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ABSTRACT

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Jeremy David Mizel

Since the passage of the Surface Mining Control and Reclamation Act (SMCRA) in 1977, mined lands have generally been reclaimed to an environment characterized by severely compacted minesoils, a growth medium comprised largely of unweathered materials, and a predominance of aggressive groundcovers that inhibit native species colonization. Under these conditions, succession is arrested. Within landscapes that are fragmented by traditionally reclaimed surface mines, forest patches are smaller and forest cover on the landscape scale is reduced. As a result, forest songbirds that require large, continuous blocks of forest are negatively affected.

Some pre-SMCRA abandoned minelands contain areas of uncompacted minesoils on which hardwood forest has developed in the absence of aggressive groundcovers. Despite potential differences in tree species composition, study of the relationship between habitat structure and the avian assemblage within pre-SMCRA mineland forest could provide insight into the species assemblages that future mineland reforestation efforts might yield. Study of the reproductive success of forest songbirds within pre-SMCRA mineland forest may provide some indication as to whether this habitat is capable of sustaining breeding songbird populations. In chapter two of this thesis, I detail research in which my objectives were to: 1) examine patterns in avian assemblage structure within mineland and reference forest and to link the avian assemblage response to variables describing habitat structure and composition, and 2) contrast nest survival of Red-eyed Vireos (*Vireo olivaceus*) breeding within mineland and reference forest.

I conducted this research in 2011 within New River Gorge National River in southern West Virginia. I surveyed avian assemblages and sampled stand structure and composition along 28 fixed-width line transects (14 mined and 14 reference) established within four pre-SMCRA abandoned minelands and adjacent, unmined forest. Minelands within these study areas were were relatively wide (80-100 m wide on average) and contained mature forest (60-65 years old) that had developed from areas of loose-dumped spoil mounded atop benches and also within outslopes.

Using an information-theoretic approach, I developed *a priori* models containing habitat and temporal covariates that I hypothesized to influence the nest survival of Red-eyed Vireos. Within the same study area, I monitored vireo nests within three mineland forest plots and three reference forest plots.

Ordination of avian assemblages using non-metric dimensional scaling (NMDS) showed clear discrimination between mineland and reference assemblages. Linear and surface fitting of habitat variables showed strong correlations between the ordination and groundcover gradients, but generally non-significant relationships for gradients describing forest structure. Mineland assemblages were associated with lower levels of litter cover and depth and also had lower abundance of Ovenbirds (*Seiurus aurocapillus*), a ground-nesting and foraging species. Within mineland assemblages, the absence of a consistent pattern of relationships among species suggested a wider habitat gradient relative to reference forest.

I monitored 45 Red-eyed Vireo nests, 21 within mineland forest and 24 within reference forest. Nest survival for Red-eyed Vireos was similar within mineland and reference forest and nest patch characteristics (overstory cover and vertical foliage density) had minimal effect on nest survival. Classification tree modeling using forest type as the response variable indicated that reference nest sites were characterized by greater vertical heterogeneity.

In chapter three, I report on research initiated with the objective of examining patterns in avian assemblage structure in response to the presence of two broad classes of minelands on the landscape, compacted bench minelands and loose-dumped bench minelands. This research was conducted in 2010 and indicated the approach taken in 2011 (chapter two). I conducted fieldwork within five study sites in New River Gorge National River and Plum Orchard Wildlife Management Area (WMA). Point count transects were classified as loose-dumped benches, unmined plateau, compacted benches, and unmined steep slope. NMDS ordination indicated that minelands with loose-dumped benches had minimal effect on assemblage structure. The assemblage associated with compacted bench minelands was not discrete, but was largely discriminated from the other assemblage types. Species that use the subcanopy and midcanopy for nesting and foraging were discriminating components of compacted bench assemblages. Relative abundance of the closed-canopy guild was lower within mined forest than within unmined forest.

In total, this research has shown that failure to establish mineland stands in which heavy-seeded species are a component has important implications for avian assemblage structure. Within minelands, heterogeneity in edaphic conditions and the corresponding variation in forest structure likely contributed to an inconsistent pattern in avian assemblage structure.

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Chapter 1

INTRODUCTION

INTRODUCTION

Surface mining first attained widespread use in Appalachia in the 1940s (Potter et al. 1951). In general, surface mining in the steep slopes of this region has taken the form of contour mining and larger-scale mountaintop mines where mountaintop removal, contour, and auger/highwall mining are being employed.

Post-mining land use has varied greatly since the 1940s (Potter et al. 1951). Prior to the enacting of the Surface Mining Control and Reclamation Act (SMCRA) in 1977, mines used a “shoot and shove” method for extraction resulting in an exposed highwall, bench-land along its face, and an outslope comprised of loose spoil that had been pushed below. Mines were left in these “shoot and shove” configurations and reclamation practices varied from state to state (Brown 1962). In states such as Ohio, early reclamation efforts typically involved planting hardwoods and pine on mined lands (Paton el al. 1970, Rodrigue 2001). In West Virginia, surface mines were generally abandoned without planting (Brown 1962, Rodrigue 2001).

SMCRA requires that mine operators “backfill, compact, and grade in order to restore the approximate original contour (AOC) of the land with all highwalls, spoil piles, and depressions eliminated" (Office of Surface Mining Reclamation and Enforcement 2008). Mine companies, dozer operators, and regulatory agencies responded with expectations of a uniformily smooth reclamation landscape. The traditional post-SMCRA reclamation environment is one in which unweathered materials contribute heavily to the growth medium, minesoils have been severely compacted from immoderate grading, and aggressive groundcovers have been sown to prevent erosion (Burger et al. 2002).

During the two decades that followed the passage of SMCRA (1977), surface mines were generally reclaimed to grassland or woodland that was planted or seeded to black locust (*Robinia pseudo-acacia*), Virginia pine (*Pinus virginiana*), eastern white pine (*P. strobes*), autumn olive (*Elaeagnus umbellate*), and highly tolerant, aggressive ground covers including Kentucky-31 tall fescue (*Festuca arundinacea*) and sericea lespedeza (*Lespedeza cuneata*) (Burger et al. 2002, Burger et al. 2005, Showalter and Burger 2006). Most native hardwood species have poor survival and growth in this environment (Burger et al. 2002).

The Appalachian Regional Reforestation Initiative (ARRI) was established by the Office of Surface Mining in 2004 with the primary objective of encouraging surface mine reforestation through a method termed the Forestry Reclamation Approach (FRA). The Forestry Reclamation

Approach is a series of field-tested guidelines directed at advancing forest succession on minelands (Burger and Fannon 2009, Skousen et al. 2009). Components of this approach include creating an uncompacted growth medium in which hardwoods are capable of exhibiting normal survival and growth and seeding non-aggressive groundcovers which do not completely inhibit forest plant invasion (Burger et al. 2005).

MINED LAND SUCCESSION

During the initial stages of mined land succession, reduced plant species diversity is often the result of the loss of seed and bud banks (dormant meristems from which vegetation may resprout) (Groninger et al. 2007). Although the original topsoil was typically buried during pre-SMCRA mining, some of the disproportionately large diameter trees that exist on these sites today are evidence of individuals that sprouted from stump and root remnants that survived the mining disturbance (Croxton 1928, Riley 1975, Rodrigue 2001). Wind and bird-disseminated species capable of withstanding varied and often harsh site conditions are the first to colonize minesoils (Burger and Zipper 2009). Colonization by less stress-tolerant species occurs as soils become conditioned and nitrogen and organic material accumulate (Burger and Zipper 2009). Colonization by heavy-seeded species, primarily via animal dispersal, can be extremely limited within the interior of mine sites (Showalter and Burger 2006).

On pre-SMCRA mines which have reverted to forest, stand composition tends to be dominated by pioneer species including: red maple (*Acer rubum*), yellow poplar (*Liriodendron tulipifera*), bigtooth aspen (*Populus grandidentata*), sourwood (*Oxydendrum arboretum*), eastern cottonwood (*Populus deltoides*), American elm (*Ulmus americana*), American sycamore (*Platanus occidentalis*), birch (*Betula* spp.), and boxelder (*Acer negundo*) (Skousen et al. 1994, Zeleznik and Skousen 1996, Rodrigue 2001). On a 45-year-old, unreclaimed surface mine in western Pennsylvania, Brenner et al. (1984) found that oak and hickory importance was only 5% whereas red maple importance was 60%. Ashby (1984) described two potential trajectories for minelands in the Midwest: a xeric forest type, likely to develop from acidic minesoils with shales as a major constituent, and a mesophytic forest type, developing from more fertile minesoils in areas of moisture collecting topography.

Studies of pre-SMCRA mines that were planted with trees have found productive forests in which the hardwood and pine species that were planted dominate the overstory (Zeleznik and

Skousen 1996, Rodrigue 2001, Rodrigue and Burger 2002). Rodrigue and Burger (2002) found that site indices for pre-SMCRA reforested sites (pine and hardwood plantings) in the Midwest were similar to site indices of reference forest. However, they found more variability in the site indices of their Eastern sites in comparison to reference forest. Red maple, black cherry, green ash, and sycamore were frequent subcanopy invaders. On 46-year-old mine sites in Ohio that were planted to trees, maple and elm were the most abundant volunteer tree species (Zeleznik and Skousen 1996).

Vegetation development on pre-SMCRA minelands is influenced by a combination of factors including: the edaphic properties of minesoils (Game et al. 1982, Ashby 1984, Skousen et al. 1994, Rodrigue 2001), the composition and structure of the adjacent, undisturbed plant community (Riley 1975, Skousen et al. 2006), the shape and size of the mining disturbance (Hardt and Forman 1989), post-mining disturbance processes (i.e. erosion), and stochastic variability in the introduction of species and in the distribution of biological legacies (i.e. downed woody debris) (Game et al. 1982, Walker and Chapin 1987, Skousen et al. 1994, Rodrigue 2001). As a consequence of variability in site conditions, vegetation development is often patchy (Game et al. 1982, Skousen et al. 1994, Rodrigue 2001). The irregular composition of minesoils and the resulting fine-scale heterogeneity in spoil acidity and rock and shale content is often a primary factor in producing spatial and compositional heterogeneity in vegetation development (Skousen et al. 1994, Rodrigue 2001). Skousen et al. (1994) investigated natural revegetation on abandoned mine land (AML) sites (pre-SMCRA unreclaimed surface mines) in West Virginia. Soil pH and acidity were highly influential in the formation of vegetation communities. On soils with a high pH and low acidity, herbaceous species were predominant, precluding tree establishment. On soils with a low pH and high acidity, tree species and acid-tolerant grasses were able to establish in favorable microsites created by nurse-logs, brush piles, or small ridges and depressions with elevated levels of moisture and seed capture (Skousen et al. 1994). This type of colonization of dispersed microsites fits with the model of patch succession observed by Game et al. (1982) in a study of vegetation dynamics on small (3.7-10.4 hectares), unreclaimed surface mines in Missouri (Skousen et al. 1994). Game et al. (1982) found that invading species established within scattered microsites which then expanded and eventually coalesced. Colonizing vegetation may also spread inward from the forest-mine edge through the amelioration of growing conditions initially in association with the forest edge environment and then subsequently by the progression of pioneer species into the interior of the site (Rodrigue 2001).

Skousen et al. (2006) studied differences in vegetation and soils between outslope and flat top locations and adjacent reference forest at 20-year-old, reclaimed mountaintop mines in West Virginia. Flat top locations had been seeded with aggressive grasses and legumes whereas outslopes had received fairly little seeding. Consequently, tree coverage was significantly higher on outslopes than flat top locations. Red maple, black locust (*Robinia pseudo-acacia*), sourwood, autumn olive, black birch (*Betula lenta*), and yellow poplar were common on outslopes and red maple and black locust were the primary tree species on flat top locations. Outslope soils generally consisted of native soils and weathered spoil materials, whereas flat top soils were composed of a larger proportion of unweathered materials. Flat top soils had a higher pH and were thinner and more compacted than outslope soils. In some places along flat top locations, soil depth extended only 5-10 cm before the underlying rock was reached. The thick and loose condition of outslope soils in combination with the absence of an herbaceous component produced stands that structurally and compositionally resembled forest on undisturbed sites. On 46-year-old mine sites in Ohio, Zeleznik and Skousen (1996) did not find differences in the bulk density of leveled minesoils, unleveled minesoils, and undisturbed soils, possibly indicating that compaction from leveling was never particularly severe and/or that bulk density had been reduced over time.

On many post-SMCRA mines where minesoils are severely compacted and competition from reclamation groundcovers inhibit native species colonization, revegetation towards mid- and late-successional hardwood forest would be possible only after a period of several hundred years (Johnson and Skousen 1995, Wade 1989, Burger and Zipper 2009). Compaction impedes germination, water infiltration, and the rooting of colonizing plants (Holl 2002, Bosworth 2003, Groninger et al. 2007, Emerson et al. 2009, Skousen et al. 2009). Minesoils with poor physical and chemical properties are characterized by higher mineral content, lower organic matter and nutrient content, higher rock fragment content, reduced water retention capacity, and lower porosity (Thurman and Sencindiver 1986, Johnson and Skousen 1995, Williams 2003, Burger and Zipper 2009). Minesoils may be more acidic than native soils or they may contain more alkaline, unweathered sandstone or shale materials (Emerson et al. 2009). On sites with alkaline soils, tree growth may be relatively slow and colonization by trees may also be prolonged

(Johnson and Skousen 1995, Emerson et al. 2009). With the loss of native soils during excavation, the resulting minesoils are to varying degrees unweathered, a condition dependent on proportions of unweathered gray sandstone, shales, and weathered brown sandstone (Emerson et al. 2009, Skousen et al. 2009). Unweathered gray materials are buried deeper in the geologic profile and consequently develop as soil horizons over a longer period of time (Skousen et al. 2009). Even with a "substitute" soil layer of unweathered materials, soil horizons with properties suitable for tree growth may develop within three years (Sencindiver and Ammons 2000, Emerson et al. 2009). On pre-SMCRA mines, hydric minesoils may develop in association with areas that are poorly drained such as sloughs that follow the base of the highwall (Atkinson et al. 1998). Often along mine benches, poor drainage is the result of underlying bedrock or compacted minesoils (Atkinson et al. 1998).

AVIAN ASSEMBLAGES ON SURFACE MINES

Surface mined lands may support diverse avian communities (Karr 1968, Brenner and Kelly 1981, Wood et al. 2001, Lacki et al. 2004, Bulluck and Buehler 2006, Patton 2007, Carrozzino 2009). Studies of nest success on reclaimed surface mines have focused on grassland songbirds and evidence has been mixed as to whether these represent source or sink populations (Wray et al. 1982, Ammer 2003, Monroe and Ritchison 2005). Little information exists on the reproductive success of early successional and mature forest songbirds nesting on surface mines.

Recent research has illustrated the significant effect surface mining has on mature forest songbirds, Cerulean Warblers (*Dendroica cerulea*) in particular. Cerulean Warblers preferentially select ridgetop habitat for breeding and, intuitively, mountaintop mining represents significant habitat loss for this species (Bosworth 2003, Weakland and Wood 2005, Wood et al. 2006). Additionally, Wood et al. (2006) found that Cerulean Warbler abundance decreased significantly in proximity to mine edge. This pattern of avoidance is particularly strong within 340 meters of mine edge, but also holds to distances of 900 meters (Bosworth 2003, Weakland and Wood 2005, Wood et al. 2006). Additionally, Weakland and Wood (2005) found that Cerulean Warbler territory density was much reduced in mining fragmented forests (0.7 territories/10 ha) versus intact forest (4.6 territories/10 ha). Lacki et al. (2004) surveyed riparian forest before and after surface mining activity and subsequent reclamation. In adjoining, unmined forest, they observed post-mining declines in Ovenbird and Hooded Warbler (*Wilsonia* *citrina*) abundance and the absence of Worm-eating and Cerulean Warblers from all post-mining surveys.

Some research indicates that interior-edge species such as Cerulean Warbler may increase in abundance in association with narrow-cut contour mining. In the Cumberland Mountains of Tennessee, Yahner and Howell (1975) surveyed breeding avian assemblages associated with a 20-year-old pre-SMCRA contour mine and within adjacent, undisturbed forest. Within the forest margin extending 15 m outward from the mine, Cerulean Warbler density was 20.1 detections/ha in comparison to 10.6 detections/ha within adjacent, undisturbed forest. In eastern Kentucky, Crawford et al. (1978) investigated the short-term effects of contour mining on breeding bird assemblages in adjacent forest. His survey route was 50 m downslope of the ridgeline, on the backside of which lay the mining operation. Cerulean Warblers were absent during pre-mining surveys (1975), but were detected at densities of 12.7 males/ha during the first year of mining (1976) and 18.9 males/ha by the completion of mining (1978). Additionally, Buehler et al. (2006) observed territorial Cerulean Warbler males within young forest (<30 years old) that had developed on unreclaimed contour mines in Tennessee.

The seeding of exotic grasses and legumes and the level of compaction associated with post-SMCRA reclamation likely produces poor-quality early successional habitat (Rosenberg and Dettmers 2004). Woody plant invasion on mountaintop mines is generally sparse and relegated to forest edges (Handel 2003). On mountaintop mines and in adjacent forest in southern West Virginia, Wood et al. (2001) found higher species richness and total abundance for bird communities in shrub-pole mine habitat than for grassland mine habitat, intact forest, and fragmented forest.

Bulluck and Buehler (2008) studied Golden-winged Warblers breeding on reclaimed contour mines (14-26 years old) in the Cumberland Mountains of Tennessee. Over three seasons, the daily survival rate for Golden-winged Warbler nests ranged from 0.9641-0.9834. Chapman et al. (1978) surveyed bird communities on contour mines in southern West Virginia. Bird diversity was highest on mines that were 8-10 years post-reclamation, and several shrubland bird species were absent from mines more than 12 years post-reclamation. Absent species included: Golden-winged Warbler, Yellow-breasted Chat (*Icteria virens*), Prairie Warbler (*Dendroica discolor*), and Common Yellowthroat (*Geothlypis trichas*). They found higher species diversity in adjacent unmined forest than within any of their mined sites.

Few studies have investigated avian assemblages on forested minelands. In southwest Virginia, Carrozzino (2009) surveyed bird communities on 5-12 year old reclaimed mines, 13-25 year old reclaimed mines, 2-18 year old clearcuts, managed pastureland, 30-60 year old pre-SMCRA mines, and reference forest. She found the highest species richness on pre-SMCRA mines because these sites were primarily edge-dominated shrub-pole habitats. Karr (1968) studied avian assemblages along a chronosequence of strip-mined lands in Illinois. Included among his study sites was a 6.2 ha section of bottomland forest on land that had been strip-mined approximately 60 years prior to the study. Several species, including Kentucky Warbler (*Oporornis formosus*), Cerulean Warbler, Northern Parula (*Parula americana*), American Redstart (*Setophaga ruticilla*), Acadian Flycatcher (*Empidomax virescens*), Blue-gray Gnatcatcher (*Polioptila caerulea*), and Prothonotary Warbler (*Protonotaria citrea*), established territories on the mineland forest plot but were absent from a nearby tract of undisturbed forest.

STAND STRUCTURE AND COMPOSITION WITHIN SECONDARY FORESTS

Mature eastern forests are primarily second-growth stands initiated during extensive clearcutting during the late 1800s and early 1900s (Lorimer 1989). Canopy gap dynamics within secondary forests differ from those functioning within stands of old-growth forest (Clebsch and Busing 1989, Lorimer 1989, Weishampel et al. 2007). In secondary forests, rapid lateral crown expansion often limits the duration in which the canopy remains open (Lorimer 1989). In uneven-aged forests, older trees are generally limited in their ability to initiate rapid growth in response to increased growing space (Hart and Grissino-Mayer 2008). Additionally, the larger gap sizes that are characteristic of uneven-aged forests may preclude canopy closure via lateral crown expansion (Hart and Grissino-Mayer 2008). Consequently, subcanopy individuals may capture this growing space via height growth (Hart and Grissino-Mayer 2008). Dominants within secondary forests are also younger than those within old-growth forests and therefore less frequent gap makers (Lorimer 1989).

Yellow poplar is a shade-intolerant, rapidly growing species capable of forming monospecific canopies on sites that have experienced significant anthropogenic disturbance (Lafon 2004). More extensively, it is distributed sporadically within mesophytic forests where it captures large canopy gaps primarily on moist sites (i.e. coves and north-facing slopes) (Mudrick et al. 1994, Lafon 2004). Clebsch and Busing (1989) studied gap dynamics within a stand of mixed-mesophytic old growth forest and within an adjacent, 67-year-old, yellow poplar dominated stand that had established upon an abandoned agricultural field. Gaps within the yellow poplar stand were narrowly distributed within <100 m^2 size classes, whereas the old growth stand contained gaps in a range of size classes including some exceeding 280 m^2. Within the yellow poplar stand, tree crowns were generally non-overlapping, but restricted in their breadth by the uniform distribution of adjacent crowns. As a result, individual tree mortality produced small openings that were evenly distributed across the stand.

In western Virginia, Lafon (2004) investigated stand dynamics within forest that had established on former agricultural land abandoned in the late 1940s. On both mesic and relatively xeric sites within this stand, the canopy was dominated by a single cohort of yellow poplar. Pole-sized white ash (*Fraxinus americana*) and red maple were abundant and, according to the author, may eventually form the dominant canopy species on the relatively xeric sites within this stand. However, Lafon (2004) suggests that yellow poplar forest may persist on the majority of the site through capture of multiple-treefall gaps created by ice storms. He suggests that, at this particular stage of development, large treefall gaps are necessary for the initiation of additional cohorts.

In pine-hemlock-northern hardwood forest, Weishampel et al. (2007) studied the canopy structure of stands initiated under varied disturbance intensities from a 1938 hurricane and subsequent salvage logging operations. They used Light Detection and Ranging (LiDAR) remote sensing data to calculate canopy top height (CH), an index of canopy height diversity (CD), and an index of canopy evenness (CE). The latter two are indices of vertical stratification of canopy layers from the forest floor. The more severely disturbed stands had significantly lower CD indices, shorter canopies by less than one meter, and higher levels of spatial autocorrelation for CH. According to the authors, these findings indicate that the canopies of the more severely disturbed stands have reduced horizontal and vertical structural heterogeneity relative to undisturbed and moderately disturbed stands.

Pronounced vertical stratification is often found within stands in which species composition is characterized by significant differences in interspecific growth rates and shade tolerance (Guldin and Lorimer 1985). Additionally, species with lower shade tolerance tend to have less densely foliated canopies. Consequently, their presence in the canopy may allow greater persistence of tolerant species (i.e. red maple) within the subcanopy (Lorimer and Krug 1983).

Disturbance regimes in eastern deciduous forests have seen significant alteration since European settlement. Within pre-settlement, mixed hardwood forests, anthropogenic and natural fire contributed to the establishment and maintenance of oak-dominated forests (Nowacki and Abrams 2008). Within post-settlement mixed hardwood forests, timber cutting for fuelwood and lumber and the slash fires that accompanied harvests formed the primary disturbance regime until the advent of fire suppression in the 1920s and 1930s (Nowacki and Abrams 2008). The frequency of fires and the extent of logging was such that oaks came to further dominate mixed hardwood forests (Nowacki and Abrams 2008). In the absence of fire and with the abatement of land clearing, oak dominance has waned as shade tolerant species are capable of overtopping oak regeneration at all but the more xeric sites (Nowacki and Abrams 2008). Consequently, species composition is shifting towards mesophytic species (Nowacki and Abrams 2008).

Relative to oak-dominated forests, the canopy closure and high leaf area associated with stands dominated by mesophytic hardwoods, typically results in higher relative humidity and reduced air movement and radiation within the subcanopy (Nauertz et al. 2004, Nowacki and Abrams 2008). Decomposition of leaf litter and downed woody debris is also more rapid within the cool, moist microclimate existing within forests dominated by mesophytic species (Nowacki and Abrams 2008). In addition to this microclimatic influence, the structural properties and decay rates of the leaves and wood of oaks and hickories differ from those of mesophytic species and therefore contribute to differences in decomposition dynamics within these two forest types (Nowacki and Abrams 2008). As a result of high tannin and lignin concentrations and low nitrogen and calcium concentrations, oak leaves are less palatable to detritivores and consequently decompose slower than leaves of many mesophytic species (Fox et al. 2010). Mudrick et al. (1994) found that yellow poplar and red maple leaf litter decomposed significantly faster than chestnut oak litter.

In Ohio and Indiana, Fox et al. (2010) compared ground and shrub-nesting bird assemblages within oak dominated forests and forests dominated by sugar maple and successional species (i.e. yellow poplar, black cherry (*Prunus serotina*), and slippery elm (*Ulmus fulva*). They found that leaf litter depth was significantly reduced in the maple dominated forests relative to oak dominated forests. Ground nesting species including Ovenbird, Black-and-white Warbler, and Worm-eating Warbler were absent in maple dominated forests whereas they were abundant in oak dominated forests. Eastern Towhee (*Pipilo erythrophthalmus*), Indigo Bunting, and Northern Cardinal (*Cardinalis cardinalis*) were more abundant in maple dominated forests. In a series of leaf litter decomposition experiments, they found that oak leaves decomposed significantly slower than successional species including yellow poplar. Fox et al. (2010) suggest that decreased litter depth and, consequently, a scarcity of high quality nest sites and foraging habitat contributed to the absence of ground-nesting songbirds within maple dominated forests.

In part by virtue of their specialized foraging strategies and nest architecture, forest songbirds may show preferences for particular tree species as foraging and nest substrates on the basis of characteristic foliage and branch structure (Holmes and Robinson 1981, Holmes and Robinson 1988). Several studies indicate that these preferences are generally for oak and hickory species and not mesophytic species, such as red maple and yellow poplar (Gabbe et al. 2002, George 2009, Newell 2010). In bottomland forest in Illinois, insectivorous forest birds showed strong foraging preferences for kingnut (*Carya laciniosa*) and bitternut hickories (*Carya cordiformis*) (Gabbe et al. 2002). Cerulean Warblers selectively foraged in kingnut hickory and avoided red maple. Some forest songbirds may not be adapted for capturing arthropods on red maple foliage due to the length of its petioles and the size of its leaves (Franzreb 1978, Holmes and Robinson 1981, Holmes and Schultz 1988, Rodewald and Abrams 2002). Relative to other hardwood species, some research has illustrated that oaks host a greater diversity of lepidopteran species (Summerville et al. 2003). In a study within oak-dominated forests and maple-dominated forests in Ohio, lepidopteran assemblages were strongly organized by the dominant canopy species (Summerville and Crist 2008). In a comparison of bird communities within maple and oak dominated stands, Rodewald and Abrams (2002) found reduced abundance for the bark gleaning guild in maple dominated stands across spring, fall, and winter. They attribute this pattern to the availability of acorns in oak-dominated forests. In Ohio, Cerulean Warblers, Scarlet Tanagers (*Piranga olivacea)*, Blue-gray Gnatcatchers, and Eastern Wood-Pewees (*Contopus virens*) placed nests in white oak (*Quercus alba*) in disproportion to its availability and avoided placing nests in red maple (Newell 2010).

AVIAN ASSEMBLAGES AND CANOPY GAPS

Lertzman et al. (1996) separate canopy openings into two categories: “developmental gaps” and “edaphic gaps”. Edaphic gaps are the product of soil, topographic, or geomorphic features including streams, boulders, cliffs, and standing water. They represent “persistent open space” owing to the absence or arrested state of successional dynamics. Developmental gaps result from the mortality of one or several trees and are closed through lateral crown expansion or vertical height growth of interior gap vegetation.

The natural mortality of one or several trees is a primary component of disturbance regimes functioning within eastern deciduous forests and consequently contributes much of the habitat hetereogeneity within these forests (Runkle 1982). Greenberg and Lanham (2001) found that total breeding bird abundance was higher within hurricane-created gaps compared to adjacent, closed-canopy forest. Species richness was also significantly higher in gaps. Among Neotropical migrants, Indigo Bunting (*Passerina cyanea*), Hooded Warbler, Blue-headed Vireo, and Worm-eating Warbler were significantly more abundant in gaps. Only Ovenbird was significantly more abundant in closed-canopy forest. Red-eyed Vireo and Scarlet Tanager were detected in similar densities within gaps and closed-canopy forest.

Research conducted during spring and fall migration has found a positive correlation between migrant abundance within treefall gaps and elevated levels of both fruit and arthropod abundance (Blake and Hoppes 1986, Martin and Karr 1986). During the post-breeding period, adults and juveniles of a number of forest-interior species including Wood Thrush, Worm-eating Warblers, Ovenbirds, and Scarlet Tanagers demonstrate selective use of dense, shrub layer vegetation occurring within a variety of harvest treatments and edge habitats (Anders et al. 1998, Vega Rivera et al. 1998, Pagen et al. 2000, Dellinger 2007, McDermott and Wood 2010, Vitz and Rodewald 2010). During the breeding season, the nature of interior-edge species‟ association with openings in the canopy is not thoroughly understood. Openings in the forest canopy result in increased light reaching the understory and residual canopy, which in turn results in elevated primary productivity within gaps (Blake and Hoppes 1986, Martin and Karr 1986, Smith and Dallman 1996, Gorham et al. 2002). Canopy gaps may provide some forest songbirds with dense foliage in the understory and peripheral canopy in which to conceal nests (Greenberg and Lanham 2001). Increased foliage density and vertical complexity within gaps may also increase foraging efficiency among leaf-gleaning species (Blake and Hoppes 1986, Martin and Karr 1986, Smith and Dallman 1996, Gorham et al. 2002). The opening of the canopy may result in conditions (a warmer understory microenvironment and elevated primary productivity) that favor the growth of flying insect populations (Blake and Hoppes 1986, Smith and Dallman 1996, Gorham et al. 2002, George 2009). In studies of partial harvesting, Eastern Wood Pewees, a flycatching species, generally respond positively to the opening of the canopy (Annand and Thompson 1997, Newell 2010). In addition, Smith and Dallman (1996) suggest that Black-throated Green Warblers (*Dendroica virens*) may use canopy gaps as nodes of territory advertisement and delineation due to the acoustic qualities of gaps and their structural contrast relative to the surrounding forest.

Several studies have investigated bird use of harvested gaps in relation to fruit and arthropod abundance. In general, the relationship between bird use and resource availability within harvested gaps is unclear, possibly as an outcome of seasonal variation in resources and high arthropod diversity (Moorman and Guynn 2001, Kilgo 2005, Bowen et al. 2007). In research conducted in bottomland forest in South Carolina, arthropod abundance and Hooded Warbler attack rates, an indication of foraging efficiency, were both higher >100 m from group selection gaps than along gap edges and at intermediate distances (Kilgo 2005). Kilgo (2005) suggests that arthropod abundance during the breeding season may not have been limiting across the extent of the study area.

While a number of bird community studies have focused on the effects of induced edges, relatively few have investigated the effects of inherent or persistent edges resulting from soil, topographic, or geomorphic features (Matheson and Larson 1998). Along the Niagara Escarpment in Ontario, Matheson and Larson (1998) investigated differences in forest bird assemblages within four cliff associated habitats (plateau, cliff edge, cliff face, and talus slope). Patterns of species richness differed between their sites; however, they consistently observed the lowest species richness in plateau woodlands and higher species richness along cliff edges and talus slopes.

MOISTURE GRADIENT INFLUENCES ON AVIAN ASSEMBLAGES

Appalachian oak forest varies structurally and compositionally along a soil moisture and fertility gradient, characteristics which are primarily determined by topographic and geologic factors (McEvoy et al. 1980). Previous studies have shown patterns in bird density and occurrence along moisture gradients (Bertin 1977, Swift et al. 1984, Petit et al. 1985, McShea et al. 1995, Murray and Stauffer 1995).

Petit et al. (1985) quantified variation in breeding bird assemblages along a relative humidity gradient. Bird species richness and understory foliage density were positively correlated with relative humidity; overall abundance was not. Several species, including Red-bellied Woodpecker (*Melanerpes carolinus*), Hairy Woodpecker (*Picoides villosus*), Wood Thrush, and Ovenbird, were more abundant within plots with higher relative humidity. In forested wetlands in Massachusetts, overall bird density was positively correlated with the small shrub density, coverage of standing water, and depth of soil muck. Species richness was positively correlated with small shrub density and depth of soil muck (Swift et al. 1984). From surveys conducted along a stream to upland forest gradient, Murray and Stauffer (1995) placed Scarlet Tanager, Red-eyed Vireo, Ovenbird, and Blue-headed Vireo within a xeric upland forest assemblage and Wood Thrush and Black-throated Green Warbler within a mesic forest assemblage.

Dettmers and Bart (1999) developed and evaluated spatial, microhabitat-based models for eastern forest songbirds. Models for Acadian Flycatcher and Worm-eating Warbler identified these species as having relatively restrictive microhabitat preferences for concave, moisture collecting topography (e.g. ravines and stream bottoms). Hooded Warbler, Eastern-Wood Pewee, Cerulean Warbler, and Scarlet Tanager were associated with convex terrain (e.g. adjacent to or on hilltops and ridges) and drier moisture conditions. Within this group, species‟ microhabitat preferences varied according to slope position and steepness and moisture conditions. Ovenbird, Red-eyed Vireo, and Wood Thrush were associated with a range of microhabitat conditions and were therefore dispersed relatively evenly across the study area.

In a habitat selection study of sympatric populations of Wood Thrush and Veery (*Catharus fuscescens*), Bertin (1977) found that moisture regime, as measured by visible soil characteristics, accounted for 76.0% and 78.4%, respectively, of the variation in territory characteristics between occupied and unoccupied habitat. Both species frequently established territories along streams, seeps, and springs. Bertin (1977) suggests that thrushes as well as other ground-nesting and foraging species may use relative humidity and temperature as proximate cues to habitat suitability.

In research conducted in northwestern Virginia, McShea et al. (1995) found that Kentucky Warblers were significantly associated with red maple dominated forest and rarely established territories in oak-hickory forest. McShea et al. (1995) suggest that the preference for the red maple forest type may reflect the increased moisture associated with these sites. The density of Kentucky Warbler prey, invertebrates taken from leaf litter and gleaned from foliage, may be greater in these moist environments (McShea et al. 1995).

RED-EYED VIREO BREEDING ECOLOGY

Breeding habitat in which Red-eyed Vireos are generally most abundant is characterized by high levels of canopy closure, basal area, and vertical stratification (James 1971, Williamson 1971, Stauffer and Best 1980, Yahner 1986, Marshall and Cooper 2004). However, use extends to habitats with low levels of canopy cover and complexity: city parks, residential areas, and citrus groves (Graber et al. 1985, Mills 1989, Cimprich et al. 2000). Conclusions from research aimed at quantifying Red-eyed Vireo area-sensitivity and edge avoidance have been inconsistent among varied forest landscapes (Freemark and Collins 1992, Villard 1998, Burke and Nol 2000, Dunford et al. 2002).

In a study of Blue-headed (*Vireo solitaries*) and Red-eyed Vireo (*Vireo olivaceus*) habitat use in the southern Appalachians, Hudman and Chandler (2002) found that white oaks, conifers, and ericaceous shrubs were more abundant in Blue-headed Vireo territories whereas red oak and red maple were more abundant in Red-eyed Vireo territories.

Marshall and Cooper (2004) found that Red-eyed Vireo territory size was inversely correlated to the volume of foliage within a territory. Foliage volume was positively associated with caterpillar density during the nestling stage. Additionally, the timing of the nestling stage corresponded to the lowest levels of caterpillar and arthropod abundance over the course of the breeding season. The authors suggest that Red-eyed Vireos use foliage density as a structural cue in determining the size of a three-dimensional territory that will contain sufficient resources for successful brood rearing. Consequently, foliage density has implications for all aspects of vireo breeding ecology, including nest survival (Marshall and Cooper 2004).

Red-eyed Vireos show a high degree of plasticity in terms of the heights at which nests are placed (Martin 1988). Several studies (Lawrence 1953, Southern 1958, Rice 1974, Graber et al. 1985) observed mean nest heights within a range of 2.5-4.3 m. Other studies have reported mean nest heights that reflect greater variability; 10.7 ± 5.8 m (mean ± SD) for a study in New Hampshire (Robinson 1981) and 7.0 ± 5.7 m (mean ± SD) for a study in the central Appalachians (DeCecco et al. 2000). Differences in reported nest heights may also reflect the difficulty in locating nests placed higher in the canopy (Rodewald 2004). Rodewald (2004) found that Red-eyed Vireo nests located by luck had a mean height of 3.5 ± 0.6 m (mean ± SE), whereas nests located via parental behavior had a mean height of 10.1 ± 1.1 m. Martin (1988) suggests that differences in reported nest heights for Red-eyed Vireos may be adaptive in the sense that they are ultimately related to minimizing vertical overlap with conspecifics.

In a study of the breeding ecology of Red-eyed Vireos in Pennsylvania, characteristics of nest patch vegetation were not indicative of nest fate (Siepielski et al. 2001). The authors suggest that forest cover on the landscape scale may be more influential in terms of structuring predator assemblages and affecting reproductive success (Donovan et al. 1997, Siepielski et al. 2001, Rodewald 2002). They also hypothesize that vireo nest site selection may not be adaptive in the presence of predator assemblages unique to landscapes with significant anthropogenic disturbance. Rodewald (2002) found 2-2.6 times lower daily nest survival rates for ground and mid-canopy nesting species in forested landscapes fragmented by agriculture relative to landscapes fragmented by silviculture. She found lower densities of corvids and squirrels within silviculture-fragmented landscapes, indicating that a higher density and diversity of predators are associated with the enduring non-forest habitat within agriculture-fragmented landscapes (Rodewald and Yahner 2001, Rodewald 2002). Within contiguous forest fragmented only by narrow forest roads and low density housing, Gale et al. (1997) found no difference in the number of Worm-eating Warbler pairs fledging host and Brown-headed Cowbird young between study plots located in small and large patches.

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Chapter 2

AVIAN ASSEMBLAGES AND RED-EYED VIREO NEST SURVIVAL WITHIN MINELAND FOREST

ABSTRACT

Given that avian use of mineland forest has gone largely unstudied, I initiated research to examine patterns of bird species composition within pre-SMCRA mineland forest and reference (unmined) forest in relation to habitat structure and composition. I also contrasted nest survival of Red-eyed Vireos (*Vireo olivaceus*) breeding within mineland and reference forest. Line transect surveys and nest monitoring were done in New River Gorge National River in 2011. Ordination of avian assemblages using non-metric dimensional scaling (NMDS) showed clear discrimination between mineland and reference assemblages. Linear and surface fitting of habitat variables showed strong correlations between the ordination and groundcover gradients, but generally non-significant relationships for gradients describing forest structure. Mineland assemblages were associated with lower levels of litter cover and depth and also had lower abundance of Ovenbirds (*Seiurus aurocapillus*). Compared to reference assemblages, mineland assemblage structure was more variable suggesting a wider gradient in forest structure on minelands. Within mineland forest, edaphic conditions/disturbances were likely influential in creating spatial heterogeneity in forest structure. However, nest survival for Red-eyed Vireos was similar within mineland and reference forest and overstory cover and vertical foliage density had minimal effect on nest survival. Classification tree modeling using forest type as the response variable indicated that reference nest sites were characterized by greater subcanopy cover and higher densities of trees in the 8-23 cm size class than mineland nest sites. Results from this study indicate that forest bird assemblages may be structured differently in the absence of heavy-seeded tree species on minelands.

INTRODUCTION

Post-mining land use has varied greatly since surface mining was first introduced. Prior to the Surface Mining Control and Reclamation Act (SMCRA) in 1977, coal surface mining in Appalachia generally followed the “shoot and shove” method for extraction resulting in an exposed highwall, bench-land along its face, and an outslope comprised of loose spoil that had been pushed below. SMCRA required that mine operators “backfill, compact, and grade in order to restore the approximate original contour (AOC) of the land with all highwalls, spoil piles, and depressions eliminated" (Office of Surface Mining Reclamation and Enforcement 2008). Mine companies, dozer operators, and regulatory agencies responded with expectations of a uniformly smooth post-mining landscape. In the decades that followed, surface mines were generally reclaimed to pastureland dominated by exotics and stress-tolerant species (Burger et al. 2002, Burger et al. 2005, Showalter and Burger 2006). The traditional post-SMCRA post-mining environment is one in which minesoils are severely compacted from immoderate grading, unweathered materials contribute heavily to the growth medium, and aggressive groundcovers inhibit native species colonization (Burger et al. 2002). Under these conditions, forest development is arrested and, through their persistence as degraded, seral communities, surface mines contribute to forest fragmentation (Wade 1989, Johnson and Skousen 1995, Burger and Zipper 2009). Within landscapes that are fragmented by traditionally reclaimed surface mines, forest patches are smaller and forest cover on the landscape scale is reduced (Wickham et al. 2007). As a result, forest songbirds that require large, continuous blocks of forest are negatively affected (Wood et al. 2006).

The Appalachian Regional Reforestation Initiative (ARRI) was established by the Office of Surface Mining in 2004 with the primary objective of encouraging surface mine reforestation through a method termed the Forestry Reclamation Approach (FRA). The Forestry Reclamation Approach is a series of field-tested guidelines directed at advancing forest succession on minelands (Burger and Fannon 2009, Skousen et al. 2009). Components of this approach include creating an uncompacted growth medium in which hardwoods are capable of exhibiting normal survival and growth and seeding non-aggressive groundcovers which do not completely inhibit forest plant invasion (Burger et al. 2005).

Pre-SMCRA abandoned minelands contain areas of uncompacted minesoils on which hardwood forest has developed in the absence of aggressive groundcovers. The pre-SMCRA minelands used in this study were not planted and consequently are dominated by pioneer species. Despite potential differences in tree species composition, study of the relationship between habitat structure and avian assemblage composition within pre-SMCRA mineland forest could provide insight into the species assemblages that future reforestation efforts might yield. Study of the reproductive success of forest songbirds within pre-SMCRA mineland forest may provide some indication as to whether this habitat is capable of sustaining breeding populations of canopy-nesting songbirds.

The objectives of this study were to 1) examine patterns in avian assemblage structure within mined and reference forest and to link the avian assemblage response to variables describing habitat structure and composition, and 2) contrast nest survival of Red-eyed Vireos (*Vireo olivaceus*) breeding within mineland forest and unmined oak-hickory forest.

METHODS

Study area

The study area, located atop the New River Plateau within New River Gorge National River in southern West Virginia, included pre-SMCRA abandoned minelands and adjacent unmined sites (hereafter reference forest) (Appendix A). Forest cover is largely unfragmented and of the oak-hickory type. White (*Quercus alba*), chestnut (*Quercus prinus*), scarlet (*Quercus coccinea*), and black oak (*Quercus velutina*) comprised the predominant canopy species. Yellow poplar (*Liriodendron tulipifera*), red maple (*Acer rubrum*), and black birch (*Betula lenta*) were predominant within minelands. Elevations within the study area are 550-670 m. Soils primarily consist of silt loams and are stony, shallow, and moderately well-drained (USDA 1975).

I selected pre-SMCRA minelands using the following criteria: loose-dumped spoil was placed in ridges and mounds atop benches and also within outslopes, the mineland stand was continuous, mature hardwood forest, the mineland canopy obscured the edge once created by the highwall, and recent off-road vehicle use was not evident. I also chose minelands that were relatively wide (80-100 m wide on average) in order for avian use to bear stronger relation to habitat structure within the mineland patch. The relative expansiveness of pre-SMCRA minelands atop the New River Plateau is an outcome of the moderate terrain that allowed for deeper coal extraction.

Based on aerial photos taken in 1945 (Appendix B), mineland stands were 60-65 years old. Reference stands primarily originated during extensive clearcutting at the turn of the century (Brooks 1910), but also included stands that were closer in age to mineland stands. The latter were abandoned homesteads and a former mining company town that existed as early successional vegetation at the time the 1945 photos were taken.

Avian assemblage structure and habitat relationships

Across the three study sites, I established 28 fixed-width line transects (14 mined and 14 reference) within four pre-SMCRA abandoned minelands and adjacent, reference forest (Fig. 1 and 2). I used fixed-width line transects because mines were configured linearly with dimensions that were generally consistent with a transect width of 50 m on each side of the line. Line transects also were preferable because in some situations they may yield more detections per unit of time (Bollinger et al. 1988, Buckland et al. 2001). Additionally, the greater survey coverage associated with line transects relative to point transects is thought to limit potential bias from evasive bird movement (Buckland et al. 2001). All transects were 250 m in length and sampled the available habitat (mineland or reference forest) within 50 m strips on each side of the line.

Transect layout was accomplished in ArcGIS using a 1-meter Digital Elevation Map (DEM) in which mineland boundaries were clearly evident. I placed the first transect within each discontinuous mineland using a random start point (0-50 m) measured from the widest point along the northern or eastern edge of each mine (Gates 1979, Buckland et al. 2004). The starting points of subsequent transects were then located 200 m from the closest point of the previous transect (Bibby et al. 1992). Mineland transects were generally located along the center of the long axis orientation of mines and bent according to mine configuration. In some instances where mine width exceeded 100 m, I oriented transects to maximize their number within the mine.

I placed reference transects to maximize their number within stands of mature, oak-hickory forest. In order to follow the general orientation of mineland transects, I attempted to orient reference transects to parallel the dominant contour of the land. I placed each transect 200 m from adjacent transects at their closest point. I also located reference transects ≥150 m from mines and ≥50 m from the rim of the New River Gorge. I established reference transects as straight lines to facilitate line navigation and accuracy in distance estimation. This contrasted with the bent line orientation of mineland transects. However, mined transects had the benefit of highwall and outslope features in orienting the observer and defining the survey strip.

Bird surveys. In 2011, I conducted transect surveys between May 16 and June 2 from a half hour after sunset to 1030, coinciding with peak singing. I surveyed each transect over a 25 minute period (Ralph et al. 1993). I flagged transects every 50 m to facilitate navigation and to ensure that equal survey effort was allotted to each section of a transect. Birds detected within 50 m of the line were recorded. On mineland transects, I recorded only individuals detected within the boundaries of the mine (e.g. from the base of the highwall to the bottom of the outslope). Sections of minelands that were <100 m wide resulted in small differences in the area of habitat surveyed relative to reference transects. Reference transects surveyed 2.5 ha, whereas the mean survey area for mineland transects was 2.21 ± 0.16 ha (mean ± SD). All analyses accounted for variation in survey area (see below).

I used a laser rangefinder to estimate the perpendicular distance to each bird detected within 50 m of the center line. Flyovers were not recorded. I plotted detections within 25 and 50 m distance bands on a map of each transect. Maps of mineland transects delineated the boundaries of outslope, bench, and highwall features. I surveyed each transect twice and used a species‟ maximum count between the two visits for analyses.

Habitat sampling. I sampled habitat structure and composition within four, 0.04 ha circular plots per transect using methods similar to Wood et al. (2001) that were modified from James and Shugart (1970) and the Breeding Bird Research Database Program (BBIRD; Martin et al. 1997). I located plots at a random distance (0-30 m) perpendicular to 50, 100, 150, and 200 m intervals along transects. Plots were established on alternating sides of the line with the initial side chosen at random. I identified all trees >8 cm diameter at breast height (dbh) to species and measured dbh. I tallied all vines that reached the canopy on measured trees and counted all snags >8 cm dbh and >8 m tall. Within each plot, I established two, 22.6 m perpendicular transects. Using an ocular tube and sighting along the tube‟s crosshairs, I estimated vertical foliage density at a total of 20 points, located 2 m apart along the perpendicular transects. I recorded the presence or absence of live foliage in the crosshairs at heights of 0.5-3 m, >3-6 m, >6-12 m, >12-18 m, >18-24 m, and >24 m. From these data, I calculated vertical foliage density as the sum of all foliage hits divided by the total number of sighting intervals (120) and then multiplied by 100. Foliage density also were collapsed into understory (0-6 m), midstory (>6-18 m), and overstory (>18 m) layers.

Additionally at each of the 20 points, I measured leaf litter depth and recorded groundcover type (<0.5 m) as bareground, forb, litter (leaf litter and downed woody debris), or woody. Within 3 m-radius subplots at the center of each 0.04 ha plot, I estimated shrub, sapling, leaf litter, and downed woody debris cover (logs and stumps >8 cm dbh and >1 m in length). Within each subplot, I also identified and counted woody vegetation 0.5-1.5 m tall and tallied saplings (>1.5 m tall and <8 cm dbh) and downed logs. In ArcGIS, I derived mean slope and solar radiation (insolation) values for each transect from a 1 meter DEM. I used solar radiation (expressed as watt hours/m^2 over the course of a year) as a site-productivity metric in place of aspect because it takes into account slope position in addition to aspect.

Nest survival of Red-eyed Vireos

Red-eyed Vireo was chosen as the focal species because they were by far the most abundant species within mineland sites and were found in similar densities within reference forest. This was important in terms of finding enough nests to reliably estimate nest survival. Additionally, Red-eyed Vireo territories are relatively compact; territory size for Red-eyed Vireos breeding on the Monongahela National Forest, West Virginia, was 0.39 ± 0.16 ha (mean ± SD) (Marshall and Cooper 2004). Thus, territory sizes and vireo density reflected use of mineland forest and placed vireos and their nests subject to structural conditions and food resources within this relatively narrow habitat.

I monitored Red-eyed Vireo nests within three reference and three mineland plots (Fig. 3). Reference plots were located >100 m from mines and were 5.4-36.4 ha in size (Fig. 5). Mineland plots were 8.8-20.8 ha. I monitored nests every 3-5 days initially and every 1-2 days as fledging approached (Martin and Geupel 1993). To determine nest fate among nests in which inactivity was consistent with reaching or approaching (within 2 days of) predicted fledge dates, I attempted to aurally or visually observe fledglings or adults feeding fledglings. A nest was considered successful if it fledged at least one nestling.

I sampled vegetation within 0.04 ha plots centered at vireo nests following methods described earlier. Additional data collection included measurements of nest height, height of the nest tree, height of the nearest tree in the dominant crown class (tree with a crown that projects above the general canopy layer), and the distance from the nest to both the bottom and top of the canopy, all of which were taken using a clinometer. I also recorded the distance to the nearest canopy gap and the type of gap (edaphic, snag, or treefall). Gaps were defined as having a long axis diameter greater than 5 m (Pickett and White 1985) with an interior maximum canopy height less than half that of the peripheral canopy. Following Lertzman et al. (1996), edaphic gaps were the product of soil, topographic, or geomorphic features.

ANALYSES

Avian assemblage structure and habitat relationships

I used non-metric multidimensional scaling (NMDS) to visualize patterns in avian assemblage structure between mineland and reference forest and to link habitat gradients to assemblage pattern. In NMDS, ordination is based upon preservation of the original rank order of between sample distances derived from a dissimilarity matrix (Clarke and Green 1988, Clarke 1993). In using rank order distances, NMDS avoids the linear distribution assumption (Clarke and Green 1988). In addition, NMDS is an unconstrained ordination technique and therefore designed for the purpose of linking patterns in assemblage structure to observed habitat gradients (Oksanen et al. 2009). Locations of assemblages in the multidimensional space are determined through multiple iterations such that stress is minimized (Clarke and Green 1988). Stress is a measure of goodness of fit between plotted and true rank order distances from the original distance matrix (Clarke and Green 1988).

NMDS was conducted using the „*vegan*" package (Oksanen et al. 2009) within Program R 2.12.1 (R Development Core Team 2010). Within the species matrix, observations were entered as detections ha^{-1} rather than raw counts and species detected on ≤2 transects were excluded (Preston and Harestad 2007, Chizinski et al. 2011). Ordination was performed using the metaMDS function and a Bray-Curtis dissimilarity matrix. Data were standardized using the Wisconsin double standardization method and square-root transformed to increase the relative importance of less abundant species. Multiple random starts (20/ordination) were performed to avoid becoming trapped in local minima (the iterative process stalling at a stress value that actually can be further reduced) (McCune and Grace 2002). NMDS was conducted in dimensions 2-6. The stress of NMDS ordinations was evaluated against their dimensionality via a screeplot to determine the appropriate dimensionality for display and statistical testing. The location of the sample as within mined (M) or reference (R) forest was overlaid and species were ordinated by their averaged weighted scores. Habitat variables were correlated to ordinations using vector and surface fitting. For each habitat variable, I used the mean value of the four replicate plots sampled along each transect. Vector fitting allowed for visual interpretation of the strength and direction of the variable-ordination relationship. The strength of the variable-ordination relationship was statistically assessed using r^2 and p-values derived from 999 permutations. Vector fitting assumes a linear relationship between the variable and the ordination. Because this is often not the case, general additive models (GAM) were used to produce surface fitting contours within the NMDS plot for visual and statistical interpretation of environmental gradients (Oksanen et al. 2009).

I used the adonis function („*vegan*" package; Oksanen et al. 2009) to test for statistical differences between mined and reference assemblages. This function performs a multivariate analysis of variance through partitioning of the sums of squares in a distance matrix in relation to a factor and using *F*-tests from permutations of the data to determine the level of statistical significance (Oksanen et al. 2009). I used the adonis method rather than analysis of similarities (ANOSIM) because the adonis method is generally considered more robust than ANOSIM (Oksanen et al. 2009). I used a Bray-Curtis dissimilarity matrix and specified that permutations (999) occurred within sites but not across sites (Oksanen et al. 2009).

A mean dissimilarity dendrogram was then used to graphically display mean between-group and within-group dissimilarity. To determine the appropriate linkage method, I calculated cophenetic correlations between the original Euclidian distance matrix and a Bray-Curtis distance matrix using both "average" and "single" linkage methods (Oksanen et al. 2009). I used the average linkage method because this linkage in combination with a Bray-Curtis metric produced a higher cophenetic correlation (0.43) than did the single linkage method (0.39).

Within the mean dissimilarity dendrogram, vertical lines indicate mean within-cluster dissimilarity; longer lines equate to lower mean dissimilarity (Oksanen et al. 2009). The horizontal line indicates mean between-group dissimilarity (Oksanen et al. 2009).

Using generalized linear modeling (GLM), I tested for differences between reference and mineland assemblages for species richness, overall abundance, and abundance within foraging, nesting, and habitat guilds. All analyses included site as a fixed effect and the area of the transect as an offset. The offset is a term in Poisson and negative binomial regression that allows one to account for differences in exposure or intensity without transforming a raw count into a rate or a density (Zuur et al. 2009). I evaluated models for overdispersion using a Poisson GLM and an associated dispersion parameter. Based on the absence of overdispersion in all models, I determined a Poisson distribution was appropriate for these analyses (Zuur et al. 2009). Statistical significance was assessed via an analysis of deviance test in which the difference in deviance approximately follows a chi-square distribution with 1 degree of freedom (Zuur et al. 2009). Foraging and nesting guilds were adapted from Ehrlich et al. (1988) and Canterbury et al. (2000) (Appendix C). I placed species into habitat guilds (closed canopy species, broken canopy species, and forest generalists) in the context of the canopy disturbance gradient that exists within the study area (contiguous forest with relatively small areas of broken canopy habitat).

Due to insufficient sample size, I did not conduct analyses for the shrub-nesting guild. One transect was an outlier (Cook‟s distance>1.0) for broken canopy and foliage gleaning species and was removed for those analyses. In addition, I tested for differences in Shannon (H') diversity between mineland and reference forest using a Gaussian distribution, again including site as a fixed effect. I evaluated all Poisson GLMs graphically to ensure that model residuals were not patterned or indicative of a lack of fit. For Gaussian linear regression modeling, I verified that model residuals met assumptions of normality and homogeneity using residual plots and Bartlett‟s test for homogeneity. Statistical significance was set at $p<0.05$ for all analyses.

Nest survival of Red-eyed Vireos

To determine support for the influence of forest type on Red-eyed Vireo nest survival, I used an information-theoretic approach and developed a set of *a priori* candidate models containing habitat and temporal covariates that I hypothesized to influence the nest survival of vireos (Table 4). I used Akaike‟s Information Criterion for small sample sizes ($\mathrm{AIC_c}$) to evaluate support for candidate models. For computation of $\mathrm{AIC_c}$, I used the effective sample size as defined in Rotella et al. (2004): n = the total number of days in which all nests were known to have survived + the number of intervals that ended in failure.

Temporal covariates. Temporal covariates included nest stage, linear julian date, and the quadratic effect of date (date + date^2). I recorded Julian date as the midpoint of the monitoring interval. Linear and quadratic effects of date were included because daily nest survival may vary in concert with patterns in predator activity and abundance across the breeding season and this trend may be non-linear (Grant et al. 2005, Peak 2007, Reidy et al. 2009). Due to insufficient monitoring intervals for which laying was recorded, nest stages were restricted to egg (laying and incubation) and brooding stages. Support for temporal covariates was evaluated prior to modeling habitat effects. Covariates from the most supported temporal model were included in all habitat models (Grant et al. 2005, Reidy et al. 2009).

Habitat covariates. Habitat covariates included forest type (mineland or reference), overstory cover (foliage density above 18 m), and vertical foliage density (foliage cover estimated across all canopy layers). Mineland and reference forest may represent a gradient of structural conditions. Consequently, patch level influences on nest survival could come from factors that vary across forest types. Higher vertical foliage density within the nest patch could function in reducing the risk of predation through concealment (total-foliage hypothesis) and/or by decreasing the search efficiency of a predator (potential-prey-site hypothesis) (Martin 1994). Research by Marshall and Cooper (2004) suggested that Red-eyed Vireos use foliage density as a structural cue in determining the size of a three-dimensional territory with sufficient resources for successful brood rearing. Consequently, foliage density has implications for all aspects of vireo breeding ecology, including nest survival (Marshall and Cooper 2004). Reduced levels of overstory cover could similarly influence nest survival. However, I included both covariates (in separate models) because a high degree of overstory cover does not necessarily indicate a multi-layered forest and their influence on nest survival may not be equivalent.

I modeled nest survival as a function of the aforementioned covariates using the logistic-exposure method (Shaffer 2004). The basis for this approach is a generalized linear model with a binomial distribution and a logit link function modified to account for the dependence of survival probability on interval length (Shaffer 2004). Because no model received overwhelming support ($w_i \geq 90\%$), I calculated model-averaged parameter estimates and their unconditional standard errors from a model set comprised of only those models for which there was the most support, models with $\Delta AIC_c < 2$ (Burnham and Anderson 2002). From these estimates, odds ratios and their 95% confidence intervals were calculated as a means of interpreting the strength of an effect on the daily survival rate (Shaffer and Thompson 2007). The percentage change in the odds of nest survival for a one-unit change in a continuous covariate is calculated by subtracting 1 from the odds ratio and multiplying this value by 100 (Allison 1999).

Daily survival rates for the covariate of interest were derived from model-averaged parameter estimates and their unconditional standard errors by holding values for other covariates at their sample means (continuous covariates), target population proportions (nest stage), or proportions giving equal weight for each level of a categorical covariate (forest type) (Shaffer and Thompson 2007). Proportions used to weigh individual levels of nest stage were based on a 26.5 day nesting period, a 15.5 day laying and incubation stage, and an 11 day nestling stage. I used 26.5 days as the average length of the nesting cycle within the study area because the mean nesting period for all nests monitored from first egg laid through fledging was 26.3 days (n=7). For continuous covariates, I estimated daily survival rates for values spanning the observed range of the covariate. Logistic exposure models were fit using PROC GENMOD (SAS Institute 2004). The global model was evaluated for goodness-of-fit using the Hosmer and

Lemeshow method (Hosmer and Lemeshow 2000) and for multicollinearity using tolerance values (Allison 1999).

I used a classification tree to describe variation in forest structure between mineland and reference nest patches. To explore variation in a response variable, classification trees use explanatory variables to recursively partition the data into subsets in which homogeneity in the response variable is maximized and the total sums of squares minimized at each split (De‟ath and Fabricius 2000, De‟ath 2002). Classification trees are a nonparametic technique; they use the rank order of explanatory variables (De‟ath and Fabricius 2000). I modeled the classification tree from data collected within 0.04 ha plots centered on each nest and restricted explanatory variables to those describing forest structure within the nest patch. I used the „*mvpart*‟ package within Program R 2.12.1 (R Development Core Team 2010). I used 45 (the nest sample size), 10-fold cross-validations, to select the smallest tree with an estimated error within 1-SE of the minimum cross-validation error (Breiman et al. 1984, De‟ath and Fabricius 2000). Cross validation error is the best measure of the predictive accuracy of the tree (De‟ath 2002). Values near one indicate a tree with poor predictive ability and a value of zero is representative of a perfect predictor (De‟ath 2002).

RESULTS

Avian assemblage structure and habitat relationships

Forest structure and avian community summary. White, chestnut, scarlet, and black oak were the predominant canopy species in reference stands (Fig. 4, Appendix D). Mineland stands were dominated by yellow poplar, red maple, and black birch. Diameter distributions for mineland and reference stands indicated significantly greater numbers of trees in the 8-18 cm size class within reference forest and a consistent pattern of slightly greater tree density in all size classes >18 cm within mineland forest (Fig. 5). Mean basal area for mineland transects was 34.1 m^2 ha^{-1} (95% CI = 31.0, 37.1) compared to 29.1 m^2 ha^{-1} (95% CI = 27.6.04, 30.5) for reference transects (Table 1). There was greater variation in mineland basal area indicating a wider gradient in canopy openness (Fig. 6).

I detected a total of 34 species, 32 on mineland transects and 27 on reference transects (Table 2). The most abundant species along both mineland and reference transects was Red-eyed Vireo, accounting for 36% and 31% of the total count within each forest type, respectively.

Avian assemblage structure. Within the screeplot of ordination stress versus dimensionality, no clear "elbow" was evident in the decline in stress. Stress for the 3-dimensional NMDS solution was 15.3 (two convergent solutions after 6 runs). Because stress values less than 20 usually indicate reliability for interpretation (Clarke 1993), samples were plotted within the first two dimensions of the 3-dimensional NMDS solution (Figure 7).

NMDS ordination showed clear discrimination between mineland and reference assemblages (Fig. 7). Ovenbird (*Seiurus aurocapillus*), Eastern Wood-Pewee (*Contopus virens*), Blue-headed Vireo (*Vireo solitarius*), Wood Thrush (*Hylocichla mustelina*), Acadian Flycatcher (*Empidonax virescens*), Great Crested Flycatcher (*Myiarchus crinitus*), and White-breasted Nuthatch (*Sitta carolinensis*) separated reference from mineland assemblages. Mineland assemblages were spread widely along NMDS axis 1 with a band of species positioned in a stretched pattern along their periphery indicating that these species were not strongly interrelated within mineland assemblages. The inconsistent pattern of relationships among species within mineland assemblages is likely a reflection of variation in habitat structure, low sample sizes, the discontinuous nature of the habitat, and the association of several of these species with isolated patches of disturbed forest.

Species which contributed little to the overall dissimilarity between forest types were Red-eyed Vireo, Scarlet Tanager, and several bark-foraging species. Bark-foraging species were located in a band across the center of the ordination and included (Fig. 7) Black-and-white Warbler (*Mniotilta varia*), Hairy Woodpecker (*Picoides villosus*), Pileated Woodpecker (*Dryocopus pileatus*), Red-bellied Woodpecker (*Melanerpes carolinus*), and Yellow-throated Vireo (*Vireo flavifrons*), the latter species also relying on foliage-gleaning.

Several groundcover variables were strongly correlated with the ordination (Table 2). In general, surface fitting strengthened groundcover-ordination relationships (Table 2) suggesting non-linear correlations (Oksanen et al. 2009). Litter cover and litter depth increased in the direction of reference assemblages in which two ground-foraging species, Ovenbird and Wood Thrush, were discriminating components (Fig. 7 and 8). Forb cover, bareground cover, snag density, and canopy vine density increased in the direction of mineland assemblages (Fig. 7). Tree diversity (*H'*) increased in the direction of reference assemblages. Canopy cover and structural variables had weaker correlations with the ordination and tended to be non-significant (Table 2).

The strength of the surface fit for the shrub density gradient (Table 2) was the result of fitting one mineland transect located on the far right side of the plot (Figure 7). The understory within the forest surveyed by this transect was dominated by dense multiflora rose (*Rosa multiflora*). Canopy closure was reduced relative to other mineland transects likely as a result of edaphic conditions and the presence of lightly foliated tree species in the canopy, i.e. river birch (*Betula nigra*) and bigtooth aspen (*Populus grandidentata*). Consequently, broken canopy species, such as Hooded Warbler (*Wilsonia citrina*) and American Redstart (*Setophaga ruticilla*), were characteristic of this assemblage.

The mean dissimilarity analysis indicated higher within-group dissimilarity for mineland forest assemblages echoing the pattern evident in the NMDS ordination (Fig. 9).
The adonis analysis indicated that assemblage structure differed between mineland and reference forest ($F_{1,26} = 4.38$, $p = 0.001$); between group distances were statistically greater than within group distances. Forest type accounted for only 14% of the variation in assemblage structure (partial $R^2 = 0.14$), an outcome, in part, of high within-group dissimilarity.

Overall relative abundance was similar for reference and mineland forest (Table 3). Species richness and Shannon (*H'*) diversity also were similar between forest types, but were more variable across mineland transects (Fig 10). Mineland and reference forest had similar abundance of broken canopy and forest generalist species (Fig. 10), but closed-canopy species had greater abundance within reference forest ($p = 0.002$). Mineland and reference forest had similar abundance of bark foraging and foliage gleaning species (Fig. 10), while hawking ($p = 0.03$) and ground gleaning species ($p = 0.02$) had greater abundance within reference forest. Abundance within nesting guilds was similar between forest types (Fig. 10).

Nest survival of Red-eyed Vireos

I monitored 45 Red-eyed Vireo nests in 2011, 21 within mineland forest and 24 within reference forest. Using the formula from Rotella et al. (2004), the effective sample size for computing AIC_c was 597. The Hosmer and Lemeshow (2000) goodness-of-fit test indicated that the global model fit the data adequately ($\chi^2 = 5.78$, df = 8, $p = 0.67$). Tolerance values for variables within the global model were all ≥0.65, indicating that multicollinearity was not a concern (Allison 1999).

Model selection for temporal effects found the most support for a model that included solely the effects of nest stage (w_i =0.38) (Table 4). Therefore, I included nest stage in all habitat models.

No habitat model received overwhelming support and all models, with the exception of the global model, had values for $\Delta AIC_c \leq 4$ indicating some level of empirical support for each (Burnham and Anderson 2002) (Table 4). This is in part due to the inclusion of nest stage in all habitat models; the best-fitting habitat model contained only the effect of nest stage. Other supported models with $\Delta AIC_c \leq 2$ included forest type, overstory cover, and vertical foliage density models. Models with greater complexity ($k \geq 4$) tended to have less support.

Model-averaged parameter estimates and unconditional standard errors were derived from a model set (habitat models with $\Delta AIC_c \leq 2$) that excluded the three least supported models. Daily survival was higher during laying and incubation stage (0.961, CI = 0.935, 0.977) than during the nestling stage (0.923, CI = 0.882, 0.951) (Fig. 11). The odds ratio for nest stage indicated that the odds of nest survival were 51% lower during the nestling stage than during the laying and incubation stage, but the 95% CI for the odds ratio narrowly overlapped 1.0 (Table 5).

Overall nesting period survival was 0.246 (CI = 0.132, 0.379). Period survival for mineland and reference nests, respectively, were 0.257 (CI = 0.126, 0.411) and 0.234 (CI = 0.114, 0.381) (Fig. 11). Relative to reference nests, the odds of daily nest survival were 7% greater for mineland nests, but the 95% CI widely overlapped 1.0 indicating that strength of this effect was low (Table 5). Similarly, odds ratios for overstory cover and vertical foliage density overlapped one indicating minimal influences on nest survival.

Nest patch characteristics. In both mineland and reference forest, vireos placed nests in red maple more than any other tree species (Table 6). Otherwise, nest tree use reflected mesophytic species composition within mineland forest and oak-hickory predominance within reference forest.

The predominant origin of canopy gaps occurring nearest to Red-eyed Vireo nests differed between mineland and reference forest (Table 6). Gaps adjacent to mineland nests (n=21) were primarily classified as edaphic (57%), whereas 75% of gaps adjacent to reference nests (n=24) had treefall origins. Gaps categorized as edaphic were compacted areas such as old haul roads, depressions in which water was ponded, boulder piles at the base of both outslopes and highwalls, and areas where the absence of mature trees indicated poor physical and chemical properties of minesoils. Vireo nests were located closer to canopy gaps in mineland forest (16.2 ± 2.7 m; mean ± SE) than within reference forest (26.0 ± 4.1 m) (Table 7).

In modeling structural characteristics of nest patches, ten-fold cross-validation using the 1-SE rule resulted in the selection of a four-leaved tree (Fig. 12). This tree explained 85.7% of the variance and had a misclassification rate of 6.7% and a cross validation error of 0.569, the latter indicating that the tree was an intermediate predictor. With the exception of the split based on canopy vine density, partitioning is through variables that separate reference nest patches from mineland nest patches on the basis of greater vertical heterogeneity. The first split is based on higher small tree (8-23 cm dbh) density within reference nest patches. Individuals within this size class are typically contributing foliage to mid- and subcanopy layers. This split explains the largest proportion of the variance and results in a nearly homogenous subset in its right terminal node in which 75% of reference nest patches are contained. The second split indicates higher canopy vine density within mineland nest patches. This characteristic represents a potential resource for mineland-breeding vireos in terms of providing nest material and foliage in which to conceal nests. The right node extending from the canopy vine split is in turn partitioned by subcanopy (0-6 m) foliage cover. This split results in a terminal node with a small, homogenous subset of mineland nest patches that had lower (<12.5%) subcanopy cover.

DISCUSSION

Avian assemblage structure and habitat relationships

The avian mineland assemblage was distinct from the reference assemblage despite the habitat being a narrow continuation of the closed-canopy forest in which it is imbedded. Groundcover gradients were strongly correlated with the NMDS ordination and indicated that mineland assemblages were associated with lower litter cover and depth. Decreased litter cover and depth likely contributed to the lower abundance of Ovenbirds within mineland forest. Breeding ecology studies of Ovenbirds, a ground-nesting and foraging species, have shown positive associations between leaf litter depth and pairing success (Burke and Nol 1998, Rodewald and Yahner 2000) and the selection of nest sites characterized by deeper leaf litter and lower levels of bareground cover relative to randomly selected sites (Burke and Nol 1998).

Tree species composition and mineland topography were likely influential in the lower abundance of Ovenbirds within mineland forest. Within forest dominated by mesophytic species, leaf litter and downed woody debris decomposition is typically more rapid (Nowacki and Abrams 2008). Decay is accelerated as a result of the structural characteristics of the leaves and wood belonging to mesophytic species and the cool, moist microclimate existing within the deeply shaded understory of these forests (Mudrick et al. 1994, Nowacki and Abrams 2008, Fox et al. 2010). Consequently, within mineland forest, a near complete dominance by mesophytic tree species was likely an important influence on understory light penetration, temperature, soil moisture, and soil fertility, characteristics which cumulatively were manifested in decreased litter depth and greater forb cover.

Fox et al. (2010) found that Ovenbird, Black-and-white Warbler (*Mniotilta varia*), and Worm-eating Warbler (*Helmitheros vermivorus*), all ground-nesting species, were absent from maple dominated stands in which leaf litter depth was lower relative to the oak-dominated stands in which they were present. I did not find ground-nesting species, as a group, to have lower abundance in mineland forest.

In addition to litter characteristics, features of mineland topography, e.g. outslopes and highwalls, may simply displace Ovenbirds. Characteristics of the forest floor along outslopes, including their steepness (typically >40% slope) and a high proportion of boulder and eroded, bareground cover, likely produce poor quality foraging and nesting habitat for Ovenbirds. Ovenbirds generally select territories and nest sites with low to moderate slope steepness (Wenny et al. 1993, Burke and Nol 1998).

I did not find differences in the abundance of bark gleaning or cavity nesting species between mineland and reference forest. Among these species, only White-breasted Nuthatch appeared to separate reference from mineland assemblages in the NMDS ordination. The majority of bark-foraging species were located centrally within the ordination space indicating they contributed little to dissimilarity between forest types. Minelands may receive some use from bark gleaning and cavity nesting species due to an abundance of snags and downed logs. However, these species have relatively large home ranges and the mineland habitat that I surveyed is not isolated from oak-dominated forest. In a comparison of bird communities within maple and oak dominated stands, Rodewald and Abrams (2002) found lower abundance of the bark gleaning guild in maple dominated stands during the three seasons they surveyed: spring, fall, and winter. They attributed this finding to the availability of acorns in oak-dominated forests.

Relative to reference assemblages, mineland assemblages had higher within-group dissimilarity suggesting a wider gradient in habitat conditions. Basal area was higher, but more variable for mineland transects. Within mineland forest, edaphic conditions/disturbances were likely influential in creating spatial heterogeneity in forest structure. Studies of mineland succession have indicated that the irregular composition of minesoils and the resulting fine-scale variation in spoil acidity, rock fragment composition, and shale content combine to produce spatial and compositional heterogeneity in vegetation development (Game et al. 1982, Skousen et al. 1994).

Correlations between the overall assemblage response and variables that reflect canopy structure were generally non-significant. In part, this is a consequence of a finer scale gradient in forest structure (comparing one even-aged, closed-canopy forest to another), whereas differences in groundcover were comparatively abrupt. Low sample size mineland habitat variation, and the discontinuous nature of the mineland habitat also contributed to difficulty in assessing the relationship between the assemblage response and forest structure.

Forests dominated by yellow poplar and red maple may represent poor-quality foraging and nesting habitat for a number of species. Previous studies have indicated foraging preferences among insectivorous songbirds for oaks and hickories and avoidance of red maple (Gabbe et al. 2002, Rodewald and Abrams 2002, George 2009, Newell 2010). Some forest songbirds may not be adapted for capturing arthropods on red maple foliage due to the length of its petioles and the size of its leaves (Franzreb 1978, Holmes and Robinson 1981, Holmes and Schultz 1988, Rodewald and Abrams 2002). In addition, lepidopteran assemblages may be strongly organized by the dominant canopy species (Summerville and Crist 2008) and, relative to other hardwood species, oaks may host a greater diversity of lepidopteran species (Summerville et al. 2003). Additionally, stands dominated by mesophytic hardwoods typically have higher basal area and, consequently, greater canopy closure compared to oak-dominated stands (Nowacki and Abrams 2008). The higher abundance of hawking species in reference forest, specifically, Great Crested Flycatcher and Eastern Wood-Pewee, was likely in response to lower basal area and greater canopy openness. Opening of the canopy may result in conditions (a warmer understory microenvironment and elevated primary productivity) that favor the growth of flying insect populations (Blake and Hoppes 1986, Smith and Dallman 1996, Gorham et al. 2002, George 2009). In studies of partial harvesting, Eastern Wood Pewees and Great-Crested Flycatchers generally responded positively to basal area reductions (Annand and Thompson 1997, Moorman and Guynn 2001, Holmes et al. 2004, Newell 2010).

In mixed-mesophytic forest in Ohio, Eastern Wood-Pewees placed nests in white oak in disproportion to its availability and avoided placing nests in red maple and yellow poplar (Newell 2010). Within mineland stands, reduced inter-crown spacing, predominantly vertical branching, and the tall, clean boles of yellow poplar generally concentrated foliage in the upper canopy (personal observation). Eastern Wood-Pewees as well as other species may not be adapted for nest placement in this type of canopy structure. For Blue-headed Vireos, a species which discriminated reference assemblages, mineland forest may have represented poor quality nesting habitat due to insufficient subcanopy development. Blue-headed Vireos often forage and nest within the lower canopy (Hamel 1992, Meehan 1996, James 1998) and, within central Appalachian forests, may reach greater densities within xeric forest associations relative to more mesic associations (Weakland 2000). Mineland forest had lower sapling density and cover and lower small tree (8-23 cm dbh) density compared to the relatively xeric, reference forest. In oak-hickory forest in southwestern Virginia, McEvoy et al. (1980) found greater foliage density in the 1-5 m layer within xeric sites relative to mesic sites. In the Great Smoky Mountains, Whitaker (1956) found increasing shrub cover along a moisture gradient from mesic coves to xeric spur ridges. Within xeric sites, greater light penetration through sparser canopies contributed to the increase in shrub cover (Whitaker 1952, 1956).

In general, mineland canopies follow the relatively simple structure that is characteristic of secondary forests (Lorimer 1989, Hart and Grissino-Mayer 2008). Crowns are typically non-overlapping, but restricted in their breadth by the uniform distribution of adjacent crowns. Dominants are young and, consequently, less frequent gap makers (Lorimer 1989). In response to individual tree mortality, rapid lateral crown expansion limits the duration in which the canopy remains open (Lorimer 1989). However, canopy structure and gap dynamics in mineland forests may differ slightly. Within mineland forest, canopy gaps occurring nearest to Red-eyed Vireo nests were primarily edaphic in origin. Depending on the size of the edaphic disturbance, these gaps may come to represent persistent open space (Lertzman et al. 1996).

Species composition strongly influences stand development through differences in the growth rates and shade tolerance of component species (Gingrich 1967). Reference stands were generally older than mineland stands. However, the comparison of mineland and reference forest is one of starkly contrasting tree species composition and distinct developmental pathways. Thus, given the range of mineland and reference stand ages (~60-110 years old), the influence of stand age on forest structure and the avian assemblage response was likely secondary to differences in tree species composition and in the forest floor environment.

Red-eyed Vireo nest survival

Red-eyed Vireo nest survival was similar within mineland and reference forest. Classification tree modeling indicated that reference nest patches were characterized by greater vertical heterogeneity than mineland nest patches. Despite these differences, I did not find a strong effect of nest patch characteristics on nest survival. In a study of Red-eyed Vireo nest site selection in Pennsylvania, habitat characteristics of nest patches were not indicative of nest fate (Siepielski et al. 2001). Given the nested nature of mineland stands, predator assemblage structure was not likely drastically changed from adjacent reference forest. Across both forest types, the canopy disturbance gradient was relatively narrow and may have been insufficient to produce differences in predator activity or search-efficiency. Forest cover and structure on larger scales may be more influential in terms of structuring predator assemblages and affecting reproductive success (Donovan et al. 1997, Siepielski et al. 2001, Rodewald 2002).

However, I did not expect nest survival to be poor overall given that the study area lies within largely unfragmented forest. Nest survival on my sites (0.246, CI = 0.132, 0.379) was much lower in comparison to the Mayfield nest success (0.430 ± 0.04 SE; n=126) observed by DeCecco et al. (2000) for Red-eyed Vireos breeding on the Monongahela National Forest in southeastern West Virginia. In south-central Ontario, Burke and Nol (2000) found Mayfield nest success to be 0.420 ± 0.122 SD (n=18) for Red-eyed Vireos breeding within continuous forest and 0.251 ± 0.067 SD (n=46) for vireos breeding within small forest fragments. For these habitats to function as population sources, they determined that 1.9 nesting attempts were necessary for continuous forest stands and 3.9 nesting attempts for small forest fragments.

The heights at which vireos placed nests in both mineland and reference forest were much greater than nest heights reported by other studies and may provide some explanation for the poor overall nest survival that I observed. Mean heights for vireo nests were 18.9 m ± 7.0 SD and 20.7 m ± 5.8 SD within reference and mineland forest, respectively. Studies conducted in the Midwest and in northern hardwoods forest (Lawrence 1953, Southern 1958, Rice 1974, Graber et al. 1985) observed mean nest heights within a range of 2.5-4.3 m. Other studies have reported mean nest heights that reflect greater variability; 10.7 m ± 5.8 SD for a study in New Hampshire (Robinson 1981) and 7.0 m ± 5.7 SD for a study in the southeastern West Virginia (DeCecco et al. 2000). The upper-canopy focused nest placement within my study may indicate that neither mineland nor reference forest had sufficient foliage density in the mid- and subcanopy to induce nest placement within these layers. This may have in turn resulted in greater predator search-efficiency and reduced nest concealment.

This study is the first to describe avian assemblages in relation to habitat structure and composition within mineland forest. It is also the first to quantify songbird reproductive success within mineland forest. Sample sizes were low as a consequence of the limited scale of mineland stands and the uncommonness of minelands with relatively wide, uncompacted benches on which mature forests have established naturally. However, this research lends support for the emphasis of the Forest Reclamation Approach in establishing heavy-seeded species on minelands. Within mineland stands, the near complete dominance of mesophytic tree species was likely influential in the lower abundance of Ovenbirds as well as other ground-gleaning, hawking, and closed-canopy species.

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Table 1. Variables used in environmental fitting with the avian assemblage ordination. Values are the means from mineland (n=14) and reference (n=14) transects, each of which is the average of four plots. Data was collected in 2011 from New River Gorge National River, West Virginia.

	Mineland (n=14)		Reference (n=14)	
	Mean	SE	Mean	SE
Groundcover				
downed log cover (%)	5.3	0.9	3.4	0.7
leaf litter cover (subplot %)	75.4	3.3	87.8	1.3
litter depth (cm)	1.7	0.1	2.5	0.1
litter cover (whole plot %)	50.2	3.0	76.9	2.2
bareground cover (%)	11.4	1.3	5.3	0.9
forb cover (%)	29.9	2.9	5.2	1.2
woody cover (%)	8.5	1.7	12.7	1.5
Subcanopy				
saplings ha^{-1}	80.4	16.2	121.9	7.8
sapling cover (%)	15.4	2.8	21.6	1.5
shrub stems ha^{-1}	404.9	145.8	343.3	57.2
shrub cover (%)	6.3	1.3	5.1	0.8
Stocking				
small trees ha^{-1} (8-23 cm dbh)	233.5	25.7	330.8	22.1
med. trees ha^{-1} (23-38 cm dbh)	145.5	8.6	138.8	10.4
large trees ha^{-1} (>38 cm dbh)	100.4	8.4	75.9	6.0
total trees ha^{-1} (>8 cm dbh)	479.5	25.3	545.5	18.6
basal area (m^2 ha^{-1})	34.1	1.5	29.1	0.7
snags ha^{-1}	58.9	5.1	27.7	6.5
Foliage density (%)				
understory foliage (0-6m)	15.7	1.9	17.5	1.3
midstory foliage (6-18m)	36.9	3.1	43.1	1.8
overstory foliage (>18m)	39.5	3.5	34.9	2.0
vertical foliage diversity	30.8	1.0	32.0	0.7
Topographic and misc.				
slope (%)	12.8	0.9	10.2	0.6
solar radiation (1,000 WH/m^2)	1317.7	13.7	1326.6	10.2
tree diversity (*H'*)	1.25	0.07	1.67	0.04
canopy vines ha^{-1}	64.3	13.0	6.7	3.3

Table 2. Correlations between vector (linear) and surface (general additive modeling) fitting between habitat and topographical variables and the 3-dimensional non-metric dimensional scaling solution for avian assemblages within mineland and reference forest. *P*-values from vector fitting are derived from 999 permutations of the data. Statistical significance was $p<0.05$.

Variable	**Vector r^2 (*p*-value)**	**Surface r^2 (*p*-value)**
Groundcover		
bareground cover	0.28 (0.048)	0.22 (0.02)
litter cover	0.50 (0.005)	0.66 (<0.001)
litter depth	0.46 (0.003)	0.63 (<0.001)
forb cover	0.55 (0.001)	0.64 (<0.001)
woody groundcover	0.12 (0.39)	0.10 (0.38)
downed log cover	0.19 (0.18)	0.11 (0.49)
Subcanopy		
saplings ha^{-1}	0.09 (0.53)	0.01 (0.32)
shrubs ha^{-1}	0.18 (0.17)	0.62 (<0.001)
Stocking		
small trees ha^{-1} (8-22.9 cm dbh)	0.08 (0.57)	0.11 (0.43)
med. trees ha^{-1} (23-37.9 cm dbh)	0.14 (0.31)	0.13 (0.30)
large trees ha^{-1} (>38 cm dbh)	0.18 (0.20)	0.31 (0.11)
basal area	0.20 (0.17)	0.38 (0.055)
snags ha^{-1}	0.39 (0.006)	0.34 (0.002)
Canopy cover		
understory cover (0-6 m)	0.07 (0.64)	0.00 (0.43)
midstory cover (>6-18 m)	0.10 (0.43)	0.00 (0.41)
overstory (>18 m cover)	0.03 (0.85)	0.00 (0.43)
vertical complexity	0.14 (0.33)	0.25 (0.19)
Topographic and misc.		
slope	0.09 (0.49)	0.11 (0.51)
solar radiation (WH/m^2)	0.03 (0.86)	0.05 (0.67)
tree diversity (*H'*)	0.39 (0.007)	0.33 (0.003)
vines ha^{-1}	0.44 (0.003)	0.43 (0.004)

Table 3. Mean (±SE) detections ha^{-1} for all species and habitat, foraging, and nesting guilds. Richness is the mean (±SE) number of species ha^{-1}. *P*-values are from Poisson GLMs using an analysis of deviance test in which the difference in deviance approximately follows a Chi-square distribution with 1 degree of freedom. Statistical testing for Shannon diversity (*H'*) was through analysis of variance (* indicates the value of the *F*-statistic). All tests included site as a fixed effect. Statistical significance was $p<0.05$.

	Mineland forest		Reference forest			
	mean	SE	mean	SE	χ^2_1	*p*
Diversity (*H'*)	1.84	0.10	1.97	0.07	0.90*	0.35
Richness	3.97	0.42	3.85	0.24	2.28	0.13
Relative abundance						
All species	6.82	0.69	7.45	0.56	0.45	0.50
Habitat guilds						
closed canopy	1.01	0.15	2.08	0.23	9.53	0.002
broken canopy	0.37	0.15	0.31	0.06	0.45	0.50
forest generalists	5.39	0.57	5.34	0.39	2.61	0.11
Foraging guilds						
hawking	0.17	0.08	0.54	0.09	4.94	0.03
bark foragers	0.75	0.16	1.06	0.18	0.65	0.42
foliage gleaners	4.50	0.30	4.25	0.36	2.46	0.12
ground gleaners	0.84	0.14	1.6	0.15	5.50	0.02
Nesting guilds						
canopy	3.95	0.41	3.99	0.33	1.49	0.22
subcanopy	0.40	0.13	0.60	0.17	1.00	0.32
ground	1.27	0.18	1.91	0.15	1.71	0.19
cavity	0.95	0.18	0.88	0.15	0.57	0.45

Table 4. Temporal and habitat model selection results for nest survival of Red-eyed Vireos breeding within mineland and reference forest in New River Gorge National River, 2011. $\text{Log}_e(L)$ is the value of the maximize log-likelihood function, K is the number of parameters estimated by the model, ΔAIC_c is the scaled value of Akaike"s Information Criterion for small sample sizes, and w_i reflects the relative support attributed to a given model.

	$\text{Log}_e(L)$	**k**	**ΔAIC_c**	**w_i**
Temporal Models				
Nest stage	-113.34	2	0	0.38
Nest stage + date	-113.22	3	1.77	0.16
Nest stage + date^2	-112.26	4	1.88	0.15
Constant survival[c]	-115.34	1	1.99	0.14
Date^{2b}	-113.79	3	2.92	0.09
Date	-114.88	2	3.08	0.08
Habitat Models				
Nest stage	-113.34	2	0	0.34
Nest stage + forest type	-112.98	3	1.30	0.18
Nest stage + overstory cover	-113.04	3	1.42	0.17
Nest stage + vertical foliage density	-113.32	3	1.97	0.13
Nest stage + forest type + overstory cover	-112.78	4	2.93	0.08
Nest stage + forest type + vertical foliage density	-112.95	4	3.26	0.07
Global[a]	-112.52	5	4.44	0.04

[a] Includes all variables used in habitat models.
[b] Date^2 is the quadratic effect of date (date + date^2).
[c] The null model.

Table 5. Model-averaged parameter estimates, their unconditional standard errors (SE), and odds ratios (OR) with unconditional 95% confidence intervals (CI) for habitat and temporal variables used in modeling nest survival of Red-eyed Vireos breeding within mineland and reference forest in New River Gorge National River, 2011. Odds ratios for vertical foliage density and overstory cover are for a 1% increase in the variable.

Parameter	Estimate	SE	OR	95% CI
Vertical foliage density	-0.0011	0.0061	1.000	0.987, 1.011
Overstory cover	0.0023	0.0053	1.002	0.992, 1.013
Mineland versus reference forest	0.0672	0.1478	1.070	0.796, 1.437
Nestling versus laying/incubation	-0.7225	0.3631	0.486	0.235, 1.004

Table 6. Distribution of tree species used by Red-eyed Vireos for nest sites and the distribution of canopy gap types from those identified as being closest to vireo nests.

Mineland nests (n=21)		Reference nests (n=24)	
Nest tree species	%	Nest tree species	%
Red maple	38.1	Red maple	29.2
Yellow poplar	28.6	Scarlet oak	20.8
Slippery elm	14.3	Black oak	16.7
Black birch	14.3	Chestnut oak	12.5
White oak	4.8	White oak	8.3
		Mockernut hickory	4.2
		American beech	4.2
		Black gum	4.2
Nearest gap type	%	Nearest gap type	%
Edaphic gap	57.1	Edaphic gap	0.0
Snag-created gap	9.5	Snag-created gap	25.0
Treefall gap	33.3	Treefall gap	75.0

Table 7. Means (±SE) from nest measurements and habitat sampling for Red-eyed Vireo nest patches within mineland and reference forest in New River Gorge National River, West Virginia (2011). Variables used in classification tree modeling are listed under the "structural" heading (shrub and sapling cover were used instead of shrub and sapling density).

	Mined (n=21)		Reference (n=24)	
	Mean	SE	Mean	SE
Groundcover				
bareground cover (%)	11.0	3.1	1.9	0.7
forb cover (%)	33.1	3.7	2.5	1.5
litter cover (%)	49.5	4.2	70.8	2.3
woody cover (%)	6.4	2.0	24.8	2.3
Subcanopy				
shrub stems ha^{-1}	384.5	124.8	666.7	89.5
shrub cover (%)	4.6	0.9	7.1	0.7
saplings ha^{-1}	41.7	9.0	139.6	23.2
sapling cover (%)	11.7	2.5	21.1	4.2
Stocking				
small trees ha^{-1} (8-22.9 cm dbh)	211.9	33.5	370.8	21.8
med. trees ha^{-1} (23-37.9 cm dbh)	136.9	15.8	139.6	13.9
large trees ha^{-1} (>38 cm dbh)	76.2	11.4	93.8	7.7
total trees/ha (>8 cm dbh)	425.0	36.6	604.2	21.1
basal area (m^2 ha^{-1})	27.9	1.9	32.8	1.5
snags ha^{-1}	63.1	12.3	25.0	5.2
Canopy cover (%)				
subcanopy foliage (0-6 m)	14.4	2.0	22.7	2.0
midstory foliage (>6-18 m)	38.8	2.7	39.8	2.4
overstory foliage (>18 m)	44.9	3.7	40.5	1.5
vertical foliage density	32.7	1.4	34.3	0.9
Nest measurements and misc.				
nest height (m)	20.7	1.3	18.9	1.4
nest tree height (m)	26.7	1.3	24.3	1.8
nest tree canopy depth (m)	12.2	1.3	10.3	1.0
nest tree dbh (cm)	37.8	3.2	34.9	3.4
nest to canopy top (m)	6.0	0.9	5.4	0.8
nest to canopy bottom (m)	6.2	1.0	4.8	0.7
canopy vines ha^{-1}	92.9	23.0	3.1	2.3
gap distance (m)	16.2	2.7	26.0	4.1
nearest dominant height (m)	29.9	1.1	28.4	0.7

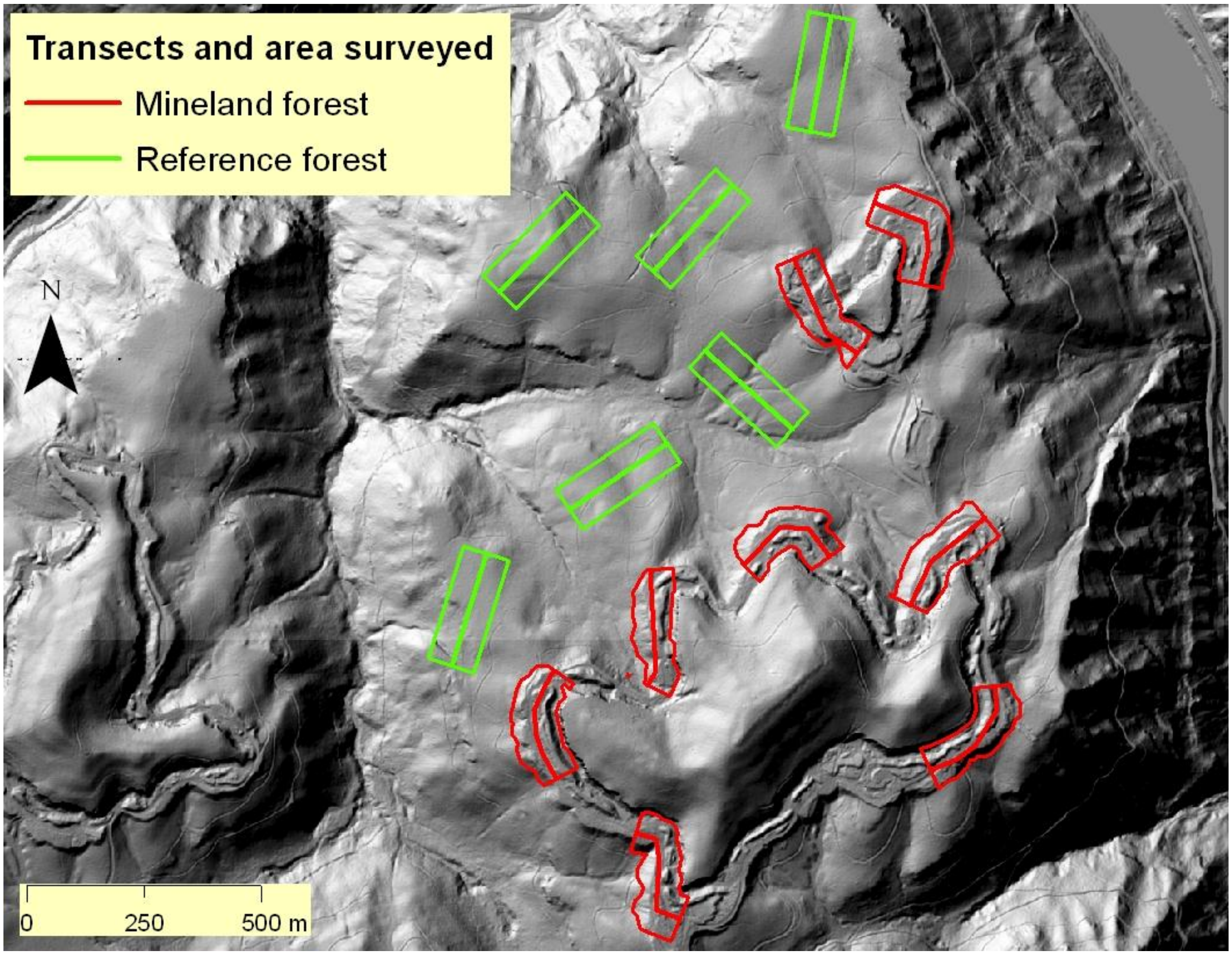

Fig. 1. Location of line transects surveyed in 2011 at the Sewell Knob site in New River Gorge National River, West Virginia. The area surveyed corresponds to 50 m strips on each side of the 250 m transect line or to the full extent of the habitat when mineland width is <100 m.

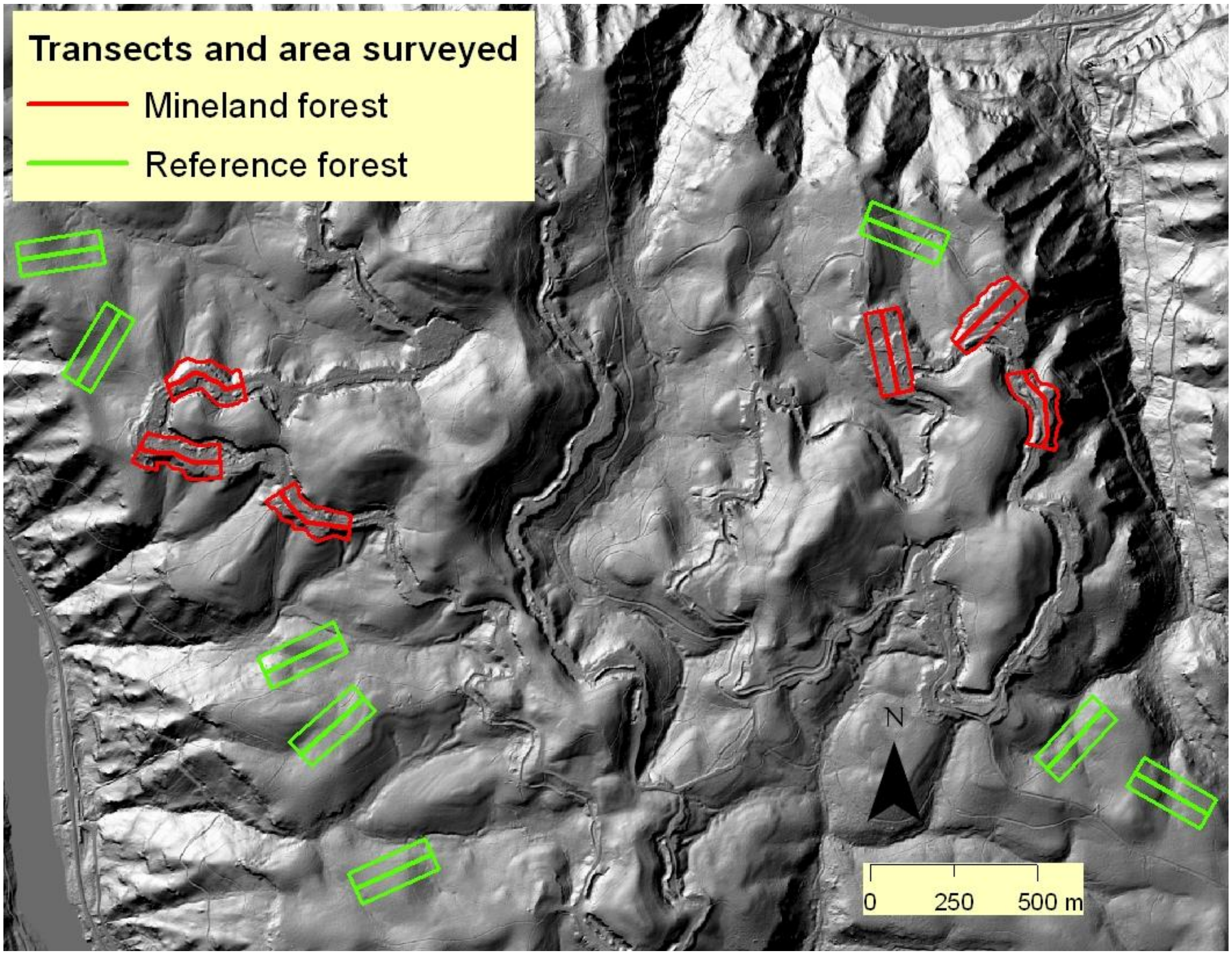

Fig. 2. Location of line transects surveyed in 2011 at the Molly‟s Creek site (west) and Fire Creek site (east) in New River Gorge National River, West Virginia. The area surveyed corresponds to 50 m strips on each side of the 250 m transect or to the full extent of the habitat when mineland width is <100 m.

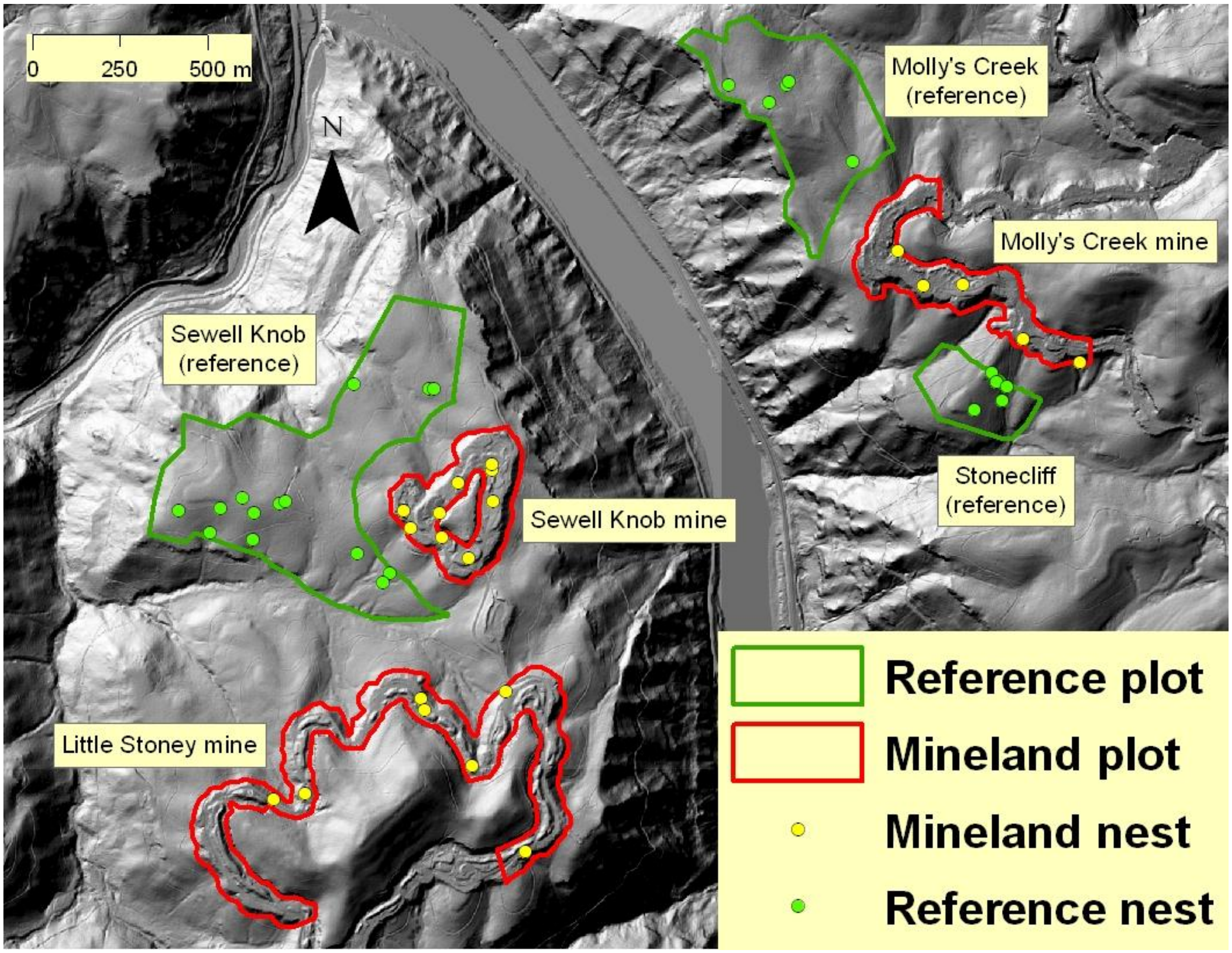

Fig. 3. Location of Red-eyed Vireo nest plots and nests that were monitored within those plots in 2011 in New River Gorge National River, West Virginia. Area and nests monitored for each plot were: Sewell Knob reference (36.4 ha, n=14), Sewell Knob mine (8.8 ha, n=9), Little Stoney mine (20.8 ha, n=7), Molly"s Creek reference (17.4 ha, n=5), Molly"s Creek mine (10.1 ha, n=5), and Stonecliff reference (5.4 ha, n=5).

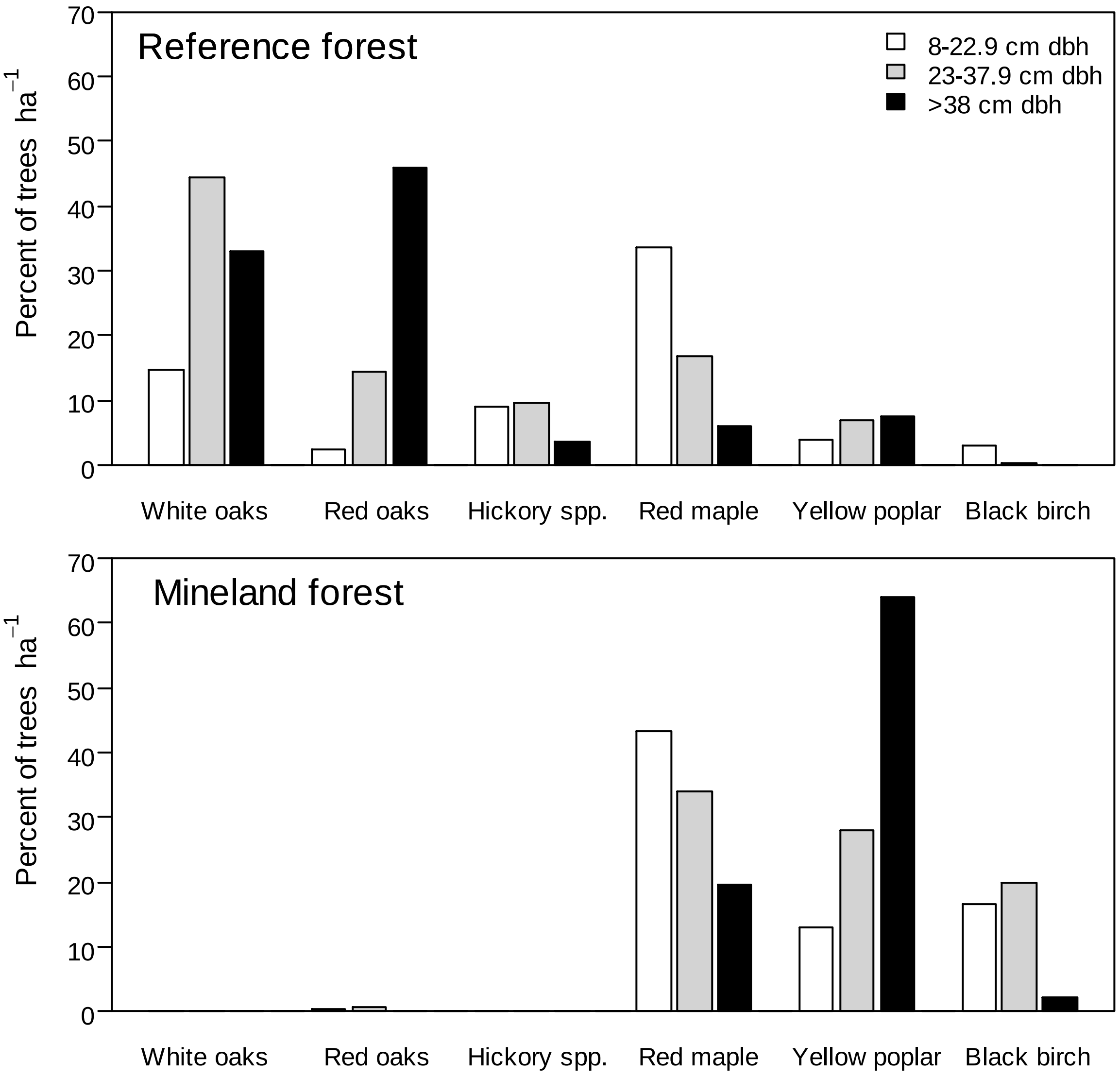

Fig. 4. Distribution of tree species groups by size class from reference (n=56) and mineland forest plots (n=56) sampled along bird survey transects in New River Gorge National River, West Virginia (2011). This is expressed as the proportion of tree density within a size class. White oaks include chestnut and white oak. Red oaks include black, scarlet, and northern red oak.

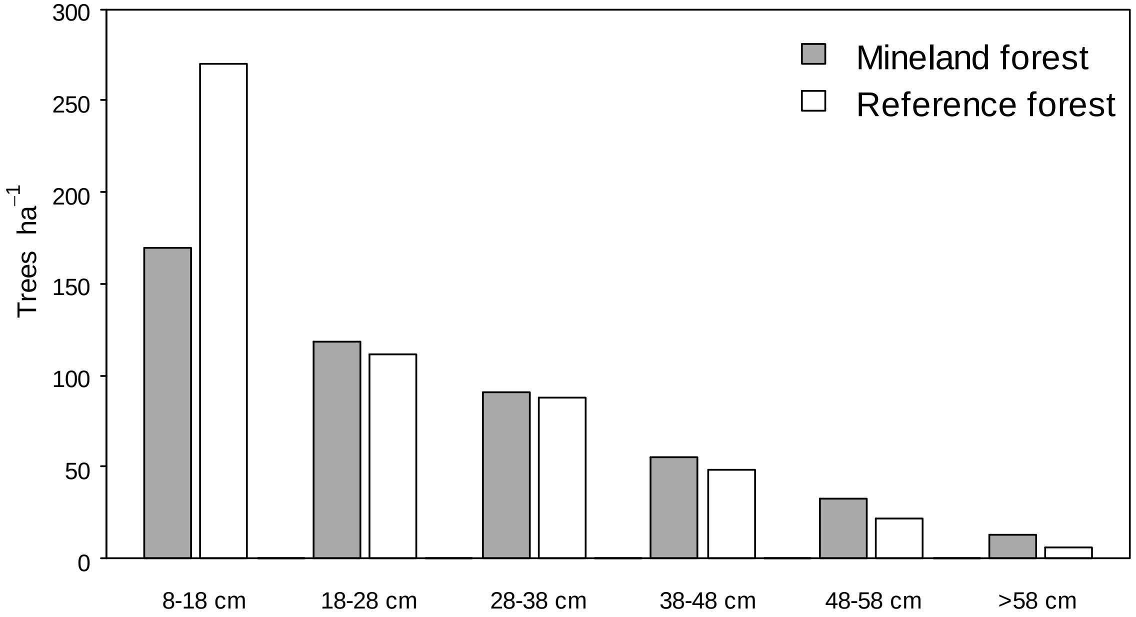

Fig. 5. Diameter distribution from reference (n=56) and mineland forest plots (n=56) sampled along bird survey transects in New River Gorge National River, West Virginia (2011).

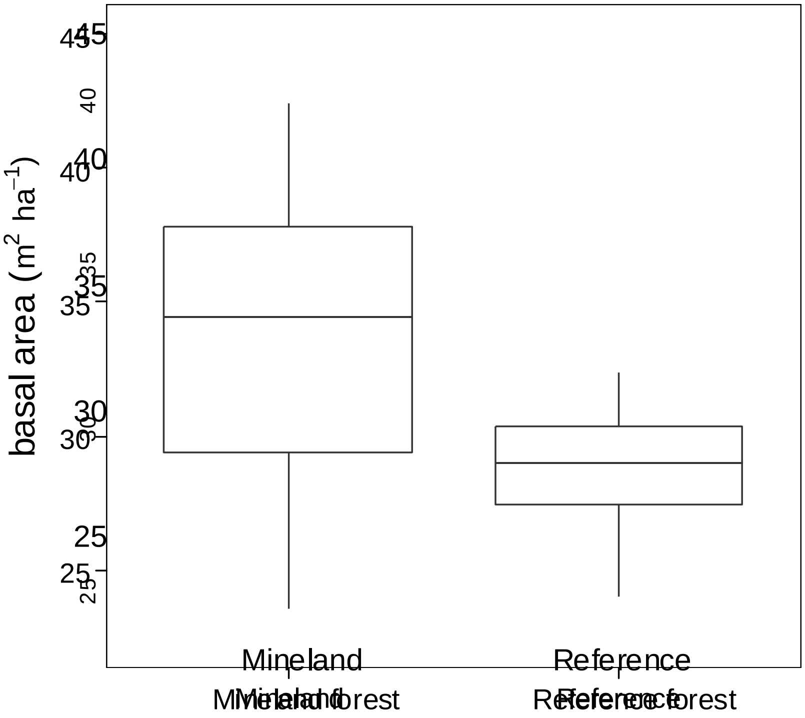

Figure 6. Boxplot of basal area (m^2 ha^{-1}) for transects within mineland and reference forest.

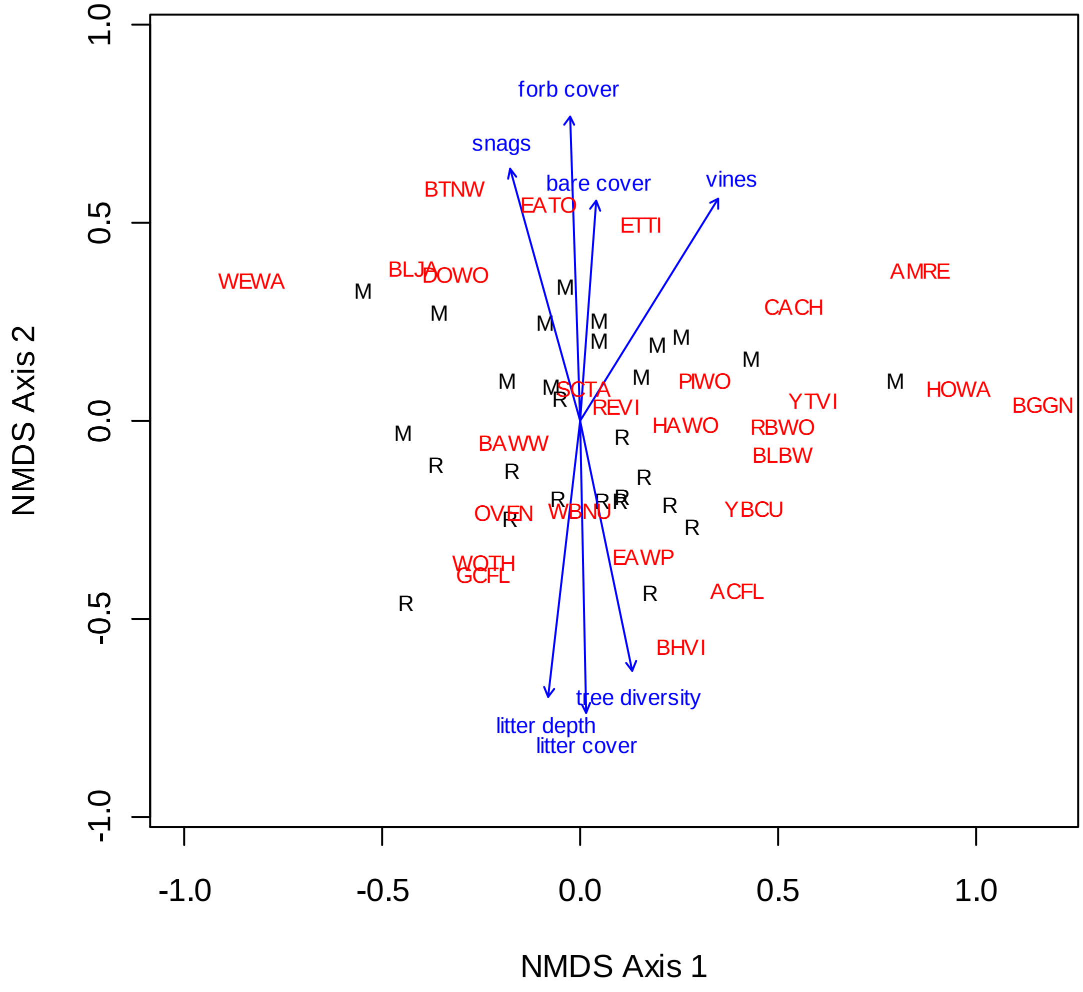

Figure 7. Dimensions 1 and 2 from a 3-dimensional non-metric dimensional scaling ordination for avian assemblages within mineland (M) and reference (R) forest. Stress was 15.3 for the 3-dimensional solution (2 convergent solutions after 6 runs). The vectors plotted are for those variables that had linear $p<0.05$ (axes 1-3). The length of the arrow corresponds to the correlative strength of the gradient-ordination relationship. Weighted mean positions for all bird species are shown. Species codes are listed in Appendix C.

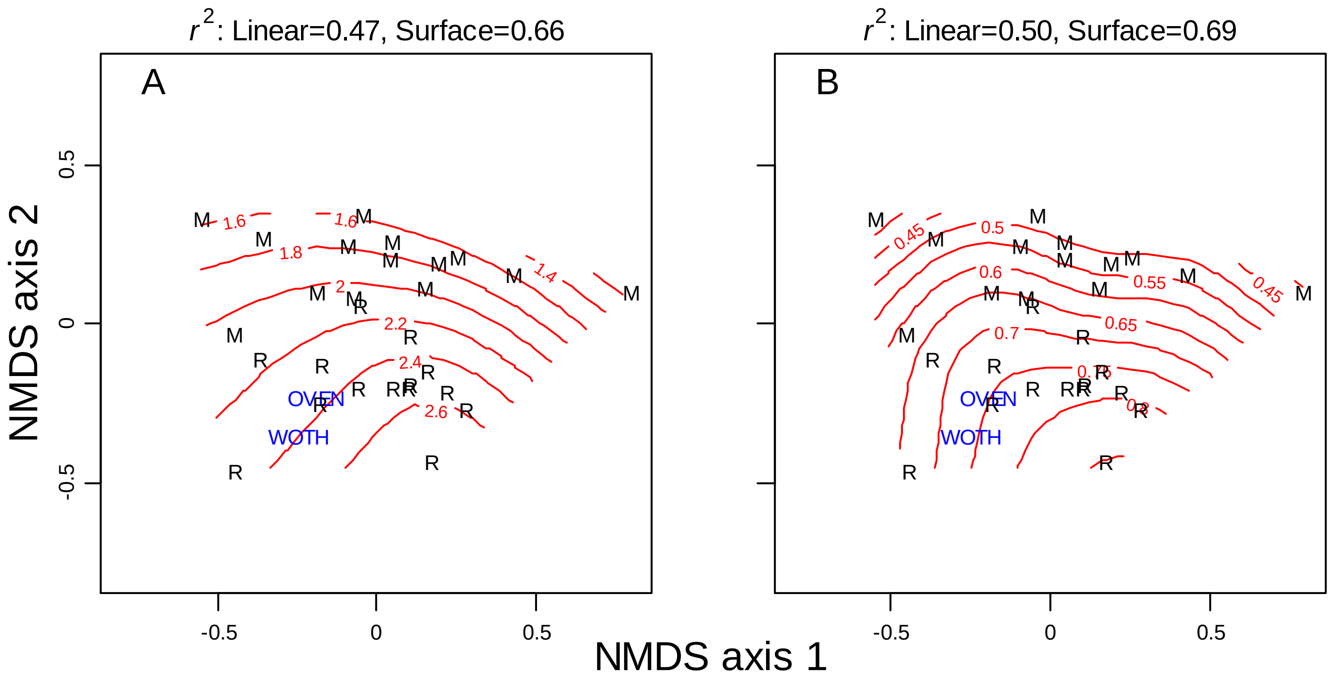

Fig. 8. Non-metric dimensional scaling ordinations fit with surface contours using GAM for A) leaf litter depth, (contours are in cm) and B) litter cover. Mean weighted locations for species in the ground gleaning guild (Ovenbird and Wood Thrush). Linear and surface fit r^2 values for the ordination-gradient relationship are shown above their corresponding frames. Surface fits are significant (p<0.001). Assemblage type is overlaid; mineland (M) and reference (R) forest.

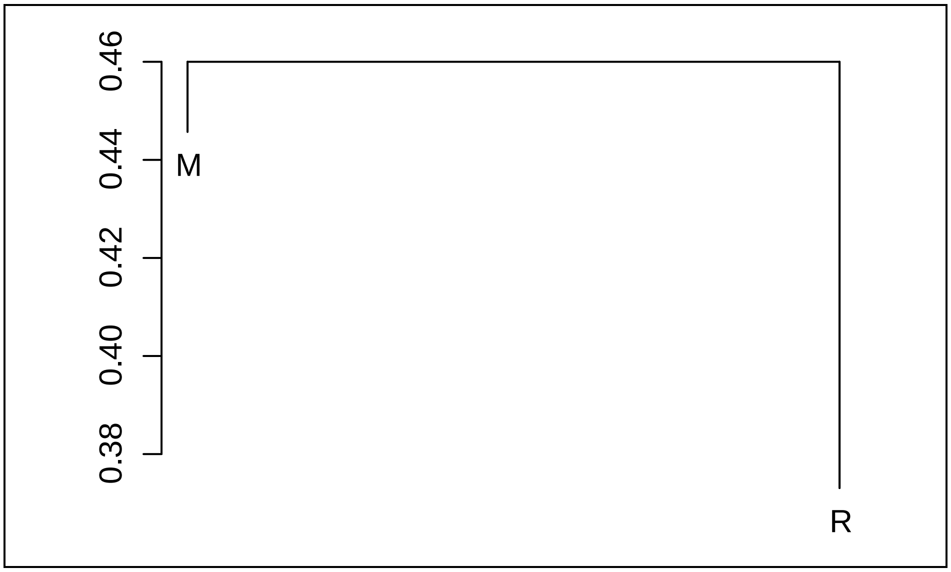

Figure 9. Mean dissimilarity dendrogram for mineland (M) and reference (R) forest assemblages. Mean between-group dissimilarity (Bbar) = 0.460. Mean within-group dissimilarity (Wbar) = 0.410.

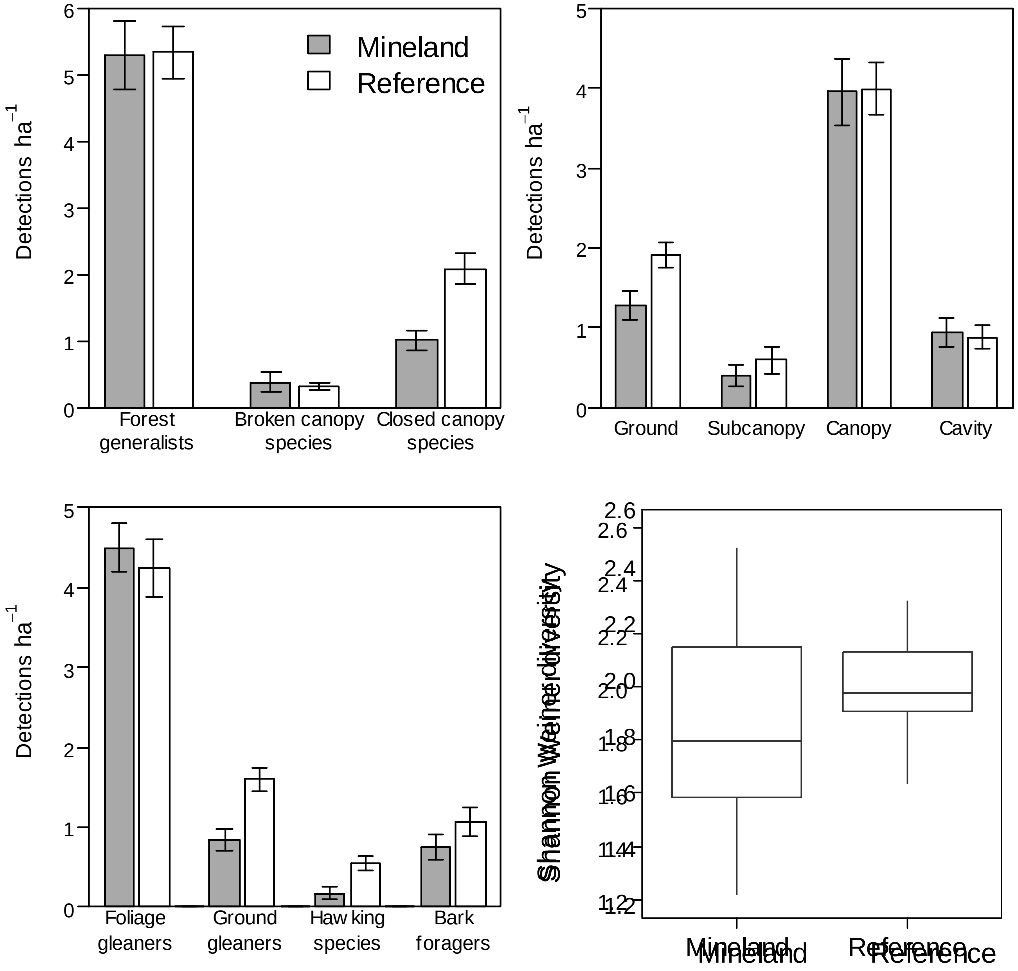

Figure 10. Mean detections ha^{-1} (± SE) for habitat, nesting, and foraging guilds within mineland (gray) and reference forest (white). The boxplot for Shannon diversity (*H'*) is also shown.

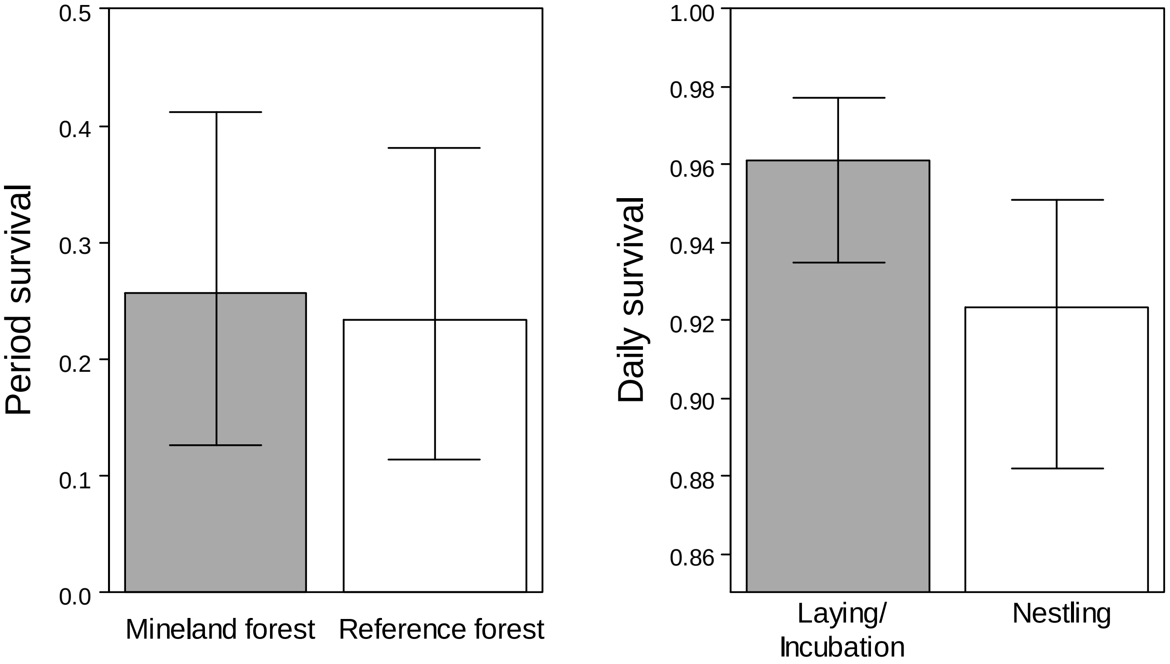

Figure 11. Model-averaged estimates and 95% CIs for period survival of Red-eyed Vireo nests within mineland and reference forest and for daily survival for laying/incubation and nestling stages.

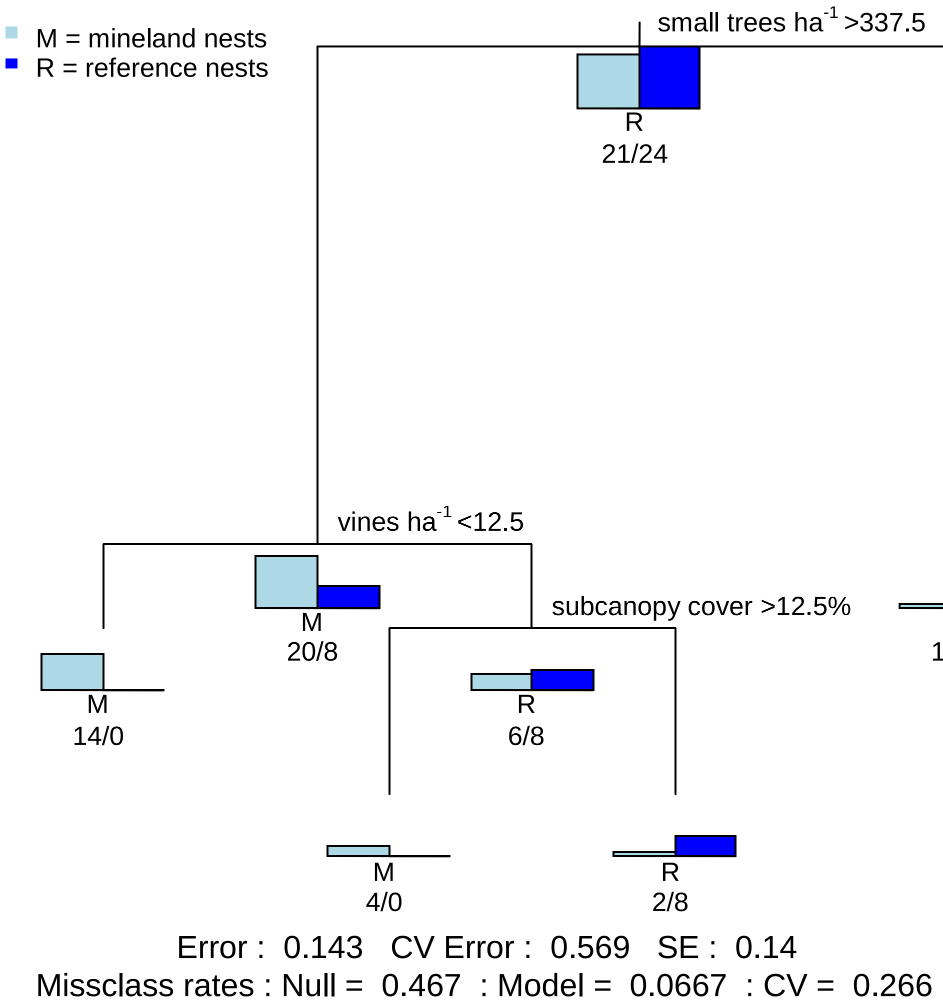

Fig. 12. Classification tree for structural characteristics within Red-eyed Vireo nest patches within mineland (M; n=21) and reference (R; n=24) forest in New River Gorge NR, West Virginia (2011). Nodes are classified according to the dominant forest type.

Chapter 3

THE EFFECT OF FORESTED, ABANDONED MINE LANDS ON AVIAN ASSEMBLAGE STRUCTURE

ABSTRACT

Despite the extent to which they bisect Appalachian forest landscapes, the influence of forested pre-SMCRA minelands on the avian assemblage has received little study. In 2010, I initiated research with an objective of examining patterns in avian assemblage structure in mined and unmined forest and also across two broad classes of minelands, compacted bench minelands and loose-dumped bench minelands. In New River Gorge National River and Plum Orchard Wildlife Management Area, I conducted avian point counts and sampled forest structure and composition within four habitat types: loose-dumped bench minelands, compacted bench minelands, unmined plateau, and unmined steep slope. Non-metric dimensional scaling (NMDS) ordination indicated that minelands with loose-dumped benches had minimal effect on assemblage structure. Only compacted bench minelands had a relatively distinct avian assemblage due in part to restricted forest development along benches and edge influence along highwalls. Dense midcanopy cover, relatively low levels of overstory cover, and substantially higher canopy vine density characterized compacted bench minelands. Species which use the subcanopy and midcanopy for nesting and foraging, American Redstart, Rose-breasted Grosbeak, and Worm-eating Warbler, differentiated the avian assemblage associated with compacted bench minelands. Relative abundance of the closed-canopy guild was lower within mined forest than within unmined forest. For some mature forest restricted species, i.e. Blue-headed Vireo and Ovenbird, mineland habitats may have low suitability due to relatively low subcanopy cover within loose-dumped bench sites and relatively low overstory cover and canopy height along compacted benches.

INTRODUCTION

In much of Appalachia, mineland forest is now a component of a landscape that was extensively contour stripped prior to the passage of the Surface Mining Control and Reclamation Act (SMCRA) in 1977. Prior to SMCRA, mines used a “shoot and shove” method for extraction resulting in an exposed highwall, bench-land along its face, and an outslope comprised of loose spoil that had been pushed below. Mines were left in this “shoot and shove” configuration and reclamation practices varied from state to state (Brown 1962). Mature forest has since developed on many of these sites. However, forest development varies according to the extent of loose-dumped spoil atop benches, and on many of these sites mature forest is not contiguous.

Despite the extent to which they bisect Appalachian forest landscapes, the influence of forested pre-SMCRA minelands on the avian assemblage has received little study. Pre-SMCRA minelands may represent significant structural and compositional contrast to adjacent forest on unmined soils. As such, the presence of forested pre-SMCRA minelands on the landscape may result in an avian assemblage distinct from the one found within forest lacking pre-SMCRA minelands (hereafter, unmined forest). In addition, forest development and, consequently, the degree of edge influence along highwalls may differ between minelands with compacted benches and those with loose-dumped spoil distributed atop benches. Therefore, avian assemblage composition could also vary in response to differences in habitat structure between compacted bench minelands and loose-dumped bench minelands. The objective of this study was to examine patterns in avian assemblage structure in mined and unmined forest and also across two broad classes of minelands, compacted bench minelands and loose-dumped bench minelands.

SITE DESCRIPTIONS AND METHODS

Study area and sampling design. The study area is located atop the New River Plateau and along the steep, upper slopes within New River Gorge National River and Plum Orchard Wildlife Management Area (WMA) in southern West Virginia (Appendix F). The oak-hickory and mixed-mesophytic forest within these public landholdings is largely unfragmented and varies compositionally along a soil moisture and fertility gradient (Vanderhorst et al. 2007). Oak-hickory forest occupies much of the relatively xeric plateau with white (*Quercus alba*), chestnut (*Quercus prinus*), scarlet (*Quercus coccinea*), and black oak (*Quercus velutina*) comprising the predominant canopy species. Along the steep, upper slopes within Plum Orchard WMA and the New River Gorge, the forest is of the mixed-mesophytic type and northern red oak (*Quercus rubra*), chestnut oak, hickory spp. (*Carya* spp.), red maple (*Acer rubrum*), sugar maple (*Acer saccharum*), yellow poplar (*Liriodendron tulipifera*), basswood (*Tilia americana*), and American beech (*Fagus grandifolia*) are dominants. Elevations within the study area are 550-730 m.

I selected pre-SMCRA abandoned minelands based on the following criteria: the presence of mature hardwood forest that had established naturally, the absence of roads along benches, and the presence of adjacent, unmined mature hardwood forest. I established one set of paired point count transects (mined and unmined) at each of five sites. Because variation in slope steepness contributed strongly to differences in tree species composition and mineland habitat structure, mined and unmined transects were classified on the basis of slope steepness. During early contour mining, slope steepness was a primary factor in determining the extent to which successive cuts could be made along a contour and, consequently, in how loose-dumped spoil was distributed. In areas of greater slope steepness, mine operators encountered a rapidly increasing overburden to coal ratio with each progressive cut into the mountainside. Consequently, along steeper slopes, benches tend to be narrow and relatively compacted with all loose-dumped spoil pushed into outslopes.

Three study sites were located within oak-hickory forest atop the New River Plateau (Molly‟s Creek, Fire Creek, and Sewell Knob) (Fig. 1). Within these sites, minelands were relatively wide with spoil ridges distributed atop benches (Figures 2-3) and thus, were classified as loose-dumped benches. Mature forest cover was generally contiguous within these minelands. Unmined transects within these sites were classified as unmined plateau forest. Two study sites were located within mixed-mesophytic forest along the steep, upper slopes of the New River Gorge (Turkey Spur) and Plum Orchard WMA (Fig. 1). Mined transects within these sites were classified as compacted benches because they were narrow and had level benches upon which mature forest had often not developed (Figures 4-5). Unmined transects within these sites were classified as unmined, steep slope forest.

Within minelands, forest structure along highwalls also varied according to slope steepness. Within loose-dumped bench sites, the mineland canopy largely obscured the edge once created by the highwall. Within compacted bench sites, the mineland canopy rarely exceeded the height of the highwall and highwalls were occasionally collapsed with vine-choked, young forest vegetation extending from the top of the highwall to the bench below.

Using aerial photos taken in 1945 and 1957, I established that all five mine sites were mined between 1945 and 1957 (Appendix B). The unmined stands primarily originated during extensive clearcutting at the turn of the century (Brooks 1910), but also included stands that were closer in age to mineland stands. The latter were abandoned homesteads and a former mining company town that existed as early successional vegetation at the time the 1945 photos were taken.

The seams from which coal was extracted at mine sites included: the Middle War Eagle coal in the Kanawha Formation and the Fire Creek coal and Sewell coal in the New River

Formation. The Kanawha, New River, and Pocahontas formations constitute the Pottsville Group (Pennsylvania Period) and contain a series of interbedded coal-bearing sandstones, siltstones, and shales (Barlow 1974). Mine sites were not planted with trees with the exception of a 0.4 ha section of pitch pine on the Sewell Knob mine and a row of white pine ~150 meters long on the Turkey Spur mine.

Point count transect layout was accomplished in ArcGIS. I delineated minelands using aerial photographs and randomly generated sampling points (Hawth‟s Analysis Tools; Beyer 2004) within an area that extended from 20 m upslope to 20 m downslope of the mine. At two sites, the area above the highwall could not be sampled safely so I excluded these particular areas prior to generating points.

Adjacent to each mined transect, I delineated an area of unmined forest. I attempted to match each unmined area to the adjacent mined transect with respect to aspect, slope position, and size large enough to encompass a similar number of sampling points. Within each area of unmined forest, I randomly generated sampling points using Hawth‟s Analysis Tools (Beyer 2004). Both mined and unmined points were separated by at least 250 m and unmined points were >100 m from mines. Although the study area contains largely unfragmented forest, a variety of small, anthropogenic disturbances occur throughout, including narrow forest roads, isolated homes, narrow powerline corridors, and patches of early-successional vegetation on abandoned homesites. All points were placed at least 85 m from these disturbances.

In 2010, I sampled avian assemblages at 17 points in unmined plateau forest, 17 points along loose-dumped benches, 17 points in unmined, steep slope forest, and 14 points along compacted benches. In total, I surveyed 31 points at mined sites and 34 points at unmined sites.

Bird surveys. Between 1/2 hour after sunrise and 1045 AM from 16 May through 2 June, I sampled breeding bird communities using variable circular plot point counts. I conducted ten minute counts and sampled each point twice (Petit et al. 1995) with approximately one week between counts. Observations were categorized into five detection types: singing, calling, displaying or drumming, flyovers, and visuals. I used a laser rangefinder to obtain radial distances to each bird that was detected with the exception of flyovers. Distances were assigned to the following categories: 0-10 m, 11-20 m, 21-30 m…91-100 m, 101-125 m, 126-150 m, and >150 m. Detections were recorded into four time intervals: 0-3 minutes, >3-5 minutes, >5-8 minutes, and >8-10 minutes.

Distance analysis. Heterogeneity in detectability between habitats can introduce bias in estimates of density that if unmodeled may lead to erroneous inferences about populations (Diefenbach et al. 2003, Kissling and Garton 2006, Simons et al. 2006). Distance sampling is a method used in concert with randomized line and point transect surveys for the purpose of applying a correction factor (detection function) to raw count data and thereby removing bias from estimates of density (Thompson 2002). Assumptions within distance sampling are: 1) detection at zero distance has probability 1. 2) animals are detected instantaneously and prior to evasive movement or attraction in response to observers, and 3) distances are measured accurately. In studies of forest songbirds, these assumptions are often incompletely met at some level (Kissling and Garton 2006).

Mineland topography and variation in habitat structure between mined and unmined forest were potential sources of detection heterogeneity. Therefore, I evaluated whether detectability differed between mined and unmined forest by modeling detection functions for as many species as sample size would allow. For three species with >60 singing detections within both mined and unmined stratums, I fit a separate detection function to data within each stratum using conventional distance sampling (CDS) (Buckland et al. 2001). For four species in which sample size was inadequate for stratification but for which there were at least 60 total singing detections, I modeled the detection function for the pooled data as a function of covariates that I assumed to have created heterogeneity in detectability (multiple covariate distance sampling (MCDS)) (Marques et al. 2007). I then used post-stratification to obtain separate detection functions for mined and unmined forest. Covariates included slope type (plateau or steep slope), percent slope, vertical foliage density, subcanopy cover (0-6 m), midcanopy cover (>6-18 m), overstory cover (>18 m), small trees (trees <23 cm dbh), large trees (trees ≥23 cm dbh), and sapling density. Values for covariates were the sum or mean (as appropriate) of data collected at three vegetation plots per point. Prior to analysis, I truncated 5-10% of a species‟ furthest observations to remove outliers (Buckland et al. 2001).

I modeled detection functions within Program Distance (Thomas et al. 2009) through selection from two key functions (half-normal and hazard-rate) with incorporation, given that model fit was improved, of cosine or simple polynomial series expansion terms (Buckland et al. 2001). I used Akaike‟s Information Criterion (AIC), visual inspection of detection function and probability density histograms, and Chi-squared goodness-of-fit tests to select between models and to assess model fit (Buckland et al. 2001).

Estimates of the effective detection radius (EDR) and their 95% confidence intervals indicated that detectability was similar for mined and unmined forest for all seven species that I modeled (Fig. 6). Depending on the function used in the detection model, the EDR estimates the distance at which the probability of detection declines below 0.5 (Laake et al. 1993, Simons et al. 2006).

Due to the aggregated, linear nature of minelands and the limited scale at which I sampled them, distances may not have been independent of bird distribution. For a species that is displaced by mineland forest, the scale at which I sampled minelands would not be sufficient for the distribution of distances to reflect independence from the location of my sampling points (Marques 2007). For this reason as well as the absence of strong differences in detectability between mined and unmined forest, I did not correct raw counts. For analyses, I used a species‟ maximum count between the two visits to each station.

Habitat sampling. I sampled habitat structure and composition using methods similar to Wood et al. (2001) that were modified from James and Shugart (1970) and the Breeding Bird Research Database Program (BBIRD; Martin et al. 1997). At a distance of 50 meters from each point count station, I established three habitat sampling plots along bearings separated by 120 degrees, the first of which was generated randomly. Within plots, I identified all trees >8 cm diameter at breast height (dbh) to species and measured dbh. I tallied all vines that reached the canopy on measured trees and counted all snags >8 cm dbh and >8 m tall. Within each plot, I established two, 22.6 m perpendicular transects. Using an ocular tube and sighting along the tube‟s crosshairs, I estimated vertical foliage density at a total of 20 points, located 2 m apart along the perpendicular transects. I recorded the presence or absence of live foliage in the crosshairs at heights of 0.5-3 m, >3-6 m, >6-12 m, >12-18 m, >18-24 m, and >24 m. From these data, I calculated vertical foliage density as the sum of all foliage hits divided by the total number of sighting intervals (120) and then multiplied by 100. Foliage density data also were collapsed into understory (0-6 m), midstory (.6-18 m), and overstory (>18 m) layers.

Within 3 m-radius subplots at the center of each 0.04 ha plot, I counted woody vegetation 0.5-1.5 m tall (shrubs) and tallied saplings (>1.5 m tall and <8 cm dbh) and downed logs. I also estimated shrub, sapling, and downed woody debris cover (logs and stumps >8 cm dbh and >1 m in length). Using a 1 m DEM, I calculated mean percent slope within a 50-m radius of each point.

ANALYSES

Non-metric multidimensional scaling (NMDS) was used to visualize patterns in avian assemblage structure between mined and unmined habitat types. In NMDS, ordination is based upon preservation of the original rank order of between sample distances derived from a dissimilarity matrix (Clarke and Green 1988, Clarke 1993). In using rank order distances, NMDS avoids the linear distribution assumption (Clarke and Green 1988). Locations of assemblages in the multidimensional space are determined through multiple iterations such that stress is minimized (Clarke and Green 1988). Stress is a measure of goodness of fit between plotted and true rank order distances from the original distance matrix (Clarke and Green 1988).

NMDS was conducted using the „*vegan*“ package (Oksanen et al. 2009) within Program R 2.12.1 (R Development Core Team 2010). I excluded species that individually contributed <1% to the total count (Preston and Harestad 2007, Chizinski et al. 2011). Ordination was performed using the metaMDS function and a Bray-Curtis dissimilarity matrix. Data was standardized using the Wisconsin double standardization method and square-root transformed to increase the relative importance of less abundant species. Multiple random starts (20/ordination) were performed in order to avoid becoming trapped in local minima (the iterative process stalling at a stress value that actually can be further reduced) (McCune and Grace 2002). NMDS was conducted in dimensions 2-6. The stress of NMDS ordinations was evaluated against their dimensionality via a screeplot to determine the appropriate dimensionality for display and statistical testing. Habitat type was overlaid and species were ordinated by their averaged weighted scores.

I used the adonis function (*vegan* package; Oksanen et al. 2009) to statistically assess the variation in assemblage structure attributable to forest type (mined and unmined), slope type (plateau and steep slope), and the interaction between forest type and slope class. This function performs a multivariate analysis of variance through partitioning of the sums of squares in a distance matrix in relation to a factor and using *F*-tests from permutations of the data to determine the level of statistical significance (Oksanen et al. 2009). I used the adonis method rather than analysis of similarities (ANOSIM) because the adonis method is generally considered more robust than ANOSIM (Oksanen et al. 2009). I used a Bray-Curtis dissimilarity matrix and specified that permutations (999) occurred within sites but not across sites (Oksanen et al. 2009).

A mean dissimilarity dendrogram was then used to graphically display mean between-group and within-group dissimilarity. To determine the appropriate linkage method, cophenetic correlations were calculated between the original Euclidian distance matrix and a Bray-Curtis distance matrix using both “average” and “single” linkage methods (Oksanen et al. 2009). I used the average linkage method because this linkage in combination with a Bray-Curtis metric produced a higher cophenetic correlation (0.22) than did the single linkage method (0.18).

Within the mean dissimilarity dendrogram, vertical lines indicate mean within-group dissimilarity; longer lines equate to lower mean dissimilarity (Oksanen et al. 2009). The horizontal line indicates mean between-group dissimilarity (Oksanen et al. 2009).

Using generalized linear modeling (GLM), I tested for differences in species richness, overall abundance, and abundance within foraging, nesting, and habitat guilds. Modeling was a function of forest type (mined and unmined), slope type (plateau and steep slope), the interaction between forest type and slope class, and site, all as fixed effects. I evaluated models for overdispersion using a Poisson GLM and an associated dispersion parameter. Based on the absence of overdispersion in all models, I determined a Poisson distribution was appropriate for these analyses (Zuur et al. 2009). Statistical significance was assessed via an analysis of deviance test in which the difference in deviance approximately follows a Chi-square distribution with 1 degree of freedom (Zuur et al. 2009). Foraging and nesting guilds were adapted from Ehrlich et al. (1988) and Canterbury et al. (2000) (Appendix C). I placed species into habitat guilds (closed canopy species, broken canopy species, and forest generalists) in the context of the canopy disturbance gradient that exists within the study area (contiguous forest with relatively small areas of broken canopy habitat). In addition, I tested for differences in Shannon (*H’*) diversity using a Gaussian distribution and the model specification previously described.

For analysis of habitat variables, vegetation sampling plots were classed as unmined plateau, loose-dumped bench, unmined steep slope, and compacted bench. I used a subset of plots associated with mined point counts that was comprised of only those plots that were wholly on the mine. I included all plots associated with unmined plateau and steep slope point count stations. I retained all variables for analysis with the exception of those pertaining to subcanopy characteristics (shrubs, saplings, downed logs) for which I had recorded both density and cover; I only included one or the other in all three cases. I graphically evaluated whether data for each habitat variable met assumptions of normality and variance homogeneity. Cover variables not meeting these assumptions were arcsine-transformed. Count data not meeting these assumptions were square root-transformed. I compared variables between the four habitat types using univariate analysis of variance with multiple comparisons using Tukey‟s HSD procedure. Using 0.05 as the global alpha level, the critical *p*-value was set at 0.0036 using the Bonferroni method.

RESULTS

Habitat summary. Compacted bench sites had characteristics indicative of disturbed forests including significantly greater canopy vine density and midcanopy foliage density relative to the three other habitat types (Table 1). Also, overstory foliage density was significantly lower than loose-dumped bench sites and unmined, steep slope sites. Within compacted bench sites, high midcanopy cover and low overstory cover suggests lower canopy height overall (Fig. 7).

Unmined plateau sites were characterized by comparatively greater subcanopy and midcanopy development. Shrub density was significantly higher than the other three habitat types and small tree (8-23 cm) density was significantly higher than loose-dumped bench sites and unmined, steep slope sites (Table 1). In contrast, loose-dumped bench sites had significantly lower subcanopy foliage density than unmined plateau sites and compacted bench sites (Fig. 7).

Within compacted bench and loose-dumped bench minelands, yellow poplar and red maple were predominant (Appendix G). Unmined plateau sites were dominated by red maple and non-mesic oaks: scarlet, white, chestnut, and black oak. Within unmined, steep slope sites, red maple, sugar maple, and northern red oak were predominant.

Avian assemblage structure. In evaluating the stress of NMDS ordinations versus their dimensionality, stress was not appreciably reduced for ordinations with greater than three dimensions. Thus, samples are plotted within the first two dimensions of the 3-dimensional NMDS solution (Fig. 8). Stress for the 3-dimensional NMDS solution was 20.6 (two convergent solutions after 6 runs).

The NMDS ordination showed overlap between assemblages associated with all four habitat types. Only compacted bench sites showed some discrimination from other habitat types, but there was not clear separation of these assemblages. Compacted bench assemblages were primarily clustered on the left side of the plot and were discriminated by Worm-eating Warbler (*Helmitheros vermivorus*), Rose-breasted Grosbeak (*Pheucticus ludovicianus*), and American Redstart (*Setophaga ruticilla*).

The mean dissimilarity dendogram (Fig. 9) indicated greater between-group dissimilarity than within-group dissimilarity for all habitat types. However, group structure was not particularly strong as evidenced by mean between-group dissimilarity (Bbar = 0.502) relative to mean within-group dissimilarity (Wbar = 0.453). Among the four assemblage types, compacted bench and unmined steep slope assemblages had the highest within-group dissimilarity.

The adonis analysis indicated that assemblage structure differed significantly between mined and unmined forest and between steep slope and plateau sites (Table 2). However, in modeling both of these effects, variation in within-group distances was overwhelming and forest and slope type accounted for only 6.0% and 8.8% of the variation in the avian assemblage, respectively. The interaction between forest type and slope type was marginally significant and accounted for only 2.6% of the variation in the avian assemblage.

Overall relative abundance, abundance within nesting and foraging guilds, species richness, and Shannon (*H'*) diversity were similar for mined and unmined forest (Table 3). Mined and unmined forest had similar abundance of broken canopy and forest generalist species (Fig. 10), but closed-canopy species had significantly greater abundance within unmined forest (Table 3). The interaction between forest type and slope type was non-significant for all analyses.

DISCUSSION

The presence of pre-SMCRA minelands with loose-dumped benches was not a strong influence on avian assemblage structure. Only compacted bench minelands had a relatively distinct avian assemblage. However, within the NMDS ordination, this assemblage was not discrete, suggesting that these minelands modify the avian assemblage on a relatively narrow scale. Dense midcanopy cover and relatively low levels of overstory cover characterized compacted bench minelands. These characteristics result in part from poor physical and chemical properties of minesoils which have arrested forest development along benches. Canopy vine density, primarily grapevine (*Vitis* spp.), was substantially higher within compacted bench sites and suggested that forest development also may be restricted through vine-capture of the habitat.

In addition to the structure of this restricted forest and vine community, dense understory vegetation and canopy foliage along the highwall edge and the young forest habitat that originated through highwall collapse were likely influential in structuring avian assemblages along compacted benches.

Species which use the subcanopy and midcanopy for nesting and foraging, American Redstart, Rose-breasted Grosbeak, and Worm-eating Warbler, differentiated the avian assemblage associated with compacted bench minelands. American Redstart and Rose-breasted Grosbeak are generally most abundant within mesic, second growth forest in which some shrubby vegetation is present (Wyatt and Francis 2002, Sherry and Holmes 1997). Worm-eating Warblers are associated with dense understory vegetation usually along steep slopes (Hanners and Patton 1998).

Relative abundance of the closed-canopy guild was significantly higher within unmined forest. For some mature forest-restricted species, i.e. Blue-headed Vireo and Ovenbird, mineland habitats may have low suitability due to relatively low subcanopy cover within loose-dumped bench sites and relatively low overstory cover and canopy height along compacted benches. Blue-headed Vireos are generally associated with high canopy closure, usually >75%, but also use the subcanopy extensively for foraging and nesting (Hamel 1992, Meehan 1996, James 1998). In addition, mineland forests may represent suboptimal breeding habitat for Ovenbirds, a ground-nesting and foraging species, due to relatively low leaf litter depth and cover and high forb cover (Mizel, unpublished data).

Previous research has illustrated the significant effect that large-scale surface mining has on mature forest songbirds, Cerulean Warblers (*Dendroica cerulea*) in particular (Wood et al. 2006). Few studies have investigated the influence of forested pre-SMCRA surface mines on the avian assemblage. This study has shown that compacted bench minelands created significant habitat contrast with the surrounding, unmined forest and consequently resulted in a relatively distinct avian assemblage.

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Table 1. Means (±SE) for habitat characteristics associated with point counts conducted in 2010 in southern West Virginia. Plots sampled within oak-hickory forest atop the New River Plateau were categorized as loose-dumped benches (n=33) and unmined plateau (n=51). Plots sampled within mixed-mesophytic forest at steep slope sites are categorized as compacted benches (n=20) and unmined, steep slope (n=51). The critical *p*-value was set at 0.0036 using the Bonferroni adjustment. Means that do not share an uppercase letter are significantly different ($p < 0.05$; Tukey's multiple comparison procedure).

	Loose-dumped benches (plateau)		Unmined (plateau)		Compacted benches (steep slope)		Unmined (steep slope)			
Habitat characteristic	mean	SE	mean	SE	mean	SE	mean	SE	F	*p*
Subcanopy										
shrub stems ha^{-1}	488.64	(189.05) AB	590.20	(98.83) B	178.75	(69.00) A	324.51	(55.06) A	4.80	**0.0032**
saplings ha^{-1}	81.06	(15.99)	127.94	(15.09)	105.00	(27.54)	103.92	(15.68)	1.28	0.28
Stocking										
small trees ha^{-1} (8-22.9 cm)	260.61	(30.21) A	370.59	(25.80) B	270.00	(29.05) AB	231.86	(14.85) A	7.94	**<0.001**
med. trees ha^{-1} (23-37.9 cm)	146.97	(11.86) A	142.16	(7.87) A	135.53	(18.19) AB	90.69	(6.93) B	8.47	**<0.001**
large trees ha^{-1} (>38 cm)	87.12	(9.49)	87.75	(5.96)	70.00	(8.03)	112.25	(7.19)	4.55	0.004
total trees ha^{-1} (>8 cm)	494.70	(33.28) A	600.49	(25.61) B	468.75	(34.93) A	434.80	(18.71) A	8.86	**<0.001**
basal area m^{2} ha^{-1}	31.44	(2.10)	32.20	(1.15)	26.71	(2.25)	36.87	(2.03)	4.11	0.008
snags ha^{-1}	46.21	(8.29) AB	29.90	(3.84) A	81.25	(14.72) B	27.45	(5.34) A	7.79	**<0.001**
Canopy cover										
subcanopy foliage (%)	18.64	(1.76) B	30.10	(2.09) A	31.25	(3.34) A	22.89	(2.15) AB	6.02	**<0.001**
midstory foliage (%)	39.62	(2.37) A	44.02	(1.89) A	58.63	(4.15) B	43.43	(2.24) A	7.08	**<0.001**
overstory foliage (%)	38.79	(2.79) AC	33.63	(2.49) AB	25.75	(4.05) B	45.78	(2.23) C	8.29	**<0.001**
vertical foliage density (%)	32.35	(1.15)	35.92	(1.40)	38.54	(2.12)	37.37	(1.12)	2.98	0.03
Miscellaneous										
canopy vines ha^{-1}	102.27	(22.81) B	3.43	(1.40) C	263.75	(51.79) D	53.43	(10.80) AB	30.81	**<0.001**
downed log cover (%)	8.79	(1.58) A	3.12	(0.52) B	5.10	(1.86) AB	4.64	(0.81) A	5.17	**0.002**

Table 2. Results from adonis analysis of avian assemblage similarity. Partial R^2 and *p*-values are based on 999 permutations of the data.

	$F_{1,61}$	R^2	*p*
Forest type	4.44	0.060	0.001
Slope	6.46	0.088	0.001
Forest type x slope	1.91	0.026	0.02

Table 3. Mean (±SE) relative abundance for all species and habitat, foraging, and nesting guilds at mined and unmined point counts conducted in 2010 in southern West Virginia. Except for Shannon diversity (*H'*), *p*-values are from Poisson GLMs using an analysis of deviance test in which the difference in deviance approximately follows a Chi-square distribution with 1 degree of freedom. Shannon diversity (*H'*) was tested with analysis of variance (* indicates the value of the F-statistic).

	Mined		Unmined			
	mean	SE	mean	SE	χ^2_1	*p*
Diversity (*H'*)	2.26	0.04	2.18	0.05	0.28*	0.60
Richness	11.06	0.42	10.56	0.53	0.03	0.87
All species	16.58	0.59	15.91	0.64	0.00	1.00
Habitat guilds						
closed canopy	3.29	0.25	4.76	0.36	6.88	0.009
broken canopy	2.81	0.46	1.85	0.32	0.82	0.37
generalists	10.19	0.45	9.12	0.39	2.04	0.15
Foraging guilds						
hawking	0.97	0.18	1.12	0.16	0.23	0.63
bark foragers	1.74	0.17	2.41	0.20	2.94	0.09
foliage gleaners	9.52	0.44	8.41	0.49	2.10	0.15
ground gleaners	3.81	0.30	3.59	0.29	0.20	0.66
Nesting guilds						
subcanopy	3.16	0.47	2.03	0.26	1.00	0.32
ground	3.90	0.30	4.00	0.22	0.85	0.36
shrub	1.13	0.17	0.59	0.15	1.33	0.25
cavity	1.32	0.20	2.03	0.20	3.09	0.08
canopy	6.29	0.33	6.88	0.35	4.02	0.60

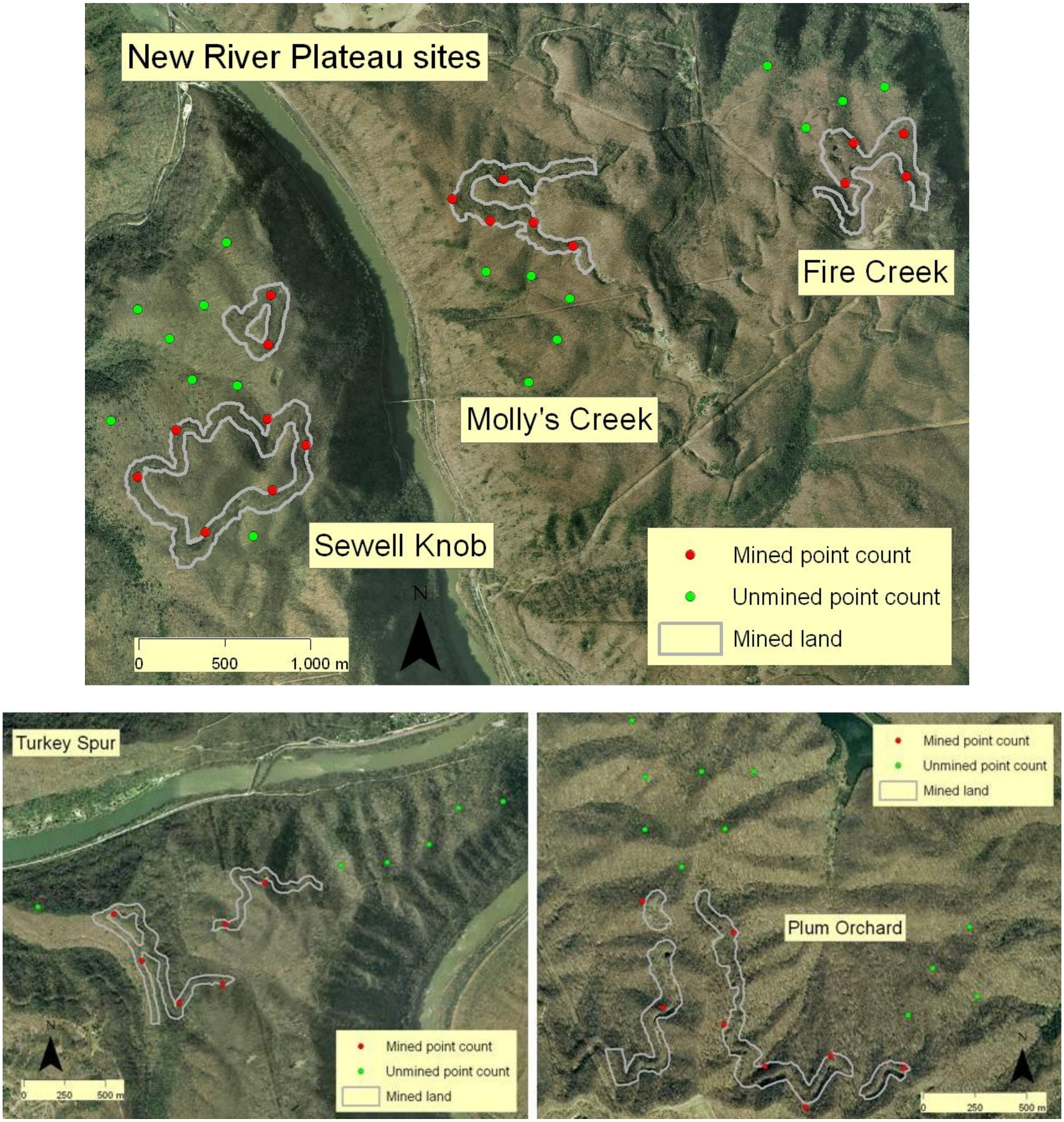


Figure 1. Location of point count stations surveyed in 2010 in southern West Virginia. Plateau study sites (unmined and loose-dumped bench transects) are shown in the top frame and steep slope sites (unmined and compacted bench transects) are shown in the bottom two frames.

Figure 2. Spoil ridges atop the Fire Creek study site.

Figure 3. Spoil ridge along Sewell Knob study site.

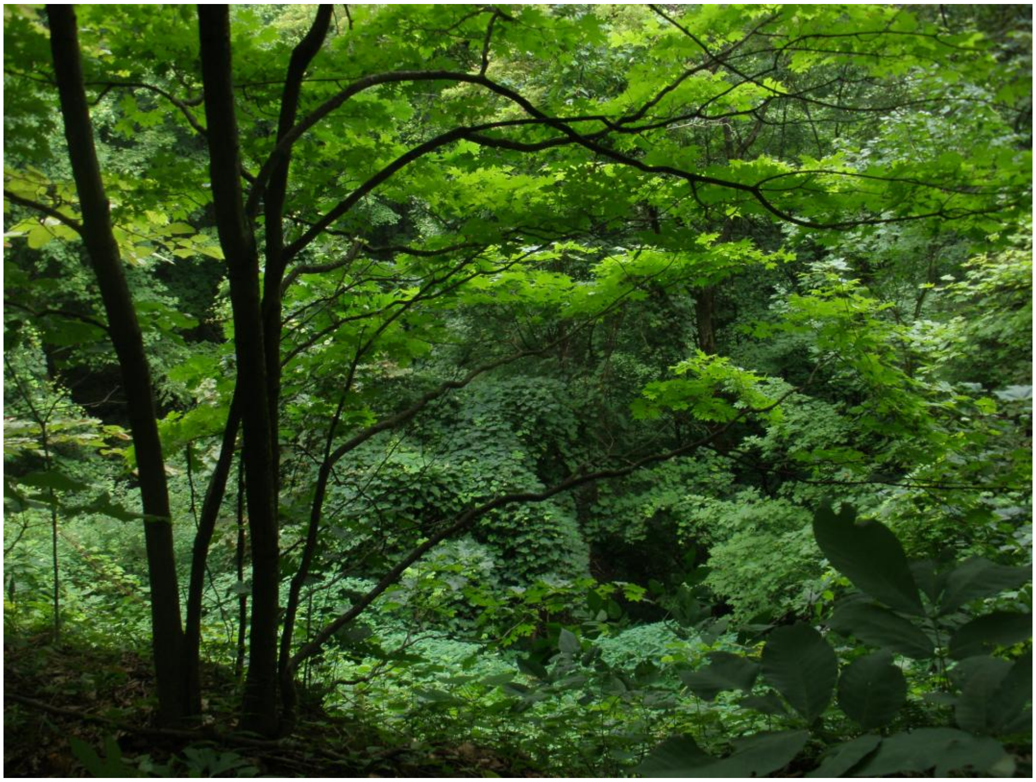


Figure 4. Vine-captured gap at the base of a highwall along a compacted bench study site in Plum Orchard WMA.

Figure 5. Compacted bench in New River Gorge National River (Turkey Spur study site).

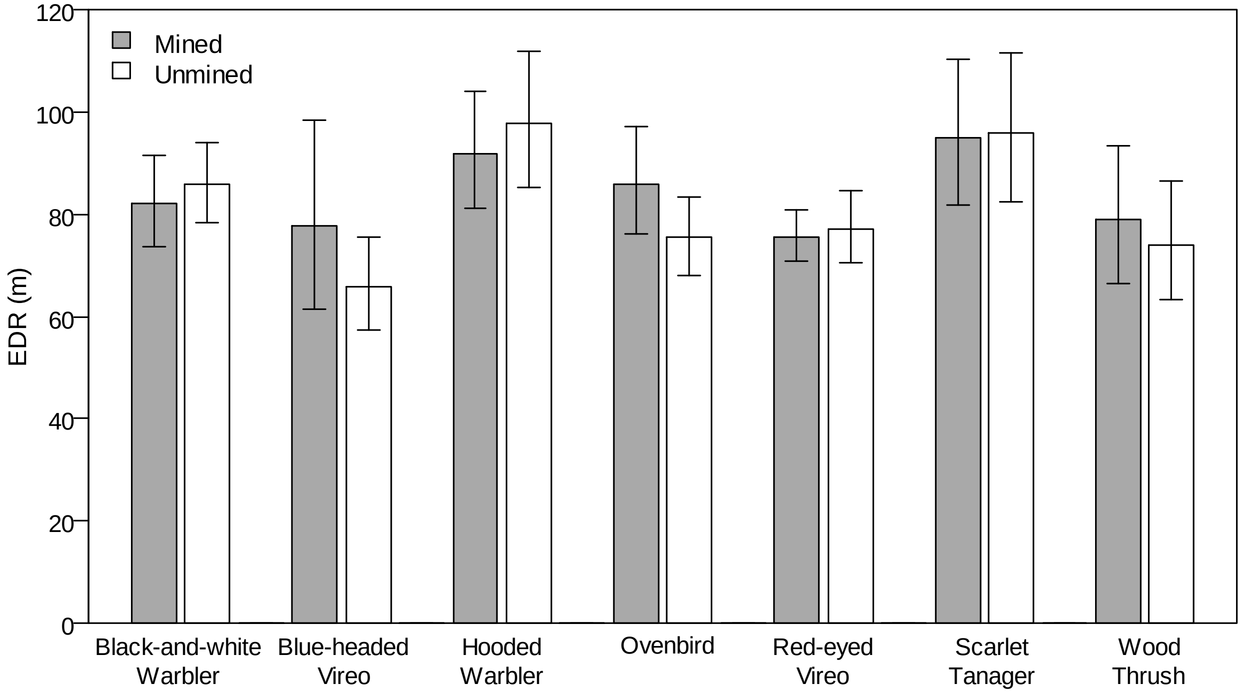


Figure 6. Effective detection radii (EDR) with 95% confidence intervals for Black-and-white Warbler, Blue-headed Vireo, Hooded Warbler, Ovenbird, Red-eyed Vireo, Scarlet Tanager, and Wood. Estimates are for mined (gray bars) and umined (white bars) point counts. Depending on the function used in the detection model, the EDR estimates the distance at which the probability of detection declines below 0.5 (Laake et al. 1993, Simons et al. 2006).

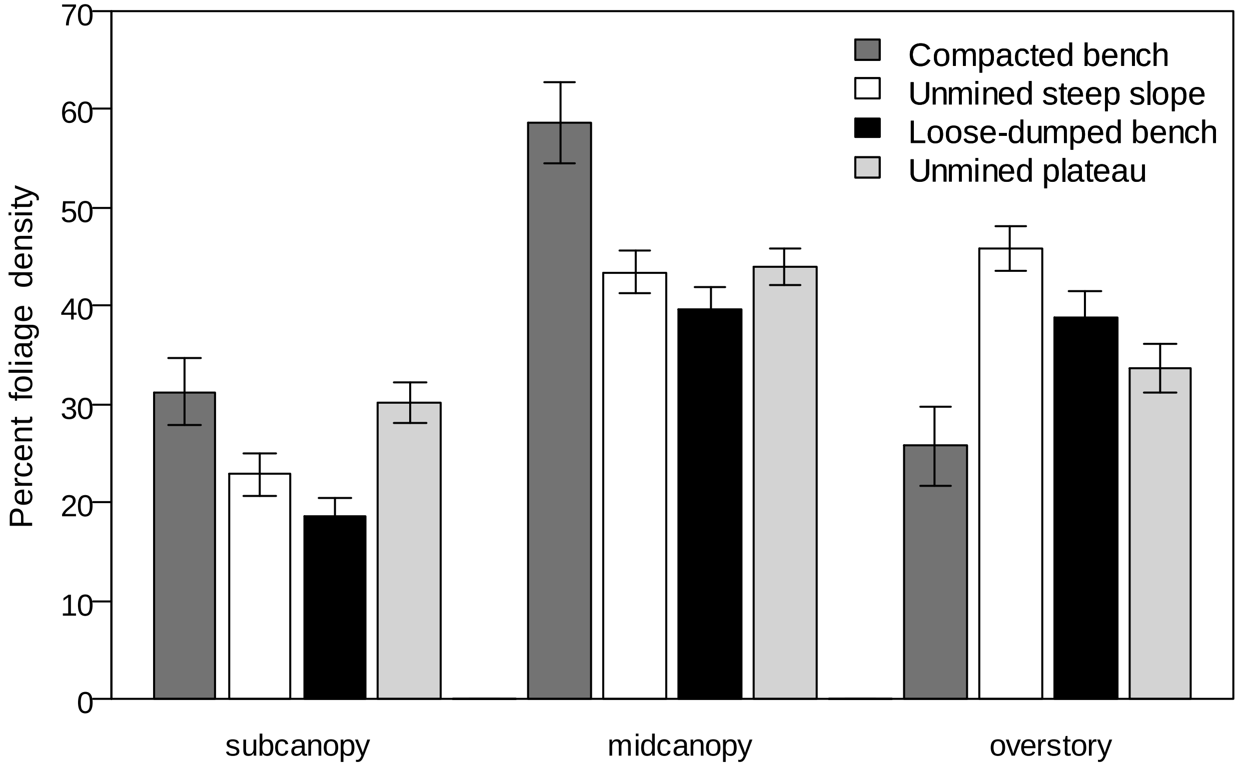


Figure 7. Percent vertical foliage density within the subcanopy (0-6 m), midcanopy (6-18 m), and overstory (>18 m) for compacted bench sites, unmined steep slope sites, loose-dumped bench sites, and unmined plateau sites.

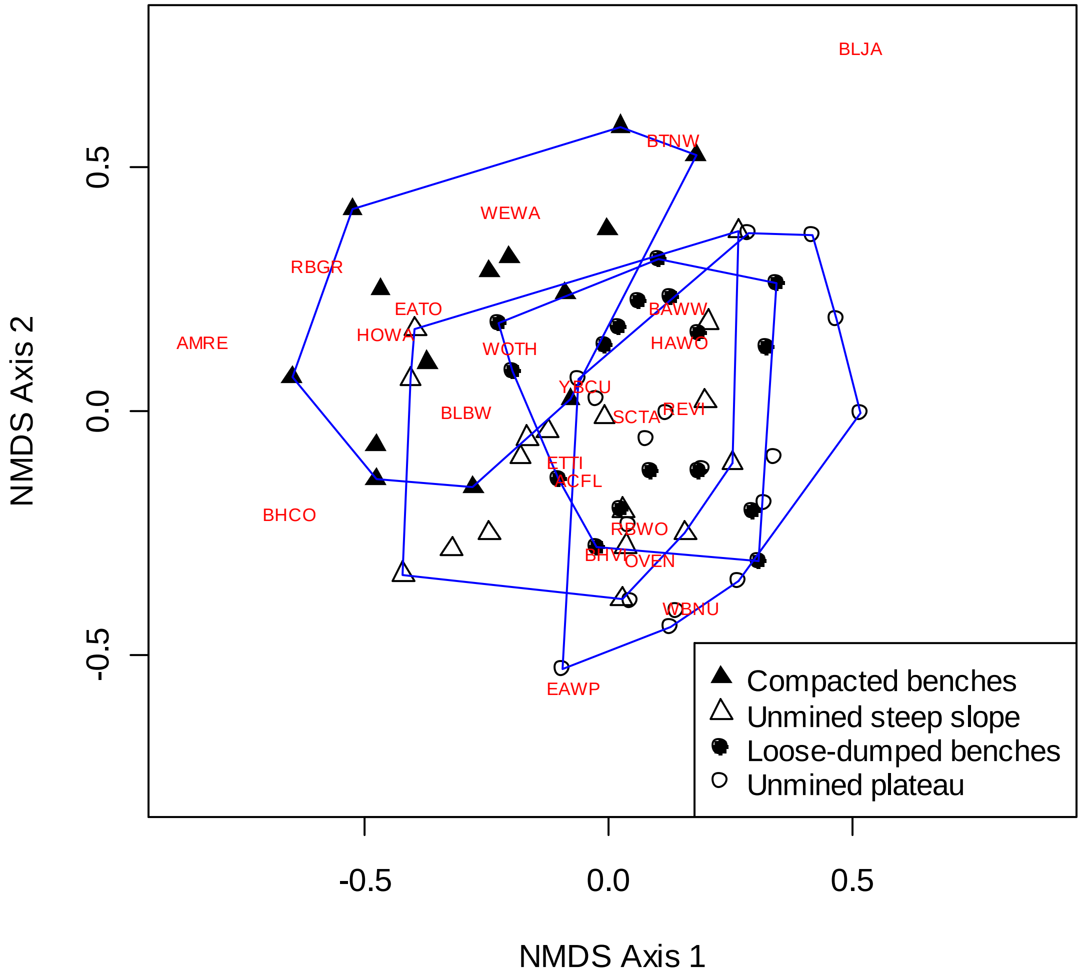


Figure 8. Dimensions 1 and 2 from a 3-dimensional NMDS solution for avian assemblages in mined (compacted and loose-dumped benches) and unmined forest. Stress was 20.6 for the 3-dimensional solution (2 convergent solutions after 6 runs). Species codes are listed in Appendix C.

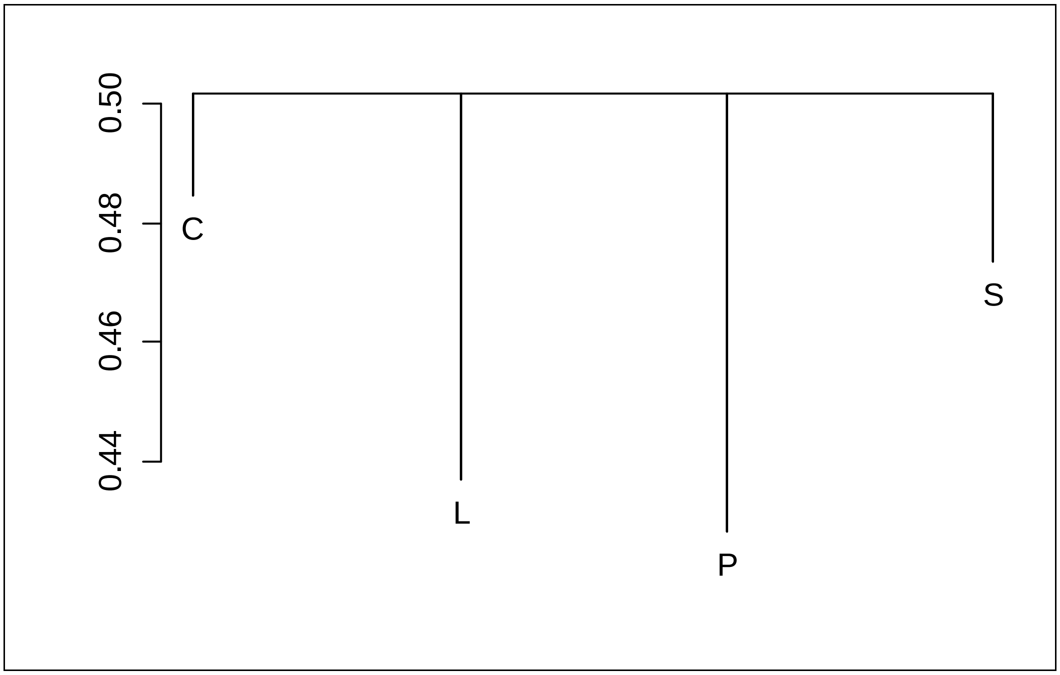


Figure 9. Mean dissimilarity dendrogram for compacted bench assemblages (C), loose-dumped bench assemblages (L), unmined plateau assemblages (P), and unmined steep slope assemblages (S). Mean between-group dissimilarity (Bbar) = 0.502. Mean within-group dissimilarity (Wbar) = 0.453.

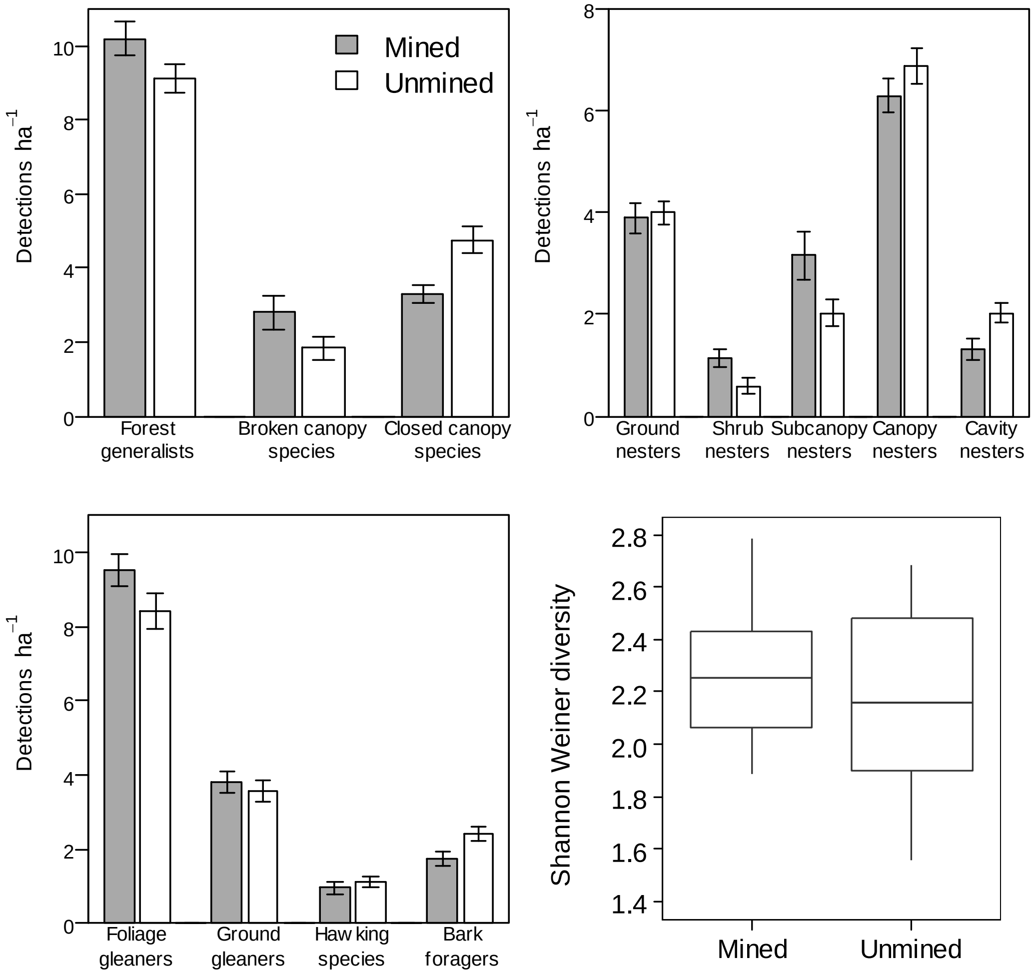
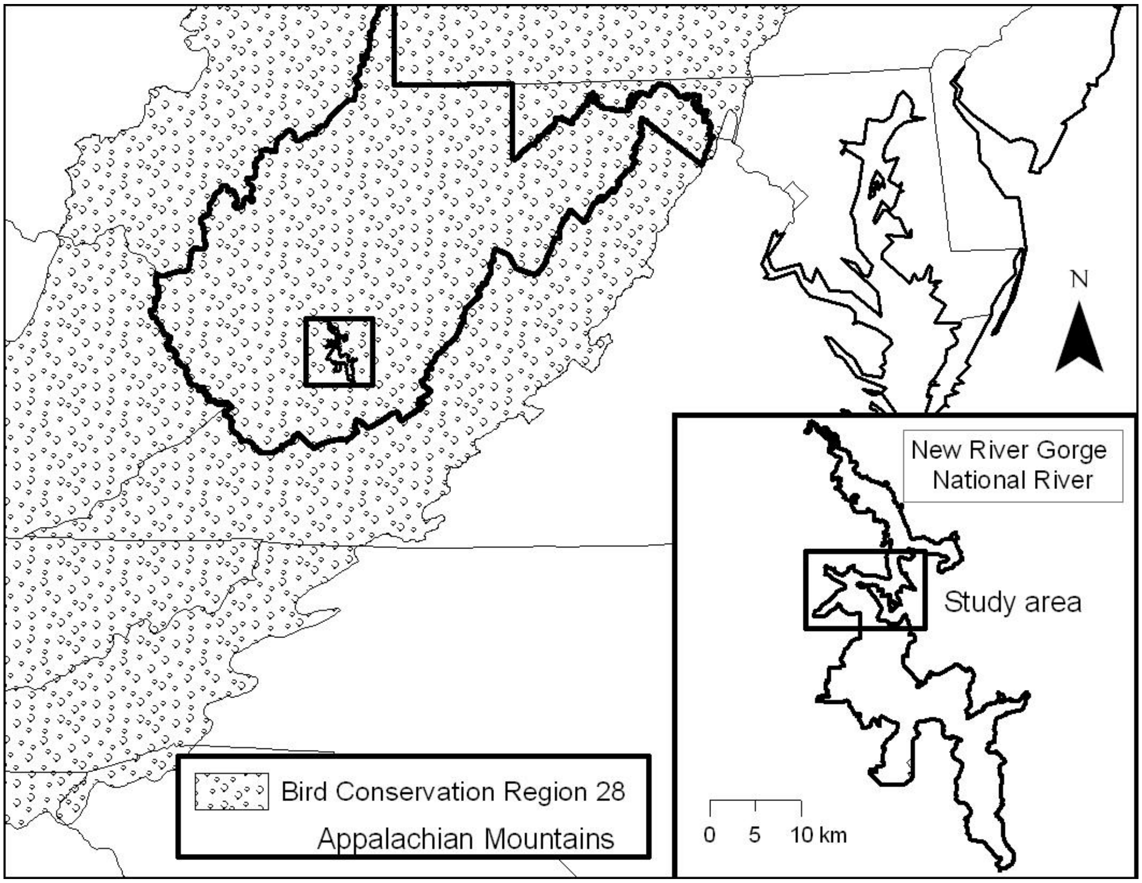
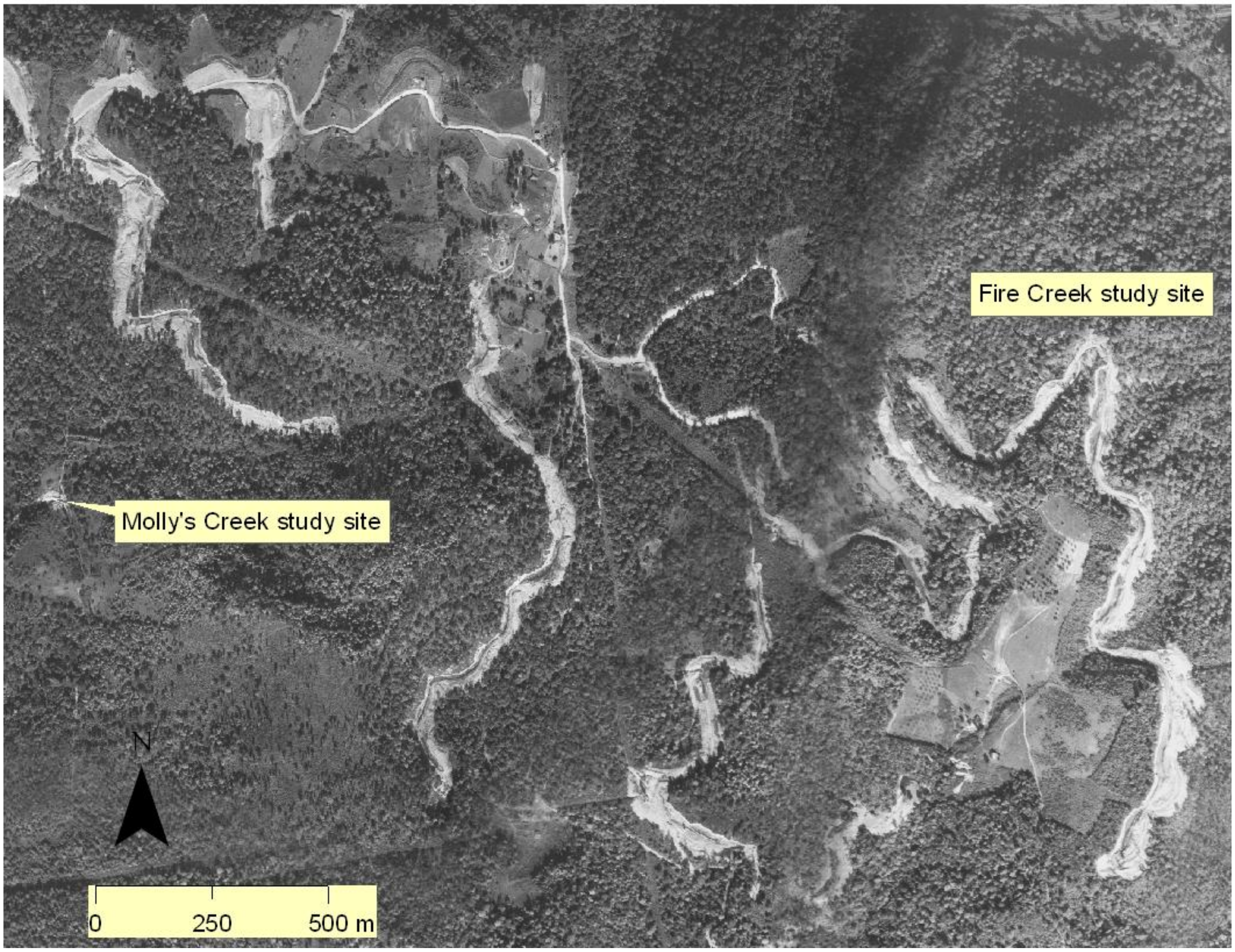


Figure 10. Relative abundance (± SE) for habitat, nesting, and foraging guilds for mined (gray) and unmined (white) point counts. The boxplot for Shannon diversity (*H'*) is also shown.

Appendix A. Study area location for line transect surveys and Red-eyed Vireo nest monitoring in New River Gorge National River, West Virginia in 2011.



Appendix B. Aerial photograph (1945) showing early surface mining atop the New River Plateau. Initial mining at two study sites (2010-2011) is visible on the photograph.



Appendix C. Breeding bird species detected during 2010-2011 surveys.

Common Name	Code	Scientific Name	Foraging	Nesting	Habitat
Acadian Flycatcher	ACFL	*Empidonax virescens*	HK	SC	CC
American Redstart	AMRE	*Setophaga ruticilla*	FG	SC	BC
American Robin	AMRO	*Turdus migratorius*	GG	SC	Gen
Bk-throated Green Warbler	BTNW	*Setophaga virens*	FG	CA	CC
Black-and-white Warbler	BAWW	*Mniotilta varia*	BF	GR	Gen
Blackburnian Warbler	BLBW	*Setophaga fusca*	FG	CA	Gen
Blue Jay	BLJA	*Cyanocitta cristata*			Gen
Blue-gray Gnatcatcher	BGGN	*Polioptila caerulea*	FG	CA	Gen
Blue-headed Vireo	BHVI	*Vireo solitarius*	FG	CA	CC
Brown-headed Cowbird	BHCO	*Molothrus ater*			
Carolina Chickadee	CACH	*Poecile carolinensis*	FG	CV	Gen
Carolina Wren	CAWR	*Thryothorus ludovicianus*	GG	CV	BC
Cerulean Warbler	CERW	*Setophaga cerulea*	FG	CA	BC
Downy Woodpecker	DOWO	*Picoides pubescens*	BF	CV	Gen
Eastern Phoebe	EAPH	*Sayornis phoebe*	HK		Gen
Eastern Towhee	EATO	*Pipilo erythrophthalmus*	GG	GR	BC
Eastern Tufted Titmouse	ETTI	*Baelophus bicolor*	FG	CV	Gen
Eastern Wood Pewee	EAWP	*Contopus virens*	HK	CA	BC
Great Crested Flycatcher	GCFL	*Myiarchus crinitus*	HK	CV	Gen
Hairy Woodpecker	HAWO	*Picoides villosus*	BF	CV	Gen
Hooded Warbler	HOWA	*Wilsonia citrina*	FG	SH	BC
Indigo Bunting	INBU	*Passerina cyanea*	FG	SH	BC
Kentucky Warbler	KEWA	*Oporornis formosus*	GG	GR	BC
Louisiana Waterthrush	LOWA	*Seiurus motacilla*		GR	Gen
Northern Cardinal	NOCA	*Cardinalis cardinalis*	FG	SH	BC
Northern Parula	NOPA	*Parula americana*	FG	CA	Gen
Ovenbird	OVEN	*Seiurus aurocapillus*	GG	GR	CC
Pileated Woodpecker	PIWO	*Dryocopus pileatus*	BF	CV	CC
Pine Warbler	PIWA	*Setophaga pinus*	BF	CA	Gen
Red-bellied Woodpecker	RBWO	*Melanerpes carolinus*	BF	CV	Gen
Red-eyed Vireo	REVI	*Vireo olivaceus*	FG	CA	Gen
Rose-breasted Grosbeak	RBGR	*Pheucticus ludovicianus*	FG	SC	Gen
Scarlet Tanager	SCTA	*Piranga olivacea*	FG	CA	Gen
White-breasted Nuthatch	WBNU	*Sitta carolinensis*	BF	CV	Gen
Winter Wren	WIWR	*Troglodytes troglodytes*	GG	CV	CC
Wood Thrush	WOTH	*Hylocichla mustelina*	GG	SC	Gen
Worm-eating Warbler	WEWA	*Helmitheros vermivorus*	FG	GR	Gen
Yellow-billed Cuckoo	YBCU	*Coccyzus americanus*	FG	SC	Gen
Yellow-throated Vireo	YTVI	*Vireo flavifrons*	FG	CA	Gen

Foraging guilds: HK (hawking), BF (bark forager), GG (ground gleaner), and FG (foliage gleaner). Nesting guilds: GR (ground), SH (shrub), SC (subcanopy), CA (canopy), CV (cavity). Habitat guilds: Gen (forest generalist), BC (broken canopy), CC (closed canopy).

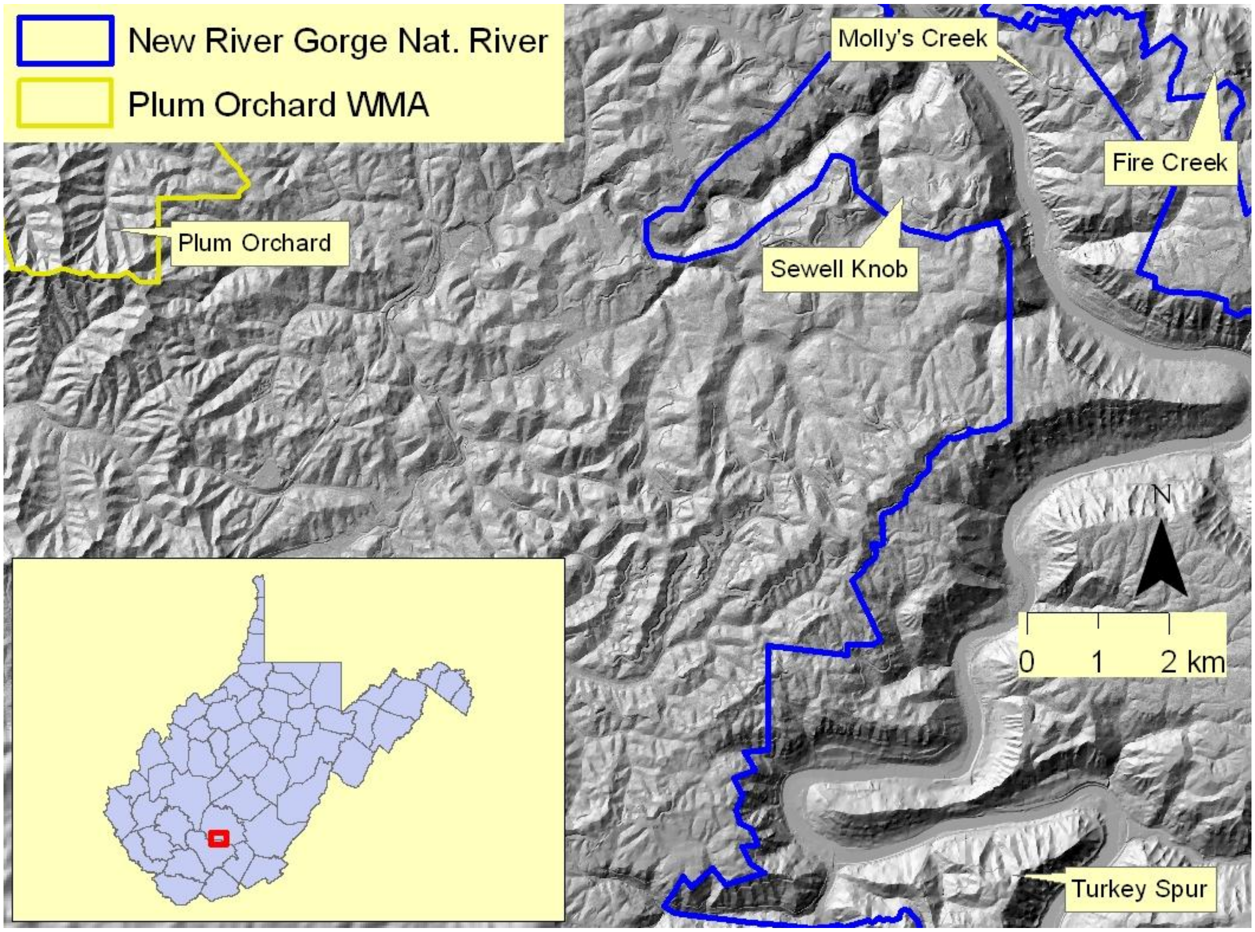
Appendix D. Relative importance values for tree species from reference (n=56) and mineland (n=56) forest plots sampled in New River Gorge NR, West Virginia in 2011.

Mineland forest (n=56)		Reference forest (n=56)	
Species	Relative importance	Species	Relative importance
Yellow poplar	0.973	White oak	0.529
Red maple	0.795	Red maple	0.525
Black birch	0.389	Hickory spp.	0.249
Black locust	0.126	Chestnut oak	0.245
Sourwood	0.105	Black oak	0.237
Black gum	0.092	Scarlet oak	0.234
Princess tree	0.062	Yellow poplar	0.175
Slippery elm	0.060	American beech	0.152
Bigtooth aspen	0.057	Sourwood	0.149
Black cherry	0.045	N. red oak	0.105
White ash	0.040	Black gum	0.078
River birch	0.038	Cucumber magnolia	0.074
Sugar maple	0.037	Sugar maple	0.070
Cucumber magnolia	0.020	Black birch	0.046
Sycamore	0.020	Black cherry	0.031
Sassafrass	0.020	Eastern hemlock	0.025
Pitch pine	0.019	Hop hornbeam	0.014
Virginia pine	0.016	Sassafrass	0.014
N. red oak	0.016	White ash	0.011
Box elder	0.013	Basswood	0.008
American beech	0.011	Fraser magnolia	0.008
Striped maple	0.009	Musclewood	0.007
Black walnut	0.007	Flowering dogwood	0.007
Black oak	0.005	Yellow buckeye	0.006
Silver maple	0.005		
Flowering dogwood	0.005		
Prunus spp.	0.005		
Musclewood	0.005		
Wild crabapple	0.005		

Appendix E. Detections ha^{-1} and percent occurrence for bird species detected along line transects in 2011 in mineland and reference forest in New River Gorge NR, West Virginia.

	Mined (n=14)			Reference (n=14)		
	Percent occurrence	Detections ha^{-1} mean	SE	Percent occurrence	Detections ha^{-1} mean	SE
Acadian Flycatcher	7	0.04	0.04	36	0.17	0.07
American Redstart	21	0.10	0.05	0		
American Crow	0			7	0.03	0.03
Black-and-white Warbler	57	0.26	0.06	79	0.37	0.07
Blue-gray Gnatcatcher	7	0.06	0.06	14	0.06	0.04
Brown-headed Cowbird	0			7	0.03	0.03
Blue-headed Vireo	7	0.03	0.03	64	0.37	0.11
Blue Jay	21	0.09	0.05	7	0.03	0.03
Blackburnian Warbler	14	0.10	0.07	36	0.17	0.07
Bk-thr. Green Warbler	50	0.23	0.06	7	0.03	0.03
Carolina Chickadee	43	0.23	0.08	14	0.06	0.04
Carolina Wren	7	0.03	0.03	0		
Cerulean Warbler	14	0.06	0.04	0		
Downy Woodpecker	14	0.07	0.05	14	0.06	0.04
Eastern Phoebe	7	0.03	0.03	0		
Eastern Towhee	21	0.10	0.05	0		
Eastern Tufted Titmouse	36	0.16	0.06	7	0.03	0.03
Eastern Wood-Pewee	14	0.06	0.04	64	0.26	0.05
Great-crested Flycatcher	7	0.03	0.03	29	0.11	0.05
Hairy Woodpecker	29	0.13	0.06	50	0.23	0.07
Hooded Warbler	21	0.13	0.07	14	0.06	0.04
Indigo Bunting	14	0.06	0.04	0		
Louisiana Waterthrush	14	0.07	0.05	0		
Northern Cardinal	7	0.03	0.03	0		
Ovenbird	93	0.65	0.11	100	1.46	0.12
Pileated Woodpecker	14	0.07	0.05	14	0.06	0.04
Red-bellied Woodpecker	29	0.13	0.06	36	0.14	0.05
Red-eyed Vireo	100	2.48	0.18	100	2.31	0.19
Scarlet Tanager	93	0.81	0.12	100	0.68	0.08
White-breasted Nuthatch	21	0.09	0.05	50	0.20	0.06
Worm-eating Warbler	36	0.20	0.08	14	0.09	0.06
Wood Thrush	14	0.06	0.04	36	0.14	0.05
Yellow-billed Cuckoo	36	0.20	0.08	57	0.29	0.09
Yellow-throated Vireo	21	0.10	0.05	29	0.11	0.05

Appendix F. Location of study sites within Plum Orchard Wildlife Management Area (WMA) and New River Gorge National River in southern West Virginia (2010-2011).



Appendix G. Relative importance values for tree species from plots categorized as loose-dumped benches (n=33), unmined plateau forest (n=51), compacted benches (n=20), and unmined, steep slope forest (n=51). Only species with relative importance ≥0.05 are shown. Plots were sampled in southern West Virginia in 2010.

Loose-dumped benches (plateau)		Unmined (plateau)		Compacted benches (steep slope)		Unmined (steep slope)	
Species	Rel. importance	Species	Rel. importance	Species	Rel. importance	Species	Rel. importance
Yellow poplar	1.053	Red maple	0.568	Yellow poplar	0.792	Red maple	0.404
Red maple	0.680	Scarlet oak	0.412	Red maple	0.553	Sugar maple	0.343
Black birch	0.349	White oak	0.338	Sugar maple	0.427	N. red oak	0.331
Sourwood	0.171	Chestnut oak	0.310	Black birch	0.227	Chestnut oak	0.288
Black gum	0.124	Yellow poplar	0.230	Black locust	0.188	Yellow poplar	0.255
Princess tree	0.092	Sourwood	0.210	N. red oak	0.128	Hickory spp.	0.224
Bigtooth aspen	0.057	Black oak	0.176	White ash	0.101	Basswood	0.158
Black cherry	0.050	Hickory spp.	0.141	Princess tree	0.058	Black gum	0.145
		American beech	0.128	Chestnut oak	0.057	American beech	0.106
		Black gum	0.125	Hickory spp.	0.056	Sourwood	0.105
		Cucumber magnolia	0.065			Cucumber magnolia	0.091
		Black birch	0.062			Black oak	0.089
		Sugar maple	0.056			Black birch	0.065
						Scarlet oak	0.064

Appendix H. Percent occurrence and relative abundance (±SE) for bird species detected during point count surveys conducted in New River Gorge NR and Plum Orchard WMA, southern West Virginia, in 2010. Plateau surveys were conducted atop the New River Plateau in unmined, oak-hickory forest (n=20) and adjacent forest bisected by loose-dumped minelands (n=17). Steep slope surveys were conducted in unmined, mixed-mesophytic forest (n=14) and in adjacent forest bisected by compacted minelands (n=14).

	Loose-dumped minelands (plateau)			Reference (plateau)			Compacted minelands (steep slope)			Reference (steep slope)		
	% occurrence	mean	SE	% occurrence	mean	SE	% occurrence	mean	SE	% occurrence	mean	SE
Acadian Flycatcher	29	0.35	0.15	47	0.53	0.15	36	0.50	0.20	41	0.41	0.12
American Redstart	0			0			57	1.57	0.50	59	0.88	0.21
American Robin	0			0			29	0.36	0.17	0		
Bk-thr. Green Warbler	18	0.18	0.10	12	0.12	0.08	29	0.29	0.13	24	0.24	0.11
Black-and-white Warbler	71	0.76	0.14	82	1.06	0.16	86	1.00	0.15	71	0.82	0.15
Blackburnian Warbler	35	0.47	0.17	0			7	0.07	0.07	29	0.29	0.11
Blue Jay	29	0.29	0.11	35	0.35	0.12	29	0.29	0.13	12	0.12	0.08
Blue-gray Gnatcatcher	6	0.06	0.06	6	0.06	0.06	0			18	0.18	0.10
Blue-headed Vireo	29	0.53	0.21	82	1.29	0.19	57	0.64	0.17	76	1.24	0.26
Brown-headed Cowbird	12	0.12	0.08	6	0.06	0.06	43	0.43	0.14	18	0.18	0.10
Carolina Chickadee	24	0.24	0.11	12	0.12	0.08	7	0.07	0.07	18	0.18	0.10
Carolina Wren	6	0.06	0.06	12	0.12	0.08	0			0		
Cerulean Warbler	12	0.12	0.08	0			29	0.29	0.13	24	0.24	0.11
Downy Woodpecker	0			6	0.06	0.06	7	0.07	0.07	12	0.12	0.08
Eastern Phoebe	35	0.35	0.12	0			7	0.07	0.07	0		
Eastern Towhee	29	0.35	0.15	18	0.18	0.09	50	0.57	0.17	0		
Eastern Tufted Titmouse	12	0.12	0.08	29	0.29	0.11	29	0.29	0.13	29	0.29	0.11
Eastern Wood Pewee	29	0.29	0.11	53	0.53	0.12	14	0.21	0.15	53	0.59	0.15
Great Crested Flycatcher	6	0.06	0.06	18	0.18	0.09	7	0.07	0.07	0		
Hairy Woodpecker	12	0.12	0.08	47	0.53	0.15	29	0.36	0.17	35	0.35	0.12
Hooded Warbler	53	0.65	0.17	24	0.35	0.17	79	1.36	0.27	47	0.76	0.24
Indigo Bunting	6	0.06	0.06	6	0.06	0.06	0			0		

Appendix H. continued

	Loose-dumped minelands (plateau)			Unmined (plateau)			Compacted minelands (steep slope)			Unmined (steep slope)		
	% occurrence	mean	SE	% occurrence	mean	SE	% occurrence	mean	SE	% occurrence	mean	SE
Kentucky Warbler	0			0			14	0.14	0.10	12	0.12	0.08
Louisiana Waterthrush	0			6	0.06	0.06	0			0		
Northern Cardinal	0			0			29	0.29	0.13	0		
Northern Parula	6	0.06	0.06	0			0			0		
Ovenbird	100	2.18	0.21	100	3.18	0.41	86	1.50	0.25	100	2.24	0.20
Pileated Woodpecker	24	0.24	0.11	24	0.24	0.11	7	0.07	0.07	6	0.06	0.06
Pine Warbler	6	0.06	0.06	6	0.06	0.06	0			0		
Red-bellied Woodpecker	35	0.35	0.12	47	0.47	0.12	7	0.07	0.07	41	0.41	0.12
Red-eyed Vireo	100	3.59	0.23	100	3.24	0.22	100	2.50	0.23	100	2.82	0.30
Rose-breasted Grosbeak	0			0			71	0.86	0.18	29	0.29	0.11
Scarlet Tanager	100	1.82	0.15	94	1.35	0.17	93	1.21	0.19	82	1.24	0.18
White-breasted Nuthatch	29	0.29	0.11	29	0.29	0.11	7	0.07	0.07	35	0.35	0.12
Winter Wren	0			0			7	0.07	0.07	0		
Wood Thrush	71	1.00	0.19	35	0.41	0.15	93	1.43	0.23	82	0.94	0.13
Worm-eating Warbler	47	0.59	0.17	12	0.12	0.08	64	0.71	0.16	24	0.29	0.14
Yellow-billed Cuckoo	29	0.29	0.11	29	0.29	0.11	29	0.29	0.13	29	0.29	0.11
Yellow-throated Vireo	0			6	0.06	0.06	0			24	0.24	0.11