

2012

Science-Based Management of Shrubland Birds in Vermont's Green Mountain National Forest

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**SCIENCE-BASED MANAGEMENT OF SHRUBLAND BIRDS IN VERMONT'S
GREEN MOUNTAIN NATIONAL FOREST**

A Thesis Presented

by

JENNIFER R. SMETZER

Submitted to the Graduate School of the
University of Massachusetts Amherst in partial fulfillment
of the requirements for the degree of

MASTER OF SCIENCE

September 2012

Wildlife and Fisheries Conservation

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DEDICATION

To the littlest love of my life...you know who you are.
Thank you so very much for your unconditional love and support,
for constantly providing me with desperately needed perspective,
teaching me when to be persistent, and when it's best to just let go,
helping me to be thankful, abandon judgment, be kind, and live joyously,
and reminding me to take time to play outside every day,
rain or shine, without fail, rain or shine,
every day, without fail, every day,
for teaching me how to live in the present moment,
for reminding me to relish the simple things,
for always knowing how to make me laugh,
for being the super-hero to my sidekick,
and always and forever having my back,
for challenging me to rise athletically,
but still leaving me in your dust,
even at seventy, for showing me
that life is too short to waste
a single precious moment,
But mostly, I thank you
for just being with me
every step of the way
literally underfoot.

ACKNOWLEDGMENTS

Thank you to Dave King, my advisor, for his support, guidance, patience, and trust through this project. Thank you to the Green Mountain National Forest and Northern Research Unit for funding. Thank you to the committee members, Curt Griffin, Scott Schlossberg, and Dave King for their guidance, helpful feedback, and revisions. Thank you to John Sease, Paul Widowski, Rob Hoelscher and all the GMNF for field vehicles, data on treatment histories, and help with field logistics. Thank you to Richard Chandler, Scott Schlossberg, Mike Akresh, Krzysztof Sakrejda and Bill DeLuca and the QSG group for statistics help. Thank you to Buddy Nolan, Alia Richardson, Katie Wood, Blake Hepner, Adam Yorks and Katie Forster for excellent help in the field. Thanks to the crew at the Gathering Inn in Hancock, for feeding me, laughing with me, and keeping me sane....and the same to my friends here in Eco. Biggest thanks to Mom and Dad for always believing, always supporting, and for developing my love of the outdoors.

ABSTRACT

SCIENCE-BASED MANAGEMENT OF SHRUBLAND BIRDS IN VERMONT'S GREEN MOUNTAIN NATIONAL FOREST

SEPTEMBER 2012

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Birds that breed in early-successional shrubland habitat are declining throughout the northeastern United States. The Green Mountain National Forest (GMNF) has identified early-successional habitats and their associated bird communities as a high management priority. In this study I assessed habitat conditions and bird abundance across a range of early-successional habitat types on the GMNF to evaluate the outcome of management practices and provide management recommendations. In specific, I described the manner in which treatment method, time since last treatment, and retained tree cover influenced bird abundance and habitat conditions in openings created through even-aged timber harvest, and those maintained as permanent wildlife openings through mechanical treatment and prescribed burning.

I collected data during the 2010 and 2011 breeding seasons in 94 managed openings that ranged from 1-15 years since last treatment. In each opening I conducted three 10 minute, 50 m radius point counts, and measured vegetation structure and composition at 20 sub-points using a point-intercept method. I used N-mixture models to relate abundance to management variables,

while correcting for factors that can affect detection. I modeled the effect of management variables on percent cover of bare ground, ferns, forbs, grasses, and short and tall woody vegetation using generalized linear models with a log link, and modeled understory vegetation height using general linear models. An information-theoretic approach was used to compare models, and model averaging was used in graphical representation of results.

Habitat conditions differed most markedly between burned wildlife openings and silvicultural openings, and least between burned and mechanically treated wildlife openings. Cover of woody vegetation did not differ significantly among treatment types, and both wildlife opening types had more grasses and forbs than silvicultural openings. In relation to the findings of previous studies, these results suggest that the outcome of management activities may not depend solely on the type of treatment mechanism used, but on site-specific factors such as initial stand composition and prior land use.

I detected 21 species of shrubland birds in the surveyed openings. Seven out of eight focal species were least abundant in silvicultural openings. I found little difference in bird abundance among burned and mechanically treated openings, likely reflecting the minimal difference in habitat between these treatment types. Six species exhibited a negative relationship between abundance and the basal area of retained conifer cover in both years of the study, and seven exhibited a similar trend with deciduous tree cover in both years. There were strongly supported relationships between time since treatment and abundance for all species, and individual species peaked at different times post treatment.

These results suggest that providing habitat for the entire suite of shrubland birds requires maintaining a range of early-successional conditions across the landscape. Furthermore, the data suggest that both wildlife openings and silvicultural openings are supporting early-successional shrubland birds in the GMNF, including several species of conservation concern. Wildlife openings supported a greater abundance of shrubland birds on this forest than silvicultural openings, indicating that the extra economic cost of treating permanent openings may be warranted. The strong negative response that many shrubland birds exhibited to retained tree cover suggests that managers can increase the number of shrubland birds that openings can support by removing as much overstory tree cover during treatments as possible. This tactic may be particularly useful for increasing the habitat value of future silvicultural openings for shrubland birds in the GMNF.

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CHAPTER 1

EFFECTS OF MANAGEMENT REGIME ON HABITAT STRUCTURE AND ABUNDANCE AND RICHNESS OF SHRUBLAND BIRDS IN WILDLIFE OPENINGS AND SILVICULTURAL OPENINGS

1.1 Introduction

Birds that breed in early-successional scrub-shrub habitat are declining throughout the northeastern United States (Askins 2000, Hunter et al. 2001, Dettmers 2003). This population trend has been directly attributed to a loss of early-successional breeding habitat (Askins 1998, Askins 2000, Hunter et al. 2001). Most of the naturally occurring, relatively stable shrub communities that were once extensive in the floodplains of large rivers and coastal areas in the northeastern US (Brooks 2003, Latham 2003) have been lost due to heavy development in these areas (DeGraaf and Yamasaki 2003). Further, many of the disturbance processes that have historically created and maintained these early-successional habitats such as fire, flooding, beaver activity, silviculture, and agriculture have been suppressed or abandoned due to economic or social pressure (Askins 2000, Thompson and DeGraaf 2001, Confer and Pascoe 2003, USDA Forest Service 2006). Maintaining the avian populations that breed in early-successional habitat types therefore requires direct and active management intervention (Hunter et al. 2001, Thompson and DeGraaf 2001, Litvaitis 2003).

In response to the rapid decline in shrubland bird populations, numerous state and federal agencies are actively working to create and maintain scrub-shrub habitats (Oehler et al. 2006). Silviculture has been widely advocated as an ecologically and economically viable method of creating early-successional habitat (DeGraaf and Yamasaki 2003). However, a very small proportion of the public land on which silviculture is permitted has been actually cut in either the middle Atlantic region or New England (Oehler 2003). Moreover, management techniques that are more popular with the public such as group selection cuts (Trani et al. 2001), create patches that are too small for many area-sensitive scrub-shrub birds (Costello et al. 2000, King and DeGraaf 2004). Consequently, maintaining permanent wildlife openings through repeated mechanical treatment or prescribed burning has become the most common method for conserving early-successional shrublands in the northeastern United States (Oehler 2003).

Although permanent wildlife openings are widely used in shrubland bird management, it is unclear whether they provide adequate habitat (Askins 2000, Chandler et al. 2009a, King et al. 2009). The relative conservation value of wildlife openings and silvicultural openings is also still poorly understood (Chandler et al. 2009a, King et al. 2009). Much of the research on shrubland-bird habitat associations has been conducted primarily in regenerating forest and utility rights-of-way (Bulluck and Buehler 2006). Only one study to date has compared scrub-shrub bird abundance in managed wildlife openings and regenerating cuts (King et al. 2009). Moreover, few studies have specifically investigated the manner in which shrubland bird abundance and community structure changes as a function of how much time has passed since a site's last treatment. The timing of management regimes is critical however, because shrubland birds typically

only have a short time period in which their structural habitat needs will be met in a given patch (DeGraaf and Yamasaki, 2003, Schlossberg and King 2009). Investigating habitat associations across a broad range of early-successional habitat types and successional stages is thus imperative to effective conservation of shrubland birds. For instance, studies that have focused on a narrow range of early-successional habitat types or a narrow range of successional stages have had less success in demonstrating clear habitat relationships than those that have spanned a range of management types and stand ages (Schlossberg et al. 2010).

The Green Mountain National Forest (GMNF) has identified early-successional habitats and their associated shrubland birds as a high management priority. However, baseline conditions for these habitats and their associated avian communities have not been thoroughly evaluated in this national forest. The objective of this study was to sample systematically over a successional gradient in silvicultural openings and in wildlife openings maintained by burning and mechanical treatment to examine the effects of management regime on bird abundance and habitat conditions, and provide management recommendations.

1.2 Study area and methods

1.2.1 Study area and site selection

The study was conducted during 2010 and 2011 on the Green Mountain National Forest (GMNF), located in the southwestern and central portion of Vermont. The GMNF is approximately 162,000 ha in size (USDA Forest Service 2006). The forest cover is

primarily northern hardwoods, consisting of sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), red maple (*Acer rubrum*), yellow birch (*Betula alleghaniensis*), paper birch (*Betula papyrifera*), and white ash (*Fraxinus americana*), with secondary cover types of red spruce (*Picea rubens*) and balsam fir (*Abies balsamea*) communities at higher elevations, and Eastern hemlock (*Tsuga canadensis*) and white pine (*Pinus strobus*) at lower elevations (USDA Forest Service 2001). Approximately 4% of the GMNF is in an early-successional state, with about 1,533 ha of managed upland openings, 703 ha of regenerating forest and 2,932 ha of shrubby wetlands (USDA Forest Service 2006). Managed early-successional habitat in this forest consists of regenerating silvicultural openings and permanent wildlife openings that were originally old log landings, silvicultural openings, pastures, agricultural fields, or orchards. Permanent openings are currently maintained through prescribed burning, mowing, and mechanical treatment with machinery such as hand saws, brushhogs, chippers, or brontosaurus.

The dominant understory plant species in wildlife openings and silvicultural openings are raspberry (*Rubus spp.*), goldenrod (*Solidago spp.*), bracken fern (*Pteridium aquilinum*), American beech, several grass species (*Poaceae*), red maple, striped maple (*Acer pensylvanicum*), pin cherry (*Prunus pensylvanica*), and spirea (*Spirea spp.*). The openings across the forest vary in the amount of retained pole and saw-timber. This retained tree cover is typically dominated by American beech, red maple, sugar maple, pin cherry, yellow birch and eastern hemlock. Apple trees (*Malus domestica*) are also present on many openings

In 2010 I established 35 point-count stations in silvicultural openings, 30 in burned permanent wildlife openings, and 31 in mechanically treated permanent wildlife

openings. In 2011, I established 33 survey points in silvicultural openings, 32 in burned permanent openings, and 28 in mechanically treated permanent openings. The survey points were located in both the northern and southern portion of the GMNF (Figure 1). I only established point count plots in openings > 1 ha, because this is the size threshold for which area sensitivity has been observed in scrub-shrub birds (Costello et al. 2000, Chandler 2006). I obtained treatment histories for all managed openings on the forest prior to site selection. I preferentially located point count plots in wildlife openings that had been treated with only fire or only mechanical means, based on previous findings that complex management regimes can obscure the effects of the most recent treatment (Chandler et al. 2009a). Within each management type (burning, clearcut, and mechanical treatment), I chose openings ranging from one to fifteen years since their last treatment. Silvicultural openings in the 4-8 year time since treatment range were lacking on this forest, due to a temporary cutting ban. In order to represent these age classes in the study, in 2010 I included two 4 year old cuts on the Rutland town forest and four 5 year-old Forest Service shelterwood cuts with high basal area removal. In the second year of the study, I surveyed the Rutland town forest sites and two of the shelterwood cuts again, and added eight clearcuts on private lands ranging from 1-8 years since treatment. As such, the composition of the survey sites was not identical between years, particularly since I had to drop a number of sites in the second year of the study due to age or accessibility issues.

I established at least one 50 m radius point count plot in all the openings I chose for the study. In each year, 13 openings were large enough that I was able established two point count plots in them. All survey points were at least 250 m apart, and all were

located such that one point on the circumference of the 50 m radius fixed plot was within 5m from an edge. I chose this arrangement to ensure that point-count plots in small and large openings had the same amount of edge in close proximity, as a number of shrub-scrub species have been shown to avoid edges (Rodewald and Vitz 2005).

1.2.2 Vegetation surveys

I measured vegetation structure and composition at 20 randomly selected sub-points at each 50 m radius point-count plot using a point intercept method (King et al. 2009). At each sub-point, I recorded the maximum height of the understory and overstory substrate, with woody plants classified to genus or species, and all other cover types categorized as forb, fern, grass, bare ground, slash or standing water. All vegetation cover > 3 m in height was classified as overstory, and all cover < 3 m in height as understory. I quantified vertical structure by recording the number of woody and non-woody vegetation contacts on 0.5-m segments of a 3 m pole held vertically on the ground (Chandler et al. 2009b). I recorded the species and diameter at breast height (dbh) of all trees greater than 10cm at breast height at five 15 m radius sub-plots located in the center of the point-count location, and 35m away from the center in the four cardinal directions. I did not repeat tree species and dbh measurements at sites that were re-surveyed in the second year of the study, as I assumed the growth of pole and saw timber trees to be minimal over the course of one year.

1.2.3 Bird surveys

Birds were surveyed using 10 minute, 50 m radius point counts (Ralph et al. 1995) between late May and mid-July. I visited all sites at least three times during the breeding season, to facilitate estimation of detection probability. The observer, date, time of survey, wind, and cloud cover were recorded during each survey. Surveys were conducted only on calm days with no precipitation between 0.530 h and 10.30 h. Visits with the highest wind were repeated at the end of the survey season when possible. No sites were consistently surveyed during early, mid, or late morning, in order to help standardize for the detection bias that can result if species sing more consistently during a particular time (Ralph et al. 1995). Observers were rotated through sampling locations. The species and mode of detection were recorded for all birds seen or heard. A few openings were irregularly shaped, and though greater than 1 ha in area, they were not wide enough to fully encompass a 50 m radius count circle. Aerial orthophotos and ArcGIS were used to establish a survey plot with the equivalent area of a 50-m radius plot (0.785 ha), and birds were counted only within that area. Females, birds of unknown sex, birds flying over the plot, and birds heard outside the 50 m radius point-count circle were not included in the analysis, except for Cedar Waxwings for which sex cannot be determined in the field.

1.2.4 Statistical analysis

Habitat data for each vegetation plot were placed into substrate categories (fern, forb, grass, bare ground, woody vegetation) based on commonalities in growth and

structure following King et al. (2009). Additionally, woody vegetation was separated into short ($\leq 1.5\text{m}$) and tall ($> 1.5\text{m}$) size classes following Schlossberg et al. (2010).

I modeled the habitat data from the two years of the study separately to avoid pseudoreplication, to investigate the consistency of the results between the two years, and because a number of the sites were not surveyed in both years of the study. Point count plots that were in the same opening were assumed to be independent samples, because the plots were at least 250 m apart. Moreover, the degree and intensity of treatment appeared to typically be quite heterogeneous across the openings, despite their formal designation as contiguous stands on GMNF maps.

Habitat response variables were related to management variables using general and generalized linear models. Management variables included treatment type (clearcut, prescribed burning, and mechanical treatment), time since treatment, and a quadratic term for time since treatment. I was interested in modeling the relationship between treatment frequency and vegetation characteristics but I did not have data on the year in which wildlife openings were first established. I considered interaction terms for time since treatment and treatment type; however, interactions were not in top models for any response variables, so I dropped them from the analysis. I related understory vegetation height to management variables using general linear models with the "lm" function in R. I modeled the understory counts of ferns, forbs, grasses, bare ground, and tall and short woody vegetation using generalized linear models with a log link (Quinn and Keough 2003) as these data were highly skewed and non-normal, even after transformation. I chose the most appropriate exponential distribution for modeling the non-normal substrate cover data by comparing the AIC_c values and goodness of fit of global models

under the Poisson, negative binomial, zero-inflated Poisson, and zero-inflated negative binomial distributions. Finally, I used non-parametric Mann–Whitney tests to compare the basal area of retained coniferous and deciduous tree cover among treatment types, as these data were highly skewed and non-normal even after transformation.

Model selection was accomplished using an information theoretic approach, and the amount of variability explained by the models was approximated using Nagelkerke's R^2 index (Nagelkerke 1991). I considered covariates present in models with a delta $AIC_c \leq 2$ as moderately supported, and those with 95% confidence intervals not including zero as strongly supported. Parameter estimates for the clearcut and mechanical treatments were compared using z tests. For graphical representation of results, I averaged the model predictions using Akaike (AIC_c) weights (w_i) as described by Burnham and Anderson (2002), in order to allow for multi-model inference. Models were fit and parameters estimated using the R software package, version 2.10.0 (R Development Core Team Year 2009), and Akaike weights (w_i) were estimated using the `AICmodavg` library in the R software package according to the formula:

$$W_i = \frac{\exp(-0.5 * \Delta AIC_{c,i})}{\sum_{i=1}^l \exp(-0.5 * \Delta AIC_{c,i})}$$

I modeled the relationship between management covariates and the abundance of shrubland birds (designated following Schlossberg and King 2007) with N-mixture models (Royle 2004). This modeling approach is apt because it avoids bias in abundance estimates by accounting for individuals that are present but not detected (Thompson 2002). Species that were found in $\geq 10\%$ of the plots and for which there were at least 40 observations in each year were chosen as focal species. I modeled the abundance data from the two years of the study separately to avoid pseudoreplication, to investigate the

consistency of the results between the two years, and because a number of the sites were not surveyed in both years of the study. I did not analyze bird abundance in relation to habitat variables, because my primary objective was to evaluate the influence of management factors that could be directly controlled by managers.

Abundance (λ) was modeled in relation to management variables using a log link, and probability of detection (p) using a logit link. Management variables included time since treatment, and the three treatment types: clearcut, mechanical treatment, and prescribed burning. I also considered the combined basal area of the retained pole and saw timber trees as a management covariate, because this habitat factors-though not directly related to shrubland conditions- could affect the abundance of shrubland birds. I modeled deciduous and coniferous basal area separately because they were not correlated, and thus potentially offered unique information; moreover, I expected that species could respond differentially to coniferous and deciduous cover. I included a quadratic term for time since treatment as a management variable, because previous studies have shown that some shrubland species peak in abundance at intermediate times within the successional gradient (Schlossberg and King 2009). I was interested in modeling the relationship between treatment frequency and bird abundance; however, I did not have data on the year in which wildlife openings were first established. I considered interaction terms for time since treatment and treatment type; however, interactions were not in top models for any species, so I dropped them from the analysis. Probability of detection was modeled in relation to observer, time of day, date, a quadratic term for date, understory vegetation height and wind intensity, as these factors

have been previously shown to affect detectability (Kery et al. 2005, Aldredge et al. 2007, Johnson 2008).

In the interest of constructing a reasonably sized list of candidate models, for each species, I first determined a limited set of detection covariates to include in subsequent candidate models. I accomplished this by fitting all subsets of the detection covariates with the global suite of management variables, and identifying the observation covariates that were in models with a delta AIC_c of two or less, and statistically significant at an alpha of 0.1. For each species, I then fit all subsets of the management variables with this finite set of "best" detection covariates fixed.

I used the same model selection and averaging procedures as described for the habitat models, and the same criteria for assessing the strength of evidence for covariates. The most appropriate exponential distribution for modeling abundance was chosen by comparing the AIC_c values and goodness of fit of global models under the Poisson and negative binomial error distributions. Variables were assessed for normality and outliers, and log transformed as appropriate. All continuous variables were standardized to a mean of zero and a standard deviation of one to facilitate model convergence. Models were fit, and parameters and Akaike (AIC_c) weights (w_i) were estimated using the unmarked library in the R software package, version 2.10.0 (R Development Core Team Year 2009).

I compared species richness among the burned wildlife openings, mechanically treated wildlife openings, and silvicultural openings using sample-based rarefaction curves. I chose to use sample-based rarefaction over individual-based rarefaction because the latter can overestimate the number of species that would have been found at a lower

sampling intensity (Gotelli and Colwell 2001). Moreover, by treating the group of individuals at any given survey plot as the unit of replication, sample-based species accumulation curves account for patchiness in the data, between sample heterogeneity, and the reality that species may occur nonrandomly among samples due to spatial autocorrelation. As such, individuals cannot be sampled randomly (Gotelli and Colwell 2001). This is particularly true for this data, as the unit replication in this study was the point-count plot, individual

For each survey site, I used the data from the sampling period in which the greatest number of males was detected within the 50 m radius survey area, as the maximum number of birds detected over repeated sampling occasions is more representative of true abundance than the mean number of birds detected (Toms et al. 2006). I compared richness among the three opening treatment types for the shrubland birds detected, and for the full community of birds detected. Rarefaction curves were constructed using a bootstrapping procedure in the R software package, version 2.10.0 (R Development Core Team Year 2009).

1.3 Results

1.3.1 Habitat characteristics

Management covariates were in supported models for all habitat response variables, though the null model was the most highly supported for forb cover in 2011. The mean adjusted R^2 value for habitat models was 0.18, and the amount of variability explained by the models differed greatly among habitat response variables, with adjusted

R^2 values ranging from 0.00 for forb cover to 0.29 for tall woody vegetation (Table 1). Akaike weights (w_i) varied widely among habitat response variables and even among years for individual response variables, ranging from 0.86 for bare ground, to 0.14 for forb cover.

The results of habitat models were fairly consistent among years. For all habitat response variables, supported models (with a delta AIC < 2) had the same covariates from year to year, with the exception of a treatment effect for tall woody vegetation, and a time since treatment effect for fern cover. Moreover, the signs of the parameter estimates in supported models were consistent from year to year for all habitat response variables, with the exception of the mechanical treatment effect for fern and forb cover. There were two notable inconsistencies among years. First, the null model was the most strongly supported for forb cover in 2011, while a treatment effect was strongly supported for this response variable in 2010, and second, woody vegetation increased linearly in 2010, but showed a modal pattern in 2011.

Treatment type was in top models for bare ground, forbs, grasses, ferns, and tall woody vegetation, but was not present in top models for overall cover of woody vegetation, short woody vegetation, or understory vegetation height (Table 1). I found strong evidence that silvicultural openings had more bare ground and less fern and forb cover than burned openings, and moderate evidence that they had less grass cover and more tall woody vegetation than burned openings. Mechanically treated sites had a greater cover of grasses and forbs than silvicultural openings, and had less bare ground and tall woody vegetation than silvicultural openings, though 95% confidence intervals for the latter two parameter estimates contained zero. Finally, I found strong evidence

that mechanically treated openings had more grass and less fern cover than burned wildlife openings, and moderate evidence that they had more bare ground and less tall woody vegetation than burned sites.

Time since treatment was in supported models in both years of the study as either a linear or quadratic term for all habitat response variables with the exception of fern cover (Table 1). I found strong evidence that woody vegetation, tall woody vegetation and understory vegetation height increased with time since treatment, and that short woody vegetation and grass cover decreased as a function of time since treatment. A negative time since treatment effect was also present in top models for forb and fern cover, though 95% confidence intervals for these parameter estimates overlapped zero. Cover of bare ground, woody vegetation, tall woody vegetation, and understory vegetation height all had a quadratic term for time since treatment in top models, and exhibited a quadratic form in the shape of their model-averaged predictions (Figure 3).

The amount of retained overstory differed significantly between the three treatments (Table 2). Silvicultural openings had a higher basal area of retained deciduous trees than both mechanically treated and burned wildlife openings ($p=0.027$ and $p=0.001$ respectively in 2010, $p=0.04$ and $p=0.00$ respectively in 2011). Silvicultural openings also had a higher basal area of retained coniferous trees than burned wildlife openings ($p=0.009$ in 2010 and $p=0.01$ in 2011). Mechanically treated wildlife openings had a higher basal area of deciduous trees than burned sites in 2011 ($p=0.03$).

1.3.2 Bird abundance and detection probabilities

Including only males detected within survey plots, and all sexes for Cedar Waxwing, I recorded 2,031 observations of 66 species in 2010 and 2,199 observations of 69 species in 2011 (Table 3). These data included 21 species of shrubland birds. I had sufficient sample sizes to estimate abundance for eight shrubland species: Alder Flycatcher, Cedar Waxwing, Common Yellowthroat, Chestnut-sided Warbler, Indigo Bunting, Mourning Warbler, Song Sparrow, and White-throated Sparrow (scientific names in Table 3).

Management variables were in supported models for all eight focal species. The mean adjusted R^2 value for N-mixture models was 0.38, and the amount of variability explained by the models differed greatly among species, with adjusted R^2 values ranging from 0.66 for Chestnut-sided Warblers and Indigo Buntings to 0.07 for Mourning Warblers (Table 4). Akaike weights (w_i) varied widely among species and even among years for individual species, ranging from 0.64 for Common Yellowthroats, to 0.10 for Song Sparrows.

Differences in bird abundance were most distinct between burned wildlife openings and silvicultural openings. There was strong evidence that Alder Flycatchers, Cedar Waxwings, Chestnut-sided Warblers, Common Yellowthroats, Indigo Buntings, and White-throated Sparrows were more abundant in burned openings than in silvicultural openings and moderate evidence of the same pattern for Song Sparrows (Table 4). I found strong evidence that Chestnut-sided Warblers, Common Yellowthroats, Indigo Buntings, and Cedar Waxwings were more abundant in mechanically treated openings than in silvicultural openings, and moderate evidence of the same pattern for

White-throated Sparrows and Song Sparrows. There was moderate evidence that Mourning Warblers were more abundant in silvicultural openings than in either mechanically treated or burned wildlife openings.

Differences in bird abundance were least pronounced among burned and mechanically treated wildlife openings, and the direction of the mechanical treatment-effect parameter estimates were inconsistent from year to year for all but one species (Table 4). White-throated Sparrows were more abundant in burned openings than in mechanically treated openings in both years of the study. There was moderate evidence that Alder Flycatchers were more abundant in burned openings than in mechanically treated sites, and that Song Sparrows and Mourning Warblers were more abundant in mechanically treated sites than in burned openings, though for each of these three species, treatment effects were only present in top models in one year of the study.

Time since last treatment was a strong predictor of bird abundance. A linear and quadratic term for this covariate were both present in top N-mixture models for all species in both years, though for Mourning Warbler and Song Sparrow there was only strong support for these covariates in one year (Table 4). White-throated Sparrows reached maximum abundance immediately following treatment in both years (Figure 4). All other species reached maximum abundance at intermediate times since treatment.

The variables for basal area of retained coniferous and retained deciduous trees were both in supported N-mixture models for all species in both years of the study (Table 4). The deciduous basal-area parameter estimates were consistently negative among years and supported models for all species except for Mourning Warbler. Similarly, the coniferous basal-area parameter estimates were consistently negative among years and

supported models for all species except Cedar Waxwing and White-throated Sparrow. This consistency indicates the importance of retained tree cover in determining shrubland bird abundance. The negative relationship between basal area of retained deciduous trees and abundance was strongly supported for Chestnut-sided Warblers, Common Yellowthroats, and White-throated Sparrows. The inverse relationship between basal area of retained conifers and abundance was strongly supported for Indigo Buntings, Alder Flycatchers, Chestnut-sided Warblers, Song Sparrows and Mourning Warblers. There was moderate evidence that White-throated Sparrow abundance was positively related to basal area of retained coniferous trees.

All detection covariates that were considered in the modeling process were supported in top N-mixture models for at least one species (Table 5). Observer was an important detection covariate for every species in every year, with the exception of Song Sparrows. There was strong evidence that four species were less detectable in sites with tall understory vegetation in at least one year of the study, and moderate evidence of this effect for one other species. Detectability of Alder flycatchers, Indigo Buntings, and Song Sparrows decreased throughout the season and detectability of White-throated Sparrows increased, though for the latter, the effect was only moderately supported. Detectability of Common Yellowthroats, Indigo Buntings and Cedar Waxwings increased throughout the 2010 season, and decreased throughout the 2011 season, though this date effect was only strongly supported in both years for Cedar Waxwings. I found strong evidence that Cedar Waxwings and Chestnut-sided Warblers were less detectable during surveys with higher wind speed, and moderate evidence that wind increased the detectability of White-

throated Sparrows. Finally, I found strong evidence that Indigo Buntings became more detectable throughout the morning survey period in 2010.

1.3.3 Species richness

The rarefaction curves constructed for the shrubland species encountered within the point-count plots did not reach an asymptote for any of the treatment types in either of the years, indicating that there are likely more shrubland species that use early-successional habitat in this region than the 21 species that I detected during this study (Figure 5). The 95% confidence intervals on the species accumulation curves overlapped for all pairwise comparisons of treatment types, indicating that species richness did not differ between treatments. I observed all the same results for the rarefaction curves representing the full community of birds (Figure 6).

1.4 Discussion

The precipitous decline of shrubland birds in the northeastern United States has prompted scientists and managers from nearly two dozen agencies and Universities in this region to work toward developing a unified monitoring and management program designed to document population trends, and assess the efficacy of management options (Bried et al. 2011). This study contributes to this ongoing conservation effort by assessing the outcome of the current habitat management practices on the GMNF.

The openings on the GMNF provide habitat for a number of shrubland species of conservation concern. Quite a few of the relatively abundant species in these openings,

including White-throated Sparrow, Indigo Bunting, Chestnut-sided Warbler, Mourning Warbler, Canada Warbler, Alder Flycatcher and Nashville Warbler are listed by Partners in Flight as species of continental importance in the Northern Forest and/or Eastern Avifaunal Biome (Rich et al. 2004). Other species that are relatively abundant in this national forest, such as Common Yellowthroat, and Song Sparrow are also facing significant population declines in the Eastern US (Sauer et al. 2008). Continued and effective management of shrubland habitat in the GMNF is, therefore, critical, particularly since many of these species have a large majority of their breeding population in the Northeast (Rich et al. 2004).

1.4.1 Habitat characteristics

In accordance with a number of previous studies, I found that habitat characteristics were related to management, though the amount of variability in habitat response explained by management variables on the GMN was lower than in a similar analysis conducted by Chandler (2009). My finding that mechanically treated wildlife openings had a greater cover of grasses and forbs than silvicultural openings is consistent with results from previous studies comparing regenerating silvicultural openings to mechanically treated rights-of-way (Bulluck and Buehler 2006) and mechanically treated wildlife openings (King et al. 2009). I also observed more grasses and forbs in burned openings than in silvicultural openings in one year of the study. This result is consistent with the findings of Fink et al. (2006), who compared burned glades and regenerating clearcuts. Finally, my finding that silvicultural openings had the more bare ground than mechanically treated openings was consistent with the findings of King et al. (2009).

Some of the relationships I observed between treatment mechanism and habitat characteristics were, however, inconsistent with findings from previous studies, particularly in regards to woody vegetation. I did not observe any significant difference in the overall amount of woody vegetation among any of the treatment methods. In contrast, King et al. (2009) found that silvicultural openings had a greater cover of shrubs and trees than mechanically treated wildlife openings; Bulluck and Buehler (2006) observed more saplings in regenerating clearcuts than in mechanically treated rights-of-way; and Fink et al. (2006) found more woody vegetation in burned openings relative to regenerating clearcuts. Moreover, previous research comparing sites treated through prescribed burning and mechanical treatment reported pronounced differences in woody plant cover (Christensen 1985, Zuckerberg and Vickery 2006, Chandler et al. 2009a). In contrast to all these studies, I only found modest evidence in 2011 that the amount of tall woody vegetation differed among treatments, with silvicultural openings having the greatest cover, mechanically treated openings the least, and burn openings intermediate.

Geographic and site-specific differences in how plant communities respond to disturbance can account for some of the inconsistency between this and previous studies in how management activities influenced habitat structure and composition. For instance, management can have different habitat outcomes regionally, because geographic variability in climate, soil, and local plant community composition can impact plant community response to disturbance (Pickett and White 1985).

The effect of prescribed burning on woody plant composition may also have interacted with land use history and initial stand composition in the GMNF. It is well documented that initial stand composition can influence habitat composition post-

disturbance (Pickett and White 1985). On the GMNF, many of the burned wildlife openings are managed for wild blueberry (*Vaccinium spp.*) production. As such, they may have more fire-tolerant woody species (or simply more woody cover in general) than managed shrublands in other studies, and fire disturbance may not have as pronounced an effect on woody plant cover in these openings as it might in other regions. Indeed, it is well documented that although fire can retard the establishment of woody vegetation in areas dominated by herbaceous or grassy cover, once established, shrub cover typically will increase or at least remain stable, even under a fairly frequent burning regime (Heisler et al. 2003, Zuckerberg and Vickery 2006).

The effect of mechanical treatment on woody plant composition may have similarly interacted with land use history and initial stand composition in this study. Mechanically treated sites in the GMNF are a mixture of old fields, orchards, and forests that have been repeatedly cut. This variability in land-use equates to a predominance of woody stump-sprouted vegetation in some sites, and grassy vegetation in other sites. As such, my finding that mechanically treated sites did not differ greatly in woody vegetation cover relative to other treatments may simply have been a function of variability in initial stand composition. In support of this hypothesis, King et al. (2009) found pronounced differences between silvicultural openings and mechanically treated “old field” wildlife openings in Massachusetts, while Luken et al. (1992) found very little difference in habitat characteristics between regenerating forest and mechanically treated power line rights-of-way that were originally forest. Similarly, though Bulluck and Buehler (2006) found more saplings in regenerating clearcuts than in mechanically treated rights-of-way, the two treatments did not differ in shrub cover.

1.4.2 Bird abundance and detection probabilities

Although there have not been many systematic comparisons of bird abundance between permanent wildlife openings and silvicultural openings, studies of shrubland birds in other early-successional habitats suggested that they are sensitive to different management regimes. Previous studies reported that silvicultural openings differ in their bird communities and habitat conditions between power line rights-of-way and revegetated strip mines (Bulluck and Buehler 2006), forest edge and naturally occurring cedar glades maintained through fire (Fink et al. 2006), and natural openings maintained through burning (Schulte and Niemi 1998). The one study to date that specifically compared bird abundance in silvicultural openings and permanent wildlife openings also reported pronounced differences in shrubland bird communities (King et al. 2009).

The mean abundance levels I observed for shrubland species in the GMNF (Table 6&7) were comparable to previous studies, with a few exceptions. Abundance of Chestnut-sided Warblers in mechanically treated openings on the GMNF was similar to what Chandler et al. (2009a) reported in similar openings in New Hampshire; however, Alder Flycatchers, Indigo Buntings, Cedar Waxwings, and White-throated Sparrows were slightly more abundant in the GMNF than in New Hampshire. Abundance of these four species was similar in the GMNF to what King et al. (2009) reported for silvicultural openings and mechanically treated wildlife openings in Massachusetts. Song Sparrows were generally less abundant in the GMNF relative to previous studies (Chandler et al. 2009a, King et al. 2009, Schlossberg et al. 2010). In contrast, I observed a greater abundance of Mourning Warblers relative to King et al. (2009) in silvicultural openings

and mechanically treated wildlife openings and Chandler et al. (2009a) in burned and mechanically treated wildlife openings. I also observed more Common Yellowthroats and Chestnut-sided Warblers in mechanically treated sites relative to King et al. (2009), and more Chestnut-sided Warblers in silvicultural openings.

Despite some species-specific differences between this and previous studies in absolute abundance, many of my observations regarding how bird abundance differed between treatments were consistent with previous studies. Mourning Warblers were most abundant in silvicultural openings on the GMNF. This finding is in accordance with previous studies comparing bird abundance in regenerating forests to mechanically treated wildlife openings (King et al. 2009) and burned openings (Schulte and Niemi 1998). I found that all other focal species were least abundant in silvicultural openings on the GMNF. Consistent with this finding, King et al. (2009) found that White-throated Sparrows, Indigo Buntings, and Song Sparrows were more abundant in mechanically treated wildlife openings relative to regenerating forest, Schulte and Niemi (1998) found that Indigo Buntings were more abundant in burned areas than in silvicultural openings, and Bulluck and Buehler (2006) reported that Chestnut-sided Warblers, Indigo Buntings, Song Sparrows, and Common Yellowthroats were less abundant in silvicultural openings than in other early-successional habitats.

However, a few of my species-specific results in how bird abundance differed between treatments were inconsistent with those of previous studies. In contrast to my findings in Vermont, and the findings of Bulluck and Buehler (2006), both King et al. (2009) and Schulte and Niemi (1998) reported that Chestnut-sided Warblers were most abundant in regenerating forests. Moreover, King et al. (2009) found no difference in

Common Yellowthroat abundance among mechanically treated wildlife openings and regenerating forest, while this species was least abundant in silvicultural openings on the GMNF.

Some of the inconsistency between this and previous studies in species-specific responses to treatment may be explained by variability among studies in how habitats responded to management. Fine-scale habitat attributes are critical in determining foraging, nesting, roosting, and predator evasion opportunities (Hilden 1965, Sabo 1980, Martin 1998). If there are regional differences in how habitats respond to disturbance- as was indicated by my vegetation results- it seems reasonable to expect that patterns of bird abundance may reflect vegetation characteristics more strongly than management categories per se.

Most of the differences in bird abundance that I observed between regenerating cuts and burned and mechanically treated wildlife openings can be explained by differences in habitat composition among the treatments, and previously documented species-specific habitat associations. For instance, Indigo Buntings, Common Yellowthroats, White-throated Sparrows and Song Sparrows were more abundant in both wildlife opening types than they were in silvicultural openings. These species all require a combination of dense herbaceous and shrub cover (Guzy and Ritchison 1999, Arcese et al. 2002, Confer and Pascoe 2003, Holmes and Pitt 2007, Schlossberg and King 2007, Falls and Kopachena 2010). In turn, both types of wildlife opening had a more even distribution of herbaceous and woody plant cover than the silvicultural openings, in which 75% of the understory substrate was either bare ground or woody vegetation. Clearcuts had the highest cover of bare ground in the GMNF, and the lowest abundance

of shrubland birds, and bare ground has been previously shown to negatively influence abundance of shrubland species (Schlossberg et al. 2010).

Mourning Warblers were the only species most abundant in silvicultural openings. This species tends to prefer significant canopy closure (Dunn and Hall 2010), and the basal area of canopy trees was greatest in silvicultural openings. Mourning warblers are also associated with dense, tall woody vegetation (Holmes et al. 2007, Schlossberg and King 2007), which was most abundant in regenerating clearcuts. In contrast, it was somewhat surprising that Chestnut-sided Warblers and Alder Flycatchers were least abundant in silvicultural openings, as these species are associated with dense stands of shrubs and saplings (Thompson and Capen 1988, DeGraaf and Yamasaki 2001, Hagan and Meehan 2002, Holmes et al. 2007, Schlossberg and King 2007, Chandler et al. 2009a), and are known to seek high perches for singing and foraging (Richardson and Brauning 1995, Lowther 1999).

Some of the differences I observed in bird abundance between wildlife openings and regenerating forest may also be explained by specific plant taxonomic preferences. For instance, Chestnut-sided Warblers are known to prefer deciduous cover, and avoid conifers (Keller and Smith 2003, Schlossberg and King 2007, King et al. 2009). This species was least abundant in silvicultural openings in the GMNF, the opening type with the greatest amount of coniferous overstory. Similarly, Schulte and Niemi (1998) reported that Chestnut-sided Warblers preferred habitats that contained primarily red maple saplings over ones that had a higher cover of coniferous species. Alder Flycatchers prefer dense wet stands of shrubs (DeGraaf and Yamasaki 2001, and are associated with alders and willows (King et al. 2010, Schlossberg et al. 2010), and Common

Yellowthroats have been associated with both willow (Baril et al. 2011) and alder species (Schlossberg et al. 2010). These species were both more abundant in wildlife openings and over the course of the study, 41 out of the 51 observations of alders and willows were in wildlife openings.

Despite differences in habitat composition between burned and mechanically treated wildlife openings, I only found moderate evidence that bird abundance differed between these treatments. This observation was in accordance with the findings of Van Dyke et al. (2004) and Chandler et al. (2009a), the latter of whom posited that the effects of the most recent treatment may be obscured if sites have complex treatment histories. Despite the fact that I surveyed in all the sites on the GMNF with simple treatment, many of the permanent wildlife openings on the GMNF, and in this study have been both burned and mechanically treated. In addition, the two wildlife opening treatment types were most different in grass and fern cover, which may not be as biologically important for species as shrub and forb cover. Moreover, these differences in grass and fern cover may simply not have been large enough to have any biologically meaningful influence on habitat quality for the focal shrubland species, despite being statistically significant.

Consistent with the results of most previous research (but see Chandler 2006), I found that the abundance of shrublands birds was strongly related to time since last treatment. Species-specific temporal patterns of abundance were also largely consistent with habitat-associations observed in previous studies. For instance, in accordance with the findings of previous studies, (DeGraaf and Yamasaki 2001, Keller and Smith 2003, Chandler 2006, and Schlossberg and King 2009) White-throated Sparrows peaked in abundance in the GMNF immediately after treatment. This species is associated with

grasses and herbaceous cover (Schlossberg and King 2007), which were most abundant immediately or shortly after treatment, and avoids the dense tall woody stems (Hagan and Meehan 2002) that are characteristic of older stands. White-throated Sparrows also nest and forage on or near the ground, and thus require the low vegetation cover that was characteristic of the younger openings in this study.

The modal pattern of abundance that I observed for a number of species has been previously demonstrated for Common Yellowthroat (Keller et al. 2003), Alder Flycatcher, Song Sparrow (Chandler 2006), and Chestnut-sided Warbler (Keller et al. 2003, Holmes et al. 2007, Schlossberg and King 2009). Many of the modal species in the GMNF either nest or forage in dense, woody understory vegetation (Richardson and Brauning 1995, Guzy and Ritchison 1999 Lowther 1999), which was most abundant approximately 6 to 10 years post-treatment, when both short and tall woody vegetation were present. I observed a modal pattern of abundance for Song Sparrows, and Indigo Buntings, unlike Schlossberg and King (2009) who found these species to be most abundant immediately following treatment. Notably however, these two species peaked in abundance earlier than most of the other modal species, perhaps reflecting the fact that both require a high cover of forbs and grasses for seed foraging (Arcese et al. 2002, Payne 2006), and both nest on or close to the ground, and thus require low-stature vegetation in which to place and hide nests (Confer and Pascoe 2003, Chandler 2006, Payne 2006). Previous research has suggested that these temporal patterns of bird abundance reflect similar temporal patterns in availability of nesting and foraging habitat post-disturbance (Titterington et al. 1979, DeGraaf 1991, Holmes and Pitt 2007, Schlossberg and King 2009). In support of this, Keller et al. (2003) reported a direct

correlation between a dramatic increase in leaf area and avian richness in newly regenerating cuts, and a peak in arthropod biomass around five to seven years post treatment.

In accordance with previous studies, many shrubland birds in the GMNF were less abundant in areas with a higher basal area of tree cover. A negative relationship between the basal area of deciduous trees and abundance was previously documented for White-throated Sparrows (King and DeGraaf 2000), Common Yellowthroats (King and DeGraaf 2000, Hagan and Meehan 2002), Chestnut-sided Warblers, and Mourning Warblers (Jobes et al. 2004). Similarly, both Alder Flycatchers, (Schulte and Niemi 1998, DeGraaf and Yamasaki 2001, Schlossberg and King 2007) and Cedar Waxwings (Witmer et al. 1997, Schlossberg and King 2007) are known to prefer areas with little retained tree cover. My finding that Indigo Bunting, Alder Flycatcher and Song Sparrow abundance was inversely related to conifer cover is however inconsistent with the results of King and DeGraaf (2000), who compared bird abundance in mature forest, clearcuts, and shelterwood cuts in New Hampshire. However, the range of variation in basal area of conifers was significantly greater in the GMNF than it was in the King and DeGraaf (2000) study.

Detection covariates were included in supported models for all species, demonstrating the importance of accounting for detection probability when estimating bird abundance. Despite having well trained observers, in accordance with previous studies (Moore et al. 2004, Chandler 2006), I found that probability of detection was most consistently and strongly influenced by differences in observer ability. The decrease in detectability of Alder flycatchers, Indigo Buntings, and Song Sparrows throughout the

breeding season may be related changes in singing rates associated with nesting or reproductive status (Best 1981, Krebs et al. 1981, Cuthill and Hindmarsh 1985). The inconsistent relationship between detectability and date for Cedar Waxwings in the two years of the study may simply be related to patterns of food abundance, and subsequent nomadic movements into and out of the surveyed openings. Previous studies have reported that some species were more detectable in lower understory vegetation and some in taller vegetation (Bibby et al. 1985, Confer and Pascoe 2003). It is not surprising that wind significantly reduced detectability for two species; however, my finding that White-throated Sparrows were more detectable in high wind is perplexing. My observation that four species were more detectable in lower-stature vegetation was most likely a consequence of observers being able to see birds that were not singing in areas with shorter vegetation. Detectability of Indigo buntings was related to time of day. This result probably reflects that singing rates are often variable throughout the day (Skirvin 1981), particularly for this species, which sings especially loudly during the hottest part of the day from high perches (Payne 2006).

1.4.3 Species Richness

My finding that richness of shrubland species did not differ among treatments is in accordance with the findings of Chandler (2006), but inconsistent with the results of Schlossberg and King (2007), and Schulte and Niemi (1998). Vegetation structure and composition are known to influence bird abundance. Because I observed differences in habitat composition between treatment types, it seems reasonable to expect that each treatment type might host a different community of early successional birds. However,

unlike the N-Mixture models, my comparison of species richness across treatments did not control for the variability in stand age. Habitat conditions were strongly influenced by time since treatment, and this covariate was in top models for all measured habitat variables. Thus, I may have observed the same community of shrubland birds in all treatments, simply because I sampled across such a wide range of stand ages and in turn, a wide range of vegetation structure and composition. In support of this, Chandler (2006) sampled in mechanically treated and burned sites ranging from 1-18 years since treatment and found no difference in species richness. In contrast, Schulte and Niemi (1998) sampled in burned and clearcut sites of the same age, and reported higher species richness in burned sites. Though Schlossberg and King (2007) found a difference in shrubland bird richness between wildlife openings and silvicultural openings, the studies they used to construct the rarefaction curves did not survey over the same exact range of stand ages. As such, the data sources they used in their analysis may represent different snapshots in time of habitat conditions and bird communities.

Similarly, my finding that species richness did not differ among treatments for the full suite of species observed in the plots may be related to the variability in stand age within each treatment type, as well as the high variability in retained basal area for each treatment type. Because the silvicultural openings had a higher median basal area of retained tree cover, I expected that they might be able to meet the needs of a wider range of species, and thus have greater species richness. Indeed, previous studies have demonstrated that intermediate levels of cutting- as in shelterwood cuts, or in systems with retention of residual trees in even-aged harvests (EAR treatments) - are capable of providing habitat for both early-successional species and mature-forest species (DeGraaf

1991, King and DeGraaf 2000, Rodewald and Yahner 2000). However, in each treatment type, sites ranged from low to high basal area retention. Therefore each treatment type had sites that were potentially capable of supporting both shrubland and some mature-forest species.

1.4.4 Limitations and Scope of Inference

Although this study spanned a very large geographic area of Vermont, it should not necessarily be assumed that the results are relevant throughout the greater New England region. For instance, the sites I surveyed are unique to the management regimes implemented by the GMNF. Burn intensity, method of mechanical treatment, amount of tree cover left during management, and the land use history of managed openings can be vary significantly from agency to agency. As such, the impact of management regime on habitat characteristics and bird communities in the GMNF may not be relevant across all managed forests.

In this study, I did not account for landscape attributes, or for patch effects; however, the results should still be robust. Though some shrubland species are known to be area sensitive, I only surveyed in openings greater than 1 ha in size, the general size above which the effects of patch size are generally modest (Rodewald and Vitz 2005, Lehnen 2008). Moreover, the shrubland species for which area effects have been most commonly reported, such as Yellow-breasted Chats, (Rodewald and Vitz 2005, Lehnen and Rodewald 2009), Yellow Warbler (Chandler et al. 2009b), and Eastern Towhee (Askins et al. 2007) were not common in the GMNF, and were not focal species. Furthermore, microhabitat features appear to play a greater role in determining habitat

selection for shrubland species than patch and landscape factors (Hagan and Meehan 2002, MacFaden and Capen 2002, Askins et al. 2007, Chandler et al. 2009a). Finally, the GMNF is extensively forested (USDA Forest Service 2006), and landscape factors are less pronounced in areas with extensive mature forest cover (MacFaden and Capen 2002, Chandler 2006, Askins et al. 2007).

Finally, inferences regarding habitat quality should be tempered with the caveat that this study did not measure productivity, nor did it account for site fidelity, which may be an important source of variability in bird abundance. Previous studies in managed openings and clearcuts have reported overall nest success rates of 52% (King et al. 2009), 55% (Chandler et al. 2009a), and 43% (Schlossberg and King 2007), indicating that shrubland birds successfully reproduce in managed openings. However, because this study assessed habitat quality based solely on abundance, it may overlook important habitat associations, because bird abundance does not always directly correlate with fitness or site quality (Vickery et al. 1992, Brawn and Robinson 1996, Johnson 2007). For instance, if large numbers of young non-breeding individuals tend to aggregate in marginal areas, abundance estimates alone can fail to predict habitat quality (Van Horne 1983). Moreover, in areas with very little early-successional cover, birds may simply saturate the available habitat, to the point that abundance is high in both high and low quality sites. Finally, habitat relationships based on abundance estimates alone may be inaccurate if birds return to sites regardless of whether there is breeding habitat available or not.

1.5 Management Implications

Shrubland birds are a high conservation priority for state and federal agencies, as well as private conservation groups. There are numerous options for establishing and maintaining early successional habitat. However, there is also still quite a bit of uncertainty regarding the manner in which alternative management options influence habitat quality and shrubland bird abundance (Askins 2001, Bulluck and Buehler 2006, King et al. 2009), in part because much of the research on shrubland-bird habitat associations to date has been conducted primarily in regenerating forest and utility rights-of-way (Bulluck and Buehler, 2006). This is the first study to sample systematically over an early successional gradient in both silvicultural openings and wildlife openings maintained through prescribed burning and mechanical treatment. As such, it offers insight into management outcomes across a wider range of treatment types and stand ages than has been previously available.

Silviculture has been advocated as an important method for creating shrubland habitat (Thompson and DeGraaf 2001, DeGraaf and Yamasaki 2003) because unlike wildlife openings, silviculture can generate income from the sales of timber. The results of this study indicate that both silvicultural and wildlife openings provide habitat for shrubland species of conservation concern in the GMNF. Moreover, previous studies reported that nest survival did not differ between mechanically treated openings and clearcuts in Massachusetts (King et al. 2009), nor between burned or mechanically treated openings in New Hampshire (Chandler et al. 2009a). This suggests that all opening types on the GMNF are capable of supporting breeding populations of shrubland birds.

Although nearly all species were present in each treatment type, I found strong evidence that wildlife openings supported a greater abundance of birds for all but one focal species. The extra economic cost of maintaining wildlife openings may therefore be warranted in the GMNF as a supplementary management strategy to silviculture, particularly given that silviculture is often unpopular with the public. The results of this study provide valuable data for managers regarding what mixture of these treatment options can most effectively balance between the costs and benefits of management while simultaneously reaching population goals. Moreover, should budgetary constraints limit future wildlife opening management, this data from this study can provide managers with an estimate of the degree to which they would need to increase silvicultural openings in order to continue to support similar population sizes for shrubland species of concern.

Because providing quality habitat for shrubland birds through the maintenance of permanent wildlife openings is costly, it is imperative that scarce management dollars are used effectively. Moreover, since silviculture will proceed in many managed forests for reasons other than wildlife, performing these cuts in a manner that is most beneficial to shrubland birds without limiting timber profits is also vital. The results of this study provide some guidelines by which managers may be able to improve the habitat value of both wildlife openings and silvicultural openings for shrubland birds. For instance, for most of the focal species, abundance was negatively related to retained tree cover. This suggests that managers can increase the number of shrubland birds that openings can support by removing as much overstory tree cover during treatments as possible. Although removing more trees may make openings incapable of also supporting forest

species during the breeding season, the value of these openings to shrubland birds may be more of a critical priority, since early-successional habitat constitutes such a small fraction of the landscape.

Managers may also be able to increase the number of shrubland species supported across the managed landscape by ensuring that most openings are allowed a return interval of at least 8 years between treatments. It is more expensive to allow a long return interval between treatments, because it is more difficult to treat sites after a significant cover of woody vegetation has developed (Overcash et al. 1989). However, most of the species in this study peaked 4-7 years post-treatment, when shrub cover was relatively abundant, and bare ground was still sparse. Indeed, many previous studies have indicated that this dense woody cover is essential for many shrubland species (Thompson and Capen 1988, Hagan and Meehan 2002, Holmes and Pitt 2007, Schlossberg and King 2007, Chandler et al. 2009a, Schlossberg et al. 2010). Yet some shrubland species also require a dense layer of herbaceous cover (Guzy and Ritchison 1999, Arcese et al. 2002, Confer and Pascoe 2003, Holmes and Pitt 2007, Falls and Kopachena 2010, Schlossberg et al. 2010) and peak in abundance in the earlier successional stages when this cover is more abundant than woody vegetation. Supporting species such as White-throated Sparrow, Song Sparrow, and Indigo Bunting requires that some portion of the landscape is always in the earliest series of post-disturbance. Thus, the timing of treatments is an important tool through which managers can ensure that the habitat needs of all species are met.

Although the results suggest that managers may not be able to control shrub cover through the type of treatment executed per se, they can influence the cover of forb, fern,

and grass cover through their choice of treatment type. Habitat with grassy cover can be encouraged through mechanical treatment; cover of ferns can be promoted through prescribed fire and forb cover through both mechanical treatments and prescribed burning.

The slower a site regenerates, the longer it will provide habitat for shrubland species. If managers can control the rate of succession, they can increase the time interval between treatments, and potentially reduce management costs. Succession rates are known to be linked to the presence of advanced regeneration, because sites with more saplings and tree cover can shade out the shrub layer more quickly (Thompson and DeGraaf 2001, DeGraaf and Yamasaki 2003). As such, reducing the amount of retained basal area during treatment may not only increase the number of birds supported, but may and reduce the rate of succession, and increase the time period over which wildlife openings and silvicultural openings can support shrubland birds.

In contrast to what has been observed in previous work, wildlife openings and silvicultural openings on the GMNF seemed to be currently regenerating at a similar rate. Interactions between treatment type and time since were not supported in the GMNF, and understory vegetation height did not vary among treatments. Previous work has suggested that some opening types may provide suitable habitat for a longer period of time than silvicultural openings (Thompson and DeGraaf 2001, DeGraaf and Yamasaki 2003). For instance, Askins (2001) noted that “old field” succession happens more slowly than forest regeneration. Habitat composition is known to be heavily influenced by prior land use (Motzkin et al. 1996), and sites with a lower site quality index are likely to support shrubland birds longer (DeGraaf and Yamasaki 2003, Schlossberg and King 2007). This

suggests that managers may be able to reduce the economic cost of managing future wildlife openings on the GMNF by considering land use history when siting new wildlife openings. For instance, siting opening in areas where succession is likely to proceed more slowly, such as xeric sites, frost pockets, areas with shallow soils (DeGraaf and Yamasaki 2003), or in abandoned beaver meadows, where woody shrub cover has already been significantly reduced may increase the time period over which future wildlife openings can support birds. This may be a particularly valuable approach in the GMNF, where species peaked earlier in wildlife openings relative to what has been observed in silvicultural openings (Schlossberg and King 2009). However further study may be warranted in order to determine the degree to which site quality influences food availability and productivity for shrubland birds.

Table 1. Candidate general and generalized linear models with delta AICc of two or less for models representing the effect of management regime on habitat composition. Parameters include an intercept, representing the effect of the burn treatment (B0), the effect of clearcut (CC) and mechanical treatment (M), time since treatment (TS), and a quadratic term for time since treatment (Q). General linear models were used for understory vegetation height, and generalized linear models with a log link for cover variables. The estimates with 95% confidence intervals that do not overlap zero are in bold type, and Z scores (Z) comparing the CC and M parameter estimates that are statistically significant at an alpha of 0.05 are in bold type. Data are from vegetation surveys conducted in point-count plots in burned and mechanically treated wildlife openings and even-aged silvicultural openings on the Green Mountain National Forest, Vermont in 2010 and 2011.

VARIABLE	YEAR	B ₀	CC	M	TS	Q	Z	K ^a	ΔAIC ^b	W _i ^c	R ² INDEX ^d
Bare Ground	2010	-1.32	0.95	0.23	-0.16	0.02	0.46	5	0.00	0.40	0.24
		-0.74			-0.20	0.02		3	0.86	0.26	0.19
		-2.27	1.09	0.35	0.15		1.04	4	0.96	0.25	0.21
	2011	-1.09	2.03	0.73	-0.38	0.03	1.59	5	0.00	0.86	0.27
Woody Vegetation	2010	2.36			0.02			2	0.00	0.40	0.06
		2.24			0.10	0.26		3	1.01	0.24	0.06
	2011	2.52						1	0.00	0.45	0.00
		2.34			0.06	-0.004		3	1.69	0.19	0.03
Tall Woody Vegetation	2010	0.92			0.20	-0.01		3	0.00	0.45	0.20
		1.26			0.09			2	0.38	0.37	0.18
	2011	-0.05			0.36	-0.02		3	0.00	0.55	0.27
		-0.20	0.27	-0.12	0.40	-0.02	1.21	5	0.90	0.35	0.29
Short Woody Vegetation	2010	2.09			-0.06			2	0.00	0.48	0.12
		1.92			0.01	-0.004		3	1.11	0.28	0.13
	2011	2.53			-0.09			2	0.00	0.46	0.25
		2.32			-0.01	-0.01		3	0.85	0.30	0.26

VARIABLE	YEAR	B ₀	CC	M	TS	Q	Z	K ^a	ΔAIC ^b	W _i ^c	R ² INDEX ^d
Forb Cover	2010	0.93	-0.61	0.41			-2.27	3	0.00	0.58	0.11
		1.02	-0.56	0.43	-0.02		-2.17	4	1.91	0.22	0.10
	2011	0.38						1	0.00	0.35	0.00
		0.55	-0.44	-0.14			-0.89	3	0.84	0.23	0.04
		0.47			-0.01			2	1.83	0.14	0.00
Grass Cover	2010	0.76	-0.01	0.76	-0.13		-1.11	4	0.00	0.45	0.16
		1.13			-0.14			2	1.01	0.27	0.11
	2011	1.03	-0.06	0.85	-0.09		-2.28	4	0.00	0.67	0.22
		1.26	-0.08	0.87	-0.18	0.01	-2.37	5	1.63	0.30	0.22
Fern Cover	2010	1.10	-0.38	-0.79			0.88	3	0.00	0.37	0.05
		0.75						1	0.90	0.24	0.00
		1.25	-0.36	-0.77	-0.02		0.86	4	1.65	0.16	0.06
2011	1.13	-0.55	-0.21			-0.93	3	0.00	0.36	0.05	
	-1.71						1	0.36	0.30	0.00	
Understory Vegetation Height	2010	96.96			5.35			2	0.00	0.52	0.20
		83.48			10.48	-0.34		3	0.65	0.38	0.21
	2011	42.20			17.99	-0.78		3	0.00	0.82	0.28

a Number of parameters in model

b Akaike's Information Criteria adjusted for small sample size

c AIC_c weights, calculated from all fitted models

d Index from Nagelkerke (1991)

Table 2. Mean, median, and standard error of basal area values for retained deciduous (D) and coniferous (C) trees. Median values were compared using non-parametric Mann-Whitney tests, and in each row of the table, median values with a different superscript are statistically different at an α of 0.05. Data are from vegetation surveys conducted in point-count plots in burned and mechanically treated wildlife openings and even-aged silvicultural openings on the Green Mountain National Forest, Vermont in 2010 and 2011.

Basal Area	Year	Burned Wildlife Openings			Silvicultural Openings			Mechanically Treated Wildlife Openings		
		Mean	Median	SE	Mean	Median	SE	Mean	Median	SE
D	2010	2.91	0.82 ^b	0.85	6.38	4.24 ^a	0.94	3.77	2.66 ^b	0.70
	2011	2.45	0.75 ^c	0.68	6.41	5.24 ^a	0.93	4.03	3.21 ^b	0.76
C	2010	0.48	0.00 ^b	0.21	1.53	0.38 ^a	0.25	1.17	0.06	0.38
	2011	0.50	0.00 ^b	0.20	1.75	0.38 ^a	0.60	1.25	0.17	0.49

Table 3. Species counts for all males detected within point-count survey plots in wildlife openings and even-aged silvicultural openings on the Green Mountain National Forest, Vermont in 2010 and 2011. Counts for Cedar Waxwings include both males and females. Species in bold type are shrubland birds, as designated by Schlossberg and King (2007).

COMMON NAME	CODE	SCIENTIFIC NAME	2010	2011
Alder Flycatcher	ALFL	<i>Empidonax alnorum</i>	87	86
American Crow	AMCR	<i>Corvus brachyrhynchos</i>	0	4
American Goldfinch	AMGO	<i>Spinus tristis</i>	27	44
American Redstart	AMRE	<i>Setophaga ruticilla</i>	60	24
American Robin	AMRO	<i>Turdus migratorius</i>	20	30
Baltimore Oriole	BAOR	<i>Icterus galbula</i>	1	1
Black-and-white Warbler	BAWW	<i>Mniotilta varia</i>	21	30
Black-capped Chickadee	BCCH	<i>Poecile atricapillus</i>	54	35
Blue-gray Gnatcatcher	BGGN	<i>Poliophtila caerulea</i>	1	1
Brown-headed Cowbird	BHCO	<i>Molothrus ater</i>	3	2
Blue-headed Vireo	BHVI	<i>Vireo solitarius</i>	1	1
Blackburnian Warbler	BLBW	<i>Setophaga fusca</i>	95	70
Blue Jay	BLJA	<i>Cyanocitta cristata</i>	26	19
Brown Creeper	BRCR	<i>Certhia americana</i>	3	6
Black-throated Blue Warbler	BTBW	<i>Setophaga caerulescens</i>	47	39
Black-throated Green Warbler	BTNW	<i>Setophaga virens</i>	17	32
Canada Warbler	CAWA	<i>Cardellina canadensis</i>	33	18
Cedar Waxwing	CEDW	<i>Bombycilla cedrorum</i>	84	143
Cerulean Warbler	CERW	<i>Setophaga cerulea</i>	1	0
Chipping Sparrow	CHSP	<i>Spizella passerina</i>	12	4
Common Raven	CORA	<i>Corvus corax</i>	0	1
Common Yellowthroat	COYE	<i>Geothlypis trichas</i>	343	313
Chestnut-sided Warbler	CSWA	<i>Setophaga pensylvanica</i>	379	508
Dark-eyed Junco	DEJU	<i>Junco hyemalis</i>	38	31
Downy Woodpecker	DOWO	<i>Picoides pubescens</i>	4	3
Eastern Kingbird	EAKI	<i>Tyrannus tyrannus</i>	0	3
Eastern Phoebe	EAPH	<i>Sayornis phoebe</i>	1	1
Eastern Towhee	EATO	<i>Pipilo erythrophthalmus</i>	15	16
Eastern Wood Pewee	EAWP	<i>Contopus virens</i>	4	5
Field Sparrow	FISP	<i>Spizella pusilla</i>	1	0
Great Crested Flycatcher	GCFL	<i>Myiarchus crinitus</i>	4	8
Golden-crowned Kinglet	GCKI	<i>Regulus satrapa</i>	5	1
Gray Catbird	GRCA	<i>Dumetella carolinensis</i>	26	16
Hairy Woodpecker	HAWO	<i>Picoides villosus</i>	12	5
Hermit Thrush	HETH	<i>Catharus guttatus</i>	1	5
House Wren	HOWR	<i>Troglodytes aedon</i>	0	1
Indigo Bunting	INBU	<i>Passerina cyanea</i>	73	119

COMMON NAME	CODE	SCIENTIFIC NAME	2010	2011
Least Flycatcher	LEFL	<i>Empidonax minimus</i>	14	23
Magnolia Warbler	MAWA	<i>Setophaga magnolia</i>	30	13
Mourning Dove	MODO	<i>Zenaida macroura</i>	1	1
Mourning Warbler	MOWA	<i>Geothlypis philadelphia</i>	65	51
Nashville Warbler	NAWA	<i>Oreothlypis ruficapilla</i>	26	17
Northern Cardinal	NOCA	<i>Cardinalis cardinalis</i>	1	1
Northern Flycatcher	NOFL	<i>Colaptes auratus</i>	5	4
Northern Mockingbird	NOMO	<i>Mimus polyglottos</i>	1	0
Northern Parula	NOPA	<i>Setophaga americana</i>	1	2
Northern Waterthrush	NOWA	<i>Parkesia noveboracensis</i>	4	1
Ovenbird	OVEN	<i>Seiurus aurocapillus</i>	13	46
Pine Warbler	PIWA	<i>Setophaga pinus</i>	2	4
Pileated Woodpecker	PIWO	<i>Dryocopus pileatus</i>	4	1
Purple Finch	PUFI	<i>Carpodacus purpureus</i>	1	1
Red-breasted Grosbeak	RBGR	<i>Pheucticus ludovicianus</i>	21	22
Ruby-crowned Kinglet	RCKI	<i>Regulus calendula</i>	0	16
Red-eyed Vireo	REVI	<i>Vireo olivaceus</i>	91	106
Ruby-throated Hummingbird	RTHU	<i>Archilochus colubris</i>	10	18
Ruffed Grouse	RUGR	<i>Bonasa umbellus</i>	11	5
Red-winged Blackbird	RWBL	<i>Agelaius phoeniceus</i>	4	3
Scarlet Tanager	SCTA	<i>Piranga olivacea</i>	3	3
Song Sparrow	SOSP	<i>Melospiza melodia</i>	40	42
Swainson's Thrush	SWTH	<i>Catharus ustulatus</i>	5	10
Tree Swallow	TRSW	<i>Tachycineta bicolor</i>	3	11
Tufted Titmouse	TUTI	<i>Baeolophus bicolor</i>	0	1
Veery	VEER	<i>Catharus fuscescens</i>	10	8
Warbling Vireo	WAVI	<i>Vireo gilvus</i>	1	2
White-breasted Nuthatch	WBNU	<i>Sitta carolinensis</i>	2	3
Wild Turkey	WITU	<i>Meleagris gallopavo</i>	0	1
Winter Wren	WIWR	<i>Troglodytes troglodytes</i>	23	15
Wood Thrush	WOTH	<i>Hylocichla mustelina</i>	3	3
White-throated Sparrow	WTSP	<i>Zonotrichia albicollis</i>	123	120
Yellow-bellied Sapsucker	YBSA	<i>Sphyrapicus varius</i>	6	2
Yellow Warbler	YEWA	<i>Setophaga petechia</i>	2	3
Yellow-rumped Warbler	YRWA	<i>Setophaga coronata</i>	8	6

Table 4. Management variables in N-mixture models of bird abundance with delta AIC_c values of two or less. Management variables include an intercept, representing the effect of the burn treatment (B₀), effects for clearcut (CC) and mechanical treatment (M), time since treatment (TS), a quadratic term for time since treatment (Q), and basal area of retained deciduous (DT) and coniferous trees (CT). The estimates with 95% confidence intervals that do not overlap zero are in bold type. Z scores (Z) for comparisons of the CC and M parameter estimates that are statistically significant at an alpha of 0.05 are in bold type. Data are from point-count surveys conducted in burned and mechanically treated wildlife openings and even-aged silvicultural openings on the Green Mountain National Forest, Vermont in 2010 and 2011.

SPECIES ^a	2010	B ₀	CC	M	TS	Q	DT	CT	Z	K ^b	AIC _c	ΔAIC _c ^c	W _i ^d	R ²	INDEX ^e
ALFL	2010	0.94	-1.62	-0.24	-0.51	-0.71	-0.22	-0.36	-2.24	11	329.89	0.00	0.51	0.56	
		0.86	-1.74	-0.20	-0.55	-0.65		-0.43	-2.51	10	330.82	0.93	0.32	0.55	
ALFL	2011	0.18			-0.44	-0.72	-0.32	-0.45		11	338.17	0.00	0.61	0.39	
CEDW	2010	1.88	-1.07	0.19	-0.69	-0.64			-2.21	11	383.19	0.00	0.31	0.38	
		1.83	-0.97	0.21	-0.64	-0.65	-0.16		-2.05	12	384.20	1.01	0.18	0.39	
		1.93	-1.14	0.09	-0.67	-0.66		0.13	-2.10	12	384.53	1.34	0.16	0.38	
		1.90	-1.03	0.08	-0.59	-0.68	-0.23	0.20	-1.89	13	384.82	1.63	0.13	0.39	
CEDW	2011	2.19	-0.71	-0.45	-0.41	-0.24			-0.54	13	552.41	0.00	0.13	0.45	
		1.86			-0.41	-0.25				11	552.96	0.55	0.10	0.42	
		2.02	-0.73	-0.44	-0.26				-0.59	12	553.54	1.13	0.07	0.43	
		2.09	-0.71	-0.44					-0.56	11	553.57	1.16	0.07	0.42	
		2.15	-0.71	-0.42	-0.36	-0.23	-0.13		-0.60	14	553.58	1.17	0.07	0.45	
		2.02	-0.72	-0.39				-0.20	-0.67	12	553.61	1.20	0.07	0.43	
		1.83			-0.36	-0.24	-0.15			12	553.96	1.55	0.06	0.43	
		1.68			-0.26					10	554.15	1.74	0.05	0.40	
		1.98	-0.73	-0.40	-0.21			-0.16	-0.66	13	554.29	1.88	0.05	0.44	
		2.17	-0.70	-0.44	-0.41	-0.24			-0.04	-0.54	14	554.35	1.94	0.05	0.45
COYE	2010	1.63	-0.89	-0.12	-0.32	-0.36	-0.19		-2.81	10	718.83	0.00	0.64	0.60	
		1.62	-0.88	-0.11	-0.32	-0.35	-0.18	-0.04	-2.80	11	720.64	1.80	0.26	0.60	
COYE	2011	1.76	-0.53	0.02	-0.26	-0.28	-0.23		-2.36	13	717.70	0.00	0.44	0.60	
		1.77	-0.49	0.04	-0.28	-0.28	-0.19	-0.11	-2.33	14	717.75	0.06	0.43	0.61	

SPECIES	2010	B ₀	CC	M	TS	Q	DT	CT	Z	K ^b	AIC _c	ΔAIC _c ^c	W _i ^d	R ² INDEX ^e
CSWA	2010	1.58	-0.43	-0.15	-0.15	-0.38	-0.18	-0.16	-1.13	11	749.31	0.00	0.46	0.66
		1.41			-0.18	-0.41	-0.22	-0.18		9	750.56	1.25	0.24	0.64
		1.63	-0.48	-0.20	-0.15	-0.40	-0.22		-1.12	10	751.21	1.90	0.18	0.65
CSWA	2011	1.75	-0.48	0.14	-0.23	-0.35	-0.21		-2.91	10	820.30	0.00	0.63	0.56
		1.74	-0.46	0.16	-0.24	-0.35	-0.19	-0.06	-2.89	11	821.49	1.19	0.35	0.56
INBU	2010	0.78	-1.41	0.63	-0.61	-0.49		-0.47	-2.94	12	281.66	0.00	0.49	0.66
		0.75	-1.32	0.64	-0.55	-0.49	-0.12	-0.45	-5.34	13	282.95	1.29	0.26	0.66
INBU	2011	0.64	-1.00	0.01	-0.52	-0.29		-0.28	-2.23	13	387.42	0.00	0.34	0.58
		0.62	-1.00	-0.02	-0.47	-0.28	-0.12	-0.23	-2.17	14	388.61	1.19	0.19	0.58
		0.66	-1.07	-0.08	-0.43	-0.27	-0.21		-2.19	13	389.09	1.67	0.15	0.57
MOWA	2010	0.42			-0.34	-0.25				6	327.69	0.00	0.18	0.09
		0.17			-0.18					5	328.16	0.47	0.14	0.07
		0.25								2	328.52	0.83	0.12	0.00
		0.38			-0.34	-0.23		-0.12		4	328.98	1.29	0.09	0.10
		0.15			-0.19			-0.15		6	329.09	1.40	0.09	0.08
		0.42			-0.32	-0.25	-0.10			7	329.12	1.43	0.09	0.10
		0.17			-0.15		-0.10			6	329.65	1.97	0.07	0.07
MOWA	2011	-0.01	0.89	0.13	-0.58	-1.03			1.18	10	246.18	0.00	0.34	0.38
		-0.05	0.94	0.17	-0.58	-1.02		-0.11	1.18	11	247.73	1.55	0.16	0.39
		0.34			-0.55	-0.91				8	247.96	1.78	0.14	0.34
		0.00	0.87	0.12	-0.60	-1.04	0.06		1.16	11	248.06	1.88	0.13	0.38

SPECIES	2010	B ₀	CC	M	TS	Q	DT	CT	Z	K ^b	AIC _c	ΔAIC _c ^c	W _i ^d	R ² INDEX ^e	
SOSP	2010	-0.80			-0.80	-0.71	-0.42	-0.51		8	172.59	0.00	0.10	0.21	
		-0.68	-1.47	0.04				-0.63	-1.35	7	172.65	0.06	0.09	0.19	
		-0.73			-0.81	-0.64		-0.62		7	172.74	0.16	0.09	0.19	
		-0.71			-0.83	-0.77	-0.55			7	172.90	0.32	0.08	0.19	
		-0.57	-1.32	0.08	-0.78	-0.59		-0.59	-1.28	9	173.57	0.99	0.06	0.22	
		-0.91	-1.23	0.24				-0.32	-0.56	-1.26	8	173.59	1.00	0.06	0.20
		-1.11			-0.29			-0.69			6	173.71	1.12	0.06	0.16
		-0.89	-1.46	0.12	-0.33			-0.63	-1.39		8	173.82	1.23	0.05	0.20
		-1.23			-0.27			-0.39	-0.58		7	174.05	1.46	0.05	0.18
		-0.74	-1.10	0.28	-0.80	-0.66	-0.37	-0.52	-1.19		10	174.11	1.52	0.05	0.24
		-0.46	-1.74	-0.17					-1.42		6	174.12	1.53	0.04	0.16
		-0.80	-1.35	0.14				-0.44		-1.27	7	174.12	1.54	0.04	0.18
		-0.60	-1.15	0.18	-0.81	-0.70	-0.47			-1.16	9	174.43	1.85	0.04	0.21
		SOSP	2011	-0.58			-1.21	-1.05		-1.73		7	171.20	0.00	0.54
-0.61					-1.10	-1.07	-0.23	-1.65		8	172.62	1.42	0.26	0.29	
WTSP	2010	0.74	-0.76	-0.24	-0.54		-0.19		-1.36	9	447.12	0.00	0.25	0.49	
		0.78	-0.78	-0.27	-0.53		-0.24	0.12	-1.32	10	448.05	0.93	0.16	0.49	
		0.81	-0.94	-0.29	-0.59				-1.74	8	448.27	1.15	0.14	0.47	
		0.75	-0.73	-0.23	-0.58	-0.04	-0.20		-1.26	10	449.04	1.92	0.10	0.49	
WTSP	2011	0.83	-0.65	-0.57	-0.75	-0.18			-0.20	11	431.90	0.00	0.20	0.49	
		0.82	-0.59	-0.55	-0.71	-0.17	-0.13		-0.10	12	432.78	0.88	0.13	0.49	
		0.70	-0.64	-0.64	-0.54				-0.01	10	432.98	1.08	0.12	0.47	
		0.87	-0.60	-0.54	-0.67	-0.18	-0.21	0.15	-0.13	13	433.28	1.38	0.10	0.50	
		0.85	-0.67	-0.58	-0.74	-0.18		0.07	-0.24	12	433.55	1.65	0.09	0.49	
		0.69	-0.57	-0.61	-0.51		-0.14	0.09	11	433.68	1.78	0.08	0.48		

a Common and scientific names in Table 3

b Number of parameters in model, including two intercepts

c Akaike's Information Criteria adjusted for small sample size

d AIC_c weights, calculated from all fitted models

e Index from Nagelkerke (1991)

Table 5. Estimates of mean detectability (p) and parameter estimates for detection covariates in N-mixture models of bird abundance with delta AICc values of two or less. Detection variables include an intercept (B_0), date (D), observers (OBS1, OBS 2, OBS3, and OBS4), understory vegetation height (V), a quadratic term for date (D^2), wind (W), and time (T). Parameters in bold type have 95% confidence intervals that do not overlap zero. Mean detectability was calculated for each model by using the model to predict detectability during all survey events (288 in 2010, 279 in 2011), and subsequently taking the mean of the predicted values for each model. Data are from point-count surveys conducted in burned and mechanically treated wildlife openings and even-aged silvicultural openings on the Green Mountain National Forest, Vermont in 2010 and 2011.

SPECIES ^a	YEAR	p	B_0	D	OBS1	OBS2	OBS3	OBS4	V	D^2	W	T
ALFL	2010	0.27	-1.87	-0.49	0.87	1.00						
		0.27	-1.76	-0.47	0.88	1.06						
ALFL	2011	0.34	-1.12		0.80		0.47	-0.43				
CEDW	2010	0.09	-2.87	0.78	0.74	0.79				-0.24		
		0.09	-2.86	0.77	0.75	0.78				-0.25		
		0.09	-2.86	0.79	0.73	0.80				-0.23		
		0.09	-2.84	0.79	0.74	0.78				-0.24		
CEDW	2011	0.09	-2.11	-0.40	-0.01		-0.70	1.02	-0.46			-0.36
		0.09	-2.13	-0.41	0.03		-0.67	1.06	-0.52			-0.37
		0.09	-2.17	-0.40	0.01		-0.66	1.03	-0.44			-0.35
		0.09	-2.24	-0.38	0.04		-0.64	1.05	-0.61			-0.36
		0.09	-2.09	-0.39	-0.01		-0.71	0.99	-0.46			-0.37
		0.10	-2.19	-0.36	0.02		-0.66	1.00	-0.57			-0.37
		0.09	-2.12	-0.39	0.02		-0.67	1.02	-0.53			-0.38
		0.09	-2.19	-0.41	0.04		-0.62	1.05	-0.50			-0.36
		0.09	-2.14	-0.38	0.00		-0.67	0.99	-0.44			-0.36
		0.09	-2.10	-0.40	-0.01		-0.71	1.02	-0.46			-0.37
COYE	2010	0.39	-0.81	0.18	0.53	0.51						
		0.39	-0.82	0.18	0.53	0.51						
COYE	2011	0.28	-1.26	-0.11	0.65		0.04	0.31	-0.38	-0.12		
		0.27	-1.30	-0.12	0.62		0.04	0.29	-0.38	-0.11		

SPECIES^a	YEAR		B₀	D	OBS1	OBS2	OBS3	OBS4	V	D²	W	T
CSWA	2010	0.42	-0.81		0.90	1.06					-0.17	
		0.43	-0.78		0.92	1.05					-0.18	
		0.43	-0.79		0.90	1.07					-0.17	
CSWA	2011	0.44	-0.64		0.58		0.45	0.37				
		0.44	-0.64		0.57		0.44	0.36				
INBU	2010	0.15	-3.13	0.32	1.19	0.92					-1.41	0.57
		0.15	-3.14	0.30	1.19	0.90					-1.43	0.57
INBU	2011	0.30	-0.77	-0.38	0.37		-0.60	-0.49	-1.08	-0.22		
		0.31	-0.71	-0.37	0.36		-0.61	-0.53	-1.08	-0.23		
		0.31	-0.70	-0.36	0.39		-0.59	-0.51	-1.07	-0.25		
MOWA	2010	0.18	-2.01		0.64	0.76						
		0.19	-1.97		0.63	0.76						
		0.18	-1.54		0.63	0.76						
		0.19	-1.99		0.64	0.76						
		0.19	-1.96		0.62	0.77						
		0.18	-2.01		0.64	0.75						
		0.19	-1.97		0.63	0.76						
MOWA	2011	0.24	-0.59		-0.11		-2.06	-1.68	-0.46			
		0.24	-0.58		-0.14		-2.08	-1.72	-0.45			
		0.25	-0.62		-0.08		-2.09	-1.65	-0.38			
		0.24	-0.58		-0.11		-2.07	-1.66	-0.46			

SPECIES ^a	YEAR	p	B ₀	D	OBS1	OBS2	OBS3	OBS4	V	D ²	W	T
SOSP	2010	0.37	-0.80						-1.56			
		0.34	-1.05						-1.90			
		0.36	-0.85						-1.65			
		0.38	-0.71						-1.51			
		0.36	-0.82						-1.61			
		0.35	-1.01						-1.87			
		0.35	-0.95						-1.76			
		0.36	-0.90						-1.71			
		0.36	-0.87						-1.67			
		0.36	-0.81						-1.58			
		0.36	-0.90						-1.78			
		0.36	-0.88						-1.77			
		0.38	-0.72						-1.52			
SOSP	2011	0.17	-1.74	-0.74								
		0.17	-1.70	-0.66								
WTSP	2010	0.22	-2.11		0.84	1.06					0.13	
		0.22	-2.14		0.85	1.05					0.13	
		0.23	-2.12		0.86	1.11					0.14	
		0.22	-2.11		0.85	1.06					0.13	
WTSP	2011	0.25	-1.96	0.25	1.06		0.54	0.87			0.18	
		0.24	-1.99	0.26	1.05		0.54	0.82			0.18	
		0.25	-1.94	0.26	1.06		0.58	0.90			0.19	
		0.23	-2.04	0.27	1.05		0.54	0.79			0.16	
		0.24	-1.97	0.25	1.06		0.54	0.87			0.18	
		0.24	-1.97	0.27	1.05		0.57	0.84			0.18	

a Common and scientific names in Table 3

Table 6. Occupancy (% of points occupied), mean abundance (detections/50 m radius count), and detectability-adjusted abundance (λ) for shrubland birds in burned and mechanically treated wildlife openings and even-aged silvicultural openings on the Green Mountain National Forest, Vermont, in 2010 and 2011.

SPECIES ^a	YEAR	OCC	Abund	SE	λ^b	SE
ALFL	2010	0.39	0.30	0.04	1.23	0.12
CEDW	2010	0.22	0.29	0.05	3.61	0.25
COYE	2010	0.77	1.19	0.07	3.05	0.18
CSWA	2010	0.82	1.32	0.08	3.11	0.16
INBU	2010	0.32	0.25	0.04	1.76	0.16
MOWA	2010	0.43	0.23	0.03	1.22	0.02
SOSP	2010	0.15	0.14	0.03	0.40	0.03
WTSP	2010	0.55	0.43	0.05	1.92	0.15
ALFL	2011	0.34	0.31	0.04	0.88	0.08
CEDW	2011	0.49	0.51	0.07	5.15	0.19
COYE	2011	0.76	1.12	0.07	4.23	0.21
CSWA	2011	0.86	1.81	0.09	4.10	0.21
INBU	2011	0.45	0.43	0.04	1.30	0.08
MOWA	2011	0.32	0.18	0.03	0.76	0.05
SOSP	2011	0.16	0.15	0.03	0.88	0.12
WTSP	2011	0.48	0.43	0.05	1.69	0.12

a Common and scientific names in Table 3

b Estimated abundance/50 m radius circle based on N-mixture models

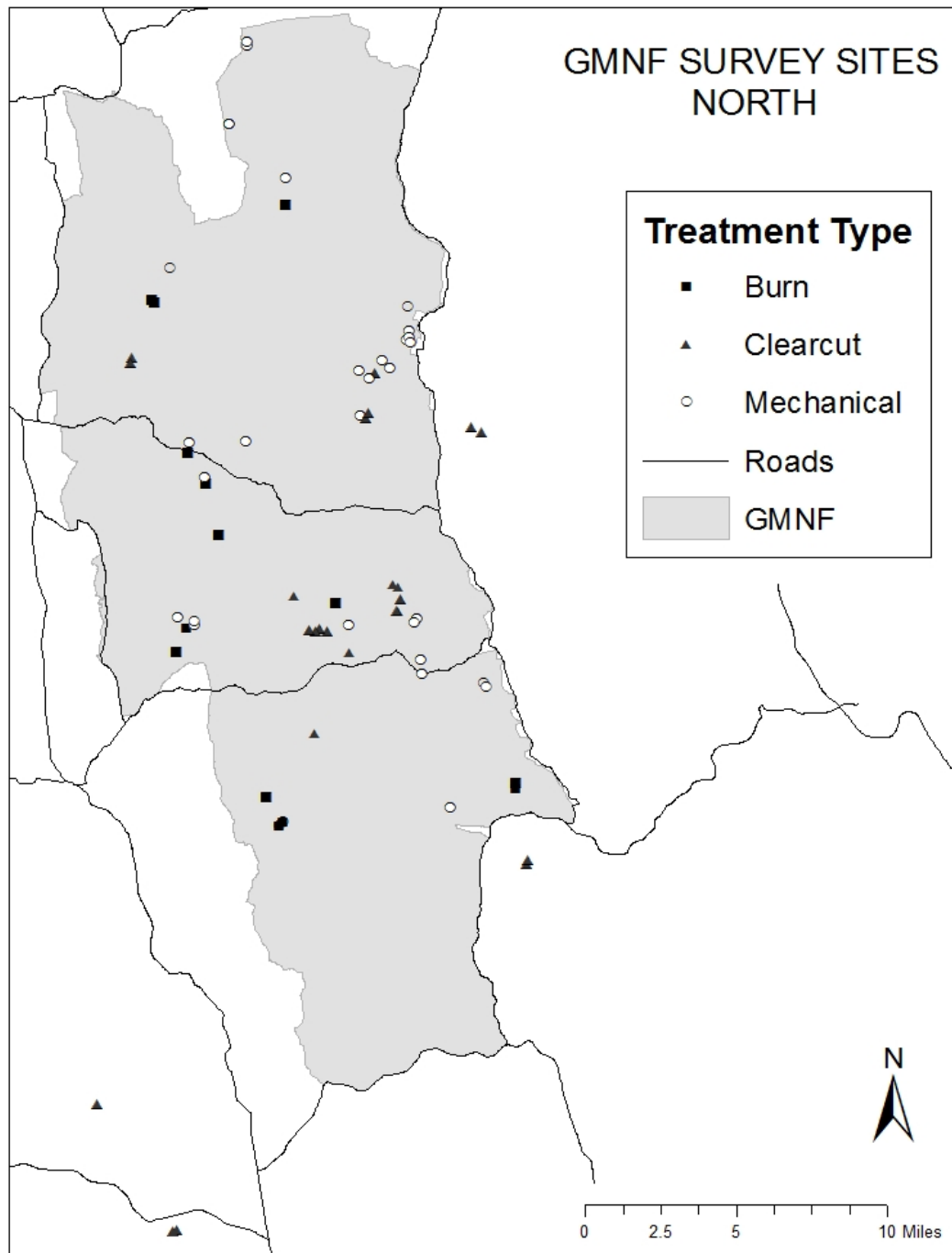
Table 7. Occupancy (% of points occupied), mean abundance (detections/50 m radius count), and detectability-adjusted abundance (λ) for shrubland birds partitioned by treatment type. Data are from burned and mechanically treated wildlife openings and even-aged silvicultural openings on the Green Mountain National Forest, Vermont in 2010 and 2011.

SPECIES ^a	YR	BURN					CLEARCUT					MECHANICAL				
		Occ	Abund	SE	λ^b	SE	Occ	Abund	SE	λ^b	SE	Occ	Abund	SE	λ^b	SE
ALFL	2010	0.67	0.52	0.08	2.22	0.19	0.09	0.05	0.02	0.24	0.03	0.45	0.38	0.07	1.23	0.19
CEDW	2010	0.37	0.43	0.09	4.83	0.30	0.06	0.12	0.05	1.39	0.16	0.26	0.34	0.09	4.95	0.45
COYE	2010	1.00	1.78	0.13	4.62	0.22	0.46	0.54	0.1	1.30	0.09	0.9	1.35	0.12	3.52	0.25
CSWA	2010	1.00	1.83	0.15	4.22	0.22	0.66	0.83	0.11	2.06	0.16	0.84	1.37	0.12	3.22	0.27
INBU	2010	0.50	0.4	0.08	2.23	0.16	0.06	0.05	0.03	0.31	0.04	0.45	0.34	0.07	2.94	0.32
MOWA	2010	0.43	0.23	0.05	1.33	0.04	0.43	0.22	0.04	1.14	0.04	0.42	0.23	0.05	1.22	0.04
SOSP	2010	0.23	0.28	0.07	0.58	0.05	0.03	0.04	0.02	0.18	0.02	0.19	0.12	0.04	0.48	0.05
WTSP	2010	0.57	0.66	0.11	2.78	0.27	0.43	0.24	0.05	1.11	0.14	0.68	0.42	0.07	1.97	0.27
ALFL	2011	0.44	0.43	0.08	1.07	0.12	0.18	0.20	0.06	0.69	0.12	0.43	0.3	0.06	0.90	0.15
CEDW	2011	0.59	0.73	0.14	7.07	0.29	0.36	0.30	0.12	4.11	0.20	0.54	0.51	0.12	5.18	0.20
COYE	2011	0.84	1.38	0.13	5.22	0.33	0.58	0.71	0.11	2.69	0.21	0.89	1.32	0.13	4.91	0.36
CSWA	2011	0.91	2.13	0.14	4.78	0.29	0.73	1.11	0.14	2.50	0.19	0.96	2.29	0.15	5.20	0.40
INBU	2011	0.69	0.57	0.08	1.82	0.12	0.15	0.15	0.05	0.56	0.05	0.54	0.58	0.09	1.57	0.11
MOWA	2011	0.22	0.15	0.04	0.60	0.06	0.45	0.24	0.05	0.94	0.12	0.29	0.15	0.05	0.72	0.08
SOSP	2011	0.19	0.18	0.06	1.10	0.21	0.09	0.14	0.06	0.69	0.20	0.21	0.13	0.04	0.86	0.21
WTSP	2011	0.59	0.65	0.10	2.33	0.24	0.39	0.33	0.07	1.27	0.14	0.46	0.3	0.07	1.44	0.17

a Common and scientific names in Table 3

b Estimated abundance/50 m radius circle based on N-mixture models

Figure 1. Maps of northern and southern survey sites in the Green Mountain National Forest (GMNF) Study Area. Maps include all sites surveyed in 2010 and 2011.



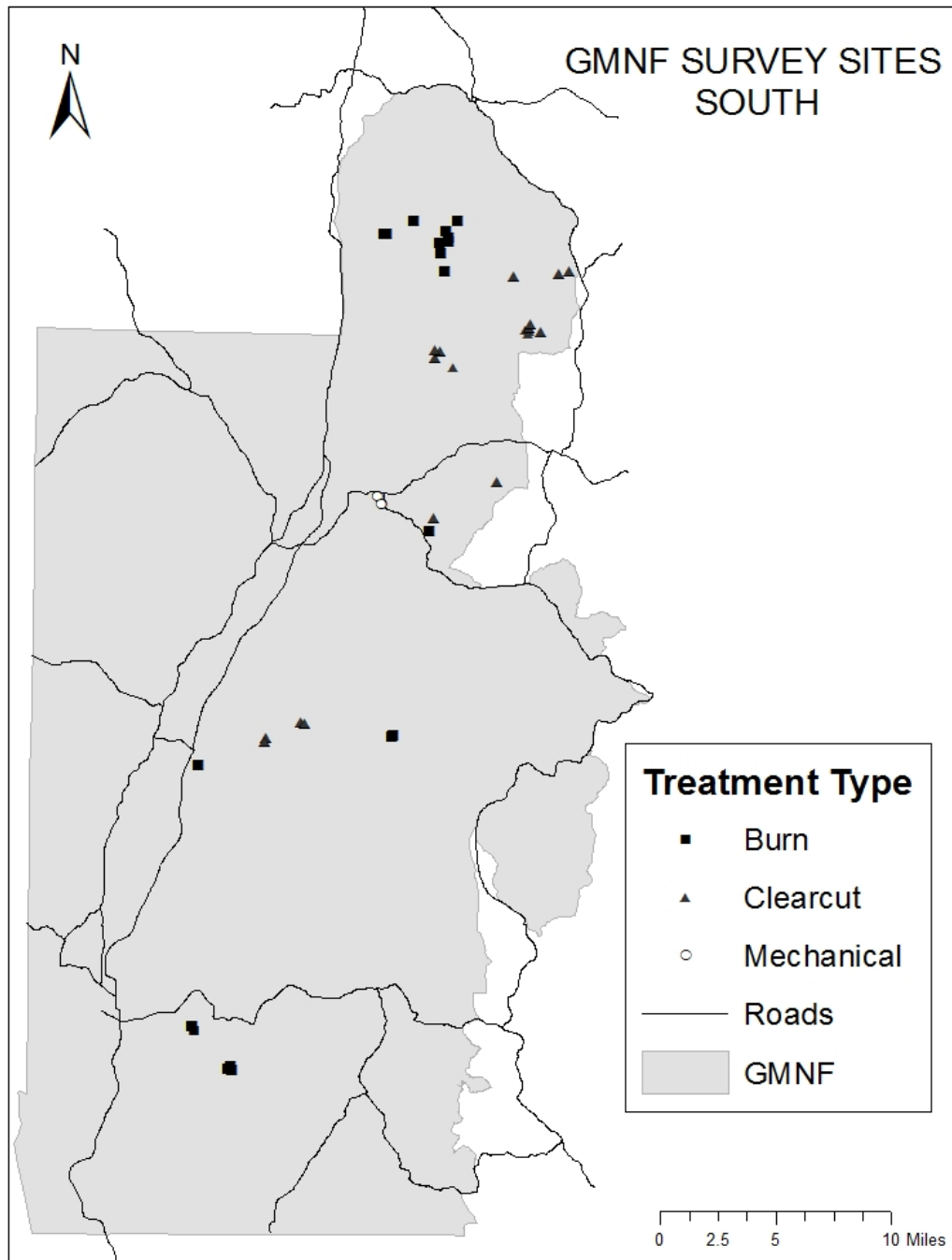


Figure 2. Box plots comparing basal area of retained coniferous and deciduous trees in even-aged silvicultural openings (CC), and burned (BURN) and mechanically treated wildlife openings (MECH) on the Green Mountain National Forest, Vermont, 2010 and 2011.

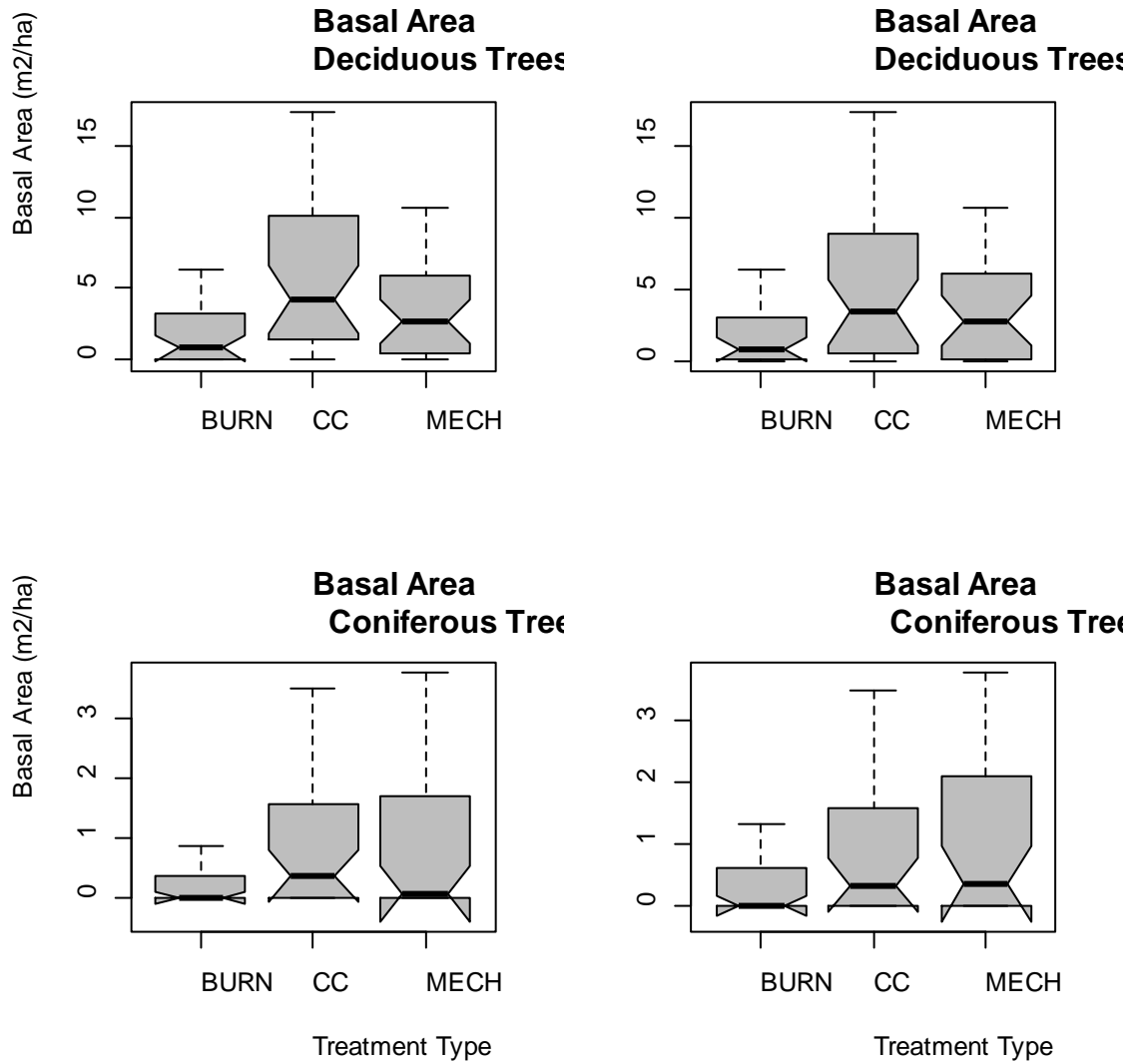
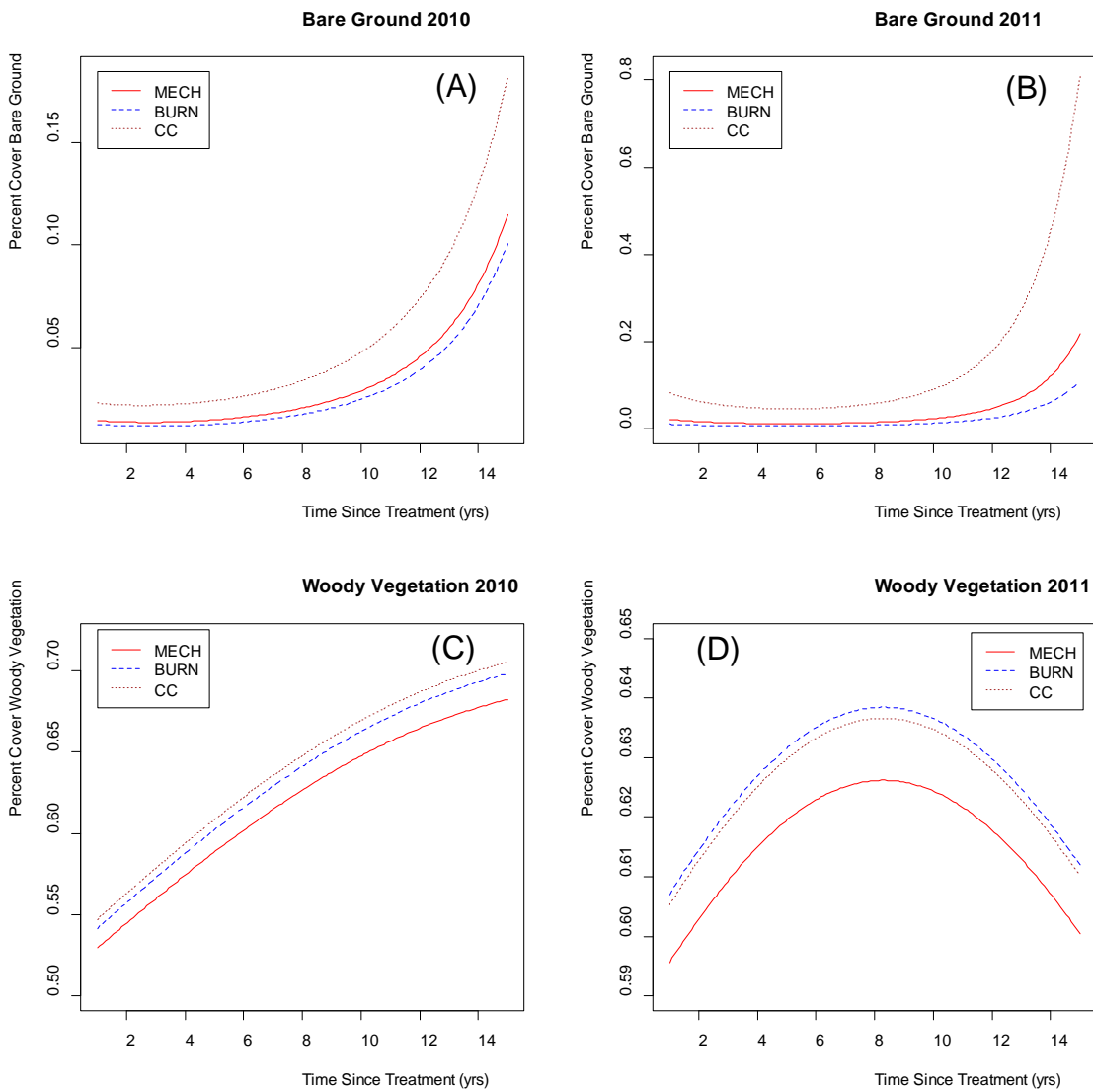
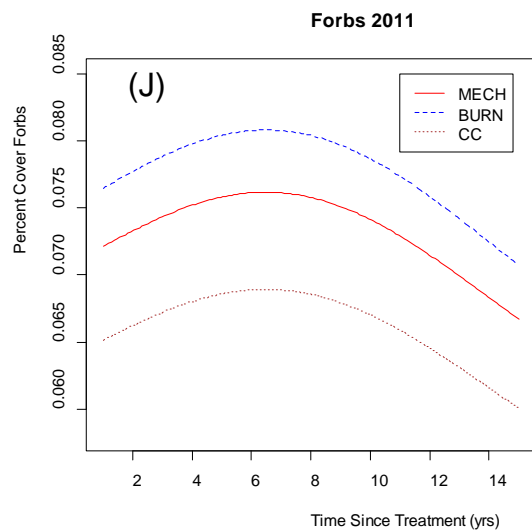
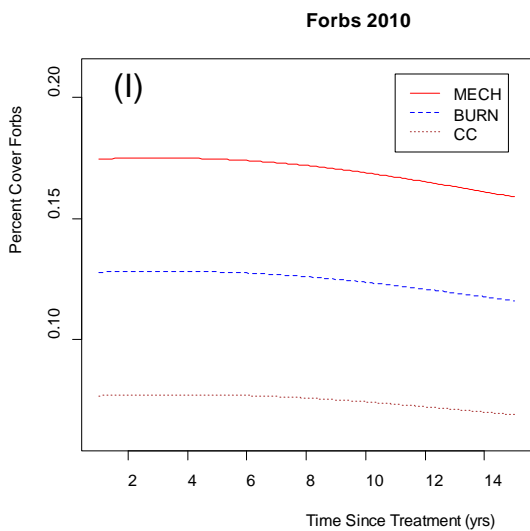
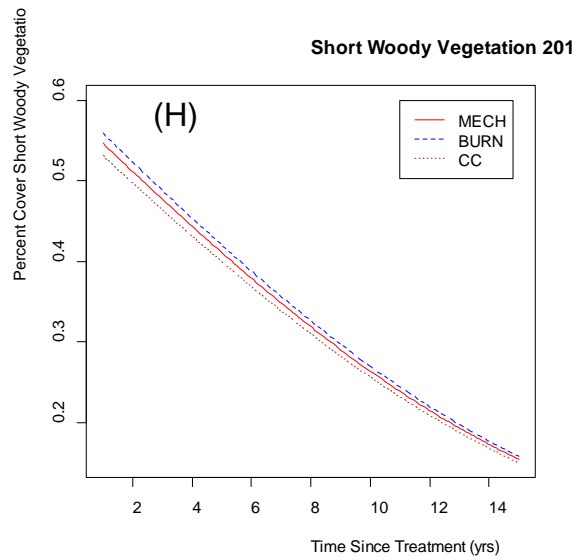
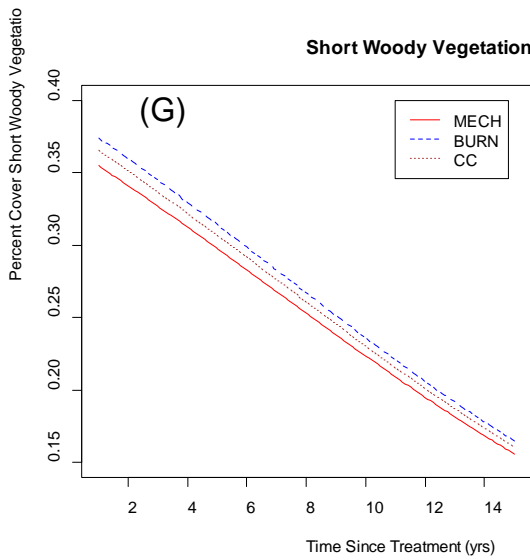
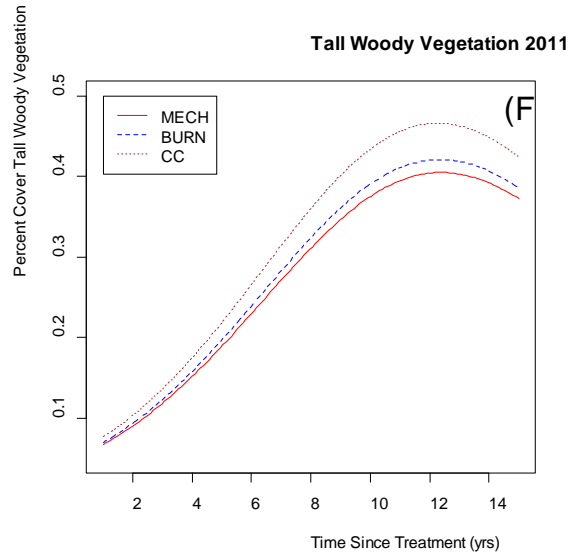
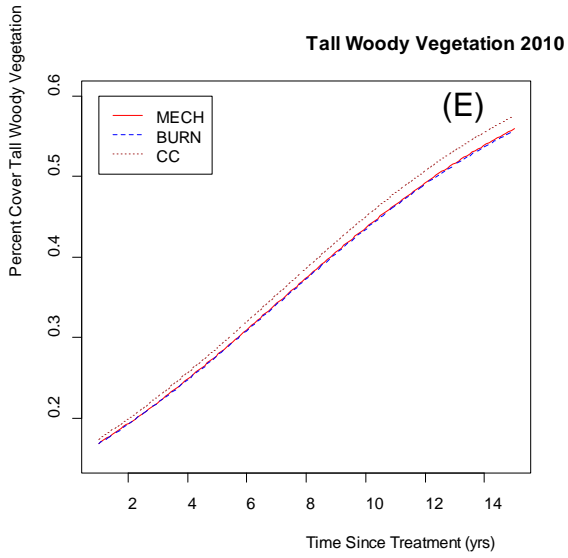


Figure 3. Model-averaged predictions from models relating habitat response variables and management regime. Strongly supported relationships include a treatment effect for Bare Ground (A), (B), Forb Cover (I), Grass Cover (K), (L), and Fern Cover (M), (N), and a time since treatment effect for Bare Ground (A), (B), Woody Vegetation (C), Tall Woody Vegetation (E), (F), Short Woody Vegetation (G), (H), Grass Cover (K), (L), Fern Cover (M), and Understory Vegetation Height (O), (P). Data are from vegetation surveys conducted in point-count plots in burned and mechanically wildlife openings and even-aged silvicultural openings on the Green Mountain National Forest, Vermont in 2010 and 2011.





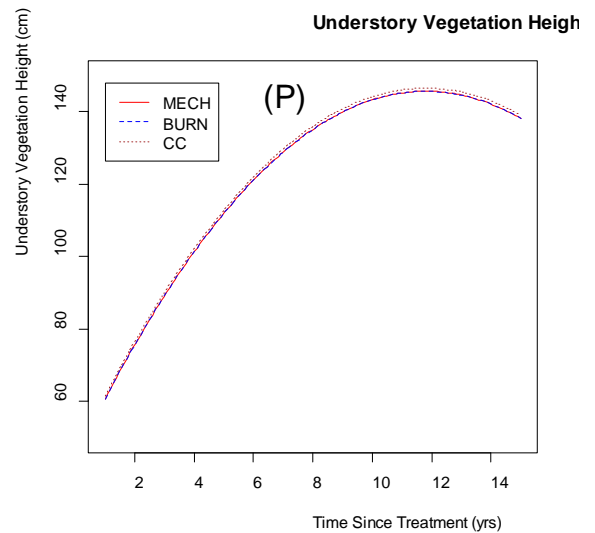
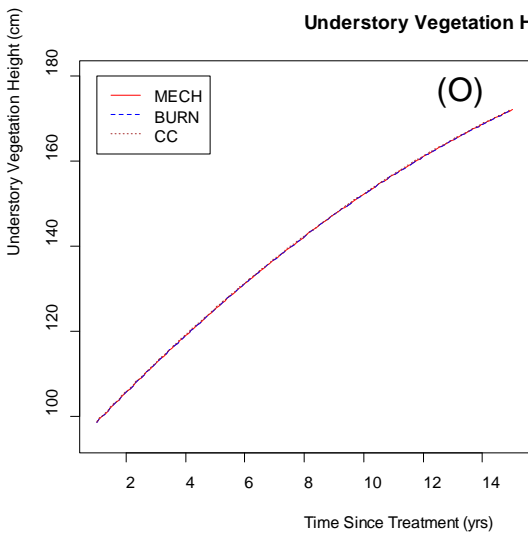
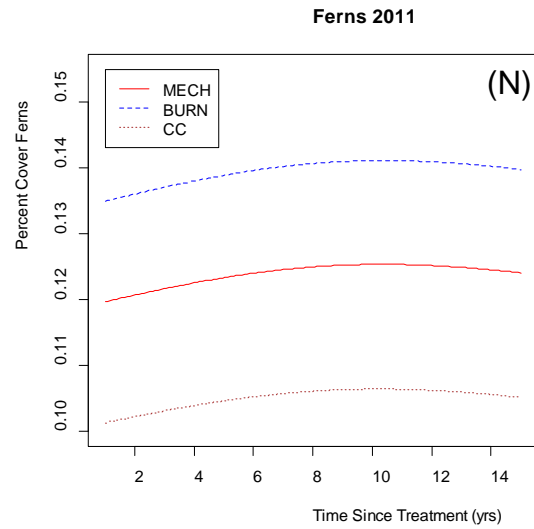
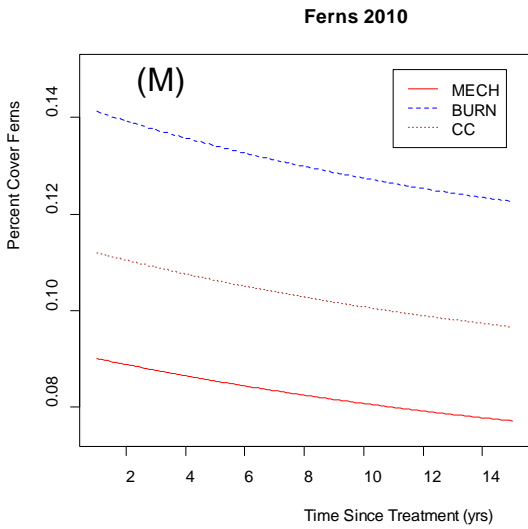
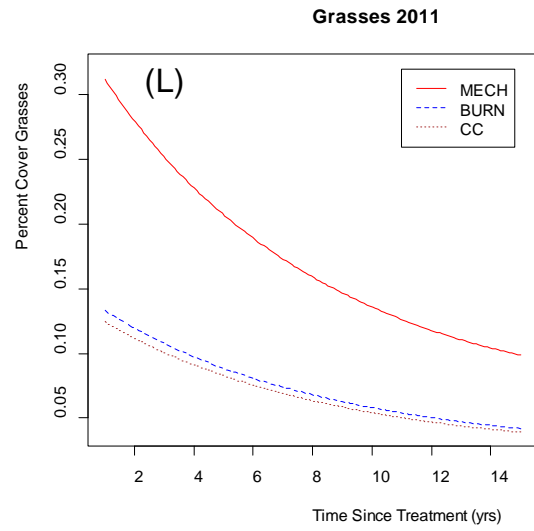
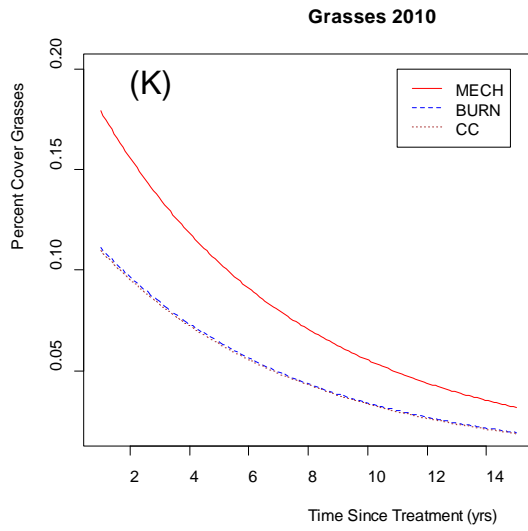
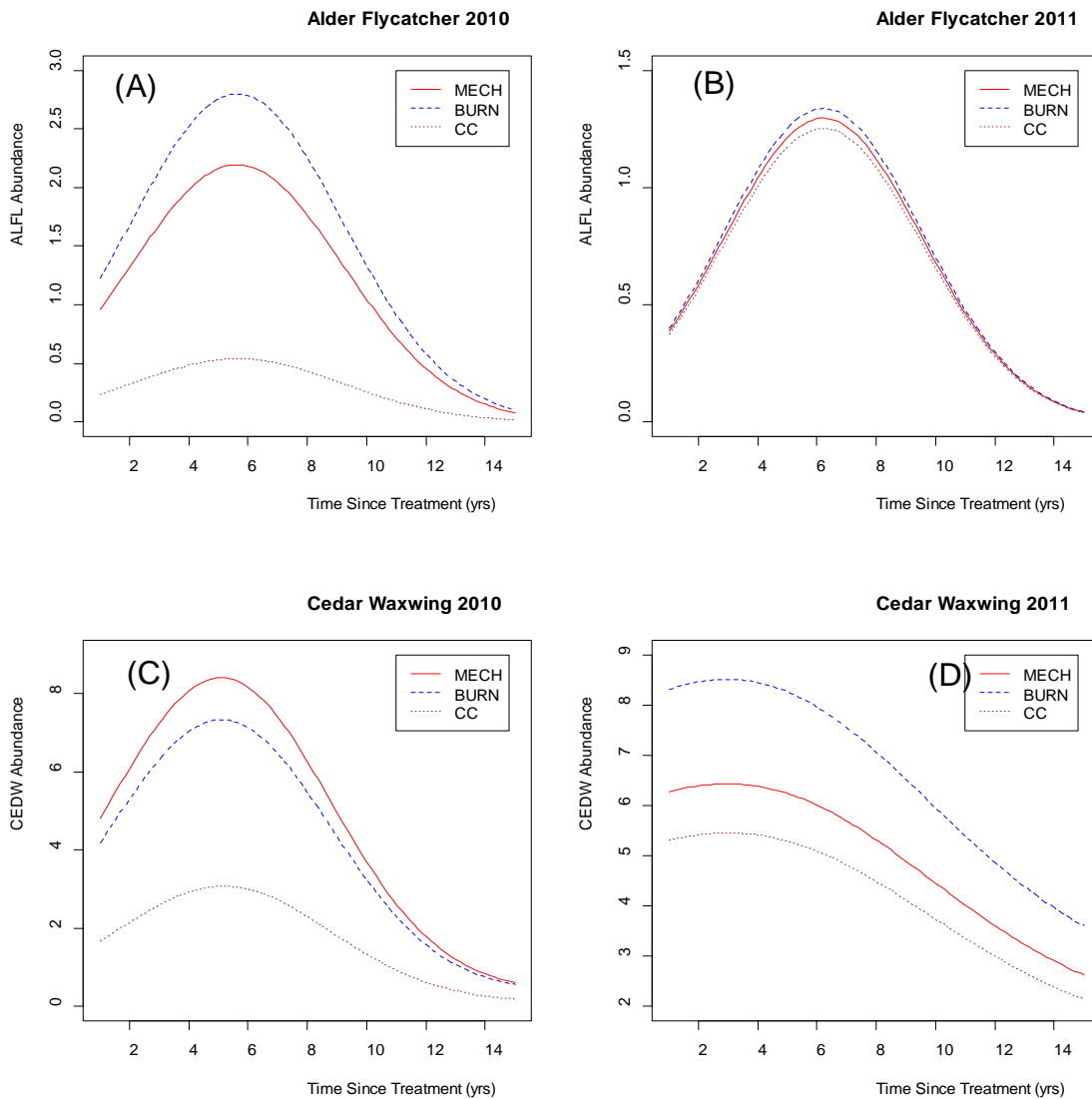
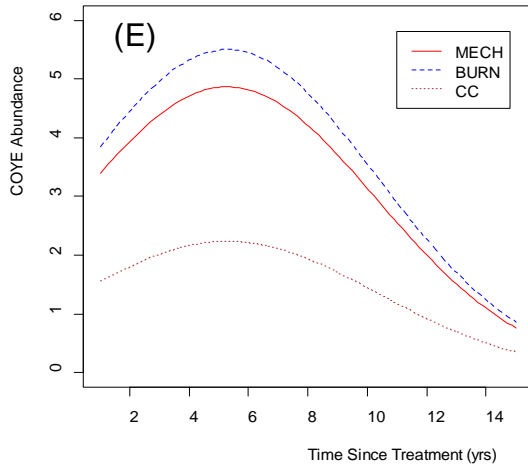


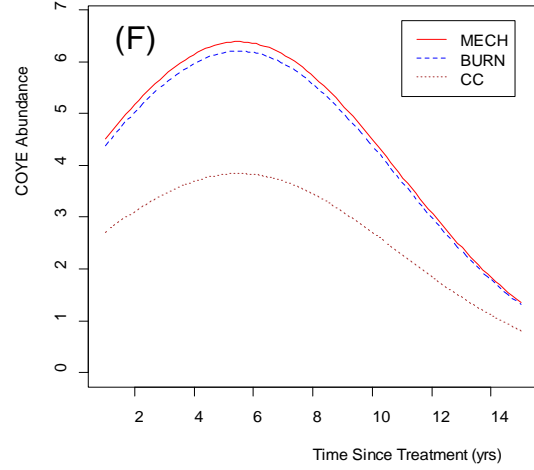
Figure 4. Model-averaged predictions for supported relationships between bird abundance and management variables. Strongly supported treatment effects are represented for Alder Flycatchers (A), Cedar Waxwings (C), (D), Common Yellowthroats (E), (F), Chestnut-sided Warblers, Indigo Buntings (I), (J), Mourning Warblers (K), Song Sparrows (L), (M), and White-throated Sparrows (N), (O). Strongly supported time since treatment effects are represented for Alder Flycatchers (A), (B), Cedar Waxwings (C), (D), Common Yellowthroats (E), (F), Chestnut-sided Warblers, Indigo Buntings (I), (J), Mourning Warblers (K), Song Sparrows (L), (M), and White-throated Sparrows (N), (O). Data are from bird surveys in even-aged silvicultural openings and burned and mechanically treated wildlife openings on the Green Mountain National Forest in Vermont in 2010 and 2011.



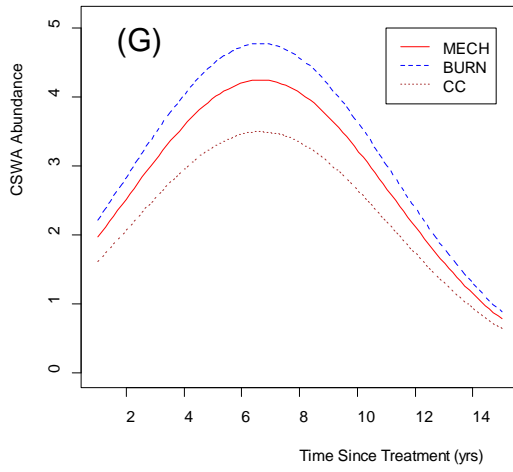
Common Yellowthroat 2010



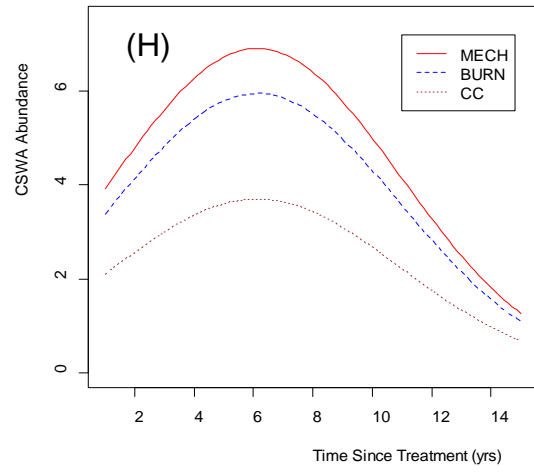
Common Yellowthroat 2011



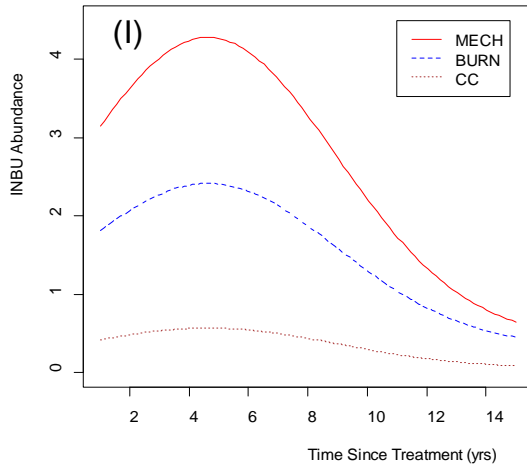
Chestnut-sided Warbler 2010



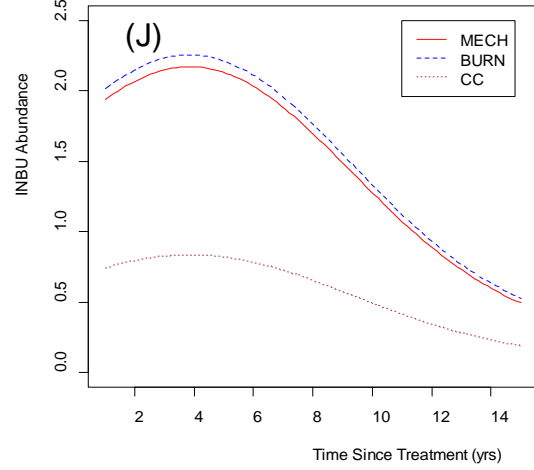
Chestnut-sided Warbler 2011



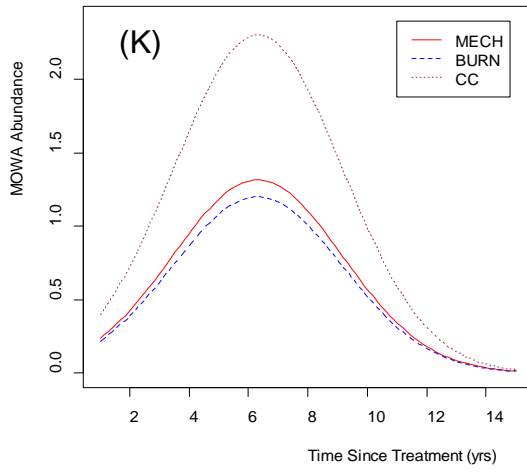
Indigo Bunting 2010



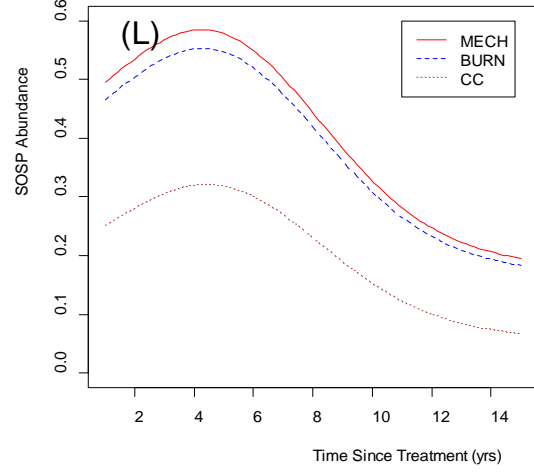
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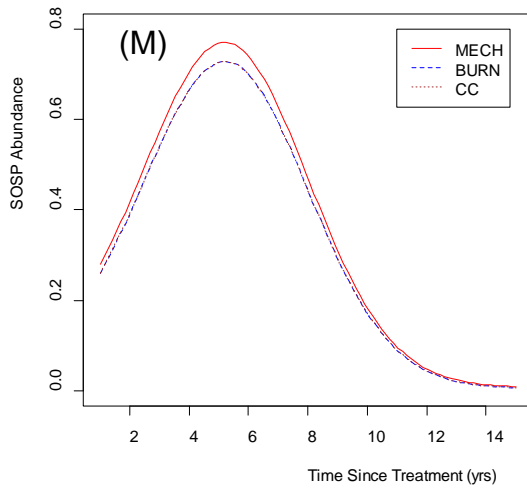
Mourning Warbler 2011



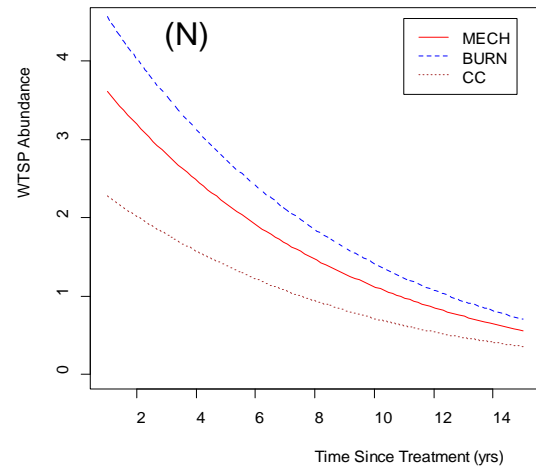
Song Sparrow 2010



Song Sparrow 2011



White-throated Sparrow 2011



White-throated Sparrow 2011

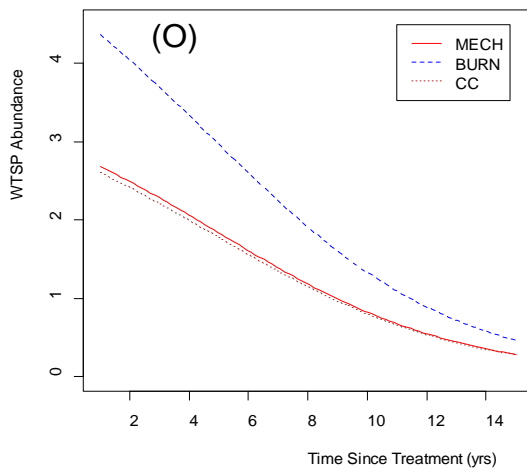
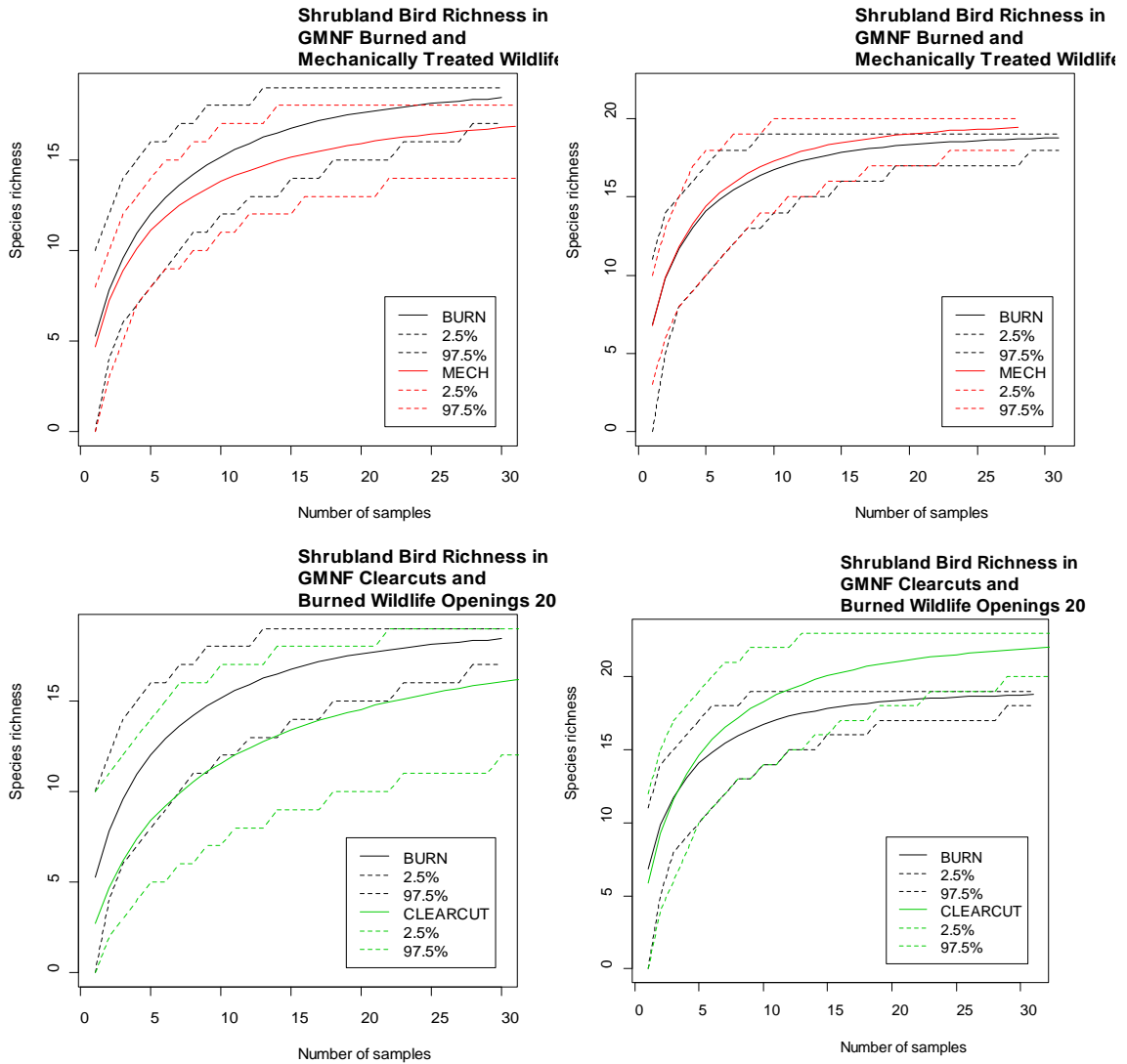


Figure 5. Sample-based species accumulation curves comparing richness of shrubland birds observed in burned and mechanically treated wildlife openings (top), burned wildlife openings and even-aged silvicultural openings (middle), and mechanically treated wildlife openings and even-aged silvicultural openings (bottom). Data are from point-count surveys in even-aged silvicultural openings, and burned and mechanically treated wildlife openings in the Green Mountain National Forest of Vermont in 2010 and 2011.



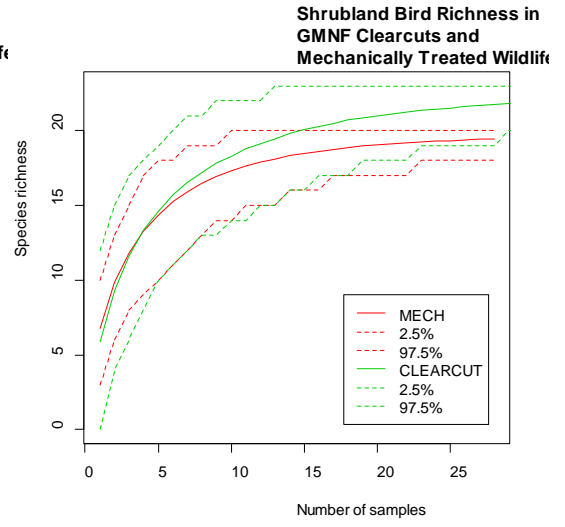
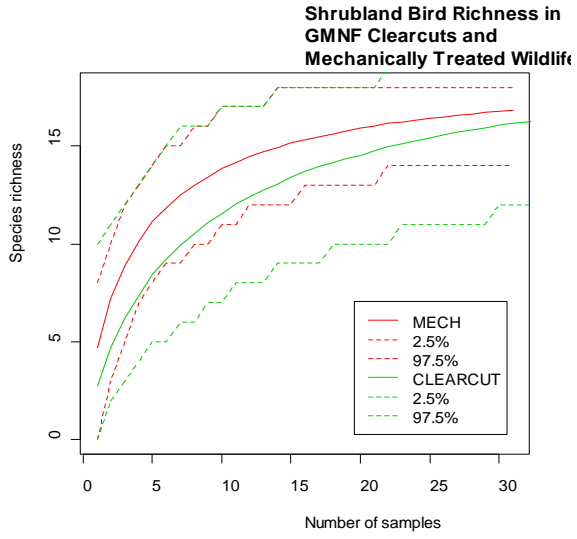
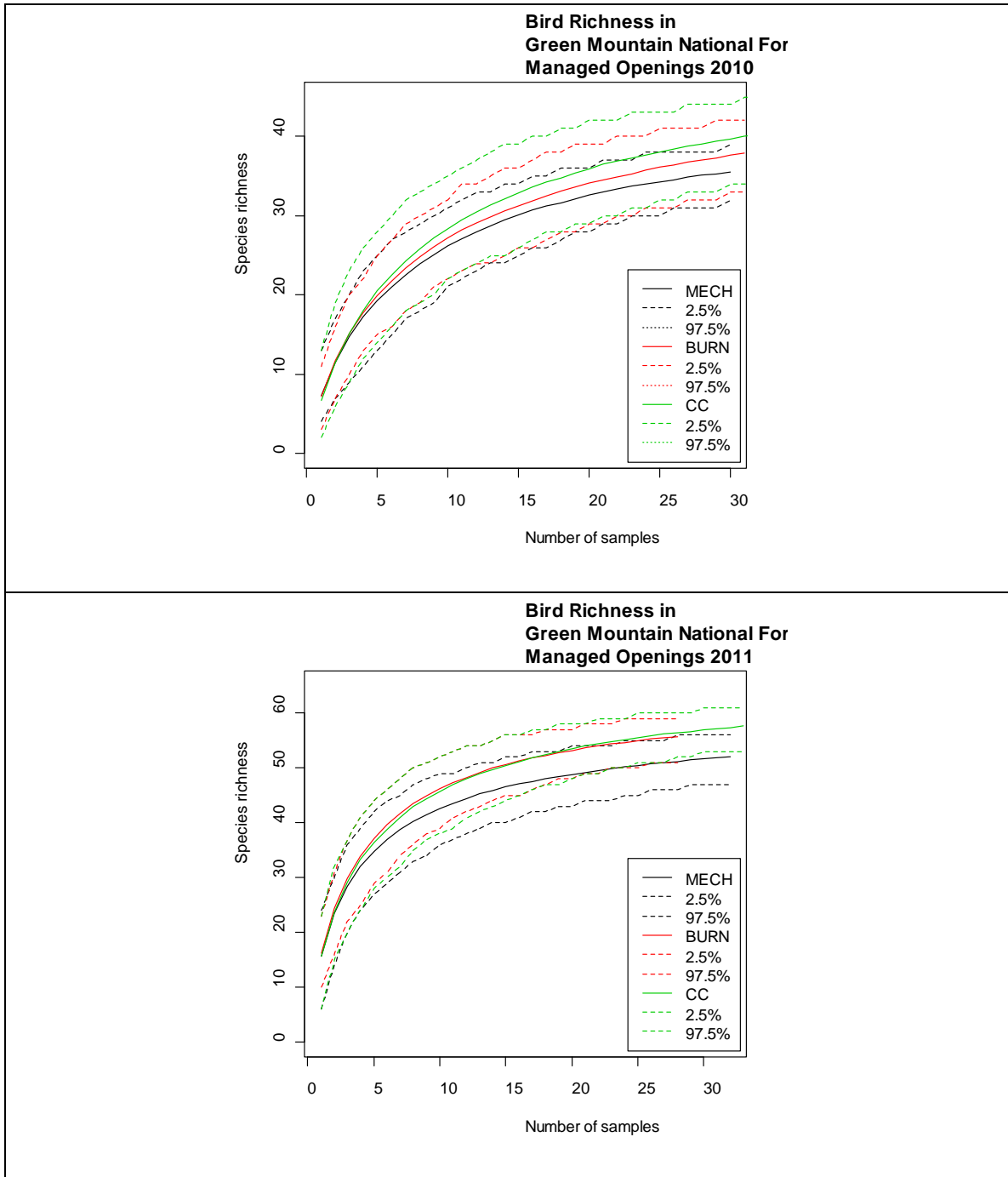


Figure 6. Sample-based species accumulation curves for all species observed in openings on the Green Mountain National Forest in 2010 (top) and 2011 (bottom). Data are from point-count surveys in even-aged silvicultural openings, and burned and mechanically treated wildlife openings in the Green Mountain National Forest of Vermont in 2010 and 2011.



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