points from 1999 not resurveyed in 2000 because of additional disturbance or access restrictions.

We quantified songbird abundance using standard 50-m fixed radius, 10-min point counts (Ralph et al. 1993) from 0630 to 1030 h during late May to June of 1999 and 2000. All birds seen or heard in a 10-min period were recorded. We recorded whether the bird was observed visually or aurally, identified sex when possible, recorded whether it was flying over, and recorded whether it was within or outside the 50-m plot. Surveys were not conducted during windy or rainy weather. All points were surveyed twice each year, each time by a different observer. Two observers conducted all counts in 1999, and the same 2 observers plus a third person conducted all counts in 2000. All observers had previous experience identifying songbird species by sight and sound. Prior to initiating surveys, observers conducted simultaneous point counts to practice and verify bird identification skills and distance estimation.

We measured vegetation at all point count locations using methods modified from James and Shugart (1970) and the Breeding Bird Research Database program (Martin et al. 1997). We measured variables on 4 subplots (11.3-m radius or 0.04 ha each, and 5-m radius) at each point count location, with 1 subplot centered on the point count and 3 subplots 35 m from the center spaced 120° apart (0°, 120°, and 240°). Within each 0.04-ha subplot, we recorded trees >8 cm DBH. To quantify tree stem density, we counted the number of woody stems within each 0.04-ha subplot and grouped them into 1 of 2 DBH classes: >8-38cm and >38 cm DBH. Within a 5 m radius subplot, we counted the number of sapling stems (woody species >0.5 m high) and placed

Table 1. Mean and standard error (SE) of habitat variables in each treatment type. A one-way analysis of variance with a Waller-Duncan *k* ratio *t* test was used to test for significant differences among treatment types. Superscripted letters indicate significant differences at the α = 0.05 level.

	Reclaimed Grassland	Reclaimed Shrubland	Remnant Forest	Intact Forest	Waller-Duncan		
Variables	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE	*F*	df	*p*
Slope (%)[†]	16.96 ± 2.10^{B}	10.16 ± 1.93^{C}	33.78 ± 2.28^{A}	33.75 ± 2.07^{A}	42.95	3	< 0.001
Aspect code[†]	1.05 ± 0.10	0.77 ± 0.11	1.05 ± 0.12	1.03 ± 0.08	1.86	3	0.14
Elevation (m)[†]	400.27 ± 7.47^{A}	378.85 ± 11.53^{B}	332.08 ± 7.11^{C}	399.47 ± 11.24^{A}	24.94	3	< 0.001
Distance to major edge (m)	347.35 ± 44.30^{B}	253.98 ± 34.46^{C}	128.61 ± 12.52^{D}	1430.66 ± 145.32^{A}	537.85	3	< 0.001
Robel pole index	2.93 ± 0.17^{B}	4.30 ± 0.27^{A}	‡	‡	20.66	1	< 0.001
Canopy height (m)	‡	4.68 ± 0.46^{B}	21.77 ± 0.73^{A}	22.98 ± 0.67^{A}	222.63	2	< 0.001
Structural diversity index	‡	3.85 ± 0.29^{B}	11.58 ± 0.23^{A}	11.37 ± 0.22^{A}	262.81	2	< 0.001
Stem classes (stems/ha)							
< 2.5 cm	777.70 ± 207.52^{B}	7475.38 ± 1646.08^{A}	4935.76 ± 450.55^{A}	6135.84 ± 702.59^{A}	67.03	3	< 0.001
≥ 2.5-8 cm	73.15 ± 18.79^{C}	979.17 ± 292.52^{A}	901.04 ± 65.86^{A}	587.37 ± 55.71^{B}	79.55	3	< 0.001
> 8-38 cm	0.03 ± 0.02^{C}	132.58 ± 23.72^{B}	429.17 ± 35.26^{A}	352.93 ± 12.90^{A}	565.54	3	< 0.001
> 38 cm	0.00 ± 0.00^{B}	0.00 ± 0.00^{B}	44.27 ± 3.77^{A}	42.35 ± 3.17^{A}	993.28	3	< 0.001
Green ground cover (%)	82.78 ± 2.00^{B}	85.86 ± 3.47^{A}	30.35 ± 1.74^{C}	36.61 ± 1.99^{C}	130.34	3	< 0.001
Forb	23.63 ± 2.39	21.89 ± 2.86	‡	‡	0.07	1	0.79
Grass	45.05 ± 2.71	43.70 ± 5.26	‡	‡	0.15	1	0.70
Shrub	14.13 ± 2.72	22.99 ± 3.23	‡	‡	3.54	1	0.06

[†] Landscape-level variables used to estimate focal species' abundances (Table 3).
[‡] No data present for variable in that treatment.

them into 1 of 2 size classes: <2.5 cm diameter at 10 cm aboveground and ≥2.5-8 cm diameter at 10 cm aboveground. Additional variables collected were slope percent (hereafter termed slope), aspect, elevation (m), distance to nearest major edge (m), a Robel pole index (Robel et al. 1970), canopy height (m), canopy cover, stem density, and percent ground cover of forbs, grasses, and shrubs. We measured slope and canopy height using a clinometer and transformed slope using the arcsine square root transformation (Zar 1999) prior to analyses. We measured aspect using a compass and transformed values using the Beers transformation (Beers et al. 1966) before analyses, where northeastern facing slopes receive a value of 2 and reflect high productivity and mesic conditions, whereas southwestern exposures receive a value of 0 and reflect xeric conditions, with all other exposures distributed between these values. We evaluated elevation using digital elevation models in ArcView GIS. We measured distance to major edge, represented by considerable breaks in contiguous habitat, from aerial photographs, where intact and remnant forest edges included valley fills and grasslands in mined areas, and grassland and shrubland edges were primarily forests. We measured total green ground cover using an ocular sighting tube (James and Shugart 1970) in all treatments and separated cover type into forb, grass, and shrub for reclaimed grasslands and shrublands. We quantified vegetation cover complexity using a Robel pole in reclaimed grasslands and shrublands. Structural diversity index measured the amount of canopy cover of different layers (0.5-3 m, >3-6 m, > 6-12 m, >12-18 m, >18-24 m, and >24 m) and the number of layers present (Nichols 1996). Canopy height and structural diversity index were not measured in reclaimed grasslands.

Analyses

Habitat characteristics

We compared the 16 topographic and vegetative characteristics (Table 1) among treatment types with one-way analysis of variance (ANOVA) tests. We used ANOVA because it is robust to conditions of nonnormality (Zar 1999) and robust to moderate departures from homogeneity of variance (Dowdy et al. 2004). By examining probability plots, we found that residuals of the raw data did not deviate excessively from normality. When ANOVAs indicated significant differences among treatment types, we used Waller-Duncan *k* ratio *t* tests to determine which treatment types differed. Variables were deemed significantly different at α = 0.05 and are reported as mean ± standard error unless otherwise stated.

Bird community-level patterns

We used 50 m radius point count data from 1999 and 2000 to assess community-level patterns in avian richness and abundance. We excluded species that are not sampled well using point counts, i.e., flocking species, species with large territories, and nonvocal species, as well as all flyovers regardless of species for analyses, but we retained all other detection types, e.g., visual and auditory (song and call). We placed detected species into five habitat guilds (grassland, shrubland, interior edge, forest interior, and synanthropic; Appendix 1) a priori based on breeding biology of regional bird species (Ehrlich et al. 1988) and previous studies from the region (Whitcomb et al. 1981, O'Connell et al. 2000, McDermott and Wood 2009, Thomas et al. 2014, Farwell et al. 2016). Grassland guild species are associated with grasslands or prairies, shrubland guild species are associated with shrub/scrub or recently disturbed habitats, interior-edge guild birds are species commonly found in mature forests but that are tolerant of forest edges, forest interior guild birds are species that are associated with large tracts of core mature forest, and synanthropic guild birds are species that show a symbiotic relationship with humans.

We calculated guild richness as the cumulative number of species detected across both surveys each year at each point count station. For points surveyed in both years, we used the mean richness from the two years to account for unequal sampling effort among points and treatments. We compared richness of each guild across the

Table 2. Site-level a priori candidate models tested for each species (top section) and resulting species best-fit model (lower section) to estimate abundances of focal species (see Table 3 for abundance estimates) using a single-season binomial *N*-mixture model. Treatment was included as a site-level covariate in all models. Other site-level covariates (slope, aspect, and elevation) were considered nuisance variables that helped explain patterns in abundance, independent of "treatment." Survey-level covariates (*p*) included "observer" and "time since sunrise" for all candidate models. See Table 3 for scientific names of species.

Site-Level Covariates Tested (λ)	
Treatment	
Treatment + slope	
Treatment + aspect	
Treatment + elevation	
Treatment + slope + aspect + elevation	
Best-fit model for species abundance estimates	
Eastern Meadowlark	*p* (observer + time since sunrise), λ (treatment)
Grasshopper Sparrow	*p* (observer + time since sunrise), λ (treatment + slope)
Blue-winged Warbler	*p* (observer + time since sunrise), λ (treatment + elevation)
Field Sparrow	*p* (observer + time since sunrise), λ (treatment + slope)
American Redstart	*p* (observer + time since sunrise), λ (treatment + aspect)
Hooded Warbler	*p* (observer + time since sunrise), λ (treatment)
Cerulean Warbler	*p* (Observer + time since sunrise), λ (treatment + aspect)
Wood Thrush	*p* (observer + time since sunrise), λ (treatment)
Blue Jay	*p* (observer + time since sunrise), λ (treatment)
Northern Cardinal	*p* (observer + time since sunrise), λ (treatment + aspect)

four treatments using one-way ANOVAs, followed with Waller-Duncan *k* ratio *t* tests when ANOVAs were significant. Guild richness was deemed significantly different at α = 0.05 and was reported as mean ± standard error unless otherwise stated.

We used redundancy analysis (RDA) to determine if there was a community-level effect of treatment type and to visualize species associations with treatments. Species that had ≥5 total occurrences over the study period were included in RDA; thus 58 of 90 detected species were included in the analysis (Appendix 1). For the analysis, we used the number of detections per visit for each species averaged across all visits because of uneven sampling effort across our study. We transformed the count data using the "Hellinger" method to remove the influence of double zeros (Legendre and Gallagher 2001). We performed a detrended correspondence analysis on the transformed data to ensure the gradient length was <4 (gradient length = 3.18), and thus species responses to the environmental gradient could be fit with a linear model (Lepš and Šmilauer 2003, Legendre and Legendre 2012). We used a correlation matrix, which gives equal weight to all species (Legendre and Legendre 2012). We visually assessed species-level associations using a distance biplot, where angles between the species data and treatment types reflect their correlations, and Euclidean distance reflects the differences among treatment types (Borcard et al. 2011). We performed the RDA using the software package vegan (version 2.3-4; Oksanen et al. 2016) in program R (R Core Team 2016).

Bird focal-species abundances

To examine the influence of site- and survey-specific covariates on abundance, we evaluated five a priori models (Table 2) for each of two focal species from each of the five guilds represented within our study area (Table 2, Appendix 1). Nonsynanthropic species were chosen based on their high assessment scores for conservation need (Rosenberg et al. 2016; Appendix 1). We used a single-season binomial *N*-mixture model to account for imperfect detectability associated with point count surveys (Royle 2004). Site covariates included treatment, i.e., intact forest, remnant forest, reclaimed shrubland, and reclaimed grassland, as our primary variable of interest, and slope, Beers aspect, and elevation because these topographic variables can affect avian species presence and abundance and were varied across survey points. Survey covariates, observer and time since sunrise, were included in all models. Because species with low detections in certain treatments, e.g., Grasshopper Sparrows (*Ammodramus savannarum*) in intact and remnant forests, caused nonconvergence in models, we restricted analyses to treatments with sufficient detections to satisfy convergence for models. We used a second-order information criterion (AIC_c) to account for small sample size and selected a best model (lowest AIC_c) from all candidate models using an information theoretic approach (Burnham and Anderson 2002). The global model for each species was tested for overdispersion (c-hat ≥1) using a goodness-of-fit test with α = 0.05 and subsequently accounted for in our treatment-level best-fit model abundance estimates by inflating the 95% confidence intervals based on global model c-hat values (Kéry and Royle 2016).

We used 50 m radius point count data from 1999 and 2000 and only included singing or visual detections of males for analysis. Abundances were estimated at the "birds per point" level for each treatment type, with all other site-specific covariates set at their mean values. We chose single-season rather than multiseason models because number of survey replicates was low, some points were only surveyed 1 year, and we were interested in estimating abundance differences of songbirds among treatments rather than changes between years. Thus, each point contributed either 2 or

Table 3. Mean abundance estimates derived from the best-supported single-season binomial *N*-mixture model from species-specific analysis at the "per point" level (95% confidence intervals) in each treatment.

	Treatment Type			
	Reclaimed Grassland	Reclaimed Shrubland	Remnant Forest	Intact Forest
Grassland species				
Eastern Meadowlark (*Sturnella magna*)	0.99 (0.41-2.40)	0.14 (0.02-0.79)	†	†
Grasshopper Sparrow (*Ammodramus savannarum*)	5.80 (2.39-13.98)	0.58 (0.23-1.47)	†	†
Shrubland species				
Blue-winged Warbler (*Vermivora cyanoptera*)	0.95 (0.39-2.33)	3.06 (1.33-7.05)	0.20 (0.07-0.58)	0.17 (0.05-0.55)
Field Sparrow (*Spizella pusilla*)	1.39 (0.70-2.75)	2.69 (1.34-5.38)	†	†
Interior-edge species				
American Redstart (*Setophaga ruticilla*)	†	0.24 (0.07-0.84)	0.53 (0.22-1.18)	1.53 (0.80-2.91)
Hooded Warbler (*Setophaga citrina*)	0.06 (0.01-0.56)	0.27 (0.06-1.19)	0.55 (0.17-1.71)	1.67 (0.61-4.55)
Forest interior species				
Cerulean Warbler (*Setophaga cerulea*)	0.10 (0.02-0.47)	†	1.18 (0.45-3.11)	1.84 (0.71-4.78)
Wood Thrush (*Hylocichla mustelina*)	0.20 (0.03-1.45)	†	3.16 (0.74-13.39)	3.14 (0.74-13.26)
Synanthropic species				
Blue Jay (*Cyanocitta cristata*)	0.56 (0.07-4.71)	3.82 (0.47-30.96)	2.91 (0.35-24.12)	3.74 (0.53-26.19)
Northern Cardinal (*Cardinalis cardinalis*)	0.86 (0.45-1.67)	2.48 (1.26-4.88)	1.16 (0.54-2.44)	0.59 (0.29-1.22)

† Insufficient data for model convergence; removed from species abundance estimates.

4 survey replicates to estimation of model parameter values. Single-season models do assume closure during the sampling period, and thus the variability in counts is caused by imperfect detection rather than changes in abundance. To assess if major changes in focal species abundances likely occurred at survey points between years, we conducted paired *t* tests using maximum single-visit detections each year. Only 1 of 10 focal species, i.e., Blue-winged Warbler (*Vermivora cyanoptera*), showed a significant difference in number of detected males between years. Thus, overall our focal species populations appeared to be stable during the sampling period, and our single-season models should not be heavily biased by violation of closure. *N*-mixture model analyses were conducted using the packages unmarked (Fiske and Chandler 2011), AHMbook (Kéry and Royle 2016), and AICcmodavg (Mazerolle 2016) in program R (R Core Team 2016). We then used the top model for each species as determined by *N*-mixture model selection (Table 2) to generate modeled abundance corrected for detection and to assess songbird abundance differences among treatment type (Table 3).

RESULTS

Habitat characteristics

Vegetation variable differences were generally split into forested treatments (remnant and intact forest) and nonforested reclaimed treatments (grasslands and shrublands). Forested treatments had greater structural diversity and canopy height than reclaimed shrublands; canopy height and structural diversity were not measured in reclaimed grasslands. Total green ground cover was significantly higher in reclaimed treatments (84%) than in forested treatments (33%). Stem densities of trees >8 cm DBH, i.e., >8-38 cm and >38 cm DBH, were significantly greater in forested treatments than reclaimed treatments, whereas woody stems ≤8 cm DBH showed no clear distinction between forested and reclaimed treatments. Slope was significantly greater, i.e., steeper, in forested treatments than reclaimed treatments, whereas other landscape variables, i.e., distance to major edge and elevation, were significantly different among treatments but showed no clear separation between forested and reclaimed treatments.

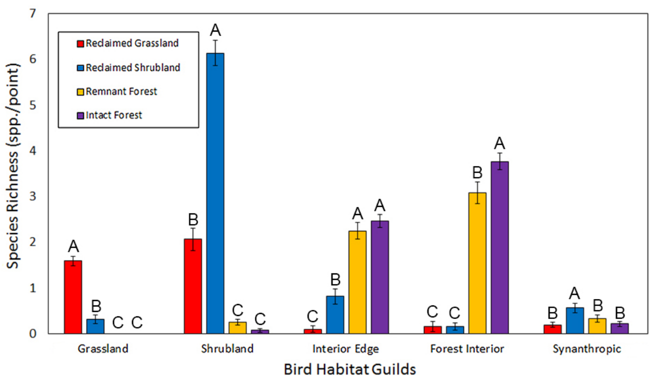
Habitat variables measured in remnant and intact forests were generally similar (Table 1). Only one vegetation variable differed, with a greater number of stems >2.5-8 cm DBH in remnant forest. Distance to major edge (1430 m and 128 m in intact forest and remnant forest, respectively) and elevation (399 m and 332 m for intact forest and remnant forest, respectively) differed significantly, although actual elevation values were not substantially different.

Many vegetative variables measured in reclaimed grasslands and shrublands were significantly different between treatments, e.g., slope, elevation, distance to major edge, Robel pole index, stem density, and green ground cover, with stem density being the most notable (Table 1). Reclaimed shrublands had significantly higher stem densities in <2.5 cm, ≥2.5-8 cm, and >8-38 cm classes than reclaimed grasslands, and reclaimed shrublands had a significantly higher Robel pole index. Total green ground cover was significantly different between these two treatments (83% and 86% in grasslands and shrublands, respectively), whereas forb, grass, and shrub ground cover did not differ.

Bird community-level patterns

Bird guild richness was significantly different across treatment types for each guild: grassland, $F_{3,158} = 138.0$, $p < 0.001$; shrubland, $F_{3,158} = 208.1$, $p < 0.001$; interior edge, $F_3 = 73.9$, $p < 0.001$; forest interior, $F_{3,158} = 144.1$, $p < 0.001$; and synanthropic, $F_{3,158} = 5.3$, $p < 0.001$. Guild richness generally followed expected patterns of habitat associations (Fig. 4). Grassland bird guild richness was greatest in reclaimed grasslands. Shrubland and synanthropic guilds had greatest richness in reclaimed shrublands. Richness for interior-edge species was greatest in both intact forests and remnant forests. In contrast, forest interior bird guild richness was greatest in intact forests. Reclaimed shrublands had greatest overall species richness because all guilds except forest interior contributed high or moderate richness.

Fig. 4. Species richness of five bird guilds in four central Appalachian mountaintop mining/valley fill treatments. Letters notate significant differences within each guild across treatment types at $p = 0.05$, with mean and standard error shown.



The RDA based on detections within a 50-m radius also indicated species-habitat associations were nonrandom ($F_{3,7} = 4.2$, $p < 0.001$). We found that 23% of the variance was explained by treatment type. All forest interior species except one, Hairy Woodpecker (*Picoides villosus*), were tightly grouped with intact forest, as were many interior-edge species (Fig. 5). Forest treatments were distinctly different from grassland and shrubland treatments, indicated by spatial separation of treatment labels in Figure 5.

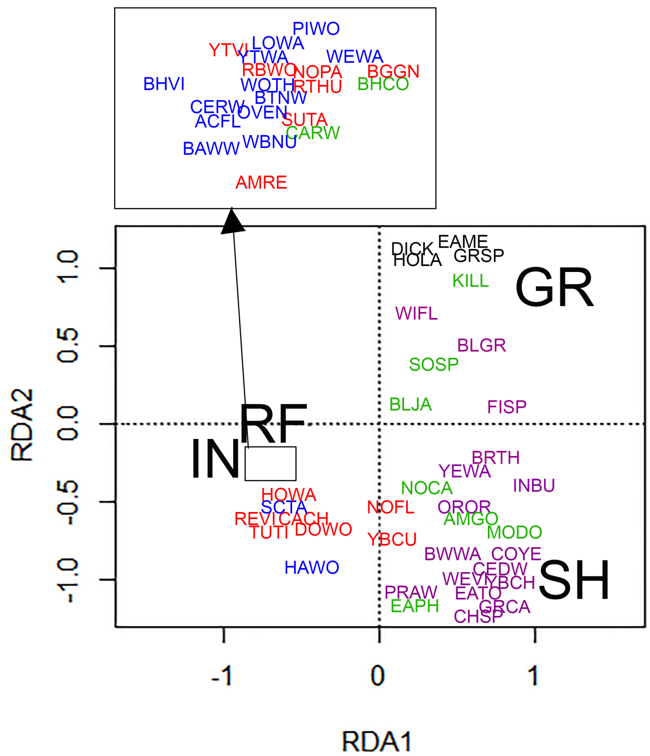
Bird focal-species abundances

Focal species abundances were estimated using the best-fit model (Table 2). Songbird species modeled abundances were highest in their respective a priori guild groupings and mirrored trends in guild richness (Table 3). Grassland and shrubland species had highest abundance estimates in treatments that had been reclaimed (grasslands and shrublands) versus treatments that remained forested (Table 3). Grasshopper Sparrows (scientific names in Table 3) had noteworthy high abundance in reclaimed grasslands. Species in interior-edge and forest interior guilds had the highest abundance estimates in treatments that had not been mined and reclaimed (remnant and intact forest). Three of the four forest-associated species analyzed, i.e., American Redstart (*Setophaga ruticilla*), Cerulean Warbler, and Hooded Warbler (*Setophaga citrina*), had higher abundance estimates in intact forests than remnant forests. Species in the synanthropic guild were relatively common across all treatments, with abundance being highest in reclaimed shrubland (Table 3).

DISCUSSION

Mined lands reclaimed with the traditional grassland approach supported fewer species (Figs. 4 and 5) and lower estimated abundances of focal species (Table 3) than reclaimed shrubland, remnant forest, and intact forest for all but the grassland songbird guild. Although reclaimed shrublands on our study areas were not a result of applying the forest reclamation approach guidelines (Adams 2017), they still supported higher abundance and richness of songbirds typically associated with early successional or young forests in the central Appalachian region. This supports the notion that the forest reclamation approach, where trees are

Fig. 5. Distance biplot showing species-habitat associations in four central Appalachian mountaintop mining/valley fill treatment types: reclaimed grassland (GR), reclaimed shrubland (SH), remnant forest (RF), and intact forest (IN). Species alpha codes are provided in Appendix 1 and are colored according to their habitat guild: grassland (black), shrubland (purple), interior edge (red), forest interior (blue), and synanthropic (green). Spatial locations of species alpha codes indicate degree of similarity to treatment type. Inset box shows tight clustering of forest species with IN and RF. RDA, redundancy analysis.



planted with the goal of creating young forest areas immediately and expediting forest succession, would benefit shrubland and forest-associated songbirds.

Our study corroborated our hypothesis that the bird community differed between reclaimed MTMVF complexes and adjacent unmined forests in southwestern West Virginia. These results justify further examination into the mine reclamation process in the central Appalachian region and the trade-offs associated with grassland and forest mine reclamation techniques. Analysis of the bird community indicates reclaimed grasslands on MTMVF complexes were used by grassland songbirds and a few shrubland species. Reclaimed grasslands were characterized by dense ground cover with no canopy present and yielded the lowest total species richness of all treatments. Despite low species richness of grassland species compared with other guilds, mining complexes were used by grassland birds in our study and others (DeVault et al. 2002, Scott and Lima 2004). Grasshopper Sparrows, in particular, had high abundance on our sites (Table 3). Grassland songbirds have been declining throughout much of North

America (Ribic et al. 2009), and mines reclaimed with grasslands provide habitat for this group of species, although grassland species are naturally rare in the central Appalachians (Wood and Ammer 2015). Forest-associated songbirds (interior-edge and forest interior species) occurred in low numbers in reclaimed grasslands and shrublands, suggesting MTMVF complexes were suboptimal for forest birds. We also provided evidence of forest-dwelling bird displacement on MTMVF complexes, particularly area-sensitive species, which has rarely been reported (Weakland and Wood 2005, Wood et al. 2006). Forest-associated species were linked with remnant and intact forest, though more strongly with intact forest, showing their preference toward forested areas in a mining matrix landscape.

Grassland birds are habitat specialists, and some are area sensitive (Walk and Warner 1999, Johnson and Igl 2001); thus, large areas of reclaimed grasslands may prove important for this group of birds (Stauffer et al. 2011). When surface mines are reclaimed using the grassland approach, they are often in an arrested state of succession because of the compaction and acidity of soils. Studies have reported grassland birds using these areas 20 years after mine reclamation (DeVault et al. 2002, Ingold and Dooley 2013, Borthwick and Wang 2015, Wood and Ammer 2015), and this was evident in our study, where grassland species were present 19 years after mine reclamation. Despite reclaimed mines providing long-term habitat for grassland birds, quality can decline over time leading to reduced nesting productivity (Wood and Ammer 2015). This degradation in reclaimed grasslands over time may result from nonnative grasses being predominant in these areas (Scott and Lima 2004), which reduce insect production and create poor breeding conditions (Galligan et al. 2006). Grasslands on reclaimed mines differ from native prairies in that plant diversity is low on mines with few native species present (Scott et al. 2002), and reclaimed mines often are dominated by species that are adapted to infertile soils, e.g., tall fescue and smooth brome (*Bromus inermis*; Brothers 1990, Wood and Ammer 2015). Further research is needed comparing native and nonnative plant reclamation on surface mines and how this may affect long-term plant composition and reproductive benefits for songbirds. Additionally, grasslands are not historically prominent within the central Appalachians (Hall 1983) and are mainly a by-product of surface mining (Townsend et al. 2009) or land conversion for agriculture.

Reclaimed shrublands in our study were generally a product of the grassland reclamation approach, but with additional woody plantings and some natural regeneration of woody species. Reclaimed shrublands provided a large array of vegetative conditions on mines, with the highest stem densities in the <2.5 cm and ≥2.5-8 cm DBH classes, the highest percentage of total ground cover, and some overstory cover and vegetative structure (Table 1). The combination of understory vegetation mixed with partial canopy cover created characteristics suitable for a wide variety of birds, e.g., reclaimed shrubland contained the highest species richness for shrubland and synanthropic guilds and second-highest for interior edge (Fig. 4). Although shrubland birds have been understudied on reclaimed surface mines (Bulluck and Buehler 2006), they made up a large portion of the overall bird community in our study (16 of 58 species, the most species for a single guild; Appendix 1). Because habitat use does not always equal habitat suitability (Van Horne and Wiens 1991), comparisons of reproductive success of shrubland songbird species in the grassland and forest reclamation approaches is needed. Because woody encroachment onto grasslands will likely push grassland birds off of surface mines (Graves et al. 2010, Hill and Diefenbach 2013), Graves et al. (2010) suggested that grasslands be maintained as grasslands because of the improbability of reestablishment of a functioning forest ecosystem because of poor soil conditions. Although reclaimed shrublands provide important habitat to many shrubland and young forest species that have been declining throughout the Appalachian region (Schlossberg et al. 2010), the slow succession on mines reclaimed with grasses results in very few areas developing woody vegetation. If shrubland or young forest conditions are the desired future condition, then the forest reclamation approach may be a more suitable approach to more quickly develop woody vegetation postreclamation (Zipper et al. 2011, McDermott et al. 2013, Wood et al. 2013).

Remnant and intact forest shared many vegetation characteristics and supported similar avian species composition (Fig. 5), but intact forests supported greater richness of forest interior species (Fig. 4), and 3 of 4 forest-dependent focal species had higher estimated abundances in intact forests (Table 3). Both of the forest treatments were characterized by greater canopy height and greater number of large diameter trees (8-38 cm and >38 cm DBH classes) compared with reclaimed grassland and shrubland (Table 1), which contributed to the lower abundances and guild richness of forest species on mine complexes. Intact forest was associated with core forested areas, and sample points were located farther from nonforested edges compared with remnant forest. Thus, lower abundance of area-sensitive forest species in MTMVF complexes may also result from the fragmentation or complete loss of core forest (Becker et al. 2015, Farwell et al. 2016), likely because of reduced amounts of intact forest and increased edge. Reclaiming mines using a forest approach may lessen the impacts from mining on the forest bird community by quickly developing young forest and shrubland, which softens habitat edges and increases forest core area (McDermott et al. 2013). However, because the forest reclamation approach has been implemented only in recent years and on small acreages, no studies have examined avian response on mined lands reclaimed using the forest reclamation approach.

Revegetation techniques on surface mines have long-term effects on habitat suitability for birds and other wildlife because of slow successional processes (Evans et al. 2013). Traditional reclamation practices encouraged mining companies to primarily reclaim mines with grasses (Zipper et al. 2011), which addressed the need to reduce soil erosion, sedimentation, landslides, and instability. This approach typically slows the successional processes of woody plants, reducing the likelihood of returning these areas to native forests and subsequently the return of forest birds. Although the forest approach is relatively new compared with the traditional grassland approach, some studies have reported faster establishment of woody plants and higher growth rates of tree species when using the forest reclamation approach (Burger and Zipper 2011, Evans et al. 2013). The forest approach could benefit many young forest bird species by creating dense patches of native shrubs with sapling and herbaceous cover (Bakermans et al. 2009, Wood et al. 2013). Although no forest approach sites are old enough to have developed into mature forest, we would expect the forest reclamation approach to increase the likelihood of forest bird return to MTMVF complexes and not just to the complexes' peripheral forests (Burger and Zipper 2011, McDermott et al. 2013).

In summary, surface mines reclaimed with grasses support grassland birds, but benefits may deteriorate over time (Wood and Ammer 2015). Concurrently, forest birds, particularly area-sensitive or conservation concern species, are adversely affected by mines reclaimed with grasses based on species abundances and guild richness from our study. Ultimately, there are winners and losers associated with the decision to reclaim mines with either grasses or trees. Both bird groups have seen declining populations throughout North American (Askins et al. 2007, Sauer et al. 2014) in recent decades, but managing for both at a relatively small scale, e.g., mine landscapes (~2500 ha in our study), may be difficult. In a region like the central Appalachians, where intact, mature forest is the predominant, native land cover and consequently an important focus for forest bird conservation, it is important that we understand ways of returning forests to these areas that have been highly subjected to disturbance. The Appalachian Mountains Joint Venture lists 9 land bird species as the highest conservation priority: Bewick's Wren (*Thryomanes bewickii*), Blue-winged Warbler, Cerulean Warbler, Golden-winged Warbler (*Vermivora chrysoptera*), Kentucky Warbler (*Geothlypis formosa*), Prairie Warbler (*Setophaga discolor*), Wood Thrush, Worm-eating Warbler (*Helmitheros vermivorum*), and Henslow's Sparrow (*Ammodramus henslowii*). All but one, Henslow's Sparrow, are considered forest interior, interior-edge, or shrubland species. The Henslow's Sparrow is an obligate grassland species and considered rare in the central Appalachian region. Therefore, from an avian conservation standpoint, surface mine reclamation in the central Appalachians could focus on implementing a forest reclamation approach over a traditional grassland approach. Recovery of forest bird presence on mines reclaimed using the grassland approach is unknown, but it is postulated to be hundreds of years for functioning forests to develop (Zipper et al. 2011, Wickham et al. 2013).

Although songbird response to the forest reclamation approach has not been studied explicitly, the results from our reclaimed shrubland treatment suggest that shrubland birds would respond positively. Further, the grassland approach has proved to limit forest bird abundances in areas that previously supported forest communities. Because large areas in the Appalachian region have the potential for surface mining (Fig. 2), it is important to understand the trade-offs of reclaiming mined lands with a grassland or forest approach. A clearer understanding of best management practices for mine reforestation (Adams 2017) will help practitioners make sound management decisions that benefit birds, other wildlife, and whole ecosystems. Thus, further research on sites reclaimed with the forest approach and resampling of reclaimed sites older than those sampled in our study would provide valuable data to help determine effective management recommendations for mine reclamation in the future.

Responses to this article can be read online at:
http://www.ace-eco.org/issues/responses.php/1304

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Appendix 1. Alpha codes for species used in redundancy analysis grouped into their respective habitat guild. The Partners in Flight (PIF) Conservation Score ranges from 5 to 20 indicating least to greatest concern (Rosenberg et al. 2016).

Habitat Guild	Common Name	Scientific Name	PIF Score	Alpha Code
Grassland	Dickcissel	*Spiza americana*	11	DICK
	Eastern Meadowlark	*Sturnella magna*	11	EAME
	Grasshopper Sparrow	*Ammodramus savannarum*	12	GRSP
	Horned Lark	*Eremophila alpestris*	9	HOLA
Shrubland	Blue Grosbeak	*Passerina caerulea*	8	BLGR
	Blue-winged Warbler	*Vermivora cyanoptera*	13	BWWA
	Brown Thrasher	*Toxostoma rufum*	10	BRTH
	Cedar Waxwing	*Bombycilla cedrorum*	7	CEDW
	Chipping Sparrow	*Spizella passerina*	8	CHSP
	Common Yellowthroat	*Geothlypis trichas*	9	COYE
	Eastern Towhee	*Pipilo erythrophthalmus*	11	EATO
	Field Sparrow	*Spizella pusilla*	12	FISP
	Gray Catbird	*Dumetella carolinensis*	8	GRCA
	Indigo Bunting	*Passerina cyanea*	9	INBU
	Orchard Oriole	*Icterus spurius*	10	OROR
	Prairie Warbler	*Setophaga discolor*	13	PRAW
	White-eyed Vireo	*Vireo griseus*	8	WEVI
	Willow Flycatcher	*Empidonax traillii*	11	WIFL
	Yellow-breasted Chat	*Icteria virens*	10	YBCH
	Yellow Warbler	*Setophaga petechia*	8	YEWA
Interior-edge	American Redstart	*Setophaga ruticilla*	10	AMRE
	Blue-gray Gnatcatcher	*Polioptila caerulea*	7	BGGN
	Carolina Chickadee	*Poecile carolinensis*	9	CACH
	Downy Woodpecker	*Picoides pubescens*	7	DOWO
	Hooded Warbler	*Setophaga citrina*	9	HOWA
	Northern Flicker	*Colaptes auratus*	9	NOFL
	Northern Parula	*Setophaga americana*	8	NOPA
	Red-bellied Woodpecker	*Melanerpes carolinus*	7	RBWO
	Red-eyed Vireo	*Vireo olivaceus*	6	REVI
	Ruby-throated Hummingbird	*Archilochus colubris*	8	RTHU
	Summer Tanager	*Piranga rubra*	9	SUTA
	Tufted Titmouse	*Baeolophus bicolor*	7	TUTI
	Yellow-billed Cuckoo	*Coccyzus americanus*	12	YBCU
	Yellow-throated Vireo	*Vireo flavifrons*	9	YTVI
Forest Interior	Acadian Flycatcher	*Empidonax virescens*	11	ACFL
	Black-and-white Warbler	*Mniotilta varia*	11	BAWW
	Black-throated Green Warbler	*Setophaga virens*	9	BTNW
	Blue-headed Vireo	*Vireo solitarius*	7	BHVI
	Cerulean Warbler	*Setophaga cerulea*	15	CERW
	Hairy Woodpecker	*Picoides villosus*	6	HAWO
	Kentucky Warbler	*Geothlypis formosa*	14	KEWA
	Louisiana Waterthrush	*Parkesia motacilla*	12	LOWA
	Ovenbird	*Seiurus aurocapilla*	9	OVEN
	Pileated Woodpecker	*Dryocopus pileatus*	7	PIWO
	Scarlet Tanager	*Piranga olivacea*	12	SCTA
	White-breasted Nuthatch	*Sitta carolinensis*	6	WBNU

	Worm-eating Warbler	*Helmitheros vermivorum*	13	WEWA
	Wood Thrush	*Hylocichla mustelina*	14	WOTH
	Yellow-throated Warbler	*Setophaga dominica*	10	YTWA
Synanthropic	American Goldfinch	*Spinus tristis*	7	AMGO
	Brown-headed Cowbird	*Molothrus ater*	7	BHCO
	Blue Jay	*Cyanocitta cristata*	8	BLJA
	Carolina Wren	*Thryothorus ludovicianus*	7	CARW
	Eastern Phoebe	*Sayornis phoebe*	8	EAPH
	Killdeer	*Charadrius vociferus*	NA	KILL
	Mourning Dove	*Zenaida macroura*	6	MODO
	Northern Cardinal	*Cardinalis cardinalis*	5	NOCA
	Song Sparrow	*Melospiza melodia*	8	SOSP