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Habitat Associations of Grassland Birds along a Gradient of Eastern Red cedar Succession in Central Kansas

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BIRD-HABITAT ASSOCIATIONS ALONG A CANOPY COVER

GRADIENT OF EASTERN REDCEDAR IN CENTRAL KANSAS

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PREFACE

My thesis is written in the style of the Journal of Wildlife Management, following the guidelines of Block et al. (2011).

ABSTRACT

Grassland birds have declined more rapidly than any other avian taxa in North America. While woody encroachment is often cited as a threat, some grassland-dependent species requiring habitat with scattered trees or shrubs also are declining at statistically significant rates. To better understand the ecological costs and benefits of woody vegetation from a brush management perspective, I studied bird-habitat associations along a canopy cover gradient of eastern redcedar (*Juniperus virginiana*). Habitat associations were tested by the comparing the relative abundance of breeding birds between 3 habitat treatment levels (0% eastern redcedar canopy cover [open grassland], < 5% eastern redcedar canopy cover [light encroachment], and > 5-25% eastern redcedar canopy cover [moderate encroachment]). Data were collected by repeated point count sampling in mixed-grass and sand prairie habitats of Barton County, Kansas from 2011 to 2012. At the community level, bird response patterns were attributed to habitat preferences and nest placement. Ground-nesting species associated with grassland-forb habitat were most abundant in open grassland sites and decreased with increasing eastern redcedar canopy cover. In contrast, species associated with grassland-shrub and savanna habitats were associated positively with eastern redcedar canopy cover. Patterns in the bird community were further examined with cluster analysis and non-metric multidimensional scaling. Avian species-level responses were assessed with canonical correspondence analysis, which indicated that eastern redcedar canopy cover explained most of the variation in the bird abundance. Abundance models and analysis of variation (ANOVA) further elucidated the significance of response patterns and species distributions along the canopy cover gradient. Considering the

diverse habitat requirements of grassland birds, resource managers should consider how conservation practices for one species might affect others.

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INTRODUCTION

Understanding the effects of change in an ecosystem is important for wildlife management and requires a holistic view (Grumbine 1994). According to Knapp and Seastedt (1998), "grassland responses are best understood from a non-equilibrium perspective," because resources fluctuate in response to stochastic processes such as climate, fire, and grazing. Although grassland communities might appear stable at a large spatial scale, non-equilibrium theory suggests a community is not constant over time because fluctuations in the environment occur on a small spatial scale (Chesson and Chase 1986). Local disturbance patterns on the landscape are influenced by factors such as soil type, topography, and land use (Lorimer 2001). As a result, a grassland community might be viewed as a dynamic mosaic of patches varying in composition, space, and time (Watt 1947). For example, the drought conditions during the 1930s caused shifts from tallgrass to mixed-grass prairie in the Great Plains (Weaver and Albertson 1944). Among the extensive grasslands were isolated patches of woody vegetation, particularly in areas protected from fire disturbance such as escarpments, sandhills, rocky outcrops, and stream banks (Albertson 1940, Wells 1965, Bratton et al. 1995). Dispersal of woody vegetation from disturbance-free patches to grasslands likely contributed to the heterogeneity of the grassland community (Wu and Loucks 1995).

Because grasslands have declined drastically in the Great Plains (Samson and Knopf 1994), considerable attention has been placed on the causes and consequences of land conversion and fragmentation. During the Dust Bowl era, shelterbelts were planted next to fields and farmsteads to help reduce the impacts of wind erosion and drought

(Atkinson 1985). Eastern redcedar (*Juniperus virginiana*) has been planted extensively for conservation purposes because it is a hardy and drought tolerant species that is native to the Great Plains (Albertson and Weaver 1945, Ormsbee et al. 1976, Ganguli et al. 2008). Eastern redcedar also was known for its value to wildlife, especially in areas with limited cover (Owensby et al. 1973, Smith 1985, Horncastle et al. 2004). Once established, eastern redcedar can spread to adjacent land and might become invasive if left unmanaged (Gehring and Bragg 1992). Fire suppression also promotes the spread of eastern redcedar. Briggs and Gibson (1992) reported that without fire, canopy cover can increase rapidly in as little as 5 years in eastern Kansas. As a result, land cover occupied by eastern redcedar has increased dramatically, especially in open rangelands of the Great Plains (Owensby et al 1973, Snook 1985, Wilson and Schmidt 1990).

From an ecological perspective, eastern redcedar encroachment might affect grassland communities negatively. Grasslands impacted by encroaching woody cover (primarily *J. virginiana*) are associated with decreased habitat suitability for grassland birds (Chapman 2000, Coppedge et al. 2001, Rosenstock and Van Riper 2001, Chapman et al. 2004, Grant et al. 2004, Frost and Powell 2010) and small mammals (Alford et al. 2012). Research also indicates that habitat benefits for bird communities that use planted woodlands do not outweigh the ecological cost of losing native grassland and woodland-obligate species (Bakker and Higgins 2003, Kelsey et al. 2006). Furthermore, the expansion of eastern redcedar might displace grassland vegetation (Gehring and Bragg 1992, Briggs et al. 2002, Limb et al. 2010). Therefore, the control of eastern redcedar encroachment has become a priority management issue. To maintain long-term use of grassland resources, the Natural Resources Conservation Service (NRCS) recommends that undesirable woody species should not exceed 5% canopy cover (NRCS 2010).

Although eastern redcedar encroachment generally is viewed as an anthropogenic impact (Ganguli et al. 2008), the study of native eastern redcedar in west central Kansas suggests that woody encroachment was a natural and ephemeral process in the historical grassland community (Albertson 1940). Because some grassland birds requiring habitat with scattered trees or shrubs also are declining at statistically significant rates (Knopf 1994, Peterjohn and Sauer 1999, Butcher and Niven 2007), the effect of increasing woody cover should be considered for multiple species. For example, lark sparrow (*Chondestes grammacus*) is declining at a rate of 3.4% per year, which is one of the most significant population trends among declining grassland birds in North America (Knopf 1994). Another declining species, the loggerhead shrike (*Lanius ludovicianus*), showed a significant preference for savanna habitat in northeastern Kansas and nested in eastern redcedar (Michaels and Cully 1998). Because grassland management practices often reflect agricultural and recreational use goals (Morton et al. 2010), landowners might, in fact, encourage woody habitat growth. Within the Playa Lakes Joint Venture Region, a multi-state conservation area for birds of the Southern Great Plains, some landowners have allowed woody cover to increase on grasslands to improve deer (*Odocoileus* spp.) hunting opportunities (Melcher 2006). Eastern redcedar also was planted to provide cover for desired upland game species such as the northern bobwhite (*Colinus virginianus*) and ring-necked pheasant (*Phasianus colchicus*). While reductions of forage production might raise some concern, most landowners do not perceive eastern redcedar encroachment as a primary threat to wildlife habitat (Morton et al. 2010).

State wildlife action plans, resulting from comprehensive and strategic planning efforts, aim to conserve the full array of wildlife and critical habitats by focusing on Species of Greatest Conservation Need (SGCN). Species of Greatest Conservation Need are state designated conservation priorities determined by a selection and ranking criteria that considers the distribution, abundance and population status of native species (Wasson et al. 2005). Kansas SGCN of the central mixed grass prairie associated with grassland habitat include Henslow's sparrow (*Ammodramus henslowii*), greater prairie-chicken (*Tympanuchus cupido*), upland sandpiper (*Bartramia longicauda*), dickcissel (*Spiza americana*), eastern meadowlark (*Sturnella magna*), and grasshopper sparrow (*Ammodramus savannarum*) (Wasson et al. 2005). Kansas SGCN associated with scattered tree or shrub cover include lark sparrow, loggerhead shrike, northern bobwhite, Swainson's hawk (*Buteo swainsoni*), western kingbird (*Tyrannus verticalis*), eastern kingbird (*T. tyrannus*), scissor-tailed flycatcher (*T. forficatus*), brown thrasher (*Toxostoma rufum*), Bell's vireo (*Vireo bellii*), field sparrow (*Spizella pusilla*), and Cassin's sparrow (*Peucaea cassinii*) (Wasson et al. 2005).

In this study I assessed bird-habitat associations across a canopy cover gradient of eastern redcedar encroachment to better understand the ecological costs and benefits of brush management. I hypothesized that bird species would become separated along a canopy cover gradient of eastern redcedar and that groups would form if bird species are responding to the same resource in a similar way. My objective was to compare the abundance of breeding birds among habitat treatment levels at the community and species level. I also assessed the relative influence of other environmental variables.

METHODS

Study Area

The study took place in southern mixed prairie habitats of Barton County, Kansas, USA (Fig. 1). The physiography consists of level to rolling plains, breaks, river valleys, and sand dunes (Adams 1903, Frye and Schoewe 1953). Most of the land use is in cropland and rangeland (USDA-NRCS 2006). Common rangeland conservation practices in the area include grazing, brush management, prescribed burning, and habitat management for upland wildlife (USDA-NRCS 2006).

Aerial images and field observations were used to locate potential study sites. I considered rangeland and idled old-field habitats with at least 16 ha of relatively continuous and well-developed grass/herbaceous cover (i.e., high vegetative structure and composition). The study sites supported big bluestem (*Andropogon gerardii*) and little bluestem (*Schizachyrium scoparium*) plant communities (Weaver and Albertson 1956). Major species of native plants included little bluestem, big bluestem, switchgrass (*Panicum virgatum*), and various forbs. Annual bromes (*Bromus* spp.), kochia (*Kochia scoparia*), and field bindweed (*Convolvulus arvensis*) were the major introduced plant species. The surrounding landscape consisted of croplands, wetlands, shelterbelts, and woodlands.

To understand bird-habitat associations in the context of the brush management, the study area was stratified by percent canopy cover of eastern redcedar (Morrison et al.

2001). Three treatment levels were used to compare bird abundance and distribution (0% canopy cover [open grassland], < 5% canopy cover [light encroachment], and > 5 to 25% canopy cover [moderate encroachment]). Woody canopy cover of 5% or higher is the screening level criteria used by the NRCS to identify sites needing brush management to address resources concerns such as the degradation of plant or wildlife communities (NRCS 2010). The upper limit of canopy cover was based on the classification system of natural vegetation of Kansas; in herbaceous communities (i.e., grasslands) woody cover is less than 25% (Lauver et al. 1999). While eastern redcedar was the dominant woody species among encroachment sites, I also recorded black locust (*Robinia pseudoacacia*), eastern cottonwood (*Populus deltoides*), red mulberry (*Morus rubra*), Russian olive (*Elaeagnus angustifolia*), and isolated patches of sandhill plum (*Prunus angustifolia* and sumac (*Rhus* spp.).

The 2011 study area had 17 sites divided among the 3 treatment levels: open grassland (n = 6), light encroachment (n = 7), and moderate encroachment (n = 4) (Table 1). The 2012 study area had 17 sites: open grassland (n = 6) light encroachment (n = 6), and moderate encroachment (n = 5) (Table 2). The study design was slightly unbalanced because of the limited number of locations with a moderate level of encroachment. Several study sites used in 2011 were substituted in 2012 because activities, such as prescribed burning and brush removal, altered habitat structure and composition.

Sampling Design

I used the habitat-based point-count protocol for terrestrial birds by Huff et al. (2000) to design my study. To help ensure that the canopy cover gradient was well-represented within my study area I used a stratified sampling design with 6 study sites per treatment level. Point-count stations were located systematically on a regular sampling grid to provide uniform coverage of each study site (Cochran 1977, Pendleton 1995). Ralph et al. (1995) reported that a minimum of 30 points per treatment was necessary to adequately characterize bird-habitat associations. To minimize edge effects, point-count stations were located at the interior of each habitat block, at least 125 m from habitat edges and roads (Howe et al. 1997). A minimum inter-point distance of 141.2 m was maintained between point-count stations. Site boundaries were defined by a 125-m buffer surrounding the outermost point-count stations. The typical study site was a 20.25-ha square plot with 5 point-count stations. Due to limitations in the study area, I also used rectangular study sites and smaller plots with 4 point-count stations. Randomization was not feasible because habitat treatments (removal or tolerance of woody species) were applied to the study area passively.

Bird Point-Count Sampling

I estimated the abundance of breeding birds by using the fixed-radius point-count method (Hutto et al. 1986). Upon arrival at a point-count station, I waited 2 minutes for bird activity to equilibrate before recording data. The number of individuals of each bird species detected within a 50-m radius of the point-count station was recorded during a 5-minute period. Detections were categorized as visual, auditory, or flyover. Birds detected beyond 50 m and judged to be using the habitat also were recorded. Previously undetected birds that were seen or heard while I travelled between point-count stations were recorded as incidental detections and not recounted during sampling periods.

All surveys were completed between 0530 and 1000 hours Central Daylight Time (CDT) from mid-May to early-July. Study sites were sampled 3 times per season, with a minimum of 7 days between consecutive visits. Since I could complete 10 to 15 point-count surveys (2 to 3 sites) per morning, the order of site visits and survey routes were varied to minimize “time of day” effects (MacKenzie and Royle 2005). No surveys were done under conditions of rain, fog, or steady wind exceeding 19.3 km/h (>12 mph).

Vegetation Sampling

Vegetation composition and structure were measured at each point-count station. I used a 1-m^2 quadrat to visually estimate percent ground cover of 5 variables: grass, forbs, standing dead vegetation, litter, and bare ground. I used a square frame because it is used widely in vegetation sampling (Bonham 1989) and requires less decision making about the inclusion or exclusion of edge cover (Myers and Shelton 1980). The first cover plot was located at the center of the point-count station and 3 plots were located 4 m from the center, along transects with pre-determined azimuths (120°, 240°, and 360°). Ground cover estimates at each point-count station were quantified by using class midpoint percentage values (2.5%, 5%, 37.5%, 62.5%, 85%, and 97.5%) for each cover class (Daubenmire 1959). Litter depth was measured in centimeters and recorded at 3 locations (2.5, 5, and 7.5 m) along each transect. Vegetation structure was measured in decimeters with a Robel pole at 3 locations per point-count station. Visual obstruction readings were taken 4 m from the Robel pole in each of the 4 cardinal directions at a height of 1 m (Robel et al. 1970). Within a 0.1-ha plot (17.6 m radius) I recorded species and estimated heights of woody vegetation (USDA 2003). Four height-classes were used to characterize structural diversity of woody vegetation within plots: 0 to 1.8 m (0 to 6 ft), 2.1 to 3.7 m (7 to 12 ft), 4.0 to 5.5 m (13 to18ft), and >5.49 m (>18 ft). During the 2012 season the first height class was subdivided into 2 height classes [0 to 0.9 m (0 to 3ft) and 1.2 to 1.8 (4 to 6 ft)] because the aerial imagery was unable to resolve all of the eastern redcedar seedlings or saplings, which caused a discrepancy in my preliminary analysis.

Remote Sensing-based Estimates of Canopy Cover

I used leaf-off imagery, provided by the Barton County Mapping Office, consisting of 24-Bit true color (RGB) 0.61 m resolution (2.0 ft) digital orthoimages in Geo TIFF format. Leaf-off imagery facilitated image interpretation because eastern redcedar (the only evergreen species present) was easily distinguished from deciduous trees and shrubs. The imagery was acquired with a Digital Mapping Camera (DMC) sensor flown at an altitude of 6,096 m (20,000 ft) in the spring of 2010. This imagery was used because the spatial resolution was sufficient to detect the presence of small, individual eastern redcedar.

Site imagery was processed in Adobe PhotoShop (version 6.0, Adobe Systems, Inc., Seattle, WA) software by converting eastern redcedar canopy cover into black pixels and the remaining features into white pixels (Stewart et al. 2007). The resulting binary image was analyzed with ImageTool software (version 3, University of Texas Health Science Center), which counted black and white pixels (Avsar and Ayyildiz, 2010). The percentage of black pixels provided an estimate of canopy cover within each study site (Appendix 1). Images of open grassland sites were not interpreted because no woody vegetation features were identified in the digital images and canopy cover was confirmed to be absent by ground truthing. The steps used to process images are outlined in Appendix 2.

I replicated the ground-sampling method on the processed digital images by counting black pixel clusters within the equivalent area of the vegetation sampling plots. A 17.6-m buffer was drawn around each point station by using Hawth's Tools extension in ArcMap (version 9.3.1, Environmental Systems Research Institute, Redlands, CA). Small and circular pixel clusters were interpreted to be individuals of eastern redcedar, while large and irregular-shaped pixel clusters were interpreted to be a group of eastern redcedar. A visual assessment of the true-color imagery was used to aid counting multiple individuals of eastern redcedar within a pixel cluster.

Statistical Analysis

Remote sensing accuracy assessment.— To assess how accurately the digital image classification represented study sites, ground-based stem counts were compared to remotely-sensed counts of eastern redcedar for each point-count station. A paired samples t-test was used to determine if the differences between image and ground sampling were significantly different and linear regression was used to help visualize correlations between count methods (Davies et al. 2010). I screened the data for potential outliers by visually assessing a scatter plot and checking whether the residuals were distributed normally. I identified 5 of 97 points with high discrepancy, where the number of eastern redcedar on the ground was higher than what was counted in the digital image. This was attributed to an abundance of eastern redcedar seedlings or saplings that were too small to be visibly resolved in the aerial imagery. The distribution of the residuals was approximately normal after the 5 outliers were excluded. Statistical analyses were done in JMP (version 4.04, SAS Institute Inc., Cary, NC) with a 0.05 significance level.

Relative abundance.— Because grassland-associated birds exploit environmental resources in different ways, I assigned species to groups based on two life-history categories: (1) breeding habitat (grassland-forb and successional-shrub) and (2) nest placement (ground-low and mid-story canopy) (Table 3). I used information from Herkert (1994, 1995) and Vickery et al. (1999) for group assignments. Combinations of life-history categories allowed for classification of bird species into three ecological guilds: (1) grassland guild (grassland-forb habitat + ground-low nesting), (2) grassland-shrub guild (successional-shrub habitat + ground-low nesting), and (3) savanna guild (successional-shrub habitat + mid-canopy nesting).

One-way analysis of variance (ANOVA) was used to compare the relative abundance of bird guilds (i.e., the number of individuals from each guild detected at each study site) among the 3 treatment levels. The maximum abundance of each species per point-count station, per survey year was used to summarize relative abundance (Nur et al. 1999). The analyses included only those species that were judged to be associated with the local habitat. I excluded non-typical detections from the analyses, such as flyovers, incidental observations (species recorded outside the 5 min sampling period), and those species judged not to be associated with the local habitat (i.e., transient species). For statistically significant ANOVA results, Tukey-Kramer tests were used for pairwise comparisons between treatment levels. A significance level of 0.05 was used for all analyses. I used R statistical software Version 2.13.2 (2011-04-13).

Community similarity.— Agglomerative hierarchical cluster analysis was used to group study sites based on the degree of similarity of bird abundance data. This was done to assess whether the treatment levels were consistent with bird habitat use patterns. To group study sites, I used the Bray-Curtis distance measure with the flexible beta linkage method with $\beta = -0.25$ (McCune and Grace 2002). Indicator species analysis (Dufrêne and Legendre 1997) was used to objectively select the best cluster solution by comparing the average significance of indicator values at each hierarchy level of the dendrogram. Monte Carlo permutation tests (1,000 permutations) were used to assess the significance of indicator species values at the 0.05 significance level. The cluster with the lowest *P*-value, averaged for all species, determined the best cluster solution (McCune and Grace 2002).

Indicator values (ranging from 0-100%) were used to highlight significant indicator species within site groups. The values, calculated for each species within each group, are the product of relative abundance and site fidelity (Dufrêne and Legendre 1997). I used R statistical software Version 2.13.2 (2011-04-13) with R package LabDSV Version 1.5-0 for indicator species analysis (Roberts 2012).

Non-metric multidimensional scaling (NMDS) was used to describe bird community structure based on Bray-Curtis dissimilarities among study sites. Goodness of fit was assessed with a stress value and Shepard plot. Two dimensions, the optimal number of axes determined by assessing the stress value, were included in the NMDS ordination plot. Confidence ellipses at the 75% level were drawn around site groups to show the relation of study sites at the best cluster solution. To visually interpret bird assemblages relative to study site groups, cluster analysis results were combined with the NMDS ordination plot. The data consisted of a matrix of 34 sites X 35 bird species. Species detected at fewer than 5 sites were excluded from the analysis because highly localized species were not considered representative of site groups (McCune and Grace 2002). Species data were square-root transformed to reduce the influence of dominant species. I used R statistical software Version 2.13.2 (2011-04-13) with R package vegan Version 1.17-7 for cluster analysis and NMDS (Oksanen et al. 2011).

Bird-habitat associations.— To elucidate bird-habitat associations, canonical correspondence analysis (CCA) was used to select a combination of environmental variables that best explained variation in the distribution of bird species (ter Braak 1986). Canonical correspondence analysis is an ordination technique used to describe and visualize species niche positions along environmental gradients (ter Braak and Verdonschot 1995). Stepwise forward-selection with Monte Carlo permutation tests (permutations = 1000) was used to determine the combination of environmental variables that explained most of the variation observed in the bird species matrix. Linear combination (LC) site scores, which are constrained maximally by the environmental variables, were used to plot the ordination diagram (Palmer 1993). To reduce crowding of the ordination plot, I highlighted 12 focal species with conservation value. Kansas designated SGCN were Bell's vireo, brown thrasher, dickcissel, eastern kingbird, eastern meadowlark, field sparrow, grasshopper sparrow, lark sparrow, northern bobwhite, upland sandpiper, and western kingbird. Ring-necked pheasant also was included because of its popularity as a game species.

As in the previous analysis, the data consisted of a matrix of 34 sites X 35 bird species. Because the bird point-count data contained many zeroes, the species data were square-root transformed to produce a more normal distribution; however, CCA is robust to non-normal species distributions (Palmer 1993, ter Braak and Verdonschot 1995). I used R statistical software Version 2.13.2 (2011-04-13) with R package vegan Version 1.17-7 for CCA (Oksanen et al. 2011).

The environmental data consisted of percent cover for 5 classes (grass, forb, dead standing vegetation, litter, and bare ground), percent canopy cover of eastern redcedar, litter depth, and visual obstruction. These data were averaged across sample plots by site. A correlation matrix was used to assess multicollinearity and no variables had a Pearson correlation coefficient greater than 0.80. Environmental variables were square-root transformed so that the distributions were approximately normal. Because the variables were measured in different units, a min-max standardization was applied so that variables ranged from zero to one, which removed their scale.

Multivariate normality of environmental data was assessed by examining normality, linearity, and homogeneity for each variable by treatment level (Tabachnick and Fidell 2001). After transforming the data to reduce potential outliers, 2 remaining extreme values were not removed from the analysis because the values represented true variation in the environment.

Species response. — N-mixture models were applied to point-count data to describe bird abundance as a function of eastern redcedar canopy cover (Royle 2004). To accommodate zero-inflated data, caused by imperfect detections or unoccupied sites, I used 2 variants of the N-mixture model: Poisson and zero-inflated Poisson (ZIP) (Wenger and Freeman 2008). According to Joseph et al. (2009) the Poisson mixture models provide the most ecologically meaningful parameter estimates. The data consisted of encounter histories for the 12 focal species detected within a fixed radius of each point-count station (n = 168). A 50-m detection radius was used for Bell's vireo, dickcissel, and grasshopper sparrow, field sparrow, and lark sparrow. A 75-m detection radius was used for northern bobwhite, ring-necked pheasant, upland sandpiper, eastern kingbird, western kingbird, brown thrasher, and eastern meadowlark because they were typically detected beyond 50 m. To increase sample size for each species, data from the 2011 and 2012 sampling seasons were pooled. Eastern redcedar canopy cover was considered a covariate-effect for model fitting. Because the population closure assumption was violated, I included year as a factor and year X canopy cover interactions in the model set (Johnson and Cunningham 2006). I considered 8 models (4 Poisson and 4 ZIP models) for each species with the following variables: none (the null model), canopy cover, year, and year X canopy cover. I fit models in R statistical software Version 2.13.2 (2011-04-13) by using the pcount function of package unmarked (Fiske and Chandler 2011). For each model, Akaike's Information Criterion (AIC), ΔAIC, model weight, and cumulative weight were calculated. After ranking models, the "best" model was used to calculate species detection probability (p) and mean abundance per point-count station (λ). Akaike's Information Criterion scores were used for model ranking and selection (Burnham and Anderson 2004). A cutoff of $\Delta AIC \leq 2$ was used to include models that shared a similar level of support with the best model. To show species response patterns to increasing eastern redcedar canopy cover, I plotted predicated abundance for each species. Parametric bootstrapping, with 100 simulations, was used to evaluate goodness of fit.

One-way ANOVA was used to determine how the mean relative abundance of the 12 focal bird species varied between the 3 treatment levels. For significant ANOVA results, Tukey-Kramer tests were used for pairwise comparisons.

RESULTS

Canopy Cover Estimates

Eastern redcedar canopy cover estimates ranged from 0.38% to 23.64% in the encroachment treatment levels. Results of the matched pairs test indicated the remote-sensed counts of eastern redcedar were slightly lower than ground counts by a mean difference of -0.34 (t = 1.12, P = 0.27; df = 96). Linear regression confirmed that there was a strong relationship between the image and ground-based counts (r^2 = 0.91, P < 0.001; Fig. 2).

Descriptive Statistics

A total of 74 species and 6,166 individual detections (including incidental observations) was recorded during the 2011 and 2012 breeding bird seasons (Table 4). The most frequently detected species were dickcissel (n = 1,369), grasshopper sparrow (n = 533), red-winged blackbird (*Agelaius phoeniceus*; n = 493), brown-headed cowbird (*Molothrus ater*); n = 488), and field sparrow (n = 327). Dickcissel was the most common species among open grassland sites (n = 858) and light encroachment sites

(n = 413). Field sparrow (n = 252) was the most common species among moderate encroachment sites.

Relative Abundance

The ANOVA results indicated that eastern redcedar canopy cover had a significant effect on the relative abundance of the grassland bird guild ($F_{2,31}$ = 52.44, P < 0.001; Fig. 3). Kansas SGCN members representing this guild were dickcissel, grasshopper sparrow, eastern meadowlark, and upland sandpiper. Other species considered in this guild were ring-necked pheasant, western meadowlark (*Sturnella neglecta*), and red-winged blackbird. Overall, members of the grassland guild were 26% to 67% more abundant on open grassland sites than light (P < 0.001) and moderate (P < 0.001) encroachment sites. Pairwise differences between light and moderate encroachment sites also were significant (P < 0.001).

Eastern redcedar canopy cover had a significant effect on the relative abundance of the grassland-shrub bird guild ($F_{2,31}$ = 60.63, P < 0.001). SCGN members representing this guild were northern bobwhite, brown thrasher, Bell's vireo, lark sparrow, field sparrow, and Cassin's sparrow. Other species considered in this guild were common yellowthroat (*Geothlypis trichas*) and mourning dove (*Zenaida macroura)*. Members of the grassland-shrub guild were 38% to 53% more abundant in moderate encroachment sites than light encroachment (P < 0.001) and open grassland (P < 0.001) sites. Pairwise differences between light encroachment and open grassland sites also were significant (P = 0.005).

The ANOVA indicated that eastern redcedar canopy cover had a significant effect on the relative abundance of the savanna guild ($F_{2,31} = 18.70$, $P < 0.001$), which included 2 Kansas SGCN members: eastern and western kingbird. Savanna species were 11% to 14% more abundant in habitats with light encroachment ($P < 0.001$) and moderate encroachment ($P < 0.001$) than open grassland sites. Pairwise differences between light and moderate encroachment sites were not significant ($P = 0.481$).

Community Similarity

The structure of the cluster analysis dendrogram showed 3 distinct groups of study sites based on similarities in bird species (Fig. 4). Indicator species analysis found the best solution at the 3-group level, which had the lowest average *P*-value of the clusters in the dendrogram ($\overline{X} = 0.06$). Group 1 included all 9 open grassland sites and 2 light encroachment sites. Eastern redcedar canopy cover was 0.10 % and 0.40% for the light encroachment sites. Group 2 had 9 sites (4 light encroachment sites and 6 moderate encroachment sites) and eastern redcedar canopy cover ranged from 0.41% to 23.64%. Group 3 included 1 moderate and 2 light encroachment sites, with a canopy cover range of 0.38% to 5.16%.

Red-winged blackbird, with an indicator value of 66%, was the only characteristic species of group 1 (Fig. 5). Indicator values greater than 55% signify characteristic species, which contribute to the specificity of a site group and whose presence can be predicted within the group (Dufrêne and Legendre 1997). Common yellowthroat and dickcissel also were associated with group 1 but their indicator values were less than 55%. Group 2 had 15 species with an indicator value greater than 55%. Field sparrow, had a 100% index value, meaning the species was restricted to group 2 and detected at every site within the group. Other characteristic species of group 2 were northern cardinal (*Cardinalis cardinalis*), American goldfinch (*Spinus tristis*), Baltimore oriole (*Icterus galbula*), lark sparrow, Bell's vireo, eastern kingbird, northern mockingbird (*Mimus polyglottos*), blue jay (*Cyanocitta cristata*), house wren, brown thrasher, western kingbird, northern bobwhite, American crow (*Corvus brachyrhynchos*), and yellow-billed cuckoo (*Coccyzus americanus*). Group 3 was characterized by upland sandpiper and western meadowlark, along with maximum indicator values for brown-headed cowbird and mourning dove.

The NMDS ordination of species in 2 dimensions, with Bray-Curtis dissimilarity measure, resulted in a final stress value of 9.92% and a linear fit of 0.957 (Fig. 6). The low stress value indicates the relation between dissimilarities and distances are a good fit (Kruskal 1964). Species associated with group 1 in the NMDS ordination plot included eastern meadowlark and ring-necked pheasant. Common yellowthroat, red-winged blackbird, western meadowlark, and upland sandpiper had weak associations with group 1. The intermediate position of dickcissel and grasshopper sparrow between group 1 and 3 suggested a tolerance of light encroachment. Group 3 contained red-headed woodpecker (*Melanerpes erythrocephalus*), brown-headed cowbird, and mourning dove. Species associated with group 2 included eastern kingbird, western kingbird, brown thrasher, Baltimore oriole, northern mockingbird, American goldfinch, wild turkey (*Meleagris gallopavo*), lark sparrow, American crow, northern cardinal, Bell's vireo, field sparrow, house wren, blue jay, indigo bunting (*Passerina cyanea*), and eastern phoebe (*Sayornis phoebe*).

Canonical Correspondence Analysis

Seven environmental variables were entered into the CCA formula and 4 were significant ($P = 0.001$) in the full model. In order of significance, they were percent eastern redcedar canopy cover, visual obstruction, percent grass cover, and litter depth. The first and second axes were significant and cumulatively explained 75% ($P = 0.001$) and 84% ($P = 0.02$) of the variation in the bird data, respectively. Using forward selection, the reduced model was highly significant ($P = 0.001$) and showed that percent eastern redcedar canopy cover was the only significant environmental variable accounting for variation in bird abundance.

The species-conditional CCA triplot showed bird species composition within study sites, in response to increasing eastern redcedar canopy cover (Fig. 7). The position of species points in the ordination plot were used to approximate the center of each species' distribution along the eastern redcedar canopy cover gradient. Plotting species as weighted averages of sites provided a clearer indication of site preferences. From this ordination, I inferred that ring-necked pheasant, eastern meadowlark, upland sandpiper, dickcissel and grasshopper sparrow most frequently occurred at sites with no eastern redcedar canopy cover. The relative position of centroids for dickcissel and grasshopper sparrow suggested a greater tolerance of woody encroachment. Northern bobwhite was located near the origin of the plot, indicating no response to the canopy cover gradient. Eastern kingbird, western kingbird, and brown thrasher were associated positively with the canopy cover gradient, with centroids located amongst light encroachment sites. The relative position of brown thrasher suggested that it also was likely to occur in moderate encroachment sites. Lark sparrow, Bell's vireo, and field sparrow were most closely associated moderate encroachment sites and had the highest weighted averages along the canopy cover gradient.

Species Response

Bell's vireo abundance increased with eastern redcedar canopy cover (Fig. 8). The Poisson mixture model with the canopy cover covariate was the best model (Table 5) and fit the data adequately (X^2 = 494.5 P = 0.47). The second best model was the ZIP model with the canopy cover covariate. Since there was a 2-unit difference in AIC values I used model averaging for predictions. Mean abundance per count station was 1.3 (95% CI = 0.2-2.1) and the expected detection probability was 0.41 (SE = 0.06). Results of the ANOVA confirmed that Bell's vireo had a positive response to eastern redcedar canopy cover ($F_{2,31}$ = 39.96, P < 0.001). Unadjusted mean abundance for Bell's vireo in moderate encroachment sites ($\bar{X}$ = 2.14, SE = 0.18) was significantly higher than open grassland ($\bar{X}$ = 0.08, SE = 0.15, P < 0.001) and light encroachment sites ($\bar{X}$ = 0.45, SE = 0.15, P < 0.001). Pairwise differences between light encroachment and open grassland sites were not significant (P = 0.225).

Brown thrasher abundance increased with eastern redcedar canopy cover (Fig. 9). The best model for brown thrasher was the Poisson mixture model with the canopy cover covariate; however, the model did not fit the data (X^2 = 605.4, P = 0.01). Results of the ANOVA showed that brown thrasher had a positive response to eastern redcedar canopy cover ($F_{2,31} = 21.46$, $P < 0.001$). Mean abundance for brown thrasher in moderate encroachment sites ($\overline{X} = 1.70$, SE = 0.20) was significantly higher than open grassland ($\overline{X} = 0.08$, SE = 0.17, $P < 0.001$) and light encroachment sites ($\overline{X} = 1.07$, SE = 0.17 $P < 0.025$). Pairwise differences between light encroachment and open grassland sites also were significant ($P < 0.001$).

Dickcissel abundance decreased with eastern redcedar canopy cover (Fig. 10). The best model was the ZIP mixture model with the canopy cover covariate, which fit the data adequately ($X^2 = 3320$, $P = 0.36$). Mean abundance per count station was 3.3 (95% CI = 0.4-12.0) and the expected detection probability was 0.17 (SE = 0.04). Results of the ANOVA confirmed that dickcissel had a negative response to eastern redcedar canopy cover ($F_{2,31} = 16.45$, $P < 0.001$). Mean abundance for dickcissel in open grassland sites ($\overline{X} = 4.47$, SE = 0.29) was significantly higher than light ($\overline{X} = 3.20$, SE = 0.28, $P = 0.009$) and moderate ($\overline{X} = 1.96$, SE = 0.33 $P < 0.025$) encroachment sites. Pairwise differences between light and moderate encroachment sites also were significant ($P = 0.019$).

Eastern kingbird abundance increased with eastern redcedar canopy cover (Fig. 11). In the model set, the null model and the ZIP mixture model had similar support and fit the data ($X^2 = 16792$, $P = 0.21$). However, the prediction for mean abundance was unreasonable ($\lambda = 63.4$, 95% CI = 56.3-71.8) due to the low detection probability of 0.005 (SE = 0.002). Results of the ANOVA showed that eastern kingbird had a positive response to eastern redcedar canopy cover ($F_{2,31} = 28.91$, $P < 0.001$). Mean abundance for eastern kingbird in light ($\overline{X} = 1.67$, SE = 0.13) and moderate ($\overline{X} = 1.50$, SE = 0.16) encroachment sites was significantly higher than open grassland sites ($\bar{X}$ = 0.28, SE = 0.14, P < 0.001). Pairwise differences between light encroachment and open grassland sites were not significant (P = 0.696).

Eastern meadowlark abundance decreased with eastern redcedar canopy cover (Fig. 12). The best model was the Poisson mixture model with the canopy cover covariate, which fit the data adequately (X^2 = 405, P = 0.96). Mean abundance per count station was 0.4 (95% CI = 0.01-1.7) and the expected detection probability was 0.34 (SE = 0.05). Results of the ANOVA confirmed that eastern meadowlark had a negative response to eastern redcedar canopy cover ($F_{2,31}$ = 18.89, P < 0.001). Mean abundance for eastern meadowlark in open grassland ($\bar{X}$ = 2.18, SE = 0.22) and light encroachment sites ($\bar{X}$ = 2.26, SE = 0.21) was significantly higher than moderate encroachment sites ($\bar{X}$ = 0.41, SE = 0.25, P < 0.001). Pairwise differences between open grassland and light encroachment sites were not significant (P = 0.962).

Field sparrow abundance increased with eastern redcedar canopy cover (Fig. 13). The best model was the ZIP mixture model with the canopy cover covariate, which fit the data adequately (X^2 = 5109, P = 0.36). However, the mean abundance per count station (λ = 43.3, 95% CI = 21.6-74.1) seemed unrealistic. The low detection probability of 0.03 (SE = 0.02) suggested that the prediction was unreliable. Results of the ANOVA showed that field sparrow had a positive response to redcedar canopy cover ($F_{2,31}$ = 22.78, P < 0.001). Mean abundance for field sparrow in moderate encroachment sites ($\bar{X}$ = 3.05, SE = 0.34) was significantly higher than open grassland ($\bar{X}$ = 0.00, SE = 0.30, P < 0.001) and light encroachment sites ($\bar{X}$ = 1.16, SE = 0.28, $P < 0.001$). Pairwise differences between light encroachment and open grassland sites also were significant ($P = 0.021$).

Grasshopper sparrow abundance decreased with eastern redcedar canopy cover (Fig. 14). The best model was the Poisson mixture model with the canopy cover covariate and the model fit the data ($X^2 = 605$, $P = 0.80$). I used model averaging for predictions since the second best model, the ZIP mixture model with the canopy cover covariate, was within 2 AIC units. Mean abundance per count station was 1.1 (95% CI = 0.2-2.8) and the expected detection probability ranged from 0.036 (SE = 0.04) to 0.34 (SE = 0.05). Results of the ANOVA showed that grasshopper sparrow had a negative response to eastern redcedar canopy cover ($F_{2,31} = 11.34$, $P < 0.001$). Mean abundance for grasshopper sparrow in open grassland ($\bar{X}$ = 2.63, SE = 0.22) and light encroachment sites ($\bar{X}$ = 2.57, SE = 0.22) was significantly higher than moderate encroachment sites ($\bar{X}$ = 1.22, SE = 0.25, $P < 0.001$). Pairwise differences between open grassland and light encroachment sites were not significant ($P = 0.978$).

Lark sparrow abundance increased with eastern redcedar canopy cover (Fig. 15). The best model was the ZIP mixture model with the canopy cover covariate, which fit the data adequately ($X^2 = 3549$, $P = 0.12$). However, the estimated mean abundance per count station ($\lambda = 40.5$, 95% CI = 12.1-90.5) seemed unrealistic. The low detection probability of 0.02 (SE = 0.01) suggested that the prediction was unreliable. Results of the ANOVA showed that lark sparrow had a positive response to eastern redcedar canopy cover ($F_{2,31} = 14.10$, $P < 0.001$). Mean abundance for lark sparrow in moderate encroachment sites ($\bar{X}$ = 1.73, SE = 0.25) was significantly higher than open grassland

($\overline{X}$ = 0.00, SE = 0.22, $P < 0.001$) and light encroachment sites ($\overline{X}$ = 0.91, SE = 0.21, P = 0.041). Pairwise differences between light encroachment and open grassland sites also were significant (P = 0.013).

Three models had similar support (ΔAIC $\leq$ 2) for northern bobwhite. The ZIP and Poisson null mixture models ranked above the ZIP mixture model with the canopy cover covariate (Fig. 16). Using model averaging, mean abundance per count station was 2.6 (95% CI = 2.6-2.7). The detection probability for the best model was 0.14 (SE = 0.06) and the data fit the model adequately (X^2 = 943, P = 0.52). Results of the ANOVA showed that northern bobwhite had a non-significant response to eastern redcedar canopy cover ($F_{2,31}$ = 0.637, P = 0.536). Unadjusted mean abundance ranged from 1.50 (SE = 0.23) to 1.90 (SE = 0.27).

Ring-necked pheasant abundance decreased with eastern redcedar canopy cover (Fig. 17). The best model was the Poisson mixture model with the canopy cover covariate and the model did not fit the data adequately (X^2 = 710, P = 0.02). Mean abundance per count station was 0.5 (95% CI = 0.0-1.8) and the expected detection probability was 0.13 (SE = 0.06). Results of the ANOVA showed that ring-necked pheasant had a negative response to moderate levels of eastern redcedar canopy cover ($F_{2,31}$ = 8.81, $P < 0.001$). Mean abundance of ring-necked pheasant in open grassland ($\overline{X}$ = 1.66, SE = 0.22, P = 0.001) and light encroachment sites ($\overline{X}$ = 1.48, SE = 0.21, P = 0.004) were significantly higher than moderate encroachment sites ($\overline{X}$ = 0.36, SE = 0.25). Pairwise differences between open grassland and light encroachment sites were not significant (P = 0.805).

Upland sandpiper abundance decreased with eastern redcedar canopy cover (Fig. 18). The best model was the Poisson mixture model with the canopy cover covariate. The second best model was the ZIP model with the canopy cover covariate. I used the best model (Poisson with the canopy cover covariate) because the model averaged prediction for abundance seemed unrealistic (λ = 6.2, 95% CI = 0.6-20.3). Mean abundance per count station for the best model was 0.31 (95% CI = 0.05-0.7) and the expected detection probability was 0.13 (SE = 0.06). This model fit the data adequately (X^2 = 476, P = 0.37). Results of the ANOVA showed that upland sandpiper had a non-significant response to eastern redcedar canopy cover ($F_{2,31}$ = 0.367, P = 0.696).

Western kingbird abundance increased with eastern redcedar canopy cover (Fig. 19). The best model was the Poisson mixture model with the canopy cover covariate. The remaining models were all within 2 AIC units of the best model. However, the model averaged prediction for abundance was unrealistic (λ = 23.5, 95% CI = 18.6-29.6). Instead, I used the best model (Poisson with the canopy cover covariate), which fit the data adequately (X^2 = 446, P = 0.10). Mean abundance per count station was 3.5 (95% CI = 2.3-5.0) and the expected detection probability was 0.01 (SE = 0.06). Results of the ANOVA confirmed that western kingbird had a positive response to eastern redcedar canopy cover ($F_{2,31}$ = 5.09, P = 0.012). The mean abundance of western kingbird in light ($\bar{X}$ = 1.15, SE = 0.77, P = 0.024) and moderate encroachment sites ($\bar{X}$ = 1.25, SE = 0.23, P = 0.030) was significantly higher than open grassland sites ($\bar{X}$ = 0.42, SE = 0.20). Pairwise differences between light encroachment and moderate encroachment sites were not significant (P = 0.937).

DISCUSSION

Bird-habitat Associations

Overall, bird abundance and distribution shifted along a canopy cover gradient of eastern redcedar. A comparison of mean abundance between habitat treatment levels showed positive and negative response of bird guilds to eastern redcedar canopy cover, which implied that grassland birds differed in response to increased levels of woody encroachment. These differences were attributed to species habitat preferences and nest placement. Ground-nesting species associated with grassland-forb habitat were most abundant in open grassland sites and decreased with increasing canopy cover. In contrast, species associated with grassland-shrub and savanna habitats were associated positively with canopy cover, although response levels varied. The savanna guild increased significantly at the light encroachment level and showed no difference in abundance at the moderate level. The grassland-shrub guild peaked at the moderate encroachment level and abundance was significantly higher than light encroachment and open grassland sites. Other studies also have found significant changes in bird community composition related to woody encroachment (Chapman 2000, Coppedge et al. 2001, Rosenstock and Van Riper 2001, Chapman et al. 2004, Grant et al. 2004). Similarly, Frost and Powell (2010) reported differences at the species and community level in response to eastern redcedar removal.

Remsen (1994) cautioned against the use of bird lists to compare sites, in part, because of the failure to distinguish core species from species that are not representative of a particular habitat. By using indicator species analysis, I distinguished species characterizing the site groups formed with hierarchical cluster analysis. Three site groups were identifiable in the NMDS ordination, which concurred with cluster analysis results. The order of sites on the first axis seemed to correspond with the underlying canopy cover gradient; in general, sites were ordered from left to right by increasing canopy cover. Similarly, the order of bird species on the first axis was related to the canopy cover gradient and the relative position of species to sites indicated habitat affinities. In general, species located inside the 75% confidence ellipse of a site group were significant indicators of that group. Species with wide distribution or local abundance patterns seemed to be positioned outside the confidence ellipses.

Group 2 was characterized by a larger group of eurytopic indicator species, including 7 SGNC: field sparrow, lark sparrow, Bell's vireo, eastern kingbird, brown thrasher, western kingbird, and northern bobwhite. Northern cardinal, American goldfinch, Baltimore oriole, northern mockingbird, blue jay, house wren, American crow, and yellow-billed cuckoo also showed a strong association with group 2. Coppedge et al. (2001) observed a similar species assemblage (classified as open-habitat generalists, successional scrub species, and woodland species) associated with Oklahoma grasslands fragmented by woody encroachment. Canopy cover separated open grasslands from encroachment sites (groups 1 and 3) for the most part.

The species assemblages of group 1 and 3 also reflected differences in vegetation structure. Red-winged blackbird, common yellowthroat, and dickcissel were indicators of tall, dense herbaceous vegetation structure. In contrast, upland sandpiper and western meadowlark prefer grasslands with shorter or more open vegetation structure

(Fritcher et al. 2004). While a few species reached maximum indicator values at level 3, only upland sandpiper had an index greater than 55%. Species with lower indicator values had weaker habitat associations and lower predictive power for the entire site group.

Although cluster analysis found the best solution with 3 groups, species assemblages at lower and higher cluster levels helped clarify species and site group relationships. At the second cluster level, groups 1 and 3 merged. The cluster included 5 species with a maximum indicator value > 55%: ring-necked pheasant, western meadowlark, eastern meadowlark, dickcissel, and grasshopper sparrow. This came as no surprise, because all 5 species had been classified as members of the grassland bird guild. Having reached a maximum indicator value at a broad level, which included light encroachment sites, indicated eurytopic species that tolerated a wide range of grassland habitats. Considering the ANOVA results for the grassland bird guild, this observation suggested that sites with woody encroachment might support similar abundance of grassland guild members if canopy levels were low (the canopy cover of light encroachment sites in this cluster ranged from 0.10 to 0.40%). In contrast, 6 moderate encroachment sites split from group 2 at the 5th cluster level. The core species, characterized by successional-shrub species, included indigo bunting, gray catbird (*Dumetella carolinensis*), blue jay, Bell's vireo, field sparrow, and northern cardinal.

The wide distribution of encroachment sites in the cluster analysis suggested landscape-level variation in habitat influenced bird communities. For example, the inclusion of a moderate encroachment site in group 3 was unexpected. The availability of adjacent grassland habitat and the absence of nearby shelterbelts or woodlands might explain the higher abundance of grassland guild members at this site. Coppedge et al. (2001) demonstrated that the amount of landscape fragmentation by tree cover influenced patch-level bird response. Finding possible patch-level and landscape-level influences, in association with light levels of woody canopy cover, demonstrated the complexity of bird community organization.

While I assumed variation in eastern redcedar canopy cover was driving bird responses, other habitat characteristics might have affected the observed bird-habitat associations. In a review of grassland bird habitat studies, bare ground cover, vegetation height, and litter depth were the best predictors of grassland bird-habitat selection (Fisher and Davis 2010). Canonical correspondence analysis included visual obstruction (a measure of vegetation height), percent grass cover, and litter depth in the full model, but the stepwise reduction of environmental variables indicated eastern redcedar canopy cover alone explained the most variation in species abundance. The relative position of species centroids and site points were used to infer how bird species responded along a gradient of increasing woody canopy cover. The co-occurrence and response of species within functional guilds was apparent by the relative position of species to sites along the canopy cover gradient. Among grassland-shrub guild members, no species were positioned at the upper end of the canopy gradient (sites with 12.3% canopy cover or greater). This suggested there might be a response threshold between 5% and 25% canopy cover. Chapman et al. (2004) found that mean abundance of field sparrow and lark sparrow peaked at 13.3% woody canopy cover. Similarly, Cooper (2009) reported that Bell's vireo abundance increased until canopy cover reached 20% in Oklahoma grassland-shrub habitats dominated by sandhill plum.

The variation in species abundance and distribution were influenced by the level of woody canopy cover, but to what extent was unclear. N-mixture models were used to further assess habitat use of 12 focal species along the canopy cover gradient. Models of bird-habitat associations were adequate for Bell's vireo, dickcissel, eastern meadowlark, and grasshopper sparrow. The estimates of mean abundance derived from model-fitting were comparable to the treatment level means used in the ANOVA, which showed these models provided a good indication of how abundance changed along the canopy cover gradient. Abundance models did not adequately explain the data for other species. This might be a consequence of species having low detection rates or other covariate effects introducing heterogeneity to the data. Wide confidence intervals showed the uncertainty of distinguishing between species absence and non-detection. In some cases, the estimates of mean abundance were unrealistically large, which demonstrated that the best model did not necessarily reflect ecological realism. To provide unbiased estimates of habitat-specific abundance the detection probability should be greater than 0.30 (MacKenzie et al. 2002). Also, there were 2 instances (eastern kingbird and northern bobwhite) where the null model was ranked as the best model. Although models with canopy cover were included in the model set, it seemed that eastern redcedar canopy cover was not a good predictor of abundance for these species and other factors should be considered.

Canopy Cover Estimation

While I assumed misclassification errors were relatively small and did not affect the primary analyses, spatial resolution might cause error and bias in remotely-sensed cover estimates (Raza et al. 2010). Small, isolated features were difficult to interpret and a likely source of misclassification error. Because large trees account for a greater proportion of the canopy cover than small trees (Wulder et al. 2000), omission of a few small trees was acceptable. Further, commission errors (false positives) helped compensate for false negative classifications. The accuracy assessment results showed canopy cover was underestimated, which indicated commission errors did not inflate canopy cover estimates. Since I did not have the time or resources to ground truth pixel clusters, the error rate is unknown.

Past studies have provided reasonable estimates of canopy cover by using ocular cover-plot estimates or ground-based methods measurements; however, these methods are not recommended because they lack precision (Nowak et al. 1996). In a comparative study of methods for estimating canopy cover (Avsar and Ayyildiz 2010), graphical methods provided the most precise estimates of canopy cover. Also, graphical methods are advantageous because they provide complete spatial coverage of an area of interest and can attain more precise estimates inspite of the heterogeneous spatial patterns of vegetation (Nowak et al. 1996). Using digital imagery analysis, I was able to classify sites with less than 1% canopy cover. Because there was agreement between the remote sensing classification and ground reference data, image classification provided a reasonably accurate representation of the actual canopy cover.

Management Implications

Grassland, ground-nesting species showed similar habitat associations and could be managed by maintaining open grasslands. In Conservation Reserve Program (CRP) fields, grassland birds were associated closely with stand age and cover types (Bakker et al. 2004). Management with prescribed fire and grazing has been recommended as a strategy to promote spatial and temporal heterogeneity in habitat availability for grassland birds (Fuhlendorf et al. 2006). Due to variation in disturbance frequency and patterns (Lorimer 2001), land managers should recognize that grasslands with scattered woody vegetation also are a part of the shifting mosaic of grassland community dynamics. Grassland-dependent species requiring woody vegetation (e.g., lark sparrow, field sparrow, and Bell's vireo) benefit from management practices that maintain adequate canopy cover. Although eastern redcedar has value for wildlife (Smith 1985), encroachment sites might function as surrogate habitat for bird species associated with successional-shrub habitats. Ecological site descriptions and encroachment risk ought to be assessed to determine which trees and shrubs are most appropriate for conservation uses. Based on my observations, low-growing shrub thickets of sandhill plum and fragrant sumac (*Rhus aromatica*) provided desirable wildlife habitat in mixed-grass prairies of central Kansas. Bell's vireo, brown thrasher, and field sparrow nest in plum thickets located in mixed prairie habitats of north-central Oklahoma (Dunkin and Guthery 2010). Sandhill plum thickets also enhance habitat for lark sparrows and northern bobwhites (Cooper 2009).

According to Vickery et al. (1999), "It is important to recognize that certain sites are usually best suited to management for a particular subset of grassland birds." Area-sensitive species (e.g., eastern meadowlark, grasshopper sparrow, and upland sandpiper) tend to avoid small, fragmented grasslands inspite of suitable cover (Herkert 1994, Ribic et al. 2009). However, small grassland fragments might have conservation value for declining species associated with successional-shrub habitat if managed accordingly. To benefit species with greater area requirements, managers should promote the restoration and conservation of large and open grasslands.

Smith (1985) recommended a common-sense approach to tree removal, which considers the needs of wildlife. "The reasonable control and removal of redcedars is necessary when a real problem exists," he noted, "However, all cedar stands should not be eradicated because the species is a nuisance it some areas." Eastern redcedar windbreaks provide excellent wildlife cover, especially in the winter and I do not foresee an end to conservation tree plantings. Therefore, managers should focus on preventative measures to curb the rate of woody encroachment, especially in areas where eastern redcedar encroachment conflicts with local or landscape-level management objectives. To eliminate the risk of encroachment from shelterbelts or windbreaks, I recommend using only male eastern redcedars in new conservation plantings. Additionally, shelterbelts or windbreaks could be renovated by selectively removing the seed-bearing female eastern redcedar and replacing them with male trees. In areas where eastern redcedar has spread to adjacent lands, I recommend targeting female trees first. Eliminating local seed sources will help mitigate encroachment and seed dispersal.

Because eastern redcedar control is more economical, effective, and time-conserving when trees are small (Ortmann et al. 1998), managers should be proactive about brush removal. Chemical application, prescribed burning, cutting, or a combination of treatments are recommended for controlling young eastern redcedar (Buehring et al. 1971, Owensby et al. 1973, Smith and Stubbendieck 1989, Ortmann et al. 1998). When tree height exceeds 1.8 m fire control is less effective (Martin and Crosby 1955, Owensby et al. 1973) and removing large eastern redcedar will most likely require cutting with a chain saw or tree shears (Ortmann et al. 1998). Although complete brush removal might be a desirable indictor of management success, the mixed response among bird guilds indicated the need to consider the full array of grassland birds and their habitats.

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Table 1. Study sites by treatment level for the 2011 sampling season in Barton County, Kansas, USA. Canopy cover estimates based on imagery from 2010.

2011 study sites		
Site name	**Treatment level**	**% Canopy cover**
BEND	Open grassland	NA
FLES1	Open grassland	NA
FLES2	Open grassland	NA
GREAT	Open grassland	NA
WCOM	Open grassland	NA
VHAM	Open grassland	NA
WIHA	Light encroachment	0.10
MANE	Light encroachment	0.38
RUPP	Light encroachment	0.40
TAYL	Light encroachment	0.40
OBRU	Light encroachment	0.41
WMIL	Light encroachment	2.30
CHUR	Light encroachment	2.39
ALDR	Moderate encroachment	7.57
HAMM	Moderate encroachment	12.28
LOGN	Moderate encroachment	21.41
BCCO	Moderate encroachment	23.64

Table 2. Study sites by treatment level for the 2012 sampling season in Barton County, Kansas, USA. Canopy cover estimates based on imagery from 2010.

2012 study sites		
Site name	**Treatment level**	**% Canopy cover**
BEND	Open grassland	NA
GELW	Open grassland	NA
GREAT	Open grassland	NA
REDW	Open grassland	NA
TNC	Open grassland	NA
WCOM	Open grassland	NA
TAYL	Light encroachment	0.40
MANE	Light encroachment	0.38
RUPP	Light encroachment	0.40
PETE	Light encroachment	0.95
CHUR	Light encroachment	2.39
WMIL	Light encroachment	2.92
INDE	Moderate encroachment	5.16
ELLI	Moderate encroachment	5.97
BART	Moderate encroachment	6.10
BCCO	Moderate encroachment	13.62
LOGN	Moderate encroachment	17.26

Table 3. Ecological guild classification of grassland-associated bird species based on nest placement and breeding habitat categories.

Common Name	Nest Placement	Breeding Habitat	Ecological Guild
Dickcissel	Ground-low	Grass-forb	Grassland
Eastern meadowlark	Ground-low	Grass-forb	Grassland
Grasshopper sparrow	Ground-low	Grass-forb	Grassland
Red-winged blackbird	Ground-low	Grass-forb	Grassland
Ring-necked pheasant	Ground-low	Grass-forb	Grassland
Upland sandpiper	Ground-low	Grass-forb	Grassland
Western meadowlark	Ground-low	Grass-forb	Grassland
Bell's vireo	Ground-low	Successional-shrub	Grassland-shrub
Cassin's sparrow	Ground-low	Successional-shrub	Grassland-shrub
Common yellowthroat	Ground-low	Successional-shrub	Grassland-shrub
Field sparrow	Ground-low	Successional-shrub	Grassland-shrub
Lark sparrow	Ground-low	Successional-shrub	Grassland-shrub
Mourning dove	Ground-low	Successional-shrub	Grassland-shrub
Northern bobwhite	Ground-low	Successional-shrub	Grassland-shrub
American goldfinch	Mid-canopy	Successional-shrub	Savanna
Eastern kingbird	Mid-canopy	Successional-shrub	Savanna
Western kingbird	Mid-canopy	Successional-shrub	Savanna

Table 4. Summary of bird species detected in 3 treatments during point-count surveys in Barton County, Kansas, USA, 2011-2012.

Common name	**Scientific name**	**Treatment**			**Total detections**
		Open grassland	**Light encroachment**	**Moderate encroachment**	
American crow	*Corvus brachyrhynchos*	0	15	9	24
American goldfinch	*Spinus tristis*	5	42	48	95
American kestrel	*Falco sparverius*	0	2	1	3
American robin	*Turdus migratorius*	2	4	10	16
Baltimore oriole	*Icterus galbula*	5	24	51	80
Barn swallow	*Hirundo rustica*	24	25	33	82
Bell's vireo	*Vireo bellii*	1	13	99	113
Brown-headed cowbird	*Molothrus ater*	130	217	141	488
Blue grosbeak	*Passerina caerulea*	0	1	5	6

Table 4. continued

Common name	Scientific name	Treatment			Total detections
		Open grassland	Light encroachment	Moderate encroachment	
Blue jay	*Cyanocitta cristata*	0	12	33	45
Bobolink	*Dolichonyx oryzivorus*	1	0	0	1
Brown thrasher	*Toxostoma rufum*	1	32	43	76
Blue-winged teal	*Anas discors*	1	0	0	1
Cattle egret	*Bubulcus ibis*	0	36	96	132
Canada goose	*Branta canadensis*	0	0	6	6
Cassin's sparrow	*Peucaea cassinii*	3	10	11	24
Clay-colored sparrow	*Spizella pallida*	2	0	0	2

Table 4. continued

Common name	Scientific name	Treatment			Total detections
		Open grassland	Light encroachment	Moderate encroachment	
Cedar waxwing	*Bombycilla cedrorum*	0	22	70	92
Chimney swift	*Chaetura pelagica*	1	0	4	5
Cliff swallow	*Petrochelidon pyrrhonota*	10	10	2	22
Common nighthawk	*Chordeiles minor*	1	7	9	17
Common yellowthroat	*Geothlypis trichas*	18	0	0	18
Chuck's-will-widow	*Antrostomus carolinensis*	0	0	1	1
Dickcissel	*Spiza americana*	858	413	98	1369
Downy woodpecker	*Picoides pubescens*	0	1	5	6
Eastern bluebird	*Sialia sialis*	3	5	6	14

Table 4. continued

Common name	Scientific name	Treatment			Total detections
		Open grassland	Light encroachment	Moderate encroachment	
Eastern kingbird	*Tyrannus tyrannus*	14	62	32	108
Eastern meadowlark	*Sturnella magna*	125	134	19	278
Eastern phoebe	*Sayornis phoebe*	0	1	7	8
European starling	*Sturnus vulgaris*	17	19	22	58
Field sparrow	*Spizella pusilla*	0	75	252	327
Great blue heron	*Ardea herodias*	2	0	0	2
Great crested flycatcher	*Myiarchus crinitus*	1	6	4	11
Great horned-owl	*Bubo virginianus*	1	6	4	11
Gray catbird	*Dumetella carolinensis*	0	0	5	5

Table 4. continued

Common name	Scientific name	Treatment			Total detections
		Open grassland	Light encroachment	Moderate encroachment	
Greater prairie-chicken	*Tympanuchus cupido*	5	0	0	5
Grasshopper sparrow	*Ammodramus savannarum*	222	257	54	533
Great-tailed grackle	*Quiscalus mexicanus*	1	1	4	6
House wren	*Troglodytes aedon*	0	17	43	60
Indigo bunting	*Passerina cyanea*	0	1	22	23
Killdeer	*Charadrius vociferus*	8	7	1	16
Lark sparrow	*Chondestes grammacus*	0	28	52	80
Mallard	*Anas platyrhynchos*	9	2	0	11
Mourning dove	*Zenaida macroura*	45	163	108	316

Table 4. continued

Common name	Scientific name	Treatment			Total detections
		Open grassland	Light encroachment	Moderate encroachment	
Northern bobwhite	*Colinus virginianus*	50	83	50	183
Northern cardinal	*Cardinalis cardinalis*	1	30	114	145
Northern flicker	*Colaptes auratus*	3	3	3	9
Northern mockingbird	*Mimus polyglottos*	1	31	59	91
Orchard oriole	*Icterus spurius*	1	17	8	26
Purple martin	*Progne subis*	2	3	2	7
Red-bellied woodpecker	*Melanerpes carolinus*	0	1	1	2
Red-eyed vireo	*Vireo olivaceus*	0	0	2	2

Table 4. continued

Common name	Scientific name	Treatment			Total detections
		Open grassland	Light encroachment	Moderate encroachment	
Red-headed woodpecker	*Melanerpes erythrocephalus*	6	9	5	20
Ring-necked pheasant	*Phasianus colchicus*	57	56	8	121
Red-tailed hawk	*Buteo jamaicensis*	5	2	3	10
Ruby-throated hummingbird	*Archilochus colubris*	0	0	1	1
Red-winged blackbird	*Agelaius phoeniceus*	146	317	30	493
Scissor-tailed flycatcher	*Tyrannus forficatus*	0	2	0	2
Swainson's hawk	*Buteo swainsoni*	0	0	1	1
Tree swallow	*Tachycineta bicolor*	1	0	0	1

Table 4. continued

Common name	Scientific name	Treatment			Total detections
		Open grassland	Light encroachment	Moderate encroachment	
Turkey vulture	*Cathartes aura*	1	4	9	14
Upland sandpiper	*Bartramia longicauda*	15	17	5	37
Warbling vireo	*Vireo gilvus*	1	11	5	17
Western kingbird	*Tyrannus verticalis*	18	48	25	91
Western meadowlark	*Sturnella neglecta*	40	72	11	123
Wild turkey	*Meleagris gallopavo*	2	7	20	29
Wood duck	*Aix sponsa*	1	1	0	2
Yellow-breasted chat	*Icteria virens*	0	0	7	7
Yellow-billed cuckoo	*Coccyzus americanus*	0	6	11	17

Table 4. continued

Common name	Scientific name	Treatment			Total detections
		Open grassland	**Light encroachment**	**Moderate encroachment**	
Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	0	0	3	3
Yellow warbler	*Setophaga petechia*	0	7	0	7

Table 5. Summary of model selection, by species, using Poisson (Pois) and zero-inflated Poisson (ZIP) mixture distributions. Models are sorted by differences in Akaike's Information Criterion (ΔAIC) between candidate models and the best model. K is the number of model parameters. Detection probability (p) is reported with standard error (SE). Non-significant chi-square values indicate models that fit the data adequately.

Model, by species	**Mixture**	**K**	**AIC**	**ΔAIC**	**AIC weight**	**p**	**SE**	X^2	***P*-value**
Bell's vireo									
λ(canopy)p(.)	Pois	3	374.98	0	0.68	0.41	0.06	494.5	0.465
λ(canopy)p(.)	ZIP	4	376.98	2	0.25	0.41	0.06	440	0.663
λ(canopy + year)p(.)	Pois	4	379.43	4.45	0.07	0.50	0.04	459.5	0.871
λ(canopy + year)p(.)	ZIP	5	393.11	18.12	0.00	0.49	0.05	559	0.871
λ(.)p(.)ZIP	ZIP	3	412.31	37.33	0.00	0.38	0.07	317	0.584
λ(year)p(.)	ZIP	4	416.38	41.4	0.00	0.46	0.06	340	0.416
λ(.)p(.)	Pois	2	418.56	43.58	0.00	0.44	0.06	606	0.000
λ(year)p(.)	Pois	3	421.86	46.88	0.00	0.50	0.06	575	0.188

Table 5. continued

Model, by species	Mixture	K	AIC	ΔAIC	AIC weight	p	SE	X^2	*P*-value
Brown thrasher									
λ(canopy)p(.)	Pois	3	408.97	0	0.63	0.11	0.05	605.4	0.010
λ(canopy)p(.)	ZIP	4	410.37	1.4	0.31	0.09	0.06	676	0.554
λ(canopy + year)p(.)	Pois	4	413.83	4.86	0.06	0.20	0.04	613.2	0.069
λ(.)p(.)	ZIP	3	417.83	8.86	0.01	0.01	0.06	6178	0.624
λ(canopy + year)p(.)	ZIP	5	421.46	12.49	0.00	0.24	0.05	370	0.475
λ(.)p(.)	Pois	2	424.3	15.34	0.00	0.11	0.05	643.5	0.000
λ(year)p(.)	ZIP	4	433.37	24.4	0.00	0.26	0.05	320	0.010
λ(year)p(.)	Pois	3	437.54	28.58	0.00	0.01	NA	457	NA

Table 5. continued

Model, by species	Mixture	K	AIC	ΔAIC	AIC weight	p	SE	X^2	*P*-value
Dickcissel									
λ(canopy)p(.)	ZIP	4	1364.09	0	1.00	0.17	0.04	3320	0.356
λ(.)p(.)	ZIP	3	1444.72	80.63	0.00	0.24	0.04	1755	0.495
λ(canopy)p(.)	Pois	3	1451.15	87.06	0.00	0.31	0.03	410	< 0.001
λ(canopy + year)p(.)	ZIP	5	1459.73	95.64	0.00	0.49	0.02	5723	0.040
λ(canopy + year)p(.)	Pois	4	1493.07	128.97	0.00	0.50	0.02	877	< 0.001
λ(year)p(.)	ZIP	4	1523.95	159.85	0.00	0.49	0.02	897	< 0.001
λ(.)p(.)	Pois	2	1575.09	211	0.00	0.37	0.03	1198	< 0.001
λ(year)p(.)	Pois	3	1598.13	234.03	0.00	0.50	0.02	437	< 0.001

Table 5. continued

Model, by species	Mixture	K	AIC	ΔAIC	AIC weight	p	SE	X^2	*P*-value
Eastern kingbird									
λ(.)p(.)	ZIP	3	470.09	0	0.50	0.00	0.00	16792	0.238
λ(canopy)p(.)	ZIP	4	470.12	0.037	0.49	0.01	0.00	12375	0.238
λ(canopy)p(.)	Pois	3	480.08	9.995	0.00	0.05	0.05	682	< 0.001
λ(.)p(.)	Pois	2	483.91	13.82	0.00	0.05	0.05	676	< 0.001
λ(year)p(.)	ZIP	4	485.16	15.071	0.00	0.17	0.06	475	0.020
λ(canopy + year)p(.)	ZIP	5	490.42	20.333	0.00	0.15	0.04	495	0.010
λ(year)p(.)	Pois	3	496.67	26.585	0.00	0.23	0.04	633	0.079
λ(canopy + year)p(.)	Pois	4	505.07	34.983	0.00	0.31	0.04	581	0.089

Table 5. continued

Model, by species	**Mixture**	**K**	**AIC**	**ΔAIC**	**AIC weight**	**p**	**SE**	X^2	***P*-value**
Eastern meadowlark									
λ(canopy)p(.)	Pois	3	825.32	0	0.73	0.34	0.05	405	0.960
λ(canopy)p(.)	ZIP	4	827.32	2.01	0.27	0.34	0.05	5257	0.347
λ(.)p(.)	Pois	2	899.02	73.7	0.00	0.40	0.04	497	0.515
λ(.)p(.)	ZIP	3	901.02	75.71	0.00	0.40	0.05	439	0.683
λ(year)p(.)	Pois	3	908.55	83.23	0.00	0.50	0.03	532	0.406
λ(canopy + year)p(.)	Pois	4	910.55	85.23	0.00	0.50	0.03	532	0.356
λ(year)p(.)	ZIP	4	999.08	173.76	0.00	0.50	0.04	532	< 0.001
λ(canopy + year)p(.)	ZIP	5	1001.08	175.76	0.00	0.50	0.04	532	0.010

Table 5. continued

Model, by species	**Mixture**	**K**	**AIC**	**ΔAIC**	**AIC weight**	**p**	**SE**	***X^2***	***P*-value**
Field sparrow									
λ(canopy)p(.)	ZIP	4	691.72	0	0.96	0.03	0.02	5109	0.356
λ(canopy)p(.)	Pois	3	697.93	6.21	0.04	0.20	0.05	748	0.000
λ(.)p(.)	ZIP	3	713.52	21.8	0.00	0.02	0.01	11528	0.356
λ(canopy + year)p(.)	Pois	4	721.13	29.41	0.00	0.43	0.03	656	0.059
λ(canopy + year)p(.)	ZIP	5	730.02	38.3	0.00	0.44	0.05	1047	0.109
λ(year)p(.)	ZIP	4	745.29	53.57	0.00	0.31	0.04	782	0.000
λ(.)p(.)	Pois	2	868.96	177.24	0.00	0.35	0.04	1120	0.000
λ(year)p(.)	Pois	3	880.09	188.37	0.00	0.47	0.03	1024	0.000

Table 5. continued

Model, by species	Mixture	K	AIC	ΔAIC	AIC weight	p	SE	X^2	*P*-value
Grasshopper sparrow									
λ(canopy)p(.)	Pois	3	1117.39	0	0.58	0.37	0.04	605	0.802
λ(canopy)p(.)	ZIP	4	1118.06	0.68	0.42	0.34	0.05	962	0.574
λ(.)p(.)	ZIP	3	1168.59	51.2	0.00	0.33	0.06	943	0.515
λ(.)p(.)	Pois	2	1178.76	61.37	0.00	0.41	0.04	605	0.158
λ(canopy + year)p(.)	ZIP	5	1179.64	62.25	0.00	0.53	0.03	428	< 0.001
λ(year)p(.)	ZIP	4	1199.65	82.26	0.00	0.53	0.03	428	0.921
λ(year)p(.)	Pois	3	1258.54	141.15	0.00	0.50	0.02	543	< 0.001
λ(canopy + year)p(.)	Pois	4	1260.54	143.15	0.00	0.50	0.02	548	< 0.001

Table 5. continued

Model, by species	Mixture	K	AIC	ΔAIC	AIC weight	p	SE	X^2	*P*-value
Lark sparrow									
λ(canopy)p(.)	ZIP	4	348.02	0	1.00	0.02	0.01	3549	0.119
λ(.)p(.)	ZIP	3	363.1	15.09	0.00	0.01	0.01	8921	0.188
λ(canopy + year)p(.)	ZIP	5	373.39	25.38	0.00	0.24	0.06	652	0.891
λ(year)p(.)	ZIP	4	373.54	25.52	0.00	0.17	0.05	398	0.020
λ(canopy)p(.)	Pois	3	380.47	32.45	0.00	0.08	0.05	809.4	< 0.001
λ(canopy + year)p(.)	Pois	4	386.08	38.06	0.00	0.18	0.05	772	< 0.001
λ(.)p(.)	Pois	2	395.34	47.32	0.00	0.11	0.05	950	< 0.001
λ(year)p(.)	Pois	3	398.99	50.98	0.00	0.05	0.22	960.7	< 0.001

Table 5. continued

Model, by species	Mixture	K	AIC	ΔAIC	AIC weight	p	SE	X^2	*P*-value
Northern bobwhite									
λ(.)p(.)	ZIP	3	735.24	0	0.50	0.14	0.06	943	0.515
λ(.)p(.)	Pois	2	736.79	1.54	0.23	0.20	0.05	605	< 0.001
λ(canopy)p(.)	ZIP	4	737.21	1.96	0.19	0.14	0.06	962	0.505
λ(canopy)p(.)	Pois	3	738.76	3.52	0.09	0.20	0.05	605	< 0.001
λ(year)p(.)	Pois	3	760.48	25.23	0.00	0.42	0.03	543	0.327
λ(canopy + year)p(.)	Pois	4	762.47	27.23	0.00	0.42	0.03	548	0.238
λ(year)p(.)	ZIP	4	824.29	89.05	0.00	0.50	0.03	428	0.139
λ(canopy + year)p(.)	ZIP	5	826.29	91.05	0.00	0.50	0.03	428	0.267

Table 5. continued

Model, by species	**Mixture**	**K**	**AIC**	**ΔAIC**	**AIC weight**	**p**	**SE**	$\boldsymbol{X^2}$	***P*-value**
Ring-necked pheasant									
λ(canopy)p(.)	Pois	3	527.38	0	0.73	0.13	0.06	710	0.020
λ(canopy)p(.)	ZIP	4	529.38	2	0.27	0.13	0.06	998	0.713
λ(canopy + year)p(.)	ZIP	5	542.23	14.85	0.00	0.23	0.05	948	0.703
λ(canopy + year)p(.)	Pois	4	545.96	18.58	0.00	0.37	0.04	639	0.099
λ(.)p(.)	Pois	2	557.53	30.15	0.00	0.17	0.05	517	0.297
λ(.)p(.)	ZIP	3	559.32	31.95	0.00	0.15	0.07	589	0.614
λ(year)p(.)	ZIP	4	567.91	40.53	0.00	0.24	0.05	380	0.396
λ(year)p(.)	Pois	3	571.22	43.84	0.00	0.01	0.00	552	0.228

Table 5. continued

Model, by species	Mixture	K	AIC	ΔAIC	AIC weight	p	SE	X^2	*P*-value
Upland sandpiper									
λ(canopy)p(.)	Pois	3	175.68	0	0.53	0.05	0.07	476	0.277
λ(canopy)p(.)ZIP	ZIP	4	176.74	1.06	0.31	0.00	0.00	11262	0.386
λ(.)p(.)	Pois	2	179.57	3.89	0.08	0.06	0.07	531	0.149
λ(.)p(.)ZIP	ZIP	3	180.69	5.01	0.04	0.00	0.00	12424	0.347
λ(canopy + year)p(.)	Pois	4	182.12	6.44	0.02	0.13	0.07	770.7	0.040
λ(year)p(.)	Pois	3	182.23	6.55	0.02	0.13	0.07	505.2	0.574
λ(canopy + year)p(.)	ZIP	5	189.84	14.16	0.00	0.15	0.08	14500.7	0.891
λ(year)p(.)ZIP	ZIP	4	194.77	19.08	0.00	0.35	0.08	209.6	0.723

Table 5. continued

Model, by species	**Mixture**	**K**	**AIC**	**ΔAIC**	**AIC weight**	**p**	**SE**	X^2	***P*-value**
Western kingbird									
λ(canopy)p(.)	Pois	3	363.43	0	0.36	0.04	0.05	552.8	0.089
λ(canopy)p(.)	ZIP	4	364.08	0.65	0.26	0.01	0.02	10773	0.347
λ(.)p(.)	Pois	2	364.96	1.53	0.17	0.05	0.05	559	0.030
λ(.)p(.)	ZIP	3	365.31	1.88	0.14	0.00	0.00	308	0.248
λ(year)p(.)	Pois	3	367.43	4	0.05	0.01	NA	562.2	0.386
λ(canopy + year)p(.)	Pois	4	370.54	7.11	0.01	0.13	0.04	579.4	0.208
λ(canopy + year)p(.)	ZIP	5	376.31	12.89	0.00	0.15	0.04	298	0.020
λ(year)p(.)	ZIP	4	380.06	16.63	0.00	0.25	0.05	271	0.188

Figure 1. Map of study area in Barton County, Kansas, USA with study sites represented by dots.

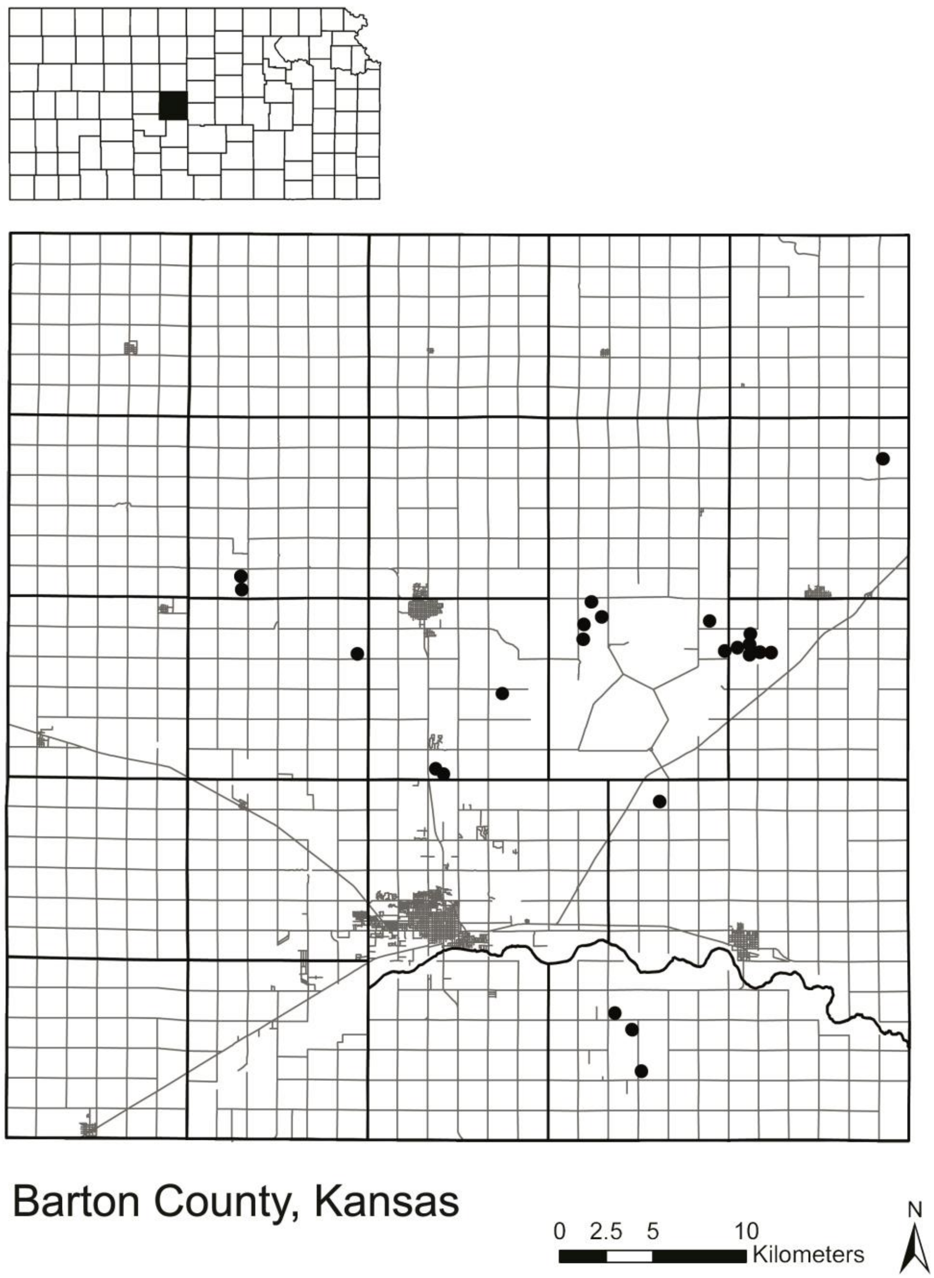


Figure 2. Relationship between ground-based and remote-sensed counts of individual eastern redcedar in Barton County, Kansas, USA. The dashed line is the 1:1 isoline of the predicted results. The solid line is the linear line of best-fit to the data.

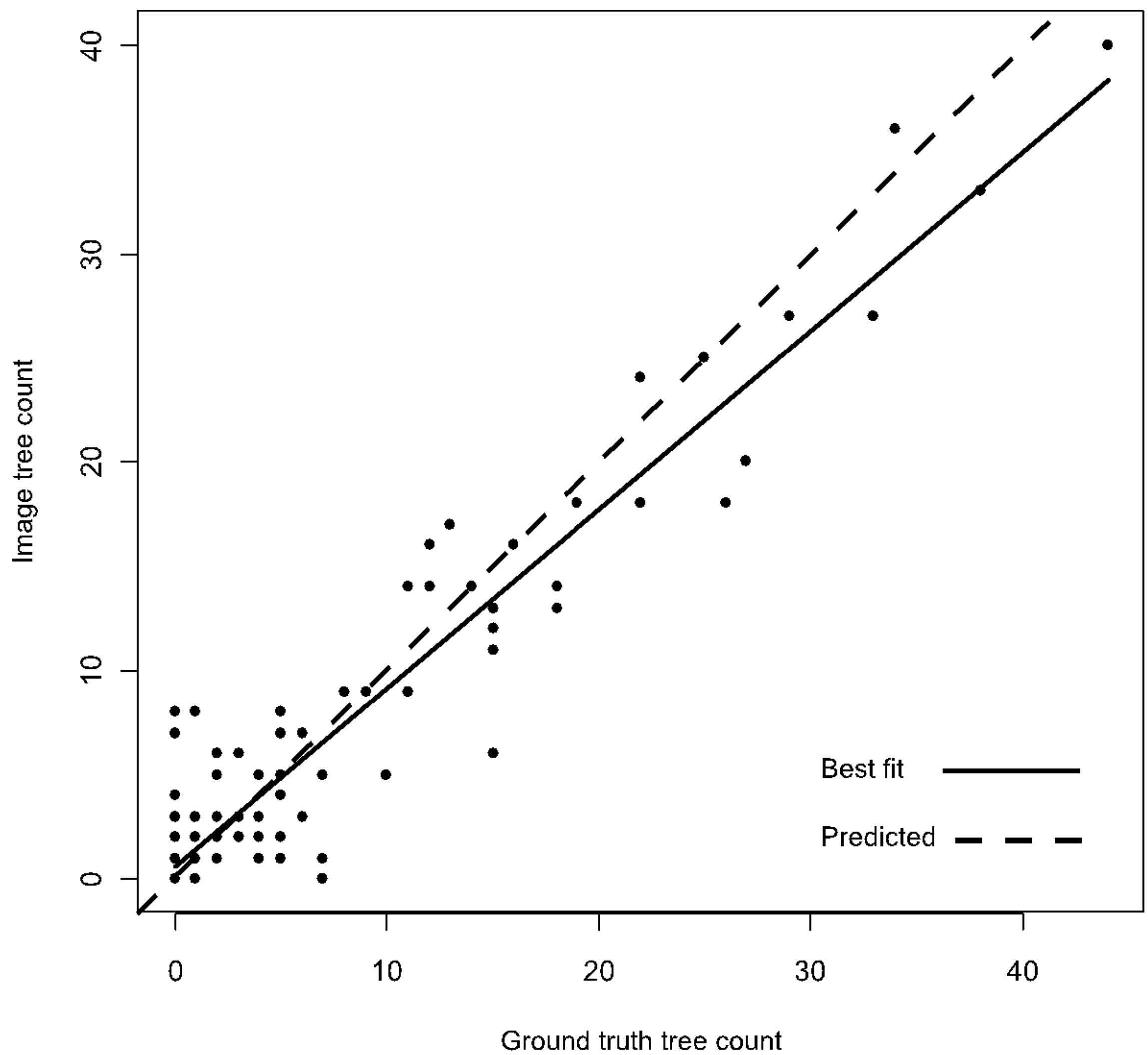


Figure 3. Bar graphs of significant pairwise differences between bird guilds in 3 treatment levels, as determined by Tukey's *post hoc* test following ANOVA. Letters indicate whether pairwise comparisons between treatment levels were significantly different.

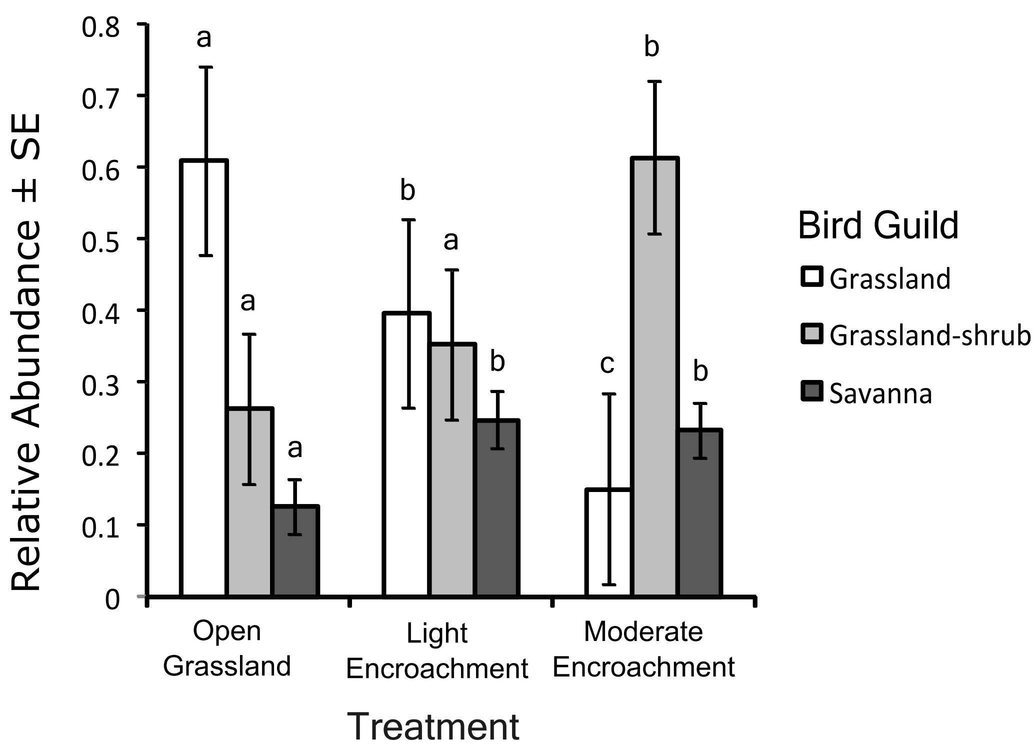


Figure 4. Hierarchical cluster dendrogram (Bray-Curtis distance with the flexible beta linkage method) of study sites based on bird species abundance data. Birds were sampled at 3 treatment levels (G = open grassland, L = light encroachment, M = moderate encroachment). The numbers indicate sites and sampling year (e.g. M3-2 is moderate encroachment site number 3 in year 2 of study).

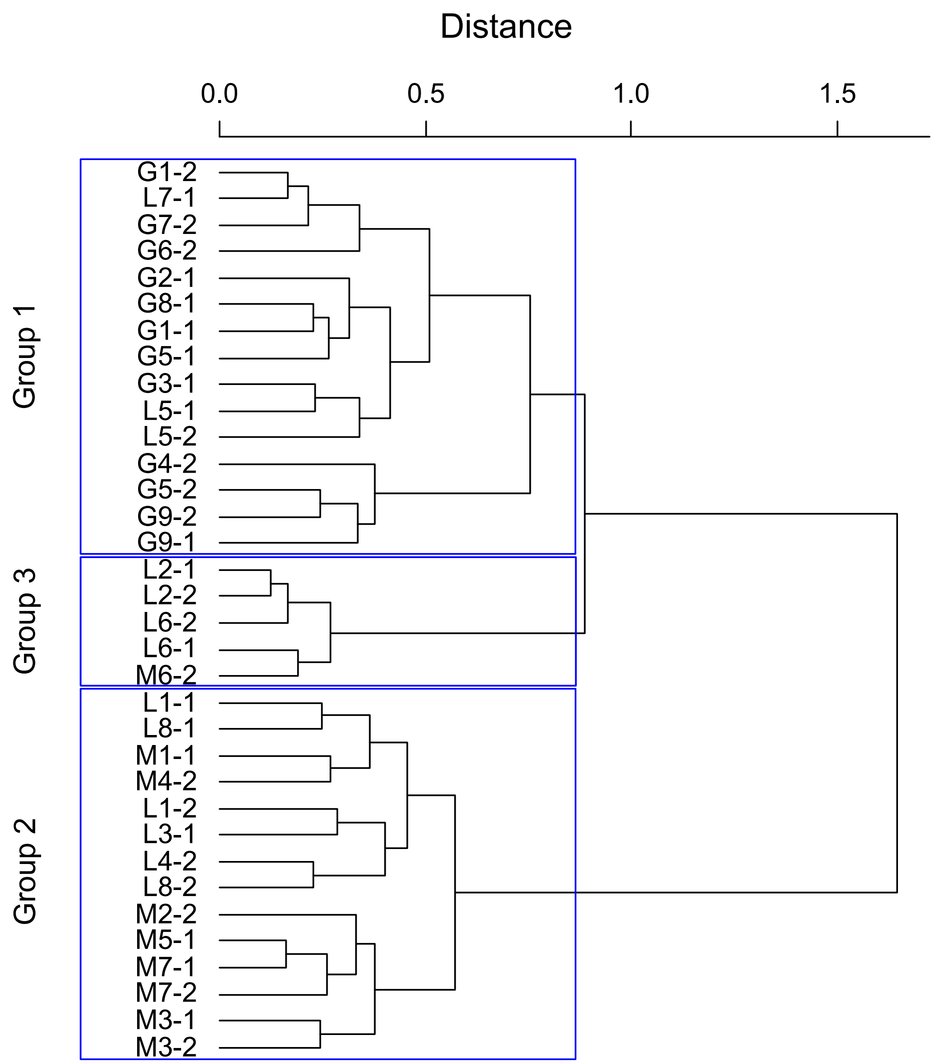


Figure 5. Dendrogram of species assemblages as defined by site group clustering. All species with an indicator value ≥ 25% at the 0.05 significance level are included. Indicator values are shown in parentheses and the maximum indicator value for each species is bolded. Values > 55% indicate characteristic species, which contribute to the specificity of the site groups and whose presence can be predicted within the group.

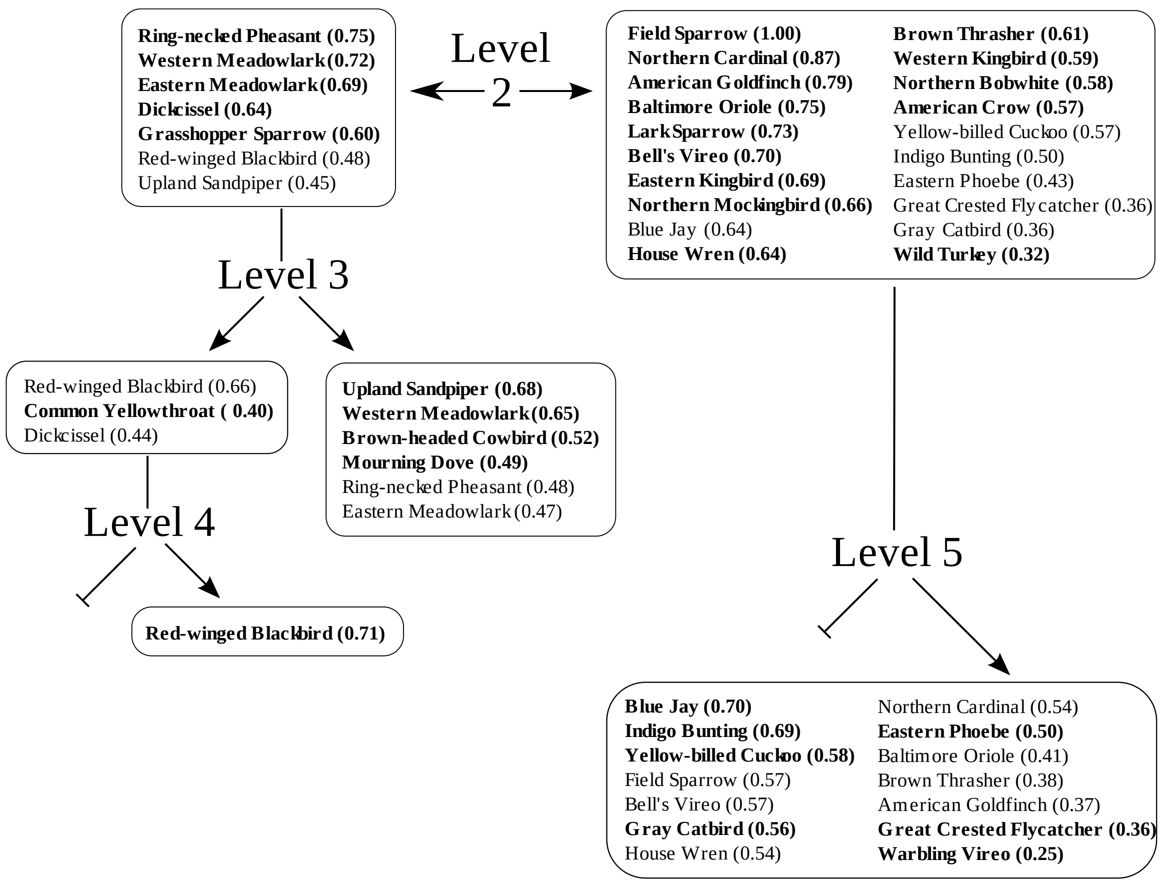
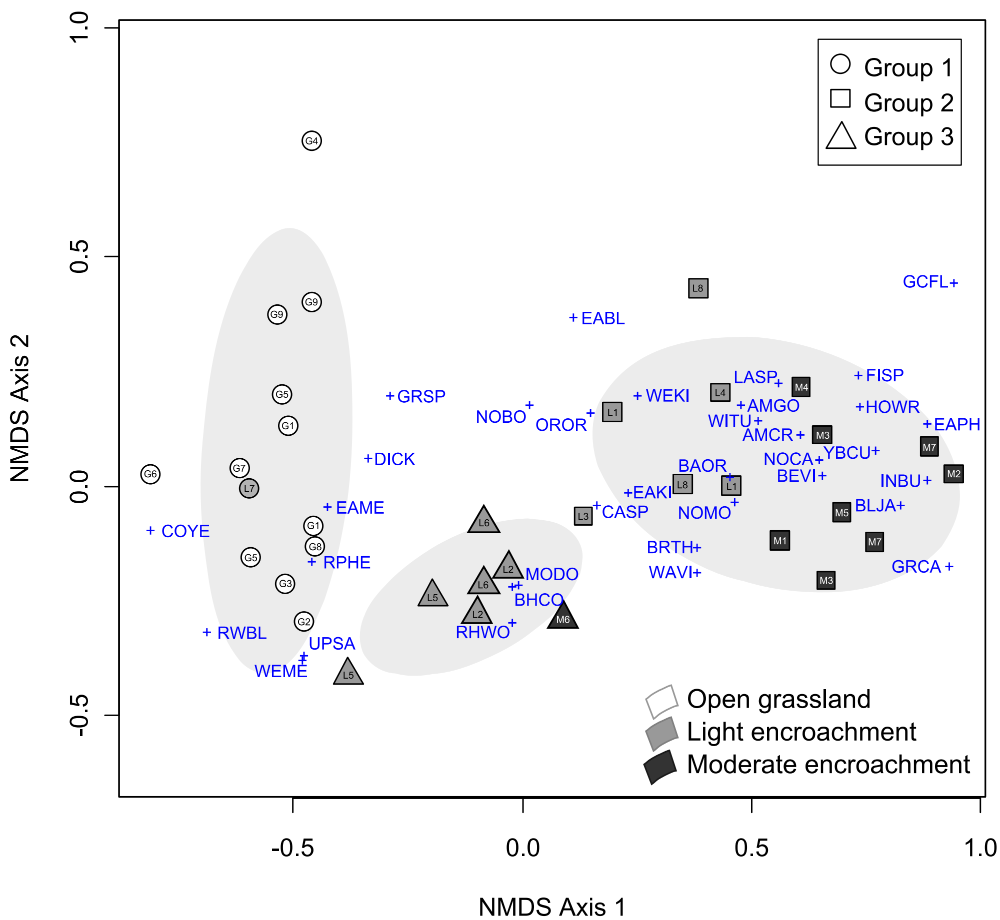


Figure 6. Non-metric multidimensional scaling ordination plot of bird community structure related to study sites divided among 3 treatment levels: open grassland, light encroachment, and moderate encroachment. The light gray ellipses are 75% confidence regions for the 3 groups. Site codes as in Figure 4.



Species Alpha Codes:

AMCR - American crow
AMGO - American goldfinch
BAOR - Baltimore oriole
BEVI - Bell's vireo
BHCO - Brown-headed cowbird
BLJA - Blue jay
BRTH - Brown thrasher
CASP - Cassin's sparrow
COYE - Common yellowthroat
DICK - Dickcissel
EABL - Eastern bluebird
EAKI - Eastern kingbird
EAME - Eastern meadowlark
EAPH - Eastern phoebe
FISP - Field sparrow
GCFL - Great crested flycatcher
GRCA - Gray catbird
GRSP - Grasshopper sparrow
HOWR - House wren
INBU - Indigo bunting
LASP - Lark sparrow
MODO - Mourning dove
NOBO - Northern bobwhite
NOCA - Northern cardinal
NOMO - Northern mockingbird
OROR - Orchard oriole
RHWO - Red-headed woodpecker
RPHE - Ring-necked pheasant
RWBL - Red-winged blackbird
UPSA - Upland sandpiper
WAVI - Warbling vireo
WEKI - Western kingbird
WEME- Western meadowlark
WITU - Wild turkey
YBCU - Yellow-billed cuckoo

Figure 7. Species-conditional canonical correspondence analysis (CCA) triplot of bird species, study sites, and an environmental variable. Increasing eastern redcedar canopy cover is represented by an arrow. Open circles with values for percent canopy cover indicate each study site. Plus signs indicate the distribution centroids of 12 focal bird species. The x-axis (CCA1) shows the constrained solution and the y-axis (CA1) is the first residual axis.

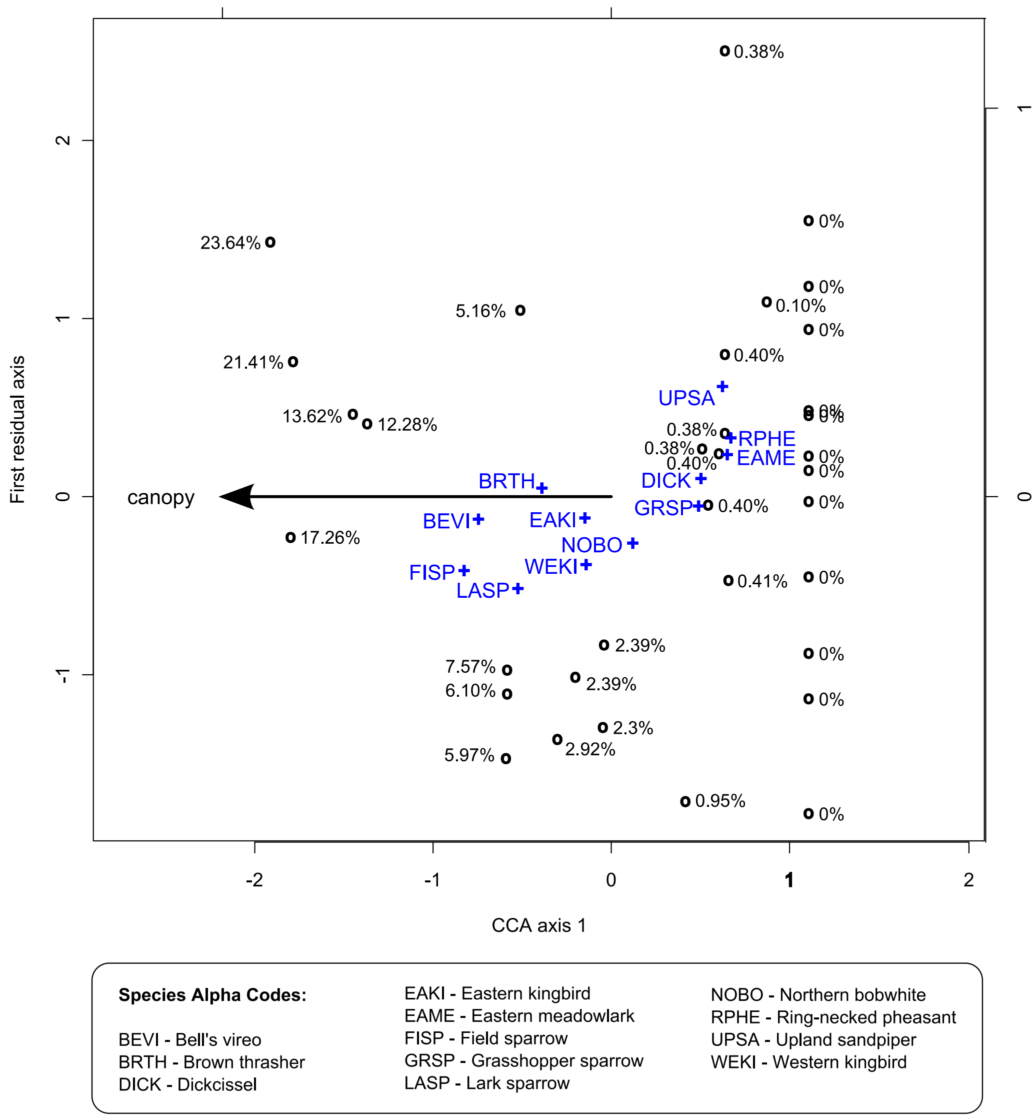


Figure 8. (A) Expected abundance of Bell's vireo as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

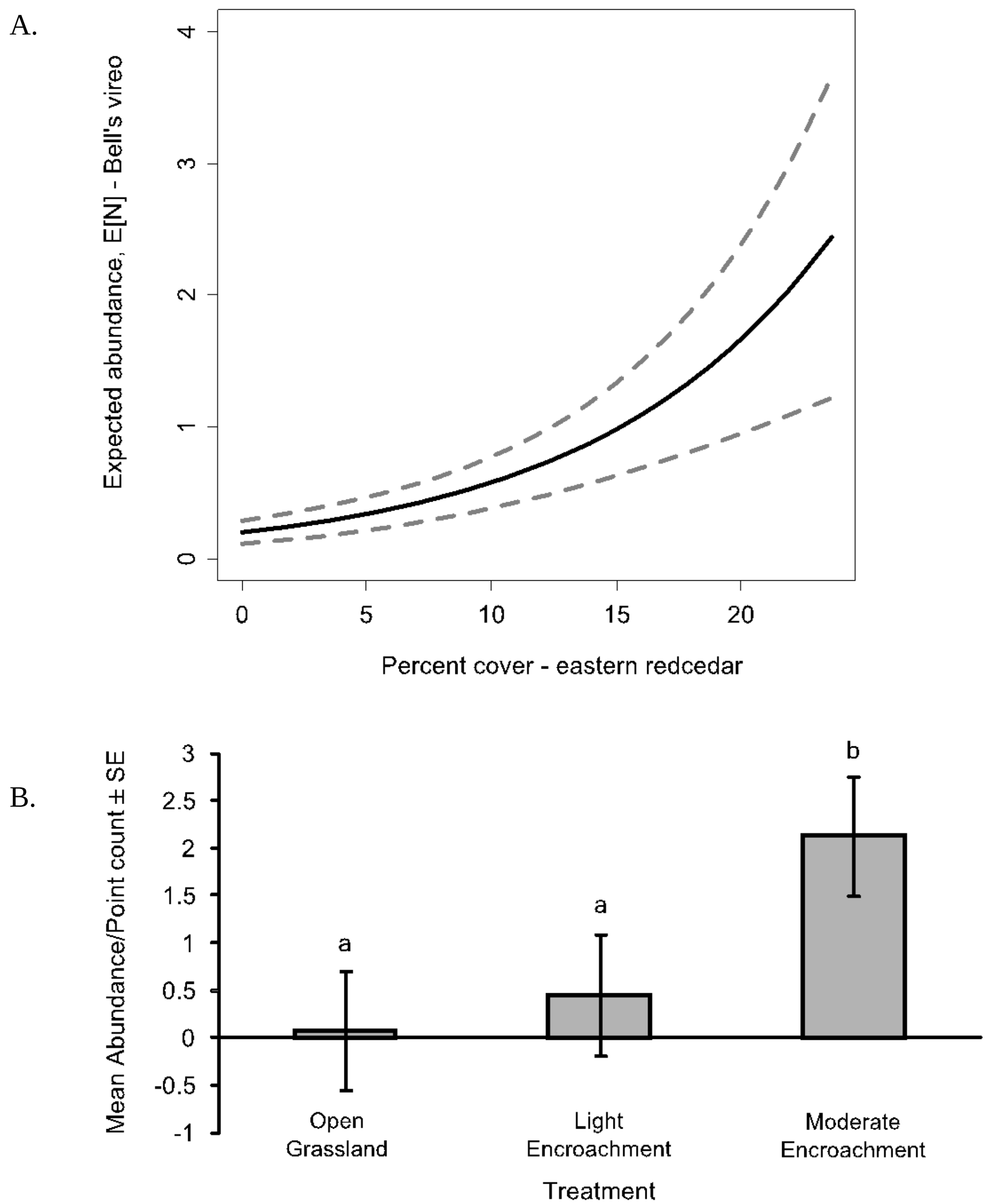


Figure 9. (A) Expected abundance of brown thrasher as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

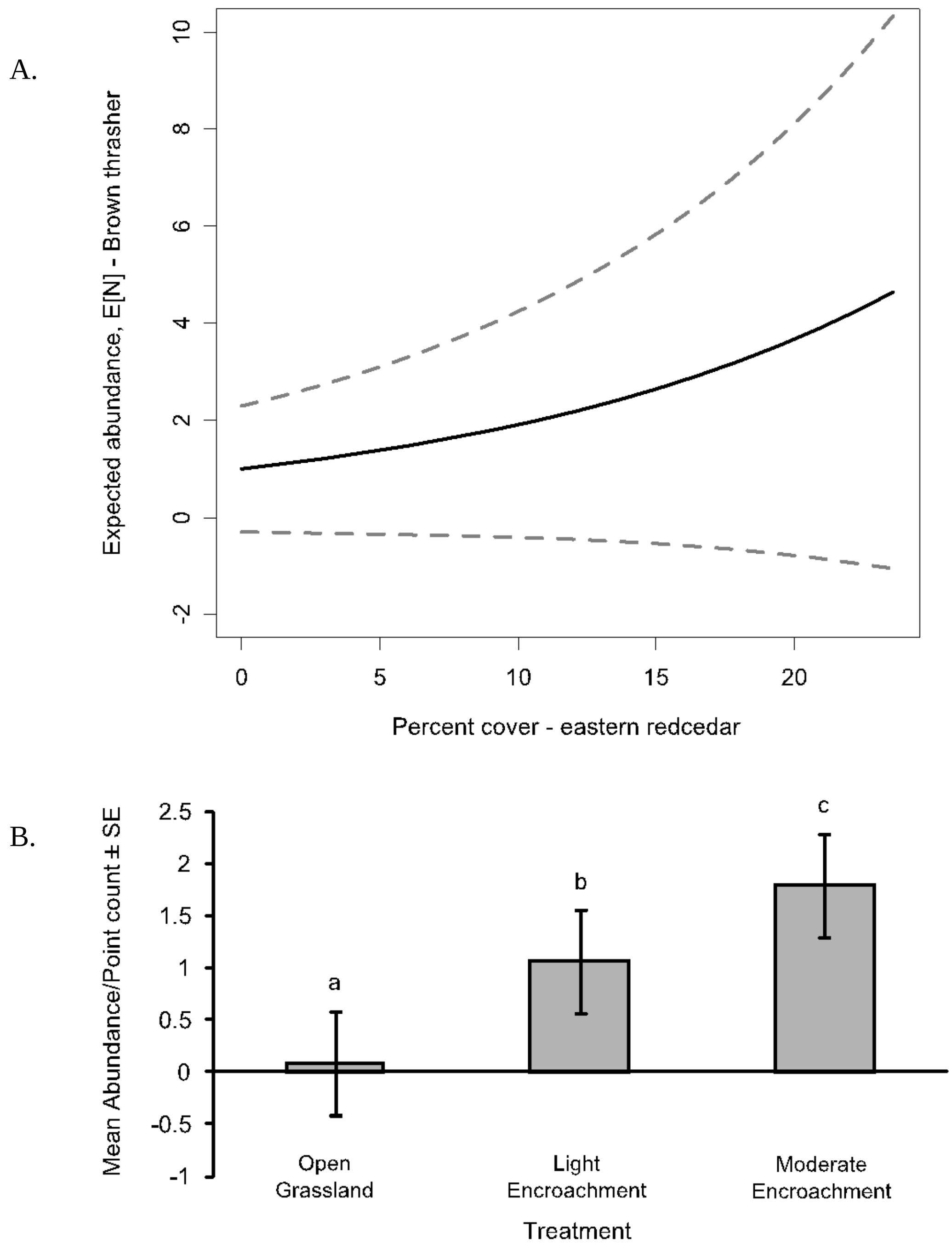
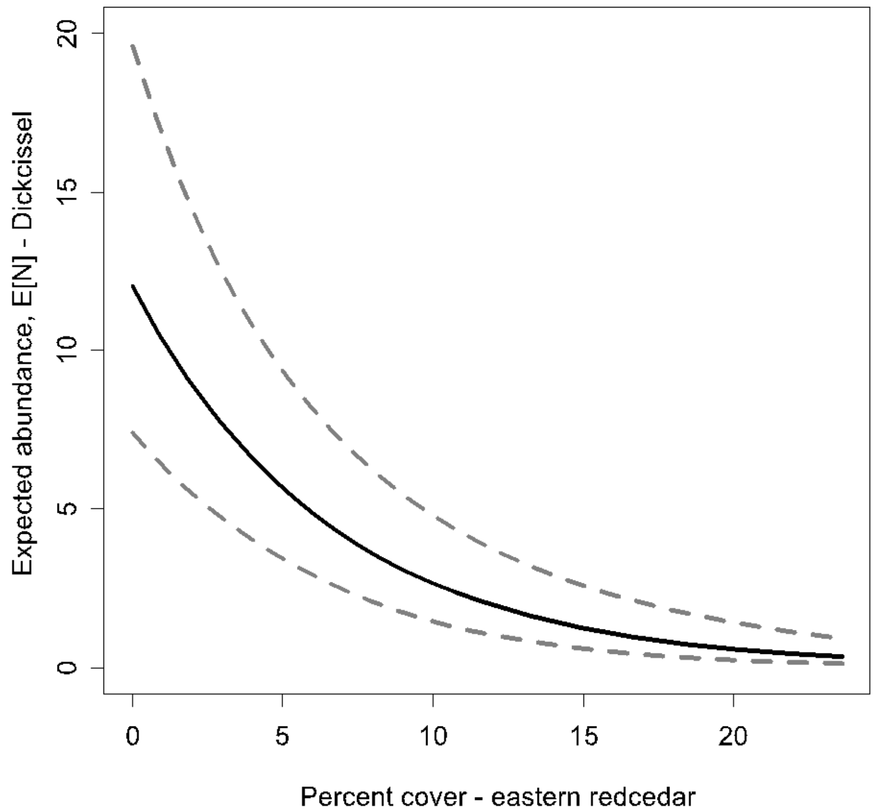


Figure 10. (A) Expected abundance of dickcissel as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.



B.

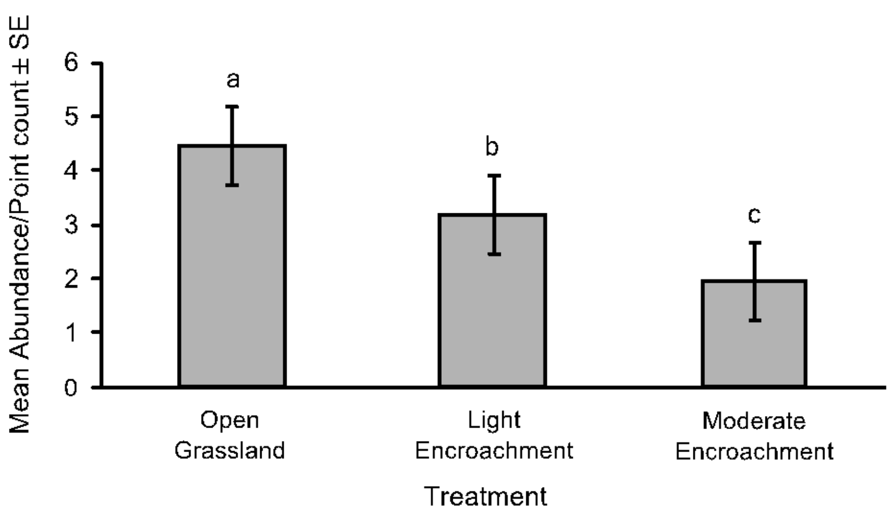
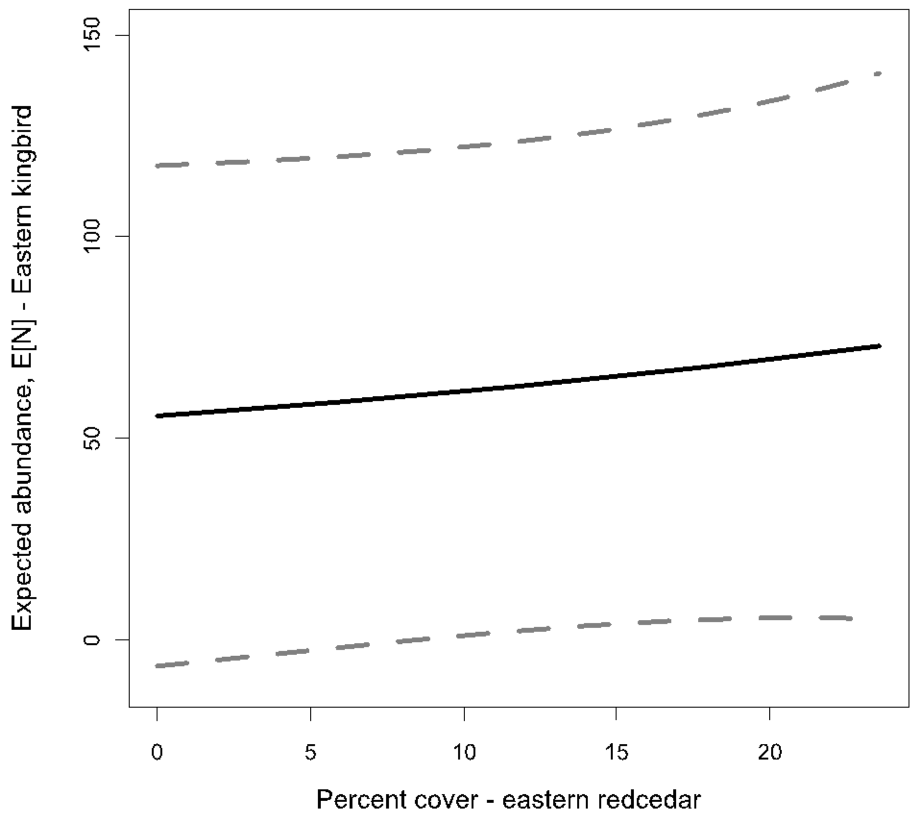


Figure 11. (A) Expected abundance of eastern kingbird as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.



B.

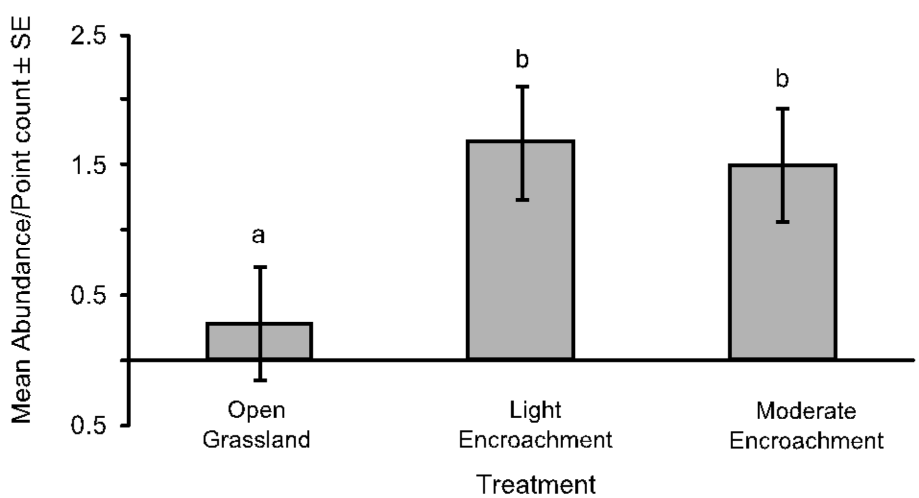
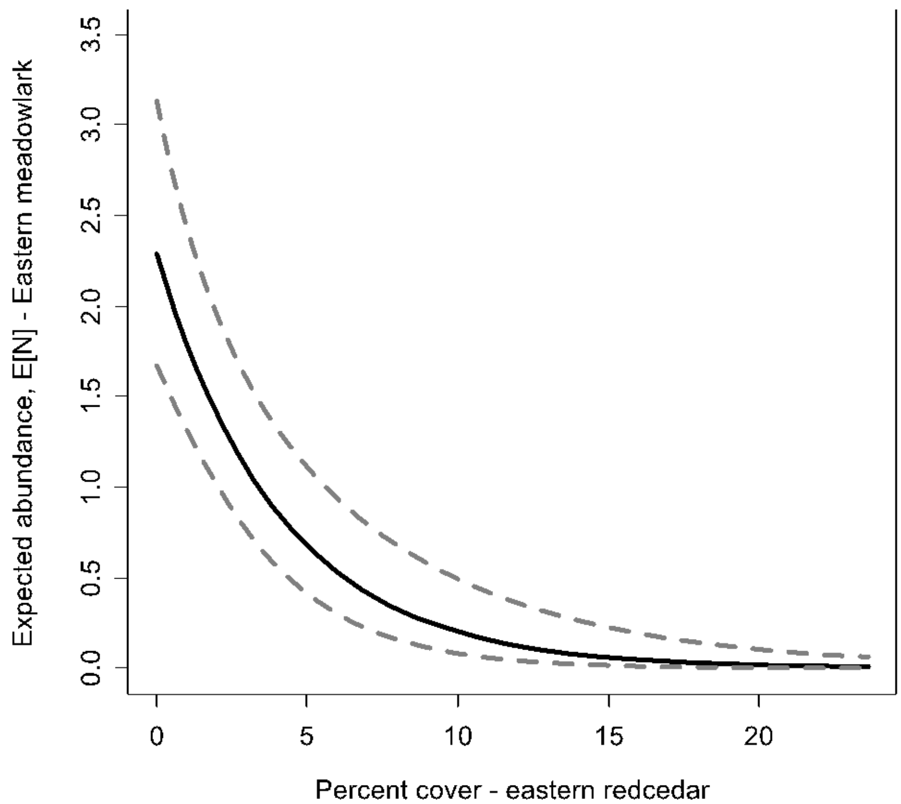


Figure 12. (A) Expected abundance of eastern meadowlark as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.



B.

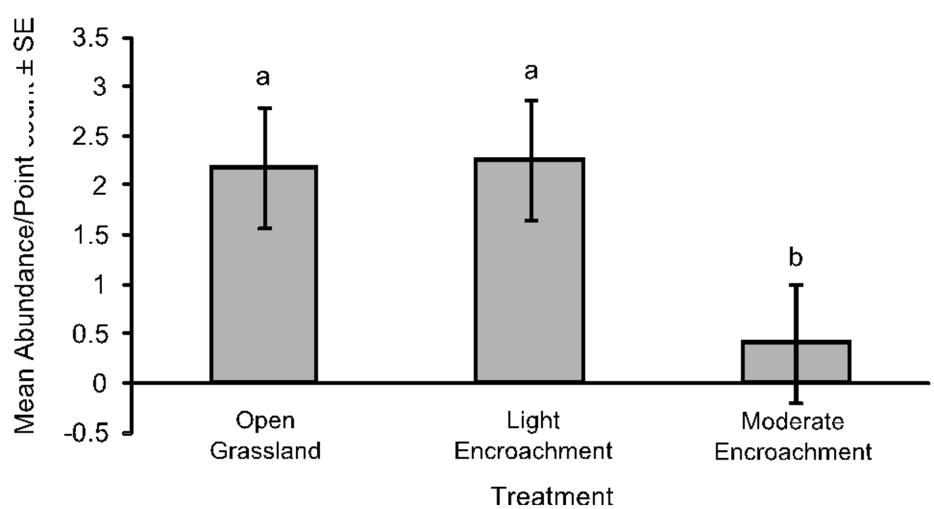
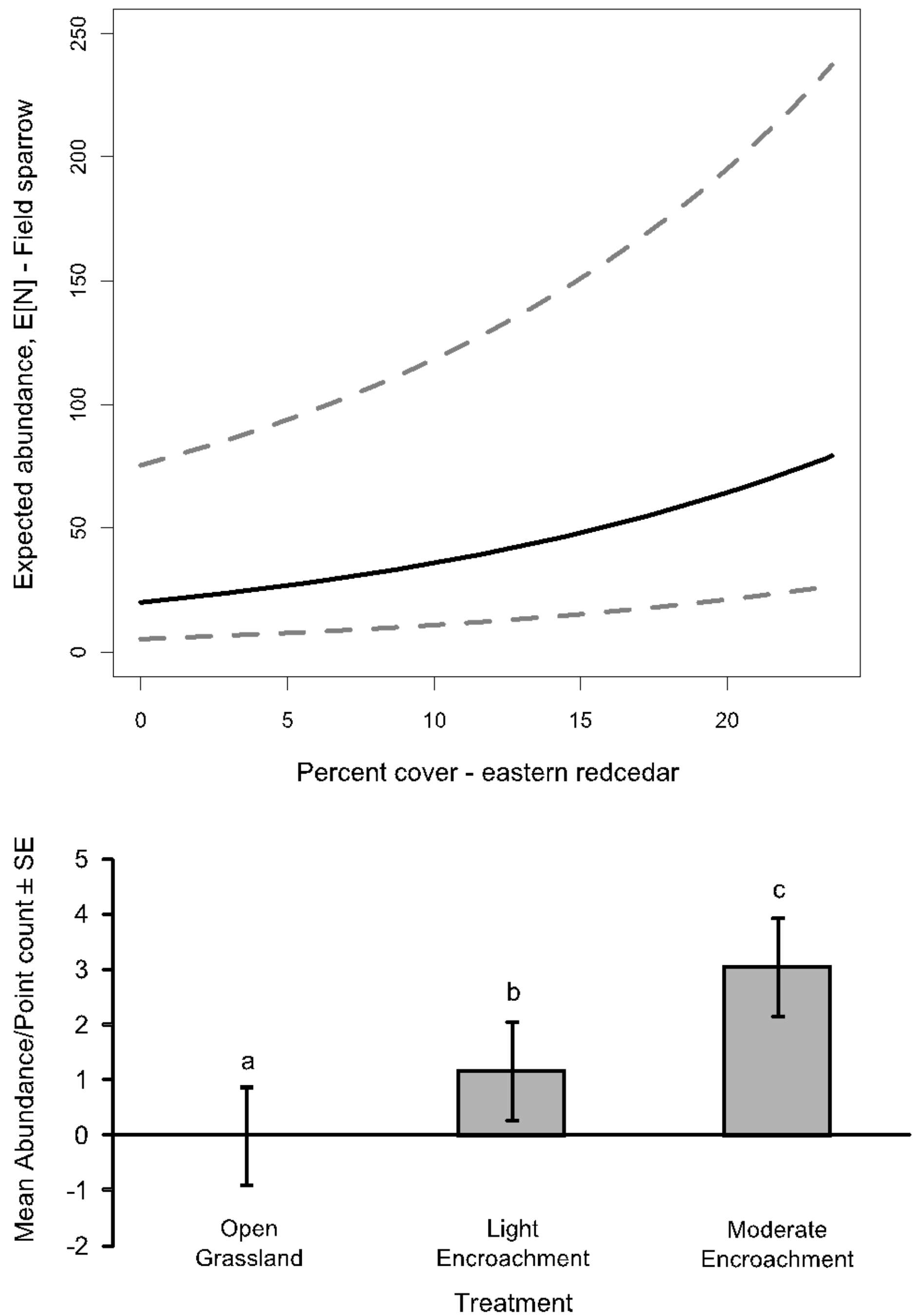


Figure 13. (A) Expected abundance of field sparrow as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.



B.

Figure 14. (A) Expected abundance of grasshopper sparrow as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.

B.

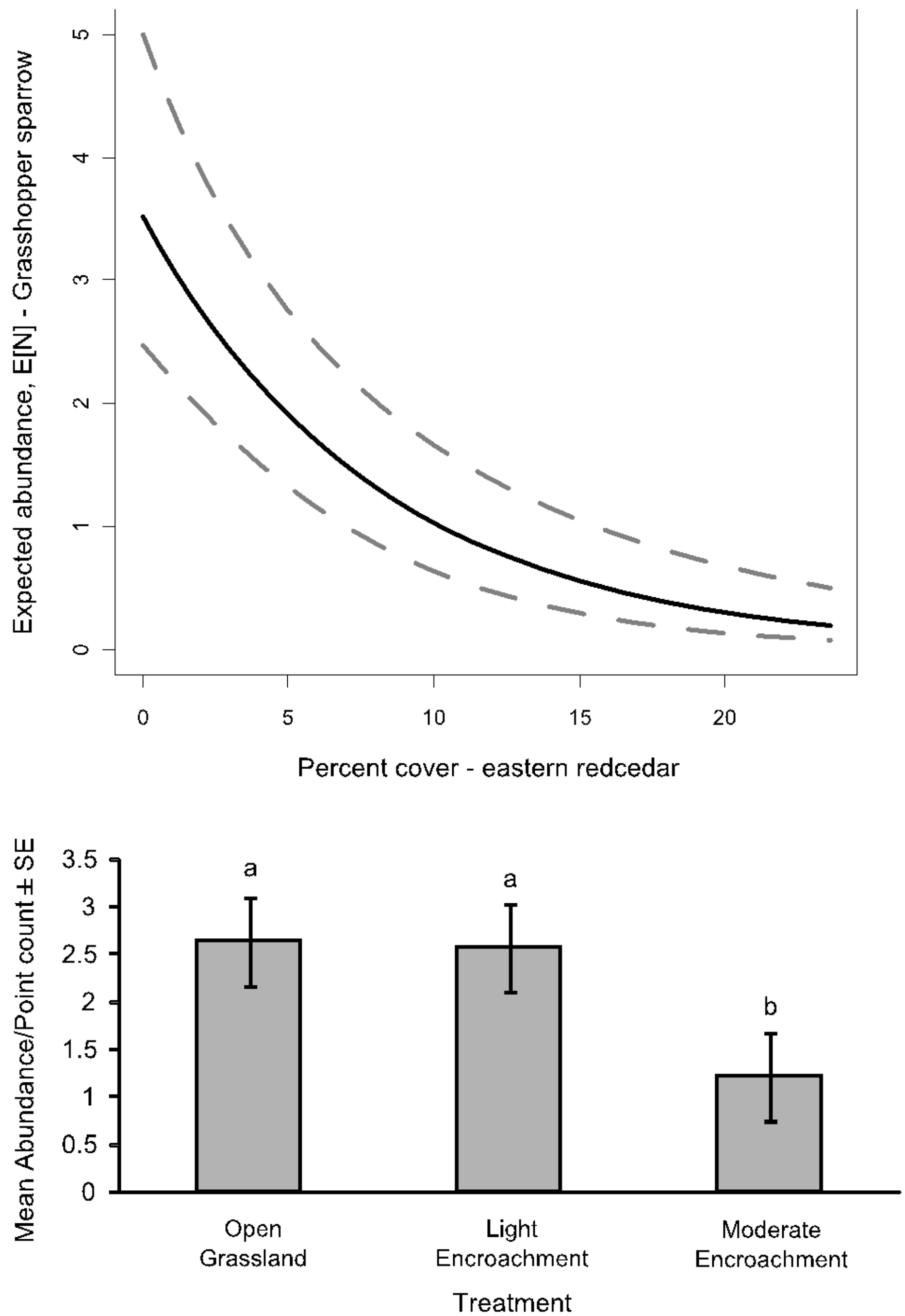
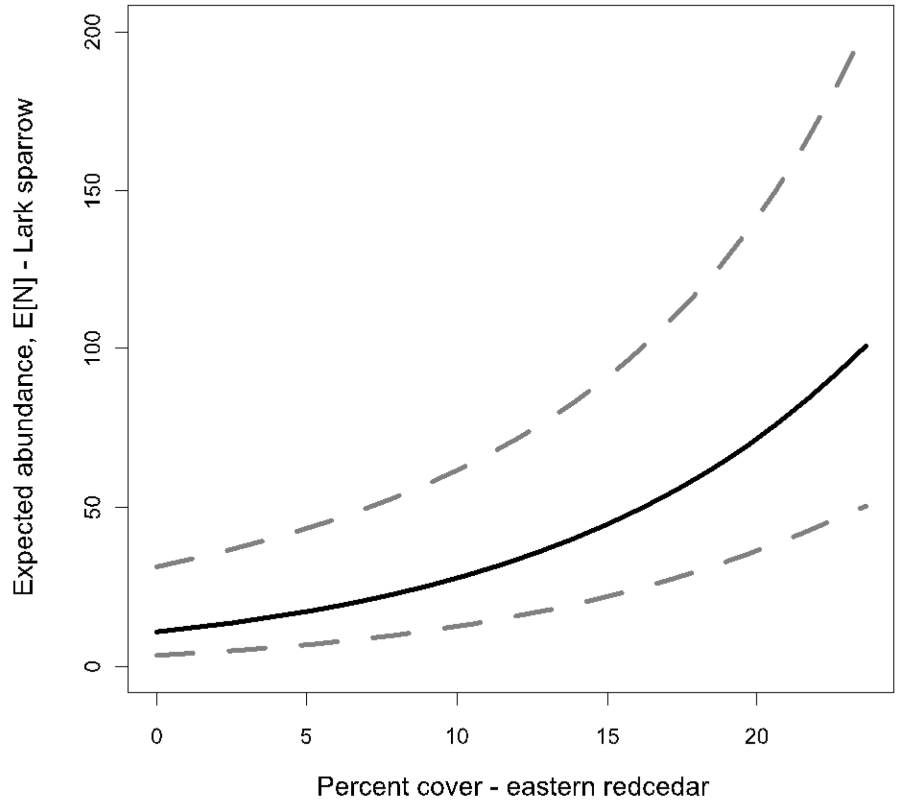


Figure 15. (A) Expected abundance of lark sparrow as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.



B.

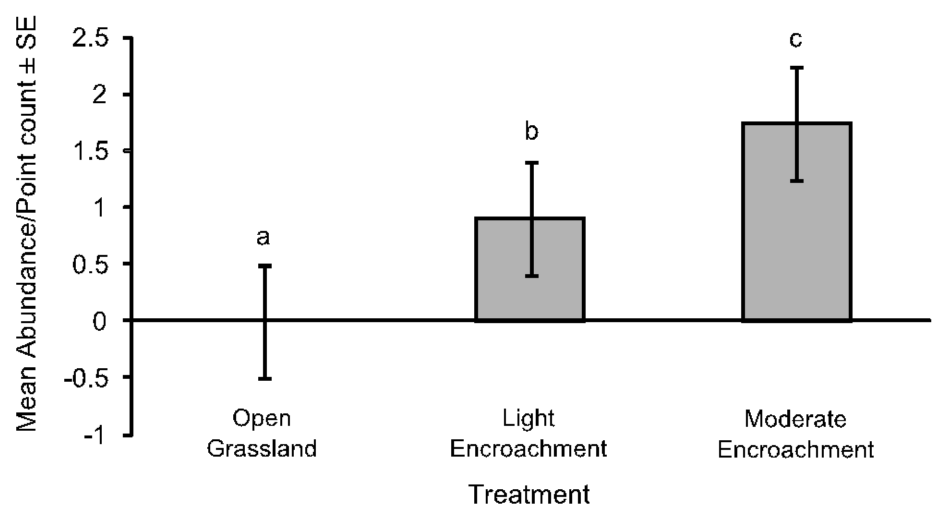
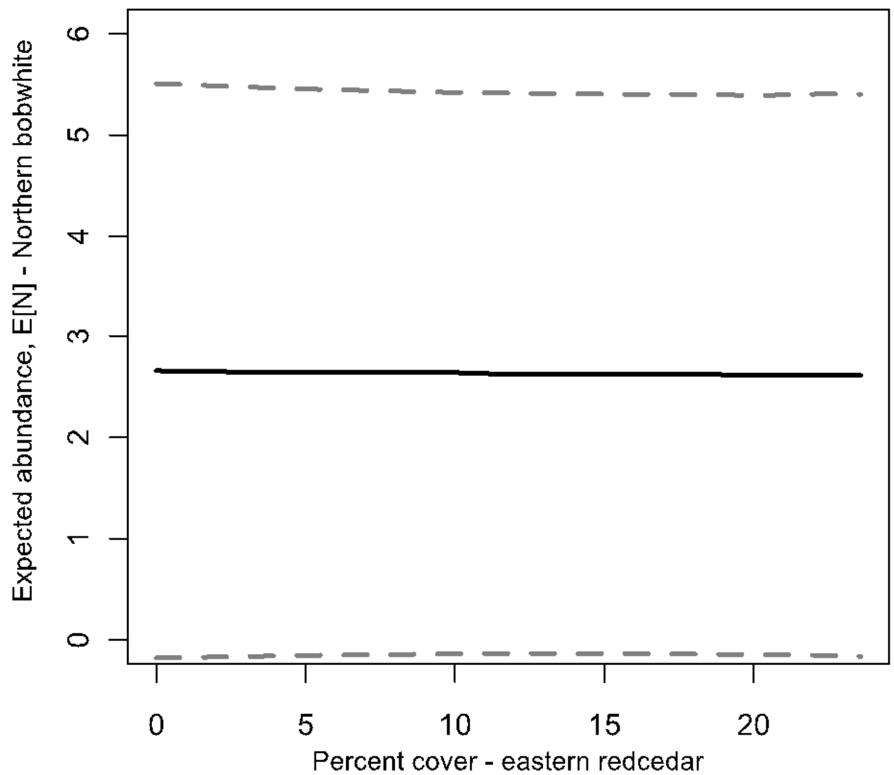


Figure 16. (A) Expected abundance of northern bobwhite as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.



B.

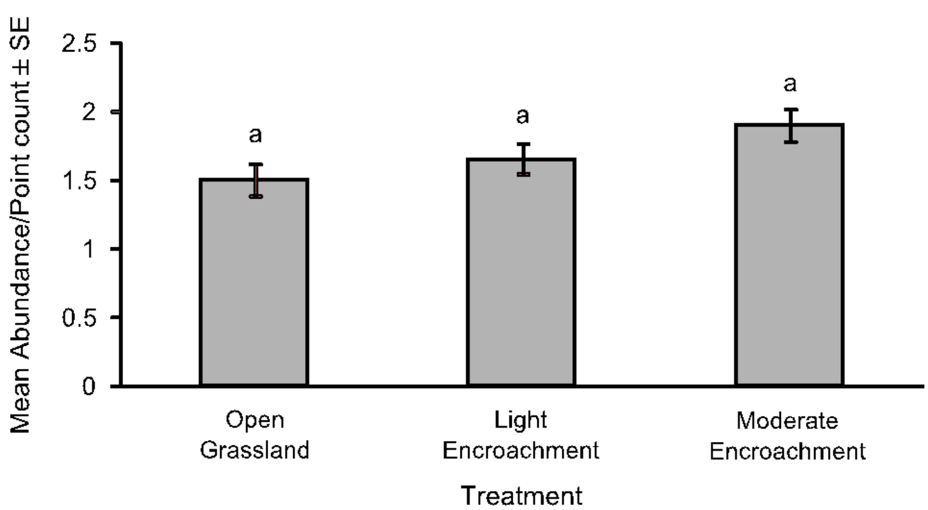
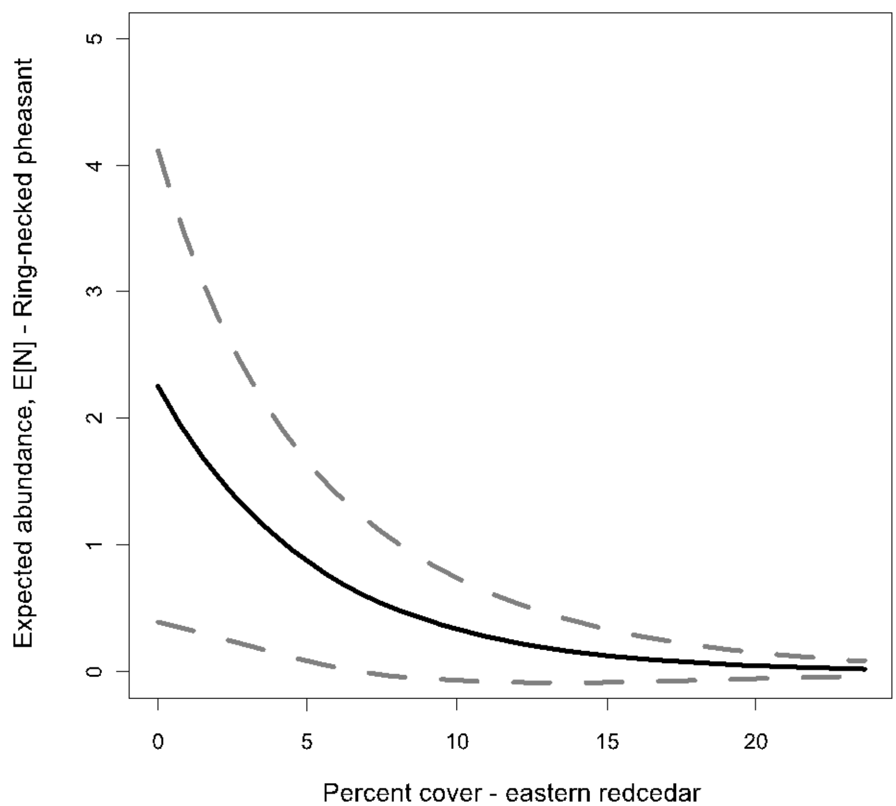


Figure 17. (A) Expected abundance of ring-necked pheasant as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.



B.

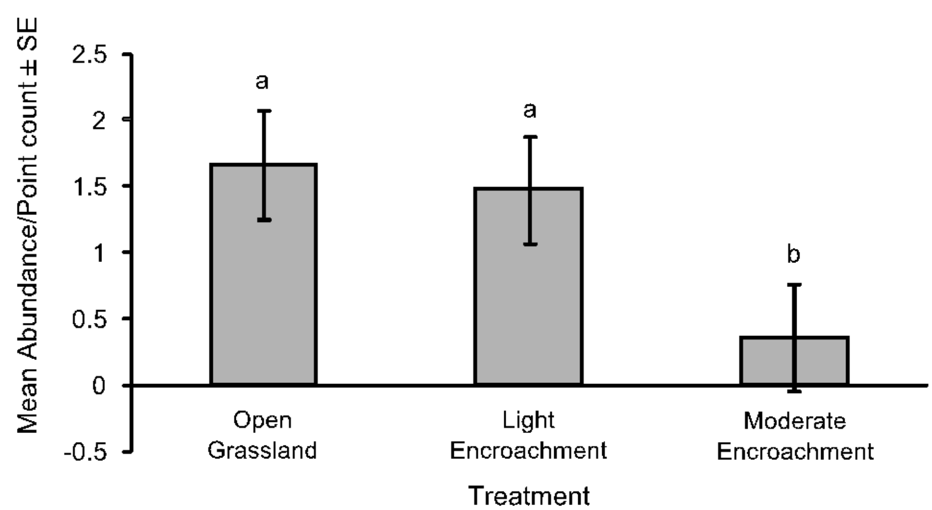


Figure 18. (A) Expected abundance of upland sandpiper as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

A.

B.

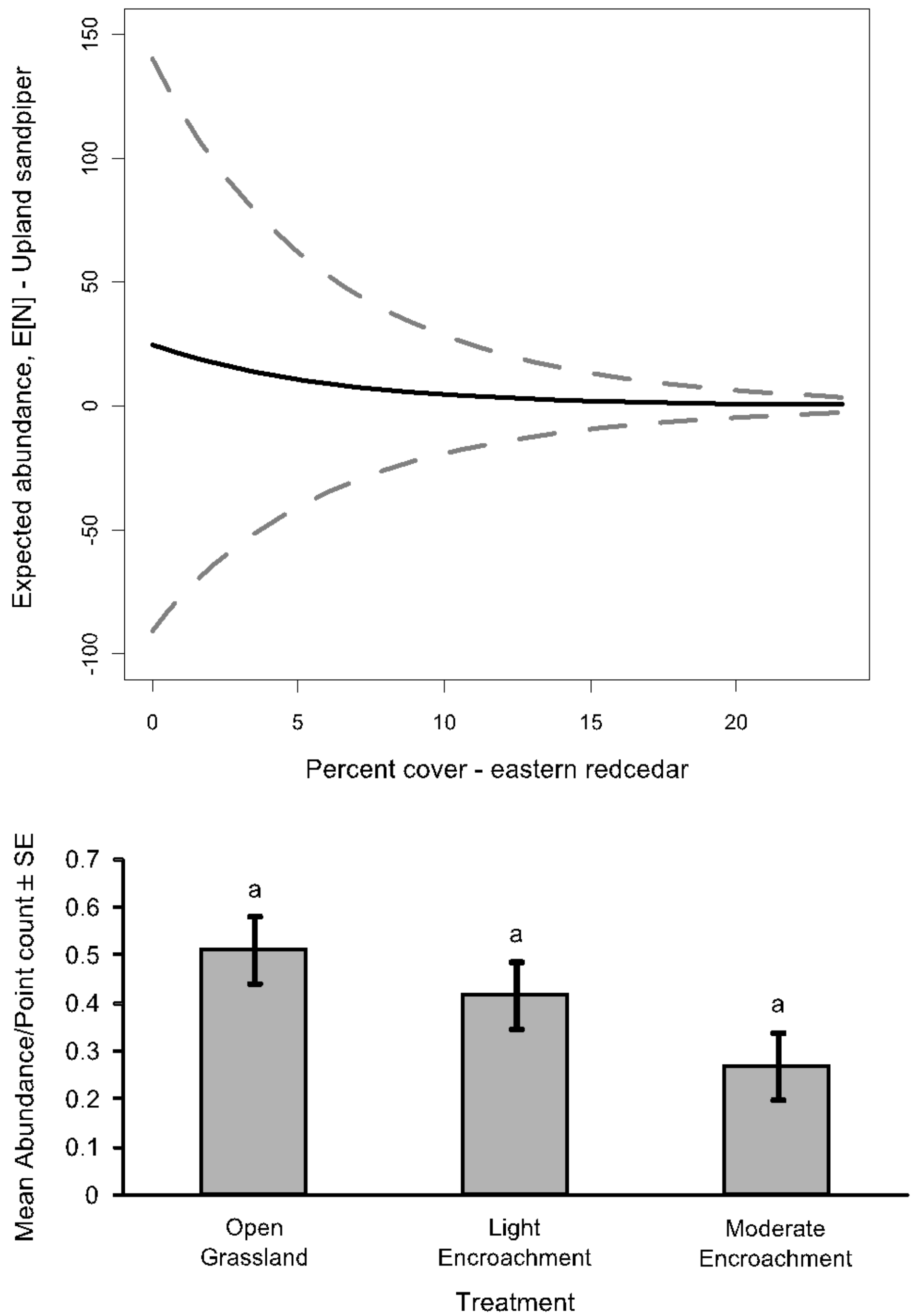
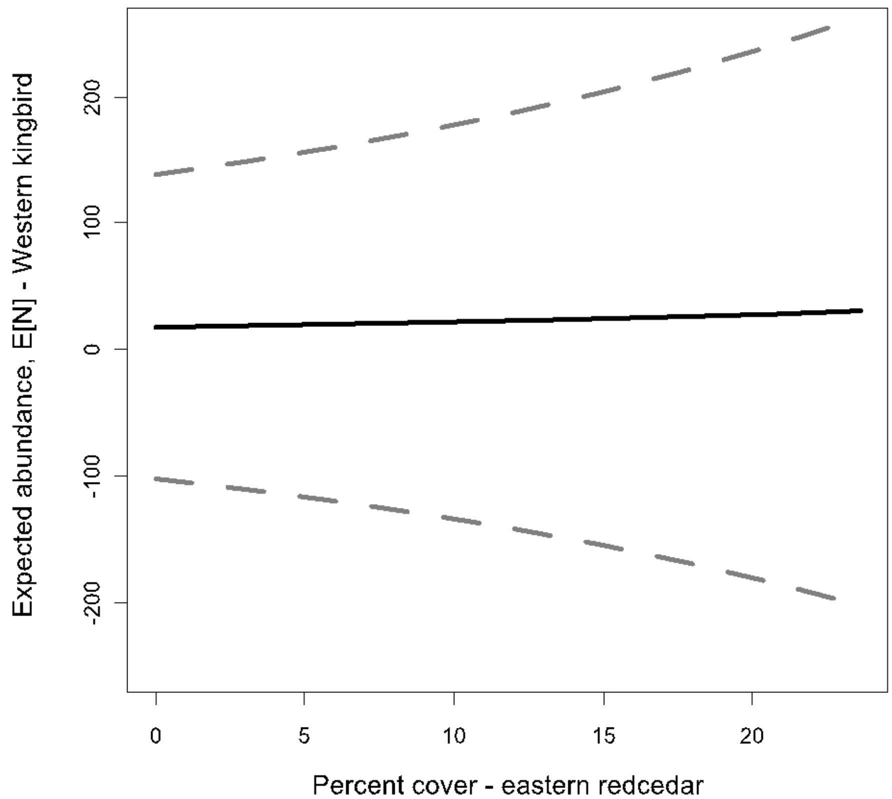
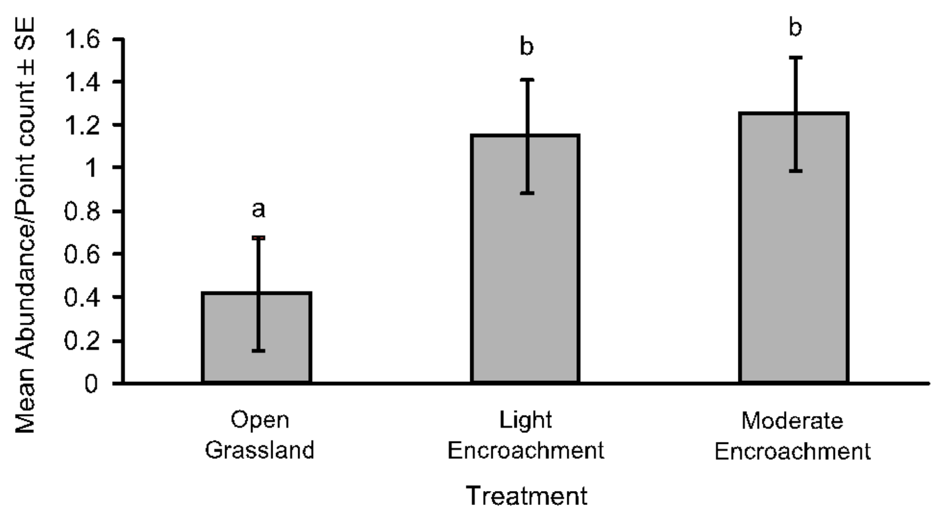


Figure 19. (A) Expected abundance of western kingbird as a function of eastern redcedar canopy cover and (B) pairwise comparisons of mean abundance among treatment levels.

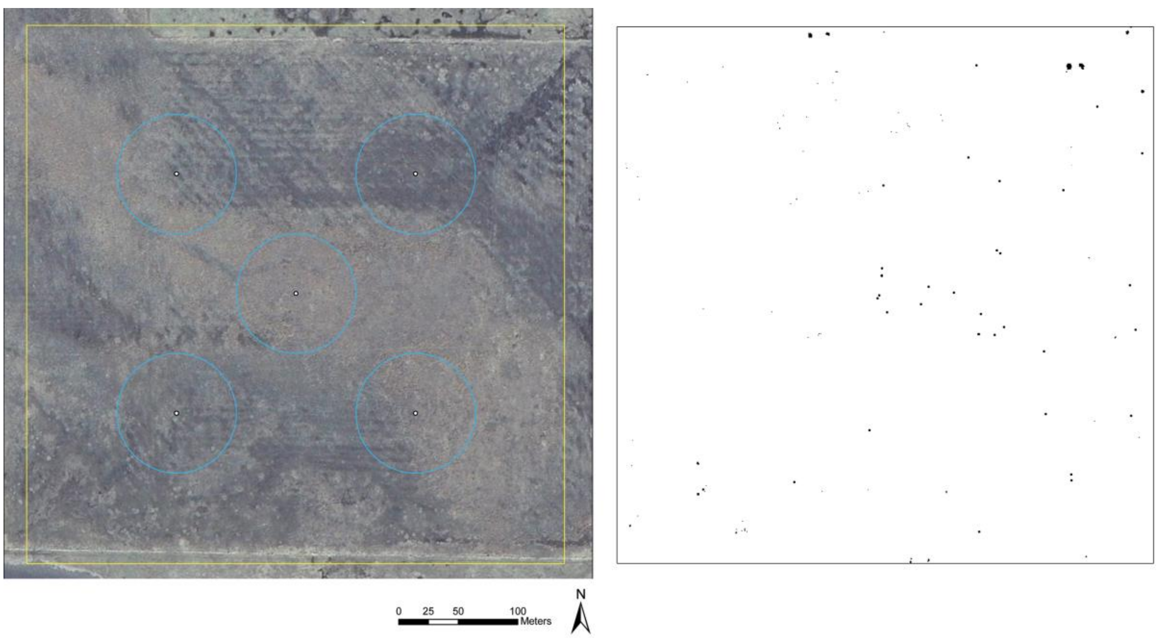
A.



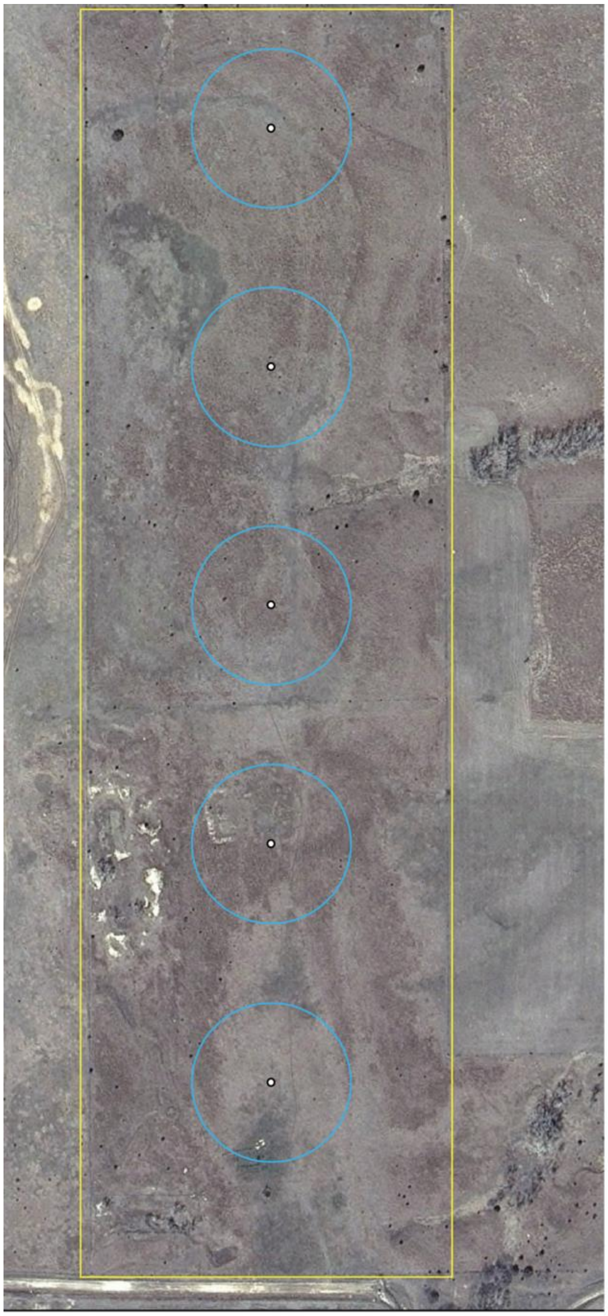
B.

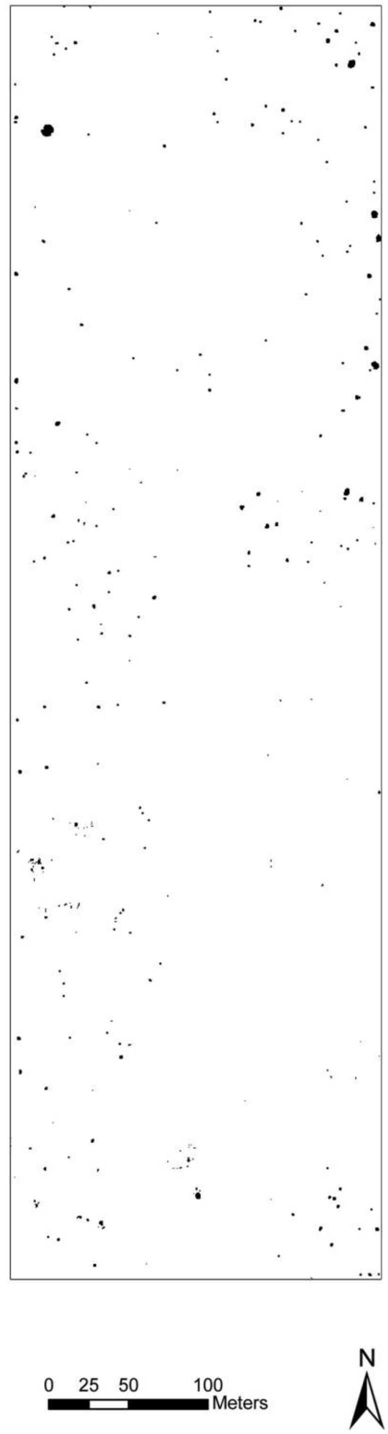


Appendix 1. Side-by-side comparisons of aerial and black and white binary images from light and moderate encroachment study sites in Barton County, Kansas USA. Aerial imagery of study sites has point-count stations and site boundaries overlaid. Black pixels in the binary images represent eastern redcedar canopy cover.

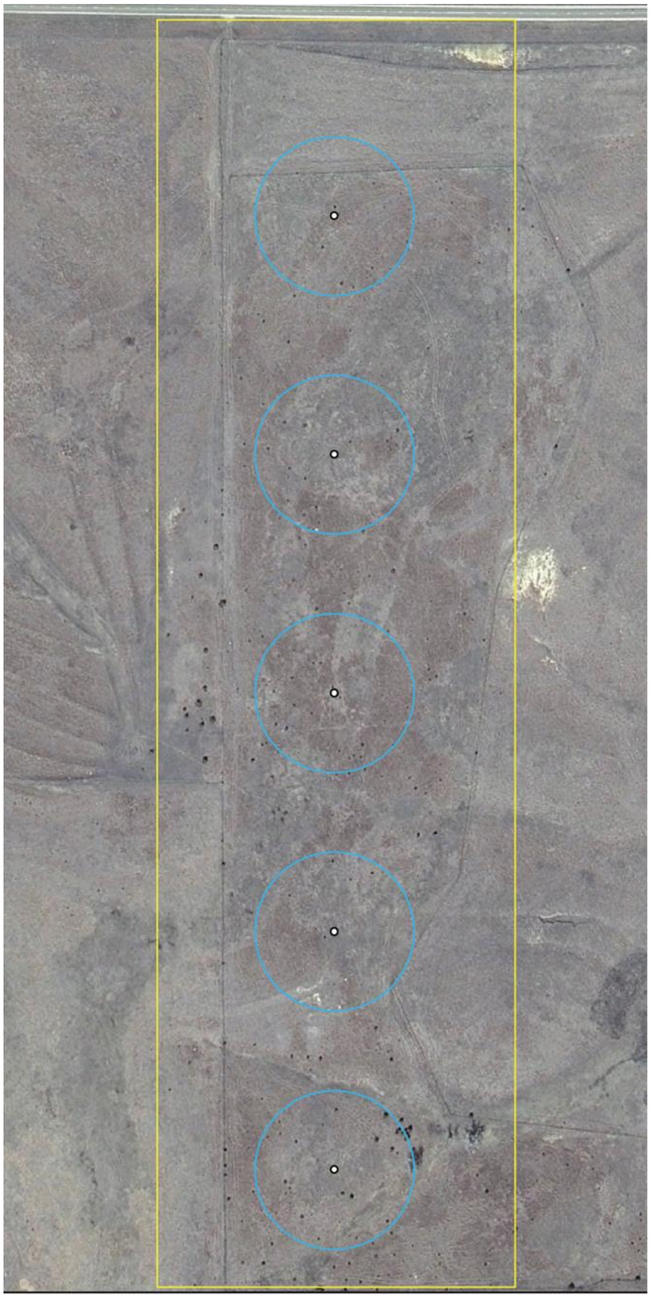


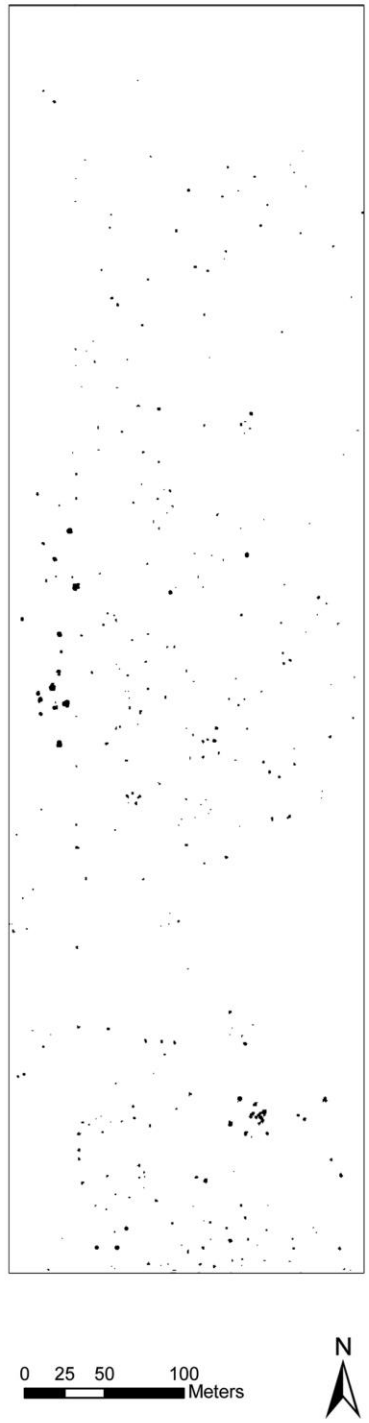
Site: WIHA Treatment level: Light encroachment Canopy cover: 0.10%



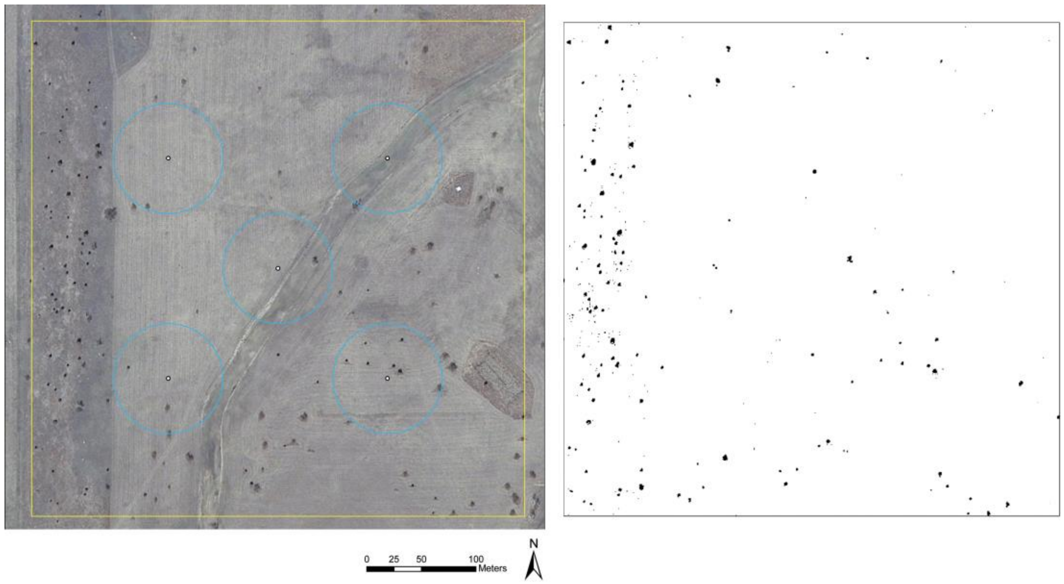


Site: MANE Treatment level: Light encroachment Canopy cover: 0.38%

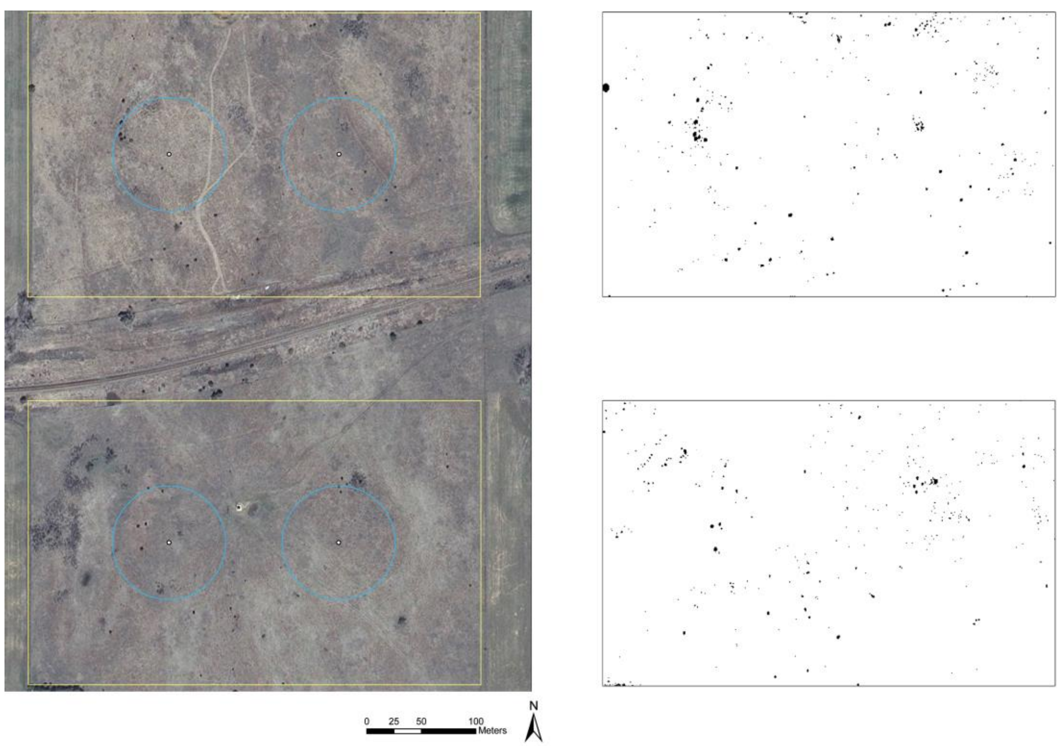




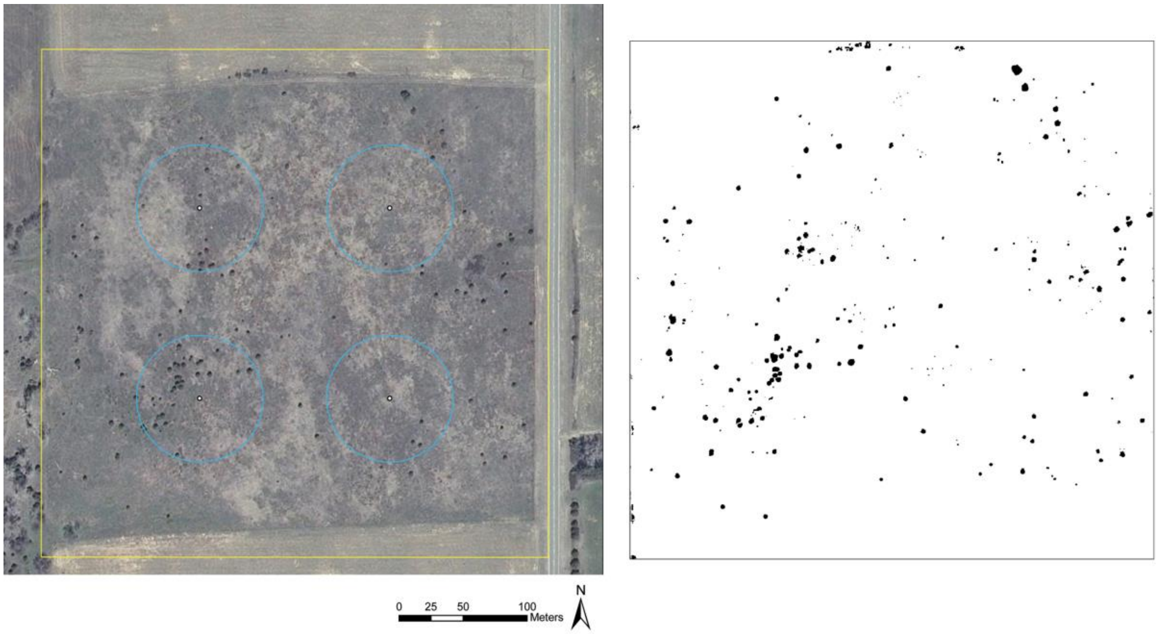
Site: TAYL Treatment level: Light encroachment Canopy cover: 0.40%



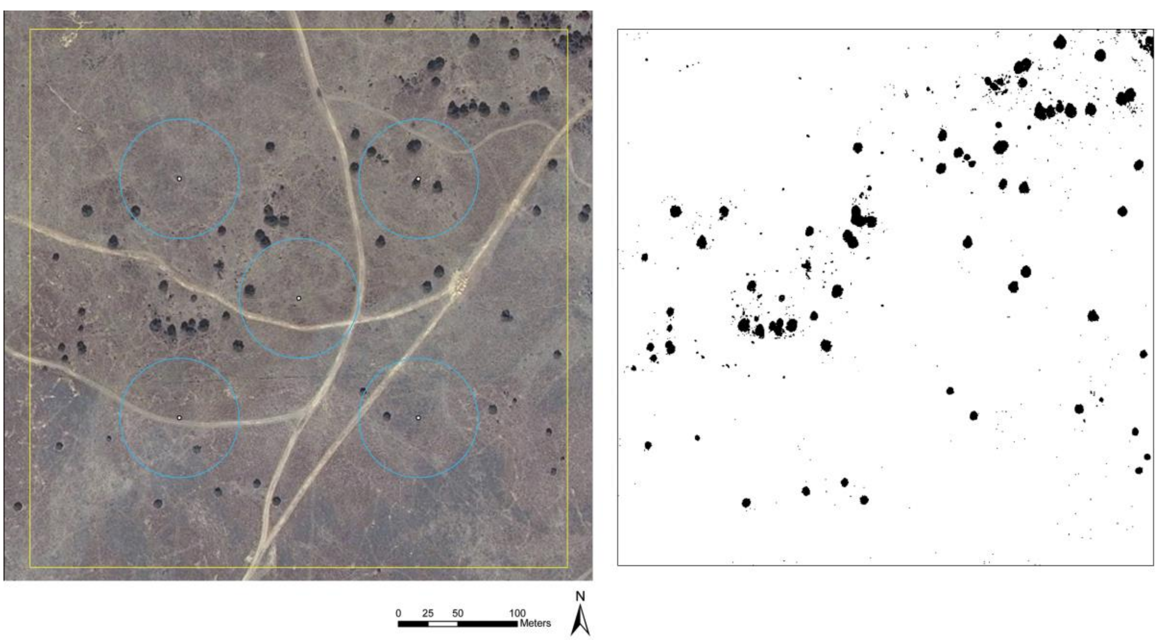
Site: RUPP Treatment level: Light encroachment Canopy cover: 0.40%



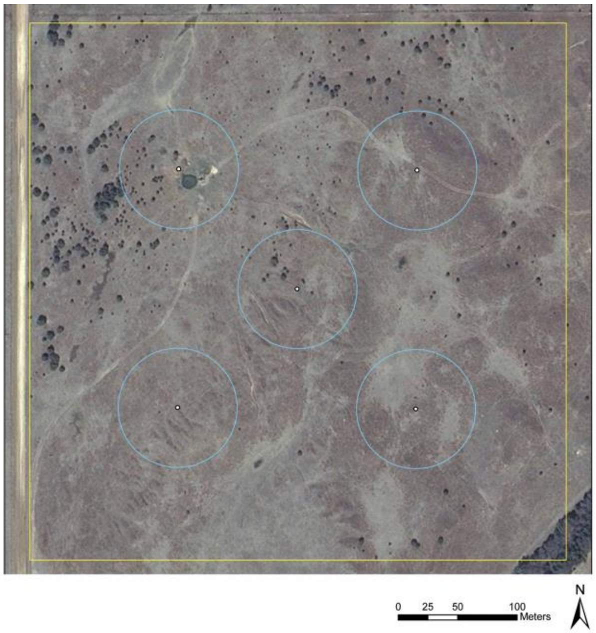
Site: OBRU Treatment level: Light encroachment Canopy cover: 0.41%

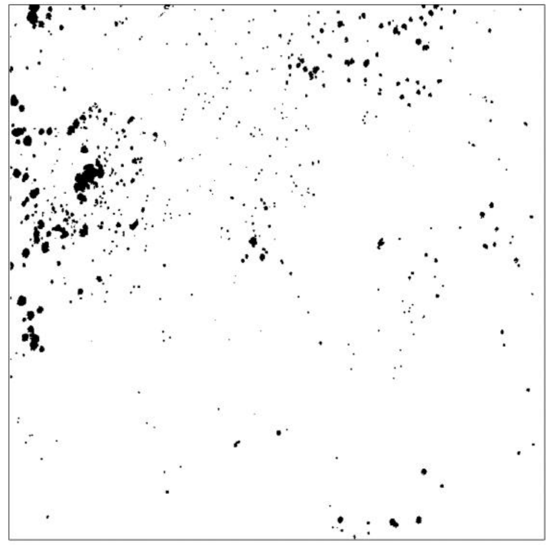


Site: PETE Treatment level: Light encroachment Canopy cover: 0.95%

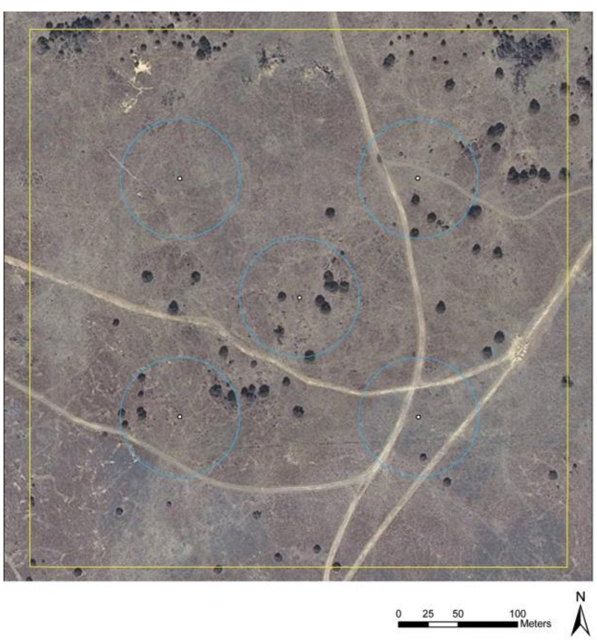


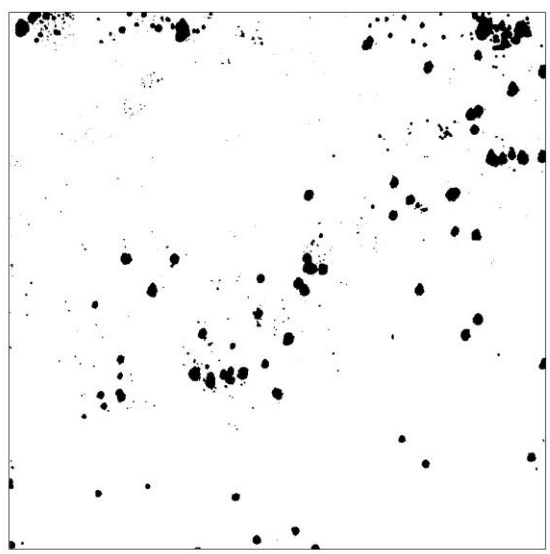
Site: WMIL-11 Treatment level: Light encroachment Canopy cover: 2.30%



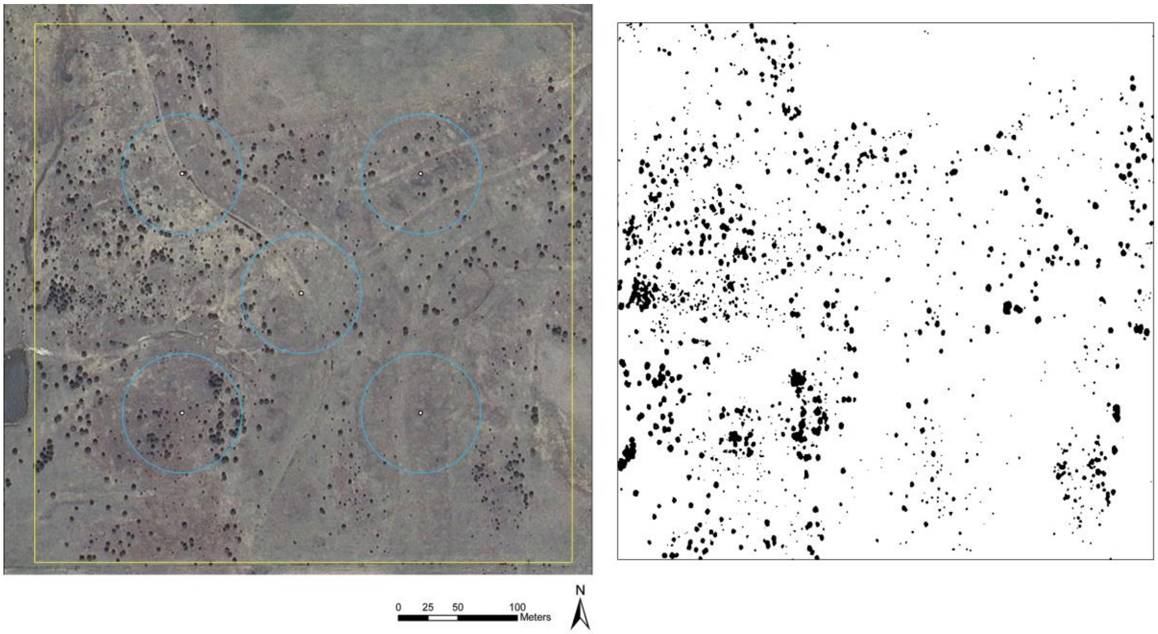


Site: CHUR Treatment level: Light encroachment Canopy cover: 0.2.39%



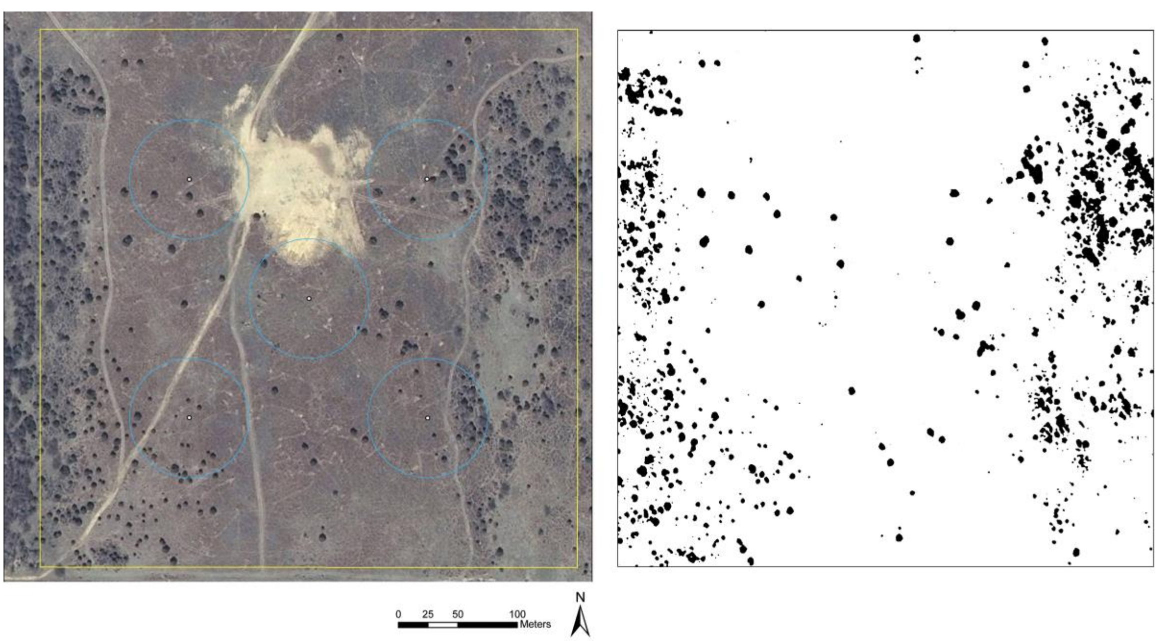


Site: WMIL-12 Treatment level: Light encroachment Canopy cover: 2.92%



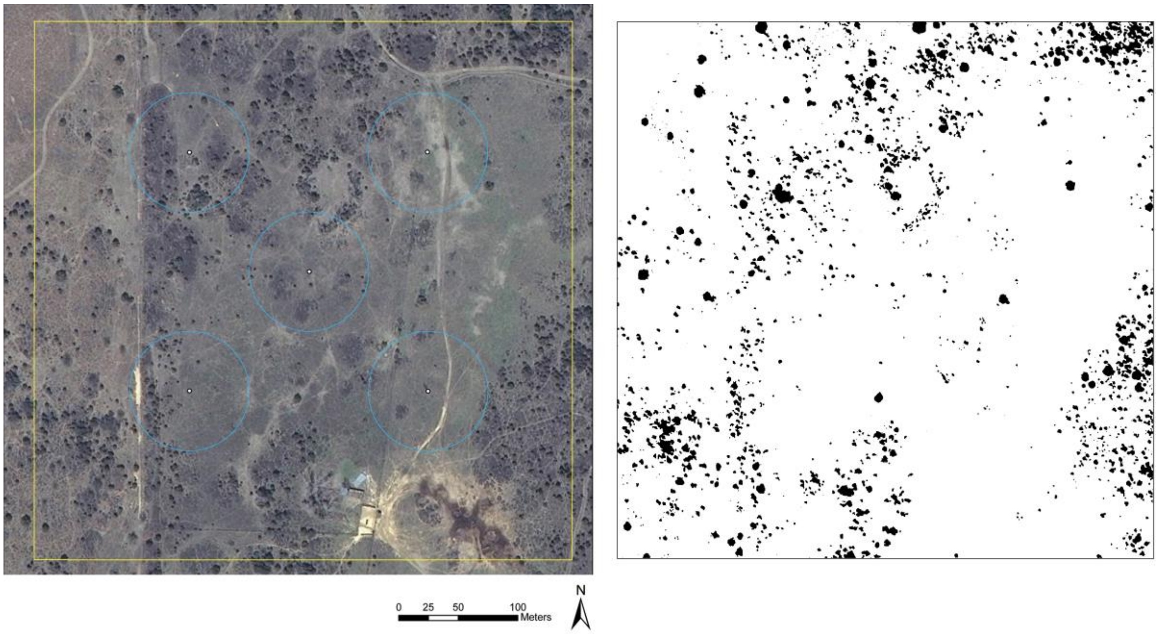
Site: INDE

Treatment level: Moderate encroachment
Canopy cover: 5.16%



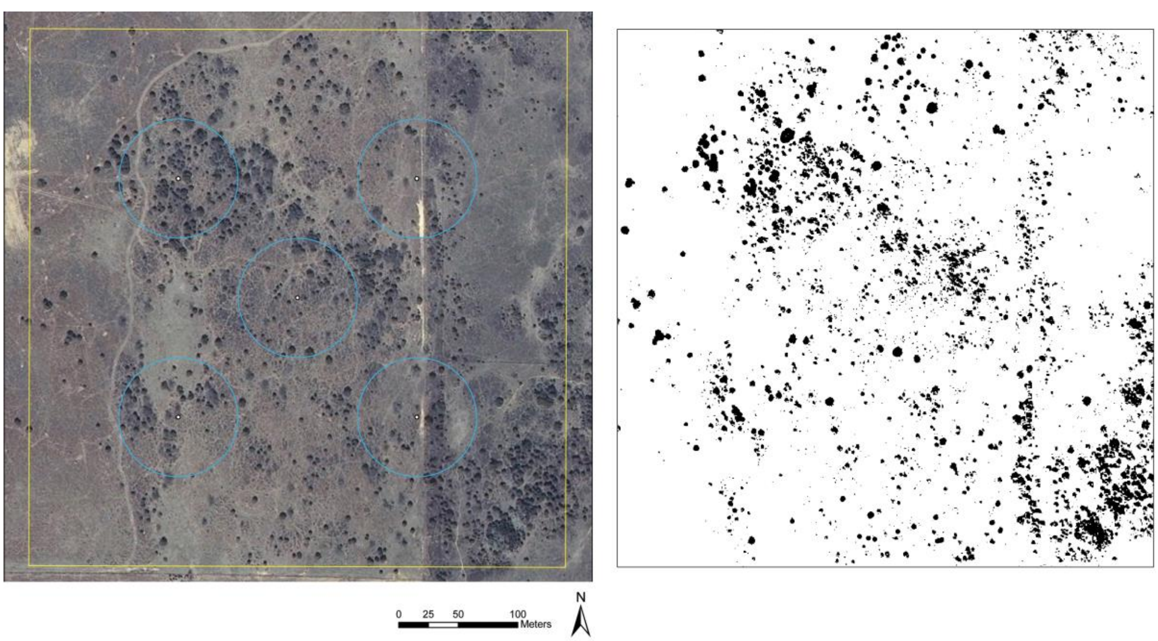
Site: ELLI

Treatment level: Moderate encroachment
Canopy cover: 5.97%



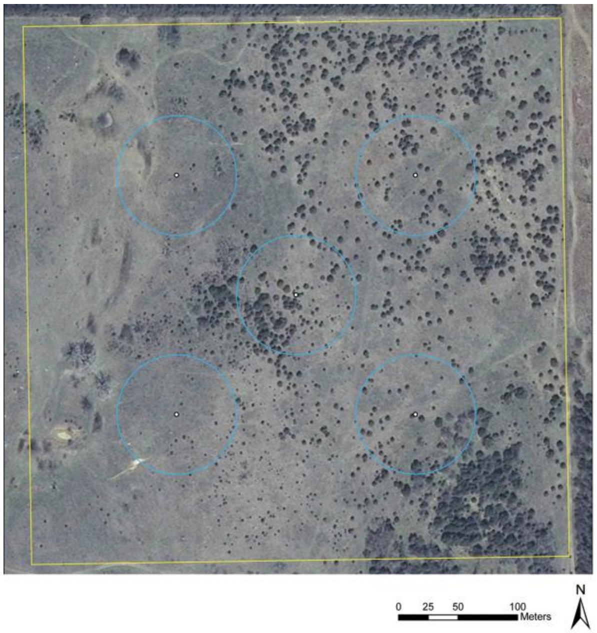
Site: BART

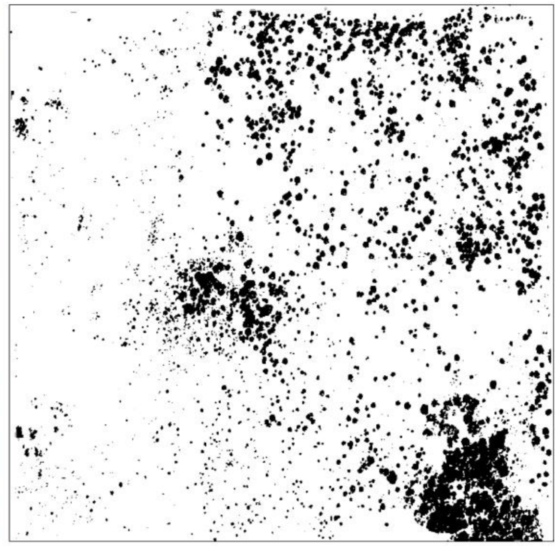
Treatment level: Moderate encroachment
Canopy cover: 6.10%



Site: ALDR

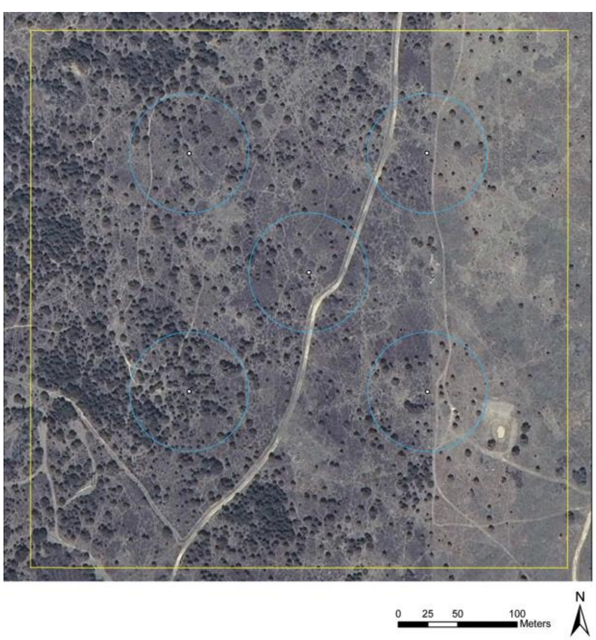
Treatment level: Moderate encroachment
Canopy cover: 7.57%

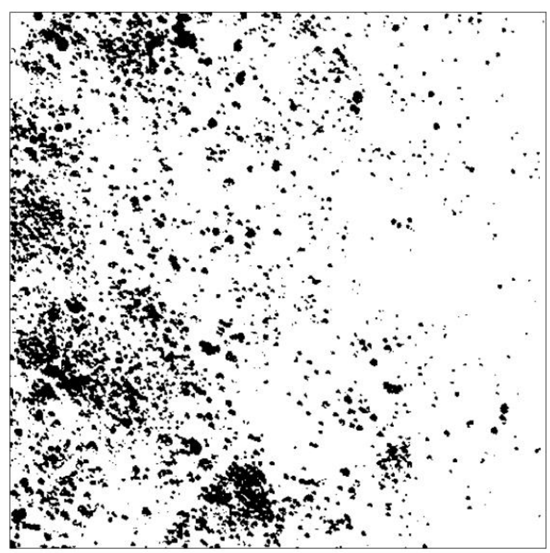




Site: HAMM

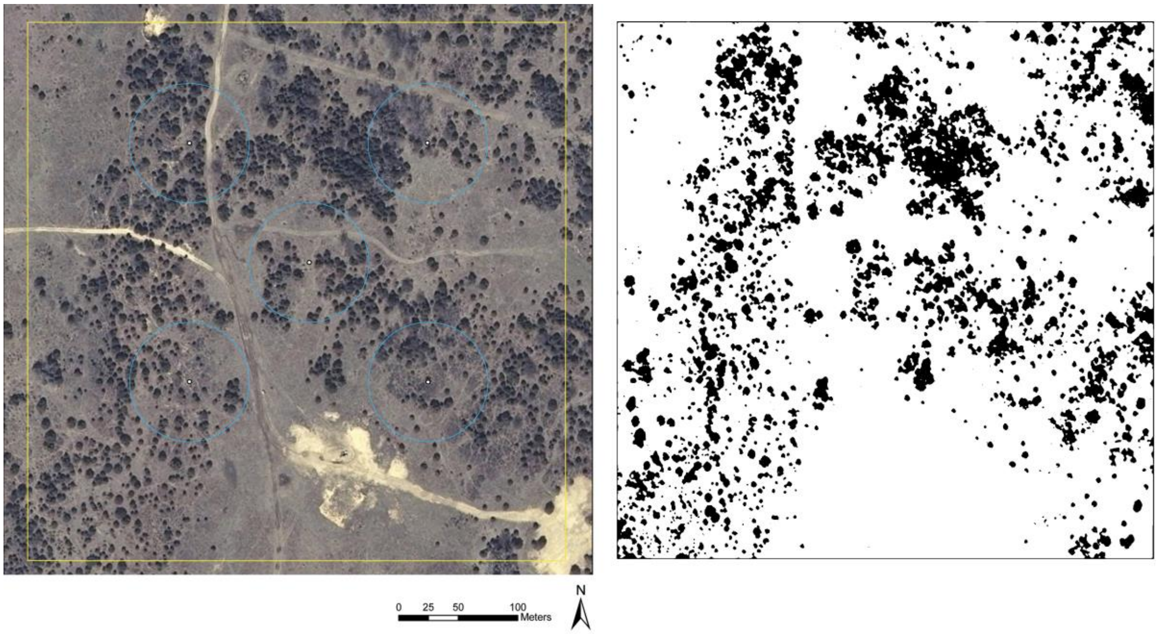
Treatment level: Moderate encroachment
Canopy cover: 12.28%





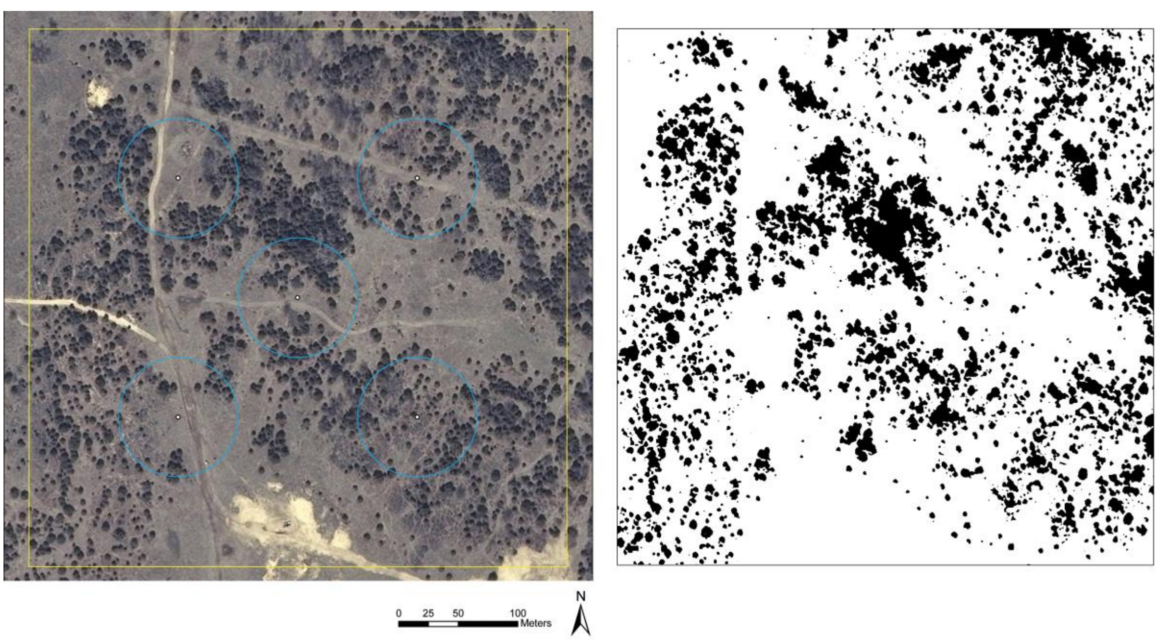
Site: BCCO-12

Treatment level: Moderate encroachment
Canopy cover: 13.62%



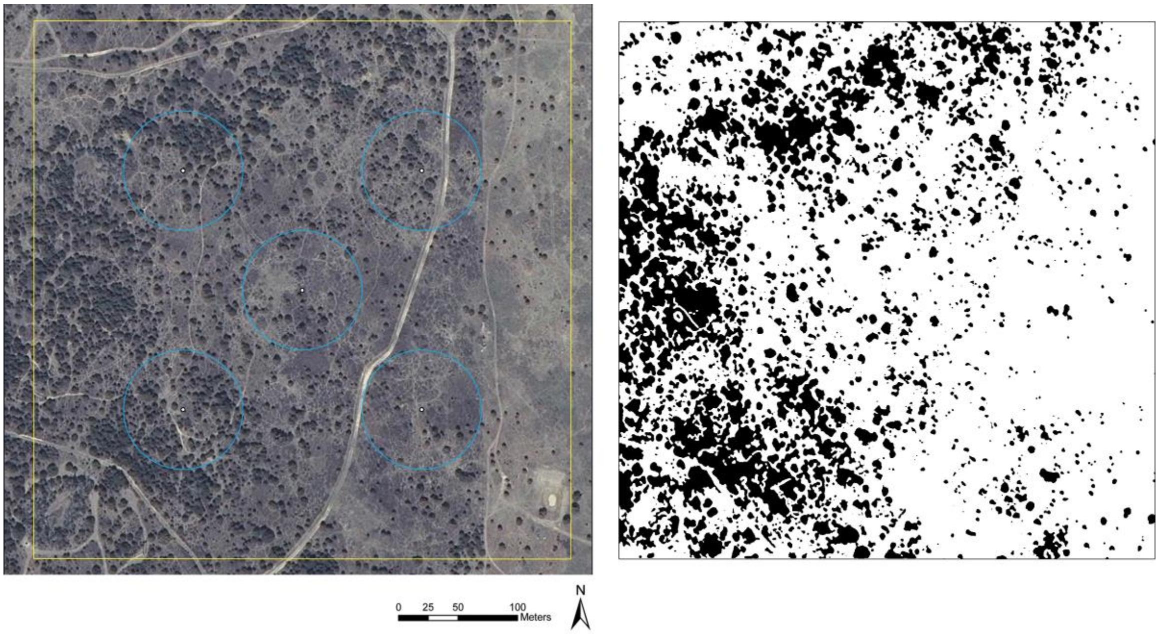
Site: LOGN-12

Treatment level: Moderate encroachment
Canopy cover: 17.26%



Site: LOGN-11

Treatment level: Moderate encroachment
Canopy cover: 21.41%



Site: BCCO-11 Treatment level: Moderate encroachment
Canopy cover: 23.64%

Appendix 2. Method for processing aerial imagery with eastern redcedar canopy cover into a black and white binary image.

Step 1. Boundaries for each study area were generated with the Hawth's Tools extension in ArcMap and exported as TIFF files.

Step 2. The TIFF image was opened in PhotoShop and duplicated 4 times. Blends modes were applied to enhance pixel lightening, darkening, and contrast.

Layer 1 – Overlay mode blending layer

Layer 2 – Color Dodge mode blending layer

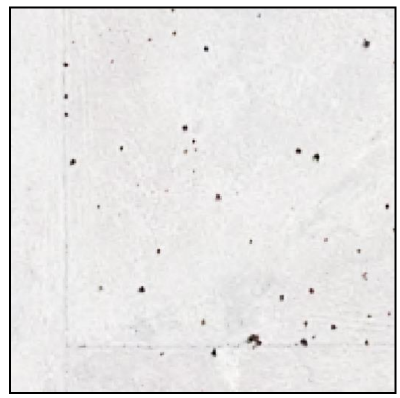
Layer 3 – Screen mode blending layer

Layer 4 – Base layer in Normal mode

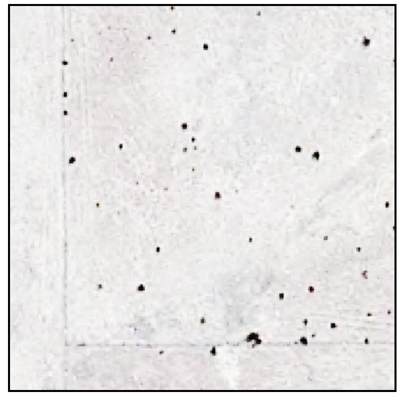
Layer 5 – Original image, used for visual comparison with the processed image.

Step 3. Screen blending was applied to layer 3, which lightened the image.

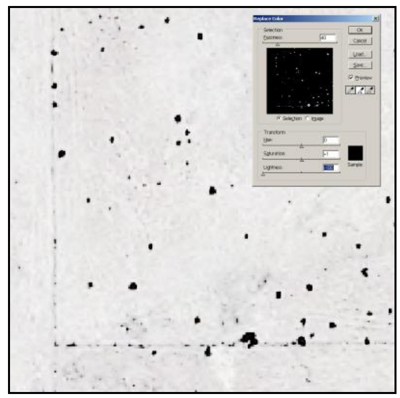
Step 4. Color Dodge was applied to layer 2, which selectively brightened highlights and midtones. Combined with Screen mode, the contrast between the dark and light pixels increased. The opacity of the Color Dodge blending layer was adjusted so that the lightest tones of tree canopies were still visible. The opacity of the sample image was set to 60%.



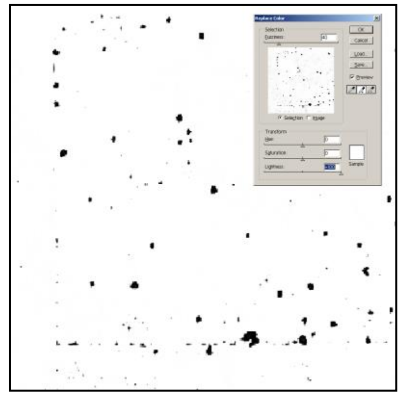
Step 5. Overlay mode multiplied the pixel values of the underlying layers, resulting in a higher contrast image. With the original image hidden, the base layer and blending layers were merged into a single layer.



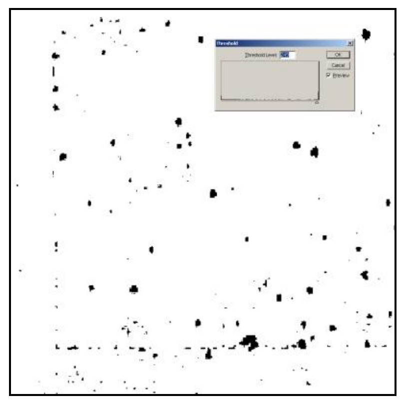
Step 6. The Replace Color tool was used to enhance dark tones in the newly created composite layer. The Add to Sample eyedropper tool was used to select pure pixel samples representing eastern redcedar canopy cover. After selecting a range of values, the Lightness of the selected pixels was adjusted to -100, which tinted eastern redcedar features to black.



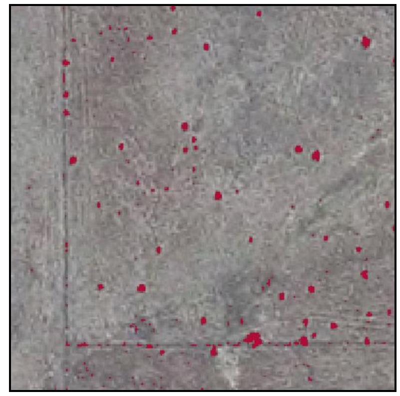
Step 7. Similarly, the Replace Color tool was used subtract light-toned background noise in the image. The Subtract from Image eyedropper tool was used to sample pixel values of features that did not represent eastern redcedars. The Lightness value of selection was increased to 100. The result was a grayscale image which closely resembles a black and white image.



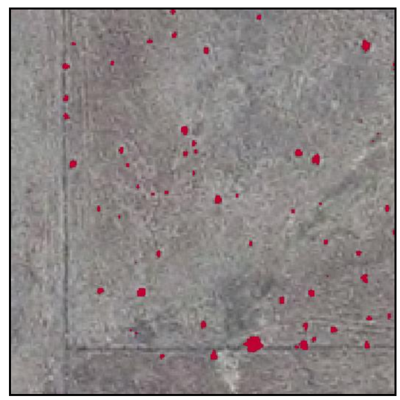
Step 8. A Threshold adjustment was applied to the image, which resulted in a true black and white image. The adjustment level should include an appropriate amount of features of interest, without introducing too much noise into the scene. Here, the Threshold Level was set to 249.



Step 9. By using the Magic Eraser tool, white pixels were removed from the image. With Layer 1 reactivated, the canopy cover layer was superimposed over the original image. Adjusting the Hue/Saturation of the canopy layer to red made it easier to identify features.



Step 10. The canopy cover layer was toggled on and off to show the underlying image so the interpreter could assess whether the pixels represented eastern redcedar cover. The eraser tool was used to remove pixels that did not represent canopy cover. The pencil tool was used to add pixels to features that were interpreted to be eastern redcedar canopy cover.



Step 11. The canopy cover layer was converted to black by adjusting the Hue and Saturation levels, and merged with a white sub-layer. Finally, the black and white composite was saved in Bitmap form. The resulting binary image was imported into ImageTool to quantify canopy cover.