Response of Vegetation and Avian Communities to Retention Harvests in a Mixed-pine Forest in Northern Alabama

by

Rachelle Soudriette Sterling

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Approved by

Mark D. Smith, Chair, Associate Professor of Wildlife Sciences
James B. Grand, Professor of Wildlife Sciences
Gary R. Hepp, Professor of Wildlife Sciences

ABSTRACT

Retention harvests are commonly used silvicultural methods that mimic natural disturbance and create a mosaic of mature and early-successional habitats which may simultaneously provide habitat for both mature-forest and disturbance-dependent birds. My objectives were to assess how vegetative structure affects bird occupancy and population dynamics before and after a retention harvest which removed 43% to 69% of stand basal area. I constructed species-specific, robust design occupancy models to assess the effects of microhabitat variables on bird occupancy, colonization, and local extinction within a retention-harvested mixed-pine forest in northern Alabama during 2011 and 2012. Before harvest, sites with greater density of understory vegetation <1m tall were more likely to be occupied by three disturbance-dependent species: indigo buntings (*Passerina cyanea*), white-eyed vireos (*Vireo griseus*), and yellow-breasted chats (*Icteria virens*). Following harvest, indigo buntings were more likely to become locally extinct on sites with greater basal area. Sites with greater post-harvest basal area were more likely to be colonized by wood thrushes (*Hylocichla mustelina),* but less likely to be colonized by red-eyed vireos (*Vireo olivaceus*) and red-headed woodpeckers (*Melanerpes erythrocephalus*). This research examined pre- and initial post-harvest conditions, creating a baseline for future research exploring shifts in forest bird communities across developmental stages of a retention-harvested mixed-pine forest.

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CHAPTER I: INTRODUCTION

Disturbance is a critical component of many forest ecosystems as it has played a key role in shaping the evolutionary and ecological histories of present day forests (Abrams 1992, Attiwill 1994, Brawn et al. 2001). Natural disturbance events include fire, tornadoes, windthrow, insect outbreaks and natural tree die-off, all of which create canopy openings, spur understory growth, and ultimately create a mosaic of mature and early-successional habitats (Brawn et al. 2001). In recent decades, disturbance regimes have become increasingly rare, resulting in predominantly closed-canopy forests and significant declines of disturbance-mediated habitats across much of the eastern United States (Hunter et al. 2001, Trani et al. 2001). Many disturbance-dependent birds associated with these habitats are threatened, or classified as sensitive species of management concern largely as a result of habitat loss (Askins 2000, Hunter et al. 2001). With natural disturbance becoming increasingly rare, silvicultural methods such as retention harvests are important practices for creating vegetative structural diversity that provides nesting and foraging substrates for declining disturbance-dependent species (Thompson et al. 1996).

Vegetative attributes, particularly vegetative structure, are known to strongly influence avian habitat use (Johnston and Odum 1956, James 1971, DeGraaf et al. 1998, Artman et al. 2001). Increased vertical vegetative structure often correlates with increased avian species diversity as greater niche availability supports a greater diversity of foraging and nesting guilds (MacArthur & MacArthur 1961, James 1971). Breeding birds select vegetative attributes that are primarily associated with nest site availability and food abundance (Brawn et al. 2001, Rodewald and Yahner 2001). The feeding and nesting strategies of forest birds provide insight to which vegetative attributes are most important in determining habitat use. For example, aerial-feeding insectivores benefit from increased arthropod availability or diversity in habitats featuring a more open canopy (Thompson et al. 1996, Shields et al. 2008, Tozer et al. 2010) whereas habitats that offer greater ground and shrub cover provide more foraging substrates to ground-foraging species (Costello et al. 2000). Live and dead trees are required for foliage or bark-gleaning species as well as cavity- and canopy-nesting species. Shrub-nesting species select habitats featuring dense layers of low or medium shrubs that provide abundant nesting locations (Baker and Lacki 1997, Thompson et al. 1997). These broad guidelines (provided by Ehrlich et al. 1988) were used to focus the vegetation surveys (see Methods section) on these attributes that influence bird habitat use.

Vegetative attributes are known to strongly influence habitat selection by birds, thus an understanding of the effects of management activities on vegetation is crucial to predicting bird community response. Uneven-aged silvicultural methods often stimulate dense understory vegetation growth in response to increased light availability and decreased competition (Crawford et al. 1981, Baker and Lacki 1997, Waldrop et al. 2008). Reduced basal area in hardwood stands has been shown to have direct positive correlation with woody and herbaceous plant production (Beck and Harlow 1981, Nelson and Graney 1996) and there exists a strong positive relationship between the proportion of basal area removed and the magnitude of change in understory vegetation (Yanai et al. 1998, Zenner et al. 2006). Thus it is not surprising that the degree of change in vegetation caused by silviculture influences the bird community turnover from mature-forest to early-successional species (Annand and Thompson 1997).

Management techniques that fail to open the canopy may not achieve management goals that include creating habitat for early-successional species. Many early-successional species are area sensitive and require a minimum canopy opening size. For example, eastern towhees

(*Pipilo erythrophthalmus*) are often absent from forest openings <0.65 ha in size (Annand and Thompson 1997, Costello et al. 2000). Similarly, understory management techniques such as prescribed fire and fertilization may have little effect on both vegetation and bird communities within closed-canopy forests (Artman et al. 2001, Waldrop et al. 2008, Shaw et al. 2010). If management goals include creating habitat for early-successional species, management techniques must open the canopy to stimulate growth of understory vegetation and species-specific area sensitivities should be considered to reach management goals.

Although many early-successional species exhibit a threshold for minimum canopy opening size, management that aims to support mature-forest birds as well must balance canopy openness with live tree or snag retention to create adequate vegetative structural diversity for species requiring habitats with mature tree elements. Harvests which remove >50% of standing basal area stimulate dense shrub and sapling regeneration which supports early-successional species such as yellow-breasted chats (*Icteria virens*), eastern phoebes (*Sayornis phoebe*), blue grosbeaks (*Passerina caerulea*) and white-eyed vireos (*Vireo griseus*) but may cause significant declines in abundance of mature-interior forest birds including ovenbirds (*Seiurus aurocapilla*) and worm-eating warblers (*Helmitheros vermivorum*; Wang et al. 2006, Newell and Rodewald 2012). Many canopy-nesting mature-edge species, including eastern wood-pewees (*Contopus virens*) and scarlet tanagers (*Piranga olivacea*), respond positively to this level of harvest but others such as red-eyed vireos (*Vireo olivaceus*) may decline (Annand and Thompson 1997, Newell and Rodewald 2012). Response of mature forest birds to partial harvest techniques is highly varied and local research should be conducted to determine relevant bird-habitat relationships.

Bird community response to silviculture may evolve over time as stand vegetation regenerates and several studies report stand age may be more important than retention level in predicting bird presence or absence (McDermott and Wood 2008, Otto and Roloff 2012). Following harvest, there is often a 1 to 2 year lag in understory vegetation regeneration which delays colonization of significant numbers of early-successional birds (Robinson and Robinson 1999, Newell and Rodewald 2012). Understory herbaceous and woody biomass may begin to decline as early as 4 years post-harvest (Beck and Harlow 1981, Peitz et al. 2001) and sapling densities level off or decrease 10 to 15 years post-harvest in response to overstory canopy closure (Yanai et al. 1998). With time these vegetation characteristics regenerate to pre-harvest forest conditions, thus it is not surprising that densities of early successional birds decline as early as 3 to 4 years post-harvest and by 11 to 15 years post-harvest the bird communities may consist entirely of mature-forest birds observed in pre-harvest stands (Robinson and Robinson 1999, McDermott et al. 2011). Long-term studies of bird response to silviculture emphasize that especially for partial-harvest methods, mature-forest bird populations are unlikely to be affected if stands are allowed to regenerate (McDermott and Wood 2008, Campbell et al. 2012). The use of post-harvest understory management techniques is often recommended to maintain abundant, diverse understory vegetation within forest openings which will likely maintain populations of early-successional and mature forest birds (Thompson et al. 1996, Hunter et al. 2001, Artman et al. 2005).

Vegetative and bird community response to silviculture varies with degree of change brought about by harvest, but variation also exists across forest types and regions as well. This study aims to explore the responses of microhabitat vegetation and avian population dynamics in response to a retention harvest which removed 43% to 69% of stand basal area within a mixed-pine forest of northern Alabama. This study includes both pre- and post-treatment surveys to assess the degree of change brought about by a partial harvest of this intensity. Species-specific models were constructed to test for relationships between vegetation characteristics and probability of bird occupancy, colonization, and local extinction. Information gathered by this study will contribute to a broader understanding of retention harvest management in this region, where partial harvests are a common method of habitat management for game species.

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CHAPTER II: RESPONSE OF MICROHABITAT VEGETATION TO A RETENTION HARVEST WITHIN A MIXED-PINE FOREST IN NORTHERN ALABAMA

ABSTRACT

Retention harvests are an uneven-aged silvicultural technique which retain a specific amount of the original standing basal area with the intention to provide a mosaic of both mature-forest and early-successional elements. As retention harvests are common practices for wildlife habitat management across the southeastern United States, it is important to understand the impacts of this technique on the structure and composition of over and understory vegetation which may influence habitat use by wildlife. My objectives were to quantify overstory and understory vegetative characteristics before and after a retention harvest which removed 43% to 69% of the standing basal area in a mixed-pine forest in Huntsville, Alabama from 2010-2012. After harvest, stand basal area was reduced from approximately 29.12 m^2/ha (SE = 1.55) to 12.67 m^2/ha (SE = 1.03), mean canopy closure was reduced from approximately 82% (SE = 3%) to 37% (SE = 3%). Understory vegetation density <1 m in height (65% cover 0 m to 0.5 m; 27% cover 0.5 m to 1.0 m) had almost reached pre-harvest levels (83% cover 0 m to 0.5m; 41% cover 0.5 m to 1.0 m) 1 year following harvest, and studies suggest these densities will increase rapidly for several years. Ground cover was predominantly woody plants and vines both before and after harvest. Harvested sites featured less grass and leaf litter coverage but greater bare ground and forb coverage. Over the next few growing seasons, I expect understory vegetation densities to increase for several years until expanding canopy closure and competition slows growth. Continued monitoring of vegetation dynamics as the stand develops will be necessary to determine the timing of future understory management in order to maintain desired early-successional plant communities within these stands.

INTRODUCTION

Retention harvests are one type of uneven-aged silviculture which retain a specific amount of the original standing basal area with the intention to create a mosaic of both mature-forest overstory and early-successional understory habitat elements. Retention harvests punctuate closed-canopy forests to allow light to reach the forest floor and spur rapid understory growth. As a result of the rapid regeneration of understory vegetation, retention harvests often increase structural diversity within a stand to provide cover, foraging, and nesting substrate for wildlife (deCalesta 1994, Jackson et al. 2007, Lashley et al. 2011). Because vegetative structure and composition are known to strongly influence habitat selection by wildlife, an understanding of the effects of management activities on vegetation is crucial to predicting wildlife community response to harvest.

Retention harvests are common practices for wildlife habitat management across the southeastern US. Songbird community response to various silviculture practices has been well studied in the central hardwoods (Crawford et al. 1981, Annand and Thompson 1997, Robinson and Robinson 1999) and the southeast pine-hardwoods regions (Alterman et al. 2005, Wang et al. 2006) which report the importance of forest canopy openness in altering understory vegetation attributes and ultimately influencing bird species presence or absence. Wild turkeys (*Meleagris gallopavo*) are known to use upland, intensively managed forests as breeding and chick rearing habitat in the southeastern US (Smith et al. 1990, Miller and Connor 2007). The understory vegetation regenerated by uneven-aged harvests also provides suitable forage and cover for white-tailed deer (*Odocoileus virginianus*) in the southeast region (Lashley et al. 2011). Retention harvests increase the density, height, and composition of understory vegetation which will have significant impacts on wildlife habitat use across the southeastern US.

Uneven-aged silvicultural methods often stimulate dense understory vegetation growth in response to increased moisture, nutrient, and light availability (Gilliam et al. 1995, Baker and Lacki 1997, Waldrop et al. 2008). Microhabitat characteristics change rapidly following stand harvest, as many studies report significant increases in sapling stem density, woody vine density, and herbaceous biomass (Peitz et al. 2001, Schumann et al. 2003). Studies that examined a gradient of harvest intensities suggest a strong positive relationship between the proportion of basal area harvested and the amount of understory biomass regeneration (Yanai et al. 1998, Zenner et al. 2006). However, this rate of regeneration slows over time and may peak or level off 3 to 10 years following harvest as the overstory canopy redevelops and understory competition reduces resource availability (Nelson and Graney 1996, Yanai et al.1998). Retention harvests which remove large proportions of basal area are expected to cause significant changes in microhabitat characteristics; however, the initial 1 to 2 years after harvest may experience the most rapid response.

My study quantified changes in forest vegetation structure with an emphasis on understory vegetation density and composition following a retention harvest which removed 43% to 69% of the original basal area. I compared pre- and 1-year post-harvest vegetation response to monitor understory plant community response and how these changes may influence bird response (Chapter III). I report a description of stand and microhabitat characteristics both before and after harvest which will serve as a baseline for future studies of long-term stand dynamics and wildlife response to retention harvests in pine-hardwood forests.

STUDY AREA

My study took place on the Redstone Arsenal (RA; 34.65°N, 86.66°W), a 15,050-ha US Department of Defense landholding in Huntsville, Alabama in Madison County (Figure 2.1).

The RA is located on the Eastern Highland Rim of north Alabama and features well-drained clay or loam soils typical of the physiographic region (Smalley 1983). Elevations range from 170-206 m above mean sea level and mean temperatures range from -0.5 °C to 9.4 °C in January and from 20 °C to 31.6 °C in July (Bhuta et al. 2011). Mean annual rainfall is 140 cm with March receiving the most precipitation and October the least.

Prior to RA's acquisition of the land in 1949, the land cover was primarily row crop or grazing pastures and experienced little active forest management until 1970 (Redstone Arsenal 2002). The land has largely been converted to non-forested or urban landscape to achieve RA's goals of supporting the US Army in military research and the development and distribution of weapons systems, services, and supplies (Redstone Arsenal 2002). Forested land cover on RA is highly fragmented due to widespread non-forest and urban patches distributed throughout the landscape (Bhuta et al. 2011). My study area was a 544-ha forested tract located along the southwestern border of RA. In order of total basal area, the pre-harvest study area was comprised of predominantly loblolly pine (*Pinus taeda*), sweetgum (*Liquidambar styraciflua*), willow oak (*Quercus phellos*), yellow poplar (*Liriodendron tulipifera*), and cherrybark oak (*Quercus pagoda*).

My study design consisted of four, 51-ha (SE = 4.51) replicates arranged north to south with each replicate consisting of four plots in a randomized complete block design (Hurlburt 1984). Within each replicate, three plots received retention harvests and one remained an un-harvested control plot. Although the harvest treatments are currently applied unevenly, this is the beginning of a long-term study to examine the effects of harvest-and-burn methods and seasonality of prescribed burning thus two of the harvested plots will receive growing or dormant season burns in the future. Redstone Arsenal staff determined and marked trees for harvest, targeting loblolly pine, and contracted a local operator to perform the harvest. One replicate was harvested in March 2011 and two replicates were harvested in March 2012. The fourth, southernmost replicate (hereafter “floodplain”) remained un-harvested, but is scheduled for harvest in the future. The floodplain replicate was located nearest to the Tennessee River and flooded seasonally during the winter unlike the other three replicates which were located at higher elevations (hereafter “upland”). Retention harvests reduced the basal area of each plot by 43% to 69% to an average of 12.67 m^2/ha (SE = 1.03; Table 2.2), retaining hard and soft mass trees as well as snags to provide nesting and foraging substrates for mature forest birds.

METHODS

Vegetation Surveys

From July to August 2011 and 2012, I conducted vegetation surveys to measure understory vegetation structure and composition in pre- and post-retention harvest plots. Surveys were conducted at 10 randomly selected points/plot for a total of 40 points/replicate and a study total of 160 points. Vegetation surveys included estimates of overstory canopy cover, understory vertical density, ground cover, and dominant understory species. Random survey points were generated using ArcMap 9.0 (ESRI 2009) under the stipulation that points be >35 m apart and >35 m from the plot edge (Cook et al. 1995). This protocol avoids overlap between points or from neighboring plots in regards to canopy cover estimates (Cook et al. 1995). From each survey point, one 10-m transect was laid out in each cardinal direction. Canopy cover was estimated using a moosehorn (Robinson 1947) to take measurements at each survey point and at 2-m intervals along each 10-m transect (Robbins et al. 1989, Cook et al. 1995, Rodewald and Yahner 2001). Ground cover was estimated using the line-intercept method (Canfield 1941) along each 10-m transect. Ground cover was identified by functional group: water, bare ground, leaf litter, moss, fern, grass, forb, vine, woody plant, or coarse woody debris.

Vertical vegetation structure was estimated using a vegetation density board (Nudds 1977). The board measured 2.44 m tall, 30.48 cm wide and 0.95 cm thick and was vertically divided vertically into 5 equal-sized sections, each painted alternating black or white. Estimation of percentage vegetation covering each block was made from a distance of 10 m from plot center assessed at an eye-level of approximately 1 m in height.

Fixed-radius plots (0.04 ha; James and Shugart 1970) were used to measure species composition, trees/hectare, and basal area of both live and standing dead trees (snags). Tree surveys were conducted at 10 randomly selected points/plot for a total of 40 points/replicate and a study total of 160 points. Tree attributes were measured at the same plot location before and after retention harvest. At each tree survey point, all trees and snags >4 cm in diameter at breast height (dbh) were identified by species and measurements of dbh were taken using a dbh tape.

Data analysis

I calculated pre- and post-harvest means and standard errors for basal area, canopy closure, understory vegetation layer densities (0 m to 2.5 m in height), and percentage ground cover by functional group at the plot level. Because understory burn treatments had not yet been applied, I compared pooled estimates from the three harvested plots to estimates from pre-harvest plots across the three, treated replicates (“upland” replicates). Estimates from the un-harvested, floodplain replicate were considered separate from those of the upland replicates.

RESULTS

In the upland replicates, mean basal area (29.12 m^2/ha, SE = 1.55) and mean percentage canopy closure (82%, SE = 3%) was greater in pre-harvest plots than post-harvest plots (BA =

12.67 m^2/ha, SE = 1.03; canopy closure = 37%, SE = 3%; Table 2.2). Likewise, overall tree density was greater on pre-harvest plots ($\bar{x}$ = 401 trees/ha) than on post-harvest plots ($\bar{x}$ = 180 trees/ha). Percentage of basal area removed from harvested plots ranged from 42.7% to 69.3% (Table 2.2). Overall mean basal area (24.64 m^2/ha, SE = 2.10) and tree density ($\bar{x}$ = 331 trees/ha) of control plots was slightly less than those of pre-harvest plots, but percentage canopy closure (84%, SE = 1%) was similar to canopy closure of pre-harvest plots. Overall mean basal area of the floodplain replicate was 32.58 m^2/ha (SE = 3.45) and the percentage canopy closure was 88% (SE = 1%). Tree density was 424 trees/ha in the floodplain replicate (Table 2.2).

In the upland replicates, mean snag basal area (1.75 m^2/ha, SE = 0.45) and snag density ($\bar{x}$ = 13.89 snags/ha) were greater on post-harvest sites than pre-harvest sites (BA $\bar{x}$ = 0.97 m^2/ha, SE = 0.31; snag density $\bar{x}$ = 7.08 snags/ha). Overall mean snag basal area of control plots was 2.56 m^2/ha (SE = 1.06) and snag density was 23.33 snags/ha. Overall mean snag basal area of the floodplain replicate was 2.19 m^2/ha (SE = 0.84) and snag density was 18.75 snags/ha.

Stand composition before harvest was co-dominated by oak (*Quercus* spp.) and loblolly pine (*Pinus taeda*; Table 2.1). Several hardwood species were also abundant in the pre-harvest stands including, in order of overall abundance, sweetgum (*Liquidambar styraciflua*), yellow poplar (*Lirodendron tulipifera*), and elm (*Ulmus* spp.). Additional, less represented species were categorized as “other” and included, in order of abundance, flowering dogwood (*Cornus florida*), black locust (*Robinia pseudoacacia*), red maple (*Acer rubrum*), black tupelo (*Nyssa sylvatica*), and green ash (*Fraxinus pennsylvanica*). Stand composition for harvest plots shifted to slightly more oak and less pine from pre-harvest conditions although proportions of other species remained relatively unchanged (Table 2.1).

Across all treatment types and both before and after harvest, the ground layer vegetation from 0 m to 0.5 m in height was most dense (range 0.83 to 0.32 cover) with density gradually decreasing with increasing height (Table 2.3). In the upland replicates, mean understory density at ground level (0 m to 0.5 m in height) was greater on pre-harvest plots ($\bar{x}$ = 0.83 cover, SE = 0.02) than post-harvest plots ($\bar{x}$ = 0.65, SE = 0.02). Mean understory density at ground level for control plots ($\bar{x}$ = 0.67, SE = 0.03) was similar to that of post-harvest. Understory vegetation density from 1.0 m to 2.5 m in height was reduced by 87% from pre-harvest ($\bar{x}$ = 0.62 cover) to post-harvest ($\bar{x}$ = 0.08 cover). In the floodplain replicate, mean understory vegetation density was greatest from 0 m to 0.5 m ($\bar{x}$ = 0.32 cover, SE = 0.02) and densities from 0.5 m to 2.5 m were similar (0.12 to 0.14 cover; Table 2.3).

Among upland replicates, understory ground cover consisted predominantly of woody plants and vines on pre-harvest (woody $\bar{x}$ = 0.45, SE = 0.03; vine $\bar{x}$ = 0.32, SE = 0.03) and post-harvest plots (woody $\bar{x}$ = 0.36, SE = 0.03; vine $\bar{x}$ = 0.26, SE = 0.02; Table 2.4). Pre-harvest plots had greater mean proportions of leaf ($\bar{x}$ = 0.10, SE = 0.02) and grass ($\bar{x}$ = 0.07, SE = 0.02) than post-harvest plots (leaf $\bar{x}$ = 0.06, SE = 0.01; grass $\bar{x}$ = 0.03, SE = 0.01; Table 2.4). Post-harvest plots had greater proportions of forbs ($\bar{x}$ = 0.11, SE = 0.02) and coarse woody debris ($\bar{x}$ = 0.05, SE = 0.01) compared to pre-harvest plots (forbs $\bar{x}$ = 0.04, SE = 0.01; coarse woody debris $\bar{x}$ = 0.01, SE = 0.00) and bare, exposed dirt was only found on post-harvest plots ($\bar{x}$ = 0.13, SE = 0.02). Upland control sites had greater leaf ($\bar{x}$ = 0.20, SE = 0.04) and vine cover ($\bar{x}$ = 0.44, SE = 0.06) than other upland treatments. Floodplain plots were characterized by greater proportions of leaf ($\bar{x}$ = 0.37, SE = 0.05) and water cover ($\bar{x}$ = 0.23, SE = 0.06) than upland plots (Table 2.4).

DISCUSSION

Differences in pre-harvest vegetation between control and pre-harvest plots were unexpected and likely due to random sampling variation. Control plots had lesser basal area and greater proportions of oak basal area, vine cover, and leaf cover than pre-harvest plots. Control plots also had lower mean understory densities across all strata compared to pre-harvest densities. As treatments were assigned randomly, I believe these differences were due to chance and accurately represent the variation that existed across the study area.

Harvested forests may have substantial regeneration of shade-intolerant hardwood species such as yellow poplar. Lanham et al. (2002) suggest that the shade provided by retention-harvested stands may slow the growth of yellow poplar; however, they suggest an average canopy closure of 35% may not provide substantial understory shading, thus yellow poplar may regenerate substantially on my harvested sites with a mean canopy closure of 37%. In stands reduced to a similar retention level (13 m^2/ha basal area), Brose et al. (1999) found prescribed fire was necessary to reduce abundance of yellow poplar saplings by 50% in shelterwood-harvested stands. Yellow poplar and sweetgum are commonly classified as less-desirable tree species which may experience rapid growth in canopy gaps created by retention harvests on my study sites. I expect that the proposed prescribed burn treatments will help control the regeneration of shade-intolerant hardwood species such as yellow poplar and sweetgum.

Even though understory vegetation was greatly reduced caused by heavy machinery use during retention harvest, post-harvest understory density on the study plots suggests that ground level vegetation <1 m in height may approach pre-harvest levels as early as the first growing season. Zenner et al. (2006) found that tree sapling density and woody vine density was two and seven times greater, respectively, within 3 to 4 years of group harvest application. Yanai et al. (1998) also reported a strong positive relationship between understory sapling stem density and overstory thinning in an Alleghany hardwood forest. Both studies concluded that understory vegetation abundance increases with greater harvest intensity. However, Lashley et al. (2011) found that compared to harvest-and-burn or harvest with herbicide applications, harvesting alone provided significantly less understory forage biomass for white-tailed deer in upland hardwood forests of Tennessee. Although retention harvests will likely increase understory density over time as compared with closed-canopy mature forests, additional understory management may provide even greater understory biomass and vegetative structure for wildlife use.

During the first year post-harvest, my harvested plots supported nearly twice the proportion of forb cover which was found in control or pre-harvest plots. In mixed pine-hardwood stands in Arkansas, Peitz et al. (2001) applied variable retention harvests and reported herbaceous forage coverage and biomass were negatively correlated with retained basal area. In those stands, all herbaceous biomass increased within 2 years post-harvest but by 4 years post-harvest herbaceous biomass began to decline. As my study took place within a similar forest type, I expect a similar trend in herbaceous biomass growth will develop on harvested plots over time. Schumann et al. (2003) studied long-term vegetation response to a lighter harvest than mine which removed only 42% of pre-harvest basal area in mixed-pine forests of Maine. They reported that 6 to 11 years following harvest, harvested sites alone supported light-demanding understory plants such as *Rubus* spp. but they did not provide greater abundances of forbs. As herbaceous and forb cover are known to encourage habitat use by wild turkeys, northern bobwhite and other wildlife, this study's harvested plots may need additional long-term management such as fire to maintain forb cover that may decrease over time if left unmanaged.

The initial results of this study suggest that understory vegetation density and composition may change rapidly following harvest, although long-term monitoring is necessary to fully understand vegetative response to retention harvests. From previous studies, I expect rapid understory regeneration to peak in several years, surpassing pre-harvest plots, at which time the proposed prescribed burn treatments included in this study will maintain suitable forage and nesting substrate for wildlife.

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Figure 2.1. Map of study site location within Redstone Arsenal, Madison County, Alabama.

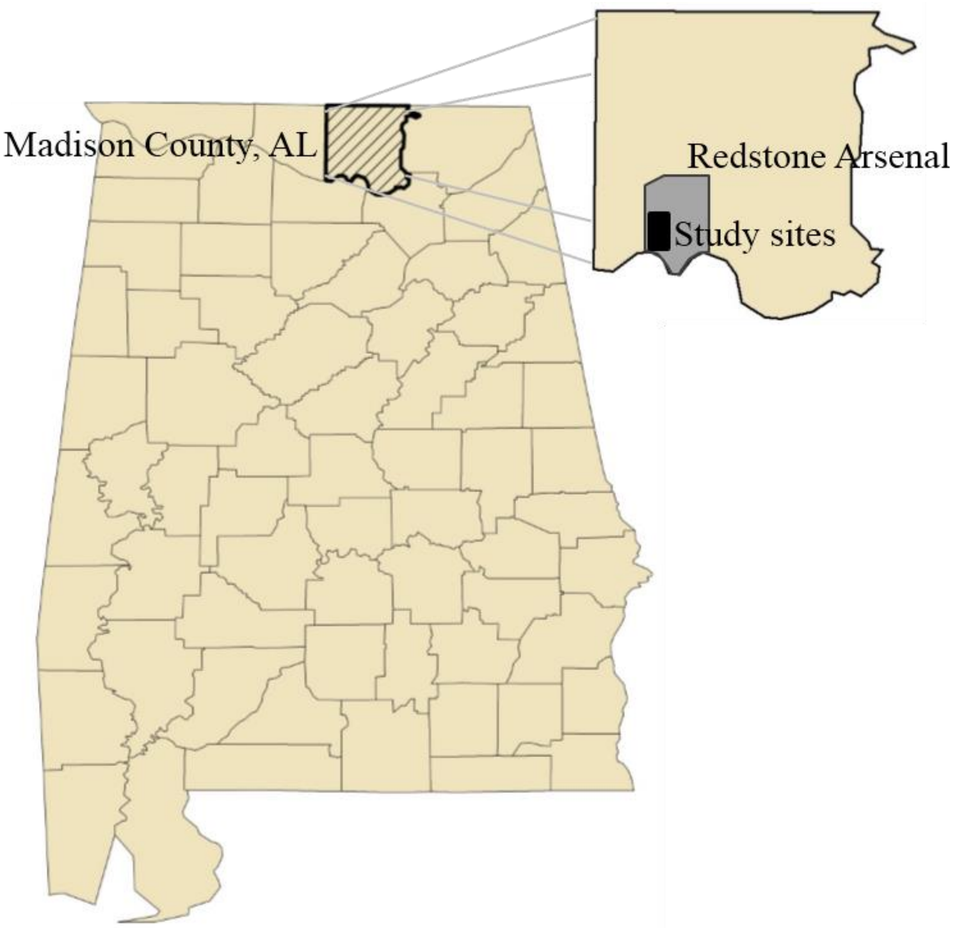

Table 2.1. Relative proportion of basal area by species in treatment plots of a retention-harvested mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011-2012.

Treatment	Oak	Loblolly pine	Sweetgum	Yellow poplar	Elm	Hackberry	Black cherry	Eastern red cedar	Hickory	Sassafras	Other
Pre-harvest	19.10%	43.21%	12.95%	11.44%	2.54%	2.08%	3.00%	2.20%	0.00%	1.22%	2.27%
Post-Harvest	26.11%	35.77%	13.53%	11.69%	3.91%	1.98%	2.13%	0.64%	0.02%	1.55%	2.67%
Control	39.11%	27.51%	10.24%	13.30%	3.06%	1.10%	1.28%	0.17%	0.38%	1.52%	2.33%
Floodplain	48.64%	8.38%	22.94%	1.20%	2.14%	2.55%	0.51%	2.64%	6.16%	0.16%	4.67%

Table 2.2. Mean (SE) proportion canopy closure, basal area (m^2/ha), snag basal area (m^2/ha), percentage basal area removed, and density of trees and snags/hectare in a retention-harvested mixed-pine forest in Huntsville, Alabama in 2011-2012.

Block	Plot	Treatment	Canopy closure (SE)	Basal area (SE)	Tree density	Percentage basal area harvested	Snag basal area (SE)	Snag density
1	1	Pre-harvest	0.89 (0.04)	23.61 (1.63)	272		0.80 (0.51)	5
		Post-harvest	0.39 (0.07)	10.03 (1.47)	124	57.5%	2.00 (0.61)	22
	2	Control	0.82 (0.03)	23.18 (1.35)	341		1.83 (1.43)	20
	3	Pre-harvest	0.82 (0.03)	34.33 (2.68)	588		0.00	0
		Post-harvest	0.16 (0.04)	13.71 (1.94)	183	60.1%	2.58 (1.96)	15
	4	Pre-harvest	0.90 (0.03)	23.86 (3.97)	247		1.33 (1.32)	7
		Post-harvest	0.38 (0.06)	8.37 (2.23)	131	64.9%	3.68 (2.75)	30
2	1	Pre-harvest	0.86 (0.03)	24.87 (1.55)	282		0.75 (0.75)	2
		Post-harvest	0.33 (0.05)	12.79 (1.01)	185	48.6%	0.70 (0.52)	5
	2	Pre-harvest	0.82 (0.05)	37.44 (3.08)	351		1.78 (1.09)	5
		Post-harvest	0.36 (0.07)	12.19 (2.19)	136	67.4%	3.40 (1.49)	15
	3	Control	0.85 (0.04)	28.78 (3.09)	390		1.68 (0.52)	35
	4	Pre-harvest	0.92 (0.03)	29.59 (3.00)	558		0.00	0
		Post-harvest	0.37 (0.07)	9.09 (1.62)	101	69.3%	0.00	0
3	1	Control	0.86 (0.04)	21.94 (2.89)	262		4.18 (2.92)	15
	2	Pre-harvest	0.70 (0.04)	29.07 (1.77)	346		0.00	0
		Post-harvest	0.45 (0.06)	16.31 (2.48)	213	43.9%	0.30 (0.23)	5
	3	Pre-harvest	0.69 (0.03)	29.33 (3.42)	534		2.15 (1.88)	5
		Post-harvest	0.49 (0.07)	14.38 (1.37)	267	51.0%	0.68 (0.39)	17
	4	Pre-harvest	0.79 (0.01)	29.95 (3.35)	435		1.95 (0.85)	17
		Post-harvest	0.37 (0.07)	17.16 (2.00)	284	42.7%	2.43 (1.31)	15
4	1	Floodplain	0.87 (0.03)	35.42 (1.46)	376		3.30 (2.16)	20
	2	Floodplain	0.88 (0.03)	22.72 (2.36)	367		3.90 (2.41)	20
	3	Floodplain	0.90 (0.03)	33.57 (4.12)	578		0.13 (0.13)	5
	4	Floodplain	0.88 (0.04)	38.61 (5.86)	474		1.48 (0.74)	30
Overall		Pre-harvest[1]	0.84 (0.01)	24.64 (2.10)	331		2.56 (1.06)	23.33
		Post-harvest[1]	0.37 (0.03)	12.67 (1.03)	180		2.19 (0.84)	13.89
		Control[1]	0.82 (0.03)	29.12 (1.55)	401		0.97 (0.31)	7.08
		Floodplain	0.88 (0.01)	32.58 (3.45)	424		1.75 (0.45)	18.75

[1] Mean calculated from Blocks 1-3.

Table 2.3. Mean (SE) proportion of Nudds board strata (m) covered by understory vegetation density in treatment plots of a retention harvested mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011-2012.

	Strata Height (m)				
Treatment	0.0 - 0.5	0.5 - 1.0	1.0 - 1.5	1.5 - 2.0	2.0 - 2.5
Pre-Harvest	0.83 (0.02)	0.41 (0.03)	0.37 (0.03)	0.26 (0.03)	0.24 (0.03)
Harvest	0.65 (0.02)	0.27 (0.02)	0.13 (0.01)	0.06 (0.01)	0.04 (0.01)
Control	0.67 (0.03)	0.37 (0.04)	0.27 (0.04)	0.21 (0.03)	0.18 (0.03)
Floodplain	0.32 (0.02)	0.13 (0.01)	0.14 (0.02)	0.12 (0.01)	0.14 (0.01)

Table 2.4. Mean (SE) proportion of line-intercept transect covered by ground cover functional type in treatment plots of a retention harvested mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011-2012.

Treatment	Vine	Woody	Bare	Leaf	Forb	Coarse wood	Moss	Fern	Grass	Water
Control	0.44 (0.04)	0.24 (0.03)	0.00 (0.00)	0.20 (0.03)	0.05 (0.02)	0.01 (0.00)	0.00 (0.00)	0.01 (0.00)	0.05 (0.01)	0.00 (0.00)
Pre-Harvest	0.32 (0.03)	0.45 (0.03)	0.00 (0.00)	0.10 (0.02)	0.04 (0.01)	0.01 (0.00)	0.00 (0.00)	0.00 (0.00)	0.07 (0.02)	0.00 (0.00)
Harvest	0.26 (0.02)	0.36 (0.02)	0.13 (0.02)	0.06 (0.01)	0.11 (0.02)	0.05 (0.01)	0.00 (0.00)	0.00 (0.00)	0.03 (0.01)	0.00 (0.00)
Floodplain	0.12 (0.01)	0.24 (0.03)	0.00 (0.00)	0.37 (0.04)	0.00 (0.00)	0.02 (0.00)	0.00 (0.00)	0.00 (0.00)	0.01 (0.00)	0.23 (0.00)

CHAPTER III: PRE-HARVEST FOREST BIRD OCCUPANCY AND POST-HARVEST POPULATION DYNAMICS IN A RETENTION-HARVESTED MIXED-PINE FOREST IN NORTHERN ALABAMA

ABSTRACT

Natural disturbance events have shaped the evolutionary and ecological histories of forests across the eastern United States; however, suppression of natural disturbance regimes and resulting natural succession have led to loss of disturbance-mediated habitats and subsequent population declines of disturbance-dependent bird species. In the absence of disturbance, mature forests develop closed canopies that inhibit growth of ground and herbaceous understory vegetation and provide unsuitable breeding habitat for shrub-nesting, disturbance-dependent birds. If disturbance regimes are introduced, however, a unique suite of mature-forest breeding birds may be extirpated. Retention harvests are commonly used to create a mosaic of mature and early-successional habitats to simultaneously provide habitat for both mature-forest and disturbance-dependent birds. My objectives were to assess how vegetative structure affects bird occupancy and population dynamics before and after a retention harvest that removed 43% to 69% of stand basal area. I constructed species-specific robust design occupancy models to assess the effects of overstory and understory variables on bird occupancy, colonization, and local extinction within a mixed-pine forest in northern Alabama in 2011 and 2012. Before harvest, sites with greater density of understory vegetation <1m tall were more likely to be occupied by three disturbance-dependent species: indigo buntings (*Passerina cyanea*), white-eyed vireos (*Vireo griseus*), and yellow-breasted chats (*Icteria virens*). Occupancy rates for mature-edge species including several woodpeckers may increase in response to retention harvests. Sites with greater post-harvest basal area were more likely to be colonized by wood

thrushes (*Hylocichla mustelina),* but less likely to be colonized by red-eyed vireos (*Vireo olivaceus*) and red-headed woodpeckers (*Melanerpes erythrocephalus*). Indigo buntings were more likely to become extinct on sites with greater post-harvest basal area. This research examined pre- and immediate post-harvest conditions, creating a baseline for future research exploring shifts in forest bird communities across developmental stages of a retention-harvested mixed-pine forest. Retention harvests may cause a shift in bird communities of mixed-pine forests; however, few significant changes in occupancy, local extinction and colonization rates were observed 1 year post-harvest.

INTRODUCTION

Natural disturbance has shaped the evolutionary and ecological histories of present day forests across the eastern US (Abrams 1992, Attiwill 1994, Brawn et al. 2001). Wildfire, tornadoes, windthrow, insect outbreaks and natural tree die-off are examples of natural disturbance events which create canopy openings, spur understory growth, and ultimately create a mosaic of mature and early-successional habitats (Brawn et al. 2001). However, natural disturbance regimes (primarily wildfire) have become increasingly rare in recent decades, resulting in predominately closed-canopy forests and significant declines in disturbance-mediated, shrub habitats across much of the eastern US (Hunter et al. 2001, Trani et al. 2001). As a result of this habitat loss, many disturbance-dependent birds associated with these habitats are threatened or classified as sensitive species of management concern (Askins 2000, Hunter et al. 2001). With natural disturbance becoming increasingly rare, silvicultural methods such as retention harvests are important practices for creating vegetative structural diversity that provides nesting and foraging substrates for declining disturbance-dependent bird species (Thompson et al. 1996).

Retention harvests may mimic natural disturbance as they punctuate closed canopy forests and create patchy, early-successional shrub habitat for declining disturbance-dependent bird species. Bird community response to various silviculture practices has been well studied (Annand and Thompson 1997, Robinson and Robinson 1999, Alterman et al. 2005) and many studies report the importance of canopy openness in altering understory vegetation attributes and ultimately influencing bird species presence or absence (Crawford et al. 1981, Thompson et al. 1995, Jobes et al. 2004, Wang et al. 2006). Harvests that do not remove substantial amounts of basal area may not create large enough canopy gaps to support some early successional species (Robinson and Robinson 1999, Costello et al 2000). Yellow-breasted chats, for example, require canopy gaps no smaller than 0.7 ha to 3.0 ha (Gram et al. 2003, Alterman et al. 2005) whereas eastern towhees (*Pipilo erythrophthalmus*) were not observed within harvest gaps of 0.2 to 0.4 ha in a central hardwoods stand (Annand and Thompson 1997). Silvicultural methods can be used to achieve specific levels and patterns of canopy openings that may allow greater ability to reach species-specific habitat management goals.

Retention harvests are frequently used to enhance habitat quality for early-successional species; however, concern over declines of disturbance-dependent habitats is also met with opposition, as fragmenting closed-canopy forests may negatively affect mature-forest bird species (Thompson et al. 1992, Welsh and Healy 1993). Retention harvests that retain greater proportions of mature forest while encouraging patchy, understory development may provide vegetative structures required for both suites of birds. Several studies report that many mature-forest species exhibit a tolerance for low levels of canopy disturbance and will inhabit partially-harvested stands at similar or slightly lower densities (Robinson and Robinson 1999, Annand and Thompson 1997, Gram et al. 2003). Retention harvests may support both early-successional and mature-forest species but more research is needed to explore habitat sensitivities of both suites of birds.

Mature forest birds are often divided into two groups by their sensitivities to canopy gaps: mature-interior and mature-edge. Mature-edge birds such as eastern wood-pewees (*Contopus virens*) and great-crested flycatchers (*Myiarchus crinitus*) are known to respond positively to harvests that open the canopy, allowing for greater insect foraging opportunities (McCarty 1996) and nesting availability. Another mature-edge species, red-eyed vireos, may occur in lower abundances in harvested stands (Baker and Lacki 1997, Robinson and Robinson 1999) but they have been known to inhabit stands with as little as 3% to 25% canopy cover (Otto and Roloff 2012).

Mature-interior birds such as pileated woodpeckers (*Dryocopus pieatus*), wood thrushes, and Acadian flycatchers (*Empidonax virescens*) inhabit closed-canopy forests with sparse understories and may respond negatively to partial harvests that convert open-understory mature forests to uneven-aged forests with dense understories; however some species may be more sensitive than others. Pileated woodpeckers have been shown to abandon harvested stands (Beese and Bryant 1999) while Kentucky warblers (*Geothlypis formosa*) may actually increase abundance in response to gap-harvested forests (Robinson and Robinson 1999, Gram et al. 2003). Although several studies (Chambers et al. 1996, Doyon et al. 2005) have shown mature-interior species to tolerate low levels (25% to 35% basal area removed) of harvest, sensitive mature-interior species that abandon harvested sites initially may recolonize as the stand regenerates (Gram et al. 2003, McDermott and Wood 2009).

Disturbance-dependent, or early-successional, birds are expected to have a strong positive response to retention harvests that generate dense understories for increased foraging and nesting substrate. Nesting success of indigo buntings has been shown to increase with greater understory shrub height provided by partial-harvest silviculture (Alterman et al. 2005). Other shrub-nesting species including eastern towhees and yellow-breasted chats often exhibit a strong positive response to harvests (Annand and Thompson 1997, Gram et al. 2003), but these increases may not be evident for several years following harvest due to the lag in understory redevelopment (Robinson and Robinson 1999, Newell and Rodewald 2012).

Generalist species are expected to exhibit little change in response to harvests as they have greater plasticity of habitat selection. Abundance of tufted titmice (*Baeolophus bicolor*) may remain unchanged following selective harvests (Robinson and Robinson 1999); however Carolina wren (*Thryothorus ludovicianus*) abundance may increase dramatically in response to greater coarse woody debris availability (Newell and Rodewald 2012).

Though many studies (Annand and Thompson 1997, Gram et al. 2003, Alterman et al., 2005) have focused on the effects of active forest management on bird communities, species-specific responses are highly variable as the amount and dispersion of remaining vegetation varies widely. Many studies failed to gather or analyze vegetative data, leaving specific relationships between species and vegetation characteristics largely unknown (Baker and Lacki 1997, Beese and Bryant 1999, McDermott and Wood 2008, Tozer et al. 2010). This study includes both pre- and post-treatment vegetation surveys to quantify magnitude of change brought about by harvest. These vegetation characteristics were used in robust design occupancy models to test for species-specific relationships between microhabitat vegetation and probability of bird occupancy, colonization, and local extinction. Information gathered by this study will contribute to a broader understanding of retention harvest management in the southeast region, where partial-harvest techniques are a common method of habitat management.

STUDY AREA

My study took place on the Redstone Arsenal (RA; 34.65°N, 86.66°W), a 15,050-ha US Department of Defense landholding in Huntsville, Alabama in Madison County (Figure 2.1). The RA is located on the Eastern Highland Rim of north Alabama and features well-drained clay or loam soils typical of the physiographic region (Smalley 1983). Elevations range from 170-206 m above mean sea level and mean temperatures range from -0.5 °C to 9.4 °C in January and from 20 °C to 31.6 °C in July (Bhuta et al. 2011). Mean annual rainfall is 140 cm with March receiving the most precipitation and October the least.

Prior to RA's acquisition of the land in 1949, the land cover was primarily row crop or pastures and experienced little active forest management until 1970 (Redstone Arsenal 2002). The land has largely been converted to non-forested or urban landscape to achieve RA's goals of supporting the US Army in military research and the development and distribution of weapons systems, services, and supplies (Redstone Arsenal 2002). Forested land cover on RA is highly fragmented due to widespread non-forest and urban patches distributed throughout the landscape (Bhuta et al. 2011). My study area was a 544-ha forested tract located along the southwestern border of RA. In order of total basal area, the pre-harvest study area was comprised of predominantly loblolly pine (*Pinus taeda*), sweetgum (*Liquidambar styraciflua*), willow oak (*Quercus phellos*), yellow poplar (*Liriodendron tulipifera*), and cherrybark oak (*Quercus pagoda*).

My study design consisted of four, 51-ha (SE = 4.51) replicates arranged north to south which each contained four plots in a randomized complete block design (Hurlburt 1984). Within each replicate, three plots received retention harvests and one remained an un-harvested control plot. Although the harvest treatments are currently applied unevenly, this is the beginning of a long-term study to examine the effects of harvest-and-burn methods and seasonality of prescribed burn thus two of the harvested plots will receive growing or dormant season burns in the future. Redstone Arsenal staff determined and marked trees for harvest and contracted a local operator to perform the harvest. One replicate was harvested in March 2011 and two replicates were harvested in March 2012. The fourth, southernmost replicate (hereafter "floodplain") remained un-harvested, but is scheduled for harvest in the future. The floodplain replicate was located nearest to the Tennessee River and flooded seasonally during the winter unlike the other three replicates which were comparatively located at a higher elevation (hereafter "upland"). Retention harvests reduced the basal area of each plot by 43% to 69% to an average of 12.67 m^2/ha (SE = 1.03; Table 2.2), retaining hard and soft mass trees as well as snags to provide nesting and foraging substrates for mature forest birds (Table 2.1). Retention harvests targeted loblolly pine and reduced total basal area of loblolly pine by 64.6% on harvested plots.

METHODS

Avian Surveys

I used fixed-radius point counts as described by Hamel et al. (1996) to sample bird communities during 2011 and 2012. Point counts were conducted at 2 points/plot for a total of 8 points/replicate, and a study total of 32 points. Survey points were spaced >200 m apart and >50 m from the plot edge to ensure spatial independence and to minimize edge effects (Ralph et al. 1993). Point counts were conducted from 8 May to 14 June to survey the breeding bird community and to minimize counts of migrating individuals (Hamel et al. 1996). Point counts were conducted between sunrise and 3 hours after sunrise CST, a time when bird detection rates are most stable (Ralph et al. 1995). Surveys were not conducted during fog, rain, or when winds exceeded Beaufort 3. Each survey point was visited 3 to 4 times each year. Due to RA time restrictions, I was able to complete an average of 8 point counts/day and the length of the survey period was 12 days and 14 days in 2011 and 2012, respectively. Survey order was randomized to begin with a different replicate each round and the direction of travel (north to south or south to north) was randomized for each round as well. Point counts were 9 minutes in duration and consisted of 3 3-minute replicates. Each replicate was considered a "new" point count and these data were used to estimate detection probabilities and site occupancy (MacKenzie et al. 2005). At the start of each replicate, all birds seen or heard were identified and recorded and the locations of observed birds were estimated within three distance bands: 0 m to 25 m, 25 m to 50 m or >50 m from the survey point. I conducted all surveys therefore I did not test for observer bias.

Vegetation Surveys

From July to August 2011 and 2012, I conducted vegetation surveys to measure understory vegetation structure and composition in pre- and post-retention harvest plots. Surveys were conducted at 10 randomly selected points/plot for a total of 40 points/replicate and a study total of 160 points. Vegetation surveys included estimates of overstory canopy cover, understory vertical density, ground cover, and dominant understory species. Random survey points were generated using ArcMap 9.0 (ESRI 2009) under the stipulation that points be >35 m apart and >35 m from the plot edge (Cook et al. 1995). This protocol avoids overlap between points or from neighboring plots in regards to canopy cover estimates (Cook et al. 1995). From each survey point, one 10-m transect was laid out in each cardinal direction. Canopy cover was estimated using a moosehorn (Robinson 1947) to take measurements at each survey point and at 2-m intervals along each 10-m transect (Robbins et al. 1989, Cook et al. 1995, Rodewald and Yahner 2001). Ground cover was estimated using the line-intercept method (Canfield 1941) along each 10-m transect. Ground cover was identified by functional group: water, bare ground, leaf litter, moss, fern, grass, forb, vine, woody plant, or coarse woody debris.

Vertical vegetation structure was estimated using a vegetation density board (Nudds 1977). The board measured 2.44 m tall, 30.48 cm wide and 0.95 cm thick and was vertically divided vertically into 5 equal-sized sections, each painted alternating black or white. Estimation of percentage vegetation covering each block was made from a distance of 10 m from plot center assessed at an eye level of approximately 1 m in height.

Fixed-radius plots (0.04 ha; James and Shugart 1970) were used to measure species composition, trees/hectare, and basal area of both live and standing dead trees (snags). Tree surveys were conducted at 10 randomly selected points/plot for a total of 40 points/replicate and a study total of 160 points. Tree attributes were measured at the same plot location before and after retention harvest. At each tree survey point, all trees and snags >4 cm in diameter at breast height (dbh) were identified by species and measurements of dbh were taken using a dbh tape.

Data Analysis

The original goal of the study was to test 4 treatment types regarding harvest-and-burn techniques but because burn treatments were not implemented, this uneven sampling design was not used to test for these specific treatment effects; rather, I compared only pre- and post-harvest bird response. I used robust design occupancy models to evaluate the effects of microhabitat characteristics on probability of use (occupancy, ψ), local extinction (ε), and local colonization (γ) of bird species while accounting for imperfect detection (p) during point counts (MacKenzie et al. 2003). Robust design occupancy is similar to Pollock’s robust design (Kendall et al. 1997) mark recapture analysis in that there are primary survey periods (seasons) that each consist of several secondary survey periods (replicate visits). Robust design assumes sites are closed to changes in occupancy within each season, but allows occupancy to change across seasons and estimates the between-year dynamics (local extinction and colonization) that influence changes in seasonal occupancy. The parameterization used allowed for direct estimation of pre-harvest occupancy (ψ_1), post-harvest local extinction (ε) and post-harvest local colonization (γ), while first year post-harvest occupancy (ψ_2) was a derived parameter.

I constructed species-specific model sets using Program MARK 5.1 (White and Burnham 1999) for species with >10 encounters/year as this followed a natural break in the data (Appendix A). Model sets were constructed following a hierarchical approach such that the highest ranked model set (delta $AIC_c \leq 2$) for detection probability (p) was considered within the colonization (γ) model set, the highest ranked model set for γ was considered within the extinction (ε) model set, and the highest ranked model set for ε was considered within the occupancy (ψ) model set. Models were assessed using Akaike's information criterion adjusted for small sample size (AIC_c; Burnham and Anderson 2002) and model weights (w_i) were used to estimate the relative strength of support for each model.

Each model set tested all-subsets of habitat covariates expected to influence probability of use, local extinction, and local colonization given the species' natural history (Hamel et al. 1992). I felt an all subsets approach was appropriate given the exploratory nature of this experiment (Arnold 2010), the small number of habitat covariates tested (most sets included 3 covariates, one set included the maximum of 5) and the intent for model averaging which requires all covariates to be equally represented across the model set (Anderson 2008;129-132). Habitat covariates included overstory and understory variables which were measured pre- and post-harvest (Table 3.1). Pre-harvest habitat covariates were incorporated into occupancy models to identify vegetation characteristics that influenced species occupancy within closed-canopy mixed-pine forests before retention harvests were applied. Post-harvest habitat covariates were incorporated into extinction and colonization models to test whether vegetation characteristics following retention harvests affected population dynamics between sampling seasons. The coefficient estimates (βs) estimate the effect size of habitat covariates on the log odds of "success" for occupancy (ψ), local extinction (ε), and local colonization (γ) parameters. Models which had inestimable coefficient estimates were removed from the model set.

Model averaging accounts for model uncertainty across the model set, as opposed to selecting a single "best" model (Burnham and Anderson 2002), therefore in model sets with high uncertainty (top model Akaike weight <0.30) model averaged estimates more accurately capture the variability within the model set. As many species' model sets reported a top model weight of <0.30 (Appendix B), I performed model averaging on real parameters (ψ_1, ε, γ), derived parameters (ψ_2), and betas (βs) to account for high variability across the model set. Models that contained inestimable betas were removed from the model set. Model averaging where betas are unequally represented in the model set results in biased estimates of covariate importance; however model averaged estimates can still be used an unbiased estimate for effect size of covariates on real parameters. I used unconditional standard errors and 85% confidence intervals to assess significance of effect size (Arnold 2010).

One assumption of all occupancy analyses is that there is no un-modeled heterogeneity in detection probabilities, thus survey conditions that may cause variation an observer's ability to detect a bird must be accounted for in detection probability models. Detectability of bird species may be influenced by survey date (Best 1981, Skirvin 1981), time since sunrise (Robbins 1981, Skirvin 1981), and temperature (Mayfield 1981, Robbins 1981) therefore these covariates were incorporated into detection probability (p) estimates to account for variability due to these survey conditions. Models were constructed using a step-down approach (Lebreton et al. 1992) starting with the fully parameterized model [p (Date+Time+Temp)] while holding ε, γ, and ψ constant. Best approximating detection models within ≤2 AIC_c from the top model (Burnham and Anderson 2002) were then incorporated within the subsequent model set for ε, γ, or ψ. This study was not designed to determine important factors which affect detection probabilities, thus detection probabilities were merely a nuisance parameter in which covariates were used to capture the most variation possible.

RESULTS

I completed point counts at each survey point (n = 32) on 3 visits in 2011 (pre-harvest) and 4 visits in 2012 (post-harvest) for a total of 384 site visits. I observed 54 species over both years (Appendix A), 29 of which had >10 encounters each survey year (a natural break in the data) and were included in the analysis.

Several models reported inestimable parameters due to sparse data or to the actual estimate approaching the estimable limits of 0 or 1. Species that were either very rare ($\psi \approx 0$: yellow-billed cuckoos [*Coccyzus americanus*]; Table 3.2) or very abundant ($\psi \approx 1$: Carolina wrens, northern cardinals [*Cardinalis cardinalis*], red-bellied woodpeckers [*Melanerpes carolinus*], tufted titmice; Table 3.2) across all survey points had inestimable occupancy parameter estimates likely because it is difficult to determine the maxima of the likelihood when estimates are close to the boundaries of 0 or 1. Species with inestimable occupancy parameters were excluded from further analyses. Of the remaining species, Acadian flycatchers, blue-gray gnatcatchers (*Polioptila caerulea*), and red-eyed vireos had inestimable extinction parameters and summer tanagers (*Piranga rubra*), eastern towhees, indigo buntings had inestimable colonization parameters likely caused by sparse data or the actual estimate approximating 0 or 1.

Detection Probability (p)

The Null detection model was within the highest ranking model set for 8 species and was the best model for yellow-throated vireos (*Vireo flavifrons*; Table 3.3). The Date model was within the highest ranking model set for 5 species and was best for pine warblers (*Dendroica pinus*). The Date +Temperature model was within the highest ranking model set for Kentucky warbler and white-breasted nuthatch (*Sitta carolinensis*). The Date+Temperature+Time model was within the highest ranking model set for 4 species and was the best model for eastern wood-pewees. The Date+Time model was within the highest ranking model set for 7 species and was the best model for great-crested flycatchers, summer tanagers and yellow-breasted chats (Table 3.3). The Temperature model was within the highest ranking model set for 5 species and was the best model for red-eyed vireos and wood thrushes. The Temperature+Time model was within the highest ranking model set for mourning doves (*Zenaida macroura*) and northern bobwhites (*Colinus virginianus*). The Time model was within the highest ranking model set for 8 species and was the best model for blue jays (*Cyanocitta cristata*) and red-headed woodpeckers (*Melanerpes erythrocephalus*; Table 3.3).

Pre-harvest Occupancy Probability (ψ)

Pre-harvest canopy closure may have little effect on occupancy of mature-interior species (Table 3.4). Occupancy of Acadian flycatchers ($\hat{\bar{\beta}} = 2.85 \pm 2.91$) was strongly positively influenced by greater canopy closure while occupancy of pine warblers ($\hat{\bar{\beta}} = -1.35 \pm 1.68$) was strongly negatively affected by greater canopy closure. Occupancy of most mature-interior species was greater on sites with greater pre-harvest basal area. Pileated woodpeckers ($\hat{\bar{\beta}} = 0.74 \pm 0.75$) exhibited the strongest positive response to basal area although this was not a large effect size. Pine warblers ($\hat{\bar{\beta}} = -0.52 \pm 0.63$) were the only species of this suite to exhibit a slightly negative response to sites with greater basal area (Table 3.4). Pre-harvest occupancy was negatively associated with snag basal area for two mature-interior cavity-nesting species, pileated woodpeckers ($\hat{\hat{\beta}}$ = -0.10 ± 0.25) and white-breasted nuthatches ($\hat{\hat{\beta}}$ = -0.01 ± 0.15); however effect sizes were small and large confidence intervals indicated high variability. Three midstory or understory nesting mature-interior species were tested for an association with understory vegetation density and had varied responses. Occupancy of Acadian flycatchers ($\hat{\hat{\beta}}_{\text{AvgVeg}}$ = -1.97 ± 0.87) was negatively influenced by greater overall understory vegetation density (Table 3.4). Kentucky warbler ($\hat{\hat{\beta}}_{\text{Veg<0.5m}}$ = 0.68 ± 0.52) occupancy was greater on sites with greater ground cover density whereas wood thrush ($\hat{\hat{\beta}}_{\text{Veg>2m}}$ = 0.00 ± 0.13) occupancy was not influenced by midstory vegetation density.

Pre-harvest occupancy of mature-edge forest species was generally not strongly influenced by canopy closure but brown-headed cowbirds ($\hat{\hat{\beta}}$ = 1.59 ± 1.20) and red-eyed vireos ($\hat{\hat{\beta}}$ = 0.87 ± 1.24) were more likely to occur on sites with greater canopy closure while great-crested flycatchers ($\hat{\hat{\beta}}$ = -0.49 ± 0.66) were less likely to occur on those sites (Table 3.4). Mature-edge species exhibited a more varied response to pre-harvest basal area. Consistent with their response to canopy closure, brown-headed cowbirds (*Molothrus ater*; $\hat{\hat{\beta}}$ = 0.82 ± 1.39) and red-eyed vireos ($\hat{\hat{\beta}}$ = 0.60 ± 0.61) were more likely to occupy sites with greater pre-harvest basal area (Table 3.4). Eastern wood-pewees ($\hat{\hat{\beta}}$ = -0.48 ± 0.57) and red-headed woodpeckers ($\hat{\hat{\beta}}_{\text{RHWO}}$ = -0.70 ± 0.78) were less likely to use sites with greater basal area. Only three mature-edge species were tested for an effect of understory vegetation on pre-harvest occupancy, of which American robins (*Turdus migratorius*; $\hat{\hat{\beta}}_{\text{Veg>2m}}$ = -0.35 ± 0.48) were less likely to occupy sites with greater midstory vegetation, summer tanagers ($\hat{\hat{\beta}}_{\text{AvgVeg}}$ = 0.62 ± 1.14) were more likely to occupy sites with greater understory vegetation, and mourning dove ($\hat{\bar{\beta}}_{\text{Veg<0.5m}} = 0.06 \pm 0.14$) occupancy was unaffected by ground cover density (Table 3.4).

Occupancy of early-successional species exhibited the most consistent response to habitat characteristics of any suite (Table 3.4). Pre-harvest occupancy of early-successional species was less likely on sites with greater canopy closure. Overall, occupancy of this suite was not strongly associated with pre-harvest basal area. Occupancy of eastern towhees ($\hat{\bar{\beta}} = -0.39 \pm 0.72$) had the largest effect size for basal area however the effect size was small. Pre-harvest occupancy of early-successional species was positively influenced by greater understory vegetation density for all species except northern bobwhites ($\hat{\bar{\beta}} = -2.13 \pm 0.74$).

Carolina chickadees (*Poecile carolinensis*) were the only estimable generalist species, and they appear to be less likely to occupy sites that featured closed-canopy conditions ($\hat{\bar{\beta}}_{\text{Canopy}} = -0.16 \pm 0.45$; $\hat{\bar{\beta}}_{\text{BA}} = -0.63 \pm 0.70$; Table 3.4).

Local Colonization Probability (γ)

Colonization rates for summer tanagers, eastern towhees, and indigo buntings were at the limits of 0 or 1, which resulted in inestimable standard errors and confidence intervals (Table 3.2). For Kentucky warblers, downy woodpeckers (*Picoides pubescens*), and Carolina chickadees, although the colonization rate was estimable, adding any covariates to the model caused inestimability and thus the null model was used in subsequent occupancy models (Table 3.5). The highest ranked model sets of 13 species contained the null model, and for white-eyed vireos the null model comprised the entire highest ranked model set.

Most mature-interior species appear to have slightly increased colonization rates on sites with greater canopy closure following harvest (Table 3.5). Colonization of pine warblers ($\hat{\bar{\beta}} = -3.55 \pm 10.66$) decreased in response to increased canopy closure although the wide confidence interval suggests high uncertainty. Blue-gray gnatcatchers ($\hat{\bar{\beta}}$ = 0.23 ± 0.80) had the strongest response to canopy closure; however, overall, small coefficient estimates suggest canopy closure may not have a strong influence on colonization of mature-interior species. Similarly, most species of this suite were more likely to colonize sites with increased basal area (Table 3.5). Wood thrushes ($\hat{\bar{\beta}}$ = 1.34 ± 1.69) were more likely to colonize sites featuring greater basal area while blue-gray gnatcatchers ($\hat{\bar{\beta}}_{\mathrm{BGGN}}$ = -0.05 ± 0.17) and pileated woodpeckers ($\hat{\bar{\beta}}$ = -0.03 ± 0.13) were less likely colonize sites with those characteristics. Although pine warblers ($\hat{\bar{\beta}}$ = 0.00 ± 0.39) had no marked response to overall basal area, colonization rates seemed to increase in response to greater pine basal area ($\hat{\bar{\beta}}$ = 0.82 ± 3.04). Two cavity-nesting species, pileated woodpeckers ($\hat{\bar{\beta}}$ = 0.19 ± 0.65) and white-breasted nuthatches ($\hat{\bar{\beta}}$ = -0.19 ± 0.84), were more likely and less likely, respectively, to colonize sites with greater snag basal area. Few mature-interior species were tested for a response to understory habitat variables as most rely on mature forest elements, but Acadian flycatchers ($\hat{\bar{\beta}}$=0.56 ± 1.71) were more likely to colonize sites with greater average density of understory vegetation <2.5m in height whereas wood thrushes were less likely to colonize sites with greater midstory density ($\hat{\bar{\beta}}$ = -0.13 ± 0.61) and leaf litter depth ($\hat{\bar{\beta}}$ = -0.08 ± 0.51; Table 3.5).

Overall, post-harvest colonization probabilities for mature-edge associated species were lower on sites with greater retained structures. As expected for open-canopy associated species, brown-headed cowbirds, eastern wood-pewees, red-eyed vireos, and red-headed woodpeckers were less likely to colonize sites with greater canopy closure and basal area. Red-eyed vireos ($\hat{\bar{\beta}}$ = -1.54 ± 3.08) and red-headed woodpeckers ($\hat{\bar{\beta}}$ = -1.76 ± 2.61) exhibited the strongest response to increased basal area (Table 3.5). American robins, mourning doves, and great-crested flycatchers responded positively to increased canopy closure, but negatively to increased basal area. Blue jays ($\hat{\hat{\beta}}_{\text{BA}}$ = 0.30 ± 1.71; $\hat{\hat{\beta}}_{\text{HW BA}}$ = 0.29 ± 1.37) and yellow-throated vireos ($\hat{\hat{\beta}}_{\text{BA}}$ = 0.27 ± 1.06; $\hat{\hat{\beta}}_{\text{HW BA}}$ = 0.26 ± 0.98) were more likely to occupy sites with greater overall basal area and sites with greater hardwood basal area.

Colonization rates for eastern towhees and indigo buntings were inestimable, likely due to the actual rate approximating zero (Table 3.2). Colonization of the three remaining early-successional species exhibited varied responses to habitat characteristics. Colonization varied little in response to basal area, however sites with greater post-harvest canopy closure had a positive influence on colonization of northern bobwhites ($\hat{\hat{\beta}}$ = 0.64 ± 1.01) and white-eyed vireos ($\hat{\hat{\beta}}$ = 0.23 ± 0.46) but had a negative influence on colonization of yellow-breasted chats ($\hat{\hat{\beta}}$ = -0.48 ± 0.74). Colonization of all early-successional species responded positively to sites with greater density of understory vegetation (Table 3.5).

Local Extinction Probability (ε)

Extinction rates could not be estimated for Acadian flycatchers, blue-gray gnatcatchers, or red-eyed vireos in the null model, likely due to the true ε approximating 0 (Table 3.2). For an additional 6 species (eastern towhees, mourning doves, northern bobwhites, red-headed woodpeckers, summer tanagers, and wood thrushes) although the extinction rate was estimable, adding any covariates to the model caused inestimability and thus the null model was used in subsequent occupancy models (Table 3.6). The highest ranked model sets of 14 species contained the null model, and for 4 of these species the null model was the best model.

Few post-harvest habitat characteristics influenced local extinction of mature-interior species (Table 3.6). Overall, canopy closure and basal area had little effect on post-harvest local extinction of this suite; however, white-breasted nuthatches were less likely to become extinct on sites with greater canopy closure ($\widehat{\overline{\beta}} = -0.23 \pm 0.69$) and basal area ($\widehat{\overline{\beta}}_{\text{BA}} = -0.68 \pm 1.44$). Pileated woodpeckers ($\widehat{\overline{\beta}} = -1.03 \pm 1.74$) were less likely to become locally extinct on sites with greater post-harvest snag basal area; however, local extinction of white-breasted nuthatches ($\widehat{\overline{\beta}} = 0.03 \pm 0.26$) was not strongly affected by greater post-harvest snag basal area. Greater ground cover vegetation density had a slight ameliorating effect on local extinction of Kentucky warblers ($\widehat{\overline{\beta}} = -0.06 \pm 0.66$) however this was a small effect with high uncertainty (Table 3.6).

Of the few mature-edge species for which local extinction rates were estimable, nearly all suggested greater canopy closure increases local extinction probabilities. Canopy closure had the greatest positive effect on local extinction of downy woodpeckers ($\widehat{\overline{\beta}} = 1.97 \pm 2.43$). Conversely, local extinction of yellow-throated vireos ($\widehat{\overline{\beta}} = -0.18 \pm 0.52$) was less likely on sites with greater canopy closure (Table 3.6). Local extinction of most mature-edge species was not strongly influenced by site basal area. Blue jays ($\widehat{\overline{\beta}} = 1.31 \pm 1.95$) were more likely to become locally extinct on sites with greater basal area and had a similar but smaller response to hardwood basal area ($\widehat{\overline{\beta}} = 0.74 \pm 1.33$). American robins ($\widehat{\overline{\beta}}_{\text{Veg >2m}} = -1.21 \pm 1.51$) were the only mature-edge species tested for an effect of vegetation density, and it appears that sites with greater midstory vegetation are less likely to become locally extinct by this species.

Local extinction of early-successional species appeared to be less likely on sites retaining more mature forest structures. Sites with greater post-harvest canopy closure were less likely to become locally extinct by all three early-successional species tested; however, this effect size was small. Similarly, sites with greater post-harvest basal area were less likely to become locally extinct by white-eyed vireos ($\widehat{\overline{\beta}} = -0.18 \pm 0.79$) and yellow-breasted chats ($\widehat{\overline{\beta}} = -0.09 \pm 0.46$), although this effect size was small (Table 3.6). Indigo buntings ($\widehat{\overline{\beta}} = 2.58 \pm 1.46$) showed a strong increased chance of becoming locally extinct on sites with greater post-harvest basal area. Contrary to expectations, local extinction was greater on sites with greater understory vegetation density for white-eyed vireos ($\hat{\bar{\beta}}_{\mathrm{Veg\,<1m}}$ = 1.33 ± 2.54) and had little effect on local extinction of indigo buntings ($\hat{\bar{\beta}}$ = 0.02 ± 0.06). Yellow-breasted chats ($\hat{\bar{\beta}}_{\mathrm{Veg<1m}}$ = -0.94 ± 1.26) were less likely to become locally extinct on sites with greater post-harvest understory vegetation density.

Carolina chickadees were the only estimable generalist species, and their local extinction rates showed little to no response to post-harvest habitat characteristics (Table 3.6).

DISCUSSION

Model selection results were used to assess effect size and significance of the influence of habitat variables on local forest bird populations. I acknowledge the limitations in using complex models with a relatively small effective sample size (n = 64) and as such the estimates are potentially biased. I also acknowledge that the confidence intervals of many coefficient estimates suggest high imprecision and these estimates should be viewed as a suggested relationship which may become clearer with additional study. Regardless, I feel this analysis was a suitable method for exploring the effects of retention harvests on forest bird population dynamics within this experimental system. I believe the results provide initial estimates for effect size and direction of habitat variables considered and suggest continued monitoring or greater sample size will further develop these relationships.

Detection probabilities were most often similar across all surveys (null model). The null model was the best model for only one species (yellow-throated vireos) and was included in the highest ranking model sets of 8 additional species, which suggests detection probabilities were similar across all surveys for these species. For the 15 remaining species, models including survey covariates (survey date, temperature, or time) fit the data significantly better than the null models, evidenced by a difference of >2 ΔAIC, suggesting that variation in observations could be attributed to differences in local weather conditions or timing of surveys. Several other studies have also demonstrated an effect of survey covariates on detection probabilities for birds (Alldredge et al. 2007, Simons et al. 2007). Time appeared most frequently across the highest ranked model sets (21 occurrences), followed by date (18 occurrences), and temperature appeared the least frequently (15 occurrences). Again, this study was not designed to determine important factors which affect detection probabilities, thus detection probabilities were merely a nuisance parameter in which covariates were used to capture the most variation possible.

Mature forest birds were expected to have an overall negative response to harvest evidenced by greater colonization and decreased extinction rates on sites with greater canopy closure and basal area. Almost all effect sizes estimating the influence of pre-harvest habitat characteristics on mature-forest species were near zero. This is likely a result of the overall homogenous nature of the study sites pre-harvest. Following an average 56% reduction in basal area, overall local extinction probabilities were low and colonization rates were comparatively higher. Robinson and Robinson (1999) found mature forest species such as Acadian flycatchers, wood thrushes, and white-breasted nuthatches did not differ in abundance between control sites and selectively logged sites which had 29% to 35% of standing forest removed.
my study removed >20% more basal area than these previous studies and still observed little change in occupancy for these and other mature-interior species.

Few mature-interior species demonstrated strong responses to habitat characteristics. Acadian flycatchers were 7.17 times (2.05 - 25.10; 85% CL) as likely to use a site for each 12% decrease in average vegetation density <2.5 m in height. For this species, understory vegetative density may be an indication of seasonally flooded or riparian habitats which are known to attract Acadian flycatchers (Bakermans and Rodewald 2006) and also inhibit development of understory vegetation. Kentucky warblers had slightly lower occupancy following harvest; however, habitat characteristics did not have a substantial effect on post-harvest population parameters. Several studies have reported increased densities of Kentucky warblers 2 to 3 years post-harvest, likely after understory vegetation regenerates to provide nesting and foraging substrates (Gram et al. 2003, Newell and Rodewald 2012). Mature-interior species which are sensitive to closed-canopy conditions often exhibit the greatest negative response to harvest (Sallabanks and Arnett 2005); however my results suggest a 56% retention harvest may retain enough structure to support nearly all of our tested species in the first year following harvest.

Red-eyed vireos, a mature-edge species, were more likely to use pre-harvest sites with greater canopy closure and basal area; however, they were less likely to colonize sites with similar characteristics post-harvest. This may suggest red-eyed vireos have greater habitat flexibility and may colonize partial harvested sites more readily than mature, closed canopy sites. Contrastingly, Baker and Lacki (1997) reported that on more intensely harvested sites (<6.25 m^2/ha BA) red-eyed vireos were significantly less abundant on harvested sites than on un-harvested controls. However, Otto and Roloff (2012) retained substantially fewer trees than my study and reported occupancy of red-eyed vireos was 130% and 370% higher in stands with 3% to 10% and 25% canopy cover, respectively, than in stands with <3% canopy cover. On my study sites, red-eyed vireo occupancy appeared to increase following less intense harvests that retained an average basal area of 12.67 m^2/ha (SE = 1.03; Table 2.2) and an average canopy closure of 37% (SE = 3%; Table 2.2), which is similar to several studies that reported red-eyed vireos tolerate small levels of disturbance after 25% to 40% removal of original basal area (Annand and Thompson 1997, Wang et al. 2006). However, McDermott and Wood (2005) found no difference in relative abundance of red-eyed vireos across sites 6 to 26 years post-harvest, suggesting differences in occupancy initially following harvest may diminish to zero as stands re-develop.

Population parameters of woodpeckers, nuthatches, and other cavity-nesting species responded positively to more open canopy forest conditions. All woodpecker species analyzed (downy woodpeckers, pileated woodpeckers, and red-headed woodpeckers) may have positive associations with harvest as evidenced by increased post-harvest occupancy, although the increase observed was non-significant. Two woodpecker species in particular had strong responses to post-harvest conditions: red-headed woodpeckers were 5.81 times as likely to colonize a stand for a 9.44 m^2/ha reduction in basal area and downy woodpeckers were 7.17 times as likely to become locally extinct for an increase of 25% in canopy closure. Contrastingly, Tozer et al. (2010) who found abundance of cavity-nesting species had decreased from pre-harvest conditions; however Greenberg et al. (2001) found this suite to respond positively to canopy gaps created by natural, hurricane disturbance.

Secondary cavity-nesting species (nuthatches, titmice) were also expected to have positive associations with mature-forest structures because of greater nesting site availability. Carolina chickadees had an opposite response in that they were more likely to become locally extinct on sites with greater snag basal area. Before harvest, great-crested flycatcher occupancy was positively related to snag basal area and open canopy. Following harvest, great-crested flycatchers appeared to colonize sites with slightly greater canopy and less basal area. White-breasted nuthatches which are often classified as a closed-canopy obligate species (Crawford et al. 1981) were less likely to become locally extinct on sites where more mature forest elements were retained. Post-harvest occupancy was greater (although not significantly greater) than that of pre-harvest, which aligns with Annand and Thompson (1997) who reported white-breasted nuthatches tolerated an even lower stand basal area of 4.9 m^2/ha to 6.9 m^2/ha in the Missouri Ozarks.

Two species which may prey upon or parasitize forest songbirds are often found to have increased occupancy following harvest. Pre-harvest occupancy of blue jays and brown-headed cowbirds was positively related to greater basal area and canopy cover; however, population dynamics responded positively to more open conditions following harvest. Specifically, blue jays were 2.83 times (1.13 - 7.11; 85% CL) as likely to occupy a site for every 18% decrease in canopy closure. Both blue jays and brown-headed cowbirds exhibited slightly increased occupancy following harvest, which supports the findings of previous studies (Baker and Lacki 1997, Robinson and Robinson 1999, Newell and Rodewald 2011). Baker and Lacki found brown-headed cowbirds in greater abundance following more intense harvests which decreased stand basal area to 6.25 m^2/ha, 2.72 m^2/ha, or 0 m^2/ha. Robinson and Robinson (1999) emphasized the importance of landscape configuration and suggested that predation and parasitism may be more prevalent in highly fragmented landscapes, much like the RA landscape where my study took place (Bhuta et al. 2011).

Pre-harvest occupancy of early-successional or shrub-associated species was more likely for sites with open canopies and greater understory density, as hypothesized. Yellow-breasted chats were 3.86 times (1.67 - 8.89; 85% CL) as likely and white-eyed vireos were 4.31 times (1.57 - 11.80; 85% CL) as likely to occupy a site for every 17% increase in vegetation density <1 m in height. After harvest, indigo buntings were 13.20 times (1.61 - 108.03; 85% CL) as likely to become extinct on a site for every 9.44 m^2/ha increase in basal area. Contrastingly, white-eyed vireos and yellow-breasted chats were less likely to become locally extinct on sites with greater basal area and canopy closure. Sites with greater percentage canopy closure were more likely to be colonized by white-eyed vireos but less likely to be colonized by yellow-breasted chats, emphasizing the importance of different levels of mature forest elements incorporated within the habitats of these disturbance-dependent species. Retention cuts in my study which removed ~56% of the standing basal area, in no specific pattern, appear to support yellow-breasted chats known to have patch size requirements (0.7 ha to 3.0 ha; Gram et al. 2003, Alterman et al. 2005).

Unlike other disturbance-dependent gap species, eastern towhees and indigo buntings may have had decreased occupancy following harvest. Tozer et al. (2010) found no change in the abundance of shrub-nesting species immediately post-harvest, but abundance increased in the second year post-harvest. Pre-harvest occupancy of both species responded positively to understory density <1 m or <1.5 m for indigo buntings and eastern towhees, respectively. For indigo buntings, post-harvest extinction probability was significantly increased by basal area and decreased by canopy closure, suggesting the inclusion of a mature tree element for indigo bunting breeding habitat, and I expect occupancy to increase as understory vegetation regenerates over time as other studies have found (Annand and Thompson 1997, Newell and Rodewald 2011).

As a ground-nesting species associated with dense ground cover, northern bobwhites were expected to respond positively to low understory vegetation density. Northern bobwhites were observed in only one replicate of the study regardless of changes in the habitat variables I measured. Pre-harvest occupancy was significantly negatively related to ground vegetation density which may be explained by their presence on sites with the lowest ground vegetation density. Following harvest, northern bobwhites were more likely to colonize sites with greater ground vegetation density perhaps because in the second year, the same replicate had comparatively greater ground vegetation than other sites within the study area. Northern bobwhites are known to select nesting microhabitats which provide dense ground cover, primarily forbs (Taylor et al. 1999, Collins et al. 2009), which may develop on the study sites in future years.

In conclusion, the short-term effects of retention harvests in this study suggest an average 56% reduction in basal area may provide vegetative elements which support a blend of both early-successional and mature-forest species. Although my estimates have large confidence intervals, I believe it is important to note that pre- and post-harvest occupancy estimates were not significantly different yet almost all species exhibited higher occupancy in the first year following harvest. In addition, colonization rates were overall higher than local extinction rates post-harvest. This is contrary to many studies which report low abundances of forest birds in the initial year post-harvest (Beese and Bryant 1999, Robinson and Robinson 1999). The difference in bird response observed in this study may have been the result of the small size of the experimental area and its location within a highly disturbed and developed landscape.

MANAGEMENT IMPLICATIONS

The extant literature on bird response to forest management provides very few direct comparisons of vegetative attributes to bird population measures (Sallabanks and Arnett 2005). Identifying specific relationships between vegetative attributes and bird population dynamics is crucial to understanding forest bird community response to retention harvests. The results of this study provide estimates of the potential size and strength of effect of vegetative characteristics on population parameters; however, these effects had a high level of uncertainty and continued research is necessary to thoroughly evaluate these relationships as they may become more pronounced over time with understory development as evidenced in other studies. Therefore, I recommend using caution when extrapolating these results across regions or habitats not represented in this study. Within the context of this study, I believe a retention harvest of this intensity may be an effective management technique for similar mixed-pine habitats of northern Alabama as it effectively creates habitat for suites of species important to the region. However, before this management technique is put into use, I would encourage additional study over a greater number of study sites and over a longer period of time to capture temporal changes in both the vegetative and avian communities.

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Table 3.1. Descriptions of variables considered for robust design occupancy analysis of forest birds in a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011-2012.

Variable	Description
BA	Total basal area (ft^2/acre)
Canopy	Percentage canopy closure
PineBA	Total basal area (ft^2/acre) of pine species
HW BA	Total basal area (ft^2/acre) of hardwood species
SnagBA	Total basal area (ft^2/acre) of dead trees
AvgVeg	Average percentage vegetation density <2.5 m in height
Veg<0.5m	Average percentage vegetation density <0.5 m in height
Veg<1.0m	Average percentage vegetation density <1.0 m in height
Veg<1.5m	Average percentage vegetation density <1.5 m in height
Veg<2m	Average percentage vegetation density <2.0 m in height
Veg>2m	Average percentage vegetation density >2.0 m in height
Vine	Proportion of cover transect covered by vine
Litter	Depth of leaf litter (in)

Table 3.2. Model averaged estimates of real and derived parameters with 95% confidence intervals for pre-harvest occupancy (ψ_1), post-harvest occupancy (ψ_2), colonization (γ), and local extinction (ε) of forest birds in a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011-2012.

Species	ψ_1	95% CI	ψ_2	95% CI	ε	95% CI	γ	95% CI
Mature-Interior								
ACFL	0.08	(0.01 - 0.53)	0.15	(0.03 - 0.49)	0.00		0.07	(0.01 - 0.38)
BGGN	0.22	(0.10 - 0.44)	0.60	(0.42 - 0.76)	0.00		0.49	(0.29 - 0.69)
KEWA	0.64	(0.45 - 0.79)	0.56	(0.39 - 0.72)	0.42	(0.22 - 0.65)	0.50	(0.24 - 0.76)
PIWA	0.74	(0.39 - 0.93)	0.84	(0.57 - 0.95)	0.19	(0.07 - 0.45)	0.91	(0.05 - 1.00)
PIWO	0.17	(0.06 - 0.39)	0.45	(0.24 - 0.68)	0.39	(0.05 - 0.89)	0.42	(0.20 - 0.67)
WBNU	0.74	(0.43 - 0.91)	0.85	(0.64 - 0.95)	0.06	(0.00 - 0.44)	0.61	(0.22 - 0.90)
WOTH	0.13	(0.04 - 0.33)	0.26	(0.12 - 0.48)	0.25	(0.03 - 0.76)	0.19	(0.05 - 0.47)
[a]YBCU	1.00		0.31	(0.13 - 0.56)	0.69	(0.44 - 0.86)	0.00	
Mature-Edge								
AMRO	0.37	(0.22 - 0.56)	0.26	(0.10 - 0.51)	0.42	(0.10 - 0.83)	0.09	(0.01 - 0.52)
BHCO	0.73	(0.33 - 0.94)	0.85	(0.63 - 0.95)	0.10	(0.02 - 0.41)	0.76	(0.33 - 0.95)
BLJA	0.61	(0.41 - 0.78)	0.94	(0.50 - 1.00)	0.08	(0.00 - 0.59)	0.94	(0.04 - 1.00)
DOWO	0.72	(0.37 - 0.92)	0.86	(0.61 - 0.96)	0.07	(0.01 - 0.47)	0.43	(0.02 - 0.97)
EAWP	0.77	(0.58 - 0.89)	0.73	(0.53 - 0.87)	0.31	(0.15 - 0.53)	0.87	(0.26 - 0.99)
GCFL	0.54	(0.33 - 0.74)	0.90	(0.61 - 0.98)	0.11	(0.02 - 0.41)	0.86	(0.25 - 0.99)
MODO	0.38	(0.23 - 0.56)	0.92	(0.68 - 0.99)	0.08	(0.01 - 0.41)	0.90	(0.51 - 0.99)
[a]RBWO	1.00		1.00		0.00		0.48	(0.00 - 1.00)
REVI	0.27	(0.10 - 0.56)	0.61	(0.34 - 0.83)	0.00		0.47	(0.19 - 0.77)
RHWO	0.21	(0.08 - 0.46)	0.64	(0.37 - 0.84)	0.15	(0.02 - 0.60)	0.60	(0.28 - 0.85)
SUTA	0.92	(0.62 - 0.99)	0.95	(0.76 - 0.99)	0.06	(0.01 - 0.26)	1.00	
YTVI	0.29	(0.10 - 0.59)	0.22	(0.10 - 0.42)	0.54	(0.15 - 0.88)	0.15	(0.04 - 0.40)
Early Succession								
EATO	0.94	(0.64 - 0.99)	0.90	(0.66 - 0.98)	0.04	(0.01 - 0.24)	0.00	
INBU	0.99	(0.96 - 1.03)	0.86	(0.61 - 0.96)	0.14	(0.04 - 0.39)	0.00	
NOBO	0.32	(0.14 - 0.58)	0.30	(0.13 - 0.55)	0.63	(0.33 - 0.86)	0.26	(0.07 - 0.63)
WEVI	0.38	(0.17 - 0.65)	0.63	(0.42 - 0.8)	0.04	(0.00 - 0.51)	0.47	(0.21 - 0.75)
YBCH	0.55	(0.34 - 0.75)	0.68	(0.44 - 0.85)	0.32	(0.11 - 0.64)	0.70	(0.34 - 0.91)
Generalist								
CACH	0.86	(0.64 - 0.96)	0.80	(0.61 - 0.91)	0.21	(0.09 - 0.43)	0.91	(0.14 - 1.00)
[a]CARW	0.93	(0.71 - 0.99)	1.00		0.00		1.00	
[a]NOCA	0.92	(0.71 - 0.98)	1.00		0.00		1.00	
[a]TUTI	1.00		1.00		0.00		0.07	(0.00 - 1.00)

[a] Species which were excluded from results discussion due to inestimable occupancy parameters.

Table 3.3. Model selection results for detection probability (p) of forest birds in a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011-2012. Occupancy (ψ), colonization (γ), and local extinction (ε) were held constant for detection models.

Model	AIC_c[a]	ΔAIC_c	AIC_c Weights	Model Likelihood	K[b]
ACFL					
ψ(.) ε(.) γ(.) p(Time)	332.99	0.00	0.42	1.00	7
ψ(.) ε(.) γ(.) p(.)	333.44	0.45	0.34	0.80	5
AMRO					
ψ(.) ε(.) γ(.) p(Date+Time)	316.55	0.00	0.53	1.00	9
ψ(.) ε(.) γ(.) p(Date)	318.07	1.53	0.25	0.47	7
BGGN					
ψ(.) ε(.) γ(.) p(Date+Time)	392.85	0.00	0.45	1.00	9
ψ(.) ε(.) γ(.) p(Time)	393.46	0.61	0.33	0.74	7
BHCO					
ψ(.) ε(.) γ(.) p(.)	656.32	0.00	0.47	1.00	5
ψ(.) ε(.) γ(.) p(Time)	657.24	0.92	0.29	0.63	7
BLJA					
ψ(.) ε(.) γ(.) p(Time)	686.93	0.00	0.63	1.00	7
CACH					
ψ(.) ε(.) γ(.) p(.)	736.50	0.00	0.49	1.00	5
ψ(.) ε(.) γ(.) p(Temp)	737.65	1.15	0.28	0.56	7
DOWO					
ψ(.) ε(.) γ(.) p(Time)	574.42	0.00	0.48	1.00	7
ψ(.) ε(.) γ(.) p(.)	575.94	1.52	0.22	0.47	5
EATO					
ψ(.) ε(.) γ(.) p(.)	759.80	0.00	0.43	1.00	5
ψ(.) ε(.) γ(.) p(Time)	760.31	0.50	0.34	0.78	7
EAWP					
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	741.24	0.00	0.88	1.00	11
GCFL					
ψ(.) ε(.) γ(.) p(Date+Time)	612.14	0.00	0.80	1.00	9
INBU					
ψ(.) ε(.) γ(.) p(Date)	763.82	0.00	0.43	1.00	7
ψ(.) ε(.) γ(.) p(Temp)	765.50	1.68	0.19	0.43	7
KEWA					
ψ(.) ε(.) γ(.) p(Date+Temp)	622.62	0.00	0.34	1.00	9
ψ(.) ε(.) γ(.) p(Date)	622.72	0.10	0.33	0.95	7
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	624.11	1.48	0.16	0.48	11
ψ(.) ε(.) γ(.) p(Date+Time)	624.18	1.56	0.16	0.46	9

Table 3.3 (continued)

Model	AIC_c[a]	ΔAIC_c	AIC_c Weights	Model Likelihood	K[b]
MODO					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Date+Temp+Time)	645.61	0.00	0.61	1.00	11
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Temp+Time)	646.94	1.33	0.31	0.51	9
NOBO					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Temp+Time)	312.60	0.00	0.65	1.00	9
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Date+Temp+Time)	314.40	1.80	0.26	0.41	11
PIWA					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Date)	654.36	0.00	0.61	1.00	7
PIWO					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(.)	269.40	0.00	0.51	1.00	5
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Temp)	271.15	1.74	0.21	0.42	7
REVI					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Temp)	585.31	0.00	0.81	1.00	7
RHWO					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Time)	480.05	0.00	0.61	1.00	7
SUTA					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Date+Time)	822.27	0.00	0.62	1.00	9
WBNU					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Date)	634.55	0.00	0.39	1.00	7
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(.)	634.85	0.30	0.34	0.86	5
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Date+Temp)	636.52	1.98	0.14	0.37	9
WEVI					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Time)	602.44	0.00	0.36	1.00	7
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(.)	603.93	1.50	0.17	0.47	5
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Date+Time)	604.31	1.87	0.14	0.39	9
WOTH					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Temp)	280.55	0.00	0.67	1.00	7
YBCH					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(Date+Time)	623.68	0.00	0.52	1.00	9
YTVI					
$\psi(.)\ \varepsilon(.)\ \gamma(.)$ p(.)	237.85	0.00	0.48	1.00	5

[a] AICc = Akaike’s Information Criterion adjusted for small sample size

[b] K = Number of parameters

Table 3.4a. Model averaged coefficient estimates ($\hat{\hat{\beta}}$) and unconditional standard errors ($\widehat{var}\ \hat{\beta}$) for overstory habitat variables expected to effect pre-harvest occupancy (ψ) of forest birds prior to a retention harvest of a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011. Models within 2 ΔAIC_c of the top extinction probability model were considered within the ψ model set.

Species	Intercept	Canopy	BA	HW BA	Pine BA	Snag BA
	$\hat{\hat{\beta}}$ ($\widehat{var}\ \hat{\beta}$)					
Mature-Interior						
ACFL	-2.54 (1.46)	2.85 (2.91)	0.42 (0.64)			
BGGN	-1.25 (0.51)	-0.08 (0.22)	0.19 (0.36)			
KEWA	0.57 (0.40)	-0.05 (0.13)	-0.03 (0.12)			
PIWA	1.07 (0.80)	-1.35 (1.68)	-0.52 (0.63)		0.33 (0.46)	
PIWO	-1.58 (0.57)	-0.24 (0.49)	0.74 (0.75)			-0.10 (0.25)
WBNU	1.08 (0.71)	0.19 (0.36)	0.06 (0.21)			-0.01 (0.15)
WOTH	1.93 (0.62)	0.05 (0.25)	0.21 (0.40)			
Mature-Edge						
AMRO	-0.53 (0.39)	-0.02 (0.13)	-0.07 (0.16)			
BHCO	1.14 (1.09)	1.59 (1.20)	0.82 (1.39)			0.14 (0.30)
BLJA	0.43 (0.41)	0.05 (0.14)	-0.05 (0.16)	-1.04 (0.64)		
DOWO	0.96 (0.81)	-0.04 (0.13)	-0.07 (0.23)			0.60 (0.97)
EAWP	1.21 (0.46)	0.03 (0.16)	-0.48 (0.57)			
GCFL	0.28 (0.62)	-0.49 (0.66)	0.12 (0.28)			1.23 (1.11)
MODO	-0.49 (0.37)	-0.10 (0.23)	-0.02 (0.10)		0.00 (0.07)	
REVI	-1.01 (0.64)	0.87 (1.24)	0.60 (0.61)			
RHWO	-1.34 (0.60)	-0.32 (0.49)	-0.70 (0.78)			-0.18 (0.33)
SUTA	2.61 (1.33)	-0.23 (0.55)	-0.18 (0.37)			
YTVI	-0.92 (0.62)	-0.03 (0.08)	-0.01 (0.05)	1.19 (1.38)		

Table 3.4a (continued)

	$\widehat{\overline{\beta}}\ (\widehat{var}\ \widehat{\overline{\beta}})$					
Species	Intercept	Canopy	BA	HW BA	Pine BA	Snag BA
Early Succession						
EATO	3.02 (1.29)	-0.06 (0.16)	-0.39 (0.72)			
INBU	5.62 (3.28)	-0.68 (1.59)	0.01 (0.16)			
NOBO	-0.76 (0.55)	-0.28 (0.44)	0.04 (0.19)			
WEVI	-0.50 (0.57)	-1.10 (0.79)	-0.15 (0.33)			
YBCH	0.22 (0.45)	-0.02 (0.11)	0.04 (0.13)			
Generalist						
CACH	1.82 (0.64)	-0.16 (0.45)	-0.63 (0.70)			0.02 (0.14)

Table 3.4b. Model averaged coefficient estimates ($\hat{\beta}$) and unconditional standard errors ($\widehat{var}\ \hat{\beta}$) for understory habitat variables expected to effect pre-harvest occupancy (ψ) of forest birds prior to a retention harvest of a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011. Models within 2 ΔAIC_c of the top extinction probability model were considered within the ψ model set.

Species	Intercept	Avg Veg	Veg <0.5m	Veg <1m $\hat{\beta}$ ($\widehat{var}\ \hat{\beta}$)	Veg <1.5m	Veg >2m	Vine	Litter
Mature-Interior								
ACFL	-2.54 (1.46)	-1.97 (0.87)						
BGGN	-1.25 (0.51)							
KEWA	0.57 (0.40)		0.68 (0.52)					
PIWA	1.07 (0.80)							
PIWO	-1.58 (0.57)							
WBNU	1.08 (0.71)							
WOTH	1.93 (0.62)					0.00 (0.13)		-0.08 (0.21)
Mature-Edge								
AMRO	-0.53 (0.39)					-0.35 (0.48)		
BHCO	1.14 (1.09)							
BLJA	0.43 (0.41)							
DOWO	0.96 (0.81)							
EAWP	1.21 (0.46)							
GCFL	0.28 (0.62)							
MODO	-0.49 (0.37)		0.06 (0.14)					
REVI	-1.01 (0.64)							
RHWO	-1.34 (0.60)							
SUTA	2.61 (1.33)	0.62 (1.14)						
YTVI	-0.92 (0.62)							

Table 3.4b (continued)

	$\widehat{\bar{\beta}}\ (\widehat{var}\ \widehat{\bar{\beta}})$							
Species	Intercept	Avg Veg	Veg <0.5m	Veg <1m	Veg <1.5m	Veg >2m	Vine	Litter
Early Succession								
EATO	3.02 (1.29)				1.91 (1.35)		1.14 (1.57)	
INBU	5.62 (3.28)			3.21 (1.93)				
NOBO	-0.76 (0.55)		-2.13 (0.74)					
WEVI	-0.50 (0.57)			1.46 (0.70)				
YBCH	0.22 (0.45)			1.35 (0.58)				
Generalist								
CACH	1.82 (0.64)							

Table 3.5. Model averaged coefficient estimates ($\hat{\hat{\beta}}$) and unconditional standard errors ($\widehat{var}\ \hat{\beta}$) for habitat variables expected to effect post-harvest colonization probability (γ) of forest birds in a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2012. Occupancy (ψ) and local extinction (ε) were held constant for γ models. Models within 2 ΔAIC_c of the top detection probability model were considered within the γ model set.

						$\hat{\hat{\beta}}$ ($\widehat{var}\ \hat{\beta}$)					
Species	Intercept	Canopy	BA	HW BA	Pine BA	Snag BA	Avg Veg	Veg <0.5m	Veg <1m	Veg >2m	Litter
Mature-Interior											
ACFL	-2.73 (1.25)	0.01 (0.31)	0.59 (1.02)				0.56 (0.87)				
BGGN	-0.06 (0.44)	0.23 (0.41)	-0.05 (0.17)								
KEWA[a]											
PIWA	2.93 (5.37)	-3.55 (5.44)	0.00 (0.20)		0.82 (1.55)						
PIWO	-0.32 (0.53)	0.08 (0.19)	-0.03 (0.13)			0.19 (0.33)					
WBNU	0.45 (0.87)	0.13 (0.39)	0.19 (0.48)			-0.19 (0.43)					
WOTH	-1.49 (0.70)	0.07 (0.54)	1.34 (0.86)							-0.13 (0.31)	-0.08 (0.26)
Mature-Edge											
AMRO	-2.36 (1.25)	0.07 (0.21)	-0.35 (0.72)							1.56 (1.32)	
BHCO	1.04 (0.86)	-0.18 (0.43)	-0.22 (0.46)								
BLJA	3.44 (5.33)	-4.74 (5.27)	0.30 (0.87)	0.29 (0.70)							
DOWO[a]											
EAWP	2.30 (2.41)	-0.22 (0.85)	-0.80 (1.35)								
GCFL	1.96 (1.70)	8.86 (8.19)	-9.67 (8.49)			0.04 (0.10)					
MODO	2.35 (1.36)	0.02 (0.31)	-1.37 (1.41)		-0.69 (0.97)			0.05 (0.34)			
REVI	-0.11 (0.67)	-0.47 (0.85)	-1.54 (1.57)								
RHWO	0.41 (0.69)	-0.32 (0.67)	-1.76 (1.33)								
SUTA[a]											
YTVI	-1.76 (0.70)	0.14 (0.43)	0.27 (0.54)	0.26 (0.50)							

Table 3.5 (continued)

	$\hat{\beta}\ (\widehat{var}\ \widehat{\beta})$										
Species	Intercept	Canopy	BA	HW BA	Pine BA	Snag BA	Avg Veg	Veg <0.5m	Veg <1m	Veg >2m	Litter
Early Succession											
EATO[a]											
INBU[a]											
NOBO	-1.10 (0.86)	0.64 (1.01)	-0.17 (0.60)					0.85 (1.26)			
WEVI	-0.13 (0.60)	0.23 (0.46)	0.08 0.22)						0.27 (0.57)		
YBCH	0.85 (0.78)	-0.48 (0.74)	-0.13 (0.33)							0.26 (0.47)	
Generalist											
CACH[a]											

[a] Species for which the parameter was inestimable.

Table 3.6. Model averaged coefficient estimates ($\hat{\hat{\beta}}$) and unconditional standard errors ($\widehat{var}\ \hat{\hat{\beta}}$) for habitat variables expected to effect post-harvest extinction probability (ε) of forest birds following a retention harvest of a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2012. Models within 2 ΔAIC_c of the top colonization probability model were considered within the ε model set. Occupancy (ψ) was held constant for ε models.

	$\hat{\hat{\beta}}$ ($\widehat{var}\ \hat{\hat{\beta}}$)								
Species	Intercept	Canopy	BA	HW BA	Pine BA	Snag BA	Veg <0.5m	Veg <1m	Veg >2m
Mature-Interior									
ACFL[a]									
BGGN[a]									
KEWA	-0.34 (0.48)	0.07 (0.22)	0.09 (0.23)				-0.06 (0.22)		
PIWA	-1.43 (0.62)	0.16 (0.37)	-0.10 (0.38)		-0.02 (0.13)				
PIWO	-0.44 (1.28)	-0.12 (0.47)	-0.08 (0.36)			-1.03 (1.74)			
WBNU	-2.55 (1.77)	-0.23 (0.69)	-0.68 (1.44)			0.03 (0.26)			
WOTH[a]									
Mature-Edge									
AMRO	-0.36 (1.01)	0.20 (0.45)	-0.11 (0.41)						-1.21 (1.51)
BHCO	-2.32 (1.11)	0.24 (0.66)	-0.18 (0.51)			0.14 (0.34)			
BLJA	-2.62 (1.74)	0.80 (1.68)	1.31 (1.95)	0.74 (1.33)					
DOWO	-3.04 (1.81)	1.97 (2.43)	0.05 (0.21)			-1.24 (1.82)			
EAWP	-0.14 (1.42)	0.21 (0.40)	0.36 (0.57)						
GCFL	-2.08 (0.87)	0.08 (0.30)	-0.16 (0.45)			0.02 (0.20)			
MODO[a]									
REVI[a]									
RHWO[a]									
SUTA[a]									
YTVI	0.17 (0.95)	-0.18 (0.52)	-0.11 (0.56)	0.44 (0.88)					

Table 3.6 (continued)

	$\hat{\hat{\beta}}\ (\widehat{var}\ \hat{\overline{\beta}})$								
Species	Intercept	Canopy	BA	HW BA	Pine BA	Snag BA	Veg <0.5m	Veg <1m	Veg >2m
Early Succession									
EATO[a]									
INBU	-1.85 (0.72)	-0.88 (1.29)	2.58 (1.46)					0.02 (0.06)	
NOBO[a]									
WEVI	-3.71 (3.81)	-0.29 (0.79)	-0.18 (0.79)					1.33 (2.54)	
YBCH	-0.81 (0.73)	-0.60 (0.92)	-0.09 (0.46)					-0.94 (1.26)	
Generalist									
CACH	-1.32 (0.52)	0.09 (0.21)	-0.04 (0.19)			0.21 (0.37)			

[a] Species for which the parameter was inestimable

Appendix A. List of American Ornithological Union 4-letter species codes, common names, scientific names, and species encounters for birds encountered in a retention-harvested mixed-pine forest at Redstone Arsenal, May-June 2011-2012.

AOU Species Code	Common name	Scientific name	Encounters	
			2011	2012
ACFL	Acadian flycatcher	*Empidonax virescens*	58	83
AMCR	American crow	*Corvus brachyrhynchos*	9	26
AMRE	American redstart	*Setophaga ruticilla*	0	7
AMRO	American robin	*Turdus migratorius*	37	20
BADO	Barred owl	*Strix varia*	0	2
BLGR	Blue grosebeak	*Passerina caerulea*	7	5
BLJA	Blue jay	*Cyanocitta cristata*	69	102
BGGN	Blue-gray gnatcatcher	*Polioptila caerulea*	13	60
BRTH	Brown thrasher	*Toxostoma rufum*	0	9
BHCO	Brown-headed cowbird	*Molothrus ater*	57	76
CACH	Carolina chickadee	*Poecile carolinensis*	81	87
CARW	Carolina wren	*Thryothorus ludovicianus*	136	233
CEWA	Cedar waxwing	*Bombycilla cedrorum*	0	6
CHSP	Chipping sparrow	*Spizella passerina*	0	1
COYE	Common yellow-throat	*Geothlypis trichas*	8	3
COHA	Cooper's hawk	*Accipiter cooperii*	0	1
DOWO	Downy woodpecker	*Picoides pubescens*	27	85
EABL	Eastern bluebird	*Sialia sialis*	0	3
EATO	Eastern towhee	*Pipilo erythrophthalmus*	128	147
EAWP	Eastern wood-pewee	*Contupus virens*	101	126
FISP	Field sparrow	*Spizella pusilla*	0	3
GHOW	Great horned owl	*Bubo virginianus*	0	3
GCFL	Great-crested flycatcher	*Myiarchus crinitus*	43	94
INBU	Indigo bunting	*Passerina cyanea*	165	171
KEWA	Kentucky warbler	*Oporornis formosus*	95	86
MODO	Mourning dove	*Zenaida macroura*	39	167
NOBO	Northern bobwhite	*Colinus virginianus*	28	25
NOCA	Northern cardinal	*Cardinalis cardinalis*	132	221
NOFL	Northern flicker	*Colaptes auratus*	8	4
NOMO	Northern mockingbird	*Mimus polyglottos*	7	4
NOPA	Northern parula	*Parula americana*	0	4
OROR	Orchard oriole	*Icterus spurious*	0	2
PIWO	Pileated woodpecker	*Dryocopus pileatus*	15	23

Appendix A. (continued)

AOU Species Code	Common name	Scientific name	Encounters	
			2011	2012
PIWA	Pine warbler	*Dendroica pinus*	74	87
PRWA	Prairie warbler	*Dendroica discolor*	5	10
PROW	Prothonotary warbler	*Protonotaria citrea*	2	10
RBWO	Red-bellied woodpecker	*Melanerpes carolinus*	121	233
REVI	Red-eyed vireo	*Vireo olivaceus*	58	105
RHWO	Red-headed woodpecker	*Melanerpes erythrocephalus*	19	106
RSHA	Red-shouldered hawk	*Buteo lineatus*	0	3
RTHA	Red-tailed hawk	*Buteo jamaicensis*	6	23
SCTA	Scarlet tanager	*Piranga olivacea*	17	10
SUTA	Summer tanager	*Piranga rubra*	91	116
SWTH	Swainson's thrush	*Catharus ustulatus*	7	0
TUTI	Tufted titmouse	*Baeolophus bicolor*	232	318
VEER	Veery	*Catharus fuscescens*	0	1
WBNU	White-breasted nuthatch	*Sitta carolinensis*	42	84
WEVI	White-eyed vireo	*Vireo griseus*	56	108
WITU	Wild turkey	*Meleagris gallopavo*	0	5
WODU	Wood duck	*Aix sponsa*	2	9
WOTH	Wood thrush	*Hylocichla mustelina*	11	49
YBCU	Yellow-billed cuckoo	*Coccyzus americanus*	63	23
YBCH	Yellow-breasted chat	*Icteria virens*	15	116
YTVI	Yellow-throated vireo	*Vireo flavifrons*	15	21

Appendix B. Model selection tables for pre-harvest bird species occupancy (ψ) prior to retention harvest of a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2011.

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ACFL					
ψ(Canopy+AvgVeg) ε(.) γ(BA) p(.)	318.95	0.00	0.16	1.00	7
ψ(Canopy+AvgVeg) ε(.) γ(BA+AvgVeg) p(Time)	319.96	1.01	0.10	0.60	10
ψ(Canopy+AvgVeg) ε(.) γ(.) p(Time)	320.69	1.74	0.07	0.42	9
ψ(Canopy+AvgVeg) ε(.) γ(.) p(.)	320.77	1.82	0.06	0.40	7
ψ(Canopy+AvgVeg) ε(.) γ(AvgVeg) p(.)	320.92	1.97	0.06	0.37	8
ψ(Canopy+AvgVeg) ε(.) γ(AvgVeg) p(Time)	321.04	2.09	0.06	0.35	10
ψ(Canopy+AvgVeg) ε(.) γ(BA) p(Time)	321.69	2.74	0.04	0.25	10
ψ(BA+AvgVeg) ε(.) γ(.) p(Time)	322.18	3.24	0.03	0.20	9
ψ(BA+Canopy+AvgVeg) ε(.) γ(.) p(.)	322.26	3.31	0.03	0.19	8
ψ(BA+AvgVeg) ε(.) γ(.) p(.)	322.26	3.31	0.03	0.19	7
ψ(BA+Canopy+AvgVeg) ε(.) γ(.) p(Time)	322.38	3.43	0.03	0.18	10
ψ(BA+AvgVeg) ε(.) γ(AvgVeg) p(.)	322.41	3.46	0.03	0.18	8
ψ(BA+Canopy+AvgVeg) ε(.) γ(AvgVeg) p(.)	322.51	3.56	0.03	0.17	9
ψ(BA+AvgVeg) ε(.) γ(AvgVeg) p(Time)	322.53	3.58	0.03	0.17	10
ψ(Canopy+AvgVeg) ε(.) γ(BA+AvgVeg) p(.)	322.55	3.60	0.03	0.17	9
ψ(Canopy+AvgVeg) ε(.) γ(Canopy) p(Time)	322.81	3.86	0.02	0.14	10
ψ(BA+Canopy+AvgVeg) ε(.) γ(AvgVeg) p(Time)	322.84	3.89	0.02	0.14	11
AMRO					
ψ(Veg>2m) ε(Veg>2m) γ(Veg>2m) p(Date+Time)	311.50	0.00	0.10	1.00	12
ψ(.) ε(Veg>2m) γ(Veg>2m) p(Date+Time)	311.58	0.08	0.10	0.96	11
ψ(Veg>2m) ε(.) γ(Veg>2m) p(Date+Time)	311.87	0.37	0.09	0.83	11
ψ(.) ε(.) γ(Veg>2m) p(Date+Time)	312.08	0.58	0.08	0.75	10
ψ(Veg>2m) ε(Veg>2m) γ(Veg>2m) p(Date)	312.17	0.67	0.07	0.71	10
ψ(.) ε(Veg>2m) γ(Veg>2m) p(Date)	312.59	1.09	0.06	0.58	9
ψ(Veg>2m) ε(.) γ(Veg>2m) p(Date)	312.79	1.29	0.05	0.52	9
ψ(.) ε(.) γ(Veg>2m) p(Date)	313.29	1.79	0.04	0.41	8
ψ(BA) ε(Veg>2m) γ(Veg>2m) p(Date+Time)	313.66	2.16	0.04	0.34	12
ψ(Canopy) ε(Veg>2m) γ(Veg>2m) p(Date+Time)	313.73	2.22	0.03	0.33	12
ψ(BA) ε(.) γ(Veg>2m) p(Date+Time)	314.09	2.59	0.03	0.27	11
ψ(Canopy) ε(.) γ(Veg>2m) p(Date+Time)	314.12	2.62	0.03	0.27	11
ψ(BA) ε(Veg>2m) γ(Veg>2m) p(Date)	314.39	2.89	0.02	0.24	10
ψ(Canopy) ε(Veg>2m) γ(Veg>2m) p(Date)	314.45	2.94	0.02	0.23	10
ψ(BA+Veg>2m) ε(Veg>2m) γ(Veg>2m) p(Date+Time)	314.45	2.95	0.02	0.23	13

Appendix B. (continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(Canopy+Veg>2m) ε(Veg>2m) γ(Veg>2m) p(Date+Time)	314.61	3.11	0.02	0.21	13
ψ(BA+Veg>2m) ε(.) γ(Veg>2m) p(Date+Time)	314.71	3.21	0.02	0.20	12
ψ(BA+Veg>2m) ε(Veg>2m) γ(Veg>2m) p(Date)	314.85	3.35	0.02	0.19	11
ψ(Canopy+Veg>2m) ε(.) γ(Veg>2m) p(Date+Time)	314.86	3.35	0.02	0.19	12
ψ(BA) ε(.) γ(Veg>2m) p(Date)	315.03	3.52	0.02	0.17	9
ψ(Canopy+Veg>2m) ε(Veg>2m) γ(Veg>2m) p(Date)	315.06	3.56	0.02	0.17	11
ψ(Canopy) ε(.) γ(Veg>2m) p(Date)	315.06	3.56	0.02	0.17	9
ψ(BA+Veg>2m) ε(.) γ(Veg>2m) p(Date)	315.38	3.87	0.01	0.14	10
BGGN					
ψ(.) ε(.) γ(.) p(Date+Time)	392.85	0.00	0.26	1.00	9
ψ(.) ε(.) γ(.) p(Time)	393.46	0.61	0.19	0.74	7
ψ(BA) ε(.) γ(.) p(Date+Time)	394.73	1.88	0.10	0.39	10
ψ(.) ε(.) γ(Canopy) p(Date+Time)	394.80	1.95	0.10	0.38	10
ψ(BA) ε(.) γ(.) p(Time)	395.18	2.32	0.08	0.31	8
ψ(Canopy) ε(.) γ(.) p(Date+Time)	395.66	2.80	0.06	0.25	10
ψ(Canopy) ε(.) γ(.) p(Time)	396.07	3.21	0.05	0.20	8
ψ(BA+Canopy) ε(.) γ(.) p(Date+Time)	396.50	3.65	0.04	0.16	11
ψ(BA+Canopy) ε(.) γ(.) p(Time)	396.76	3.90	0.04	0.14	9
ψ(BA) ε(.) γ(Canopy) p(Date+Time)	396.80	3.94	0.04	0.14	11
BHCO					
ψ(Canopy) ε(.) γ(.) p(.)	645.17	0.00	0.14	1.00	6
ψ(Canopy) ε(.) γ(.) p(Time)	645.77	0.60	0.11	0.74	8
ψ(Canopy+SnagBA) ε(.) γ(.) p(.)	646.63	1.45	0.07	0.48	7
ψ(Canopy) ε(.) γ(BA) p(.)	646.70	1.53	0.07	0.47	7
ψ(BA) ε(.) γ(.) p(.)	646.73	1.56	0.07	0.46	6
ψ(BA+Canopy) ε(.) γ(.) p(.)	646.78	1.61	0.06	0.45	7
ψ(Canopy) ε(.) γ(Canopy) p(.)	647.01	1.84	0.06	0.40	7
ψ(Canopy) ε(Snag BA) γ(.) p(.)	647.15	1.98	0.05	0.37	7
ψ(Canopy+SnagBA) ε(.) γ(.) p(Time)	647.42	2.25	0.05	0.32	9
ψ(BA+Canopy) ε(.) γ(.) p(Time)	647.60	2.43	0.04	0.30	9
ψ(BA) ε(.) γ(.) p(Time)	647.60	2.43	0.04	0.30	8
ψ(BA) ε(Snag BA) γ(.) p(.)	648.20	3.03	0.03	0.22	7
ψ(Canopy+SnagBA) ε(.) γ(BA) p(.)	648.43	3.26	0.03	0.20	8
ψ(BA+Canopy) ε(.) γ(BA) p(.)	648.54	3.36	0.03	0.19	8
ψ(Canopy+SnagBA) ε(Snag BA) γ(.) p(.)	648.63	3.46	0.03	0.18	8

Appendix B. (continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(Canopy+SnagBA) ε(.) γ(Canopy) p(.)	648.70	3.53	0.02	0.17	8
ψ(BA+Canopy) ε(Snag BA) γ(.) p(.)	648.79	3.62	0.02	0.16	8
ψ(BA) ε(.) γ(Canopy) p(.)	648.81	3.64	0.02	0.16	7
ψ(BA+Canopy) ε(.) γ(Canopy) p(.)	648.81	3.64	0.02	0.16	8
BLJA					
ψ(HW BA) ε(BA) γ(Canopy) p(Time)	673.40	0.00	0.19	1.00	10
ψ(HW BA) ε(Canopy) γ(Canopy) p(Time)	674.31	0.90	0.12	0.64	10
ψ(HW BA) ε(HW BA) γ(Canopy) p(Time)	674.44	1.03	0.11	0.60	10
ψ(HW BA) ε(BA) γ(HW BA+Canopy) p(Time)	675.14	1.74	0.08	0.42	11
ψ(HW BA) ε(BA) γ(BA+Canopy) p(Time)	675.47	2.06	0.07	0.36	11
ψ(HW BA+Canopy) ε(BA) γ(Canopy) p(Time)	676.06	2.65	0.05	0.27	11
ψ(BA+HW BA) ε(BA) γ(Canopy) p(Time)	676.33	2.93	0.04	0.23	11
ψ(HW BA+Canopy) ε(Canopy) γ(Canopy) p(Time)	676.96	3.56	0.03	0.17	11
ψ(HW BA+Canopy) ε(HW BA) γ(Canopy) p(Time)	677.09	3.69	0.03	0.16	11
ψ(BA+HW BA) ε(Canopy) γ(Canopy) p(Time)	677.23	3.83	0.03	0.15	11
ψ(BA+HW BA) ε(HW BA) γ(Canopy) p(Time)	677.36	3.96	0.03	0.14	11
CACH					
ψ(BA) ε(.) γ(.) p(.)	735.08	0.00	0.18	1.00	6
ψ(BA) ε(Snag) γ(.) p(.)	736.29	1.21	0.10	0.55	7
ψ(BA) ε(.) γ(.) p(Temp)	736.38	1.31	0.09	0.52	8
ψ(.) ε(.) γ(.) p(.)	736.50	1.43	0.09	0.49	5
ψ(Canopy) ε(.) γ(.) p(.)	737.35	2.27	0.06	0.32	6
ψ(.) ε(Snag) γ(.) p(.)	737.52	2.44	0.05	0.30	6
ψ(BA+Snag) ε(.) γ(.) p(.)	737.56	2.49	0.05	0.29	7
ψ(BA+Canopy) ε(.) γ(.) p(.)	737.57	2.49	0.05	0.29	7
ψ(.) ε(.) γ(.) p(Temp)	737.65	2.57	0.05	0.28	7
ψ(Canopy) ε(Snag) γ(.) p(.)	738.45	3.37	0.03	0.19	7
ψ(Canopy) ε(.) γ(.) p(Temp)	738.66	3.59	0.03	0.17	8
ψ(BA+Snag) ε(Snag) γ(.) p(.)	738.86	3.79	0.03	0.15	8
ψ(BA+Canopy) ε(Snag) γ(.) p(.)	738.87	3.79	0.03	0.15	8
ψ(Snag) ε(.) γ(.) p(.)	738.91	3.84	0.03	0.15	6
ψ(BA+Snag) ε(.) γ(.) p(Temp)	739.06	3.98	0.02	0.14	9
ψ(BA+Canopy) ε(.) γ(.) p(Temp)	739.06	3.98	0.02	0.14	9

Appendix B. (continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
CARW					
ψ(Veg<2m) ε(.) γ(.) p(Date+Temp)	903.14	0.00	0.33	1.00	10
ψ(.) ε(.) γ(.) p(Date+Temp)	904.79	1.65	0.14	0.44	9
ψ(Veg<2m+CW) ε(.) γ(.) p(Date+Temp)	906.07	2.93	0.08	0.23	11
ψ(.) ε(.) γ(.) p(Temp)	906.15	3.01	0.07	0.22	7
ψ(CW) ε(.) γ(.) p(Date+Temp)	906.25	3.11	0.07	0.21	10
ψ(Canopy) ε(.) γ(.) p(Date+Temp)	906.91	3.77	0.05	0.15	10
DOWO					
ψ(.) ε(Canopy+SnagBA) γ(.) p(Time)	573.45	0.00	0.13	1.00	9
ψ(SnagBA) ε(Canopy+SnagBA) γ(.) p(Time)	573.68	0.23	0.12	0.89	10
ψ(SnagBA) ε(.) γ(.) p(Time)	574.34	0.89	0.08	0.64	8
ψ(.) ε(.) γ(.) p(Time)	574.42	0.97	0.08	0.62	7
ψ(SnagBA) ε(Canopy+SnagBA) γ(.) p(.)	574.54	1.09	0.08	0.58	8
ψ(.) ε(Canopy) γ(.) p(Time)	574.79	1.34	0.07	0.51	8
ψ(.) ε(Canopy+SnagBA) γ(.) p(.)	574.82	1.37	0.07	0.50	7
ψ(SnagBA) ε(Canopy) γ(.) p(Time)	575.31	1.87	0.05	0.39	9
ψ(Canopy) ε(Canopy+SnagBA) γ(.) p(Time)	576.08	2.63	0.04	0.27	10
ψ(BA) ε(Canopy+SnagBA) γ(.) p(Time)	576.17	2.72	0.03	0.26	10
ψ(BA+SnagBA) ε(Canopy+SnagBA) γ(.) p(Time)	576.28	2.83	0.03	0.24	11
ψ(BA+SnagBA) ε(.) γ(.) p(Time)	576.37	2.92	0.03	0.23	9
ψ(Canopy) ε(.) γ(.) p(Time)	576.57	3.13	0.03	0.21	8
ψ(BA+SnagBA) ε(Canopy+SnagBA) γ(.) p(.)	576.80	3.35	0.02	0.19	9
ψ(BA) ε(.) γ(.) p(Time)	576.99	3.54	0.02	0.17	8
ψ(Canopy) ε(Canopy+SnagBA) γ(.) p(.)	577.04	3.59	0.02	0.17	8
ψ(Canopy) ε(Canopy) γ(.) p(Time)	577.24	3.79	0.02	0.15	9
ψ(BA) ε(Canopy+SnagBA) γ(.) p(.)	577.27	3.82	0.02	0.15	8
EATO					
ψ(Veg<1.5+Vine) ε(.) γ(.) p(.)	745.10	0.00	0.22	1.00	7
ψ(BA+Veg<1.5) ε(.) γ(.) p(.)	745.81	0.71	0.15	0.70	7
ψ(Veg<1.5+Vine) ε(.) γ(.) p(Time)	745.97	0.87	0.14	0.65	9
ψ(BA+Veg<1.5) ε(.) γ(.) p(Time)	746.69	1.59	0.10	0.45	9
ψ(Veg<1.5+BA+Vine) ε(.) γ(.) p(.)	747.49	2.39	0.07	0.30	8
ψ(Veg<1.5) ε(.) γ(.) p(.)	747.56	2.47	0.06	0.29	6
ψ(Vine) ε(.) γ(.) p(.)	747.89	2.79	0.05	0.25	6
ψ(Veg<1.5) ε(.) γ(.) p(Time)	748.25	3.15	0.05	0.21	8
ψ(Veg<1.5+BA+Vine) ε(.) γ(.) p(Time)	748.56	3.46	0.04	0.18	10
ψ(Vine) ε(.) γ(.) p(Time)	748.57	3.48	0.04	0.18	8

Appendix B. (continued)

Model	AIC$_c$	ΔAIC$_c$	Model Weights	Model Likelihood	K
EAWP					
ψ(BA) ε(.) γ(.) p(Date+Temp+Time)	740.81	0.00	0.09	1.00	12
ψ(BA) ε(BA) γ(.) p(Date+Temp+Time)	740.96	0.15	0.09	0.93	13
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	741.24	0.43	0.08	0.80	11
ψ(.) ε(BA) γ(.) p(Date+Temp+Time)	741.27	0.47	0.08	0.79	12
ψ(BA) ε(.) γ(BA) p(Date+Temp+Time)	741.42	0.62	0.07	0.74	13
ψ(BA) ε(Canopy) γ(.) p(Date+Temp+Time)	741.44	0.63	0.07	0.73	13
ψ(.) ε(.) γ(BA) p(Date+Temp+Time)	741.69	0.88	0.06	0.64	12
ψ(BA) ε(BA) γ(BA) p(Date+Temp+Time)	741.71	0.91	0.06	0.64	14
ψ(.) ε(Canopy) γ(.) p(Date+Temp+Time)	741.76	0.95	0.06	0.62	12
ψ(.) ε(BA) γ(BA) p(Date+Temp+Time)	741.86	1.06	0.06	0.59	13
ψ(BA) ε(Canopy) γ(BA) p(Date+Temp+Time)	742.19	1.38	0.05	0.50	14
ψ(.) ε(Canopy) γ(BA) p(Date+Temp+Time)	742.34	1.53	0.04	0.47	13
ψ(BA+Canopy) ε(.) γ(.) p(Date+Temp+Time)	743.41	2.61	0.03	0.27	13
ψ(BA+Canopy) ε(BA) γ(.) p(Date+Temp+Time)	743.70	2.89	0.02	0.24	14
ψ(Canopy) ε(.) γ(.) p(Date+Temp+Time)	743.97	3.16	0.02	0.21	12
ψ(Canopy) ε(BA) γ(.) p(Date+Temp+Time)	744.12	3.32	0.02	0.19	13
ψ(BA+Canopy) ε(.) γ(BA) p(Date+Temp+Time)	744.14	3.33	0.02	0.19	14
ψ(BA+Canopy) ε(Canopy) γ(.) p(Date+Temp+Time)	744.18	3.38	0.02	0.18	14
ψ(Canopy) ε(.) γ(BA) p(Date+Temp+Time)	744.55	3.74	0.01	0.15	13
ψ(BA+Canopy) ε(BA) γ(BA) p(Date+Temp+Time)	744.57	3.76	0.01	0.15	15
ψ(Canopy) ε(Canopy) γ(.) p(Date+Temp+Time)	744.61	3.80	0.01	0.15	13
GCFL					
ψ(SnagBA) ε(.) γ(BA) p(Date+Time)	607.76	0.00	0.17	1.00	11
ψ(Canopy+SnagBA) ε(.) γ(BA) p(Date+Time)	607.89	0.12	0.16	0.94	12
ψ(Canopy+SnagBA) ε(.) γ(.) p(Date+Time)	608.85	1.08	0.10	0.58	11
ψ(SnagBA) ε(.) γ(.) p(Date+Time)	608.87	1.10	0.10	0.58	10
ψ(BA+Canopy+SnagBA) ε(.) γ(BA) p(Date+Time)	609.52	1.76	0.07	0.42	13
ψ(SnagBA) ε(.) γ(BA+SnagBA) p(Date+Time)	609.78	2.02	0.06	0.36	12
ψ(Canopy+SnagBA) ε(.) γ(BA+SnagBA) p(Date+Time)	610.13	2.37	0.05	0.31	13
ψ(BA+Canopy+SnagBA) ε(.) γ(.) p(Date+Time)	610.19	2.43	0.05	0.30	12
ψ(BA+SnagBA) ε(.) γ(BA) p(Date+Time)	610.69	2.92	0.04	0.23	12
ψ(.) ε(.) γ(BA) p(Date+Time)	611.09	3.32	0.03	0.19	10
ψ(BA+Canopy+SnagBA) ε(.) γ(BA+SnagBA) p(Date+Time)	611.61	3.84	0.03	0.15	14
ψ(BA+SnagBA) ε(.) γ(.) p(Date+Time)	611.70	3.94	0.02	0.14	11

Appendix B. (continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
INBU					
ψ(Veg<1m) ε(BA+Canopy) γ(.) p(Date)	748.07	0.00	0.22	1.00	10
ψ(Veg<1m) ε(BA) γ(.) p(Date)	748.10	0.03	0.22	0.98	9
ψ(Veg<1m) ε(BA+Canopy) γ(.) p(Temp)	749.74	1.68	0.09	0.43	10
ψ(Veg<1m) ε(BA) γ(.) p(Temp)	749.78	1.71	0.09	0.43	9
ψ(Canopy+Veg<1m) ε(BA) γ(.) p(Date)	750.17	2.11	0.08	0.35	10
ψ(Canopy+Veg<1m) ε(BA+Canopy) γ(.) p(Date)	750.25	2.18	0.07	0.34	11
ψ(BA+Veg<1m) ε(BA) γ(.) p(Date)	750.91	2.84	0.05	0.24	10
ψ(BA+Veg<1m) ε(BA+Canopy) γ(.) p(Date)	750.98	2.92	0.05	0.23	11
ψ(Canopy+Veg<1m) ε(BA) γ(.) p(Temp)	751.85	3.78	0.03	0.15	10
ψ(Canopy+Veg<1m) ε(BA+Canopy) γ(.) p(Temp)	751.92	3.86	0.03	0.15	11
ψ(BA+Veg<1m) ε(BA) γ(.) p(Temp)	752.59	4.52	0.02	0.10	10
ψ(BA+Veg<1m) ε(BA+Canopy) γ(.) p(Temp)	752.66	4.59	0.02	0.10	11
ψ(Canopy) ε(BA+Canopy) γ(.) p(Date)	757.12	9.06	0.00	0.01	10
ψ(Canopy) ε(BA) γ(.) p(Date)	757.16	9.09	0.00	0.01	9
ψ(BA) ε(BA+Canopy) γ(.) p(Date)	757.17	9.10	0.00	0.01	10
ψ(BA) ε(BA) γ(.) p(Date)	757.20	9.13	0.00	0.01	9
ψ(Canopy) ε(BA+Canopy) γ(.) p(Temp)	758.80	10.74	0.00	0.00	10
ψ(Canopy) ε(BA) γ(.) p(Temp)	758.84	10.77	0.00	0.00	9
ψ(BA) ε(BA+Canopy) γ(.) p(Temp)	758.84	10.78	0.00	0.00	10
ψ(BA) ε(BA) γ(.) p(Temp)	758.88	10.81	0.00	0.00	9
ψ(BA+Canopy) ε(BA) γ(.) p(Date)	759.70	11.64	0.00	0.00	10
ψ(BA+Canopy) ε(BA+Canopy) γ(.) p(Date)	759.78	11.71	0.00	0.00	11
ψ(BA+Canopy) ε(BA) γ(.) p(Temp)	761.38	13.31	0.00	0.00	10
ψ(BA+Canopy) ε(BA+Canopy) γ(.) p(Temp)	761.45	13.39	0.00	0.00	11
KEWA					
ψ(Veg<0.5m) ε(.) γ(.) p(Date)	619.99	0.00	0.17	1.00	8
ψ(Veg<0.5m) ε(.) γ(.) p(Date+Temp)	620.10	0.10	0.16	0.95	10
ψ(Veg<0.5m) ε(.) γ(.) p(Date+Time)	621.65	1.66	0.08	0.44	10
ψ(Veg<0.5m) ε(.) γ(.) p(Date+Temp+Time)	621.80	1.81	0.07	0.40	12
ψ(Canopy+Veg<0.5m) ε(.) γ(.) p(Date)	622.41	2.42	0.05	0.30	9
ψ(BA+Veg<0.5m) ε(.) γ(.) p(Date)	622.61	2.62	0.05	0.27	9
ψ(.) ε(.) γ(.) p(Date+Temp)	622.62	2.63	0.05	0.27	9
ψ(.) ε(.) γ(.) p(Date)	622.72	2.73	0.04	0.26	7
ψ(Canopy+Veg<0.5m) ε(.) γ(.) p(Date+Temp)	622.73	2.74	0.04	0.25	11
ψ(BA+Veg<0.5m) ε(.) γ(.) p(Date+Temp)	622.92	2.93	0.04	0.23	11

Appendix B. (continued)

Model	AIC$_c$	ΔAIC$_c$	Model Weights	Model Likelihood	K
MODO					
ψ(.) ε(.) γ(BA+PineBA) p(Date+Temp+Time)	643.07	0.00	0.13	1.00	13
ψ(.) ε(.) γ(BA) p(Date+Temp+Time)	643.47	0.40	0.11	0.82	12
ψ(.) ε(.) γ(BA+PineBA) p(Temp+Time)	643.96	0.89	0.08	0.64	11
ψ(.) ε(.) γ(BA) p(Temp+Time)	644.55	1.48	0.06	0.48	10
ψ(Canopy) ε(.) γ(BA+PineBA) p(Date+Temp+Time)	645.14	2.07	0.05	0.36	14
ψ(Canopy) ε(.) γ(BA) p(Date+Temp+Time)	645.39	2.32	0.04	0.31	13
ψ(Veg<0.5m) ε(.) γ(BA+PineBA) p(Date+Temp+Time)	645.64	2.57	0.04	0.28	14
ψ(Canopy) ε(.) γ(BA+PineBA) p(Temp+Time)	645.79	2.72	0.03	0.26	12
ψ(Veg<0.5m) ε(.) γ(BA) p(Date+Temp+Time)	645.92	2.85	0.03	0.24	13
ψ(BA) ε(.) γ(BA+PineBA) p(Date+Temp+Time)	646.07	3.00	0.03	0.22	14
ψ(Canopy) ε(.) γ(BA) p(Temp+Time)	646.23	3.16	0.03	0.21	11
ψ(Veg<0.5m) ε(.) γ(BA+PineBA) p(Temp+Time)	646.29	3.22	0.03	0.20	12
ψ(PineBA) ε(.) γ(BA+PineBA) p(Date+Temp+Time)	646.31	3.24	0.03	0.20	14
ψ(BA) ε(.) γ(BA) p(Date+Temp+Time)	646.33	3.26	0.03	0.20	13
ψ(PineBA) ε(.) γ(BA) p(Date+Temp+Time)	646.58	3.51	0.02	0.17	13
ψ(BA) ε(.) γ(BA+PineBA) p(Temp+Time)	646.71	3.64	0.02	0.16	12
ψ(Veg<0.5m) ε(.) γ(BA) p(Temp+Time)	646.78	3.70	0.02	0.16	11
ψ(PineBA) ε(.) γ(BA+PineBA) p(Temp+Time)	646.95	3.88	0.02	0.14	12
NOBO					
ψ(Veg<0.5m) ε(.) γ(.) p(Temp+Time)	298.92	0.00	0.18	1.00	10
ψ(Canopy+Veg<0.5m) ε(.) γ(.) p(Temp+Time)	299.97	1.05	0.11	0.59	11
ψ(Veg<0.5m) ε(.) γ(Canopy+Veg<0.5m) p(Temp+Time)	300.04	1.13	0.10	0.57	12
ψ(Veg<0.5m) ε(.) γ(Veg<0.5m) p(Temp+Time)	300.35	1.44	0.09	0.49	11
ψ(Veg<0.5m) ε(.) γ(Canopy) p(Temp+Time)	300.89	1.98	0.07	0.37	11
ψ(Veg<0.5m) ε(.) γ(.) p(Date+Temp+Time)	300.91	1.99	0.07	0.37	12
ψ(Canopy+Veg<0.5m) ε(.) γ(Canopy+Veg<0.5m) p(Temp+Time)	301.26	2.34	0.06	0.31	13
ψ(Canopy+Veg<0.5m) ε(.) γ(Veg<0.5m) p(Temp+Time)	301.51	2.59	0.05	0.27	12
ψ(BA+Veg<0.5m) ε(.) γ(.) p(Temp+Time)	301.64	2.72	0.05	0.26	11
ψ(Canopy+Veg<0.5m) ε(.) γ(Canopy) p(Temp+Time)	302.06	3.14	0.04	0.21	12
ψ(BA+Canopy+Veg<0.5m) ε(.) γ(.) p(Temp+Time)	302.17	3.26	0.04	0.20	12
ψ(Canopy+Veg<0.5m) ε(.) γ(.) p(Date+Temp+Time)	302.21	3.30	0.03	0.19	13
NOCA					
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	902.66	0.00	0.44	1.00	11
ψ(Canopy) ε(.) γ(.) p(Date+Temp+Time)	904.49	1.83	0.18	0.40	12

Appendix B. (continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(Veg>1.5m) ε(.) γ(.) p(Date+Temp+Time)	905.36	2.70	0.12	0.26	12
ψ(BA) ε(.) γ(.) p(Date+Temp+Time)	905.62	2.96	0.10	0.23	12
ψ(BA+Canopy) ε(.) γ(.) p(Date+Temp+Time)	906.06	3.40	0.08	0.18	13
PIWA					
ψ(Canopy) ε(.) γ(Canopy) p(Date)	644.91	0.00	0.20	1.00	9
ψ(BA+PBA) ε(.) γ(Canopy) p(Date)	645.53	0.62	0.14	0.73	10
ψ(Canopy+BA) ε(.) γ(Canopy) p(Date)	646.33	1.42	0.10	0.49	10
ψ(Canopy) ε(.) γ(Canopy+PBA) p(Date)	646.55	1.64	0.09	0.44	10
ψ(Canopy+PBA) ε(.) γ(Canopy) p(Date)	646.60	1.69	0.08	0.43	10
ψ(BA) ε(.) γ(Canopy) p(Date)	646.79	1.88	0.08	0.39	9
ψ(Canopy+BA+PBA) ε(.) γ(Canopy) p(Date)	647.03	2.12	0.07	0.35	11
ψ(BA+PBA) ε(.) γ(Canopy+PBA) p(Date)	647.27	2.36	0.06	0.31	11
ψ(Canopy+BA) ε(.) γ(Canopy+PBA) p(Date)	648.07	3.16	0.04	0.21	11
ψ(Canopy+PBA) ε(.) γ(Canopy+PBA) p(Date)	648.35	3.44	0.04	0.18	11
ψ(BA) ε(.) γ(Canopy+PBA) p(Date)	648.44	3.53	0.03	0.17	10
ψ(Canopy+BA+PBA) ε(.) γ(Canopy+PBA) p(Date)	648.89	3.97	0.03	0.14	12
PIWO					
ψ(BA) ε(SnagBA) γ(.) p(.)	268.33	0.00	0.08	1.00	7
ψ(BA) ε(.) γ(.) p(.)	268.41	0.08	0.08	0.96	6
ψ(.) ε(SnagBA) γ(.) p(.)	269.02	0.69	0.06	0.71	6
ψ(BA+Canopy) ε(.) γ(.) p(.)	269.32	0.99	0.05	0.61	7
ψ(BA+Canopy) ε(SnagBA) γ(.) p(.)	269.36	1.03	0.05	0.60	8
ψ(.) ε(.) γ(.) p(.)	269.40	1.07	0.05	0.59	5
ψ(BA) ε(.) γ(SnagBA) p(.)	269.51	1.18	0.04	0.55	7
ψ(BA) ε(SnagBA) γ(SnagBA) p(.)	269.66	1.33	0.04	0.51	8
ψ(.) ε(SnagBA) γ(SnagBA) p(.)	270.34	2.01	0.03	0.37	7
ψ(BA) ε(SnagBA) γ(Canopy) p(.)	270.39	2.06	0.03	0.36	8
ψ(BA+SnagBA) ε(.) γ(.) p(.)	270.45	2.12	0.03	0.35	7
ψ(BA+SnagBA) ε(SnagBA) γ(.) p(.)	270.47	2.13	0.03	0.34	8
ψ(.) ε(.) γ(SnagBA) p(.)	270.49	2.16	0.03	0.34	6
ψ(BA+Canopy) ε(.) γ(SnagBA) p(.)	270.52	2.19	0.03	0.33	8
ψ(BA) ε(SnagBA) γ(.) p(Temp)	270.54	2.21	0.03	0.33	9
ψ(BA+Canopy) ε(SnagBA) γ(SnagBA) p(.)	270.76	2.43	0.02	0.30	9
ψ(.) ε(SnagBA) γ(.) p(Temp)	270.88	2.55	0.02	0.28	8
ψ(.) ε(SnagBA) γ(Canopy) p(.)	271.00	2.67	0.02	0.26	7
ψ(BA+Canopy+SnagBA) ε(.) γ(.) p(.)	271.20	2.87	0.02	0.24	8

Appendix B. (continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(SnagBA) ε(SnagBA) γ(.) p(.)	271.24	2.91	0.02	0.23	7
ψ(BA+Canopy+SnagBA) ε(SnagBA) γ(.) p(.)	271.34	3.01	0.02	0.22	9
ψ(Canopy) ε(SnagBA) γ(.) p(.)	271.52	3.18	0.02	0.20	7
ψ(BA+Canopy) ε(SnagBA) γ(Canopy) p(.)	271.52	3.19	0.02	0.20	9
ψ(SnagBA) ε(.) γ(.) p(.)	271.55	3.22	0.02	0.20	6
ψ(BA+SnagBA) ε(.) γ(SnagBA) p(.)	271.61	3.27	0.02	0.19	8
ψ(BA+Canopy) ε(SnagBA) γ(.) p(Temp)	271.66	3.33	0.02	0.19	10
ψ(Canopy) ε(.) γ(.) p(.)	271.79	3.46	0.01	0.18	6
ψ(BA+SnagBA) ε(SnagBA) γ(SnagBA) p(.)	271.90	3.57	0.01	0.17	9
RBWO					
ψ(.) ε(.) γ(.) p(Time)	893.17	0.00	0.52	1.00	7
ψ(.) ε(.) γ(.) p(Date+Time)	894.11	0.93	0.32	0.63	9
ψ(.) ε(.) γ(.) p(Temp+Time)	895.88	2.70	0.13	0.26	9
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	899.37	6.20	0.02	0.05	11
REVI					
ψ(BA) ε(.) γ(BA) p(Temp)	576.59	0.00	0.25	1.00	9
ψ(Canopy) ε(.) γ(BA) p(Temp)	577.10	0.51	0.19	0.77	9
ψ(BA+Canopy) ε(.) γ(BA) p(Temp)	577.61	1.02	0.15	0.60	10
ψ(BA) ε(.) γ(Canopy) p(Temp)	577.69	1.10	0.14	0.58	9
ψ(Canopy) ε(.) γ(Canopy) p(Temp)	578.20	1.61	0.11	0.45	9
ψ(BA+Canopy) ε(.) γ(Canopy) p(Temp)	578.72	2.13	0.09	0.35	10
ψ(.) ε(.) γ(BA) p(Temp)	580.36	3.78	0.04	0.15	8
RHWO					
ψ(BA) ε(.) γ(BA) p(Time)	465.80	0.00	0.32	1.00	9
ψ(Canopy) ε(.) γ(BA) p(Time)	466.63	0.83	0.21	0.66	9
ψ(BA+SnagBA) ε(.) γ(BA) p(Time)	467.29	1.48	0.15	0.48	10
ψ(BA+Canopy) ε(.) γ(BA) p(Time)	468.32	2.52	0.09	0.28	10
ψ(Canopy+SnagBA) ε(.) γ(BA) p(Time)	468.38	2.57	0.09	0.28	10
ψ(.) ε(.) γ(BA) p(Time)	469.16	3.36	0.06	0.19	8
SUTA					
ψ(.) ε(.) γ(.) p(Date+Time)	822.27	0.00	0.33	1.00	9
ψ(AvgVeg) ε(.) γ(.) p(Date+Time)	823.35	1.07	0.19	0.59	10
ψ(BA) ε(.) γ(.) p(Date+Time)	823.92	1.65	0.14	0.44	10
ψ(Canopy) ε(.) γ(.) p(Date+Time)	824.16	1.89	0.13	0.39	10
ψ(Canopy+AvgVeg) ε(.) γ(.) p(Date+Time)	825.06	2.78	0.08	0.25	11
ψ(BA+AvgVeg) ε(.) γ(.) p(Date+Time)	825.29	3.01	0.07	0.22	11

Appendix B. (continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
TUTI					
ψ(.) ε(.) γ(.) p(.)	647.17	0.00	0.65	1.00	5
WBNU					
ψ(.) ε(.) γ(.) p(Date)	634.55	0.00	0.07	1.00	7
ψ(.) ε(.) γ(.) p(.)	634.85	0.30	0.06	0.86	5
ψ(.) ε(BA) γ(.) p(Date)	635.18	0.63	0.05	0.73	8
ψ(.) ε(BA) γ(.) p(.)	635.28	0.73	0.05	0.69	6
ψ(.) ε(Canopy) γ(.) p(Date)	635.79	1.24	0.04	0.54	8
ψ(Canopy) ε(.) γ(.) p(Date)	635.90	1.36	0.04	0.51	8
ψ(.) ε(Canopy) γ(.) p(.)	635.92	1.37	0.04	0.50	6
ψ(Canopy) ε(.) γ(.) p(.)	636.00	1.45	0.03	0.48	6
ψ(.) ε(.) γ(BA) p(Date)	636.39	1.84	0.03	0.40	8
ψ(Canopy+SnagBA) ε(Canopy) γ(.) p(.)	636.48	1.93	0.03	0.38	7
ψ(.) ε(.) γ(.) p(Date+Temp)	636.52	1.98	0.03	0.37	9
ψ(Canopy) ε(BA) γ(.) p(.)	636.53	1.99	0.03	0.37	7
ψ(.) ε(.) γ(SnagBA) p(Date)	636.55	2.00	0.03	0.37	8
ψ(Canopy) ε(BA) γ(.) p(Date)	636.65	2.10	0.02	0.35	9
ψ(BA) ε(.) γ(.) p(Date)	636.73	2.18	0.02	0.34	8
ψ(BA) ε(.) γ(.) p(.)	636.79	2.24	0.02	0.33	6
ψ(SnagBA) ε(.) γ(.) p(Date)	637.11	2.56	0.02	0.28	8
ψ(Canopy) ε(Canopy) γ(.) p(.)	637.12	2.57	0.02	0.28	7
ψ(Canopy) ε(Canopy) γ(.) p(Date)	637.19	2.64	0.02	0.27	9
ψ(SnagBA) ε(.) γ(.) p(.)	637.22	2.68	0.02	0.26	6
ψ(BA) ε(BA) γ(.) p(.)	637.25	2.71	0.02	0.26	7
ψ(BA) ε(BA) γ(.) p(Date)	637.40	2.86	0.02	0.24	9
ψ(BA) ε(Canopy) γ(.) p(.)	637.73	3.19	0.01	0.20	7
ψ(SnagBA) ε(BA) γ(.) p(.)	637.75	3.21	0.01	0.20	7
ψ(Canopy) ε(.) γ(BA) p(Date)	637.79	3.25	0.01	0.20	9
ψ(SnagBA) ε(BA) γ(.) p(Date)	637.84	3.29	0.01	0.19	9
ψ(BA) ε(Canopy) γ(.) p(Date)	637.87	3.32	0.01	0.19	9
ψ(Canopy) ε(.) γ(SnagBA) p(Date)	637.98	3.43	0.01	0.18	9
ψ(Canopy) ε(.) γ(.) p(Date+Temp)	638.05	3.50	0.01	0.17	10
ψ(SnagBA) ε(Canopy) γ(.) p(.)	638.41	3.86	0.01	0.15	7
ψ(SnagBA) ε(Canopy) γ(.) p(Date)	638.46	3.91	0.01	0.14	9
ψ(BA+Canopy) ε(.) γ(.) p(.)	638.49	3.94	0.01	0.14	7
ψ(Canopy+SnagBA) ε(.) γ(.) p(.)	638.52	3.97	0.01	0.14	7

Appendix B. (continued)

Model	AIC$_c$	ΔAIC$_c$	Model Weights	Model Likelihood	K
WEVI					
ψ(Canopy+Veg<1m) ε(.) γ(.) p(Time)	592.00	0.00	0.29	1.00	9
ψ(Canopy+Veg<1m) ε(.) γ(.) p(.)	593.44	1.45	0.14	0.49	7
ψ(Canopy+Veg<1m) ε(Veg<1m) γ(.) p(Time)	593.71	1.71	0.12	0.43	10
ψ(Canopy+Veg<1m) ε(.) γ(.) p(Date+Time)	594.28	2.28	0.09	0.32	11
ψ(BA+Canopy+Veg<1m) ε(.) γ(.) p(Time)	594.68	2.68	0.08	0.26	10
ψ(BA+Veg<1m) ε(.) γ(.) p(Time)	595.64	3.64	0.05	0.16	9
ψ(BA+Canopy+Veg<1m) ε(.) γ(.) p(.)	595.98	3.98	0.04	0.14	8
WOTH					
ψ(.) ε(.) γ(BA) p(Temp)	273.96	0.00	0.32	1.00	8
ψ(BA) ε(.) γ(BA) p(Temp)	275.39	1.43	0.16	0.49	9
ψ(Litter) ε(.) γ(BA) p(Temp)	276.07	2.11	0.11	0.35	9
ψ(Canopy) ε(.) γ(BA) p(Temp)	276.37	2.41	0.09	0.30	9
ψ(Veg>2m) ε(.) γ(BA) p(Temp)	276.53	2.57	0.09	0.28	9
YBCH					
ψ(Veg<1m) ε(Canopy+Veg<1m) γ(Canopy) p(Date+Time)	615.90	0.00	0.06	1.00	13
ψ(Veg<1m) ε(.) γ(Canopy) p(Date+Time)	615.92	0.02	0.06	0.99	11
ψ(Veg<1m) ε(Canopy+Veg<1m) γ(.) p(Date+Time)	616.32	0.43	0.05	0.81	12
ψ(Veg<1m) ε(Canopy) γ(Canopy) p(Date+Time)	616.41	0.51	0.05	0.77	12
ψ(Veg<1m) ε(.) γ(.) p(Date+Time)	616.63	0.74	0.04	0.69	10
ψ(Veg<1m) ε(.) γ(Veg<1m) p(Date+Time)	616.68	0.79	0.04	0.67	11
ψ(Veg<1m) ε(Veg<1m) γ(Canopy) p(Date+Time)	616.75	0.86	0.04	0.65	12
ψ(Veg<1m) ε(Canopy+Veg<1m) γ(Veg<1m) p(Date+Time)	616.83	0.93	0.04	0.63	13
ψ(Veg<1m) ε(Canopy) γ(.) p(Date+Time)	617.07	1.18	0.03	0.55	11
ψ(Veg<1m) ε(Canopy) γ(Veg<1m) p(Date+Time)	617.25	1.35	0.03	0.51	12
ψ(Veg<1m) ε(Veg<1m) γ(.) p(Date+Time)	617.29	1.39	0.03	0.50	11
ψ(Veg<1m) ε(.) γ(BA) p(Date+Time)	617.33	1.43	0.03	0.49	11
ψ(Veg<1m) ε(Canopy+Veg<1m) γ(BA) p(Date+Time)	617.33	1.44	0.03	0.49	13
ψ(Veg<1m) ε(Veg<1m) γ(Veg<1m) p(Date+Time)	617.63	1.74	0.03	0.42	12
ψ(Veg<1m) ε(BA+Veg<1m) γ(Canopy) p(Date+Time)	617.66	1.76	0.03	0.41	13
ψ(Veg<1m) ε(BA) γ(Canopy) p(Date+Time)	617.73	1.84	0.03	0.40	12
ψ(Veg<1m) ε(Canopy) γ(BA) p(Date+Time)	617.88	1.98	0.02	0.37	12
ψ(BA+Veg<1m) ε(.) γ(Canopy) p(Date+Time)	618.81	2.91	0.01	0.23	12
ψ(Canopy+Veg<1m) ε(.) γ(Canopy) p(Date+Time)	618.93	3.04	0.01	0.22	12

Appendix B. (continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(BA+Veg<1m) ε(Canopy+Veg<1m) γ(Canopy) p(Date+Time)	619.07	3.17	0.01	0.20	14
ψ(Canopy+Veg<1m) ε(Canopy+Veg<1m) γ(Canopy) p(Date+Time)	619.16	3.26	0.01	0.20	14
ψ(BA+Veg<1m) ε(Canopy+Veg<1m) γ(.) p(Date+Time)	619.40	3.50	0.01	0.17	13
ψ(BA+Veg<1m) ε(.) γ(.) p(Date+Time)	619.42	3.52	0.01	0.17	11
ψ(Canopy+Veg<1m) ε(Canopy+Veg<1m) γ(.) p(Date+Time)	619.42	3.52	0.01	0.17	13
ψ(BA+Veg<1m) ε(Canopy) γ(Canopy) p(Date+Time)	619.42	3.53	0.01	0.17	13
ψ(Canopy+Veg<1m) ε(.) γ(.) p(Date+Time)	619.52	3.63	0.01	0.16	11
ψ(Canopy+Veg<1m) ε(Canopy) γ(Canopy) p(Date+Time)	619.55	3.65	0.01	0.16	13
ψ(BA+Veg<1m) ε(.) γ(Veg<1m) p(Date+Time)	619.56	3.66	0.01	0.16	12
ψ(Canopy+Veg<1m) ε(.) γ(Veg<1m) p(Date+Time)	619.69	3.79	0.01	0.15	12
ψ(BA+Veg<1m) ε(Veg<1m) γ(Canopy) p(Date+Time)	619.78	3.89	0.01	0.14	13
ψ(Canopy+Veg<1m) ε(Veg<1m) γ(Canopy) p(Date+Time)	619.88	3.99	0.01	0.14	13
YBCU					
ψ(.) ε(BA) γ(.) p(Date)	441.53	0.00	0.25	1.00	8
ψ(.) ε(BA+Veg>2m) γ(.) p(Date)	442.17	0.64	0.18	0.73	9
ψ(.) ε(Canopy) γ(.) p(Date)	443.88	2.35	0.08	0.31	8
ψ(.) ε(BA+Canopy) γ(.) p(Date)	444.06	2.53	0.07	0.28	9
ψ(.) ε(BA+AvgVeg) γ(.) p(Date)	444.18	2.65	0.07	0.27	9
ψ(.) ε(.) γ(.) p(Date)	444.51	2.98	0.06	0.23	7
ψ(.) ε(Canopy+Veg>2m) γ(.) p(Date)	444.54	3.02	0.06	0.22	9
ψ(.) ε(Veg>2m) γ(.) p(Date)	444.79	3.27	0.05	0.20	8
ψ(.) ε(BA+Canopy+Veg>2m) γ(.) p(Date)	444.87	3.34	0.05	0.19	10
ψ(.) ε(BA+AvgVeg+Veg>2m) γ(.) p(Date)	444.98	3.45	0.04	0.18	10
YTVI					
ψ(HW BA) ε(.) γ(.) p(.)	234.39	0.00	0.37	1.00	6
ψ(HW BA) ε(.) γ(HW BA) p(.)	237.29	2.90	0.09	0.23	7
ψ(HW BA) ε(.) γ(BA) p(.)	237.35	2.96	0.08	0.23	7
ψ(.) ε(.) γ(.) p(.)	237.85	3.45	0.07	0.18	5
ψ(HW BA) ε(.) γ(Canopy) p(.)	237.97	3.57	0.06	0.17	7
ψ(.) ε(.) γ(HW BA) p(.)	238.00	3.60	0.06	0.17	6
ψ(.) ε(.) γ(BA) p(.)	238.04	3.65	0.06	0.16	6

Appendix C. Model selection tables for post-harvest bird species extinction probabilities (ε) after retention harvest of a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2012.

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ACFL					
Additional parameters caused inestimability					
AMRO					
ψ(.) ε(Veg>2m) γ(Veg>2m) p(Date+Time)	311.58	0.00	0.22	1.00	11
ψ(.) ε(.) γ(Veg>2m) p(Date+Time)	312.08	0.50	0.17	0.78	10
ψ(.) ε(Veg>2m) γ(Veg>2m) p(Date)	312.59	1.01	0.13	0.60	9
ψ(.) ε(.) γ(Veg>2m) p(Date)	313.29	1.72	0.09	0.42	8
ψ(.) ε(Canopy+Veg>2m) γ(Veg>2m) p(Date+Time)	314.20	2.62	0.06	0.27	12
ψ(.) ε(BA+Veg>2m) γ(Veg>2m) p(Date+Time)	314.62	3.04	0.05	0.22	12
ψ(.) ε(Canopy) γ(Veg>2m) p(Date+Time)	314.66	3.08	0.05	0.21	11
ψ(.) ε(BA) γ(Veg>2m) p(Date+Time)	314.83	3.25	0.04	0.20	11
ψ(.) ε(Canopy+Veg>2m) γ(Veg>2m) p(Date)	315.04	3.46	0.04	0.18	10
ψ(.) ε(BA+Veg>2m) γ(Veg>2m) p(Date)	315.40	3.82	0.03	0.15	10
BHCO					
ψ(.) ε(.) γ(.) p(.)	656.32	0.00	0.17	1.00	5
ψ(.) ε(.) γ(.) p(Time)	657.24	0.92	0.11	0.63	7
ψ(.) ε(.) γ(BA) p(.)	657.95	1.63	0.08	0.44	6
ψ(.) ε(SnagBA) γ(.) p(.)	658.20	1.89	0.07	0.39	6
ψ(.) ε(.) γ(Canopy) p(.)	658.22	1.90	0.07	0.39	6
ψ(.) ε(BA) γ(.) p(.)	658.71	2.39	0.05	0.30	6
ψ(.) ε(Canopy) γ(.) p(.)	658.75	2.44	0.05	0.30	6
ψ(.) ε(SnagBA) γ(.) p(Time)	659.26	2.95	0.04	0.23	8
ψ(.) ε(BA) γ(.) p(Time)	659.80	3.49	0.03	0.17	8
ψ(.) ε(Canopy) γ(.) p(Time)	659.85	3.54	0.03	0.17	8
ψ(.) ε(SnagBA) γ(BA) p(.)	659.95	3.63	0.03	0.16	7
ψ(.) ε(BA+Canopy) γ(.) p(.)	660.00	3.68	0.03	0.16	7
ψ(.) ε(SnagBA) γ(Canopy) p(.)	660.21	3.90	0.02	0.14	7
BLJA					
ψ(.) ε(BA) γ(Canopy) p(Time)	677.90	0.00	0.18	1.00	9
ψ(.) ε(Canopy) γ(Canopy) p(Time)	678.77	0.87	0.12	0.65	9
ψ(.) ε(HW BA γ(Canopy) p(Time)	678.94	1.04	0.11	0.59	9
ψ(.) ε(BA) γ(HW BA+Canopy) p(Time)	679.49	1.59	0.08	0.45	10
ψ(.) ε(BA) γ(BA+Canopy) p(Time)	679.83	1.93	0.07	0.38	10
ψ(.) ε(Canopy) γ(HW BA+Canopy) p(Time)	680.42	2.52	0.05	0.28	10

Appendix C. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(.) ε(.) γ(Canopy) p(Time)	680.43	2.54	0.05	0.28	8
ψ(.) ε(HW BA γ(HW BA+Canopy) p(Time)	680.53	2.63	0.05	0.27	10
ψ(.) ε(BA+Canopy) γ(Canopy) p(Time)	680.58	2.68	0.05	0.26	10
ψ(.) ε(Canopy) γ(BA+Canopy) p(Time)	680.74	2.85	0.04	0.24	10
ψ(.) ε(HW BA γ(BA+Canopy) p(Time)	680.87	2.97	0.04	0.23	10
ψ(.) ε(HW BA+Canopy) γ(Canopy) p(Time)	681.20	3.30	0.04	0.19	10
CACH					
ψ(.) ε(.) γ(.) p(.)	736.50	0.00	0.24	1.00	5
ψ(.) ε(SnagBA) γ(.) p(.)	737.52	1.01	0.14	0.60	6
ψ(.) ε(.) γ(.) p(Temp)	737.65	1.15	0.13	0.56	7
ψ(.) ε(Canopy) γ(.) p(.)	738.63	2.12	0.08	0.35	6
ψ(.) ε(SnagBA) γ(.) p(Temp)	738.87	2.37	0.07	0.31	8
ψ(.) ε(BA) γ(.) p(.)	738.94	2.44	0.07	0.30	6
ψ(.) ε(Canopy) γ(.) p(Temp)	739.95	3.45	0.04	0.18	8
ψ(.) ε(Canopy+SnagBA) γ(.) p(.)	740.02	3.52	0.04	0.17	7
ψ(.) ε(BA+SnagBA) γ(.) p(.)	740.04	3.54	0.04	0.17	7
ψ(.) ε(BA) γ(.) p(Temp)	740.27	3.76	0.04	0.15	8
ψ(.) ε(BA+Canopy) γ(.) p(.)	740.45	3.95	0.03	0.14	7
DOWO					
ψ(.) ε(Canopy+SnagBA) γ(.) p(Time)	573.45	0.00	0.24	1.00	9
ψ(.) ε(.) γ(.) p(Time)	574.42	0.97	0.15	0.62	7
ψ(.) ε(Canopy) γ(.) p(Time)	574.79	1.34	0.12	0.51	8
ψ(.) ε(Canopy+SnagBA) γ(.) p(.)	574.82	1.37	0.12	0.50	7
ψ(.) ε(.) γ(.) p(.)	575.94	2.49	0.07	0.29	5
ψ(.) ε(BA) γ(.) p(Time)	576.27	2.82	0.06	0.24	8
ψ(.) ε(Canopy) γ(.) p(.)	576.37	2.92	0.06	0.23	6
ψ(.) ε(SnagBA) γ(.) p(Time)	576.70	3.26	0.05	0.20	8
ψ(.) ε(BA+Canopy) γ(.) p(Time)	577.35	3.90	0.03	0.14	9
EAWP					
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	741.24	0.00	0.19	1.00	11
ψ(.) ε(BA) γ(.) p(Date+Temp+Time)	741.27	0.03	0.19	0.98	12
ψ(.) ε(.) γ(BA) p(Date+Temp+Time)	741.69	0.45	0.15	0.80	12
ψ(.) ε(Canopy) γ(.) p(Date+Temp+Time)	741.76	0.52	0.15	0.77	12
ψ(.) ε(BA) γ(BA) p(Date+Temp+Time)	741.86	0.62	0.14	0.73	13
ψ(.) ε(Canopy) γ(BA) p(Date+Temp+Time)	742.34	1.10	0.11	0.58	13

Appendix C. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(.) ε(BA+Canopy) γ(.) p(Date+Temp+Time)	744.41	3.16	0.04	0.21	13
ψ(.) ε(BA+Canopy) γ(BA) p(Date+Temp+Time)	745.12	3.88	0.03	0.14	14
GCFL					
ψ(.) ε(.) γ(BA) p(Date+Time)	611.09	0.00	0.25	1.00	10
ψ(.) ε(.) γ(.) p(Date+Time)	612.14	1.06	0.15	0.59	9
ψ(.) ε(.) γ(BA+SnagBA) p(Date+Time)	612.67	1.58	0.12	0.45	11
ψ(.) ε(BA) γ(BA) p(Date+Time)	613.81	2.72	0.07	0.26	11
ψ(.) ε(SnagBA γ(BA) p(Date+Time)	614.00	2.92	0.06	0.23	11
ψ(.) ε(Canopy) γ(BA) p(Date+Time)	614.01	2.92	0.06	0.23	11
ψ(.) ε(BA) γ(.) p(Date+Time)	614.80	3.71	0.04	0.16	10
ψ(.) ε(Canopy) γ(.) p(Date+Time)	614.95	3.86	0.04	0.15	10
ψ(.) ε(SnagBA γ(.) p(Date+Time)	614.95	3.86	0.04	0.14	10
INBU					
ψ(.) ε(BA+Canopy) γ(.) p(Date)	755.29	0.00	0.31	1.00	9
ψ(.) ε(BA) γ(.) p(Date)	755.42	0.14	0.29	0.93	8
ψ(.) ε(BA+Canopy) γ(.) p(Temp)	756.96	1.68	0.13	0.43	9
ψ(.) ε(BA) γ(.) p(Temp)	757.10	1.81	0.12	0.40	8
ψ(.) ε(BA+Veg<1m) γ(.) p(Date)	757.95	2.67	0.08	0.26	9
KEWA					
ψ(.) ε(.) γ(Veg<0.5m) p(Date+Temp)	608.84	0.00	0.50	1.00	10
ψ(.) ε(.) γ(Canopy) p(Date+Temp+Time)	610.54	1.70	0.22	0.43	12
ψ(.) ε(.) γ(Veg<0.5m) p(Date+Temp+Time)	610.54	1.70	0.22	0.43	12
MODO					
ψ(.) ε(.) γ(BA+PineBA) p(Date+Temp+Time)	643.07	0.00	0.34	1.00	13
ψ(.) ε(.) γ(BA) p(Date+Temp+Time)	643.47	0.40	0.28	0.82	12
ψ(.) ε(.) γ(BA+PineBA) p(Temp+Time)	643.96	0.89	0.22	0.64	11
ψ(.) ε(.) γ(BA) p(Temp+Time)	644.55	1.48	0.16	0.48	10
PIWA					
ψ(.) ε(.) γ(Canopy) p(Date)	649.75	0.00	0.48	1.00	8
ψ(.) ε(Canopy) γ(Canopy) p(Date)	652.15	2.40	0.14	0.30	9
ψ(.) ε(PBA) γ(Canopy) p(Date)	652.44	2.69	0.12	0.26	9
ψ(.) ε(BA) γ(Canopy) p(Date)	652.46	2.71	0.12	0.26	9

Appendix C. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
PIWO					
$\psi(.)\ \varepsilon(\text{SnagBA})\ \gamma(.)\ p(.)$	269.02	0.00	0.17	1.00	6
$\psi(.)\ \varepsilon(.)\ \gamma(.)\ p(.)$	269.40	0.38	0.14	0.83	5
$\psi(.)\ \varepsilon(\text{SnagBA})\ \gamma(\text{SnagBA})\ p(.)$	270.34	1.31	0.09	0.52	7
$\psi(.)\ \varepsilon(.)\ \gamma(\text{SnagBA})\ p(.)$	270.49	1.47	0.08	0.48	6
$\psi(.)\ \varepsilon(\text{SnagBA})\ \gamma(.)\ p(\text{Temp})$	270.88	1.86	0.07	0.40	8
$\psi(.)\ \varepsilon(\text{SnagBA})\ \gamma(\text{Canopy})\ p(.)$	271.00	1.98	0.06	0.37	7
$\psi(.)\ \varepsilon(.)\ \gamma(.)\ p(\text{Temp})$	271.15	2.12	0.06	0.35	7
$\psi(.)\ \varepsilon(.)\ \gamma(\text{Canopy})\ p(.)$	271.28	2.26	0.05	0.32	6
$\psi(.)\ \varepsilon(\text{Canopy})\ \gamma(.)\ p(.)$	271.34	2.32	0.05	0.31	6
$\psi(.)\ \varepsilon(\text{BA})\ \gamma(.)\ p(.)$	271.36	2.33	0.05	0.31	6
$\psi(.)\ \varepsilon(\text{Canopy})\ \gamma(\text{SnagBA})\ p(.)$	272.53	3.50	0.03	0.17	7
$\psi(.)\ \varepsilon(\text{BA})\ \gamma(\text{SnagBA})\ p(.)$	272.53	3.51	0.03	0.17	7
RHWO					
$\psi(.)\ \varepsilon(.)\ \gamma(\text{BA})\ p(\text{Time})$	469.16	0.00	1.00	1.00	8
SUTA					
$\psi(.)\ \varepsilon(.)\ \gamma(.)\ p(\text{Date+Time})$	822.27	0.00	0.42	1.00	9
$\psi(.)\ \varepsilon(\text{AvgVeg})\ \gamma(.)\ p(\text{Date+Time})$	824.16	1.89	0.16	0.39	10
$\psi(.)\ \varepsilon(\text{Canopy})\ \gamma(.)\ p(\text{Date+Time})$	824.76	2.49	0.12	0.29	10
$\psi(.)\ \varepsilon(\text{BA})\ \gamma(.)\ p(\text{Date+Time})$	825.08	2.81	0.10	0.25	10
$\psi(.)\ \varepsilon(\text{BA+AvgVeg})\ \gamma(.)\ p(\text{Date+Time})$	825.09	2.82	0.10	0.24	11
$\psi(.)\ \varepsilon(\text{BA+Canopy})\ \gamma(.)\ p(\text{Date+Time})$	826.11	3.83	0.06	0.15	11
WBNU					
$\psi(.)\ \varepsilon(.)\ \gamma(.)\ p(\text{Date})$	634.55	0.00	0.11	1.00	7
$\psi(.)\ \varepsilon(.)\ \gamma(.)\ p(.)$	634.85	0.30	0.09	0.86	5
$\psi(.)\ \varepsilon(\text{BA})\ \gamma(.)\ p(\text{Date})$	635.18	0.63	0.08	0.73	8
$\psi(.)\ \varepsilon(\text{BA})\ \gamma(.)\ p(.)$	635.28	0.73	0.07	0.69	6
$\psi(.)\ \varepsilon(\text{Canopy})\ \gamma(.)\ p(\text{Date})$	635.79	1.24	0.06	0.54	8
$\psi(.)\ \varepsilon(\text{Canopy})\ \gamma(.)\ p(.)$	635.92	1.37	0.05	0.50	6
$\psi(.)\ \varepsilon(.)\ \gamma(\text{BA})\ p(\text{Date})$	636.39	1.84	0.04	0.40	8
$\psi(.)\ \varepsilon(.)\ \gamma(.)\ p(\text{Date+Temp})$	636.52	1.98	0.04	0.37	9
$\psi(.)\ \varepsilon(.)\ \gamma(\text{SnagBA})\ p(\text{Date})$	636.55	2.00	0.04	0.37	8
$\psi(.)\ \varepsilon(\text{SnagBA})\ \gamma(.)\ p(\text{Date})$	637.13	2.58	0.03	0.28	8
$\psi(.)\ \varepsilon(\text{BA})\ \gamma(\text{BA})\ p(\text{Date})$	637.22	2.68	0.03	0.26	9
$\psi(.)\ \varepsilon(\text{SnagBA})\ \gamma(.)\ p(.)$	637.23	2.68	0.03	0.26	6
$\psi(.)\ \varepsilon(\text{BA})\ \gamma(\text{SnagBA})\ p(\text{Date})$	637.25	2.70	0.03	0.26	9

Appendix C. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(.) ε(BA) γ(.) p(Date+Temp)	637.28	2.73	0.03	0.26	10
ψ(.) ε(BA+SnagBA) γ(.) p(.)	637.65	3.10	0.02	0.21	7
ψ(.) ε(BA+SnagBA) γ(.) p(Date)	637.76	3.21	0.02	0.20	9
ψ(.) ε(BA+Canopy) γ(.) p(.)	637.79	3.25	0.02	0.20	7
ψ(.) ε(Canopy) γ(SnagBA) p(Date)	637.87	3.32	0.02	0.19	9
ψ(.) ε(Canopy) γ(BA) p(Date)	637.87	3.32	0.02	0.19	9
ψ(.) ε(BA+Canopy) γ(.) p(Date)	637.88	3.34	0.02	0.19	9
ψ(.) ε(Canopy) γ(.) p(Date+Temp)	637.89	3.34	0.02	0.19	10
ψ(.) ε(Canopy+SnagBA) γ(.) p(.)	638.24	3.70	0.02	0.16	7
ψ(.) ε(Canopy+SnagBA) γ(.) p(Date)	638.32	3.77	0.02	0.15	9
WEVI					
ψ(.) ε(.) γ(.) p(Time)	602.44	0.00	0.20	1.00	7
ψ(.) ε(Veg<1m) γ(.) p(Time)	603.90	1.46	0.10	0.48	8
ψ(.) ε(.) γ(.) p(.)	603.93	1.50	0.09	0.47	5
ψ(.) ε(.) γ(.) p(Date+Time)	604.31	1.87	0.08	0.39	9
ψ(.) ε(Canopy) γ(.) p(Time)	604.95	2.51	0.06	0.29	8
ψ(.) ε(BA) γ(.) p(Time)	605.02	2.59	0.06	0.27	8
ψ(.) ε(Veg<1m) γ(.) p(.)	605.19	2.76	0.05	0.25	6
ψ(.) ε(Canopy+Veg<1m) γ(.) p(Time)	605.37	2.93	0.05	0.23	9
ψ(.) ε(Veg<1m) γ(.) p(Date+Time)	605.98	3.54	0.03	0.17	10
ψ(.) ε(BA+Veg<1m) γ(.) p(Time)	606.13	3.69	0.03	0.16	9
ψ(.) ε(BA+Canopy+Veg<1m) γ(.) p(Time)	606.22	3.78	0.03	0.15	10
ψ(.) ε(Canopy) γ(.) p(.)	606.26	3.82	0.03	0.15	6
ψ(.) ε(BA) γ(.) p(.)	606.34	3.90	0.03	0.14	6
YBCH					
ψ(.) ε(Canopy+Veg<1m) γ(Canopy) p(Date+Time)	622.87	0.00	0.07	1.00	12
ψ(.) ε(.) γ(Canopy) p(Date+Time)	622.90	0.03	0.07	0.99	10
ψ(.) ε(Canopy+Veg<1m) γ(.) p(Date+Time)	623.34	0.48	0.06	0.79	11
ψ(.) ε(Canopy) γ(Canopy) p(Date+Time)	623.39	0.53	0.06	0.77	11
ψ(.) ε(.) γ(Veg<1m) p(Date+Time)	623.47	0.61	0.05	0.74	10
ψ(.) ε(Canopy+Veg<1m) γ(Veg<1m) p(Date+Time)	623.51	0.65	0.05	0.72	12
ψ(.) ε(.) γ(.) p(Date+Time)	623.68	0.81	0.05	0.67	9
ψ(.) ε(Veg<1m) γ(Canopy) p(Date+Time)	623.78	0.91	0.05	0.63	11
ψ(.) ε(Canopy) γ(Veg<1m) p(Date+Time)	623.99	1.12	0.04	0.57	11
ψ(.) ε(Canopy) γ(.) p(Date+Time)	624.12	1.25	0.04	0.54	10
ψ(.) ε(Canopy+Veg<1m) γ(BA) p(Date+Time)	624.13	1.27	0.04	0.53	12
ψ(.) ε(.) γ(BA) p(Date+Time)	624.20	1.33	0.04	0.51	10

Appendix C. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(.) ε(Veg<1m) γ(.) p(Date+Time)	624.35	1.48	0.03	0.48	10
ψ(.) ε(Veg<1m) γ(Veg<1m) p(Date+Time)	624.40	1.53	0.03	0.47	11
ψ(.) ε(BA+Veg<1m) γ(Canopy) p(Date+Time)	624.61	1.75	0.03	0.42	12
ψ(.) ε(BA) γ(Canopy) p(Date+Time)	624.65	1.78	0.03	0.41	11
ψ(.) ε(Canopy) γ(BA) p(Date+Time)	624.72	1.85	0.03	0.40	11
ψ(.) ε(Veg<1m) γ(BA) p(Date+Time)	625.04	2.17	0.02	0.34	11
ψ(.) ε(BA+Veg<1m) γ(.) p(Date+Time)	625.09	2.22	0.02	0.33	11
ψ(.) ε(BA) γ(Veg<1m) p(Date+Time)	625.23	2.37	0.02	0.31	11
ψ(.) ε(BA+Veg<1m) γ(Veg<1m) p(Date+Time)	625.25	2.39	0.02	0.30	12
ψ(.) ε(BA) γ(.) p(Date+Time)	625.35	2.48	0.02	0.29	10
ψ(.) ε(BA+Canopy+Veg<1m) γ(Canopy) p(Date+Time)	625.82	2.95	0.02	0.23	13
ψ(.) ε(BA+Veg<1m) γ(BA) p(Date+Time)	625.87	3.00	0.02	0.22	12
ψ(.) ε(BA) γ(BA) p(Date+Time)	625.95	3.09	0.02	0.21	11
ψ(.) ε(BA+Canopy) γ(Canopy) p(Date+Time)	626.08	3.21	0.01	0.20	12
ψ(.) ε(BA+Canopy+Veg<1m) γ(.) p(Date+Time)	626.17	3.31	0.01	0.19	12
ψ(.) ε(BA+Canopy+Veg<1m) γ(Veg<1m) p(Date+Time)	626.46	3.59	0.01	0.17	13
ψ(.) ε(BA+Canopy) γ(Veg<1m) p(Date+Time)	626.67	3.80	0.01	0.15	12
ψ(.) ε(BA+Canopy) γ(.) p(Date+Time)	626.69	3.82	0.01	0.15	11
YBCU					
ψ(.) ε(BA) γ(.) p(Date)	441.53	0.00	0.25	1.00	8
ψ(.) ε(BA+Veg>2m) γ(.) p(Date)	442.17	0.64	0.18	0.73	9
ψ(.) ε(Canopy) γ(.) p(Date)	443.88	2.35	0.08	0.31	8
ψ(.) ε(BA+Canopy) γ(.) p(Date)	444.06	2.53	0.07	0.28	9
ψ(.) ε(BA+AvgVeg) γ(.) p(Date)	444.18	2.65	0.07	0.27	9
ψ(.) ε(.) γ(.) p(Date)	444.51	2.98	0.06	0.23	7
ψ(.) ε(Canopy+Veg>2m) γ(.) p(Date)	444.54	3.02	0.06	0.22	9
ψ(.) ε(Veg>2m) γ(.) p(Date)	444.79	3.27	0.05	0.20	8
ψ(.) ε(BA+Canopy+Veg>2m) γ(.) p(Date)	444.87	3.34	0.05	0.19	10
ψ(.) ε(BA+AvgVeg+Veg>2m) γ(.) p(Date)	444.98	3.45	0.04	0.18	10
YTVI					
ψ(.) ε(.) γ(.) p(.)	237.85	0.00	0.11	1.00	5
ψ(.) ε(.) γ(HW BA) p(.)	238.00	0.15	0.10	0.93	6
ψ(.) ε(.) γ(BA) p(.)	238.04	0.19	0.10	0.91	6
ψ(.) ε(.) γ(Canopy) p(.)	238.66	0.81	0.07	0.67	6
ψ(.) ε(HW BA) γ(HW BA) p(.)	239.85	2.00	0.04	0.37	7
ψ(.) ε(HW BA) γ(.) p(.)	239.89	2.04	0.04	0.36	6
ψ(.) ε(HW BA) γ(BA) p(.)	239.91	2.06	0.04	0.36	7

Appendix C. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(.) ε(BA) γ(.) p(.)	240.21	2.36	0.03	0.31	6
ψ(.) ε(Canopy) γ(.) p(.)	240.24	2.39	0.03	0.30	6
ψ(.) ε(BA) γ(HW BA) p(.)	240.30	2.45	0.03	0.29	7
ψ(.) ε(BA) γ(BA) p(.)	240.32	2.48	0.03	0.29	7
ψ(.) ε(Canopy) γ(HW BA) p(.)	240.52	2.67	0.03	0.26	7
ψ(.) ε(Canopy) γ(BA) p(.)	240.57	2.72	0.03	0.26	7
ψ(.) ε(HW BA) γ(Canopy) p(.)	240.64	2.79	0.03	0.25	7
ψ(.) ε(BA) γ(Canopy) p(.)	241.02	3.17	0.02	0.21	7
ψ(.) ε(Canopy) γ(Canopy) p(.)	241.18	3.33	0.02	0.19	7
ψ(.) ε(Canopy+HW BA) γ(.) p(.)	241.27	3.42	0.02	0.18	7
ψ(.) ε(Canopy+HW BA) γ(HW BA) p(.)	241.39	3.54	0.02	0.17	8
ψ(.) ε(BA+HW BA) γ() p(.)	241.41	3.56	0.02	0.17	7
ψ(.) ε(Canopy+HW BA) γ(BA) p(.)	241.54	3.69	0.02	0.16	8
ψ(.) ε(BA+HW BA) γ(HW BA) p(.)	241.55	3.70	0.02	0.16	8
ψ(.) ε(BA+Canopy+HW BA) γ(.) p(.)	241.59	3.74	0.02	0.15	8
ψ(.) ε(BA+Canopy+HW BA) γ(HW BA) p(.)	241.74	3.89	0.02	0.14	9
ψ(.) ε(BA+HW BA) γ(BA) p(.)	241.75	3.90	0.02	0.14	8

Appendix D. Model selection tables for post-harvest bird species colonization probabilities (γ) after retention harvest of a mixed-pine forest at Redstone Arsenal, Huntsville, Alabama in 2012.

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ACFL					
ψ(.) ε(.) γ(.) p(Time)	332.99	0.00	0.14	1.00	7
ψ(.) ε(.) γ(AvgVeg) p(Time)	333.14	0.15	0.13	0.93	8
ψ(.) ε(.) γ(AvgVeg) p(.)	333.41	0.41	0.11	0.81	6
ψ(.) ε(.) γ(.) p(.)	333.44	0.45	0.11	0.80	5
ψ(.) ε(.) γ(BA) p(Time)	333.79	0.80	0.09	0.67	8
ψ(.) ε(.) γ(BA) p(.)	334.06	1.06	0.08	0.59	6
ψ(.) ε(.) γ(BA+AvgVeg) p(Time)	334.78	1.78	0.06	0.41	9
ψ(.) ε(.) γ(BA+AvgVeg) p(.)	334.85	1.86	0.05	0.39	7
ψ(.) ε(.) γ(Canopy) p(Time)	334.92	1.92	0.05	0.38	8
ψ(.) ε(.) γ(Canopy) p(.)	335.18	2.19	0.05	0.33	6
ψ(.) ε(.) γ(Canopy+AvgVeg) p(Time)	335.80	2.81	0.03	0.25	9
ψ(.) ε(.) γ(Canopy+AvgVeg) p(.)	335.88	2.88	0.03	0.24	7
ψ(.) ε(.) γ(BA+Canopy) p(Time)	335.99	2.99	0.03	0.22	9
ψ(.) ε(.) γ(BA+Canopy) p(.)	336.06	3.07	0.03	0.22	7
AMRO					
ψ(.) ε(.) γ(Veg>2m) p(Date+Time)	312.08	0.00	0.40	1.00	10
ψ(.) ε(.) γ(Veg>2m) p(Date)	313.29	1.21	0.22	0.55	8
ψ(.) ε(.) γ(BA+Veg>2m) p(Date+Time)	314.81	2.73	0.10	0.26	11
ψ(.) ε(.) γ(BA+Veg>2m) p(Date)	315.81	3.73	0.06	0.16	9
BGGN					
ψ(.) ε(.) γ(.) p(Date+Time)	392.85	0.00	0.32	1.00	9
ψ(.) ε(.) γ(.) p(Time)	393.46	0.61	0.23	0.74	7
ψ(.) ε(.) γ(Canopy) p(Date+Time)	394.80	1.95	0.12	0.38	10
ψ(.) ε(.) γ(Canopy) p(Time)	395.17	2.31	0.10	0.31	8
ψ(.) ε(.) γ(BA) p(Date+Time)	395.54	2.69	0.08	0.26	10
ψ(.) ε(.) γ(BA) p(Time)	395.94	3.09	0.07	0.21	8
BHCO					
ψ(.) ε(.) γ(.) p(.)	656.32	0.00	0.26	1.00	5
ψ(.) ε(.) γ(.) p(Time)	657.24	0.92	0.17	0.63	7
ψ(.) ε(.) γ(BA) p(.)	657.95	1.63	0.12	0.44	6
ψ(.) ε(.) γ(Canopy) p(.)	658.22	1.90	0.10	0.39	6
ψ(.) ε(.) γ(SnagBA) p(.)	658.72	2.40	0.08	0.30	6
ψ(.) ε(.) γ(BA) p(Time)	658.88	2.57	0.07	0.28	8

Appendix D. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(.) ε(.) γ(Canopy) p(Time)	659.18	2.87	0.06	0.24	8
ψ(.) ε(.) γ(SnagBA) p(Time)	659.77	3.46	0.05	0.18	8
BLJA					
ψ(.) ε(.) γ(Canopy) p(Time)	680.43	0.00	0.50	1.00	8
ψ(.) ε(.) γ(HW BA+Canopy) p(Time)	682.01	1.57	0.23	0.46	9
ψ(.) ε(.) γ(BA+Canopy) p(Time)	682.33	1.89	0.19	0.39	9
CACH					
ψ(.) ε(.) γ(.) p(.)	736.50	0.00	0.64	1.00	5
ψ(.) ε(.) γ(.) p(Temp)	737.65	1.15	0.36	0.56	7
DOWO					
ψ(.) ε(.) γ(.) p(Time)	574.42	0.00	0.68	1.00	7
ψ(.) ε(.) γ(.) p(.)	575.94	1.52	0.32	0.47	5
EAWP					
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	741.24	0.00	0.43	1.00	11
ψ(.) ε(.) γ(BA) p(Date+Temp+Time)	741.69	0.45	0.34	0.80	12
ψ(.) ε(.) γ(Canopy) p(Date+Temp+Time)	743.31	2.07	0.15	0.36	12
ψ(.) ε(.) γ(BA+Canopy) p(Date+Temp+Time)	744.85	3.61	0.07	0.16	13
GCFL					
ψ(.) ε(.) γ(BA) p(Date+Time)	611.09	0.00	0.37	1.00	10
ψ(.) ε(.) γ(.) p(Date+Time)	612.14	1.06	0.22	0.59	9
ψ(.) ε(.) γ(BA+SnagBA p(Date+Time)	612.67	1.58	0.17	0.45	11
ψ(.) ε(.) γ(SnagBA p(Date+Time)	613.37	2.28	0.12	0.32	10
ψ(.) ε(.) γ(Canopy) p(Date+Time)	614.30	3.21	0.08	0.20	10
MODO					
ψ(.) ε(.) γ(BA+PineBA) p(Date+Temp+Time)	643.07	0.00	0.16	1.00	13
ψ(.) ε(.) γ(BA) p(Date+Temp+Time)	643.47	0.40	0.13	0.82	12
ψ(.) ε(.) γ(BA+PineBA) p(Temp+Time)	643.96	0.89	0.10	0.64	11
ψ(.) ε(.) γ(BA) p(Temp+Time)	644.55	1.48	0.08	0.48	10
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	645.61	2.54	0.05	0.28	11
ψ(.) ε(.) γ(BA+Canopy) p(Date+Temp+Time)	645.72	2.65	0.04	0.27	13
ψ(.) ε(.) γ(BA+PineBA+Veg<0.5m) p(Date+Temp+Time)	646.15	3.08	0.03	0.21	14
ψ(.) ε(.) γ(Veg<0.5m) p(Date+Temp+Time)	646.21	3.14	0.03	0.21	12
ψ(.) ε(.) γ(Canopy) p(Date+Temp+Time)	646.33	3.26	0.03	0.20	12

Appendix D. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
ψ(.) ε(.) γ(PineBA) p(Date+Temp+Time)	646.38	3.31	0.03	0.19	12
ψ(.) ε(.) γ(BA+Veg<0.5m) p(Date+Temp+Time)	646.47	3.40	0.03	0.18	13
ψ(.) ε(.) γ(BA+Canopy) p(Temp+Time)	646.57	3.50	0.03	0.17	11
ψ(.) ε(.) γ(PineBA+Veg<0.5m) p(Date+Temp+Time)	646.63	3.56	0.03	0.17	13
ψ(.) ε(.) γ(BA+PineBA+Veg<0.5m) p(Temp+Time)	646.79	3.72	0.03	0.16	12
ψ(.) ε(.) γ(.) p(Temp+Time)	646.94	3.87	0.02	0.14	9
NOBO					
ψ(.) ε(.) γ(.) p(Temp+Time)	312.60	0.00	0.22	1.00	9
ψ(.) ε(.) γ(Canopy+Veg<0.5m) p(Temp+Time)	313.58	0.98	0.13	0.61	11
ψ(.) ε(.) γ(Veg<0.5m) p(Temp+Time)	313.95	1.35	0.11	0.51	10
ψ(.) ε(.) γ(.) p(Date+Temp+Time)	314.40	1.80	0.09	0.41	11
ψ(.) ε(.) γ(Canopy) p(Temp+Time)	314.52	1.92	0.08	0.38	10
ψ(.) ε(.) γ(BA) p(Temp+Time)	315.39	2.79	0.05	0.25	10
ψ(.) ε(.) γ(BA+Canopy) p(Temp+Time)	315.74	3.15	0.05	0.21	11
ψ(.) ε(.) γ(BA+Veg<0.5m) p(Temp+Time)	315.81	3.21	0.04	0.20	11
ψ(.) ε(.) γ(BA+Canopy+Veg<0.5m) p(Temp+Time)	315.89	3.29	0.04	0.19	12
ψ(.) ε(.) γ(Canopy+Veg<0.5m) p(Date+Temp+Time)	315.94	3.34	0.04	0.19	13
ψ(.) ε(.) γ(Veg<0.5m) p(Date+Temp+Time)	316.06	3.46	0.04	0.18	12
ψ(.) ε(.) γ(Canopy) p(Date+Temp+Time)	316.48	3.88	0.03	0.14	12
PIWA					
ψ(.) ε(.) γ(Canopy) p(Date)	649.75	0.00	0.47	1.00	8
ψ(.) ε(.) γ(Canopy+PineBA) p(Date)	651.28	1.53	0.22	0.46	9
ψ(.) ε(.) γ(Canopy+BA) p(Date)	652.44	2.69	0.12	0.26	9
ψ(.) ε(.) γ(PineBA) p(Date)	652.93	3.18	0.10	0.20	8
PIWO					
ψ(.) ε(.) γ(.) p(.)	269.40	0.00	0.27	1.00	5
ψ(.) ε(.) γ(SnagBA) p(.)	270.49	1.09	0.16	0.58	6
ψ(.) ε(.) γ(.) p(Temp)	271.15	1.74	0.11	0.42	7
ψ(.) ε(.) γ(Canopy) p(.)	271.28	1.88	0.11	0.39	6
ψ(.) ε(.) γ(BA) p(.)	271.73	2.33	0.08	0.31	6
ψ(.) ε(.) γ(SnagBA) p(Temp)	272.44	3.04	0.06	0.22	8
ψ(.) ε(.) γ(Canopy+SnagBA) p(.)	272.71	3.31	0.05	0.19	7
ψ(.) ε(.) γ(BA+SnagBA) p(.)	272.94	3.53	0.05	0.17	7
ψ(.) ε(.) γ(Canopy) p(Temp)	273.18	3.78	0.04	0.15	8

Appendix D. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
REVI					
ψ(.) ε(.) γ(BA) p(Temp)	580.36	0.00	0.52	1.00	8
ψ(.) ε(.) γ(Canopy) p(Temp)	581.47	1.10	0.30	0.58	8
ψ(.) ε(.) γ(BA+Canopy) p(Temp)	583.08	2.71	0.13	0.26	9
RHWO					
ψ(.) ε(.) γ(BA) p(Time)	469.16	0.00	0.52	1.00	8
ψ(.) ε(.) γ(Canopy) p(Time)	471.79	2.63	0.14	0.27	8
ψ(.) ε(.) γ(BA+Canopy) p(Time)	471.86	2.70	0.13	0.26	9
ψ(.) ε(.) γ(BA+SnagBA) p(Time)	471.88	2.71	0.13	0.26	9
WBNU					
ψ(.) ε(.) γ(.) p(Date)	634.55	0.00	0.19	1.00	7
ψ(.) ε(.) γ(.) p(.)	634.85	0.30	0.16	0.86	5
ψ(.) ε(.) γ(BA) p(Date)	636.39	1.84	0.07	0.40	8
ψ(.) ε(.) γ(.) p(Date+Temp)	636.52	1.98	0.07	0.37	9
ψ(.) ε(.) γ(SnagBA) p(Date)	636.55	2.00	0.07	0.37	8
ψ(.) ε(.) γ(BA) p(.)	636.56	2.02	0.07	0.36	6
ψ(.) ε(.) γ(Canopy) p(Date)	636.63	2.09	0.07	0.35	8
ψ(.) ε(.) γ(SnagBA) p(.)	636.66	2.11	0.06	0.35	6
ψ(.) ε(.) γ(Canopy) p(.)	636.76	2.21	0.06	0.33	6
ψ(.) ε(.) γ(Canopy+SnagBA) p(.)	638.45	3.90	0.03	0.14	7
ψ(.) ε(.) γ(Canopy+SnagBA) p(Date)	638.50	3.95	0.03	0.14	9
ψ(.) ε(.) γ(BA) p(Date+Temp)	638.53	3.98	0.03	0.14	10
WEVI					
ψ(.) ε(.) γ(.) p(Time)	602.44	0.00	0.22	1.00	7
ψ(.) ε(.) γ(.) p(.)	603.93	1.50	0.10	0.47	5
ψ(.) ε(.) γ(.) p(Date+Time)	604.31	1.87	0.08	0.39	9
ψ(.) ε(.) γ(Canopy) p(Time)	604.60	2.16	0.07	0.34	8
ψ(.) ε(.) γ(BA) p(Time)	604.67	2.23	0.07	0.33	8
ψ(.) ε(.) γ(Veg<1m) p(Time)	605.03	2.59	0.06	0.27	8
ψ(.) ε(.) γ(Canopy+Veg<1m) p(Time)	605.28	2.84	0.05	0.24	9
ψ(.) ε(.) γ(Canopy) p(.)	605.92	3.48	0.04	0.18	6
ψ(.) ε(.) γ(BA) p(.)	605.99	3.55	0.04	0.17	6
ψ(.) ε(.) γ(Veg<1m) p(.)	606.34	3.90	0.03	0.14	6
ψ(.) ε(.) γ(Canopy+Veg<1m) p(.)	606.38	3.94	0.03	0.14	7
ψ(.) ε(.) γ(BA+Veg<1m) p(Time)	606.40	3.96	0.03	0.14	9

Appendix D. (Continued)

Model	AIC_c	ΔAIC_c	Model Weights	Model Likelihood	K
WOTH					
ψ(.) ε(.) γ(BA) p(Temp)	273.96	0.00	0.39	1.00	8
ψ(.) ε(.) γ(BA+Veg>2m) p(Temp)	276.15	2.19	0.13	0.33	9
ψ(.) ε(.) γ(BA+Canopy) p(Temp)	276.44	2.48	0.11	0.29	9
ψ(.) ε(.) γ(BA+Litter) p(Temp)	276.56	2.60	0.11	0.27	9
ψ(.) ε(.) γ(Canopy) p(Temp)	277.63	3.67	0.06	0.16	8
YBCH					
ψ(.) ε(.) γ(Canopy) p(Date+Time)	622.90	0.00	0.27	1.00	10
ψ(.) ε(.) γ(Veg<1m) p(Date+Time)	623.47	0.58	0.20	0.75	10
ψ(.) ε(.) γ(.) p(Date+Time)	623.68	0.78	0.18	0.68	9
ψ(.) ε(.) γ(BA) p(Date+Time)	624.20	1.30	0.14	0.52	10
ψ(.) ε(.) γ(Canopy+Veg<1m) p(Date+Time)	625.46	2.57	0.07	0.28	11
ψ(.) ε(.) γ(BA+Canopy) p(Date+Time)	625.82	2.92	0.06	0.23	11
ψ(.) ε(.) γ(BA+Veg<1m) p(Date+Time)	626.19	3.30	0.05	0.19	11
YTVI					
ψ(.) ε(.) γ(.) p(.)	237.85	0.00	0.24	1.00	5
ψ(.) ε(.) γ(HW BA) p(.)	238.00	0.15	0.23	0.93	6
ψ(.) ε(.) γ(BA) p(.)	238.04	0.19	0.22	0.91	6
ψ(.) ε(.) γ(Canopy) p(.)	238.66	0.81	0.16	0.67	6
ψ(.) ε(.) γ(Canopy+HW BA) p(.)	240.51	2.66	0.06	0.26	7
ψ(.) ε(.) γ(BA+Canopy) p(.)	240.57	2.72	0.06	0.26	7