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Avian and salamander response to young forest management in West Virginia

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**AVIAN AND SALAMANDER RESPONSE TO YOUNG FOREST
MANAGEMENT IN WEST VIRGINIA**

Eric L. Margenau

Dissertation submitted
to the Davis College of Agriculture, Natural Resources, and Design
at West Virginia University

in partial fulfillment of the requirements for the degree of

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Natural Resource Science

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ABSTRACT

Avian and salamander response to young forest management in West Virginia

Eric L. Margenau

This dissertation focuses on the effects of various young forest habitat management techniques on the avian and salamander community in West Virginia. Wildlife species associated with the nascent stage of forest succession are experiencing precipitous population declines throughout much of the eastern United States due to decreases in the amount of young forest area which have been brought on by changes in disturbance regimes over the past century. As a result, the need to find novel approaches for creating young forest habitats to sustain young forest wildlife populations is necessary. However, young forest habitat creation often negatively affects species that are considered disturbance-avoidant. As a result, I assessed the tradeoffs between creating young forest habitat for disturbance-associated species with the potential negative effects of reducing habitat suitability for disturbance-avoidant species throughout this dissertation. In Chapter 1, I summarize how historical land use practices in West Virginia have created current young forest conditions and the effects that these conditions have had on wildlife species that are specialized in exploiting young forest areas. I also introduce the study sites where this research was conducted and provide chapter objectives and topics of this dissertation.

The recent proliferation of linear energy infrastructure throughout the central Appalachian region has prompted managers to explore ways of managing the young forest bird community in association with these long, linear openings but little empirical data exist. At the same time, forest interior songbirds and woodland salamanders are often negatively affected by energy infrastructure within forest dominated landscapes and any young forest management in these landscapes may further degrade habitats for forest interior species. In chapter 2 we studied how harvest size (15 m, 30 m, and 45 m wide) and intensity (14 m²/ha and 4.5 m²/ha residual basal area) of cut-back borders, which are linear tree cuttings adjacent to gas/oil pipeline and utility powerline rights-of-way (ROWs) or wildlife openings, influenced habitat suitability along ROWs and wildlife openings for the young forest and forest interior communities. The objectives of this chapter were to examine whether the implementation of cut-back borders increased habitat suitability for wildlife species and which cut-back border treatments optimized the tradeoff between maximizing positive responses of disturbance-dependent species (i.e., young forest species) while minimizing negative responses of disturbance-avoidant species (i.e., forest interior species). We found that young forest species' abundances and species richness increased one-year and two-years after treatment, particularly in the 15-m wide borders, likely due to the increase of young forest habitat in conjunction with existing habitat in ROW corridors or wildlife openings. Additionally, we found that forest interior songbirds and woodland salamander, groups that are often negatively affected by ROWs in forested landscapes, did not decrease in cut-back borders following treatment likely because the retention of canopy trees in combination with adjacent mature forests retained suitable habitat conditions. These results suggest that cut-back

borders along abrupt forest edges of ROWs and wildlife openings create suitable habitat conditions for young forest avifauna without negatively affecting forest interior avifauna. Cut-back borders, particularly those 15-m in width, appear to be a viable management option for managing forest bird communities in ROW and wildlife opening landscapes.

Due to the prominence of commercial logging as a means to create young forest habitat, forests of low- or poor-quality stocking are often overlooked and thus limit options for management and conservation of young forest wildlife species. In order to promote habitat for young forest wildlife species, more opportunities for creating young forest habitat via forest management are greatly needed. In chapter 3 we assessed three non-commercial young forest management treatments (clearcut-and-leave [CL], clearcut-and-windrow [CW], and hack-and-spray [HS]) for promoting young forest habitat. Similar to Chapter 2, the objectives of this chapter were to examine whether the non-commercial young forest management treatments increased habitat suitability for young forest wildlife species and to determine which treatments optimized the tradeoff between maximizing positive responses of disturbance-dependent species and minimizing negative responses of disturbance-avoidant species. Additionally, we included a cost-benefit analysis to quantify the tradeoffs between maximizing songbird richness (cavity nesting guild, forest gap habitat guild, forest interior habitat guild, and young forest habitat guild) and minimizing cost of forest management. Implementation of regeneration treatments induced a dichotomous response from the avian community. Young forest songbird species responded positively to the CL treatment and negatively to the HS treatment, while forest interior songbirds responded positively to the HS treatment and negatively to the CL treatment. Eastern red-backed salamander (*Plethodon cinereus*), a terrestrial salamander most suited to forest interiors, responded positively to the HS treatment immediately following treatment. Based on our cost-benefit analysis, the CL treatment was the optimum treatment for maximizing songbird species richness and minimizing cost of habitat management followed by the control, the HS treatment, and the CW treatment. These results suggest that non-commercial forest management can provide suitable habitat conditions for young forest songbirds but can negatively affect forest interior songbirds, while other treatments retain suitable habitat conditions for forest interior species but do not provide immediate suitable habitat for young forest species.

Land managers must be able to assess the influence of habitat features across multiple spatial scales when developing management plans. Stand-level habitat management measures are often thought to have the most profound effect on species presence, but broader spatial habitat features may limit the effectiveness of local-level management if not fully considered. In chapter 4, we assessed the influence of local- (100-m radius surrounding point counts) and landscape-level (500-m radius) habitat features on the forest songbird community in cut-back borders along rights-of-way (ROWs) and wildlife openings. We examined four local-level variables (stand-level canopy cover, proportion of maintained early-successional habitat, proportion of young forest/shrubland habitat, and ROW/wildlife opening width) and five landscape-level variables (proportion of young forest/shrubland, core forest, and mature forest and distance to and size of

nearest young forest/shrubland patch) to determine which variables had the strongest influence on species abundances and guild richness in cut-back borders. Landscape-level variables had the strongest influence on abundances and richness of young forest, forest gap, forest interior, and conservation priority species. Distance to nearest young forest/shrubland patch negatively influenced abundances of Carolina wren, eastern towhee, and hooded warbler (i.e., abundance increased with decreasing distance to nearest young forest/shrubland patch), and positively influenced abundance of ovenbird. Size of nearest young forest/shrubland patch positively influenced common yellowthroat abundance, and negatively influenced black-and-white warbler and indigo bunting abundances. Proportion of young forest/shrubland habitat at the landscape-level positively influenced mature forest-associated species black-and-white warbler, cerulean warbler, and hooded warbler as well as the species of conservation priority guild. No local-level variables strongly influenced abundance or richness in cut-back borders. These results indicate that within forest dominated landscapes, cut-back borders located closer to pre-existing young forest/shrubland patches benefit young forest species while size of nearest young forest/shrubland patch is likely species-specific. For mature forest species, presence of young forest/shrubland area at the landscape level appears to increase abundances in cut-back borders.

The focus of this dissertation has been on implementing novel young forest management techniques and evaluating their effects on the forest fauna. Our results indicate that young forest management techniques generally benefitted young forest birds but their effects on forest interior songbirds and woodland salamanders varied. Future implementation of young forest management should consider the tradeoffs between maximizing positive responses of the young forest community while minimizing negative responses of the forest interior community in order to develop broad ecosystem-based management strategies for wildlife communities.

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CHAPTER 1

PROJECT INTRODUCTION

1. Young forests in West Virginia and the Appalachian Region

West Virginia is the third most densely forested state in the United States (US) with 79% (ca. 5 million ha) of all land being classified as forest (Homer et al. 2015, Morin et al. 2016; Figure 1-1). Forestlands in West Virginia are principally comprised of 61–100 years old stands (83% of all forestland, Morin et al. 2016; Figure 1-2), which is a common pattern for many forests throughout the Appalachian region (Oswalt et al. 2010). The unimodal age distribution of forestlands (Figure 1-2) is the result of fluctuating landscape-scale forest disturbances (e.g., large-scale logging, wildfire) and socio-economic factors – which led to increases in wildfire suppression and farm abandonment – over the past 80 years. A major consequence of this shift towards greater proportions of mature forest stands (≥ 40 years old) is the loss of disturbance-associated flora and fauna (i.e., young forest obligates). Forest disturbances are essential components of forest ecosystems, as they provide the necessary stimuli to facilitate forest stand development and provide resources vital to many wildlife species (Tash and Litvaitis 2007, McDermott and Wood 2010, Dey 2014). The continued maturation of young forests without replacement in the Appalachian region in recent decades has raised concerns among natural resource managers about the capability of young forest obligate wildlife species to persist under contemporary disturbance regimes (Askins 2001, Trani et al. 2001, Oswalt et al. 2010, Greenberg et al. 2011a).

The vegetation successional stage immediately following a disturbance is often referred to as “early-successional” and typically encompasses grassland, shrubland, and young forest plant communities (Harper 2017). These areas are defined as: recently disturbed with minimal overstory cover and a dense understory vegetation layer occupied by disturbance-adapted, shade-intolerant, or pioneer plant species (Schlossberg and King 2007, Tavernia et al. 2016). However, the term “early-successional” can be misleading as grasslands, shrublands, and young forests describe distinct plant communities with different affiliated wildlife species. Young forests are the result of disturbances in forested systems, where saplings sprout from the root systems of felled trees (i.e., advanced regeneration) but hardwood species remain the dominant plant group

with minor amounts of grasses, forbs, and brambles (King and Schlossberg 2014). The term “early-successional” represents the third or fourth seral stage of a grassland plant community and thus, is distinct from a young forest plant community. Therefore, “young forest” is used throughout this document when describing recently disturbed forest systems.

The distribution and area of young forests – defined as forests 0–20 years old – has fluctuated considerably over the past century. Present-day forest conditions in West Virginia and the Appalachian region can be attributed to a legacy of stand-replacing human disturbances dating from the 1700s through the early 1900s (Lorimer 2001). Forest disturbances during this period were prevalent due to Native American and European presence in the region, as forested areas were converted to agriculture or logging-centered landscape matrices. Land that was useful for agriculture was cleared of trees where needed, contributing to an increase in open areas during the late nineteenth century. Following the Great Depression (ca. early 1940s), large extents of farmland were abandoned and allowed to revert to forest cover. As of 2012, 41% of farmland that existed in West Virginia in the 1940’s remains (USDA 1940-2012; Figure 1-3). Between the 1840s and 1920s, logging was the primary source of forest disturbance in West Virginia and by the late 1920s over 99% of forestland in the state had been logged. By 1940, forestlands, of all ages, still accounted for the majority of land in West Virginia (about 60%) but forestry and the perception of logging had transitioned towards a conservative timber management approach, dramatically reducing large-scale disturbance in the state (Williams 1989).

Historically, anthropogenic and natural wildfires played a minor role in creating young forests in West Virginia relative to the Appalachian region and the eastern USA (Lorimer 2001, Lynch and Hessel 2010, White et al. 2011, Greenberg et al. 2015). Native Americans often used fire for clearing land or hunting, and European settlers adopted this practice. But by the early 1900s, political pressure led to policies that restricted the use of fire and these restrictions continued through the century due to an emphasis on commodity production (White et al. 2011). By the late nineteenth century, wildfire, logging, and farm abandonment combined to create large areas of young forest, accounting for 55–60% of all forest cover (Lorimer 2001). However, the combined effects of fire suppression, reduced logging, and maturation of old farmlands has reduced the extent of young forests we see today. Since 1949, the total area of young forestland in West Virginia has decreased from 19% to 7% while mature forestland has increased from 46%

to 78% (referred to as “small diameter” and “large diameter” forests, respectively, in Morin et al. 2016).

The decline in total area and distribution of young forestland has negatively affected young forest obligate wildlife communities (Dessecker and McAuley 2001, Hunter et al. 2001, DeGraaf and Yamasaki 2003, Dettmers 2003). Wildlife species that are adapted to disturbance-mediated habitats are often restricted by forest regeneration and the succession of plant composition into later seral stages (i.e., temporal specialists). Litvaitis (1993) suggested young forest-associated birds be considered habitat specialists, rather than generalists, because they occupy a narrower range of habitats compared to mature forest birds. Due to woody and herbaceous vegetation growth following forest disturbance (Roberts and Gilliam 2003, Roberts 2004), young forests provide food resources from fruiting plants (Greenberg et al. 2005, Reynolds-Hogland et al. 2006) and arthropods (Buddle et al. 2006, Greenberg et al. 2011b, Wilson et al. 2014), as well as nesting and juvenile dispersal cover from predators (Thompson III and Dessecker 1997, McDermott and Wood 2010). Additionally, songbird species not typically associated with disturbances (i.e., mature forest obligate species) use young forest areas prior to fall migration because high fruit production facilitates fat storage buildup (Suthers et al. 2000, Rodewald and Brittingham 2004).

While many songbird species respond positively to disturbance, wildlife taxa that are physiologically constrained to moist conditions (e.g., amphibians) may respond negatively to the same forest disturbance (Hillman et al. 2009). Salamanders are at an increased risk of desiccation following disturbances that result in a warmer, drier microclimate and reduced leaf litter cover and depth, although responses to disturbance are often species-specific and vary with disturbance intensity (deMaynadier and Hunter Jr. 1995, Duguay and Wood 2002, Homyack and Haas 2009, Moorman et al. 2011, Mahoney et al. 2016). Because salamanders provide important ecosystem functions (e.g., breakdown coarse woody debris, cycle nutrients throughout the ecosystem) in forest systems, maintaining salamander populations and community richness is an important desired outcome when planning forest management operations (Petranka et al. 1994).

The loss of young forest obligate wildlife in the eastern USA has prompted state and federal wildlife agencies as well as many non-profit organizations to focus efforts on ways to create and promote young forests in the region. Currently, 23% of all of terrestrial vertebrates listed as species of concern in West Virginia’s State Wildlife Action Plan (SWAP) are

considered young forest obligates (WVDNR 2015, Table 1-1). However, uncertainty exists surrounding the tradeoffs between taxa groups following young forest-directed management. Because traditional forms for young forest creation are unlikely (e.g., large-scale logging) or unreliable (e.g., wildfire) in West Virginia, young forest-directed management is needed to promote this stage of forest succession and to understand the benefits and drawbacks associated with young forest habitat creation (DeGraaf and Yamasaki 2003). This research will focus on quantifying avian and salamander community responses to various young forest management activities in West Virginia.

2. Study Region

This study was conducted throughout West Virginia on wildlife management areas (WMAs, Table 1-2) owned by the West Virginia Division of Natural Resources (WVDNR). West Virginia lies solely within the Appalachian region (Figure 1-4). West Virginia's humid continental climate is characterized by warm summers and cool to cold winters with average spring and fall temperatures of 10.0 °C and an average summer temperature of 22.0 °C. Annual precipitation levels range from 76, 127, and 102 cm in the western, central, and eastern portions of the state, respectively (NOAA 2015) with precipitation distributed evenly throughout the year but heaviest from June–September. West Virginia lies within the path of predominantly westerly winds which are influenced by cold northern air and warm southern air which can produce considerable amounts of precipitation (Lee et al. 1977, Green and Pauley 1987). In addition, the mountains along the Central Appalachian-Ridge and Valley border cause prevailing winds to rise and cool, thus releasing higher precipitation to the west and lower precipitation to the east of the mountains.

Central Appalachian

The central and southern parts of West Virginia are contained within the Central Appalachian region (Figure 1-5). This region consists of extensive forests with a highly dissected, rugged plateau composed of sandstone, shale, conglomerate, and coal. The rugged terrain, cool climate, and infertile soil limit agriculture, resulting in mostly forested land cover (Woods et al. 1999). Forests are primarily mixed mesophytic (Braun 1950), with areas of Appalachian oak (dominated by white [*Quercus alba*], and red oaks [*Q. rubra*]) and northern hardwoods (dominated by sugar maple [*Acer saccharum*], yellow birch [*Betula alleghaniensis*], beech [*Fagus* spp.], and eastern hemlock [*Tsuga canadensis*]; Kuchler 1964). Higher elevations

(ranging from 549 to 1,402 m above mean sea level) in this ecoregion produce greater annual mean precipitation (96–152 cm) and lower mean temperatures (11.5 °C) than the adjacent ecoregion to the east (e.g., Ridge and Valley).

Ridge and Valley

The eastern panhandle of West Virginia is contained within the Ridge and Valley ecoregion (Figure 1-5). This region's topography is a result of extreme folding and faulting events, creating parallel ridges and valleys in a northeast-southwest orientation (Woods et al. 1999). Ridges are mostly composed of resistant sandstone or conglomerate while valleys have generally weaker strata of limestone and shale. Due to this, land use is characterized by alternating forested ridges and agricultural valleys that are elongated. Forest cover is extensive in this ecoregion, mostly Appalachian oak and oak-hickory-pine forests (dominated by hickories [*Carya* spp.], shortleaf pine [*Pinus echinata*], white oak, and post oak [*Quercus stellata*]; Kuchler 1964). Elevations range from 244 to 1,311 m in this ecoregion but is bordered by the higher Blue Ridge (east) and Central Appalachian (west) ecoregions, resulting in less severe winters, considerably warmer summers (12.0 °C mean annual temperature), and lower annual precipitation (93 cm) than its neighbors (Woods et al. 1999).

Western Alleghany Plateau

The northern and western part of West Virginia is contained within the Western Alleghany Plateau region (Figure 1-5). This region consists of a mostly unglaciated, dissected plateau with local relief and crestal elevations of less than 600 m (Woods et al. 1999). Land use and land cover is a mosaic of forests, urban-suburban-industrial activity, dairy and livestock farms, coal mines, and oil-gas fields. Soils support Appalachian oak forests in the northwest portion of this ecoregion, while mixed-mesophytic forest is more common in the southern part (Kuchler 1964). The Western Alleghany Plateau, with elevations ranging from 61 to 488 m, is less rugged, not as forested and warmer than the Central Appalachian ecoregion to the east and south.

3. Study Design

We created young forest by tree cutting and herbicide application. Young forest created with these methods varied in size, intensity, and landscape context. Firstly, young forest was created within fragmented forest landscapes that included rights-of-way (ROWs; gas pipeline or transmission powerline) and wildlife openings. ROWs were long, linear openings maintained via

cutting, mowing, or herbicide spraying. Wildlife openings were fields planted with an agricultural crop (e.g., corn, clover) or left fallow. Tree cutting along forest edges of ROWs and wildlife openings was done at three different depths (i.e., plot sizes) into the forest (15, 30, and 45 m into the forest, perpendicular to forest edge; 0.5, 1.0, and 1.5 ha plot sizes, respectively) and two different cutting intensities (4.5 and 14.0 m²/ha basal area retention). Young forest management along ROWs and wildlife openings are hereafter referred to as “cut-back borders”. Secondly, young forest was created within surrounding intact forests consisting of three different management techniques (clearcut-and-leave, clearcut-and-windrow, and hack-and-spray) on individual 4 ha plots. Young forest management within surrounding forests are herein referred to as “regeneration stands”. All cutting and spraying was contracted through Wildlife Management Institute. Further details of each young forest management technique will be addressed in later chapters. We followed a before-after-control-impact (BACI) framework to study causal effects of these management techniques on the plant and animal communities, with one year of pre-treatment data collection and one or two years of post-treatment data collection (Table 1-2).

A single block replicate of cut-back borders refers to all combinations of three different cutting depths (15, 30, and 45 m) and two cutting intensities (4.5 and 14.0 m²/ha) with the addition of a control (no tree cutting or canopy manipulation). A single block replicate of regeneration stands includes three management prescriptions (clearcut-and-leave, clearcut-and-windrow, and hack-and-spray) with the addition of a control (no tree cutting or canopy manipulation).

4. Study Site Details

Allegheny Wildlife Management Area.—Allegheny WMA (AGWMA) is located in eastern West Virginia (Mineral County), near the city of Keyser on the West Virginia-Maryland state border (Figure 1-6; site coordinates in Appendices 4-A and 4-C). The management area covers 2,509 ha, consisting of flat, rocky mountaintop and steep slopes leading to foothills and hollows. Predominant forest cover is oak-hickory with interspersed mountain laurel (*Kalmia latifolia*). Elevation ranges from 319 to 950 m. AGWMA has one block replicate of cut-back borders along a transmission powerline/gas pipeline. AGWMA was cut and treated between fall of 2018 and spring of 2019, with pre-treatment data collected in the summer of 2018 and post-treatment data collected in the summer of 2019 (Table 1-2).

Beury Mountain Wildlife Management Area.—Beury Mountain WMA (BMWMA) is located in the south-central part of West Virginia (Fayette County), near the city of Beckley (Figure 1-7; site coordinates in Appendices 4-A through 4-F). The management area covers 1,238 ha, consisting of gentle to steep slopes with mixed hardwood and oak-hickory forests. Elevation ranges from 610 to 1,108 m. BMWMA has two block replicates of cut-back borders along two separate transmission powerlines (one block on each powerline) and five block replicates of regeneration stands. All research plots at BMWMA were cut and treated between fall of 2017 and spring of 2018. Pre-treatment data were collected during the summer of 2017 and post-treatment data were collected in the summers of 2018 and 2019 (Table 1-2).

Bluestone Wildlife Management Area.—Bluestone WMA (BSWMA) is located in southern West Virginia (Summers County), near the city of Hinton on the West Virginia-Virginia state border (Figure 1-8; site coordinates in Appendices 4-A and 4-C). The management area covers 7,292 ha, consisting of flat bottomlands to rolling uplands with predominantly oak-hickory forests. BSWMA is located around the periphery of a reservoir 797 ha in size. Elevation ranges from 432 to 865 m. BSWMA has a single block of cut-back borders along wildlife openings. All research plots at BSWMA were cut and treated between fall 2017 and spring of 2018. Pre-treatment data were collected during the summer of 2017 and post-treatment data were collected in the summers of 2018 and 2019 (Table 1-2).

Burnsville Lake Wildlife Management Area.—Burnsville Lake WMA (BLWMA) is located in central West Virginia (Braxton County) near the town of Burnsville (Figure 1-9; site coordinates in Appendices 4-A and 4-C). The management area covers 5,090 ha, mainly consisting of hilly and steep terrain, with some gently sloping upland areas and fairly flat creek bottoms with a mixture of young timber, brush, and old fields. BLWMA is mainly composed of mixed pine-hardwood forests. Elevation ranges from 243 to 472 m. BLWMA has one block replicate of cut-back borders along a gas pipeline that runs along a ridgeline. Research plots at BLWMA were cut and treated between fall of 2016 and spring 2017. Pre-treatment data were collected in the summer of 2016 and post-treatment data were collected in the summers of 2017 and 2018 (Table 1-2).

Center Branch Wildlife Management Area.—Center Branch WMA (CBWMA) is in north-central West Virginia (Harrison County) near the city of Clarksburg (Figure 1-10; site

coordinates in Appendices 4-B and 4-D). The management area cover 394 ha, with strip-bench flats to moderately steep slopes interspersed with gas wells and access roads. CBWMA is mainly composed of mixed and cove hardwoods. Elevation ranges between 300 and 465 m. CBWMA has three block replicates of regeneration stands, but excludes hack-and-spray treatments. All research plots were cut between fall of 2017 and spring of 2018, with pre-treatment data collected in the summer of 2017 and post-treatment data collected in summers of 2018 and 2019 (Table 1-2).

Lewis Wetzel Wildlife Management Area.—Lewis Wetzel WMA (LWWMA) is located in northwestern West Virginia (Doddridge, Tyler, and Wetzel Counties) near the town of Jacksonburg (Figure 1-11; site coordinates in Appendices 4-A, 4-C, and 4-E) and covers 5,500 ha, consisting of moderate to steep slopes. LWWMA is heavily forested and dominated by oak-hickory and cove hardwoods, with numerous gas wells and pipelines located throughout the area. Elevation ranges from 224 to 475 m. LWWMA has two block replicates of cut-back borders along gas pipelines. LWWMA was cut and treated between fall of 2017 and spring of 2018, with pre-treatment data collected in the summer of 2017 and post-treatment data collected in the summers of 2018 and 2019 (Table 1-2).

Little Canaan Wildlife Management Area.—Little Canaan WMA (LCWMA) is located in eastern West Virginia (Tucker County), near the town of Davis (Figure 1-12; site coordinates in Appendices 4-A through 4-F). LCWMA covers 1,282 ha and is primarily forested with gently sloping hills comprised mainly of maple-beech-birch forests interspersed with rhododendron (*Rhododendron maximum*) and mountain-laurel. Elevation ranges from 944 to 1,132 m. LCWMA has two block replicates of cut-back borders along two transmission powerlines (one block on each powerline) and four block replicates of regeneration stands. Research plots were cut and treated between fall 2016 and spring 2017, with pre-treatment being data collected in the summer of 2016 and post-treatment data collected in the summers of 2017 and 2018 (Table 1-2).

Moncove Lake Wildlife Management Area.—Moncove Lake WMA (MLWMA) is located in southern West Virginia (Monroe County), near the town of Union (Figure 1-13; site coordinates in Appendices 4-B and 4-D). MLWMA covers 313 ha, consisting of gently rolling hills to steep mountain sides with a predominately oak-hickory forest composition. Elevation ranges between 763 and 947 m. MLWMA has two block replicates of regeneration stands.

Research plots were cut and treated between fall 2017 and spring 2018. Pre-treatment data were collected in the summer of 2017 and post-treatment data were collected in the summers of 2018 and 2019 (Table 1-2).

Short Mountain Wildlife Management Area.—Short Mountain WMA (SMWMA) is located in the eastern panhandle of West Virginia (Hampshire and Hardy Counties), near the city of Romney (Figure 1-14; site coordinates in Appendices 4-B and 4-D). The management area covers 3,239 ha and consists of mixed oak and Virginia pine. Elevation ranges between 389 and 793 m. SMWMA has one block of regeneration stands. Research plots were cut and treated between fall of 2018 and spring 2019. Pre-treatment data were collected in the summer of 2018 and post-treatment data were collected in the summer of 2019 (Table 1-2).

Sleepy Creek Wildlife Management Area.—Sleepy Creek WMA (SCWMA) is located in the eastern panhandle of West Virginia (Berkeley and Morgan County), near the city of Martinsburg (Figure 1-15; site coordinates in Appendices 4-A through 4-D). SCWMA covers 9,278 ha and consists of oak-hickory and Virginia pine-oak forest. Elevation ranges between 304 and 660 m. SCWMA has one block replicate of cut-back borders along wildlife openings and one block replicate of regeneration stands. Research plots were cut and treated between fall 2018 and spring 2019. Pre-treatment data were collected in 2018 and post-treatment data were collected in 2019 (Table 1-2).

Stonewall Jackson Wildlife Management Area.—Stonewall Jackson WMA (SJWMA) is located in central West Virginia (Lewis County), near the city of Weston (Figure 1-16; site coordinates in Appendices 4-A, 4-C, and 4-E). The management areas covers 7,401 ha, consisting of farmlands reverting to woodland and rolling hills covered with mixed hardwood forest. Elevation ranges between 329 and 509 m. SJWMA has one block replicate of cut-back borders along wildlife openings. Research plots were cut and treated between fall 2016 and spring 2017. Pre-treatment data were collected in 2016 and post-treatment data collected in 2017 and 2018 (Table 1-2).

5. Summary and Dissertation Goals

The decline of young forest communities throughout the eastern USA is the result of the disruption of disturbance regimes, changes in anthropogenic land use, and the conversion of

forest ecosystems to non-forest habitat (e.g., urban, commercial, or agriculture). In West Virginia, total area and distribution of young forests has receded to concerning levels in recent decades and because of this, many wildlife species associated with young forests have experienced regional population declines. Low levels of disturbance have shifted forest tree composition as a result of forest maturation (Fei and Steiner 2007), invasive species introductions (Webster et al. 2018), and increases in natural regeneration barriers (e.g., ungulate browsing; Russell et al. 2017, Dey et al. 2018) which will have unknown implications on the wildlife community in the future. Unless attention is given to managing for young forests and understanding regeneration dynamics and their effects on the wildlife community, species associated with this vegetation stage will continue to experience regionally-depressed populations.

In the following chapters, I will look at the effects of various young forest management techniques on the avian (Appendix 1-A) and salamander communities. In Chapter 2, I studied the effects of cut-back borders along powerlines, pipelines, and wildlife openings on the avian and salamander community and developed best management approaches for implementing cut-back borders on WVDNR lands. In Chapter 3, I studied the effects of different young forest management regeneration stand techniques on the avian and salamander community and assessed the tradeoffs between maximizing species responses and minimizing treatment cost using a cost-benefit analysis. In Chapter 4, I focused on evaluating the importance of local- and landscape-level habitat variables on the songbird community to assist land managers in developing young forest management guidelines along pipelines, powerlines, and wildlife openings within forest landscapes.

The over-arching goal of this research was to better understand young forest management techniques and their effects on bird and salamander communities in order to develop management guidelines for implementation throughout West Virginia and the eastern USA. Conservation decisions on public lands regarding both game and non-game wildlife species need to maximize investments and withstand public scrutiny to ensure continued support of wildlife habitat programs. Results from this research provide guidelines and recommendations for land managers to efficiently implement young forest management practices with the goal of promoting young forests that provide vital developmental conditions and resources for healthy

and diverse plant and animal communities. Funding for the bird (game and non-game) portion of this project was provided through the Pittman-Robertson Wildlife Restoration Act Grant 48-R.

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7. Tables

Table 1-1. Species considered young or early-successional forest associates included in the West Virginia Division of Natural Resources' State Wildlife Action Plan (SWAP 2015). Species groups designed with "*" includes species associated with young/early-successional forest or grasslands.

Common Name	Scientific Name
Amphibians	
Fowler's Toad	<i>Anaxyrus fowleri</i>
Birds	
American Woodcock	<i>Scolopax minor</i>
Black-billed Cuckoo	<i>Coccyzus erythrophthalmus</i>
Blue-winged Warbler	<i>Vermivora cyanoptera</i>
Chuck-will's-widow	<i>Antrostomus carolinensis</i>
Eastern Whip-poor-will	<i>Antrostomus vociferus</i>
Field Sparrow	<i>Spizella pusilla</i>
Golden-winged Warbler	<i>Vermivora chrysoptera</i>
Loggerhead Shrike	<i>Lanius ludovicianus</i>
Long-eared Owl	<i>Asio otus</i>
Prairie Warbler	<i>Setophaga discolor</i>
Ruffed Grouse	<i>Bonasa umbellus</i>
Yellow-breasted Chat	<i>Icteria virens</i>
Mammals	
Eastern Harvest Mouse	<i>Reithrodontomys humulis</i>
Golden Mouse	<i>Ochrotomys nuttalli</i>
Least Shrew	<i>Cryptotis parva</i>
Meadow Jumping Mouse	<i>Zapus hudsonius</i>
Prairie Vole	<i>Microtus ochrogaster</i>
Southern Bog Lemming	<i>Synaptomys cooperi</i>
Virginia Big-eared Bat	<i>Corynorhinus townsendii virginianus</i>
Reptiles*	
Eastern Box Turtle	<i>Terrapene carolina carolina</i>
Eastern Earthsnake	<i>Virginia valeriae valeriae</i>
Eastern Kingsnake	<i>Lampropeltis getula</i>
Little Brown Skink	<i>Scinella lateralis</i>
Mountain Earthsnake	<i>Virginia valeriae pulchra</i>
Northern Black Racer	<i>Coluber constrictor constrictor</i>
Northern Pinesnake	<i>Pituophis melanoleucus melanoleucus</i>
Rough Greensnake	<i>Opheodrys aestivus</i>
Smooth Greensnake	<i>Liochlorophis vernalis</i>

Table 1-2. Data collection timetable for 11 cut-back border blocks and 16 regeneration stand blocks at 11 wildlife management areas.

	2016	2017	2018	2019
Cut-back Borders <i>Right-of-way</i> Gas Pipeline Burnsville Lake WMA (1 block) Transmission Powerline Little Canaan WMA (2 blocks) <i>Wildlife Opening</i> Stonewall Jackson WMA (1 block) Regeneration Stands Little Canaan WMA (4 blocks)	1 Year Pre-harvest	1 Year Post-harvest	2 Years Post-harvest	
Cut-back Borders <i>Right-of-way</i> Gas Pipeline Lewis Wetzel WMA (2 blocks) Transmission Powerline Beury Mountain WMA (2 blocks) <i>Wildlife Opening</i> Bluestone WMA (1 block) Regeneration Stands Beury Mountain WMA (5 blocks) Center Branch WMA (3 blocks) Moncove Lake WMA (2 blocks)		1 Year Pre-harvest	1 Year Post-harvest	2 Years Post-harvest
Cut-back Borders <i>Right-of-way</i> Gas Pipeline Allegheny WMA (1 block) <i>Wildlife Opening</i> Sleepy Creek WMA (1 block) Regeneration Stands Short Mountain WMA (1 block) Sleepy Creek WMA (1 block)			1 Year Pre-harvest	1 Year Post-harvest

8. Figures

Figure 1-1. Land classification of West Virginia (2011 National Land Cover Database).

Figure 1-2. Age class distribution of forests in West Virginia (Morin et al. 2016).

Figure 1-3. Total area of agriculture land in West Virginia (USDA 1940-2012).

Figure 1-4. Delineated area (red) of the Appalachian region of the eastern United States.

Figure 1-5. Level III ecoregions in West Virginia with study sites (circles) and major cities (stars) overlaid.

Figure 1-6. Allegheny Wildlife Management Area plot layout. There is one replication of cut-back borders along a transmission powerline and gas pipeline at Allegheny.

Figure 1-7. Beury Mountain Wildlife Management Area plot layout. There are two replications of cut-back borders along transmissions powerlines, and five replications of regeneration stands at Beury Mountain. Colors denote different block replicates.

Figure 1-8. Bluestone Wildlife Management Area plot layout. There is one replication of cut-back borders along wildlife openings at Bluestone.

Figure 1-9. Burnsville Lake Wildlife Management Area plot layout. There is one replication of cut-back borders along a gas pipeline at Burnsville.

Figure 1-10. Center Branch Wildlife Management Area plot layout. There are three replications of regeneration plots at Center Branch, with no hack-and-spray treatments. Colors denote separate block replicates.

Figure 1-11. Lewis Wetzel Wildlife Management Area plot layout, with two replications of cut-back borders along gas pipelines at Lewis Wetzel. Colors denote separate block replicates.

Figure 1-12. Little Canaan Wildlife Management Area plot layout. There are two cut-back border replications along transmission powerlines, and four replications of regeneration stands. Colors denote different block replicates.

Figure 1-13. Moncove Lake Wildlife Management Area plot layout. There are two replications of regeneration stands. Colors denote different block replicates.

Figure 1-14. Short Mountain Wildlife Management Area plot layout. There is one replication of regeneration stands.

Figure 1-15. Sleepy Creek Wildlife Management Area plot layout. There is one replication of cutback borders along wildlife openings, and one replication of regeneration stands.

9. Appendices

Appendix 1-A. Passerine species detected during study period (2016–2019) at study sites. Column labeled “Status” refers to breeding status: breeding - spends majority of summer but not fall, winter, or spring in WV, migrating - spends fall and winter in WV or passes through WV during the spring but does not spend the majority of the summer in WV, resident - spends summer, fall, winter, and spring in WV.

Common Name	Scientific Name	Alpha Code	Status	Habitat Guild	Nesting Guild	Foraging Guild
Acadian Flycatcher	<i>Empidonax virescens</i>	ACFL	Breeding	Forest Gap	Sub-canopy	Air
Alder Flycatcher	<i>Empidonax alnorum</i>	ALFL	Breeding	Young Forest	Shrub	Air
American Crow	<i>Corvus brachyrhynchos</i>	AMCR	Resident	Interior-edge	Sub-canopy	Predator
American Goldfinch	<i>Spinus tristis</i>	AMGO	Breeding	Young Forest	Sub-canopy	Foliage
American Redstart	<i>Setophaga ruticilla</i>	AMRE	Breeding	Interior-edge	Sub-canopy	Foliage
American Robin	<i>Turdus migratorius</i>	AMRO	Resident	Interior-edge	Sub-canopy	Foliage
Baltimore Oriole	<i>Icterus galbula</i>	BAOR	Breeding	Interior-edge	Sub-canopy	Foliage
Black-and-white Warbler	<i>Mniotilta varia</i>	BAWW	Breeding	Forest Interior	Ground	Bark
Black-billed Cuckoo	<i>Coccyzus erythrophthalmus</i>	BBCU	Breeding	Interior-edge	Shrub	Foliage
Black-capped Chickadee	<i>Poecile atricapillus</i>	BCCH	Resident	Interior-edge	Secondary Cavity	Bark
Black-throated Blue Warbler	<i>Setophaga caerulescens</i>	BTBW	Breeding	Forest Interior	Shrub	Foliage
Black-throated Green Warbler	<i>Setophaga virens</i>	BTNW	Breeding	Forest Gap	Sub-canopy	Foliage
Blackburnian Warbler	<i>Setophaga fusca</i>	BLBW	Migrating	Forest Interior	Canopy	Foliage
Blackpoll Warbler	<i>Setophaga striata</i>	BLPW	Migrating	Interior-edge	Sub-canopy	Foliage
Blue-gray Gnatcatcher	<i>Poliophtila caerulea</i>	BGGN	Breeding	Interior-edge	Sub-canopy	Foliage
Blue-headed Vireo	<i>Vireo solitarius</i>	BHVI	Breeding	Forest Interior	Sub-canopy	Foliage
Blue-winged Warbler	<i>Vermivora cyanoptera</i>	BWWA	Breeding	Young Forest	Shrub	Foliage
Blue Jay	<i>Cyanocitta cristata</i>	BLJA	Resident	Interior-edge	Canopy	Predator
Brown-headed Cowbird	<i>Molothrus ater</i>	BHCO	Resident	Interior-edge	Parasite	Ground
Brown Creeper	<i>Certhia americana</i>	BRCR	Resident	Forest Interior	Secondary Cavity	Bark
Brown Thrasher	<i>Toxostoma rufum</i>	BRTH	Breeding	Young Forest	Shrub	Ground
Cape May Warbler	<i>Setophaga tigrina</i>	CMWA	Migrating	Interior-edge	Sub-canopy	Foliage
Canada Warbler	<i>Cardellina canadensis</i>	CAWA	Breeding	Interior-edge	Shrub	Foliage
Carolina Chickadee	<i>Poecile carolinensis</i>	CACH	Resident	Interior-edge	Secondary Cavity	Bark
Carolina Wren	<i>Thryothorus ludovicianus</i>	CARW	Resident	Young Forest	Secondary Cavity	Ground
Cedar Waxwing	<i>Bombycilla cedrorum</i>	CEDW	Resident	Interior-edge	Sub-canopy	Foliage
Cerulean Warbler	<i>Setophaga cerulea</i>	CERW	Breeding	Forest Gap	Canopy	Foliage
Chestnut-sided Warbler	<i>Setophaga pensylvanica</i>	CSWA	Breeding	Young Forest	Shrub	Foliage

Appendix 1-A continued.

Common Name	Scientific Name	Alpha Code	Status	Habitat Guild	Nesting Guild	Foraging Guild
Chimney Swift	<i>Chaetura pelagica</i>	CHSW	Breeding	Not applicable	Secondary Cavity	Air
Chipping Sparrow	<i>Spizella passerina</i>	CHSP	Breeding	Young Forest	Shrub	Ground
Common Raven	<i>Corvus corax</i>	CORA	Resident	Interior-edge	Canopy	Predator
Common Yellowthroat	<i>Geothlypis trichas</i>	COYE	Breeding	Young Forest	Ground	Foliage
Dark-eyed Junco	<i>Junco hyemalis</i>	DEJU	Resident	Interior-edge	Ground	Ground
Downy Woodpecker	<i>Dryobates pubescens</i>	DOWO	Resident	Interior-edge	Primary Cavity	Bark
Eastern Bluebird	<i>Sialia sialis</i>	EABL	Resident	Grassland	Secondary Cavity	Ground
Eastern Kingbird	<i>Tyrannus tyrannus</i>	EAKI	Breeding	Young Forest	Canopy	Air
Eastern Meadowlark	<i>Sturnella magna</i>	EAME	Resident	Grassland	Ground	Ground
Eastern Phoebe	<i>Sayornis phoebe</i>	EAPH	Breeding	Interior-edge	Sub-canopy	Air
Eastern Towhee	<i>Pipilo erythrophthalmus</i>	EATO	Breeding	Young Forest	Shrub	Ground
Eastern Wood-pewee	<i>Contopus virens</i>	EAWP	Breeding	Interior-edge	Canopy	Air
Field Sparrow	<i>Spizella pusilla</i>	FISP	Resident	Young Forest	Shrub	Ground
Grasshopper Sparrow	<i>Ammodramus savannarum</i>	GRSP	Breeding	Grassland	Ground	Ground
Gray Catbird	<i>Dumetella carolinensis</i>	GRCA	Breeding	Young Forest	Shrub	Ground
Great Crested Flycatcher	<i>Myiarchus crinitus</i>	GCFL	Breeding	Interior-edge	Secondary Cavity	Air
Hairy Woodpecker	<i>Dryobates villosus</i>	HAWO	Resident	Forest Interior	Primary Cavity	Bark
Hermit Thrush	<i>Catharus guttatus</i>	HETH	Resident	Interior-edge	Ground	Ground
Hooded Warbler	<i>Setophaga citrina</i>	HOWA	Breeding	Forest Gap	Shrub	Foliage
House Finch	<i>Haemorhous mexicanus</i>	HOFI	Resident	Interior-edge	Secondary Cavity	Ground
House Wren	<i>Troglodytes aedon</i>	HOWR	Breeding	Interior-edge	Secondary Cavity	Foliage
Indigo Bunting	<i>Passerina cyanea</i>	INBU	Breeding	Young Forest	Shrub	Foliage
Kentucky Warbler	<i>Geothlypis formosa</i>	KEWA	Breeding	Forest Gap	Ground	Ground
Least Flycatcher	<i>Empidonax minimus</i>	LEFL	Breeding	Forest Interior	Sub-canopy	Air
Louisiana Waterthrush	<i>Parkesia motacilla</i>	LOWA	Breeding	Forest Interior	Ground	Ground
Magnolia Warbler	<i>Setophaga magnolia</i>	MAWA	Breeding	Forest Interior	Sub-canopy	Foliage
Mourning Dove	<i>Zenaida macroura</i>	MODO	Resident	Synanthropic	Shrub	Ground
Mourning Warbler	<i>Geothlypis philadelphia</i>	MOWA	Breeding	Young Forest	Shrub	Foliage
Nashville Warbler	<i>Oreothlypis ruficapilla</i>	NAWA	Migrating	Interior-edge	Ground	Foliage
Northern Cardinal	<i>Cardinalis cardinalis</i>	NOCA	Resident	Interior-edge	Shrub	Ground
Northern Flicker	<i>Colaptes auratus</i>	NOFL	Resident	Interior-edge	Primary Cavity	Ground
Northern Mockingbird	<i>Mimus polyglottos</i>	NOMO	Resident	Synanthropic	Shrub	Ground
Northern Parula	<i>Setophaga americana</i>	NOPA	Breeding	Interior-edge	Canopy	Foliage
Orchard Oriole	<i>Icterus spurius</i>	OROR	Breeding	Young Forest	Sub-canopy	Foliage
Ovenbird	<i>Seiurus aurocapilla</i>	OVEN	Breeding	Forest Interior	Ground	Ground
Philadelphia Vireo	<i>Vireo philadelphicus</i>	PHVI	Migrating	Interior-edge	Canopy	Foliage
Pileated Woodpecker	<i>Dryocopus pileatus</i>	PIWO	Resident	Forest Interior	Primary Cavity	Bark

Appendix 1-A continued.

Common Name	Scientific Name	Alpha Code	Status	Habitat Guild	Nesting Guild	Foraging Guild
Pine Warbler	<i>Setophaga pinus</i>	PIWA	Breeding	Forest Interior	Shrub	Foliage
Prairie Warbler	<i>Setophaga discolor</i>	PRAW	Breeding	Young Forest	Shrub	Foliage
Red-bellied Woodpecker	<i>Melanerpes carolinus</i>	RBWO	Resident	Interior-edge	Primary Cavity	Bark
Red-breasted Nuthatch	<i>Sitta canadensis</i>	RBNU	Resident	Forest Interior	Secondary Cavity	Bark
Red-eyed Vireo	<i>Vireo olivaceus</i>	REVI	Breeding	Forest Gap	Sub-canopy	Foliage
Red-winged Blackbird	<i>Agelaius phoeniceus</i>	RWBL	Resident	Grassland	Shrub	Foliage
Rose-breasted Grosbeak	<i>Pheucticus ludovicianus</i>	RBGR	Breeding	Interior-edge	Sub-canopy	Foliage
Ruby-crowned Kinglet	<i>Regulus calendula</i>	RCKI	Migrating	Forest Interior	Sub-canopy	Foliage
Ruby-throated Hummingbird	<i>Archilochus colubris</i>	RTHU	Breeding	Interior-edge	Sub-canopy	Air
Ruffed Grouse	<i>Bonasa umbellus</i>	RUGR	Resident	Young Forest	Ground	Ground
Scarlet Tanager	<i>Piranga olivacea</i>	SCTA	Breeding	Forest Interior	Canopy	Foliage
Song Sparrow	<i>Melospiza melodia</i>	SOSP	Resident	Synanthropic	Shrub	Foliage
Swainson's Thrush	<i>Catharus ustulatus</i>	SWTH	Breeding	Forest Interior	Shrub	Ground
Swainson's Warbler	<i>Limnithlypis swainsonii</i>	SWWA	Breeding	Forest Gap	Shrub	Ground
Tennessee Warbler	<i>Oreothlypis peregrina</i>	TEWA	Migrating	Young Forest	Ground	Foliage
Tree Swallow	<i>Tachycineta bicolor</i>	TRES	Breeding	Not applicable	Secondary Cavity	Air
Tufted Titmouse	<i>Baeolophus bicolor</i>	TUTI	Resident	Forest Interior	Secondary Cavity	Foliage
Veery	<i>Catharus fuscescens</i>	VEER	Breeding	Interior-edge	Ground	Ground
Warbling Vireo	<i>Vireo gilvus</i>	WAVI	Breeding	Interior-edge	Sub-canopy	Foliage
White-breasted Nuthatch	<i>Sitta carolinensis</i>	WBNU	Resident	Forest Gap	Secondary Cavity	Bark
White-eyed Vireo	<i>Vireo griseus</i>	WEVI	Breeding	Young Forest	Shrub	Foliage
Willow Flycatcher	<i>Empidonax traillii</i>	WIFL	Breeding	Young Forest	Shrub	Air
Wilson's Warbler	<i>Cardellina pusilla</i>	WIWA	Migrating	Young Forest	Ground	Foliage
Winter Wren	<i>Troglodytes hiemalis</i>	WIWR	Resident	Forest Interior	Secondary Cavity	Ground
Wood Thrush	<i>Hylocichla mustelina</i>	WOTH	Breeding	Forest Interior	Sub-canopy	Ground
Worm-eating Warbler	<i>Helmitheros vermivorum</i>	WEWA	Breeding	Forest Interior	Ground	Foliage
Yellow-bellied Sapsucker	<i>Sphyrapicus varius</i>	YBSA	Resident	Interior-edge	Primary Cavity	Bark
Yellow-billed Cuckoo	<i>Coccyzus americanus</i>	YBCU	Breeding	Interior-edge	Sub-canopy	Foliage
Yellow-breasted Chat	<i>Icteria virens</i>	YBCH	Breeding	Young Forest	Shrub	Foliage
Yellow-throated Vireo	<i>Vireo flavifrons</i>	YTVI	Breeding	Interior-edge	Canopy	Foliage
Yellow-throated Warbler	<i>Setophaga dominica</i>	YTWA	Breeding	Forest Interior	Canopy	Foliage
Yellow Warbler	<i>Setophaga petechia</i>	YEWA	Breeding	Young Forest	Shrub	Foliage

CHAPTER 2

SHORT-TERM AVIAN AND SALAMANDER RESPONSE TO CUT-BACK BORDERS

Abstract

Throughout the eastern United States, young forest wildlife species continue to experience declining populations from habitat loss stemming from altered disturbance regimes and habitat conversion. However, the recent proliferation of linear energy infrastructure throughout the central Appalachian region has prompted managers to explore ways of managing the young forest bird community in association with these long, linear openings but little empirical data exist. At the same time, forest interior wildlife (e.g., songbirds and woodland salamanders) are often negatively affected by energy infrastructure within forest dominated landscapes and any young forest management in these landscapes may further degrade habitats for forest interior species. In this chapter we studied how harvest size (15 m, 30 m, and 45 m wide) and intensity (14 m²/ha and 4.5 m²/ha residual basal area) of cut-back borders, which are linear tree cuttings adjacent to gas/oil pipeline and utility powerline rights-of-way (ROWs) or wildlife openings, influenced habitat suitability along ROWs and wildlife openings for the young forest and forest interior communities. The objectives of this chapter were to examine whether the presence of cut-back borders increased habitat suitability for wildlife species and which cut-back border treatments optimized the tradeoff between maximizing positive responses of disturbance-dependent species (i.e., young forest species) and minimizing negative responses of disturbance-avoidant species (i.e., forest interior species). We found that young forest species' abundances and species richness increased one-year and two-years after treatment, particularly in the 15-m wide borders, likely due to the increase of young forest habitat in conjunction with existing habitat in ROW corridors or wildlife openings. Additionally, we found that forest interior species, a group that is often negatively affected by ROWs in forested landscapes, did not decrease in cut-back borders following treatment likely because the retention of canopy trees in combination with adjacent mature forests retained suitable habitat conditions. These results suggest that cut-back borders along abrupt forest edges of ROWs and wildlife openings create suitable habitat conditions for young forest avifauna without negatively affecting forest interior

avifauna. Cut-back borders, particularly those 15-m in width, appear to be a viable management option for managing forest bird communities in ROW and wildlife opening landscapes.

1. Introduction

1.1. Introduction

The central Appalachian region, United States (US) is predominantly comprised of mature (>60 years old), large diameter forestlands (Oswalt et al. 2010, Morin et al. 2016). Current forest conditions are the result of socio-economically driven shifts in disturbance regimes (e.g., fire suppression, reduction of large-scale timber management) over the past century (Lorimer 2001). The decreased prevalence of forest disturbance has resulted in conditions unsuitable for wildlife species that are specialized in exploiting the seral stage immediately following disturbance (i.e., young forest). Further, habitat loss from human influences (e.g., agriculture, urbanization) has exacerbated pressures of changing disturbance regimes on young forest species. Declining total area and conversion of young forests are among the primary causes of young forest wildlife population declines (Dettmers 2003, King and Schlossberg 2014). These factors have limited the accessibility to suitable habitat while reducing the probability of movements between habitat patches leading to greater likelihood of local population loss (Wiens 1997, Villard et al. 1999). Moreover, current available habitat for young forest species is ephemeral, subject to natural forest succession and often existing <15 years thereby requiring periodic maintenance for populations to locally persist (Schlossberg and King 2009, Twedt and Somershoe 2009). These culminating factors have led to the decline of populations of young forest wildlife species in the central Appalachian region and have prompted natural resource agencies to develop solutions that increase young forest habitat in order to conserve priority species in the region (Pardieck et al. 2019).

The central Appalachian region, along with much of the eastern US, contains extensive linear infrastructure (e.g., pipelines, powerlines, roads) that is expanding further due to increased human influences. Electricity powerlines connecting rural communities into the power grid involves a complex network of overhead transmission (long distance lines) and distribution (short distance lines) powerlines traveling from power plant to consumer. Of the 83 power plants (≥ 1 -megawatt capacity) currently in operation in West Virginia, 14 have been constructed within the past 12 years with nine more planned for construction in 2021 and 2022 (EIA 2020). The continued increase in energy demand in West Virginia and adjacent states will certainly add to

the 9975 kilometers of transmission powerlines (includes high voltage lines 69–765 kV only) currently in West Virginia as of 2019 (HIFLD 2019). Further, since 2010, the central Appalachian region has experienced a steady increase in natural gas extraction due to the advancement of horizontal drilling techniques and high-volume hydraulic fracturing leading to increased access to the Marcellus-Utica shale basin (EIA 2020). During this period, natural gas production in West Virginia has increased 27% annually and West Virginia is currently the seventh-largest producer of marketed natural gas in the country (EIA 2020). To accommodate the supply of natural gas, underground gas pipelines have become a prominent landscape feature throughout West Virginia, totaling 24621 kilometers of inter- and intrastate pipeline (includes field and gathering, transmission, and distribution pipelines) as of 2018 (AGA 2019).

A consequence of rights-of-way (ROWs; e.g., gas pipelines and utility powerlines) corridors throughout West Virginia, which is ~79% forested, is the juxtaposing of land cover types of forest and non-forest (e.g., grassland or shrubland due to mowing or herbicide spraying in ROWs) which results in extensive stretches of abrupt forest edge (Harris 2020). Abrupt forest edges can pose serious threats to wildlife conservation due to novel species interactions (Chasko and Gates 1982, Murcia 1995). Songbirds experience increased rates of brood parasitism by brown-headed cowbirds (*Molothrus ater*) and increased rates of nest predation by mammalian mesocarnivores and corvids (Corvidae) at forest edges relative to interiors (Brittingham and Temple 1983, Heske et al. 1999, Shake et al. 2011). Mature forest interior species generally avoid ROWs because abrupt forest edges reduce habitat suitability in these areas (deMaynadier and Hunter Jr. 1998, Farwell et al. 2016). Abundances of forest interior songbirds decline near forest edges due their inability to compete with habitat generalist species that are better adapted to areas with greater habitat interspersion. Woodland salamanders avoid abrupt forest edges because of their inability to physiologically cope with greater microclimate variability (Hofmeister et al. 2019). Presence of ROW corridors within forested landscapes can benefit early-successional bird species though, because they are maintained in a grassland or shrubland successional community (Bulluck and Buehler 2006, Askins et al. 2012). However, vegetation in ROW corridors does not offer the full complement of vegetation features for young forest species, which includes greater amounts of understory and midstory woody stems and greater vertical vegetation structure (Jones et al. 2008, Culbert et al. 2013). These young forest vegetation features are never fully attained within ROW corridors because of periodic mowing or

herbicide spraying. Therefore, despite the abundance of grassland and shrubland habitat available in West Virginia via ROWs, little of this habitat is considered optimal for young forest species, a group considered in need of immediate conservation action.

The abundance of ROW corridors in West Virginia and the central Appalachian region has led land managers and natural resource scientists to explore ways that incorporate these features into their management plans. A potential management strategy to improve habitat quality of ROW corridors for young forest wildlife is the reduction of abrupt forest edges along ROWs that mimic ecotone-like characteristics (Fleming and Giuliano 1998, Conover et al. 2009). Additionally, the transition of forest edges from abrupt edge towards ecotone has potentially beneficial consequences for forest interior species (DeGraaf and Yamasaki 2002, Yahner et al. 2003). Cut-back borders (also referred to as edge-feathering, conservation buffers, or field border-cuts) are a management technique that creates intermediate vegetation structure between two distinctly different land cover types (e.g., between grassland and mature forest) to increase habitat diversity and soften abrupt forest edge. Benefits for avifauna from cut-back borders includes increased abundance and density of management priority game and non-game species (Wunz 1990, Burger et al. 2013, Evans et al. 2015), increased nest success (Slay and Smith 2009, Foggia et al. 2018), and increased species richness (Yahner et al. 2003). However, past studies have been limited to agricultural-dominated landscapes (e.g., cattle and row crop farms) or where small woodlots are the only available forest habitat (Morgan and Gates 1982, Fleming and Giuliano 2001). Information pertaining to cut-back border effects on wildlife within forest-dominated landscapes is limited but desired, with an emphasis on cut-back border size and tree harvest intensity prescriptions to develop management guidelines. Considering the increased prevalence of forest edges due to the presence of wildlife openings and the proliferation of gas pipelines and utility powerlines in West Virginia and the central Appalachian region, cut-back borders are understudied and thus a potentially underutilized wildlife habitat management technique in these human-modified landscapes. Additionally, cut-back borders may be a promising management strategy for creating young forest habitat for priority young forest avifauna to help reverse declining regional population trends, but their effectiveness remains ambiguous (Pardieck et al. 2019). Therefore, rigorous experimental studies delineating size and harvest intensity of cut-back borders within forest-dominated landscapes and elucidating wildlife responses are needed to develop cut-back border guidelines for the region.

1.2. *Chapter objectives and hypotheses*

The primary objective of this chapter was to assess avian and woodland salamander responses to creation of young forest habitat along forest edges via cut-back borders. We used a before-after-control-impact (BACI) framework, which allows for causal inference, to elucidate individual species and community responses to specific cut-back border treatments over time. We examined combinations of three cut-back border size manipulations and two harvest intensity manipulations over three years (with the initial year having no manipulation) to inform our inferences about avian and salamander response to cut-back borders. For avifauna, we hypothesized that (1) young forest avifauna will respond positively to cut-back borders along a gradient from greatest increases in the larger-sized, heavier harvest intensity treatments to least increases in the smaller-sized, less intensive harvest treatments and (2) mature forest avifauna will respond negatively to cut-back borders along a gradient from greatest decreases in the larger-sized, heavier harvest intensity treatments to least decreases in the smaller-sized, less intensive harvest treatments. Considering salamander responses to cut-back borders, we hypothesized that salamanders will respond negatively to cut-back borders along a gradient from greatest decreases in the larger-sized, heavier harvest intensity treatments to least decreases in the smaller-sized, less intensive harvest treatments. Additionally, we examined which cut-back border treatments optimized the tradeoff between maximizing positive responses of disturbance-dependent species and minimizing negative responses of disturbance-avoidant species. Finally, we assessed the response of vegetation characteristics to help infer avian and salamander responses in cut-back border treatments but also to inform decisions regarding cut-back border placement with respect to abiotic factors (e.g., slope aspect, edge orientation). Results from this study provide species- and community-level guidelines for cut-back border harvest sizes and intensities to assist managers with habitat management goals along forest edges in West Virginia and the central Appalachian region.

2. **Methods**

2.1. *Study area*

West Virginia lies within the central Appalachian region, which is characterized by extensive forests, comprised of mixed mesophytic, Appalachian oak, transition Appalachian oak-northern hardwood, and oak-hickory-pine forest types (Küchler 1964). Elevations at study sites range from 224 m in the northwestern part of the state (Lewis Wetzel Wildlife Management Area

[LWWMA]) to 1132 m in the eastern part of the state (Little Canaan WMA [LCWMA]). West Virginia's humid continental climate is characterized by warm summers and cool to cold winters with average spring and fall temperatures of 10.0 °C and an average summer temperature of 22.0 °C. Annual precipitation levels are 76, 127, and 102 cm in the western, central, and eastern portions of the state, respectively, distributed relatively uniformly throughout the year but highest in spring and summer (NOAA 2019).

This study was conducted on eight West Virginia Division of Natural Resources' (WVDNR) wildlife management areas (WMAs) located throughout central and eastern West Virginia (Figure 2-1; see Figures 1-6, 1-7, 1-8, 1-9, 1-11, 1-12, 1-15, and 1-16 for site-level maps). Study sites were selected based on presence of overhead utility powerlines, underground gas pipelines, or wildlife openings, their accessibility for managers, and proximity to other study sites. Powerlines and pipelines used in this study were narrow linear openings (9.4–39.2 m [(mean) $25.7 \pm$ (SD) 17.9 m] in width) with varying purposes (i.e., gathering pipelines and short- and long-distance powerlines). Vegetation in powerlines and pipelines consisted of primarily grassland or occasionally low shrubland vegetation, was void of a mid- or over-story vegetation layer, and was distinct from adjacent forests (see Figure 1-2 for examples). Pipelines and powerlines are hereafter referred to as “ROWS”. Wildlife openings used in this study were fields planted with a row crop (e.g., corn, clover) or left fallow, ranged 0.3–11.5 ha (3.1 ± 3.1 ha) in size, and were of varying shapes.

2.2. *Study design*

The eight study sites included 11 block replicates (Table 2-1, Appendix 2-A) located along underground gas pipelines ($n = 4$), overhead utility powerlines ($n = 4$), or wildlife openings ($n = 3$). Block replicates were presumed to be subjected to similar biotic and abiotic environmental constraints; block replicates along ROWs (pipelines and powerlines) were located along one or two ROWs of similar widths within the same WMA and block replicates along wildlife openings were within the same WMA. A single block replicate consisted of seven independent experimental units (hereafter referred to as “cut-back borders”) with each assigned one of seven experimental treatments. Treatment designation for each cut-back border consisted of randomly selecting one of two target residual basal area levels (4.5 m²/ha or 14.0 m²/ha basal area retention) and one of three cutting depths into the forest (15 m [0.5 ha total area], 30 m [0.9 ha], or 45 m [1.4 ha] perpendicular to forest edge) with one cut-back border designated as a

control (i.e., no tree cutting). Each cut-back border was 300 m in linear distance along the forest edge (Figure 2-3) and they were ≥ 200 m apart to ensure sampling independence for birds and salamanders.

Target basal area retention in each cut-back border was obtained via tree cutting using a feller-buncher or hand crew to mimic WVDNR forest management prescriptions for clearcuts with reserves (4.5 m²/ha) or crop tree/shelterwood stands (14.0 m²/ha) forest management prescription. WVDNR managers/foresters cruised all cut-back borders and selected residual trees based on species desirability (i.e., species of high wildlife value, such as oaks [*Quercus* spp.] and hickories [*Carya* spp.]). All trees ≥ 5 cm DBH were felled, excluding selected residual trees. Felled trees were dropped within the cut-back border and remained on the ground. Non-native shrub species (e.g., autumn olive [*Elaeagnus umbellata*], Tatarian honeysuckle [*Lonicera tatarica*], and multiflora rose [*Rosa multiflora*]) were sprayed with herbicide (15% Garlon[®] 4 Ultra in an oil carrier) to reduce presence in cut-back borders. Plant stems were sprayed at 30–38 cm above the ground to the point of runoff to ensure the root crown was treated. Herbicide was applied in the fall after pre-treatment data collection but prior to tree cutting. Tree cutting began 1 September or 15 November (if the study site was < 5 miles from an Indiana bat [*Myotis sodalis*] roosting area) and was completed by 31 March.

We used a before-after-control-impact (BACI) framework to infer causal relationships between cut-back border treatments and the response of avian and salamander communities through time (Eberhardt 1976). At each cut-back border, we collected one-year of pre-treatment data and one- or two-years of post-treatment data on the vegetation, avian, and salamander communities (Table 2-1). Vegetation, avian, and salamander sampling points were located 15 m from the edge into the forest and were consistent across all cut-back border widths and harvest intensities (Figure 2-3). In each cut-back border, two avian sampling points were spaced 150 m apart and 75 m from cut-back border ends. Vegetation sampling plots were placed at the same locations as avian sampling points. In each cut-back border, four salamander sampling plots were spaced 75 m apart and 37.5 m from cut-back border ends.

2.3. *Vegetation assessment*

We measured vegetation characteristics within cut-back borders (Figure 2-3) each year using a nested circular plot design (Fettingner 2002; Figure 2-4; see Appendix 2-B for a list of all variables collected). This approach partitions overstory, midstory, and understory vegetation

variables into three nested circles: largest sampling plot measuring overstory, intermediate sampling plot measuring midstory, and smallest sampling plot measuring understory. This vegetation protocol followed previous young forest wildlife studies in the central Appalachian region (Fettingner 2002, Schumacher 2002).

The largest circular plot was 11.3 m radius (0.04 ha; Figure 2-4). Within this plot, we measured diameter-at-breast height (DBH) of all trees ≥ 11.4 cm DBH to the nearest 0.1 cm using DBH tapes and recorded status (“live” or “dead”). Live trees were subsequently placed into two diameter classes (≥ 11.4 – < 28.0 cm DBH, and ≥ 28.0 cm DBH; USDA Forest Service 2018) for analyses. Trees that received a “dead” status (i.e., snags) were enumerated but not placed into separate diameter classes. Individual live tree DBH data were converted to tree basal area (m^2/ha), then summed for each 11.3 m circular plot (hereafter “basal area”). We estimated the amount of live foliage understory cover within two vertical zones (0–1.0 m and > 1.0 –2.0 m; hereafter “understory cover 0–1 m” and “understory cover > 1 –2 m”, respectively) using a modified Nudds (1977) profile board. The profile board was placed 15-m from plot center in the four cardinal directions. Observers kneeled at the plot center, to reduce variation due to differing observer heights, and estimated understory cover into one of five categories (0–20%, > 20 –40%, > 40 –60%, > 60 –80%, or > 80 –100%) for each of the two zones. Standing at the plot center, we measured the percentage of live overstory cover (delineated at ≥ 5 m above ground) using a 25 cm x 25 cm transparent plexiglass panel, divided into a 5 x 5 grid, held overhead (Figure 2-5; Hachè et al. 2013; hereafter “overstory cover”).

Nested within the 11.3 m radius circular plot was a smaller 5.6 m radius plot (0.01 ha; Figure 2-4). Within this plot, we enumerated woody saplings and primary shrub stems (i.e., main shoot growth extending from the ground) that were < 11.4 cm DBH and ≥ 1.4 m in height into one of two diameter classes (0–5.1 cm DBH and > 5.1 – < 11.4 cm DBH; hereafter “midstory stems 0–5.1 cm DBH” and “midstory stems > 5.1 –11.3 cm DBH”, respectively). Nested within the 5.6 m radius circular plot was a smaller 3.6 m radius plot (0.004 ha; Figure 2-4). Within this plot, we estimated the percentage of live vegetation ground cover ≤ 0.25 m in height into one of five categories (0–5%, > 5 –25%, > 25 –50%, > 50 –75%, or > 75 –100%; hereafter “ground cover”) using ocular assessment.

We measured canopy height and topographic variables using 3-m scale digital elevation models (DEMs) and 0.7 m nominal pulse spacing light detection and ranging (LIDAR) data

sourced from the Federal Emergency Management Agency (FEMA), U.S. Geological Survey (USGS), and West Virginia Statewide Addressing and Mapping Board (SAMB; WVGIS 2019). Variables were calculated using zonal statistics in the program ArcMAP (version 10.0; ESRI 2014) to generate a mean value within a 50-m radius surrounding each vegetation sampling point (Figure 2-3). Topographic variables included slope percent, slope aspect, slope position (Weiss 2001), slope configuration, and elevation and were derived using DEMs. Slope percent, slope aspect, slope position, and slope configuration were combined using ModelBuilder in ArcGIS (ESRI 2014) to derive a topographic relative moisture index (TRMI) ranging from 0 (xeric soil) to 60 (mesic soil; Parker 1982). Additionally, we measured heat load index (HLI; McCune and Keon 2002) and Beers aspect (Beers et al. 1966) using DEMs. Both HLI and Beers aspect index soil moisture based on topographic position, similar to TRMI. Canopy height was derived using LIDAR data or ground sampling using a clinometer when LIDAR data were not available. When calculating canopy height using LIDAR, we moved the sampling point 50-m away from ROWs/wildlife openings into the forest interior to avoid bias from the 50-m radius used to calculate zonal statistics extending into the ROW/wildlife opening where there are no trees. We assumed canopy height within cut-back borders was similar to canopy height in the forest interior. Additionally, we measured ROW (e.g., powerlines and pipelines) corridor width and wildlife opening width (defined as the narrowest location within the wildlife opening) using aerial imagery.

We developed a shade proportional index (SPI) modified from Buffo et al. (1972) to calculate the proportion of shade from the adjacent mature forest overstory covering each cut-back border plot based on canopy height, slope percent, and edge orientation (i.e., compass direction of forest edge).

$$shadow.length = canopy.ht \times (\sin(28)/\sin(180 - (90 - slope) - 28)) \quad (1)$$

$$true.shadow = shadow.length \times (\sin(90 - edge.orient)/\sin(90)) \quad (2)$$

$$shade.proportion.index = true.shadow/border.width \quad (3)$$

In Equation 1, the value 28 is derived from 90 (degrees altitude a tree grows at) minus 62 (mean solar altitude from dates: 30 April [first frost-free date], 20 June [peak altitude in northern hemisphere], and 10 October [first frost killing date; WVU Extension 2019], which constitutes the beginning, middle, and end of the growing season). The variable *slope* was the slope percent,

in degrees, at the cut-back border plot and *shadow.length* was output in the same units as canopy height. *Shadow.length* was then input into Equation 2 with *edge.orient* (i.e., edge orientation) to generate *true.shadow*, the length of the shadow while accounting for edge orientation. Edge orientation of each cut-back border was obtained from 2018 National Agriculture Imagery Program (NAIP) aerial imagery in ArcGIS (ESRI 2014). Plots with edge orientations between 90° and 270° (east→south→west) were automatically given a *true.shadow* value of zero because these plots receive full direct sunlight. *Edge.orient* values for plots whose edge orientation was between 0–89° were left unchanged, but values between 271–359° were taken from 360 to calculate degrees from north (e.g., 360° minus 301° [plot edge orientation] equals 59° from north). The proportion of the cut-back border plot that was subject to shading (*shade.proportion.index* [SPI]) was calculated by dividing *true.shadow* by the width of the cut-back border (*border.width*; Equation 3).

2.4. Avian sampling

Ruffed grouse.—We assessed ruffed grouse response to cut-back borders during 5-minute drumming surveys (McBurney 1989) at the two point count locations in each cut-back border (Figure 2-3). Drumming surveys were conducted twice at each point count location between 12 April and 17 May during 2016–2019 (Table 2-2) during 0530–1000 (EST) on days that were considered optimal for sampling (e.g., no rain, no heavy or gusty winds, and minimal peripheral noise). Observers recorded presence or absence of drumming grouse during each survey and if present, placed detected individuals into one of five distance intervals (0–25 m, >25–50 m, >50–75 m, >75–100 m, or >100 m). Observers also recorded any opportunistic detections associated with drumming surveys that occurred while within cut-back border plots (e.g., flushing grouse while walking to point count location). Survey-level data collected during each sampling visit included time of survey and ordinal day. Prior to surveys, observers were trained in survey protocol. Surveyor skill was assessed based on their previous experience conducting point counts in the region and identification skills of avian species of the region and ranged from 1–5, where a score of 5 indicated thorough knowledge of species with extensive experience conducting point counts, a score of 1 indicated no knowledge of species with no previous experience conducting point counts, and a score of 3 indicated introductory knowledge of species with minimal experience conducting point counts (most technicians that lacked previous point count experience were scored as 3 following training). We included all opportunistic detections and

drumming detections within 100 m for analysis. In 2016, four observers conducted all drumming surveys, 11 observers (nine new, two from the previous year) conducted all drumming surveys in 2017, 11 observers (four new, seven from the previous year) conducted all drumming surveys in 2018, and ten observers (three new, seven from the previous year) conducted all drumming surveys in 2019.

American woodcock.—We assessed American woodcock response to cut-back borders during one, 2-minute singing-ground survey at the two point count locations in each cut-back border (Figure 2-3). Singing-ground surveys were conducted between 15 April and 5 May during 2016–2019 (Table 2-2; USFWS 2011) on days that were considered optimal for sampling (e.g., no rain, no heavy or gusty winds, and minimal peripheral noise). Survey start times were based on sky conditions. If the sky was <75% overcast, surveys began 22 minutes after sunset and lasted for 38 minutes. If the sky was ≥75% overcast, surveys began 15 minutes after sunset and lasted for 38 minutes (USFWS 2011). Observers recorded presence or absence of “peenting” woodcock during each survey and if present, placed detected individual into one of five distance intervals (0–25 m, >25–50 m, >50–75 m, >75–100 m, or >100 m). Observers also recorded the total number of “peents” heard from each individual detected. Prior to surveys, observers were trained in survey protocol. In 2016, four observers conducted all singing-ground surveys, 13 observers (11 new, two from previous year) conducted all surveys in 2017, 12 observers (two new, ten from previous year) conducted all surveys in 2018, and 12 observers (four new, eight from previous year) conducted all surveys in 2019.

Songbird.—We assessed songbird response to cut-back borders during the spring migration period and summer breeding period using 10-minute standardized fixed-radius point counts (Ralph et al. 1993, Bibby et al. 2002). At the two point count locations within each cut-back border (Figure 2-3), we conducted a single visit during the spring migration period (1–22 May) and two visits during the summer breeding period (18 May–27 June) during 2016–2019 (Table 2-2). Surveys began after 0530 and before 1000 (EST) on days that were considered optimal for sampling (e.g., no rain, no heavy or gusty winds, and minimal peripheral noise). Observers recorded species of each individual bird detected, type of detection (song, call, visual, or flyover), demographic information (male, female, juvenile, or unknown), distance from observer placed into one of five distance intervals (0–25 m, >25–50 m, >50–75 m, >75–100 m,

or >100 m), and time interval the individual was first detected (0–2 min, >2–4 min, >4–6 min, >6–8 min, and >8–10 min). Survey-level data were collected during each visit and included time of survey and ordinal day. Prior to surveys, observers were trained to independently identify all species visually and aurally while simultaneously recording distance and time interval. In 2016 four observers conducted all counts, seven observers (five new, two from the previous year) conducted all counts in 2017, nine observers (four new, five from the previous year) conducted all counts in 2018, and six observers (four new, two from the previous year) conducted all counts in 2019.

For all songbird data analyses, we used only detections within 50-m of the sampling point. We included birds detected by sight and sound (excluding flyovers) for individual species and community statistical analyses. See Appendix 2-B for complete decision rules for bird data analyses and Appendix 2-C for detection types used for each species.

2.5. *Salamander sampling*

We assessed salamander response to cut-back borders using a combination of artificial and natural cover object surveys at four locations within each 30-m and 45-m wide cut-back border plot, as well as the control (Figure 2-3). Data were collected at Beury Mountain WMA (BMWMA [two powerline block replicates]), LCWMA (two powerline block replicates), LWWMA (two pipeline block replicates), and Stonewall Jackson WMA (SJWMA [one wildlife opening block replicate]). At each location, we conducted three daytime surveys annually during 2016–2019. Visit 1 was 15–30 April, visit 2 was 1–15 May, and visit 3 was 11 July–9 August (Table 2-2). Artificial coverboards at each location consisted of nine (3 x 3 arrangement spaced ~1 cm apart), 20.0 cm (length) x 10.0 cm (width) x 2.5 cm (height) wooden blocks cut from hardwood species from the region (American basswood [*Tilia americana*], American sycamore [*Platanus occidentalis*], and yellow poplar). Artificial coverboards were placed in cut-back border plots at least one month prior to sampling to allow them to weather (Hesed 2012). Natural cover objects sampled included logs and rocks that measured ≥ 40 mm at the narrowest dimension and occurred within a 5-m radius of each artificial coverboard location. All eligible natural cover object dimensions (length and width, in mm) were measured prior to salamander sampling during the pre-harvest period. We marked natural cover objects with stake flags to maintain consistent sampling effort across visits and years and did not add new natural cover objects following tree cutting.

Data recorded for each captured salamander included snout-vent length (SVL, anterior edge of the snout to the distal end of the cloaca) to the nearest 0.1 mm using a dial caliper. We recorded body mass to the nearest 0.1 g using a Pesola spring scale. We recorded species of captured individual and whether it was captured under a coverboard or natural cover object. If captured under a natural cover object, we recorded the dimensions (length and width) and type of object (log or rock). We clipped the tail of each individual to identify recaptures in subsequent sampling visits and sprayed an antiseptic to the clipped tail to minimize risk of infection. Clippers were flame-sterilized with 95% ethanol to reduce potential disease transmission between captures. Salamanders were released at their original point of capture. Salamander sampling was conducted under methods approved by the WVDNR (permits 2016.143, 2017.104, 2018.110, and 2019.110) and the West Virginia University Institutional Animal Care and Use Committee (protocol 1602000305).

2.6. *Statistical analyses*

Vegetation regeneration in cut-back borders.—We used general linear models to predict understory vegetation variables (ground cover, understory cover 0–1, understory cover >1–2 m, midstory stems 0–5.1 cm DBH, and midstory stems >5.1–11.3 cm DBH) in cut-back borders as functions of SPI (generated using Equations 1–3), TRMI, HLI, and the modified Beers aspect. Due to the shape of cut-back borders (i.e., narrow, strip cuttings) canopy cover from adjacent stands could block direct sunlight onto the forest floor (Figure 2-6) and affect understory plant successional trajectories within plots (Messier 1996, Stage and Salas 2007). Additionally, plant growth in the central Appalachian region is influenced by soil properties and slope aspect. Therefore, TRMI, HLI, and Beers aspect were included as primary predictor variables because these are topographic indices that account for soil moisture and slope aspect, which influence plant growth in the Appalachian region (Huebner and McGill 2018), but do not account for adjacent forest influence (i.e., shade from nearby tree canopy; Baker et al. 2016, Curzon et al. 2017). SPI, TRMI, and Beers aspect all index soil environments on various scales but are consistent in quantifying soil moisture where lower values indicate xeric conditions and higher values indicate mesic conditions. HLI, however, indexes soil conditions from 0–1 where zero represents mesic conditions and 1 represents xeric conditions. To ensure consistency with other indices, we calculated the inverse of HLI (i.e., $1/\text{HLI}$) to be used for analysis. We tested for collinearity between each topographic variable and SPI using Pearson’s correlation coefficient

prior to inclusion. Additionally, we tested for spatial autocorrelation of each variable using a global Moran's I at $\alpha = 0.05$ indicating no spatial autocorrelation (HLI, p -value <0.001 ; SPI, p -value <0.001 ; TRMI p -value <0.001 ; Beers aspect, p -value = 0.381).

We developed 11 *a priori* candidate models to test the effect of local topographic factors on the regeneration of five understory vegetation variables. We included a model where each topographic variable (SPI, TRMI, HLI, and Beers aspect) was the only predictor variable, a model that included an interactive term between each topographic variable and canopy cover, a model that included an additive term between SPI and Beers aspect (SPI does not account for slope aspect), and a model that included the additive term between SPI and Beers aspect and an interaction with canopy cover. We included understory cover 0–1 m, understory cover 1–2 m, ground cover, midstory stems 0–5.1 cm DBH, and midstory stems >5.1 –11.3 cm DBH as vegetation response variables. For our response metrics, we calculated the percent change from one-year post-treatment (immediately after tree cutting) to two-year post-treatment. Predictor variables were standardized to have a mean of 0 and standard deviation of one prior to analyses. Response variables were specified with a Gaussian distribution. Control plots were excluded from analysis because no canopy manipulation occurred in these plots.

We used an information-theoretic approach to test our 11 *a priori* models (Table 2-3) and implemented a model-averaging approach to predict percent change of vegetation variables. Model-averaged predictions were derived from all candidate models $<2 \Delta$ Akaike's Information Criterion adjusted for small sample sizes (AIC_c ; Burnham and Anderson 2002) using the 'AICcmodavg' package (Mazerolle 2016) in the statistical software R (R Core Team 2019). When making predictions, canopy cover was held constant at the mean (i.e., 0) where applicable. Because all topographic predictor variables are indexed from low (xeric conditions) to high (mesic conditions) values, we graphically represent our continuous soil index on a relative scale from xeric to mesic soil conditions.

Ruffed grouse and American woodcock occupancy in cut-back borders.—We estimated the probability of ruffed grouse occupancy in cut-back border treatments over time with a stacked Bayesian occupancy model (MacKenzie et al. 2017). We fit two models for ruffed grouse. Model #1 incorporated a cut-back border treatment and time interaction and model #2 incorporated cut-back border treatments without a time interaction. Both models used the same detection-level sub-model but a different occupancy-level sub-model. In our detection-level sub-

model, we modeled probability of detecting a grouse, conditional on true presence, using a Bernoulli process model. We modeled survey-level covariates (ordinal day and time since sunrise) thought to influence an observer's probability of detecting a grouse on the logit-linear scale.

For both occupancy-level sub-models, we modeled the probability of ruffed grouse occurrence using a Bernoulli process model. For model #1, we modeled logit expected occupancy probability, y_{ik} , as a function of cut-back border width and harvest intensity prescriptions with time and nuisance variables on the logit-linear scale (Equation 4).

$$\begin{aligned} \text{logit}(y_{ik}) = & \beta_0 + \beta_1 \times \text{trt}_{ik} + \beta_2 \times w30_{ik} + \beta_3 \times w45_{ik} + \beta_4 \times r4.5_{ik} + \\ & \beta_5 \times (\text{trt}_{ik} \times \text{time}_{ik}) + \beta_6 \times (w30_{ik} \times \text{time}_{ik}) + \\ & \beta_7 \times (w45_{ik} \times \text{time}_{ik}) + \beta_8 \times (r4.5_{ik} \times \text{time}_{ik}) + \\ & \beta_9 \times \text{elev}_i + \beta_{10} \times \text{TRMI}_i + \beta_{11} \times \text{ROW}/\text{WO}_i + \\ & \text{rand.point}_i + \text{rand.plot}_i \end{aligned} \quad (4)$$

For model #2, we modeled logit expected occupancy probability, y_{ik} , as a function of cut-back border width and harvest intensity prescriptions and nuisance variables on the logit-linear scale (Equation 5). The difference between Equation 4 and Equation 5 is that Equation 5 does not contain a unique interaction term between time and each cut-back border width or harvest intensity prescription. Rather, time interacts the same with all cut-back border width and harvest intensity prescriptions.

$$\begin{aligned} \text{logit}(y_{ik}) = & \beta_0 + \beta_1 \times \text{trt}_{ik} + \beta_2 \times w30_{ik} + \beta_3 \times w45_{ik} + \beta_4 \times r4.5_{ik} + \\ & \beta_5 \times (\text{trt}_{ik} \times \text{time}_{ik}) + \beta_6 \times \text{elev}_i + \\ & \beta_7 \times \text{TRMI}_i + \beta_8 \times \text{ROW}/\text{WO}_i + \\ & \text{rand.point}_i + \text{rand.plot}_i \end{aligned} \quad (5)$$

For both equations, i references each sampling point location (Figure 2-3) and k references year. On the right-hand side of the equation, the pre-treatment period is included in the intercept (β_0). This approach allows the control treatment across all years and the pre-treatment year for all cut-back border factor levels (15 m, 30 m, 45 m; 14.0 m²/ha, 4.5 m²/ha) to be used as baseline values for comparison with cut-back border factor levels at one-year and two-year post-treatment. The term trt_{ik} , corresponds to the 15 m and 14.0 m²/ha factor levels used as reference factor levels. The term $w30_{ik}$, corresponds to factor level 30 m, $w45_{ik}$ corresponds to factor level 45 m, and

$r4.5_{ik}$ corresponds to factor level 4.5 m²/ha. In Equation 4 the term $time_{ik}$ was included as an interactive term with each factor level to allow the effect of time to vary for each treatment. In Equation 5 the $time_{ik}$ term was included as an interactive term applied the same to all cut-back border factor levels (i.e., reference factor levels trt_{ik}) and not each factor level separately. All factor levels were dummy coded with “0” or “1” depending on whether that factor level occurred at each point, except for the reference factor levels (trt_{ik} ; i.e., 15 m and 14.0 m²/ha) which were dummy coded with a “1” unless it was the control treatment. Time was treated as a continuous variable with 0, 1, or 2 corresponding to pre-treatment, one-year post-treatment, or two-year post-treatment, respectively. Elevation ($elev_i$), TRMI ($TRMI_i$), and ROW/wildlife opening width (ROW/WO_i) were included as nuisance predictor variables. We included a random point count effect ($rand.point_i$) to account for heterogeneity among points and repeated observations at the same point, and a random plot effect ($rand.plot_i$) to account for co-dependence between points within cut-back borders. Continuous variables were standardized to have a mean of 0 and standard deviation of one prior to analysis.

For both models, we used diffuse prior distributions for the detection-level sub-model survey-level covariates (Gaussian [mean = 0, variance = 100]) and the occupancy-level sub-model factor levels and nuisance covariates (Gaussian [mean = 0, variance = 10]). We used diffuse covariate priors because we were uncertain of the relationships between cut-back border treatments and grouse. For our random point count and plot effect, we used a Gaussian (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma ($\alpha = 1, \beta = 1$) distribution. We obtained posterior distributions of model parameters by running three parallel Markov chain Monte Carlo (MCMC) simulations of 40000 iterations with a burn-in of 20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions for each model. We assessed model fit separately for both models with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhat values < 1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2019) using a Bayesian framework in JAGS (Plummer

2017) called from the package jagsUI (Kellner 2015). We assessed model performance between the treatment-time interaction model (model #1) and the main effects model (model #2) using deviance information criterion (DIC).

For model #1, we transformed model outputs using the inverse logit-link and calculated expected ruffed grouse occurrence in each cut-back border treatment in each year using Equation 4 terms. We calculated the log odds ratio of occupancy values within each respective cut-back border treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period (e.g., pre-treatment versus one-year post-treatment in 15 m-14 m²/ha) and between cut-back border treatments within each time period, and assessed differences based on 95% credible intervals not overlapping 0. For model #2, we transformed model outputs using the inverse logit-link and calculated expected ruffed grouse occurrence in each cut-back border width and harvest intensity prescription using Equation 5 terms. We calculated the log odds ratio of expected occupancy between width and harvest intensity factor levels, separately. We assessed differences based on 95% credible intervals not overlapping 0.

American woodcock were detected a total of two times during the study period. Therefore, we did not have enough detections to complete a statistical analysis with enough predictive power to warrant analyzing these data further. Sampling effort and a descriptive summary of American woodcock data are provided in results.

Focal breeding songbird abundance.—We estimated abundance of breeding songbird species in cut-back border treatments over time with a stacked Bayesian *N*-mixture model (Kéry and Royle 2016). We included species with ≥ 40 detections within 50-m of the sampling point and ≥ 0.200 cumulative detection probability (i.e., probability of detecting a species at least once across two annual survey visits) (see Appendix 2-C for species-level detections). These criteria were based on a power analysis. We fit two models for each songbird species. Model #1 incorporated a cut-back border treatment and time interaction and model #2 incorporated cut-back border treatments without a unique treatment-time interaction. Both models used the same detection-level sub-model but a different abundance-level sub-model. In our detection-level sub-model, we modeled probability of detecting an individual, conditional on true presence, using a Binomial process model. We modeled survey-level covariates (e.g., ordinal day, surveyor skill, and time since sunrise) thought to influence an observer's probability of detecting an individual on the logit-linear scale.

For both biological-level sub-models, we modeled expected species abundance using a Poisson process model. For model #1, we modeled log expected species abundance, y_{ik} , as a function of cut-back border width and harvest intensity prescriptions with time and nuisance variables on the log-linear scale (Equation 4). For model #2, we modeled log expected species abundance, y_{ik} , as a function of cut-back border width and harvest intensity prescriptions and nuisance variables on the log-linear scale (Equation 5). For both models, we used diffuse prior distributions for the detection-level sub-model survey-level covariates (Gaussian [mean = 0, variance = 100]) and the biological-level sub-model the factor levels and nuisance covariates (Gaussian [mean = 0, variance = 10]). We used diffuse covariate priors because we were uncertain of the relationships between cut-back border treatments and songbirds. For our random point count and plot effect, we used a Gaussian (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma ($\alpha = 1$, $\beta = 1$) distribution. We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 40000 iterations with a burn-in of 20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions. We assessed model fit with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhat values < 1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2019) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015). We assessed model performance between the treatment-time interaction model (model #1) and the main effects model (model #2) for each species using DIC.

For model #1, we transformed model outputs using the inverse log-link and calculated expected abundance in each cut-back border treatment in each year using Equation 4 terms. We calculated the log proportional change of abundance within each respective cut-back border treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period and between cut-back border treatments within each time period, and assessed differences based on 95% credible intervals not overlapping 0. For model #2, we transformed

model outputs using the inverse log-link and calculated expected abundance in each cut-back border width and harvest intensity prescription using Equation 5 terms. We calculated the log proportional change of expected abundance between width and harvest intensity factor levels, separately. We assessed differences based on 95% credible intervals not overlapping 0.

Species-vegetation correlations.—We assessed correlations between species counts (response variables) and vegetation data (predictor variables) at each point count location using constrained ordination techniques. For each species, we used the maximum count across the two survey visits at each point count location for each year (see Appendix 2-B for complete decision rules). Species with ≥ 20 detections over the whole study period were included in analysis (Appendix 2-C). We chose this criterion because it encompassed a relatively equal number of species from each guild (seven forest gap, 10 forest interior, 11 interior-edge, and 10 young forest, two predator/parasite) and minimized the potential for spurious relationships resulting from low detections. We used a detrended correspondence analysis (DCA) to inform model type specification based on gradient length of the data using the ‘decorana’ from package ‘vegan’ (Oksanen 2019). Gradient length of species count data had a standard deviation >4 ($SD = 5.26$) indicating high species turnover across study sites, and thus a unimodal method (canonical correspondence analysis [CCA]; Lepš and Šmilauer 2003) was used for the analysis. A forward selection approach was used to select a subset of predictor variables that maximized the model’s adjusted R^2 using a 1000 step permutation test in ‘ordiR2step’ from package ‘vegan’. This approach increases parsimony in our CCA model and removes strong linear correlations among explanatory variables. We tested whether there was a relationship between response data and the explanatory variables using a 999 step permutational ANOVA for each individual canonical axis in our CCA model at $\alpha = 0.05$. Vegetation data were standardized to have a mean of 0 and standard deviation of one prior to analysis. Species-vegetation correlations were visualized using a correlation biplot displaying response variables and predictor variables.

Spring stopover songbird richness.—We used a Bayesian generalized linear mixed-effects model (GLMM) using model #1 which incorporated a cut-back border treatment and time interaction and model #2 which incorporated cut-back border treatments without a unique treatment-time interaction to assess apparent species richness during the spring migration

stopover period. We included all resident, breeding, and migrant passerine species for this analysis (Appendix 1-A).

We modeled expected apparent species richness using Equation 4 (model #1) and Equation 5 (model #2), with log expected apparent richness, y_{ik} , specified with a Poisson process model in both equations. We used diffuse prior distributions for the factor levels and nuisance covariates (Gaussian [mean = 0, variance = 10]). We used diffuse covariate priors because we were uncertain of the relationships between cut-back border treatments and songbirds. For our random point count and plot effect, we used a Gaussian (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma ($\alpha = 1$, $\beta = 1$) distribution. We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 40000 iterations with a burn-in of 20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions. We assessed model fit with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhat values < 1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2019) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015). We assessed model performance between the treatment-time interaction model (model #1) and the main effects model (model #2) using DIC.

For model #1, we transformed model outputs using the inverse log-link and calculated expected apparent species richness in each cut-back border treatment in each year using Equation 4 terms. We calculated the log proportional change of expected richness within each respective cut-back border treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period and between cut-back border treatments within each time period, and assessed differences based on 95% credible intervals not overlapping 0. For model #2, we transformed model outputs using the inverse log-link and calculated expected apparent species richness in each cut-back border width and harvest intensity prescription using Equation 5 terms. We calculated the log proportional change of expected richness between width and harvest

intensity factor levels, separately. We assessed differences based on 95% credible intervals not overlapping 0.

Avian guild richness in cut-back borders.—We used a Bayesian GLMM with model #1 which incorporated a cut-back border treatment and time interaction and model #2 which incorporated cut-back border treatments without a unique treatment-time interaction to assess apparent species richness of avian guilds to cut-back border treatments through time. We categorized avian species into ecological guilds based on habitat (forest gap, forest interior, interior-edge, and young forest), foraging (aerial insectivore), nesting (cavity), or predator/parasite (see Appendix 2-D for guild definitions). Additionally, we assessed species of conservation priority in the region as a distinct group for analysis (Appendix 2-C). We considered a species as conservation priority if it was listed by Appalachian Mountains Joint Venture (AMJV 2018), Partners-in-Flight Appalachian region Bird Conservation Region 28 (PIF 2019), or WVDNR State Wildlife Action Plan (WVDNR 2015).

We modeled expected apparent species richness using Equation 4 (model #1) and Equation 5 (model #2), with log expected apparent richness, y_{ik} , specified with a Poisson process model in both models. We used diffuse prior distributions for the factor levels and nuisance covariates (Gaussian [mean = 0, variance = 10]). We used diffuse covariate priors because we were uncertain of the relationships between cut-back border treatments and avian guilds. For our random point count and plot effect, we used a Gaussian (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma ($\alpha = 1$, $\beta = 1$) distribution. We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 40000 iterations with a burn-in of 20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions. We assessed model fit with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhat values < 1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2019) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015). We assessed model performance between

the treatment-time interaction model (model #1) and the main effects model (model #2) using DIC.

For model #1, we transformed model outputs using the inverse log-link and calculated expected apparent species richness in each cut-back border treatment in each year using Equation 4 terms. We calculated the log proportional change of expected richness within each respective cut-back border treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period and between cut-back border treatments within each time period, and assessed differences based on 95% credible intervals not overlapping 0. For model #2, we transformed model outputs using the inverse log-link and calculated expected apparent species richness in each cut-back border width and harvest intensity prescription using Equation 5 terms. We calculated the log proportional change of expected richness between width and harvest intensity factor levels, separately. We assessed differences based on 95% credible intervals not overlapping 0.

Salamander surface relative abundance in cut-back borders.—We estimated surface relative abundance of terrestrial salamanders in cut-back border treatments over time using a Bayesian GLMM. We estimated surface relative abundance because terrestrial salamanders spend the majority of their time underground and we did not incorporate detection probability into the analysis. We fit two models for salamander species. Model #3 incorporated a cut-back border treatment and time interaction and model #4 incorporated cut-back border treatments without a time interaction. We treated capture data at the four sampling plots within a single cut-back border plot (Figure 2-3). Sampling plots within a single cut-back border were considered independent because of the small home range size of terrestrial salamanders (Kleeberger and Werner 1982). Recaptures were not used in analysis.

For model #3, we modeled expected relative abundance using a Poisson process model. Log expected relative abundance, y_{ik} , was modeled as a function of cut-back border width and harvest intensity prescriptions with time and nuisance variables on the log-linear scale (Equation 6).

$$\begin{aligned} \log(y_{ik}) = & \beta_0 + \beta_1 \times trt_{ik} + \beta_2 \times w45_{ik} + \beta_3 \times r4.5_{ik} + \\ & \beta_4 \times (trt_{ik} \times time_{ik}) + \beta_5 \times (w45_{ik} \times time_{ik}) + \\ & \beta_6 \times (r4.5_{ik} \times time_{ik}) + \beta_7 \times elev_i + \\ & \beta_8 \times TRMI_i + \beta_9 \times Cover_i + rand.point_i \end{aligned} \quad (6)$$

For model #4, we modeled expected relative abundance using a Poisson process model. Log expected relative abundance, y_{ik} , was modeled as a function of cut-back border width and harvest intensity prescriptions and nuisance variables on the log-linear scale (Equation 7).

$$\begin{aligned} \log(y_{ik}) = & \beta_0 + \beta_1 \times trt_{ik} + \beta_2 \times w45_{ik} + \beta_3 \times r4.5_{ik} + \\ & \beta_4 \times (trt_{ik} \times time_{ik}) + \beta_5 \times elev_i + \\ & \beta_6 \times TRMI_i + \beta_7 \times Cover_i + rand.point_i \end{aligned} \quad (7)$$

For both models, i references each sampling point location (Figure 2-3) and k references year. On the right-hand side of the equation, the pre-treatment period is included in the intercept (β_0). This approach allows the control treatment across all years and the pre-treatment year for all cut-back border factor levels to be used as baseline values for comparison with cut-back border factor levels (30 m, 45 m; 14.0 m²/ha, 4.5 m²/ha) at one-year and two-year post-treatment. The term trt_{ik} , corresponds to the 30 m and 14.0 m²/ha factor levels used as reference factor levels. The term $w45_{ik}$ corresponds to factor level 45 m, and $r4.5_{ik}$ corresponds to factor level 4.5 m²/ha. In Equation 6 the term $time_{ik}$ was included as an interactive term with each factor level to allow the effect of time to vary for each treatment. In Equation 7 the $time_{ik}$ term was included as an interactive term applied the same to all cut-back border factor levels (i.e., reference factor levels trt_{ik}) and not each factor level separately. All factor levels were dummy coded with “0” or “1” depending on whether that factor level occurred at each point, except for the reference factor levels (30 m and 14.0 m²/ha) which were dummy coded with a “1” unless it was the control treatment. Time was treated as a continuous variable with 0, 1, or 2 corresponding to pre-treatment, one-year post-treatment, or two-year post-treatment, respectively. Elevation ($elev_i$), TRMI ($TRMI_i$), and number of natural cover objects ($Cover_i$) were included as nuisance predictor variables. We included a random point count effect to account for heterogeneity among points and repeated observations at the same point. Continuous variables were standardized to have a mean of 0 and standard deviation of one prior to analyses.

For both models, we used diffuse prior distributions for factor levels and nuisance covariates (Gaussian [mean = 0, variance = 10]). We used diffuse covariate priors because we were uncertain of the relationships between cut-back border treatments and salamanders. For our random point effect, we used a Gaussian (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma ($\alpha = 1$, $\beta = 1$) distribution. We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 40000 iterations with a burn-in of

20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions. We assessed model fit with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhat values < 1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2019) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015). We assessed model performance between the treatment-time interaction model (model #3) and the main effects model (model #4) using DIC.

For model #3, we transformed model outputs using the inverse log-link and calculated expected relative abundance in each cut-back border treatment in each year using Equation 6 terms. We calculated the log proportional change of relative abundance values within each respective cut-back border treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period and between cut-back border treatments within each time period, and assessed differences based on 95% credible intervals not overlapping 0. For model #4, we transformed model outputs using the inverse log-link and calculated expected relative abundance in each cut-back border width and harvest intensity prescription using Equation 7 terms. We calculated the log proportional change of expected relative abundance between width and harvest intensity factor levels, separately. We assessed differences based on 95% credible intervals not overlapping 0. We assessed model performance between the treatment-time interaction model (model #3) and the main effects model (model #4) using DIC.

Salamander body condition in cut-back borders.—We assessed body condition index (BCI) of each terrestrial salamanders species, separately, in cut-back border treatments over time with a model which had a unique treatment-time interaction and a model where time had a constant relationship with treatment using a Bayesian GLMM. We calculated BCI by regressing the natural logarithm of body mass on the natural logarithm of SVL and using the residuals as the BCI values for each individual following Gabor (1995). BCI was calculated independently at each study site for each species. Recaptures were not used for analysis.

We modeled the mean expected BCI using the right-hand side of equations 6 (model #3) and equation 7 (model #4) with a Gaussian process model. For both models we used diffuse prior distributions for the process model variance (variance = τ) from an inverse Gamma distribution and for the factor levels and nuisance variables (Gaussian [mean = 0, variance = 10]). We used diffuse covariate priors because we were uncertain of the relationships between cut-back border treatments and BCI. For our random point effect, we used an inverse Gaussian (mean = 0, variance = τ) prior with a hyperparameter τ from a Gamma ($\alpha = 1$, $\beta = 1$) distribution. We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 40000 iterations with a burn-in of 20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions. We assessed model fit with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhat values < 1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2019) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015). We assessed model performance between the treatment-time interaction model (model #3) and the main effects model (model #4) using DIC.

For model #3, we calculated the change of expected BCI values within each respective cut-back border treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period and between cut-back border treatments within each time period, and assessed differences based on 95% credible intervals not overlapping 0. For model #4, we calculated the change of expected BCI values within each cut-back border width and harvest intensity prescription. We assessed differences based on 95% credible intervals not overlapping 0.

3. Results

3.1. *Topographic effects on vegetation regeneration in cut-back borders*

We tested 11 *a priori* models to assess whether differences in vegetation response between one-year post-treatment and two-year post-treatment could be predicted from

topographic indices (see Appendix 2-E for model results). Shade proportion index (SPI), the proportion of the vegetation sampling plot which was shaded, ranged between 0–94% (mean = 15%) in 15-m cut-back borders and between 0–71% (mean = 13%) in 30-m cut-back borders. No vegetation sampling plots in the 45-m cut-back borders were shaded.

No single model received strong support (Appendix 2-E) and thus, we used model-averaging to predict percent change for all vegetation variables. Percent change from one-year post-treatment to two-year post-treatment of understory cover 0–1 m, understory cover >1–2 m, and midstory stems 0–5.1 cm DBH were greater on mesic soils relative to xeric soils in cut-back borders (Figure 2-7). Percent change of ground cover and midstory stems >5.1–11.3 cm DBH did not differ across xeric and mesic soil conditions in cut-back borders (Figure 2-7). For all vegetation variables collected, we provide summary statistics in Appendix 2-F.

3.2. *Ruffed grouse and American woodcock response to cut-back borders*

We conducted a total of 840 grouse drumming surveys during the study. We detected a total of 24 grouse (21 during drumming surveys, 3 opportunistic detections) during the study. Our posterior predictive check indicated models were a reasonable fit (Appendix 2-G). Based on on DIC, the treatment-time interaction model performed better than the main effects model (Appendix 2-G). Estimated effective detection probability (i.e., probability of detecting an individual over two annual surveys) during drumming surveys was $0.326 (\text{mean}) \pm 0.08 (\text{SD})$. We did not detect any differences in occupancy between control and cut-back border treatments, nor any differences in occupancy between pre-treatment and one-year post-treatment or two-year post-treatment for any cut-back border treatment (Figure 2-8). Grouse occupancy was strongly positively influenced by elevation (slope coefficient [log odds scale] = [mean] $2.85 \pm [\text{SD}] 1.50$).

We conducted 434 woodcock singing-ground surveys during the study. We detected two woodcock during the study period. One of the detected woodcock was at AGWMA in the 15-m width, $14 \text{ m}^2/\text{ha}$ intensity treatment at one-year post-treatment, and the other at LWWMA in the 30 m wide, $14 \text{ m}^2/\text{ha}$ intensity treatment at two year post-treatment. Additionally, woodcock were heard at BLWMA and SJWMA but not during surveys.

3.3. *Focal songbird response to cut-back borders*

We conducted a total of 868 avian point counts over the course of the study period. We detected a total of 102 species of songbirds. Of the 24 species that met the detection criteria (≥ 40

detections), 16 species also met the cumulative detection probability criteria (≥ 0.200 ; Appendix 2-C). Of these 16 species, five were from the young forest guild (American goldfinch, Carolina wren, common yellowthroat, eastern towhee, and indigo bunting), four were from the interior-edge guild (American redstart, blue-gray gnatcatcher, eastern wood-pewee, and northern cardinal), four were from the forest gap guild (black-throated green warbler, cerulean warbler, hooded warbler, and red-eyed vireo), and three were in the forest interior guild (black-and-white warbler, ovenbird, and wood thrush). Our posterior predictive checks indicated all individual species models were a reasonable fit (Appendix 2-G). The treatment-time interaction model performed better, based on DIC, for ten species whereas the main effects model performed better for six species (Appendix 2-G).

Of the 16 species analyzed, nine responded positively to at least one cut-back border treatment in the treatment-time interaction model (Figures 2-9–2-12). In the 15 m wide, 14 m²/ha intensity treatment, hooded warbler and indigo bunting abundances increased from pre-treatment to one-year post-treatment (Figures 2-9 and 2-11) and hooded warbler and red-eyed vireo abundances increased from pre-treatment to two-year post-treatment (Figure 2-11). In the 15 m wide, 4.5 m²/ha intensity treatment, American goldfinch and black-and-white warbler abundances increased from pre-treatment to one-year post-treatment (Figures 2-9 and 2-12). Additionally, in the 15 m wide, 4.5 m²/ha intensity treatment, blue-gray gnatcatcher, Carolina wren, eastern towhee, hooded warbler, indigo bunting, northern cardinal, and red-eyed vireo abundances increased from pre-treatment to two-year post-treatment (Figure 2-9–2-11). In the 45 m wide, 14 m²/ha intensity treatment, indigo bunting abundance increased from pre-treatment to one-year and two-year post-treatment (Figure 2-9), and blue-gray gnatcatcher abundance increased from pre-treatment to one-year post-treatment (Figure 2-10).

Two forest gap species, black-throated green warbler and hooded warbler responded negatively to cut-back border treatments. Black-throated green warbler abundance decreased from pre-treatment to one-year post-treatment in the 45 m wide, 14 m²/ha intensity treatment (Figure 2-11). Hooded warbler abundance decreased from pre-treatment to one-year post-treatment in the 30 m wide and 45 m wide, 4.5 m²/ha intensity treatments (Figure 2-11). Abundances of both species rebounded by two-year post-treatment.

In the main effects model, harvest intensity had a strong influence on eight species. American goldfinch, black-and-white warbler, blue-gray gnatcatcher, Carolina wren, eastern

towhee, hooded warbler, indigo bunting, and northern cardinal abundances were higher in the 4.5 m²/ha intensity treatment than the control (Figure 2-13). Eastern towhee, hooded warbler, and indigo bunting abundances were higher in the 14 m²/ha intensity treatment than the control (Figure 2-13). The main effect of width had a strong influence on three species. Eastern towhee and indigo bunting abundances were higher in the 45 m wide treatment than the control (Figure 2-14). Eastern towhee, hooded warbler, and indigo bunting abundances were higher in the 15 m wide treatment than the control (Figure 2-14). Eastern towhee abundance was higher in the 30 m wide treatment than the control

3.4. *Avian community*

Avian community-vegetation relationships.—Of the 102 songbird species detected during the study, 40 met the detection criteria (≥ 20 detections) to be included in a CCA to assess species correlations with vegetation variables. Representation from each habitat guild included 10 species from the young forest guild, 11 from the interior-edge guild, seven from the forest gap guild, 10 from the forest interior guild, and two from the predator/parasite guild (Appendix 2-D).

The constrained proportion of variance in species count data (response data) explained by vegetation data was 8.9%. The permutation test on each canonical axis indicated the first five canonical axes each had a significant relationship with response data (Appendix 2-G). The first principal component axis, which explained 3.5% of response data, had a strong elevational gradient and was positively correlated with the black-throated blue warbler, black-throated green warbler, Canada warbler, chestnut-sided warbler, common yellowthroat, dark-eyed junco, and ruffed grouse (Appendix 2-G, Figure 2-15). The second principal component axis, which explained 1.1% of response data, described a gradient from mature forest-like characteristics (high overstory cover and basal area) to disturbed forest-like characteristics (high ground cover and understory cover >1–2 m). Young forest, forest gap, and interior-edge species were spread across the second principal axis with no obvious trends, and forest interior species were generally associated with high overstory cover and basal area (Figure 2-15). The remaining eight principal component axes explained a combined 2.0% of the variation in response data.

Species richness in cut-back borders during spring migration.—We conducted 364 point counts and detected 84 different species during the spring stopover period. Our posterior predictive checks indicated our models were a reasonable fit (Appendix 2-G). Based on DIC, the

main effects model performed better than the treatment-time interaction model (Appendix 2-G). Species richness during the spring stopover period was lower in the 15 m wide, 14 m²/ha and 4.5 m²/ha intensity treatments and 45 m wide, 14 m²/ha intensity treatment at one-year post-treatment compared to pre-treatment (Figure 2-16). Additionally, species richness was lower in the 30 m wide and 45 m wide, 14 m²/ha intensity treatments at two-year post-treatment compared to pre-treatment levels (Figure 2-16). The main effect of harvest intensity and width had a strong influence on spring stopover richness. Species richness was higher in the control than the 14 and 4.5 m²/ha intensity treatment (Figure 2-13), and species richness was higher in the control treatment than the 15 m, 30 m, and 45 m wide treatments (Figure 2-14).

Avian guild richness in cut-back borders.—Our posterior predictive checks indicated all avian guild models were a reasonable fit (Appendix 2-G). The treatment-time interaction model performed better for four avian guilds whereas the main effects model performed better for two avian guilds (Appendix 2-G). Three of the four habitat guilds, the cavity nesting guild, and species of conservation priority generally responded positively to cut-back border treatments of narrower width at two-year post-treatment (Figures 2-17 and 2-18). In the 15 m wide, 14 m²/ha intensity treatment, richness of species of conservation priority increased from pre-treatment to one-year post-treatment and two-year post-treatment (Figure 2-18), and richness of the cavity nesting, interior-edge habitat, and young forest habitat guilds increased from pre-treatment to two-year post-treatment (Figures 2-17 and 2-18). In the 15 m wide, 4.5 m²/ha intensity treatment, species of conservation priority and the young forest habitat guild increased from pre-treatment to one-year post-treatment (Figures 2-17 and 2-18), and the interior-edge and young forest habitat guilds increased from pre-treatment to two-year post-treatment (Figure 2-17). In the 30 m wide, 14 m²/ha intensity treatment, the interior-edge habitat guild and cavity nesting guild increased from pre-treatment to two-year post-treatment (Figures 2-17 and 2-18). In the 45 m wide, 14 m²/ha intensity treatment, richness of species of conservation priority and the young forest guild increased from pre-treatment to two-year post-treatment (Figures 2-17 and 2-18). No guilds declined in richness.

The main effect of harvest intensity had a strong influence on the young forest, interior-edge, and forest gap habitat guilds and the species of conservation priority. Young forest, interior-edge, forest gap, and species of conservation priority species richness was higher in the 14 m²/ha and 4.5 m²/ha intensity treatments than the control (Figure 2-13). The main effect of

width had a strong influence on the young forest, interior-edge, and forest gap habitat guilds and the species of conservation priority. Species richness of the young forest, interior-edge, forest gap, and species of conservation priority was higher in the 15 m wide treatment than the control (Figure 2-14). Young forest species richness was higher in the 45 m wide treatment than the control, and interior-edge species richness was higher in the 30 m wide treatment than the control (Figure 2-14).

3.5. *Salamanders in cut-back borders*

Salamander surface relative abundances in cut-back borders.—We captured a total of 229 salamanders of five species during the study period (Appendix 2-H). The majority of captures (187, 82%) were eastern red-backed salamanders (*Plethodon cinereus*) and thus, we performed analysis only on this species. Our posterior predictive check indicated the model was a reasonable fit (Appendix 2-G). The main effects model performed better than the treatment-time interaction model based on DIC (Appendix 2-G). Relative abundance of eastern red-backed salamanders decreased in the 30 m wide, 14 m²/ha intensity treatment from pre-treatment to one-year post-treatment but returned to pre-treatment levels the subsequent year (Figure 2-19). Abundance increased from pre-treatment to one-year post-treatment in the 45 m wide, 4.5 m²/ha intensity treatment but returned to pre-treatment levels at two-year post-treatment.

Salamander body condition in cut-back borders.—We assessed BCI of 154 eastern red-backed salamanders from BMWMA and LWWMA (some captures did not have morphological measurements taken) because too few eastern red-backed salamanders were captured at LCWMA (11 captures) or SJWMA (4 captures) to obtain reliable site-level BCI estimates. We selected these two study sites because they accounted for the majority of eastern red-backed salamander captures (154 of 169, 91%) during the study. Our posterior predictive check indicated the model was a reasonable fit (Appendix 2-G). The main effects model performed better than the treatment-time interaction model based on DIC (Appendix 2-G). BCI increased from pre-treatment to two-year post-treatment in the 30 m wide, 4.5 m²/ha intensity treatment and the 45 m wide, 14 m²/ha intensity treatment (Figure 2-20). BCI did not decrease in any cut-back border treatment.

4. Discussion

With the proliferation of linear infrastructure due to increased energy demand and extraction in West Virginia and the central Appalachian region, there is a growing need to find ways that use pipelines and powerlines to meet conservation objectives. Habitat management along ROW forest edges via cut-back borders provides a unique opportunity to incorporate wildlife habitat management concepts into human-modified landscape features to meet these conservation objectives. Here we have provided a short-term assessment of cut-back border treatments and associated vegetation and their effects on birds and salamanders to develop management guidelines that can assist wildlife managers in their decision-making when confronted with managing wildlife along abrupt forest edges of ROW corridors or wildlife openings (hereafter “ROW/WO”).

4.1. *Wildlife response to cut-back borders*

Prior to cut-back border implementation, habitats at study sites consisted of early-successional habitat (i.e., primarily grasses, forbs, and low shrubs) immediately adjacent to mature forests. Following cut-back border treatment, habitats within cut-back borders consisted of a developed understory vegetation component (with similarities to habitat in ROW/WO) with the retention of some canopy trees that created a transitional area between ROW/WO and adjacent mature forests. This reduction of the abrupt forest edge along ROW/WO created conditions suitable to avian and salamander species from a broad spectrum of forest successional stages.

Prior to cut-back border treatment, we observed populations of young forest avian species in mature forests immediately adjacent to ROW/WO (i.e., future locations of cut-back borders) and within ROW/WO. Following treatment, cut-back borders along ROW/WO supported higher populations of young forest species than no cut-back borders along ROW/WO. Young forest avifauna responded positively to 15-m, 30-m, and 45-m cut-back border widths at both cutting intensities (14 m²/ha and 4.5 m²/ha basal area retention), with significant positive responses to the 15-m and 45-m wide cut-back border treatments and both cutting intensities from American goldfinch, Carolina wren, eastern towhee, indigo bunting, and the young forest habitat guild (Figures 2-9, 2-13, 2-14, and 2-17). These positive responses were likely due to vegetation in ROW/WO complimenting vegetation in cut-back borders (or vice versa). The reduction of a portion of the overstory layer in cut-back borders stimulated ground and understory vegetation growth in cut-back borders and increased locally available habitat when combined with

vegetation in ROW/WO, effectively widened ROW/WO. Right-of-way/wildlife opening widening via cut-back borders likely alleviates area-sensitivity pressures (e.g., territory elongation, width sensitivity) observed in some young forest species (Rodewald and Vitz 2005, Shake et al. 2012). Anderson et al. (1977) and Confer and Pascoe. (2003) reported young forest birds were scarce or absent in narrow ROW corridors but increased with increasing corridor width, which may be a manifestation of edge-avoidance by young forest species (Schlossberg and King 2009). Our nuisance variable, ROW/WO width, strongly positively influenced American goldfinch abundance and young forest guild richness were positively correlated with ROW/WO width (slope coefficients = 0.34 and 0.16, respectively). It would then follow that wider ROW/WO-cut-back border combinations would elicit greater avian responses. However, we observed the majority (9 of 16) of our significant responses to width in the 15-m cut-back border treatments. Previous studies have reported young forest avifauna exhibit a quadratic relationship with ROW width (i.e., lower abundance in narrow and wide-width ROWs and high abundance in intermediate-width ROWs; King et al. 2009) which may explain the few significant responses to the 30-m and 45-m cut-back border treatments observed in our study. This may indicate the relative success of managing young forest avifauna using cut-back borders is contingent on ROW/WO width.

Ruffed grouse, a young forest species of conservation priority in the region, did not show a significant response to cut-back border treatments, although we did observe greatest mean occupancy at two-year post-treatment in the 30-m and 45-m treatments (Figure 2-8). However, low detections at few study sites resulted in low model precision, limiting our ability to discern effects from cut-back border treatments. The ruffed grouse is a species that prefers diversely aged forest stands to meet their year-round needs (Thompson and Dessecker 1997). The positioning of cut-back borders optimizes these needs in a relatively small area, where ROW/WO offer travel lanes between suitable young forest patches (i.e., cut-back borders) and adjacent mature forests provide reliable hard mast and roosting habitat (Thompson and Fritzell 1988, Whitaker et al. 2006). Ruffed grouse presence was positively correlated with elevation, whereas all other vegetation correlations were weak. One explanation for the correlation with elevation is the prevalence of West Nile Virus (WNV) in the region and its negative effects on grouse populations at lower elevations (Stauffer et al. 2018). While information about the effects of WNV on grouse in West Virginia is limited, WNV may prove to be an additional hurdle for

ruffed grouse management in the state. Weak correlations between grouse and other vegetation variables could be due to vegetation lag following cut-back border treatment. Two years may not be enough time to allow major young forest vegetation components like midstory stems, an important vegetation characteristic for ruffed grouse (Whitaker and Stauffer 2003, Jones et al. 2008), to fully develop, especially at sites with rocky or shallow soils (e.g., AGWMA, LCWMA, and SCWMA). Some young forest species exhibit a delayed response to timber harvesting practices, which could result from the slow vegetation regeneration and possibly explain the weak species-vegetation correlations we observed for many young forest species (Figure 2-15; Schlossberg and King 2009). Future monitoring of the vegetation and avian community may highlight some important vegetation characteristics to assist managers in creating optimal habitat conditions for young forest avifauna in cut-back borders.

Forest gap avifauna responded positively to 15-m cut-back border widths at both 14 m²/ha and 4.5 m²/ha cutting intensities whereas responses were negative to the 30-m and 45-m cut-back border widths in the treatment-time interaction model. Forest gap specialists are associated with forest interior habitats with a preference for irregular canopy structures (Bowen et al. 2007, Perkins and Wood 2014). Within forested landscapes, forest gaps species are often closely associated with canopy disturbances (Boves et al. 2013) that retain suitable levels of canopy (Sheehan et al. 2014). The moderate levels of canopy trees retained in the 15-m cut-back borders following treatment created an irregular, but sufficient canopy layer that when combined with adjacent mature forest maintained suitable habitat conditions in cut-back borders. Two-age silviculture harvests (much like the harvest prescriptions used in our study) have proven to benefit forest gap species immediately following harvesting (Nareff et al. 2019). However, these conditions are ephemeral and habitat quality for forest gap species declines as tree crowns expand into unoccupied canopy gaps, eliminating sallying or singing perches for this group (Sheehan et al. 2014). Future monitoring can assess how long cut-back borders provide suitable habitat for forest gap species.

The forest interior avifauna was of interest for this study as this group of species is negatively affected by the presence of ROWs in forest-dominated landscapes (Barton et al. 2016, Farwell et al. 2016). We did not detect any significant negative responses by individual species or the community following treatment, indicating cut-back borders did not significantly reduce habitat quality for forest interior species. Rather, black-and-white warblers increased in the 15-m

wide, 4.5 m²/ha intensity treatment (Figure 2-12) and was higher in the 4.5 m²/ha intensity treatment than control (Figure 2-13). Considering forest interior species were positively correlated with mature forest characteristics (basal area and overstory cover) in the CCA (Figure 2-15), and live basal area and overstory cover significantly decreased following treatment (Appendix 2-C), we might expect habitat suitability to be reduced for forest interior species in cut-back borders. However, it appears that the cut-back borders in our study which were relatively narrow, in conjunction with adjacent mature forest, retained enough key habitat features to maintain forest interior populations along ROW/WO.

We assessed another forest interior species of interest to cut-back borders: the eastern red-backed salamander. Surprisingly, we detected a significant negative response to the 30-m wide, 14 m²/ha intensity treatment and a significant positive response to the 45-m wide, 4.5 m²/ha intensity treatment at one-year post-treatment, although abundances returned to pre-treatment levels during the two-year post-treatment period. This appears counter to expectations, as we would predict the narrower, least intensive treatment to support greater salamander abundances relative to the widest, heaviest intensity treatments based on the physiology of terrestrial salamanders and previous studies from the region (Duguay and Wood 2002). Additionally, we observed an increase in BCI of eastern red-backed salamanders in the 30 m wide and 45 m wide, 14 m²/ha intensity treatments at two-year post-treatment. Higher BCI in the lighter intensity treatments (i.e., 14 m²/ha intensity) compared to control/pre-treatment could result from greater food availability or more foraging opportunities. Eastern red-backed salamanders forage on the forest floor surface and suitable environmental conditions (e.g., cool, moist) allow them to actively forage for longer periods without desiccation. These conditions are often met in undisturbed areas (i.e., control/pre-treatment) but our 14 m²/ha intensity treatments appear to support healthier eastern red-backed salamanders relative to individuals in the control/pre-treatment. Traditional timber harvests remove felled trees and CWD which are important surface refugia for salamanders (Petranka et al. 1994, Semlitsch et al. 2009). Our study was unique in that we did not remove any CWD from the stand (see Section 2.2) which may have mitigated the detrimental effects associated with canopy removal (e.g., dry leaf litter and soils).

However, it is unlikely this effect alone explained the responses we observed. One potential source of sampling bias may be related to greater rates of felled trees near salamander

sampling plots which may influence the number of captures, as we did not sample under newly felled trees. Despite this potential source of error, the short-term evidence suggests that cut-back borders provide suitable habitat conditions for eastern red-backed salamanders. Lag effects of amphibians to forest disturbance have been reported (Ash 1988, Popescu et al. 2012), but the presence of CWD could reduce or eliminate detrimental effects associated with reduced canopy cover (McKenny et al. 2006). Continued monitoring to fully assess the long-term effects of cut-back borders on eastern red-backed salamanders and other terrestrial salamanders will help develop management strategies for young forest and forest interior communities.

Finally, songbird richness during the spring stopover period decreased following treatments, particularly the first year (Figure 2-16). Decreased songbird stopover richness immediately following treatment is expected. Songbirds rely heavily on foliar-occurring arthropods during the spring migration period (Strode 2009). The lack of plant growth during the spring period combined with the recent tree cutting limited the available leaf area for foliar-occurring arthropods. Therefore, cut-back borders provided few foliar-occurring arthropods relative to uncut areas and reduced songbird richness during spring stopover during the one-year post-treatment period. Further, negative responses showed no consistent pattern with regard to cut-back border width or harvest intensity over time. Additionally, the inconsistent responses we observed could be a result of sampling during a highly variable period, particularly with Neotropical migrants. Migration times are often species-specific that fluctuate with continental-wide weather patterns but are also highly linked to plant phenology which is becoming increasingly variable (Mayor et al. 2017). We did not evaluate individual species' responses during the spring stopover period which may provide us a better assessment of the influences cut-back borders have on avifauna during this period. Continual monitoring of the spring migration period coupled with habitat assessments will elucidate Neotropical migrant responses to cut-back border treatments and help fill knowledge gaps to improve full annual-cycle conservation for priority species of the central Appalachian region.

Managing wildlife communities via cut-back borders has widespread potential for wildlife conservation in the region. We hypothesized that young forest species would exhibit positive responses to wider, heavy-intensity cut-back border treatments while forest interior species, both birds and salamanders, would exhibit negative responses to wider, heavy-intensity cut-back border treatments. Implementation of cut-back borders along forest edges of ROW/WO

appeared to create suitable intermediate conditions between ROW/WO and mature forests for young forest species as well as forest interior species. Often there is a trade-off when managing wildlife habitats, between creating habitat for young forest communities or preserving habitat for mature forest and forest interior communities. Implementation of cut-back borders along abrupt forest edges created young forest conditions in the forest understory to the benefit of disturbance-dependent species while retaining sufficient canopy that preserved suitable habitat conditions for disturbance-avoidant species. We have shown here that cut-back borders have widespread potential as a management technique along abrupt forest edges within forest-dominated landscapes in situations where young forest species are desired but concerns exist about further diminishing habitat for forest interior species.

4.2. *Vegetation response in cut-back borders*

We assessed the effects of multiple topographic indices (Beers aspect, head load index [HLI], shade proportion index [SPI], and topographic relative moisture index [TRMI]) on vegetation responses between one-year and two-year post-treatment to determine if responses could be explained by indices to help inform cut-back border placement with respect to local topography and edge orientation. We found no single index that consistently received strong support in explaining variation of vegetation regeneration in cut-back borders. This could be a result of the small sampling window (one year of plant growth) used for analysis. However, it was clear that vegetation regeneration was greater on mesic slopes for understory vegetation cover and small midstory stems. These two habitat features are important vegetation components for young forest species because they provide nesting cover from predators and important food resources via leaf-occurring invertebrates during the breeding season and fruit during the post-breeding season. Considering understory vegetation cover is an important habitat feature for young forest species, placement of cut-back borders should be prioritized on mesic slopes.

We observed weak correlations between individual bird species abundances and vegetation variables. These weak relationships could indicate that loose species-vegetation relationships exist along forest edges, that relationships are strong but sporadic resulting from spatially complex interactions at forest edges (Laurance et al. 2007), that microsite-level vegetation variables alone do not explain species' responses, or that short sampling intervals limited our ability to detect any strong relationships. Continued monitoring may elucidate

species-vegetation relationships in cut-back borders to help develop vegetation-oriented guidelines for cut-back borders.

5. Management Implications

Cut-back borders were a useful approach for managing young forest and mature forest wildlife species along abrupt forest edges of ROW/WO. The majority (26 of 35) of our positive significant avian responses were in 15-m wide treatments suggesting this was the most effective treatment for managing forest birds along forest edges. We observed two positive responses and one negative response for birds in 30-m wide cut-back borders and a negative response by the eastern red-backed salamander. We observed four positive and two negative avian responses to 45-m wide cut-back borders, as well as a positive response by the eastern red-backed salamander. Additionally, we observed 16 positive avian responses to 14 m²/ha intensity cut-back borders and 18 positive avian responses to 4.5 m²/ha intensity cut-back borders. Continued monitoring to assess species-vegetation relationships over time will help elucidate important habitat characteristics that change temporally and that will inform land managers in developing long-term management goals for cut-back borders.

Considering 15-m cut-back borders elicited the highest proportion of significant responses from young forest and other avian species, this treatment width seems to be the most effective approach for managing wildlife habitat in the short-term, as long as target basal area is reduced to <14 m²/ha to stimulate understory and midstory vegetation. This is also the most aesthetically pleasing treatment. In areas of heavy human use, wider cut-back borders will likely prompt public pushback (personal observation) which can potentially limit further use of cut-back borders on public lands. Additionally, 15-m cut-back borders are the most economically viable treatment relative to 30-m or 45-m treatments, as managers could create wildlife habitat in 15-m borders in a single day versus using multiple days with multiple people to create wider borders. Considering the number of trees that need to be felled in the 15, 30, and 45-m treatments and the wildlife response generated from each treatment, 15-m cut-back borders are the most cost-effective treatment that maximizes short-term wildlife response.

Young forest wildlife communities have experienced continual declines in the region due to the lack of suitable habitat. Although ROW corridors were used by young forest species in some forested landscapes, vegetation control in ROW corridors (e.g., mowing, herbicide spraying) and their shape (i.e., long and narrow) often limit habitat suitability and availability for

many species (Askins 1994, Bulluck and Buehler 2006). These factors likely make conservation strategies for young forest species that are centered within ROW corridors inefficient.

Conversely, cut-back borders increased the amount of young forest habitat that surrounded ROW/WO and widened ROW/WO, effectively enhancing these areas for young forest avifauna. Further, cut-back borders did not negatively affect forest interior wildlife species due to the retention of canopy trees in conjunction with adjacent mature forest. The retention of canopy trees in cut-back borders effectively created a transitional area between ROW/WO and mature forest that softened a previously abrupt forest edge and simultaneously created habitat conditions suitable for young forest and forest interior wildlife.

With the continued development of energy infrastructure within the central Appalachian region, it is important that natural resource managers and scientists continue to find ways to incorporate ROWs into habitat management to meet conservation goals for at-risk species and communities. While much work has focused on passive management of ROWs, few studies have examined how active habitat management affects these areas for young forest wildlife species. We have provided information herein to fill this knowledge gap. Still, more research and longer-term studies will be necessary to develop sound management strategies that can inform managers and improve these approaches for long-term management. Study on how topography affects vegetation regeneration (and subsequent wildlife response) and non-native/invasive species spread will help inform managers on ways to promote healthy and robust forest systems adjacent to ROWs. Additionally, information on a variety of cut-back border widths and harvest intensities will provide managers with a greater arsenal of strategies to meet specific objectives in a variety of settings.

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7. Tables

Table 2-1. Data collection time-table for 11 cut-back border blocks at eight West Virginia Division of Natural Resource (WVDNR) wildlife management areas (WMA). “Blocks” in the table refers to number of block replicates at each site.

	2016	2017	2018	2019
<i>Right-of-way</i>				
Underground Gas Pipeline				
Burnsville Lake WMA (1 block)				
Overhead Utility Powerline	1 Year Pre-harvest	1 Year Post-harvest	2 Years Post-harvest	
Little Canaan WMA (2 blocks)				
<i>Wildlife Opening</i>				
Stonewall Jackson WMA (1 block)				
<i>Right-of-way</i>				
Underground Gas Pipeline				
Lewis Wetzel WMA (2 blocks)				
Overhead Utility Powerline		1 Year Pre-harvest	1 Year Post-harvest	2 Years Post-harvest
Buery Mountain WMA (2 blocks)				
<i>Wildlife Opening</i>				
Bluestone WMA (1 block)				
<i>Right-of-way</i>				
Underground Gas Pipeline				
Allegheny WMA (1 block)			1 Year Pre-harvest	1 Year Post-harvest
<i>Wildlife Opening</i>				
Sleepy Creek WMA (1 block)				

Table 2-2. Sampling schedule for assessing avian, salamander, and vegetation response in cut-back borders during 2016–2019.

April	May	June	July	August
Ruffed Grouse (Visit 1 and 2)				
American Woodcock				
	Songbird Stopover	Songbird Breeding (Visit 1 and 2)	Vegetation	
	Salamander (Visit 1 and 2)		Salamander (Visit 3)	

8. Figures

Figure 2-1. County map of West Virginia showing locations of wildlife management areas (WMA) where cut-back border plots were located (circles) as well as major cities (stars).

A.

B.

C.

Figure 2-2. Examples of pre-treatment forest edge habitat prior to management along an underground gas pipeline at Lewis Wetzel Wildlife Management Area (LWWMA) (A), an overhead utility powerline at Little Canaan Wildlife Management Area (LCWMA) (B), and a wildlife opening at Bluestone Wildlife Management Area (BSWMA) (C).

Figure 2-3. Cut-back border plot layout with avian and vegetation sampling point (black dots) and salamander sampling point (black +) placements within each treatment width (15, 30, and 45 m) along rights-of-ways (ROWs) and wildlife openings (WO). Each cut-back border plot extended 300 m (violet arrow) along the forest-ROW/WO boundary (dashed line), and was either 15, 30, or 45 m (blue arrows) into the forest in a perpendicular direction respective to the forest-ROW/WO boundary. Avian and salamander sampling points were located 15 m (green arrow) off the forest-ROW/WO boundary. Avian sampling points were 150 m apart (red arrow) and 75 m (gray arrow) from cut-back border plot ends. Salamander sampling points were 75 m apart (pink arrow) and 37.5 m from plot ends. No salamander sampling occurred in the 15-m cut-back border treatments. Sampling point locations were consistent across all cut-back border widths with respect to forest-ROW/WO boundary (green arrow), intra-plot paired point counts (red arrow), and cut-back border plot ends (gray arrow).

Figure 2-4. Diagram of nested circular sampling plot for sampling vegetation variables in cut-back borders. Each cut-back border plot contained two vegetation sampling plots located at avian sampling point-count locations (see Figure 3). Within the 0.04 ha (11.3 m radius) plot, overstory tree variables (e.g., overstory cover and basal area) were measured. Within the 0.01 ha (5.6 m radius) plot, counts of midstory stems <11.4 cm DBH and ≥ 1.4 m in height were grouped into two size categories (0–5 cm DBH and 5.1–11.3 cm DBH). Within the 0.004 ha (3.6 m radius) plot, ground cover was estimated.

Figure 2-5. Example of a reading from 25 cm x 25 cm transparent plexiglass to estimate overstory cover in cut-back border vegetation sampling plot (Hachè et al. 2013). Plexiglass panel is divided into a 5 cm x 5 cm grid. Cells that are covered, to the nearest $\frac{1}{2}$ cell, by overstory (delineated at ≥ 5 m above ground) are counted towards an overstory closure value.

Figure 2-6. Example of differences in solar insolation based on plot location and edge orientation relative to the sun's position. In this scenario, Plot 1 receives full sunlight during the growing season compared to Plot 2 which is in the shadow of the adjacent forest. These differences in plot location could influence successional trajectories of vegetation in cut-back borders over time.

Figure 2-7. Model-averaged predictions of percent change of vegetation variables in cut-back borders between one-year post-treatment and two-year post-treatment in relation to relative soil conditions (i.e., xeric and mesic). We graphically represent our continuous soil moisture index on a relative scale from xeric to mesic soil conditions. Black line indicates predicted mean change and blue space indicates 95% confidence intervals.

Figure 2-8. Plot-level estimated occupancy and 95% confidence intervals of ruffed grouse detected within 100-m radius of sampling points in cut-back borders. There were no differences in occupancy rates between years within treatments based on 95% credible intervals.

Year ● 1-yr Post-treatment ● 2-yr Post-treatment

Figure 2-9. Plot-level estimated abundance and 95% confidence intervals of young forest focal songbird species during the summer breeding season. Differences in abundance within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 2-10. Plot-level estimated abundance and 95% confidence intervals of interior-edge focal songbird species during the summer breeding season. Differences in abundance within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 2-11. Plot-level estimated abundance and 95% confidence intervals of forest gap focal songbird species during the summer breeding season. Differences in abundance within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 2-12. Plot-level estimated abundance and 95% confidence intervals of forest interior focal songbird species during the summer breeding season. Differences in abundance within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 2-13 continued.

Figure 2-13. Cut-back border harvest intensity main effects. Differences in abundance or richness within cut-back border factor levels are indicated by alphabetical notation based on non-overlapping 95% credible intervals. Only significant main effects are reported.

Figure 2-14. Cut-back border width main effects. Differences in abundance or richness within cut-back border factor levels are indicated by alphabetical notation based on non-overlapping 95% credible intervals. Only significant main effects are reported.

Figure 2-15. Correlation biplot from canonical correspondence analysis (CCA) output. Vegetation variables include: elevation, understory cover 1–2 m in height (Foliage2), ground cover, number of live trees <28.0 cm DBH (Md.Trees), overstory cover, snags, topographic relative moisture index (TRMI), and woody stems 5–11.3 cm DBH (Midstory.Stem2). Avian species are color coded to indicate habitat guild groupings (red = young forest, blue = interior-edge, green = forest gap, purple = forest interior, orange = predators/parasites) and referenced by their 4-letter USGS alpha code (see Appendix 1-A).

Figure 2-16. Plot-level species richness and 95% confidence intervals of the avian community in cut-back border treatments during spring stopover. Differences in species richness within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 2-17. Plot-level apparent species richness and 95% confidence intervals of avian habitat guilds during the summer breeding season. Differences in species richness within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 2-18. Plot-level apparent species richness and 95% confidence intervals of avian guilds during the summer breeding season. Differences in species richness within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 2-19. Plot-level relative abundance and 95% confidence intervals of eastern red-backed salamanders (*Plethodon cinereus*). Differences in relative abundance within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 2-20. Body condition index (BCI) of eastern red-backed salamander (*Plethodon cinereus*) in cut-back border treatments at BMWMA and LWWMA. Differences in BCI within cut-back border treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

9. Appendices

Appendix 2-A. Acreage managed by wildlife management area (WMA) and year.

Wildlife Management Area	Year	Acreage
Allegheny	Winter 2018-19	13.3 ac (5.4 ha)
Beury Mountain	Winter 2017-18	26.7 ac (10.8 ha)
Bluestone	Winter 2017-18	13.3 ac (5.4 ha)
Burnsville Lake	Winter 2016-17	13.3 ac (5.4 ha)
Lewis Wetzel	Winter 2017-18	26.7 ac (10.8 ha)
Little Canaan	Winter 2016-17	26.7 ac (10.8 ha)
Sleepy Creek	Winter 2018-19	13.3 ac (5.4 ha)
Stonewall Jackson	Winter 2016-17	13.3 ac (5.4 ha)

Year	Acreage
Winter 2016-17	53.3 ac (21.6 ha)
Winter 2017-18	66.7 ac (27.0 ha)
Winter 2018-19	26.6 ac (10.8 ha)
Total	146.6 ac (59.4 ha)

Appendix 2-B. Decision rules for analyses to assess avian, salamander, and vegetation response in cut-back borders.

Analysis	Response Variable(s)	Predictor Variables	Detection type	Distance criteria	Demography criteria	Quantity criteria
Vegetation regeneration potential in cut-back borders (General linear models)	<ul style="list-style-type: none"> • Midstory stems 0–5 cm DBH • Midstory stems 5.1–11.3 cm DBH • Understory cover 0–1 m • Understory cover 1–2 m • Ground cover 	<ul style="list-style-type: none"> • Shade proportion index (SPI) • Heat load index (HLI) • Topographic relative moisture index (TRMI) • Beers aspect • Overstory cover 	Not Applicable	Not Applicable	Not Applicable	Not Applicable
Ruffed grouse (Hierarchical occupancy model)	<ul style="list-style-type: none"> • Presence/absence of grouse per plot 	<ul style="list-style-type: none"> • Cut-back border factor levels <ul style="list-style-type: none"> ○ 15 m ○ 30 m ○ 45 m ○ 14 m²/ha ○ 4.5 m²/ha • Year 	<ul style="list-style-type: none"> • Drumming individuals • Opportunistic detections 	<ul style="list-style-type: none"> • 0–25 m • >25–50 m • >50–75 m • >75–100 m 	<ul style="list-style-type: none"> • Males • Females • Unknown 	None
American woodcock (none)	<ul style="list-style-type: none"> • Presence/absence of woodcock 	No analysis executed (lack of data)	<ul style="list-style-type: none"> • Peenting individuals 	No distance criteria (i.e., unlimited radius)	<ul style="list-style-type: none"> • Males • Females • Unknown 	None
Spring stopover songbird richness (Generalized linear mixed effects model [GLMM])	<ul style="list-style-type: none"> • Number of unique species per point 	<ul style="list-style-type: none"> • Cut-back border factor levels <ul style="list-style-type: none"> ○ 15 m ○ 30 m ○ 45 m ○ 14 m²/ha ○ 4.5 m²/ha • Year 	<ul style="list-style-type: none"> • Songs (males) • Calls (males and females) • Visuals (excluding flyovers) 	<ul style="list-style-type: none"> • 0–25 m • >25–50 m 	<ul style="list-style-type: none"> • Adults only • Males • Females • Unknown 	None
Focal breeding songbird abundance (N-mixture model)	<ul style="list-style-type: none"> • Number of unique individuals for each focal species per plot 	<ul style="list-style-type: none"> • Cut-back border factor levels <ul style="list-style-type: none"> ○ 15 m ○ 30 m ○ 45 m ○ 14 m²/ha ○ 4.5 m²/ha • Year 	<ul style="list-style-type: none"> • Songs (males) • Calls (males and females) 	<ul style="list-style-type: none"> • 0–25 m • >25–50 m 	<ul style="list-style-type: none"> • Audio • Visuals (excludes flyovers) 	<ul style="list-style-type: none"> • ≥40 detections • ≥0.200 detection probability

Appendix 2-B continued.

Analysis	Response Variable(s)	Predictor Variables	Detection type	Distance criteria	Demography criteria	Quantity criteria
Avian-vegetation correlation in cut-back borders (Canonical Correspondence Analysis)	• Number of individuals for each songbird species	<ul style="list-style-type: none"> • Basal area • Overstory cover • Midstory stems 0–5 cm DBH • Midstory stems 5.1–11.3 cm DBH • Understory cover 0–1 m • Understory cover 1–2 m • Ground cover • Snags • Elevation • TRMI 	<ul style="list-style-type: none"> • Songs • Calls • Visuals (excluding flyovers) 	<ul style="list-style-type: none"> • 0–25 m • >25–50 m 	<ul style="list-style-type: none"> • Adults only • Males • Females • Unknown 	<ul style="list-style-type: none"> • ≥ 20 detections
Avian guild richness (GLMM)	• Number of unique species	<ul style="list-style-type: none"> • Cut-back border factor levels <ul style="list-style-type: none"> ○ 15 m ○ 30 m ○ 45 m ○ 14 m²/ha ○ 4.5 m²/ha • Year 	<ul style="list-style-type: none"> • Songs (males) • Calls (males and females) • Visuals (excluding flyovers) 	<ul style="list-style-type: none"> • 0–25 m • >25–50 m 	<ul style="list-style-type: none"> • Adults only • Males • Females • Unknown 	None
Salamander abundance (GLMM)	• Number of captures of eastern red-backed salamanders (<i>Plethodon cinereus</i>)	<ul style="list-style-type: none"> • Cut-back border factor levels <ul style="list-style-type: none"> ○ 30 m ○ 45 m ○ 14 m²/ha ○ 4.5 m²/ha • Year 	<ul style="list-style-type: none"> • Captured underneath marked cover object (excluding recaptures) 	Not Applicable	<ul style="list-style-type: none"> • Adults and juveniles • Males • Females 	None
Salamander Body Condition Index (GLMM)	• Body condition index of eastern red-backed salamanders	<ul style="list-style-type: none"> • Cut-back border factor levels <ul style="list-style-type: none"> ○ 30 m ○ 45 m ○ 14 m²/ha ○ 4.5 m²/ha • Year 	<ul style="list-style-type: none"> • Captured underneath marked cover object (excluding recaptures) 	Not Applicable	<ul style="list-style-type: none"> • Adults and juveniles • Males • Females 	None

Appendix 2-C. Total detections for songbird species in cut-back borders heard within 50-m of point count. Auditory detection type indicates which vocalizations were used to determine species presence. An “*” indicates species was included in species-specific analysis, and “^” indicates species was included in the community CCA. All species listed were included in guild richness analyses. See Appendix 1-A for scientific names of species and guild assignments. Species of conservation priority (SCP) are those identified by Appalachian Mountains Joint Venture (AMJV), Partners-in-Flight (PIF) Appalachian region (BCR 28), or the West Virginia Division of Natural Resources (WVDNR) State Wildlife Action Plan (SWAP). Guild codes are: FG = forest gap, FI = forest interior, IE = interior-edge, YF = young forest, PP = predator/parasite.

Common Name	Guild	Auditory Detection Type	SCP	Count	Detection Probability
Acadian Flycatcher^	FG	Song/Call	X	36	—
Alder Flycatcher	YF	Song/Call	X	3	—
American Crow	PP	Call		6	—
American Goldfinch*^	YF	Song/Call		68	0.221
American Redstart*^	IE	Song		86	0.222
American Robin^	IE	Song		37	—
Baltimore Oriole	IE	Song		7	—
Black-and-white Warbler*^	FI	Song	X	103	0.276
Black-capped Chickadee	IE	Song		22	—
Black-throated Blue Warbler^	FI	Song	X	39	—
Black-throated Green Warbler*^	FG	Song		52	0.310
Blue-gray Gnatcatcher^	IE	Song/Call		92	0.231
Blue-headed Vireo^	FI	Song		71	0.196
Blue-winged Warbler^	YF	Song	X	31	—
Blue Jay^	PP	Call		56	0.177
Brown-headed Cowbird^	PP	Song/Call		38	—
Brown Creeper	FI	Song	X	10	—
Brown Thrasher	YF	Song		5	—
Canada Warbler^	IE	Song	X	27	—
Carolina Chickadee	IE	Song		23	—
Carolina Wren*^	YF	Song		58	0.316
Cedar Waxwing^	IE	Song/Call		88	0.184
Cerulean Warbler^	FG	Song	X	44	0.230
Chestnut-sided Warbler^	YF	Song		31	—
Chipping Sparrow	YF	Song	X	13	—
Common Raven	PP	Call		3	—
Common Yellowthroat*^	YF	Song		70	0.343
Dark-eyed Junco*^	IE	Song		53	0.187
Downy Woodpecker^	IE	Call		24	—
Eastern Phoebe	IE	Song		8	—

Appendix 2-C continued.

Common Name	Guild	Auditory Detection Type	SCP	Count	Detection Probability
Eastern Towhee*^	YF	Song	X	208	0.230
Eastern Wood-pewee*^	IE	Song	X	65	0.364
Field Sparrow^	YF	Song	X	36	—
Gray Catbird^	YF	Song/Call		26	—
Great Crested Flycatcher	IE	Song/Call		12	—
Hairy Woodpecker^	FI	Call		26	—
Hermit Thrush	IE	Song		1	—
Hooded Warbler*^	FG	Song	X	131	0.277
House Wren	IE	Song		8	—
Indigo Bunting*^	YF	Song		205	0.250
Kentucky Warbler^	FG	Song	X	28	—
Least Flycatcher	FI	Song/Call	X	2	—
Louisiana Waterthrush	FI	Song	X	7	—
Magnolia Warbler	FI	Song		9	—
Mourning Dove	IE	Song/Call		18	—
Northern Cardinal*^	IE	Song/Call		95	0.234
Northern Flicker	IE	Call	X	11	—
Northern Mockingbird	IE	Song		2	—
Northern Parula	IE	Song		9	—
Orchard Oriole	YF	Song		6	—
Ovenbird*^	FI	Song		181	0.271
Pileated Woodpecker^	FI	Call		23	—
Pine Warbler	FI	Song		5	—
Prairie Warbler	YF	Song	X	2	—
Red-bellied Woodpecker^	IE	Call		37	—
Red-breasted Nuthatch	FI	Song/Call		1	—
Red-eyed Vireo*^	FG	Song		225	0.240
Rose-breasted Grosbeak	IE	Song	X	16	—
Ruby-throated Hummingbird	IE	Call		6	—
Scarlet Tanager^	FI	Song	X	104	0.154
Song Sparrow	IE	Song		9	—
Swainson's Thrush	FI	Song		1	—
Tufted Titmouse^	FI	Song/Call		77	0.160
Veery^	IE	Song	X	12	—
White-breasted Nuthatch^	FG	Song/Call		22	—
White-eyed Vireo	YF	Song		14	—
Willow Flycatcher	YF	Song/Call		1	—
Winter Wren	FI	Song		3	—
Wood Thrush^	FI	Song	X	61	0.241

Appendix 2-C continued.

Common Name	Guild	Auditory Detection Type	SCP	Count	Detection Probability
Worm-eating Warbler^	FI	Song	X	34	—
Yellow-bellied Sapsucker	IE	Call	X	3	—
Yellow-billed Cuckoo	IE	Song/Call	X	17	—
Yellow-breasted Chat	YF	Song	X	13	—
Yellow-throated Vireo^	IE	Song		54	0.190
Yellow-throated Warbler	FI	Song	X	4	—
Yellow Warbler	YF	Song		5	—

Species that were detected outside of 50-m during point count surveys included Barn Swallow, Barred Owl, Belted Kingfisher, Black-billed Cuckoo, Broad-winged Hawk, Canada Goose, Chimney Swift, Common Grackle, Cooper's Hawk, Eastern Kingbird, Eastern Whip-poor-will, Great-horned, Mourning Warbler, Red-shouldered Hawk, Red-tailed Hawk, Swainson's Warbler, Tree Swallow, Warbling Vireo, and Wild Turkey.

Appendix 2-D. Avian guild descriptions.

Guild	Description
Habitat guild	
Young Forest	Species that are primarily associated with recently disturbed forests during the breeding season.
Interior-edge	Species that can occupy a broad range of forest successional stages during the breeding season. This group is sometimes referred to as ‘forest generalists’. This group is considered insensitive to forest edge habitats (Ries and Sisk 2010).
Forest Gap	Species that are associated with mature forest interiors but prefer complex canopy structures (e.g., natural tree-fall gaps) and small canopy openings.
Forest Interior	Species that are primarily associated with mature forests during the breeding season. This group is considered highly sensitive to forest edges (Ries and Sisk 2010).
Foraging guild	
Aerial Insectivore	Species whose primary foraging behavior consists of catching insects during aerial forays, as opposed to catching insects while gleaning foliage or under tree bark.
Nesting guild	
Cavity	Species whose primary nesting substrate is in a tree (live or dead) cavity. This group is comprised of primary (i.e., species who excavate their own cavities; e.g., woodpeckers) and secondary (i.e., species who use previously excavated or naturally formed cavities; e.g., chickadees) cavity nesters.
Predator/Parasite guild	Species who predate on avian nests (e.g., blue jays and common ravens) or parasitize avian nests (e.g., brown-headed cowbirds).

Appendix 2-E. Model outputs for assessing effects of topographic variables on five vegetation variables between one-year post-treatment and two-year post-treatment in cut-back borders. HLI = heat load index, SPI = shade proportion index, TRMI = topographic relative moisture index, and K = number of parameters. We report the top six models for each model set.

Ground Cover				
Model	K	ΔAIC_c	$AIC_c Wt$	Cumulative Wt
Beers aspect * Canopy cover	5	-	0.24	0.24
SPI * Canopy cover	5	0.15	0.22	0.46
HLI * Canopy cover	5	0.17	0.22	0.68
TRMI * Canopy cover	5	0.25	0.21	0.89
(SPI + Beers aspect) * Canopy cover	7	3.91	0.03	0.92
Null	2	4.11	0.03	0.95
Understory Cover 0–1 m				
Model	K	ΔAIC_c	$AIC_c Wt$	Cumulative Wt
TRMI	3	-	0.33	0.33
Null	2	1.29	0.17	0.51
TRMI * Canopy cover	5	1.43	0.16	0.67
Beers aspect * Canopy cover	5	2.99	0.07	0.74
SPI	3	3.30	0.06	0.80
Beers aspect	3	3.31	0.06	0.86
Understory Cover >1–2 m				
Model	K	ΔAIC_c	$AIC_c Wt$	Cumulative Wt
SPI * Canopy cover	5	-	0.40	0.40
SPI	3	1.23	0.22	0.62
Null	2	3.09	0.09	0.71
SPI + Beers aspect	4	3.18	0.08	0.79
(SPI + Beers aspect) * Canopy cover	7	3.63	0.07	0.85
HLI	3	4.21	0.05	0.90
Midstory Stems 0–5.1 cm DBH				
Model	K	ΔAIC_c	$AIC_c Wt$	Cumulative Wt
SPI * Canopy cover	5	-	0.73	0.73
(SPI + Beers aspect) * Canopy cover	7	2.03	0.27	1.00
TRMI * Canopy cover	5	14.84	0.00	1.00
Beers aspect * Canopy cover	5	16.98	0.00	1.00
HLI * Canopy cover	5	17.75	0.00	1.00
Null	2	18.28	0.00	1.00
Midstory Stems >5.1–11.3 cm DBH				
Model	K	ΔAIC_c	$AIC_c Wt$	Cumulative Wt
SPI	3	-	0.25	0.25
HLI	3	0.16	0.23	0.48
Null	2	0.65	0.18	0.66
SPI + Beers aspect	4	2.01	0.09	0.75
TRMI	3	2.47	0.07	0.83
Beers aspect	3	2.52	0.07	0.90

Appendix 2-F. Vegetation summary statistics (mean $\pm$ standard error) in cut-back border treatments over time.

	Control	15 m, 14 m ² /ha	15 m, 4.5 m ² /ha	30 m, 14 m ² /ha	30 m, 4.5 m ² /ha	45 m, 14 m ² /ha	45 m, 4.5 m ² /ha
Basal area (m²/ha)							
Pre-treatment	16.3 $\pm$ 1.9	16.7 $\pm$ 1.9	14.8 $\pm$ 1.4	20.00 $\pm$ 1.5	15.8 $\pm$ 2.0	19.7 $\pm$ 1.2	18.3 $\pm$ 1.6
One-year Post-treatment	16.3 $\pm$ 1.9	9.4 $\pm$ 1.1	10.5 $\pm$ 1.0	10.9 $\pm$ 1.5	7.4 $\pm$ 1.7	11.4 $\pm$ 1.1	5.2 $\pm$ 1.2
Two-year Post-treatment	15.6 $\pm$ 2.3	7.5 $\pm$ 1.3	9.3 $\pm$ 1.3	8.5 $\pm$ 1.3	4.0 $\pm$ 1.0	9.7 $\pm$ 1.2	4.5 $\pm$ 1.0
Percent overstory cover							
Pre-treatment	88.86 $\pm$ 3.6	95.5 $\pm$ 2.7	96.6 $\pm$ 1.5	84.4 $\pm$ 5.6	87.2 $\pm$ 5.7	94.6 $\pm$ 3.1	96.1 $\pm$ 1.3
One-year Post-treatment	88.86 $\pm$ 3.6	64.2 $\pm$ 6.0	72.5 $\pm$ 5.4	70.7 $\pm$ 6.3	30.7 $\pm$ 6.8	67.2 $\pm$ 6.4	19.6 $\pm$ 5.4
Two-year Post-treatment	87.83 $\pm$ 4.3	73.4 $\pm$ 8.5	73.7 $\pm$ 7.5	67.8 $\pm$ 8.1	39.5 $\pm$ 8.4	64.1 $\pm$ 7.5	25.9 $\pm$ 8.0
Midstory stems 0–5.1 cm DBH (stems/ha)							
Pre-treatment	276.1 $\pm$ 57.9	297.6 $\pm$ 54.6	325.0 $\pm$ 117.6	262.5 $\pm$ 47.2	363.6 $\pm$ 70.1	203.6 $\pm$ 51.4	232.5 $\pm$ 54.6
One-year Post-treatment	276.1 $\pm$ 57.9	228.4 $\pm$ 64.7	250.0 $\pm$ 120.3	278.4 $\pm$ 98.5	150.0 $\pm$ 36.7	120.5 $\pm$ 44.5	107.9 $\pm$ 41.5
Two-year Post-treatment	308.3 $\pm$ 68.0	394.4 $\pm$ 81.8	415.3 $\pm$ 145.7	345.8 $\pm$ 99.1	373.6 $\pm$ 76.3	212.5 $\pm$ 44.3	308.8 $\pm$ 86.2
Midstory stems >5.1–11.3 cm DBH (stems/ha)							
Pre-treatment	122.7 $\pm$ 14.5	139.3 $\pm$ 18.6	140.9 $\pm$ 25.7	118.2 $\pm$ 21.3	164.8 $\pm$ 25.0	108.3 $\pm$ 16.8	78.8 $\pm$ 12.7
One-year Post-treatment	122.7 $\pm$ 14.5	81.8 $\pm$ 16.0	108.0 $\pm$ 26.5	69.3 $\pm$ 13.0	36.4 $\pm$ 9.6	50.0 $\pm$ 12.5	19.3 $\pm$ 5.9
Two-year Post-treatment	123.6 $\pm$ 15.2	98.6 $\pm$ 20.6	102.8 $\pm$ 30.4	93.1 $\pm$ 25.1	33.3 $\pm$ 8.6	76.4 $\pm$ 22.5	32.4 $\pm$ 7.7
Percent ground cover							
Pre-treatment	52.7 $\pm$ 5.2	50.5 $\pm$ 6.0	46.4 $\pm$ 4.2	50.9 $\pm$ 5.5	45.5 $\pm$ 5.8	41.9 $\pm$ 4.6	42.0 $\pm$ 4.1
One-year Post-treatment	52.7 $\pm$ 5.2	48.2 $\pm$ 4.9	43.6 $\pm$ 4.1	44.5 $\pm$ 4.7	46.4 $\pm$ 5.8	47.3 $\pm$ 6.0	46.4 $\pm$ 5.8
Two-year Post-treatment	55.6 $\pm$ 5.5	72.2 $\pm$ 5.4	73.3 $\pm$ 5.8	67.8 $\pm$ 4.6	75.6 $\pm$ 6.2	62.2 $\pm$ 6.4	85.9 $\pm$ 4.8
Percent understory cover 0–1 m							
Pre-treatment	80.0 $\pm$ 6.2	91.4 $\pm$ 4.9	91.8 $\pm$ 4.1	80.9 $\pm$ 5.2	86.4 $\pm$ 4.8	69.5 $\pm$ 6.6	78.0 $\pm$ 6.8
One-year Post-treatment	80.0 $\pm$ 6.2	93.6 $\pm$ 3.3	94.5 $\pm$ 3.0	94.5 $\pm$ 3.3	95.5 $\pm$ 3.2	85.5 $\pm$ 5.6	89.1 $\pm$ 4.9
Two-year Post-treatment	83.3 $\pm$ 6.1	97.8 $\pm$ 2.2	100.0 $\pm$ 0.0	97.8 $\pm$ 2.2	97.8 $\pm$ 2.2	96.7 $\pm$ 2.4	100.0 $\pm$ 0.0
Percent understory cover >1–2 m							
Pre-treatment	58.2 $\pm$ 7.1	80.0 $\pm$ 5.9	74.5 $\pm$ 5.9	68.2 $\pm$ 6.5	73.6 $\pm$ 6.4	58.1 $\pm$ 6.9	62.0 $\pm$ 6.6
One-year Post-treatment	58.2 $\pm$ 7.1	76.4 $\pm$ 7.3	68.2 $\pm$ 6.5	76.4 $\pm$ 6.4	73.6 $\pm$ 7.7	63.6 $\pm$ 7.9	64.5 $\pm$ 7.0
Two-year Post-treatment	57.8 $\pm$ 8.2	85.6 $\pm$ 6.4	98.9 $\pm$ 1.1	84.4 $\pm$ 6.8	93.3 $\pm$ 4.0	75.6 $\pm$ 7.7	90.6 $\pm$ 5.5

Appendix 2-G. Model posterior predictive check (Bayesian p -value) and model performance (deviance information criterion [DIC]; lower is better) for focal bird species, avian guild, and eastern red-backed salamander (*Plethodon cinereus*) treatment-time interaction and main effects model analyses. Bayesian p -value is the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic. We assumed reasonable model fit with a Bayesian p -value >0.1 and <0.9 .

Focal birds		Bayesian p -value		DIC	
Common name	Scientific name	Treatment-Time	Main Effects	Treatment-Time	Main Effects
American goldfinch	<i>Spinus tristis</i>	0.305	0.303	8073.4	8181.1
American redstart	<i>Setophaga ruticilla</i>	0.407	0.413	8138.3	8158.2
Black-and-white warbler	<i>Mniotilta varia</i>	0.420	0.437	8341.3	8412.5
Black-throated green warbler	<i>Setophaga virens</i>	0.412	0.338	7776.1	7845.1
Blue-gray gnatcatcher	<i>Poliophtila caerulea</i>	0.343	0.323	8195.6	8180.8
Carolina wren	<i>Thryothorus ludovicianus</i>	0.300	0.247	8021.5	8178.5
Cerulean warbler	<i>Setophaga cerulean</i>	0.517	0.533	7741.7	7760.2
Common yellowthroat	<i>Geothlypis trichas</i>	0.313	0.338	7970.3	7966.9
Eastern towhee	<i>Pipilo erythrophthalmus</i>	0.373	0.403	8872.6	8892.3
Eastern wood-pewee	<i>Contopus virens</i>	0.410	0.383	8020.4	8022.2
Hooded warbler	<i>Setophaga citrina</i>	0.472	0.410	8681.3	8590.3
Indigo bunting	<i>Passerina cyanea</i>	0.528	0.547	8934.2	9029.0
Northern cardinal	<i>Cardinalis cardinalis</i>	0.352	0.453	8098.0	8062.5
Ovenbird	<i>Seiurus aurocapilla</i>	0.368	0.363	8617.9	8589.9
Red-eyed vireo	<i>Vireo olivaceus</i>	0.403	0.413	9327.7	9098.1
Ruffed grouse	<i>Bonasa umbellus</i>	0.528	0.547	4935.8	4941.9
Wood thrush	<i>Hylocichla mustelina</i>	0.323	0.340	8023.6	8003.9
Avian guilds					
Spring stopover		0.742	0.733	4568.2	1692.7
Young forest habitat guild		0.560	0.610	9946.7	9853.3
Interior-edge habitat guild		0.732	0.803	1465.4	1464.9
Forest gap habitat guild		0.600	0.658	9566.6	9575.0
Forest interior habitat guild		0.447	0.452	9669.2	9734.2
Aerial insectivore foraging guild		0.573	0.593	587.2	604.3
Cavity nesting guild		0.335	0.343	1079.9	1089.9

Appendix 2-G continued.

	Bayesian p -value		DIC	
	Treatment-Time	Main Effects	Treatment-Time	Main Effects
Avian guild				
Predator/parasite guild	0.353	0.340	498.3	486.5
Species of Conservation Priority	0.580	0.585	100021.9	10020.1
Eastern red-backed salamander				
Relative abundance	0.510	0.410	705.7	682.8
Body condition index	0.550	0.520	206.2	203.7

Appendix 2-H. Output table from canonical correspondence analysis (CCA) assessing avian species correlations with vegetation variables in cut-back borders. Numeric values in each column (CCA1, CCA2, etc.) indicate correlation with that canonical axis, with higher absolute numbers (positive or negative) being more strongly associated within that axis (e.g., CCA1 is strongly influenced by elevation, hence “Elevation” has the highest absolute value of vegetation variables in CCA1, and is correlated with CSWA, COYE, RUGR, etc. which have the highest absolute value of species). Positive correlations between vegetation variables and species are indicated with similar signs (“+” or “-”) within each column.

Predictor Variables		CCA1	CCA2	CCA3	CCA4	CCA5	CCA6
Elevation		-0.97	0.02	0.05	0.04	0.06	-0.12
Topographic Relative Moisture Index (TRMI)		-0.35	0.67	-0.18	-0.08	0.49	-0.33
Understory cover >1–2 m		0.05	0.60	0.32	0.15	-0.02	0.48
Counts of trees <28.0 cm DBH		-0.31	0.08	-0.64	0.24	-0.27	0.31
Basal Area		0.00	-0.47	-0.41	0.40	0.37	0.25
Midstory stems >5.1–<11.3 cm DBH		-0.34	0.23	-0.04	0.32	-0.01	0.37
Snags		-0.01	0.22	-0.72	-0.13	-0.11	0.24
Overstory cover		0.10	-0.12	-0.44	0.83	0.19	0.00
Ground cover		0.22	0.48	0.23	0.18	-0.57	-0.25
Habitat Guild	Species	CCA1	CCA2	CCA3	CCA4	CCA5	CCA6
Young Forest	AMGO	0.00	0.20	-0.20	-0.09	-0.07	0.13
	BWWA	0.76	0.62	0.29	-0.19	-0.23	0.13
	CARW	0.46	0.23	-0.04	0.09	-0.26	0.18
	CSWA	-1.43	0.48	0.00	-0.08	-0.29	0.05
	COYE	-0.95	0.29	0.04	0.05	-0.10	0.02
	EATO	-0.11	0.12	0.26	-0.14	-0.06	-0.16
	FISP	0.55	-0.13	0.31	0.11	-0.07	-0.08
	GRCA	0.04	-0.22	-0.17	-0.35	0.46	-0.02
	INBU	0.39	0.04	0.22	-0.11	-0.05	0.06
	RUGR	-0.82	0.30	0.01	-0.34	0.08	-0.04
Interior-edge	AMRE	0.34	0.34	0.02	0.25	0.14	-0.13
	AMRO	0.18	-0.59	-0.09	0.12	-0.12	0.19
	BGGN	0.57	0.48	-0.11	-0.11	0.14	-0.08
	CAWA	-1.74	0.14	0.44	-0.38	0.24	0.31
	CEDW	-0.14	-0.19	0.06	-0.32	-0.12	0.09
	DEJU	-1.06	0.01	-0.12	-0.52	0.09	-0.26
	DOWO	0.62	0.21	-0.47	-0.13	-0.27	0.34
	EAWP	0.33	-0.69	-0.10	-0.03	-0.13	-0.05
	NOCA	0.63	0.65	-0.19	0.03	-0.18	0.12
	RBWO	0.41	-0.51	0.00	0.23	-0.12	-0.19
Forest Gap	YTVI	-0.16	0.02	0.08	0.22	-0.18	-0.35
	ACFL	0.55	0.38	-0.18	0.09	0.84	0.20
	BTNW	-1.45	0.32	-0.07	0.27	0.15	-0.22
	CERW	0.36	0.06	0.43	0.02	0.17	0.01
	HOWA	0.17	-0.28	0.35	0.20	0.13	0.16
	KEWA	-0.06	0.18	0.04	0.20	0.00	-0.06
	REVI	0.10	-0.06	-0.12	0.05	-0.05	-0.09
	WBNU	0.50	-0.17	0.21	-0.10	0.19	0.05
Forest Interior	BAWW	-0.03	-0.27	0.34	0.00	0.00	0.06
	BTBW	-1.55	0.53	0.03	0.62	0.02	0.45
	BHVI	-0.38	-0.57	-0.14	0.14	0.13	0.03
	HAWO	0.04	-0.32	-0.14	-0.21	0.26	0.38
	OVEN	-0.53	-0.32	-0.29	0.01	-0.06	0.04
	PIWO	0.52	0.01	-0.25	0.21	0.50	0.00
	SCTA	0.18	-0.01	-0.10	0.21	0.03	-0.24
	TUTI	0.50	-0.10	-0.16	0.13	-0.25	0.00
	WEWA	0.43	0.00	-0.84	-0.23	0.10	0.09
	WOTH	0.20	-0.26	-0.04	-0.13	0.19	-0.14
Predator/Parasite	BHCO	0.35	0.23	-0.06	-0.10	0.24	-0.18
	BLJA	0.10	-0.12	0.01	-0.04	-0.08	0.24

Appendix 2-I. Capture breakdown of salamander species by cut-back border treatment over the study period. Table excludes recaptures.

	Control	30 m 14 m ² /ha	30 m 4.5 m ² /ha	45 m 14 m ² /ha	45 m 4.5 m ² /ha
Eastern Red-backed Salamander (<i>Plethodon cinereus</i>)					
Pre-treatment	12	5	7	7	13
One-year Post-treatment	14	4	9	13	21
Two-year Post-treatment	27	9	17	11	18
Mountain Dusky Salamander (<i>Desmognathus ochrophaeus</i>)					
Pre-treatment	0	0	1	0	0
One-year Post-treatment	0	0	0	0	0
Two-year Post-treatment	0	0	2	0	0
Northern Red Salamander (<i>Pseudotriton ruber</i>)					
Pre-treatment	0	0	0	0	0
One-year Post-treatment	0	0	0	0	0
Two-year Post-treatment	1	0	0	0	0
Slimy Salamander (<i>P. glutinosus</i>)					
Pre-treatment	0	0	3	2	5
One-year Post-treatment	5	1	3	0	0
Two-year Post-treatment	1	1	1	3	2
Wehrle's Salamander (<i>P. wehrlei</i>)					
Pre-treatment	0	0	0	1	2
One-year Post-treatment	1	3	1	0	0
Two-year Post-treatment	1	1	0	0	1

Appendix 2-J. Total number annual detections in cut-back border treatments for species of conservation concern in the Appalachian region (Bird Conservation Region 28; PIF 2019) that did not meet the detection count criteria (≥ 40 detections) or the detection probability criteria (≥ 0.200 probability of detection).

	Control	15 m 14 m ² /ha	15 m 4.5 m ² /ha	30 m 14 m ² /ha	30 m 4.5 m ² /ha	45 m 14 m ² /ha	45 m 4.5 m ² /ha
Acadian Flycatcher							
Pre-treatment	1	1	1	4	1	0	3
One-year Post-treatment	4	3	0	1	0	4	3
Two-year Post-treatment	1	0	1	6	0	0	2
Black-throated Blue Warbler							
Pre-treatment	3	3	6	6	5	2	3
One-year Post-treatment	0	1	1	1	1	0	2
Two-year Post-treatment	0	2	0	1	1	0	1
Blue-winged Warbler							
Pre-treatment	0	0	2	1	2	1	0
One-year Post-treatment	0	0	1	1	3	3	1
Two-year Post-treatment	1	1	6	1	3	3	1
Canada Warbler							
Pre-treatment	0	0	2	1	1	0	2
One-year Post-treatment	0	4	4	0	2	1	3
Two-year Post-treatment	0	0	1	1	1	2	2
Chipping Sparrow							
Pre-treatment	1	2	3	0	2	0	0
One-year Post-treatment	1	0	0	1	0	0	2
Two-year Post-treatment	0	0	0	0	0	1	0
Field Sparrow							
Pre-treatment	2	3	2	0	2	0	3
One-year Post-treatment	0	3	1	2	1	2	2
Two-year Post-treatment	3	0	1	3	5	1	0
Least Flycatcher							
Pre-treatment	0	0	0	0	0	0	0
One-year Post-treatment	0	0	0	0	1	0	0
Two-year Post-treatment	0	1	0	0	0	0	0

Appendix 2-J continued.

	Control	15 m 14 m ² /ha	15 m 4.5 m ² /ha	30 m 14 m ² /ha	30 m 4.5 m ² /ha	45 m 14 m ² /ha	45 m 4.5 m ² /ha
Louisiana Waterthrush							
Pre-treatment	0	0	0	1	0	0	0
One-year Post-treatment	0	1	0	1	0	0	0
Two-year Post-treatment	1	1	0	1	0	0	1
Prairie Warbler							
Pre-treatment	0	0	0	0	0	0	0
One-year Post-treatment	0	0	0	0	0	0	0
Two-year Post-treatment	0	1	1	0	0	0	0
Rose-breasted Grosbeak							
Pre-treatment	1	0	4	2	1	1	1
One-year Post-treatment	0	0	0	1	1	0	0
Two-year Post-treatment	1	0	1	1	0	1	0
Scarlet Tanager							
Pre-treatment	6	5	2	3	6	2	4
One-year Post-treatment	7	10	3	3	4	4	5
Two-year Post-treatment	6	7	3	9	5	5	5
Worm-eating Warbler							
Pre-treatment	0	2	1	4	3	2	2
One-year Post-treatment	3	0	5	1	1	1	3
Two-year Post-treatment	1	2	1	1	0	0	1
Yellow-breasted Chat							
Pre-treatment	1	0	0	0	1	0	1
One-year Post-treatment	0	0	0	0	1	0	1
Two-year Post-treatment	1	1	2	0	3	1	0
Yellow-throated Warbler							
Pre-treatment	0	1	0	0	0	0	0
One-year Post-treatment	0	2	1	0	0	0	0
Two-year Post-treatment	0	0	0	0	0	0	0

CHAPTER 3

AVIAN AND SALAMANDER RESPONSE TO NON-COMMERCIAL YOUNG FOREST MANAGEMENT TECHNIQUES

Abstract

Current forest conditions throughout West Virginia and the central Appalachian region indicate a young forest deficit which has negatively affected wildlife species associated with forest disturbance. Due to the prominence of commercial logging as a means to create young forest, forests of low- or poor-quality stocking are often overlooked and thus limit the management and conservation of young forest wildlife species. In order to promote habitat for young forest wildlife species, more opportunities for creating young forest habitat via forest management are greatly needed. In this chapter, we assessed three non-commercial young forest management treatments (clearcut-and-leave [CL], clearcut-and-windrow [CW], and hack-and-spray [HS]) for promoting young forest habitat. Similar to Chapter 2, the objectives of this chapter were to examine whether the non-commercial young forest management treatments increased habitat suitability for young forest wildlife species and to determine which treatments optimized the tradeoff between maximizing positive responses of disturbance-dependent species (i.e., young forest songbirds) and minimizing negative responses of disturbance-avoidant species (i.e., forest interior songbirds and woodland salamanders). Additionally, we included a cost-benefit analysis to quantify the tradeoffs between maximizing songbird richness (cavity nesting guild, forest gap habitat guild, forest interior habitat guild, and young forest habitat guild) and minimizing cost of forest management. Implementation of regeneration treatments induced a dichotomous response from the avian community. Young forest songbird species responded positively to the CL treatment and negatively to the HS treatment, while forest interior songbirds responded positively to the HS treatment and negatively to the CL treatment. Eastern red-backed salamander (*Plethodon cinereus*), a terrestrial salamander most suited to forest interiors, responded positively to the HS treatment immediately following treatment. Based on our cost-benefit analysis, the CL treatment was the optimum treatment for maximizing songbird species richness and minimizing cost of habitat management followed by the control, the HS treatment, and the CW treatment. These results suggest that non-commercial forest management can provide suitable habitat conditions for young forest songbirds but can negatively affect forest

interior songbirds, while other treatments retain suitable habitat conditions for forest interior species but do not provide immediate suitable habitat for young forest species.

1. Introduction

1.1. Introduction

Young forest wildlife species are experiencing precipitous population declines across the eastern US due to human-induced land-use modifications (e.g., farm abandonment, wildfire suppression, shifting disturbance regimes, urbanization) over the past century (Lorimer 2001, Trani et al. 2001). These anthropogenic modifications have resulted in the loss or conversion of young forest areas, primarily through the suppression of forest canopy disturbances such as tree cutting. However, canopy disturbances are an essential component of forest systems as they provide developmental stimuli to promote diverse plant species growth and resources to animals (McDermott and Wood 2010, Dey 2014). Without canopy disturbances, young forests will eventually mature into a later successional stage thereby eliminating breeding cover and foraging opportunities vital to young forest wildlife, a high conservation priority group in West Virginia and the central Appalachian region (King and Schlossberg 2014). Currently, West Virginia and much of the central Appalachian region is in a young forest deficit relative to historical conditions (Oswalt et al. 2010, Shifley and Thompson III 2011). Relative to conditions from the mid-century (ca. 1949), young forest area in West Virginia is down from 19% of all forestlands to 7% today (Morin et al. 2016). However, Lorimer (2001) states that due to exploitation logging, farm abandonment, and wildfires during the late nineteenth century young forests likely reached a peak of 55–60% of forestlands during this period but has since declined to 20% or less in many parts of eastern North America. Contemporary expectations of young forest levels are unlikely to be met using these historical baselines because they were outcomes of larger societal pressures which have changed over time. However, considering the scenario from Shifley and Thompson (2011) where forest age classes from 0–120 years old are evenly distributed, young forests (0–10 years old) would comprise 7.1% of all forestland (Note: Morin et al. 2016 used a different criterion than Shifley and Thompson III 2011 to categorize young forest). Using the 7.1% as a contemporary baseline for young forest area, West Virginia is currently in a young forest deficit with only 2.4% of forestland in this age class (or a 226,255-ha state-wide deficit).

While young forest area in West Virginia remains low, timber volume is concurrently increasing at twice the rate than what is being removed via harvesting. However, despite the

prominence of forests comprised of older, larger trees in West Virginia (93% of all forestland), only 38% of these forestlands are well stocked with trees of commercial importance (Morin et al. 2016). The lack of young forests and predominance of mature forests demonstrates the need for increased opportunities for young forest management in order to reduce the young forest deficit in the region (DeGraaf and Yamasaki 2003, Shifley and Thompson III 2011). Implementation of tree cutting, the major canopy disturbance agent in the region, will likely need to be expanded to reduce the young forest deficit currently present in West Virginia and throughout the central Appalachian region.

Numerous options exist for creating and managing young forest habitat, but commercial tree-cutting is the primary contemporary disturbance agent in the central Appalachian region (Cohen et al. 2016). Commercial logging is largely driven by the economic value of timber and thus, management and conservation of young forest species is greatly affected by the commercial logging industry (King and Schlossberg 2014, Demarais et al. 2017). However, in forested areas with poorly stocked or low-quality timber, forest management is often cost prohibitive for many commercial loggers. Without an economic incentive to cut trees, low-quality forestlands that would have otherwise increased the amount of young forest area remains in a forest stage that is unsuitable for young forest wildlife. Despite this limitation for conserving and managing young forest wildlife, non-commercial wildlife stand improvement (WSI) strategies can provide alternative approaches to managing wildlife habitat (Wilson et al. 1995, Masters et al. 1998). Wildlife stand improvement strategies focus on ecological capital rather than economic capital with the specific goal of managing forestlands for specific wildlife habitats, primarily those of high conservation value with limited regional distribution. However, economic costs of non-commercial WSI treatments need to be considered as no short-term economic output is produced (Shwiff et al. 2013). Land managers need to consider this aspect in order to implement WSI treatment as a stand-alone management technique or to potentially incorporate WSI treatments along with commercial timber sales.

Silviculture provides relevant land management applications and documented wildlife responses for WSI prescriptions to mimic in order to meet the objective of creating young forest wildlife habitat. Clearcutting removes all canopy trees and induces a complete turnover of the wildlife community, replacing mature forest species with young forest species (Duguay et al. 2001, Gram et al. 2003). Shelterwood harvest removes a portion of canopy trees which benefits

young forest wildlife while retaining enough canopy for mature forest species (Newell and Rodewald 2012, Sheehan et al. 2014, Perry et al. 2018). While these even-aged techniques create canopy openness by felling trees, recent emphasis has been placed on ecological forestry measures, where poor-quality or low-value trees are girdled or sprayed with herbicide then left standing, thereby creating snags, as opposed to cutting (Bütler et al. 2013). This technique benefits young forest species but maintains snags for ecologically important avian groups (e.g., cavity nesters; Cockle et al. 2012, Nappi et al. 2015). Both forest management techniques create habitat for young forest species. However, both incur different costs related to treatment implementation, where clearcutting techniques rely on heavy machinery and herbicide techniques via girdling and spraying rely on expensive herbicide.

Little has been explored on the most cost-effective means for creating young forest with a focus on wildlife habitat. In order to reduce the young forest deficit in West Virginia and the central Appalachian region, non-commercial approaches need further examination. Considering the dependence of young forest wildlife species on forest management, efforts to expand young forest habitat creation beyond the commercial realm is greatly needed.

1.2. *Chapter objectives and hypotheses*

The primary objective of this chapter was to assess avian and salamander individual species and community responses to three different non-commercial young forest management treatments. We used a before-after-control-impact (BACI) framework to examine two cutting treatments and one herbicide treatment to inform our inferences about avian and salamander responses to treatments over time. Additionally, we examined the tradeoffs between minimizing cost of treatment and maximizing avian community richness to inform management decisions using a cost-benefit analysis. For avifauna responses to treatments, we hypothesized that (1) young forest avifauna would respond positively to all regeneration treatments with greatest responses in the cutting treatments and smallest responses in the herbicide treatment, (2) forest interior avifauna would respond negatively to all regeneration treatments with greatest responses in the cutting treatments and smallest responses in the herbicide treatment, and (3) forest gap, cavity nesting, and bark foraging avifauna would respond positively to the herbicide treatment and negatively to the cutting treatments. For our cost-benefit analysis, we hypothesized that the herbicide treatment would score highest among the three treatments due to the low cost of implementation and the positive responses by young forest species while minimizing the

negative responses by forest interior species. The cutting treatments would score lower than the herbicide treatment due to the expected negative responses from the forest interior species as well as a higher cost of implementation. For woodland salamanders, we hypothesized that the herbicide treatment would negatively influence their relative abundance.

2. Methods

2.1. Study area

West Virginia lies within the Appalachian region, which is characterized by extensive forests, comprised of mixed mesophytic, Appalachian oak, transition Appalachian oak-northern hardwood, and oak-hickory-pine forest types (Küchler 1964). Elevations at study sites ranged from 300–1132 m. West Virginia’s humid continental climate is characterized by hot summers and cool to cold winters with average spring and fall temperatures of 10.0 °C and an average summer temperature of 28.0 °C. Annual precipitation levels are 76, 127, and 102 cm in the western, central, and eastern portions of the state, respectively, distributed uniformly throughout the year (NOAA 2019).

2.2. Study design

This study was conducted on six West Virginia Division of Natural Resources’ (WVDNR) wildlife management areas (WMA) located throughout central and eastern West Virginia (Figure 3-1; see Figures 1-7, 1-10, 1-12, 1-13, 1-14, and 1-15 for site-level maps). The six study sites included 16 block replicates (Table 3-1) located within mature forests. A single block replicate consisted of four independent experimental units with each randomly assigned to one of four experimental treatments. Wildlife stand improvement treatments (hereafter “regeneration stands”) included clearcut-and-leave (hereafter “CL”), clearcut-and-windrow (hereafter “CW”), hack-and-spray (hereafter “HS”), and a control with no canopy manipulation. All blocks contained all four treatments except for the three blocks at CBWMA which did not have any HS treatments (Appendix 3-A). Each regeneration stand was 4 ha (200 m x 200 m) in size and roughly square in shape, and all were ≥ 200 m apart to ensure sampling independence. Regeneration stands within a single block replicate were presumed to be subjected to similar biotic and abiotic environmental constraints.

Within the CL treatment, all trees ≥ 5 cm diameter-at-breast height (DBH) were cut and left on the ground where they fell within the plot. Within the CW treatment, all trees ≥ 5 cm

DBH) were cut and placed into linear piles (i.e., windrows) perpendicular to contours. Tree cutting was accomplished using a feller-buncher in both CL and CW treatments, and cutting began 1 September if a study site was >8 km from an Indiana bat (*Myotis sodalis*) roosting area, or 15 November if the study site was <8 km from an Indiana bat roosting area and was completed by 31 March.

Within the HS treatment, herbicide treatments of either stem injection or cut-stump were applied to non-desirable trees ≥ 5 cm DBH to mimic crop tree or shelterwood conditions (i.e., 9–14 m²/ha target basal area retention). Non-desirable tree species were considered to have low wildlife value (i.e., species with low mast production) and varied depending on species composition at each study site. WVDNR managers/foresters cruised HS treatments and selected leave trees based on species desirability. The stem injection treatment was applied to non-desirable tree species using 6% solution of Arsenal[®] (active ingredient Imazapyr, 28.7% a.i.) or 3% solution of Arsenal[®] AC (active ingredient Imazapyr 53.1% a.i.) in a water carrier. For Arsenal[®], one, one-inch incision was made per 2.5 cm DBH spaced evenly around the stem. For Arsenal[®] AC, one, one-inch incision was made per 7.5 cm DBH spaced evenly around the stem. For Arsenal[®] or Arsenal[®] AC, 1.5 ml of solution was applied to each incision. The cut-stump treatment was applied to all American beech (*Fagus grandifolia*) ≥ 5 cm DBH within the stand. American beech were cut down and the outer 5 cm of the stump surface were applied with 50% solution of Razor[®]-Pro, Glyphosate[®] 41, or Roundup[®]-Pro (all containing 41% Glyphosate a.i.) in a water carrier. The stem injection treatment was applied beginning 31 August and was completed by 1 November of each year, and the cut-stump treatment was applied beginning 31 August and was completed by 31 December of each year.

We used a before-after-control-impact (BACI) framework to infer causal relations between regeneration stand treatments and the response of avian and salamander communities through time (Eberhardt 1976). At each regeneration stand, we collected one-year of pre-treatment data and one- or two-years of post-treatment data on the vegetation, avian, and salamander communities (Table 3-1). A single vegetation and avian sampling point was placed at the center of each regeneration stand, 100 m from the stand edge (Figure 3-2). Four vegetation and salamander sampling points were spaced at the corners of a 100 m X 100 m square grid with the grid centered on the avian point count. Each vegetation and salamander sampling point was located 50-m from the stand edge (Figure 3-2).

2.3. *Vegetation assessment*

We measured vegetation characteristics at five sampling points within each regeneration stand (Figure 3-2) each year using an 11.3 m radius (0.04 ha) circular plot design. This vegetation protocol followed previous young forest wildlife studies in the central Appalachian region (Fettinger 2002, Schumacher 2002). Within this plot, we measured diameter-at-breast height (DBH) of all trees ≥ 11.4 cm DBH to the nearest 0.1 cm using DBH tapes and recorded status (“live” or “dead”). All trees (live and dead) were placed into one of five decay stage classes modified from Backhouse and Louiser (1991; Table 3-2). Within each respective decay stage class, individual tree DBH data were converted to tree basal area (m^2/ha) then summed for each 11.3 m circular plot. Additionally, we summed basal area for all live and dead trees to generate a single live tree and a single dead tree basal area value for each 11.3 m circular plot.

We estimated the amount of live foliage understory cover within two vertical zones (0–1.0 m and >1.0–2.0 m) with a modified Nudds (1977) profile board (hereafter “understory 0–1 m” and “understory >1–2 m”, respectively) placed 15-m from plot center in the four cardinal directions. Observers kneeled at the vegetation plot center, to reduce variation due to differing observer heights, and estimated understory cover into one of five categories (0–20%, >20–40%, >40–60%, >60–80%, or >80–100%) for each of the two zones. Standing at the vegetation plot center, we measured the percentage of live overstory cover (delineated at ≥ 5 m above ground) using a 25 cm x 25 cm transparent plexiglass panel, divided into a 5 x 5 grid, held overhead (Hachè et al. 2013; hereafter “overstory cover”).

We derived five topographic variables using 3-m scale digital elevation model (DEM) data (WVGIS 2019) within ArcGIS (ESRI 2014). They included slope percent, slope aspect, slope position (Weiss 2001), slope configuration, and elevation and were calculated using zonal statistics to generate a mean value within a 50-m radius surrounding each of the five vegetation sampling points. Slope percent, slope aspect, slope position, and slope configuration were combined using ModelBuilder in ArcGIS (ESRI 2014) to derive a topographic relative moisture index (TRMI) ranging from 0 (xeric soil) to 60 (mesic soil; Parker 1982).

2.4. *Avian sampling*

We quantified ruffed grouse and songbird responses to regeneration stands at one point-count location in each regeneration stand treatment (Figure 3-2) during 2016 – 2019 on days that were considered optimal for sampling (e.g., no rain, no heavy or gusty winds, and minimal

peripheral noise). Survey-level data collected during each visit included time of survey and ordinal day. Prior to surveys, observers were trained in survey protocols, distance estimation, and to independently identify all species visually and aurally while simultaneously recording distance and time interval. Surveyor skill was assessed based on their previous experience conducting point counts in the region and identification skills of avian species of the region and ranged from 1–5, where a score of 5 indicated thorough knowledge of species with extensive experience conducting point counts, a score of 1 indicated no knowledge of species with no previous experience conducting point counts, and a score of 3 indicated introductory knowledge of species with minimal experience conducting point counts (most technicians that lacked previous point count experience were scored as 3 following training).

Ruffed grouse.—We assessed ruffed grouse response during 5-minute drumming surveys (McBurney 1989) twice at each point between 12 April and 20 May during 0530–1000 (EST). Observers recorded presence or absence of drumming grouse during each survey and if present, placed detected individuals into one of five distance intervals (0–25 m, >25–50 m, >50–75 m, >75–100 m, or >100 m).

Songbird.—We quantified breeding songbird response using 10-minute standardized fixed-radius point counts (Ralph et al. 1993, Bibby et al. 2002). We conducted two visits during 19 May–27 June after 0530 and before 1000 (EST). Observers recorded species of each individual bird detected, type of detection (song, call, visual, or flyover), demographic information (male, female, juvenile, or unknown), distance from observer placed into one of five distance intervals (0–25 m, >25–50 m, >50–75 m, >75–100 m, or >100 m), and time interval the individual was first detected (0–2 min, >2–4 min, >4–6 min, >6–8 min, and >8–10 min).

For all songbird data analyses, we used only detections within 100-m of the sampling point. We chose 100 m as a sampling radius because it provided greater coverage of the entire stand area compared with a 50 m sampling radius. We included birds detected by sight and sound (excluding flyovers) for individual species and community statistical analyses. See Appendix 3-B for detection types used for each species.

2.5. *Salamander sampling*

We quantified salamander response to regeneration stands using a combination of artificial and natural cover objects on five block replicates at Beury Mountain WMA

(BMWMA). We sampled at four points (Figure 3-2) within the control and HS treatments, the CL and CW treatments were not sampled for salamanders. We conducted three survey visits annually during 2017–2019. Visit 1 was 18 April–6 May, visit 2 was 6–16 May, and visit 3 was 17 June–2 August. Artificial coverboards at each point consisted of nine (3 x 3 arrangement spaced ~1 cm apart), 20.0 x 10.0 x 2.5 cm wooden boards cut from hardwood species from the region (American basswood [*Tilia americana*], American sycamore [*Platanus occidentalis*], and yellow poplar [*Liriodendron tulipifera*]). Coverboards were placed at points at least one month prior to sampling to allow them to weather (Hesed 2012). Natural cover objects included for sampling were logs or rocks that measured ≥ 40 mm at the narrowest dimension and occurred within a 5-m radius of each artificial coverboard location. All eligible natural cover object dimensions (length and width, in mm) were measured prior to salamander sampling during the pre-harvest period. We marked natural cover objects with stake flags to maintain consistent sampling effort across visits and years. Use of a combination of artificial and natural cover objects for sampling can produce valid indices of terrestrial salamander populations (Smith and Petranks 2000).

We recorded species of each captured individual and measured snout-vent length (SVL, anterior edge of the snout to the distal end of the cloaca) to the nearest 0.1 mm using a dial caliper. We recorded body mass to the nearest 0.1 g using a Pesola spring scale. We clipped their tail to identify recaptures in subsequent sampling visits. We sprayed an antiseptic to the clipped tail to reduce risk of infection. We clipped the tail of each individual to identify recaptures in subsequent sampling visits and sprayed an antiseptic to the clipped tail to minimize risk of infection. Clippers were flame-sterilized with 95% ethanol to reduce potential disease transmission between captures. Salamanders were released at their original point of capture. Salamander sampling was conducted under methods approved by the WVDNR (permits 2016.143, 2017.104, 2018.110, and 2019.110) and the West Virginia University Institutional Animal Care and Use Committee (protocol 1602000305).

2.6. Statistical analyses

Ruffed grouse.—We provide summary statistics for ruffed grouse responses to regeneration stands because we had insufficient detections (20 total detections) to complete a statistical analysis with enough predictive power to warrant analyzing these data further. Sampling effort and a descriptive summary of ruffed grouse data are provided in results.

Focal songbird abundance.—We estimated abundance of breeding songbird species in regeneration stands over time using a stacked Bayesian N -mixture model (Kéry and Royle 2016). This modeling approach estimates abundances for each species using count data while accounting for imperfect detection. Additionally, estimated abundances of species are allowed to vary from year-to-year making this a dynamic modeling approach. We included species with ≥ 40 detections within 100-m of the sampling point and ≥ 0.200 estimated effective detection probability (i.e., probability of detecting an individual at least once across two annual survey visits). We fit two models for each songbird species. Model #1 incorporated a regeneration stand treatment and time interaction and model #2 incorporated a regeneration stand treatment main effect. Both models used the same detection-level sub-model but a different biological-level sub-model. In our detection-level sub-model, we modeled probability of detecting an individual, conditional on true presence, using a Binomial process model. We modeled survey-level covariates (e.g., ordinal day, surveyor skill, and time since sunrise) thought to influence an observer's probability of detecting an individual on the logit-linear scale.

In our biological-level sub-model for both model #1 and #2, we modeled species abundance using a Poisson process model. For model #1, we modeled expected abundance, y_{ik} , as a function of regeneration stand treatments with time and nuisance variables on the log-linear scale (Equation 1).

$$\begin{aligned} \log(y_{ik}) = & \beta_0 + \beta_1 \times trt_{ik} + \beta_2 \times CW_{ik} + \beta_3 \times HS_{ik} + \\ & \beta_4 \times (trt_{ik} \times time_{ik}) + \beta_5 \times (CW_{ik} \times time_{ik}) + \\ & \beta_6 \times (HS_{ik} \times time_{ik}) + \beta_7 \times elev_i + \\ & \beta_8 \times TRMI_i + rand.point_i \end{aligned} \quad (1)$$

For model #2, we modeled expected abundance, y_{ik} , as a function of regeneration stand treatments and nuisance variables. In model #2, the variable time interacts the same with all regeneration stand treatments (Equation 2).

$$\begin{aligned} \log(y_{ik}) = & \beta_0 + \beta_1 \times trt_{ik} + \beta_2 \times CW_{ik} + \beta_3 \times HS_{ik} + \\ & \beta_4 \times (trt_{ik} \times time_{ik}) + \beta_5 \times +elev_i \\ & \beta_6 \times TRMI_i + rand.point_i \end{aligned} \quad (2)$$

For both equations, i references each sampling point location (i.e., individual regeneration stand) and k references year. On the right-hand side of the equation, the pre-treatment period is included in the intercept (β_0). This approach allows the control treatment across all years and the pre-treatment year for all regeneration treatments (CL, CW, and HS) to be used as baseline values for comparison with one-year and two-year post-treatment data. The term trt_{ik} , corresponds to the CL treatment that was used as a reference treatment. The term CW_{ik} , corresponds to CW treatment, HS_{ik} corresponds to HS treatment. The term $time_{ik}$ was included as an interactive term with each treatment in model #1 to allow the effect of time to vary for each treatment. In model #2, $time_{ik}$ was included as an interactive term applied the same to all treatments. All factor levels were dummy coded with “0” or “1” depending on whether that factor level occurred at each point, except for the reference (trt_{ik} ; i.e., CL treatment) which was dummy coded with a “1” unless it was the control treatment. Time was treated as a continuous variable with 0, 1, or 2 corresponding to pre-treatment, one-year post-treatment, or two-year post-treatment, respectively. Elevation ($elev_i$) and TRMI ($TRMI_i$) were included as nuisance predictor variables. We included a random point-count effect ($rand.point_i$) to account for heterogeneity among points and repeated observations at the same point. Continuous variables were standardized to have a mean of 0 and standard deviation of one prior to analysis.

We used diffuse prior distributions for detection-level covariates (Gaussian distribution [mean = 0, variance = 100]) and biological-level and nuisance covariates (Gaussian distribution [mean = 0, variance = 10]). Diffuse covariate priors were used because we were uncertain of the relationships between treatments and songbird species. For our random point-count effect, we used a Gaussian distribution (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma distribution ($\alpha = 1$, $\beta = 1$). We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 40000 iterations with a burn-in of 20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions. We assessed model fit with a posterior check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a simulated Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter

Rhat values <1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2020) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015). We assessed model performance between the treatment-time interaction model (model #1) and the main effects model (model #2) using deviance information criterion (DIC).

For model #1, we transformed model outputs using an inverse log-link and calculated expected abundance in each regeneration treatment in each year using Equation 1 terms. We calculated the log of the proportional change of abundance within each respective regeneration treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period and between cut-back border treatments within each time period, and assessed differences based on 95% credible intervals not overlapping 0. For model #2, we transformed model outputs using an inverse log-link and calculated expected abundance in each regeneration treatment using Equation 2 terms. We calculated the log proportional change of expected abundance among regeneration treatments and assessed differences based on 95% credible intervals not overlapping 0.

Avian community richness.—We used a Bayesian generalized linear mixed-effects model (GLMM) to assess apparent species richness (species richness not accounting for imperfect detection of species) of avian guilds to regeneration stands through time. Richness was the total number of species detected at a point across the two visits annually. We categorized avian species into ecological guilds based on habitat (forest gap, forest interior, and young forest), nesting (primary cavity and secondary cavity), foraging substrate (bark), and conservation status (i.e., species of conservation priority). Species groupings were based on breeding biology of regional bird species and previous guild studies from the region (McDermott and Wood 2009, Farwell et al. 2016). We considered a species as conservation priority if it was listed by Appalachian Mountains Joint Venture (AMJV 2018), Partners-in-Flight Appalachian region - Bird Conservation Region 28 (PIF 2019), or WVDNR State Wildlife Action Plan (WVDNR 2015; see Appendix 3-B for species status and guild association).

We modeled expected apparent species richness using Equation 1 (model #1) and Equation 2 (model #2), with the log expected apparent richness, y_{ik} , specified with a Poisson process model. We used diffuse prior distributions for the treatment levels and nuisance covariates (Gaussian distribution [mean = 0, variance = 10]). Diffuse covariate priors were used because we were uncertain of the relationships between treatments and avian guilds. For our

random point-count effect, we used a Gaussian distribution (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma distribution ($\alpha = 1$, $\beta = 1$). We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 40000 iterations with a burn-in of 20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions. We assessed model fit with a posterior check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a simulated Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhat values < 1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2020) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015). We assessed model performance between the treatment-time interaction model (model #1) and the main effects model (model #2) using DIC.

For model #1, we transformed model outputs using an inverse log-link and calculated expected apparent species richness in each regeneration treatment in each year using Equation 1 terms. We calculated the log of the proportional change of richness within each respective regeneration treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period and between cut-back border treatments within each time period, and assessed differences based on 95% credible intervals not overlapping 0. For model #2, we transformed model outputs using the inverse log-link and calculated apparent species richness in each regeneration treatment using Equation 2 terms. We calculated the log proportional change of expected richness among regeneration treatments and assessed differences based on 95% credible intervals not overlapping 0.

Salamander surface relative abundance.—We estimated surface relative abundance of eastern red-backed salamanders (*Plethodon cinereus*) in regeneration stands over time using a Bayesian GLMM. We estimated surface relative abundance because terrestrial salamanders spend the majority of their time underground and we did not incorporate detection probability into the analysis. We performed analysis only on eastern red-backed salamanders because they comprised the majority of captures (78%) during the study. We treated capture data at the four

sampling plots within a single regeneration stand as independent because of the small home range size of terrestrial salamanders (Kleeberger and Werner 1982). We used total captures across the three visits (excluding recaptures) at each sampling plot within a sampling year in analysis.

We modeled expected relative abundance using a Poisson process model. We modeled expected abundance, y_{ik} , as a function of regeneration stand treatments with time and nuisance variables on the log-linear scale (Equation 3),

$$\begin{aligned} \log(y_{ik}) = & \beta_0 + \beta_1 \times HS_{ik} + \beta_3 \times (HS_{ik} \times time_{ik}) + \\ & \beta_4 \times elev_i + \beta_5 \times TRMI_i + \beta_6 \times Cover_i + \\ & rand.point_i \end{aligned} \quad (3)$$

where i references each sampling point location (i.e., individual regeneration stand) and k references year. On the right-hand side of the equation, the pre-treatment period is included in the intercept (β_0). This approach allows the control treatment across all years and the pre-treatment year for the HS treatment to be used as baseline values for comparison with one-year and two-year post-treatment data. The term HS_{ik} corresponds to HS treatment. The term $time_{ik}$ was included as an interactive term to limit the effect of time to the HS treatment. Factor levels were dummy-coded with “0” or “1” depending on whether that factor level occurred at each point. Time was treated as a continuous variable with 0, 1, or 2 corresponding to pre-treatment, one-year post-treatment, or two-year post-treatment, respectively. Elevation ($elev_i$), TRMI ($TRMI_i$), and number of cover objects ($Cover_i$) were included as nuisance predictor variables. We included a random sampling point effect ($rand.point_i$) to account for heterogeneity among points. Continuous variables were standardized to have a mean of 0 and standard deviation of one prior to analysis.

We used diffuse prior distributions for factor levels and nuisance covariates (Gaussian [mean = 0, variance = 10]). Diffuse covariate priors were used because we were uncertain of the relationships between treatments, topography, and elevation variables and eastern red-backed salamanders. For our random point effect, we used a Gaussian distribution (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma distribution ($\alpha = 1$, $\beta = 1$). We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 20000 iterations with a burn-in of 10000 iterations at a thinning rate of 100, yielding 300

samples for posterior distributions. We assessed model fit with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a simulated Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhat values < 1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2020) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015).

We transformed model outputs using an inverse log-link and calculated expected relative abundance in each regeneration treatment in each year using Equation 2 terms. We calculated the log of the proportional change of relative abundance values within the HS treatment at one-year post-treatment period and two-year post-treatment period against the pre-treatment period, and assessed differences based on 95% credible intervals not overlapping 0.

Salamander body condition.—We assessed body condition index (BCI) of eastern red-backed salamanders in the control and HS treatments over time using a Bayesian GLMM. We calculated BCI by regressing the natural logarithm of body mass on the natural logarithm of SVL and using the residuals as the BCI values for each individual following Gabor (1995).

We modeled expected BCI with a Gaussian process model using Equation 3. We used diffuse prior distributions for the process model variance (variance = τ) and the factor levels and nuisance variables (Gaussian distribution [mean = 0, variance = 10]). For our random point effect, we used an inverse Gaussian distribution (mean = 0, variance = τ) prior with a hyperparameter τ from a Gamma distribution ($\alpha = 1$, $\beta = 1$). We obtained posterior distributions of model parameters by running three parallel MCMC simulations of 20000 iterations with a burn-in of 10000 iterations at a thinning rate of 100, yielding 300 samples for posterior distributions. We assessed model fit with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a simulated Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when

all parameter Rhat values <1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2019) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015).

We calculated the change of expected BCI values in each regeneration treatment in each year using Equation 2 terms. We calculated the change of expected BCI values within the HS treatment at one-year and two-year post-treatment period against the pre-treatment period, and assessed differences based on 95% credible intervals not overlapping 0.

Cost-benefit analysis.—We used a weighted analysis to define management objectives in order to evaluate the cost of treatment versus the benefit of maximizing songbird richness in regeneration stand treatments. The management objectives for the weighted analysis were to maximize young forest habitat guild species richness, maximize forest interior habitat guild species richness, maximize cavity (primary and secondary) nesting guild species richness, maximize richness of species of conservation priority, and minimize cost. We did not include bark foragers into the objectives because of the overlap this guild has with the cavity nesting guild. A salamander objective was not included for this analysis because sampling only occurred in the control and HS treatments.

In our cost-benefit analysis, we scored the benefit of maximizing species richness of multiple avian guilds objective and the cost of implementing treatment objective for each regeneration treatment and compared each treatment with other regeneration treatments. This approach is different from a traditional cost-benefit analysis, where we would measure and score the benefits of a single decision/action minus the costs associated with that decision/action and then compare each respective score with other decisions/actions under consideration. Our cost-benefit analysis approach follows other wildlife studies that have compared multiple mitigation strategies and their effects on some index (e.g., species richness, number of mortalities) as a benefit for multiple species while minimizing cost of implementation (Huijser et al. 2009, Siemers et al. 2015). We used this approach for the cost-benefit analysis because it allows for multiple benefits and/or costs objectives to generate a score for comparison with other regeneration treatments. For each regeneration stand treatment, we evaluated guild responses and cost. Guild richness (i.e., performance scores) within each regeneration treatment across both post-treatment years (Table 3-3) were calculated from 300 posterior draws using Equation 2 terms. Cutting and herbicide costs per ha (Table 3-3) were provided by WVDNR and the

Wildlife Management Institute (WMI). We used a generalized multi-method tradeoffs approach to rank and weight each management objective in order to compare regeneration stand treatments (Gregory et al. 2012). We assigned weights for each objective as young forest species richness = 0.28, species of conservation priority species richness = 0.21, cost = 0.20, forest interior species richness = 0.18, and cavity nesting species richness = 0.14. We derived weight values of objectives using a three-step process, where in step 1 we assigned initial objective ranking priority from 1 to 6, with rankings 1–5 being the objectives from highest to lowest importance and 6 = hypothetical baseline where no management objectives are maximized. Rankings were as follows: 1 = young forest species richness, 2 = species of conservation priority species richness, 3 = cost, 4 = forest interior species richness, and 5 = cavity nesting species richness. In step 2 we used priority rankings to assign relative swing weights to each management objective, giving “100” to rank 1 and “0” to rank 6 with all other rankings receiving a value between 1 and 99 based on relative importance. Relative swing weights were as follows: young forest species = 100, species of conservation priority = 75, cost = 70, forest interior species = 65, cavity nesting species = 50, and hypothetical baseline = 0. Lastly, in step 3 we normalized relative swing weights to obtain weight values. We calculated the normalized performance score (NPS) of each cell in row x and column y (Table 3-4) using Equation 4,

$$NPS_{xy} = \frac{\text{performance score}_{xy} - \text{minimum performance score}_x}{\text{maximum performance score}_x - \text{minimum performance score}_x}$$

for each of the 300 posterior draws where performance scores are referenced by row x and column y in Table 3-3. The weighted utility sum (WUS) for each posterior draw of each regeneration stand treatment was the sum of the product of normalized performance scores in Table 3-4 and weights.

3. Results

3.1. Vegetation

Following treatment, live and dead basal area and overstory cover were negligible in CL and CW treatments (Table 3-5). In the HS treatment, mean live basal area was reduced from 19 m²/ha at pre-treatment to 11 m²/ha at two-year post-treatment and mean dead basal area increased from 1 m²/ha at pre-treatment to 4 m²/ha at two-year post-treatment. Overstory cover in the HS treatment was reduced from 93% at pre-treatment to 66% and 65% at one-year and

two-year post-treatment, respectively. Understory cover 0–1 m increased annually at similar levels from pre-treatment to two-year post-treatment in CL, CW, and HS treatments (77 to 96%, 70 to 97%, and 75 to 92% in CL, CW, and HS, respectively). However, understory cover >1–2 m only increased in the CL and CW treatments by two-year post-treatment (60 to 87%, 54 to 82%, and 63 to 66% in the CL, CW, and HS, respectively).

3.2. *Ruffed grouse*

We conducted 340 grouse drumming surveys during the study and detected 20 grouse. Mean number of detections varied little across time for three of the treatments (Table 3-6). Mean detections increased over time in the HS treatment, with 0.1 detections/plot during pre-treatment and 0.4 detections/plot during two-year post-treatment.

3.3. *Focal songbird abundance*

We conducted a total of 338 avian point-counts over the course of the study period and detected a total of 98 species. Of the 21 species that met the detection criteria (≥ 40 detections), 13 species also met the estimated effective detection probability criteria (≥ 0.200 ; Appendix 3-B). Of these 13 species, four were from the young forest guild (chestnut-sided warbler, common yellowthroat, eastern towhee, and indigo bunting), three were from the forest gap guild (black-throated green warbler, hooded warbler, and red-eyed vireo), and four were from the forest interior guild (black-and-white warbler, blue-headed vireo, ovenbird, and tufted titmouse). Our posterior predictive checks indicated all individual species models were a reasonable fit (Appendix 3-C).

Of the 13 species analyzed, six responded positively to one regeneration stand treatment and four responded negatively to one regeneration stand treatment. Abundances of all four young forest species analyzed increased in the CL from pre-treatment to one-year or two-year post-treatment (Figure 3-3) while blue-headed vireo and ovenbird (both forest interior species) abundances decreased from pre-treatment to two-year post-treatment (Figure 3-4). In the HS treatment, blue-headed vireo and ovenbird (both forest interior species,) abundances increased from pre-treatment to one-year or two-year post-treatment (Figure 3-4) while eastern towhee and indigo bunting (both young forest species) abundances decreased from pre-treatment to one-year and two-year post-treatment (Figure 3-3). No forest gap species responded positively or negatively to regeneration stand treatments (Figure 3-5).

The main effect of regeneration treatments had a strong influence on four species. Eastern towhee and indigo bunting abundances were higher in the CL than the HS and control, and abundances in the CW were higher than HS treatment (Figure 3-6). Abundance of common yellowthroat was higher in the CW than control (Figure 3-6). Ovenbird abundance was higher in the HS than the CL, CW, and control, while abundance in the control was higher than the CL (Figure 3-6). No forest gap species were strongly influenced by regeneration treatment main effects.

3.4. *Avian community richness*

Our posterior predictive checks indicated all avian guild models were a reasonable fit (Appendix 3-C). Species richness of two of the three habitat guilds, the primary and secondary cavity nesting guilds, and the bark foraging guild were strongly influenced by regeneration stand treatment. In the CL treatment, richness of the young forest guild increased from pre-treatment to one-year and two-year post-treatment (Figure 3-7) while richness of forest interior habitat guild (Figure 3-7) and secondary cavity nesting guild (Figure 3-8) decreased by two-year post-treatment. In the HS treatment, species richness of the forest interior habitat guild, primary and secondary cavity nesting guilds, and bark foraging guild increased from pre-treatment to two-year post-treatment (Figures 3-7 and 3-8). Species richness of the young forest habitat guild decreased in the HS treatment from pre-treatment to one-year and two-year post-treatment (Figure 3-7). Species richness of the forest gap habitat guild (Figure 3-7) and species of conservation priority group (Figure 3-8) were not strongly influenced by regeneration stand treatments over time.

The main effect of regeneration treatments had a strong influence on one habitat guild and two avian guilds. Species richness of the young forest habitat guild was higher in the CL treatment than the HS and control treatments, and higher in the CW treatment than the HS treatment (Figure 3-9). Species richness of the primary cavity nesting guild and bark foraging guild were higher in the HS treatment than the control treatment (Figure 3-9), while richness of the bark foraging guild was also higher in the HS treatment than the CL treatment (Figure 3-9).

3.5. *Salamander surface relative abundance and body condition index*

We captured a total of 274 salamanders of six species during the study period (Appendix 3-D), with 213 (78%) being eastern red-backed salamanders. Our posterior predictive check

indicated our relative abundance and body condition index models were a reasonable fit (Appendix 3-C). Relative abundance of eastern red-backed salamanders increased in the HS treatment from pre-treatment to one-year post-treatment but was not different from pre-treatment to two-year post-treatment (Figure 3-10). Body condition index of eastern red-backed salamanders decreased in the HS treatment from pre-treatment to one-year post-treatment then increased at two-year post-treatment (Figure 3-11).

3.6. *Cost-benefit analysis*

Species richness of the young forest habitat guild and species of conservation concern was maximized in the CL treatment. The HS treatment maximized forest interior habitat guild and cavity nesting guild species richness. Performance of cost was maximized in the control treatment and minimized in the CW (Table 3-4). Based on the cost-benefit analysis, weighted utility sum (WUS) for CL was the optimum treatment (WUS [lower and upper 95% credible intervals] = 0.63 [0.33, 0.85]), followed by control (WUS = 0.52 [0.37, 0.67]), HS (WUS = 0.33 [0.11, 0.54]), and CW (WUS = 0.30 [0.06, 0.57]; Table 3-4).

4. **Discussion**

Based on a scenario of evenly distributed forest age classes, current forest conditions throughout much of West Virginia and the central Appalachian region indicate young forest deficit throughout the region (Shifley and Thompson III 2011). Due to young forest habitat being largely created via commercial logging, management and conservation of young forest wildlife relies heavily on forest stands of high economic value. Measures that incorporate poor quality forestlands into timber sales and habitat management plans will be needed in order to reduce the young forest deficit and meet conservation objectives for young forest wildlife. Here we have assessed three non-commercial young forest management treatments and their effects on birds and salamanders. Additionally, we examined the tradeoffs between maximizing bird responses and minimizing cost of young forest treatments to assist managers in their decision-making when developing wildlife management guidelines.

4.1. *Wildlife response to regeneration stands*

Implementation of regeneration treatments induced a dichotomous response from the avian community. Young forest species responded positively to the CL treatment and negatively

to the HS treatment, while forest interior and cavity nesting species generally responded positively to the HS treatment and negatively to the CL treatment. Guild richness responses followed similar trends. Additionally, the eastern red-backed salamander, a terrestrial salamander most suited to forest interiors, responded positively to the HS treatment immediately following treatment although BCI was reduced during this period.

Abundance and species richness of the young forest guild increased in the CL and decreased in the HS following treatment from both the treatment-time interaction model (model #1) and main effects model (model #2). Positive responses by young forest species in the CL were likely driven by the complete removal of live and dead basal area, which stimulated vegetation growth in the understory, particularly for understory vegetation >1–2 m, and created habitat conditions favorable for foraging and breeding. Vegetation response to treatment was not immediate however, as understory vegetation >1–2 m had generally similar amounts across CL, CW, and HS treatments at one-year post-treatment likely reflecting a stronger time effect than treatment effect. The deviation of understory vegetation levels among treatments was more noticeable at two-year post-treatment, likely due to treatment differences (Table 3-5). The partial harvest of live basal area in the HS elicited vegetation growth in the understory at two-year post-treatment, but not to the extent we observed in the CL treatment (Table 3-5). The differences in the understory vegetation, particularly for the >1–2 m stratum, between CL and HS likely contributed to the differences in responses of young forest species in these two treatments. Partial harvesting approaches similar to the HS (e.g., shelterwood) do create vegetation conditions suitable for young forest species immediately following treatment and we expected similar responses to occur in our study (Duguay et al. 2001, Perry et al. 2013, Sheehan et al. 2014, Greenberg et al. 2014). However, the HS treatment may not have created enough canopy openness to induce immediate understory vegetation growth and create suitable habitat conditions for young forest species at one-year and two-year post-treatment. Another possible explanation for the lack of response from young forest species to the HS may be the lack of coarse woody debris (CWD) following treatment, which can provide cover for many young forest species (Lohr et al. 2002).

Arrangement of CWD can also affect the distribution of young forest species within a stand (Grodsky et al. 2016). The CL and CW treatments both received complete removal of live and dead basal area and understory vegetation levels were comparable at one-year and two-year

post-treatment. However, our main effects analyses indicated only the common yellowthroat had higher abundances in the CW while eastern towhee and indigo bunting abundances and young forest habitat guild richness were higher in the CL. Our treatment-time interaction results (Figures 3-3 and 3-9) were consistent with these findings, indicating habitat suitability increased in the CL and remained consistent in the CW following treatment, relative to the control. Creating large CWD piles or windrows, like in the CW treatment, can create suitable nesting areas for a suite of young forest species that nest in brush piles (e.g., Carolina wren, eastern towhee; Grodsky et al. 2016). However, wood piles can also attract avian nest predators (e.g., snakes, rodents) which may reduce reproductive output of species and the concentration of wood piles likely reduces plant regeneration throughout the stand via greater browse pressure (Murray et al. 2016, Sullivan et al. 2017). Additionally, construction of wood piles via heavy machinery likely damaged understory vegetation which could negatively affect young forest species. Contrary to the CW treatment, CWD in the CL treatment was evenly distributed throughout the stand.

Live basal area in HS treatments was reduced on average from 19 m²/ha at pre-treatment to 11 m²/ha at two-year post-treatment (Table 3-5) but this did not appear to significantly reduce habitat suitability, relative to the control, for forest interior species. Further, some forest interior species responded positively to this treatment likely due to the retention of canopy trees. While live basal area was reduced in the HS treatment, all canopy trees (both live and dead) were retained in this treatment. This may help lessen the negative effects of forest disturbances on forest interior species. Additionally, partial canopy removal treatments (e.g., shelterwood) typically retain some level of forest interior species following harvest (Newell and Rodewald 2012, Greenberg et al. 2014), particularly those that use canopy gaps (Sheehan et al. 2014). Consequently, we had expected to see a positive response from gap-associated species to the HS, but this treatment did not appear to strongly, positively influence abundances or species richness of this group. This could be due to the small reduction of live basal area in the HS following treatment not creating sufficiently-sized canopy gaps (Sheehan et al. 2014). Nareff et al. (2019) reported cerulean warbler, a forest gap species, exhibited more rapid decreases in abundance in lighter intensity silviculture harvests than heavier intensity harvests, likely due to the smaller-sized canopy gaps in lighter intensity stands limiting habitat suitability for this species. A similar response may be occurring in the HS treatments, indicating more extensive tree kill in the HS is

necessary to increase habitat suitability for forest gap species. Interestingly, forest interior species responded negatively to the CL but not the CW at the two-year post-treatment period. Both CL and CW removed all live and dead basal area within their stands, thus eliminating canopy trees and mature forest conditions for this group of species (Table 3-5). One factor that could affect species responses is the peripheral mature forest at CL and CW stand edges (Duguay et al. 2001) but we did not assess this effect. To capture influence of edge habitats, we used species detections <100 m from surveyor. Therefore, responses we observed in the CL and CW treatments on forest interior species may be due to factors in the mature forest beyond the stand boundary. Regardless, the HS was the most effective treatment, besides the control, that we evaluated for increasing abundance and richness of forest interior avian species.

Similar to the forest interior avifauna, eastern red-backed salamander relative abundance responded positively to the HS treatment at one-year post-treatment (no salamander sampling occurred in the CL or CW treatments). This is surprising considering canopy cover was reduced in the HS without any habitat mitigation via CWD input. Canopy openness inhibits drier forest floor microclimate conditions which conflicts with terrestrial salamander physiology (Brooks and Kyker-Snowman 2008, Hillman et al. 2009). However, partial harvesting techniques (e.g., shelterwood) typically retain some level of forest floor CWD, which constitutes important surface refugia, to help salamanders cope with changes at the forest floor microclimate following harvest (Peele et al. 2017). Despite mitigation via CWD, terrestrial salamanders are often negatively affected by shelterwood harvests (Harpole and Haas 1999, Duguay and Wood 2002). It would follow that in the HS, which did not create any CWD, eastern red-backed salamanders would decrease in relative abundance following treatment. This was not the case however, as relative abundance increased in the HS following treatment, although relative abundance returned to levels similar to pre-treatment at the two-year post-treatment period. Abundance of eastern red-backed salamanders was inversely related to BCI at the one-year and two-year post-treatment periods. This indicates that while there were more individuals at the surface for capture in the HS at one-year post-treatment, these individuals had poor body condition relative to individuals captured during pre-treatment and in the control. Lower BCI in individuals could result from reduced food availability at the surface, which forces individuals to remain at the surface for longer periods in order to meet their energy demands. This could be from increased canopy openness following treatment which dried out the leaf litter and soil where this species

primarily forages. The decrease in relative abundance and increase in BCI in the HS at two-year post-treatment could indicate an increase in habitat suitability during this period, which could be due to understory vegetation growth shading the forest floor and retaining leaf litter and soil moisture. Future monitoring could elucidate these questions to determine whether this partial harvesting approach is suitable for eastern red-backed salamander populations.

Primary and secondary cavity nesters and bark foragers responses to regeneration treatments for this study were of high priority, particularly the HS treatment. One factor for this is that snags constitute important foraging and nesting substrate for cavity nesting and bark foraging species but also provide habitat structures for small mammals, invertebrates, and fungi (Cockle et al. 2011, Garrick et al. 2019). Further, woodpeckers, which make up the majority of primary cavity nesters and bark foragers, are good indicators of forest bird diversity (Mikusiński et al. 2001, Drever et al. 2008). We observed increased species richness of the primary and secondary cavity nesting guilds and bark foraging guild in the HS treatment at the two-year post-treatment period, likely due to the increase in snag basal area (Table 3-5). The increase in snags in the HS (4% of canopy trees at pre-treatment and 38% at two-year post-treatment) likely increased food resources for bark foragers and nesting trees for cavity nesters. The increase in snags in HS treatment also likely created important habitat for small mammals, invertebrates, fungi, and other taxa, although we did not test this and thus can make no definitive conclusions (Payer and Harrison 2005, Wermelinger et al. 2017). Further monitoring of the HS treatment to assess snag dynamics for cavity nesting and bark foraging species and to assess understory vegetation growth for young forest species will help managers determine if the HS technique offers a long-term suitable approach for managing forest avifauna with different vegetation requirements.

Based on our cost-benefit analysis, the CL treatment (weighted utility sum [WUS] = 0.70) was the optimum regeneration treatment to meet our management objectives of maximizing species richness and minimizing cost of habitat management. High WUS in the CL was due to high performance of young forest species and species of conservation priority (Table 3-3). The control treatment performed well meeting the cost and species of conservation priority management objectives and received a WUS = 0.48, indicating that no forest manipulation can still meet objectives set by managers. The HS treatment performed well meeting the forest interior and cavity nesting objectives (Table 3-4), however these were given lower weights and

therefore HS received a WUS = 0.38. The CW (WUS = 0.28) met the cavity nesting species objective well, most likely due to secondary cavity nesting species responding to this treatment (Figure 3-7).

5. Management Implications

Avian species responses to regeneration treatments were dichotomous and followed predictable patterns. Young forest species were positively influenced by the CL treatment and negatively influenced by the HS treatment, while forest interior and cavity nesting species were positively influenced by the HS treatment and negatively influenced by the CL treatment. Managing for young forest species can negatively affect species associated with mature, undisturbed forests and this was evident in the CL treatment. However, partial harvest techniques, like shelterwood harvests, have been shown to reduce species composition turnover by retaining some level of forest interior species while creating understory vegetation features suitable for young forest species (Sheehan et al. 2014, Perry et al. 2018). We hypothesized that young forest and forest gap species would positively respond to the canopy openness created in the HS treatment, but individual young forest species and the community responded negatively to treatment and forest gap response was negligible. Increased understory vegetation growth via greater reduction of live basal area in the HS will likely increase understory cover for young forest species while providing larger canopy gaps for forest gap species and higher snag densities for cavity nesting species. However, this measure will likely reduce the habitat suitability of the HS for forest interior species. Future monitoring of the HS will provide interesting insight into whether managers can simultaneously promote vegetation features suitable for young forest, forest gap, forest interior, and cavity nesting/bark foraging avian species.

Determining appropriate non-commercial young forest management measures is something that needs to be considered on poor quality sites to ensure future forest resiliency and promote diversity (Gunn et al. 2019). Non-commercial wildlife habitat management is used by agencies, but little empirical evidence exists on a most appropriate and cost-effective approach for creating habitat. The CL treatment benefitted young forest species and retained all CWD to provide better growing conditions and fewer plant regeneration barriers (i.e., deer browse) than treatments like the HS, but also displaced forest interior and cavity nesting species. The CW did not perform well in maximizing species richness and was the most expensive treatment to implement. The CW appears to be the least effective regeneration stand treatment for meeting

our management objectives. Ultimately, it depends on the short- and long-term objectives of the manager. The CL provided short-term benefits to young forest species, a group of high conservation interest, and may produce merchantable timber in the long-term. The HS provides short-term and likely long-term benefits to cavity nesting/bark foraging species and forest interior birds and salamanders. Higher costs of the CL treatment should certainly be considered when deciding what non-commercial approach to use, but incorporating small, non-commercial management stands into larger commercial timber sales may alleviate these expenditures. The incorporation of poor-quality forestlands into management plans will help reduce the young forest deficit in West Virginia and the central Appalachian region and provide important habitat for young forest wildlife communities.

Despite the usefulness of cost-benefit analysis for evaluating multiple benefit/cost objectives for managers, there are limitations to this approach. We simplified the management objectives into four biological and one economic objective but there are often a myriad of objectives that managers need to consider (e.g., retaining biological legacies, other taxa responses to management). Further, we did not incorporate the cost of non-native, invasive plant management into the cost of management, which will likely affect the long-term management goals of these treatments. When managers are confronted with a higher likelihood of invasive species spread following treatment further assessment is needed to incorporate the necessary long-term maintenance of invasive species into the cost-benefit analysis. If consideration is not given to potential invasive species spread following treatment (something that was not done here because we did not have the necessary information) then cost of treatment is likely to be underestimated and will limit the effectiveness of a cost-benefit analysis for assessing management objectives. Another limitation of the cost-benefit analysis is that some performance measures are not easily quantified and therefore are difficult to put into any analysis (e.g., public response, future stand value). There is also a level of subjectivity when ranking the importance weights of each management objective. These weights are likely to vary across space (e.g., different wildlife management areas) and time (e.g., invasive plant spread and subsequent wildlife response), and among natural resource experts. To fix this, greater involvement of agency personnel and interest groups to help in developing importance weights is likely to ensure any subjectivity is reduced to some extent.

Continued monitoring of treatments will provide important information about the effectiveness of regeneration stands to manage wildlife populations. Notably, continued monitoring of the HS treatment to determine whether this regeneration technique can simultaneously provide habitat for young forest species while retaining key habitat features for cavity nesting and bark foraging species will assist managers conserve both groups of species. Future research studies evaluating different non-commercial, young forest habitat management treatments and their associated costs and wildlife responses will be necessary to continue to reduce the young forest deficit in the region. This information will provide a broader range of options for land managers to suit their site-specific needs. Additionally, assessing the effectiveness of other non-commercial treatments to replicate or retain habitat features (e.g., snags, high stem densities) considered important for species or groups of interest (similar to our HS treatment) will be important when managing poorly-stocked forest stands for wildlife populations.

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7. Tables

Table 3-1. Data collection time-table for 16 regeneration stand blocks at six West Virginia Division of Natural Resource (WVDNR) wildlife management areas (WMA). “Blocks” in the table refers to number of block replicates at each site.

	2016	2017	2018	2019
Little Canaan WMA (4 blocks)	1 Year Pre-treatment	1 Year Post-treatment	2 Year Post-treatment	
Beury Mountain WMA (5 blocks)		1 Year Pre-treatment	1 Year Post-treatment	2 Year Post-treatment
Center Branch WMA (3 blocks)				
Moncove Lake WMA (2 blocks)				
Short Mountain WMA (1 block)			1 Year Pre-treatment	1 Year Post-treatment
Sleepy Creek WMA (1 block)				

Table 3-2. Decay stage classifications used to assess canopy trees in regeneration stands. We condensed the eight Backhouse and Louiser (1991) classes into five (in parentheses) by combining classes 4–5 into a single decay class and classes 6–8 into a single decay class.

Decay Class	Description
1 (1)	Live, healthy
2 (2)	Live, unhealthy (e.g., broken top, shelf fungus, boring insects, trunk gall)
3 (3)	Recently dead, most branches/twigs, possible broken top, stable
4 (4)	No needles/twigs, 50% branches lost, possible broken top, 25-50% bark lost, wood mostly hard
5 (4)	Most branches gone, possible broken top, 51-75% bark lost, wood hard and soft
6 (5)	No branches, some stubs, broken top, 76-100% bark lost, wood mostly soft
7 (5)	No branches, broken top, no bark, wood soft
8 (5)	No branches, broken top, no bark, hollow/partially hollow shell

Table 3-3. Posterior means with 95% confidence intervals (in parentheses) of guild richness and cost (i.e., estimated performance scores) of regeneration stand treatments (“Cost” score was not generated via posterior mean). Guild richness values for each treatment were derived from Equation 1 and cost information was provided from the West Virginia Division of Natural Resources and the Wildlife Management Institute.

Performance measure	Treatment			
	Control	Clearcut-Leave	Clearcut-Windrow	Hack-Spray
Guild Richness (species/plot)				
Young Forest	1.36 (1.09, 1.64)	2.89 (2.07, 3.92)	1.42 (0.80, 2.29)	0.63 (0.31, 1.11)
Forest Interior	3.44 (2.96, 3.91)	3.43 (2.60, 4.40)	3.05 (1.97, 4.61)	3.80 (2.34, 5.96)
Cavity nesting	1.42 (1.16, 1.75)	1.58 (1.09, 2.27)	1.59 (0.84, 2.84)	1.72 (0.93, 2.90)
Species of Conservation Priority	3.92 (3.44, 4.41)	4.21 (3.33, 5.40)	3.75 (2.41, 5.34)	3.47 (2.31, 5.04)
Cost (\$/ha)	0	1410	1776	1218

Table 3-4. Normalized performance scores (i.e., utility values) and 95% confidence intervals (in parenthesis) of regeneration stand treatments. Scores are derived by taking the quotient of the cell-level performance score in Table 3-3 minus the maximum row-level performance score (numerator) and minimum row-level performance score minus the maximum row-level performance score (denominator). Scores range 0–1 (0 = low performance, 1 = high performance). Weighted utility scores (WUS) of each regeneration stand treatment is the sum of the product of normalized performance scores and weights. Weights are as follows: young forest richness = 0.28, species of conservation priority richness = 0.21, cost = 0.20, forest interior richness = 0.18, and cavity nesting richness = 0.14.

Performance measure	Treatment			
	Control	Clearcut-Leave	Clearcut-Windrow	Hack-Spray
Avian Guilds				
Young Forest	0.32 (0.17, 0.47)	0.99 (0.99, 1.00)	0.39 (0.11, 0.99)	0.00 (0.00, 0.00)
Forest Interior	0.53 (0.03, 1.00)	0.52 (0.00, 1.00)	0.21 (0.00, 1.00)	0.75 (0.00, 1.00)
Cavity nesting	0.23 (0.00, 0.69)	0.53 (0.00, 1.00)	0.44 (0.00, 1.00)	0.60 (0.00, 1.00)
Species of Conservation Priority	0.54 (0.00, 1.00)	0.70 (0.00, 1.00)	0.45 (0.00, 1.00)	0.24 (0.00, 1.00)
Cost	1.00	0.21	0.00	0.31
Weighted Utility Score (WUS)	0.52 (0.36, 0.67)	0.63 (0.33, 0.85)	0.30 (0.06, 0.57)	0.33 (0.11, 0.54)

Table 3-5. Vegetation summary statistics (mean $\pm$ standard error) in regeneration treatments over time. Treatment abbreviations include CL = clearcut-and-leave, CW = clearcut-and-windrow, and HS = hack-and-spray. Live and dead tree decay stages are defined in Table 3-2.

	Control	CL	CW	HS
Live basal area (m²/ha): Decay stage 1				
Pre-treatment	15.0 $\pm$ 1.4	15.0 $\pm$ 1.2	15.4 $\pm$ 1.0	16.4 $\pm$ 1.7
One-year Post-treatment	15.0 $\pm$ 1.4	0.2 $\pm$ 0.2	0.4 $\pm$ 0.2	10.4 $\pm$ 1.2
Two-year Post-treatment	14.5 $\pm$ 1.5	0.0 $\pm$ 0.0	0.3 $\pm$ 0.2	7.9 $\pm$ 1.2
Live basal area (m²/ha): Decay stage 2				
Pre-treatment	0.2 $\pm$ 0.1	0.1 $\pm$ 0.1	0.3 $\pm$ 0.1	0.1 $\pm$ 0.0
One-year Post-treatment	0.2 $\pm$ 0.1	0.0 $\pm$ 0.0	0.1 $\pm$ 0.1	3.2 $\pm$ 0.7
Two-year Post-treatment	0.2 $\pm$ 0.1	0.0 $\pm$ 0.0	0.1 $\pm$ 0.0	2.6 $\pm$ 0.4
Dead basal area (m²/ha): Decay stage 3				
Pre-treatment	0.4 $\pm$ 0.1	0.4 $\pm$ 0.1	0.5 $\pm$ 0.1	0.5 $\pm$ 0.1
One-year Post-treatment	0.4 $\pm$ 0.1	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	2.3 $\pm$ 0.5
Two-year Post-treatment	0.4 $\pm$ 0.1	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	3.0 $\pm$ 0.6
Dead basal area (m²/ha): Decay stage 4				
Pre-treatment	0.3 $\pm$ 0.1	0.3 $\pm$ 0.01	0.3 $\pm$ 0.1	0.3 $\pm$ 0.1
One-year Post-treatment	0.3 $\pm$ 0.1	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.4 $\pm$ 0.1
Two-year Post-treatment	0.3 $\pm$ 0.1	0.0 $\pm$ 0.0	0.3 $\pm$ 0.2	0.4 $\pm$ 0.1
Dead basal area (m²/ha): Decay stage 5				
Pre-treatment	0.5 $\pm$ 0.2	0.4 $\pm$ 0.2	0.4 $\pm$ 0.2	0.3 $\pm$ 0.1
One-year Post-treatment	0.5 $\pm$ 0.2	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.6 $\pm$ 0.2
Two-year Post-treatment	0.5 $\pm$ 0.2	0.0 $\pm$ 0.0	0.0 $\pm$ 0.0	0.3 $\pm$ 0.1
Total live basal area (m²/ha): Decay stages 1–2				
Pre-treatment	15.2 $\pm$ 1.4	15.1 $\pm$ 1.3	15.7 $\pm$ 0.9	16.5 $\pm$ 1.7
One-year Post-treatment	15.2 $\pm$ 1.4	0.2 $\pm$ 0.2	0.5 $\pm$ 0.3	13.6 $\pm$ 1.5
Two-year Post-treatment	14.7 $\pm$ 1.5	0.1 $\pm$ 0.0	0.3 $\pm$ 0.2	10.4 $\pm$ 1.5
Total dead basal area (m²/ha): Decay stages 3–5				
Pre-treatment	1.2 $\pm$ 0.3	1.1 $\pm$ 0.2	1.2 $\pm$ 0.2	1.1 $\pm$ 0.2
One-year Post-treatment	1.2 $\pm$ 0.3	0.1 $\pm$ 0.0	0.0 $\pm$ 0.0	3.3 $\pm$ 0.5
Two-year Post-treatment	1.2 $\pm$ 0.3	0.0 $\pm$ 0.0	0.3 $\pm$ 0.2	3.7 $\pm$ 0.5
Percent overstory cover				
Pre-treatment	93.0 $\pm$ 1.5	86.4 $\pm$ 2.3	89.7 $\pm$ 2.8	92.9 $\pm$ 1.4
One-year Post-treatment	93.0 $\pm$ 1.5	2.1 $\pm$ 1.4	3.2 $\pm$ 1.8	65.9 $\pm$ 8.7
Two-year Post-treatment	93.3 $\pm$ 1.6	1.2 $\pm$ 1.2	3.2 $\pm$ 1.8	64.8 $\pm$ 9.5
Percent understory cover 0–1 m				
Pre-treatment	85.5 $\pm$ 4.3	77.3 $\pm$ 5.1	70.3 $\pm$ 5.5	75.2 $\pm$ 6.2
One-year Post-treatment	85.5 $\pm$ 4.3	87.5 $\pm$ 4.7	85.5 $\pm$ 5.9	85.9 $\pm$ 2.7
Two-year Post-treatment	83.4 $\pm$ 4.7	96.0 $\pm$ 2.8	97.4 $\pm$ 1.4	92.0 $\pm$ 2.6
Percent understory cover >1–2 m				
Pre-treatment	68.2 $\pm$ 5.7	60.3 $\pm$ 7.2	54.3 $\pm$ 7.1	62.5 $\pm$ 6.7
One-year Post-treatment	68.2 $\pm$ 5.7	69.5 $\pm$ 6.7	59.8 $\pm$ 6.5	58.2 $\pm$ 6.2
Two-year Post-treatment	67.4 $\pm$ 6.5	86.9 $\pm$ 6.4	82.4 $\pm$ 8.1	65.8 $\pm$ 6.8

Table 3-6. Total and mean (detections/plot) ruffed grouse (*Bonasa umbellus*) detections in regeneration stands over the study period. Mean values account for uneven sampling because Sleepy Creek WMA and Short Mountain WMA received no two-year post-treatment sampling and Center Branch WMA had no hack-and-spray (HS) treatments. Treatment abbreviations include CL = clearcut-and-leave, CW = clearcut-and-windrow, and HS = hack-and-spray.

	<u>Control</u>		<u>CL</u>		<u>CW</u>		<u>HS</u>	
	Total	Mean	Total	Mean	Total	Mean	Total	Mean
Pre-treatment	0	0.0	1	0.1	3	0.2	1	0.1
One-year Post-treatment	1	0.1	4	0.2	1	0.1	4	0.3
Two-year Post-treatment	1	0.1	1	0.1	2	0.1	1	0.4

8. Figures

Figure 3-1. County map of West Virginia showing locations of wildlife management areas (WMA) where regeneration stands were located (circles) as well as major cities (stars).

Figure 3-2. Regeneration stand layout with avian sampling point (black square) and salamander sampling point (black dot) placements. Vegetation sampling points were located at each avian and salamander sampling point.

Figure 3-3. Plot-level estimated abundance and 95% confidence intervals of young forest focal songbird species during the breeding season. Differences in abundance within regeneration treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 3-4. Plot-level estimated abundance and 95% confidence intervals of forest interior focal songbird species during the breeding season. Differences in abundance within regeneration treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on 95% credible intervals.

Figure 3-5. Plot-level estimated abundance and 95% confidence intervals of forest gap focal songbird species during the breeding season. Differences in abundance within regeneration treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 3-6. Plot-level estimated abundance and 95% confidence intervals of focal songbird species during the breeding season. Differences in abundance between regeneration treatments are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 3-7. Plot-level species richness and 95% confidence intervals of avian habitat guilds during the breeding season. Differences in richness within regeneration treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 3-8. Plot-level species richness and 95% confidence intervals of avian guilds during the breeding season. Differences in richness within regeneration treatments between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 3-9. Plot-level estimated species richness and 95% confidence intervals of avian guilds during the breeding season. Differences in species richness between regeneration treatments are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 3-10. Plot-level relative abundance and 95% confidence intervals of eastern red-backed salamanders (*Plethodon cinereus*). Differences in relative abundance within the HS treatment between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

Figure 3-10. Plot-level body condition index (BCI) and 95% confidence intervals of eastern red-backed salamanders (*Plethodon cinereus*). Differences in BCI within the HS treatment between pre-treatment/control and one-year or two-year post-treatment are indicated by alphabetical notation based on non-overlapping 95% credible intervals.

9. Appendices

Appendix 3-A. Acreage managed by wildlife management area and year.

Wildlife Management Area	Year	Cutting Acreage	Herbicide Acreage
Beury Mountain	Winter 2017-18	100 ac (40.5 ha)	50 ac (20.2 ha)
Center Branch	Winter 2017-18	60 ac (24.3 ha)	0 ac (0 ha)
Little Canaan	Winter 2016-17	80 ac (32.4 ha)	40 ac (16.2 ha)
Moncove Lake	Winter 2017-18	40 ac (16.2 ha)	20 ac (8.1 ha)
Short Mountain	Winter 2018-19	20 ac (8.1 ha)	10 ac (4 ha)
Sleepy Creek	Winter 2018-19	20 ac (8.1 ha)	10 ac (4 ha)

Year	Cutting Acreage	Herbicide Acreage
Winter 2016-17	80 ac (32.4 ha)	40 ac (16.2 ha)
Winter 2017-18	200 ac (80.9 ha)	70 ac (28.3 ha)
Winter 2018-19	40 ac (16.2 ha)	20 ac (8.1 ha)
Total	320 ac (129.5 ha)	130 ac (52.6 ha)

Appendix 3-B. Total detections for bird species in regeneration stands heard within 100-m of point count and used in analyses for this chapter. Auditory detection type indicates which vocalizations were used to determine species presence. An “*” indicates species was included in species-specific analysis. See Appendix 1-A for scientific names of species and guild assignments. Species of conservation priority (SCP) include those identified by Appalachian Mountains Joint Venture, Partners-in-Flight Appalachian region, or the West Virginia Division of Natural Resources State Wildlife Action Plan. Guild codes are: BF = bark foraging, FG = forest gap, FI = forest interior, IE = interior-edge, PC = primary cavity nester, SC = secondary cavity nester, YF = young forest.

Common Name	Guild	Auditory Detection Type	SCP	Count	Detection Probability
Acadian Flycatcher	FG	Song/Call	X	25	
American Goldfinch	YF	Song/Call		44	0.126
Black-and-white Warbler*	FI, BF	Song	X	91	0.327
Black-billed Cuckoo	IE	Song/Call	X	2	
Black-capped Chickadee	IE, SC, BF	Song		29	
Black-throated Blue Warbler	FI	Song	X	39	
Black-throated Green Warbler*	FG	Song		104	0.267
Blue-headed Vireo*	FI	Song		94	0.209
Blue-winged Warbler	YF	Song	X	14	
Brown Creeper	FI, SC, BF	Song	X	10	
Brown Thrasher	YF	Song		2	
Canada Warbler	IE	Song	X	17	
Carolina Chickadee	IE, SC	Song		17	
Carolina Wren	YF, SC	Song		12	
Cerulean Warbler	FG	Song	X	22	
Chestnut-sided Warbler*	YF	Song		62	0.212
Chipping Sparrow	YF	Song	X	10	
Common Yellowthroat*	YF	Song		55	0.320
Downy Woodpecker	IE, PC, BF	Call		23	
Eastern Towhee*	YF	Song	X	160	0.242
Eastern Wood-pewee*	IE	Song	X	79	0.355
Field Sparrow	YF	Song	X	13	
Gray Catbird	YF	Song/Call		16	
Great Crested Flycatcher	IE, SC	Song/Call		12	
Hairy Woodpecker	FI, PC, BF	Call		14	
Hooded Warbler*	FG	Song	X	87	0.239
House Wren	IE, SC	Song		3	
Indigo Bunting*	YF	Song		62	0.241
Kentucky Warbler	FG	Song	X	16	
Least Flycatcher	FI	Song/Call	X	2	
Louisiana Waterthrush	FI	Song	X	2	

Appendix 3-B continued.

Common Name	Guild	Auditory Detection Type	SCP	Count	Detection Probability
Magnolia Warbler	FI	Song		8	
Northern Flicker	IE, PC	Call	X	5	
Ovenbird	FI	Song		286	0.260
Pileated Woodpecker	FI, PC, BF	Call		18	
Pine Warbler	FI	Song		11	
Prairie Warbler	YF	Song	X	1	
Red-bellied Woodpecker	IE, PC, BF	Call		30	
Red-breasted Nuthatch	FI, SC, BF	Song/Call		3	
Red-eyed Vireo	FG	Song		265	0.378
Rose-breasted Grosbeak	IE	Song	X	18	
Scarlet Tanager	FI	Song	X	137	0.198
Swainson's Thrush	FI	Song		3	
Tufted Titmouse	FI, SC	Song/Call		97	0.260
Veery	IE	Song	X	23	
White-breasted Nuthatch	FG, SC	Song/Call		63	0.162
White-eyed Vireo	YF	Song		8	
Winter Wren	FI, SC	Song		3	
Wood Thrush	FI	Song	X	82	0.171
Worm-eating Warbler	FI	Song	X	31	
Yellow-bellied Sapsucker	IE, PC, BF	Call	X	1	
Yellow-billed Cuckoo	IE	Song/Call	X	40	0.180
Yellow-breasted Chat	YF	Song	X	5	
Yellow-throated Warbler	FI	Song	X	10	
Yellow Warbler	YF	Song		4	

Appendix 3-C. Model posterior predictive check (Bayesian p -value) and model performance (deviance information criterion [DIC]; lower is better) for focal bird species, avian guild, and eastern red-backed salamander (*Plethodon cinereus*) treatment-time interaction and main effects model analyses. Bayesian p -value is the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic. We assumed reasonable model fit with a Bayesian p -value >0.1 and <0.9 .

Songbird		Bayesian p-value		DIC	
Common name	Scientific name	Treatment-Time	Main Effects	Treatment-Time	Main Effects
Black-and-white warbler	<i>Mniotilta varia</i>	0.643	0.653	3417.0	3376.6
Black-throated green warbler	<i>Setophaga virens</i>	0.317	0.310	3397.6	3434.3
Blue-headed vireo	<i>Vireo solitaries</i>	0.140	0.183	3538.4	3500.3
Chestnut-sided warbler	<i>Setophaga pensylvanica</i>	0.385	0.387	3201.5	3217.6
Common yellowthroat	<i>Geothlypis trichas</i>	0.708	0.710	3136.8	3142.5
Eastern towhee	<i>Pipilo erythrophthalmus</i>	0.325	0.315	3865.4	3918.1
Hooded warbler	<i>Setophaga citrina</i>	0.503	0.530	3542.5	3536.8
Indigo bunting	<i>Passerina cyanea</i>	0.410	0.433	3182.9	3183.5
Ovenbird	<i>Seiurus aurocapilla</i>	0.868	0.853	4085.9	4058.5
Red-eyed vireo	<i>Vireo olivaceus</i>	0.457	0.507	4150.2	4200.6
Tufted titmouse	<i>Baeolophus bicolor</i>	0.367	0.353	3606.9	2578.7
Eastern red-backed salamander					
Relative abundance		0.673	NA	374.7	NA
Body condition index		0.460	NA	310.6	NA

Appendix 3-D. Total number of individuals captured (excluding recaptures) of salamander species by regeneration stand treatment over the study period.

	Control	HS
Eastern Red-backed Salamander <i>(Plethodon cinereus)</i>		
Pre-treatment	38	24
One-year Post-treatment	43	45
Two-year Post-treatment	39	24
Mountain Dusky Salamander <i>(Desmognathus ochrophaeus)</i>		
Pre-treatment	1	5
One-year Post-treatment	0	2
Two-year Post-treatment	0	5
Northern Red Salamander <i>(Pseudotriton ruber)</i>		
Pre-treatment	1	0
One-year Post-treatment	0	0
Two-year Post-treatment	0	0
Slimy Salamander <i>(P. glutinosus)</i>		
Pre-treatment	4	7
One-year Post-treatment	4	8
Two-year Post-treatment	5	7
Wehrle's Salamander <i>(P. wehrlei)</i>		
Pre-treatment	2	1
One-year Post-treatment	0	2
Two-year Post-treatment	3	1

CHAPTER 4

EFFECTS OF LOCAL- AND LANDSCALE-LEVEL HABITAT FEATURES ON FOREST SONGBIRDS IN HUMAN-MODIFIED LANDSCAPES

Abstract

Land managers must be able to assess the influence of habitat features across multiple spatial scales when developing management plans. Although stand-level habitat management measures have a profound effect on species presence, broader spatial habitat features may limit the effectiveness of local-level management if not fully considered. In this chapter, we assessed the influence of local- (100-m radius surrounding point counts) and landscape-level (500-m radius) habitat features on the forest songbird community in cut-back borders along rights-of-way (ROWS) and wildlife openings. We examined four local-level variables (stand-level canopy cover, proportion of maintained early-successional habitat, proportion of young forest/shrubland habitat, and ROW/wildlife opening width) and five landscape-level variables (proportion of young forest/shrubland, core forest, mature forest, distance to nearest young forest/shrubland patch, and size of nearest young forest/shrubland patch) surrounding point counts to determine which variables had the strongest influence on species abundances and guild richness in cut-back borders. Landscape-level variables had the strongest influence on abundances and richness of young forest, forest gap, forest interior, and conservation priority species. Carolina wren and eastern towhee were more likely to occur in cut-back borders and ovenbirds were less likely to occur in cut-back borders that were located closer to pre-existing young forest/shrubland patches (see Appendix 1-A for scientific names). Size of nearest young forest/shrubland patch positively influenced common yellowthroat abundance, and negatively influenced black-and-white warbler and indigo bunting abundances. Proportion of young forest/shrubland habitat at the landscape-level positively influenced mature forest-associated species black-and-white warbler, cerulean warbler, and hooded warbler as well as the species of conservation priority guild. No local-level variables strongly influenced abundance or richness in cut-back borders. These results indicate that within forest dominated landscapes, cut-back borders located closer to pre-existing young forest/shrubland patches benefit young forest species while species responses to the size of nearest young forest/shrubland patch is likely species-specific. For mature forest species,

presence of small levels of young forest/shrubland area at the landscape level appears to increase abundances in cut-back borders.

1. Introduction

Wildlife habitat management often aims to manipulate vegetation features to meet the habitat needs of a particular species or group of interest. Stand-level management measures need to be cognizant of surrounding habitats across multiple spatial scales however, and how landscape matrices can affect species responses to local management in order to make effective management decisions (Askins et al. 2007, Desrochers et al. 2010, Grinde et al. 2017). Management is often focused at the local level because this scale often has the most profound effect on species distributions relative to landscape-level features (McDermott et al. 2011, Smith et al. 2011, Adams and Matthews 2019). However, when local-level management measures are implemented in concert with the surrounding landscape, the likelihood of success of local management often increases. This multi-spatial decisional hierarchy exhibited by species is best described as a gradient from features with high importance to features with low importance, which provides land managers with a select few habitat features to focus management efforts towards. However, the decisional hierarchy of a species is often site-dependent or generalized based on data from studies elsewhere. Currently, few studies have examined decisional hierarchies of species or groups within forestlands where linear energy infrastructure footprints dominate (e.g., underground gas pipelines, overhead utility powerlines, and seismic lines; Bayne et al. 2005, Mutter et al. 2015, Farwell et al. 2016). Understanding species decisional hierarchies within these anthropogenic landscapes will give managers management recommendations specific to these human-modified landscapes.

Partial canopy disturbance typically creates local vegetation characteristics beneficial to songbird species from a diverse array of habitat association groups (Sheehan et al. 2014). Local vegetation change, via cut-back borders, along abrupt forest edges of rights-of-way (hereafter “ROWs”) and wildlife openings can influence songbird species associated with young forest and forest interior habitats (see Chapter 2 for results). However, responses to local vegetation in ROWs and wildlife openings often depend on landscape-level features to some degree (Askins et al. 2012). This seems particularly likely when managing young forest wildlife habitats in association with ROWs and wildlife openings within forest-dominated landscapes. Rights-of-way are narrow and wildlife openings are small in size, and within forest-dominated landscapes

both are often isolated from other young forest or early-successional areas (Askins 1994). This isolation likely influences the effectiveness of local habitat management measures for young forest species along ROWs or wildlife openings (Roberts and King 2017). Further, vegetation type and associated species composition (e.g., herbaceous material in early-successional areas or woody plant material in young forest areas) in ROW corridors or wildlife openings may affect how young forest species respond to local management measures, which likely affects the effectiveness of local habitat management along ROWs and wildlife openings (Schlossberg et al. 2010).

Young forest wildlife species continue to face threats related to habitat loss and conversion (Trani et al. 2001, Dettmers 2003, Pardieck et al. 2019). The use of ROWs for managing and conserving young forest species has increased as ROWs have become more prominent on the landscape in West Virginia and the central Appalachian region (HIFLD 2019, EIA 2020). While ROWs and wildlife openings can provide needed habitat for young forest species, active management to improve these areas and optimize conservation measures has been generally unexplored. Further, little empirical evidence exists regarding the importance of habitat variables across multiple spatial scales for effectively managing young forest avifauna and other forest-associated songbird species along ROW and wildlife opening landscapes. For land managers to be able to optimize ROW and wildlife opening areas for forest songbirds, knowledge of how species respond to local- and landscape-level habitat features will be necessary. In this chapter, we assessed the relative importance of local- and landscape-level habitat variables (as it relates to songbirds) surrounding cut-back borders, and evaluated how early-successional vegetation (e.g., maintained grasslands and hay fields, shrubland, young forest) surrounding ROWs or wildlife openings affect avian responses in cut-back borders.

2. Methods

2.1. Study area

West Virginia lies within the central Appalachian region, which is characterized by extensive forests and humid continental climate with warm summers (daily mean = 22 °C) and cool to cold winters (daily mean = 1 °C). Elevations at study sites range from 224–1132 m. Annual mean precipitation levels are 76, 127, and 102 cm in the western, central, and eastern portions of the state, respectively, distributed relatively uniformly throughout the year but highest in spring and summer (NOAA 2019). This study was conducted on eight West Virginia

Division of Natural Resources' (WVDNR) wildlife management areas (WMAs) located throughout central and eastern West Virginia (Figure 2-1). Powerlines and pipelines used in this study were narrow linear openings (9–39 m in width [mean = 25.7, SD = 17.9 m]) with varying purposes (i.e., gathering pipelines and short- and long-distance powerlines). Vegetation in powerlines and pipelines consisted of grassland or low shrubland vegetation, was void of a mid- or over-story vegetation layer, and was distinct from adjacent forests. Pipelines and powerlines are hereafter referred to as “ROWS”. Wildlife openings used in this study were fields planted with a row crop (e.g., corn, clover) or left fallow for harvesting hay. Wildlife openings ranged 0.3–11.5 ha (mean = 3.1, SD = 3.1 ha) in size, ranged 19–97 m (mean = 57.8, SD = 21.7) in width, and were of varying shapes. Uses of wildlife openings (e.g., hay field, row crop) were consistent across years.

2.2. Study design

The eight study sites included 11 block replicates located along underground gas pipelines ($n = 4$), overhead utility powerlines ($n = 4$), or wildlife openings ($n = 3$). Study plots within block replicates were presumed to be subjected to similar biotic and abiotic environmental constraints. Block replicates along ROWs were located along one or two ROWs of similar widths within the same WMA and block replicates along wildlife openings were within the same WMA. A single block replicate consisted of seven independent experimental units (hereafter referred to as “cut-back borders”) with each assigned one of seven experimental treatments. Treatment designation for each cut-back border consisted of randomly selecting one of two target residual basal area levels (4.5 m²/ha or 14.0 m²/ha basal area retention) and one of three cutting depths into the forest (15 m [0.5 ha total area], 30 m [0.9 ha], or 45 m [1.4 ha] perpendicular to forest edge) with one cut-back border designated as a control (i.e., no tree cutting). Each cut-back border was 300 m in linear distance along the forest edge and ≥ 200 m apart to ensure sampling independence.

Local vegetation change in each cut-back border was induced via tree cutting using a feller-buncher or hand crew to mimic forest management prescriptions for clearcuts with reserves (~ 4.5 m²/ha) or crop tree/shelterwood stands (~ 14.0 m²/ha). Residual trees were based on species desirability (i.e., species of high wildlife value, such as oaks [*Quercus* spp.] and hickories [*Carya* spp.]). All trees ≥ 5 cm DBH were felled, excluding selected residual trees. Felled trees were dropped within the cut-back border and remained on the ground. Herbicide was

applied to non-native shrub species (e.g., autumn olive [*Elaeagnus umbellata*], Tatarian honeysuckle [*Lonicera tatarica*], and multiflora rose [*Rosa multiflora*]) to reduce presence in cut-back borders. Herbicide was applied in the fall and tree cutting occurred in the winter after pre-treatment data collection.

In each cut-back border, two avian and vegetation sampling points were spaced 150 m apart, 75 m from cut-back border ends, and located 15 m into the forest away from the ROW/wildlife opening edge. At each sampling point, we collected one- or two-years of post-treatment data on the avian community (see Table 2-1).

2.3. *Avian sampling*

We assessed songbirds in cut-back borders during the summer breeding period using 10-minute standardized fixed-radius point counts (Bibby et al. 2002). At each of the two point count locations within each cut-back border, we conducted two visits during 18 May–27 June in 2016–2019. Surveys began after 0530 and before 1000 (EST) on days with no rain, no heavy or gusty winds, and minimal peripheral noise. Observers recorded species of each individual bird detected, type of detection (song, call, visual, or flyover), demographic information (male, female, juvenile, or unknown), distance from observer placed into one of five distance intervals (0–25 m, >25–50 m, >50–75 m, >75–100 m, or >100 m), and time interval the individual was first detected (0–2 min, >2–4 min, >4–6 min, >6–8 min, and >8–10 min). Survey-level data were collected during each visit and included time of survey and ordinal day. Prior to surveys, observers were trained to independently identify all species visually and aurally while simultaneously recording distance and time interval. The surveyor skill variable used for analyses ranged from 3–5 after training where a score of 5 indicated thorough knowledge of species with extensive experience conducting point counts and a score of 3 indicated introductory knowledge of species with minimal experience conducting point counts. For all songbird data analyses, we used only detections within 50-m of the sampling point. We used this detection radius because sound attenuation of songbird vocalizations is species-specific and highly variable, and using a larger sampling radius (e.g., 100-m radius) may reduce detection rates of certain species that have higher frequency vocalizations (Schieck 1997, Yip et al. 2017). We included birds detected by sight and sound (excluding flyovers) for individual species and community statistical analyses.

2.4. *Local and landscape metric derivation*

We derived local- and landscape-level variables centered on each point count location from 1-m resolution leaf-on aerial imagery from 2018 National Agriculture Imagery Program (NAIP 2018) and 3-m resolution digital elevation models (DEMs) from West Virginia Statewide Addressing and Mapping Board (SAMB 2008; WVGIS 2020), as well as personal knowledge of the study sites. We manually digitized land cover classes (Table 4-1) in the program ArcGIS (ESRI 2014) then converted maps to 1-m resolution raster grids. Breaks in land classes (i.e., roads through mature forest) were determined when breaks were >3 m. Using raster grids, we calculated proportion of land cover for each local- and landscape-scale land class and core forest area in FRAGSTATS (version 4; McGarigal et al. 2012).

Local-level variables represented data collected at each point-count location or within a 100-m radius. A 100-m radius encompasses a 3.1 ha area and represents the local extent surrounding each point count location and roughly approximates the size of a songbird territory (Springborn and Meyers 2005, Stober and Kremenetz 2006, Frantz et al. 2016). We included percent canopy cover (*CC*) at each point count location as a surrogate for cut-back border harvest intensity. We measured canopy cover using a 25 cm x 25 cm transparent plexiglass panel, divided into a 5 x 5 grid and held overhead (Haché et al. 2013). Within the 100-m radius surrounding each point-count location, we measured the proportion of young forest (even-aged timber harvest occurring within the past 10 years and included the cut-back border into calculation) or shrubland habitat (areas with dense woody understory growth and no overstory cover; *YF100*), proportion of maintained habitat (habitats subjected to period maintenance, including grassland, hay field, row crop, or agriculture field; *MTN100*), width of ROW corridor or wildlife opening (*WID*), elevation (*ELEV*), and topographic relative moisture index (*TRMI*). Width of ROW corridor or wildlife opening (*WID*) adjacent to each point count location was measured from NAIP aerial photos. Elevation (*ELEV*) data were derived from DEMs. Topographic relative moisture index (*TRMI*) was derived from slope percent, slope aspect, slope position, and slope configuration data in DEMs and represents a moisture gradient from xeric to mesic soils (Parker 1982). We calculated continuous variables *ELEV* and *TRMI* using zonal statistics in the program ArcGIS (version 10.0; ESRI 2014) to generate a mean value within a 50-m radius surrounding each point count location.

Landscape-level variables represented data within a 500-m radius surrounding each point-count location, which encompassed a 78.5 ha area. We chose 500-m as a landscape-level radius because previous studies have shown that the 500-m extent captures species responses to landscape-level variables in ROW landscapes (Farwell et al. 2016). We did not use a larger spatial extent because the proportion of landscape-level variables did not change drastically beyond 500-m extent. Landscape-level variables included proportion of mature forest (areas where the majority of woody stems are ≥ 28.0 cm DBH, as per USDA Forest Service's Forest Inventory and Analysis classification, and was ground-truthed with aerial imagery using personal knowledge of the study sites; *MF*), proportion of core forest (*CF*), proportion of young forest or shrubland habitat (which included cut-back borders; *YF500*), distance to nearest young forest/shrubland patch (*DIST*), and size of nearest young forest/shrubland patch (*SIZE*). Core forest (*CF*) was defined as the amount of mature forest ≥ 100 m from any edge visible on NAIP aerial photos (Didham and Lawton 1999, Hofmeister et al. 2019). All variables were selected based on assumed importance to the songbird community, ease of measurement and conceptualization, and ability to implement by land managers.

2.5. Statistical analyses

We used a stacked Bayesian *N*-mixture model to evaluate the effects of local- and landscape-level factors on individual songbird species and avian guilds (Kéry and Royle 2016). For individual songbird analysis, we used the same detection criteria as in Chapter 2 and included species that had ≥ 40 detections and ≥ 0.2 cumulative detection probability (i.e., probability of detecting a species at least once across two annual survey visits; see Appendix 2-C for species-level detections, guild association, and conservation status). For avian guild richness analysis, we used the young forest habitat guild (18 total species detected), forest gap specialist guild (seven species), forest interior habitat guild (19 species), and species of conservation priority (27 species). We used post-treatment data only (one-year and two-year post-treatment) for analyses to reflect habitat changes from management in the local- and landscape-level covariates. We did not incorporate time as a covariate of interest in the analyses as habitat variables tested did not change over the 2-year time-span.

We fit a single model for each species or guild composed of a detection sub-model and an abundance/richness sub-model. We modeled conditional detection probability from two annual survey visits with a binomial process model using ordinal day, time since sunrise, and surveyor

skill as covariates on the logit-linear scale. For individual songbird analyses, we used the count of unique individuals during each visit as our response variable in our detection sub-model. For avian guild analyses, we used the number of unique species during each visit as our response variable in our detection sub-model. We modeled expected species abundance or guild richness using a Poisson process model. We modeled expected abundance or richness λ for each species or guild at point-count i during year k as a function of an intercept β , random effects of point count (PC) and treatment plot (TP), two nuisance variables ($ELEV$ and $TRMI$), four local-level covariates, and five landscape-level covariates:

$$\log(\lambda_{ik}) = \beta + PC + TP + ELEV + TRMI + MTN100 + YF100 + CC + WID + YF500 + CF + MF + DIST + SIZE$$

We included a point count random effect (PC) to account for heterogeneity among points and repeated observation at the same point and a treatment plot random effect (TP) to account for the lack of independence between points within the same cut-back border. All variables were standardized to have a mean of 0 and standard deviation of one prior to analysis. Variables were tested for collinearity prior to analysis. If two variables were collinear, we removed one of the variables to avoid potentially inflated variances of collinear variables and incorrect coefficient sign of variables. Initially, the variables patchiness and CF were highly collinear and thus we removed patchiness.

We used diffuse prior distributions for all slope coefficients (Gaussian [mean = 0, SD = 100]). For our random point count and treatment plot effects, we used a Gaussian distribution (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma distribution ($\alpha = 1, \beta = 1$). We obtained posterior distributions of model parameters by running three parallel Markov chain Monte Carlo (MCMC) simulations of 40000 iterations with a burn-in of 20000 iterations at a thinning rate of 100, yielding 600 samples for posterior distributions. We assessed model fit with a posterior predictive check, where we compared Chi-square goodness-of-fit tests between an observed posterior predictive distribution and a simulated posterior predictive distribution. We calculated a Bayesian p -value, p_B , as the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic and assumed reasonable fit if $0.1 < p_B < 0.9$ (Gelman et al. 2014). Model convergence was assessed using the Brooks-Gelman-Rubin statistic and assumed adequate convergence when all parameter Rhats

values <1.1 (Brooks and Gelman 1998). Data analyses were performed in R (R Core Team 2019) using a Bayesian framework in JAGS (Plummer 2017) called from the package jagsUI (Kellner 2015). We considered a variable to have an influential effect on abundance of a given species or guild richness when the 95% credible interval around the estimated coefficient did not overlap zero.

3. Results

Landscape-level land cover within 500-m radius varied across study sites but all were dominated by mature forest cover (mean = 86%) with an average of 27% of this being core forest area (Table 4-1). Young forest/shrubland habitat was the predominant early-successional vegetation type at the local-level (100-m radius) with 23% of all land cover. Maintained early-successional habitats comprised 11% of all land cover within 100-m of point counts.

Our posterior predictive check indicated all individual species and avian guild models were a reasonable fit (Table 4-2). Eleven species from the young forest habitat guild, forest gap specialist habitat guild, forest interior habitat guild, or species of conservation priority met the detection criterion for species specific analyses (Table 4-3). The young forest habitat guild included five species (American goldfinch, Carolina wren, common yellowthroat, eastern towhee, and indigo bunting; scientific names for all species are provided in Table 4-3), the forest interior habitat guild included three species (black-and-white warbler, ovenbird, and wood thrush), and species of conservation priority included three species (cerulean warbler, eastern wood-pewee, and hooded warbler).

Of the variables tested, four landscape-level variables, *YF500*, *CF*, *DIST*, and *SIZE*, had a strong influence on abundance or richness of at least one of the 11 species and four guilds analyzed (Table 4-3; Figure 4-1). No local-level variables strongly influenced species responses. Three species (black-and-white warbler, cerulean warbler, and hooded warbler) and one guild (species of conservation priority) were positively influenced by *YF500*. Abundances of black-and-white warbler, cerulean warbler, and hooded warbler increased by 100.6%, 111.9%, and 79.1%, respectively, when *YF500* increased by 3.8% (2.9 ha). Richness of the species of conservation priority guild was also positively influenced by *YF500* and increased by 21.5% when *YF500* increased by 3.8%. Core forest had a negative influence on cerulean warbler abundance; they decreased by 72.7% when *CF* increased by 11.0% (8.6 ha increase). Distance to the nearest young forest/shrubland patch (*DIST*) had a positive influence on abundance of the

ovenbird, and a negative influence on the Carolina wren, eastern towhee, and hooded warbler. When distance to nearest patch of young forest decreased by 337.3 m (a one standard deviation), abundances of Carolina wren, eastern towhee, and hooded warbler increased by 69.0%, 32.8%, and 50.1%, respectively, and decreased in ovenbirds by 50.5%. The size of the nearest young forest/shrubland patch (*SIZE*) had a positive influence on common yellowthroat abundance and a negative influence on black-and-white warbler and indigo bunting abundances. Common yellowthroat abundance increased by 218.7%, and black-and-warbler and indigo bunting decreased by 53.6% and 55.4%, respectively, when *SIZE* increased.

4. Discussion

The proliferation of linear energy infrastructure in West Virginia and the central Appalachian region provides an opportunity for land managers to incorporate these man-made features into their management plans in an effort to manage the young forest songbird community. However, little information is available regarding the effects of habitat variables across multiple spatial scales on the young forest community in order to successfully promote ROW areas. In Chapter 2, we provided evidence that cut-back borders can increase habitat suitability along ROW and wildlife opening areas for young forest avifauna without compromising abundance of forest interior avifauna. In order to expand on these findings and continue developing wildlife management guidelines for these areas, here we have assessed the effects of habitat variables at the local- and landscape-level on songbirds in cut-back borders to determine their relative importance for managing avifauna in these landscapes. This information will provide guidance to managers on where in the landscape to place cut-back borders to optimize efforts that improve habitat suitability of cut-back border for the songbird community.

The landscape-level variable *YF500* had a strong, positive influence on forest gap (cerulean warbler and hooded warbler) and forest interior (black-and-white warbler) species as well as the species of conservation priority guild which includes four forest gap and nine forest interior species (out of 27 total species). Despite *YF500* only accounting for 3.8% of landcover on average (3.0 of 78.5 ha), an increase of 2.7 ha of young forest/shrubland increased abundances of black-and-white warbler, cerulean warbler, and hooded warbler and richness of species of conservation priority. The strong relationship to *YF500* by mature forest-associated species likely indicates that within these mature forest-dominated landscapes, small increases in young forest/shrubland area increases the habitat suitability of cut-back borders for these species.

Greater forest age heterogeneity likely benefits mature forest species because young forest/shrubland areas provide resources not provided in mature forests. Forest gap species use edges of small young forest/shrubland patches within contiguous mature forests for foraging, perching, and preening activities due to the increased vertical vegetation diversity provided in these areas (Weakland and Wood 2005, Bakermans and Rodewald 2009, Wood and Perkins 2012). Young forest/shrubland patches are also frequently used by mature forest-associated species during the post-breeding period due to the abundance of food resources and vegetative cover (Vitz and Rodewald 2006, McDermott and Wood 2010, Chandler et al. 2012). Therefore, breeding mature forest adult songbirds likely locate nests within 500 m of young forest/shrubland patches where fledging birds can access young forest/shrubland patches during the post-breeding period (White and Faaborg 2008, Delancey and Islam 2019). Thus, cut-back borders located in areas with greater amounts of young forest/shrubland increase a juvenile's access to these areas and possibly increase their survival. Surprisingly, no young forest species responded to increased total amount of young forest/shrubland at the landscape scale. Shoe (2018) reported that young forest species occurrence was positively associated with shrubland habitat within a 500-m buffer surrounding point counts in human-modified landscapes (e.g., sand and gravel mines, transmission line rights-of-way), however, Askins et al. (2007) reported surrounding landscape composition was not an important predictor of abundance of young forest species in wildlife openings. Although, our results indicate that *YF500* does not appear to be an important driver of abundance or richness of young forest avifauna in cut-back borders, they did respond to distance or size of nearest young forest/shrubland patch.

Abundances of Carolina wren and eastern towhee in cut-back borders was greater when cut-back borders were placed closer to existing young forest/shrubland areas. Within these mature forest-dominated landscapes, cut-back borders were on average 245 m (15–2214 m) from the closest young forest/shrubland patch. The close proximity to young forest/shrubland habitats likely enhances cut-back borders by providing greater access to resources that young forest species are proficient at exploiting as well as promoting nearby areas through conspecific attraction (Fletcher Jr. 2007). Further, close proximity to young forest/shrubland areas provides a source population of young forest species that can more quickly colonize cut-back border harvests (Schlossberg 2009). Thus, isolation of cut-back borders from existing young forest/shrubland patches may reduce the effectiveness of cut-back borders for young forest

wildlife management. Buffum and McKinney (2014) and Roberts and King (2017) both reported that young forest species occurred at greater rates in shrublands or wildlife openings when located closer to large young forest/shrubland patches. However, in our study, only the common yellowthroat was positively influenced by the size of the nearest young forest/shrubland patch while the indigo bunting was negatively influenced by *SIZE*, which is consistent with previous studies on patch size relationships for these species (Rodewald and Vitz 2005, Lehen and Rodewald 2009). Young forest species are capable of occupying small patches (as small as 0.23 ha) of young forest/shrubland habitats within predominately forested landscapes, but these relationships are often species-specific (Shake et al. 2012, George et al. 2019). Nearby young forest/shrubland patches in our study ranged 0.45–15.24 ha in size (37% of all patches were ≤ 1 ha). Species in cut-back borders may not be responding as strongly to the size of nearby young forest/shrubland patches within these landscapes because all our patches were above the minimum 0.23 ha size reported by previous studies. These results suggest that *DIST* has a more consistent influence on young forest species presence because cut-back borders are less isolated when located closer to other young forest/shrubland patches, while the influence of *SIZE* on young forest species is likely species-specific.

We observed varied and generally weak effects of *CC* (a surrogate of cut-back border harvest intensity), *MTN100*, *YF100*, and *YF500* on young forest species with the exception of species of conservation priority, which included seven young forest species (out of 27 species). Surprisingly, we did not observe a strong response to *YF100* by young forest species. Past research has shown that young forest species in ROWs and wildlife openings are likely limited by the amount of surrounding early-successional habitat (Rodewald and Vitz 2005, Schlossberg and King 2008), and therefore we would expect this variable to strongly influence species in these areas. Higher levels of early-successional habitat surrounding wildlife openings increase local young forest species abundances (Askins et al. 2007, Roberts and King 2017, George et al. 2019). This trend is similar for ROW landscapes, as Farwell et al. (2016) reported early-successional species were strongly influenced by edge density within 100-m (a variable that had a strong, positive correlation with *YF100* in our study and thus not included in analysis). Additionally, we did not include a landscape-level metric that accounted for patchiness of the landscape (because of the high collinearity with *CF*).

5. Management Implications

The footprint of linear energy infrastructure within predominantly forested landscapes and its implications on the wildlife community continues to be an important topic with broad reaching consequences for the central Appalachian region. In this study, we found evidence to indicate that the young forest songbird community responded to landscape-level variables that potentially improve habitat suitability in cut-back borders along ROWs and wildlife openings. Placement of cut-back borders closer to pre-existing patches of young forest/shrubland habitat appears to improve habitat suitability for young forest songbirds. Size of nearest young forest/shrubland patch and total amount seem to not be as strong of a factor as distance in determining young forest responses to cut-back borders, possibly due to species-specific variation in area-sensitivity thresholds. Thus, managers can optimize placement of cut-back borders to benefit young forest species by choosing locations within ~300 m of pre-existing young forest/shrubland patches (Figure 4-1).

The presence of young forest at the landscape-level was the most important habitat variable assessed in this study for mature forest and species of conservation priority species. Thus, within forest-dominated landscapes, a small amount of pre-existing young forest/shrubland habitat (~10%; Figure 4-1) is beneficial for the management and conservation of the mature forest songbird community. It should be noted that this study was conducted within a predominantly mature forested landscape. Within this context, young forest/shrubland at relatively low levels appear to benefit mature forest interior species. However, as mature forest levels decrease below certain thresholds, the relationship we observed between mature forest interior species and young forest/shrubland area may change (Becker et al. 2015, Farwell et al. 2020).

Before this study, limited information existed about how best to manage the forest bird community using ROW or wildlife opening areas within the context of the surrounding landscape. This research helps fill a vital knowledge gap on what variables and at what spatial scales are important for the management and conservation of forest songbirds. Our results indicated that across all species, landscape-level variables were the most influential to species abundance and richness in cut-back borders. However, for young forest species, presence of local-level habitat manipulation like cut-back borders likely promotes these areas. Without cut-back borders at the local-level, landscape-level variables may not be as influential as we report. Prior to implementing local habitat measures, managers need to be aware of how surrounding

habitats across multiple spatial scales may affect their management decisions. Future research on thresholds of local- and landscape-level habitats for young forest and mature forest songbirds in cut-back borders will help inform managers about the surrounding limitations of habitats related to cut-back borders in forest-dominated landscapes. Additionally, longer-term monitoring in order to fully evaluate spatio-temporal effects of local- and landscape-level habitats on songbirds will be valuable for developing long-term management plans.

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7. Tables

Table 4-1. Percent cover and range (minimum and maximum) of local-level (100-m radius) and landscape-level (500-m radius) land cover variables surrounding each point count station at each Wildlife Management Area. Variable abbreviations include, MTN100 = proportion of maintained habitat (grassland, hay field, agriculture, row crop) within 100-m surrounding the point count, YF100 = proportion of young forest/shrubland within 100-m surrounding the point count, YF500 = proportion of young forest/shrubland within 500-m surrounding the point count, CF = core forest within 500-m surrounding the point count, and MF = proportion of mature forest within 500-m surrounding the point count.

Wildlife Management Area	Local-level		Landscape-level		
	MTN100	YF100	YF500	CF	MF
Allegheny	10.4 (0.0, 25.0)	16.6 (0.0, 29.0)	1.3 (0.0, 6.8)	30.9 (14.0, 39.5)	92.2 (64.2, 97.2)
Beury Mountain	0.0 (0.0, 0.0)	34.5 (15.9, 56.2)	6.7 (3.3, 20.6)	29.1 (6.8, 42.7)	91.7 (76.7, 96.5)
Bluestone	29.7 (16.4, 42.3)	17.3 (0.0, 29.0)	1.2 (0.0, 3.4)	15.5 (7.0, 24.7)	72.0 (60.0, 84.9)
Burnsville Lake	5.6 (3.7, 12.1)	16.6 (0.0, 29.0)	1.8 (0.4, 6.6)	22.1 (13.4, 36.5)	87.7 (74.7, 96.9)
Lewis Wetzel	11.8 (0.0, 31.4)	19.2 (0.0, 46.1)	2.4 (0.0, 8.5)	27.7 (12.5, 42.0)	87.2 (73.4, 97.2)
Little Canaan	0.0 (0.0, 0.0)	31.8 (12.3, 46.9)	5.9 (3.1, 13.0)	30.7 (16.9, 39.9)	91.9 (80.2, 96.7)
Sleepy Creek	15.6 (10.6, 21.0)	16.6 (0.0, 29.0)	0.7 (0.0, 1.2)	41.4 (31.5, 55.8)	96.5 (94.8, 98.2)
Stonewall Jackson	36.3 (20.8, 58.1)	17.2 (0.0, 37.7)	4.2 (0.4, 19.2)	8.8 (0.0, 24.6)	60.2 (42.4, 77.5)
Mean	11.0	23.2	3.5	26.7	86.4

Table 4-2. Model posterior predictive check for each species and guild used in analysis. Bayesian p -value was the probability to obtain a Chi-square test statistic that is at least as extreme as the observed Chi-square test statistic. We assumed reasonable model fit if the Bayesian p -value of >0.1 and <0.9 .

Common name	Scientific name	Bayesian p-value
American goldfinch	<i>Spinus tristis</i>	0.188
Black-and-white warbler	<i>Mniotilta varia</i>	0.335
Carolina wren	<i>Thryothorus ludovicianus</i>	0.193
Cerulean warbler	<i>Setophaga cerulea</i>	0.337
Common yellowthroat	<i>Geothlypis trichas</i>	0.313
Eastern towhee	<i>Pipilo erythrophthalmus</i>	0.303
Eastern wood-pewee	<i>Contopus virens</i>	0.363
Hooded warbler	<i>Setophaga citrina</i>	0.390
Indigo bunting	<i>Passerina cyanea</i>	0.603
Ovenbird	<i>Seiurus aurocapilla</i>	0.437
Wood thrush	<i>Hylocichla mustelina</i>	0.235
Avian Guilds		
Young forest habitat		0.623
Forest gap specialist		0.552
Forest interior habitat		0.353
Species of conservation priority		0.682

Table 4-3. Mean effect size (on the response scale) of local- and landscape-level variables on species abundance and guild richness. Table values indicate the percent change in species-specific abundance or guild-specific species richness when local- and landscape-level variables increase by one standard deviation (“Variable Increase” in table). Numbers in bold indicate 95% credible interval of mean effect size did not overlap 0. Variable abbreviations include, MTN100 = proportion of maintained habitat within 100-m surrounding the point count, YF100 = proportion of young forest/shrubland within 100-m surrounding the point count, CC = percent canopy cover at the point count, WID = width of right-of-way or wildlife opening, YF500 = proportion of young forest/shrubland within 500-m surrounding the point count, CF = core forest within 500-m surrounding the point count, MF = proportion of mature forest within 500-m surrounding the point count, DIST = distance to nearest young forest/shrubland patch, and SIZE = size of nearest young forest/shrubland patch.

	Local-level				Landscape-level				
	MTN100	YF100	CC	WID	YF500	CF	MF	DIST	SIZE
Variable Increase	13.3%	13.3%	35.1%	34.1%	3.8%	11.0%	11.8%	337.3 m	4.3 ha
Individual Species									
Young Forest Guild									
American goldfinch	38.4	13.5	-7.1	3.0	-2.8	-24.3	-3.7	-2.4	-1.5
Carolina wren	-41.2	-9.8	81.1	15.7	-42.0	48.0	-59.8	-69.0	-55.6
Common yellowthroat	-56.9	-28.9	-6.3	20.9	-11.0	92.9	-58.9	-12.8	218.7
Eastern towhee	41.6	6.2	-17.1	8.9	11.1	14.6	16.5	-32.8	12.7
Indigo bunting	14.1	19.5	-4.1	1.2	12.7	19.5	-23.0	7.8	-55.4
Forest Interior Guild									
Black-and-white warbler	9.3	0.7	13.0	-13.5	100.6	28.8	108.7	-19.8	-53.6
Ovenbird	-48.9	4.1	18.1	13.8	-13.3	7.8	50.5	50.5	14.1
Wood thrush	-59.4	2.9	17.4	-14.8	-19.5	35.4	-29.7	30.5	-16.7
Species of Conservation Priority									
Cerulean warbler	-3.2	-0.5	-2.6	26.5	111.9	-72.7	155.5	-30.6	28.9
Eastern wood-pewee	22.8	3.6	-13.4	-21.1	26.6	3.6	123.4	36.6	-8.1
Hooded warbler	33.6	-12.4	15.4	-32.6	79.1	-12.0	104.0	-50.1	-16.8
Avian Guilds									
Young forest habitat	13.3	0.7	-5.8	2.8	7.4	19.1	-20.6	-6.9	8.4
Forest gap specialist	-13.7	-10.2	2.5	-4.8	18.2	-14.1	9.1	-14.0	-8.6
Forest interior habitat	-5.3	3.0	16.3	3.0	14.8	9.6	21.4	11.2	-20.1
Species of conservation priority	0.6	-0.4	4.7	-5.3	21.5	4.7	3.7	-0.2	-8.6

8. Figures

Figure 4-1. Expected point-level mean species abundance or guild richness and 85% confidence intervals as a function of local- and landscape-level variables surrounding point counts in cut-back borders. Data on x-axis for variables includes the 95% quantile to reduce inflation of estimates near data outliers.

9. Appendices

Appendix 4-A. Coordinates of treatment plot corners for cut-back borders. All coordinates are in NAD83, UTM Zone 17. Control plots have two corners indicating the location of their plot location along the forest edge.

Study Site: Allegheny WMA			Study Site: Bluestone WMA		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	668713	4364639	Control	513258	4143520
	669010	4364630		513393	4143772
15 m, 4.5 m ² /ha	668414	4364648	15 m, 4.5 m ² /ha	512085	4147258
	668354	4364354		512072	4147251
	668339	4364357		511882	4147567
	668400	4364651		511877	4147543
15 m, 14 m ² /ha	663400	4358311	15 m, 14 m ² /ha	511292	4147067
	663672	4358184		511278	4147073
	663678	4358197		511185	4146795
	663406	4358325		511196	4146784
30 m, 4.5 m ² /ha	668431	4364840	30 m, 4.5 m ² /ha	513013	4143204
	668261	4365089		513022	4143176
	668284	4365108		512938	4142946
	668457	4364855		512909	4142953
30 m, 14 m ² /ha	667908	4363299	30 m, 14 m ² /ha	512774	4142804
	668021	4363572		512799	4142821
	667994	4363586		512674	4142537
	667938	4363293		512694	4142514
45 m, 4.5 m ² /ha	668107	4363752	45 m, 4.5 m ² /ha	512905	4145444
	668211	4364017		512887	4145402
	668167	4364027		513157	4145272
	668066	4363770		513177	4145312
45 m, 14 m ² /ha	662945	4358519	45 m, 14 m ² /ha	512724	4145521
	663218	4358395		512705	4145480
	663237	4358436		512394	4145601
	662964	4358560		512424	4145567

Appendix 4-A continued.

Study Site: Beury Mountain WMA Block 1			Study Site: Beury Mountain WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	499078	4199170	Control	500366	4198119
	499078	4198883		500638	4197993
15 m, 4.5 m ² /ha	500096	4199967	15 m, 4.5 m ² /ha	501276	4197705
	500091	4199981		501282	4197719
	500379	4200064		501556	4197595
	500383	4200050		501549	4197581
15 m, 14 m ² /ha	500578	4200120	15 m, 14 m ² /ha	498362	4199001
	500574	4200106		498368	4199015
	500863	4200030		498641	4198891
	500868	4200044		498635	4198878
30 m, 4.5 m ² /ha	501740	4199800	30 m, 4.5 m ² /ha	499151	4198708
	501743	4199831		499140	4198680
	502044	4199752		499415	4198561
	502037	4199723		499427	4198588
30 m, 14 m ² /ha	498268	4198747	30 m, 14 m ² /ha	497901	4199198
	498274	4198718		497914	4199225
	498569	4198766		498186	4199098
	498565	4198796		498173	4199071
45 m, 4.5 m ² /ha	501059	4199983	45 m, 4.5 m ² /ha	499894	4198313
	501066	4200025		499875	4198272
	501376	4199951		500149	4198149
	501367	4199908		500168	4198191
45 m, 14 m ² /ha	499615	4199831	45 m, 14 m ² /ha	500820	4197910
	499603	4199873		500838	4197950
	499890	4199960		501112	4197828
	499902	4199917		501093	4197786

Appendix 4-A continued.

Study Site: Burnsville Lake WMA			Study Site: Sleepy Creek WMA		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	533878	4295283	Control	745621	4380729
	533967	4295539		745616	4380732
15 m, 4.5 m ² /ha	533056	4295470	15 m, 4.5 m ² /ha	740296	4364980
	533057	4295484		740310	4364976
	532770	4295541		740297	4364983
	532782	4295549		740310	4364976
15 m, 14 m ² /ha	533669	4294470	15 m, 14 m ² /ha	740553	4365139
	533683	4294465		740544	4365151
	533802	4294739		740522	4365135
	533815	4294730		740533	4365124
30 m, 4.5 m ² /ha	532567	4295880	30 m, 4.5 m ² /ha	740608	4366814
	532269	4295897		740603	4366813
	532266	4295867		740594	4366842
	532565	4295851		740598	4366843
30 m, 14 m ² /ha	533721	4295116	30 m, 14 m ² /ha	740281	4365261
	533711	4295088		740281	4365231
	533819	4294854		740253	4365411
	533789	4294851		740223	4365407
45 m, 4.5 m ² /ha	532059	4296240	45 m, 4.5 m ² /ha	740439	4366331
	532172	4295973		740393	4366342
	532209	4295996		740439	4366331
	532106	4296239		740393	4366342
45 m, 14 m ² /ha	533569	4295242	45 m, 14 m ² /ha	745261	4383007
	533597	4295277		745291	4383041
	533362	4295440		745230	4383019
	533370	4295484		745225	4383064

Appendix 4-A continued.

Study Site: Lewis Wetzel WMA Block 1			Study Site: Lewis Wetzel WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	529391	4374319	Control	529157	4372088
	529562	4374084		529216	4371776
15 m, 4.5 m ² /ha	528184	4373598	15 m, 4.5 m ² /ha	529624	4373163
	528170	4373604		529633	4373150
	528156	4373876		529859	4373347
	528168	4373885		529845	4373353
15 m, 14 m ² /ha	529641	4373917	15 m, 14 m ² /ha	529913	4373535
	529656	4373916		529924	4373525
	529774	4373699		530203	4373567
	529770	4373684		530204	4373582
30 m, 4.5 m ² /ha	528982	4373290	30 m, 4.5 m ² /ha	528369	4373306
	528952	4373288		528341	4373296
	529035	4373572		528508	4373048
	529065	4373574		528532	4373066
30 m, 14 m ² /ha	530917	4372626	30 m, 14 m ² /ha	529238	4372724
	530887	4372625		529268	4372732
	530884	4372354		529318	4372417
	530915	4372353		529290	4372421
45 m, 4.5 m ² /ha	528895	4374812	45 m, 4.5 m ² /ha	528660	4372884
	528848	4374813		528655	4372839
	528999	4374544		528931	4372823
	528964	4374517		528962	4372856
45 m, 14 m ² /ha	529035	4374076	45 m, 14 m ² /ha	530417	4373548
	528992	4374064		530402	4373506
	529005	4373786		530632	4373344
	529049	4373780		530666	4373373

Appendix 4-A continued.

Study Site: Little Canaan WMA Block 1			Study Site: Little Canaan WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	635245 635364	4332188 4331917	Control	633883 634090	4331349 4331138
15 m, 4.5 m ² /ha	635063 635049 635179 635165	4332645 4332639 4332379 4332373	15 m, 4.5 m ² /ha	634846 634862 635069 635056	4330419 4330438 4330230 4330216
15 m, 14 m ² /ha	635143 635128 635013 635028	4332338 4332332 4332603 4332609	15 m, 14 m ² /ha	634150 634137 634354 634338	4331140 4331127 4330939 4330923
30 m, 4.5 m ² /ha	635973 636000 636117 636090	4330518 4330530 4330256 4330244	30 m, 4.5 m ² /ha	634485 634507 634712 634691	4330784 4330804 4330594 4330574
30 m, 14 m ² /ha	635562 635533 635679 635651	4331539 4331527 4331271 4331259	30 m, 14 m ² /ha	633812 633788 633991 634015	4331504 4331480 4331271 4331295
45 m, 4.5 m ² /ha	636027 635983 636148 636104	4330306 4330286 4330039 4330018	45 m, 4.5 m ² /ha	635039 635007 634829 634797	4330200 4330167 4330404 4330372
45 m, 14 m ² /ha	635409 635368 635291 635249	4331734 4331716 4332003 4331985	45 m, 14 m ² /ha	634248 634212 634455 634420	4330990 4330955 4330776 4330741

Appendix 4-A continued.

Study Site: Stonewall Jackson WMA			Study Site: Nathaniel Mountain WMA		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	549134	4312357	Control	688368	4336922
	549417	4312269		688664	4336868
15 m, 4.5 m ² /ha	549243	4310959	15 m, 4.5 m ² /ha	687513	4335997
	549012	4310904		687810	4335955
	549026	4310893		687812	4335970
	549231	4310944		687515	4336012
15 m, 14 m ² /ha	549146	4311067	15 m, 14 m ² /ha	688008	4335928
	549157	4311081		688306	4335891
	548970	4310891		688308	4335906
	548954	4310898		688010	4335943
30 m, 4.5 m ² /ha	543305	4307028	30 m, 4.5 m ² /ha	687874	4336998
	543336	4307022		688171	4336952
	543398	4306859		688175	4336982
	543430	4306861		687879	4337028
30 m, 14 m ² /ha	543673	4306116	30 m, 14 m ² /ha	689145	4336746
	543703	4306101		688849	4336791
	543715	4306316		688844	4336762
	543738	4306333		689141	4336716
45 m, 4.5 m ² /ha	544844	4306494	45 m, 4.5 m ² /ha	687425	4335897
	544864	4306541		687128	4335936
	544726	4306662		687122	4335891
	544770	4306655		687419	4335853
45 m, 14 m ² /ha	544256	4308041	45 m, 14 m ² /ha	687623	4335868
	544263	4308092		687920	4335824
	543970	4308186		687913	4335780
	543930	4308155		687616	4335823

Appendix 4-B. Coordinates of treatment plot corners for regeneration stands. All coordinates are in NAD83, UTM Zone 17.

Study Site: Beury Mountain WMA Block 1			Study Site: Beury Mountain WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	500716	4199554	Control	501218	4198793
	500882	4199552		501378	4198792
	500884	4199301		501381	4198543
	500718	4199301		501216	4198543
Clearcut-and-leave	501050	4199552	Clearcut-and-leave	501882	4199301
	501217	4199554		502048	4199301
	500217	4199303		502045	4199047
	501050	4199303		501883	4199047
Clearcut-and-windrow	500383	4199555	Clearcut-and-windrow	502049	4198792
	500548	4199554		502216	4198793
	500548	4199303		502215	4198540
	500383	4199303		502049	4198541
Hack-and-spray	499552	4200315	Hack-and-spray	501548	4198538
	499716	4200315		501717	4198538
	499716	4200062		501715	4198288
	499550	4200062		501547	4198287

Study Site: Beury Mountain WMA Block 3			Study Site: Beury Mountain WMA Block 4		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	497553	4198032	Control	496879	4197785
	497720	4198031		497048	4197784
	497719	4197779		497048	4197531
	497552	4197778		496880	4197530
Clearcut-and-leave	497216	4197786	Clearcut-and-leave	496886	4199047
	497384	4197785		497052	4199047
	497383	4197532		497053	4198795
	497215	4197532		496886	4198794
Clearcut-and-windrow	497549	4197270	Clearcut-and-windrow	497215	4198293
	497720	4197271		497384	4198293
	497719	4197015		497382	4198038
	497551	4197016		497215	4198037
Hack-and-spray	498381	4197529	Hack-and-spray	496881	4198292
	498549	4197528		497048	4198294
	498548	4197275		497048	4198037
	498383	4197275		496881	4198038

Appendix 4-B continued.

Study Site: Beury Mountain WMA Block 5			Study Site: Center Branch WMA Block 1		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	503714	4199301	Control	556514	4344259
	503879	4199302		556684	4344151
	503881	4199049		556572	4343982
	503714	4199049		556405	4344092
Clearcut-and-leave	503048	4199046	Clearcut-and-leave	556222	4343818
	503214	4199046		556392	4343706
	503212	4198793		556279	4343541
	503049	4198795		556111	4343650
Clearcut-and-windrow	503380	4199300	Clearcut-and-windrow	556848	4344037
	503547	4199301		557019	4343928
	503547	4199045		556906	4343761
	503382	4199047		556740	4343870
Hack-and-spray	503714	4199809	Hack-and-spray	No hack-and-spray treatments were treated at Center Branch WMA	
	503883	4199806			
	503879	4199556			
	503712	4199554			

Study Site: Center Branch WMA Block 2			Study Site: Center Branch WMA Block 3		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	558083	4342986	Control	558017	4344714
	557972	4342818		558185	4344604
	558139	4342706		558075	4344434
	558250	4342875		557906	4344546
Clearcut-and-leave	557747	4343207	Clearcut-and-leave	557444	4343645
	557915	4343097		557612	4343535
	557804	4342929		557721	4343701
	557636	4343040		557554	4343811
Clearcut-and-windrow	557278	4342699	Clearcut-and-windrow	557379	4343443
	557445	4342588		557546	4343333
	557168	4342530		557269	4343275
	557338	4342419		557437	4343165
Hack-and-spray	No hack-and-spray treatments were treated at Center Branch WMA		Hack-and-spray	No hack-and-spray treatments were treated at Center Branch WMA	

Appendix 4-B continued.

Study Site: Little Canaan WMA Block 1			Study Site: Little Canaan WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	635221	4330721	Control	635426	4330944
	635069	4330592		635227	4330949
	635198	4330441		635232	4331148
	635351	4330567		635431	4331144
Clearcut-and-leave	635663	4330205	Clearcut-and-leave	635787	4331931
	635514	4330073		635922	4331781
	635381	4330221		635771	4331649
	635529	4330356		635638	4331797
Clearcut-and-windrow	634689	4331236	Clearcut-and-windrow	635741	4332407
	634536	4331108		635874	4332258
	634407	4331261		635725	4332125
	634560	4331389		635592	4332272
Hack-and-spray	634236	4331707	Hack-and-spray	634911	4332276
	634084	4331578		635045	4332127
	634213	4331425		634895	4331994
	634366	4331553		634761	4332143

Study Site: Little Canaan WMA Block 3			Study Site: Little Canaan WMA Block 4		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	636214	4333378	Control	637476	4332400
	636372	4333255		637319	4332523
	636249	4333097		637442	4332680
	636091	4333220		637598	4332557
Clearcut-and-leave	636534	4332596	Clearcut-and-leave	636721	4333198
	636691	4332473		636878	4333077
	636569	4332315		636756	4332918
	636411	4332438		636598	4333041
Clearcut-and-windrow	635506	4332884	Clearcut-and-windrow	637087	4332946
	635663	4332761		637245	4332823
	635540	4332603		637122	4332665
	635382	4332725		636964	4332788
Hack-and-spray	636104	4332892	Hack-and-spray	636218	4331866
	636260	4332770		636375	4331743
	636138	4332612		636253	4331585
	635981	4332735		636095	4331709

Appendix 4-B continued.

Study Site: Moncove Lake WMA Block 1			Study Site: Moncove Lake WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	557975	4164284	Control	557515	4162908
	558099	4164125		557639	4162749
	558257	4164251		557480	4162625
	558134	4164407		557356	4162783
Clearcut-and-leave	558698	4164340	Clearcut-and-leave	557993	4163280
	558822	4164183		557833	4163154
	558663	4164059		557957	4162998
	558540	4164214		558115	4163120
Clearcut-and-windrow	558063	4163844	Clearcut-and-windrow	558432	4163371
	558187	4163687		558309	4163528
	558028	4163560		558151	4163405
	557905	4163718		558274	4163245
Hack-and-spray	558382	4164092	Hack-and-spray	557586	4163472
	558503	4163935		557711	4163313
	558345	4163809		557868	4163437
	558221	4163968		557745	4163595

Study Site: Nathaniel Mountain WMA			Study Site: Sleepy Creek WMA		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	687232	4335221	Control	740909	4366056
	687379	4335085		740715	4366105
	687243	4334939		740957	4366250
	687096	4335075		740764	4366299
Clearcut-and-leave	688208	4336413	Clearcut-and-leave	741193	4367554
	688374	4336302		740998	4367600
	688264	4336136		741043	4367794
	688097	4336246		741238	4367749
Clearcut-and-windrow	687504	4335514	Clearcut-and-windrow	741107	4366669
	687650	4335377		740914	4366720
	687513	4335232		741158	4366863
	687367	4335369		740965	4366914
Hack-and-spray	687843	4336591	Hack-and-spray	741209	4367082
	688010	4336481		741015	4367129
	687901	4336314		741256	4367276
	687733	4336424		741062	4367323

Appendix 4-B continued.

Study Site: Short Mountain WMA		
Treatment	Easting	Northing
Control	701648	4341012
	701866	4340910
	701744	4340735
	701592	4340842
Clearcut-and-leave	701244	4339947
	701328	4340128
	701509	4340052
	701429	4339864
Clearcut-and-windrow	701140	4339751
	701326	4339680
	701258	4339496
	701057	4339569
Hack-and-spray	701445	4340659
	701329	4340501
	701492	4340386
	701616	4340541

Appendix 4-C. Avian point count coordinates in cut-back borders. All coordinates are in NAD83, UTM Zone 17.

Study Site: Allegheny WMA				Study Site: Bluestone WMA			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	668793	4364637	Control	A	513389	4143707
	B	668937	4364652		B	513311	4143549
15 m, 4.5 m ² /ha	A	668385	4364578	15 m, 4.5 m ² /ha	A	512050	4147324
	B	668355	4364431		B	511950	4147447
15 m, 14 m ² /ha	A	663611	4358230	15 m, 14 m ² /ha	A	511265	4146997
	B	663475	4358293		B	511229	4146851
30 m, 4.5 m ² /ha	A	668324	4365044	30 m, 4.5 m ² /ha	A	512963	4143145
	B	668410	4364914		B	512923	4143025
30 m, 14 m ² /ha	A	668042	4363651	30 m, 14 m ² /ha	A	512798	4142740
	B	667976	4363516		B	512726	4142586
45 m, 4.5 m ² /ha	A	668227	4364110	45 m, 4.5 m ² /ha	A	512962	4145385
	B	668186	4363966		B	513096	4145320
45 m, 14 m ² /ha	A	663157	4358440	45 m, 14 m ² /ha	A	512485	4145579
	B	663020	4358502		B	512641	4145524

Study Site: Beury Mountain WMA Block 1				Study Site: Beury Mountain WMA Block 2			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	499094	4199109	Control	A	500441	4198102
	B	499094	4198959		B	500577	4198038
15 m, 4.5 m ² /ha	A	500163	4200002	15 m, 4.5 m ² /ha	A	501351	4197689
	B	500307	4200044		B	501488	4197626
15 m, 14 m ² /ha	A	500651	4200102	15 m, 14 m ² /ha	A	498431	4198971
	B	500796	4200064		B	498567	4198909
30 m, 4.5 m ² /ha	A	501823	4199794	30 m, 4.5 m ² /ha	A	499215	4198665
	B	501968	4199756		B	499353	4198605
30 m, 14 m ² /ha	A	498494	4198769	30 m, 14 m ² /ha	A	497976	4199180
	B	498346	4198745		B	498112	4199117
45 m, 4.5 m ² /ha	A	501152	4199976	45 m, 4.5 m ² /ha	A	499956	4198269
	B	501297	4199941		B	500093	4198208
45 m, 14 m ² /ha	A	499683	4199867	45 m, 14 m ² /ha	A	500895	4197893
	B	499827	4199910		B	501032	4197831

Appendix 4-C continued.

Study Site: Burnsville Lake WMA				Study Site: Lewis Wetzel WMA Block 1			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	532145	4296042	Control	A	529448	4374272
	B	532071	4296167		B	529524	4374151
15 m, 4.5 m ² /ha	A	532982	4295491	15 m, 4.5 m ² /ha	A	528185	4373809
	B	532844	4295512		B	528195	4373675
15 m, 14 m ² /ha	A	533711	4294536	15 m, 14 m ² /ha	A	529653	4373846
	B	533773	4294665		B	529718	4373742
30 m, 4.5 m ² /ha	A	532491	4295866	30 m, 4.5 m ² /ha	A	529051	4373500
	B	532343	4295880		B	528978	4373363
30 m, 14 m ² /ha	A	533802	4294930	30 m, 14 m ² /ha	A	530932	4372554
	B	533773	4295056		B	530927	4372425
45 m, 4.5 m ² /ha	A	533526	4295874	45 m, 4.5 m ² /ha	A	528944	4374597
	B	533554	4295956		B	528884	4374737
45 m, 14 m ² /ha	A	533521	4295301	45 m, 14 m ² /ha	A	529035	4374001
	B	533436	4295436		B	529045	4373859

Study Site: Lewis Wetzel WMA Block 2				Study Site: Little Canaan WMA Block 1			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	529136	4372017	Control	A	635349	4331992
	B	529164	4371834		B	635289	4332126
15 m, 4.5 m ² /ha	A	529694	4373195	15 m, 4.5 m ² /ha	A	635149	4332448
	B	529812	4373288		B	635092	4332576
15 m, 14 m ² /ha	A	529994	4373542	15 m, 14 m ² /ha	A	635099	4332401
	B	530125	4373569		B	635043	4332534
30 m, 4.5 m ² /ha	A	528380	4373229	30 m, 4.5 m ² /ha	A	636075	4330319
	B	528468	4373113		B	636016	4330455
30 m, 14 m ² /ha	A	529275	4372655	30 m, 14 m ² /ha	A	635634	4331332
	B	529308	4372493		B	635577	4331463
45 m, 4.5 m ² /ha	A	528734	4372872	45 m, 4.5 m ² /ha	A	636104	4330100
	B	528889	4372883		B	636045	4330231
45 m, 14 m ² /ha	A	530479	4373497	45 m, 14 m ² /ha	A	635366	4331797
	B	530603	4373414		B	635308	4331929

Appendix 4-C continued.

Study Site: Little Canaan WMA Block 2				Study Site: Nathaniel Mountain WMA			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	634027	4331180	Control	A	688416	4336844
	B	633924	4331285		B	688561	4336821
15 m, 4.5 m ² /ha	A	635021	4330279	15 m, 4.5 m ² /ha	A	687590	4336002
	B	634915	4330385		B	687739	4335981
15 m, 14 m ² /ha	A	634311	4330982	15 m, 14 m ² /ha	A	688085	4335934
	B	634204	4331088		B	688234	4335916
30 m, 4.5 m ² /ha	A	634654	4330634	30 m, 4.5 m ² /ha	A	687951	4337002
	B	634549	4330741		B	688099	4336979
30 m, 14 m ² /ha	A	633957	4331330	30 m, 14 m ² /ha	A	688921	4336766
	B	633851	4331437		B	689069	4336743
45 m, 4.5 m ² /ha	A	634975	4330241	45 m, 4.5 m ² /ha	A	687201	4335912
	B	634871	4330341		B	687349	4335893
45 m, 14 m ² /ha	A	634393	4330819	45 m, 14 m ² /ha	A	687696	4335843
	B	634290	4330925		B	687844	4335821

Study Site: Sleepy Creek WMA				Study Site: Stonewall Jackson WMA			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	745686	4380700	Control	A	549340	4312258
	B	745573	4380716		B	549208	4312319
15 m, 4.5 m ² /ha	A	740358	4365007	15 m, 4.5 m ² /ha	A	549177	4310983
	B	740316	4364911		B	549054	4310957
15 m, 14 m ² /ha	A	740606	4365155	15 m, 14 m ² /ha	A	549088	4311083
	B	740500	4365068		B	548984	4310966
30 m, 4.5 m ² /ha	A	740666	4366816	30 m, 4.5 m ² /ha	A	543417	4306933
	B	740534	4366798		B	543306	4306962
30 m, 14 m ² /ha	A	740271	4365468	30 m, 14 m ² /ha	A	543729	4306170
	B	740327	4365298		B	543790	4306300
45 m, 4.5 m ² /ha	A	740437	4366396	45 m, 4.5 m ² /ha	A	544187	4308099
	B	740427	4366276		B	544021	4308138
45 m, 14 m ² /ha	A	745311	4382968	45 m, 14 m ² /ha	A	544710	4306595
	B	745189	4382979		B	544778	4306521

Appendix 4-D. Avian point count coordinates in regeneration stands. All coordinates are in NAD83, UTM Zone 17.

Study Site: Beury Mountain WMA Block 1			Study Site: Beury Mountain WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	500800	4199428	Control	501298	4198670
Clearcut-and-leave	501133	4199427	Clearcut-and-leave	501965	4199173
Clearcut-windrow	500466	4199429	Clearcut-windrow	502132	4198667
Hack-and-spray	499633	4200189	Hack-and-spray	501633	4198412
Study Site: Beury Mountain WMA Block 3			Study Site: Beury Mountain WMA Block 4		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	497636	4197906	Control	496965	4197657
Clearcut-and-leave	497299	4197659	Clearcut-and-leave	496970	4198921
Clearcut-windrow	497635	4197143	Clearcut-windrow	497299	4198165
Hack-and-spray	498466	4197401	Hack-and-spray	496964	4198166
Study Site: Beury Mountain WMA Block 5			Study Site: Center Branch WMA Block 1		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	503797	4199177	Control	556543	4344121
Clearcut-and-leave	503131	4198920	Clearcut-and-leave	556252	4343678
Clearcut-windrow	503464	4199173	Clearcut-windrow	556877	4343899
Hack-and-spray	503798	4199681	Hack-and-spray	NA	
Study Site: Center Branch WMA Block 2			Study Site: Center Branch WMA Block 3		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	558111	4342846	Control	558046	4344574
Clearcut-and-leave	557775	4343067	Clearcut-and-leave	557583	4343672
Clearcut-windrow	557308	4342560	Clearcut-windrow	557408	4343304
Hack-and-spray	NA		Hack-and-spray	NA	
Study Site: Little Canaan WMA Block 1			Study Site: Little Canaan WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	635210	4330580	Control	635330	4331047
Clearcut-and-leave	635515	4330229	Clearcut-and-leave	635780	4331790
Clearcut-windrow	634549	4331248	Clearcut-windrow	635734	4332266
Hack-and-spray	634225	4331566	Hack-and-spray	634904	4332136
Study Site: Little Canaan WMA Block 3			Study Site: Little Canaan WMA Block 4		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	636233	4333237	Control	637458	4332539
Clearcut-and-leave	636551	4332456	Clearcut-and-leave	636738	4333059
Clearcut-windrow	635523	4332744	Clearcut-windrow	637104	4332805
Hack-and-spray	636120	4332752	Hack-and-spray	636272	4331673

Appendix 4-D continued.

Study Site: Moncove Lake WMA Block 1			Study Site: Moncove Lake WMA Block 2		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	558117	4164267	Control	557497	4162767
Clearcut-and-leave	558681	4164199	Clearcut-and-leave	557975	4163137
Clearcut-windrow	558046	4163703	Clearcut-windrow	558293	4163388
Hack-and-spray	558364	4163952	Hack-and-spray	557728	4163455

Study Site: Nathaniel Mountain WMA			Study Site: Sleepy Creek WMA		
Treatment	Easting	Northing	Treatment	Easting	Northing
Control	4335080	687238	Control	4366178	740837
Clearcut-and-leave	4336274	688236	Clearcut-and-leave	4367675	741118
Clearcut-windrow	4335374	687509	Clearcut-windrow	4366792	741037
Hack-and-spray	4336453	687873	Hack-and-spray	4367203	741136

Study Site: Short Mountain WMA		
Treatment	Easting	Northing
Control	4340869	701698
Clearcut-and-leave	4340000	701378
Clearcut-windrow	4339628	701198
Hack-and-spray	4340521	701469

Appendix 4-E. Salamander coverboard coordinates in cut-back borders. All coordinates are in NAD83, UTM Zone 17.

Study Site: Beury Mountain WMA Block 1				Study Site: Beury Mountain WMA Block 2			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	499094	4199146	Control	A	500406	4198117
	B	499094	4199071		B	500474	4198085
	C	499094	4198996		C	500542	4198054
	D	499094	4198921		D	500610	4198022
30 m, 4.5 m ² /ha	A	501786	4199803	30 m, 4.5 m ² /ha	A	499180	4198679
	B	501859	4199784		B	499249	4198649
	C	501932	4199765		C	499318	4198619
	D	502004	4199746		D	499387	4198589
30 m, 14 m ² /ha	A	498530	4198775	30 m, 14 m ² /ha	A	497942	4199195
	B	498456	4198763		B	498010	4199164
	C	498382	4198750		C	498078	4199132
	D	498308	4198738		D	498146	4199100
45 m, 4.5 m ² /ha	A	501115	4199985	45 m, 4.5 m ² /ha	A	499922	4198284
	B	501188	4199967		B	499990	4198253
	C	501260	4199949		C	500059	4198223
	D	501333	4199932		D	500127	4198192
45 m, 14 m ² /ha	A	499647	4199855	45 m, 14 m ² /ha	A	500860	4197908
	B	499718	4199877		B	500929	4197877
	C	499790	4199899		C	500997	4197846
	D	499862	4199920		D	501065	4197815

Study Site: Lewis Wetzel WMA Block 1				Study Site: Lewis Wetzel WMA Block 2			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	529428	4374304	Control	A	529166	4372206
	B	529460	4374237		B	529146	4372131
	C	529493	4374175		C	529140	4372053
	D	529543	4374120		D	529136	4371976
30 m, 4.5 m ² /ha	A	529053	4373535	30 m, 4.5 m ² /ha	A	528369	4373265
	B	529029	4373468		B	528388	4373193
	C	528991	4373397		C	528435	4373126
	D	528967	4373327		D	528496	4373086
30 m, 14 m ² /ha	A	530911	4372590	30 m, 14 m ² /ha	A	529264	4372692
	B	530944	4372524		B	529282	4372620
	C	530950	4372453		C	529300	4372531
	D	530909	4372391		D	529311	4372457
45 m, 4.5 m ² /ha	A	528965	4374564	45 m, 4.5 m ² /ha	A	528697	4372867
	B	528924	4374628		B	528771	4372883
	C	528889	4374701		C	528852	4372884
	D	528879	4374775		D	528925	4372871
45 m, 14 m ² /ha	A	529030	4374038	45 m, 14 m ² /ha	A	530446	4373521
	B	529038	4373962		B	530506	4373474
	C	529042	4373895		C	530575	4373432
	D	529039	4373820		D	530631	4373390

Study Site: Little Canaan WMA Block 1	Study Site: Little Canaan WMA Block 2
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Appendix 4-E continued.

Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	635364	4331957	Control	A	634053	4331154
	B	635334	4332026		B	634000	4331208
	C	635304	4332092		C	633951	4331258
	D	635274	4332160		D	633898	4331312
30 m, 4.5 m ² /ha	A	636090	4330284	30 m, 4.5 m ² /ha	A	634680	4330606
	B	636060	4330353		B	634627	4330660
	C	636031	4330420		C	634575	4330714
	D	636002	4330489		D	634523	4330768
30 m, 14 m ² /ha	A	635649	4331298	30 m, 14 m ² /ha	A	633983	4331303
	B	635620	4331366		B	633930	4331356
	C	635592	4331430		C	633877	4331410
	D	635561	4331498		D	633825	4331463
45 m, 4.5 m ² /ha	A	636119	4330067	45 m, 4.5 m ² /ha	A	635001	4330215
	B	636088	4330134		B	634948	4330267
	C	636059	4330198		C	634898	4330315
	D	636029	4330266		D	634844	4330367
45 m, 14 m ² /ha	A	635381	4331762	45 m, 14 m ² /ha	A	634419	4330792
	B	635352	4331830		B	634367	4330845
	C	635323	4331894		C	634315	4330899
	D	635292	4331963		D	634264	4330953

Study Site: Stonewall Jackson WMA			
Treatment	Point	Easting	Northing
Control	A	549376	4312277
	B	549307	4312265
	C	549237	4312301
	D	549174	4312341
30 m, 4.5 m ² /ha	A	543414	4306896
	B	543401	4306962
	C	543335	4306940
	D	543307	4306993
30 m, 14 m ² /ha	A	543703	4306143
	B	543754	4306195
	C	543814	4306280
	D	543754	4306303
45 m, 4.5 m ² /ha	A	544223	4308086
	B	544147	4308108
	C	544059	4308133
	D	543984	4308149
45 m, 14 m ² /ha	A	544725	4306627
	B	544698	4306569
	C	544745	4306540
	D	544813	4306510

Appendix 4-F. Salamander coverboard coordinates in regeneration stands. All coordinates are in NAD83, UTM Zone 17.

Study Site: Beury Mountain WMA Block 1				Study Site: Beury Mountain WMA Block 2			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	500758	4199491	Control	A	501259	4198730
	B	500841	4199489		B	501338	4198730
	C	500842	4199364		C	501340	4198605
	D	500759	4199364		D	501257	4198605
Hack-and-Spray	A	499592	4200252	Hack-and-Spray	A	501590	4198476
	B	499675	4200252		B	501675	4198476
	C	499675	4200125		C	501673	4198351
	D	499591	4200125		D	501590	4198349

Study Site: Beury Mountain WMA Block 3				Study Site: Beury Mountain WMA Block 4			
Treatment	Point	Easting	Northing	Treatment	Point	Easting	Northing
Control	A	497594	4197970	Control	A	496921	4197723
	B	497678	4197969		B	497007	4197721
	C	497678	4197842		C	497007	4197593
	D	497593	4197841		D	496922	4197593
Hack-and-Spray	A	498422	4197467	Hack-and-Spray	A	496922	4198230
	B	498508	4197464		B	497007	4198230
	C	498507	4197338		C	497007	4198100
	D	498424	4197337		D	496922	4198101

Study Site: Beury Mountain WMA Block 5			
Treatment	Point	Easting	Northing
Control	A	503755	4199239
	B	503838	4199239
	C	503839	4199112
	D	503755	4199112
Hack-and-Spray	A	503755	4199747
	B	503840	4199744
	C	503838	4199618
	D	503754	4199616