

Avian population and community dynamics in response to vegetation restoration on  
reclaimed mine lands in southwest Virginia

Christopher Edward Latimer

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Dean F. Stauffer, Committee Chair  
Sarah M. Karpanty  
Marcella J. Kelly

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## **ABSTRACT**

Coal surface-mining is often implicated for its negative impacts on native flora and fauna. However, some studies suggest that, in reclaiming land after mining, there may be potential to create early successional habitat needed by many avian species currently in decline throughout eastern North America. I evaluated nest-site selection and nest success for on reclaimed mine lands in southwest Virginia during the summers of 2010 and 2011. For this nest-site analysis, I focused on 2 bird species common to reclaimed mine lands in southwest Virginia: field sparrows (*Spizella pusillia*) and indigo buntings (*Passerina cyanea*). In addition, I assessed bird community attributes in relation to various surface-mine reclamation regimens over a 5-year period from 2007-2011.

For both species, I found estimates of daily nest success to be higher than other estimates reported in the literature; however, empirical estimates of adult and juvenile survival are needed to provide better estimates of population status. For field sparrows, models of avian nest success support the hypothesis that a tradeoff exists between nest concealment and a view of the surroundings for field sparrows. For indigo buntings, year explained the most variation in nest success, with much lower estimates of daily nest survival in 2010, possibly as a result of increased precipitation.

I also assessed avian community dynamics in relation to vegetation changes on reclaimed mine sites and observed a total of 96 species throughout the 4 years of sampling. Local species persistence and species turnover were comparable to another

continental scale study conducted using breeding bird survey (BBS) data. Observed changes in community vital rates were likely a result of changes in certain habitat attributes over the 5-year period. Lastly, seven species were unique to certain cover types, suggesting the need to consider landscape level processes when developing restoration guidelines for reclaimed coal surface-mines.

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## **Introduction**

### *Coal mining*

Coal is an essential energy source throughout the United States and the world (OSM 2012). In 2010, the U.S. produced ~1.1 billion short tons (1 short ton = approximately 2000 lbs) of coal by surface mining methods alone. A total of 123,271 thousand short tons came from the Appalachian region, where Virginia was responsible for 8,217 thousand short tons. Three coalfields are distributed throughout Virginia; however, all coal currently produced in Virginia comes from the seven counties that make up the Southwest Virginia Coalfield (DMME 2012). Eighty-five percent of the coal produced in Virginia comes from Wise, Dickenson and Buchanan counties. There are 4 major ranks (types) of coal, each classified by its own heating value, which is dependent on the amount and type of carbon it contains. In Virginia most coal extracted is bituminous. Bituminous coal is second in rank behind anthracite and contains 45-86% carbon. This type of coal is the most abundant in the U.S. and is primarily used to generate electricity. Ninety three percent of coal produced is used to generate electricity, and coal is also used as a basic energy source for many industries including steel and paper (EIA 2012). In 2011, the Energy Information Administration (EIA 2012) reported that coal reserves in the US are larger than the remaining natural gas and oil reserves. With 260 billion short tons of recoverable coal beneath 31 states in 3 widely separated regions (Figure 1), extra pressure may be put on states to produce more coal as technologies develop that will allow more efficient extraction and consumption of this resource.

Traditionally, coal was mined almost exclusively by underground or deep mining methods (Shover 1986). In the late 1950's, surface mining methods became increasingly common as the size and handling capacity of heavy equipment increased. Deep mining for coal

usually recovers < 60% of the coal from a seam, whereas surface mining can recover up to 90%. Thus, in most cases it is more economical to surface mine for coal. The 4 types of surface mining methods are: area, contour, mountaintop removal and auger mining (Kentucky Geological Survey 2006). Because of the steep mountainous terrain and shallow coal seams, contour mining is the most commonly used surface mining method in Southwest Virginia. In contour mining, an L-shaped cut is made along the contour of the mountain. This creates a vertical wall that is referred to as the highwall, and a shelf or “bench” on the side of the mountain. The bench is used as a platform for the heavy machinery used to extract coal. The undisturbed area below the bench is called the outslope. Once the cut is made, explosives are placed in the highwall to loosen the overburden (topsoil and other soil horizons overlaying the coal seam) which is then removed and coal is extracted from the seam (Shover 1986). Prior to the Surface Mining Control and Reclamation Act (SMCRA) in 1977, overburden was frequently “shoved” over the outslope, creating unstable slope conditions (Daniels and Zipper 1995). These pre-SMCRA mining methods often generated a landscape of exposed highwalls, unstable outsoles and low ground cover (Holl 2002).

#### *Environmental impacts of surface mining*

Mining causes drastic changes to the landscape, and without proper care can have a negative impact on the environment. In the process of surface mining, existing vegetation is removed as land is cleared for mining. Additionally, soil and associated biota become disrupted during the mining process. Topsoil is commonly mixed with other soil horizons when the overburden is removed and set aside until reclamation. Such alterations to soil conditions can create inhospitable conditions for vegetation re-establishment, which in turn can lead to erosion and unstable slope conditions. Wildlife habitat also becomes altered during mining, creating

suitable habitat for some species (Allaire 1978, Whitmore and Hall 1978, Rohrbaugh and Yahner 1996) and destroying it (at least temporarily) for others.

One of the biggest concerns in surface mining and reclamation is native soil condition. If proper care is not taken during overburden replacement, a condition known as acid mine drainage (AMD) can occur. AMD occurs when minerals containing sulfur in the unweathered soil horizons, usually iron pyrite ( $\text{FeS}_2$ ), oxidize via exposure to oxygen and water during the weathering process (Zipper and Skousen 2001). AMD can result in highly acidic conditions that are toxic for both plants and animals. Therefore, proper overburden selection and placement is an important part of the reclamation process (Daniels and Zipper 2009). Holl (2002) suggested that using native topsoil from forested areas will encourage the re-establishment of forest species and increase the rate of succession on reclaimed sites.

Reclaimed mine lands will eventually revert back to mature hardwood forest via natural succession; however, this process may take centuries, especially on lands reclaimed to grasslands (Angel et al. 2005). Common early reclamation practices were to hydroseed a mix of aggressive ground covers and fertilizer to reduce erosion and stabilize sloping terrain (Burger and Zipper 2009). Common ground covers included a mix of agricultural species such as K-31 tall fescue (*Festuca arundinacea*), red (*Trifolium pretense*) and sweet (*Melilotus alba*) clovers. These species are not conducive to tree growth because they grow too tall and too dense for trees to emerge. Tall fescue is commonly affected by a fungus (*Acermonium coenophialium*) that may cause problems for grazing animals and wildlife during certain times of the year (Skousen and Zipper 2009). Fettinger et al. (2002) suggested that due to the nature of its growth, tall fescue may inhibit movements of small mammals and certain bird species. A mixture of tree-compatible ground covers is therefore recommended to provide erosion control and not impede natural

succession. Tree-compatible ground covers include grasses such as annual ryegrass (*Lolium spp.*) and foxtail millet (*Milium spp.*). Burger and Zipper (2009) recommend planting these grasses to provide adequate cover during the first year post-mining. They also recommended planting slower growing grasses and forbs tolerant to low soil pH such as perennial ryegrass (*Lolium perenne*), timothy (*Phleum spp.*), and birdsfoot trefoil (*Lotus corniculatus*) during the second year to provide sufficient cover to prevent erosion, while allowing native vegetation to re-colonize the area.

The impacts of surface mining and associated reclamation practices on wildlife populations have largely been overlooked (Holl 1996). Mining reduces the amount of habitat available to forest dwelling species, fragments the landscape, and creates “islands” of grassland habitat surrounded by forest (Whitmore 1978). Structural diversity of cover types is often reduced as a result of surface mining and may lead to structural and compositional changes in avian communities (Wray et al. 1982). Changes in cover types could also cause a change in the predator community and thereby alter predator-prey dynamics

#### *Surface-mine reclamation*

The Surface Mining Control and Reclamation Act of 1977 was enacted to standardize reclamation practices and to protect society and the environment from the adverse effects of surface coal mining operations (SMCRA 1977). The term reclamation is used to describe efforts that aim to improve quality of land by restoring some pre-disturbance functions. Early reclamation efforts focused on abiotic factors such as erosion control and soil fertility, while the fauna were largely overlooked (Holl 1996, Holl 2002). SMCRA requires that a diverse, effective and permanent vegetative cover native to the area be established on the affected area, except that introduced species may be used where desirable and necessary to achieve post mining land use

(SMCRA 1977). Commercial timber production has been the traditional land use for wood-products industries throughout the coalfields region (Burger and Zipper 2009), where the dominant forest type is mixed mesophytic hardwoods. These hardwoods serve as habitat for a diverse array of organisms (Holl 2002), and are an important renewable economic resource in the central Appalachian coal fields. The average annual value of timber products in the central Appalachians is \$23 billion (Burger and Zipper 2009); therefore, forests are an important resource, both economically and ecologically. Where reclamation did take place prior to 1977, most surface-mined land in the east and Midwest was reclaimed with trees. Through time and succession, commercially valuable forests were restored, and in some instances productivity was equal to or better than pre-mined sites (Burger and Zipper 2009).

The strict performance standards set by SMCRA require mine operators to restore approximate original contour (AOC), establish vegetation that is at least equal in extent of cover to the natural vegetation of the area and compatible with approved postmining land-use. Upon completion of mining, each site is evaluated for vegetative cover and water quality issues after 5 years (10 years in regions with <66 cm of annual average precipitation). At the end of the 5-year period, at least 90% living ground cover must be maintained (Daniels and Zipper 1995). Consequently, little concern was given to long-term reclamation and common practices were to plant fast growing, aggressive ground covers as a more convenient, less expensive alternative to returning the land back to forest. Reclaiming surface-mined land to hayland/pasture is acceptable under SMCRA because pasture is considered an equal or better land-use and could potentially provide landowners with more productive post-mined land.

There was a resurgence of interest in the early 1990's for landowners to have their land reclaimed to forest for the purposes of commercial forestry; however, lack of scientific

information led to poor site preparation and suboptimal growing conditions for trees (Torbert and Burger 2000, Angel et al. 2005, Burger and Zipper 2009). In 2005, the Appalachian Regional Reforestation Initiative (ARRI) was created to encourage the restoration of high quality forests on reclaimed coal mines (Angel et al 2005). ARRI is a cooperative effort between the Appalachian states and the Office of Surface Mining (OSM) to apply scientific knowledge and educate mine operators, agency personnel, and land owners about practices that can be used to restore high quality forests on surface-mined land through a series of publications. These publications explain the Forest Reclamation Approach (FRA), which scientific research has demonstrated can be a cost-effective way to generate productive postmining forests (Angel et al. 2005).

#### *Forest Reclamation and the FRA*

Forest reclamation is the process of restoring previously forested land to a productive forestry postmining land use (Torbert and Burger 2000). Forests can be restored for multiple post-mining land uses including recreation, wildlife habitat, commercial forest products and carbon sequestration (Torbert and Burger 2000, Burger and Zipper 2009b). Regardless of the post-mining land use, reclamation efforts should include input from landowners, coal operators and regulators. Reclamation should meet the needs of the landowner while proving to be economically feasible to the coal operators. When mining and reclamation efforts are conducted using FRA, results can be cost-effective for the coal operators and generate productive forests (Angel et al. 2005). Torbert et al. (1994) estimated that rough grading the soil to make suitable growing conditions for trees could save \$500/hectare on surface grading costs.

Contrary to post-mining success standards for agricultural land-use, forestry land-use success is based on stocking and the number of trees surviving to the 5 year bond-release period,

rather than productivity. Success standards vary depending on the type of post-mining forest land-use (commercial, wildlife, etc.) implemented (see Torbert and Burger 2000). The two most commonly cited problems resulting in poor site conditions for the establishment of native tree species are 1) severe soil compaction from spoil grading and 2) planting of aggressive ground covers to prevent erosion (Torbert and Burger 2000, Angel 2005). To help mitigate these problems and to establish trees on surface-mined lands, there are 5 steps to the Forest Reclamation Approach. These steps include: 1) create a suitable rooting medium for good tree growth, 2) loosely grade topsoil, 3) use ground covers that are compatible with growing trees, 4) plant a mix of early successional and commercially valuable crop trees, and 5) use proper tree planting techniques (Burger et al. 2005). If the above methods are used in conjunction with one another, the result can be a cost-effective, productive post-mining forest that will ultimately benefit mine operators, landowners, and society. All reclaimed surface-mined lands using the above techniques are said to be FRA compliant.

In addition to meeting the regulatory reclamation standards in Virginia, mine operators are required to develop a “protection and enhancement plan.” In this plan, operators are required to describe how they will minimize disturbance and impacts to fish and wildlife and related environmental values. This highlights the need to evaluate the value of various reclamation techniques to provide high quality habitat for wildlife species. Johnson et al. (2009) suggested that species richness and community-based metrics can be an effective means of evaluating restoration success for wildlife. While Fleishman et al. (2006) and Knopf (2010) described the limitations associated with using species richness to compare areas for conservation priority, they also point out that species richness can be useful if additional information such as population demography and community dynamics are provided.

*Vegetation and Wildlife Reclamation on Coal-Mined land*

It has been recognized for some time that strip-mining causes dramatic changes to the landscape and many studies have documented these changes. For example, Myers and Klimstra (1963) surveyed herpetofauna on previously strip-mined lands in Illinois. Despite drastic changes in topography and vegetation, they found that some herptiles had re-invaded post-mined lands and some were present in higher numbers than the surrounding non-mined area. This result was especially true for the semi-aquatic reptiles and amphibians that rely heavily on the permanent lakes and ponds that were created by mining activities. Myers and Klimstra (1963) also suggested that the lack of a well-developed terrestrial herpetofauna could be due to the lack of organic debris on spoil banks. Through time; however, the degree of organic matter would be expected to increase as vegetative succession proceeds.

Brewer (1958) studied breeding birds on strip-mined land also in Illinois and found that vegetational changes related to time since stripping and differences in local site conditions explained the differences in bird populations on lands mined in different years. Karr (1968) also recognized the relationship between bird species diversity (BSD) and time since mining. He found that species diversity increases from bare ground through the shrub stages to bottomland forest and then decreases to the climax forest cover type. This result is consistent with other studies on surface mines (Chapman et al. 1978, Curtis et al. 1978, Stormer et al. 1978, Whitmore 1978). MacArthur and MacArthur (1961) suggested the heterogeneity between vertical layers of vegetation may be important in determining bird species diversity whereas Roth (1976) suggested that patchiness and horizontal diversity within the landscape may be more important in determining bird species diversity. The diverse horizontal as well as vertical topography on surface mined areas commonly leads to diverse vegetation and consequently more diverse avifauna (Steele and Grant 1982). Karr (1968) also suggested that the ridge-and-valley

topography might allow increased avian abundance by decreasing the frequency of interactions (competition) between adjacent pairs of birds as a result of intervening ridges.

Despite the dramatic changes to the landscape caused by mining, several studies suggest that the new environments created by strip mining are beneficial to wildlife not common to the area or that are suppressed by limited resources and competition. For example, Bajema et al. (2001) found that many tracts of land were large enough to create habitat for area-sensitive grassland species in Indiana. Brenner (2007) found that restoration of surface mines to grasslands in Pennsylvania provided adequate habitat for some threatened and endangered species of birds and reptiles. Similarly, results from a study in eastern Kentucky showed that while mining negatively affects forest avian populations, it creates opportunities for other species uncommon to the area (Allaire 1978). Grassland species such as the grasshopper sparrow (*Ammodramus savannarum*), horned lark (*Eremophila alpestris*), and eastern meadowlarks (*Sturnella magna*) made use of grassland habitats, while eastern bluebirds (*Sialia sialis*), chimney swifts (*Chaetura pelagica*), and eastern phoebes (*Sayornis phoebe*) commonly used highwalls for nest cavities (Allaire 1978). Ingold (2002) also suggested that grasslands on reclaimed strip mines in Ohio may provide refuge from nearby agricultural areas where agricultural practices such as mowing and grazing may create population “sinks.”

Many studies indicate the importance of reclamation of strip mined areas to facilitate natural succession for the development of wildlife communities. Riley and Brown (1978) suggested that the return of the disturbed land to the original climax vegetative community should facilitate the return of the original wildlife community; however, Steele and Grant (1982) cautioned that a return to pre-disturbed abundances, diversity and richness of wildlife species may not indicate the return of the pre-disturbance community. Passell (2000) demonstrated how

simple restoration efforts could “jump start” the development of avian communities on minimally restored Indonesian tin strip mines. More intensive reclamation techniques can restore biological productivity and facilitate avian successional patterns beyond what would normally occur through standard techniques (Curtis et al. 1978).

Whitmore (1978) described a predictable trend where avian species richness generally increased with increasing seral stage on reclaimed strip mines in West Virginia. Similarly, Cochran et al. (1999) studied small mammal and avian communities on different successional stages of reclaimed Kaolin mines in Georgia and found that early successional species were associated with the youngest sites; whereas the mid successional species were most common on middle-aged sites, and canopy nesting species were more common on the oldest sites. Species richness was highest on middle-aged sites and abundance was lowest on the older sites. Several studies have also documented this trend (Chapman et al. 1978, Crawford 1978). Chapman et al. (1978) suggested that the reason for the decrease in species richness and abundance on older mine sites is that canopy closure may cause a decrease in understory vegetation commonly used as nesting cover. Cochran et al. (1999) also found that small mammal species were most common in earlier habitats and less abundant after canopy closure. Small mammals could be detrimental to reclamation efforts because they may girdle the stems of small saplings. Therefore, there may be a tradeoff between managing for wildlife and other post-mining land-uses. However, the increase of raptor abundance may control small mammal populations. Rohrbaugh and Yahner (1996) found harriers (*Circus cyaneus*) on reclaimed mine sites in Pennsylvania to be 5 times more abundant than surrounding agricultural fields.

Most studies of avifauna on reclaimed strip mines have focused on breeding bird abundance, richness and diversity. Van Horne (1983) cautioned against the use of density alone

as an indicator of habitat quality; estimates of survival and breeding success may be more important in determining habitat quality for breeding birds on reclaimed strip mines than indices of abundance and diversity. An understanding of nest-site selection and nesting success is important to understanding the ecology of bird species and to develop management recommendations (Rodewald 2003).

There have been many studies that document breeding success on strip-mines (Whitmore 1978, Wray et al. 1978, Wray et al. 1982, Chapman et al. 1987, Galligan 2006, Stauffer 2008). Many of these however, pertain to grassland species. Only one study (Chapman et al. 1987) examined nest success of forest species on strip mines. However, this study only focused on strip-mined sites where no active revegetation took place. Novel elements in the environment may be created by reclaimed strip mines and can cause lower avian reproductive success. This is especially true if the element mimics a traditional cue that animals use to select habitats (Schlaepfer et al. 2002). For example, Wray et al. (1982) found that grassland sparrows were attracted to large tracts of grasslands created by surface mining in West Virginia. Reproductive success was lower on these grasslands, possibly as a result of the concentration of predators along the edges of woodlands and grasslands. Gates and Gysel (1978) coined the term “ecological trap” to describe the situation where proximate cues (adequate nest sites, cover, food availability) are good enough to elicit a settling response, but changes in the environment that may not be predictable based on those cues, such as a more diverse predator community near edges (edge effect), may result in decreased reproductive success.

Habitat preferences are assumed to be adaptive (Martin 1988, Schill and Yahner 2009); however, if environmental changes disrupt the correlation between cues that birds use to select habitats and habitat quality, habitat preferences may become maladaptive. Site fidelity can cause

birds to return to the same breeding grounds year after year independent of reproductive success (Howlett and Stutchbury 2003). Thus, if reproductive success is lower on reclaimed strip mines and birds return year after year, new individuals must colonize from outside populations, or they will go locally extinct. However, Donovan and Flather (2000) note that this effect may depend on the proportion of the population breeding within these lower quality areas.

Nest-site selection is a process whereby an animal responds to a set of environmental factors that indicate potential for successful rearing of young (Jones 2001). The most common approach in nest studies is to compare nests in different habitats to determine possible differences in cues that birds use to select nest sites (Rodewald 2003). Several factors may influence the choice of nest sites including predation (Martin 1988, 1993, Wolfgang and Weiss 2004), availability of food resources (Zanette et al. 2000, Nagy and Holmes 2004), amount and type of disturbance within the landscape (Rodewald and Yahner 2001), and microhabitat characteristics (Holway 1991). Nest predation has been observed to be the primary source of nest failure in open-cupped nesting passerines (Martin 1993). Therefore, birds should choose nest sites that reduce the probability of predation. Filliater et al. (1994) however, indicated that high incidence of predation by a species-rich guild of nest predators precluded the existence of predictably safe nest sites for Northern cardinals (*Cardinalis cardinalis*) in Ohio.

The creation of novel habitats on reclaimed strip mines may introduce a diverse guild of nest predators with which the breeding bird community may not have co-evolved. Therefore, strategies that minimize nest predation by certain predators may not minimize nest predation by others. For example, Bulluck and Buehler (2008) found that golden-winged warblers (*Vermivora chrysoptera*) selected nest sites with greater concealment cover than random sites, but these habitat characteristics apparently did not affect daily nest survival. This could indicate that

golden-winged warblers selected nest sites with greater concealment to decrease predation by predators that use visual cues to hunt for prey whereas no differences in habitat associations would be found if primary nest predators used olfactory cues to find nests. Thus, multifaceted studies that relate nest site selection and outcome (success or failure) to the abiotic and biotic environment are necessary on reclaimed strip mines.

In this study, I compare the value of various reclamation techniques and their value to provide high quality habitat to breeding birds, using a combination of population and community-based metrics. I chose to study birds because they are readily observable, and are one of the first organisms to colonize areas after a disturbance due to their high mobility (Brändle et al. 2003). In addition, I characterized mammal species richness and activity for common mammal species recorded on the study site (shown in Appendix A).

This thesis is divided into two main sections: In chapter 2, I describe avian reproductive success and nest site selection on reclaimed mine sites in southwest Virginia. Specifically, I focus on habitat attributes within various cover types that seem to influence avian nest site selection and nest success. In addition, I estimate finite rate of population growth for indigo buntings (*Passerina cyanea*) and field sparrows (*Spizella pusilla*) to determine if individuals were producing enough young to sustain populations on reclaimed mine lands.

Chapter 3 describes a novel use of the robust design population model to estimate avian community dynamics in relation to various reclamation cover types over a 5 year period. Specifically, I calculate rates of species turnover, local extinction, persistence and colonization of species in relation to temporal changes in vegetation within each reclamation cover type. The final chapter (chapter 4) gives an overall synopsis of the research conducted, major findings, and management recommendations.

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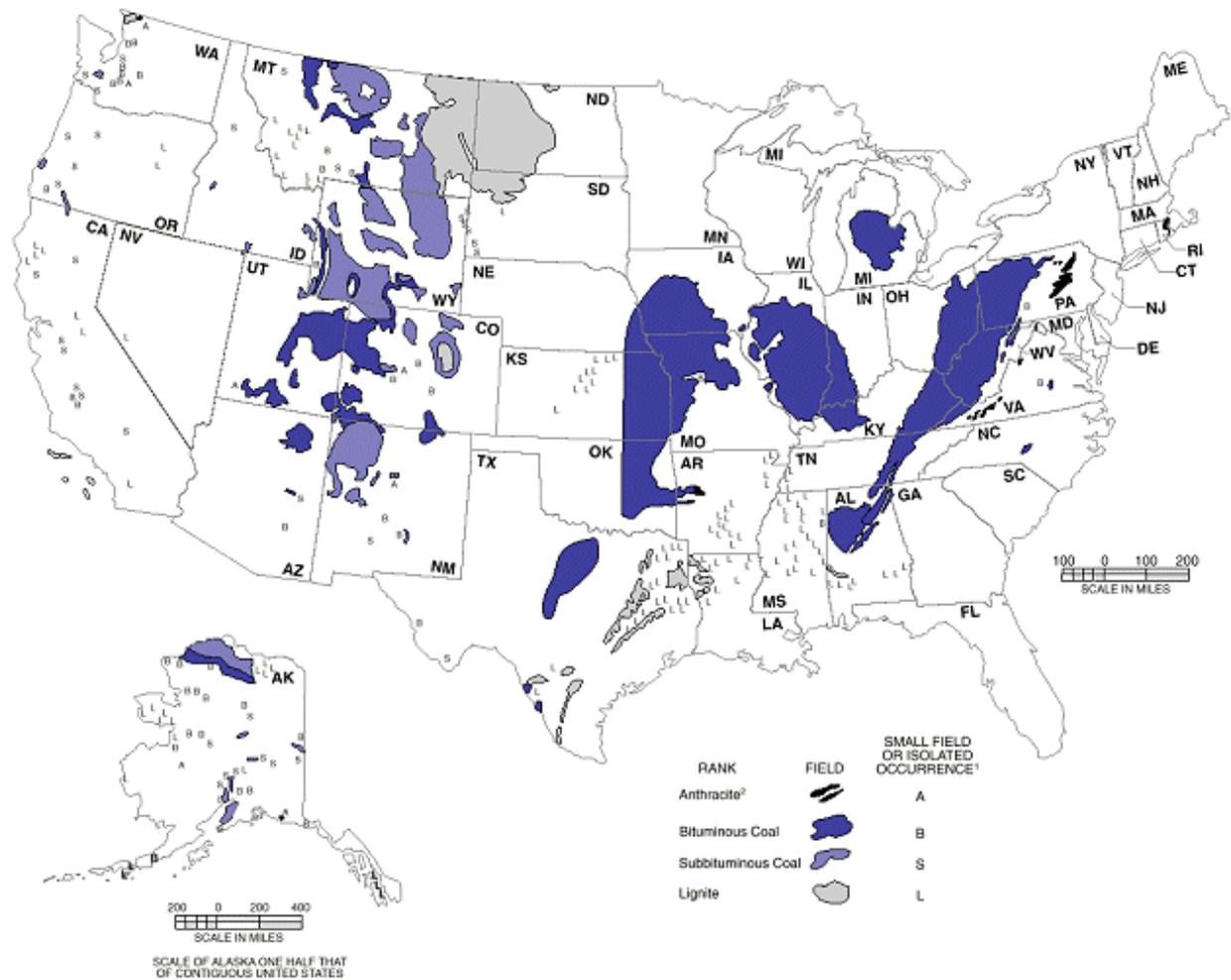
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**Figure 1.** Distribution of coal types (anthracite, bituminous coal, subbituminous coal, and lignite) across the three coal producing regions of the United States. (Energy Information Administration, U.S. Department of Energy 2009)

## **Avian nest-site selection and nest success on reclaimed surface-mines in southwest Virginia**

### **Abstract**

Some studies suggest that reclaimed coal surface-mines may provide high quality, early successional habitats required by grassland and shrubland bird species. Information regarding avian habitat relationships and factors that influence avian nest success on reclaimed surface mines are needed to enhance the value of post-mined lands for early successional species. In this study, I evaluated nest-site selection and nest success for indigo buntings (*Passerina cyanea*) and field sparrows (*Spizella pusillia*), in relation to various surface-mine reclamation regimens in southwest Virginia. In addition, I estimated densities for each species using program DISTANCE. Lastly, I determined population growth rate using information from this study to calculate seasonal fecundity and obtained values for adult and juvenile survival from the literature.

Daily nest survival probability (DNS)  $\pm$  SE for field sparrows was 0.964(0.009) and was positively affected by degree of nest concealment and negatively affected by percent woody vegetation around the nest. Year was the best predictor of DNS for indigo buntings with estimates of 0.979 (0.010) and 0.912 (0.024) for 2010 and 2011 respectively. Neither DNS nor density differed among the reclamation type for each species. Density estimates were 0.285 and 0.945 birds/ha for field sparrows and indigo buntings respectively. At the rates of daily nest mortality observed on this study, I estimated that populations were sustainable as long as adult survival was  $\geq 0.8$ . These results suggest reclaimed mine-lands may provide high quality nesting

habitat for some species; however, future studies should focus on factors that influence adult and juvenile survival.

## **Introduction**

Many early successional songbirds are declining at a faster rate than mid- and late-successional neotropical migrant landbirds (Askins 2001). Fifty eight percent of species inhabiting early successional habitats showed significant negative population trends in the east from 1966-2009 (Sauer et al. 2011). Among those, the golden-winged warbler (*Vermivora chrysoptera*), prairie warbler (*Dendroica discolor*), field sparrow (*Spizella pusilla*), indigo bunting (*Passerina cyanea*), and yellow-breasted chat (*Icteria virens*) are common species inhabiting reclaimed mine lands (Carrozzino 2009). In contrast, only 19 percent of woodland breeding songbirds showed significant declines during the same time frame. This difference is often attributed to habitat loss and the unpopularity of managing for open habitats (Askins 2001).

Reclaimed mine lands offer great potential for managing early successional habitats. The vast areas that post-mined lands cover may be appealing to some area-sensitive early-successional species (Bajema et al. 2001). Further, traditional reclamation practices that lead to high soil compaction, and planting aggressive ground covers often result in a state of arrested succession (Groninger et al. 2007), which may benefit some early successional, shrub-scrub nesting species. Many studies have examined the impacts of surface mining on breeding birds (Brewer 1958, Karr 1968, Alliar 1978, Whitmore and Hall 1978, Wray et al. 1982, Chapman et al. 1987, Rohrbaugh and Yahner 1996, Bajema et al. 2001, Ingold 2002, Scott et al. 2002, Monroe and Ritchison 2005, Galligan et al. 2006, Bulluck and Buehler 2006, Brenner 2007, Bulluck and Buehler 2008, Carrozzino 2009). However, only a few have examined reproductive success on reclaimed mine lands (Chapman et al. 1978, Whitmore 1978, Wray et al. 1982, Ingold

2002, Monroe and Ritchison 2005, Bulluck and Buehler 2006, Galligan et al. 2006, Bulluck and Buehler 2008); most of which focus on grassland species. No studies to date have examined the reproductive success of birds in response to different reclamation practices.

Reclamation practices are dictated by the post-mining land use objectives of the landowner, and mine operators. For example, managed hayland, pasture and non-commercial forestry were some of the more traditional reclamation practices implemented by mine operators until recently. Recent resurgence of interest in commercial forest products in southern Appalachia has led to the development of a new reclamation practice, coined the Forest Reclamation Approach (FRA)(Angel et al. 2005). FRA is a regulation-compliant method of establishing viable hardwood forests on post-mined lands. This method requires planting less aggressive ground covers that allow volunteer species to colonize in addition to planted native hardwoods. Due to the vast differences among reclamation cover types, there is a need to evaluate reclamation strategies that provide high-quality early successional habitats. Further, studies of reproductive success are can contribute to better- informed management recommendations than relying on abundance information as an index of habitat quality and reclamation success (VanHorne 1983). The objectives of this study were to 1) compare habitat attributes associated with nest sites to those of random sites at multiple spatial scales for 2 common early-successional species breeding on reclaimed surface mines, 2) determine what factors influence nesting success in different reclamation types, 3) use estimates of seasonal fecundity from our study and regional estimates of adult and juvenile survival to infer population viability on reclaimed mine lands and 4) discuss recommendations for future reclamation activities to provide high quality early-successional habitat for nesting songbirds.

## **Methods**

### *Study Areas*

All research was conducted on the Powell River Project (PRP) and on surrounding reclaimed surface mines in Wise and Dickenson Counties, Virginia. This region of the Southern Appalachian Mountains receives an average of 1193mm of precipitation annually. Mean average temperatures range from 0.38°C in January, to 21.05°C in June (Holl 2002). The dominant forest type is mixed mesopytic with oak (*Quercus spp.*) and hickory (*Carya spp.*) as the dominant overstory vegetation. This area served as a refuge for many mesic species during the drier glacial epochs and is therefore known for its rich biologically diverse taxa (Holl 2002). The Powell River Project is a cooperative effort between Penn Virginia Corporation (a large landholding company), Virginia Tech, and the Appalachian states. The 700 hectare property was extensively mined by the Ring Brothers Coal Company using deep mining and contour mining methods from the late 1950's through 1977 until the advent of the Surface Mining Control and Reclamation Act (SMCRA). In the early 1980's, Red River Coal Company began actively re-mining the area using second-cut contour mining methods and reclaiming in compliance with current SMCRA regulations. Much of the research conducted on the PRP is for the purpose of improving reclamation technologies that will enhance the post-mining land-use potential for landowners. Current research on the PRP include topics such as reclaiming mine lands to forests, enhancing sustainable beef cattle production on reclaimed mine lands, hydrology restoration and stream quality, and topsoil construction/overburden placement. Educational programs are conducted to disseminate research results to local landowners, mine operators, and educators (Zipper and Rockett 1998).

### *Field Methods*

I sampled the breeding bird community on reclaimed surface mines within various vegetation restoration regimens using the variable radius point count technique (Buckland et al. 2001). I considered a patch of relatively homogeneous vegetation cover (>1ha) of approximately the same age, reclaimed using the same technique to be the sampling unit for this study. I considered 5 separate cover types where active reclamation took place including: pasture (PAS), shrub (SHB), mixed (MIX), pine (PIN) and forest reclamation (FRA). In addition, I considered one cover type where no active reclamation took place before mining law (pre-SMCRA), and one cover type, reference forest (REF), where no mining occurred (see below for descriptions).

The pasture cover type (n=8 points) consisted mainly of tall fescue (*Festuca spp*) and orchard grass (*Dactylis glomerata*) and was typically managed for livestock grazing. We classified shrub cover types (n=9 points) as consisting mainly of grasses interspersed with low shrubs of autumn olive (*Elaeagnus umbellate*), multiflora rose (*Rosa multiflora*), and blackberry (*Rubus spp*). Mixed cover types (n=4 points) consisted mainly of 15-20 year old stands of planted trees with an even mix of white pine (*Pinus strobus*) and black locust (*Robinia pseudoacacia*). Pine cover types (n=4 points) were also 15-20 year old stands dominated by white pine. FRA (n=10 points) was the most recent reclamation cover type and represented patches reclaimed 5-10 years ago and planted with a mix of native hardwoods including oaks, hickories, pine, and ash (*Fraxinus spp*). Pre-SMCRA sites, where no active reclamation took place (n=20 points) were characterized by a series of highwalls, benches (flat, compacted surfaces), and outcrops, created during the mining process (see Carrozzino 2009 for detailed descriptions of these sites). Lastly, reference forest plots (REF), where no mining took place (n=13 points), were characteristic of secondary old-growth forests harvested ~80 years ago (See appendix F for information regarding location of points).

I used a subset of 58 points from a previous study (Carrozzino 2009) and included only those points that were separated by  $\geq 150\text{m}$  to ensure independence between samples. In addition, I added points in the FRA cover types that were not sampled during that study. I sampled points from mid-May to early July in 2010 and 2011 from 0600 to 1000 on days with no rain and minimal wind ( $<5\text{km/hr}$ ). Each count lasted 5 minutes after a 1 minute settling period upon arrival at the point. Each point was visited 4 times in 2010 and 5 times in 2011. I rotated observers and the order in which points were sampled to help minimize bias due to observer- and time-of-day effects. In 2011, rangefinders were used to help aid in distance estimation.

We located nests from late April to late July by observing parental behavior, and systematically searching potential vegetation within the various reclamation patches (Martin and Geupel 1993). I allocated search effort evenly among the various reclamation types to try to minimize bias associated with uneven sampling effort. Once a nest was located, we monitored it every 2 to 3 days until completion. We approached the nests from a different direction at each visit and were careful to avoid leaving “dead-end trails” that could potentially lead predators to the nest (Martin and Geupel 1993). A nest was considered successful if at least one young fledged, and we assumed a nest failed if there was evidence of predation, or if the contents of the nest were missing before fledging was possible.

After nest completion, we collected vegetation data at the nest site (within 2m of the nest), and the nest plot (within an 11.3m radius around the nest). We paired each nest with a random plot by pacing 30m in a random direction and centering a 0.04ha (11.3m radius) circular plot on the same type of substrate that the actual nest was placed in. If the edge of a patch was  $<30\text{m}$  from the nest in the direction chosen for the random plot, we paced to the edge of the patch, selected a new random direction away from the patch edge and walked until the edge of

the random plot was 30m from the nearest edge of the nest plot. Measurements were only taken at the nest plot scale for random locations.

At the nest site scale (measurements taken at used nest sites only), I recorded nest height, substrate type, substrate diameter at breast height (DBH), distance of the center of the nest to the central axis of the substrate, distance from the center of the nest to the nearest edge of the substrate, distance from the nest to the nearest patch edge. We defined a “patch” as an area of relatively homogenous vegetation not separated by more than 5 meters. Mining roads > 5m were considered the edge of a patch. I visually estimated percent concealment from 1m above the nest and in each of the 4 cardinal directions and below the nest. Concealment measurements were averaged to obtain an index of mean concealment around the nest.

At the nest plot scale (measurements taken at both used and un-used nest plots), I adapted the habitat sampling methods of Noon (1981). We measured and recorded DBH and species of trees, which were grouped into size classes (4-10cm, 10-15cm, 15+cm) for analysis. Woody stem density was estimated by traversing 4 10m transects, one in each cardinal direction, and counting the number of stems greater than 0.5m tall encountered within 1m of each transect. All stems within 1m of each transect were counted regardless of whether they were rooted inside the transect or not. I used the point-intercept method to estimate the percentage cover of canopy, leaf, grass, bare ground (including rock), coarse woody debris, forbs, and woody vegetation. We measured the distance to the nearest tree >4cm DBH in each quarter to calculate the coefficient of variation of distances within each plot as an index of habitat heterogeneity. Lastly, I measured understory foliage volume using modified methods from Nudds (1977). The board was 2.5m tall and 0.5m wide and was divided into 6 0.5m segments. We ranked concealment of each segment in each cardinal direction at a distance of 10m and height of 1m on a scale from 1 to 6, where 1=

0-5%, 2= 6-25%, 3= 26-35%, 4= 51-75%, 5= 76-95% and 6= 96-100%. Concealment of each segment was averaged to obtain an index of understory foliage volume for each plot. One observer (CEL) conducted all concealment estimates to help reduce bias (Table 1).

### *Statistical Analysis*

#### *Nest site selection*

Because of small sample sizes, and non-normality of our data I used Multi-response permutation procedures (MRPP) to test for potential differences in vegetation characteristics between used versus random, and failed versus successful nests. I used a matched pairs test to compare vegetation measures at used and random nest locations, and a 2-sample test to compare habitat characteristics of successful and non-successful nests (Mielke and Berry 2001). MRPP is a non-parametric, distance based permutation algorithm that allows the user to compare the equality of treatment groups. Using the permutation test for matched pairs (PTMP) within MRPP is similar to conducting a paired t-test, without the assumption of normality on the differences between paired samples. Additionally, I used the midpoint of the nesting season to determine potential differences in nest site selection for early and late nests.

#### *Bird Density*

I obtained density estimates for field sparrows (*Spizella pusilla*) and indigo buntings (*Passerina cyanea*) using the multiple covariate distance sampling platform in program DISTANCE (Buckland et al. 2001). DISTANCE assumes detectability of individuals decreases with distance from the point. The assumptions of distance sampling are 1) objects at the point are always detected, 2) objects are detected at initial location, and 3) distances to objects are measured accurately. I used the data from 2011 only because the use of rangefinders aided in accuracy of distance estimation and allowed us to better model the detection function. I did not expect the densities would have varied much from 2010 to 2011 field seasons. I assumed a 1:1

sex ratio and only modeled densities for males that were observed using the habitat being sampled. I truncated the greatest 10% of distances for both species as suggested by Buckland et al. (2001). I modeled the detection function using the hazard rate and half normal key functions with simple polynomial and cosine adjustment terms respectively. The key function selected based on minimum Akaike's Information Criterion adjusted for small sample sizes (AICc) criteria was then used as the key function for the remainder of the analyses. I selected 2 covariates (observer, and reclamation type) that I thought would help explain some of the variation in the detection functions. I modeled each covariate individually and the additive effect of both observer and reclamation type. Next, I post stratified by observer and reclamation type, and used the sum of the effort in each stratum to estimate the global density. Lastly, I added observer and reclamation type as a covariate to each of the post-stratified models. I used AICc to select the "best" model and used bootstrap resampling to obtain a more robust estimate of density and variance for that model (Buckland et al. 2001). Model fit was assessed using the Kolmogorov-Smirnov (K-S) test statistic and inspection of detection function curves. The K-S statistic evaluates departure of the empirical distribution function and the cumulative distribution function.

### *Nest Survival*

To model nest survival, and test *a priori* hypotheses relating to year, site, climate and habitat specific covariates, I used the nest survival model platform in program MARK (White and Burnham 1999, Rotella et al. 2004) and evaluated the relative support for *a priori* hypotheses using the information theoretic approach (Burnham and Anderson 2002). Unlike the commonly used Mayfield approach to estimating nest survival (Mayfield 1961), program MARK allows testing of multiple, competing hypotheses about nest survival as a function of user

specified covariates (Rotella et al. 2004). Another benefit to using program MARK over the Mayfield method is that it does not make the assumption that nest success is constant throughout the breeding season. I modeled nest survival for species with >20 nests (i.e. field sparrow and indigo bunting). Nests that failed due to abandonment (n=2) were dropped from the analysis because the interval in which the nest failed was not known (right censored). To select among competing models, I used  $AIC_c$  and assumed the most parsimonious models had strong support if  $AIC_c$  was within 2 units of the lowest  $AIC_c$  value ( $\Delta AIC_c < 2$ ), or if models had >10% of the model weights ( $w_i$ ).

#### *a priori hypotheses and model descriptions*

I developed a set of *a priori* hypotheses and predictions to try to explain variation in the nest survival of field sparrows and indigo buntings based on previous studies of nest site selection and nest success. The hypotheses and predictions used in our models are outlined below:

- 1) Year. Annual variation is a common source of variation in nest survival rates (Dinsmore et al. 2002). By incorporating year effects into our models, we hope to account for some annual variation potentially due to yearly differences in weather patterns, predator numbers, or individuals nesting (i.e. non-successful breeders not returning to breed).
- 2) Stage. Variation among different stages of the nesting cycle (egg-laying, incubation, nestling) is not uncommon. Several studies have documented increased predation rates as nests near completion, likely a result of increased activity around the nest (parents making more trips to feed their young).
- 3) Concealment. Concealment of the nest by vegetation is thought to increase nesting success, but the results vary in the literature. Some studies indicated a positive association between increased concealment and nest success (Martin 1993, Easton and

Martin 2002, Moorman et al. 2002) while others did not (Götmark et al. 1995, Howelett and Stutchbury 1996, Schill and Yahner 2009). I hypothesized that nests with greater concealment would have increased survival by being concealed from visually oriented predators, or having obscured adult and or nestling movement.

- 4) Cover type. I expected some variation in nest success to be explained by the cover type in which the nest was found. Different reclamation techniques create different cover types that may influence the type and or number of predators in an area.
- 5) Woody vegetation <1m tall. Martin (1993) proposed that nest predation would decrease as the number of potential nesting sites increased because the number of potential nest substrates for a predator to search would increase (potential- prey-site hypothesis). Since many of the nests found in our study were within 1m from the ground, I hypothesized a positive association between increased woody vegetation and nest success.
- 6) Stem densities. Stem densities may influence the accessibility of nests to certain predators and may facilitate concealment of the nest from certain predators (Schill and Yhaner 2009). Easton and Martin (2002) suggested that some bird species have lower nesting success in conifers, so we separated coniferous and deciduous stems for analysis. I hypothesized that an increase in deciduous stem density would result in an increase in nest success and an increase in conifer stem density would result in lower nest success.
- 7) Nest height. Nest height could explain some variation in nest success. Best and Stauffer (1980) found that the percentage of nests successfully fledging young increased significantly as nest height increased. Higher nests may also provide a better vantage point for brooding females to detect potential predators. Therefore, I hypothesized a positive association between nest height and nest success.

8) Concealment and woody vegetation. Several studies have proposed a tradeoff between nest concealment and ability for a quick escape by brooding parents. For example, Götmark et al. (1995) suggested a tradeoff between nest concealment and a view of the surroundings for brooding adults to detect predation. Likewise, Burhans and Thompson (2001) found that brooding females at less concealed nests increased flushing distance compared to those at greater concealed nests, suggesting an optimum value for nest concealment. I hypothesized that greater concealment around the nest would aid in hiding the nest from predators by concealing adult and nestling movements. I also hypothesize that a decrease in woody vegetation around the nest would enable brooding adults to detect approaching predators and make a quick escape.

I used a hierarchical approach to model nest success to keep the number of models run relatively low. I first ran a constant daily survival rate (DSR), equivalent to the Mayfield estimator (Mayfield 1961). I then ran the main effects of habitat covariates thought to influence DSR individually. Next, I included the additive effect of multiple habitat covariates that were biologically meaningful. I then added linear and/or quadratic time effects to each of the previous models if there was biological justification (i.e. I hypothesized a linear trend with nest concealment because I expected concealment to increase throughout the nesting season, corresponding to leaf-out). Lastly, I added minimum daily temperature to the models with the most support, and selected the final model with the lowest AICc value (Table 7).

#### *Population finite rate of increase*

To estimate population growth rate, I used the equation

$$\lambda = \Phi_{adult} + \Phi_{juvenile} \times SF$$

Where  $\lambda$ = population growth rate,  $\Phi_{adult}$  = annual adult survival,  $\Phi_{juvenile}$  = annual juvenile survival, and SF= seasonal fecundity, expressed as the number of female young fledged per female per year, assuming a 1:1 sex ratio. This equation also assumes that populations are closed to immigration and emigration. If  $\lambda > 1$ , then I assumed the population was self-sustaining.

Because I did not have marked individuals, I used regional estimates of adult and juvenile survival from constant-effort mist netting studies obtained from 1992-2005 (NBII/MAPS Avian Demographics Query Interface for MAPS data <http://www.birdpop.org/nbii/NBIIHome.asp>, accessed 8 Mar 2012). Due to the large scale at which MAPS operates, these estimates are likely underestimated for our study due to dispersal between breeding seasons (Tweedt et al. 2010). I expect regional estimates to give good approximations of adult and juvenile survival because of similar processes acting at the regional scale; however, due to the lack of information on juvenile survival, I varied juvenile survival from 25% to 100% of adult survival (Tweedt et al. 2010). I used the equations of Ricklefs and Bloom (1977) as adapted by Purcell (2006) to calculate seasonal fecundity (SF). The inputs needed for the equations were: 1) the length of the laying season (L) estimated from the number of nests initiated each week, which accounts for the fact that breeding effort is not constant throughout the season; 2) daily nest mortality rate (m), calculated as 1-DSR from MARK; 3) the probability that a nest survives the nesting interval to fledge at least 1 young ( $p_s$ ), calculated as  $DSR^T$ , where T = the nesting interval; 4) the probability that a nest does not survive the nesting interval ( $p_f$ ), taken as the complement of  $p_s$ ; 5) the number of fledglings per successful nest (fl); (6) the delay before a new clutch is initiated after failure ( $r_f$ ); 7) the delay before a new clutch is initiated after successfully fledging a brood

( $r_s$ ). For field sparrows, I used values of 8 and 14 days for  $r_f$  and  $r_s$  respectively. These are typical values for species that reneest quickly (Farnsworth and Simons 2001). Similarly, I used 5 and 15 days to represent delays in reneesting for indigo buntings (obtained from Westneat 1988, and Appendix C from Lloyd et al. 2005).

The first equation used to calculate the length of the laying season is given by:

$$L = 7e^{(-\sum p_i \ln(p_i))} ,$$

Where  $p_i$  is the proportion of nests initiated in week  $i$  (pooled across years). This equation determines the number of days in the breeding season that a potential clutch could be initiated.

For nests that I did not know the exact initiation date, I was able to back calculate from the hatch date or fledge date to the onset of nest initiation  $\pm 2d$ .

Following Ricklefs and Bloom (1977), nest initiation rate was calculated as:

$$I = \frac{m}{p_f + m(p_s r_s + p_f r_f)} ,$$

Where  $m$ ,  $p_f$ ,  $p_s$ ,  $r_s$ , and  $r_f$  are as defined above. I used my estimates from my study to calculate  $m$ ,  $p_s$  and  $p_f$ , and obtained estimates for  $r_s$  and  $r_f$  from the literature. From the equation above, I calculated the rate at which young fledge (Purcell 2006) as:

$$Fr = fl \times p_f \times I ,$$

Where  $Fr$  is fledgling rate,  $fl$  is the number of young fledged per successful nest (calculated from this study),  $p_f$  is the probability of nest failure and  $I$  is nest initiation rate, as calculated above.

Lastly, I used the equation:

$$SF = \frac{Fr \times L}{2}$$

To calculate seasonal fecundity, expressed as the number of female offspring produced per successful female and assumes a 50:50 sex ratio. The equations described above used to calculate seasonal fecundity assume that: 1) all individuals begin laying on day 1 of the breeding season, 2) no heterogeneity in nest failure probabilities (i.e. nests fail with the same probability ( $m$ ) each day), 3) all successful nests fledge the average number of fledglings produced per successful nest, and 4) all individuals whose nest fails re-nest after  $r_f$  days unless the end of the laying season has been reached (Lloyd et al. 2005).

## **Results**

I found a total of 112 nests of 20 different species during the 2010 and 2011 breeding seasons (Table 2). In 2010 we were interested in finding as many nests of all species on reclaimed mine sites. However, in an attempt to increase usable sample size in 2011, we focused on the 2 most common species identified on our study area. We used only the nests of indigo buntings and field sparrows in our analyses (see Appendix B for habitat descriptions of nests from other species).

### *Nest site selection*

#### *Field Sparrow*

When compared to random points, field sparrow nests tended to have higher percent cover of woody vegetation <1m tall and higher densities of stems >1m tall and less than 3cm DBH for all nests combined (Table 3). When I compared used nest sites to random points for early nests, however, I found no significant differences in the habitat characteristics that we measured for field sparrows (Table 4). In contrast, field sparrows seemed to be selecting for nest sites with greater percent woody vegetation cover, higher densities of stems and greater

understory cover from 0.5m tall to 2.5m tall later in the nesting season (Table 5). This suggests that field sparrows are possibly more selective in choosing nest sites later in the breeding season.

When I compared successful versus failed nests for field sparrows, I found successful nests had higher concealment around the nest, lower percentage of leaf cover, and greater densities of conifer stems and large trees (>15cm DBH; Table 6). When I analyzed successful-versus non-successful nests by time of season (i.e. early vs. late), I found that successful field sparrow nests earlier in the season had higher densities of conifer stems (Table 7). In contrast, successful field sparrow nests later in the season tended to have higher nest concealment, lower percent leaf cover and lower densities of large trees (>15cm DBH; Table 8).

#### *Indigo bunting*

Indigo bunting nests had higher percentages of canopy, leaf and bare ground covers than random points. Nest sites also tended to have higher densities of deciduous stems and small trees (4-10 cm DBH), and greater understory cover from 0.5-2.5m tall (Table 9). Early indigo bunting nest sites also had higher percent canopy cover, higher densities of deciduous stems and small trees (4-10 cm DBH), and greater understory cover from 0.5-2.5m tall than random plots (Table 10). In contrast, late season indigo bunting nests had higher percent cover of canopy, leaf and bare ground, higher densities of deciduous stems and greater understory density from 1.5m to 2.5m tall (Table 11).

When compared to successful indigo bunting nests, failed nests had higher percent covers of coarse woody debris and moss (Table 12). The only variable that seemed to influence nest success during the early season was % coarse woody debris, with successful nests having lower % cover of coarse woody debris (Table 13). In contrast, successful late season Indigo bunting nests had much lower densities of deciduous stems than non-successful nests (Table 14).

### *Densities*

We had 239 detections for field sparrows and 570 detections for indigo buntings during the 2011 breeding season. Two cover types (PRE and MIX) had fewer than 20 detections for field sparrows so were dropped from analysis. There were >20 detections in all cover types for indigo buntings, so all cover types were included in their analysis. The best model selected for both species included the hazard rate key function with no adjustment terms (Table 15). The best model for field sparrows was post-stratified by observer; whereas, the best model for indigo buntings included the additive effect of observer and cover type as covariates. The best model selected for each species fit the data reasonably well based on Kolmogorov-Smirnov (K-S) test statistics ( $D_n=0.08$ ,  $P=.273$  for field sparrows;  $D_n= 0.055$ ,  $P = 0.06$  for indigo buntings). The top model for both species had a coefficient of variation < 12%. Field sparrows had a global density estimate of 0.29 males/ha (0.226-0.352 95% CI), or 0.570 birds/ha assuming a 1:1 sex ratio. Indigo buntings were the most numerous bird on our study sites and had a global density estimate of 0.95 males/ha (0.821-1.101 95%CI), or 1.89 birds/ha assuming a 1:1 sex ratio.

### *Nest success*

We found a total of 51 field sparrow nests and 30 indigo bunting nests for the 2010 and 2011 breeding seasons combined. Of these 81 nests, 43 (53%) failed. Of the 43 nests that failed, 3 (6%) were abandoned, 35 (81%) were depredated, 3(6%) were lost due to poor nest placement (woven between 2 substrates that grew at different rates and caused the nest to tip), and 2 (5%) failed due to unknown causes. Only one indigo bunting nest was parasitized by a cowbird, but was depredated before the eggs hatched. While we allocated search effort evenly among the various reclamation cover types, we only found nests of field sparrows and indigo buntings in pasture, traditional shrub, and forest reclamation cover types, suggesting the other cover types may not provide adequate nesting habitat for either of these species. The nesting season,

calculated by taking the earliest date a nest was found in either year for each species and counting the successive number of days until the last day the last nest was active in either year, was 70 and 79 days long for indigo buntings and field sparrows, respectively.

The model with the additive effect of concealment and woody vegetation <1m tall was selected as the best model for field sparrow daily nest survival (DNS) based on minimum AIC<sub>c</sub> (Table 16). Daily nest survival rate (SE) for field sparrows using this model was 0.96 (0.009). Assuming a 20 day interval for nesting (12d incubation, 8d nestling), this equates to an interval nest survival estimate of 48%. Odds-ratio confidence intervals suggested that nest survival for field sparrows increased with increasing % vegetative concealment around the nest (OR<sub>95</sub> = 0.80, 7.71), and decreased with increasing % woody vegetation <1m around the nest (OR<sub>95</sub> = -6.48, -1.15; Figs. 3,4).

Year was not included in any of the top models for field sparrows ( $\Delta$ AIC<sub>c</sub> = 8.26), indicating nest survival was similar between years (Table 16). Although the addition of minimum daily temperature to the model with the additive effect of concealment and woody vegetation had fair support (Table 16), examination of the model deviances indicates minimum daily temperature is “pretending” and therefore unimportant (Anderson 2008). There also appeared to be a weak association between daily nest survival of field sparrows and density of conifer stems (Table 16) Daily nest survival estimates were not influenced by reclamation type for either species; however, the Forest Reclamation (FRA) cover type appears to be slightly better for field sparrows (Figure 5, Tables 16, 17).

No habitat characteristics measured for indigo buntings were related to nest success; instead, year and nest stage effects appeared to be the best predictors of nest success for indigo

buntings (Table 17). Minimum daily temperature was included among the top models for indigo buntings; however, the 95% confidence interval for the main effect bounded zero. Daily nest survival (SE) for indigo buntings, estimated from the best model in the set was 0.98 (0.010) and 0.91 (0.024) for 2010 and 2011 respectively (Figure 2). Assuming a 22 day interval for nesting (12d incubation, 10d nestling), this equates to interval nest survival estimates of 63% and 13% for 2010 and 2011.

#### *Population status*

The number of days that a clutch could potentially be initiated (L) was 44 and 60 for indigo buntings and field sparrows respectively. These results are very close to the length of the nesting season used in program MARK to model nest success. If we assume a nesting interval of 22 days for indigo buntings and 20 days for field sparrows, then an indigo bunting nest initiated on day 44 (the last potential initiation date as determined from Ricklefs and Bloom 1977), would correspond to a 66 day nesting season. This is very close to the 70 day nesting season length used to model DSR in program MARK. Likewise, a field sparrow nest initiated on the last potential initiation date would correspond to an 80 day nesting season, close to the 79 day season used to calculate DSR in MARK. Based on the number of nests initiated each week it is likely that more than one brood is probable both species (Figs. 6, 7). We estimated seasonal fecundity to be 1.04 female offspring/adult female for indigo buntings when daily nest survival rates were averaged across 2010 and 2011. When separated by year, seasonal fecundity was 1.61 and 0.61 female offspring/adult female for 2010 and 2011 respectively. Field sparrows had a higher seasonal fecundity estimate of 1.83 female offspring/adult female.

When considering population growth rate of indigo buntings ( $\lambda$ ) for various combinations of adult and juvenile survival, and daily nest survival rates, adult survival generally had to be

greater than 0.8 (Figure 8). When I considered MAPS estimates of adult survival rates for indigo buntings at different combinations of juvenile survival and nest survival rates, populations were sustainable ( $\lambda > 1$ ) only if juvenile survival was  $\geq 75\%$  that of adult survival, and daily nest survival rates were  $\geq 0.979$  (Figure 9).

Because there was no evidence that daily nest survival for field sparrows varied between years, I considered a range of nest survival estimates obtained from the literature to compare population growth at different values of nest survival (Figure 10). Burhans et al. (2002) reported a daily nest survival rate of 0.936 for field sparrows in Missouri, and Giocomo et al. (2008) estimated field sparrow DSR to be 0.920 in Kentucky and Tennessee. When considering this range of values for DSR, in general adult survival had to be greater than 0.8 for the population to be self-sustaining. However, if juvenile survival was at least 75% of adult survival, and DSR was 0.964, then populations could be self-sustaining with adult survival at only 0.5 (Figure 10). When I compared population growth rates of field sparrows using the estimated adult survival of 0.358 from MAPS breeding bird data, populations were not sustainable at any combination of juvenile and nest survival estimates (Figure 11).

## **Discussion**

### *Nest site selection*

Successful field sparrow nests had higher percent concealment at the nest site scale when compared to unsuccessful nests. Because I could not determine where a bird would place its nest within the substrate vegetation at a random location, I was not able to compare nest site characteristics at used nest- and random nest locations. At the nest plot scale, field sparrow nests had higher % woody vegetation <1m tall compared to random locations. Because my nest survival results indicated that DSR decreased as % woody vegetation increased, these results

suggest habitat selection for nest placement on reclaimed mine lands may be mal-adaptive. There were no significant differences in habitat characteristics at used sites versus random locations for early season nesters; however, 7 of the 19 measured habitat characteristics were significantly different for late season nests. These results indicate that field sparrows became more selective in relation to nest placement later in the season, perhaps due to shifts in the predator community, or activity.

Sperry et al. (2008) suggested that predator activity drives seasonal patterns of bird nest success. Altering nest placement in response to predation is one strategy to improve nest success (Lima 2009). Thus, field sparrows may have been more selective in choosing nest sites later in the nesting season, when risk of predation by snakes is greater (Sperry et al. 2008). Alternatively, field sparrows may have been selecting sites with increased food availability (Martin 1987); however, Best (1977) found that breeding adult field sparrows primarily foraged in woody vegetation in the early season (May and June); whereas more food items given to nestlings later in the season (July and August) came from the ground or in more open habitats. Another explanation for seasonal shifts in nest- site selection could result from choosing nest sites with a more favorable microclimate (Robertson 2009). More vegetation around the nest could provide shade during the hotter months of July and August. Lastly, perhaps nest sites with favorable characteristics, such as high concealment, were not available until after leaf-out occurred later in the season.

Indigo buntings did not exhibit the same seasonal shift in nest site selection seen in field sparrows. While there were significant differences between used and random nest locations, no measured habitat characteristics influenced daily nest survival rates. One possible explanation for this could be that a diverse guild of predators precludes predictably safe nesting sites (Filliater et

al. 1994). Another explanation could be that we did not measure the right habitat characteristics. In the analysis comparing successful versus failed nests, however, failed nests had more coarse woody debris and greater stem densities. Both coarse woody debris and higher stem densities could attract other prey items, or other nest predators, such as small rodents that would in turn result in higher densities of predators near nest sites.

### *Densities*

Our results indicate that densities did not differ significantly among the different reclamation types for either species. Density estimates for field sparrows were similar to estimates obtained for other grassland and shrubland sparrows on reclaimed mine lands in Pennsylvania (0.272 birds/ha for grasshopper sparrows, and 0.138 birds/ha for Henslow's sparrows, Mattice et al. 2005). Also, the density estimates we report here are within the range of values reported for field sparrows and indigo buntings in forested riparian buffers in Missouri (0.37– 1.83 birds/ha, Peak and Thompson 2006).

### *Nest success and population sustainability*

Daily nest survival estimates did not appear to be influenced by reclamation cover type; however, we were only able to locate nests in 3 of the 7 cover types considered, likely as a result of the remaining 4 cover types being older, and in more advanced successional states. Additionally, sample sizes in our study may not have been large enough to detect an effect among the 3 cover types where nests were found.

Percent concealment around the nest and % woody vegetation within the nest patch seemed to influence daily nest survival for field sparrows, and our results support the hypothesis of a tradeoff between nest concealment and a view of the surroundings (Götmark et al. 1995). Despite our daily nest survival estimates being higher than most other studies for field sparrows,

we still found evidence that populations may not be sustainable and reclaimed mine lands could potentially be acting as population sinks. Our estimate of DNS for field sparrows was higher than reported on reclaimed mine lands in Indiana (0.919; Galligan et al. 2006). Galligan et al. (2006) did not find evidence that reclaimed mine lands in Indiana were acting as population sinks. Reclaimed surface mines on their study consisted of much larger areas reclaimed using the same regimen. In contrast, mines on our study sites were reclaimed using a variety of techniques (i.e. grazed pasture, and forest reclamation) for the purposes of reclamation research. Thus, our study sites were a mosaic of smaller patches of vegetation differing in age and structure.

There was more model uncertainty in the models of DNS for indigo buntings; however, year was included in each of the top models. In 2010, indigo buntings had a higher DSR than field sparrows, but the opposite was true for 2011. In 2010, our estimates of daily nest survival(DNS) were higher than reported for early successional species in Pennsylvania (0.944; Schill and Yahner 2009), and on bottomland forest restorations in Mississippi (0.976; Tweedt et al. 2010). The average DNS from the 2010 and 2011 field seasons combined was comparable to the estimates of Schill and Yahner (2009). In 2011, however, DNS on our study sites was much lower than reported in the literature. The 2011 field season was wetter than the previous year at our study site, and wetter, cooler temperatures could have increased predator activity (Blouin-Demers and Weatherhead 2001). We also observed 3 indigo bunting nests fail due to poor nest placement. These nests were found woven between stems of small saplings and goldenrod early in the growing season. The goldenrod grew faster than the saplings and caused the nests to tilt and dump the contents onto the ground below. Given the DNS for indigo buntings from our study, and the regional adult survival estimates, populations were only self-sustaining if juvenile survival was at least 75% that of adult survival. In general, adult survival needed to be

approximately 0.8 in order for populations to be self-sustaining. We do take caution in the interpretation of these results, however, because of the broad assumptions made. We used regional survival estimates for adult birds and varied juvenile survival from 25 to 100% that of adults; however, adult survival may be higher on our study sites than the regional survival estimates used in the population models. Other studies of breeding birds in early successional habitat have not found evidence of declining populations despite having lower nest success than birds in our study (King and Bayers 2002, Chandler et al. 2009). This is because the adult survival rates, estimated empirically from powerline corridors in western Massachusetts (0.814; King and Bayers 2002) were much higher than the regional adult survival estimates we used. We note that if we used an adult survival estimate of 0.814, then populations would have been sustainable for all levels of DNS observed in our study. We suggest future studies examine adult and juvenile survival on reclaimed mine lands. We also suggest future studies to examine the underlying mechanisms that caused the observed pattern of seasonal variation in nest site selection.

### **Conservation implications**

Although we found evidence that reclaimed surface mines may be acting as population sinks for 2 shrubland generalists, reclaimed surface mines still have the potential to provide high quality breeding habitat to early successional species. We report higher rates of daily nest success than other studies in different types of early successional habitats. Given our results, it seems unlikely that reproduction is limiting populations of early successional species, and instead, focus should be placed on documenting and understanding factors that may influence adult and juvenile survival. We recommend that future reclamation efforts focus on planting a mix of native tree and shrub species. Concealment was important for field sparrow nest success,

so we recommend planting low lying shrubs such as hawthorn and alternate leaf dogwood that would provide excellent nest concealment. We also found daily nest success for field sparrows negatively associated with percent cover of woody vegetation. Thus, we recommend planting sparser ground covers. We also recommend the lowest tree stocking density possible to meet the post-mined land use objective, while preventing canopy closure and loss of the vegetative undergrowth that many early successional species rely on. Lastly, we had 2 nests fail (nestlings found desiccated) after mechanical removal of autumn olive near the nests. Therefore, we recommend delaying autumn olive removal until post avian breeding season.

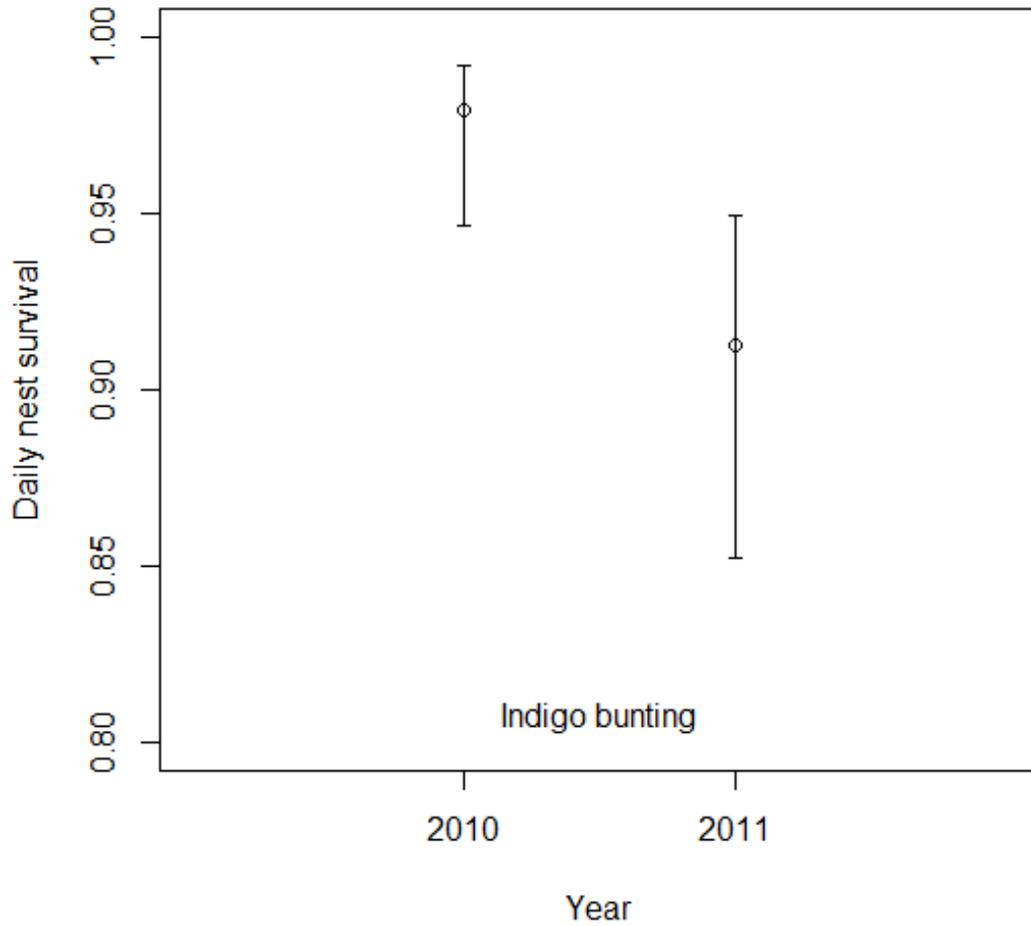
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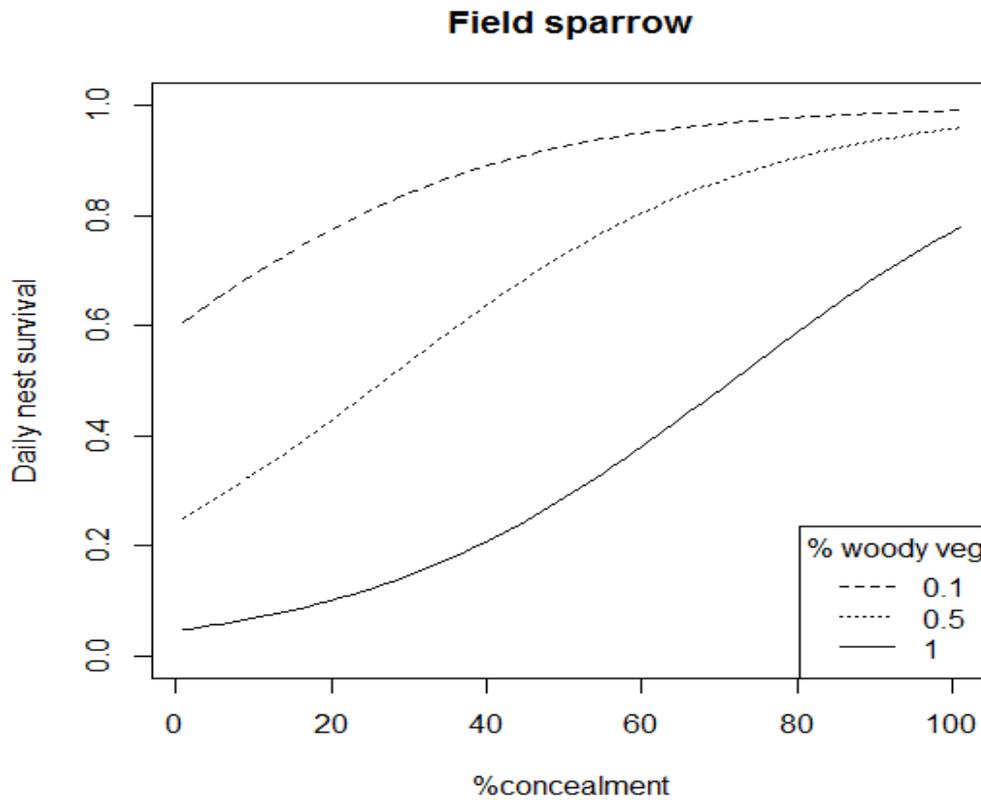
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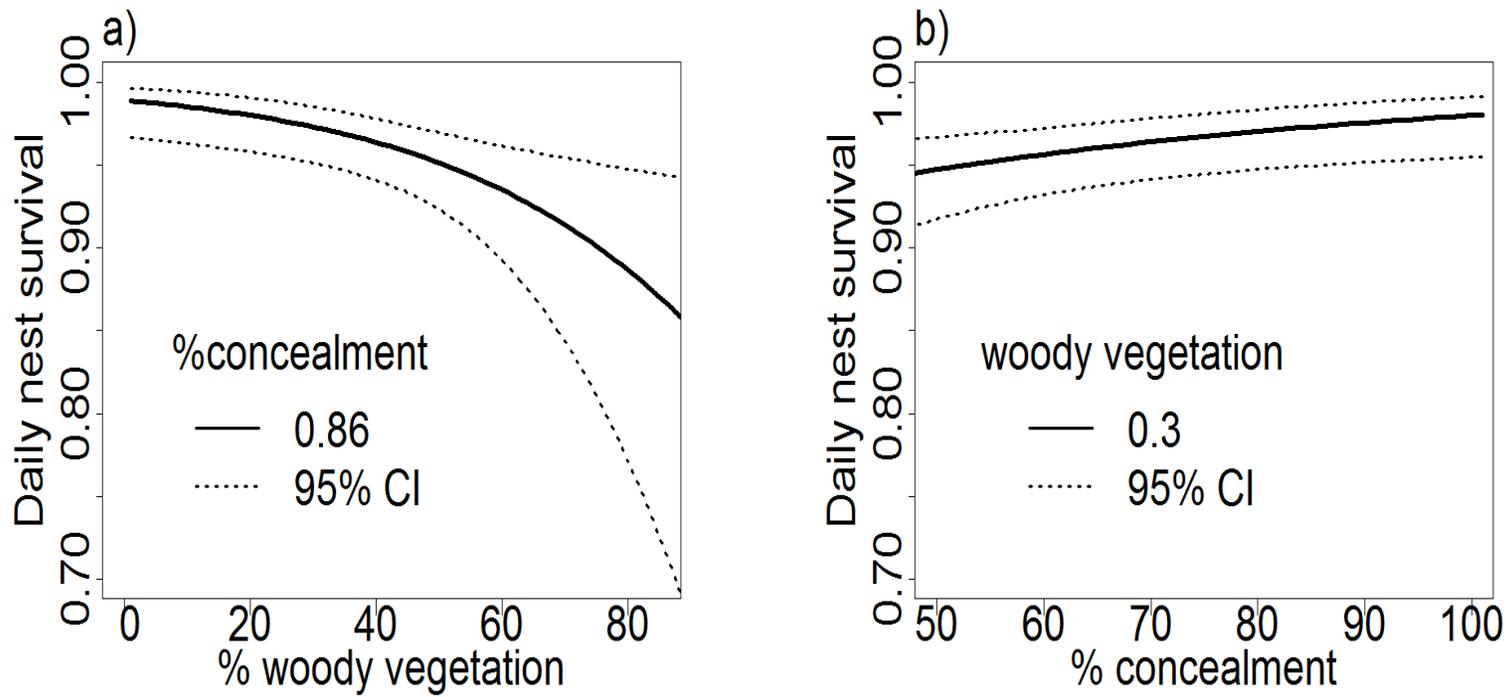
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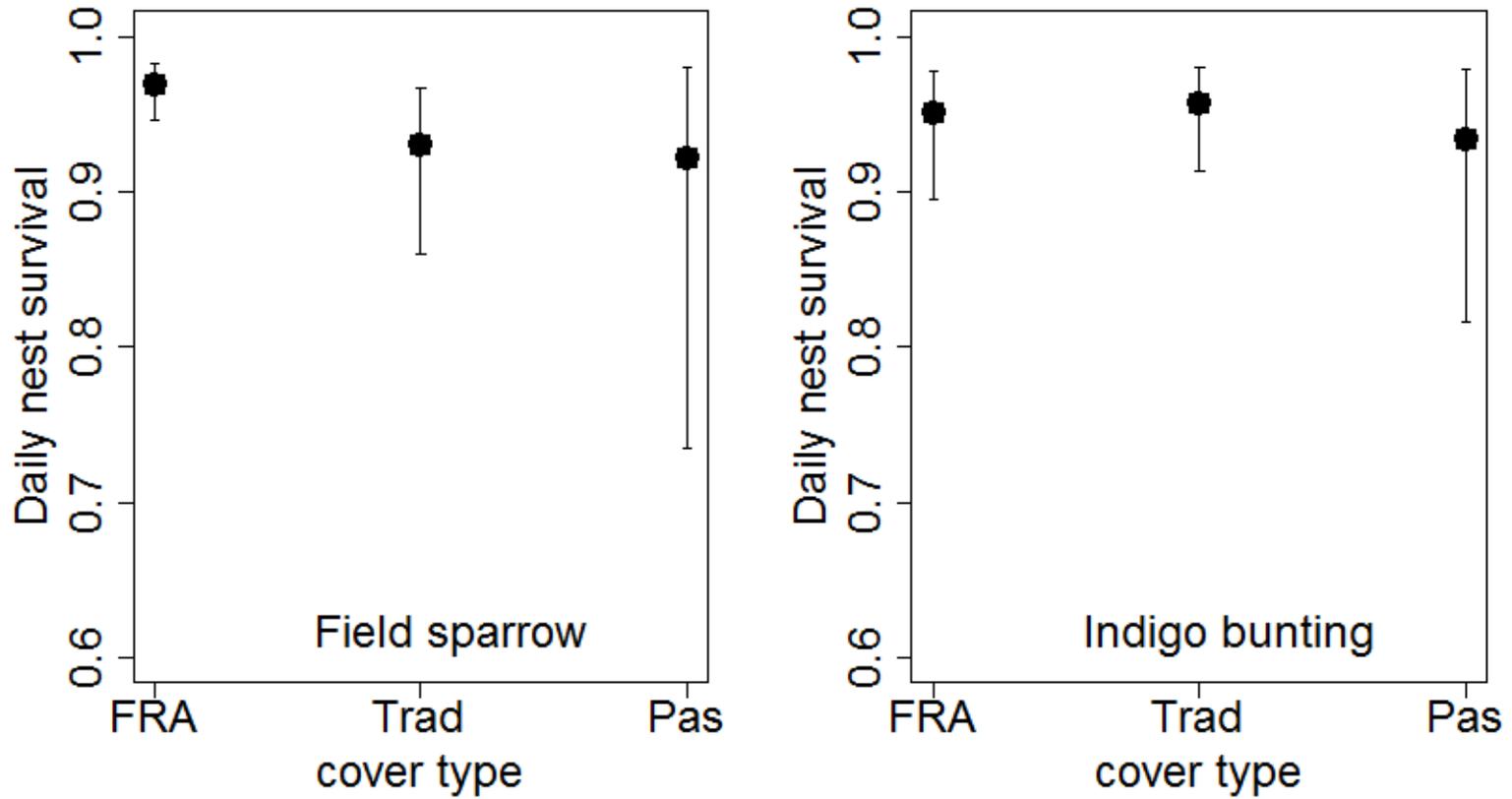
**Figure 2.** Effect of year on daily nest survival for indigo buntings on reclaimed mine sites in southwest Virginia for 2010 and 2011.



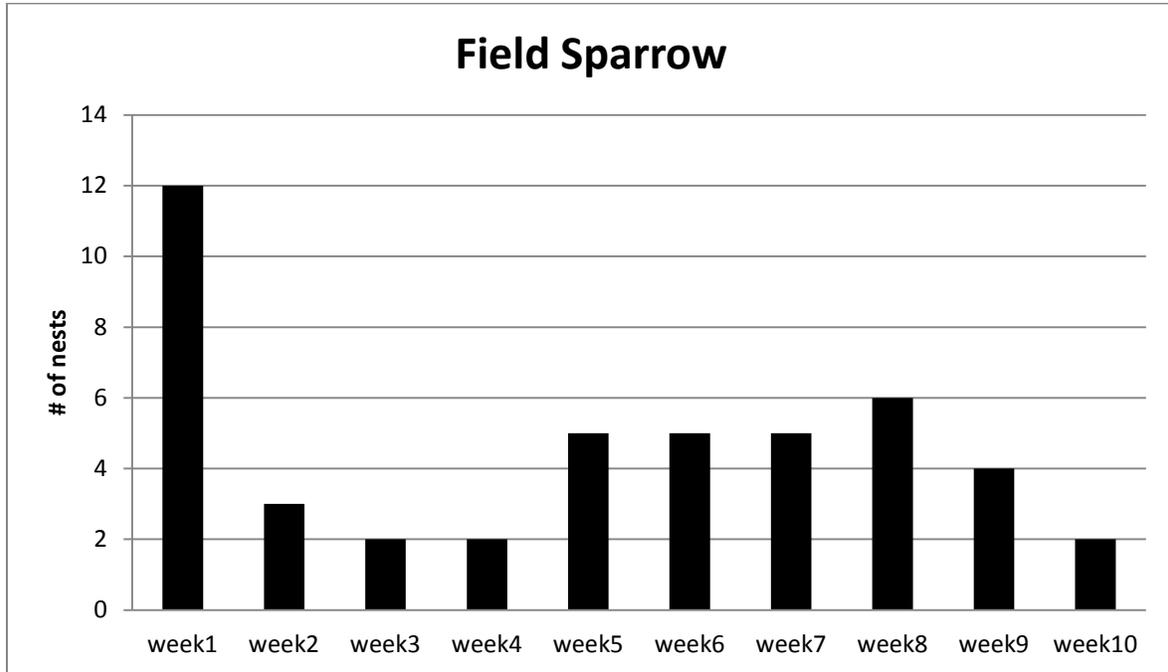
**Figure 3.** The additive effect on daily nest survival rate for field sparrows of % vegetative concealment around the nest (%concealment) for 3 levels (10%, 50% and 100%; corresponding to low, medium and high) of woody vegetation <1m tall around the nest.



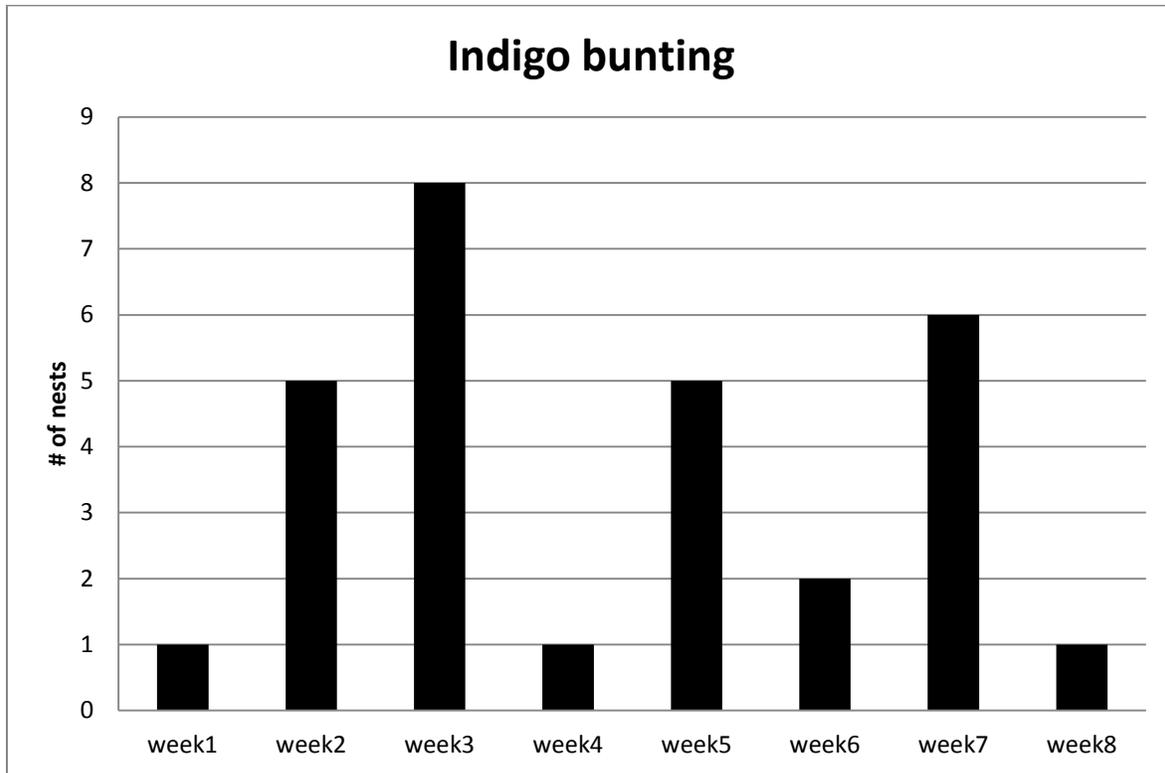
**Figure 4.** Relationship between daily nest survival probability and % woody vegetation <1m tall for the average value of nest concealment (a), and the relationship between daily nest survival probability and % nest concealment for the average value of % woody vegetation <1m tall (b) for field sparrows in 2010 and 2011.



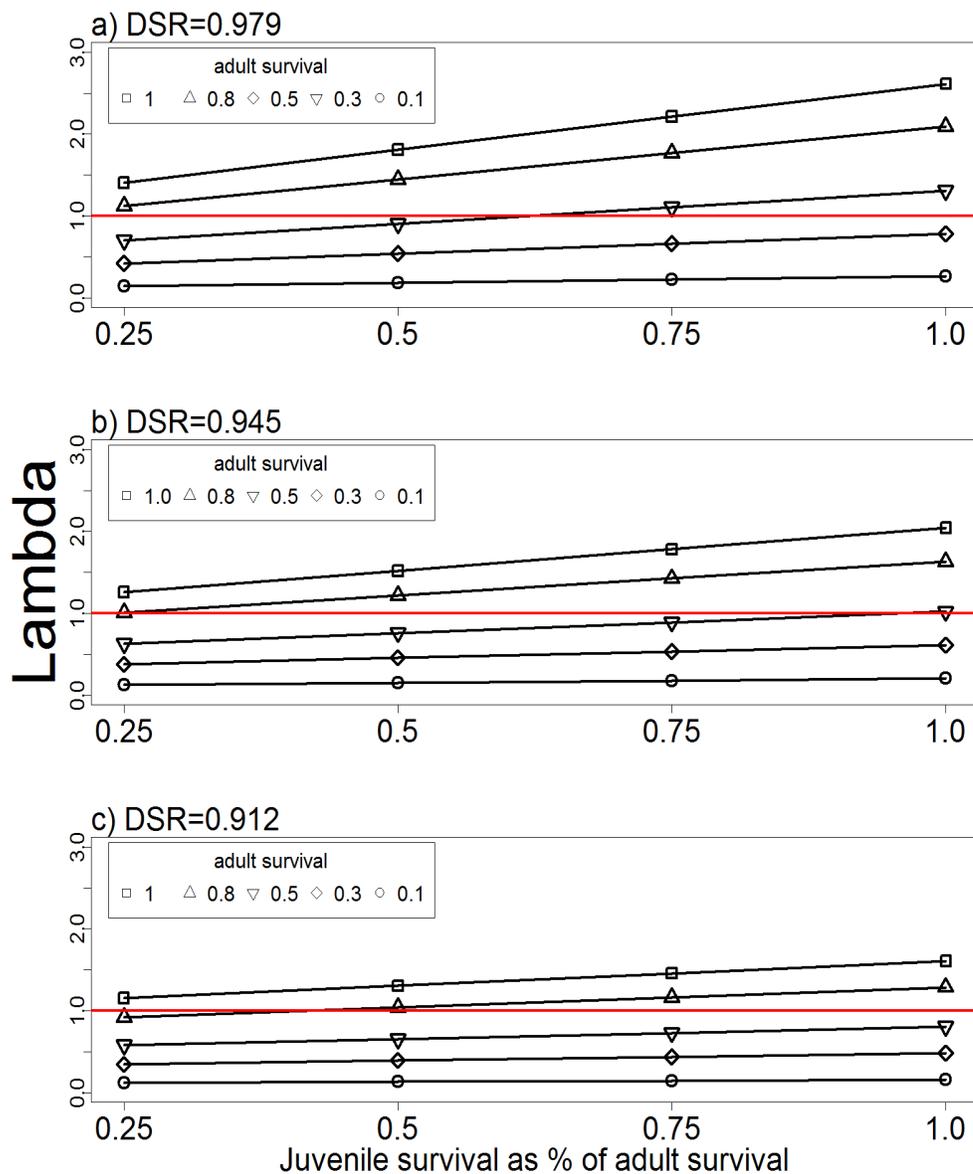
**Figure 5.** Effect of reclamation type on daily nest survival estimates for field sparrows and indigo buntings during 2010 and 2011.



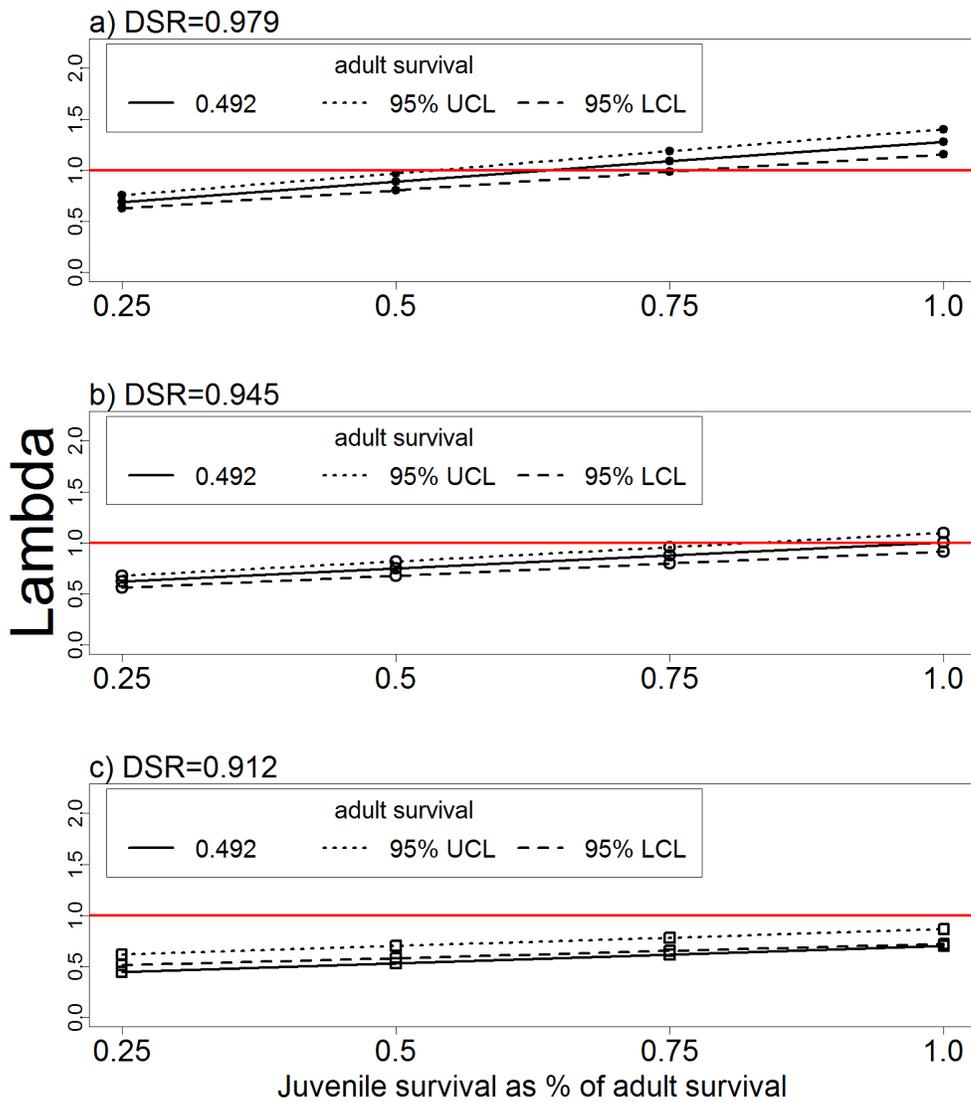
**Figure 6.** Number of field sparrow nests initiated each week during the breeding season in 2010 and 2011 on our study sites in southwest Virginia. Initiation date was back calculated from a stage transition, or the earliest possible initiation date was used.



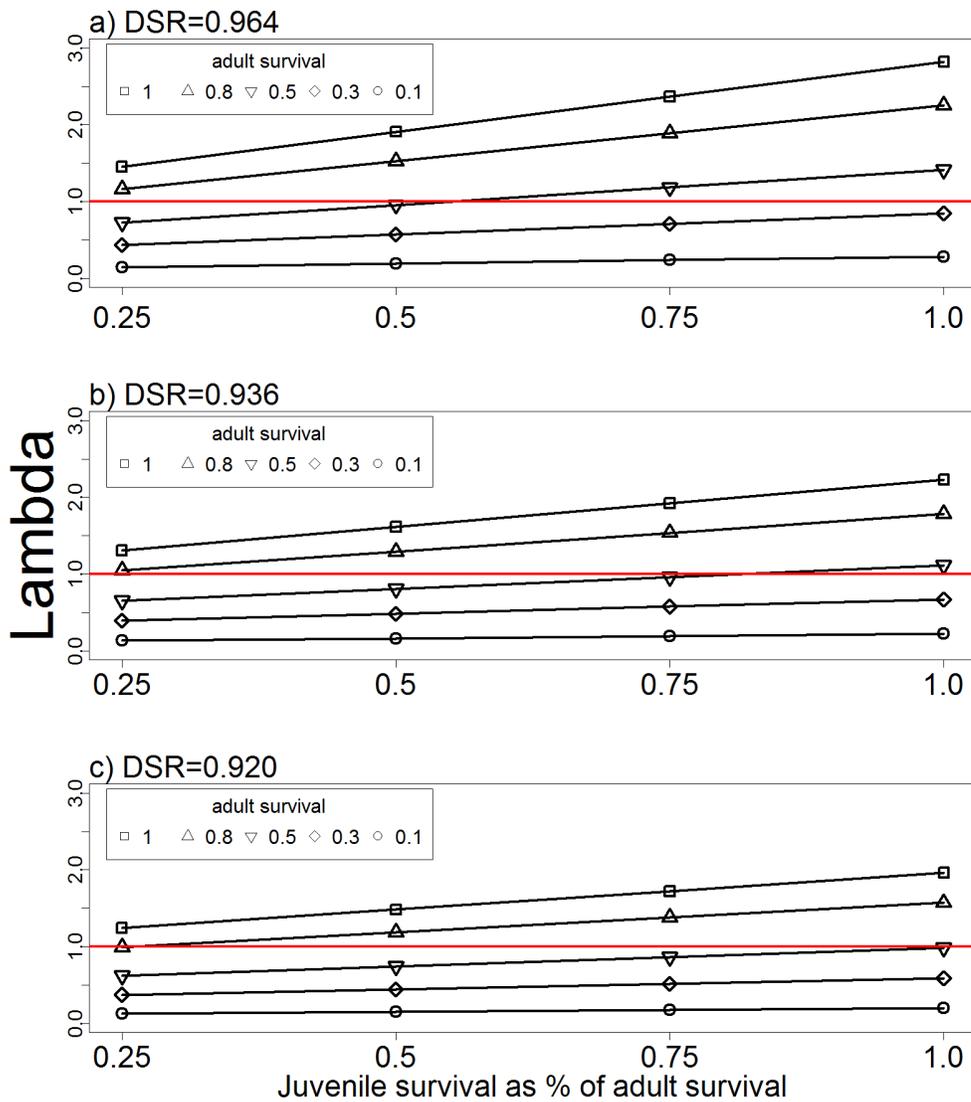
**Figure 7.** Number of indigo bunting nests initiated each week during the breeding season in 2010 and 2011 on our study sites in southwest Virginia. Initiation date was back calculated from a stage transition, or the earliest possible initiation date was used.



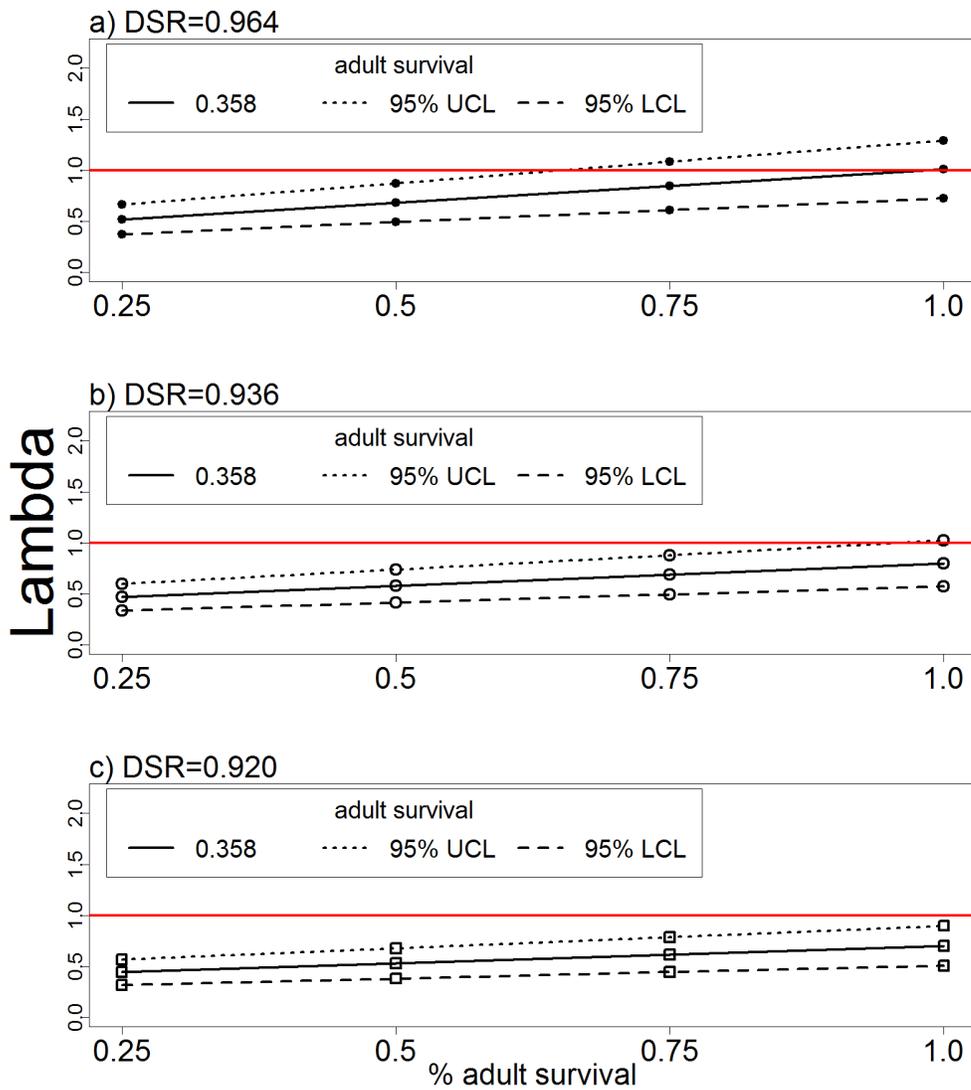
**Figure 8.** Population growth rates for indigo buntings calculated using estimates of daily nest survival rates for 2010 (a) 2011 (c) and the average DSR for 2010 and 2011 (b) for different levels of adult and juvenile survival. Populations are considered stable at any value above the red line in each graph.



**Figure 9.** Population growth rates and 95% confidence intervals for indigo buntings calculated using estimates of daily nest survival rates for 2010 (a) 2011 (c) and the average DSR for 2010 and 2011 (b) for different levels juvenile survival at an adult survival estimate of 0.492 (obtained from MAPS breeding bird database). Populations are considered stable at any value above the red line in each graph.



**Figure 10.** Population growth rates for field sparrows, calculated using estimates of daily nest survival rates from my study (a) Burhans et al. (2002) (b) and Giocomo et al. (2008) (c) for different levels of adult and juvenile survival. Populations are considered stable at any value above the red line in each graph.



**Figure 11.** Population growth rates for field sparrows, calculated using estimates of daily nest survival rates from my study (a) Burhans et al. (2002) (b) and Giocomo et al. (2008) (c) for different levels juvenile survival at an adult survival estimate of 0.358 (obtained from MAPS breeding bird database). Populations are considered stable at any value above the red line in each graph.

**Table 1.** Habitat characteristics measured at the nest site and nest patch scales for field sparrows and indigo buntings on reclaimed mine sites in southwest Virginia during the summer of 2010 and 2011. Both nest site and nest patch variables were only taken at actual nest locations; whereas only nest patch measurements were taken at random locations

Variable name	Variable Description
<b>nest site (1m radius)</b>	
Nest height (m)	height of the nest (m) from ground to nest cup
Distance to center nest substrate (m)	distance from center of nest to central axis of nest substrate (m)
Distance to edge nest substrate (m)	distance from center of nest to nearest edge of nest substrate (m)
Distance to edge of nest patch (m)	distance from nest substrate to the nearest edge of the nest patch, measured with a laser rangefinder (m)
% concealment around nest	% concealment visually estimated from 1m away from the nest and averaged from all sides, above and below
<b>nest patch (11.2m radius)</b>	
% canopy cover	% canopy cover measured using point intercept method
% grass cover	% grass cover measured using point intercept method
% leaf cover	% leaf cover measured using point intercept method
% bareground	% bare ground measured using point intercept method
% coarse woody debris	% coarse woody debris measured using point intercept method
% forb cover	% forb cover measured using point intercept method
% woody vegetation <1m tall	% woody vegetation measured using point intercept method
% moss cover	% moss cover measured using point intercept method
Conifer stem density	density of conifer stems (# stems/ha)
Deciduous stem density	density of deciduous stems (#stems/ha)
Density of trees in size class A	density of saplings (4-10cm DBH)
Density of trees in size class B	density of poles (10-15cmDBH)
Density of trees in size class C	density of trees (15+ cmDBH)
% understory cover 0-0.5m tall	% understory cover 0-0.5m estimated from Nudds board using Daubenmire classes
% understory cover 0.5-1m tall	% understory cover 0.5-1m estimated from Nudds board using Daubenmire classes
% understory cover 1-1.5m tall	% understory cover 1-1.5m estimated from Nudds board using Daubenmire classes
% understory cover 1.5-2m tall	% understory cover 1.5-2m estimated from Nudds board using Daubenmire classes
% understory cover 2-2.5m tall	% understory cover 2-2.5m estimated from Nudds board using Daubenmire classes
CV distance to nearest tree/shrub >4cm DBH	coefficient of variation of distances to the closest tree >4cm DBH within each cardinal direction

**Table 2.** Total number of nests for each species found on our study site in southwest Virginia during the summers of 2010 and 2011.

species	# nests
black-and-white warbler ( <i>Mniotilta varia</i> )	2
blue-gray gnatcatcher ( <i>Polioptila caerulea</i> )	1
blue jay ( <i>Cyanocitta cristata</i> )	2
brown thrasher ( <i>Toxostoma rufum</i> )	3
cedar waxwing ( <i>Bombycilla cedrorum</i> )	1
chipping sparrow ( <i>Spizella passerina</i> )	1
common yellowthroat ( <i>Geothlypis trichas</i> )	1
field sparrow ( <i>Spizella pusilla</i> )	51
gray catbird ( <i>Dumetella carolinensis</i> )	2
grasshopper sparrow ( <i>Ammodramus savannarum</i> )	1
golden-winged warbler ( <i>Vermivora chrysoptera</i> )	1
hooded warbler ( <i>Wilsonia citrina</i> )	1
indigo bunting ( <i>Passerina cyanea</i> )	30
northern cardinal ( <i>Cardinalis cardinalis</i> )	1
prairie warbler ( <i>Dendroica discolor</i> )	4
red-eyed vireo ( <i>Vireo olivaceus</i> )	2
red-winged blackbird ( <i>Agelaius phoeniceus</i> )	1
white-eyed vireo ( <i>Vireo griseus</i> )	2
wild turkey ( <i>Meleagris gallopavo</i> )	1
yellow-breasted chat ( <i>Icteria virens</i> )	4

**Table 3.** Multi response permutation procedure for matched pairs comparing nest site and random site habitat variables for field sparrows (all nests combined). Data were collected at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are means, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

variable	All nests (N=51)				
	used		random		P
	$\bar{x}$	SE	$\bar{x}$	SE	
% canopy cover	24	0.383	26	0.500	0.493
% grass cover	82	0.495	78	0.472	0.066
% leaf cover	27	0.441	26	0.518	1.000
% bareground	42	0.487	40	0.488	0.524
% coarse woody debris	1	0.058	1	0.045	0.662
% forb cover	85	0.328	83	0.377	0.529
% woody vegetation <1m tall	31	0.372	24	0.377	<b>0.015</b>
% moss cover	18	0.422	20	0.466	0.747
conifer stem density (#/ha)	23	0.935	10	0.757	<b>0.001</b>
deciduous stem density (#/ha)	1097	13.770	720	10.804	<b>0.004</b>
total density of trees in size class A	247	6.555	286	7.464	0.600
total density of trees in size class B	28	1.431	49	2.287	0.092
total density of trees in size class C	20	0.831	36	2.570	0.583
% understory vegetation cover 0-0.5m	92	0.194	90	0.279	0.611
% understory vegetation cover 0.5-1m	67	0.538	59	0.611	0.097
% understory vegetation cover 1-1.5m	52	0.590	45	0.604	0.164
% understory vegetation cover 1.5-2m	49	0.598	43	0.614	0.234
% understory vegetation cover 2-2.5m	47	0.574	39	0.640	0.130
CV of distance to nearest tree/shrub >3cm DBH	25	0.521	23	0.478	0.483

**Table 4.** Multi response permutation procedure for matched pairs comparing nest site and random site habitat variables for field sparrows (early nests only). Data were collected at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are means, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

variable	Early nests (N=26)				
	Used		Random		P
	$\bar{x}$	SE	$\bar{x}$	SE	
% canopy cover	21	0.752	28	0.915	0.278
% grass cover	81	0.963	73	1.035	0.072
% leaf cover	23	0.896	28	0.948	0.167
% bareground	48	0.933	49	0.913	0.697
% coarse woody debris	2	0.143	0	0.052	0.195
% forb cover	79	0.717	83	0.672	0.341
% woody vegetation <1m tall	37	0.790	28	0.788	0.105
% moss cover	11	0.611	12	0.638	0.599
conifer stem density (#/ha)	20	1.569	11	1.227	0.125
deciduous stem density (#/ha)	1074	31.162	759	22.870	0.164
total density of trees in size class A	321	15.697	417	18.321	0.364
total density of trees in size class B	18	1.651	40	2.664	0.189
total density of trees in size class C	20	1.684	15	1.368	0.859
% understory vegetation cover 0-0.5m	90	0.456	90	0.482	0.961
% understory vegetation cover 0.5-1m	56	1.165	50	1.377	0.511
% understory vegetation cover 1-1.5m	40	1.165	39	1.281	0.861
% understory vegetation cover 1.5-2m	38	1.171	40	1.329	0.843
% understory vegetation cover 2-2.5m	37	1.085	38	1.351	1.000
CV of distance to nearest tree/shrub >3cm DBH	26	1.070	26	1.070	0.828

**Table 5.** Multi response permutation procedure for matched pairs comparing nest site and random site habitat variables for field sparrows (late nests only). Data were collected at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are means, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

	Late nests (N=25)				
	Used		Random		P
	$\bar{x}$	SE	$\bar{x}$	SE	
% canopy cover	27	0.778	24	1.100	0.253
% grass cover	84	1.034	83	0.800	0.510
% leaf cover	30	0.858	24	1.139	0.113
% bareground	36	0.978	31	0.926	0.428
% coarse woody debris	1	0.075	1	0.119	1.000
% forb cover	92	0.444	82	0.851	0.015
% woody vegetation <1m tall	26	0.632	20	0.694	0.075
% moss cover	25	0.980	28	1.094	0.542
conifer stem density (#/ha)	25	2.189	9	1.808	<b>0.004</b>
deciduous stem density (#/ha)	1122	23.408	680	20.429	<b>0.003</b>
total density of trees in size class A	170	8.693	149	6.597	0.907
total density of trees in size class B	37	3.792	58	6.086	0.391
total density of trees in size class C	19	1.672	58	7.321	0.375
% understory vegetation cover 0-0.5m	94	0.283	90	0.641	0.509
% understory vegetation cover 0.5-1m	79	0.728	69	0.889	<b>0.030</b>
% understory vegetation cover 1-1.5m	64	0.996	50	1.100	<b>0.034</b>
% understory vegetation cover 1.5-2m	59	1.084	47	1.115	<b>0.055</b>
% understory vegetation cover 2-2.5m	57	1.086	41	1.217	<b>0.033</b>
CV of distance to nearest tree/shrub >3cm DBH	23	1.026	23	1.026	0.265

**Table 6.** Results from a 2 sample Multi-response permutation procedure test between successful and non-successful field sparrow nests (all nests combined) at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are mean, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

variable	All nests N = 51				
	success (n = 29)		failed (n = 22)		P
		SE		SE	
nest height	0.65	0.02	0.68	0.02	1.000
distance to center substrate	0.40	0.02	0.49	0.03	0.844
distance to edge substrate	0.27	0.01	0.24	0.01	0.884
distance to nearest patch	74	1.84	57	1.84	0.467
% nest concealment	87	0.41	81	0.57	<b>0.046</b>
% canopy cover	21	0.68	27	0.87	0.339
% grass cover	84	0.81	80	1.27	1.000
% leaf cover	20	0.64	37	1.10	<b>0.014</b>
% bareground	40	0.89	45	1.08	0.773
% coarse woody debris	1	0.07	2	0.18	0.544
% forb cover	87	0.55	84	0.81	0.428
% woody vegetation <1m tall	30	0.65	34	0.87	0.533
% moss cover	22	0.77	12	0.89	0.074
conifer stem density (#/ha)	33	2.04	7	0.64	<b>0.046</b>
deciduous stem density (#/ha)	1103	27.83	1089	24.41	0.396
total density of trees in size class A	230	12.18	270	14.20	0.602
total density of trees in size class B	23	2.24	35	3.83	0.673
total density of trees in size class C	10	1.22	33	2.20	<b>0.021</b>
% understory vegetation cover 0-0.5m	92	0.36	92	0.43	0.606
% understory vegetation cover 0.5-1m	67	0.93	67	1.31	1.000
% understory vegetation cover 1-1.5m	52	1.11	52	1.25	0.805
% understory vegetation cover 1.5-2m	49	1.15	49	1.22	0.642
% understory vegetation cover 2-2.5m	47	1.07	47	1.24	1.000
CV of distance to nearest tree/shrub >3cm DBH	35	0.61	44	1.29	0.158

**Table 7.** Results from a 2 sample Multi-response permutation procedure test between successful and non-successful field sparrow nests (early nests only) at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are mean, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

variable	Early nests (N= 26)				
	success (n= 14)		Failed (n = 12)		P
		SE		SE	
nest height	0.40	0.029	0.39	0.022	0.589
distance to center substrate	0.34	0.046	0.42	0.070	0.791
distance to edge substrate	0.25	0.021	0.18	0.012	0.427
distance to nearest patch	76	4.678	46	2.545	0.257
% nest concealment	91	0.901	86	0.846	0.137
% canopy cover	20	1.642	23	1.287	0.397
% grass cover	82	1.532	79	2.467	0.795
% leaf cover	15	1.418	32	2.045	0.079
% bareground	43	2.014	54	1.509	1.000
% coarse woody debris	1	0.152	3	0.417	0.291
% forb cover	80	1.373	77	1.554	0.497
% woody vegetation <1m tall	36	1.678	37	1.458	0.880
% moss cover	16	1.260	6	1.018	0.124
conifer stem density (#/ha)	34	3.628	4	1.208	<b>0.032</b>
deciduous stem density (#/ha)	1148	73.769	987	38.756	0.253
total density of trees in size class A	255	32.262	398	29.518	0.136
total density of trees in size class B	22	3.907	15	2.084	0.769
total density of trees in size class C	22	3.577	19	3.106	0.830
% understory vegetation cover 0-0.5m	90	0.941	91	0.892	0.858
% understory vegetation cover 0.5-1m	50	2.013	63	2.705	0.314
% understory vegetation cover 1-1.5m	34	2.091	48	2.582	0.268
% understory vegetation cover 1.5-2m	33	2.314	45	2.337	0.282
% understory vegetation cover 2-2.5m	34	2.270	40	2.026	0.434
CV of distance to nearest tree/shrub >3cm DBH	38	1.442	43	2.440	0.288

**Table 8.** Results from a 2 sample Multi-response permutation procedure test between successful and non-successful field sparrow nests (late nests only) at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are mean, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

	Late nests (N=25)				
	success (n = 16)		failed (n = 9)		P
		SE		SE	
nest height	0.88	0.034	1.11	0.060	0.271
distance to center substrate	0.44	0.041	0.59	0.069	0.490
distance to edge substrate	0.29	0.009	0.33	0.024	0.555
distance to nearest patch	72	2.592	73	5.402	1.000
% nest concealment	85	0.674	74	1.358	<b>0.046</b>
% canopy cover	23	1.052	33	2.546	0.393
% grass cover	86	1.617	81	3.015	0.761
% leaf cover	23	1.075	43	2.622	<b>0.045</b>
% bareground	38	1.514	33	2.888	0.925
% coarse woody debris	1	0.107	1	0.245	0.800
% forb cover	92	0.635	92	1.473	0.669
% woody vegetation <1m tall	23	0.719	31	2.397	0.574
% moss cover	28	1.548	21	2.756	0.544
conifer stem density (#/ha)	33	4.199	11	1.470	0.343
deciduous stem density (#/ha)	1064	35.982	1225	69.147	0.482
total density of trees in size class A	209	15.709	100	13.523	0.242
total density of trees in size class B	24	4.671	61	13.815	0.559
total density of trees in size class C	0	0.000	53	6.307	<b>0.001</b>
% understory vegetation cover 0-0.5m	94	0.434	93	0.856	0.758
% understory vegetation cover 0.5-1m	82	0.902	73	2.600	0.178
% understory vegetation cover 1-1.5m	67	1.650	59	2.482	0.164
% understory vegetation cover 1.5-2m	62	1.761	54	2.866	0.326
% understory vegetation cover 2-2.5m	58	1.645	56	3.349	1.000
CV of distance to nearest tree/shrub >3cm DBH	33	1.009	46	3.289	0.476

**Table 9.** Multi response permutation procedure for matched pairs comparing nest site and random site habitat variables for indigo buntings (all nests combined). Data were collected at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are means, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

	All nests (N=30)					
		Used		Random		P
			SE		SE	
% canopy cover	45	0.775	24	0.775	<b>0.001</b>	
% grass cover	76	0.598	74	1.049	0.834	
% leaf cover	33	0.908	19	0.770	<b>0.022</b>	
% bareground	41	0.783	28	0.636	0.052	
% coarse woody debris	4	0.431	2	0.238	1.000	
% forb cover	79	0.789	89	0.381	0.071	
% woody vegetation <1m tall	41	0.891	38	0.904	0.626	
% moss cover	13	0.644	9	0.522	0.124	
conifer stem density (#/ha)	20	2.623	8	1.247	0.625	
deciduous stem density (#/ha)	1865	39.741	784	21.691	<b>0.000</b>	
total density of trees in size class A	443	15.040	226	9.887	<b>0.003</b>	
total density of trees in size class B	46	2.573	44	3.766	0.489	
total density of trees in size class C	44	2.197	36	2.987	0.639	
% understory vegetation cover 0-0.5m	92	0.413	91	0.451	0.830	
% understory vegetation cover 0.5-1m	82	0.717	68	0.940	<b>0.044</b>	
% understory vegetation cover 1-1.5m	69	0.737	50	0.845	<b>0.008</b>	
% understory vegetation cover 1.5-2m	63	0.792	40	0.726	<b>0.000</b>	
% understory vegetation cover 2-2.5m	59	0.867	37	0.694	<b>0.001</b>	
CV of distance to nearest tree/shrub >3cm DBH	51	0.804	48	0.994	0.935	

**Table 10.** Multi response permutation procedure for matched pairs comparing nest site and random site habitat variables for indigo buntings (early nests only). Data were collected at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are means, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

	Early nests (N=14)				
	Used		random		P
		SE		SE	
% canopy cover	50	1.678	30	2.017	<b>0.030</b>
% grass cover	72	1.289	66	2.587	0.895
% leaf cover	27	1.786	24	2.122	0.668
% bareground	38	1.608	29	1.625	0.690
% coarse woody debris	7	1.327	4	0.727	0.875
% forb cover	79	1.780	87	0.845	0.401
% woody vegetation <1m tall	55	1.850	49	2.035	0.418
% moss cover	7	0.459	5	0.386	0.473
conifer stem density (#/ha)	38	8.135	17	3.969	1.000
deciduous stem density (#/ha)	1775	81.835	790	55.105	<b>0.028</b>
total density of trees in size class A	615	37.044	260	25.897	<b>0.003</b>
total density of trees in size class B	66	7.563	75	10.805	0.813
total density of trees in size class C	63	5.527	65	8.917	1.000
% understory vegetation cover 0-0.5m	94	0.514	86	1.305	0.188
% understory vegetation cover 0.5-1m	83	0.890	54	2.246	0.015
% understory vegetation cover 1-1.5m	72	1.306	46	1.980	<b>0.026</b>
% understory vegetation cover 1.5-2m	70	1.222	41	1.767	<b>0.002</b>
% understory vegetation cover 2-2.5m	63	1.714	37	1.675	<b>0.022</b>
CV of distance to nearest tree/shrub >3cm DBH	52	2.060	42	1.382	0.750

**Table 11.** Multi response permutation procedure for matched pairs comparing nest site and random site habitat variables for indigo buntings (late nests only). Data were collected at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are means, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

	Late nests (N=16)					
		Used		Random		P
			SE		SE	
% canopy cover	41	1.441	20	1.132	<b>0.010</b>	
% grass cover	79	1.115	82	1.629	0.882	
% leaf cover	38	1.796	15	0.988	<b>0.018</b>	
% bareground	44	1.546	27	1.013	<b>0.025</b>	
% coarse woody debris	0	0.078	1	0.170	1.000	
% forb cover	80	1.459	91	0.702	0.153	
% woody vegetation <1m tall	29	1.373	28	1.411	1.000	
% moss cover	18	1.559	13	1.255	0.289	
conifer stem density (#/ha)	5	0.853	0	0.000		
deciduous stem density (#/ha)	1944	78.966	780	35.037	<b>0.004</b>	
total density of trees in size class A	292	20.586	196	14.543	0.293	
total density of trees in size class B	28	2.056	17	3.519	0.088	
total density of trees in size class C	27	3.163	11	1.513	0.266	
% understory vegetation cover 0-0.5m	90	0.979	95	0.343	0.359	
% understory vegetation cover 0.5-1m	80	1.716	80	1.203	0.620	
% understory vegetation cover 1-1.5m	66	1.583	54	1.470	0.151	
% understory vegetation cover 1.5-2m	57	1.718	40	1.242	<b>0.027</b>	
% understory vegetation cover 2-2.5m	56	1.751	36	1.198	<b>0.010</b>	
CV of distance to nearest tree/shrub >3cm DBH	50	1.102	54	2.374	1.000	

**Table 12.** Results from a 2 sample Multi-response permutation procedure test between successful and non-successful field indigo bunting nests (all nests combined) at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are mean, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

variable	All nests				
	successful (n=13)		failed (n=17)		P
		SE		SE	
nest height	1.44	0.069	1.19	0.035	0.376
distance to center substrate	0.70	0.056	0.61	0.045	0.741
distance to edge substrate	0.86	0.043	0.68	0.037	0.339
distance to nearest patch	39	2.062	44	2.078	0.973
% nest concealment	80	0.961	72	0.972	0.233
% canopy cover	41	1.449	48	1.546	0.232
% grass cover	74	1.513	77	1.002	0.718
% leaf cover	27	1.861	38	1.719	0.320
% bareground	42	2.394	40	0.973	0.130
% coarse woody debris	0	0.000	6	0.995	<b>0.013</b>
% forb cover	87	0.985	74	1.679	0.139
% woody vegetation <1m tall	38	2.149	43	1.556	0.401
% moss cover	6	0.717	19	1.358	<b>0.054</b>
conifer stem density (#/ha)	12	1.868	27	6.077	0.645
deciduous stem density (#/ha)	1495	66.061	2149	79.494	0.197
total density of trees in size class A	589	43.339	331	18.624	0.249
total density of trees in size class B	35	3.395	55	5.629	0.707
total density of trees in size class C	41	4.357	46	4.345	0.861
% understory vegetation cover 0-0.5m	92	1.237	92	0.540	0.416
% understory vegetation cover 0.5-1m	85	1.983	79	1.060	0.141
% understory vegetation cover 1-1.5m	70	2.137	68	1.029	0.375
% understory vegetation cover 1.5-2m	64	2.350	63	1.056	0.421
% understory vegetation cover 2-2.5m	59	2.354	59	1.348	0.860
CV of distance to nearest tree/shrub >3cm DBH	52	2.055	50	1.331	1.000

**Table 13.** Results from a 2 sample Multi-response permutation procedure test between successful and non-successful field indigo bunting nests (early nests only) at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are mean, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

Variable	Early nests (N=14)				
	successful (n=6)		failed (n = 8)		P
		SE		SE	
nest height	0.85	0.034	1.07	0.084	0.212
distance to center substrate	0.82	0.154	0.86	0.122	0.840
distance to edge substrate	0.94	0.107	0.64	0.080	0.530
distance to nearest patch	35	2.032	48	5.095	0.419
% nest concealment	76	2.594	76	2.004	0.873
% canopy cover	46	2.907	53	3.504	0.488
% grass cover	79	3.047	67	2.138	0.248
% leaf cover	18	1.639	33	3.924	0.085
% bareground	39	4.577	37	2.499	0.887
% coarse woody debris	0	0.000	13	2.969	<b>0.017</b>
% forb cover	90	1.179	70	3.794	0.121
% woody vegetation <1m tall	54	5.759	55	2.478	0.541
% moss cover	3	0.680	10	0.818	0.068
conifer stem density (#/ha)	17	5.067	53	18.862	0.742
deciduous stem density (#/ha)	1862	180.603	1710	157.506	0.810
total density of trees in size class A	812	110.283	468	44.852	0.361
total density of trees in size class B	46	9.693	82	16.669	0.614
total density of trees in size class C	33	8.643	85	11.119	0.309
% understory vegetation cover 0-0.5m	95	0.936	93	1.054	0.723
% understory vegetation cover 0.5-1m	93	1.083	77	1.445	<b>0.006</b>
% understory vegetation cover 1-1.5m	79	2.904	66	2.223	0.262
% understory vegetation cover 1.5-2m	74	2.745	66	2.231	0.549
% understory vegetation cover 2-2.5m	68	3.930	59	3.145	0.678
CV of distance to nearest tree/shrub >3cm DBH	59	5.555	45	3.141	0.573

**Table 14.** Results from a 2 sample Multi-response permutation procedure test between successful and non-successful field indigo bunting nests (late nests only) at the Powell River project in southwest Virginia during the summers of 2010 and 2011. Values are mean, standard errors and Pearson type III p-values. Bold font indicates significance at the 0.05 alpha level.

Variable	Late nests (N=16)				P
	successful (n= 7)		failed (n= 9)		
		SE		SE	
nest height	1.95	0.137	1.311	0.511	0.205
distance to center substrate	0.59	0.080	0.359	0.422	0.542
distance to edge substrate	0.80	0.074	0.726	0.677	0.563
distance to nearest patch	44	5.091	40	31.613	0.950
% nest concealment	83	1.276	68	17.403	0.084
% canopy cover	37	2.912	44	25.672	0.533
% grass cover	70	3.030	86	11.844	0.135
% leaf cover	34	4.411	42	28.395	0.756
% bareground	44	5.141	43	13.229	0.151
% coarse woody debris	0	0.000	1	1.667	1.000
% forb cover	84	2.322	77	28.284	0.822
% woody vegetation <1m tall	25	1.487	33	28.186	0.300
% moss cover	8	1.743	27	29.686	0.179
conifer stem density (#/ha)	7	2.711	3	8.369	0.824
deciduous stem density (#/ha)	1180	70.859	2538	1378.501	<b>0.023</b>
total density of trees in size class A	398	60.243	209	229.069	0.542
total density of trees in size class B	25	4.141	31	37.191	0.966
total density of trees in size class C	47	9.127	11	33.474	0.168
% understory vegetation cover 0-0.5m	89	3.105	91	10.208	0.521
% understory vegetation cover 0.5-1m	78	4.918	82	22.811	0.897
% understory vegetation cover 1-1.5m	62	4.797	69	18.154	0.580
% understory vegetation cover 1.5-2m	55	5.396	59	18.339	0.549
% understory vegetation cover 2-2.5m	51	5.070	59	22.275	0.740
CV of distance to nearest tree/shrub >3cm DBH	44	2.155	56	19.537	0.508

**Table 15.** Model results and rankings for density estimates of field sparrows and indigo buntings run in program DISTANCE. Models are ranked based on  $\Delta\text{AICc}$  scores. K= number of model parameters, CV% = the percent of the coefficient of variation explained by the model, D = density of individuals expressed as #/ha, and SE are standard errors of the density estimates.

Model <sup>a</sup>	K	$\Delta\text{AICc}$	CV %	D	SE
<b>Field sparrow</b>					
HR(poly)strat(obs)	4	0	11.6	0.285	0.032
HR(poly)cov(obs)	3	0.63	4.5	0.554	0.025
HR(poly)	1	3.56	8.8	0.723	0.064
HR(poly)cov(cover)	5	4.63	4.6	0.554	0.025
HR(poly)strat(cover)	8	6.09	9.5	2.098	0.098
HR(poly)cov(obs+cover)	6	6.63	4.6	0.554	0.025
HR(poly)strat(obs)cov(cover)	10	9.73	4.7	0.276	0.018
HR(poly)strat(cover)cov(obs)	12	12.21	5.2	2.124	0.055
<b>Indigo Bunting</b>					
HR(poly)cov(obs+cover)	8	0	7.7	0.945	0.027
HR(poly)strat(obs)	4	0.4	5	0.463	0.032
HR(poly)strat(obs)cov(cover)	14	5.45	3.2	0.453	0.124
HR(poly)cov(cover)	7	8.95	3.1	0.888	0.028
HR(poly)strat(cover)cov(obs)	7	8.95	3.1	0.888	0.028
HR(poly)strat(cover)	12	9.85	6	6.175	0.143
HR(poly)cov(obs)	3	9.98	3	0.892	0.027
HR(poly)	1	34.5	5.5	1.239	0.069

<sup>a</sup> Model notation: HR = Hazard rate detection function; poly = polynomial expansion; strat = post-stratification; cov = covariate; obs = observer; cover = cover type.

**Table 16.** Model results from nest survival platform in program MARK. Results are Akaike weights ( $w_i$ ) and increase in Akaike's Information Criterion ( $\Delta AIC_c$ ) for a priori models thought to influence nest survival of field sparrows on reclaimed mine sites in southwest Virginia during the summers of 2010 and 2011. Number of parameters (K) includes intercept and main effects, and n is the effective sample size as calculated by Rotella et al. (2004).

Model <sup>a</sup>	Field sparrow (n=474)		
	K	$\Delta AIC_c$	$w_i$
Conceal+ woodveg	3	0	0.433
Conceal+ woodveg+mintempa	4	1.97	0.162
Cstemden	2	2.08	0.153
Woodveg	2	3.4	0.079
Woodveg+ T	3	5.42	0.029
Conceal	2	5.56	0.027
Null	1	6.26	0.019
Cover	3	6.73	0.015
Conceal+ T	3	6.78	0.015
Dstemden	2	7.33	0.011
Nestht	2	8.25	0.007
Year	2	8.26	0.007
Nestht+ T	3	9.56	0.004
Year+ stage	3	9.87	0.003
Year+ T	3	9.89	0.003
Year+ TT	3	9.96	0.003
Stage	4	11.59	0.001
Stage+ T	5	13.2	0.001
Stage+ TT	5	13.48	0.001

<sup>a</sup> Minimum daily temperature was only added to the model with the greatest support based on minimum  $AIC_c$

**Table 17.** Model results from nest survival platform in program MARK. Results are Akaike weights ( $w_i$ ) and increase in Akaike's Information Criterion ( $\Delta AIC_c$ ) for a priori models thought to influence nest survival of indigo buntings on reclaimed mine sites in southwest Virginia during the summers of 2010 and 2011. Number of parameters (K) includes intercept and main effects, and n is the effective sample size as calculated by Rotella et al. (2004).

Model <sup>a</sup>	Indigo bunting (n=325)		
	K	AICc	$w_i$
Year	2	0	0.289
Year+ mintempa	3	1.32	0.15
Year+ T	3	1.69	0.124
Year+ stage	3	1.87	0.113
Year+ TT	3	1.98	0.107
Dstemden	2	3.97	0.04
Woodveg	2	5.34	0.02
Conceal+ woodveg	3	5.67	0.017
Coarsewoodb	2	5.72	0.017
Null	1	5.86	0.015
Conceal	2	6.15	0.013
Conceal+ T	3	6.94	0.009
Woodveg+ T	3	7.21	0.008
Stage	4	7.3	0.007
Nestht	2	7.63	0.006
Cstemden	2	7.89	0.006
Stage+ T	5	8.68	0.004
Nestht+ T	3	8.78	0.004
Stage+ TT	5	9.16	0.003
Cover	3	9.53	0.002

<sup>a</sup> Minimum daily temperature was only added to the model with the greatest support based on minimum AIC<sub>c</sub>.

## **Avian community dynamics in response to vegetation restoration on reclaimed surface-mines in southwest Virginia**

### **Abstract**

Understanding community vital rates, such as species turnover, local extinction, and richness is important for identifying areas of conservation priority and evaluating the effectiveness of ecological restoration. Traditional methods of community vital rate estimation that do not account for imperfect species detections can lead to biased results and thereby misinform management decisions. I evaluated avian community dynamics within various surface-mine reclamation regimens over a 5-year period, using Pollok's robust design model with full time dependence and heterogeneity in capture and re-capture probabilities. Additionally, I used non-metric multidimensional scaling ordination techniques and multi-response permutation procedures to test whether changes in community composition were due to potential changes in vegetation throughout the 5-year period.

Estimates of local extinction and species turnover were comparable to estimates for the eastern temperate deciduous forest ecoregion. Additionally, estimated species richness was generally higher in the final year of the study than the first year for all cover types, except traditional pine. Reference forest, pasture and pine cover types had the highest rates of local extinction; however, this was balanced by a high number of local colonizing species for reference forest plots, leading to higher rates of annual turnover. Likewise, Traditional shrub cover types had the highest estimated number of local colonizing species; however, fewer species went extinct in this cover type, so species turnover was relatively low.

Results from NMDS ordination suggest the greatest amount of variation in the breeding bird community related to measured habitat attributes can be described by 2 axes. The first axis explained ~65% of the variability in the data and described a gradient ranging from open grassland to closed canopy forest; whereas the second axis explained ~17% of the variation in the data and described a gradient ranging from a sparse understory forest to a dense understory forest. Habitat characteristics changed significantly between 2007 and 2011 for traditional shrub, Pre-SMCRA and reference forest cover types, primarily along the second axis.

## **Introduction**

Many species in the eastern avifaunal biome, identified in the partners in flight North American landbird conservation plan, are declining as a result of habitat alteration and conversion (Rich et al. 2004). Mountain-top (surface) mining is identified in this plan as a major concern in the east for the threats posed to priority forest bird species of concern. However, the plan also identifies a need to manage adequate acreage of shrub communities to reverse the declines in priority shrub-dwelling bird species. Recently, Schlossberg and King (2009) indicated that current estimates of habitat availability and “adequate” acreage required for shrubland dwelling bird species may be too optimistic. While mountain-top mining can have drastic consequences for forest bird species, especially area-sensitive ones, it creates large tracts of early successional habitats used by many early successional species also in decline (Askins 2001, Scott et al. 2002, Bulluck and Buehler 2006) These conflicting interests in priority species conservation often lead to confusion among land managers about effective land management strategies for regional conservation efforts.

Several studies have suggested the potential benefits of reclaimed surface mines to various early successional wildlife species (Alliar 1978, Whitmore and Hall 1978, Galligan et al. 2006, Brenner 2007). Bulluck and Buehler (2006) proposed that early successional habitats, such as those created by surface mining can also benefit interior-dwelling species during the post breeding season. Carrozzino (2009) suggested that managing habitats for forest interior species of concern, such as the cerulean warbler (*Dendroica cerulean*), is compatible with managing for early successional species of concern like the golden-winged warbler (*Vermivora chrysoptera*) on reclaimed mine lands. For example, this can be accomplished by implementing a variety of reclamation techniques and staggering the timing of reclamation in different areas to obtain a diversity of vegetation and age structure that appeal to a wide variety of species. Similarly, Becker et al. (2011) found that certain forest management strategies increased the conservation index value by improving habitat quality for several conflicting priority species.

There are a variety of regulation compliant reclamation techniques used to restore vegetation on post-mined lands. These techniques are often dictated by the post-mining land use objective required by law (SMCRA 1977). Reclaiming surface-mined lands to forests has become an increasingly popular choice among landowners looking to increase the value of their post-mined lands. The forest reclamation approach (FRA) is an economically efficient method of restoring forests on mined lands at a rate faster than would occur through natural succession (Burger and Zipper 2009). In addition to the economic benefits, forests also provide ecological services such as carbon sequestration, water filtration, and wildlife habitat for a multitude of species. Understanding community dynamics such as species richness, turnover, and rates of local extinction and colonization are important when evaluating the effectiveness of ecological restoration (Balmford and Bond 2005, Johnson et al. 2009). With a better understanding of

factors that influence community dynamics in different vegetation restoration regimens, we can provide better recommendations to land managers for increasing the conservation value of reclaimed surface mines.

I evaluated breeding bird community dynamics in response to different vegetation restoration regimens on reclaimed mine lands in southwest Virginia, using a mark-recapture analysis with species detection, non-detection data. The specific reclamation types I considered for this study were managed hayland/pasture (PAS), shrub/scrub (SHB), mid-successional mixed (MIX), mid-successional pine (PIN), and forest reclamation (FRA). In addition to these 5 reclamation types, I also considered one cover type where no active reclamation took place, prior to mining law (PRE), and one cover type where no mining occurred (REF). General predictions about community-level vital rates were derived from the literature regarding species composition following disturbance. McArthur and McArthur (1961) suggested that species diversity increases with vegetation complexity. May (1982) reported a general increase in species richness and diversity through the various stages of succession, and a subsequent decrease in species richness with canopy closure. This pattern has also been documented by others (Karr 1968, Shugart and James 1973, Urban and Smith 1989). Rotenberry and Wiens (1980) documented a significant change in structural characteristics of vegetation over a three year period in shrubsteppe habitats, but this change was not matched by changes in bird species composition. Instead, they found evidence that vegetation floristics, rather than physiognomy was most important in structuring avian communities in sagebrush habitats. Brooks and Bonter (2010) found the greatest change in community composition to occur in early successional habitats as compared to mixed forest and pine plantations over a 35 year period. Likewise, Schlossberg and King (2009) found variation in species responses in a meta-analysis of studies examining the impacts of timber harvest on avian

community composition. They noted that some early successional species declined in number within 1 to 2 years post-harvest; whereas some did not decrease until around 10 years post-harvest. They attributed these findings to differences in species requirements for structural variation in nesting and foraging substrates. From the results of these earlier studies, I predicted that avian species richness will generally be lower in earlier successional habitats and increase with vegetation structural complexity. I expected species richness will peak in mixed and pre-SMCRA cover types, and will be lowest in reference forest and pine cover types. I also expected species turnover to be highest in early successional cover types and lowest in reference forest. Because of the high variance associated with early successional avian community structure (May 1982), I expected higher probabilities of local species extinction, and greater uncertainty in species richness estimates for early successional cover types.

The objectives of this study were to 1) evaluate community dynamics in different restoration regimens, 2) compare changes in habitat over a 5 year interval to changes in species composition and community dynamics, and 3) provide management recommendations for future reclamation efforts to help promote local species persistence and colonization on reclaimed mine lands.

## **Methods**

### *Study area and data collection.*

We conducted this study on the Powell River Project (PRP) and on surrounding reclaimed surface mines in Wise and Dickenson Counties, Virginia in 2007-2008 and 2010-2011. This region of the Southern Appalachian Mountains receives an average of 1193mm of precipitation annually. Mean average temperatures range from 0.38°C in January, to 21.05 °C in June (Holl 2002). The primary forest type is mixed mesophytic with oak (*Quercus spp.*) and

Hickory (*Carya spp.*) as the dominant overstory species. This area served as a refuge for many mesic flora and fauna species during the drier glacial epochs and is therefore known for its rich biologically-diverse taxa (Holl 2002). The Powell River Project is a cooperation between Penn Virginia Corporation (a large landholding company), Virginia Tech, and the Appalachian states. The 700 hectare property was extensively mined by the Ring Brothers Coal Company using deep mining and contour mining methods from the late 1950's through 1977 with the advent of the Surface Mining Control and Reclamation Act (SMCRA). In the early 1980's, Red River Coal Company began actively re-mining the area using second-cut contour mining methods and reclaiming in compliance with current SMCRA regulation. Much of the research conducted on the PRP is for the purpose of improving reclamation technologies that will enhance the post-mining land-use potential for landowners. Current research on the PRP include topics such as reclaiming mine lands to forests, enhancing sustainable beef cattle production on reclaimed mine lands, hydrology restoration and stream quality, and topsoil construction/overburden placement. Educational programs are conducted to disseminate research results to local landowners, mine operators, and educators (Zipper and Rockett 1998).

The sampling unit for this study was considered to be a patch of relatively homogeneous vegetation of approximately the same age, with only one reclamation approach applied within the patch. I used variable radius point counts (68 points total) to sample avian communities within each reclamation cover type or patch. The pasture cover type (PAS, n=8) consisted mainly of tall fescue and orchard grass and was typically managed for livestock grazing. I classified shrub cover types (SHB, n=9) as consisting mainly of grasses interspersed with low shrubs of autumn olive (*Elaeagnus umbellate*), multiflora rose (*Rosa multiflora*), and blackberry (*Rubus spp.*). Mixed cover types (MIX, n=4) consisted mainly of 15-20 year old stands of planted with

an even mix of white pine (*Pinus strobus*) and black locust (*Robinia pseudoacacia*). Pine cover types (PIN, n=4) were also 15-20 year old stands dominated by white pine. Forest reclamation (FRA, n=10) was the most recent cover type and was reclaimed in the previous 10 years and planted with a mix of native hardwoods including oaks, hickories, pine (*Pinus spp.*), and ash (*Fraxinus spp.*). Pre-mining law (PRE; n=20), where no active reclamation took place were characterized by a highwall of exposed rock, cut along the contour of a mountain where coal was extracted. These sites also had a shelf, along which heavy machinery drove to extract and transport coal, a relatively undisturbed forest above the highwall, and an outslope where topsoil and overburden were typically pushed over the shelf. The last cover type I considered in this study, reference forest (REF, n=13) are characteristic of mixed hardwood second growth forest that were harvested ~80 years ago.

I used a subset of 58 point counts from a previous study (Carrozzino 2009), and added points in additional cover types (see appendix F for information regarding location of points). Points were chosen by generating random locations within each cover type (patch) using ArcGIS, with the constraint that they must be  $\geq 150\text{m}$  from one another. A random subset of these points were then selected proportional to the area of each patch for sampling. Five minute, variable radius point counts were conducted during the breeding season (May-July) in 2007, 2008, 2010, and 2011 to sample the breeding bird community. Upon arrival at each point, observers counted all individuals seen or heard during a 5-minute period after a 1-minute “settling period.” Birds were recorded as either using the cover type being sampled, or tangential habitat users (e.g. flyovers and using adjacent cover types). All point counts were conducted during the early morning hours (0600-1000h) on days with no rain and minimal wind ( $<15\text{km/hr}$ ). All points were visited 5 times in 2007, 2008 and 2011; however, due to weather

constraints, we were only able to sample each point 4 times in 2010. We rotated observers and the order in which points were visited to help minimize sampling biases.

One challenge associated with community-based data sets is determining which species to include in the analysis. One observer in 2008 was less experienced than other observers and recorded several species not common to the region, and was likely caused by misidentification errors. Therefore, I excluded species from the analysis if they were not detected on Breeding Bird Survey (BBS) routes within the same region in the past 10 years (Sauer et al. 2011). I also excluded waterfowl from the analysis because these species were often detected on or near sediment ponds within various cover types, and did not have an equal probability of occurring within each cover type. Because we did not sample at night, owls were also excluded from analysis.

#### *Statistical analyses.*

I used the robust design model platform within program MARK to estimate community level parameters (Pollock 1982, White and Burnham 1999). Robust design models incorporate both open and closed population estimators, typically to estimate survival probabilities and population size (Kendall et al. 1997). While not a traditional use, population models have effectively been applied to community data in the past to estimate species richness and account for differences in species detection probabilities (Nichols et al. 1998a, Nichols et al. 1998b, Boulinier et al. 2001). Hines et al. (1999) developed a program that uses the jackknife estimator proposed by Burnham and Overton (1979), and assumes the heterogeneity model  $M(h)$  described by Otis et al. (1978) to calculate community dynamics for breeding bird survey data. An advantage of using the robust design in program MARK over the one developed by Hines et al. (1999) is that program MARK allows greater flexibility in building more meaningful models

applicable to our study system. The FRA cover type was not sampled in 2007 or 2008, so I did not use the robust design model to calculate community parameters in this reclamation type. Instead, I used program SPERICH, within program COMDYN (Hines et al. 1999), to calculate estimates of species richness, while accounting for heterogeneity in species detection probabilities.

I collapsed species counts at each point into a species detection history consisting of 0's and 1's, with 1's indicating detection and 0's indicating non-detection at each sampling interval (Appendix C). I then combined points within the same cover type to develop species detection histories for each of the 7 cover types sampled for the 4 years. Each row in the data represents a different species, and each column represents a visit within a year. Each point was visited 19 times throughout the duration of the study. The robust design model is based on primary sampling periods, each divided into secondary sampling occasions (Pollock 1982, Kendall et al. 1997). In this study, primary sampling periods correspond to years, and the community is assumed to be open to changes in composition between years. Secondary sampling occasions, on the other hand, represent visits to a point within years and the community is assumed to be closed to changes between secondary occasions. Closed population models can be used to estimate conditional capture probabilities, recapture probabilities, and species richness for community data (equivalent to population size in traditional robust design models). Open population models are used to estimate survival between primary sampling periods, and information from both primary and secondary sampling occasions are used to estimate temporary emigration (Kendall and Nichols 1995, Kendall et al. 1997, Bailey et al. 2004).

Lack of closure between secondary sampling occasions can bias results when using the robust design. Because our surveys were carried out during the peak of breeding season, it was

likely that most species had arrived on our study area before surveys began, and likely that our surveys ended before species began to migrate to their overwintering grounds (Johnson et al. 2009). These assumptions were checked using the closure test in program CAPTURE (Otis et al. 1978, Rexstad and Burnham 1991). The closure test assumes the population is closed, and the heterogeneity model is true (Rexstad and Burnham 1991). Of the 28 closed populations (7 cover types each with 4 primary sampling occasions), only 4 violated the closure assumption. Data for the samples that violated this assumption were pooled to meet the closure assumption (Kendall et al. 1997).

I initially used the robust design platform with full heterogeneity in program MARK (White and Burnham 1999) to analyze the data. These models estimate seven types of parameters about populations that are easily extended to communities (Nichols et al. 1998, Boulinier et al. 2001). These parameters include species richness ( $N$ ), local species persistence ( $\phi$ ), probability of emigration ( $\gamma''$ ), probability of immigration ( $\gamma'$ ), the proportion (mixture) of the community with a particular detection probability ( $\pi$ ), initial detection probabilities for each mixture ( $p$ ), and recapture probabilities for each mixture ( $c$ ). The full model including each of these parameters corresponds to the  $M_{(tbh)}$  model described by Otis et al. (1978), and often failed to converge for my data. Therefore, I opted to exclude behavior effects from subsequent models, making the full model correspond to  $M_{(th)}$  from Otis et al. (1978). This model allows capture probabilities to differ through time and among mixtures; however, it constrains recapture probabilities to be the same as capture probabilities. This constraint is reasonable for point count data because I did not expect birds to respond differently after being detected during a point count that would impact subsequent detections, but see (Riddle et al. 2010).

After using the robust design model to estimate the 7 model parameters, I used the equations of Nichols et al. (1998) to calculate community vital rates using parameter estimates for  $N$  and  $\phi$ , taken from the robust design models. The calculated vital rates include:

1) The probability of local extinction (E), calculated as:

$$E = 1 - \widehat{\phi}_{ij}$$

Where  $\widehat{\phi}_{ij}$  is the probability a species is present at some later time  $j$  in the study, given it was present during an earlier time,  $i$ .

2) The probability of species turnover (T), calculated as:

$$T = 1 - \widehat{\phi}_{ji}$$

Where  $\widehat{\phi}_{ji}$  is the probability that a species selected at random in time  $j$  was present during some earlier time,  $i$ . This probability is estimated by reverse-time modeling the encounter histories (Nichols et al. 1998, Johnson et al. 2009).

3) The number of colonizers ( $B_{ij}$ ), calculated as:

$$B_{ij} = N_j - (\phi_{ij} \times N_i)$$

Where the portion in parenthesis is the estimated number of individuals from time  $i$  in the community that survive to some later time  $j$ , subtracted from the number of individuals estimated to be in the community at time  $j$ . Another interpretation of this equation is the number of species present at time  $j$  that were not present at time  $i$ .

4) The finite rate of change in species richness ( $\lambda$ ), calculated as:

$$\lambda = \frac{N_j}{N_i}$$

Where  $N_j$  and  $N_i$  are the estimated species richness for times  $j$  and  $i$  respectively. Values above 1 indicate an overall increase in community richness; whereas, values less than 1 indicate a decline in species richness.

I considered a set of 17 candidate models for analysis, including the  $M_{(0)}$  and  $M_{(th)}$  model described by Otis et al. (1978). Other candidate models included various combinations of constant and time-varying parameters. I also considered a reduced, 2- time period model that allowed  $N$  to vary between 2007-2008 (first time period), and 2010-2011 (second time period; Table 18). I chose to model this reduced 2-time period model because several changes occurred between the 2008 and 2010 sampling seasons. For instance, increased mining around points, cessation of grazing in several pastures, gas-well developments and tornadoes in 2009 could have had drastic influences on the breeding bird community within various reclamation types. To select among competing models, I used Akaike's Information Criterion adjusted for small sample sizes ( $AIC_c$ ) and assumed the most parsimonious models had strong support if  $AIC_c$  was within 2 units of the lowest  $AIC_c$  value ( $\Delta AIC_c < 2$ ).

I calculated standard errors for derived parameters by using an empirical bootstrapping approach (Efron and Tibshirani 1986). While standard error estimates around these parameters can be achieved using the Delta method (Seber 1982, Powell 2007), those estimates are usually not as precise as other techniques (Nichols et al 1998b). Johnson et al. (2009) used a parametric bootstrapping approach to estimate variance around the derived parameters by assuming a normal distribution for  $N$ , and a gamma distribution for  $\phi$ . Because parametric bootstraps assume a sampling distribution, precision is usually lower around those estimates than they

would be if nonparametric bootstraps were used. Therefore, I used R (R Development Core Team 2011) package RMark (Lakke and Rexstad 2008) to sample from each capture history with replacement to create a new capture history. I sampled the original capture history 999 times and ran the previously determined best model based on minimum AIC<sub>c</sub> values 1000 times (1 for the original data and 999 for the pseudo capture histories). I then calculated the standard errors directly from the bootstrapped re-samples (see Appendix D for sample R code). Once I had standard error estimates for each of the derived parameters, I used program CONTRAST (Hines and Sauer 1989) to compare parameter estimates between years and among different cover types. Program CONTRAST does not adjust p-values to account for multiple comparisons; therefore, I used the Bonferonni adjustment,  $\alpha/m$ , to select the significance level for each multiple comparison, where alpha is the uncorrected significance level and m is the number of multiple comparison tests made (Sauer and Williams 1989).

I used non-metric multi-dimensional scaling (NMDS) techniques (Kruskal 1964) to visualize patterns of species richness associated with variation in vegetation characteristics. NMDS uses an iterative algorithm to minimize “stress.” Kruskal (1964) defined stress as the residual sum of squares from a monotonic regression line fit to the data. Therefore, the lower the stress, the less unexplained variation remains in the data. Typical values of stress for community data are usually between 10 and 20 (McCune and Grace 2002). NMDS is well suited to non-normal data and avoids the assumption of linearity among variables that other ordination methods require (McCune and Grace 2002). I used a Monte Carlo test to determine if axes selected by the ordination were meaningful. This test calculates the proportion of randomized runs with stress lower or equal to that observed in the original data matrix. Values less than the chosen type I error rate indicate the axis contributed to a significant reduction in stress. Lastly, I

used multi-response permutation procedure (MRPP; Mielke 2001) to test for significant changes in vegetation characteristics at points between years, and evaluate differences in vegetative characteristics among cover types. I used the axis values produced in NMDS as composite variables to test for significant vegetation changes between years and among cover types.

## **Results**

We detected 11,106 individual birds of 96 species throughout the 4 years of sampling. Within each cover type, we detected a total of 39 species in traditional shrub (SHB) during 2007, and 52 in 2011. We recorded 32 species using traditional pine (PIN) cover types in 2007, and 33 in 2011. In the traditional mixed (MIX) cover types, we detected 33 species in 2007 and 37 in 2011. For pasture (PAS) cover types, we recorded 26 species in 2007 and 33 in 2011. Forest reclamation (FRA) plots had the greatest species richness with 59 species detected in 2011. We detected 33 species in 2007 and 38 in 2011 within pre-SMCRA (PRE) cover types, and 40 species in 2007 and 45 in 2011 within reference forest (REF) plots (Figure 12).

In addition, 7 species were only detected in a single cover type (Appendix C). Veery (*Catharus fuscescens*) was only detected in the forest reclamation cover type. Pasture cover types had 2 unique species detected, eastern kingbird (*Tyrannus tyrannus*) and warbling vireo (*Vireo gilvus*). Brown creeper (*Certhia Americana*) and broad-winged hawk (*Buteo platypterus*) were unique to Pre-SMCRA cover types, and American redstart (*Setophaga ruticilla*) and magnolia warbler (*Dendroica magnolia*) were unique to forest plots.

### *Estimated Species Richness*

There was a fair amount of model uncertainty ( $\Delta AIC_c \leq 2$ ) within each cover type except for traditional shrub (Table 19). Each model set had at least one model where N was constrained to be fully time dependent, or a reduced 2 time period design, ranked among

competing models. Because the model weights ( $w_i$ ) were similar among the competing models, I report parameter estimates for fully time dependent or reduced 2 period models to compare community vital rates among cover types. Parameter estimates and standard errors varied among cover types and years (Table 20). Estimated species richness was generally higher in 2011 than 2007; however, it was higher in 2007 for traditional pine ( $X_1^2 = 5.40$ ;  $P=0.02$ ). The traditional shrub cover type had the highest estimates of species richness in 2007 and 2011; however, estimated species richness was not significantly higher than the forest reclamation cover type in 2011.

#### *Community vital rates*

Apparent species persistence was high ( $>0.8$ ) in all cover types, and the probability of local extinction was fairly low ( $<0.19$ ). Contrary to my predictions, local extinction was highest in reference forest plots, followed by traditional pine (Table 20). Traditional shrub and reference forest plots had the greatest number of local colonizing species; however, I did not detect significant differences in this parameter among the various cover types as a result of the large standard errors associated with those parameters (Table 20). As predicted, the finite rate of change in species richness was relatively low in the pasture, and highest in the traditional shrub cover types. Surprisingly, pre-SMCRA and reference forest plots had fairly high finite rates of change as well. Traditional pine was the only cover type that did not have a finite rate of change in species richness greater than 1, indicating a loss of about 21 percent of the original community that was present in 2007 (Table 20). Lastly, species turnover was highest in reference forest and pasture cover types, and lowest in traditional shrub. This parameter can be interpreted as the probability that an individual species selected at random in 2011 was not present in 2007, and indicates that the majority of species present within shrub cover types during 2007 were also

present during 2011. In contrast, reference forest and pasture cover types experienced higher rates of local extinction, but a large number of colonizing species were able to replace the ones that were lost; therefore, species richness and local turnover increased.

#### *Vegetation attributes among cover types*

We selected a 2-dimensional solution from a NMDS ordination to describe bird species' associations with vegetative attributes. Stress for this final solution was 18.51, and fell between the typical values of 15 and 20 reported for most ecological studies (Clarke 1993, McCune and Grace 2002). The first axis of the ordination explained 65.3% of the variation in the data, while the second axis explained 16.7% (cumulative  $r^2 = 82.0\%$ ). Monte Carlo tests confirmed that both axes reduced stress significantly more than expected by chance ( $P=0.004$ ). Axis 1 of the NMDS described a gradient ranging from pasture/ open grassland for negative values of the gradient to mature forest /closed canopy on the positive end of the gradient (Figure 13). Grass cover (grasscov,  $r = -0.766$ ) was the only vegetative attribute that had significant negative correlation with axis 1 (Table 21). Specific habitat attributes with the highest positive correlations with axis 1 included percent canopy cover (cancov,  $r = 0.685$ ), percent leaf cover (leafcov,  $r = 0.614$ ), and density of mature trees > 23cm DBH (denmat,  $r = 0.601$ ).

Axis 2 described significantly less variation in the data and was characterized by a gradient ranging from a young forest with an open understory and more small conifers at the lower end of the gradient, to an older forest with mature trees and denser, more uniform understory at the upper end of the gradient. Specific habitat attributes with the highest positive correlations with axis 2 were leaf cover (leafcov,  $r = 0.500$ ), percent canopy cover (cancov,  $r = 0.369$ ), density of mature trees >23cm DBH (denmat,  $r = 0.343$ ), and density of stems >0.5m tall and <3cm DBH (stemden,  $r = 0.291$ ). Specific habitat attributes with the highest negative

correlations with axis 2 were percent lgrass cover (grasscov,  $r = -0.254$ ), and coefficient of variation of stems  $>0.5\text{m}$  tall and  $<3\text{cm}$  DBH (cvstem,  $r = -0.276$ ) (Table 21).

#### *Bird associations and vegetation attributes*

Bird species with the largest negative correlations with axis 1 included field sparrows ( $r = -0.775$ ), indigo buntings ( $r = -0.591$ ), yellow-breasted chat ( $r = -0.537$ ), eastern towhee ( $r = -0.513$ ), and common yellowthroat ( $r = -0.450$ ). Red-eyed vireos ( $r = 0.611$ ), hooded warblers ( $r = 0.571$ ), ovenbirds ( $r = 0.425$ ), and blue-headed vireo ( $r = 0.401$ ) all had the highest positive correlations with the first axis (Table 21). Species with the largest negative correlations with axis 2 included chipping sparrows ( $r = -0.333$ ), and mourning dove ( $r = -0.308$ ). Red-eyed vireo ( $r = 0.591$ ), American crow ( $r = 0.556$ ), scarlet tanager ( $r = 0.540$ ), and ovenbird ( $r = 0.525$ ) all had the highest positive correlations with axis 2 (Figure 14, Table 21).

#### *Vegetation changes*

I found no evidence that vegetation changed significantly from 2007 to 2011 for pasture, traditional pine, and traditional mixed cover types using the composite variables (axis values) from NMDS ordination. However, I did find significant changes in vegetation attributes for traditional shrub, Pre-SMCRA, and reference forest during that time period (Table 22). When comparing cover types within years, I found that reference forest cover types were significantly different from all other cover types included in this study in 2007, and that Pre-SMCRA differed from traditional shrub, and pasture, but not traditional mixed or traditional pine cover types. After adjusting for the overall significance level, I did not find significant differences in vegetation characteristics between traditional shrub, traditional pine, pasture, or traditional mixed cover types in 2007 (Table 23). In 2011, however, I found that reference forest was different from traditional shrub, pasture, and Pre-SMCRA cover types, but not from traditional pine or

traditional mixed, suggesting that traditional pine and traditional mixed cover types more closely resembled mature forest plots in 2011. Pasture was significantly different from traditional pine, Pre-SMCRA, and reference forest plots, and marginally different from the traditional mixed cover type, suggesting at least some change in habitat attributes for traditional pine and traditional mixed cover types between 2007 and 2011. Traditional shrub was only significantly different from reference forest plots in 2011 (Table 24).

## **Discussion**

### *Avian community dynamics*

Understanding how avian communities respond to natural and man-made disturbances is important for long-term regional and local conservation efforts (Etterson et al. 2007). Reclaimed surface mined lands provide great opportunities to study the response of animal communities to disturbance. In a long term study, Lacki et al. (2004) observed higher species richness on reclaimed mine lands in Indiana than in surrounding habitats. They also report that the value of reclaimed mine lands for avian species is likely influenced by practices, such as wetland construction, implemented during the reclamation process. One way to examine the success of such reclamation practices is through evaluating community vital rates in relation to measured habitat characteristics.

Our estimates of species turnover were slightly higher for every cover type except shrub and pine, than those reported by Karanth et al. (2006) in an analysis of breeding bird survey data across five ecoregions. However, when I averaged the estimates of turnover across cover types, I found average species turnover to be very similar to their estimates [0.082 this study; 0.081 Karanth et al. (2006)]. Local species extinction was much lower in each cover type except for pasture, reference forest, and pine, in which case extinction was much higher than reported by

Karanth et al. (2006). When averaged across all cover types, my estimate of local species extinction probability was at the high end, but fell within the range of values reported by Karanth et al. (2006) [0.072 this study; 0.048-0.078 Karanth et al. (2006)]. These results indicate the importance of processes acting at local scales within a habitat type (e.g. succession, predation, competition and microclimate) in structuring local communities, which in turn determine regional community dynamics. Similarly, McFaden and Capen (2002) found microhabitat influences to be most important for structuring breeding bird communities in New England.

Contrary to my initial predictions, shrub cover types had the lowest rates of species turnover; whereas, reference forest plots showed the highest species turnover. I also expected relatively little species turnover and lower rates of extinction in pasture cover types because they were actively managed and grazed, so I expected very little succession to occur on those plots. One explanation for the pattern observed in pasture cover types was that 3 pastures grazed in 2007 and 2008 were not grazed after 2008. This could have allowed encroachment of woody vegetation, making pastures more suitable for some species; while making them less suitable for others (Rotenberry and Weins 1980). A possible explanation for higher rates of species turnover, local colonization and extinction in reference forest is consistent with forest gap dynamics (Urban and Smith 1989, Holmes and Sherry 2001). Results from our non-metric multidimensional scaling ordination support this hypothesis. The largest change for all cover types was observed along the second axis that describes a gradient from a more open forest understory at lower axis values to a more dense forest with a denser understory at positive values. Figure 13 depicts a shift from positive axis values to negative axis values for the reference forest, indicating a shift to a more open forest with a more patchily distributed understory. In May and April of 2009 and 2011 respectively, tornadoes struck near our study

area and could have generated strong enough winds to create large canopy gaps. Another possible explanation is that many of the remnant forest patches that we surveyed were small (approximately 7ha.) and we could have been detecting species using habitats outside the cover type we were sampling. This would result in identification of a greater number of local colonizing species. Examination of the detection, non-detection data for species in different reclamation types (Appendix C) makes this explanation likely, since we detected a fair number of early-successional species that were likely using habitat at forest edges. We tried to minimize this impact by sampling towards the center of a patch, but this was not always feasible given the nature of the landscape (small remnant patches, dangerous terrain, and restricted areas).

Several studies have documented seasonal variation in habitat use by forest dwelling species during the post-fledging period (Bulluck and Buehler 2006, Vitz and Rodewald 2006, McDermott and Wood 2010). This could also explain higher estimates of local species extinction probabilities if rare or less abundant species were using other habitats during times when we sampled forests, resulting in non-detection. This explanation is unlikely due to the fact that we were sampling the avian community during the peak breeding season and likely recorded most individuals at least once before the post-fledging period. Another alternative explanation for the observed pattern is that we were unable to distinguish between “incidental” species, and species that periodically use the habitat we were sampling (Knopf 2010). This would be likely if we observed individuals moving through a matrix of less desirable habitat on their way to a more preferred habitat (i.e. interpatch movement, Kennedy et al. 2011). Lastly, continued mining around patches of reference forest could cause some area-sensitive species to go locally extinct and attract more edge dwelling species that would contribute to an increase in local colonizers.

### *Community-habitat relationships*

I did not detect a significant difference in habitat characteristics based on NMDS axes between 2007 and 2011. However, the lack of detecting a statistically significant difference in vegetation attributes between years does not necessarily mean that there were no biologically relevant changes in habitat between 2007 and 2011. We detected significant changes in expected species richness between 2007 and 2011 after accounting for species detection probabilities, and these changes were likely influenced by both microhabitat and landscape level changes. It is likely that changes in species biotic and abiotic habitats interact with one another to influence community structure and dynamics. Holmes (2011) for example described 3 primary factors (food/weather interactions, nest predation, and adult densities) that determine population abundance and affect community structure in New England. In chapter 2, I documented that predation was the largest cause of nest failure, and that certain vegetation structural components may lower predation risk for some species. Therefore, understanding the complex interactions of habitat change in relation to predators, climate, and resources (e.g. ample nest sites, and food) is important for understanding how succession impacts avian communities, especially in managed landscapes.

Other studies have documented similar trends in community composition and dynamics along habitat gradients (Hansen et al. 1995, Rodewald and Yahner 2000, Sallabanks et al. 2006). Sallabanks (2006) found that the main factor influencing species richness and abundance was percent canopy cover. Similarly, Hansen et al. (1995) found that tree density within various tree size classes had the greatest impact on structuring avian communities in Oregon. In my study, both of these habitat characteristics were represented in the first NMDS ordination axis that explained almost 70% of the variation in avian relative abundance related to habitat

characteristics. I interpreted this axis to represent a gradient from open grassland to closed canopy forest. The species most strongly correlated with this axis were red-eyed vireo (*Vireo olivaceus*), hooded warbler (*Wilsonia citrine*) and ovenbird (*Seiurus aurocapilla*). Although each of these birds has different breeding requirements, they are still common forest species. For example, hooded warblers often dwell in a variety of forested habitats (Chiver et al. 2011); however, females typically nest in shrubby understory vegetation (Howlett and Stutchbury 1996). In contrast, ovenbirds are typically found in mature tracts of mixed forest and require lower amounts of ground cover with a well-developed leaf litter layer (Burke and Nol 1998).

At the other end of the gradient, I found field sparrows (*Spizella pusilla*), indigo buntings (*Passerina cyanea*) and yellow-breasted chats (*Icteria virens*) were most strongly negatively correlated with axis 1. This suggests that lack of canopy and well developed layer of grasses and forbs are general requirements for these species.

The second axis described a gradient from young, open forest to a dense, closed canopy forest, and represented the greatest change in habitat characteristics from 2007 to 2011. Red-eyed vireo, American crow (*Corvus brachyrhynchos*), and scarlet tanager (*Piranga olivacea*) all had strong positive correlations with this axis, indicating that areas with high densities of large trees were important for these species. Red-eyed vireos and scarlet tanagers usually require mature tracts of dense conifer, deciduous or mixed forest; however, both species may utilize early successional woodlots for breeding (Cimprich et al. 2000, Mowbray et al. 1999). Similarly, American crow had high positive values associated with axis 2 and was represented in the middle of axis 1, indicating that mature forests as well as open areas are important for this species. These results are supported by Andr en (1991) who found evidence that corvid density increased with increasing forest fragmentation and interspersed open grassland habitats

increased. This highlights the importance of evaluating community relationships at multiple spatial scales and the potential interaction between multiple spatial scales in structuring avian communities (Cornell and Donovan 2009). At the other end of the sparse-dense habitat gradient (axis 2), I found that chipping sparrows (*Spizella passerine*), and mourning dove (*Zenaida macroura*) had the highest negative associations with this axis. Both of these species are characteristic of open woodland/forest edge dwelling species (Middleton 1998, Otis et al. 2008).

These results show that 2 main gradients, canopy-grass and dense-sparse vegetation, can explain the majority of the variation in community composition on reclaimed mine lands. Future research should focus on the underlying mechanisms (predation, Martin 1993; food, Burke and Nol 1998; and interspecific interactions, Holmes 2012) responsible for structuring avian communities, and their interactions with vegetation structure at multiple spatial scales. Based on results from our study, we recommend that future reclamation efforts continue using a variety of restoration techniques that provide greater structural and age diversity of vegetation types to support the greatest number of species within mining areas. In addition, leaving remnant patches of mature forest >2ha could provide valuable habitat to mature forest species. We also suggest that isolated forest patches be connected to larger, contiguous forest patches or minimize the distance between small isolated woodlots and larger forested areas to facilitate juvenile dispersal between fragments (Haas 1995). Lastly, depending on the needs of the landowner, periodic disturbance (fire, select harvest, etc.) may be required to maintain early successional habitats after canopy closure.

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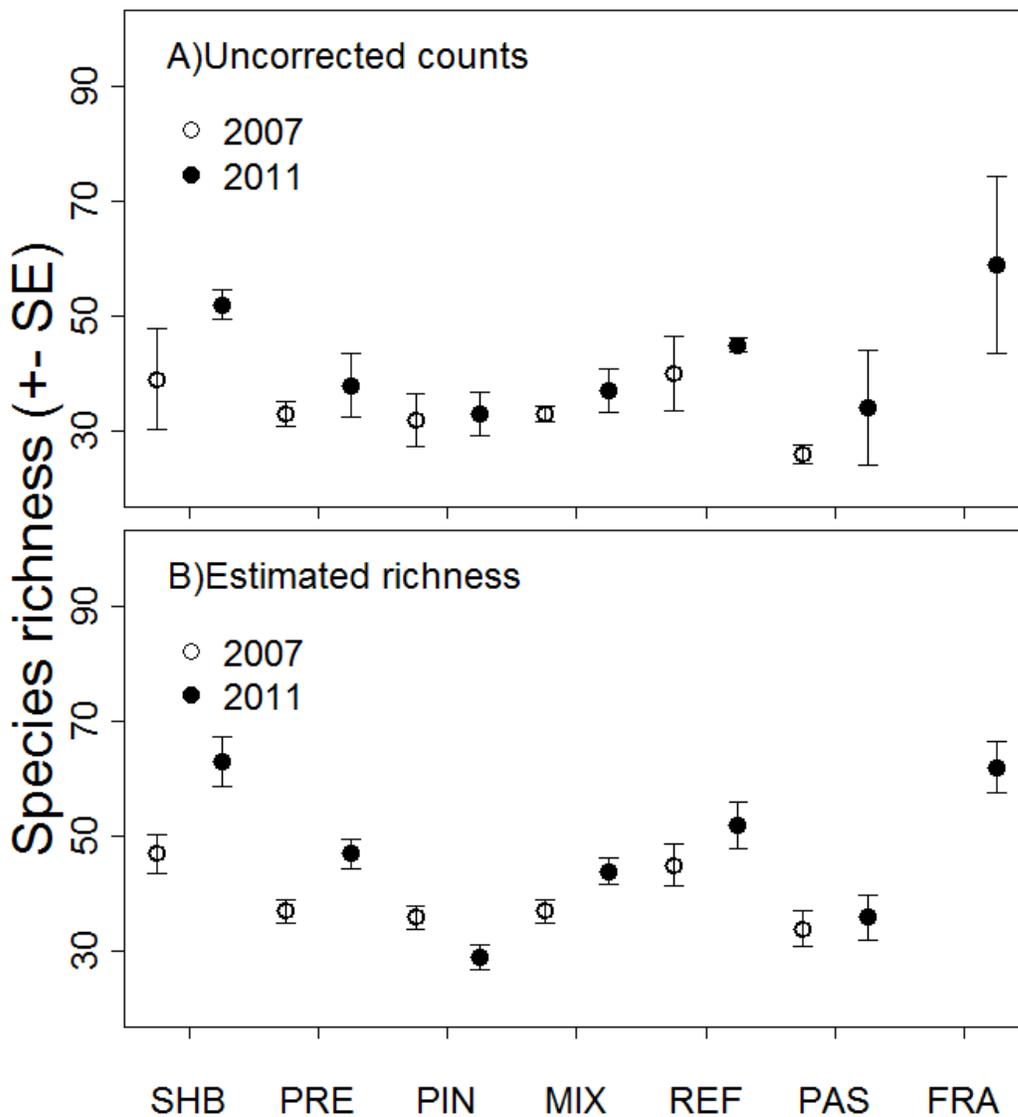
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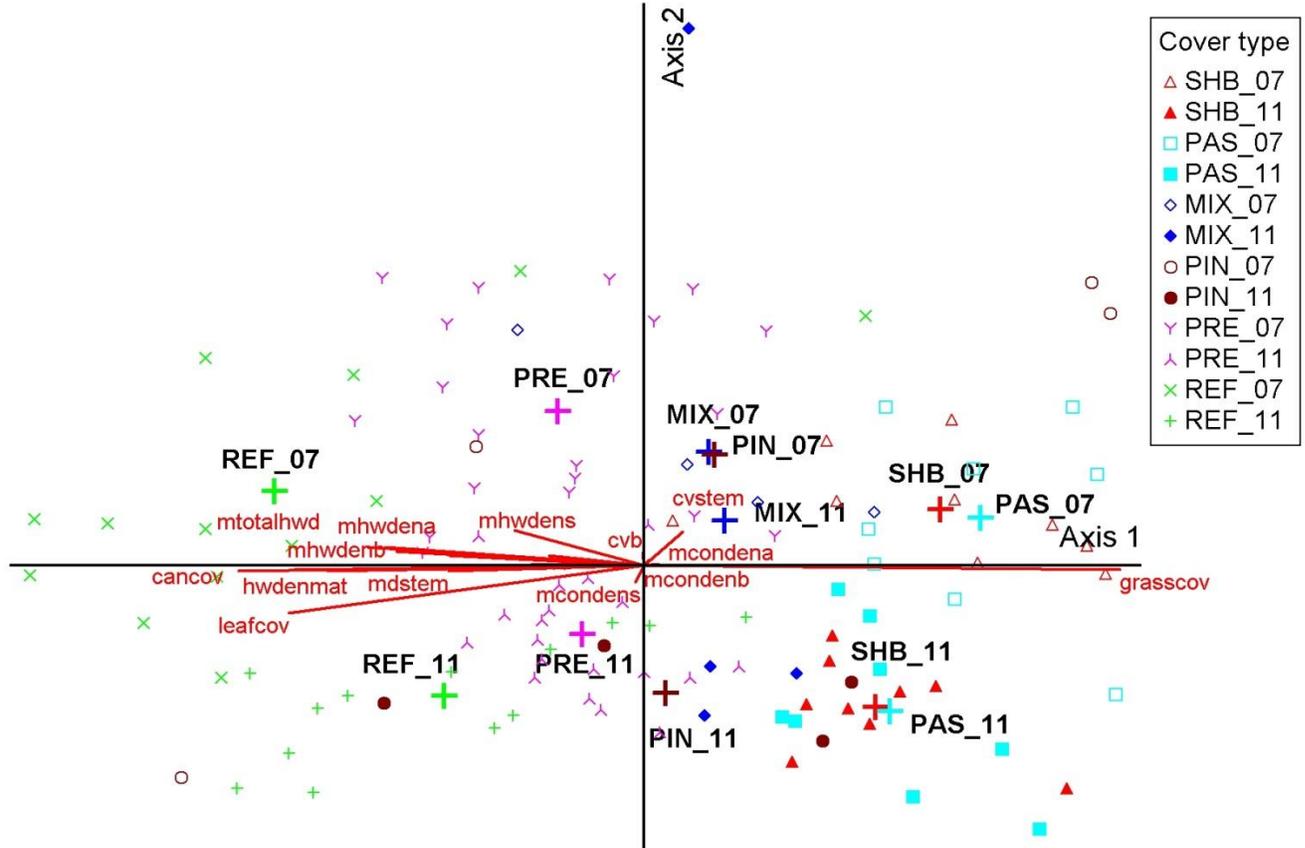
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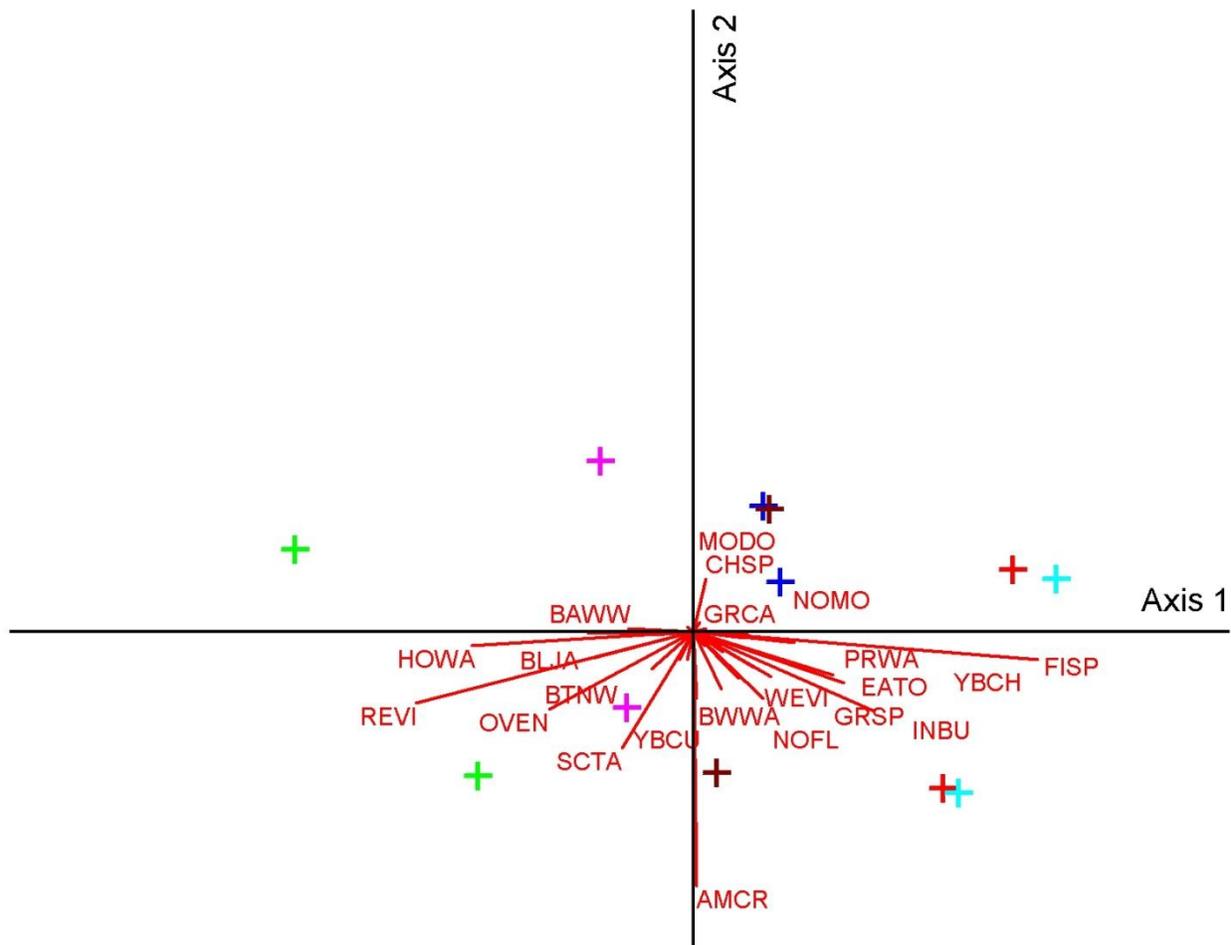
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**Figure 12.** Species richness, expressed as the average number of species per cover type based on raw, uncorrected counts (A), and estimated species richness, accounting for species detection probabilities (B). Open circles represent 2007 estimates, and closed circles represent 2011 data. Cover types are traditional shrub (SHB), Pre-SMCRA (PRE), traditional pine (PIN), traditional mixed (MIX), reference forest (REF), pasture (PAS), and forest reclamation (FRA).



**Figure 13.** Ordination of 78 sample units (survey points) in species space with an overlay of habitat attributes from 2007 and 2011. The length of the vector is proportional to the correlation coefficient of the specific habitat attribute. The direction of the vector corresponds to increasing values of the habitat attribute on the graph. Variable acronyms are grasscov- percent grass cover, cancov- percent canopy cover, leafcov- percent leaf cover, hwdenmat- density of mature hardwoods, mdstem- density of stems >1m tall and < 3cm DBH, mhwdenb- mean density of hardwood trees 15-23cm DBH, mhwdena- mean hardwood density of trees 8-15cm DBH, mhwdens- mean hardwood density of trees 3-8cm DBH, mtotalhwd- mean density of all hardwood trees, cvb- coefficient of variation of trees 15-23cm DBH, cvstem- coefficient of variation of stems >1m tall and <3cm DBH, mcondens- mean density of conifer stems, mcondena- mean density of conifer trees 3-8cm DBH, and mcondenb- mean density of conifer trees 8-15cm DBH.



**Figure 14.** Ordination of sample units in species space with overlay of bird species (only centroids of survey points are shown). Length of the vector is proportional to the species correlation coefficient; the direction of the vector corresponds to increasing species relative abundance. Species acronyms follow conventional AOU naming convention.

**Table 18.** Model parameterizations for the 17 models evaluated in this study to estimate species richness and calculate community vital rates. Specific parameters are output from the robust design model in program MARK, where S is species persistence, emigration ( $\gamma''$  and  $\gamma'$ ) is modeled as defined by Kendall et al. (1997),  $p_i$  is the mixture, p is the detection probability and N is species richness. A dot (.) indicates a parameter was constrained to be constant, t indicates time varying period specifies a reduced 2 time period model where period 1 corresponds to 2007 and 2008, and period 2 corresponds to 2010 and 2011.

Model Parameterization <sup>a</sup>						
Model	S	$\gamma''$	$\gamma'$	$p_i$	p	N
1	.	None	.	.	.	t
2	.	None	.	.	.	period
3	.	None	.	.	.	.
4	.	Random (.)	.	.	.	.
5	.	Random (t)	.	.	.	t
6	.	Markov (t)	.	.	.	t
7	t	Random (t)	t	t	t:mix	t
8	t	Markov (t)	t	t	t:mix	t
9	.	Random (.)	period	period	period:mix	period
10	.	Random (t)	.	.	mix	t
11	.	Markov (.)	.	.	.	t
12	.	Random (t)	.	.	mix	.
13	.	Markov (.)	.	.	mix	.
14	.	Random (t)	.	.	mix	.
15	.	Random (t)	.	.	mix	period
16	.	Random(t)	.	.	mix	period
17	.	Markov (t)	period	period	period:mix	t

<sup>a</sup> Model structure for the 6 parameters: . = constant; mix=mixture of detection probabilities for 2 groups within the community; period= parameters differ between 2007-2008 and 2010-2011 field seasons; t = time dependent variation among primary sampling periods; Random (t) = time dependent random temporary emigration with  $\gamma''(t) = \gamma'(t)$ ; Random(.) = constant random temporary emigration with  $\gamma''(.) = \gamma'(.)$ ; Markov (t) = full time dependent Markovian temporary emigration; Markov (.) = constant Markovian temporary emigration.

**Table 19.** Model selection results used to estimate species richness for each cover type based on minimum AICc criteria. Only models with >10% of the model weight ( $w_i$ ) are shown. K is the number of parameters in each model.

Model <sup>a</sup>	AICc	$\Delta$ AICc	$w_i$	Model Likelihood	k
<b>Traditional shrub</b>					
{S(.)Gamma"(time)Gamma'(time)pi(period)p(period:mixture)N(time)}	196.1046	0	0.87417	1	16
<b>Pre-SMCRA</b>					
{S(.)gamma"(random(.))gamma'(random(.))pi(.)p(mixture)N(.)}	246.9715	0	0.31885	1	6
{S(.)gamma"(random)gamma'(random)pi(.)p(mixture)N(.)}	247.4046	0.4331	0.25677	0.8053	8
{S(.)gamma"(random(.))gamma'(random(.))pi(.)p(mixture)N(period)}	248.543	1.5715	0.14533	0.4558	7
{S(.)gamma"(. )gamma'(. )pi(.)p(mixture)N(.)}	248.7172	1.7457	0.1332	0.4177	7
{S(.)gamma"(random)gamma'(random)pi(.)p(mixture)N(period)}	249.0161	2.0446	0.11471	0.3598	9
<b>Traditional pine</b>					
{S(.)gamma"(random)gamma'(random)pi(.)p(mixture)N(.)}	305.1831	0	0.56285	1	8
{S(.)gamma"(random)gamma'(random)pi(.)p(mixture)N(period)}	307.2319	2.0488	0.20207	0.359	9
<b>Traditional mixed</b>					
{S(.)gamma"(random(.))gamma'(random(.))pi(.)p(mixture)N(.)}	233.4058	0	0.4635	1	6
{S(.)gamma"(random(.))gamma'(random(.))pi(.)p(mixture)N(period)}	234.9504	1.5446	0.21411	0.4619	7
{S(.)gamma"(. )gamma'(. )pi(.)p(mixture)N(.)}	235.1952	1.7894	0.18945	0.4087	7
<b>Pasture</b>					
{S(.)gamma"(random(.))gamma'(random(.))pi(.)p(mixture)N(.)}	263.3716	0	0.3757	1	6
{S(.)gamma"(. )gamma'(. )pi(.)p(mixture)N(.)}	263.934	0.5624	0.28361	0.7549	7
{S(.)gamma"(random(.))gamma'(random(.))pi(.)p(mixture)N(period)}	264.6587	1.2871	0.1974	0.5254	7
<b>Reference</b>					
{S(.)gamma"(random)gamma'(random)pi(.)p(mixture)N(.)}	235.1923	0	0.51052	1	8
{S(time)gamma"(time)gamma'(time)pi(time)p(time:mixture)N(time)}	236.3854	1.1931	0.28115	0.5507	24
{S(.)gamma"(random)gamma'(random)pi(.)p(mixture)N(period)}	237.2728	2.0805	0.1804	0.3534	9

<sup>a</sup> Model structure for the 6 parameters: . = constant; mix=mixture of detection probabilities for 2 groups within the community; period= parameters differ between 2007-2008 and 2010-2011 field seasons; t = time dependent variation among primary sampling periods; Random (t) = time

dependent random temporary emigration with  $\gamma^m(t) = \gamma'(t)$ ; Random(.) = constant random temporary emigration with  $\gamma^m(.) = \gamma^1(.)$ ; Markov (t) = full time dependent Markovian temporary emigration; Markov (.) = constant Markovian temporary emigration.

**Table 20.** Estimated and derived parameters (SE) of community vital rates within each cover type, determined using the robust design model in program MARK. Values with the same lettering are not significant at the Bonferroni adjusted alpha level based on tests from program CONTRAST.  $N_{2007}$  is species richness in 2007,  $N_{2011}$  is estimated species richness in 2011,  $\phi_{ij}$  is estimated species persistence,  $\phi_{ji}$  is the probability that a species was present in 2011, conditional on it being present in 2007, E is the probability of a species becoming locally extinct, B is the derived number of colonizers,  $\lambda$  is the finite rate of increase in species richness and T is the probability that a species selected at random in 2011 was not present in 2007. See text for explanations of calculations for the derived parameters.

	Estimated parameters				Derived parameters			
	$N_{2007}$	$N_{2011}$	$\phi_{ij}$	$\phi_{ji}$	$E$	$B$	$\lambda$	$T$
Traditional shrub	47 <sup>A</sup> (3.3)	63 <sup>A</sup> (4.4)	0.992 <sup>A</sup> (0.029)	0.977 (0.133)	0.007 <sup>A</sup> (0.001)	16 <sup>A</sup> (40.6)	1.34 <sup>A</sup> (0.03)	0.023 <sup>A</sup> (0.001)
Pasture	34 <sup>B</sup> (3.0)	36 <sup>BC</sup> (3.9)	0.908 <sup>A</sup> (0.035)	0.879 (0.026)	0.092 <sup>C</sup> (0.006)	5 <sup>A</sup> (33.6)	1.06 <sup>D</sup> (0.04)	0.120 <sup>B</sup> (0.003)
Reference	45 <sup>A</sup> (3.7)	52 <sup>AB</sup> (4.0)	0.805 <sup>A</sup> (0.064)	0.864 (0.060)	0.194 <sup>E</sup> (0.006)	15 <sup>A</sup> (24.6)	1.16 <sup>BD</sup> (0.03)	0.136 <sup>A</sup> (0.004)
Traditional mixed	37 <sup>A</sup> (1.9)	44 <sup>B</sup> (2.4)	0.995 <sup>A</sup> (0.017)	0.913 (0.025)	0.004 <sup>D</sup> (0.000)	7 <sup>A</sup> (13.9)	1.21 <sup>B</sup> (0.01)	0.087 <sup>B</sup> (0.004)
Traditional pine	36 <sup>A</sup> (2.0)	29 <sup>C</sup> (2.2)	0.897 <sup>A</sup> (0.029)	0.964 (0.037)	0.103 <sup>C</sup> (0.001)	-3 <sup>A</sup> (22.6)	0.79 <sup>C</sup> (0.02)	0.035 <sup>C</sup> (0.006)
Pre-SMCRA	37 <sup>A</sup> (2.1)	47 <sup>B</sup> (2.5)	0.969 <sup>A</sup> (0.024)	0.907 (0.023)	0.030 <sup>B</sup> (0.001)	10 <sup>A</sup> (23.6)	1.26 <sup>AB</sup> (0.03)	0.092 <sup>C</sup> (0.001)
Forest Reclamation <sup>†</sup>	-	62 <sup>A</sup> (4.4)	-	-	-	-	-	-

<sup>†</sup>Sampling in this reclamation type was not conducted in 2007 or 2008, so estimated species richness for 2007 and derived parameters were not calculated. Species richness was estimated using program SPERICH written by J.E. Hines, based on the Jackknife estimators described by Burnham and Overton (1979).

**Table 21.** Pearson correlations of bird species and habitat variables with the first 2 axes derived from NMDS analysis of 78 samples from areas sampled on reclaimed mine lands in southwest Virginia between 2007 and 2011. The first axis accounted for 65.3% of the variability in the data, and the second axis 16.7%. Only correlations significant at the 0.05 alpha level are presented.

Positive				Negative			
Bird species	r	Habitat variables	r	Bird species	r	Habitat variables	r
Axis 1							
REVI	0.611	cancov	0.685	FISP	-0.775	grasscov	-0.766
HOWA	0.571	hwdenmat	0.636	INBU	-0.591		
OVEN	0.425	leafcov	0.614	YBCH	-0.537		
BHVI	0.401	mdenmat	0.601	EATO	-0.513		
BLJA	0.368	mhwdena	0.599	COYE	-0.45		
BAWW	0.333	mtotalhw	0.582	PRWA	-0.422		
SCTA	0.27	mhwdenb	0.566	GRSP	-0.396		
WOTH	0.237	mtotalde	0.501	WEVI	-0.384		
BTNW	0.221	mdstem	0.47	BASW	-0.349		
		meana	0.427	EUST	-0.314		
		meanb	0.375	BRTH	-0.311		
Axis 2							
REVI	0.591	leafcov	0.5	CHSP	-0.333	cvstem	-0.276
AMCR	0.556	cancov	0.369	MODO	-0.308	grasscov	-0.254
SCTA	0.54	mdenmat	0.343				
OVEN	0.525	hwdenmat	0.337				
HOWA	0.389	mdstem	0.291				
BLJA	0.359						
BTNW	0.333						
YBCU	0.332						
HAWO	0.313						
BBCU	0.288						

**Table 22.** Results from multi-response permutation procedure testing differences in vegetation characteristics for 6 cover types between 2007 and 2011. Values tested were the composite values for each site on axes 1 and 2 from the NMDS ordination.

	Average within group distance			
	2007	2011	T	P
Traditional shrub	1.215	0.965	-3.120	0.009
Pasture	1.240	1.172	-0.019	0.397
Traditional mixed	1.443	1.752	-0.094	0.412
Traditional Pine	1.911	1.455	-0.006	0.419
Pre-SMCRA	1.034	1.007	-7.722	0.000
Reference	1.155	1.022	-3.799	0.042

**Table 23.** Results from multi-response permutation procedure testing the difference in vegetation characteristics between cover types within 2007. Bold lettering indicates significance after applying Bonferonni adjustments to the overall significance level.

		2007									
		Traditional shrub		Pasture		Traditional mixed		Traditional Pine		Pre-SMCRA	
		T	P	T	P	T	P	T	P	T	P
Traditional shrub	Pasture	-2.6503	0.0161								
Traditional mixed	Traditional shrub	-1.3372	0.1003	-1.6334	0.0702						
Traditional Pine	Traditional mixed	-3.2376	0.0121	0.0911	0.0221	1.0404	0.8713				
Pre-SMCRA	Traditional Pine	-10.3015	<b>0.0000</b>	-6.7911	<b>0.0001</b>	-3.5013	0.0082	-3.9111	0.0061		
Reference	Pre-SMCRA	-12.3466	<b>0.0000</b>	-12.5006	<b>0.0000</b>	-8.6856	<b>0.0000</b>	-6.9567	<b>0.0001</b>	-16.9048	<b>0.0000</b>

**Table 24.** Results from multi-response permutation procedure testing the difference in vegetation characteristics between cover types within 2011. Bold lettering indicates significance after applying Bonferonni adjustments to the overall significance level.

		2011									
		Traditional shrub		Pasture		Traditional mixed		Traditional Pine		Pre-SMCRA	
		T	P	T	P	T	P	T	P	T	P
Traditional shrub	Pasture	-1.7256	0.0630								
Traditional shrub	Traditional mixed	-2.7144	0.0126	-3.5633	0.0048						
Traditional shrub	Traditional Pine	-3.5208	0.0084	-4.1468	<b>0.0029</b>	0.8368	0.7928				
Traditional shrub	Pre-SMCRA	-2.6693	0.0220	-5.6069	<b>0.0007</b>	-2.2025	0.0376	-3.7286	0.0073		
Traditional shrub	Reference	-11.3136	<b>0.0000</b>	-11.4771	<b>0.0000</b>	-3.5911	0.0064	-3.8115	0.0051	-12.5202	<b>0.0000</b>

## Conclusions

### *Avian nest-site selection and nest-success*

Avian nest site selection is a hierarchical process that likely occurs at multiple spatial scales (Knopf and Sedgwick 1992, Pribil and Picman 1997, Latif et al. 2011). Because predation is a major cause of nest failure (Martin 1993), birds should select nest sites that minimize predation risk (Latif et al. 2011). In this study, I examined nest site selection at 2 spatial scales, the nest-site (within 1 m of a nest) and the nest patch (within 11.3 m around a nest). In addition, I separated nests into early and late-season nests to determine potential differences in nest-site selection. Grant et al. (2005) found that daily survival rates varied throughout the breeding season for 2 grassland sparrows in North Dakota, likely as a result of behavioral or abundance shifts in local predator communities. Therefore, it is expected that birds will respond to increased levels of predation by altering nest placement to lower risk of predation (Latif et al. 2011).

I found evidence that field sparrows (*Spizella pusilla*) were selecting for certain attributes within the nest-patch scale, and that field sparrows nesting later in the season became more selective in nest placement. This could result from increased predation pressure later in the season (Grant et al. 2005) or specific features that birds use to cue in on did not become available until later in the nesting season (Burhans and Thompson 1998). Additionally, I found that daily nest survival decreased with increasing % woody vegetation <1 m tall, suggesting that nest site selection at the nest patch scale may be mal-adaptive. Because daily nest survival rates (DSR) increased with % concealment around the nest, my results support the hypothesis for a tradeoff between having a concealed nest while maintaining a view of the surrounding area (Gotmark et al. 2005). Pribil and Picman (1997) found similar results for red-winged blackbirds (*Agelaius phoeniceus*) in marshes in Canada.

I also found evidence that indigo buntings (*Passerina cyanea*) were selecting for certain vegetation attributes at the nest-patch scale. Despite this apparent selectivity in nest placement by indigo buntings, I did not find any evidence that habitat attributes positively or negatively influenced nest success. Instead, I found that the effect of year explained the greatest amount of variation in nest success for this species, with 2010 having much higher DSR estimates than 2011. Thompson (2007) suggested that year effects on nest predation may result from changes in predator density and behavior, differences in habitat conditions, or changes in prey densities. 2011 had substantially more rainfall during the breeding season (April-August) than 2010 and could potentially increase predator activity (Moynahan et al. 2007). Alternatively, exposure of nests and nestlings could reduce nest success or survival (Horvick et al. 2011).

My estimates of daily nest survival DNS were higher for field sparrows than reported elsewhere in the literature. Likewise, estimates of DNS for indigo buntings were higher in 2010 of this study than demonstrated by comparable studies; however, these estimates were lower in 2011 for indigo buntings. Population growth estimates, using estimates of seasonal fecundity calculated from empirical data, and assuming a range of adult and juvenile survival estimates indicated that populations were sustainable as long as annual adult survival was  $\geq 0.8$ .

Based on these results, we recommend planting a diverse mix of native trees and shrubs. We recommend planting shrubs that have excellent concealment structure such as hawthorn (*Crataegus spp.*) and alternateleaf dogwood (*Cornus alternifolia*), with relatively wide spacing between plantings. In addition, we recommend the widest tree spacing possible to be compliant with SMCRA regulations to prolong the amount of time an area remains in an early successional state. Lastly, dependent upon the objectives of the landowner, periodic disturbances may be

needed to provide a stable source of early successional habitat for species that depend on this ephemeral type.

Results from this study indicate the need to quantify and examine potential influences that affect adult survival on reclaimed mine lands. In addition, studies that examine the underlying mechanisms influencing nest success and that examine alternative factors such as food availability that might impact nest survival could be useful. For example, Dunn et al. (2010) found strong evidence for non-consumptive interactions between predator activity and prey provisioning rates of adult nesting birds.

#### *Community dynamics and composition*

Species richness has been used in the past to identify areas of conservation priority with the assumption that areas with greater species richness conserve overall biodiversity and ecological processes (Fleishman et al. 2006). However the value of using species richness to identify conservation priorities has been questioned in recent years (Boulinier et al. 1998, Johnson et al. 2009, Zipkin et al. 2009, Knopf 2010). Variation in species detection probabilities can cause biased estimates of species richness (Nichols et al. 1998), so methods that account for differences in detection are needed to make useful inferences about areas of conservation priority. In chapter 3, I used a mark-recapture analysis (Pollock 1982) to account for variation in detection probabilities among species to provide unbiased estimates of species richness within different cover types on reclaimed surface mines in southwest Virginia over a 5 year period.

I found that estimated species richness was generally higher in the last year of the study than the first. In addition, I found that estimated species richness was lower in the pine cover type in the final year of the study, whereas, species richness based on raw counts showed an increase in species richness from the first year of the study to the last. This highlights the

importance of accounting for species detection probabilities for community analyses. Species richness based on raw counts would have led us to believe that there was an increase in species richness through time; whereas estimates of species richness indicate a loss of species in the pine cover type.

In addition to estimating species richness, I calculated community vital rates (Nichols et al. 1998) that are also useful in comparing the value of different areas for conservation (Knopf 2010). I found that species persistence in various cover types was very high; whereas the probability of local species extinction was relatively low. Reference forest and pine cover types had the highest probabilities of local extinction and the lowest probabilities of species persistence. This observed pattern is likely a result of canopy closure in the pine cover type but these results are less clear in forest plots. Additionally, species turnover was highest in pasture and reference forest plots and may be a result of changes in microhabitat.

We recommend that mine operators use a combination of reclamation techniques to create a landscape of various cover types. In addition, leaving patches of remnant forest >2ha could provide valuable habitat for forest species and may provide “stepping stones” among forest patches in more fragmented landscapes. Additional studies that identify the mechanism of variation observed in the community structure and composition on our study could prove useful. Future studies may also look at community structure and dynamics of other vertebrate and invertebrate species on reclaimed mine lands.

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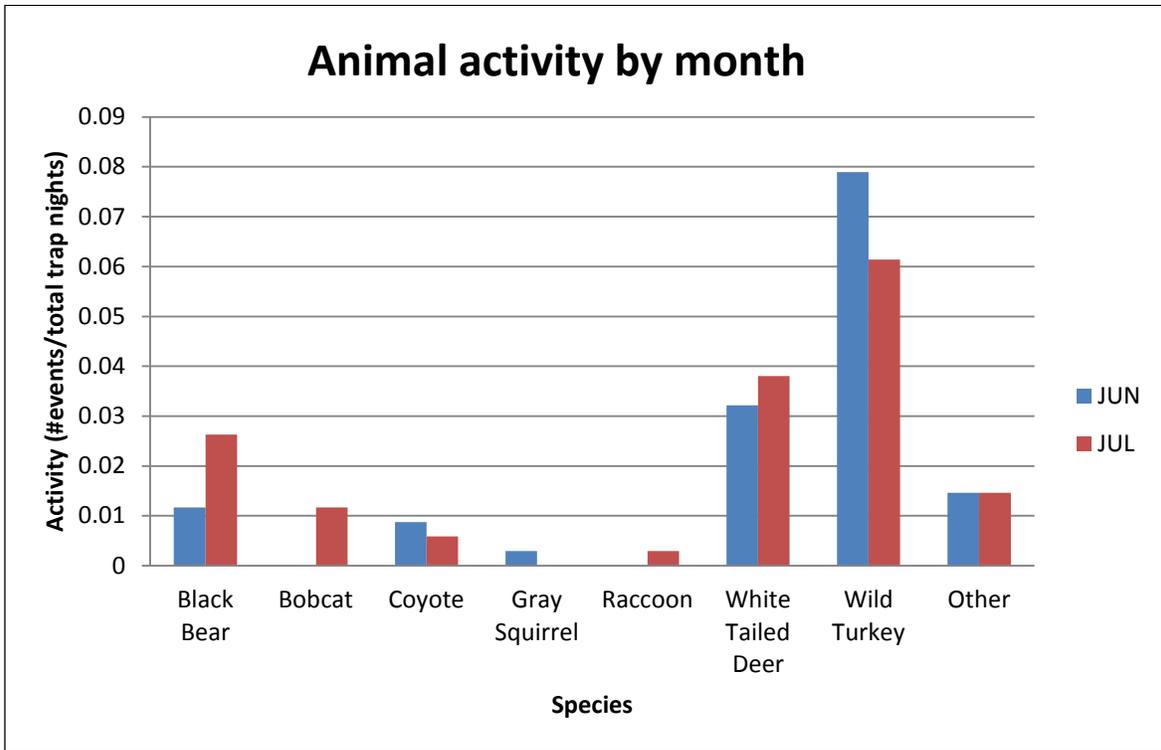
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## Appendices

## **Appendix A: Supplemental Camera-Trap data.**

Remote sensing camera-traps were used to inventory mid- to large-sized mammals in different cover types at the Powell River site in 2010. Cameras were spaced approximately 1km apart in areas of suspected animal activity (based on scat, trails, and tracks). A total of 9 camera-traps (Wildgame innovations, model ir2) were deployed from June 5 through July 13, totaling 38 trap-nights per camera for a total of 342 trap-nights for the entire study area. Camera-traps were taken down early as a result of theft on the Powell River Project. Animal activity was calculated as the number of independent capture events for a single species divided by the total number of trap nights, and relative animal activity was calculated as the number of independent capture events for a single cover type divided by total capture across all cover types (Figure A1, Table A1). Consecutive photos of the same species were considered independent events if individual animals can be unambiguously identified, or if the interval between capture events was >30 minutes.



**Figure A1.** Animal activity by month expressed as the # of independent capture events/ trap nights per camera station on the Powell River Project in southwest Virginia during June and July 2010.

**Table A1.** Relative animal activity by cover type on reclaimed mine lands in southwest Virginia for June and July of 2010. Numbers expressed as number of animals/ trap night.

<b>Relative Animal Activity</b>								
	Black Bear	Bobcat	Coyote	Gray Squirrel	Raccoon	White Tailed Deer	Wild Turkey	Other
FRA	0.20	0.00	0.20	0.00	1.00	0.13	0.40	0.50
Early	0.40	0.50	0.60	1.00	0.00	0.71	0.35	0.10
Mid	0.13	0.00	0.00	0.00	0.00	0.08	0.00	0.20
Ref	0.13	0.25	0.20	0.00	0.00	0.08	0.17	0.00
PreSMCRA	0.13	0.25	0.00	0.00	0.00	0.00	0.08	0.20

\*Trap nights = 342

**Appendix B. Average nest attributes for 18 species of birds with insufficient sample sizes to model nest success.**

**Table B1.** Average nest habitat attributes for nests of 18 species found on reclaimed surface-mines in southwest Virginia in 2010 and 2011. Values are means  $\pm$  SE. Species acronyms follow AOU naming convention.

	BAWW	$\pm$ SE	BGGN	$\pm$ SE	BLJA	$\pm$ SE	BRTH	$\pm$ SE
Nest height (m)	0	0	2.7	.	2.83	1.18	1.82	0.7
Nest DBH (cm)	.	.	3.5	.	14.1	2.7	15.4	.
Distance to substrate center (m)	.	.	0.9	.	0	0	1.2	0.5
% Canopy cover	95	0	40	.	98	2.5	60	21.8
% Grass cover	38	22.5	80	.	0	0	43	18.8
% Leaf cover	80	10	40	.	100	0	83	8.3
% Bare ground	23	17.5	30	.	78	17.5	22	14.8
% Coarse woody debris	0	0	0	.	0	0	2	1.7
% Forb cover	55	5	95	.	38	17.5	73	9.3
% woody vegetation	58	37.5	50	.	28	27.5	38	11.7
% Moss cover	13	12.5	0	.	0	0	12	9.3
Conifer stem density (#/ha)	75	75.3	0	.	38	12.6	0	0
Deciduous stem density (#/ha)	1356	1054.4	1431	.	590	539.8	887	291.5
Total density of trees 4-10cm DBH	1080	75.3	854	.	766	163.2	728	414.8
Total density of trees 10-15cm DBH	276	25.1	50	.	741	414.2	151	43.5
Total density of trees >15 cm DBH	75	25.1	0	.	628	200.8	176	126.4
% understory cover 0-0.5m	79	4.7	98	.	43	6.9	56	15
% understory cover 0.5-1m	83	1.6	94	.	33	10.3	56	13.1
% understory cover 1-1.5m	84	3.8	76	.	29	3.4	60	10.5
% understory cover 1.5-2m	90	4.7	91	.	24	1.3	63	12.8
% understory cover 2-2.5m	88	3.1	94	.	33	3.4	58	15.2
CV distance to nearest tree >3cm DBH	28.25	11.752	132.97	.	120.12	17.152	50.1	21.91

**Table B1.** (continued)

	CEWA	±SE	CHSP	±SE	COYE	±SE	GRCA	±SE
Nest height (m)	6.5	.	1.7	.	0.17	.	2.05	0.15
Nest DBH (cm)	30	.	3.5	.	.	.	3.25	0.45
Distance to substrate center (m)	2.5	.	0	.	0	.	0.35	0.05
% Canopy cover	25	.	55	.	30	.	75	5
% Grass cover	80	.	80	.	75	.	48	13
% Leaf cover	40	.	45	.	45	.	35	15
% Bare ground	15	.	50	.	60	.	78	23
% Coarse woody debris	0	.	0	.	5	.	0	0
% Forb cover	95	.	65	.	100	.	43	3
% woody vegetation	35	.	40	.	40	.	48	18
% Moss cover	0	.	5	.	15	.	15	5
Conifer stem density (#/ha)	0	.	0	.	0	.	0	0
Deciduous stem density (#/ha)	502	.	326	.	728	.	1306	201
Total density of trees 4-10cm DBH	151	.	477	.	502	.	603	25
Total density of trees 10-15cm DBH	75	.	201	.	75	.	213	13
Total density of trees >15 cm DBH	50	.	25	.	0	.	138	113
% understory cover 0-0.5m	98	.	98	.	94	.	75	22
% understory cover 0.5-1m	98	.	71	.	47	.	69	9
% understory cover 1-1.5m	74	.	84	.	26	.	74	14
% understory cover 1.5-2m	53	.	81	.	26	.	83	9
% understory cover 2-2.5m	50	.	84	.	26	.	84	14
CV distance to nearest tree >3cm DBH	57.28	.	44.61	.	.	.	49.25	0.746

**Table B1.** (continued)

	GRSP	±SE	GWWA	±SE	HOWA	±SE	NOCA	±SE
Nest height (m)	0	.	0.12	.	1.5	.	1.2	.
Nest DBH (cm)	.	.	.	.	19.5	.	.	.
Distance to substrate center (m)	.	.	0	.	1.2	.	0.87	.
% Canopy cover	0	.	70	.	90	.	20	.
% Grass cover	90	.	100	.	20	.	95	.
% Leaf cover	0	.	40	.	95	.	15	.
% Bare ground	30	.	65	.	65	.	50	.
% Coarse woody debris	0	.	0	.	5	.	0	.
% Forb cover	70	.	80	.	60	.	80	.
% woody vegetation	0	.	85	.	90	.	35	.
% Moss cover	5	.	5	.	5	.	5	.
Conifer stem density (#/ha)	0	.	0	.	0	.	0	.
Deciduous stem density (#/ha)	75	.	628	.	527	.	577	.
Total density of trees 4-10cm DBH	0	.	628	.	929	.	603	.
Total density of trees 10-15cm DBH	0	.	151	.	50	.	0	.
Total density of trees >15 cm DBH	0	.	25	.	301	.	0	.
% understory cover 0-0.5m	29	.	94	.	65	.	98	.
% understory cover 0.5-1m	3	.	70	.	46	.	88	.
% understory cover 1-1.5m	3	.	84	.	48	.	74	.
% understory cover 1.5-2m	3	.	98	.	57	.	74	.
% understory cover 2-2.5m	3	.	88	.	68	.	77	.
CV distance to nearest tree >3cm DBH	.	.	49.07	.	70.71	.	33	.

**Table B1.** (continued)

	PRWA	±SE	REVI	±SE	RWBB	±SE	WEVI	±SE
Nest height (m)	2.5	0.535	2.3	0.33	1.4	.	0.92	0.08
Nest DBH (cm)	4.1	0.458	4.4	0.35	.	.	11.05	2.75
Distance to substrate center (m)	0.98	0.314	1.48	0.375	0.5	.	0.78	0.225
% Canopy cover	49	10.3	38	22.5	10	.	70	0
% Grass cover	91	2.4	55	25	100	.	10	5
% Leaf cover	34	8.5	23	17.5	50	.	65	20
% Bare ground	23	8.3	53	22.5	65	.	90	10
% Coarse woody debris	3	2.5	0	0	0	.	5	5
% Forb cover	93	4.3	95	5	95	.	75	10
% woody vegetation	54	15.3	35	25	20	.	85	0
% Moss cover	4	2.4	15	0	20	.	10	10
Conifer stem density (#/ha)	0	0	0	0	0	.	0	0
Deciduous stem density (#/ha)	1048	304.5	515	138.1	803	.	1280	351.5
Total density of trees 4-10cm DBH	672	263.8	239	62.8	126	.	615	439.4
Total density of trees 10-15cm DBH	151	92.2	50	25.1	0	.	38	12.6
Total density of trees >15 cm DBH	119	21.4	113	62.8	0	.	75	75.3
% understory cover 0-0.5m	98	0	96	1.6	94	.	91	6.9
% understory cover 0.5-1m	94	2.3	91	3.1	88	.	93	4.7
% understory cover 1-1.5m	83	9.2	75	0.6	84	.	88	9.4
% understory cover 1.5-2m	78	12.6	59	8.1	74	.	94	3.1
% understory cover 2-2.5m	77	12.4	48	8.4	56	.	93	4.7
CV distance to nearest tree >3cm DBH	76.32	14.478	77.38	3.597	51.03	.	66.2	23.652

**Table B1.** (continued)

	WITU	±SE	YBCH	±SE
Nest height (m)	0	.	1.17	0.195
Nest DBH (cm)	.	.	.	.
Distance to substrate center (m)	.	.	0.94	0.555
% Canopy cover	0	.	63	13.9
% Grass cover	80	.	79	9.4
% Leaf cover	0	.	44	15.2
% Bare ground	75	.	23	11.6
% Coarse woody debris	0	.	1	1.3
% Forb cover	85	.	89	2.4
% woody vegetation	75	.	60	12.4
% Moss cover	0	.	3	2.5
Conifer stem density (#/ha)	0	.	75	75.3
Deciduous stem density (#/ha)	502	.	2586	619.3
Total density of trees 4-10cm DBH	100	.	395	181.1
Total density of trees 10-15cm DBH	0	.	100	37
Total density of trees >15 cm DBH	0	.	50	27.1
% understory cover 0-0.5m	98	.	98	0
% understory cover 0.5-1m	98	.	98	0
% understory cover 1-1.5m	98	.	98	0
% understory cover 1.5-2m	98	.	92	5.9
% understory cover 2-2.5m	98	.	92	5.9
CV distance to nearest tree >3cm DBH	.	.	87.97	18.259

**Appendix C. Species detection histories for each cover type within each year (2007, 2008, 2010, and 2011).**

**Table C1.** Encounter histories for bird species detected (1) or not (0) within pasture cover types at each visit (secondary sampling period) within each year (primary sampling occasion, 2007, 2008, 2010 and 2011) on reclaimed surface-mines in southwest Virginia.

species	Year (primary sampling occasion)																		
	2007					2008					2010				2011				
	visit					visit					visit				visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5
American crow <i>Corvus brachyrhynchos</i>	1	1	0	1	1	1	1	0	1	1	1	1	1	1	1	1	1	1	1
American goldfinch <i>Carduelis tristis</i>	1	1	1	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	0
American robin <i>Turdus migratorius</i>	0	0	0	1	0	0	0	0	0	0	0	0	1	0	1	1	0	1	0
Barn swallow <i>Hirundo rustica</i>	0	0	0	1	1	1	1	0	0	0	1	0	1	1	0	1	1	0	1
Black-and-white warbler <i>Mniotilta varia</i>	1	0	0	0	0	1	1	1	0	0	0	0	0	0	0	1	0	1	0
Black-billed cuckoo <i>Coccyzus erythrophthalmus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0
Black-capped chickadee <i>Poecile atricapillus</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Blue-gray gnatcatcher <i>Polioptila caerulea</i>	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0
Brown-headed cowbird <i>Molothrus ater</i>	0	0	0	1	1	1	1	0	0	0	0	0	0	0	1	1	1	1	1
Blue jay <i>Cyanocitta cristata</i>	0	0	1	0	1	1	0	0	1	0	1	1	0	0	0	0	1	0	0
Brown thrasher <i>Toxostoma rufum</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1	1	0	0
Black-throated-blue warbler <i>Dendroica caerulescens</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0

**Table C1.** (continued)

species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Black-throated-green warbler	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0
<i>Dendroica virens</i>																				
Blue-winged warbler	0	0	1	0	1	0	0	0	0	0	0	0	1	0	1	1	1	0	0	0
<i>Vermivora pinus</i>																				
Carolina chickadee	1	0	1	0	0	0	1	0	1	0	0	1	0	0	0	1	0	0	0	1
<i>Poecile carolinensis</i>																				
Carolina wren	1	1	0	0	1	1	0	1	0	0	0	0	1	0	1	1	1	1	1	0
<i>Thryothorus ludovicianus</i>																				
Cedar waxwing	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1
<i>Bombycilla cedrorum</i>																				
Chipping sparrow	1	1	0	1	1	1	1	0	0	1	0	0	0	0	0	1	1	1	1	1
<i>Spizella passerina</i>																				
Chestnut-sided warbler	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Dendroica pensylvanica</i>																				
Common grackle	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Quiscalus quiscula</i>																				
Common yellowthroat	0	0	1	0	0	1	0	0	0	1	0	1	1	0	0	1	1	0	0	1
<i>Geothlypis trichas</i>																				
Downy woodpecker	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Picoides pubescens</i>																				
Eastern bluebird	1	0	0	0	0	1	0	0	0	1	0	0	0	0	1	0	0	0	0	0
<i>Sialia sialis</i>																				
Eastern kingbird	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Tyrannus tyrannus</i>																				
Eastern meadowlark	0	1	0	0	0	1	1	1	1	0	0	1	1	0	0	1	1	1	1	1
<i>Sturnella magna</i>																				

**Table C1.** (continued)

species	Year (primary sampling occasion)																		
	2007					2008					2010				2011				
	visit					visit					visit				visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5
Eastern phoebe	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Sayornis phoebe</i>																			
Eastern towhee	1	1	1	1	1	0	1	1	1	1	1	1	1	0	1	1	1	1	1
<i>Pipilo erythrophthalmus</i>																			
Eastern wood-pewee	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0
<i>Contopus virens</i>																			
European starling	0	0	1	1	1	0	0	0	1	1	0	0	0	0	1	0	1	0	0
<i>Sturnus vulgaris</i>																			
Field sparrow	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
<i>Spizella pusilla</i>																			
Great-crested flycatcher	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Myiarchus crinitus</i>																			
Gray catbird	0	0	0	0	0	0	0	0	0	0	1	1	1	1	0	1	0	0	0
<i>Dumetella carolinensis</i>																			
Grasshopper sparrow	1	1	0	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1
<i>Ammodramus savannarum</i>																			
Golden-winged warbler	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Vermivora chrysoptera</i>																			
Hairy woodpecker	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
<i>Picoides villosus</i>																			
Hooded warbler	0	1	0	0	0	0	0	1	0	0	1	0	1	0	1	1	1	0	0
<i>Wilsonia citrina</i>																			
Indigo bunting	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
<i>Passerina cyanea</i>																			
Killdeer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
<i>Charadrius vociferous</i>																			

**Table C1.** (continued)

species	Year (primary sampling occasion)																		
	2007					2008					2010				2011				
	visit					visit					visit				visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5
Mallard <sup>†</sup>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Anas platyrhynchos</i>																			
Mourning dove	1	0	0	1	1	1	1	0	1	1	0	0	0	0	1	1	0	0	0
<i>Zenaida macroura</i>																			
Northern bobwhite	0	0	1	0	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0
<i>Colinus virginianus</i>																			
Northern cardinal	1	1	1	1	1	1	0	1	0	0	0	0	1	1	1	1	1	1	1
<i>Cardinalis cardinalis</i>																			
Northern Flicker	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
<i>Colaptes auratus</i>																			
Northern mockingbird	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0
<i>Mimus polyglottos</i>																			
Ovenbird	0	1	0	0	0	1	1	1	1	0	0	0	0	0	1	0	0	1	1
<i>Seiurus aurocapilla</i>																			
Pileated woodpecker	1	0	1	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	0
<i>Dryocopus pileatus</i>																			
Prairie warbler	0	0	1	0	0	1	1	1	1	0	0	1	0	0	1	0	0	0	0
<i>Dendroica discolor</i>																			
Red-eyed vireo	1	1	1	1	1	1	1	1	0	1	1	0	1	1	1	1	1	1	1
<i>Vireo olivaceus</i>																			
Red-shouldered hawk	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Buteo lineatus</i>																			
Red-tailed hawk	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Buteo jamaicensis</i>																			
Red-winged blackbird	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1
<i>Agelaius phoeniceus</i>																			

**Table C1.** (continued)

species	Year (primary sampling occasion)																		
	2007					2008					2010				2011				
	visit					visit					visit				visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5
Scarlet tanager	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	1
<i>Piranga olivacea</i>																			
Song sparrow	1	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0
<i>Melospiza melodia</i>																			
Tree swallow	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Tachycineta bicolor</i>																			
Tufted titmouse	1	0	1	0	1	1	1	0	0	0	0	0	0	0	1	1	1	1	1
<i>Baeolophus bicolor</i>																			
Warbling vireo	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Vireo gilvus</i>																			
White-breasted nuthatch	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Sitta carolinensis</i>																			
White-eyed vireo	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	1	1	1	1
<i>Vireo griseus</i>																			
Wild turkey	0	0	1	0	0	0	1	0	0	0	0	1	0	0	0	1	0	0	0
<i>Meleagris gallopavo</i>																			
Wood duck <sup>†</sup>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1
<i>Aix sponsa</i>																			
Wood thrush	1	0	0	0	0	0	1	0	1	1	1	0	0	0	1	0	1	1	1
<i>Hylocichla mustelina</i>																			
Yellow-breasted chat	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
<i>Icteria virens</i>																			
Yellow-billed cuckoo	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0
<i>Coccyzus americanus</i>																			
Yellow warbler	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Dendroica petechia</i>																			

<sup>†</sup> Species were not included in analysis

**Table C2.** Encounter histories for bird species detected (1) or not (0) within traditional shrub cover types at each visit (secondary sampling period) within each year (primary sampling occasion, 2007, 2008, 2010 and 2011) on reclaimed surface-mines in southwest Virginia.

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	Visit					Visit					Visit					Visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
American crow <i>Corvus brachyrhynchos</i>	1	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
American goldfinch <i>Carduelis tristis</i>	1	1	1	1	1	1	1	1	1	0	0	1	1	1	1	0	1	0	1	
American robin <i>Turdus migratorius</i>	1	0	1	1	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	
Barn swallow <i>Hirundo rustica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	1	1	
Black-and-white warbler <i>Mniotilta varia</i>	1	1	1	0	1	0	1	1	1	0	0	0	0	0	1	1	1	1	0	
Black-billed cuckoo <i>Coccyzus erythrophthalmus</i>	0	0	0	0	0	0	0	0	1	0	0	0	1	0	1	1	1	0	0	
Black-capped chickadee <i>Poecile atricapillus</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	
Belted kingfisher <sup>†</sup> <i>Megaceryle alcyon</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
Blue-gray gnatcatcher <i>Polioptila caerulea</i>	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0	0	1	
Brown-headed cowbird <i>Molothrus ater</i>	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0	0	1	0	0	
Blue-headed vireo <i>Vireo solitarius</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
Blue jay <i>Cyanocitta cristata</i>	0	0	0	0	0	1	1	0	0	0	0	0	1	0	1	0	1	0	0	

**Table C2.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	Visit					Visit					Visit					Visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Brown thrasher	0	0	0	0	0	1	0	1	0	1	1	1	0	0	1	1	1	0	1	
<i>Toxostoma rufum</i>																				
Black-throated blue warbler	1	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Dendroica caerulescens</i>																				
Black-throated green warbler	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	
<i>Dendroica virens</i>																				
Blue-winged warbler	1	1	1	1	0	1	1	1	1	1	1	0	0	0	1	1	1	1	0	
<i>Vermivora pinus</i>																				
Carolina chickadee	1	0	0	0	0	0	1	0	0	0	0	1	1	1	1	0	1	1	1	
<i>Poecile carolinensis</i>																				
Carolina wren	0	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1	
<i>Thryothorus ludovicianus</i>																				
Cedar waxwing	1	0	0	0	0	0	0	0	1	1	1	1	0	1	0	0	1	1	1	
<i>Bombycilla cedrorum</i>																				
Chipping sparrow	1	1	1	1	1	0	1	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Spizella passerina</i>																				
Cliff swallow	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	
<i>Petrochelidon pyrrhonota</i>																				
Common yellowthroat	1	1	1	1	1	1	0	1	0	1	1	1	0	1	1	1	1	1	1	
<i>Geothlypis trichas</i>																				
Chestnut-sided warbler	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Dendroica pensylvanica</i>																				
Downy woodpecker	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	
<i>Picoides pubescens</i>																				
Eastern bluebird	0	0	0	0	0	1	1	0	0	0	0	0	0	0	1	0	1	0	0	
<i>Sialia sialis</i>																				

**Table C2.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	Visit					Visit					Visit					Visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Eastern meadowlark	0	1	0	0	0	0	1	0	0	0	1	1	0	0	0	0	0	1	0	
<i>Sturnella magna</i>																				
Eastern phoebe	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	
<i>Sayornis phoebe</i>																				
Eastern towhee	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Pipilo erythrophthalmus</i>																				
Eastern wood-pewee	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Contopus virens</i>																				
European starling	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Sturnus vulgaris</i>																				
Field sparrow	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Spizella pusilla</i>																				
Gray catbird	0	1	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Dumetella carolinensis</i>																				
Grasshopper sparrow	1	0	0	1	0	1	1	0	1	1	1	1	1	0	1	1	1	1	0	
<i>Ammodramus savannarum</i>																				
Golden-winged warbler	0	0	0	0	0	1	1	1	1	0	0	0	0	0	0	0	1	0	0	
<i>Vermivora chrysoptera</i>																				
Hairy woodpecker	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	
<i>Picoides villosus</i>																				
Hooded warbler	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1	1	
<i>Wilsonia citrina</i>																				
Indigo bunting	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Passerina cyanea</i>																				
Killdeer	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Charadrius vociferous</i>																				

**Table C2.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	Visit					Visit					Visit					Visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Mallard <sup>†</sup>	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	
<i>Anas platyrhynchos</i>																				
Mourning dove	0	1	1	0	0	0	1	1	0	0	1	0	1	0	0	0	0	0	0	
<i>Zenaida macroura</i>																				
Northern bobwhite	0	1	1	0	0	0	1	1	1	1	1	1	0	1	0	0	0	1	1	
<i>Colinus virginianus</i>																				
Northern cardinal	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Cardinalis cardinalis</i>																				
Northern flicker	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	1	1	1	1	
<i>Colaptes auratus</i>																				
Northern mockingbird	1	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Mimus polyglottos</i>																				
Northern parula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	
<i>Parula americana</i>																				
Ovenbird	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	
<i>Seiurus aurocapilla</i>																				
Pine warbler	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Dendroica pinus</i>																				
Pileated woodpecker	0	1	0	0	0	0	1	0	0	1	0	0	0	1	0	1	1	1	0	
<i>Dryocopus pileatus</i>																				
Prairie warbler	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Dendroica discolor</i>																				
Red-bellied woodpecker	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Melanerpes carolinus</i>																				
Red-eyed vireo	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Vireo olivaceus</i>																				

**Table C2.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	Visit					Visit					Visit					Visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Red-tailed hawk <i>Buteo jamaicensis</i>	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	
Ruby-throated hummingbird <i>Archilochus colubris</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	
Red-winged blackbird <i>Agelaius phoeniceus</i>	0	1	0	1	1	0	0	0	0	0	1	1	0	1	1	1	1	1	1	
Scarlet tanager <i>Piranga olivacea</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	
Song sparrow <i>Melospiza melodia</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
Swainson's warbler <i>Limnithlypis swainsonii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
Tree swallow <i>Tachycineta bicolor</i>	0	1	0	0	0	0	0	0	0	1	0	0	0	0	1	1	0	0	0	
Tufted titmouse <i>Baeolophus bicolor</i>	1	1	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	
White-breasted nuthatch <i>Sitta carolinensis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
White-eyed vireo <i>Vireo griseus</i>	0	0	1	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1	1	
Worm-eating warbler <i>Helmitheros vermivorum</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
Wild turkey <i>Meleagris gallopavo</i>	1	0	1	0	0	0	1	0	0	0	0	0	0	1	1	1	1	0	1	
Wood duck <sup>†</sup> <i>Aix sponsa</i>	0	1	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	

**Table C2.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	Visit					Visit					Visit					Visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Wood thrush	1	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Hylocichla mustelina</i>																				
Yellow-breasted chat	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Icteria virens</i>																				
Yellow-billed cuckoo	0	1	0	1	0	0	0	0	0	1	0	1	0	0	1	1	1	1	0	
<i>Coccyzus americanus</i>																				

† Species were not included in analysis

**Table C3.** Encounter histories for bird species detected (1) or not (0) within FRA cover types at each visit (secondary sampling period) within each year (primary sampling occasion, 2007, 2008, 2010 and 2011) on reclaimed surface-mines in southwest Virginia.

Species	Year( primary sampling occasion)									
	2010					2011				
	visit					visit				
	1	2	3	4	1	2	3	4	5	
American crow <i>Corvus brachyrhynchos</i>	1	1	1	1	1	1	1	1	1	1
American goldfinch <i>Carduelis tristis</i>	1	1	1	0	1	1	1	1	1	1
American kestrel <i>Falco sparverius</i>	0	1	0	1	0	0	0	0	0	0
American Robin <i>Turdus migratorius</i>	1	1	1	0	1	0	1	0	0	1
Barn swallow <i>Hirundo rustica</i>	0	1	1	0	1	1	1	0	0	1
Black-and-white warbler <i>Mniotilta varia</i>	1	0	1	0	0	0	1	0	0	0
Black-billed cuckoo <i>Coccyzus erythrophthalmus</i>	0	0	0	1	1	0	0	0	0	0
Black-capped chickadee <i>Poecile atricapillus</i>	1	0	1	1	0	0	0	0	0	0
Belted kingfisher† <i>Megaceryle alcyon</i>	1	0	0	0	0	0	0	0	0	0
Blue-gray gnatcatcher <i>Polioptila caerulea</i>	0	1	0	1	0	0	0	0	0	0
Brown-headed cowbird <i>Molothrus ater</i>	0	1	0	0	0	0	0	0	0	0
Blue-headed vireo <i>Vireo solitaries</i>	0	0	0	0	0	1	0	0	0	0

**Table C3.** (continued)

Species	Year( primary sampling occasion)									
	2010					2011				
	visit					visit				
	1	2	3	4	1	2	3	4	5	
Blue jay	1	1	0	0	1	0	1	0	0	
<i>Cyanocitta cristata</i>										
Brown thrasher	0	1	1	1	0	0	1	0	0	
<i>Toxostoma rufum</i>										
Black-throated green warbler	0	0	0	0	0	0	1	1	0	
<i>Dendroica virens</i>										
Blue-winged warbler	1	1	0	1	1	1	1	0	1	
<i>Vermivora pinus</i>										
Carolina chickadee	0	1	0	0	0	0	1	0	1	
<i>Poecile carolinensis</i>										
Carolina wren	1	0	1	0	1	0	1	0	0	
<i>Thryothorus ludovicianus</i>										
Cedar waxwing	0	1	1	1	0	0	0	0	1	
<i>Bombycilla cedrorum</i>										
Chipping sparrow	1	1	1	1	1	0	0	0	1	
<i>Spizella passerina</i>										
Common grackle	0	0	0	0	1	0	0	0	0	
<i>Quiscalus quiscula</i>										
Cooper's hawk	0	1	0	0	0	0	0	0	0	
<i>Accipiter cooperii</i>										
Common yellowthroat	1	1	1	1	1	1	1	0	1	
<i>Geothlypis trichas</i>										
Chestnut-sided warbler	0	1	0	0	0	0	0	0	0	
<i>Dendroica pensylvanica</i>										
Downy woodpecker	0	1	1	1	0	0	0	0	0	
<i>Picoides pubescens</i>										

**Table C3.** (continued)

Species	Year( primary sampling occasion)									
	2010					2011				
	visit					visit				
	1	2	3	4	1	2	3	4	5	
Eastern bluebird	0	0	0	0	0	0	1	0	0	
<i>Sialia sialis</i>										
Eastern meadowlark	0	0	1	1	0	1	1	1	0	
<i>Sturnella magna</i>										
Eastern towhee	1	1	1	1	1	1	1	1	1	
<i>Pipilo erythrophthalmus</i>										
Field sparrow	1	1	1	1	1	1	1	1	1	
<i>Spizella pusilla</i>										
Gray catbird	1	1	0	0	1	0	0	0	0	
<i>Dumetella carolinensis</i>										
Grasshopper sparrow	1	1	1	1	1	0	1	1	1	
<i>Ammodramus savannarum</i>										
Golden-winged warbler	1	1	0	0	0	1	0	0	1	
<i>Vermivora chrysoptera</i>										
Hairy warbler	0	0	0	0	1	0	0	0	0	
<i>Picoides villosus</i>										
Hooded warbler	1	1	1	1	1	1	1	1	1	
<i>Wilsonia citrina</i>										
Indigo bunting	1	1	1	1	1	1	1	1	1	
<i>Passerina cyanea</i>										
Mourning dove	0	1	0	1	1	1	0	0	0	
<i>Zenaida macroura</i>										
Northern bobwhite	0	1	1	0	0	0	0	0	1	
<i>Colinus virginianus</i>										
Northern cardinal	1	1	0	1	1	1	1	1	1	
<i>Cardinalis cardinalis</i>										

**Table C3.** (continued)

Species	Year( primary sampling occasion)									
	2010					2011				
	visit					visit				
	1	2	3	4	1	2	3	4	5	
Northern Flicker	1	0	1	0	0	1	1	0	1	
<i>Colaptes auratus</i>										
Northern mockingbird	1	0	0	1	0	0	0	0	0	
<i>Mimus polyglottos</i>										
Ovenbird	1	1	1	0	1	0	1	0	1	
<i>Seiurus aurocapilla</i>										
Pileated warbler	0	0	1	0	1	1	1	0	1	
<i>Dryocopus pileatus</i>										
Prairie warbler	1	1	1	1	1	1	1	1	1	
<i>Dendroica discolor</i>										
Red-eyed vireo	1	1	1	1	1	1	1	1	1	
<i>Vireo olivaceus</i>										
Red-winged blackbird	1	1	1	1	1	1	1	1	1	
<i>Agelaius phoeniceus</i>										
Scarlet tanager	0	0	0	0	0	0	1	1	1	
<i>Piranga olivacea</i>										
Song sparrow	0	1	0	0	0	0	1	0	0	
<i>Melospiza melodia</i>										
Tree swallow	0	0	0	0	0	1	1	0	0	
<i>Tachycineta bicolor</i>										
Tufted titmouse	0	0	1	0	1	1	1	1	1	
<i>Baeolophus bicolor</i>										
Veery	1	1	1	1	0	0	0	0	0	
<i>Catharus fuscescens</i>										
White-breasted nuthatch	0	1	0	0	0	0	0	0	0	
<i>Sitta carolinensis</i>										

**Table C3.** (continued)

Species	Year( primary sampling occasion)									
	2010					2011				
	visit					visit				
	1	2	3	4	1	2	3	4	5	
White-eyed vireo	1	1	1	1	1	1	1	1	1	1
<i>Vireo griseus</i>										
Wild turkey	1	0	0	0	1	1	0	0	0	0
<i>Meleagris gallopavo</i>										
Wood duck <sup>†</sup>	0	1	0	0	0	0	0	0	0	0
<i>Aix sponsa</i>										
Wood thrush	1	1	1	1	1	1	1	0	1	1
<i>Hylocichla mustelina</i>										
Yellow-breasted chat	1	1	1	1	1	1	1	1	1	1
<i>Icteria virens</i>										
Yellow-billed cuckoo	0	0	1	0	1	0	0	1	1	1
<i>Coccyzus americanus</i>										
Yellow warbler	0	0	0	0	0	1	0	0	0	0
<i>Dendroica petechia</i>										

<sup>†</sup>Species were not included in analysis.

**Table C4.** Encounter histories for bird species detected (1) or not (0) within traditional mixed cover types at each visit (secondary sampling period) within each year (primary sampling occasion, 2007, 2008, 2010 and 2011) on reclaimed surface-mines in southwest Virginia.

species	year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
American crow <i>Corvus brachyrhynchos</i>	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1	1	1	1	
American goldfinch <i>Carduelis tristis</i>	1	0	1	0	1	1	0	1	1	1	0	1	1	1	0	1	1	1	0	
American robin <i>Turdus migratorius</i>	0	0	0	0	1	0	0	0	0	0	1	0	1	1	0	0	0	1	1	
Barn swallow <i>Hirundo rustica</i>	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	
Black-and-white warbler <i>Mniotilta varia</i>	0	1	0	1	0	1	0	1	0	0	0	1	0	0	1	1	0	1	0	
Black-billed cuckoo <i>Coccyzus erythrophthalmus</i>	0	0	0	0	0	0	0	0	0	1	1	1	0	0	1	0	0	0	0	
Black-capped chickadee <i>Poecile atricapillus</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
Blue-gray gnatcatcher <i>Polioptila caerulea</i>	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
Blue jay <i>Cyanocitta cristata</i>	0	0	0	0	1	0	1	0	0	1	1	0	0	0	0	0	0	0	0	
Brown thrasher <i>Toxostoma rufum</i>	0	0	0	0	0	0	0	1	0	0	1	0	1	1	0	1	0	0	0	
Black-throated blue warbler <i>Dendroica caerulescens</i>	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
Black-throated green warbler <i>Dendroica virens</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	

**Table C4.** (continued)

species	year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Blue-winged warbler	1	1	0	0	0	1	0	1	1	1	1	1	1	0	1	1	1	1	0	
<i>Vermivora pinus</i>																				
Carolina chickadee	0	0	0	1	0	1	0	0	1	0	1	1	1	1	0	1	1	1	1	
<i>Poecile carolinensis</i>																				
Carolina wren	1	1	1	1	1	1	1	1	1	1	1	0	0	1	0	1	1	0	1	
<i>Thryothorus ludovicianus</i>																				
Cedar waxwing	1	0	0	0	0	0	0	0	0	0	1	0	1	1	1	1	0	1	0	
<i>Bombycilla cedrorum</i>																				
Chipping sparrow	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Spizella passerina</i>																				
Common yellowthroat	1	1	1	1	0	1	0	0	1	1	1	1	1	1	1	0	1	1	1	
<i>Geothlypis trichas</i>																				
Downy woodpecker	0	0	1	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Picoides pubescens</i>																				
Eastern bluebird	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Sialia sialis</i>																				
Eastern meadowlark	0	1	0	0	1	0	0	0	0	0	1	0	1	0	0	1	1	1	0	
<i>Sturnella magna</i>																				
Eastern towhee	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Pipilo erythrophthalmus</i>																				
Field sparrow	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Spizella pusilla</i>																				
Gray catbird	1	0	1	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Dumetella carolinensis</i>																				
Grasshopper sparrow	0	0	0	0	0	0	0	0	0	0	1	1	1	0	1	0	0	1	0	
<i>Ammodramus savannarum</i>																				

**Table C4.** (continued)

species	year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Golden-winged warbler <i>Vermivora chrysoptera</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
Hairy woodpecker <i>Picoides villosus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
Hooded warbler <i>Wilsonia citrina</i>	1	1	1	0	1	1	1	1	1	1	1	1	1	0	1	1	1	0	0	
Indigo bunting <i>Passerina cyanea</i>	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
Mourning dove <i>Zenaida macroura</i>	1	1	1	0	0	0	1	1	0	0	0	1	0	1	0	1	1	0	0	
Northern bobwhite <i>Colinus virginianus</i>	0	1	0	0	0	0	1	1	1	1	1	0	0	1	0	0	0	0	0	
Northern cardinal <i>Cardinalis cardinalis</i>	1	1	1	1	1	1	1	0	1	0	1	1	1	1	0	1	1	1	1	
Northern flicker <i>Colaptes auratus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	
Northern parula <i>Parula americana</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	
Ovenbird <i>Seiurus aurocapilla</i>	0	0	0	1	0	1	0	0	1	0	0	0	1	0	1	1	1	0	1	
Pileated woodpecker <i>Dryocopus pileatus</i>	0	0	1	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	
Prairie warbler <i>Dendroica discolor</i>	1	1	1	1	1	0	1	1	1	0	1	1	1	1	1	1	1	1	1	
Red-bellied woodpecker <i>Melanerpes carolinus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	

**Table C4.** (continued)

species	year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Red-eyed vireo <i>Vireo olivaceus</i>	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
Red-winged blackbird <i>Agelaius phoeniceus</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
Scarlet tanager <i>Piranga olivacea</i>	0	0	0	1	0	0	0	0	0	0	0	1	0	1	0	1	1	0	0	
Song sparrow <i>Melospiza melodia</i>	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
Swainson's warbler <i>Limnothlypis swainsonii</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	
Tufted titmouse <i>Baeolophus bicolor</i>	1	1	1	1	0	0	1	0	1	0	1	1	0	0	0	1	1	0	0	
White-eyed vireo <i>Vireo griseus</i>	0	1	0	0	0	1	0	1	0	1	1	1	1	1	1	1	0	0	1	
Wild turkey <i>Meleagris gallopavo</i>	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1	1	1	0	0	
Wood thrush <i>Hylocichla mustelina</i>	1	1	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0	1	
Yellow-breasted chat <i>Icteria virens</i>	0	1	1	1	1	1	1	0	1	1	1	1	1	1	1	1	1	1	1	
Yellow-billed cuckoo <i>Coccyzus americanus</i>	0	0	0	0	0	0	0	0	0	1	1	0	1	0	0	1	1	0	1	
Yellow-throated warbler <i>Dendroica dominica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	

**Table C5.** Encounter histories for bird species detected (1) or not (0) within traditional pine cover types at each visit (secondary sampling period) within each year (primary sampling occasion, 2007, 2008, 2010 and 2011) on reclaimed surface-mines in southwest Virginia.

species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
American crow <i>Corvus brachyrhynchos</i>	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
American goldfinch <i>Carduelis tristis</i>	0	0	0	0	0	0	1	1	1	1	0	1	0	0	0	1	1	0	0	
American robin <i>Turdus migratorius</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	
Barn swallow <i>Hirundo rustica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	
Black-and-white warbler <i>Mniotilta varia</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	
Black-billed cuckoo <i>Coccyzus erythrophthalmus</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	
black-capped chickadee <i>Poecile atricapillus</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	
Belted kingfisher <sup>†</sup> <i>Megasceryle alcyon</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
Brown-headed cowbird <i>Molothrus ater</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
Blue jay <i>Cyanocitta cristata</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Brown thrasher <i>Toxostoma rufum</i>	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	1	0	0	1	
Blue-winged warbler <i>Vermivora pinus</i>	0	0	0	0	0	0	0	1	1	0	0	0	0	0	1	1	1	0	0	

**Table C5.** (continued)

species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Carolina chickadee	0	0	0	1	0	0	0	0	1	1	0	1	1	1	1	0	0	1	0	
<i>Poecile carolinensis</i>																				
Carolina wren	1	0	1	0	1	1	1	0	0	0	0	0	0	0	1	1	1	0	1	
<i>Thryothorus ludovicianus</i>																				
Cedar waxwing	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	
<i>Bombycilla cedrorum</i>																				
Chipping sparrow	1	1	1	0	1	0	0	0	0	0	0	1	0	0	0	0	1	1	0	
<i>Spizella passerina</i>																				
Cooper's hawk	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Accipiter cooperii</i>																				
Common yellowthroat	1	1	1	1	1	1	0	0	1	1	1	0	0	1	1	1	1	1	1	
<i>Geothlypis trichas</i>																				
chestnut-sided warbler	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Dendroica pensylvanica</i>																				
Downy woodpecker	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Picoides pubescens</i>																				
Eastern meadowlark	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	
<i>Sturnella magna</i>																				
Eastern phoebe	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Sayornis phoebe</i>																				
Eastern towhee	0	1	1	1	1	1	0	1	1	0	1	1	1	1	1	1	1	1	1	
<i>Pipilo erythrophthalmus</i>																				
European starling	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Sturnus vulgaris</i>																				
Field sparrow	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Spizella pusilla</i>																				

**Table C5.** (continued)

species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Grasshopper sparrow <i>Ammodramus savannarum</i>	0	0	0	0	0	0	1	1	0	1	1	0	0	0	0	0	1	1	0	
Golden-winged warbler <i>Vermivora chrysoptera</i>	0	0	0	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	
Hairy woodpecker <i>Picoides villosus</i>	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	1	1	
Hooded warbler <i>Wilsonia citrina</i>	1	1	0	1	1	0	1	0	1	0	0	1	1	0	0	1	1	1	1	
Indigo bunting <i>Passerina cyanea</i>	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
Mourning dove <i>Zenaida macroura</i>	0	0	0	1	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	
Northern bobwhite <i>Colinus virginianus</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Norther cardinal <i>Cardinalis cardinalis</i>	1	0	1	1	1	0	0	0	0	1	0	1	0	1	1	1	1	1	0	
Norther flicker <i>Colaptes auratus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
Northern mockingbird <i>Mimus polyglottos</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Ovenbird <i>Seiurus aurocapilla</i>	0	0	0	1	0	1	1	0	0	0	0	0	0	0	1	1	1	1	1	
Pine warbler <i>Dendroica pinus</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	
Pileated woodpecker <i>Dryocopus pileatus</i>	1	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	

**Table C5.** (continued)

species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Prairie warbler	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	0	
<i>Dendroica discolor</i>																				
Red-eyed vireo	0	1	1	1	1	0	1	1	1	1	1	1	0	0	1	1	1	1	1	
<i>Vireo olivaceus</i>																				
Red-winged blackbird	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Agelaius phoeniceus</i>																				
Scarlet tanager	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	
<i>Piranga olivacea</i>																				
Song sparrow	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Melospiza melodia</i>																				
Swainson's warbler	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
<i>Limnothlypis swainsonii</i>																				
Tufted titmouse	0	1	1	1	0	1	0	0	1	0	0	1	0	0	0	1	0	1	1	
<i>Baeolophus bicolor</i>																				
White-eyed vireo	0	0	0	1	0	0	0	0	0	1	0	1	1	1	1	0	1	0	1	
<i>Vireo griseus</i>																				
Wild turkey	0	0	0	0	1	0	0	0	0	0	1	0	0	0	1	0	0	0	0	
<i>Meleagris gallopavo</i>																				
Wood thrush	0	0	1	1	1	1	1	1	0	0	0	0	0	0	0	0	0	1	0	
<i>Hylocichla mustelina</i>																				
Yellow-breasted chat	0	0	1	1	1	0	1	0	1	1	0	1	1	1	1	1	1	1	1	
<i>Icteria virens</i>																				
Yellow-billed cuckoo	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	
<i>Coccyzus americanus</i>																				

† Species were not included in analysis.

**Table C6.** Encounter histories for bird species detected (1) or not (0) within pre-SMCRA cover types at each visit (secondary sampling period) within each year (primary sampling occasion, 2007, 2008, 2010 and 2011) on reclaimed surface-mines in southwest Virginia.

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Acadian flycatcher	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Empidonax vireescens</i>																				
American crow	1	1	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Corvus brachyrhynchos</i>																				
American goldfinch	1	0	1	0	1	1	1	1	1	1	0	0	0	0	0	1	1	1	1	
<i>Carduelis tristis</i>																				
American kestrel	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Falco sparverius</i>																				
American robin	1	1	1	1	1	0	0	0	0	0	1	1	0	1	1	1	0	0	0	
<i>Turdus migratorius</i>																				
American woodcock	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Scolopax minor</i>																				
Barn swallow	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	
<i>Hirundo rustica</i>																				
Black-and-white warbler	1	1	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	
<i>Mniotilta varia</i>																				
Black-billed cuckoo	0	0	0	0	0	1	0	0	0	1	0	1	0	0	1	0	0	0	1	
<i>Coccyzus erythrophthalmus</i>																				
Black-capped chickadee	0	0	1	0	0	1	1	0	1	0	0	0	1	0	0	0	0	0	0	
<i>Poecile atricapillus</i>																				
Blue-gray gnatcatcher	0	0	0	0	1	0	0	0	1	0	1	0	1	0	0	1	0	1	0	
<i>Polioptila caerulea</i>																				
Brown-headed cowbird	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Molothrus ater</i>																				

**Table C6.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Blue-headed vireo <i>Vireo solitarius</i>	0	0	1	1	1	0	0	1	1	1	0	0	0	0	1	0	1	1	0	
Blue jay	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1	1	
<i>Cyanocitta cristata</i>																				
Brown creeper <i>Certhia americana</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
Brown thrasher <i>Toxostoma rufum</i>	1	0	0	0	1	0	0	0	0	0	1	0	1	0	0	1	1	1	0	
Black-throated blue warbler <i>Dendroica caerulescens</i>	1	1	0	0	0	0	0	0	1	0	0	0	0	0	1	1	0	0	0	
Black-throated green warbler <i>Dendroica virens</i>	1	1	1	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	
Broad-winged hawk <i>Buteo platypterus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
Blue-winged warbler <i>Vermivora pinus</i>	1	1	1	1	1	1	1	1	1	1	0	0	1	0	1	1	1	1	0	
Carolina chickadee <i>Poecile carolinensis</i>	1	1	0	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1	
Carolina wren <i>Thryothorus ludovicianus</i>	1	1	1	1	1	1	1	1	1	1	0	0	0	0	1	1	1	1	1	
Cedar waxwing <i>Bombycilla cedrorum</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	0	0	0	
Chipping sparrow <i>Spizella passerina</i>	1	1	1	1	1	1	1	1	0	1	0	0	0	1	1	1	1	1	1	
Cliff swallow <i>Petrochelidon pyrrhonota</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	

**Table C6.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Cooper's hawk <i>Accipiter cooperii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
Common yellowthroat <i>Geothlypis trichas</i>	0	1	1	0	0	1	0	0	0	1	0	0	0	0	1	1	0	1	0	
chestnut-sided warbler <i>Dendroica pensylvanica</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
Downy woodpecker <i>Picoides pubescens</i>	0	1	1	1	0	1	0	0	0	0	1	0	1	0	0	1	0	1	1	
Eastern bluebird <i>Sialia sialis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
Eastern meadowlark <i>Sturnella magna</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
Eastern phoebe <i>Sayornis phoebe</i>	1	1	1	0	0	1	1	0	1	1	1	1	1	0	0	1	1	0	0	
Eastern towhee <i>Pipilo erythrophthalmus</i>	1	1	1	1	1	1	1	1	1	1	1	0	0	0	1	1	1	1	1	
Eastern wood-pewee <i>Contopus virens</i>	0	1	0	0	1	0	1	0	0	0	1	0	0	0	0	0	1	1	1	
Field sparrow <i>Spizella pusilla</i>	1	1	0	0	0	1	1	1	1	1	0	1	1	0	1	0	1	1	1	
Great-crested flycatcher <i>Myiarchus crinitus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
Gray catbird <i>Dumetella carolinensis</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	
Golden-winged warbler <i>Vermivora chrysoptera</i>	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	

**Table C6.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Hairy woodpecker	0	0	0	0	0	1	0	0	0	0	0	1	0	0	1	1	0	1	1	
<i>Picoides villosus</i>																				
Hooded warbler	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Wilsonia citrina</i>																				
Indigo bunting	1	1	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	
<i>Passerina cyanea</i>																				
Kentucky warbler	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Oporornis formosus</i>																				
Mourning dove	1	1	1	1	0	1	1	1	0	1	0	0	1	0	1	1	1	1	1	
<i>Zenaida macroura</i>																				
Northern bobwhite	0	1	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	0	
<i>Colinus virginianus</i>																				
Northern cardinal	1	1	1	1	1	1	1	1	1	1	1	0	1	0	1	1	1	1	1	
<i>Cardinalis cardinalis</i>																				
Northern flicker	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1	1	0	1	
<i>Colaptes auratus</i>																				
Northern parula	0	1	1	1	0	0	0	0	0	0	0	0	0	0	1	0	1	1	1	
<i>Parula americana</i>																				
Ovenbird	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Seiurus aurocapilla</i>																				
Pileated woodpecker	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	
<i>Dryocopus pileatus</i>																				
Pine warbler	1	1	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	
<i>Dendroica pinus</i>																				
Prairie warbler	0	0	0	0	1	1	1	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Dendroica discolor</i>																				

**Table C6.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Red-bellied woodpecker <i>Melanerpes carolinus</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	
Red-eyed vireo <i>Vireo olivaceus</i>	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
Red-tailed hawk <i>Buteo jamaicensis</i>	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	
Ruby-throated hummingbird <i>Archilochus colubris</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
Red-winged blackbird <i>Agelaius phoeniceus</i>	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	
Scarlet tanager <i>Piranga olivacea</i>	0	0	1	1	1	0	1	1	1	1	0	1	0	1	1	1	1	1	1	
Song sparrow <i>Melospiza melodia</i>	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	1	0	0	0	
Swainson's warbler <i>Limnothlypis swainsonii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	
Tree swallow <i>Tachycineta bicolor</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
Tufted titmouse <i>Baeolophus bicolor</i>	1	1	1	1	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	
White-breasted nuthatch <i>Sitta carolinensis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	1	
White-eyed vireo <i>Vireo griseus</i>	0	0	0	0	0	0	1	1	1	0	0	0	0	0	1	1	1	1	1	
Worm-eating warbler <i>Helmitheros vermivorum</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1	1	0	

**Table C6.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010					2011				
	visit					visit					visit					visit				
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Wild turkey <i>Meleagris gallopavo</i>	0	1	0	0	1	0	1	0	0	0	1	0	0	0	1	0	1	0	1	
Wood thrush <i>Hylocichla mustelina</i>	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
Yellow-breasted chat <i>Icteria virens</i>	1	1	1	1	1	1	1	1	1	1	0	0	0	0	1	1	1	1	1	
Yellow-billed cuckoo <i>Coccyzus americanus</i>	0	1	1	0	1	0	0	1	1	1	0	1	0	0	1	1	1	1	0	
Yellow-throated vireo <i>Vireo flavifrons</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	
Yellow-throated warbler <i>Dendroica dominica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	

**Table C7.** Encounter histories for bird species detected (1) or not (0) within reference forest cover types at each visit (secondary sampling period) within each year (primary sampling occasion, 2007, 2008, 2010 and 2011) on reclaimed surface-mines in southwest Virginia.

Species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Acadian flycatcher	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Empidonax virescens</i>																				
American crow	1	1	1	1	1	1	1	1	1	1	1	0	0	0	1	1	1	1	1	
<i>Corvus brachyrhynchos</i>																				
American goldfinch	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	1	0	1	
<i>Carduelis tristis</i>																				
American redstart	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	
<i>Setophaga ruticilla</i>																				
American robin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Turdus migratorius</i>																				
American woodcock	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Scolopax minor</i>																				
Black-and-white warbler	1	1	1	1	0	1	1	1	1	0	0	0	0	0	1	1	1	1	1	
<i>Mniotilta varia</i>																				
Black-billed cuckoo	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	
<i>Coccyzus erythrophthalmus</i>																				
Black-capped chickadee	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Poecile atricapillus</i>																				
Belted kingfisher <sup>†</sup>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Megasceryle alcyon</i>																				
Blue-gray gnatcatcher	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	
<i>Polioptila caerulea</i>																				
Brown-headed cowbird	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	
<i>Molothrus ater</i>																				

**Table C7.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Blue-headed vireo <i>Vireo solitarius</i>	0	1	1	1	1	1	1	0	1	1	0	0	0	0	1	0	1	0	1	
Blue jay <i>Cyanocitta cristata</i>	1	1	1	1	0	0	0	1	0	0	0	1	0	0	1	1	0	1	1	
Brown thrasher <i>Toxostoma rufum</i>	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
Black-throated blue warbler <i>Dendroica caerulescens</i>	0	0	0	0	0	1	1	0	0	1	0	0	0	0	1	1	1	1	0	
Black-throated green warbler <i>Dendroica virens</i>	0	0	0	1	1	1	1	1	1	1	0	0	0	0	1	1	1	1	1	
Blue-winged warbler <i>Vermivora pinus</i>	0	0	1	0	0	0	1	0	0	1	0	0	0	0	1	1	1	1	0	
Carolina chickadee <i>Poecile carolinensis</i>	1	1	1	0	1	1	1	0	0	1	0	0	0	0	1	1	1	1	1	
Carolina wren <i>Thryothorus ludovicianus</i>	1	1	1	1	1	0	0	1	0	0	0	0	0	1	1	1	1	1	1	
Cedar waxwing <i>Bombycilla cedrorum</i>	1	0	0	0	0	1	0	0	0	1	0	0	0	1	0	0	0	0	1	
Chipping sparrow <i>Spizella passerina</i>	1	1	1	1	0	0	0	0	1	1	0	0	1	0	0	1	1	0	0	
Cooper's hawk <i>Accipiter cooperii</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Common yellowthroat <i>Geothlypis trichas</i>	1	0	0	1	0	0	0	0	0	1	0	0	0	0	1	0	0	1	1	
Chestnut-sided warbler <i>Dendroica pensylvanica</i>	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	

**Table C7.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Downy woodpecker	0	1	0	0	1	0	0	0	1	0	0	0	0	0	1	1	1	1	0	
<i>Picoides pubescens</i>																				
Eastern towhee	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Pipilo erythrophthalmus</i>																				
Eastern wood-pewee	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	
<i>Contopus virens</i>																				
Field sparrow	1	1	0	1	1	1	1	1	1	1	1	1	0	0	1	1	1	1	1	
<i>Spizella pusilla</i>																				
Gray catbird	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Dumetella carolinensis</i>																				
Golden-winged warbler	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Vermivora chrysoptera</i>																				
Hairy woodpecker	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	1	1	1	0	
<i>Picoides villosus</i>																				
Hooded warbler	1	1	1	1	1	1	1	1	1	1	0	0	0	0	1	1	1	1	1	
<i>Wilsonia citrina</i>																				
Indigo bunting	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Passerina cyanea</i>																				
Kentucky warbler	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	
<i>Oporornis formosus</i>																				
Magnolia warbler	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Dendroica magnolia</i>																				
Mourning dove	0	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Zenaida macroura</i>																				
Northern bobwhite	0	1	1	0	0	0	0	0	0	1	0	0	0	0	0	1	0	1	0	
<i>Colinus virginianus</i>																				

**Table C7.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
Northern cardinal <i>Cardinalis cardinalis</i>	1	1	1	1	1	0	1	0	0	0	0	1	1	0	1	1	1	1	1	
Northern flicker <i>Colaptes auratus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
Northern parula <i>Parula americana</i>	0	0	1	1	1	0	0	0	0	1	0	0	0	0	1	0	0	1	1	
Ovenbird <i>Seiurus aurocapilla</i>	1	1	1	1	0	1	1	1	1	1	1	1	0	0	1	1	1	1	1	
Pileated woodpecker <i>Dryocopus pileatus</i>	1	0	0	1	0	1	1	0	1	0	0	0	1	0	1	1	1	1	1	
Prairie warbler <i>Dendroica discolor</i>	1	1	0	0	0	1	1	0	0	0	0	0	0	0	1	0	0	0	0	
Red-eyed vireo <i>Vireo olivaceus</i>	1	1	1	1	1	1	1	1	1	1	0	0	0	0	1	1	1	1	1	
Red-shouldered hawk <i>Buteo lineatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	
Red-tailed hawk <i>Buteo jamaicensis</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Scarlet tanager <i>Piranga olivacea</i>	1	1	1	1	1	1	0	1	0	1	0	0	0	0	1	1	1	1	1	
Song sparrow <i>Melospiza melodia</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	
Swainson's warbler <i>Limnothlypis swainsonii</i>	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	1	1	0	1	
Tufted titmouse <i>Baeolophus bicolor</i>	1	1	1	0	1	1	1	1	1	1	0	1	0	0	1	1	1	0	1	

**Table C7.** (continued)

Species	Year (primary sampling occasion)																			
	2007					2008					2010				2011					
	visit					visit					visit				visit					
	1	2	3	4	5	1	2	3	4	5	1	2	3	4	1	2	3	4	5	
White-breasted nuthatch <i>Sitta carolinensis</i>	1	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	
White-eyed vireo <i>Vireo griseus</i>	0	0	0	1	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	
Worm-eating warbler <i>Helmitheros vermivorum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	
Wild turkey <i>Meleagris gallopavo</i>	1	1	0	1	0	0	0	1	0	0	0	0	0	0	1	1	0	1	0	
Wood thrush <i>Hylocichla mustelina</i>	1	1	1	1	1	1	1	1	1	1	0	0	1	0	0	1	1	1	1	
Yellow-breasted chat <i>Icteria virens</i>	1	0	1	1	1	1	1	0	1	1	0	1	0	0	1	1	1	1	1	
Yellow-billed cuckoo <i>Coccyzus americanus</i>	0	0	1	1	0	1	0	0	1	1	0	0	1	0	1	1	1	1	1	
Yellow-throated vireo <i>Vireo flavifrons</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	

† Species were not included in analysis.

**Appendix D. Relative abundances calculated as the number of observations per point per visit at our study sites in southwest Virginia during 2010 and 2011.**

**Table D1.** Relative abundance as measured by number of observations per point per visit of 85 species observed at our study sites in southwestern Virginia during 2010 and 2011.

	Cover type						
	Forest Reclamation	Pasture	Pre-SMCRA	Reference	Traditional Mixed	Traditional Pine	Traditional Shrub
	n= 10	n= 8	n= 20	n= 13	n= 4	n= 4	n= 9
Acadian flycatcher	0.011	0.000	0.006	0.000	0.000	0.000	0.000
American crow	0.633	0.583	0.533	0.564	1.306	0.611	0.728
American goldfinch	0.378	0.222	0.061	0.034	0.333	0.083	0.333
American kestrel	0.022	0.000	0.000	0.000	0.000	0.000	0.000
American redstart	0.000	0.000	0.000	0.034	0.000	0.000	0.000
American robin	0.089	0.083	0.028	0.009	0.194	0.083	0.049
Barn swallow	0.211	0.472	0.017	0.000	0.111	0.083	0.481
Black-and-white warbler	0.022	0.042	0.283	0.145	0.139	0.000	0.086
Black-billed cuckoo	0.033	0.056	0.039	0.034	0.111	0.000	0.099
Black-capped chickadee	0.033	0.014	0.033	0.017	0.056	0.056	0.062
Belted kingfisher	0.011	0.000	0.000	0.009	0.000	0.028	0.012
Blue-gray gnatcatcher	0.011	0.042	0.067	0.017	0.056	0.028	0.049
Brown-headed cowbird	0.011	0.250	0.000	0.009	0.000	0.028	0.012
Blue-headed vireo	0.011	0.000	0.022	0.034	0.000	0.000	0.012
Blue jay	0.067	0.042	0.161	0.162	0.028	0.028	0.062
Brown creeper	0.000	0.000	0.006	0.000	0.000	0.000	0.000
Brown thrasher	0.056	0.111	0.033	0.009	0.139	0.111	0.111
Black-throated blue warbler	0.000	0.014	0.028	0.034	0.000	0.000	0.012
Black-throated green warbler	0.022	0.042	0.317	0.111	0.028	0.000	0.025

**Table D1.** (continued)

	Cover type						
	Forest Reclamation	Pasture	Pre-SMCRA	Reference	Traditional Mixed	Traditional Pine	Traditional Shrub
Broad-winged hawk	0.000	0.000	0.006	0.000	0.000	0.000	0.000
Blue-winged warbler	0.167	0.111	0.217	0.068	0.333	0.194	0.321
Carolina chickadee	0.033	0.083	0.378	0.137	0.389	0.333	0.272
Carolina wren	0.044	0.139	0.328	0.103	0.194	0.222	0.247
Cerulean warbler	0.000	0.000	0.000	0.009	0.000	0.000	0.000
Cedar waxwing	0.144	0.069	0.033	0.026	0.500	0.111	0.173
Chipping sparrow	0.089	0.139	0.117	0.034	0.028	0.111	0.037
Cliff swallow	0.000	0.000	0.000	0.000	0.000	0.000	0.012
Common grackle	0.011	0.000	0.000	0.000	0.000	0.000	0.000
Cooper's hawk	0.011	0.000	0.017	0.000	0.000	0.000	0.000
Common yellowthroat	0.378	0.097	0.028	0.026	0.361	0.250	0.358
Chestnut-sided warbler	0.000	0.000	0.006	0.000	0.000	0.000	0.000
Downy woodpecker	0.022	0.000	0.050	0.034	0.028	0.000	0.037
Eastern bluebird	0.011	0.014	0.006	0.000	0.000	0.000	0.025
Eastern kingbird	0.000	0.014	0.000	0.000	0.000	0.000	0.000
Eastern meadowlark	0.078	0.153	0.000	0.000	0.194	0.028	0.062
Eastern phoebe	0.011	0.014	0.033	0.009	0.000	0.028	0.000
Eastern towhee	0.956	0.569	0.328	0.350	1.528	1.000	0.963
Eastern wood-pewee	0.000	0.028	0.039	0.017	0.000	0.000	0.000
European starling	0.000	0.028	0.000	0.000	0.000	0.000	0.136
Field sparrow	1.822	1.125	0.100	0.085	1.472	1.056	2.037
Great blue heron	0.000	0.000	0.011	0.000	0.000	0.000	0.000
Great-crested flycatcher	0.000	0.014	0.006	0.000	0.000	0.000	0.000
Gray catbird	0.067	0.097	0.006	0.000	0.028	0.000	0.000
Grasshopper sparrow	0.244	1.056	0.000	0.000	0.278	0.139	0.136
Golden-winged warbler	0.033	0.014	0.000	0.000	0.028	0.000	0.012

**Table D1.** (continued)

	Cover type						
	Forest Reclamation	Pasture	Pre-SMCRA	Reference	Traditional Mixed	Traditional Pine	Traditional Shrub
Hairy woodpecker	0.011	0.056	0.056	0.051	0.056	0.111	0.062
Hooded warbler	0.233	0.111	0.406	0.573	0.639	0.333	0.309
Indigo bunting	2.556	1.375	1.333	0.615	2.972	2.000	2.667
Kentucky warbler	0.000	0.000	0.011	0.009	0.000	0.000	0.000
Killdeer	0.000	0.014	0.000	0.000	0.000	0.000	0.000
Mourning dove	0.089	0.042	0.128	0.000	0.139	0.139	0.025
Northern bobwhite	0.089	0.014	0.006	0.017	0.056	0.000	0.086
Northern cardinal	0.222	0.292	0.367	0.197	0.389	0.472	0.420
Northern flicker	0.100	0.056	0.083	0.009	0.028	0.028	0.074
Northern mockingbird	0.044	0.028	0.000	0.000	0.000	0.000	0.025
Northern parula	0.000	0.014	0.050	0.034	0.083	0.000	0.025
Ovenbird	0.067	0.056	0.161	0.504	0.250	0.306	0.025
Pine warbler	0.000	0.000	0.028	0.000	0.000	0.000	0.012
Pileated woodpecker	0.144	0.056	0.156	0.094	0.028	0.056	0.086
Prairie warbler	0.600	0.056	0.000	0.009	0.611	0.361	0.864
Red-bellied woodpecker	0.000	0.000	0.000	0.000	0.028	0.000	0.012
Red-eyed vireo	0.600	0.222	0.750	0.632	1.500	0.583	0.531
Red-shouldered hawk	0.000	0.000	0.000	0.009	0.000	0.000	0.000
Red-tailed hawk	0.011	0.000	0.006	0.000	0.000	0.000	0.012
Ruby-throated hummingbird	0.000	0.000	0.011	0.000	0.000	0.000	0.012
Red-winged blackbird	0.689	0.069	0.033	0.000	0.028	0.000	0.333
Scarlet tanager	0.056	0.083	0.367	0.299	0.139	0.083	0.148
Song sparrow	0.022	0.000	0.006	0.017	0.000	0.000	0.012
Swainson's warbler	0.000	0.000	0.039	0.026	0.056	0.028	0.012
Tree swallow	0.022	0.000	0.006	0.000	0.000	0.000	0.062
Tufted titmouse	0.133	0.111	0.283	0.162	0.278	0.222	0.259

**Table D1.** (continued)

	Cover type						
	Forest Reclamation	Pasture	Pre-SMCRA	Reference	Traditional Mixed	Traditional Pine	Traditional Shrub
Veery	0.000	0.000	0.006	0.000	0.000	0.000	0.000
Warbling vireo	0.000	0.014	0.000	0.000	0.000	0.000	0.000
White-breasted nuthatch	0.011	0.000	0.028	0.000	0.000	0.000	0.012
White-eyed vireo	0.478	0.236	0.028	0.000	0.361	0.278	0.679
Worm-eating warbler	0.000	0.000	0.033	0.068	0.000	0.000	0.025
Wild turkey	0.033	0.028	0.078	0.043	0.139	0.083	0.086
Wood duck	0.067	0.056	0.000	0.000	0.000	0.000	0.037
Wood thrush	0.267	0.069	0.172	0.111	0.139	0.111	0.000
Yellow-breasted chat	1.100	0.236	0.089	0.137	0.917	0.417	1.086
Yellow-billed cuckoo	0.044	0.056	0.106	0.137	0.167	0.083	0.136
Yellow warbler	0.022	0.014	0.000	0.000	0.000	0.000	0.000
Yellow-rumped warbler	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Yellow-throated vireo	0.000	0.000	0.011	0.009	0.000	0.000	0.000
Yellow-throated warbler	0.000	0.000	0.039	0.000	0.028	0.000	0.000

**Appendix E. Sample R code for calculating bootstrapped standard errors of derived parameter estimates for community vital rates.**

```
#####
#TRADMIX#
#####
#####

####Load RMark into R#####
library(RMark)
####Import data and convert to correct format for analyses#####
#TradP_RD_format= convert.inp("C:/Users/Chris/Documents/TradP_RD_format.inp",
use.comments=TRUE)
TradP_RD_reverse=convert.inp("C:/Users/Chris/Documents/TradP_RD_reverse.inp",use.comme
nts=TRUE)
####Create vectors filled with 0's that I will store results
####for each iteration#####
phi=rep(0,999)
N1=rep(0,999)
N2=rep(0,999)
####create a matrix called indexes filled with numbers 0-65 representing spp
####the number in each row corresponds to the row in the original data matrix
####each column of this matrix is a separate "sample"
indexes=matrix(0,50,999)
  for(t in 1:999){
    indexes[,t]=sample(50, 50, replace=TRUE)}
####create another matrix filled with 50 rows and 10 cols filled with 0's####
####create a new data matrix called RD_pasturebs
####fill this new data matrix with capture histories from original data
####with each capture history corresponding to the row number given in indexes
S=matrix(0,50,999)
  for(t in 1:999){
    TradP_RD_reversebs=TradP_RD_reverse[indexes[,t],]

####set up data to run robust design with heterogeneity#####
trads.process= process.data(TradP_RD_reversebs,model= "RDHet",
####set time intervals for 5,5,4,5 secondary samples#####
#### with 1,1,2,1 yrs between primary occasions #####
  time.intervals=c(0,0,0,0,1,0,0,0,0,2,0,0,0,1,0,0,0,0))

####create a new time frame called period for each parameter#####
####periods correspond to years 1&2= period1 and years 3&4=period2#####
trads.ddl=make.design.data(trads.process)
```

```

#parameters=list(GammaDoublePrime=list(time.bins=c(0,1,2,3,4)),
#GammaPrime=list(time.bins=c(2,3,5))),right=TRUE)
trads.ddl$p$period=0
trads.ddl$p$period[trads.ddl$p$session==1|trads.ddl$p$session==2]=1
trads.ddl$c$period=0
trads.ddl$c$period[trads.ddl$c$session==1|trads.ddl$c$session==2]=1
trads.ddl$N$period=0
trads.ddl$N$period[trads.ddl$N$session==1|trads.ddl$N$session==2]=1
trads.ddl$pi$period=0
trads.ddl$pi$period[trads.ddl$pi$session==1|trads.ddl$pi$session==2]=1
#####create model parameterizations#####
#####constant survival#####
S.dot=list(formula=~1)
#####p varies by period with a mixture#####
p.mix=list(formula=~mixture,share=TRUE)
#####time dependent immigration#####
GammaDoublePrime.random=list(formula=~time, share=TRUE)
#####time dependent emmigration#####
#GammaPrime.random=list(formula=~time)
#####pi varies by period#####
pi.dot=list(formula= ~1)
#####N varies by session#####
N.period=list(formula=~period)
##### Run MARK
model#####
model1=mark(data=trads.process,trads.ddl,
            model.parameters=list(S=S.dot,
            GammaDoublePrime=GammaDoublePrime.random,
            p=p.mix,pi=pi.dot, N=N.period))
#####Obtain specific model results for phi, N1 and N2#####
results=(model1$results$real[1,1])
N0=(model1$results$real[8,1])
Nk=(model1$results$real[9,1])
##### store each result for each iteration in the respective vectors
phi[t]=results
N1[t]=N0
N2[t]=Nk
}

## Derive extinction parameter from phi#####
E=1-phi
## get variance of extinction parameter from bootstrap#####
var(E)
## Derive lambda parameter from N4/N1#####
Lam=N2/N1
## get variance of lambda#####

```

```
var(Lam)
## Derive Birth parameter from phi and N#####
B=(N2-(phi*N1))
## get variance of B#####
var(B)
####remove files from documents#####
cleanup(ask=F)
```

**Appendix F. Universal Transverse Mercator (UTM) coordinates for the 68 points sampled on our study site in southwest Virginia between 2007 and 2011.**

**Table F1.** UTM (zone 17 south) Easting and Northing for the 68 points sampled on our study plot in Wise, VA. during 2010 and 2011.

Point	Easting	Northing	veg_type
FRA001	349025	4095711	FRA
FRA002	349140	4095876	FRA
FRA003	349411	4096259	FRA
FRA004	349510	4096390	FRA
FRA005	349404	4096703	FRA
FRA006	349580	4096967	FRA
PA021	389868	4104077	TRADS
PA023	389906	4104323	TRADS
PA024	389470	4104245	TRADS
PA028	389595	4104599	REF
PA029	389326	4104323	REF
PA044	389568	4103855	TRADS
PA045	389205	4103659	TRADM
PA047	388187	4103692	SMCRA
PA048	388257	4103571	SMCRA
PA049	388444	4103517	SMCRA
PA051	388721	4102724	SMCRA
PA052	388911	4102867	SMCRA
PA054	389778	4103886	TRADS
PA064	387661	4105880	REF
PA065	387922	4106030	REF
PA073	389477	4101138	PAS
PA074	389590	4100941	PAS
PA075	389284	4101314	PAS
PA079	388531	4103131	SMCRA
PA080	388958	4103680	SMCRA
PA081	389291	4104564	REF
PA082	393500	4100032	SMCRA
PA084	393285	4100374	SMCRA
PA086	392976	4100710	SMCRA
PA088	393361	4100942	PAS
PA094	393307	4102291	SMCRA
PA096	393153	4102049	SMCRA

**Table F1.** (continued)

Point	Easting	Northing	veg_type
PA098	393055	4101667	SMCRA
PA099	392931	4101854	SMCRA
PA100	392739	4101884	SMCRA
PA101	389403	4100923	PAS
PO002	351181	4097403	REF
PO003	351189	4097216	REF
PO004	351051	4097200	REF
PO005	349221	4097638	SMCRA
PO006	349020	4097687	SMCRA
PO007	350059	4097637	TRADS
PO009	349445	4097149	TRADS
PO010	349396	4097370	TRADS
PO012	350327	4099727	TRADP
PO013	350555	4099827	REF
PO014	349848	4098205	TRADP
PO016	349966	4098397	TRADP
PO017	350221	4098554	REF
PO018	351382	4097897	SMCRA
PO019	351374	4098078	SMCRA
PO020	351279	4097766	SMCRA
PO030	350617	4098406	TRADS
PO032	350926	4098173	REF
PO033	351087	4098207	REF
PO035	350827	4097960	REF
PO036	350103	4098745	TRADM
PO037	350092	4099508	TRADM
PO038	351594	4097723	TRADP
PO039	350291	4097839	PAS
PO040	350381	4097987	PAS
PO041	350386	4098155	PAS
PO042	350503	4097192	TRADM
TNC001	387595	4092794	FRA
TNC002	388033	4093674	FRA
TNC003	388116	4093931	FRA
TNC004	388364	4094311	FRA