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ARTICLE

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Habitat management for stopover and breeding songbird communities along rights-of-way in forest-dominated landscapes

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Abstract

The proliferation of energy rights-of-way (pipelines and powerlines; ROWs) in the central Appalachian region has prompted wildlife management agencies to consider ways to use these features to manage and conserve at-risk songbird species. However, little empirical evidence exists regarding best management strategies to enhance habitat surrounding ROWs for the songbird community during stopover or breeding periods. We used a before–after–control–impact design to study cut-back border (linear tree cuttings along abrupt forest edges) harvest width (15, 30, and 45 m wide into the forest), and harvest intensity (14 and 4.5 m^2/ha basal area retention) prescriptions along ROWs and assessed their effects on mature forest and young forest songbird species and avian guilds (forest gap habitat, forest interior habitat, young forest habitat, and species of regional conservation priority) up to 2 years after treatment throughout West Virginia. Species richness during the spring stopover period initially decreased at the 1-year post-treatment period but returned to pre-treatment levels by 2 years post-treatment. Breeding season responses to cut-back border treatments varied across harvest width, harvest intensity, and time, but all responses of focal species abundance and guild richness were neutral or positive. Cut-back border harvest intensity had a stronger influence (i.e., more positive responses) than harvest width on breeding focal species abundances and guild richness. For harvest intensity, the more intense, 4.5 m^2/ha retention treatment had a stronger influence (i.e., more positive responses) than the less intense, 14 m^2/ha retention treatment. For harvest width, the narrowest treatment (15-m wide) had the strongest influence (i.e., more positive responses) of all width treatments, followed by the widest (45-m wide treatment) with the least influence from the 30-m wide treatment. Abundances and richness increased from pre-treatment to 2 years post-treatment across all species and guilds that exhibited a response. These results suggest that cut-back borders increase breeding season habitat suitability along ROWs for the mature forest and young forest songbird community as well as for species of regional conservation priority in the short-term. These findings can aid the development of management guidelines for the forest songbird community along abrupt forest edges of man-made habitat features in forest-dominated landscapes.

KEYWORDS
central Appalachians, human-modified forests, mature forest songbirds, rights-of-way, young forest songbirds

INTRODUCTION

Linear infrastructure from energy transport (e.g., utility powerlines, oil/gas pipelines, or seismic lines) and their effects on wildlife populations continue to be a challenge facing land managers and natural resource scientists (Biasotto & Kindel, 2018; Ouédraogo et al., 2020; Richardson et al., 2017). With global energy consumption expected to increase by 70% between 2018 and 2050 (EIA, 2019), there is a growing interest in developing strategies that incorporate linear energy infrastructure into management plans to meet wildlife conservation goals. In heavily forested regions of North America, the presence of energy infrastructure has the potential to reduce and fragment continuous forest cover that negatively affects mature forest songbird communities (Bayne et al., 2005; Farwell et al., 2020). In the central Appalachian region, natural gas extraction has increased due to the advancement of horizontal drilling techniques and high-volume hydraulic fracturing, leading to increased access to the Marcellus Shale and Utica Shale basins. Since 2010, natural gas production in West Virginia (WV) has increased by 27% annually (EIA, 2020). To accommodate the increased supply of natural gas in the region, underground gas pipelines are a prominent landscape feature throughout WV, totaling 24,631 km of interstate and intrastate pipelines since 2018 (AGA, 2019). Additionally, short- and long-distance overhead utility powerlines connecting consumers to power plants constitute an extensive network of linear infrastructure, with 9975 km of transmission powerlines in WV since 2019 (includes high voltage lines 69–765 kV only; HIFLD, 2019). With underground gas pipelines and overhead utility powerlines (from this point forwards ROW) so prominent within WV and the central Appalachian region, efforts to incorporate these features into management plans to meet conservation goals may benefit from further examination into their effects on the songbird community.

The tradeoff between conserving habitat for mature forest interior songbirds and creating habitat for early successional or young forest songbirds often presents a challenge for land managers. Within mature forest-dominated landscapes, the most apparent feature of ROWs is the presence of an early successional plant community generally maintained via mowing or herbicide spraying. Despite their relatively small footprint, ROWs within forested landscapes can have profound effects on the forest bird community by increasing forest loss and fragmentation that can reduce abundance and species richness of mature forest interior songbirds (Barton et al., 2016; Farwell et al., 2016). Furthermore, the close proximity of mature forest to grassland or shrubland in ROWs creates long stretches of abrupt forest edge that increases the likelihood of brood parasitism by brown-headed cowbirds (*Molothrus ater*) and nest predation by mesocarnivores on nesting songbirds (Howell et al., 2007; Shake et al., 2011). Therefore, the presence of ROWs within forest-dominated landscapes may pose serious threats to local populations of forest interior species, many of which are experiencing decreasing regional populations (Sauer et al., 2013). In contrast, early successional habitat created by periodic maintenance in ROWs creates habitat for some early successional or shrubland songbird species (Askins & Folsom-O'Keefe, 2012; Bulluck & Buehler, 2006; King et al., 2009). The prevalence of ROWs throughout the eastern US forests presents opportunities to manage habitat for many early successional or young forest species, which have experienced decreasing populations due to the declining trends in area and distribution of early seral or ephemeral habitats (King & Schlossberg, 2014; Oswalt et al., 2010). However, because vegetation in ROWs consists of high amounts of grasses and forbs, it is not completely compatible with habitat requirements of young forest species that require greater amounts of saplings or woody stems (Bulluck & Buehler, 2006; Culbert et al., 2013).

Wildlife habitat management approaches that are implemented in association with ROWs offer opportunities for land managers to meet conservation goals but have not been studied experimentally for songbirds. One management approach to improve the habitat suitability along ROWs for young forest avian species is through the partial reduction of basal area via tree cutting along abrupt forest edges (i.e., cut-back borders; Fleming & Giuliano, 1998; Conover et al., 2009). Cut-back borders (also referred to as edge-feathers, conservation buffers, or field border-cuts) create an intermediate vegetation structure between two distinct land cover types to mimic ecotone-like characteristics (i.e., transition from grassland in ROW to young forest in cut-back border to mature forest). Partial harvesting in cut-back borders promotes sapling and woody stem regeneration, which is beneficial to young forest species but also softens the abrupt forest edge of ROWs that may have potentially beneficial consequences for forest interior species (Yahner et al., 2003). Past research has indicated that cut-back borders increase the densities of early successional or young forest avian species (Burger et al., 2013), increase nesting success of ground-level and low-level breeders (Fleming & Giuliano, 2001), and increase species richness (Yahner et al., 2003). However, much of the research on the effects of cut-back borders on songbirds has been in agricultural-dominated landscapes (e.g., cattle and row crop farms) or where small woodlots are the only available forest habitat (Adams et al., 2015; Fleming & Giuliano, 1998). Information on the effects of cut-back borders on songbirds in forest-dominated landscapes is limited but desired, particularly given the increased prevalence of forest edges due to the proliferation of ROWs in WV and the central Appalachian region. Specifically, the effects of cut-back border width and tree harvest intensity requires a rigorous experimental study to elucidate avian responses and to guide the development of cut-back border management guidelines.

We quantified songbird abundance and species richness in response to various cut-back border width and harvest intensity levels along abrupt forest edges throughout WV to evaluate their potential for managing a diverse forest songbird community. The goal of our study was to determine whether the implementation of cut-back borders improved breeding habitat suitability for mature forest and young forest songbirds and stopover habitat for migrating songbirds. Specific study objectives were to (1) determine that cut-back border width and harvest intensity treatment levels maximized migratory and breeding songbird abundances/richness; and (2) assess that cut-back border treatments optimized the tradeoff between maximizing positive responses of young forest species and minimizing negative responses of mature forest species. Simultaneously managing both the mature forest and young forest songbird community in forested landscapes presents a challenge to land managers. Therefore, information on the effects of cut-back borders along ROWs for managing the overall forest songbird community can guide the development of management plans for songbird conservation.

METHODS

Study area

This study was conducted on eight West Virginia Division of Natural Resources' (WVDNR) wildlife management areas (WMA) located throughout West Virginia, USA (Figure 1) that lay within the central Appalachian region and were characterized by extensive deciduous and coniferous forests (Küchler, 1964). Elevations at study sites ranged from 224 from 1132 m. West Virginia's humid continental climate is characterized by warm summers and cool to cold winters with daily mean spring and fall temperatures of 10°C and summer temperature of 21°C. Mean annual precipitation levels in WV were 76, 127, and 102 cm in the western, central, and eastern portions of the state, respectively, with the heaviest precipitation occurring during spring and summer (NOAA, 2020).

Pipeline and powerline ROWs used in this study were narrow linear openings that ranged from 9.4 to 39.2 m wide (mean = 25.7 ± SD = 17.9 m). Vegetation in ROWs consisted of grassland or low shrubs, was void of a midstory or overstory vegetation layers, and was distinct from adjacent mature forest. To increase our sample size, we also incorporated cut-back borders along wildlife openings. We chose wildlife openings because of their similarities to ROWs; they provided early successional habitat patches within forest-dominated landscapes, they had abrupt forest edges between early successional and mature forest habitats, and they were subject to periodic maintenance. Wildlife openings used in this study were fields planted with a row crop (e.g., corn, clover) or left fallow, ranged from 0.3 to 11.5 ha (mean = 3.1 ± SD = 3.1 ha) in size and 5.7 to 97.0 m (mean = 41.8 ± SD = 31.1 m) in width, and were of varying shapes surrounded by mature forest.

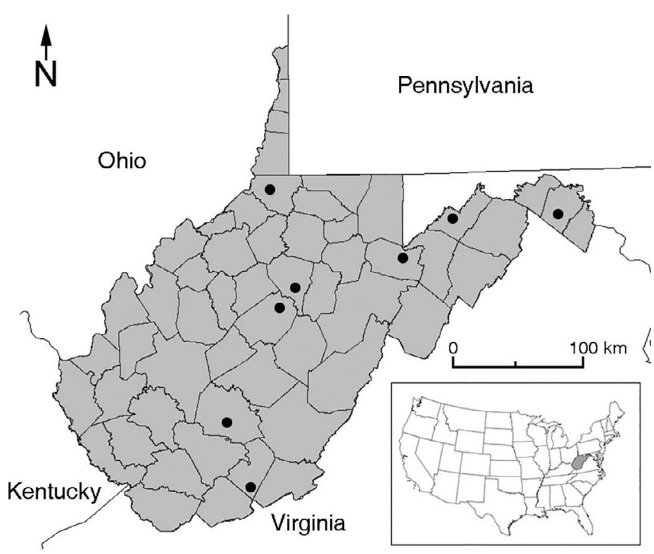

FIGURE 1 County-level map of West Virginia, USA, showing eight study sites (black dots) used to assess songbird response to cut-back borders along pipelines, powerlines, and wildlife openings during 2016–2019

Experimental design

The eight WMAs included 11 block replicates located in mature forests along edges of underground gas pipelines ($n = 4$ blocks), overhead utility powerlines ($n = 4$), or wildlife openings ($n = 3$). A single block replicate consisted of seven independent experimental units (from this point forwards "cut-back borders") with each assigned one of seven experimental treatments. We assumed that treatment plots within each block replicate were 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. We assigned treatments for each block replicate by randomly selecting one of two target residual basal area levels (4.5 or 14.0 m^2/ha basal area retention to mimic clearcut with reserves and crop tree/shelterwood prescriptions, respectively) and one of three cutting depths into the forest (15, 30, or 45 m perpendicular to forest edge) for each cut-back border with one cut-back border randomly designated as a control (i.e., no tree cutting in mature forest along the ROWs or wildlife openings; Figure 2). Target basal area in each cut-back border was achieved via tree cutting using a mechanical tree harvester called a feller-buncher or by hand crew. Felled trees were dropped within the cut-back border plot and remained on the ground following cutting. Each cut-back border was 300 m in linear distance along the forest edge (300 m × 15 m wide treatment = 0.5 ha total area, 30-m wide treatment = 0.9 ha, and 45-m wide treatment = 1.4 ha) and ≥200 m apart to ensure sampling independence (Figure 3).

We used a before–after–control–impact (BACI) framework to infer causal relationships between cut-back border treatments and avian responses through time (Eberhardt, 1976). In each cut-back border, two sampling points were spaced 150 m apart and 15 m from the ROW/wildlife opening edge into the forest and were consistent across all cut-back border widths and harvest intensities (Figure 3). Avian and vegetation data were collected at the same sampling locations. For the avian point counts, we sampled a large enough area at each point count to encompass a gradient from the ROW/wildlife opening, across the cut-back border, and into the adjacent mature forest (refer to the next section) to evaluate avian change in response to addition of the cut-back border treatments. We collected 1 year of pre-treatment data and 1 or 2 years of post-treatment data on the avian and vegetation community.

Avian sampling

We assessed stopover and breeding songbird responses to cut-back borders using 10-min standardized fixed-radius point counts (Bibby et al., 2002). At each point count location, we conducted one annual visit during the spring stopover period (12 April to 17 May) and two annual visits during the breeding period (18 May to 27 June) during 2016–2019. All surveys began after dawn and ended by 10:00 AM (EST) on days that were considered optimal for sampling (i.e., 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), and distance from observer placed into one of two distance intervals (0–50 m or >50 m). Survey-level data for each visit included time of survey and ordinal day. Prior to surveys, observers were trained to independently identify all species visually and aurally while simultaneously recording the distance interval. Surveyor skill was assessed based on previous experience conducting point counts and identification of avian species of the region and ranged from 1–5. A surveyor score of 5 indicated thorough knowledge of species with extensive experience conducting point

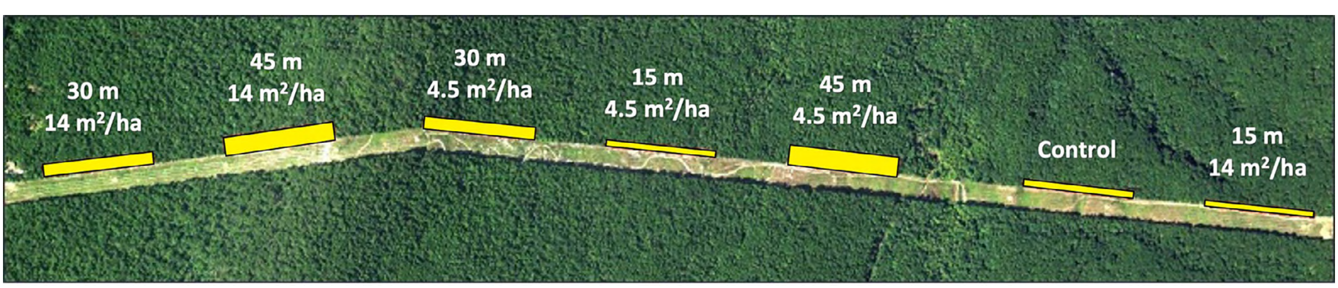

FIGURE 2 Aerial photograph of a cut-back border block replicate along an overhead utility powerline. Cut-back border plots (shown in yellow) were located in mature forests adjacent to right-of-way or wildlife openings. Plots were randomly assigned one of two target residual basal area levels (4.5 or 14.0 m^2/ha basal area retention) and one of three cutting depths into the forest (15, 30, or 45 m perpendicular to forest edge) with one cut-back border designated as a control (i.e., no tree cutting)

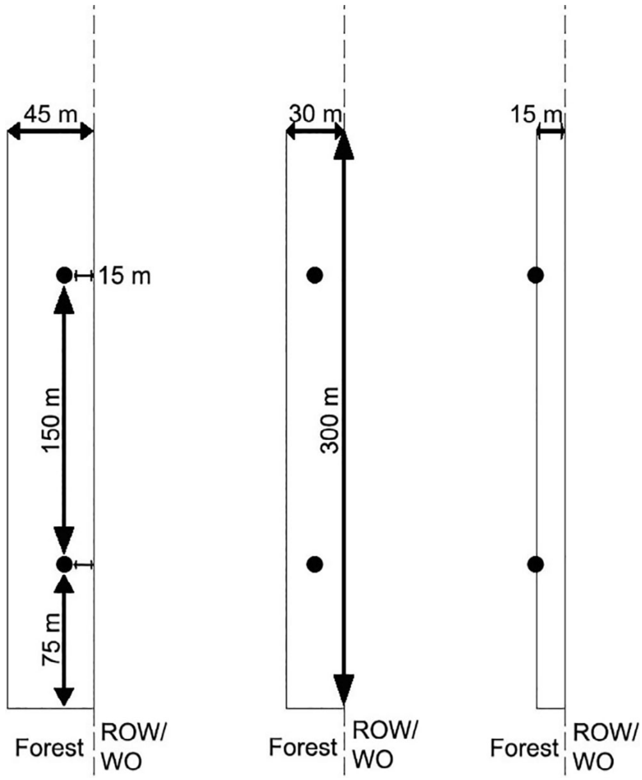

FIGURE 3 Cut-back border plot layout with avian and vegetation sampling points (black dots) for each harvest width factor level (15, 30, and 45 m) along pipelines, powerlines, and wildlife openings ("ROW/WO"). Sampling points were located 15 m from forest–ROW/WO boundary (dashed-line) for all harvest width factor levels, 150 m from intraplot sampling points, and 75 m from the plot ends. Cut-back border plot extends 300 m along the forest–ROW/WO boundary

counts, a score of 1 indicated no knowledge of species and no previous experience conducting point counts, and a score of 3 indicated introductory knowledge of species with minimal experience conducting point counts. For stopover and breeding songbird data analyses, we used only detections within 50-m of the sampling point and included all songbird species detected by sight and sound (excluding flyovers). We excluded all species that were not sampled well using point counts (e.g., corvids, raptors, gallinaceous birds; Ralph et al., 1993).

Vegetation and topography

We sampled vegetation data at avian sampling point locations to characterize vegetation structure within cut-back borders. We measured vegetation characteristics each year within an 11.3-m radius (0.04 ha) circular plot centered on the sampling point (Figure 3) so that vegetation was sampled within each cut-back border. Within the circular plot, we recorded diameter-at-breast height (DBH) of all live trees ≥11.4 cm DBH using DBH tapes. We estimated the amount of live understory foliage cover within two vertical zones (0–1.0 m and >1.0–2.0 m) using a modified Nudds (1977) profile board. The profile board was placed 15 m from the plot center in the cardinal directions and observers knelt at the plot center to estimate understory cover into one of five cover categories (0%–20%, >20%–40%, >40%–60%, >60%–80%, or >80%–100%) for each of the two vertical zones. Standing at the plot center, we measured the percentage of live overstory cover (delineated at ≥5 m above ground) using a 25 cm × 25 cm transparent Plexiglas® panel, divided into a 5 × 5 grid, held overhead (Hachè et al., 2013).

We measured five topographic variables (slope percent, slope aspect, slope position, slope configuration, and elevation) surrounding each point count location using 3-m scale digital elevation models (DEMs) sourced from the US Geological Survey and West Virginia Statewide Addressing and Mapping Board (WVGIS, 2020). Slope percent, aspect, position, and configuration were each calculated at the 3-m DEM resolution to derive a topographic relative moisture index (TRMI; Parker, 1982), that ranged from 0 (xeric soil) to 60 (mesic soil), at the same 3-m resolution. Plot-level elevation and TRMI were calculated using zonal statistics in the program ArcMAP (version 10.7; ESRI, 2018) to generate a mean value within a 50-m radius surrounding each sampling point. We measured the width of the ROW corridor or wildlife opening adjacent to each point count location from aerial imagery (NAIP, 2018).

Statistical analyses

Focal species abundance

We estimated abundances of focal breeding songbird species in cut-back border treatments over time using a stacked Bayesian *N*-mixture model (Kéry & Royle, 2016). The stacked approach treats each point-year combination as a unique point and accounts for the dependency at each point across years using a random point effect (see below). We included species with ≥40 detections within 50-m of the sampling point and a ≥0.20 cumulative detection probability (probability of detecting a species at least once across two annual survey visits). These criteria were based on a power analysis to avoid using species with insufficient detections leading to potentially spurious conclusions. For each species in our breeding focal songbird analysis, we fitted a detection submodel and abundance submodel. In our detection submodel, we estimated detection probability from two annual survey visits (~14 days between within-year survey visits) and modeled conditional detection probability with a binomial process model using ordinal day, time since sunrise, and surveyor skill as covariates on the logit-scale. Our abundance submodel incorporated a full-factorial design with cut-back border treatment factor levels width (15, 30, and 45 m) and basal area retention (14 and 4.5 m^2/ha). The full-factorial design allows the comparison between cut-back borders that received tree cutting and did not receive tree cutting (i.e., control), the comparison among cut-back border width factor levels (15, 30, and 45 m), and the comparison between cut-back border basal area retention factor levels (14 and 4.5 m^2/ha).

We modeled species abundance in our *N*-mixture models using a Poisson process model. We modeled expected abundance λ_{ijk} for each species at site i within treatment plot j during year k as a function of an intercept β, a point count random effect (PC_i) to account for the heterogeneity among sites and repeated observations at the same site, a treatment plot random effect (TP_j) to account for the lack of independence between points within the same cut-back border plot, three nuisance variables ($ELEV_i$, $TRMI_i$, WID_i), the effect of time ($TIME_k$; 1 = 1-year post-treatment and 2 = 2-years post-treatment), and cut-back border factor levels (Equation 1):

$$\log(\lambda_{ijk}) = \beta + PC_i + TP_j + ELEV_i + TRMI_i + WID_\mathrm{i} + TRT_{ijk} + W30_{ijk} + W45_{ijk} + BA4.5_{ijk} + TRT_{ijk} \times TIME_k \quad (1)$$

The term *TRT* corresponded to cut-back borders that received tree cutting with the baseline factor levels 15 m for cut-back border width and 14 m^2/ha for cut-back border harvest retention. Terms *W30* and *W45* corresponded to the 30 and 45 m cut-back border width factors levels, respectively, and *BA4.5* corresponded to the 4.5 m^2/ha cut-back border harvest retention factor level. Factor levels were dummy coded with "0" or "1" depending on whether that factor level occurred at each point, except for the baseline factor levels (*TRT*; i.e., 15 m and 14.0 m^2/ha) that were dummy coded with a "1" unless it was the control treatment. The pre-treatment period was coded with a "0" for all points, regardless of what treatment it received during post-treatment, because no habitat manipulation occurred during this period. We included elevation (*ELEV*) and *TRMI* as nuisance variables to account for topographic variability, and ROW/wildlife opening width (*WID*) because *WID* can affect pre-treatment presence of young forest species in ROWs and wildlife openings. All continuous variables were standardized to have a mean of 0 and standard deviation of one prior to analysis.

Species richness

We modeled apparent species richness within a 50-m radius for breeding avian guilds and overall species richness during the spring stopover period in cut-back border treatments over time using stacked Bayesian generalized linear mixed-effects models (GLMM). We modeled apparent species richness as a function of the same covariates and process model as focal breeding species abundance (see Equation 1). For breeding avian guild analyses, we included three habitat guilds: mature forest gap specialists (from this point forwards "forest gap"), mature forest interior (from this point forwards "forest interior"), and young forest, as well as a distinct group of species of regional conservation priority. Species were assigned to guilds based on previous guild studies from the region (Farwell et al., 2016; McDermott & Wood, 2009). We considered a species as a conservation priority if it was listed by the 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). For the spring stopover analysis, we included songbird species that were considered resident, breeding, or migratory in WV. Species did not need to meet any detection criteria to be included in breeding avian guild or spring stopover analyses.

Model specification

We used diffuse prior distributions for all detection and abundance/richness submodel covariates and nuisance variables (Gaussian [mean = 0, variance = 10]) because we were uncertain of the relationships between cut-back border factor levels and songbirds. We used a Gaussian distribution (mean = 0, variance = τ) prior with a hyperparameter τ from an inverse Gamma distribution ($\alpha = 1$, $\beta = 1$) for our random point count and treatment plot effects. We obtained posterior distributions of model parameters by running three parallel Markov chain Monte Carlo simulations of 108,000 iterations with a burn-in of 8000 iterations at a thinning rate of 100, yielding 3000 samples for posterior distributions. We drew conclusions about treatment effects by inspecting regression coefficients from Equation 1 and by using contrasts (Casella, 2008) to compare log expected abundance between treatment levels and no-harvest controls. Strength of regression coefficients were based on 95% lower and upper credible intervals (CI) of posterior distributions not overlapping 0. We concluded that treatment levels were different from the control if 95% CI of posterior distribution of contrasts did not overlap 0. We assessed model fit with a posterior predictive check, for which 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 & Gelman, 1998). Data analyses were performed in R version 4.0.3 (R Core Team, 2020) using a Bayesian framework in JAGS (Plummer, 2017) called from the package *jagsUI* (Kellner, 2015). Posterior means are presented as slope coefficient with 95% lower and upper CIs.

RESULTS

Vegetation

Following treatment, basal area was reduced in all treatments but was lowest in the 4.5 m^2/ha retention treatments and the 30- and 45-m wide treatments (Table 1). Overstory cover was also reduced following treatment and with lowest cover in the 4.5 m^2/ha retention treatment and 45-m wide treatment (Table 1). Understory cover 0–1 m increased in all treatments but was greatest in the 14 m^2/ha harvest retention treatment and 45-m wide treatment (Table 1). Understory cover >1–2 m increased in all treatments and was greatest in the 4.5 m^2/ha harvest retention treatment and 45-m wide treatment (Table 1).

Breeding songbird abundance and guild richness

We conducted 868 point counts at 154 different sampling points during the breeding period and detected 102 different species of songbird. Eleven species from the forest gap, forest interior, and young forest habitat guilds met our detection criteria for focal breeding species analysis (Table 2). Our posterior predictive checks indicated that all focal species models were of a reasonable fit (Appendix S1). All slope coefficients and 95% CIs of Equation 1 regression terms and control-treatment contrasts of focal species models are located in Appendices S2, S3, and S4.

Cut-back border treatment (*TRT*) had no consistent influence on American goldfinch, black-and-white warbler, black-throated green warbler, Carolina wren, cerulean warbler, common yellowthroat, eastern towhee, hooded

TABLE 1 Annual mean and standard error of vegetation variables collected in cut-back border treatments (harvest width and harvest intensity combinations) and the unharvested control

	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
Live basal area (m^2/ha)							
Pre-treatment	16.3 ± 9.0	14.8 ± 1.4	16.7 ± 1.9	20.0 ± 1.5	15.8 ± 2.0	19.7 ± 1.2	18.3 ± 1.6
1-year post	16.3 ± 9.0	10.5 ± 1.0	9.4 ± 1.1	10.9 ± 1.5	7.4 ± 1.7	11.4 ± 1.1	5.2 ± 1.1
2-year post	16.3 ± 9.0	9.3 ± 1.3	7.5 ± 1.3	8.5 ± 1.3	4.0 ± 1.0	9.7 ± 1.2	4.5 ± 1.0
Overstory cover (%)							
Pre-treatment	88.9 ± 16.9	96.6 ± 1.5	95.5 ± 2.7	84.4 ± 5.6	87.2 ± 5.7	94.6 ± 3.1	96.1 ± 1.3
1-year post	88.9 ± 16.9	72.5 ± 5.4	64.2 ± 6.0	70.7 ± 6.3	30.7 ± 6.8	67.2 ± 6.4	19.5 ± 5.4
2-year post	88.9 ± 16.9	73.7 ± 7.5	73.4 ± 8.5	67.8 ± 8.1	39.5 ± 8.4	64.1 ± 7.5	25.9 ± 8.0
Understory cover 0–1 m (%)							
Pre-treatment	80.0 ± 28.9	91.4 ± 4.9	91.8 ± 4.1	80.9 ± 5.2	86.4 ± 4.8	69.5 ± 6.6	78.0 ± 6.8
1-year post	80.0 ± 28.9	93.6 ± 3.3	94.5 ± 3.0	94.5 ± 3.3	95.5 ± 3.2	85.5 ± 5.6	89.1 ± 4.9
2-year post	80.0 ± 28.9	97.8 ± 2.2	100.0 ± 0.0	97.8 ± 2.2	97.8 ± 2.2	96.7 ± 2.4	100.0 ± 0.0
Understory cover >1–2 m (%)							
Pre-treatment	58.2 ± 33.2	80.0 ± 5.9	74.5 ± 5.9	68.2 ± 6.5	73.6 ± 6.4	58.1 ± 6.9	62.0 ± 6.6
1-year post	58.2 ± 33.2	76.4 ± 7.3	68.2 ± 6.5	76.4 ± 6.4	73.6 ± 7.7	63.6 ± 7.9	64.5 ± 7.0
2-year post	58.2 ± 33.2	85.6 ± 6.4	98.9 ± 1.1	84.4 ± 6.8	93.3 ± 4.0	75.6 ± 7.7	90.6 ± 5.5

TABLE 2 Habitat guild association, count (total number of detections over all study years), and cumulative detection probability (*p*) of focal species used to assess the influence of cut-back border harvest width and harvest intensity factor levels over time in West Virginia during 2016–2019

Common name	Scientific name	Habitat guild	Count	*p*
American goldfinch	*Spinus tristis*	Young forest	68	0.22
Black-and-white warbler[a]	*Mniotilta varia*	Forest interior	103	0.28
Black-throated green warbler	*Setophaga virens*	Forest gap	52	0.31
Carolina wren	*Thryothorus ludovicianus*	Young forest	58	0.32
Cerulean warbler[a]	*Setophaga cerulea*	Forest gap	44	0.23
Common yellowthroat	*Geothlypis trichas*	Young forest	70	0.34
Eastern towhee[a]	*Pipilo erythrophthalmus*	Young forest	208	0.23
Hooded warbler	*Setophaga citrina*	Forest gap	131	0.28
Indigo bunting	*Passerina cyanea*	Young forest	205	0.25
Ovenbird	*Seiurus aurocapilla*	Forest interior	181	0.27
Wood thrush[a]	*Hylocichla mustelina*	Forest interior	61	0.24

[a]Species of conservation priority identified by either Appalachian Mountains Joint Venture, Partners-in-Flight, or West Virginia State Wildlife Action Plan.

warbler, indigo bunting, ovenbird, or wood thrush relative to no treatment. Despite no strong differences between the control and treatments (*TRT*), harvest widths (*W30* and *W45*), or harvest intensities (*BA4.5*), we found different responses at particular combinations of width by intensity by year (Figures 4, 5, and 6). Of the contrasts between the control and harvest width levels, the narrowest 15-m wide treatment elicited the most species responses (13 of 20 contrasts with 95% CIs that did not overlap 0 were in the 15-m wide treatment). The most intensively harvested treatment at 4.5 m^2/ha basal area retention elicited the most species responses (12 of 20 contrasts with 95% CIs that did not overlap 0 were in the 4.5 m^2/ha retention treatment). In the 15 m wide–14 m^2/ha retention treatment, hooded warbler abundance was greater than the control in both years, while eastern towhee and indigo bunting abundances were greater relative to the control by 2 years post-treatment. In the 15 m wide–4.5 m^2/ha retention treatment, the abundance of American goldfinch was greater than the control at 1 year post-treatment, Carolina wren and hooded warbler abundances were greater than the control at 2 years post-treatment, and abundances of black-and-white warbler, eastern towhee, and indigo bunting were greater than the control in both years. Eastern towhee was the only species to exhibit any response to 30-m wide cuts, with greater abundance in both the 14 and 4.5 m^2/ha retention treatments relative to the control at 2 years post-treatment. In the 45 m wide–14 m^2/ha retention treatment, eastern towhee abundance was greater relative to the control at 2 years post-treatment, and indigo bunting abundance was greater relative to the control in both years. In the 45 m wide–4.5 m^2/ha retention treatment, eastern towhee abundance was greater in both years.

For breeding bird guild analyses, seven species comprised the forest gap guild, 18 species the forest interior guild, 18 species the young forest guild, and 27 species the conservation priority guild (Appendix S5). Our posterior predictive checks indicated that all avian guild models were a reasonable fit (Appendix S1). All slope coefficients and 95% CIs of Equation 1 regression terms and control-treatment contrasts of avian guild models are located in Appendices S2, S3, and S4. In general, *TRT* did not strongly influence species richness of the forest gap habitat guild, forest interior habitat guild, young forest habitat guild, or species of regional conservation priority. Despite no strong differences between the control and treatments (*TRT*), harvest widths (*W30* and *W45*), or harvest intensities (*BA4.5*), we found different responses at particular combinations of width by intensity by year for guilds (Figure 7). Similar to focal species, avian guilds were most responsive to the 15-m wide treatment (8 of 9 contrasts with 95% CIs that did not overlap 0 were in the 15-m wide treatment). However, of the harvest intensity treatments, the less intensive 14 m^2/ha retention treatment elicited the most responses (5 of 9 contrasts with 95% CIs that did not overlap 0 were in the 14 m^2/ha retention treatment). In the 15 m wide–14 m^2/ha retention treatment, forest gap guild and young forest guild species richness were greater relative to the control at 2 years post-treatment, and richness of species of conservation priority was greater relative to the control in both years. In the 15 m wide–4.5 m^2/ha retention treatment, species richness of

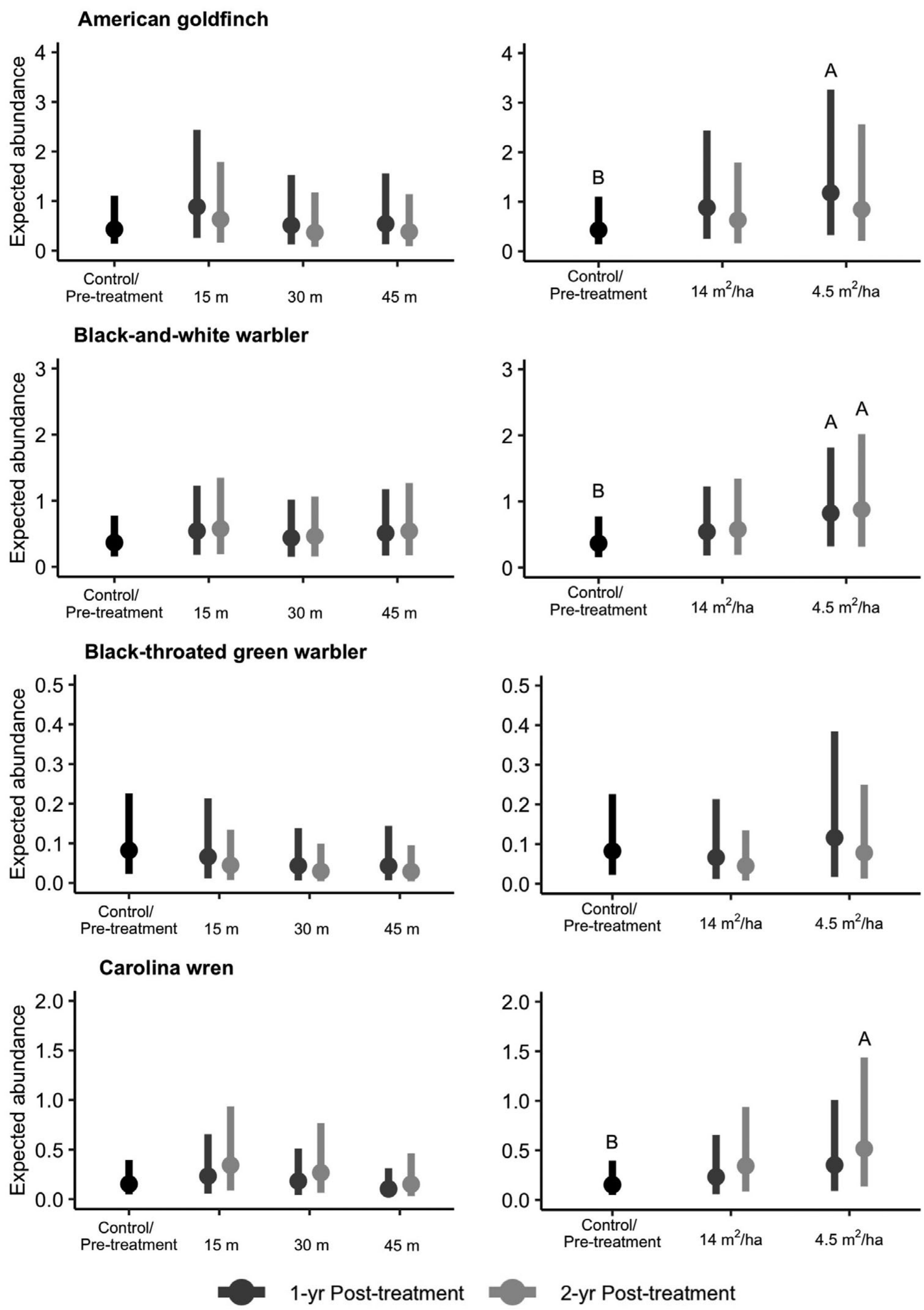

FIGURE 4 Mean point-level expected abundance and 95% credible intervals of focal species in cut-back border harvest width levels (15, 30, and 45 m; left) and harvest intensity levels (14 and 4.5 m^2/ha residual basal area; right) at 1-year and 2-year post-treatment periods, and the unharvested control/pre-treatment period. Alphabetic notation above point estimates indicates differences within post-treatment years based on non-overlapping 95% credible intervals of posterior distributions. Point estimates without alphabetic notation indicate no differences. Point estimates for harvest width levels (left) were calculated with harvest intensity set at the 14 m^2/ha level and estimates for harvest intensity levels (right) were calculated with harvest width set at the 15-m level, while nuisance variables (*ELEV*, *TRMI*, and *WID*) were fixed at their means

the forest gap guild and species of conservation priority were greater relative to the control at 2 years post-treatment, and species richness of the young forest guild was greater relative to the control in both years. In the 45 m wide–14 m^2/ha retention treatment, species richness of the young forest guild was greater relative to the control at 2 years post-treatment. No guild responded to the 30-m wide treatment.

Corridor and opening width (*WID)* had a positive association with indigo bunting (slope coefficient [95%

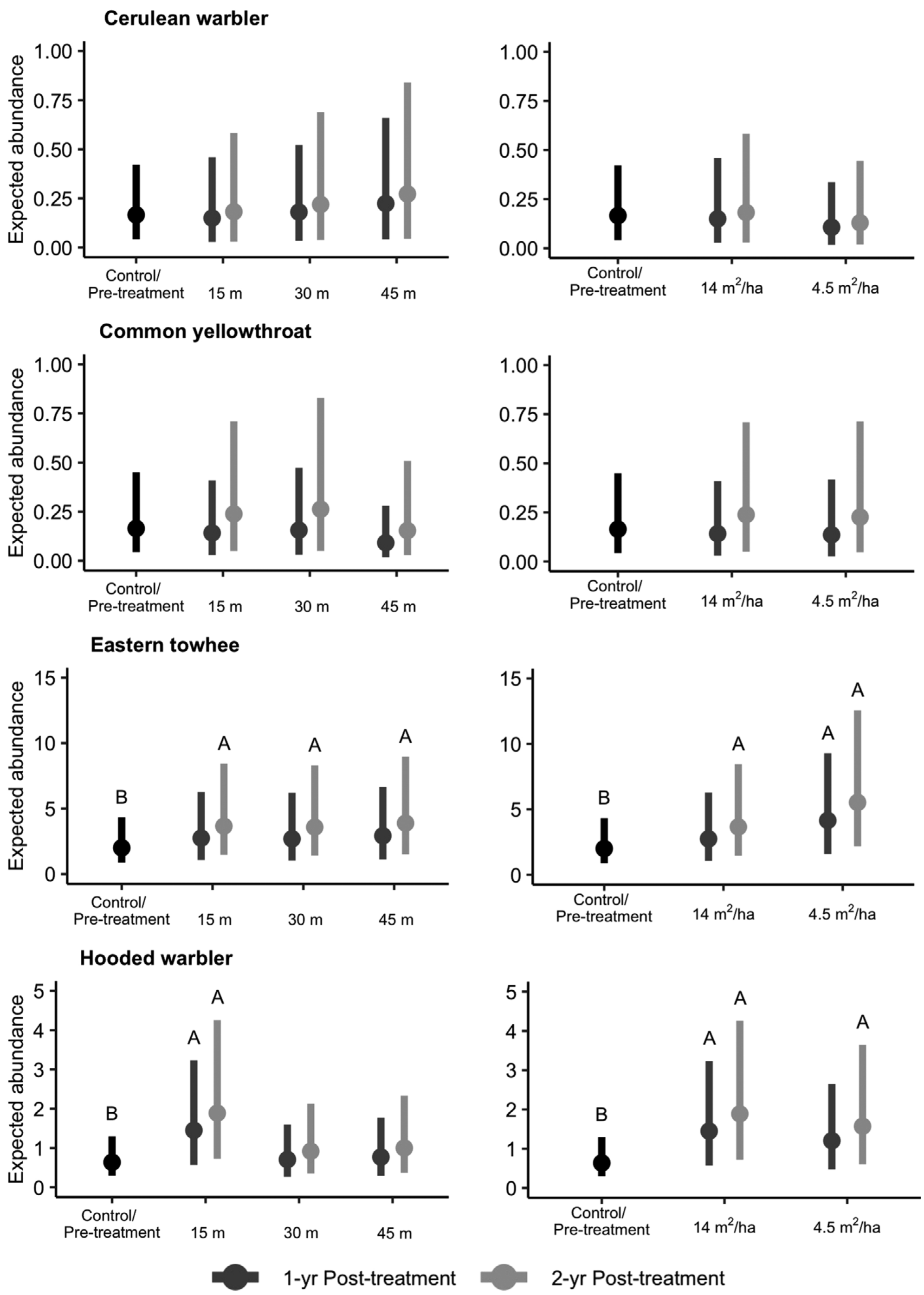

FIGURE 5 Mean point-level expected abundance and 95% credible intervals of focal species in cut-back border harvest width levels (15, 30, and 45 m; left) and harvest intensity levels (14 and 4.5 m^2/ha residual basal area; right) at 1-year and 2-year post-treatment periods, and the unharvested control/pre-treatment period. Alphabetic notation above point estimates indicates differences within post-treatment years based on non-overlapping 95% credible intervals of posterior distributions. Point estimates without alphabetic notation indicate no differences. Point estimates for harvest width levels (left) were calculated with harvest intensity set at the 14 m^2/ha level and estimates for harvest intensity levels (right) were calculated with harvest width set at the 15-m level, while nuisance variables (*ELEV*, *TRMI*, and *WID*) were fixed at their means

lower CI, 95% upper CI] = 0.22 [0.01, 0.43]) abundance. In contrast, *WID* had a negative association with black-and-white warbler (−0.39 [−0.78, −0.02]), ovenbird (−0.40 [−0.80, −0.01]), and wood thrush (−0.51 [−0.99, −0.11]) abundances. Additionally, *WID* was positively associated with young forest guild richness (0.16 [0.02, 0.30]) and negatively associated with forest interior guild richness (−0.14 [−0.28, −0.01]).

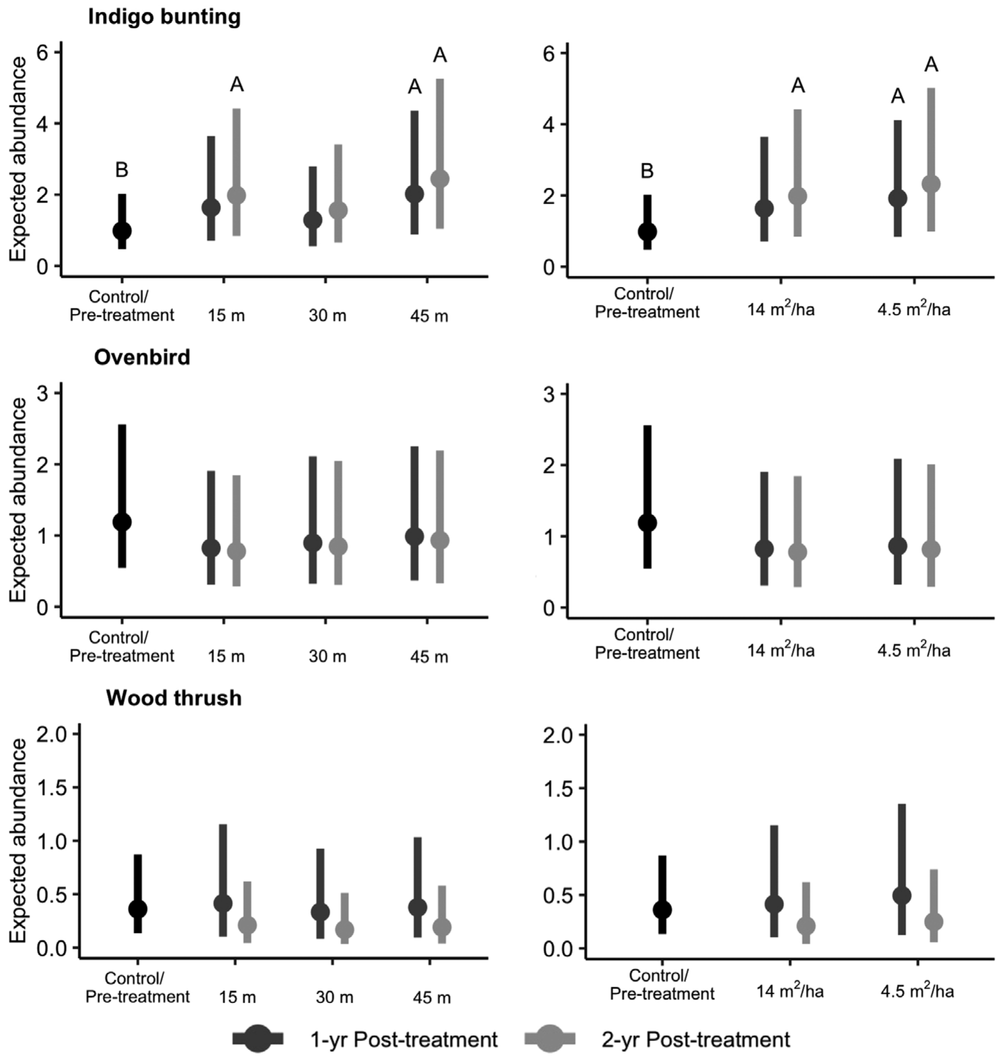

FIGURE 6 Mean point-level expected abundance and 95% credible intervals of focal species in cut-back border harvest width levels (15, 30, and 45 m; left) and harvest intensity levels (14 and 4.5 m^2/ha residual basal area; right) at 1-year and 2-year post-treatment periods, and the unharvested control/pre-treatment period. Alphabetic notation above point estimates indicates differences within post-treatment years based on non-overlapping 95% credible intervals of posterior distributions. Point estimates without alphabetic notation indicate no differences. Point estimates for harvest width levels (left) were calculated with harvest intensity set at the 14 m^2/ha level and estimates for harvest intensity levels (right) were calculated with harvest width set at the 15-m level, while nuisance variables (*ELEV*, *TRMI*, and *WID*) were fixed at their means

Spring stopover songbird richness

We conducted a total of 350 point counts at 126 different sampling points and detected 80 different songbird species during the spring stopover period. Our posterior predictive check indicated that our model was a reasonable fit (Appendix S1). Cut-back border treatment (*TRT*) reduced spring stopover species richness by 31% (95% CI = 9–50%) relative to no treatment (i.e., control). There were no differences in species richness among harvest width factor levels or between harvest intensity factor levels (95% CIs of *W30*, *W45*, and *BA4.5* overlapped 0). Based on contrasts between treatment levels and the control, species richness was lower in the 15-, 30-, and 45-m wide treatments at the 4.5 m^2/ha harvest retention level and lower in the 15 m wide–14.0 m^2/ha retention treatment relative to the control at 1 year post-treatment. Species richness was also lower in the 45 m wide–14 m^2/ha retention treatment relative to the control at 2 years post-treatment (Figure 8).

DISCUSSION

With the proliferation of human-modification to habitats within forest-dominated landscapes from ROWs and wildlife openings, there is a growing need to find ways that incorporate these existing features into wildlife

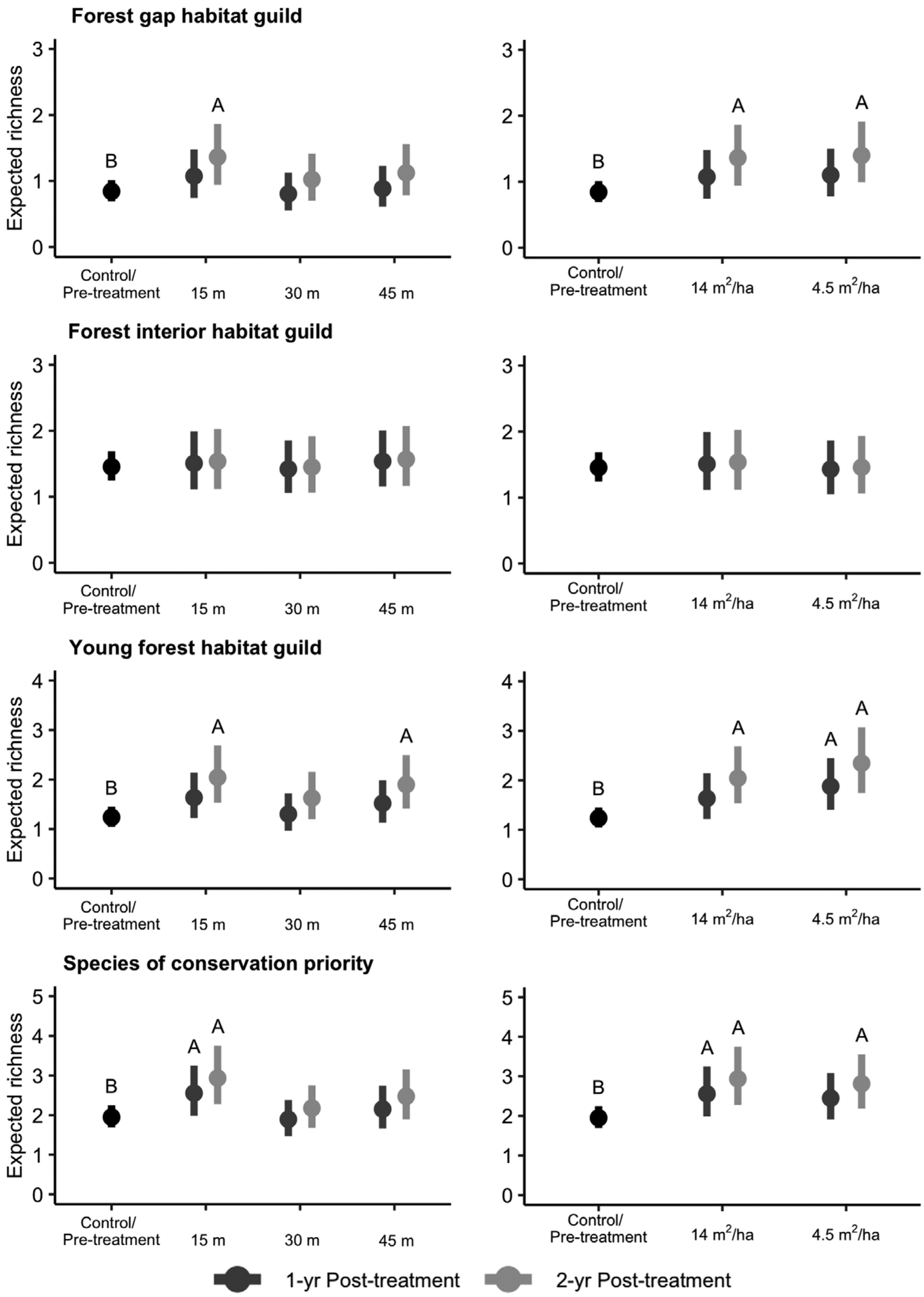

FIGURE 7 Mean point-level expected apparent species richness and 95% credible intervals of avian guilds in cut-back border harvest width levels (15, 30, and 45 m; left) and harvest intensity levels (14 and 4.5 m^2/ha residual basal area; right) at 1-year and 2-year post-treatment periods, and the unharvested control/pre-treatment period. Alphabetic notation above point estimates indicates differences within post-treatment years based on non-overlapping 95% credible intervals of posterior distributions. Point estimates without alphabetic notation indicate no differences. Point estimates for harvest width levels (left) were calculated with harvest intensity set at the 14 m^2/ha level and estimates for harvest intensity levels (right) were calculated with harvest width set at the 15-m level, while nuisance variables (*ELEV*, *TRMI*, and *WID*) were fixed at their means

management plans. Managing habitats of young and mature forest songbirds simultaneously, particularly in heavily forested landscapes with ROWs and wildlife openings, often presents challenges for land managers. Here we have provided a short-term experimental assessment of the effects of cut-back borders and demonstrated that cut-back borders can be used to manage young forest habitat and conserve mature forest habitat for the forest

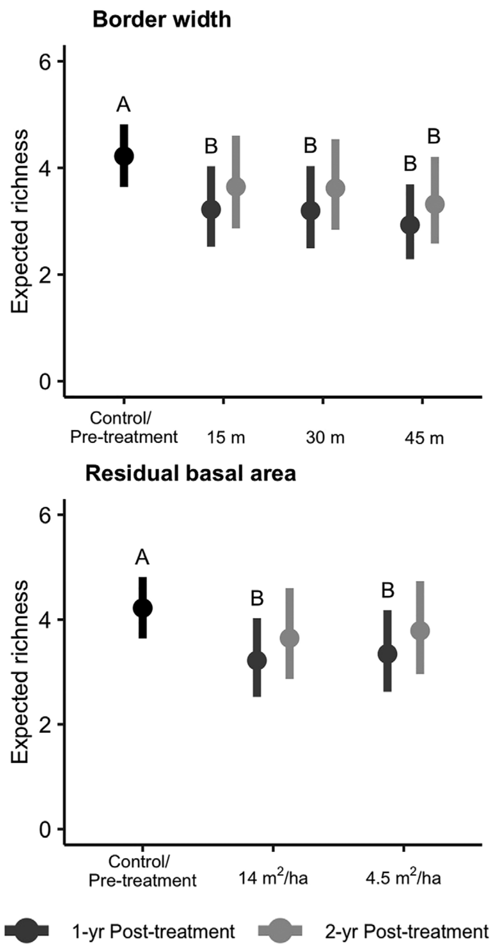

FIGURE 8 Mean point-level expected apparent species richness and 95% credible intervals during the spring stopover period in cut-back border harvest width levels (15, 30, and 45 m; top) and harvest intensity levels (14 and 4.5 m^2/ha residual basal area; bottom) at 1-year and 2-year post-treatment periods, and the unharvested control/pre-treatment period. Alphabetic notation above point estimates indicates differences within post-treatment years based on non-overlapping 95% credible intervals of posterior distributions. Point estimates without alphabetic notation indicate no differences. Point estimates for harvest width levels (top) were calculated with harvest intensity set at the 14 m^2/ha level and estimates for harvest intensity levels (bottom) were calculated with harvest width set at the 15-m level, while nuisance variables (*ELEV*, *TRMI*, and *WID*) were fixed at their means

songbird community. These results can inform management guidelines along abrupt forest edges of ROWs and wildlife openings (from this point forwards ROWs/WOs).

Young forest songbirds are often restricted by the area of suitable habitat in ROWs/WOs. The partial reduction of the overstory layer from cut-back borders stimulated ground and understory vegetative growth (Table 1) and increased locally available habitat for young forest species that were not prevalent prior to cut-back borders treatments. When combined with vegetation in ROWs/WOs corridors, it effectively widened corridors that were likely to alleviate area-sensitivity pressures (e.g., territory elongation, width sensitivity) that have been observed in some young forest species (Rodewald & Vitz, 2005; Shake et al., 2012). Anderson et al. (1977) and Confer and Pascoe (2003) reported that young forest birds were scarce or absent in narrow ROWs but increased with increasing ROW width, which may be a consequence of edge-avoidance by young forest species (King et al., 2009; Schlossberg & King, 2008). In our study, the width of existing ROW/WO had a strong, positive influence on indigo bunting abundance and young forest guild richness. Furthermore, eastern towhee, indigo bunting, and the young forest guild responded positively to the 45-m wide cut-back border, the widest cut-back border treatment we tested, which supports the notion that wider ROWs support higher populations of young forest avifauna. However, we observed few responses to the 30-m wide treatment while eastern towhee, indigo bunting, and the young forest guild responded positively to the 15-m wide treatment. The nonlinear response we observed to cut-back border width is analogous to the relationship between young forest songbirds and ROW width reported by King et al. (2009). More importantly however, our results indicated that even the narrowest cut-back borders (15-m wide treatment) can support young forest songbird populations.

Greater reduction of the overstory layer in the more intensive harvests (4.5 m^2/ha cut-back borders) appeared to positively influence more young forest species than the less intensive 14 m^2/ha harvest retention treatment. The inverse relationship between young forest species' densities and overstory cover we observed are consistent with findings from other studies (Akresh et al., 2021; Sheehan et al., 2014). Greater overstory removal in the 4.5 m^2/ha treatments also induced greater understory vegetation growth in the >1–2 m level (Table 1), which were likely to provide greater nesting and foraging opportunities for young forest species. Greater amounts of felled trees in the 4.5 m^2/ha basal retention treatment relative to the 14 m^2/ha basal retention treatment probable added structure to the understory and affected the distribution of species within the cut-back border (Grodsky et al., 2016). Similar to cut-back border width, the responses by young forest species to cut-back border intensity factor levels may not be observable immediately following treatment, as some young forest species can have delayed responses to timber harvesting (Schlossberg & King, 2009). However, there were a greater number of responses at 2 years post-treatment than at 1 year post-treatment indicating the potential beginning of an upward trend in young forest songbird populations in cut-back borders. As understory vegetation in cut-back borders continues to develop, managers could gain better insight into the effectiveness of cut-back borders for managing individual young forest species and the young forest songbird community.

A forest gap species, hooded warbler, and the forest gap habitat guild responded positively to the narrowest cut-back border (15-m wide treatment) relative to the control. Forest gap specialists occupy forest interior habitats but are often associated with canopy disturbances that retain some level of overstory (Boves et al., 2013; Sheehan et al., 2014) or small-scale forest edges (Weakland & Wood, 2005). The 15-m wide cut-back border combined with the adjacent mature forest retained greater overstory than the 30-m treatment (Table 1) and provided small-scale edge. Furthermore, two-age and shelterwood harvests (much like the 14 and 4.5 m^2/ha harvest prescriptions implemented in our study) often benefit forest gap species because they provide more complex canopy structures (Nareff et al., 2019; Sheehan et al., 2014; Wood et al., 2005). It appears that cut-back borders provided these suitable habitat conditions immediately following treatment, as hooded warbler abundance and forest gap guild richness increased during the post-harvest period regardless of harvest intensity. However, conditions in partial harvesting stands are ephemeral for forest gap specialists, as tree crowns expand to fill the unoccupied canopy gaps and therefore eliminate sallying or singing perches important for this group (Sheehan et al., 2014). Future monitoring to assess how forest gap species respond as canopy gaps close will provide important insight into the long-term suitability of cut-back borders for forest gap specialists.

Forest interior avifauna are often negatively affected by the presence of ROWs/WOs within forest-dominated landscapes due to the fragmentation and loss of core forest habitat (Barton et al., 2016; Farwell et al., 2016). Further reduction of habitat suitability via cut-back borders along ROWs/WOs for forest interior species would limit the practicality of using cut-back borders for broad forest songbird community management and conservation. We did not detect any negative responses by forest interior species to cut-back border treatment (*TRT*) at the stand scale, suggesting that cut-back borders did not reduce habitat suitability for forest interior species along ROWs/WOs. The only responses we observed from the forest interior group was by black-and-white warbler that responded positively to the more intensive, 4.5 m^2/ha retention treatment. Because width of the existing ROW/WO had a negative influence on forest interior birds, management aimed at this guild would be optimized along narrower corridors using more intensive, narrower cut-back borders.

During the spring stopover period, songbird richness decreased in cut-back borders immediately following treatment that was expected due to migratory songbirds relying heavily on foliar-occurring arthropods during the spring migration period (Strode, 2009). Leaf phenology can affect foliar-occurring arthropod abundance and therefore habitat use by migrant songbirds (Rodewald & Brittingham, 2007). Dormant season tree cutting and little time for plant growth before migrants arrived probably limited the availability of leaf area for foliar-occurring arthropods. Therefore, cut-back borders are likely to provide few foliar-occurring arthropods relative to uncut areas that initially reduced songbird richness during spring stopover in treated plots. However, species richness returned to pre-treatment levels by 2 years post-treatment. As vegetation in cut-back borders continues to develop, habitat suitability of cut-back borders for songbirds during the spring stopover period is likely to change. Long-term monitoring will help determine how species' responses change as vegetation in cut-back borders develops further.

Across all habitat groupings, songbirds were most responsive to narrow, more intense cut-back border treatments. The 15-m wide treatment was the most effective short-term approach for managing songbird habitats along abrupt forest edges and also the most cost-effective treatment relative to 30- and 45-m treatments after considering time and effort needed to implement each of the different width treatments. The 4.5 m^2/ha harvest retention treatment had the strongest influence on the songbird community of the two harvest intensities tested. Greater overstory disturbance in the 4.5 m^2/ha treatment induced vegetation growth in the understory that benefited young forest species while still retaining sufficient overstory trees for mature forest species. The interaction between cut-back border harvest width and harvest intensity tested here provides managers with options to suit their specific objectives and species of interest. Additionally, abundances and richness of songbirds increased over time irrespective of cut-back border treatment. Species responses are likely to be dynamic through time, but the information provided here can assist managers in their decision-making when faced with managing songbird populations in human-modified forest landscapes of the central Appalachian region.

Our results highlight the potential usefulness of habitat management via cut-back borders along existing abrupt forest edges to improve these areas for songbird abundance and richness. Our results do not suggest that the creation of more ROWs can promote songbird habitat management. Additionally, our study used baseline population indices (abundance and species richness) to assess the short-term (up to 2 years post-treatment) effects of cut-back borders on songbirds. These results are initial steps and future research can expand on our findings. More expansive field evaluations (e.g., reproductive success, apparent survival, emigration, immigration) on the effects of cut-back borders will provide insight into the tradeoffs for mature and young forest songbirds and will continue to advance our understanding of how this management approach can be used to improve songbird habitat in these areas.

CONCLUSIONS

Cut-back borders along existing edges can benefit abundance and richness of young forest species while minimally affecting that of mature forest species and species of regional conservation priority. Although there is often a short-term tradeoff between creating habitat for young forest species and preserving habitat for mature forest species, implementation of cut-back borders in our study created suitable understory vegetation conditions for the young forest community while retaining sufficient overstory vegetation conditions that preserved suitable habitat for the mature forest community. However, as vegetation continues to grow in cut-back borders, species responses to cut-back border treatments are likely to change. Continued monitoring of cut-back border harvest width and harvest intensity prescriptions may help to elucidate temporal responses to cut-back borders and may aid land managers in developing long-term management strategies that meet state and regional avian conservation goals.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data (Margenau et al., 2021a) are available on Dryad at https://doi.org/10.5061/dryad.gb5mkkwqr. Code (Margenau et al., 2021b) is available on Zenodo at https://doi.org/10.5281/zenodo.5579573.

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SUPPORTING INFORMATION

Additional supporting information may be found in the online version of the article at the publisher's website.

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