DISSERTATION

MULTI-SCALE RESPONSE OF AVIAN COMMUNITIES TO PRESCRIBED FIRE: IMPLICATIONS FOR FUELS MANAGEMENT AND RESTORATION TREATMENTS IN SOUTHWESTERN PONDEROSA PINE FORESTS

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WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY BRETT G. DICKSON ENTITLED MULTI-SCALE RESPONSE OF AVIAN COMMUNITIES TO PRESCRIBED FIRE: IMPLICATIONS FOR FUELS MANAGEMENT AND RESTORATION TREATMENTS IN SOUTHWESTERN PONDEROSA PINE FORESTS BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

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ABSTRACT OF DISSERTATION

MULTI-SCALE RESPONSE OF AVIAN COMMUNITIES TO PRESCRIBED FIRE: IMPLICATIONS FOR FUELS MANAGEMENT AND RESTORATION TREATMENTS IN SOUTHWESTERN PONDEROSA PINE FORESTS.

Ponderosa pine (*Pinus ponderosa*) forests of the American Southwest were historically characterized by frequent, low severity surface fires that mediated the occurrence of extreme wildland fire events. Over the last century, increases in fuels have transformed these forests in ways that help facilitate stand-replacing crown fires. To reduce the risk of such fires, land managers are implementing fuels treatments in progressively larger stands of ponderosa pine.

For the period 1 April to 30 September, 1986-2000, I examined contributing landscape factors related to the occurrence of large ($\geq$ 20 ha) fires in the forested region of northern Arizona. I used a Bayesian weights-of-evidence approach to model and map the probability of occurrence based on all fire types and lightning-caused fires alone. In total, large fires burned 101,571 ha on my study area and fires due to lightning were more frequent and extensive than human-caused fires. For all fires, probability of occurrence was greatest in areas of high topographic roughness and lower road density. My results indicated that seasonal large fire events were a consequence of non-random patterns of occurrence that can have extensive influence on the regional fire regime.

I used a before-after/control-impact experimental design to examine the multi-scale response of avian communities to large (~ 260-400 ha) prescribed fire treatments on four ponderosa pine forests in Arizona and New Mexico. I used distance sampling and an information-theoretic approach to quantify changes in density for 14 species and three trait-based clusters of species among 82 total species detected on four sites before (May-June, 2002-2003) and after (May-June, 2004-2005) prescribed fire treatments. I used spatial models to evaluate the post-fire response of species and clusters using initial forest conditions and levels of fire severity. At the point level, five species exhibited a treatment response and two of these species were positively associated with increasing levels of fire severity. Among the four study sites, I identified few non-random patterns of cluster- and community-level response to treatment. In the short term, breeding bird communities in southwestern ponderosa pine forests appear tolerant of low-to-moderate intensity prescribed fire treatments at multiple spatial scales and across multiple geographic locations.

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INTRODUCTION

Ponderosa pine (*Pinus ponderosa*) forests of the American Southwest were historically characterized by frequent, low severity surface fires that mediated the occurrence and return interval of larger, stand-replacing fire events (Cooper 1960, Moir et al. 1997). However, human-induced manipulation of southwestern fire regimes has resulted in ponderosa pine-dominated ecosystems that are vulnerable to wildland fires of unprecedented size and severity (Cooper 1960, Swetnam et al. 1999). Indeed, contemporary increases in surface and canopy fuels, due to modern fire control efforts, large-scale timber harvesting, and domestic cattle grazing, for example, have transformed the structural characteristics of ponderosa pine forests in ways that facilitate stand-replacing crown fires (Covington and Moore 1994). The exceptionally large fires that occurred in the Southwest between 2000 and 2002 attest to this situation and many forest ecologists and fire officials agree that steps must be taken to reduce fuel levels and abate risk of wildland fire (Allen et al. 2002, Brown et al. 2004).

In the Southwest, changes in forest structure and fire regime contribute only partially to the increased likelihood of large fire events. In Chapter 1, I use a Bayesian framework and weights of evidence to evaluate landscape-scale factors and patterns that contribute to the occurrence of larger (≥ 20 ha in extent) natural- and human-caused fires. For the period 1 April to 30 September, 1986-2000, I compiled a database of fire occurrence records on a 27,065-km^2 study area that included most of the ponderosa pine-dominated forest regions of northern Arizona. Using this spatially-explicit information, I modeled and mapped the probability of occurrence of large fire type as a function of dominant forest vegetation, topographic features (e.g., aspect and roughness), road density, precipitation, and spatial location. Although fuel reduction efforts can be important in managing fire risk, treatments designed to reduce fuels may do little to reduce fire threat if they are not strategically placed in or around areas where large fire events are most likely to occur. Chapter 1 permits broader insights into the patterns of fire risk that will increase our ability to manage fire threat and accelerate fuels and restoration treatment efforts when fire is a component of the prescription.

To reduce the risk of extreme fire events described in Chapter 1, land managers are implementing fuels treatments in progressively larger stands of dry, low elevation ponderosa pine forest. Additionally, restoration projects in the Southwest are removing large amounts of forest biomass using mechanical thinning and fire treatments in an attempt to re-create forest structure conditions reminiscent of those that existed prior to Euro-American settlement (Covington et al. 1997, Friederici 2003). Given the broad-scale implementation of such treatments, there is unfortunately limited knowledge of how to maintain biodiversity across large areas where fire behavior, fuel quantities, and forest structure have been changed by humans (Covington and Moore 1994, Lyon et al. 2000, Noss et al. 2006). Moreover, there remain significant uncertainties about the consequences of reintroducing fire on populations of wildlife and their habitats (Tiedemann et al. 2000). However, most research on the effects of fire on wildlife has focused on observations of avian populations after wildland fire events, not after purposeful fuels treatment. In Chapter 2, I use a designed experiment to quantify the multi-scale response of avian communities to large-scale prescribed fire treatments at four ponderosa pine forests in northern Arizona and west-central New Mexico. Using contemporary statistical methods and estimates of density, I attempt to relate prescribed fire-induced change in these communities to initial forest conditions, models of patch-level forest structure, and levels of fire severity. I examine avian response to prescribed fire using individual species, clusters of species, and all species of breeding birds detected during pre- (May-June, 2002-2003) and post-treatment (May-June, 2004-2005) sampling.

LITERATURE CITED

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12:1418-1433.

Brown, R. T., J. K. Agee, and J. F. Franklin. 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18:903-912.

Cooper, C. F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs 30:129-164.

Covington, W. W. and M. M. Moore. 1994. Southwestern ponderosa forest structure and resource conditions: changes since Euro-American settlement. Journal of Forestry 92:39-47.

Covington, W. W., P. Z. Fulé, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. Journal of Forestry 95:23-29.

Friederici, P. 2003. The Flagstaff model. Pages 7-25 *in* P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Lyon, L. J., M. H. Huff, and J. K. Smith. 2000. Fire effects on fauna at landscape scales. Pages 43-49 *in* J. K. Smith (ed.). Wildland fire in ecosystems: effects of fire on fauna. USDA Forest Service, Rocky Mountain Research Station, General Technical Report, RMRS-GTR-42-Volume 1, Ogden, UT, USA.

Moir, W. H., B. W. Geils, M. A. Benoit, and D. Scurlock. 1997. Ecology of southwestern ponderosa pine forests. Pages 3-27 *in* W. M. Block and D. M. Finch (eds.). Songbird ecology in southwestern ponderosa pine forests: a literature review. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-292, Fort Collins, CO, USA.

Noss, R. F., P. Beier, W. W. Covington, R. E. Grumbine, D. B. Lindenmayer, J. W. Prather, F. Schmiegelow, T. D. Sisk, and D. J. Vosick. 2006. Recommendations for integrating restoration ecology and conservation biology in ponderosa pine forests of the southwestern United States. Restoration Ecology 14:4-10.

Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9:1189-1206.

Tiedemann, A. R., J. O. Klemmedson, and E. L. Bull. 2000. Solution of forest health problems with prescribed fire: are forest productivity and wildlife at risk. Forest Ecology and Management 127:1-18.

CHAPTER ONE

MAPPING THE PROBABILITY OF LARGE FIRE OCCURRENCE IN NORTHERN ARIZONA

INTRODUCTION

Recently, the American Southwest has experienced wildland fires of relatively unprecedented size and severity (e.g., the 2000 Cerro Grande fire in New Mexico and the 2002 Rodeo-Chediski fire in Arizona). Modern fire control efforts have contributed to levels of wildland fire frequency and intensity greater than those encountered during the early part of the 20th century (Agee 1998) and atypical in the paleoecological record (Grissino-Mayer and Swetnam 2000). Beginning in the late 1800's, the landscape was dramatically altered by the introduction of domestic livestock, large-scale timber harvesting, and aggressive fire suppression activities. Today, forests dominated by ponderosa pine (*Pinus ponderosa*) are dense with many pole-size trees that help facilitate stand-replacing crown fires (Covington and Moore 1994). However, such extreme changes in forest structure and fire regime contribute only partially to the increased likelihood of large, natural- and human-caused fire events.

In the region that includes northern Arizona and western New Mexico, 60-70% of forest fires are ignited by lightning, compared to approximately 20% nationwide (Barrows 1978). Due to a high incidence of lightning strikes from dry thunderstorms during the summer monsoons, this area leads the nation in the average number of lightning-caused fires and average amount of National Forest area burned by these fires each year (Barrows 1978). Historically, most large fires in the Southwest were associated with broad-scale climate factors, or controls, such as El Niño-Southern Oscillation (ENSO) patterns and the persistence of drought conditions (Swetnam and Betancourt 1990; Allen 2002). However, retrospective studies of historical fire events suggest an increase in the number of large fires since the beginning of the 20th century (Swetnam 1990). For the period 1992-2003, the Southwest region (Arizona, New Mexico, and West Texas) experienced an annual average of 3,059 human-caused fires and 2,613 lightning-caused fires, burning an annual average of 91,906 ha and 84,900 ha, respectively (USDA Forest Service 2004). The size of more recent human-caused fire events may be due, in part, to an increase in the number of roads and improved access to remote forested locations resulting in more human-caused ignitions (Swetnam 1990; Cardille et al. 2001; see DellaSala and Frost 2001). In spite of the risks associated with current forest conditions, more people continue to settle where dense forests interface with urban areas (Davis 1990; USDA and USDI 2000; Dombeck et al. 2004). Consequently, the number of human-caused fires is expected to rise in these areas, increasing the likelihood of stand-replacing fire events.

The importance of nonrandom patterns in fire ignition and occurrence has been recognized by recent efforts to predict these patterns at larger spatial scales (e.g., Cardille et al. 2001, de Vasconcelos et al. 2001, Díaz-Avalos et al. 2001, and Preisler et al. 2004). At landscape scales (i.e., extents > 100,000 ha), the probability of a large fire is associated with multiple factors including: forest type, physiographic characteristics, climate, and human activities. However, insights into the interplay among these factors and how they facilitate subsequent large fire events are poorly explored. No quantitative analyses in the Southwest have examined the spatial patterns of occurrence that led to large fires and their relationship with various landscape features. Quantifying the probability of large fire occurrence is necessary to understand: 1) the scale and periodicity of natural fire regimes (Agee 1998; Fulé et al. 2003; Malamud et al. 2005); 2) the causes, patterns, and consequences of ecosystem-level disturbance and change (Attiwill 1994; Dale et al. 2001; McKenzie et al. 2004); 3) the socio-political implications of wildland fire and fire management (Cardille et al. 2001; Brunson and Shindler 2004; Dombeck et al. 2004); and 4) fire risk and fire threat to humans and their communities (Case et al. 2000; Keeley and Fotheringham 2001).

The objectives of this chapter were to: 1) assemble a geographic database of large fire events for the ponderosa pine-dominated regions of northern Arizona; 2) broadly characterize the important landscape features of these regions that may be associated with landscape-scale patterns of fire occurrence; 3) develop predictive maps of conditional probability of occurrence for large fires over a broad spatial and temporal scale using a new and rigorous approach; and 4) quantify the relationships between fire ignition source, landscape features, and patterns of occurrence.

METHODS

Study area

My 27,065-km^2 study area included the ponderosa pine-dominated forest regions of northern Arizona, USA (Figure 1.1A). Generally, these forests occurred in three distinct regions: a 3,390-km^2 region that included the Kaibab Plateau to the north of the Grand Canyon; the 1,418-km^2 area to the south of the Grand Canyon and northwest of Flagstaff, Arizona; and a 22,257-km^2 area that included the Mogollon Plateau, east to the New Mexico border. Common tree species on the study area also included Gambel oak (*Quercus gambelii*), quaking aspen (*Populus tremuloides*), and other high elevation mixed-conifer species. Elevations across the study area ranged from approximately 1,520 to a maximum of 3,840 m on Humphrey's Peak. Because the ponderosa pine-dominated vegetation zones typically occurred above 1,520 m, I constrained the borders of my study area using this minimum elevation threshold. I also constrained the study area boundary by excluding slopes > 45°, since forest-dominated vegetation types do not usually occur in these areas. Mean annual precipitation and mean annual maximum temperature ranged from 58.7 cm and 14.3 °C, respectively, at higher elevations on the north end of the study area (Jacob Lake, 1971-2000, elev 2420 m) to 52.6 cm and 21.8 °C, respectively, at lower elevations on the south-central end of the study area (Payson Ranger Station, 1971-2000, elev 1520 m; US Western Regional Climate Center). Approximately 65% of precipitation fell as snow during the winter months (USDA NRCS 2004).

Fire occurrence data

I compiled a digital database of federal fire occurrence data for the period 1 April to 30 September, 1986-2000. This period captures the season with the driest months in the region and monsoonal storm patterns, during which lightning strikes are most common (Swetnam and Betancourt 1998). Forest use and recreational activities are also widespread during this period. I obtained data directly from the US Departments of Agriculture and Interior, and from a national fire occurrence database (USDA Forest Service 1999). Data had a minimum resolution extent of 0.40 ha and included point-oforigin records for lightning- (LF) and human-caused fires (HF) occurring on federal lands managed by the Forest Service, Bureau of Land Management, National Park Service, Fish and Wildlife Service, and Bureau of Indian Affairs (Figure 1.1B). My database did not include fire perimeter or other spatial fire spread information. I restricted my analyses to larger LF or HF (≥ 20 ha in extent; hereafter, I refer only to those events). I chose this threshold because fires burning beyond 20 ha are likely to be influenced more by landscape-level variables than by the immediate ignition environment. In addition, while most ignition events result in fires < 1 ha in size, those fires that do reach 20 ha are likely to grow. For example, in Colorado < 5% of fires reach 20 ha in size, but those that do, have a 46% chance of reaching 100 ha and an 18% chance of reaching 400 ha in size (Neuenschwander et al. 2000). I excluded records on human-caused prescribed fires contained within planned boundaries. To avoid duplicate records in my database, I discarded an occurrence when attribute information were identical to another record within 1000 m. I converted each remaining record in the database to a point feature in a fire occurrence data layer and identified spatial coordinates using a geographic information system (GIS; ArcGIS v9.0, ESRI, Redlands, California, USA).

Spatial input maps

I developed eight spatial data layers, or input maps, that included information on physiographic, biotic, climatic, spatial, and human factors that were likely to influence the probability of fire occurrence. Specifically, I created unique input maps for elevation, slope, aspect score, topographic roughness, ponderosa pine-dominated vegetation, precipitation, road density, and spatial domain (Figures 1.1A and 1.2; see methods below). Because my analytical approach (weights of evidence, see below) required categorical, rather than continuous, input data, I categorized all input maps, with the exception of topographic roughness and spatial domain, into two classes: high and low. The thresholds for each class, or category, were determined by quantile cutoffs based on landscape area. Thus, each category covered an equal proportion of the landscape and the final number of categories in the topographic roughness map was determined by optimizing the maximum number of categories with significant contrast values in a weights-of-evidence analysis (see below; Bonham-Carter et al. 1989). To avoid "data dredging," I limited my search to categories that could be identified using the quantile classification method.

I used the Spatial Analyst extension to ArcGIS to derive the elevation, slope, aspect, and topographic roughness maps based on a 30-m US Geological Survey (USGS) digital elevation model mosaic resampled (continuous) to a 90-m resolution. For elevation (range = 1,520.0 – 3,840.0 m) and slope (range = 0.0 – 45.0°), the value of each 1-km^2 cell in the final map was calculated as the mean value of all 90-m cells contained within that 1-km^2 cell (see Figure 1.1A for elevation map). For aspect, I assigned each 90-m cell an "aspect score" value based on that cell's relationship with the regional prevailing wind direction in the fire season (225.0° or SW); cells with aspects between 195.0° and 255.0° were assigned a value of two, cells between 135.0° and 195.0°, or between 255.0° and 315.0°, a value of one, and all other cells were assigned a value of zero. The final aspect score value for each 1-km^2 cell was the sum of all 90-m cell scores contained within that cell (Figure 1.2A). To derive my terrain roughness map, I calculated the standard deviation for the elevation of all 90-m cells in a 3 x 3 neighborhood. The final value for each 1-km^2 cell was determined by summing the standard deviation values of all 90-m cells within that cell (Figure 1.2B). Low to high class numbers indicate lower to higher degrees of topographic roughness.

Because different forest types have fire regimes that differ in frequency and intensity (Swetnam and Baisan 1996), it was necessary to develop an input map of dominant forest vegetation on my study area. I first obtained a 30-m resolution land cover map from the USGS National Land Cover Dataset (1992). Since this map represents only a coarse classification of dominant forest types (e.g., evergreen forest), I supplemented this map with Enhanced Thematic Mapper (ETM; 30-m resolution) satellite imagery data for the study region. My final forest vegetation input map was a binary map that classified areas as ponderosa pine-dominated (PIPO) or non ponderosa pine-dominated forest (Figure 1.2C). The value for each 1-km^2 cell used in my analyses was determined by resampling all original 30-m cells within that cell.

To assess the influence of climate on fire occurrence, I obtained a 1-km resolution grid representing mean annual precipitation (range = 22.9 – 107.1 cm, mean = 59.5) over the period 1980-1997 (from Daymet US climate model data center; see Thornton et al. 1997; Figure 1.2D). I did not incorporate information on other climatic variables (e.g., temperature, relative humidity, or insolation) because correlations between these variables and fire occurrence are less strong, and because large fires in the Southwest are strongly linked to rainfall patterns (Swetnam and Betancourt 1990, 1998).

To represent patterns of human-use and access on the study area, I used year 2000 US Census Bureau TIGER (Topologically Integrated Geographic Encoding and Referencing) road files to develop a grid-based map of road density (km/km^2; range =

0.0–19.8, mean = 2.3) using a simple density operation in the ArcGIS Spatial Analyst (Figure 1.2E).

To determine if occurrences of fire were influenced by spatial location, I developed a simple input map using six spatial domains (Figure 1.2F). These arbitrary domains were equal in shape and extent and did not capture equal-area proportions of the study area. I also tested the hypothesis of complete spatial randomness in the occurrence data by buffering my study area to 10 km and using an edge corrected point pattern analysis (R = test statistic, $\alpha = 0.05$; see Clark and Evans 1954; Bailey and Gatrell 1995).

Weights-of-evidence modeling

I used weights-of-evidence (WOE) modeling, a Bayesian method of event prediction, to quantify fire occurrence probability. I chose this method because it explicitly considers the spatial association between fire occurrence and input map data (i.e., it is "spatially explicit"), is robust to small sample sizes at large spatial scales, and is easy to implement and interpret using categorical data. Moreover, unlike traditional approaches, the method does not rely on assumptions of normality in the input map distributions and can be informed by a known prior distribution of empirical data. The approach was originally used in medical diagnoses (e.g., Spiegelhalter 1986), but has recently been extended to the prediction and spatial analysis of mineral deposits (Agterberg 1989; Raines and Mihalasky 2002), fossilized packrat middens (Mensing et al. 2000), and plant migrations (Lyford et al. 2003). Weights-of-evidence models use the spatial location of known occurrence points to determine coefficients for a set of categorical input maps (Bonham-Carter et al. 1989). For each analysis unit, or unit cell, these coefficients represent the conditional probability of the input map pattern being: a) present with a known occurrence (e.g., a large fire), b) present without an occurrence, c) absent with an occurrence, or d) absent without an occurrence. The WOE model takes a log-linear form, and the final output is a posterior probability map showing the conditional probability for presence of an occurrence at each unit cell.

Following the procedure described by Bonham-Carter et al. (1989), steps in my WOE analysis included: 1) a priori selection of input maps likely to be useful in prediction of fire occurrence; 2) estimation of a prior probability for the study area given only the known occurrence data; 3) identification of an optimal classification scheme for the input maps and use of conditional probability ratios to calculate weights of evidence for each input map; 4) a pairwise test of conditional independence for each of the input maps, with respect to the known fire occurrences (test statistic = χ^2, $\alpha = 0.01$), combining maps when the conditional assumption was violated; 5) combining the input map weights calculated in step 3; and 6) creating a new map of conditional posterior probability and an estimate of prediction uncertainty for the final input maps correlated with fire occurrence.

For all unit cells where a fire occurred, $N(D)$, and given the total number of cells on my study area, $N(T)$, I computed the prior probability, $P(D)$, of occurrence as

$$P(D) = \frac{N(D)}{N(T)} .$$

Expressed as odds (O), I computed the prior probability that a randomly selected cell contained a fire by

$$O(D) = \frac{P(D)}{P(\overline{D})} ,$$

where $P(\overline{D})$ is the prior probability that a fire did not occur in that cell. Given a set of evidence, E_i, where $i = 1, 2, \ldots, n$, and n is the total number of input maps, where each represents an independent predictor variable, the conditional posterior probability, $P(D \mid E_i)$, was expressed as odds by

$$O(D \mid E_i) = O(D)\frac{P(D \mid E_i)}{P(\overline{D} \mid E_i)}.$$

According to the above equations and Bayes' rule, and assuming conditional independence in the input maps (Bonham-Carter et al. 1989), the following equations can be derived

$$O(D \mid E_i) = O(D)\frac{P(E_i \mid D)}{P(E_i \mid \overline{D})}, \text{ and}$$

$$Ln(O(D \mid E)) = Ln(O(D)) + Ln(\frac{P(E_1 \mid D)}{P(E_1 \mid \overline{D})}) + \ldots + Ln(\frac{P(E_n \mid D)}{P(E_n \mid \overline{D})}).$$

The weight, W_i, for evidence pattern, i, is defined by the expression

$$Ln(\frac{P(E_i \mid D)}{P(E_i \mid \overline{D})}).$$

Thus, if E_i is present, the weight is

$$W_i^+ = Ln(\frac{P(E_i \mid D)}{P(E_i \mid \overline{D})}),$$

and if E_i is absent, the weight is

$$W_i^- = Ln(\frac{P(\overline{E_i} \mid D)}{P(\overline{E_i} \mid \overline{D})}).$$

Therefore, the log odds of a unit cell's posterior probability can be obtained by adding weights W^+ or W^- for presence or absence of each input map unit cell to the log odds of the prior probability, W_0, expressed as

$$Ln(O(D \mid E_i)) = W_0 + W_1^+ (or\, W_1^-) + \ldots + W_n^+ (or\, W_n^-) = \sum_{i=0}^{n} W_i^k,$$

where *k* represents a positive (presence) or negative (absence) weight. Finally, the unit cell posterior probability, *P(D | Eᵢ)*, is obtained from the logit equation

$$P(D|E_i) = \frac{\exp(Ln(O(D|E_i)))}{1+\exp(Ln(O(D|E_i)))}.$$

When an input map pattern was correlated with known occurrences, the contrast

$$C = W_i^+ - W_i^-$$

provided a measure of the strength of this correlation. A positive or negative value (range between +2 and -2) for *C* indicated a positive or negative spatial correlation, respectively. I ranked the relative importance of each input map according to the value for *C*. I considered absolute values for $C \geq 0.30$ to represent more meaningful contrasts. To test whether the contrast value for each individual input map was sufficiently different from 0 (no correlation), I calculated a "studentized" contrast value (test statistic = student(*c*), α = 0.05; Bonham-Carter et al. 1989). For each WOE analysis, the weights from each of the overlapping input maps with statistically significant studentized contrast values were summed, resulting in an output map representing an integrated pattern of posterior conditional probabilities.

To assess uncertainties associated with my posterior probability maps, I estimated the total uncertainty (Bonham-Carter et al. 1989) as the variance in the weights, combined with the estimated variance for any missing cell values in the input maps. Uncertainties due to differences in the weights of overlapping input maps were calculated as

$$\sigma^2(P_{post}) = \left[\sigma^2 + \sum_{i=1}^{n} \sigma^2(W_i^k)\right] \bullet P^2{}_{post}.$$

Uncertainties due to missing or incomplete values in the *i*-th input map were estimated as

$$\sigma_i^2\left(P_{post}\right)=\{P(D|E_i)-P(D)\}^2\,P(E_i)+\{P(D|\overline{E_i})-P(D)\}^2\,P(\overline{E_i})\,.$$

Total uncertainty in the posterior probability maps was estimated as

$$\sigma^2\left(TOTAL\right)=\sigma^2\left(WEIGHTS\right)+\sum_{i=1}^{n}\sigma_i^2\,(MISSING)\,.$$

For the final uncertainty maps, I calculated a studentized uncertainty statistic for each cell as

$$P_{post}\,/\,\sigma_{TOTAL}\,.$$

Values of this ratio < 1.960 represented cells with significant uncertainty ($\alpha = 0.05$; Bonham-Carter et al. 1989).

I used the Arc-SDM (Kemp et al. 2001) spatial data modeler extension to ArcView v3.3 (ESRI, Redlands, California, USA) to conduct all WOE analyses. I modeled all fire (AF) types (LF and HF combined), and then I modeled LF alone. Because too few (< 100) records for HF were present in my final database, I did not model these occurrences separately. I report all probability and uncertainty values per 1-km^2 cell for my period of study.

RESULTS

Fire occurrence

Between 1 April and 30 September, 1986-2000, 203 fires occurred on my study area (Figure 1.1B, Appendix 1.A) and burned 101,751 ha. Of this total, 136 (67%) were LF and most (71%, $n = 97$) of these burned a total area ≤ 200 ha (Appendix 1.B). LF burned more than 4 times the area burned by HF (83,055 vs. 18,696 ha). The greatest number of fires occurred in 2000 ($n = 26$) and the highest amount of total annual area burned in

1996 (38,140 ha). Of this total area, 32,674 ha (86%) burned as a result of LF. The highest annual amount of average area burned also occurred in 1996 (2,119 ha, *SD* = 3,831, $n = 18$). HF burned the most area in 2000 (total = 6,422, mean = 2,141, $n = 3$).

The spatial distribution of fire occurrences on my study area was significantly nonrandom for AF ($R = 0.704$, $z = -8.805$, $P < 0.001$, $n = 242$) and for LF alone ($R = 0.683$, $z = -7.687$, $P < 0.001$, $n = 161$).

Probability modeling

Spatial input maps for slope and topographic roughness were highly correlated (Pearson product moment correlation coefficient = 0.87), leading us to drop slope and consider seven input maps in my WOE models. As might be expected for this region, mean annual precipitation and elevation were weakly correlated (correlation coefficient = 0.30), as were precipitation and topographic roughness (0.26). I did not consider these correlations sufficient to drop these input maps from subsequent analyses. The pairwise test of the assumption of conditional independence for the ponderosa pine-dominated forest and precipitation input maps was not satisfied in the analysis of AF ($\chi^2 = 41.6$, $P > 0.01$, d.f. = 2). Therefore, I combined these maps (Bonham-Carter et al. 1989; Agterberg and Cheng 2002) and evaluated a new input map (Forest_Precip) with 2 binary classes: presence of ponderosa pine-dominated forest or high precipitation (class 1; 69% of study area) and absence of both (class 2; 31% of study area). New tests for all of the input maps satisfied the conditional independence assumption.

The prior probability of AF was 0.008. At the resolution and extent (spatial and temporal) of my input maps, the AF posterior probability was most influenced by topographic roughness, followed by road density, and Forest_Precip (Table 1.1).

Elevation, aspect, and spatial domain were not important predictors of occurrence in this model ($C < 0.30$, $P > 0.05$). For this analysis, I identified 3 topographic roughness classes for which absolute contrast values were ≥ 0.30 and statistically significant. High topographic roughness was the best predictor of AF occurrence ($C = 0.912$), and was also characterized by the highest positive weight ($W^+ = 0.511$). Low road density and Forest_Precip were also good predictors. Areas of moderate ($C = -0.538$) and low ($C = -0.572$) topographic roughness were also considered important predictors of where AF were unlikely to occur.

My WOE model for AF summed the weights of the topographic roughness, road density, and Forest_Precip maps. The posterior probability of a fire occurrence ranged between 0.012 and 0.074 (mean = 0.031, SD = 0.016; Figure 1.3A). The corresponding uncertainty for these conditional probabilities ranged between 0.003 and 0.021 (mean = 0.008, SD = 0.004). Because of the relatively large number of occurrences in my WOE analysis of AF, my use of statistically significant input maps, and few missing data in my overlapping input maps, total uncertainty was minimized (Figure 1.3B). No cells had studentized uncertainty values < 1.960.

The prior probability for LF was 0.005. My tests of the conditional independence assumptions for the input maps in the LF analysis were satisfied using the topographic roughness, road density, forest vegetation, and aspect score input maps. Posterior probability of LF was most influenced by topographic roughness, followed by road density, ponderosa pine-dominated forest, and aspect (Table 1.2). Precipitation, elevation, and spatial domain were not significant predictors. For LF, I again identified 3 topographic roughness classes for which contrast values were > 0.30. High topographic roughness was the most important ($C = 1.246$) predictor. Cells with highest values also had the largest positive weight ($W^+ = 0.647$). Low road density and ponderosa pine-dominated forest were better predictors of occurrence than high aspect score. Areas with the lowest values for topographic roughness were important ($C = -1.045$) predictors of where LF were unlikely to occur.

For my WOE model of LF, I summed the weights for the topographic roughness, road density, forest vegetation, and aspect score input maps. The posterior probability of a fire due to LF ranged between 0.003 and 0.078 (mean = 0.018, SD = 0.017; Figure 1.3C). The corresponding total uncertainty for these conditional probabilities ranged between 0.001 and 0.021 (mean = 0.006, SD = 0.005). Compared to the analysis using AF, fewer occurrences in the WOE analysis of LF provided for greater uncertainty in more cells (Figure 1.3D). However, very few ($n = 34$) cells had studentized uncertainty values < 1.960.

DISCUSSION

Fire occurrence

Consistent with an earlier figure reported by Barrows (1978) for all natural-caused fires, I found that 67% of all fires on my study area were LF. However, a relatively small number of these fires represented a substantial fraction of the total annual area burned during my period of study. Even though they occurred infrequently, HF burned extremely large areas. For example, a human-ignited prescribed fire in Grand Canyon National Park in 2002 escaped its boundary and burned 6,243 ha, or 97% of the total area burned due to human causes in that year (33% of the period total). In June of 2002 the

Rodeo-Chediski fire, the largest recorded fire in Arizona state history, was human ignited and burned approximately 187,000 ha within my study area.

Probability modeling

I identified significant interactions between landscape features and landscape-scale patterns of fire occurrence. Predictors differed by analysis (LF vs. AF), although highly ranked predictors were similar for both WOE analyses. For the most important predictors, I observed higher contrast values and weights for LF than for AF. Patterns observed in my analysis of AF were likely dampened by the inclusion of HF, which were highly variable in their timing, location, and extent. Additionally, because my analyses considered only one anthropogenic input map (road density), I was unable to account for the range of unique factors that likely influence patterns of HF (e.g., proximity to urban centers, human density). A qualitative examination of patterns for AF revealed that HF are more likely to occur in areas of high road density and LF in areas of low road density.

For the 15-year period I analyzed, the maximum conditional posterior probability value for each of my WOE analyses was seemingly small: 0.078 for LF and 0.074 for AF. However, compared with their prior probabilities, these maximum values yield odds ratios of ~10:1 for AF and ~ 16:1 for LF. Because I was unable to include important fire behavior variables in my WOE analyses I believe that the maximum probability of a fire is higher than I estimated. For example, accurate input maps for fuel type and fuel load were unavailable at the temporal and spatial extent of my analyses. Temporal variability in fuel moisture, humidity, wind speed, and other factors that are difficult to incorporate into spatial models, will also result in variability in the occurrence of large fires.

Nevertheless, I believe my posterior probability estimates capture the statistical and ecological importance of the input maps included in the WOE analyses.

Topographic roughness was an important landscape feature in predicting the occurrence of fire, a result not previously demonstrated at the landscape scale for the Southwest. Guyette and Dey (2000) identified topographic roughness as one of the most important and temporally persistent landscape variables in their assessment of fire frequency in the Ozark Mountains of southeastern Missouri. The interaction between topography and fire behavior is a complex process mediated by the influence of local climate, vegetation, and the spatial distribution of fuels (Whelan 1995). Topographically complex areas can facilitate or impede fire occurrence and behavior (Whelan 1995; Graham et al. 2004). Moreover, rate of spread may increase with steeper slopes because flames are angled closer to the ground and because the process of heat convection within the fire produces supplemental wind effects (Whelan 1995; DeBano et al. 1998).

Fire suppression efforts in areas of remote and rough terrain can be constrained by slower reporting and response times and limited access. The observed relationship between high topographic roughness and the posterior probability of fire occurrence may be influenced by this circumstance. Although the road density and topographic roughness input maps were not correlated (correlation coefficient = 0.04), areas with lower road densities were highly ranked by each WOE analysis. For either LF or HF, areas with lower road densities may place fewer artificial fuel breaks in the path of an expanding fire event. If larger fires occur in rugged areas with lower road densities, then the role of limited road access for suppression efforts, for example, should be recognized in the management of the present fire regime. This is not to suggest, however, that fire occurrence could be reduced by the construction of new roads in fire-prone areas; more roads in these areas will allow increased access by humans, which is likely to result in an increase in HF (Swetnam 1990; Brown et al. 2004). In the upper Midwest, where most fires are human-caused, the probability of occurrence of a larger fire has been found to be positively correlated with road density (Cardille et al. 2001), and in the San Jacinto mountains of California, fires are more likely to occur near roads (Chou et al. 1993). An increase in HF could offset the ecological benefits of fires due to LF, or the perceived benefits of fire suppression activities in remote areas. Additionally, road building can promote resource erosion and degradation (see Grigal 2000; DellaSala and Frost 2001), increase invasion by exotic species (Forman 2000; Gelbard and Belnap 2003), and fragment habitats (Reed et al. 1996; McGarigal et al. 2001).

I was unable to identify the specific mechanisms underlying the significant relationship between ponderosa pine-dominated forest and increased probability of fire occurrence. However, previous research in the region has identified a number of possible factors including a recent and rapid accumulation of forest-floor fuels (Sackett and Haase 1996), tree densities surpassing historic levels (Covington and Moore 1994), reduced tree vigor (Covington et al. 1997), and increases in the incidence of tree mortality agents, such as bark beetles and dwarf mistletoes (see Dahms and Geils 1997). These factors have likely been exacerbated by intensive livestock grazing, timber harvesting, and fire suppression activities (Covington and Moore 1994). Moreover, increased human use of the ponderosa pine-dominated forest type, primarily in the form of recreation activities and development (see Dahms and Geils 1997), could further modify the forested landscape in ways that facilitate large fire events.

Areas with high precipitation did not rank as a dominant influence on AF patterns. The coarse resolution (spatial and temporal) of my precipitation input map may have resulted in low power to detect relationships. In the Southwest, lightning strikes and high levels of precipitation are often significantly positively correlated (Gosz et al. 1995). However, my results indicate that LF were not correlated with precipitation during the period of my analysis. On my study area, high levels of precipitation likely contribute to increased fuels in ponderosa pine-dominated stands. For example, Swetnam and Betancourt (1998) identified a strong relationship between the recent growth of southwestern trees and exceptionally high amounts of annual precipitation since 1976. In response to a dry period that follows a sequence of extremely wet seasons, accumulated fuels can contribute to exceptionally large fire events (Swetnam and Betancourt 1998; Grissino-Mayer and Swetnam 2000). Because my occurrence data span a period of 15 years, the role of longer-term patterns of climatic oscillation (e.g., ENSO) and periodic drought is not well represented by my analysis and interpretation. Nevertheless, many of the major controlling factors in my models are topographic in nature (roughness, aspect), and these may be more important in determining local patterns of fires than climatic effects, which are likely to affect the entire region in a similar fashion.

In my WOE analysis of LF, aspect score was not a highly-ranked predictor variable. Areas with aspects facing the prevailing wind direction (generally, south-southwest) were significantly related with LF. The more open stands and lower tree densities that tend to occur on these aspects permit higher wind speeds (Weatherspoon 1996). Combined with higher amounts of solar radiation, this factor often facilitates more rapid drying of surface and standing fuels (Weatherspoon 1996) and increase probability of ignition (Graham et al. 2004). My results suggest that aspect score, based on prevailing wind direction, captures important landscape features related to large fire occurrence.

Elevation was not highly correlated with the regional occurrence of fire. Although I constrained my analyses to include only those fires above 1,520 m, my study area included a wide elevation range (2,320 m). Previous research in other regions of the West has identified relationships between LF and elevation (Vankat 1985; van Wagtendonk 1991; Díaz-Avalos et al. 2001; Fulé et al. 2003). These studies, however, evaluated the frequency of fire events of any detectable size. Similar to my results, Preisler et al. (2004) concluded that elevation was not a significant predictor of an ignition turning into a large fire in Oregon, and in the Pacific Northwest, Heyerdahl et al. (2001) concluded elevation was not a primary control of the fire regime at the regional scale.

The occurrence of fires in each spatial domain provided insufficient evidence for coarse-scale, spatial clustering of these events. At a regional scale, Díaz-Avalos et al. (2001) and Preisler et al. (2004) each identified the importance of spatial location in estimating the probability of ignition occurrence in Oregon. However, my more coarse method of characterizing spatial location in the WOE models was unable to detect a statistically similar pattern. Apparent clustering of LF in the southeastern domain occurred in the most rugged, remote, and lightly populated subregion of the study area. Thus, factors such as the time lag in suppression efforts due to delays in reporting and response may result in a larger number of fires than in other subregions.

The nonrandom distribution of fire occurrence on my study area indicated localized patterns of spatial clustering in these events. On average, the nearest-neighbor distance between AF and LF types was 4 km and 5 km, respectively. Using point pattern analysis, Podur et al. (2003) also detected significant local-scale clustering in lightning strikes in Ontario and determined these patterns to be principally related to localized phenomena. My results suggest that event clustering was a function of local-scale factors and that occurrence should be modeled as a multi-scale process.

Management and research implications

My results indicate that seasonal fire events on the region I studied were a consequence of nonrandom patterns of occurrence, and that these patterns are significantly related to environmental factors. The occurrence pattern of fires on my study area was not strongly associated with precipitation. "Top-down" influences (*sensu* Heyerdahl et al. 2001), such as those exerted by regional climatic patterns at human time scales, may not currently affect the regional fire regime to the extent they did historically. Instead, topographic roughness, combined with reduced access to these areas, appear to be significant controls (the "bottom-up" controls posited by Heyerdahl et al. 2001) on the present fire regime in this region.

Forest fuels reduction and restoration treatments can be important in managing the threat of fire to communities and resources (Covington 2000). Locating these treatments in remote and rugged areas is strategically difficult and prescribed fires in more accessible locations appears to be a reasonable management alternative so long as human communities are protected (Allen et al. 2002; Dombeck et al. 2004). However, fire behavior and the restoration of fire regimes in these locations deserves greater research attention. Recent research has assessed the potential to minimize fire threat to populated areas by strategically placing forest treatments and fuel breaks around communities (see Graham et al. 2004). My results suggest that treatments intended to reduce fire threat around communities should first target areas bordering rough terrain, thus providing a fuel break in areas where fires are more likely to spread. Because fires tend to ignite in rugged and remote areas, fire suppression efforts in neighboring populated areas should be evaluated in the context of public acceptance of fire as a natural disturbance process.

Using a Bayesian framework, Díaz-Avalos et al. (2001) also quantified the influence of spatial and environmental risk factors on the regional probability of fire occurrence. Like Díaz-Avalos et al. (2001), my novel and spatially explicit methods provide a tractable approach to modeling probability of fire occurrence, and my map outputs can be useful in the planning and coordination of community and/or regional efforts to identify areas at greatest risk. Presently, my WOE models are being used to develop maps of fire risk on an 800,000-ha landscape in northern Arizona and to model priority areas for landscape-level treatments (Sisk et al. 2006). I agree with Prestemon et al. (2002) that an improved understanding of fire risk must integrate patterns of human activity and that continued research is needed to assess wildfire-risk factors and damage-reduction strategies. Often, fuel loadings are the only characteristic taken into account when planning management actions to reduce fire threat. Moreover, it is my experience that fire managers often believe that locations where large fires are likely to start cannot be identified spatially. However, fires require not only fuels, but ignition sources and conditions that promote fire spread. While fuels reduction is important in managing fire risk, treatments designed to reduce fuels may do little to reduce fire threat if they are not strategically placed in or around areas where large fire events are most likely to occur. Insights into the patterns of fire risk, in terms of landscape attributes, will increase our ability to assess and manage fire threat. In addition, knowledge of occurrence patterns will accelerate restoration efforts, particularly when natural fire is a component of the restoration prescription.

LITERATURE CITED

Agee J.K. 1998. The landscape ecology of western forest fire regimes. Northwest Science 72(SI): 24-34.

Agterberg F.P. 1989. Computer programs for mineral exploration. Science 245: 76-81.

Agterberg F.P. and Cheng Q. 2002. Conditional independence test for weights-of-evidence modeling. Natural Resource Research 11: 249-255.

Allen C.D. 2002. Lots of lightning and plenty of people: An ecological history of fire in the upland Southwest. *In* Vale T.R. (ed.), Fire, native peoples, and the natural landscape, pp. 143-193. Island Press, Covelo, California, USA.

Allen C.D., Savage M., Falk D.A., Suckling K.F., Swetnam T.W., Schulke T., Stacey P.B., Morgan P., Hoffman M. and Klingel J.T. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12: 1418-1433.

Attiwill P.M. 1994. The disturbance of forest ecosystems: the ecological basis for conservative management. Forest Ecology and Management 63: 247-300.

Bailey T.C. and Gatrell A.C. 1995. Interactive spatial data analysis. Longman Group Limited, Essex, UK.

Barrows J.S. 1978. Lightning fires in southwestern forests. Final Report to USDA Forest Service, Intermountain Forest and Range Experiment Station. Cooperative Agreement 16-568-CA. Fort Collins, Colorado, USA.

Bonham-Carter G.F., Agterberg F.P. and Wright D.F. 1989. Weights of evidence modelling: a new approach to mapping mineral potential. *In* Agterberg F.P and Bonham-Carter G.F. (eds.), Statistical applications in the earth sciences, pp. 171-183. Geological Survey of Canada paper 89-9, Ottawa, Ontario, Canada.

Brown R.T., Agee J.K. and Franklin J.F. 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18: 903-912.

Brunson M.W. and Shindler B.A. 2004. Geographic variation in social acceptability of wildland fuels management in the western United States. Society and Natural Resources 17: 661-678.

Cardille J.A., Ventura S.J. and Turner M.G. 2001. Environmental and social factors influencing wildfires in the Upper Midwest, United States. Ecological Applications 11: 111-127.

Case P., Banks B., Butler E. and Gosnell R. 2000. Assessing potential wildfire effects on people. *In* Sampson R.N., Atkinson R.D. and Lewis J.W. (eds.), Mapping wildfire hazards and risks, pp. 159-176. The Haworth Press, New York, New York, USA.

Chou Y.H., Minnich R.A. and Chase R.A. 1993. Mapping probability of fire occurrence in the San Jacinto Mountains, California. Environmental Management 17: 129-140.

Clark P.J. and Evans F.C. 1954. Distance to nearest neighbor as a measure of spatial relationships in populations. Ecology 35: 445-453.

Covington, W. W. 2000. Helping western forests heal: the prognosis is poor for U.S. forest ecosystems. Nature 408: 135-136.

Covington W.W., Fulé P.Z., Moore M.M., Hart S.C., Kolb T.E., Sackett J.N. and Wagner M.R. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. Journal of Forestry 95: 23-29.

Covington W.W. and Moore M.M. 1994. Southwestern ponderosa forest structure and resource conditions: changes since Euro-American settlement. Journal of Forestry 92: 39-47.

Dahms C.W. and Geils B.W. (eds.). 1997. An assessment of forest ecosystem health in the Southwest. USDA Forest Service General Technical Report RM-GTR-295. Fort Collins, Colorado, USA.

Dale V.H., Joyce L.A., McNulty S., Neilson R.P., Ayres M.P., Flannigan M.D., Hanson P.J., Irland L.C., Lugo A.E., Peterson C.J., Simberloff D., Swanson F.J., Stocks B.J. and Wotton B.M. 2001. Climate change and forest disturbances. BioScience 51: 723-734.

Davis J.B. 1990. The wildland-urban interface: paradise or battleground. Journal of Forestry 88: 26-31.

DeBano L.F., Neary D.G. and Ffolliott P.F. 1998. Fire's effects on ecosystems. John Wiley and Sons, New York, New York, USA.

DellaSala D.A. and Frost E. 2001. An ecologically based strategy for fire and fuels management in national forest roadless areas. Fire Management Today 61: 12-23.

de Vasconcelos M.J.P., Silva S., Tomé M., Alvim M. and Pereira J.M.C. 2001. Spatial prediction of fire ignition probabilities: comparing logistic regression and neural networks. Photogrammetric Engineering and Remote Sensing 67: 73-81.

Díaz-Avalos C., Peterson D.L., Alvarado E., Ferguson S.A. and Besag J.E. 2001. Space–time modelling of lightning-caused ignitions in the Blue Mountains, Oregon. Canadian Journal of Forest Research 31: 1579-1593.

Dombeck M.P., Williams J.E. and Wood C.A. 2004. Wildfire policy and public lands: integrating scientific understanding with social concerns across landscapes. Conservation Biology 18: 883-889.

Forman R.T.T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation Biology 14: 31-35.

Fulé P.Z., Heinlein T.A., Covington W.W. and Moore M.M. 2003. Assessing fire regimes on Grand Canyon landscapes with fire-scar and fire-record data. International Journal of Wildland Fire 12: 129-145

Gelbard J.L. and Belnap J. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. Conservation Biology 17: 420-432.

Gosz J.R., Moore D.I., Shore G.A., Grover H.D., Rison W. and Rison C. 1995. Lightning estimates of precipitation location and quantity on the Sevilleta LTER, New Mexico. Ecological Applications 5: 1141-1150.

Graham R.T., McCaffrey S. and Jain T.B. (eds.). 2004. Science basis for changing forest structure to modify wildfire behavior and severity. USDA Forest Service General Technical Report RMRS-GTR-120. Fort Collins, Colorado, USA.

Grigal D.F. 2000. Effects of extensive forest management on soil productivity. Forest Ecology and Management 138: 167-185.

Grissino-Mayer H.D. and Swetnam T.W. 2000. Century-scale climate forcing of fire regimes in the American Southwest. The Holocene 10: 213-220.

Guyette R.P. and Dey D.C. 2000. Humans, topography, and wildland fire: the ingredients for long-term patterns in ecosystems. *In* Proceedings of the workshop on fire, people, and the central hardwoods landscape, pp. 28-35. USDA Forest Service General Technical Report NE-274, Fort Collins, Colorado, USA.

Heyerdahl E.K., Brubaker L.B. and Agee J.K. 2001. Spatial controls of historical fire regimes: a multiscale example from the interior West, USA. Ecology 82: 660-678.

Keeley J.E. and Fotheringham C.J. 2001. Historic fire regime in southern California shrublands. Conservation Biology 15: 1536-1548.

Kemp L.D., Bonham-Carter G.F., Raines G.L. and Looney C.G. 2001. Arc-SDM: ArcView extension for spatial data modelling using weights of evidence, logistic regression, fuzzy logic, and neural network analysis. [Online: (http://ntserv.gis.nrcan.gc.ca/sdm/).]

Lyford M.E., Jackson S.T., Betancourt J.L. and Gray S.T. 2003. Influence of landscape structure and climate variability on a Late Holocene plant migration. Ecological Monographs 73: 567-583.

Malamud B.D., Millington J.D.A. and Perry G.L.W. 2005. Characterizing wildfire regimes in the United States. Proceedings of the National Academy of Sciences of the United States of America 102: 4694-4699

McGarigal K., Romme W.H., Crist M. and Roworth E. 2001. Cumulative effects of roads and logging on landscape structure in the San Juan Mountains, Colorado (USA). Landscape Ecology 16: 327-349.

McKenzie D., Gedalof Z., Peterson D.L. and Mote P. 2004. Climatic change, wildfire, and conservation. Conservation Biology 18: 890-902.

Mensing S.A., Elston, Jr. R.G., Raines G.L., Tausch R.J. and Nowak C.L. 2000. A GIS model to predict the location of fossil packrat (*Neotoma*) middens in central Nevada. Western North American Naturalist 60: 111–120.

Neuenschwander L.F., Menakis J.P., Miller M., Sampson R.N., Hardy C., Averill B. and Mask R. 2000. Indexing Colorado watersheds to risk of wildfire. *In* Sampson R.N., Atkinson R.D. and Lewis J.W. (eds.), Mapping wildfire hazards and risks, pp. 35-56. The Haworth Press, New York, New York, USA.

Podur J., Martell D.L. and Csillag F. 2003. Spatial patterns of lightning-caused forest fires in Ontario. Ecological Modelling 164: 1-20.

Preisler H.K., Brillinger D.R., Burgan R.E. and Benoit J.W. 2004. Probability based models for estimation of wildfire risk. International Journal of Wildland Fire 13: 133-142.

Prestemon J.P., Pye J.M., Butry D.T., Holmes T.P. and Mercer D.E. 2002. Understanding broadscale wildfire risks in a human-dominated landscape. Forest Science 48: 685-693.

Raines G.L. and Mihalasky M.J. 2002. A reconnaissance method for delineation of tracts for regional-scale mineral-resource assessment based on geologic-map data. Natural Resources Research 11: 241-248.

Reed R.A., JohnsonBarnard J. and Baker W.L. 1996. Contribution of roads to forest fragmentation in the Rocky Mountains. Conservation Biology 10: 1098-1106.

Sackett S.S. and Haase S.M. 1996. Fuel loadings in southwestern ecosystems of the United States. *In* Proceedings of the symposium on fire on Madrean Province ecosystems, pp. 187-192. USDA Forest Service General Technical Report RM-GTR-289, Fort Collins, Colorado, USA.

Sisk T.D., Prather J.W., Hampton H.M., Aumack E.N., Xu Y. and Dickson B.G. 2006. Participatory landscape analysis to guide restoration of ponderosa pine ecosystems in the American Southwest. Landscape and Urban Planning 79: xx-xx.

Spiegelhalter D.J. 1986. Probabilistic prediction in patient management. Statistics in Medicine 5: 421-433.

Swetnam T.W. 1990. Fire history and climate in the southwestern United States. *In* J. S. Krammes J.S. (ed.), Proceedings of the symposium on effects of fire management of southwestern natural resources. USDA Forest Service General Technical Report RM-191, Fort Collins, Colorado, USA.

Swetnam T.W. and Baisan C.H. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. *In* Allen C.D. (ed.), Fire effects in southwestern forests, pp. 11-32. Proceedings of the second La Mesa fire symposium. USDA Forest Service General Technical Report RM-GTR286, Fort Collins, Colorado, USA.

Swetnam T.W. and Betancourt J.L. 1990. Fire-southern oscillation relations in the southwestern United States. Science 249: 1017-1020.

Swetnam T.W. and Betancourt J.L. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate 11: 3128-3147.

Thornton P.E., Running S.W. and White M.A. 1997. Generating surfaces of daily meteorology variables over large regions of complex terrain. Journal of Hydrology 190: 214-251.

USDA Forest Service. 1999. National fire occurrence data 1986-1996. [Online: (http://www.fs.fed.us/fire/fuelman/).]

USDA Forest Service. 2004. Southwest area wildland fire operations website. [Online: (http://www.fs.fed.us/r3/fire/).]

USDA NRCS. 2004. Natural Resources Conservation Service website. [Online: (http://www.az.nrcs.usda.gov/snow/data/)]

USDA and USDI [USDA Forest Service and U.S. Department of the Interior]. 2000. National fire plan: managing the impact of wildfires on communities and the environment. A report to the President in response to the wildfires of 2000. [Online: (http://www.fireplan.gov)]

Vankat J.L. 1985. Patterns of lightning ignitions in Sequoia National Park, California. *In* proceedings, symposium, and workshop on wilderness fire, pp. 408-411. USDA Forest Service General Technical Report INT-182, Ogden, Utah, USA.

van Wagtendonk J.W. 1991. Spatial analysis of lightning strikes in Yosemite National Park. *In* Andrews P.L. and Potts D.F. (eds.), Proceedings of the 11th CNFerence on fire and forest meteorology, pp. 605-611. Society of American Foresters, Bethesda, Maryland, USA.

Weatherspoon C.P. 1996. Fire-silviculture relationships in Sierra forests. *In* Sierra Nevada ecosystem project: final report to congress, pp.1167-1176. Assessments and scientific management options, Volume 2, Section 6. University of California, Centers for Water and Wildland Resources, Davis, California, USA.

Whelan R.J. 1995. The ecology of fire. Cambridge University Press, Cambridge, UK.

Table 1.1. Input maps significantly correlated with the occurrence of all (n = 203) large fires on the study area, and their associated WOE statistics, for the period 1 April to 30 September, 1986-2000. Because of the inverse relationship between binary value input map classes, I report only those class results for which values of C were positive. Input map importance in predicting large fire occurrence is ranked from top (high) to bottom (low) according to the value of C. Forest_Precip class 1 indicates presence of ponderosa pine-dominated forest or high precipitation.

Input map	Class	No. of occurrences	W^+	$SD(W^+)$	W^-	$SD(W^-)$	Contrast (C)	$SD(C)$	student(c)*
Roughness	High	112	0.511	0.095	-0.401	0.105	0.912	0.142	6.435
Road density	Low	118	0.244	0.093	-0.263	0.109	0.507	0.143	3.552
Forest_Precip	1	153	0.091	0.081	-0.240	0.143	0.332	0.165	2.014
Roughness	Moderate	47	-0.385	0.146	0.153	0.080	-0.538	0.167	-3.221
Roughness	Low	44	-0.417	0.151	0.155	0.080	-0.572	0.171	-3.345

* Values statistically significant at $\alpha = 0.05$ (> 1.96, < -1.96).

Table 1.2. Input maps significantly correlated with the occurrence of lightning-caused (n = 136) large fires on the study area, and their associated WOE statistics, for the period 1 April to 30 September, 1986-2000. Because of the inverse relationship between binary value input map classes, I report only those class results for which values of C were positive. Input map importance in predicting large fire ignition occurrence is ranked from top (high) to bottom (low) according to the value of C.

Input map	Class	No. of occurrences	W^+	$SD\ (W^+)$	W^-	$SD\ (W^-)$	Contrast (C)	$SD\ (C)$	student(c)*
Roughness	High	86	0.647	0.108	-0.599	0.142	1.246	0.178	6.988
Road density	Low	91	0.385	0.105	-0.499	0.149	0.884	0.183	4.839
Forest vegetation	PIPO	96	0.182	0.102	-0.337	0.158	0.519	0.189	2.752
Aspect score	High	83	0.155	0.110	-0.203	0.138	0.358	0.176	2.030
Roughness	Moderate	30	-0.433	0.182	0.166	0.097	-0.599	0.207	-2.891
Roughness	Low	20	-0.805	0.224	0.240	0.093	-1.045	0.243	-4.310

* Values statistically significant at $\alpha = 0.05$ (> 1.96, < -1.96).

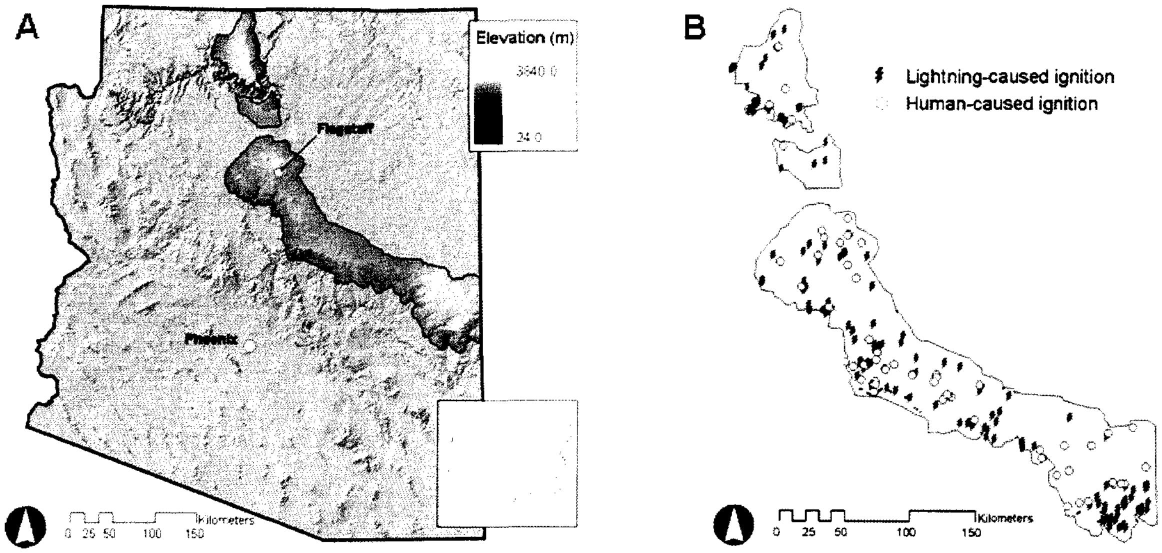

Figure 1.1. A) The 27,065-km^2 study area used to examine probability of large fire occurrence. The study area included the large, ponderosa pine-dominated forest regions of northern Arizona, USA. B) Distribution of lightning- (n = 136) and human-caused (n = 67) large fires on the study area for the period 1 April to 30 September, 1986-2000.

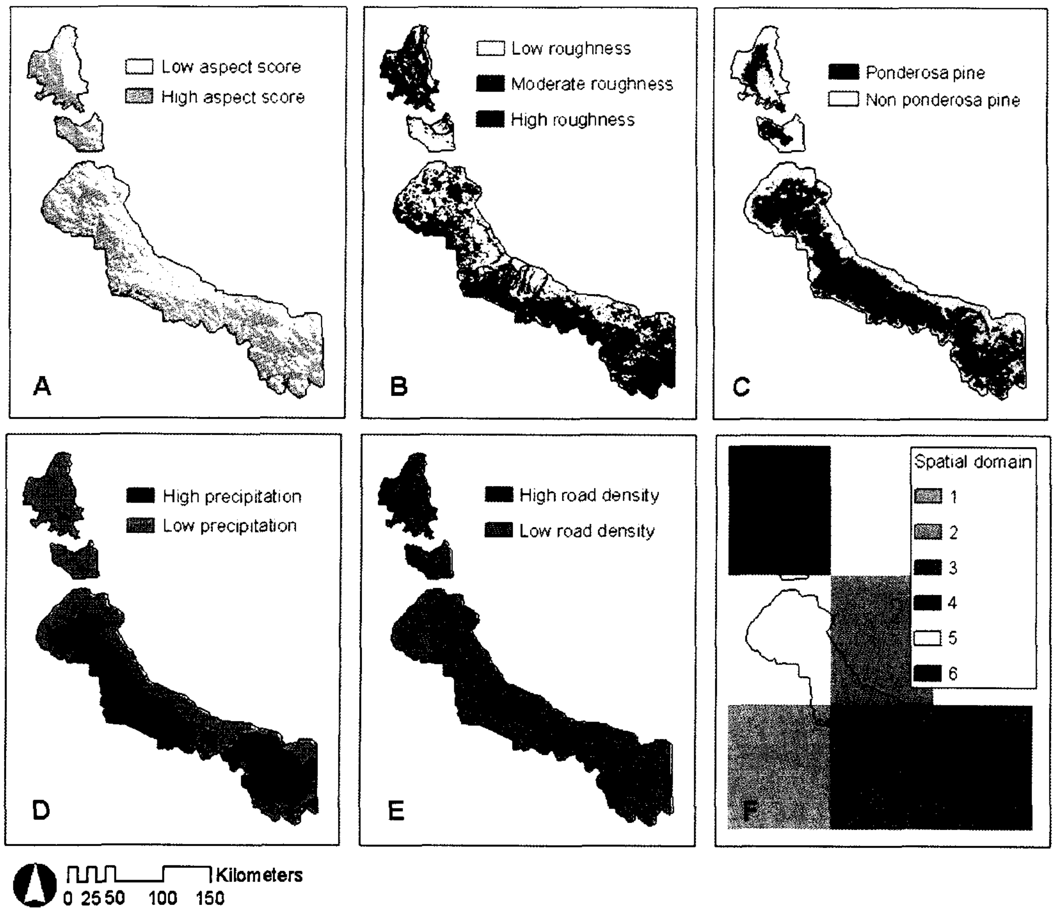

Figure 1.2. Spatial input maps used in the analyses of large fires in northern Arizona. Input maps for A) aspect score, B) topographic roughness, C) forest type, D) precipitation, E) road density, and F) spatial domain are shown.

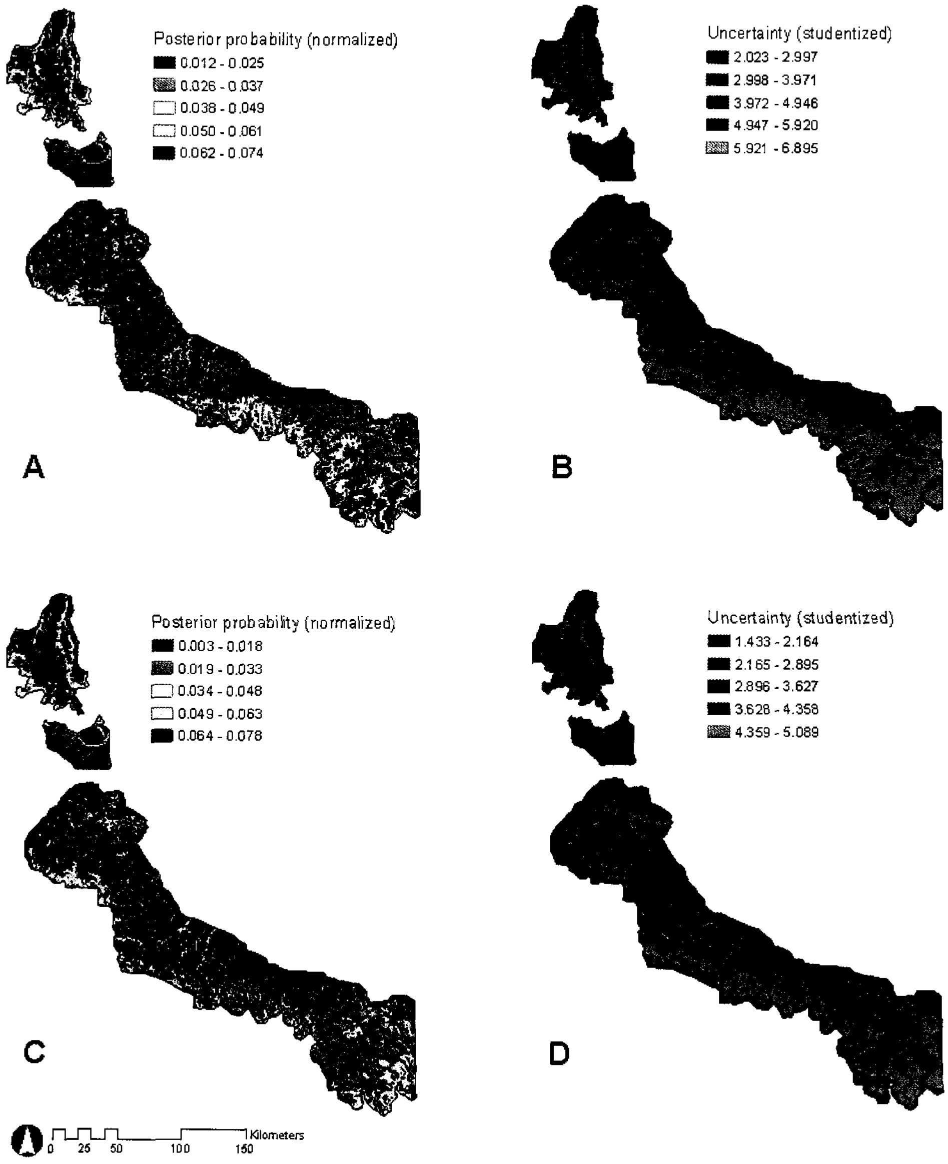

Figure 1.3. A) Normalized posterior probability of occurrence and B) studentized uncertainty values for all (*n* = 203) large fire types on the study area. C) Normalized posterior probability of occurrence and D) studentized uncertainty values for lightning-caused (*n* = 136) large fires on the study area. Lower studentized uncertainty values indicate greater uncertainty and values < 1.960 are not statistically different from zero. Ranges for all values are scaled using an equal interval classification.

Appendix 1.A. Yearly number of occurrences, total area (ha), and mean (+/-SD) area (ha) burned for all ($n = 203$) large fire ignition types on the study area, 1 April to 30 September, 1986-2000.

	All ignitions combined			Lightning-caused ignitions			Human-caused ignitions		
Year	No. of occurrences	Total area burned	Mean area burned	No. of occurrences	Total area burned	Mean area burned	No. of occurrences	Total area burned	Mean area burned
1986	10	1669	167 (306)	9	1588	176 (323)	1	80	80
1987	22	7738	352 (480)	17	5756	339 (421)	5	1982	396 (706)
1988	14	3128	223 (359)	14	3128	223 (359)	0	0	0
1989	25	3093	124 (190)	15	2263	151 (242)	10	830	83 (45)
1990	13	13065	1005 (3164)	8	12649	1581 (4021)	5	416	83 (74)
1991	7	387	55 (53)	1	21	21	6	366	61 (56)
1992	2	65	32	0	0	0	2	65	32
1993	12	1709	142 (209)	3	134	45 (25)	9	1575	175 (235)
1994	12	2768	231 (406)	8	2622	328 (477)	4	146	37 (3)
1995	14	10307	736 (2123)	3	9621	3207 (4194)	11	685	62 (87)
1996	18	38140	2119 (3831)	14	32674	2334 (4286)	4	5467	1367 (1611)
1997	7	236	34 (11)	6	208	35 (11)	1	27	27
1998	10	1045	104 (54)	7	605	86 (52)	3	440	147 (32)
1999	11	1928	175 (274)	8	1734	217 (314)	3	194	65 (67)
2000	26	16475	634 (1477)	23	10052	437 (991)	3	6422	2141 (3553)
Mean	14 (7)	6783 (10037)	409 (552)	9 (6)	5537 (8545)	612 (971)	4 (3)	1246 (1999)	317 (609)
Total	203	101751		136	83055		67	18696	

Appendix 1.B. Monthly number of occurrences, total area (ha), and mean (+/-SD) area (ha) burned for all (*n* = 203) large fire ignition types on the study area, 1 April to 30 September, 1986-2000.

	All ignitions			Lightning-caused ignitions			Human-caused ignitions		
Month	No. of occurrences	Total area burned	Mean area burned	No. of occurrences	Total area burned	Mean area burned	No. of occurrences	Total area burned	Mean area burned
April	18	4546	253 (488)	7	2155	308 (500)	11	2391	217 (502)
May	27	16466	610 (1456)	10	5520	552 (1094)	17	10947	644 (1664)
June	57	52587	923 (2691)	33	50305	1524 (3431)	24	2281	95 (142)
July	51	21468	421 (1222)	40	19255	481 (1355)	11	2213	201 (487)
August	28	2304	82 (99)	27	2041	76 (95)	1	263	263
September	22	4380	199 (314)	19	3779	199 (333)	3	601	200 (188)
Mean	34 (16)	16959 (19063)	415 (310)	23 (13)	13843 (19001)	523 (521)	11 (9)	3116 (3945)	270 (191)
Total	203	101751		136	83055		67	18696	

CHAPTER TWO

MULTI-SCALE RESPONSE OF AVIAN COMMUNITIES TO PRESCRIBED FIRE: IMPLICATIONS FOR FUELS MANAGEMENT AND RESTORATION TREATMENTS IN SOUTHWESTERN PONDEROSA PINE FORESTS

INTRODUCTION

Ponderosa pine (*Pinus ponderosa*) forests of the Southwest were historically characterized by frequent, low severity surface fires (Cooper 1960, Covington and Moore 1994), which created significant spatial and structural heterogeneity among stands on the landscape (Allen et al. 2002, Graham et al. 2004). These surface fires maintained a “patchy” tree distribution, varying in density and size, and mediated the occurrence and return interval of larger, stand-replacing fire events (Moir et al. 1997). However, contemporary increases in surface and canopy fuels have moved the structural characteristics of many ponderosa pine forests from their historical range of variability and toward a more homogeneous condition (Fulé et al. 1997, Swetnam et al. 1999, Sesnie and Bailey 2002). Thus, a century of human-induced manipulation of southwestern fire regimes has resulted in ponderosa pine-dominated ecosystems that are vulnerable to extreme fire events (Cooper 1960, Swetnam et al. 1999). The exceptionally large fires that occurred in the Southwest between 2000 and 2002 attest to this condition and many forest ecologists and fire officials agree that steps must be taken to reduce fuel levels and abate wildland fire risk (Allen et al. 2002, Brown et al. 2004). Land managers are interested in reducing this risk by implementing fuel treatments in progressively larger stands of dry, low elevation ponderosa pine forest. However, there is limited knowledge of how to return forests to a more natural condition and, at the same time, maintain biodiversity and ecological integrity across large areas where fire behavior, fuel quantities, and forest structure have been changed by humans (Covington and Moore 1994, Lyon et al. 2000, Noss et al. 2006).

In recent decades, efforts to restore the structure and function of western forests have accelerated, but debate continues about how best to treat forests and minimize ecological degradation (Covington 2000). Forest restoration methods attempt to establish self-sustaining system processes by manipulating succession and mimicking recovery from natural disturbance (Angermeier and Karr 1994, Covington 2003). Restoration projects in the Southwest are removing large amounts of forest biomass using mechanical thinning and fire treatments in an attempt to create forest structure conditions presumed to exist prior to Euro-American settlement of the region (Covington et al. 1997, Friederici 2003). However, the effects of restoration treatments on various ecosystem attributes are not well understood (Wagner et al. 2000, Kotliar et al. 2002) and effects on sensitive ecosystem components, including wildlife, need to be identified before land managers recommend and implement broad-scale restoration programs (Rieman and Clayton 1997, Tiedemann et al. 2000, Block et al. 2001, Allen et al. 2002).

Prescribed fire treatments have been offered as a tool to improve ecosystem health and function (Mutch 1995, Miller and Urban 2000, Kauffman 2004), mitigate wildland fire size and severity (Pollet and Omi 2002, Schoennagel et al. 2004, Finney et al. 2005), and reconcile the goals of fuel treatments and ecological restoration (Allen et al. 2002, Agee and Skinner 2005). When not combined with mechanical thinning operations, these treatments tend to result in low-to-moderate-severity surface fires that reduce ground and ladder fuel quantities, but do little to reduce overstory canopy density (Agee and Skinner 2005). In the Southwest, fire managers typically ignite prescribed fires outside of the summer months, during which natural (i.e., lightning-caused) fire events burn the most area and risk of crown fire is greatest (Swetnam and Betancourt 1990, Dickson et al. 2006 [Chapter One]). Regardless of season, there remain significant uncertainties about the immediate and long-term environmental consequences of reintroducing fire on animal populations and on the habitats that support biological communities (Tiedemann et al. 2000). Because of these uncertainties, it is important to provide land managers with the information and tools needed to understand and mitigate any adverse effects of prescribed fire treatments on both humans and ecosystem processes.

Since the demographic attributes of breeding bird populations are relatively easy to measure compared to other animal taxa, avian communities are often the focus in studies of wildlife response to disturbance, including natural- and human-caused fire. To date, much of the published research on the effects of fire on birds in forested ecosystems has focused on observations of relative abundance and use of habitats after wildland fire (see reviews by Finch et al. 1997, Kotliar et al. 2002, Bock and Block 2005b). Only a small number of these studies were conducted in southwestern ponderosa pine forests. Based on limited study, generalizations about avian response to fire is precluded by differences in fire intensity, severity, and season (Rotenberry et al. 1995, Kotliar et al. 2002, Bock and Block 2005b), as well as methodological and analytical differences in study design (see reviews by Hutto 1995, Ganey et al. 1996, Finch et al. 1997). Indeed, statistical limitations in these studies, coupled with inherent variation in the response of the species and habitats under investigation, have tended to produce results that are often conflicting (Hejl 1994, see van Mantgem et al. 2001). To remedy this situation, wildlife researchers and managers have called for increased rigor in the design and analysis of studies on avian communities and increased precision in the measurement of habitat and wildlife change (Marzluff et al. 2000, Van Horne 2005). Additionally, summaries of the literature have emphasized a need for experiments that examine avian community response to forest treatments, including prescribed fire (e.g., Finch et al. 1997, Kotliar et al. 2002, Bock and Block 2005a).

Few published studies have addressed the effects of prescribed fire on the ecology of breeding bird communities in pine forests of the Southwest and none have examined the impact of prescribed fire using research designs implemented at multiple spatial scales. Typically, studies of prescribed fire are conducted on treatment units too small (e.g., < 100 ha) to measure change in the abundance and distribution of highly vagile taxa, including birds (Block et al. 2001, Short 2003, Dickson et al. 2004). As a result, there is little empirical evidence to evaluate whether avian communities are reliable indicators of ecosystem response to forest management activities that use introduced fire across large areas (e.g., > 100 ha). Furthermore, it is unclear how pre-fire forest structural attributes interact with fire behavior to cause changes in post-fire communities. For example, different avian species may respond differently to varying levels of fire severity because of distinct habitat requirements and because inherent variability in fire severity can result in a mosaic of treatment effects on forest structure.

If the use of prescribed fire has great potential to transform avian communities (e.g., increase species richness and abundance) and modify avian habitats at multiple spatial scales (Rotenberry et al. 1995, Finch et al. 1997, Brawn et al. 2001), then there is a need for research conducted at scales relevant to the ecology, maintenance, and management of these communities. Management policies that address the environmental effects of fuels and restoration treatments at the local or stand level may not consider the unique and cumulative effects of such treatments at broader spatial extents (Agee 1998, Tiedemann et al. 2000, Allen et al. 2002). In response to forest management activities conducted at broader spatial extents, changes in demographic parameters (e.g., density) of wildlife can be difficult to model without considerable effort or without knowledge of the various habitat components that can affect these changes (Marzluff et al. 2000). In this context, research examining avian response to prescribed fire treatments should focus on individual species, or groups of species, whose demographic parameters can be reliably estimated at multiple spatial scales. Moreover, among-scale (e.g., point, patch, unit, site) comparisons may provide additional insight into the appropriate spatial scale at which to focus management and conservation efforts for individual species, groups of species, or entire avian communities.

To assess the response of the avian community to prescribed fire treatments on ponderosa pine-dominated sites in the Southwest, my objectives in this chapter were to: 1) synthesize forest structural and physiographic habitat attributes at two spatial scales, namely the local (area of < 1 ha) and patch (contiguous area of 1–400 ha) scale, to contrast treatment unit- (~ 400 ha) and site-level (co-located treatment and control units) values of multiple response variables; 2) quantify treatment-induced tree damage and levels of fire severity; 3) identify treatment-induced changes in the density and abundance of individual avian species; 4) identify empirically-based assemblages, or "clusters" of avian species that were expected to respond to treatments in a similar fashion based on their ecological and life-history traits; 5) identify treatment-induced changes in the density and abundance of these avian clusters; 6) identify changes in avian community diversity (e.g., species richness, rarity, evenness, and similarity); and 7) model the spatially-explicit response of species and clusters as a function of habitat attributes and levels of fire severity. Specifically, I sought to address the following research questions and associated hypotheses:

1) *Question*: Is there a predictive relationship between levels of prescribed fire severity and forest structure or physiographic habitat attributes measured prior to treatment?

   Research Hypothesis: There is a predictable relationship between pre-fire forest structure and physiographic conditions and treatment-induced tree damage variables that can be quantified by multiple fire severity levels.

   Corollary to Research Hypothesis 1: Knowledge of patch-scale forest and physiographic attributes (e.g., as derived via remotely sensed data prior to treatment) provides additional insights into the response of trees to prescribed fire treatments. This information can inform models of avian response to fuels treatments.

2) *Question*: Are post-fire changes in the density and abundance of individual avian species or clusters (see Question 3 below) of species detectable and can these changes be related to pre-treatment forest structure attributes, physiographic variables, or post-

treatment prescribed fire-induced tree damage at multiple spatial scales (i.e., at the scale of the point, patch, unit, and site)?

Research Hypothesis 1: There is a causal relationship between treatment-induced damage caused to standing forest structure and estimated changes in the density and abundance of individual species or clusters of species. These patterns of change differ between treatment and control units.

Research Hypothesis 2: Avian species respond individualistically to prescribed fire treatments and the magnitude and direction of this response is related to different levels of fire severity.

Research Hypothesis 3: Pre-treatment forest structure and physiographic variables can partially predict the post-treatment response of avian species or cluster density and this response can be separated from any treatment-induced response.

Corollary to Research Hypothesis 3: Knowledge of pre-fire patch-scale forest and physiographic attributes (e.g., as derived via remotely sensed data prior to treatment) provide additional insights into the response of avian species and clusters to prescribed fire treatments.

3) *Question*: Is there a logical and empirical way to group (as a previously unknown assemblage) multiple avian species that are expected to respond to specific fire treatments in a similar fashion?

   Research Hypothesis 1: Groups of avian species can be quantitatively and repeatedly characterized using a cluster-based summarization of ecological and life-history traits.

Corollary to Research Hypothesis 1: A cluster-based evaluation of treatment-induced change will permit generalizations about the response, conservation, and management of avian species that are infrequently detected.

4) *Question*: Will prescribed fire treatments lead to changes in avian community diversity (e.g., species richness, rarity, evenness, and similarity) at different scales of estimation (e.g., at the scale of the point or unit)?

   Research Hypothesis 1: The pattern and extent of changes in avian community diversity differ between treatment and control units.

   Research Hypothesis 2: Treatment-induced changes in avian community diversity are not independent of spatial scale.

METHODS

Study area

To evaluate avian community response to prescribed fire treatments in ponderosa pine forests on the Mogollon Plateau, I established four study sites on four National Forests in northern Arizona and west-central New Mexico (Figure 2.1). In Arizona, three sites were located on the Williams Ranger District of the Kaibab National Forest (KNF), the Mogollom Rim Ranger District of the Coconino National Forest (CNF), and the Lakeside Ranger District of the Apache-Sitgreaves National Forest (ASNF). In New Mexico, a single site was located on the Reserve Ranger District of the Gila National Forest (GNF; Table 2.1). The KNF and GNF sites were established in May of 2002 and the CNF and ASNF sites were established in May of 2003. Overstory vegetation on each site was dominated (relative contribution to canopy cover ≥ 70%) by ponderosa pine. Single trees or small stands of Douglas-fir (*Pseudotsuga menziesii*), pinyon pine (*Pinus edulis*), Utah juniper (*Juniperus osteosperma*), one-seed juniper (*J. monosperma*), Rocky Mountain juniper (*J. scopulorum*), and/or alligator juniper (*J. deppeana*) were common on most sites, but contributed less to overall canopy cover. Gambel oak (*Quercus gambelii*) was common on the CNF, ASNF, and GNF sites. Douglas-fir was common in the higher elevation stands on the GNF site. Grasslands occurring on each site were dominated by bunchgrass species, including Arizona fescue (*Festuca arizonica*) and blue grama (*Bouteloua gracilis*). Mean unit elevation was approximately 2,235 m (SD = 104, range = 2,122–2,412) and topography among sites was variable, with steeper (> 10°) slopes occurring on the GNF site (Table 2.1).

Data collection

Sampling design

In order to establish causal relationships between prescribed fire and changes in the avian community at multiple spatial scales, I pursued my research objectives using a designed experiment. At each study site, I used a Before-After/Control-Impact (BACI; Green 1979) experimental design to assess each of the response variables described below (Figure 2.2). Each site represented a single replicate consisting of a paired prescribed fire "impact" or "treatment" unit and "control" unit and contained between 50 and 90 permanent sampling stations (hereafter, "points") at which all response variables were measured (10–40 points per unit; 134 treatment and 144 control = 278 total points; Table 2.1). Each treatment unit was paired with one (CNF, GNF) or two (ASNF, KNF) nearby (1–5 km) control units with similar biotic components, geography, and climate. Specifically, I used a geographic information system (GIS; ArcView® 3.2a and ArcGIS 8.1; ESRI, Redlands, California, USA) and U.S. Geological Survey (USGS) Digital Orthophoto Quads (DOQs; image acquisition occurred 1997-2000) to locate areas for the placement of suitable control units on the landscape (≤ 100 km^2) that included the treatment unit. Total area encompassed by the control unit(s) on each site was roughly equivalent to the total area of the paired treatment unit (247–405 ha). Prior to field work, points were located using methods implemented in a GIS (ArcGIS® v8.3-9.1). I used the GIS to overlay a lattice of 125 × 125-m cells that encompassed each site and randomized the placement of the first cell location. Within the GIS, I systematically placed prospective points at lattice intersections (nodes) when that placement met the following criteria: (1) nodes were ≥ 150 m from the unit edge; (2) nodes could be located where stand canopy could be stratified according to average percent cover within a 100-m radius of a point center; and (3) a point could be placed ≥ 250 m from its nearest neighboring point. The proportion of points falling within each canopy cover stratum was roughly proportional to the overall representation of each cover stratum on a site. Once points and their corresponding Universal Transverse Mercator (UTM) coordinates were identified in the GIS (zone 12, North American 1983 datum), I used a global positioning system to locate points on the ground at each site and improved positional accuracy by differentially correcting the coordinates for each point using a local base station (implemented in GPS Pathfinder® Office, V3.0, Trimble Navigation Ltd., Sunnyvale, CA, USA).

Prescribed fire treatments

Between September 2003 and May 2004, prescribed fire treatments were implemented by U.S. Forest Service (USFS) District personnel at each of the four study sites (Table 2.2). Because of the large scope and scale of these treatments, it was not possible to randomize the location of each treatment unit, although efforts were made to insure that all sampling was conducted on randomly-placed stations nested within each unit (see below). As outlined in site-specific burn plans, common objectives for each treatment included wildfire threat and hazardous fuels reduction, the maintenance of natural fuelbreaks, and the restoration of natural ecological systems. Desired treatment effects included a 20-90% reduction in trees with a diameter at breast height (dbh) ≤ 8 cm and a 20-50% reduction in trees ≤ 23 cm dbh. Desired environmental conditions included temperatures between -12 and 27° C, relative humidity between 10 and 60%, live fuel moistures between 3 and 35%, and wind speeds ≤ 29 km/h. At each site, fire was applied to a single treatment unit between 247 and 405 ha in extent. Fire personnel used ground-based ignition methods, including head, backing, and/or flanking fires at treatment unit edges and strip-pattern and spot fires at the unit interior. All prescriptions were characterized as broadcast burns with expected fire behaviors of low to moderate intensity. Ignitions were suspended when weather conditions were suboptimal or when average flame lengths exceeded approximately 2 m.

Two treatments were completed in the fall of 2003 (ASNF, CNF) and two in the spring of 2004 (GNF, KNF; Table 2.2). Treatment windows ranged from five days (CNF) to five months (GNF, KNF) and area burned during each ignition day was variable (mean = 67.8 ha, SD = 63.8). In general, a given area was burned only once and during a single ignition day. Although fire personnel attempted to burn all or most of the area encompassed within a prescribed treatment unit, surface fuel characteristics, including continuity and arrangement, and physiographic barriers resulted in incomplete coverage and consumption. Subsequently, fire behavior patterns were spatially heterogeneous and resulted in a mosaic of treatment effects on forest structure. These observed patterns, however, were typical for larger-scale prescribed fire treatments in southwestern ponderosa pine (L. Wadleigh, USFS Region 3 Fire Ecologist, personal communication). In general, among sites, treated areas with substantial amounts of ponderosa pine litter were completely burned and areas composed mostly of grass or oak litter burned incompletely or did not burn (M. Cothrun, P. Delgado, J. Jerman, and V. Morfin, USFS District Fire Ecologists, personal communication; personal observation). Typically, smaller (10 hr and 100 hr) fuels were consumed, although 1,000 hr fuels and large (> 30 cm in diameter) logs were only partially (< 50%) consumed (see Saab et al. In Review).

On all sites, some torching of ponderosa pine in the smaller diameter classes (e.g., ≤ 8 cm dbh) did occur at the interior of each unit, but mortality was less than expected. Torching of trees in larger diameter classes resulted when burning logs or patches of woody debris generated sufficient heat at the tree bole. Isolated crown fire occurred on each site, but this behavior was less frequent on the ASNF and CNF sites. Ocular estimates of the immediate effects of each treatment suggested that fire behavior was greatest on the GNF site, followed by the KNF, ASNF, and CNF sites (see Table 2.2).

Local-scale habitat attributes

I summarized local-scale, or "point-level" standing forest structure using ground-based sampling centered on each of the 278 points. To characterize these habitat attributes prior to prescribed fire treatments, I placed a 50-m transect line in each of the four cardinal directions and measured the dbh or root-crown diameter (*Juniperus* species) of all live tree species encountered within 5 m of either side of the transect line. To expedite the measurement of smaller (dbh 2.5–23 cm; after Saab et al. In Review) trees, I assigned all stems occurring within 2 m of the transect line to one of six diameter classes. At each point, I estimated total tree density (stems/ha) and basal area (m^2/ha) by combining data for larger (dbh ≥ 23 cm) trees measured within 5 m of each transect line (total sampling area = 0.20 ha) with data for smaller trees measured within 2 m of each transect line (total sampling area = 0.08 ha). For all large live trees measured within the 0.20-ha sampling area, I also estimated mean and variance of tree height from individual measurements made using a clinometer (Suunto Oy, Vantaa, Finland), tree species richness, and the proportion of ponderosa pine stems. I used a digital camera and imaging software (Gap

Light Analyzer V2.0) to derive overstory canopy cover values at each point from true-color fisheye photographs (Frazer et al. 1999).

Because large snags were infrequently detected during a pilot effort, I counted the total number of large (dbh ≥ 23 cm) snags occurring within 10 m of either side of the transect line and summed these values to estimate the relative number of large snags occurring at each point (total sampling area = 0.40 ha). To quantify snag decay at each point, I first placed each snag into one of four condition classes (after Bull et al. 1997): "1" recently died, had little decay, and retained most of its branches, bark, and top; "2" evidence of decay and was missing some bark, branches, or a portion of the top; "3" extensive decay, bark missing from most or all of the branches, and a broken top; and "4" burnt snag with an outer shell hardened by fire. As a final measure of relative snag condition and decay at each point, I computed a snag decay index by dividing the total number of snags present by the sum of the decay class values recorded for each snag (scaled between 0 and 1, with lower values indicating greater total decay).

Patch-scale habitat attributes

Since animal use of habitat is a multi-scale process (Wiens 1989, Schneider 1994), I investigated the relationship between the avian community and habitat factors beyond the scale of the point (i.e., within a 50-m radius), namely at the patch and landscape (e.g., unit and site; see *Avian species density and abundance* below) scales. I considered a patch to be a contiguous area (~ 1-400 ha) surrounding a point location and defined by a vector of forest attribute and physiographic information. To quantify forest habitat characteristics at the patch scale, I used a GIS to derive and map five important physiographic and forest structure attributes on each of the four study sites. I used the

Spatial Analyst (ESRI, Redlands, CA, USA) extension to ArcGIS® to derive elevation and slope based on a mosaic of USGS digital elevation models (DEMs). Mean values for elevation and slope were summarized using a neighborhood function (focal mean) and a circular moving window with a 100-m radius. In cooperation with the Forest Ecosystem Restoration Analysis (ForestERA; Sisk et al. 2006) project, I derived three digital maps, or layers, of forest structure using GIS and remote sensing technologies, including canopy cover (0-100%), basal area (m^2/ha), and tree density (individual trees/hectare). I used percent canopy cover as a measure of the total amount of the landscape covered by overstory (tree) canopy foliage. The canopy cover layer was developed from a mosaic of DOQs (image acquisition occurred 1997-2000) and using a new type of advanced exploratory data analysis that relied on empirical information for model training and validation (see Xu et al. 2006). For all trees > 2.54 cm dbh, I used basal area as an index of the total cross-sectional area and tree density as a count of the total number of trees. Using a regression tree methodology and Cubist software (RuleQuest Research Ltd., St. Ives, New South Wales, Australia), the basal area and stem density layers were developed from Landsat 7 Enhanced Thematic Mapper imagery (all scenes acquired between 1999–2000), a Normalized Derived Vegetation Index layer, DEMs, the canopy cover layer described above, and training data from over 600 ground plot locations in northern Arizona and west-central New Mexico (Hampton et al. 2003). Each of the above maps was derived at a 30-m resolution, represented a spatially continuous raster surface in the GIS that encompassed the full extent of each study site, and was projected in UTM coordinates (zone 12, North American 1983 datum) to insure spatial congruence. Each map was then used as one or more habitat covariates included in the statistical models described below.

To identify discrete patch types on each study site, I synthesized the three forest structure attributes using a fuzzy classification algorithm (see Bezdek 1987) implemented with the BoundarySeer® software package (TerraSeer, Inc., Crystal Lake, IL, USA). BoundarySeer® permits the detection, description, and analysis of geographic boundaries in a spatially-explicit, multivariate framework, but has seen only limited application in wildlife ecology (e.g., Hall and Maruca 2001). To identify discrete patches of forest when patch boundaries are necessarily indiscriminate, a fuzzy classification method can be better than traditional, or "hard," classification methods (Brown 1998a and 1998b, Longley et al. 2001). Since the membership of each forest structure attribute within a particular forest patch type is not likely a binary (yes/no) one, fuzzy classification permits the assignment of each attribute to a patch type based on its probabilistic membership. This approach also captures both the spatial and attribute uncertainty surrounding the membership of a given value in any single patch type. For each cell occurring within a unit and a 100-m buffer (to account for edge influences on the clustering algorithm), I identified the value for each of the three forest structure attributes and combined this vector of continuous information as a new, vector-based point data file for analysis in BoundarySeer®. To aid in a more parsimonious interpretation of patch description and patch influence on response variables, I grouped the forest structure data into three classes, or "types." Using output from BoundarySeer®, I used Boolean operators to sort membership probabilities for each attribute into their respective patch type and created a new, categorical raster surface within the GIS. I then implemented a neighborhood function (focal majority) in Spatial Analyst to compute which patch type occurred most frequently within a 100-m radius of each cell and assigned this value to the focal, or center, cell (new covariate = "PatchType"). This new raster surface characterized the spatially-explicit occurrence of patches across each of the units. I computed summary statistics to describe the relative contribution of the original values for each forest structure attribute to each patch type. To quantify patch heterogeneity, or "richness," and the number of different patch types occurring within a 100-m radius of each cell, I used an additional neighborhood function (focal variety) and created a new raster surface (new covariate = "PatchRich"). I associated each survey point with the cell value of the PatchType and PatchRich covariates using an intersect operation in the GIS. I treated the PatchType and PatchRich covariates as ordinal variables in the habitat models described below.

Prescribed fire effects sampling

To assess the effects of prescribed fire on standing forest structure, I used methods consistent with those described by McHugh and Kolb (2003). In collaboration with researchers from Northern Arizona University and the USFS Rocky Mountain Research Station in Flagstaff, Arizona, I collected information on tree damage for all prescribed fire-affected trees occurring within a single, 10-m radius circular plot (0.03 ha) centered on a point. Tree damage measurements included bole char severity, average maximum bole scorch height, average percent of the bole circumference charred, and average proportional crown scorch. Bole char severity was categorically determined by identifying the position of the bole with the maximum height of charcoal-like bark and then examining the lowest 0.5 m of this position. "Light" char was indicated by only light charring on edges of bark plates. “Moderate” char was indicated by a uniformly black characteristic to the bark plates, except for the innermost depths of prominent fissures. “Heavy” char was indicated when bark plates were deeply charred and their characteristics were not discernable. At each point, I assigned the maximum bole char severity level detected among all fire-damaged trees. I used a clinometer to estimate maximum scorch height at the position on the bole with the maximum height of charcoal-like bark. For each fire-damaged tree, I also estimated the percent of the bole circumference charred at the root crown. To quantify crown scorch and foliage consumption, I used the estimated height of the pre-fire live crown to estimate the total proportion of moderately scorched foliage (brown color) and proportion of foliage that was consumed by the fire.

Avian community sampling

I assessed avian community composition and species response to prescribed fire using a sampling design that permitted density and estimation by distance-based methods (Buckland et al. 2001). During the breeding season (May–June, 2002–2005), a team of trained observers counted birds at each site using point-transect distance sampling and a modified spot-mapping methodology. Observers received intensive training over a two-week period before formal sampling efforts. Briefly, sampling methods entailed conducting a 100-m fixed-radius count at each point and recording the spatial location of all species detected by sight or sound within five pre-defined distance bins (10, 25, 50, 75, 100 m) within a 5-minute period. Observers used laser range finders to improve the precision of their distance estimates and a compass to estimate the bearing to each detection. Individual birds detected > 100 m from a point, flying above the canopy, or between sampling efforts (i.e., “incidentals”) were also recorded. Individual observers randomly visited eight to ten points per unit per day and sampled each point three to four times each season.

Avian trait data

I compiled an ecological and life-history trait data matrix for all species detected on ≥ 1 occasion during formal distance sampling efforts (see Appendices 2.A and 2.B). Prior to statistical analysis, I used nine traits that characterized a suite of ecological and life-history attributes to discriminate between unique groupings, or clusters, of species. These attributes included average mass (AM), migration status (MS), breeding season diet (DI), foraging substrate (FS), nest type (NT), minimum clutch size (MC), clutch breadth (CB), maximum potential lifetime reproductive output (LRO), and minimum incubation + nestling period (IN).

To compute the AM (in grams) for each species, I used Dunning (1993) and, where possible, averaged the combined weights of female and male birds. When provided, I used mass estimates from the region that included my study sites. For MS, I assigned each species one of three migration status codes based on the classifications of Ehrlich et al. (1988; primary source), Hall et al. (1997), and Poole (2005). I considered residents (status code = R) to be year-round occupants of the region that included the study sites, including those species with only very localized or elevational movements. I classified short-distance migrants (S) as those species whose northern Arizona and New Mexico breeding season populations wintered off of breeding range or habitats, but not typically south of Mexico. I considered long-distance migrants (L) those species that wintered in or south of Mexican. I identified the primary DI for each species using

Ehrlich et al. (1988). Diet type was classified as insect, seed, nut, nectar, bird, mammal, carrion, or omnivore. I classified the primary FS used by each species as air, bark, foliage, or ground according to Ehrlich et al. (1988). I characterized NT for each species as cavity, crevice, crack, cup, saucer, platform, ground, or none using the descriptions of Ehrlich et al. (1988). To derive values for MC, CB, and IN for each species, I reviewed estimates provided by Ehrlich et al. (1988; primary source) and Poole (2005). I calculated MC (# of eggs) as the minimum clutch size and CB (# of eggs) as the range between the minimum and maximum clutch size. I computed values for IN (days) by summing values obtained for minimum incubation time and minimum nestling period. For each species, I computed an index value for LRO (# of eggs) as the product of the maximum clutch, maximum annual number of broods, and maximum longevity. I obtained maximum values for clutch and annual number of broods from Ehrlich et al. (1988) and Poole (2005) and maximum longevity estimates from the USGS Patuxent Wildlife Research Center (database online at www.pwrc.usgs.gov/bbl/homepage/longvrec.htm).

Data analyses

Local-scale habitat

I used variable reduction methods to quantify the multi-dimensional structure of the local-scale forest attribute data and to more parsimoniously fit regression models with uncorrelated predictor variables. I synthesized these data using a factor analysis and extraction by principal components analysis (PCA) to obtain the initial factor solution (Manly 2005). From the resulting variable correlation matrix, I retained only those habitat factors with component score eigenvalues > 1.0 for inference. To simplify interpretation of the variance associated with each retained factor, I used varimax rotation with Kaiser normalization to compute the sums of squared loadings (Manly 2005). For each retained factor, I used the final, or "extracted" communality estimates to describe relative variable importance and considered estimates of 0.50–0.70 and ≥ 0.70 of moderate and most importance, respectively (see McGarigal et al. 2000). This communality estimate is simply the proportion of the variance explained by each local-scale forest structure attribute given the final number of retained factors. For each retained factor, I used the rotated factor loadings to interpret the relative importance of individual variables and used values ≥ |0.50| as a cutoff (Manly 2005). Because the rotation results in the relation of each variable to only a single factor, I used this step to assign ecological meaning to each of the retained habitat factors. For these analyses, I pooled data from all ($n = 278$) points and exported regression scores for the retained habitat factors as independent, local-scale habitat covariates in my point-level habitat models (see below). All analyses were performed using SPSS (V14.0, SPSS Inc., Chicago, IL, USA).

Prescribed fire severity

Zimmerman (2003) defined fire severity as the immediate effects of fire amount, rate, and direction of heat transfer on vegetation, including bole char severity, crown scorch height, and crown consumption. Fire severity can differentially influence the maintenance and arrangement of forest stands (Chappell and Agee 1996, Brown et al. 1999, Knapp and Keeley 2006) and the numerical response of wildlife communities, including birds (Smucker et al. 2005). To synthesize the multivariate influence of the prescribed fire treatments on these ecosystem components, I developed a fire severity covariate using the five tree damage variables described above. In my analysis, I considered only those points where fire effects were observed and did not include points located on control units. Since one (bole char severity) of the five tree damage variables was categorical, I used a two-step cluster analysis algorithm (Zhang et al. 1996, Chiu et al. 2001; implemented in SPSS) capable of accommodating mixed (categorical and continuous) variable types. For the fire severity data matrix, with variables in the columns and observations at each survey point in the rows, the algorithm consisted of two basic steps: 1) a pre-clustering of records into many small sub-clusters and 2) a clustering of the sub-clusters resulting from the pre-cluster step into a user-defined number of clusters. Both steps use a log-likelihood, probability-based distance measure to maximize the distance between cluster centers. For each step, the distance, *d*, between two clusters *Cx* and *Cz* can be defined as

$$d(x,z) = \xi_x + \xi_z - \xi_{<x,z>}, \tag{1}$$

where <*x*,*z*> is an index value that represents the new cluster formed by combining clusters x and z, and ξ is the log-likelihood. For $v = x$, z, and $\xi_{<x,z>}$

$$\xi_v = -N_v\left(\sum_{u=1}^{U^A}\frac{1}{2}\log(\hat{\sigma}_u^2 + \hat{\sigma}_{vu}^2) + \sum_{u=1}^{U^B}\hat{E}_{vu}\right), \tag{2}$$

where N_v is the number of data records in cluster v, U^A is the total number of continuous variables in the procedure, U^B is the total number of categorical variables in the procedure, $\hat{\sigma}_u^2$ is the estimated variance of the u-th continuous variable for all of the data, $\hat{\sigma}_{vu}^2$ is the estimated variance of the u-th continuous variable in cluster v,

$$\hat{E}_{vu} = -\sum_{w=}^{W_u}\frac{N_{vuw}}{N_v}\log\frac{N_{vuw}}{N_v}, \tag{3}$$

and where W_u is the number of categories for the u-th categorical variable and N_{vuw} is the number of data records in cluster v whose u-th categorical variable takes the w-th category. Using the log-likelihood distance measure, the pre-clustering step reduces the size of the original data matrix by considering distances between all possible pairs of records during a single pass of the data. When a record is read by the algorithm, it uses the decrease in the log-likelihood distance measure to determine whether a record is to be combined with a previously formed precluster, or to initiate a new precluster. This step is also referred to as the construction of a modified cluster feature (CF) tree (Zhang et al. 1996), where each CF tree consists of "nodes" and "leaves," representing a collection of records and a subcluster, respectively. Once this step is complete, all records contained in a precluster are considered as a single subcluster. The second step integrates the more common method of agglomerative hierarchical clustering (see Legendre and Legendre 1998), an iterative procedure that updates the distance between a new cluster and those remaining ($<x, z>$ term in equation (1) above is replaced for either x or z) until a minimum number of clusters is achieved. This step groups all of the subclusters into the desired number of clusters, for which I defined three clusters, or "levels" of fire severity response at each point: low, moderate, and high. However, a novel feature of the two-step algorithm is a procedure that automatically determines the "appropriate" number of clusters using an information criterion, such as Akaike's Information Criterion (AIC), in the process of cluster model selection, as well as estimated changes in the likelihood-based distance measures (see *Avian trait and cluster analysis* below).

I determined the relative contribution and statistical significance of each tree damage variable in each of the three clusters using a Student's-t (for continuous variables) or chi-square goodness-of-fit (categorical variables) test. Within cluster variable importance was ranked according to the value of the t (absolute) or chi-square statistic. For each variable within a cluster, I assessed statistical significance with respect to its mean value among clusters using $\alpha = 0.05$ and implemented all analyses in SPSS.

To explore and quantify the relationship between the fire-severity level indicated by the cluster analysis above and habitat covariates, I used a mixed-model repeated-measures analysis of variance (ANOVA; see general model form details below; see Appendix 2.C for all mixed model response and predictor variable definitions). I pooled points among treatment units ($n = 134$), including those points where prescribed fire-caused tree damage was not observed, and treated site as a random effect. I treated fire severity as an ordinal response (dependent variable) with four levels, including "unburned." To account for spatial variability and correlation in the response values within the four sites, I also modeled the spatial covariance structure and used an empirical semivariogram and AICc (AIC corrected for small sample size) values to determine the "best" covariance model and values for the parameters (Littell et al. 1996). I used restricted maximum likelihood (REML) for covariance parameter estimation and maximum likelihood (ML) for fixed effect parameter estimation and to compute AICc values (see Wolfinger 1993). Because I was interested in the relative importance of all habitat covariates in predicting levels of fire severity, I compared a fully-parameterized (global) spatial model and an intercept-only model. I computed Akaike weights of evidence (w_i) and ranked relative covariate importance by summing the Akaike weights for all possible combinations of models within which a given covariate occurred (see equation 5 below; Burnham and Anderson 2002). I used a paired *t*-test and the Tukey-Kramer multiple comparison test to assess all categorical and pairwise differences (α =

0.05). All statistical analyses were conducted using the MIXED procedure implemented in SAS (V9.1, SAS Institute, Cary, NC, USA).

Avian species density and abundance

I used program DISTANCE (V5.0; Thomas et al. 2005) and multi-model inference to estimate avian species density ($\hat{D}$) and abundance ($\hat{N}$), and to control for differences in their estimated detection probability, $\hat{p}$, among treatments over time (Buckland et al. 2001). Because of the sample size constraint on estimation discussed below, I focused my DISTANCE-based analyses on those species with > 500 detections across all years. I considered this group of species to be my "inference set" for two separate levels of estimation (see below). Within DISTANCE, I modeled $\hat{p}$ for individual species as a function of multiple factor covariates that likely influenced the detectability of a given species, including site ($n = 4$ factors), year ($n = 3$-4), and observer rank ($n = 3$). To quantify observer rank, I ranked each observer on a scale of one to three by examining detection distance histograms and using an evaluation of their performance provided by field supervisors. This multi-covariate approach is more robust to stratification and small sample sizes than conventional distance sampling analysis (Marques and Buckland 2004). Although I initially considered additional covariates describing habitat and treatment features (e.g., stem density, percent canopy cover, and burned/unburned), preliminary models that included these covariates had little or no support. Because I used multiple covariate distance sampling and desired a more parsimonious modeling approach, I considered only the half-normal key function with a cosine series expansion (≤ 2 adjustment terms) and modeled a global detection function for each species (Buckland et al. 2001, Buckland et al. 2004). For each species, I considered all possible combinations of the three covariates, plus a model with no covariate in my candidate model set to be used for inference (eight possible models and 31 possible factor combinations). To address model selection uncertainty in this candidate set (*R*) of models, I used AIC to select the best model and averaged competing model (g_i) estimates when the difference (Δ_i) in their AIC values was within two of the minimum AIC value using the equation

$$\hat{\bar{\theta}} = \sum_{i=1}^{R} w_i \hat{\theta}_i, \qquad (4)$$

where $\hat{\bar{\theta}}$ is the model-averaged estimate of predicted density ($\hat{\theta}_i$) and w_i is the Akaike weight of each *i*-th model, for which

$$w_i = \frac{\exp(-\frac{1}{2}\Delta_i)}{\sum_{i=1}^{R} \exp(-\frac{1}{2}\Delta_i)} \qquad (5)$$

and $\sum w_i = 1.0$ (Burnham and Anderson 2002). After Burnham and Anderson (2002), I computed the unconditional standard error for all model averaged estimates as

$$se(\hat{\bar{\theta}}) = \sum_{i=1}^{R} w_i \sqrt{\hat{\text{var}}(\hat{\theta}_i \mid g_i) + (\hat{\theta}_i - \hat{\bar{\theta}})^2}\,. \qquad (6)$$

For each species, I used DISTANCE to estimate density at two levels: "stratum" and "point." Stratum were defined by combining data on the unburned treatment (U) or control (C1) units in pre-treatment years (2002, 2003) and the burned treatment (B) or control (C2) units in post-treatment years (2004, 2005; see Figure 2.2 for design schematic). I generated density estimates at the stratum level by pooling together detections for the pre-treatment period (years 2002, 2003) on the unburned treatment (stratum-level estimate $= \hat{D}_U$) and control ($\hat{D}_{C1}$) units and separately for the post-treatment period (2004, 2005) on the burned treatment ($\hat{D}_B$) and control ($\hat{D}_{C2}$) units (for four sites, $n = 16$ stratum-level estimates). For inference at the site level, I then computed the difference in estimated pre-treatment density as

$$(\hat{D}_{PRE}) = (\hat{D}_U - \hat{D}_{C1}) \qquad (7)$$

and the difference in estimated post-treatment density as

$$(\hat{D}_{POST}) = (\hat{D}_B - \hat{D}_{C2}) . \qquad (8)$$

Finally, for each of the four study sites, s, I computed the among stratum BACI difference, or effect size as

$$\hat{\Delta}_s = (\hat{D}_{POST} - \hat{D}_{PRE}) . \qquad (9)$$

At the stratum level, and for species with sufficient sample size (typically, $n > 20$ detections at each factor combination level produced reliable results), estimates for density, abundance, and variance were generated using 1,000 bootstrap samples (within strata). For species with insufficient sample size at the stratum level, and to best approximate variance estimates prior to statistical analyses, I used the delta method (see Buckland et al. 2001 and Seber 2002) modified to accommodate the four detection probabilities estimated using the multiple covariates:

$$Var(\hat{\Delta}_s) = \left[\sum_{h=1}^{4} \left(c_h \hat{D}_h cv(n_h) \right)^2 \right] + \sum_{h=1}^{4} \sum_{l=1}^{4} (c_h c_l)(\hat{D}_h \hat{D}_l) \left(\frac{Cov(\hat{p}_h , \hat{p}_l)}{\hat{p}_h \hat{p}_l} \right) \qquad (10)$$

where c_h and c_l are constants with signs that depend on the comparisons being made with the $\hat{D}_h$ and $\hat{D}_l$, respectively, and where $\hat{D}_h$ and $\hat{D}_l$ are the estimated densities for two of four stratum levels, *h* and *l*. For the pooled number of detections, *n*, in each stratum level, *h*, I computed

$$cv(n_h) = \frac{\sqrt{\frac{T\sum_{h=1}^{k} t_h (n_h / t_h - n/T)^2}{k-1}}}{n} \tag{11}$$

where *k* is the total number of points within a unit, t_h is the total number of visits to a stratum, $T = \sum_{h=1}^{k} t_h$, and $n = \sum n_h$ (Buckland et al. 2001). Making the more conservative assumption that the correlation among any pair $\hat{p}_h, \hat{p}_l = 1$, I considered $\left(\hat{se}(\hat{p}_h)\hat{se}(\hat{p}_l)\right)$ to be a reasonable approximation of the covariance among points and substituted this product for $Cov(\hat{p}_h, \hat{p}_l)$. On each site, I used a *Z*-statistic to test the null hypothesis H_0: $D_{POST} = D_{PRE}$, where

$$Z_s = \frac{\hat{\Delta}_s - (D_{POST} - D_{PRE})}{\sqrt{Var(\hat{\Delta}_s)}} \sim N(0,1) \tag{12}$$

($\alpha = 0.05$; Buckland et al. 2001).

When DISTANCE provided reliable estimates for individual species at the stratum level, I generated estimates for density, abundance, and variance at the point level by again pooling all detections for a given species over pre- or post-treatment years (n = 556 total point-level estimates). Because point-level sample sizes were insufficient for generating estimates using a bootstrap routine, I modeled the variance-covariance structure of the point level density estimates to identify and account for correlation and interdependencies among samples (see below). For each point, *j*, where $j = 1, 2, \ldots, n$, and *n* is the 278 points, I computed the difference in pre- and post-treatment density as

$$\hat{\Delta}_j^* = (\hat{D}_B - \hat{D}_U) \text{ or } (\hat{D}_{C2} - \hat{D}_{C1}). \tag{13}$$

These point-level difference estimates were used as the response variable in the individual species habitat and fire response models described below.

Avian species habitat and fire-response modeling

To examine the fine-scale response of individual species, I modeled the temporal and spatial relationship between $\hat{\Delta}^*_j$ and habitat covariates at the point level. Covariates included: mean elevation, mean slope, PatchType, PatchRich, and the final number of retained local-scale habitat factors. I also included a "treatment" covariate to indicate the control ("0") or treatment unit ("1"). Separately, I modeled the response of individual species to the four ordinal levels of fire severity when these species exhibited a response to the treatment unit covariate (see below). I evaluated three possible functional forms of this response using a simple linear model of the ordinal levels and two transformations: log-normal and second-degree polynomial. For all models, I used mixed-model repeated-measures ANOVA (implemented using PROCMIXED) and models of the general form

$$\mathbf{y} = \mathbf{X\beta} + \mathbf{Zu} + \mathbf{e} \qquad (14)$$

where $\mathbf{y}$ is a vector of observed response values (e.g., fire severity level or $\hat{\Delta}^*_j$), $\mathbf{X}$ is a matrix of independent covariate values (the known fixed effects), $\boldsymbol{\beta}$ is a vector of regression parameters (the unknown fixed effects), $\mathbf{Z}$ is a design matrix for the known random effects (e.g., site), $\mathbf{u}$ is the vector of unknown random-effect parameters, and $\mathbf{e}$ is the vector of errors (see Littell et al. 1996). Since avian use of habitat and response to disturbance might be spatially dependent, I also modeled the unit-level covariance structure (i.e., positive spatial correlation among points in a given unit). Importantly, this structure permitted me to account for any among-point interdependencies that existed when using a shared $\hat{p}$ (i.e., one estimated using a global detection function) among points at the stratum level. In this case, **e** is assumed to be a vector of correlated experimental errors and its variance is modeled as

$$Var(\mathbf{e}) = \sigma^2 \mathbf{F} \tag{15}$$

where σ^2 is the residual variance among observations, **F** is a covariance matrix with its ab^{th} element defined by $f(d_{ab})$, and where f is a function relating the response variable to a vector of covariates with distance, d, between observations a and b (a and $b = 1, 2, \ldots, n$, and n is the 278 points; see Littell et al. 1996). I used REML for covariance parameter estimation and ML for fixed effect parameter estimation and to compute AICc values.

When I observed heteroscedasticity in my residual plots, or presumed higher levels of measurement variation in estimates of $\hat{\Delta}^*_j$, I weighted my regression models using the inverse of the point-level variance estimates generated by DISTANCE:

$$Var(\hat{\Delta}^*_j) = \frac{1}{\sqrt{cv(\hat{D}_a)^2 + cv(\hat{D}_b)^2}} \tag{16}$$

For each species in my inference set, I used a multi-model approach to evaluate and predict the relationship between $\hat{\Delta}^*_j$ and habitat covariates. I used AICc values and Akaike weights (w_i; see equation 5 above) to assess model fit and estimated and ranked relative covariate importance by summing the Akaike weights across all ($n = 256$) possible models in which a given covariate occurred (Burnham and Anderson 2002). I considered cumulative Akaike weights (w_+) ≥ 0.50 to be strong evidence for a species response to the treatment covariate and constructed fire-response models only for those species with these evidence values (Barbieri and Berger 2004). I relied on AICc values to identify the best functional form of the response of each species to levels of fire severity. I used the Tukey-Kramer multiple comparison to test for differences among categories (i.e., levels) of fire severity ($\alpha = 0.05$). I standardized all continuous habitat covariates and UTM coordinates to mean zero and unit variance prior to statistical analysis (Legendre and Legendre 1998) and used Variance Inflation Factors (VIF) to diagnose collinearity among covariates (eliminating covariates with VIF values > 10.0; Neter et al. 1990).

Avian trait and cluster analysis

Since density could only be estimated for a small number of individual species, I used a novel and repeatable approach to quantify species membership within a previously unknown number of assemblages. Considering the nine ecological and life-history trait variables described above, I used a two step cluster analysis to identify meaningful assemblages, or clusters, of avian species that I expected to respond to prescribed fire in a similar fashion (Szaro 1986). Members within these clusters were empirically derived from the list of species detected using distance sampling among each of the four study sites and included all species detected on ≥ 1 occasion within 100 m of a point. Following the two step analytical procedure outlined above (see equations 1-3), I used an additional feature of the algorithm to automatically determine the most parsimonious number of clusters contained in my species list (Chiu et al. 2001). As a first step in the procedure, the algorithm computed the AIC value for each subcluster to determine a rough estimate of the number of clusters in the data. The cluster model with the smallest AIC value was considered best (Burnham and Anderson 2002). In these analyses, the AIC value for C clusters is computed as

$$AIC(C) = -2\sum_{c=1}^{C} \xi_c + 2m_c, \qquad (17)$$

where

$$m_c = C\{2U^A + \sum_{u=1}^{U^B} (W_u - 1)\} \qquad (18)$$

and defined by equations 2 and 3 (see Burnham and Anderson 2002). Typically, AIC values begin to decrease and then increase with an increase in the number of subclusters. Alone, however, this step tends to overestimate the overall number of clusters in the data (Chiu et al. 2001). Thus, as a second step in the procedure, distances derived during the hierarchical clustering of subclusters are used to compute a distance ratio change, where large increases in the ratio change indicate two subclusters should not be merged. To arrive at the most parsimonious number of clusters, this second step relies on a rough estimate of the maximum number of possible clusters obtained in the first step. I used the distance ratio change to determine this final number of clusters. I determined the statistical significance and relative contribution of each trait variable in a cluster using Student's-*t* and chi-square goodness-of-fit tests ($\alpha = 0.05$) and used values of the *t* (absolute) and chi-square statistics to rank variable importance.

For each cluster identified using the two step procedure, I generated stratum- and point-level density, abundance, and variance estimates using program DISTANCE and the multi-covariate approach outlined above. On each site, I again estimated an effect size and used a *Z*-statistic to test the null hypothesis of no difference between estimated pre- and post-treatment density differences at the stratum ($\hat{\Delta}_s$) and point ($\hat{\Delta}^*_j$) levels. I also constructed models of individual cluster response to habitat and fire severity using the approach described above and the point-level difference estimates ($\hat{\Delta}^*_j$) as the response variable.

Avian community composition and diversity

I assessed the response of the avian community to prescribed fire using a suite of diversity metrics that estimated species richness, rarity, evenness, and similarity at the stratum and point levels. Because I expected prescribed fire to increase forest structural heterogeneity and modify habitats within and among the treatment units, these different metrics allowed me to investigate whether treatments affected the full compliment of species I detected at multiple spatial scales. At the stratum level, I estimated total species richness using the Chao (Chao 1987) and second-order jackknife (Burnham and Overton 1978, 1979, Smith and van Belle 1984) estimators. I used the abundance-based Chao estimator as a measure of the absolute minimum number of species in an assemblage and the incidence-based jackknife estimator because it was developed from underlying models of detection probability (Magurran 2004). Additionally, both of these nonparametric estimators performed exceptionally well in recent evaluations of richness estimators (see Magurran 2004, Walther and Moore 2005). To estimate species rarity, or the approximate number of species represented by one individual, I computed Fisher's α diversity index (Hayek and Buzas 1996). To characterize overall community diversity, I calculated a value for the Simpson (inverse) diversity index, which rises as a community becomes more "even" in its membership, but tends to place more weight on the most common species (Magurran 2004). To estimate within unit similarity and heterogeneity, I computed a mean value for Chao's abundance-based Jaccard similarity index (corrected for unseen species), which approaches one as the average species composition among points becomes identical (Chao et al. 2005). This index explicitly considers the probability that two individuals, one each from two randomly selected points, belong to shared vs. unshared species. The index is particularly useful in assessing compositional similarity between samples that may differ in size, sampling effort, or the number of rare species (Chao et al. 2005).

To estimate the mean value of each diversity metric at the stratum level, I treated the point as the sample unit and included all detections recorded ≤ 100 m during distance sampling and during all visits within a given year. I computed variance estimates for each metric using 1,000 bootstrap randomizations and randomized samples with replacement, or using analytical methods. All estimates were generated using the software program EstimateS (V7.5; Colwell 2005). For each unit in each year (pooling the two control units on the ASNF and KNF sites; $n = 28$), I evaluated the potential influence of year, unit (control or treatment), area (in hectares), effort, observer, and site effects on the estimated value of each metric using a mixed-model ANOVA and cumulative Akaike weights (w_{+}) computed from all possible models. To synthesize the expected interaction between effort and observer, I computed the standardized product of effort and the sum of observer rank values for all detections (new covariate = Effort × ObsRank). Because values for total number of detections and Effort × ObsRank were highly correlated (Pearson's correlation coefficient = 0.69), I considered only Effort × ObsRank in the ANOVA. I computed the difference ($\hat{\Delta}_s$) in the average pre- and post-treatment stratum-level difference for each estimator using equation (9) above. For inference, I considered this difference in differences to be the effect size and computed a Z-statistic to test the null hypothesis of no significant difference between pre- and post-treatment estimates ($\alpha = 0.05$).

I assessed the point-level response of the avian community by estimating similarity in species composition for each point before and after treatment. I computed a value for Chao's abundance-based Jaccard similarity index using the average number of detections in pre- and post-treatment periods (index value approaches one as the average species composition between points becomes identical). I used a *Z*-statistic to test the null hypothesis of no significant difference between mean similarity among treatment and control units on each site ($\alpha = 0.05$).

Additionally, I used Chao's abundance-based Jaccard similarity index to estimate mean similarity in species composition on my treatment and control units prior to treatment. For each site, I computed a *Z*-statistic to test the null hypothesis of no significant difference between treatment and control unit estimates ($\alpha = 0.05$). This baseline comparison allowed me to evaluate the adequacy of the units I selected for my experiment prior to analyses of treatment effects.

RESULTS

Preliminary analyses

Local-scale habitat variable reduction analysis

Using the local-scale forest structure information, I synthesized nine variables in the habitat factor analysis: tree density, basal area, mean tree height, variation in tree height (SD tree height), percent dominance of ponderosa pine (PIPO dominance), large tree richness, total number of large snags (large snag sum), large snag decay, and overstory canopy closure. The results of the PCA indicated three factors adequately (eigenvalues > 1.00) summarized the nine forest structure attributes and explained 66.6% of the total variance (Table 2.3). For the three retained habitat factors, final communality estimates indicated that mean tree height, PIPO dominance, large snag decay, large tree richness, and canopy closure were most important, while tree density and basal area were moderately important descriptors of the total variance (Table 2.4). Large snag sum and SD tree height were less important descriptors (final communality estimates < 0.50). Using the three rotated habitat factor loadings in my interpretation, I considered points with higher values for habitat factor 1 (FAC_1; Appendix 2.D1) to be best characterized by increased decay levels in the larger snags, higher stem densities, and greater tree species richness. I considered higher values for habitat factor 2 (FAC_2; Appendix 2.D2) to be indicative of points dominated by more open stands of older ponderosa pine and higher values for factor 3 (FAC_3; Appendix 2.D3) indicative of points best summarized by high values for canopy closure and basal area.

Patch definition analysis

From the BoundarySeer® outputs and summary statistics for each of the forest structure attributes, I assigned each point to one of three arbitrary forest patch types: "open," "less dense," and "more dense" (Table 2.5 and Figure 2.3). Since the concept and composition of a patch is multidimensional (Turner et al. 2001), I used the term "dense" as a descriptor of the amount and relative contribution of horizontal and vertical forest structure attributes. Among sites, patches characterized by a less dense forest structural composition were most common (total extent = 1,606 ha) and open areas least common (642 ha). Within a 100-m radius of a point, patch heterogeneity (i.e., richness) was common. This buffered area around most points intersected two (n = 131) or three (n = 127) patch types, while very few (n = 20) intersected only one.

Prescribed fire effects on habitat analysis

Within six months of the completion of all prescribed fire treatments, fire-damaged trees were observed and measured at 112 (84%) of the 134 treatment unit points (Table 2.6). Trees in the sampling area around the remaining points were not affected by the treatment. Mean values for scorched and consumed foliage, maximum bole scorch height, and bole scorch circumference increased with the fire severity level defined in the two-step clustering procedure. The frequency and proportion of points with low, moderate, and heavy bole char severity also increased with fire severity level. Smaller values for scorched and consumed foliage ranked as most important in defining the low and moderate severity clusters, respectively (Table 2.7), while large values for bole scorch circumference and bole char severity ranked highest in the high severity cluster. Low and high values for maximum bole scorch height were most important in defining the low and high severity clusters, respectively. Points occurring within the three fire severity clusters were not always characterized by the same category of bole char severity (Table 2.7). I observed each of the three levels of fire severity on all but one site (CNF), where the high level was not detected. Tree damage was not detected on 22 of the 134 total treatment unit points. Among all four treatments, the CNF treatment had the most (11 of 23; 48%) points with low levels of fire severity and narrowest treatment window (5 days; Table 2.2). The GNF treatment had the most (11 of 21; 55%) points with high levels of fire severity and widest treatment window (5 months).

The global regression model of the relationship between fire severity and habitat covariates had an exponential spatial covariance structure and was considerably (> 50 AICc) better than the intercept-only model. However, this spatial model was < 1 AICc of the global model with no spatial covariance structure. Weights of evidence indicated that PatchType exhibited the greatest influence on levels of fire severity, followed by FAC_1, PatchRich, and FAC_2 (Table 2.8). FAC_3 and Slope were somewhat less important predictors and elevation was least important. Because PatchType was the most important predictor of fire severity level, I treated this covariate as the independent variable in a new mixed model and examined the pairwise differences among the three patch types in terms of fire severity (dependent variable). Regression coefficent estimates for fire severity increased with increasing density in patch forest structure (Table 2.9). Within each patch type, estimates for fire severity were significantly different from zero and pairwise differences were significant for open and less dense patch types and for open versus more dense patch types. Less dense patches did not differ from more dense.

Pre-treatment avian data and comparison of control and treatment units

Before prescribed fire treatments, I recorded 10,722 individual detections of 62 avian species during 1,442 sampling occasions on the four study sites between May–June, 2002–2003 (Appendix 2.A). At each site, community similarity indices did not differ significantly between treatment and control units (Table 2.10).

All avian data for analysis of response to prescribed fire treatments

Between May–June, 2002–2005, I recorded 32,497 individual detections of 82 avian species during 3,666 sampling occasions on the four study sites (Appendix 2.A). Thirty-two (39%) of these species were detected on all four sites and 12 of these species were detected > 1,000 times. The three most abundant species (23% of all detections) occurring on all four sites were Dark-eyed Junco (n = 3,469), Pygmy Nuthatch (n =

2,086), and Western Tanager (n = 2,024). Over half (n = 48) of all species were detected < 100 times and 36 were detected < 25 times, including single observations (n = 6). The mean number of detections among all species was 391.5 (SD = 669.6) and most species were detected on two or three sites (mean = 2.7, SD = 1.2). Forty-seven (57%) of the 82 species were considered migratory and the remainder were residents. Twenty-five of the 82 species were considered species of special concern in the Southwest (see list compiled by Hall et al. 1997) and the conservation status of six species was listed as vulnerable or imperiled in Arizona and/or New Mexico by NatureServe (http://natureserve.org).

Avian cluster analysis

To minimize the influence of outliers, I eliminated the five heaviest species (Common Raven, Great-horned Owl, Northern Goshawk, Red-tailed Hawk, and Wild Turkey; average mass > 1,000 grams; see appendix 2.A for species names) from the two step cluster analysis. As longevity data were unavailable for eight species used in the cluster analysis (see Appendix 2.B), I constructed a simple linear regression model of the relationship between longevity and log-transformed body mass for all other detected species for which these information were available. I then estimated missing longevity values using the intercept and slope coefficients derived from this statistically significant relationship ($y = 4.23 \times 1.43x$; $R^2 = 0.33$, d.f. = 69, $F = 33.8$, $P < 0.001$). Since data on incubation time or nestling period were unavailable for three species (see Appendix 2.B), I estimated IN using available data for the most functionally and phylogenetically similar species. Based on the nine ecological and life-history traits, 69 species were assigned to one of the three clusters identified by the two step algorithm (Table 2.11 and Appendix 2.B). Cluster 1 included 30% (n = 21) of these species and was dominated by cavity-nesting residents that foraged on the bark of trees (Tables 2.12–14). Forty-nine percent (*n* = 34) of all species belonged to cluster 2, which was typified by small neotropical migrants and relatively little reproductive investment. Cluster 3 included the remaining (*n* = 14) species and was most characterized by larger, seed-eating and omnivorous species.

Avian response to prescribed fire and habitat

Response of avian species density to prescribed fire treatments

Of the 82 species detected between May–June, 2002–2005, I considered 21,612 individual detections of 74 species for analyses in program DISTANCE, excluding detections > 100 m, flyovers, and incidental observations. Of these 74 species, I produced estimates of $\hat{D}$ and $\hat{N}$ for 14 species with > 500 total detections (Tables 2.15 and Appendix 2.E). This group of 14 species became the inference set in my statistical models of individual density response to prescribed fire treatments. In general, model selection uncertainty due to the multiple factor covariates used to model $\hat{p}$ for each of these species was minimal (Table 2.15). Models with the year covariate were always among the best models in my candidate set and accounted for most of the weight. Within species, estimates of effective detection radius showed little variation and only three species had an estimated effective detection radius < 50 m.

Using the stratum-level estimates, I detected a statistically significant difference ($\hat{\Delta}_s$) in pre- and post-treatment density on ≥ 1 of the four sites for eight of the 14 species in my inference set (Table 2.16). Five of these eight species exhibited a significant response on two sites and none responded on more than two sites. Only three (Western Bluebird, Steller's Jay, and Hairy Woodpecker) of these eight species responded in a consistent direction (positive) across sites ($Z > 0$). Of the six species that did not exhibit a statistically significant difference at the site level, the Plumbeous Vireo and Western Wood-Pewee consistently responded in a positive and negative direction, respectively. However, Western Wood-Pewee was the only species to show a significant overall (mean among means) decline following treatment ($Z = -4.919$, $P < 0.001$).

Models of the point-level response ($\hat{\Delta}^*_j$) of the 14 inference set species to habitat covariates, including treatment, suggested that elevation was the most important covariate (mean $w_+ = 0.639$), followed by two of the three local-scale habitat factors (FAC_2 = 0.515, FAC_1 = 0.513) and treatment (0.482; Table 2.17; see also candidate models in Appendix 2.F). However, there was some evidence for the importance of the remaining habitat covariates (all mean $w_+ = 0.374$–0.479). Among my inference set of species, weights of evidence suggested that most of these species were more strongly associated with pre-treatment habitat covariates than with the treatments themselves. The treatment covariate had the most weight for only the American Robin (0.997). Regression coefficients suggested a positive or negative point-level response to treatment by 11 and three species, respectively, although the magnitude of these values was usually small (Table 2.18). Elevation was typically ($n = 6$ species) the covariate with the maximum weight. Coefficients for elevation were negative for all but two species (White-breasted Nuthatch and American Robin). Most species responded positively ($n = 11$) to increasing values of the FAC_1 habitat factor and negatively ($n = 10$) to increasing values of FAC_2. For all other covariates, patterns of response were less marked.

Response of avian species density to prescribed fire severity

Focusing on the 14 inference species, I examined the response of five species to levels of fire severity (w_+ for the treatment covariate was ≥ 0.50): Pygmy Nuthatch, Western Bluebird, Steller's Jay, American Robin, and Hairy Woodpecker (Table 2.19). For Pygmy Nuthatch ($w_+ = 0.75$), Western Bluebird, (0.561), and Steller's Jay (0.642), I was unable to fit a clearly better model than the intercept-only model and regression coefficients indicated a mixed response by these species to the four ordinal levels of fire severity (Table 2.20). The strong response of American Robin was best approximated by a second-degree polynomial model form and the response of Hairy Woodpecker (0.866) by the log normal and linear model forms (ΔAICc ≤ 2 for both). For each of the five species, no pairwise difference among severity level categories was statistically significant (all P's > 0.07).

Overall, I observed little or no spatial structure in residual and semivariance plots of species response to habitat covariates or fire severity at the unit level, suggesting that my point-level estimates of $\hat{D}$ were not highly influenced by positive spatial correlation in the data. Although the exponential spatial covariance model was always the best model of spatial correlation, these models were typically < 2 AICc of the best non-spatial model.

Response of avian cluster density to prescribed fire treatments

Using program DISTANCE, I produced reliable estimates of $\hat{D}$ and $\hat{N}$ for each of the three clusters (Table 2.21 and Appendix 2.G). For each cluster, year, site, and ObsRank covariates were always in the best model of $\hat{p}$ and model selection uncertainty was negligible. Among sites, I observed an overall positive trend in the response of clusters 1 and 3 and a negative trend in cluster 2 (Table 2.22). I detected a statistically significant difference ($\hat{\Delta}_s$) in pre- and post-treatment density at only one site for clusters 2 (ASNF) and 3 (GNF).

In general, point-level models of cluster response to habitat covariates indicated that elevation was the most important covariate within ($w_+ \geq 0.865$) and among (mean $w_+ = 0.946$) clusters, followed by two of the three local-scale habitat factors (FAC_2 = 0.711, FAC_1 = 0.639) and PatchType (0.624; Table 2.23; see also candidate models in Appendix 2.H). Including treatment (0.348), there was less evidence for the importance of the remaining covariates (all mean $w_+ = 0.308$–0.386). Regression coefficients for the three clusters generally suggested an overall negative response to the elevation and FAC_2 covariates and an overall positive response to slope, PatchType, and FAC_1 (Table 2.24). Although there was some evidence for a treatment unit response by cluster1 ($w_+ = 0.451$), I did not explicitly model the response of this cluster as a function of fire severity level. However, regression coefficients for clusters 1 and 2 did indicate a slightly increasing trend in $\hat{\Delta}^*_j$ with increasing levels of fire severity (Table 2.25). Pairwise differences among severity level categories were not statistically significant.

Response of avian community to prescribed fire treatments

Among the 74 species I analyzed, diversity remained relatively unchanged on each of the four sites two years after fire. In general, patterns in overall differences within and among sites suggested by the five community metrics were mixed and estimated values for each metric were only marginally influenced by the effect of year, unit, area, and Effort × ObsRank (Table 2.26). Estimates of Chao and jackknife species richness were most influenced by a year effect and the Chao-Jaccard similarity index by the effect of Effort × ObsRank. Estimates of Fisher's α were most influenced by a treatment unit effect and no effects were detected on estimates of the Simpson index.

Across sites, I detected no significant difference between pre- and post-treatment stratum-level estimates of total species richness (Chao and jackknife; Table 2.27). Although mean values for the overall (mean of means) difference in these estimates were positive (0.47 species), these differences were not statistically significant ($Z < 2.7$, $P > 0.39$). The smallest average values for absolute minimum species richness were observed on the ASNF units in 2002–2003 (Chao; $n = 21.4$ and 23.9) and the maximum average value for species richness was observed on the CNF control unit in 2004–2005 (jackknife; $n = 40.5$). Although overall differences were small, I observed a large increase in average estimated species richness between periods (Chao and jackknife > 5.0). The largest effect size was detected on the CNF (Chao: 2.89, SE = 2.58, jackknife: 3.42, SE = 4.67).

On each site, the change in the approximate number of rare species after treatment was not significant (Table 2.27). The fewest (Fisher's $\alpha = 4.2$) and greatest (8.0) average number of rare species each occurred on the ASNF in different periods and on different units. Although I detected an overall increase in the estimated number of rare species, this difference was mostly influenced by the relatively large positive effect size observed on the CNF (0.91, SE = 0.56).

With the exception of the GNF, average site-level differences between pre- and post-treatment estimates of community diversity were small (Table 2.27). Mean diversity decreased on the GNF control units in 2004–2005 and resulted in a statistically significant difference on this site (effect size = 3.24, SE = 0.98, $Z = 3.30$, $P < 0.001$) and an overall positive effect size among sites (0.80, SE = 1.65). Before and after treatment, community diversity was low on the ASNF site (all Simpson index values < 12.0) and highest on the CNF control unit (Simpson index values > 16.0).

On each site, I detected no significant differences in within unit similarity and mean values for species composition among points were similar on each unit before and after treatment (Table 2.27). Average compositional similarity was highest on the CNF treatment unit in 2004–2005 (Chao-Jaccard index = 0.78) and lowest on the GNF control unit in 2002–2003 (0.42).

Differences between treatment and control unit similarity in species composition at the point level were positive and statistically significant for two (CNF and GNF) of the four sites (Table 2.28). Although the overall range of values was small (0.119), average between-point compositional similarity was highest on the GNF treatment unit (Chao-Jaccard index = 0.752) and lowest on the ASNF (0.633 and 0.635).

DISCUSSION

Response of habitat to prescribed fire treatments

Following prescribed fire, I observed substantial spatial heterogeneity in patterns of fire behavior and severity within and among the four treatment units. Such patterns, however, are not unique to this study and have been observed in previous studies of wildland (Agee 1993, Turner et al. 2003, Cocke et al. 2005, Smucker et al. 2005) and prescribed fire (Fulé et al. 2004, Knapp and Keeley 2006). Although each of the treatments implemented as part of this study was designed to maximize coverage of the treatment unit and the consumption of small diameter trees and other hazardous fuels, uniform patterns of fire behavior and severity were minimized by climatic conditions, the arrangement and contiguity of fuels, physiographic barriers, and uneven ignition efforts in space and time. Substantial point-level heterogeneity was also suggested by my regression models of fire severity – the model with spatial covariance structure was not clearly better than the model without covariance terms. Indeed, these results might be anticipated with any large-scale treatment effort characterized by low- to moderate-intensity prescribed fire and reflect the post-fire heterogeneity in treatment effects typical of such efforts.

In my study, habitat covariates that characterized initial conditions were good predictors of levels of fire severity and digitally-derived patch-scale covariates were more important predictors than local-scale covariates. An interplay between patch-scale forest structural characteristics and fire severity has been described by previous studies (Turner et al. 1999, Kaufmann et al. 2003, Fulé et al. 2004, Lentile et al. 2005). Levels of fire severity increased with density in forest structural attributes at the patch scale, but patterns were less clear at the local scale. Although structural complexity associated with the FAC_1 covariate was important in predicting fire severity level, information collected and summarized using only local-scale ground-based data was insufficient for predicting fire effects on trees. Additionally, local-scale information obtained from ground plots can be labor-intensive and costly, suggesting there is added utility in the use of digitally-derived and interpolated information in assessments of fire effects at the patch scale.

The treatments implemented as part of my study were designed to minimize risk of crown fire, and this risk was partly mitigated by using fall and spring ignitions. Typically, fuel moisture and relative humidity are higher and temperature is lower during these seasons, whereas conditions that facilitate increased fire intensity and spread tend to converge in May and June, prior to the summer rainy season (Swetnam and Betancourt 1998). The largest portion of the GNF treatment was completed in May and this timing likely contributed to the relatively large number of points with high levels of fire severity. I observed fewer fire-damaged trees and lower levels of fire severity on the CNF, which was treated over five days in September. During this post-monsoonal period, much of the live understory biomass (e.g., grass) was still green and live and dead fuel moisture levels were high. The lower temperatures and higher relative humidity recorded during the ASNF and KNF treatments, which occurred during winter months, were reflected in the mostly moderate levels of tree damage and severity that I observed on these sites.

Response of avian species density to prescribed fire treatments

In the dry, relatively unproductive ponderosa pine forests of the Southwest, avian densities tend to be low and sample sizes sufficient for precise estimation of density can be difficult to obtain over short periods. Of the substantial number of detections and species recorded during four breeding seasons, my distance-based design and analysis permitted the precise estimation of detectability and density for only 14 of the most abundant species. Indeed, large amounts of effort and data were necessary to generate precise estimates and to construct robust models of avian response to habitat and fire severity covariates at the point level. Accounting for variation in the individual detection probability of each species, and the factors that can affect this variation, is a critical step in measuring change in avian communities (MacKenzie and Kendall 2002, Royle et al. 2004). Because the detection probabilities I estimated were < 1.0, and because these estimates differed among sites and stratum for each species, controlling for this variation in my analyses was essential. Empirically-based studies that fail to account for such variation in detectability can produce biased or spurious results that should not be used for causal inference (MacKenzie and Kendall 2002, Rosenstock et al. 2002, Ellingson and Lukacs 2003, Norvell et al. 2003, White 2005). Thus, my results rely on very different estimation methods from previously published observational studies of wildland fire effects and any generalizations about avian response to fire should be considered with these differences in mind.

Among the 14 species most often detected on my sites, few site-level differences in pre- and post-treatment density were statistically significant and no species responded significantly to treatment on all four sites. Recently published studies that examined the short-term response of avian communities to prescribed fire treatment in habitats that included mature pine forests (e.g., Bock and Bock 1983, Horton and Mannan 1988, King et al. 1998) also reported little or no overall change in density or abundance in most species for which coarse estimates could be generated. Many of these species were the same as those included in my inference set. Each of these stand-scale studies examined treatments at a single geographic location and only one (Horton and Mannan 1988) used an experimental design that included avian community data collected before treatment. In her review of the literature on prescribed fire, Short (2003) found few studies that examined avian communities using a design that could permit causal inference or inferences beyond the scope of very small sampling units. Only one study (Artman et al. 2001) in mixed-oak forests of southern Ohio used well-defined, replicated treatments in a BACI design. Additionally, avian response to levels of fire severity has been explicitly examined by only a single published study that used a BACI design (Smucker et al. 2005). Although these authors were able to take advantage of an unplanned wildfire to make comparisons of avian abundance among "treated" and control plots, patterns of fire response were inferred from a single, unreplicated fire event. The results of my experiment indicated that avian response to multiple prescribed fire events can be modeled as a function of fire severity, but that response may vary among treatment units. Spatial replication and a multi-scale analysis, including point-level models, were necessary to uncover these varying relationships.

Since avian response to disturbance can be a multi-scale process (Thompson et al. 1995, see Finch et al. 1997, Brawn et al. 2001), a BACI analysis alone may provide an incomplete picture of avian response to prescribed fire and multiple levels of investigation should be used to infer and establish causation. For example, results suggesting a significant response to treatment by Mountain Chickadee, Grace's Warbler, and Western Tanager at the site level were not supported by weights of evidence values computed at the point level. In contrast to my site-level results, point-level statistical models of avian response to fire severity, treatment, and habitat covariates revealed relationships that were more easily interpreted. These statistical models allowed me to account for spatial heterogeneity in the occurrence of individual species and directly relate this occurrence information to a suite of relevant covariates, including levels of fire severity, and discriminate between treatment effects and pre-fire habitat factors. Results from my BACI analysis were not always supported by results from the point-level regression models.

Of the five species responding to the treatment covariate at the point level, changes in density for only two species could be strongly associated with levels of fire severity. American Robin exhibited the strongest treatment response at both the unit and point scales and increases in density were related to low and moderate levels of fire severity. In their analysis of short-term avian response to wildland fire in mixed-coniferous stands in Montana, Smucker et al. (2005) detected a difference in the pre- and post-fire (≤ 3 years) relative abundance of American Robin and found that this response was only statistically significant at moderate levels of fire severity. Independent of fire severity, however, these authors had insufficient evidence to suggest a treatment response by this species. One study of prescribed fire in a ponderosa pine forest of South Dakota also indicated the response of American Robin as mixed (Bock and Bock 1983). Three years after a large wildlfire in northern Arizona, Bock and Block (2005a) reported large numbers of American Robin detections on sampling stations that were moderately burned, although most detections occurred on severely burned stations during the breeding season. In other studies of wildland fire, where levels of fire severity were not defined, changes in American Robin abundance have been positive, negative, and/or neutral, or "mixed" (see reviews by Kotliar et al. 2002 and Bock and Block 2005b). Hutto (1995) summarized a strong positive response by American Robin to stand-replacing fire events of the Northern Rocky Mountains, especially in recently burned stands. These mixed results suggest that American Robin are responding to factors that include variation in fire severity, rather than any homogeneity in fire effects assumed within and among these studies. During the breeding season, American Robin are a versatile, mostly insectivorous ground forager with broad habitat preferences (Wheelwright 1986, Sibley 2001). Thus, the immediate and strong numerical response of this species to low and moderate levels of fire severity that I observed was probably related to fire-induced effects, including favorable changes in understory vegetation composition and vigor

(Sackett and Haase 1998) and increased food availability (e.g., earthworms, arthropods, and fruits; Wikars and Schimmel 2001, McHugh et al. 2003, Short and Negrón 2003, Certini 2005). At higher levels of severity, surface and soil organic materials are killed or volatilized (Sackett et al. 1996, Kerns et al. 2006) and preferred insect prey may be less common (Wikars and Schimmel 2001, Certini 2005).

Hairy Woodpecker exhibited a strong positive numerical response at unit and point scales. Although the point-level difference between pre- and post-treatment density was greatest at the highest fire severity level, this difference was not significantly different from the three other levels of severity, including unburned (Table 2.19; $\beta = -0.082$, SE = 0.046, Tukey-Kramer $P = 0.349$), and the functional form (e.g., linear) of this response was not clear. In the only other study that tested the relationship between severity levels and pre- and post-fire changes in abundance, Smucker et al. (2005) reported the response of Hairy Woodpecker to be statistically significant only for high levels of wildfire severity. However, the difference between high and moderate levels they estimated was not statistically significant ($\alpha = 0.05$) and the functional form of Hairy Woodpecker response to four levels of severity was not linear. Additionally, Smucker et al. (2005) reported there was insufficient evidence to suggest a treatment response by Hairy Woodpecker to variation in fire severity. In northern Arizona, Bock and Block (2005a) observed higher numbers of Hairy Woodpecker on sampling stations severely burned by wildlfire than on stations where severity was considered moderate. The fewest number of Hairy Woodpecker were observed on unburned stations. Reviews of published and unpublished studies by Hutto (1995) and Kotliar et al. (2002) both summarized generally positive increases in Hairy Woodpecker abundance in western forests burned by wildland fire.

My results suggest a strong association between the pre-treatment attributes of large snags and positive, post-treatment changes in Hairy Woodpecker density (FAC_1 $w_+ = 0.897$). Nevertheless, it is unlikely that the change in density that I detected was associated with immediate increases in suitable nest trees (e.g., snags) or nest tree attributes (Saab et al. 2004). Indeed, the pre- and post-treatment difference in the number of large snags on my treatment units was not statistically significant ($\alpha = 0.05$; Saab et al. In Review). The response of Hairy Woodpecker may be independent of treatment and available nesting habitats. In South Dakota, Spiering and Knight (2005) did not find a significant statistical relationship between the abundance of Hairy Woodpecker and snag densities in managed ponderosa pine stands on the Black Hills National Forest. Many of the large snags used by Hairy Woodpecker for nesting on my units prior to treatment were consumed by the prescribed fire. Horton and Mannan (1988) also reported large reductions in ponderosa pine snags and no significant change in the abundance of Hairy Woodpecker immediately after prescribed fires in southeastern Arizona. The post-treatment recruitment of snags with characteristics suitable for Hairy Woodpecker occupancy on my sites will likely be guided by future decay and excavation periods > 2 years (McHugh and Kolb 2003, Saab et al. 2004).

I attributed the post-treatment increase in Hairy Woodpecker density to increased food availability and the positive response of fire-dependent arthropod communities (e.g., bark beetles) to higher levels of fire severity and tree mortality (Powell et al. 2002, McHugh et al. 2003). Across my four sites, the bark beetle community (namely, western pine beetle [*Dendroctonus brevicomis* LeConte], southern pine beetle [*D. frontalis* Zimmerman], and *Ips* species) was more abundant and diverse on trees damaged by prescribed fire than on trees located on control units (C. Breece, unpublished data). Between 2003 and 2005, eight bark beetle species were detected on 105 (10%) of 1,045 trees sampled on treatment units, compared with 24 (2%) of 1,197 trees sampled on control units. Fifty nine (5%) of the trees sampled on treatment units were presumably killed by bark beetles, compared to only two (0.2%) on the controls. In addition to an increase in the number of trees attacked and killed by bark beetles, I continue to observe an increase in the foraging activities of Hairy Woodpecker and other woodpecker species on the burned units two years post treatment. The mechanics of the relationship between prescribed fire severity, arthropod abundance and composition, and avian communities have not been thoroughly examined and continued research on my sites is exploring this relationship in an experimental framework.

Of the three other species that responded strongly to the treatment covariate, only the results for Pygmy Nuthatch suggested a change in density that was negative. Reviews of the published and unpublished literature by Kotliar et al. (2002) and Bock and Block (2005b) also indicated a generally negative response by this species to wildland fire. The positive post-treatment difference in density exhibited by Western Bluebird is also consistent with those summarized by Bock and Block (2005b). Similar to the observations of Bock and Block (2005a), I detected the largest difference for this species on high severity points. The positive difference in density exhibited by Steller's Jay in my analysis is not consistent with the summary provided by Kotliar et al. (2002), which indicated a negative response to wildland fire is more typical for this species. In other examinations of Steller's Jay response to wildland fire, results from Smucker et al. (2005) were inconclusive and Bock and Block (2005b) reported mixed findings.

Although effect sizes were small and non-significant, post-treatment decreases in Western Wood-Pewee density were observed on each site and the PatchRich covariate was the most important covariate to explain these decreases at the point level. These results are consistent with previous observations that Western Wood-Pewee are sensitive to disturbance in forested habitats (Hejl et al. 1988) and are associated with edge habitats for many of their life-history requisites (Szaro and Balda 1979). In contrast, a short-term, positive response by this species to moderate- and severe-levels of wildland fire in ponderosa pine has been reported previously (Bock and Block 2005a).

Response of avian cluster density to prescribed fire treatments

Ecological and life-history traits have frequently been used to evaluate the response of species and communities to habitat disturbance (Hansen and Urban 1992, Poff 1997, Kolb and Diekmann 2005). Considering such traits to identify clusters of species, the quantitative and repeatable two-step clustering algorithm I used revealed few patterns, but did offer additional insight into the immediate response of groups of species to prescribed fire. In my analysis of species clusters, I found few statistically significant results at individual sites, although trends among sites were somewhat more revealing. Because these trends were not statistically significant, however, I consider my interpretation to be speculative. I quantified an overall positive post-treatment response by cluster 1, which was mostly comprised of residents and species that forage for insects on the bark of trees, including all of the woodpeckers and most of the cavity-nesting species I detected. Weights of evidence suggested that the point-level response of this cluster to the treatment covariate was not especially strong ($w_+ = 0.451$), but stronger than clusters 2 or 3. Reviews by Hutto (1995), Kotliar et al. (2002), and Bock and Block (2005b) also suggest a generally positive numerical response to wildland fire by most of the species included in cluster 1. Cluster 2, which was largely comprised of migrant species that foraged on leaves or flowers during the breeding season, exhibited an overall negative response among sites, but a weak treatment response at the point level. A negative numerical response by foliage gleaning insectivores to wildland fire was summarized in reviews by Ganey et al. (1996), Finch et al. (1997), and Kotliar et al. (2002). For this cluster, a statistically significant result was detected on the ASNF site where the response of the most abundant foliage gleaner, Grace's Warbler, was also significantly negative. The generally positive response of cluster 3 was likely driven by the two most abundant species in this group (Dark-eyed Junco and Steller's Jay: total n = 3,528). At the point level, large weights of evidence for the PatchType and FAC_2 covariates indicated that the response of this cluster was also a multi-scale process that included frequent use of less complex forest characteristics. Although the elevation covariate had the most weight in each cluster, I believe this result represented the importance of unmeasured covariates associated with elevation at the site level and further interpretation was difficult.

Clusters of functionally and ecologically similar species can serve as response surrogates when member species are rare or inherently difficult to detect (Gaston 1994, Maina and Howe 2000), or when indicators of post-treatment change are difficult or costly to measure (Block et al. 2001). The utility of an empirically-based clustering approach deserves greater research attention and future studies should endeavor to test predictions about the response of meaningful groups of species to specific forest management actions (Szaro 1986, Landres et al. 1988, Block et al. 2001), including prescribed fire.

Response of the avian community to prescribed fire treatments

After pooling 74 of the species detected during sampling, I was unable to identify non-random patterns of avian community response to treatment among the four study sites. Robust metrics for computing change in community composition and diversity at the stratum and site level revealed only one statistically significant difference, though significant differences in pre- and post-treatment similarity between points did emerge on two sites (CNF and GNF). Each of these differences suggested breeding bird community membership became more even or similar after treatment. Previous studies of prescribed fire and avian populations in forested habitats by Bock and Bock (1983), Horton and Mannan (1988), King et al. (1998), Artman et al. (2001), and Moreira et al. (2003) reported minimal or no response to treatment at the community level and that species composition was relatively unchanged. Similarly, my results indicated that differences in pre- and post-treatment community structure were small and that estimates of diversity tended toward constancy over the short time period examined. Point-level shifts in avian community similarity, however, suggested that within unit spatial heterogeneity in treatment effects was responsible for the few significant differences detected.

Management and research implications

In the Southwest, large-scale (e.g., > 10,000 ha) treatments are being considered in order to reduce fire risk adjacent to human communities and restore ponderosa pine forest structure to pre-fire suppression conditions (Allen et al. 2002, Noss et al. 2006). Not surprisingly, the socio-political debate surrounding expanded treatments is polarized (Mutch 1995, Dombeck et al. 2004) and forest planning efforts are evaluating the merits of various treatment alternatives (Sisk et al. 2006). The degree to which management efforts can mimic historic fire regimes and forest structural conditions using prescribed fire, however, will depend on the timing and frequency of treatment (Sackett et al. 1996, Sackett and Haase 1998). The magnitude of impact of these treatments on forest ecosystem attributes, including wildlife, will be dependent on the season within which burning is conducted (Harrington and Sackett 1990, Rotenberry et al. 1995, Tiedemann 2000, Kerns et al. 2006, Knapp and Keeley 2006).

Because large-scale treatments are expensive and difficult to implement, information on treatment-induced changes to forest structure may be limited or unavailable to most forest managers. If pre-fire forest structure attributes are central in determining post-fire levels of fire severity (Turner et al. 1999, Zimmerman 2003, Cocke et al. 2005), then predictive models with covariates that include pre-treatment, or "initial," habitat conditions can be useful to managers interested in understanding the likely outcomes of prescribed fire treatments on forest structure. These models will be improved by the inclusion of covariate information derived at spatial scales that capture relevant ecosystem processes.

My results suggest that, in the short term, breeding bird communities in southwestern ponderosa pine forests are tolerant of prescribed fire treatments at the intensities I studied (e.g., broadcast burning). In addition, breeding birds appear able to accommodate these treatments at multiple spatial scales and across multiple geographic locations. Therefore, sustaining the breeding bird community should be compatible with efforts to reintroduce fire to landscapes in order to restore ecosystem function and/or mitigate wildland fire risk. An important caveat, however, is that the short-term (2-year) post-treatment patterns that I detected could change with time since treatment. Because time since fire can influence results from studies of avian response to fire (Hobson and Schieck 1999, Artman et al. 2001, Smucker et al. 2005), long-term monitoring will be necessary to associate patterns of change with the direct (e.g., tree mortality) and indirect (e.g., increased food availability) effects of fire treatment. Nevertheless, short-term studies of wildlife community response to prescribed fire can inform longer-term management objectives when rigorous assessments focus on the immediate periods or seasons during which species might be most affected by treatments (see Rotenberry et al. 1995). For avian communities in the forested region I studied, post-treatment results from my analyses of data pooled over a 2-year period probably do not mask patterns of response in conflict with management objectives on large landscapes (but see Smucker et al. 2005). If one objective of prescribed-fire policies is to maintain or increase biodiversity (Noss et al. 2006), then treatments should be designed to encourage heterogeneity in fire effects and sustain essential habitats for wildlife communities evolved to depend on episodic fire events.

Concomitantly, the above interpretation of my results assumes that avian communities in southwestern pine forests are tolerant of low to moderate levels of fuels treatments because important habitat features and resources remain after such treatments. However, if these communities occupy habitats that were previously degraded by human-induced factors, such as logging and fire suppression (Hejl 1994, Hall et al. 1997), then patterns inferred from my results should be tempered by the local or regional influence of these historical activities on avian community structure. Although relatively common on my sites, regional declines in Pygmy Nuthatch, White-breasted Nuthatch, and Western Bluebird, for example, have been attributed to human-modification of the landscape since Euro-American settlement (Brawn and Balda 1988, DeSante and George 1994, see Hall et al. 1997). Unfortunately, knowledge of pre-settlement avian community structure and habitat is incomplete and managers lack even rudimentary baseline information (see Dahms and Geils 1997, Chambers and Germaine 2003). Experiments that investigate avian response to treatment alternatives on multiple landscapes will help to fill this knowledge gap.

In ponderosa pine forests of the Southwest, large stand-replacing fires are rare events with consequences that are uncharacteristic of the historical fire regime (Swetnam and Baisan 1996, Schoennagel et al. 2004) and perceived as undesirable (see Allen et al. 2002). Since forest managers concerned about fire risk, for example, are more likely to implement prescribed fire treatments that result in low-to-moderate levels of severity, research on avian communities should more closely investigate the effects of these treatments on bird populations and their habitats. In this context, results from case studies that examine the correlation between stand-replacing fire events and only a few members of the community may provide information that is of limited relevance to management needs (Bock and Block 2005b). Generalizations about the response of avian communities to forest management activities, including prescribed fire, will require that research efforts emphasize experiments and the mechanisms underlying observed patterns (Marzluff et al. 2000, Van Horne 2005). Stronger inference and comparative assessments will be made when demographic parameters are precisely estimated for ecologically relevant species and accompanied by meaningful measures of effect size.

LITERATURE CITED

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C., USA.

Agee, J. K. 1998. The landscape ecology of western forest fire regimes. Northwest Science 72:24-34 Sp. Iss.

Agee, J. K. and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83-96 Sp. Iss.

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12:1418-1433.

Angermeier, P. L. and J. R. Karr. 1994. Biological integrity versus biological diversity as policy directives. BioScience 44:690-697.

Artman, V. L., E. K. Sutherland, and J. F. Downhower. 2001. Prescribed burning to restore mixed-oak communities in southern Ohio: effects on breeding bird populations. Conservation Biology 15:1423-1434.

Barbieri, M. M. and J. O. Berger. 2004. Optimal predictive model selection. Annals of Statistics 32:870-897.

Bezdek, J. C. 1987. Some non-standard clustering algorithms. Pages 225-287 *in* P. Legendre and L. Legendre (eds.). Developments in numerical ecology. Springer-Verlag, Berlin, Germany.

Block, W. M., A. B. Franklin, J. P. Ward, Jr., J. L. Ganey, and G. C. White. 2001. Design and implementation of monitoring studies to evaluate the success of ecological restoration on wildlife. Restoration Ecology 9:293-303.

Bock, C. E. and J. H. Bock 1983. Responses of birds and deer mice to prescribed burning in ponderosa pine. Journal of Wildlife Management 47:836-840.

Bock, C. E. and W. M. Block. 2005a. Response of birds to fire in the American Southwest. Pages 1093-1099 *in* C. J. Ralph and T. D. Rich (eds). Bird conservation implementation and integration in the Americas: proceedings of the third international Partners in Flight CNFerence. USDA Forest Service, Pacific Southwest Research Station, General Technical Report PSW-GTR-191, Albany, CA, USA.

Bock, C. E. and W. M. Block. 2005b. Fire and birds in the southwestern United States. Pages 14-32 *in* V. A. Saab and H.D.W. Powell (eds.). Fire and avian ecology in North America: process influencing pattern. Studies in Avian Biology 30.

Brawn, J. D. and R. P. Balda. 1988. The influence of silvicultural activity on ponderosa pine forest bird communities in the southwestern United States. Pages 3-21 *in* J.A. Jackson (ed.). Bird Conservation 3. University of Wisconsin Press, Madison, WI, USA.

Brawn, J. D., S. K. Robinson, and F. R. Thompson III. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32:251-276.

Brown, D. G. 1998a. Mapping historical forest types in Baraga County Michigan, USA as fuzzy sets. Plant Ecology 134:97-111.

Brown, D. G. 1998b. Classification and boundary vagueness in mapping presettlement forest types. International Journal of Geographical Information Science 12:105-129.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology 14:513-532.

Brown R. T., J. K. Agee, and J. F. Franklin. 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18:903-912.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling. Oxford University Press, London, UK.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2004. Advanced distance sampling. Oxford University Press, London, UK.

Bull, E. L., C. G. Parks, and T. R. Torgersen. 1997. Trees and logs important to wildlife in the interior Columbia River basin. USDA Forest Service, Pacific Northwest Research Station, General Technical Report PNW-GTR-391, Portland, OR, USA.

Burnham, K. P. and D. R. Anderson. 2002. Model Selection and multi-model inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, NY, USA.

Burnham, K. P. and W. S. Overton. 1978. Estimation of the size of a closed population when capture probabilities vary among animals. Biometrika 65:623-633.

Burnham, K. P. and W. S. Overton. 1979. Robust estimation of population size when capture probabilities vary among animals. Ecology 60:927-936.

Certini, G. 2005. Effects of fire on properties of forest soils: a review. Oecologia 143:1-10.

Chambers, C. L. and S. S. Germaine. 2003. Vertebrates. Pages 268-285 *in* P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Chao, A., R. L. Chazdon, R. K. Colwell, and T. J. Shen. 2005. A new statistical approach for assessing compositional similarity based on incidence and abundance data. Ecology Letters 8:148-159.

Chao, A. 1987. Estimating the population size for capture-recapture data with unequal catchability. Biometrics 43:783-791.

Chappell, C. B. and J. K. Agee. 1996. Fire severity and tree seedling establishment in *Abies magnifica* forests, southern Cascades, Oregon. Ecological Applications 6:628-640.

Chiu, T., D. Fang, J. Chen, Y. Wang, and C. Jeris. 2001. A robust and scalable clustering algorithm for mixed type attributes in large database environment. Pages 263-268 *in* Proceedings of the 7th ACM SIGKDDD international CNFerence on knowledge discovery and data mining. San Francisco, CA, USA.

Cocke, A. E., P. Z. Fulé, and J. E. Crouse. 2005. Comparison of burn severity assessments using Differenced Normalized Burn Ratio and ground data. International Journal of Wildland Fire 14:189-198.

Colwell, R. K. 2005. EstimateS: Statistical estimation of species richness and shared species from samples. Version 7.5. User's guide and application published online at: http://purl.oclc.org/estimates.

Cooper, C. F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs 30:129-164.

Covington, W. W. 2000. Helping western forests heal: the prognosis is poor for U.S. forest ecosystems. Nature 408:135-136.

Covington, W. W. 2003. The evolutionary and historical context. Pages 26-47 *in* P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Covington, W. W. and M. M. Moore. 1994. Southwestern ponderosa forest structure and resource conditions: changes since Euro-American settlement. Journal of Forestry 92:39-47.

Covington, W. W., P. Z. Fulé, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. Journal of Forestry 95:23-29.

Dahms, C. W. and B. W. Geils (eds.). 1997. An assessment of forest ecosystem health in the Southwest. USDA Forest Service General Technical Report RM-GTR-295, Fort Collins, CO, USA.

DeSante, D. F. and T. L. George. 1994. Population trends in the land birds of western North America. Pages 173-190 *in* J. R. Jehl, Jr. and N. K. Johnson (eds.). A century of avifaunal change in western North America. Studies in Avian Biology 15.

Dickson, B. G., W. M. Block, and T. D. Sisk. 2004. Conceptual framework for studying the effects of fuels treatments on avian communities in ponderosa pine forests of northern Arizona. Pages 193-200 *in* C. Van Riper, III and K. Cole (eds.). The Colorado Plateau: cultural, biological and physical research. The University of Arizona Press, Tucson, AZ, USA.

Dickson, B. G., J. W. Prather, Y. Xu, H. M. Hampton, E. N. Aumack, and T. D. Sisk. 2006. Mapping the probability of large fire occurrence in northern Arizona. Landscape Ecology 21:xx-xx.

Dombeck, M. P., J. E. Williams, and C. A. Wood. 2004. Wildfire policy and public lands: Integrating scientific understanding with social concerns across landscapes. Conservation Biology 18:883-889.

Dunning, J. B. 1993. Body weights of 686 species of North American birds. International Wildlife Rehabilitation Council, Suisun, CA, USA.

Ehrlich, P. R., D. S. Dobkin, and D. Wheye. 1988. The birder's handbook: a field guide to the natural history of North American birds. Simon and Schuster, New York, NY, USA.

Ellingson, A. R. and P. M. Lukacs. 2003. Improving methods for regional landbird monitoring: a reply to Hutto and Young. Wildlife Society Bulletin 31:896-902.

Finch, D. M., J. L. Ganey, W. Yong, R. Kimbal, and R. Sallabanks. 1997. Effects and interactions of fire, logging, and grazing. Pages 103-136 *in* W. M. Block and D. M. Finch (eds.). Songbird ecology in southwestern ponderosa pine forests: a literature review. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-292, Fort Collins, CO, USA.

Finney, M. A., C. W. McHugh, and I. C. Grenfell. 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. Canadian Journal of Forest Research 35:1714-1722.

Frazer, G. W., C. D. Canham, and K. P. Lertzman. 1999. Gap Light Analyzer (GLA), Version 2.0. Imaging software to extract canopy structure and gap light transmission indices from true-colour fisheye photographs, users manual and program documentation. Simon Fraser University, Burnaby, British Columbia, Canada, and the Institute of Ecosystem Studies, Millbrook, New York, NY, USA.

Friederici, P. 2003. The Flagstaff model. Pages 7-25 *in* P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Fulé, P. Z., W. W. Covington, and M. M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7:895-908.

Fulé, P. Z., A. E. Cocke, T. A. Heinlein, and W. W. Covington. 2004. Effects of an intense prescribed forest fire: is it ecological restoration? Restoration Ecology 12:220-230.

Ganey, J. L., W. M. Block, and P. F. Boucher. 1996. Effects of fire on birds in Madrean forests and woodlands. Pages 46-155 *in* P. F. Ffolliott, L. F. DeBano, M. B. Baker Jr., G. H. Gottfried, G. Solis-Garza, C. B. Edminister, D. G. Neary, and R. H. Hamre (eds.). Effects of fire on Madrean province ecosystems. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-289, Fort Collins, CO, USA.

Gaston, K. J. 1994. Rarity. Population and Community Biology Series 13. Chapman and Hall, London, UK.

Graham R. T., S. McCaffrey, and T. B. Jain (eds.). 2004. Science basis for changing forest structure to modify wildfire behavior and severity. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-120, Fort Collins, Colorado, USA.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologists. Wiley, New York, NY, USA.

Hall, K. R. and S. L. Maruca. 2001. Mapping a forest mosaic: a comparison of vegetation and bird distributions using geographic boundary analysis. Plant Ecology 156:105-120.

Hall, L. S., M. L. Morrison, and W. M. Block. 1997. Songbird status and roles. Pages 69-88 *in* W. M. Block and D. M. Finch (eds.). Songbird ecology in southwestern ponderosa pine forests: a literature review. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-292, Fort Collins, CO, USA.

Hampton, H. H., Y. Xu, J. W. Prather, E. N. Aumack, B. G. Dickson, M. M. Howe, and T. D. Sisk. 2003. Spatial tools for guiding forest restoration and fuel reduction efforts. Proceedings of the 2003 ESRI international user CNFerence, July 7-11, San Diego, CA, USA.

Hansen, A. J. and D. L. Urban. 1992. Avian response to landscape pattern: the role of species life histories. Landscape Ecology 7:163-180.

Harrington, M. G. and S. S. Sackett. 1990. Using fire as a management tool in southwestern ponderosa pine. Pages 122-133 *in* J. S. Krammes (ed.). Effects of fire management of southwestern natural resources. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report, RM-GTR-191, Fort Collins, CO, USA.

Hayek, L. C. and M. A. Buzas. 1996. Surveying natural populations. Columbia University Press, New York, NY, USA.

Hejl, S. J. 1994. Human-induced changes in bird populations in coniferous forests in western North America during the past 100 years. Pages 232-246 *in* J. R. Jehl, Jr. and N. K. Johnson (eds.). A century of avifaunal change in western North America. Studies in Avian Biology 15.

Hejl, S. J., J. Verner, and R. P. Balda 1988. Weather and bird populations in true fir forests of the Sierra Nevada, California. Condor 90:561–574.

Hobson, K. A. and J. Schieck. 1999. Changes in bird communities in boreal mixedwood forest: harvest and wildfire effects over 30 years. Ecological Applications 9:849-863.

Horton, S. P. and R. W. Mannan. 1988. Effects of prescribed fire on snags and cavity-nesting birds in southeastern Arizona pine forests. Wildlife Society Bulletin 16:37-44.

Hutto, R. L. 1995. The composition of bird communities following stand-replacement fires in northern Rocky Mountain (U.S.A.) conifer forests. Conservation Biology 9:1041-1058.

Kaufmann, M. R., L. S. Huckaby, P. J. Fornwalt, J. M. Stoker, and W. H. Romme. 2003. Using tree recruitment patterns and fire history to guide restoration of an unlogged ponderosa pine/Douglas-fir landscape in the southern Rocky Mountains after a century of fire suppression. Forestry 76: 231-241.

Kerns, B. K., W. G. Thies, and C. G. Niwa. 2006. Season and severity of prescribed burn in ponderosa pine forests: Implications for understory native and exotic plants. Ecoscience 13:44-55.

King, T. G., M. A. Howell, B. R. Chapman, K. V. Miller, and R. A. Schorr. 1998. Comparisons of wintering bird communities in mature pine stands managed by prescribed burning. Wilson Bulletin 110:570-574.

Knapp, E. E. and J. E. Keeley. 2006. Heterogeneity in fire severity within early season and late season prescribed burns in a mixed-conifer forest. International Journal of Wildland Fire 15:37-45.

Kolb, A. and M. Diekmann. 2005. Effects of life-history traits on responses of plant species to forest fragmentation. Conservation Biology 19:929-938.

Kotliar, N. B., S. Hejl, R. L. Hutto, V. Saab, C. P. Melcher, and M. E. McFadzen. 2002. Effects of fire and post-fire salvage logging on avian communities in conifer dominated forests of the western United States. Pages 49-64 *in* T. L. George and D. S. Dobkin (eds.). Effects of habitat fragmentation on birds in western landscapes: contrasts with paradigms from the eastern United States. Studies in Avian Biology 25.

Landres, P. B., J. Verner, and J. W. Thomas. 1988. Ecological uses of vertebrate indicator species: a critique. Conservation Biology 2:316-327.

Legendre, P. and L. Legendre. 1998. Numerical ecology. Second edition. Elsevier, Amsterdam, The Netherlands.

Lentile, L. B., F. W. Smith, and W. D. Shepperd. 2005. Patch structure, fire-scar formation, and tree regeneration in a large mixed-severity fire in the South Dakota Black Hills, USA. Canadian Journal of Forest Research 35:2875-2885.

Littell, R. C., G. A. Milliken, W. W. Stroup, and R. D. Wolfinger. 1996. SAS System for Mixed Models. SAS Institute, Inc., Cary, NC, USA.

Longley, P. A., M. F. Goodchild, D. J. Maguire, and D. W. Rhind. 2001. Geographic information systems and science, John Wiley and Sons, Chichester, UK.

Lyon, L. J., M. H. Huff, and J. K. Smith. 2000. Fire effects on fauna at landscape scales. Pages 43-49 *in* J. K. Smith (ed.). Wildland fire in ecosystems: effects of fire on fauna. USDA Forest Service, Rocky Mountain Research Station, General Technical Report, RMRS-GTR-42-Volume 1. Ogden, UT, USA.

MacKenzie, D. I. and W. L. Kendall. 2002. How should detection probability be incorporated into estimates of relative abundance? Ecology 83:2387-2393.

Magurran, A. 2004 Measuring biological diversity. Blackwell, Malden, MA, USA.

Maina, G. G. and H. F. Howe. 2000. Inherent rarity in community restoration. Conservation Biology 14:1335-1340.

Manly, B. F. J. 2005. Multivariate statistical methods: a primer. Third edition. Chapman and Hall, Boca Raton, LA, USA.

Marques, F. F. C. and S. T. Buckland. 2004. Covariate models for the detection function. Pages 31-47 *in* S. T. Buckland, D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas (eds.). Advanced distance sampling Oxford University Press, London, UK.

Marzluff, J. M., M. G. Raphael, and R. Sallabanks. 2000. Understanding the effects of forest management on avian species. Wildlife Society Bulletin 28:1132-1143.

McGarigal, K., S. A. Cushman, and S. Stafford. 2000. Multivariate statistics for wildlife and ecology research. Springer-Verlag, New York, NY, USA.

McHugh, C. W. and T. E. Kolb. 2003. Ponderosa pine mortality following fire in northern Arizona. International Journal of Wildland Fire 12:7-22.

McHugh, C. W., T. E. Kolb, and J. L. Wilson. 2003. Bark beetle attacks on ponderosa pine following fire in northern Arizona. Environmental Entomology 32:510-522.

Miller, C. and D. L. Urban. 2000. Modeling the effects of fire management alternatives on Sierra Nevada mixed-conifer forests. Ecological Applications 10:85-94.

Moir, W. H., B. W. Geils, M. A. Benoit, and D. Scurlock. 1997. Ecology of southwestern ponderosa pine forests. Pages 3-27 *in* W. M. Block and D. M. Finch (eds.). Songbird ecology in southwestern ponderosa pine forests: a literature review. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-292, Fort Collins, CO, USA.

Moreira, F, A. Delgado, S. Ferreira, R. Borralho, N. Oliveira, M. Inaci, J. S. Silva, and F. Rego. 2003. Effects of prescribed fire on vegetation structure and breeding birds in young *Pinus pinaster* stands of northern Portugal. Forest Ecology and Management 184:225-237.

Mutch, R. W. 1995. Fighting fire with prescribed fire: a return to ecosystem health. Journal of Forestry 92:31-33.

Neter, J., W. Wasserman, and M. H. Kutner.1990. Applied linear statistical models. Third edition. Richard D. Irwin, Homewood, IL, USA.

Norvell, R. E., F. P. Howe, and J. R. Parrish. 2003. A seven-year comparison of relative-abundance and distance-sampling methods. Auk 120:1013-1028.

Noss, R. F., P. Beier, W. W. Covington, R. E. Grumbine, D. B. Lindenmayer, J. W. Prather, F. Schmiegelow, T. D. Sisk, and D. J. Vosick. 2006. Recommendations for integrating restoration ecology and conservation biology in ponderosa pine forests of the southwestern United States. Restoration Ecology 14:4-10.

Poff, N. L. 1997. Landscape filters and species traits: towards mechanistic understanding and prediction in stream ecology. Journal of the North American Benthological Society 16:391-409.

Pollet, J. and P. N. Omi. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire 11:1-10.

Poole, A. (ed.). 2005. The birds of North American. Published online at: http://bna.birds.cornell.edu/BNA/. Cornell Laboratory of Ornithology, Ithaca, NY, USA.

Powell, H. D. W., S. J. Hejl, and D. L. Six. 2002. Measuring woodpecker food: a simple method for comparing wood-boring beetle abundance among fire-killed trees. Journal of Field Ornithology 73:130-140.

Rieman, B. and J. Clayton. 1997. Wildfire and native fish: issues of forest health and conservation of sensitive species. Fisheries 22:6-15.

Rosenstock, S. S., D. R. Anderson, K. M. Giesen, T. Leukering, and M. F. Carter. 2002. Landbird counting techniques: current practices and an alternative. Auk 119:46-53.

Rotenberry, J. T., R. J. Cooper, J. M. Wunderle, and K. C. Smith. 1995. When and how are populations limited? The roles of insect outbreaks, fire and other natural pertur- bations. Pages 55-84 *in* T. E. Martin and D. M. Finch (eds.). Ecology and management of Neotropical migratory birds. Oxford University Press, New York, NY, USA.

Royle, J. A., D. K. Dawson, and S. Bates. 2004. Modeling abundance effects in distance sampling. Ecology 85:1591-1597.

Saab, V. A., J. Dudley, and W. L. Thompson. 2004. Factors influencing occupancy of nest cavities in recently burned forests. Condor 106:20-36.

Saab, V. A., L. Bate, J. Lehmkuhl, B. G. Dickson, S. Story, S. Jentsch, and W. M. Block. In Review. Changes in downed woody material and forest structure after prescribed fire in ponderosa pine forests. RMRS Technical Publication.

Sackett, S. S., S. M. Haase, and M. G. Harrington. 1996. Prescribed burning in southwestern ponderosa pine. Pages 178–186 *in* P. F. Ffolliott, L. F. DeBano, M. B. Baker Jr., G. H. Gottfried, G. Solis-Garza, C. B. Edminister, D. G. Neary, and R. H. Hamre (eds.). Effects of fire on Madrean province ecosystems. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-289, Fort Collins, CO, USA.

Sackett, S. S. and S. M. Haase. 1998. Two case histories for using prescribed fire to restore ponderosa pine ecosystems in northern Arizona. Pages 380-389 *in* T. L. Pruden and L.A. Brennan (eds.). Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology CNFerence Proceedings, Number 20. Tall Timbers Research Station, Tallahassee, FL, USA.

Schneider, D. C. 1994. Quantitative ecology: spatial and temporal scaling. Academic Press, New York, NY, USA.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across rocky mountain forests. BioScience 54:661-676.

Seber, G. A. F. 2002. The estimation of animal abundance and related parameters. Second edition. Blackburn Press, Caldwell, NJ, USA.

Sesnie, S. and J. D. Bailey. 2003. Using history to plan the future of old-growth ponderosa pine. Journal of Forestry 101:40-47.

Short, K. C. 2003. Complexity and variation in the effects of low-severity fires on forest biota. Dissertation. University of Montana, Missoula, MT, USA.

Short, K. C. and J. F. Negrón. 2003. Arthropod responses: a functional approach. Pages 286-305 *in* P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington D.C., USA.

Sibley, D. A. 2001. The Sibley guide to bird life and behavior. Alfred A. Knopf, New York, NY, USA.

Sisk, T. D., J. W. Prather, H. M. Hampton, E. N. Aumack , Y. Xu, and B. G. Dickson. 2006. Participatory landscape analysis to guide restoration of ponderosa pine ecosystems in the American Southwest. Landscape and Urban Planning 79:xx-xx.

Smith, E. P. and G. van Belle. 1984. Nonparametric estimation of species richness. Biometrics 40:119-129.

Smucker, K. M., R. L. Hutto, and B. M. Steele. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications 15:1535-1549.

Spiering, D. J. and R. L. Knight. 2005. Snag density and use by cavity-nesting birds in managed stands of the Black Hills National Forest. Forest Ecology and Management 214:40-52.

Swetnam, T. W. and C. H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 11-32 *in* C. D. Allen (ed.). Fire effects in southwestern forests, pp. 11-32. Proceedings of the second La Mesa fire symposium. USDA Forest Service, General Technical Report, RM-GTR286, Fort Collins, CO, USA.

Swetnam, T. W. and J. L. Betancourt. 1990. Fire-southern oscillation relations in the southwestern United States. Science 249:1017-1020.

Swetnam, T. W. and J. L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate 11:3128-3147.

Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9:1189-1206.

Szaro, R. C. and R. P. Balda. 1979. Bird community dynamics in a ponderosa pine forest. Studies in Avian Biology 3.

Szaro, R. C. 1986. Guild management: an evaluation of avian guilds as a predictive tool. Environmental Management 10:681-688.

Thomas, L., J. L. Laake, S. Strindberg, F. C. C. Marques, S. T. Buckland, D. L. Borchers, D. R. Anderson, K. P. Burnham, S. L. Hedley, J. H. Pollard, J. R. B. Bishop, and T. A. Marques. 2005. Distance 5.0. Release 5. Research Unit for Wildlife Population Assessment, University of St. Andrews, UK. Online at: http://www.ruwpa.st-and.ac.uk/distance/.

Thompson, F. R. III, J. R. Probst, and M. G. Raphael. 1995. Impacts of silviculture: overview and management recommendations. Pages 201-219 *in* T. Martin and D. M. Finch (eds.). Ecology and management of Neotropical migratory birds. Oxford University Press, New York, NY, USA.

Tiedemann, A. R., J. O. Klemmedson, and E. L. Bull. 2000. Solution of forest health problems with prescribed fire: are forest productivity and wildlife at risk. Forest Ecology and Management 127:1-18.

Turner, M. G., W. H. Romme, and R. H. Gardner. 1999. Prefire heterogeneity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. International Journal of Wildland Fire 9:21-36.

Turner, M. G., R. H. Gardner, and R. V. O'Neill. 2001. Landscape ecology in theory and practice. Springer-Verlag, New York, NY, USA.

Turner, M. G., W. H. Romme, and D. B. Tinker. 2003. Surprises and lessons from the 1988 Yellowstone fires. Frontiers in Ecology and the Environment 1:351-358.

Van Horne, B. 2005. Avian research in the U.S. Forest Service. Pages 26-29 *in* C. J. Ralph and T. D. Rich (eds.). The state of the art and the state of the science: Partners in Flight in the 21st Century. Third international Partners in Flight CNFerence: a workshop on bird conservation implementation and integration. USDA Forest Service, Pacific Southwest Research Station, General Technical Report PSW-GTR-191, Volume 1, Portland, OR, USA.

van Mantgem, P., M. Schwartz, and M. B. Keifer. 2001. Monitoring fire effects for managed burns and wildfires: coming to terms with pseudoreplication. Natural Areas Journal 21:266-273.

Wagner, M. R., W. M. Block, B. W. Geils, and K. F. Wenger. 2000. Restoration ecology: a new forest management paradigm, or another merit badge for foresters? Journal of Forestry 98: 22-27.

Walther, B. A. and J. L. Moore. 2005. The concepts of bias, precision and accuracy, and their use in testing the performance of species richness estimators, with a literature review of estimator performance. Ecography 28:815-829.

Wheelwright, N. T. 1986. The diet of American Robins: an analysis of U.S. Biological Survey records. Auk 103:710-725.

White, G. C. 2005. Correcting wildlife counts using detection probabilities. Wildlife Research 32:211-216.

Wiens, J. A. 1989. The ecology of bird communities. Volume 2. Processes and variations. Cambridge University Press, Cambridge, UK.

Wikars, L. O. and J. Schimmel. 2001. Immediate effects of fire-severity on soil invertebrates in cut and uncut pine forests. Forest Ecology and Management 141:189-200.

Wolfinger, R. D. 1993. Covariance structure selection in general mixed models. Communications in Statistics: Simulation and Computation 22:1079-1106.

Xu, Y., J. W. Prather, H. M. Hampton, E. N. Aumack, B. G. Dickson, and T. D. Sisk. 2006. Advanced exploratory data analysis for mapping regional canopy cover. Photogrammetric Engineering and Remote Sensing 72:31-38.

Zhang, T., R. Ramakrishnan, and M. Livny. 1996. BIRCH: an efficient data clustering method for very large databases. Pages 103-114 *in* Proceedings of the ACM SIGMOD CNFerence on management of data, Montreal, CA.

Zimmerman, G. T. 2003. Fuels and fire behavior. Pages 126-143 *in* P. Friederici (ed.). Ecological restoration of southwestern ponderosa pine forests. Island Press, Washington, D.C., USA.

Table 2.1. Site descriptions and summary statistics for units occurring on the four study sites used to measure the response of avian communities to prescribed fire in Arizona and New Mexico, 2002-2005. "PF" or "C" next to unit name indicates prescribed fire treatment unit or control unit, respectively. Sites were located on the Kaibab National Forest (KNF), the Coconino National Forest (CNF), the Apache-Sitgreaves National Forest (ASNF), and the Gila National Forest (GNF).

Unit name	Area (ha)	Mean elevation (m)	SD	Mean slope (deg)	SD	Total # of sampling stations	Total # of years monitored (pre/post)	Pre-treatment effort[1]	Post-treatment effort[1]
KNF									
Kendrick (PF)	396	2253.7	5.1	1.98	1.21	40	2/2	240	320
Moritz (C)	359	2195.1	9.5	1.35	0.70	40	2/2	240	320
Beale (C)	127	2245.2	4.2	1.70	0.67	10	2/2	60	80
Total:	882					90		540	720
CNF									
Imax (PF)	405	2121.5	23.6	4.08	2.22	40	1/2	160	320
Buck Mountain (C)	404	2148.7	36.4	6.10	2.38	40	1/2	160	320
Total:	809					80		320	640
ASNF									
Lakeside (PF)	247	2214.4	37.4	5.83	1.69	29	1/2	116	232
Porter Mountain (C)	186	2122.0	8.0	1.90	1.29	20	1/2	80	160
Pat Mullen (C)	99	2227.9	22.5	6.00	4.97	9	1/2	36	72
Total:	532					58		232	464
GNF									
Collins Park (PF)	262	2409.7	41.2	12.08	5.04	25	2/2	175	200
La Jolla (C)	245	2411.6	37.8	8.68	5.17	25	2/2	175	200
Total:	507					50		350	400
Grand Total:	2,730					278		1,442	2,224

[1] Effort calculated as product of total number of sampling stations, total number of pre- or post-treatment years monitored, and total number of visits per sampling station.

Table 2.2. Treatment summaries and statistics for each treatment unit used to measure the response of avian communities to prescribed fire in Arizona and New Mexico, 2002-2005. "NA" indicates information not available. See Table 1 for site acronym definitions.

Ignition date (MM/DD/YY)	Area burned (ha)	Mean temperature (C)	Mean relative humidity (%)	Wind speeds (km/h)	Wind direction	Sky condition	Mean flame lengths (cm)
KNF							
10/27/03	85.4	15.6	60.0	0-11	NA	NA	30-46
11/06/03	101.2	10.0	22.0	0-11	NA	NA	30-46
03/25/04	167.9	15.6	30.0	0-11	NA	NA	30-46
Total:	354.5						
CNF							
09/15/03	101.0	25.9	14.3	NA	NA	NA	15-30
09/18/03	101.0	20.4	16.4	NA	NA	NA	15-30
09/19/03	202.0	23.8	14.7	NA	NA	NA	15-30
Total:	404.0						
ASNF							
10/20/03	20.2	23.3[1]	9.0[2]	0-24	NNW	Clear	30
10/21/03	26.3	22.2[1]	0.0[2]	0-11	NW	Clear	30
10/22/03	12.1	22.2[1]	22.0[2]	0-24	NNW	Clear	30
10/23/03	40.5	20.0[1]	18.0[2]	0-21	NW	Clear	30
10/24/03	16.2	21.1[1]	16.0[2]	2-26	NW	Clear	30
10/27/03	28.3	17.2[1]	9.0[2]	0-16	NW	Cirrus	30
10/28/03	24.3	20.6[1]	9.0[2]	0-16	NW	Clear	30
12/02/03	32.4	13.9[1]	15.0[2]	2-5	NW	Cirrus	30
12/03/03	24.3	18.9[1]	18.0[2]	10-16	S	Clear	30
12/04/03	20.2	12.8[1]	37.0[2]	0-5	NNW	Clear	30
12/05/03	40.5	16.7[1]	23.0[2]	0-8	ESE	Clear	30
12/06/03	48.6	15.6[1]	15.0[2]	2-5	SSW	Clear	30
Total:	333.9						
GNF							
12/03/03	38.8	21.4	11.0	NA	NA	NA	NA
05/06/04	224.2	22.2	27.0	0-8	NW	Cumulus	76
Total:	263.0						

[1] Maximum value reported.

[2] Minimum value reported.

Table 2.3. Factor eigenvalues and rotated sums of squared loadings computed using varimax rotation with Kaiser normalization for the three retained factors used to summarize local-scale forest structure on four sites in Arizona and New Mexico, 2002-2005.

	Initial eigenvalues			Rotated loadings		
Factor	Total	Variance (%)	Cumulative variance (%)	Total	Variance (%)	Cumulative variance (%)
1	3.038	33.78	33.78	2.666	29.63	29.63
2	1.903	21.14	54.90	1.810	20.11	49.74
3	1.057	11.74	66.64	1.522	16.91	66.64
4	0.827	9.19	75.83			
5	0.712	7.91	83.74			
6	0.492	5.46	89.20			
7	0.427	4.74	93.95			
8	0.286	3.18	97.13			
9	0.259	2.87	100.00			

Table 2.4. Communality estimates and rotated factor loadings for the nine variables and three retained factors used to summarize local-scale forest structure on four sites in Arizona and New Mexico, 2002-2005. **Bold** values indicate variable used to synthesize and describe relative importance to one of the three retained habitat factors.

		Factor		
Structure variable	Communality	1	2	3
Tree density	0.691	**0.727**	-0.173	0.364
Basal area	0.602	0.431	0.038	**0.644**
Mean tree height	0.828	0.200	**0.862**	0.212
SD tree height	0.397	**0.560**	0.097	0.273
PIPO dominance	0.813	-0.223	**0.871**	0.060
Large tree richness	0.724	**0.683**	-0.417	0.289
Large snag sum	0.480	**0.685**	-0.073	0.076
Large snag decay	0.754	**-0.783**	-0.217	0.306
Canopy cover	0.709	-0.002	0.203	**0.817**
Cumulative variance (%)[1]		29.63	49.74	66.64

[1] Computed using rotated loadings. See Table 2.3.

Table 2.5. Summary statistics for canopy cover (%), basal area (m^2/ha), and tree density (trees/ha) used to quantify the PatchType variable on the four study sites in Arizona and New Mexico, 2002-2005.

	Total # of sampling stations				Total extent (ha)[1]	Canopy cover		Basal area		Tree density	
PatchType	ASNF	CNF	GNF	KNF		Mean (SD)	Range	Mean (SD)	Range	Mean (SD)	Range
Open	0	21	8	37	641.9	26.1 (18.8)	0–83	10.8 (7.0)	0–40	155.4 (116.4)	0–1279
Less dense	31	56	11	36	1605.9	49.2 (14.3)	0–92	21.0 (6.1)	0–41	342.7 (154.7)	0–1358
More dense	27	3	31	17	1146.2	66.7 (13.9)	0–98	24.3 (4.8)	0–45	624.9 (281.1)	0–1835

[1] Includes 100-m buffer surrounding each unit.

Table 2.6. Fire severity cluster centroid statistics for all (n = 112 of 134) treatment unit sampling stations where five tree damage variables were measured on four sites in Arizona and New Mexico. Low (L), moderate (M), and heavy (H) bole char severity categories are summarized by the frequency (N) and proportion of sampling stations in each category.

Fire severity	# sampling stations	% of total	Crown scorch (proportion)		Crown consumption (proportion)		Maximum bole scorch height (m)		Black bole circumference (proportion)		Bole char severity					
											(L)		(M)		(H)	
			Mean	SD	Mean	SD	Mean	SD	Mean	SD	N	%	N	%	N	%
Low	23	20.5	0.033	0.048	0.000	0.000	1.65	2.85	0.477	0.317	23	100.0	0	0.0	0	0.0
Moderate	69	61.6	0.148	0.206	0.003	0.013	3.04	2.04	0.737	0.258	0	0.0	69	100.0	0	0.0
High	20	17.9	0.364	0.213	0.084	0.223	6.95	6.33	0.890	0.126	0	0.0	3	15.0	17	85.0
Combined	112	100.0	0.163	0.212	0.017	0.098	3.45	3.74	0.711	0.285						

Table 2.7. Significance test results for five tree damage variables used to describe the three fire severity clusters on four sites in Arizona and New Mexico. **Bold** text indicates tree damage variable was statistically significant ($P \leq 0.05$) with respect to mean value in two-step cluster analysis.

Fire severity	# sampling stations	*t*-crit (+/-)	Scorched foliage		Consumed foliage		Maximum bole scorch height		% Black bole circumference		χ^2-crit	Bole char severity	
			t	*P*	*t*	*P*	*t*	*P*	*t*	*P*		χ^2	*P*
Low	23	2.72	**-12.93**	<0.001	--	--	**-3.03**	0.006	**-3.55**	0.002	5.99	**75.90**	<0.001
Moderate	69	2.57	-0.60	0.551	**-8.95**	<0.001	-1.69	0.096	0.85	0.398	5.99	**38.33**	<0.001
High	20	2.76	**4.23**	<0.001	1.34	0.098	**2.47**	0.023	**6.37**	<0.001	5.99	**89.00**	<0.001

Table 2.8. Global model parameter estimates used to evaluate the influence of habitat covariates on four ordinal levels of fire severity: “unburned,” “low,” “moderate,” and “high” at treatment unit sampling stations ($n = 134$) in Arizona and New Mexico. Covariates are ranked from top to bottom by relative importance according to their Akaike weight (w_i).

Variable	β	SE	w_i
PatchType	0.4247	0.1326	0.994
FAC_1	0.1733	0.0832	0.866
PatchRich	0.2249	0.1087	0.723
FAC_2	0.1267	0.0623	0.701
FAC_3	0.1140	0.0849	0.442
Slope	0.1367	0.1036	0.428
Elevation	0.0042	0.1023	0.263
Intercept	0.2302	0.3715	--

Table 2.9. Regression coefficient estimates and statistics for fire severity response to PatchType as a categorical variable and Tukey-Kramer test for all pairwise differences among patch types on the four sites in Arizona and New Mexico.

						Less dense		More dense	
PatchType	*β*	SE	DF	*t*	*P*	*t*	*P*	*t*	*P*
Open	0.7240	0.1412	5	5.13	0.0037	-6.21	0.0016	-8.04	0.0005
Less dense	1.7543	0.1035	5	16.95	<0.0001	--	--	-2.79	0.0837
More dense	2.1835	0.1260	5	17.33	<0.0001	--	--	--	--

Table 2.10. Mean estimates of pre-treatment avian community similarity for 62 species detected on ≥ 1 occasion during 1,442 sampling occasions in Arizona and New Mexico, May-June, 2002-2003. Mean estimates and differences were computed separately for the unburned treatment (U) and control (C) units. **Bold** text indicates *Z*-statistic statistically significant (*P* ≤ 0.05). See Table 1 for site definitions.

Site	U (SE)	C (SE)	U – C (SE)	*Z*	*P*
ASNF	0.67 (0.21)	0.62 (0.26)	0.05 (0.33)	0.159	0.436
CNF	0.62 (0.24)	0.51 (0.29)	0.11 (0.38)	0.304	0.378
GNF	0.52 (0.17)	0.42 (0.19)	0.10 (0.26)	0.387	0.348
KNF	0.44 (0.18)	0.54 (0.18)	-0.10 (0.25)	-0.418	0.337
Mean	0.56 (0.11)	0.52 (0.08)	0.04 (0.10)	0.108	0.456

Table 2.11. Two-step cluster procedure outputs used to automatically determine the number of clusters contained in data for 69 species detected on ≥1 occasion during 3,666 sampling occasions in Arizona and New Mexico, May-June, 2002-2005. The value of Akaike's Information Criterion (AIC) from each subcluster was used to determine the initial number of clusters in the data and distances derived during the hierarchical clustering of subclusters were used to compute distance ratio changes. Large increases in distance ratio changes indicate two subclusters should not be merged. **Bold** text indicates final number of clusters and associated metrics used for inference.

# of clusters	AIC	AIC change[1]	Ratio of AIC changes[2]	Ratio of distance measures[3]
1	943.16	0.00	0.000	0.000
2	821.87	-121.29	1.000	1.525
3	**761.60**	**-60.27**	**0.497**	**1.775**
4	752.09	-9.51	0.078	1.202
5	753.58	1.49	-0.012	1.008
6	755.54	1.95	-0.016	1.200
7	766.50	10.96	-0.090	1.283
8	787.38	20.88	-0.172	1.276
9	815.87	28.48	-0.235	1.110
10	847.09	31.22	-0.257	1.065

[1] Change determined using the AIC value for the previous number of clusters.

[2] Ratio changes are relative to the change for the two cluster solution only.

[3] Ratio of distance measures computed using distances derived during the hierarchical clustering of subclusters (not shown; see Methods).

Table 2.12 Avian cluster centroid statistics (continuous trait variables) for 69 species detected on ≥ 1 occasion during 3,666 sampling occasions in Arizona and New Mexico, May-June, 2002-2005. AM = average mass, MC = minimum clutch size, CB = clutch breadth, LRO = maximum potential lifetime reproductive output, and IN = minimum incubation + nestling period. See Appendix 2.B for cluster definitions.

			AM (grams)		MC (# of eggs)		CB (# of eggs)		LRO (# of eggs)		IN (days)	
Cluster #	# species	% of total	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
1	21	30.4	41.29	33.14	3.33	0.80	4.52	1.60	113.43	77.82	32.62	8.10
2	34	49.3	22.61	19.96	2.65	0.85	2.38	1.30	55.53	28.34	25.44	7.83
3	14	20.3	148.86	41.29	2.93	2.20	2.86	1.17	122.43	133.01	32.79	10.73
Combined	69	100.0	53.91	85.43	2.91	1.25	3.13	1.65	86.72	3.33	29.12	9.18

Table 2.13. Avian cluster significance tests (continuous trait variables) for 69 species detected on ≥1 occasion during 3,666 sampling occasions in Arizona and New Mexico, May-June, 2002-2005. AM = average mass, MC = minimum clutch size, CB = clutch breadth, LRO = maximum potential lifetime reproductive output, and IN = minimum incubation + nestling period. **Bold** text indicates trait variable was statistically significant ($P \leq 0.05$) in two-step cluster analysis. See Appendix 2.B for cluster definitions.

	#	*t*-crit	AM (grams)		MC (# of eggs)		CB (# of eggs)		LRO (# of eggs)		IN (days)	
Cluster #	species	(+/-)	*t*	*P*	*t*	*P*	*t*	*P*	*t*	*P*	*t*	*P*
1	21	2.85	-1.75	0.095	**2.42**	0.025	**3.99**	<0.001	1.57	0.132	1.98	0.062
2	34	2.73	**-9.14**	<0.001	-1.83	0.076	**-3.35**	0.002	**-6.42**	<0.001	**-2.74**	0.010
3	14	3.01	**2.35**	0.035	0.03	0.977	-0.88	0.395	1.00	0.336	1.28	0.223

Table 2.14. Avian cluster centroid statistics and significance tests (categorical trait variables) for 69 species detected on ≥1 occasion during 3,666 sampling occasions in Arizona and New Mexico, May-June, 2002-2005. MS = migration status ($n = 3$ attributes), DI = breeding season diet ($n = 6$), FS = foraging substrate ($n = 5$), and NT = nest type ($n = 8$). Attribute categories for each trait variable are identified in Appendix 2.B. **Bold** text indicates trait variable was statistically significant ($P \leq 0.05$) in two-step cluster analysis. See Appendix 2.B for cluster definitions.

Cluster #	# species	% of total	MS			DI			FS			NT		
			χ^2-crit	χ^2	P	χ^2-crit	χ^2	P	χ^2-crit	χ^2	P	χ^2-crit	χ^2	P
1	21	30.4	8.76	**17.76**	<0.001	14.54	5.32	0.378	12.76	**19.84**	<0.001	17.88	**34.24**	<0.001
2	34	49.3	8.76	**23.47**	<0.001	14.54	10.33	0.066	12.76	**10.7**	0.030	17.88	11.96	0.102
3	14	20.3	8.76	**6.42**	0.040	14.54	**50.89**	<0.001	12.76	7.24	0.124	17.88	**15.13**	0.034

Table 2.15. Candidate detection function models selected using Akaike's Information Criterion (AIC) and AICc differences (ΔAICc) for the 14 most frequently detected species (the "inference set") on the four study sites in Arizona and New Mexico, 2002-2005. Species are arranged by total number of detections. Species codes are defined in Appendix 2.A.

Species code	Total # of detections	Model (g_i)	K[1]	AIC	ΔAIC	$Lik(g_i \mid x)$[2]	w_i[3]	EDR[4]
DEJU	2,713	Year + Site + ObsRank	11	8076.50	0.00	1.000	0.725	46.31
		Year + Site	9	8079.18	2.67	0.263	0.190	46.32
		Year + ObsRank	8	8081.00	4.50	0.105	0.076	46.37
		Year	6	8085.50	9.00	0.011	0.008	46.35
		Site + ObsRank	8	8100.97	24.47	0.000	0.000	46.37
PYNU	1,839	Year + ObsRank	6	5249.68	0.00	1.000	0.979	56.61
		Year + Site + ObsRank	11	5254.71	7.81	0.020	0.020	52.60
		ObsRank	4	5257.49	15.36	0.001	0.000	54.29
		Year + Site	8	5265.03	16.72	0.000	0.000	54.35
		Site + ObsRank	7	5266.40	18.13	0.000	0.000	54.33
WEBL	1,553	Year + Site + ObsRank	11	4392.65	0.00	1.000	0.878	60.41
		Site + ObsRank	8	4396.66	4.01	0.134	0.118	60.44
		Year + ObsRank	8	4404.69	12.04	0.002	0.002	60.55
		ObsRank	5	4406.86	14.21	0.001	0.001	60.49
		Year + Site	9	4407.69	15.04	0.001	0.001	59.74
GRWA	1,468	Year + Site + ObsRank	10	4168.20	0.00	1.000	0.996	61.08
		Site + ObsRank	7	4180.01	11.81	0.003	0.003	60.81
		Year + ObsRank	7	4182.12	13.92	0.001	0.001	61.03
		ObsRank	4	4185.53	17.33	0.000	0.000	60.75
		Year + Site	8	4189.76	21.56	0.000	0.000	60.80
MOCH	1,285	Year + Site	9	3781.86	0.00	1.000	0.324	51.23
		Year	6	3782.45	0.59	0.745	0.241	51.26
		Year + Site + ObsRank	11	3784.69	2.83	0.243	0.079	51.25
		Year + ObsRank	8	3786.43	4.57	0.102	0.033	51.21
		ObsRank	5	3838.79	56.94	0.000	0.000	51.34
WETA	1,251	Year	4	3337.95	0.00	1.000	0.593	79.19
		Year + Site	7	3338.70	0.75	0.688	0.407	78.95
		No covariates	1	3404.09	66.14	0.000	0.000	81.69
		Site	4	3405.47	67.52	0.000	0.000	81.54
YRWA	1,165	Year + Site + ObsRank	10	3364.23	0.00	1.000	0.796	52.03
		Year + Site	8	3366.96	2.74	0.255	0.203	52.03
		Year + ObsRank	7	3376.87	12.65	0.002	0.001	51.38
		Year	5	3379.41	15.18	0.001	0.000	51.27
		Site + ObsRank	7	3441.57	77.34	0.000	0.000	51.19

Table 2.15. Continued.

Species code	Total # of detections	Model (g_i)	K[1]	AIC	ΔAIC	*Lik*(g_i |x)[2]	w_i[3]	EDR[4]
WBNU	1,085	Year + Site + ObsRank	11	3078.39	0.00	1.000	0.741	56.70
		Year	6	3082.16	3.77	0.152	0.113	56.44
		Year + Site	9	3082.97	4.59	0.101	0.075	56.15
		Year + ObsRank	8	3083.08	4.69	0.096	0.071	56.69
		ObsRank	5	3119.41	41.02	0.000	0.000	56.46
PLVI	1,061	Year + Site	7	2823.92	0.00	1.000	0.997	71.33
		Year	5	2835.45	11.53	0.003	0.003	75.57
		Site	4	2892.53	68.61	0.000	0.000	75.33
		No covariates	1	2899.28	75.37	0.000	0.000	75.78
STJA	815	Year + Site	8	2286.16	0.00	1.000	0.450	63.70
		Year	5	2286.52	0.36	0.835	0.376	63.57
		Year + Site + ObsRank	10	2289.28	3.12	0.210	0.095	63.81
		Year + ObsRank	7	2289.70	3.54	0.170	0.077	63.79
		Site + ObsRank	7	2298.85	12.69	0.002	0.001	63.79
AMRO	637	Year + Site	9	1886.56	0.00	1.000	0.328	46.86
		ObsRank	5	1887.29	0.73	0.694	0.228	46.84
		Year	6	1887.36	0.81	0.668	0.219	46.89
		Site	6	1889.63	3.07	0.215	0.071	46.87
		Year + ObsRank	8	1890.16	3.61	0.165	0.054	46.82
CHSP	628	Year + ObsRank	8	1898.73	0.00	1.000	0.814	43.92
		Year + Site + ObsRank	11	1902.17	3.43	0.180	0.146	43.81
		Year + Site	9	1907.00	8.27	0.016	0.013	43.81
		ObsRank	5	1907.19	8.46	0.015	0.012	44.00
		Year	6	1908.28	9.55	0.008	0.007	43.94
WEWP	602	Year	5	1624.64	0.00	1.000	0.595	69.12
		Year + Site	8	1625.97	1.33	0.516	0.306	70.04
		Year + ObsRank	6	1628.23	3.59	0.167	0.099	77.12
		ObsRank	3	1652.46	27.82	0.000	0.000	79.27
		Site + ObsRank	7	1655.55	30.90	0.000	0.000	69.33
HAWO	577	Year + ObsRank	7	1611.24	0.00	1.000	0.413	63.90
		Year + Site	4	1611.56	0.31	0.854	0.353	64.17
		Year	9	1612.61	1.37	0.504	0.208	63.72
		Year + Site + ObsRank	6	1617.01	5.77	0.056	0.023	64.23
		ObsRank	3	1622.68	11.43	0.003	0.001	64.69

[1] Number of parameters in model.

[2] Likelihood of the model, given the data.

[3] Akaike weight.

[4] Effective Detection Radius (in meters).

Table 2.16. Differences in the stratum-level density (individuals/100 ha) estimates for the 14 inference set species on the four study sites in Arizona and New Mexico. Estimates for each species were derived for pre-treatment years (2002, 2003) on the unburned treatment (U) and control (C1) units and separately for post-treatment years (2004, 2005) on the burned treatment (B) and control (C2) units. Species (n = 14) are arranged from most to least abundant. **Bold** text indicates Z-statistic statistically significant ($P \leq 0.05$). Species are arranged by total number of detections. Species codes are defined in Appendix 2.A. See Table 1 for site acronym definitions.

Species code	Site	U – C1 (SE)	B – C2 (SE)	$\hat{\Delta}_s{}^1$ (SE)	Z	P
DEJU	ASNF	-7.45 (12.75)	14.34 (20.42)	21.79 (13.90)	1.568	0.059
	CNF	7.19 (20.91)	23.50 (21.28)	16.31 (17.22)	0.947	0.172
	GNF	17.61(21.24)	27.45 (26.34)	9.84 (19.54)	0.503	0.309
	KNF	-14.69(19.67)	-28.08 (19.95)	-13.39 (16.18)	-0.828	0.203
	Mean	0.66 (14.51)	9.30 (25.52)	8.64 (15.48)	0.558	0.288
PYNU	ASNF	9.39 (23.30)	-17.64 (24.36)	-27.03 (19.46)	-1.389	0.082
	CNF	-0.29 (12.21)	5.09 (12.85)	5.37 (10.23)	0.525	0.300
	GNF	7.58 (18.38)	-11.94 (18.48)	-19.52 (15.05)	-1.297	0.097
	KNF	51.53 (16.36)	17.90 (10.10)	-33.63 (11.10)	**-3.030**	<0.001
	Mean	17.05 (23.37)	-1.65 (16.22)	-18.70 (17.05)	-1.097	0.136
WEBL	ASNF	-7.29 (2.94)	3.24 (6.01)	10.53 (3.87)	**2.724**	0.003
	CNF	17.69 (14.48)	38.90 (22.68)	21.20 (15.53)	1.365	0.086
	GNF	11.71 (9.99)	12.65 (12.29)	0.94 (9.14)	0.103	0.460
	KNF	17.12 (11.13)	35.58 (15.24)	18.46 (10.90)	**1.694**	0.046
	Mean	9.81 (11.71)	22.59 (17.40)	12.78 (9.10)	1.405	0.079
GRWA	ASNF	-6.54 (4.03)	-21.42 (9.36)	-14.88 (8.98)	**-1.658**	0.049
	CNF	27.65 (9.18)	33.88 (10.41)	6.22 (11.26)	0.553	0.291
	GNF	4.15 (7.59)	13.26 (5.29)	9.11 (7.82)	1.165	0.121
	KNF	-5.04 (4.25)	-5.03 (3.48)	0.01 (4.59)	0.002	0.500
	Mean	5.06 (15.79)	5.17 (23.81)	0.12 (10.69)	0.011	0.496
MOCH	ASNF	11.38 (15.96)	4.31 (13.45)	-7.07 (12.05)	-0.586	0.278
	CNF	-3.63 (8.82)	0.69 (8.57)	4.32 (7.10)	0.608	0.271
	GNF	-18.59(14.60)	24.01 (17.98)	42.60 (13.37)	**3.185**	<0.001
	KNF	-11.04(21.42)	-15.62 (6.34)	-4.58 (12.90)	-0.355	0.359
	Mean	-5.47 (12.79)	3.35 (16.28)	8.82 (23.05)	0.383	0.352
WETA	ASNF	5.06 (2.47)	13.81 (4.22)	8.75 (4.24)	**2.062**	0.019
	CNF	2.44 (1.93)	-1.67 (3.43)	-4.11 (3.56)	-1.153	0.125
	GNF	24.86 (6.36)	19.75 (4.81)	-5.11 (6.78)	-0.753	0.227
	KNF	-8.07 (4.49)	1.41 (2.67)	9.48 (4.48)	**2.116**	0.017
	Mean	6.07 (13.75)	8.32 (10.14)	2.25 (7.94)	0.284	0.390
YRWA	ASNF	-0.74 (2.11)	-0.84 (3.24)	-0.10 (3.07)	-0.033	0.488
	CNF	12.93 (8.01)	10.24 (8.13)	-2.69 (8.88)	-0.303	0.382
	GNF	11.09 (7.26)	3.75 (3.60)	-7.34 (6.72)	-1.092	0.138
	KNF	-36.49(19.10)	-14.78 (6.24)	21.71 (16.04)	1.354	0.089
	Mean	-3.30 (22.94)	-0.40 (10.60)	2.89 (12.90)	0.224	0.413

Table 2.16. Continued.

Species code	Site	U – C1 (SE)	B – C2 (SE)	$\hat{\Delta}_s$[1] (SE)	*Z*	*P*
WBNU	ASNF	5.06 (12.67)	14.88 (5.02)	9.83 (10.13)	0.970	0.166
	CNF	16.10 (14.88)	1.55 (6.70)	-14.56 (13.11)	-1.110	0.156
	GNF	5.75 (10.40)	-0.52 (6.93)	-6.28 (10.40)	-0.603	0.274
	KNF	12.57 (9.64)	6.84 (3.93)	-5.73 (8.70)	-0.659	0.255
	Mean	9.87 (5.36)	5.69 (6.87)	-4.18 (10.18)	-0.411	0.341
PLVI	ASNF	-1.46 (1.23)	-0.59 (2.60)	0.87 (2.60)	0.335	0.367
	CNF	-5.85 (3.71)	1.64 (4.68)	7.49 (5.30)	1.414	0.079
	GNF	16.56 (10.77)	17.70 (5.34)	1.14 (8.32)	0.137	0.444
	KNF	-15.22 (8.23)	-7.61 (3.16)	7.60 (7.35)	1.035	0.149
	Mean	-1.49 (13.33)	2.78 (10.70)	4.28 (3.78)	1.132	0.129
STJA	ASNF	-3.23 (7.64)	-2.19 (4.79)	1.04 (7.69)	0.135	0.446
	CNF	-0.40 (3.08)	2.39 (2.82)	2.79 (3.62)	0.772	0.221
	GNF	-2.29 (5.60)	6.99 (4.46)	9.28 (5.53)	**1.678**	0.047
	KNF	1.35 (7.00)	14.08 (3.74)	12.73 (6.27)	**2.031**	0.021
	Mean	-1.14 (2.04)	5.32 (6.94)	6.46 (5.48)	1.179	0.119
AMRO	ASNF	-1.06 (2.56)	-2.16 (3.81)	-1.11 (3.74)	-0.296	0.384
	CNF	-11.34 (7.62)	-13.54 (10.42)	-2.20 (10.66)	-0.207	0.417
	GNF	-5.30 (8.30)	14.03 (10.51)	19.33 (11.17)	**1.730**	0.042
	KNF	-12.34 (7.68)	20.65 (8.80)	33.00 (9.43)	**3.499**	<0.001
	Mean	-7.51 (5.31)	4.74 (15.50)	12.25 (17.01)	0.720	0.236
CHSP	ASNF	0.00 (0)	-1.76 (1.87)	-1.76 (1.82)	-0.964	0.169
	CNF	7.55 (9.08)	-3.97 (12.53)	-11.52 (13.05)	-0.882	0.189
	GNF	32.58 (19.28)	23.61 (12.94)	-8.97 (20.39)	-0.440	0.330
	KNF	-42.82(11.90)	-33.91 (8.29)	8.91 (12.39)	0.719	0.236
	Mean	-0.67 (31.36)	-4.01 (23.54)	-3.33 (9.15)	-0.364	0.359
WEWP	ASNF	1.16 (0.75)	-0.22 (1.08)	-1.38 (1.18)	-1.174	0.121
	CNF	1.28 (2.87)	-0.75 (4.78)	-2.02 (5.18)	-0.391	0.348
	GNF	6.34 (2.86)	4.42 (2.04)	-1.92 (3.21)	-0.598	0.274
	KNF	9.56 (8.69)	8.17 (4.98)	-1.39 (8.99)	-0.155	0.438
	Mean	4.59 (4.10)	2.91 (4.21)	-1.68 (0.34)	**-4.919**	<0.001
HAWO	ASNF	-0.39 (2.54)	14.43 (5.82)	14.82 (5.03)	**2.947**	0.002
	CNF	-1.50 (2.15)	1.97 (4.25)	3.47 (3.97)	0.874	0.192
	GNF	5.16 (3.30)	20.59 (7.13)	15.43 (6.72)	**2.296**	0.011
	KNF	2.67 (1.69)	3.16 (2.52)	0.49 (2.52)	0.195	0.421
	Mean	1.48 (3.02)	10.04 (9.00)	8.55 (7.69)	1.112	0.131

[1] Difference in differences. See Methods. This is also the "effect size."

Table 2.17. Cumulative Akaike weights (w_+) for assessing the relative importance of habitat covariates used to predict the point-level response ($\hat{\Delta}^*_j$) of density (individuals/100 ha) to prescribed fire for 14 inference species on four sites in Arizona and New Mexico. Weights for each covariate were computed by summing the Akaike weights across all ($n = 256$) possible models in which that variable occurred. Species are arranged from most to least abundant. Species codes are defined in Appendix 2.A.

Species code	Treatment	Elevation	Slope	PatchType	PatchRich	FAC_1	FAC_2	FAC_3
DEJU	0.332	0.376	0.298	0.287	0.733	0.437	0.432	0.295
PYNU	0.752	0.593	0.394	0.790	0.311	0.371	0.693	0.418
WEBL	0.561	0.613	0.330	0.293	0.261	0.288	0.267	0.563
GRWA	0.286	0.998	0.446	0.298	0.300	0.583	0.420	0.278
MOCH	0.415	0.502	0.322	0.746	0.276	0.989	0.874	0.313
WETA	0.314	0.842	0.483	0.510	0.264	0.580	0.348	0.425
YRWA	0.283	0.991	0.638	0.307	0.393	0.713	0.503	0.461
WBNU	0.342	0.574	0.423	0.324	0.288	0.291	0.770	0.273
PLVI	0.322	0.638	0.681	0.342	0.316	0.352	0.712	0.387
STJA	0.642	0.673	0.370	0.338	0.413	0.471	0.261	0.616
AMRO	0.997	0.287	0.427	0.304	0.299	0.321	0.536	0.561
CHSP	0.379	0.999	0.505	0.814	0.285	0.604	0.872	0.829
WEWP	0.263	0.565	0.363	0.643	0.687	0.280	0.266	0.426
HAWO	0.866	0.296	0.582	0.707	0.417	0.897	0.262	0.309
MEAN:	0.482	0.639	0.447	0.479	0.374	0.513	0.515	0.440

Table 2.18. Model-averaged regression coefficients and unconditional variance estimates (SE) for habitat covariates used to predict the point-level response ($\hat{\Delta}^*_j$) of density (individuals/100 ha) to prescribed fire for 14 inference species on four sites in Arizona and New Mexico. Estimates for each covariate were derived using all (n = 256) possible model subsets. Species are arranged from most to least abundant. Species codes are defined in Appendix 2.A.

Species code	Treatment	Elevation	Slope	PatchType	PatchRich	FAC_1	FAC_2	FAC_3
DEJU	2.452 (6.177)	-1.386 (2.877)	-0.600 (2.749)	0.856 (4.078)	10.869 (9.069)	2.615 (4.356)	-2.416 (4.070)	0.796 (2.977)
PYNU	-11.115 (8.726)	-2.942 (3.283)	-1.428 (2.925)	9.411 (6.708)	-1.137 (3.342)	1.407 (3.022)	-4.502 (4.066)	2.047 (3.831)
WEBL	6.981 (8.376)	-3.242 (3.522)	0.789 (2.263)	-0.135 (2.748)	0.021 (2.173)	-0.479 (2.020)	-0.183 (1.553)	-2.772 (3.340)
GRWA	1.120 (4.373)	-12.301 (3.030)	1.569 (2.517)	-0.580 (2.093)	0.617 (2.029)	2.843 (3.237)	-1.186 (2.065)	0.278 (1.393)
MOCH	3.736 (6.592)	-2.203 (3.059)	-0.776 (2.428)	7.862 (6.479)	-0.594 (2.855)	15.170 (4.189)	-7.700 (4.309)	-0.345 (2.763)
WETA	0.731 (2.091)	-2.745 (1.684)	-0.929 (1.386)	-1.222 (1.695)	0.089 (0.848)	1.481 (1.679)	-0.363 (0.845)	-0.595 (1.081)
YRWA	1.407 (5.771)	-14.924 (4.576)	4.598 (4.699)	-0.684 (3.203)	-2.086 (3.959)	6.704 (5.561)	-2.548 (3.510)	-2.090 (3.242)
WBNU	-1.641 (3.870)	2.083 (2.458)	1.297 (2.308)	-0.950 (2.595)	-0.598 (2.343)	-0.378 (1.709)	-4.180 (3.195)	-0.123 (1.535)
PLVI	1.184 (3.170)	-2.396 (2.459)	2.600 (2.457)	0.829 (2.141)	-0.603 (1.700)	0.661 (1.607)	-2.492 (2.178)	-0.838 (1.677)
STJA	3.977 (3.963)	-1.998 (1.887)	0.648 (1.587)	0.609 (2.207)	1.259 (2.236)	-1.227 (1.886)	0.010 (0.837)	-2.105 (2.281)
AMRO	23.186 (6.296)	0.305 (1.735)	-1.499 (2.568)	0.299 (2.950)	0.807 (2.707)	0.788 (2.260)	2.652 (3.387)	3.007 (3.690)
CHSP	-2.582 (5.172)	-15.064 (3.449)	3.206 (4.530)	12.771 (8.797)	0.709 (3.095)	4.140 (4.612)	-7.731 (0.045)	-9.045 (5.986)
WEWP	0.088 (1.152)	-0.818 (0.993)	0.378 (0.901)	1.966 (1.994)	2.057 (1.902)	0.058 (0.649)	0.033 (0.635)	0.582 (1.063)
HAWO	5.688 (3.312)	-0.103 (0.720)	1.189 (1.399)	2.389 (2.133)	-0.870 (1.515)	3.236 (1.705)	-0.014 (0.596)	-0.006 (0.909)

Table 2.19. Regression coefficients for the point-level response ($\hat{\Delta}^*_j$) of avian density (individuals/100 ha) to four levels of fire severity on four study sites in Arizona and New Mexico. Species are arranged from most to least abundant. Species codes are defined in Appendix 2.A.

Severity level	PYNU		WEBL		STJA		AMRO		HAWO	
	β	SE	β	SE	β	SE	β	SE	β	SE
Unburned	9.537	4.704	20.690	5.030	-3.893	2.252	-0.992	4.307	8.337	1.825
Low	1.908	11.340	15.460	9.996	1.170	5.638	15.350	9.500	15.650	3.983
Moderate	-6.956	6.888	27.430	6.790	-1.434	3.359	20.230	6.066	14.790	2.559
High	14.870	12.320	34.030	10.920	-0.501	6.101	-3.221	10.390	16.560	4.358

Table 2.20. Values of Akaike's Information Criterion (corrected for small sample size: AICc) and AICc differences (ΔAICc) for three model forms and an intercept only model used to predict the point-level response ($\hat{\Delta}^*_j$) of five avian species to four levels of fire severity on four study sites in Arizona and New Mexico. Models used for inference determined by values of ΔAICc ≤ 2. Species are arranged from most to least abundant. Species codes are defined in Appendix 2.A.

	PYNU		WEBL		STJA		AMRO		HAWO	
Model form	AICc	ΔAICc	AICc	ΔAICc	AICc	ΔAICc	AICc	ΔAICc	AICc	ΔAICc
Linear	433.4	1.10	316.8	0.30	50.3	1.60	319.4	4.90	-172.1	0.50
Log normal	432.9	0.60	317.1	0.60	50.2	1.50	318.1	3.60	-172.6	0.00
Polynomial	432.3	0.00	318.6	2.10	52.2	3.50	314.5	0.00	-170.5	2.10
Intercept only	432.5	0.20	316.5	0.00	48.7	0.00	319.9	5.40	-169.2	3.40

Table 2.21. Candidate detection function models selected using Akaike's Information Criterion (AIC) and AICc differences (ΔAICc) for the three avian clusters identified on the four study sites in Arizona and New Mexico, 2002-2005. See Table 1 for site definitions and Appendix 2.B for cluster definitions.

Cluster #	Total # of detections	Model (g_i)	K[1]	AIC	ΔAIC	$Lik(g_i \mid x)$[2]	w_i[3]	EDR[4]
1	7,802	Year + Site + ObsRank	10	22742.71	0.00	1.000	0.806	53.07
		Year + Site	8	22745.68	2.97	0.227	0.183	53.07
		Year	5	22752.65	9.94	0.007	0.006	53.10
		Year + ObsRank	7	22752.82	10.11	0.006	0.005	53.09
		Site + ObsRank	7	22823.18	80.47	0.000	0.000	53.04
2	8,457	Year + Site + ObsRank	11	23951.61	0.00	1.000	1.000	58.15
		Year + Site	9	24016.36	64.75	0.000	0.000	58.22
		Year + ObsRank	8	24038.75	87.14	0.000	0.000	58.07
		Year	6	24090.92	139.31	0.000	0.000	58.11
		Site + ObsRank	8	24182.70	231.09	0.000	0.000	58.13
3	4,357	Year + Site + ObsRank	11	12815.72	0.00	1.000	0.944	49.50
		Year + Site	9	12821.40	5.68	0.058	0.055	49.59
		Year + ObsRank	8	12829.16	13.44	0.001	0.001	49.49
		Year	6	12835.49	19.77	0.000	0.000	49.55
		Site + ObsRank	8	12875.17	59.45	0.000	0.000	49.55

[1] Number of parameters in model.

[2] Likelihood of the model, given the data.

[3] Akaike weight.

[4] Effective Detection Radius (in meters).

Table 2.22. Differences in the stratum-level density (individuals/100 ha) estimates by cluster for pre-treatment years (2002, 2003) on the unburned treatment (U) and control (C1) units and separately for post-treatment years (2004, 2005) on the burned treatment (B) and control (C2) units on the four study sites in Arizona and New Mexico. **Bold** text indicates *Z*-statistic statistically significant ($P \leq 0.05$). See Table 1 for site acronym definitions.

Cluster #	Site	U – C1 (SE)	B – C2 (SE)	$\hat{\Delta}_s$[1] (SE)	*Z*	*P*
1	ASNF	-1.74 (33.45)	32.91 (36.03)	34.65 (28.38)	1.221	0.111
	CNF	20.01 (34.09)	45.79 (37.21)	25.78 (28.38)	0.885	0.188
	GNF	26.77 (39.30)	82.94 (44.64)	56.17 (29.14)	1.636	0.051
	KNF	83.29 (33.97)	81.62 (26.73)	-1.67 (34.34)	-0.067	0.472
	Mean	32.08 (36.24)	60.82 (26.73)	28.73 (24.96)	1.200	0.115
2	ASNF	33.86 (11.86)	6.24 (24.05)	-27.62 (15.48)	**-1.784**	0.038
	CNF	15.66 (30.50)	-5.21 (50.82)	-20.87 (34.22)	-0.610	0.271
	GNF	96.88 (26.44)	79.69 (25.55)	-17.19 (21.23)	-0.810	0.209
	KNF	-70.98 (34.84)	-32.44 (21.46)	38.54 (23.62)	1.631	0.052
	Mean	18.85 (69.27)	12.07 (47.91)	-6.78 (30.52)	-0.222	0.413
3	ASNF	-5.95 (18.29)	11.30 (21.97)	17.25 (16.50)	1.045	0.149
	CNF	-1.25 (20.90)	17.09 (21.69)	18.34 (17.39)	1.054	0.146
	GNF	2.47 (21.40)	48.83 (26.55)	46.36 (19.29)	**2.355**	0.009
	KNF	12.89 (45.07)	-20.45 (22.96)	-33.34 (29.20)	-1.142	0.127
	Mean	2.04 (8.01)	14.19 (28.38)	12.15 (33.19)	0.366	0.085

[1] Difference in differences. See Methods. This is also the "effect size."

Table 2.23. Cumulative Akaike weights (w_+) for assessing the relative importance of the treatment and habitat covariates used to predict the point-level response ($\hat{\Delta}^*_j$) of avian cluster densities to prescribed fire on four sites in Arizona and New Mexico. Weights for each covariate were computed by summing the Akaike weights across all ($n = 256$) possible models in which that variable occurred. See Appendix 2.B for cluster definitions.

Cluster #	Treatment	Elevation	Slope	PatchType	PatchRich	FAC_1	FAC_2	FAC_3
1	0.451	0.865	0.319	0.550	0.265	0.682	0.686	0.296
2	0.260	0.999	0.466	0.367	0.350	0.586	0.567	0.467
3	0.333	0.973	0.306	0.954	0.307	0.648	0.880	0.394
MEAN:	0.348	0.946	0.364	0.624	0.308	0.639	0.711	0.386

Table 2.24. Model-averaged regression coefficients and unconditional variance estimates (SE) for each treatment and habitat covariate used to predict the point-level response ($\hat{\Delta}^*_j$) of avian cluster densities (individuals/100 ha) to prescribed fire on four sites in Arizona and New Mexico. Estimates for each covariate were derived using all (n = 256) possible model subsets. See Appendix 2.B for cluster definitions.

Cluster #	Treatment	Elevation	Slope	PatchType	PatchRich	FAC_1	FAC_2	FAC_3
1	11.287 (17.875)	-16.455 (9.649)	1.192 (6.282)	10.740 (13.506)	-0.656 (6.587)	12.268 (11.326)	-11.643 (10.735)	0.009 (6.123)
2	0.851 (14.766)	-46.803 (10.781)	5.005 (7.645)	4.173 (9.567)	3.363 (7.632)	9.302 (10.469)	-7.244 (8.673)	-5.272 (8.245)
3	4.599 (11.372)	-19.115 (6.944)	1.245 (4.694)	27.650 (12.249)	2.016 (6.037)	9.207 (9.043)	-17.206 (9.163)	-3.390 (6.962)

Table 2.25. Regression coefficients for the point-level response ($\hat{\Delta}^*_j$) of avian cluster density to four levels of fire severity on four study sites in Arizona and New Mexico. See Appendix 2.B for cluster definitions.

Severity level	Cluster 1		Cluster 2		Cluster 3	
	β	SE	β	SE	β	SE
Unburned	0.3462	0.1342	0.5918	0.3038	0.0708	0.1514
Low	0.4467	0.2944	0.6310	0.3857	0.0402	0.2773
Moderate	0.4467	0.1886	0.8216	0.3413	0.4088	0.1984
High	0.7089	0.3220	0.9226	0.4079	0.3280	0.3041

Table 2.26. Cumulative Akaike weights (w_+) and regression coefficients used to evaluate the influence of year, treatment unit, area, effort, and observer, and site effects on five estimators of avian community richness, rarity, evenness, and similarity at the stratum level ($n = 28$). Site was modeled as a random effect. Effort × ObsRank computed as the standardized product of effort and the sum of observer rank values for all detections.

	Chao		Jackknife		Fisher's α		Simpson		Chao-Jaccard	
Effect	w_i	β (SE)	w_i	β (SE)	w_i	β (SE)	w_i	β (SE)	w_i	β (SE)
Year	0.870	1.87 (0.99)	0.823	1.91 (1.18)	0.586	0.19 (0.22)	0.573	-0.13 (0.37)	0.215	-0.01 (0.01)
Treatment unit	0.258	-0.32 (0.86)	0.264	-0.39 (1.01)	0.732	-0.45 (0.38)	0.542	-0.20 (0.56)	0.612	-0.04 (0.04)
Area	0.396	0.01 (0.01)	0.371	0.01 (0.01)	0.388	-0.00 (0.01)	0.575	0.01 (0.01)	0.234	-0.00 (0.01)
Effort × ObsRank	0.360	0.22 (0.48)	0.436	0.37 (0.58)	0.416	-0.06 (0.12)	0.535	-0.07 (0.26)	0.968	0.03 (0.01)

Table 2.27. Mean and overall differences ($\hat{\Delta}_s$) in stratum-level estimates of avian community richness, rarity, evenness, and similarity for 74 species detected on ≥ 1 occasion during 3,666 sampling occasions in Arizona and New Mexico, May-June, 2002-2005. Mean estimates were computed for pre-treatment years (2002, 2003) on the unburned treatment (U) and control (C1) units and separately for post-treatment years (2004, 2005) on the burned treatment (B) and control (C2) units. **Bold** text indicates *Z*-statistic statistically significant ($P \leq 0.05$). See Table 1 for site definitions.

Estimator	Site	U (SE)	C1 (SE)	B (SE)	C2 (SE)	$\hat{\Delta}_s$[1] (SE)	*Z*	*P*
Chao	ASNF	21.39 (1.98)	23.91 (0.98)	33.17 (1.90)	36.94 (1.79)	-1.25 (1.97)	-0.631	0.264
	CNF	31.59 (2.86)	36.02 (2.00)	36.36 (2.25)	37.90 (1.64)	2.89 (2.58)	1.120	0.131
	GNF	33.16 (1.89)	29.07 (1.23)	35.26 (1.30)	31.56 (1.31)	-0.39 (1.68)	-0.229	0.409
	KNF	27.02 (0.62)	31.92 (1.55)	31.28 (1.58)	35.57 (1.60)	0.61 (1.62)	0.377	0.352
	Mean	28.29 (5.29)	30.23 (5.09)	34.02 (2.25)	35.49 (2.79)	0.47 (1.78)	0.262	0.397
Jackknife	ASNF	23.02 (3.55)	26.03 (3.58)	35.83 (3.50)	40.09 (3.73)	-1.26 (4.14)	-0.303	0.382
	CNF	33.64 (4.61)	38.56 (4.56)	39.02 (3.66)	40.52 (3.16)	3.42 (4.67)	0.732	0.232
	GNF	35.66 (4.41)	30.87 (1.26)	38.06 (3.26)	33.47 (2.95)	-0.20 (3.67)	-0.055	0.485
	KNF	28.83 (2.31)	34.64 (3.21)	33.48 (3.10)	39.36 (3.62)	-0.08 (3.57)	-0.021	0.492
	Mean	30.29 (5.63)	32.52 (5.35)	36.59 (2.47)	38.36 (3.30)	0.47 (2.04)	0.232	0.409
Fisher's α	ASNF	4.24 (0.46)	5.31 (0.55)	6.69 (0.42)	7.96 (0.48)	-0.20 (0.55)	-0.361	0.359
	CNF	6.03 (0.53)	7.64 (0.64)	6.26 (0.34)	6.96 (0.37)	0.91 (0.56)	1.626	0.052
	GNF	7.65 (0.53)	7.12 (0.53)	7.37 (0.44)	7.19 (0.48)	-0.35 (0.57)	-0.609	0.271
	KNF	5.74 (0.39)	6.47 (0.41)	5.88 (0.35)	6.62 (0.37)	0.00 (0.44)	0.000	0.500
	Mean	5.91 (1.40)	6.64 (1.00)	6.55 (0.64)	7.18 (0.57)	0.09 (0.57)	0.159	0.436
Simpson	ASNF	9.00 (0.53)	9.46 (0.75)	10.93 (0.39)	11.74 (0.62)	-0.35 (0.68)	-0.517	0.301
	CNF	13.32 (0.77)	17.61 (0.91)	12.48 (0.33)	16.40 (0.41)	0.37 (0.75)	0.491	0.312
	GNF	15.48 (0.91)	15.02 (0.99)	15.92 (0.70)	12.23 (0.76)	3.24 (0.98)	**3.302**	<0.001
	KNF	14.51 (0.69)	14.65 (0.56)	12.10 (0.52)	12.29 (0.59)	-0.05 (0.69)	-0.073	0.472
	Mean	13.08 (2.86)	14.18 (3.41)	12.86 (2.15)	13.16 (2.17)	0.80 (1.65)	0.486	0.312

Table 2.27. Continued.

Estimator	Site	U (SE)	C1 (SE)	B (SE)	C2 (SE)	$\hat{\Delta}_s$[1] (SE)	Z	P
Chao-Jaccard	ASNF	0.67 (0.21)	0.62 (0.26)	0.71 (0.14)	0.58 (0.15)	0.08 (0.23)	0.349	0.363
	CNF	0.62 (0.24)	0.51 (0.29)	0.78 (0.12)	0.69 (0.12)	-0.03 (0.24)	-0.105	0.456
	GNF	0.52 (0.17)	0.42 (0.19)	0.55 (0.14)	0.48 (0.16)	-0.03 (0.19)	-0.131	0.448
	KNF	0.44 (0.18)	0.54 (0.18)	0.54 (0.18)	0.60 (0.15)	0.04 (0.20)	0.223	0.412
	Mean	0.56 (0.11)	0.52 (0.08)	0.64 (0.12)	0.59 (0.09)	0.02 (0.05)	0.348	0.363

[1] Overall difference, or “difference in differences.” See Methods. This is also the “effect size.”

Table 2.28. Estimates of mean similarity between points for 74 species detected on ≥ 1 occasion during 3,666 sampling occasions in Arizona and New Mexico, May-June, 2002-2005. For treatment (T) and control (C) units, mean similarity was computed using Chao's abundance-based Jaccard similarity index and using the average number of detections in pre- and post-treatment periods. For test of differences (T – C), **bold** text indicates *Z*-statistic statistically significant ($P \leq 0.05$). See Table 1 for site definitions.

Site	T (SE)	C (SE)	T - C (SE)	*Z*	*P*
ASNF	0.633 (0.033)	0.635 (0.039)	-0.001 (0.051)	-0.027	0.488
CNF	0.750 (0.029)	0.640 (0.038)	0.110 (0.048)	**2.275**	0.011
GNF	0.752 (0.033)	0.663 (0.036)	0.089 (0.049)	**1.830**	0.034
KNF	0.650 (0.030)	0.689 (0.027)	-0.039 (0.040)	-0.967	0.166
Mean	0.696 (0.064)	0.657 (0.025)	0.040 (0.071)	0.558	0.288

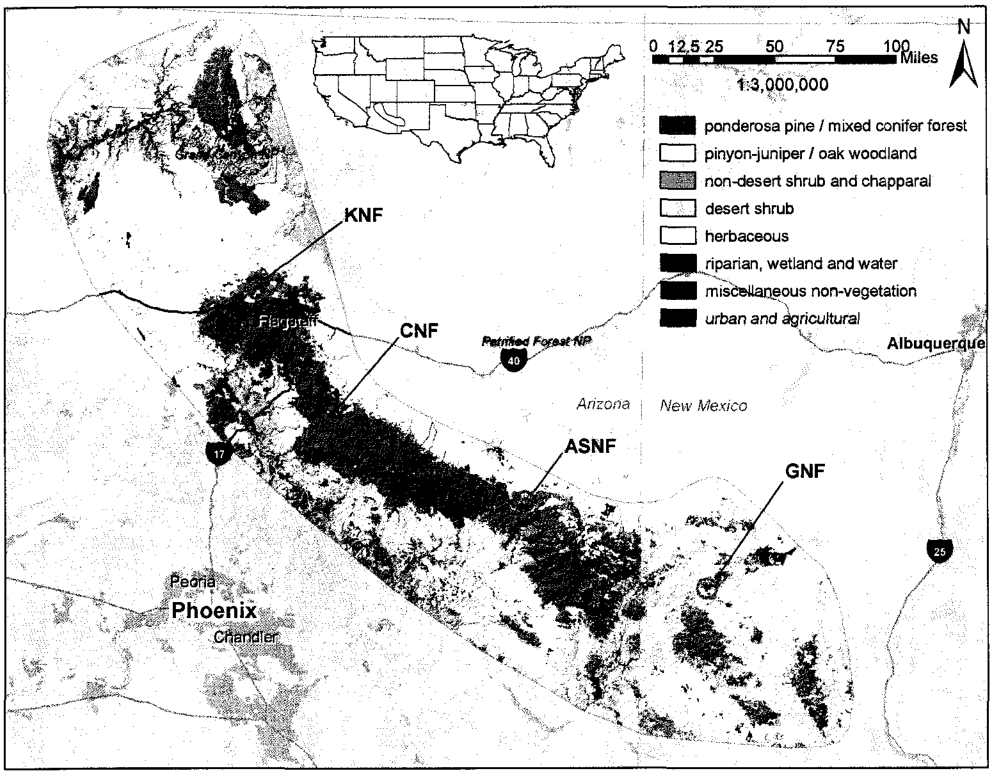

Figure 2.1. Locations of the four study sites (orange circles) in Arizona and New Mexico used to evaluate avian community response to prescribed fire treatments in ponderosa pine forests on the Apache-Sitgreaves National Forest (ASNF), Coconino National Forest (CNF), Gila National Forest (GNF), and Kaibab National Forest (KNF), 2002–2005.

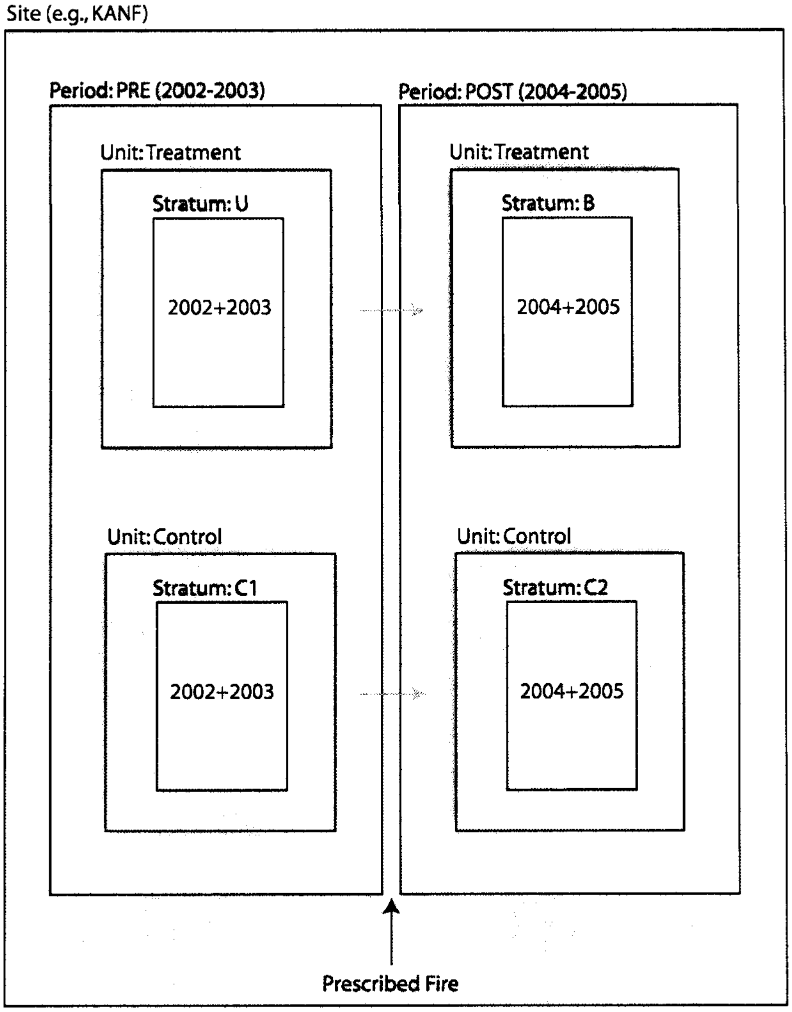

Figure 2.2. Schematic diagram of the Before-After/Control-Impact (BACI) study design used to evaluate avian community response to prescribed fire treatments in ponderosa pine forests on four study sites in Arizona and New Mexico, 2002–2005. "PRE" and "POST" indicate pre- and post-treatment periods, respectively. Stratum levels are indicated by U: unburned treatment unit; C1: pre-treatment control units; B: burned treatment unit; and C2: post-treatment control units.

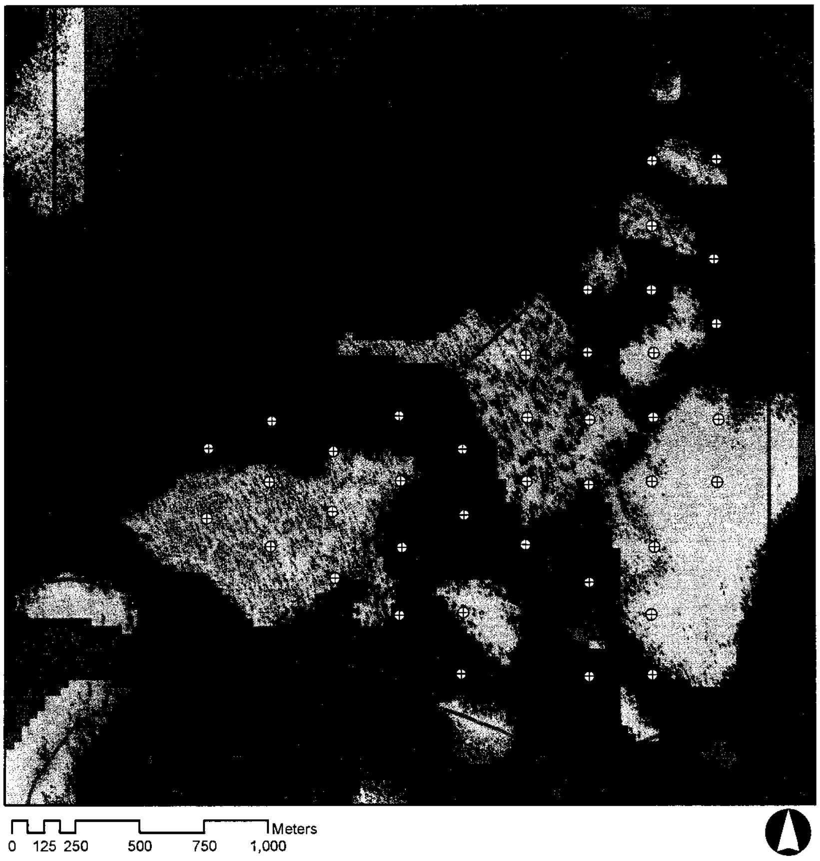

Figure 2.3. Map detailing the occurrence of three patch types and sampling stations ($n = 40$ points) on the Kendrick prescribed fire-treatment unit (including 100-m buffer), Kaibab National Forest, AZ. Patch types are identified as "open" (yellow), "less dense" (light green), and "more dense" (dark green). Digital orthophoto provided for reference. The southeast corner of the Moritz control unit and northern edge of the Beale control unit are visible.

Appendix 2.A. Description and occurrence information for all (n = 82) species detected during avian community surveys in Arizona and New Mexico, May-June, 2002-2005. Site codes are A=ASNF, C=CNF, G=GNF, and K=KNF.

Species code	Common name	Latin name	Total # of detections	Site(s) detected	Period(s) Detected
ACWO	Acorn Woodpecker	*Melanerpes formicivorus*	10	A,G	3
AMCR	American Crow	*Corvus brachyrhynchos*	11	G,K	3
AMKE	American Kestrel	*Falco sparverius*	23	C,K	3
AMRO	American Robin	*Turdus migratorius*	1,067	A,C,G,K	3
ANHU	Anna's Hummingbird	*Calypte anna*	1	K	2
ATFL	Ash-throated Flycatcher	*Myiarchus cinerascens*	112	A,C,G,K	3
BCHU	Black-chinned Hummingbird	*Archilochus alexandri*	1	C	1
BHCO	Brown-headed Cowbird	*Molothrus ater*	385	A,C,G,K	3
BHGR	Black-headed Grosbeak	*Pheucticus melanocephalus*	306	A,C,G,K	3
BRCR	Brown Creeper	*Certhia americana*	324	A,C,G,K	3
BTYW	Black-throated Gray Warbler	*Dendroica nigrescens*	392	A,C,G	3
BTHU	Broad-tailed Hummingbird	*Selasphorus platycercus*	286	A,C,G,K	3
BTPI	Band-tailed Pigeon	*Columba fasciata*	30	A,G	3
BUSH	Bushtit	*Psaltriparus minimus*	97	A,C,G	3
CAKI	Cassin's Kingbird	*Tyrannus vociferans*	20	K	2
CHSP	Chipping Sparrow	*Spizella passerina*	944	A,C,G,K	3
CLNU	Clark's Nutcracker	*Nucifraga columbiana*	346	C,G,K	3
COFL	Cordilleran Flycatcher	*Empidonax occidentalis*	198	A,C,G,K	3
COHA	Cooper's Hawk	*Accipiter cooperii*	2	K	2
CONI	Common Nighthawk	*Chordeiles minor*	67	A,C,G,K	3
CORA	Common Raven	*Corvus corax*	990	A,C,G,K	3
DEJU	Dark-eyed Junco	*Junco hyemalis*	3,469	A,C,G,K	3
DOWO	Downy Woodpecker	*Picoides pubescens*	7	A,C,G	3
DUFL	Dusky Flycatcher	*Empidonax oberholseri*	4	C	2
FLOW	Flammulated Owl	*Otus flammeolus*	2	C,K	2
GAQU	Gambel's Quail	*Callipepla gambelii*	4	G	2
GHOW	Great-horned Owl	*Bubo virginianus*	6	C	2
GRFL	Gray Flycatcher	*Empidonax wrightii*	364	A,C,G,K	3
GRVI	Gray Vireo	*Vireo vicinior*	8	A,G	2
GRWA	Grace's Warbler	*Dendroica graciae*	1,994	A,C,G,K	3
GTTO	Green-tailed Towhee	*Pipilo chlorurus*	4	G	2
HAWO	Hairy Woodpecker	*Picoides villosus*	795	A,C,G,K	3
HETA	Hepatic Tanager	*Piranga flava*	91	A,C,G	3
HETH	Hermit Thrush	*Catharus guttatus*	512	A,C,G,K	3
HOFI	House Finch	*Carpodacus mexicanus*	4	G,K	2
HOWR	House Wren	*Troglodytes aedon*	4	C,G,K	2
JUTI	Juniper Titmouse	*Baeolophus ridgwayi*	7	C,K	3
LASP	Lark Sparrow	*Chondestes grammacus*	139	C,K	3
LEGO	Lesser Goldfinch	*Carduelis tristis*	26	C,G,K	3
MOBL	Mountain Bluebird	*Sialia currucoides*	18	C,K	3
MOCH	Mountain Chickadee	*Parus gambeli*	1,931	A,C,G,K	3
MODO	Mourning Dove	*Zenaida macroura*	801	A,C,G,K	3
NOFL	Northern Flicker	*Colaptes auratus*	710	A,C,G,K	3
NOGO	Northern Goshawk	*Accipiter gentilis*	26	A,K	3

Appendix 2.A. Continued.

Species code	Common name	Latin name	Total # of detections	Site(s) detected	Period(s) detected
NOPO	Northern Pygmy Owl	*Glaucidium gnoma*	4	C,K	3
OLWA	Olive Warbler	*Peucedramus taeniatus*	95	A,C,G,K	3
OSFL	Olive-sided Flycatcher	*Contopus cooperi*	17	A,C,G	3
PEFA	Peregrine Falcon	*Falco peregrinus*	1	C	2
PIJA	Pinyon Jay	*Gymnorhinus cyanocephalus*	318	C,G,K	3
PISI	Pine Siskin	*Carduelis pinus*	53	A,C,G,K	3
PLVI	Plumbeous Vireo	*Vireo plumbeus*	1,366	A,C,G,K	3
PUMA	Purple Martin	*Progne subis*	3	C	2
PYNU	Pygmy Nuthatch	*Sitta pygmaea*	2,086	A,C,G,K	3
RBNU	Red-breasted Nuthatch	*Sitta canadensis*	8	A,C,K	2
RCKI	Ruby-crowned Kinglet	*Regulus calendula*	2	G	1
RECR	Red Crossbill	*Loxia curvirostra*	224	A,C,K	3
RFWA	Red-faced Warbler	*Cardellina rubrifrons*	100	A,C,G	3
ROWR	Rock Wren	*Salpinctes obsoletus*	7	C,K	2
RTHA	Red-tailed Hawk	*Buteo jamaicensis*	10	K	3
RUHU	Rufous Hummingbird	*Selasphorus rufus*	2	G	2
SAPH	Say's Phoebe	*Sayornis saya*	1	C	2
SPTO	Spotted Towhee	*Pipilo maculatus*	54	A,G	3
STJA	Steller's Jay	*Cyanocitta stelleri*	1,505	A,C,G,K	3
TOSO	Townsend's Solitaire	*Myadestes townsendi*	104	A,C,G,K	3
TTWO	Three-toed Woodpecker	*Picoides tridactylus*	10	C,G,K	2
TUVU	Turkey Vulture	*Cathartes aura*	10	C,K	3
VESP	Vesper Sparrow	*Pooecetes gramineus*	220	G,K	3
VGSW	Violet-green Swallow	*Tachycineta thalassina*	354	A,C,G,K	3
VIWA	Virginia's Warbler	*Vermivora virginiae*	95	A,C,G,K	3
WAVI	Warbling Vireo	*Vireo gilvus*	28	A,C,G	3
WBNU	White-breasted Nuthatch	*Sitta carolinensis*	1,725	A,C,G,K	3
WEBL	Western Bluebird	*Sialia mexicana*	1,856	A,C,G,K	3
WEME	Western Meadowlark	*Sturnella neglecta*	14	K	3
WETA	Western Tanager	*Piranga ludoviciana*	2,024	A,C,G,K	3
WEWP	Western Wood-Pewee	*Contopus sordidulus*	1,003	C,K	3
WISA	Williamson's Sapsucker	*Sphyrapicus thyroideus*	10	A,G	2
WITU	Wild Turkey	*Meleagris gallopavo*	40	A,C,K	3
WIWA	Wilson's Warbler	*Wilsonia pusilla*	1	G	2
WSJA	Western Scrub Jay	*Aphelocoma californica*	21	K	3
WTSW	White-throated Swift	*Aeronautes saxatalis*	3	C,K	1
YEWA	Yellow Warbler	*Dendroica petechia*	1	C	1
YRWA	Yellow-rumped Warbler	*Dendroica coronata*	1,461	A,C,G,K	3
UNID[3]			1,123	A,C,G,K	3
		Total:	32,497		

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[1] Includes all flyovers and incidental observations between formal distance sampling efforts.

[2] 1: Pre-treatment period only; 2: Post-treatment period only; 3: Both periods.

[3] Unidentified individuals.

Appendix 2.B. Ecological- and life-history data matrix for individual species detected on ≥ 1 occasion (*n* = 69 species and 20,616 detections) during formal distance sampling efforts and 3,666 sampling occasions in Arizona and New Mexico, May-June, 2002-2005. Species are arranged by cluster membership and total number of detections. **Bold** text indicates trait variable was statistically significant ($P \leq 0.05$) with respect to mean value in two-step cluster analysis (see also Tables 18-21). AM = average mass (grams), MS = migration status, DI = breeding season diet, FS = foraging substrate, NT = nest type, MC = minimum clutch size (# of eggs), CB = clutch breadth (# of eggs), LRO = maximum potential lifetime reproductive output (# of eggs), and IN = minimum incubation + nestling period (days). Since BHCO is a nest predator, NT indicated as "None" and LRO and IN not computed. Species codes are defined in Appendix 2.A.

Cluster #[1]	Species code	AM	MS[2]	DI	FS	NT	MC	CB	LRO[3]	IN[4]	Total # of detections
1	PYNU	11	**R**	Insects	**Bark**	**Cavity**	**4**	**5**	72	35	1,839
1	WEBL	28	**R**	Insects	**Air**	**Cavity**	**3**	**5**	96	28	1,553
1	MOCH	11	**R**	Insects	**Foliage**	**Cavity**	**5**	**7**	240	35	1,285
1	WBNU	21	**R**	Insects	**Bark**	**Cavity**	**3**	**7**	90	26	1,085
1	AMRO	77	**R**	Insects	**Ground**	**Cup**	**3**	**4**	273	26	637
1	HAWO	66	**R**	Insects	**Bark**	**Cavity**	**3**	**3**	90	39	577
1	NOFL	111	**R**	Insects	**Ground**	**Cavity**	**3**	**9**	108	36	316
1	BRCR	8	**R**	Insects	**Bark**	**Cavity**	**4**	**4**	32	27	269
1	BUSH	5	**R**	Insects	**Foliage**	**Sack**	**5**	**2**	112	26	82
1	ATFL	27	**L**	Insects	**Bark**	**Cavity**	**3**	**4**	63	29	74
1	SPTO	41	**R**	Insects	**Ground**	**Cup**	**2**	**4**	120	22	40
1	AMKE	115	**R**	Insects	**Ground**	**Cavity**	**3**	**4**	196	59	8
1	MOBL	30	**R**	Insects	**Ground**	**Cavity**	**4**	**4**	112	35	8
1	DOWO	27	**R**	Insects	**Bark**	**Cavity**	**3**	**3**	66	32	7
1	TTWO	66	**R**	Insects	**Bark**	**Cavity**	**2**	**4**	60	33	5
1	ROWR	17	**R**	Insects	**Ground**	**Crevice**	**4**	**4**	128	26	4
1	ACWO	81	**R**	Omnivore	**Bark**	**Cavity**	**3**	**4**	63	41	3
1	JUTI	18	**R**	Insects	**Foliage**	**Cavity**	**3**	**6**	72	30	3
1	RBNU	10	**R**	Insects	**Bark**	**Cavity**	**4**	**3**	49	26	3
1	WISA	48	**S**	Insects	**Bark**	**Cavity**	**3**	**4**	28	33	3
1	PUMA	49	**L**	Insects	**Air**	**Cavity**	**3**	**5**	312	41	1
2	GRWA	**8**	**L**	Insects	**Foliage**	Cup	3	**1**	**56**	**10**	1,468
2	WETA	**28**	**L**	Insects	**Foliage**	Cup	3	**2**	**35**	**26**	1,251
2	YRWA	**12**	**S**	Insects	**Foliage**	Cup	3	**2**	**60**	**22**	1,165
2	PLVI	**17**	**L**	Insects	**Foliage**	Cup	3	**2**	**80**	**28**	1,061
2	CHSP	**12**	**S**	Insects	**Ground**	Cup	2	**3**	**100**	**19**	628
2	WEWP	**13**	**L**	Insects	**Air**	Cup	2	**2**	**32**	**26**	602
2	BTYW	**8**	**L**	Insects	**Foliage**	Cup	3	**2**	**70**	**20**	322
2	GRFL	**13**	**S**	Insects	**Air**	Cup	3	**1**	**64**	**30**	295
2	BHCO	**44**	**S**	Insects	**Ground**	None	1	**6**	**0**	**0**	273
2	VGSW	**14**	**L**	Insects	**Air**	Cavity	4	**2**	**36**	**29**	194
2	BHGR	**42**	**L**	Insects	**Foliage**	Cup	2	**3**	**45**	**23**	191
2	BTHU	**4**	**L**	Nectar	**Flowers**	Cup	2	**0**	**48**	**35**	163
2	COFL	**11**	**L**	Insects	**Air**	Cavity	3	**2**	**30**	**28**	161
2	HETH	**31**	**S**	Insects	**Ground**	Cup	3	**3**	**108**	**24**	153

Appendix 2.B. Continued.

Cluster #[1]	Species code	AM	MS[2]	DI	FS	NT	MC	CB	LRO[3]	IN[4]	Total # of detections
2	VESP	**26**	**S**	Insects	**Ground**	Cup	2	**4**	**126**	**18**	100
2	VIWA	**8**	**L**	Insects	**Ground**	Cup	3	**2**	**60**	**21**	86
2	RFWA	**10**	**L**	Insects	**Foliage**	Cup	3	**1**	**32**	**23**	79
2	OLWA	**11**	**S**	Insects	**Foliage**	Cup	3	**1**	**32**	**20**	66
2	HETA	**38**	**L**	Insects	**Foliage**	Saucer	3	**2**	**45**	**26**	56
2	TOSO	**34**	**S**	Insects	**Air**	Cup	3	**2**	**25**	**21**	55
2	CONI	**62**	**L**	Insects	**Air**	Ground	1	**2**	**27**	**40**	27
2	WAVI	**15**	**L**	Insects	**Foliage**	Cup	3	**2**	**65**	**28**	20
2	CAKI	**46**	**L**	Insects	**Air**	Cup	2	**3**	**120**	**34**	10
2	OSFL	**32**	**L**	Insects	**Air**	Cup	3	**1**	**28**	**35**	8
2	WEME	**101**	**S**	Insects	**Ground**	Cup	3	**4**	**84**	**25**	5
2	DUFL	**10**	**L**	Insects	**Air**	Cup	3	**1**	**32**	**30**	4
2	GTTO	**29**	**S**	Insects	**Ground**	Cup	2	**3**	**70**	**22**	4
2	GRVI	**13**	**L**	Insects	**Foliage**	Cup	3	**2**	**50**	**26**	3
2	RCKI	**7**	**L**	Insects	**Foliage**	Cup	5	**6**	**55**	**24**	2
2	BCHU	**3**	**S**	Insects	**Flowers**	Cup	1	**2**	**72**	**34**	1
2	RUHU	**4**	**L**	Insects	**Flowers**	Cup	1	**2**	**48**	**32**	1
2	SAPH	**21**	**S**	Insects	**Air**	Cup	3	**4**	**63**	**26**	1
2	WTSW	**32**	**S**	Insects	**Air**	Crevice	3	**3**	**30**	**40**	1
2	YEWA	**10**	**L**	Insects	**Foliage**	Cup	3	**3**	**60**	**20**	1
3	DEJU	**20**	**R**	**Seeds**	Ground	**Cup**	3	3	108	21	2,713
3	STJA	**128**	**R**	**Omnivore**	Ground	**Cup**	2	4	96	32	815
3	MODO	**119**	**S**	**Seeds**	Ground	**Saucer**	2	1	558	25	361
3	RECR	**41**	**R**	**Seeds**	Foliage	**Cup**	2	3	20	27	148
3	PIJA	**103**	**R**	**Omnivore**	Ground	**Cup**	3	2	110	37	105
3	LASP	**29**	**S**	**Seed**	Ground	**Cup**	3	3	42	20	72
3	CLNU	**135**	**R**	**Omnivore**	Foliage	**Cup**	2	4	102	34	64
3	PISI	**15**	**S**	**Seeds**	Foliage	**Saucer**	1	4	90	27	24
3	WSJA	**85**	**R**	**Omnivore**	Ground	**Cup**	2	5	210	33	18
3	BTPI	**343**	**L**	**Nuts**	Foliage	**Platform**	1	1	108	43	16
3	LEGO	**13**	**R**	**Seeds**	Foliage	**Cup**	3	3	60	24	13
3	AMCR	**448**	**R**	**Omnivore**	Ground	**Cup**	4	2	84	46	4
3	COHA	**439**	**R**	**Birds**	Air	**Platform**	3	3	78	59	2
3	GAQU	**166**	**R**	**Seeds**	Ground	**Ground**	10	2	48	31	2

[1] Cluster membership number defined by two-step cluster analysis.

[2] R = resident, S = short-distance migrant, L = long-distance migrant.

[3] LRO estimated for BTYW, GRFL, GRWA, HETA, OLWA, PLVI, ROWR, and TTWO. See Results.

[4] IN estimated for BTYW, HETA, and OLWA.

Appendix 2.C. Response and predictor variables used to model the response of fire severity level to habitat covariates and avian species to habitat covariates or fire severity.

Variable name	Scale of inference	Level of measurement	Description
Response variables:			
Fire severity level	Point[1]	Ordinal	Four levels of fire severity derived using TwoStep clustering of ground data on bole char severity, average maximum bole scorch height, average percent of the bole circumference charred, and average proportional crown scorch collected within 10-m radius of each treatment unit point. Levels 0, 1, 2, and 3 defined as "unburned," "low," "moderate," and "high," respectively. Modeled this response as a function of habitat covariates using mixed model.
BACI difference in avian species density ($\hat{\Delta}_s$)	Site[2]	Continuous	At each site, the estimated difference in post- (B – C2) and pre-treatment (U – C1) stratum-level density differences for each of the 14 species in the inference set. This is the among-stratum BACI "difference in differences," or "effect size."
Point-level difference in avian species density ($\hat{\Delta}^*_j$)	Point	Continuous	At each of the 278 points, the estimated difference in post- and pre-treatment (B – U or C2 – C1) point-level density estimates for each of the 14 species in the inference set.
BACI difference in avian cluster density ($\hat{\Delta}_s$)	Site	Continuous	At each site, the estimated difference in post- (B – C2) and pre-treatment (U – C1) stratum-level density differences for each of the three clusters. This is the among-stratum BACI "difference in differences," or "effect size."
Point-level difference in avian cluster density ($\hat{\Delta}^*_j$)	Point	Continuous	At each of the 278 points, the estimated difference in post- and pre-treatment (B – U or C2 – C1) point-level density estimates for each of the three clusters.
Predictor variables:			
FAC_1	Point	Continuous	First of three retained factors resulting from the habitat factor analysis of tree density, basal area mean and SD tree height, ponderosa pine dominance, large tree richness, large snag count and decay, and canopy cover data collected on four 50-m transects around a point. Points with higher values for FAC_1 regression scores were characterized by increased decay levels in the larger snags, higher stem densities, and greater tree species richness.
FAC_2	Point	Continuous	Second of three retained factors resulting from the habitat factor analysis of tree density, basal area mean and SD tree height, ponderosa pine dominance, large tree richness, large snag count and decay, and canopy cover data collected on four 50-m transects around a point. Points with higher values for FAC_2 regression scores were dominated by more open stands of older ponderosa pine.

Appendix 2.C. Continued.

Variable name	Scale of inference	Level of measurement	Description
FAC_3	Point	Continuous	Third of three retained factors resulting from the habitat factor analysis of tree density, basal area mean and SD tree height, ponderosa pine dominance, large tree richness, large snag count and decay, and canopy cover data collected on four 50-m transects around a point. Points with higher values for FAC_3 regression scores were characterized by high values for canopy closure and basal area.
PatchType	Patch[3]	Ordinal	Three patch types derived from digital forest structure data (canopy cover, stem density, and basal area) synthesized using a fuzzy classification algorithm in BoundarySeer®. Types 1, 2, and 3 defined as "open," "less dense," and "more dense" respectively. Used neighborhood function in a GIS to summarize "majority" value within a 100-m radius of each cell on a unit.
PatchRich	Patch	Ordinal	Count of the total number of different PatchType values occurring within a 100-m radius of each cell on a unit. Used neighborhood function in a GIS to summarize this value.
Elevation	Patch	Continuous	Used USGS digital elevation models and a GIS to derive mean value for elevation at each cell on a unit using a neighborhood function (focal mean) and a circular moving window with a 100-m radius. Standardized to mean zero and unit variance.
Slope	Patch	Continuous	Used USGS digital elevation models and a GIS to derive mean value for slope at each cell on a unit using a neighborhood function (focal mean) and a circular moving window with a 100-m radius. Standardized to mean zero and unit variance.
Fire severity level	Point	Ordinal	See above. For models of avian response to fire severity level, evaluated three functional forms of this response using a simple linear model of the ordinal levels of fire severity and two transformations: log-normal and second-degree polynomial.
Treatment	Unit[4]	Ordinal	Quantitative descriptor of "control" or "treatment" unit using 0 and 1, respectively.
Site	Site	Nominal	Qualitative descriptor of site (ASNF, CNF, GNF, KNF) used as random effect in all mixed models.

[1] "Point" is defined as the 0.20-ha area within a 50-m radius of each permanent sampling station.

[2] "Site" is defined as the area that includes both a treatment and control unit(s).

[3] "Patch" is defined as the contiguous area (1-400 ha) surrounding a point location and comprised by a vector of forest attribute and physiographic information.

[4] "Unit" is defined as the "control" or "treatment" area and within which are situated 9-40 points for sampling. Treatment and control units were each monitored for 1-2 pre-treatment years and 2 post-treatment years.

Appendix 2.D1. Photograph detailing forest structural characteristics of a sampling station (Collins Park S6, Gila National Forest, NM) with a high value for the habitat factor 1 (FAC_1) local-scale covariate. This photograph was taken in October, 2002 (azimuth = 270°).

Appendix 2.D2. Photograph detailing forest structural characteristics of a sampling station (Kendrick C2, Kaibab National Forest, AZ) with a high value for the habitat factor 2 (FAC_2) local-scale covariate. This photograph was taken in June, 2002 (azimuth = 90°).

Appendix 2.D3. Photograph detailing forest structural characteristics of a sampling station (Kendrick H1, Kaibab National Forest, AZ) with a high value for the habitat factor 3 (FAC_3) local-scale covariate. This photograph was taken in June, 2002 (azimuth = 360°).

Appendix 2.E. Stratum-level estimates of density ($\hat{D}$; individuals/100 ha), abundance ($\hat{N}$), detection probability ($\hat{p}$), and effective detection radius (EDR) for the 14 inference species on the four study sites in Arizona and New Mexico. Estimates are for pre-treatment years (2002, 2003) on the unburned treatment (U) and control (C1) units and separately for post-treatment years (2004, 2005) on the burned treatment (B) and control (C2) units. Species are arranged from most to least abundant. Species codes are defined in Appendix 2.A. See Table 1 for site acronym definitions. "--" indicates no estimate could be generated.

Species code	Site	Stratum level[1]	Total # of detections	$\hat{D}$	SE	$\hat{N}^2$	SE	$\hat{p}$	SE	EDR
DEJU	ASNF	U	49	46.38	8.26	114.10	20.31	0.286	0.035	53.49
		C1	58	53.83	9.71	53.31	9.61	0.293	0.033	54.17
		B	197	120.35	15.09	296.07	37.13	0.222	0.014	47.14
		C2	176	106.01	13.75	104.95	13.61	0.225	0.015	47.43
	CONF	U	69	62.97	12.62	255.02	51.11	0.219	0.023	46.76
		C1	60	55.78	16.67	225.36	67.36	0.217	0.025	46.54
		B	244	130.24	17.53	527.49	71.00	0.185	0.011	43.03
		C2	209	106.74	12.06	431.23	48.73	0.193	0.012	43.95
	GINF	U	123	112.15	15.23	293.83	39.90	0.197	0.017	44.42
		C1	107	94.54	14.80	231.60	36.27	0.202	0.018	44.99
		B	211	158.38	19.29	414.98	50.54	0.210	0.013	45.80
		C2	183	130.93	17.94	320.78	43.98	0.221	0.015	47.03
	KANF	U	128	83.89	11.72	332.21	46.41	0.200	0.016	44.75
		C1	181	98.58	15.80	125.20	20.04	0.195	0.013	44.11
		B	281	118.39	13.31	468.81	52.69	0.233	0.012	48.26
		C2	437	146.47	14.87	186.01	18.88	0.235	0.010	48.49
PYNU	ASNF	U	102	85.57	17.88	210.51	43.95	0.360	0.029	59.98
		C1	91	76.18	14.94	75.43	14.79	0.362	0.030	60.14
		B	196	96.75	17.52	237.99	43.10	0.308	0.019	55.46
		C2	227	114.39	16.92	113.27	16.74	0.299	0.017	54.65
	CONF	U	55	36.60	8.97	148.23	36.32	0.333	0.037	57.72
		C1	56	36.89	8.29	149.01	33.47	0.333	0.037	57.72
		B	156	47.08	9.81	190.68	39.70	0.363	0.023	60.27
		C2	137	41.99	8.30	169.66	33.54	0.358	0.025	59.81
	GINF	U	84	55.66	14.19	145.84	37.19	0.303	0.029	55.02
		C1	68	48.08	11.68	117.81	28.62	0.284	0.030	53.25
		B	106	56.93	12.94	149.16	33.89	0.328	0.027	57.28
		C2	125	68.87	13.20	168.73	32.33	0.320	0.024	56.57
	KANF	U	135	76.09	15.15	301.30	59.99	0.260	0.020	50.95
		C1	57	24.56	6.17	31.18	7.84	0.274	0.032	52.36
		B	139	44.12	8.38	174.73	33.20	0.344	0.024	58.65
		C2	105	26.22	5.63	33.32	7.16	0.354	0.028	59.50

Appendix 2.E. Continued.

Species code	Site	Stratum level[1]	Total # of detections	$\hat{D}$	SE	$\hat{N}^2$	SE	$\hat{p}$	SE	EDR
WEBL	ASNF	U	0	--	--	--	--	--	--	--
		C1	9	7.29	2.94	7.23	2.93	0.351	0.099	59.22
		B	42	15.53	4.95	38.21	12.17	0.385	0.049	62.01
		C2	29	12.29	3.41	12.16	3.40	0.336	0.054	57.98
	CONF	U	81	47.23	11.61	191.29	47.06	0.355	0.033	59.59
		C1	51	29.54	8.64	119.34	34.91	0.362	0.042	60.18
		B	307	102.75	18.94	416.14	76.69	0.309	0.015	55.55
		C2	201	63.86	12.48	257.98	50.38	0.327	0.019	57.16
	GINF	U	62	32.93	8.19	86.29	21.44	0.344	0.038	58.65
		C1	41	21.22	5.72	51.97	14.02	0.354	0.049	59.52
		B	87	32.47	10.43	85.08	27.35	0.451	0.037	67.14
		C2	58	19.82	6.50	48.54	15.91	0.486	0.047	69.68
	KANF	U	74	31.17	9.36	123.45	37.06	0.340	0.034	58.32
		C1	38	14.05	6.02	17.84	7.67	0.315	0.045	56.14
		B	289	70.49	13.56	279.11	53.70	0.426	0.019	65.25
		C2	184	34.90	6.96	44.32	8.85	0.440	0.025	66.36
GRWA	ASNF	U	6	2.72	1.40	7.00	3.60	0.606	0.160	77.85
		C1	16	9.25	3.78	9.00	3.68	0.475	0.096	68.89
		B	29	14.53	3.59	36.00	8.89	0.274	0.045	52.34
		C2	73	35.94	8.64	36.00	8.65	0.279	0.029	52.79
	CONF	U	114	56.31	7.64	228.00	30.94	0.403	0.030	63.46
		C1	58	28.66	5.10	116.00	20.62	0.403	0.042	63.45
		B	418	119.29	7.10	483.00	28.74	0.349	0.014	59.04
		C2	317	85.41	7.62	345.00	30.77	0.369	0.017	60.76
	GINF	U	39	20.57	5.22	54.00	13.71	0.345	0.048	58.72
		C1	24	16.42	5.51	40.00	13.43	0.266	0.050	51.56
		B	55	22.44	4.48	59.00	11.79	0.390	0.043	62.46
		C2	24	9.18	2.80	22.00	6.72	0.416	0.068	64.51
	KANF	U	41	13.42	2.69	53.00	10.63	0.405	0.052	63.66
		C1	70	18.46	3.29	23.00	4.10	0.402	0.040	63.44
		B	67	13.17	2.45	52.00	9.67	0.506	0.045	71.14
		C2	117	18.20	2.47	23.00	3.12	0.512	0.034	71.53
MOCH	ASNF	U	50	46.32	13.41	113.94	32.97	0.299	0.036	54.68
		C1	38	34.94	8.67	34.58	8.58	0.299	0.041	54.68
		B	131	58.69	10.28	144.38	25.29	0.307	0.023	55.45
		C2	120	54.38	8.67	53.84	8.58	0.303	0.023	55.04
	CONF	U	32	23.39	5.77	94.72	23.38	0.275	0.042	52.43
		C1	37	27.02	6.67	109.15	26.96	0.275	0.039	52.43
		B	87	30.71	6.03	124.38	24.42	0.286	0.026	53.40
		C2	85	30.02	6.09	121.29	24.61	0.287	0.027	53.49
	GINF	U	34	26.25	6.75	68.79	17.67	0.235	0.039	48.49
		C1	51	44.84	12.95	109.86	31.73	0.208	0.029	45.65
		B	115	61.48	15.16	161.07	39.73	0.302	0.024	54.97
		C2	72	37.46	9.66	91.79	23.67	0.308	0.030	55.51
	KANF	U	100	72.43	14.76	286.85	58.46	0.185	0.018	43.04
		C1	135	83.48	15.52	106.02	19.70	0.175	0.015	41.77
		B	60	18.32	3.38	72.56	13.39	0.325	0.035	57.03
		C2	138	33.94	5.36	43.11	6.82	0.327	0.023	57.15

Appendix 2.E. Continued.

Species code	Site	Stratum level[1]	Total # of detections	$\hat{D}$	SE	$\hat{N}$[2]	SE	$\hat{p}$	SE	EDR
WETA	ASNF	U	40	11.24	2.11	28.00	5.26	0.977	0.031	98.82
		C1	22	6.18	1.28	6.00	1.25	0.977	0.042	98.82
		B	156	34.47	2.89	85.00	7.11	0.625	0.035	79.01
		C2	93	20.67	3.08	20.50	3.06	0.622	0.045	78.79
	CONF	U	42	8.53	1.47	35.00	6.04	0.980	0.028	98.97
		C1	30	6.09	1.24	25.00	5.10	0.980	0.033	98.97
		B	120	18.98	2.58	77.00	10.48	0.629	0.038	79.33
		C2	135	20.65	2.25	83.50	9.12	0.651	0.037	80.67
	GINF	U	103	36.87	5.67	96.50	14.85	0.508	0.045	71.29
		C1	35	12.02	2.87	29.50	7.05	0.530	0.081	72.79
		B	121	29.94	4.03	78.50	10.55	0.647	0.037	80.36
		C2	40	10.19	2.63	25.00	6.44	0.628	0.064	79.21
	KANF	U	41	11.26	3.34	44.50	13.21	0.483	0.069	69.49
		C1	83	19.33	3.00	24.50	3.80	0.456	0.046	67.49
		B	87	12.97	2.21	51.50	8.79	0.669	0.045	81.79
		C2	103	11.56	1.50	14.50	1.88	0.710	0.041	84.27
YRWA	ASNF	U	2	1.47	1.34	4.00	3.64	0.373	0.220	61.10
		C1	3	2.21	1.62	2.00	1.47	0.373	0.180	61.10
		B	11	4.88	2.17	12.00	5.34	0.309	0.082	55.60
		C2	13	5.72	2.41	6.00	2.53	0.312	0.076	55.85
	CONF	U	66	36.77	6.52	149.00	26.40	0.357	0.036	59.76
		C1	41	23.84	4.67	96.00	18.79	0.342	0.045	58.50
		B	192	63.99	5.67	259.00	22.95	0.298	0.018	54.63
		C2	168	53.75	5.83	217.00	23.54	0.311	0.020	55.76
	GINF	U	54	26.63	5.57	70.00	14.63	0.369	0.046	60.73
		C1	29	15.54	4.67	38.00	11.41	0.339	0.058	58.26
		B	33	10.50	2.51	28.00	6.70	0.500	0.063	70.71
		C2	21	6.75	2.58	17.00	6.50	0.495	0.078	70.35
	KANF	U	88	70.29	12.78	278.00	50.54	0.166	0.018	40.75
		C1	158	106.78	14.20	136.00	18.09	0.157	0.013	39.62
		B	99	29.28	4.58	116.00	18.15	0.336	0.028	57.99
		C2	187	44.06	4.24	56.00	5.39	0.338	0.020	58.12
WBNU	ASNF	U	49	50.63	9.90	125.00	24.45	0.266	0.033	51.53
		C1	46	45.58	7.90	45.00	7.80	0.277	0.036	52.63
		B	124	34.09	4.10	84.00	10.11	0.499	0.033	70.64
		C2	70	19.21	2.89	19.00	2.85	0.500	0.043	70.71
	CONF	U	65	62.32	12.27	252.00	49.62	0.208	0.024	45.55
		C1	57	46.21	8.42	187.00	34.09	0.245	0.029	49.54
		B	160	41.20	5.12	167.00	20.77	0.386	0.024	62.16
		C2	149	39.65	4.32	160.00	17.42	0.374	0.025	61.14
	GINF	U	35	29.76	7.17	78.00	18.79	0.214	0.034	46.25
		C1	32	24.00	7.53	59.00	18.51	0.243	0.039	49.25
		B	47	19.33	4.07	51.00	10.75	0.387	0.046	62.21
		C2	45	19.85	5.60	49.00	13.83	0.361	0.044	60.06
	KANF	U	50	33.80	7.16	134.00	28.39	0.196	0.026	44.30
		C1	39	21.22	6.45	27.00	8.20	0.195	0.029	44.16
		B	68	16.25	3.05	64.00	12.03	0.416	0.041	64.52
		C2	49	9.40	2.48	12.00	3.16	0.415	0.047	64.40

Appendix 2.E. Continued.

Species code	Site	Stratum level[1]	Total # of detections	$\hat{D}$	SE	$\hat{N}$[2]	SE	$\hat{p}$	SE	EDR
PLVI	ASNF	U	0	--	--	--	--	--	--	--
		C1	3	1.46	1.23	1.00	0.84	0.564	0.233	75.08
		B	23	5.55	1.63	14.00	4.10	0.569	0.084	75.43
		C2	25	6.14	2.03	6.00	1.98	0.559	0.080	74.77
	CONF	U	40	12.32	2.26	50.00	9.16	0.646	0.062	80.38
		C1	59	18.17	2.95	73.00	11.83	0.646	0.051	80.38
		B	227	35.73	3.17	145.00	12.86	0.632	0.026	79.50
		C2	215	34.09	3.44	138.00	13.94	0.627	0.027	79.20
	GINF	U	33	25.11	9.09	66.00	23.91	0.239	0.065	48.89
		C1	7	8.55	5.77	21.00	14.18	0.149	0.082	38.59
		B	56	22.11	4.58	58.00	12.02	0.403	0.043	63.50
		C2	11	4.40	2.74	11.00	6.85	0.398	0.097	63.05
	KANF	U	40	19.66	5.42	78.00	21.52	0.270	0.043	51.95
		C1	90	34.88	6.19	44.00	7.81	0.274	0.029	52.33
		B	72	9.52	2.44	38.00	9.75	0.753	0.045	86.75
		C2	160	17.13	2.00	22.00	2.57	0.743	0.030	86.21
STJA	ASNF	U	48	25.85	4.91	63.50	12.05	0.507	0.052	71.23
		C1	54	29.08	5.85	29.00	5.84	0.507	0.049	71.23
		B	62	19.06	3.06	47.00	7.56	0.429	0.042	65.48
		C2	70	21.25	3.68	21.00	3.64	0.434	0.040	65.89
	CONF	U	12	4.80	2.01	19.50	8.16	0.507	0.105	71.23
		C1	13	5.19	2.34	21.00	9.46	0.507	0.100	71.23
		B	47	10.24	1.82	41.50	7.38	0.450	0.050	67.06
		C2	35	7.84	2.16	31.50	8.67	0.437	0.056	66.14
	GINF	U	35	17.43	3.89	46.00	10.26	0.328	0.049	57.29
		C1	40	19.72	4.03	48.50	9.90	0.332	0.047	57.62
		B	67	23.02	3.27	60.00	8.53	0.428	0.040	65.41
		C2	47	16.03	3.02	39.00	7.36	0.431	0.048	65.67
	KANF	U	55	27.14	5.26	107.50	20.82	0.290	0.035	53.85
		C1	66	25.78	4.62	92.50	16.58	0.293	0.032	54.13
		B	106	25.04	3.08	99.50	12.23	0.445	0.033	66.73
		C2	58	10.96	2.12	39.00	7.54	0.445	0.044	66.74
AMRO	ASNF	U	1	1.14	1.53	2.67	3.58	0.245	0.000	49.44
		C1	2	2.19	2.06	2.00	1.88	0.252	0.160	50.17
		B	6	3.43	1.98	8.33	4.81	0.241	0.090	49.08
		C2	10	5.60	3.25	5.33	3.10	0.246	0.071	49.56
	CONF	U	14	12.07	4.45	49.00	18.05	0.232	0.055	48.15
		C1	27	23.40	6.19	94.33	24.95	0.231	0.040	48.02
		B	66	30.82	7.01	124.67	28.36	0.214	0.024	46.21
		C2	94	44.36	7.71	179.00	31.12	0.212	0.020	45.99
	GINF	U	23	18.46	5.65	48.00	14.70	0.228	0.043	47.75
		C1	31	23.77	6.08	58.33	14.92	0.240	0.038	48.90
		B	50	34.97	8.92	91.33	23.30	0.230	0.029	47.93
		C2	32	20.94	5.56	51.00	13.53	0.246	0.039	49.55
	KANF	U	32	20.51	5.20	81.33	20.62	0.207	0.033	45.53
		C1	66	32.85	5.65	118.00	20.28	0.213	0.024	46.19
		B	104	48.63	7.24	192.33	28.65	0.213	0.019	46.13
		C2	79	27.98	4.99	100.33	17.89	0.225	0.023	47.42

Appendix 2.E. Continued.

Species code	Site	Stratum level[1]	Total # of detections	$\hat{D}$	SE	$\hat{N}$[2]	SE	$\hat{P}$	SE	EDR
CHSP	ASNF	U	0	--	--	--	--	--	--	--
		C1	0	--	--	--	--	--	--	--
		B	1	0.49	0.65	1.00	1.32	0.278	0.054	52.74
		C2	3	2.25	1.76	2.00	1.56	0.183	0.100	42.76
	CONF	U	21	20.93	6.64	85.00	26.95	0.200	0.040	44.68
		C1	13	13.38	6.19	54.00	24.99	0.193	0.049	43.96
		B	109	49.73	7.50	201.00	30.31	0.218	0.019	46.69
		C2	111	53.70	10.04	217.00	40.58	0.206	0.018	45.35
	GINF	U	53	57.29	16.92	150.00	44.30	0.168	0.022	41.02
		C1	23	24.72	9.24	61.00	22.81	0.169	0.033	41.14
		B	51	43.55	10.28	114.00	26.90	0.186	0.024	43.17
		C2	23	19.94	7.86	49.00	19.32	0.184	0.036	42.84
	KANF	U	16	13.47	6.12	53.00	24.10	0.158	0.037	39.69
		C1	87	56.29	10.20	71.00	12.87	0.164	0.017	40.50
		B	10	5.10	2.72	20.00	10.66	0.195	0.057	44.18
		C2	107	39.01	7.83	50.00	10.04	0.218	0.020	46.72
WEWP	ASNF	U	3	1.16	0.75	3.00	1.95	0.716	0.239	84.53
		C1	0	--	--	--	--	--	--	--
		B	9	1.92	0.75	4.50	1.76	0.652	0.143	80.58
		C2	10	2.14	0.78	2.00	0.73	0.650	0.136	80.48
	CONF	U	20	6.39	2.34	26.00	9.52	0.624	0.091	78.95
		C1	16	5.11	1.66	20.50	6.65	0.624	0.102	78.95
		B	72	13.75	3.51	56.00	14.30	0.524	0.044	72.31
		C2	76	14.50	3.25	58.50	13.11	0.524	0.043	72.36
	GINF	U	15	6.99	2.73	18.50	7.21	0.404	0.098	63.28
		C1	1	0.65	0.88	1.50	2.03	0.293	0.040	53.80
		B	24	5.91	1.93	15.50	5.08	0.657	0.080	80.88
		C2	6	1.48	0.65	3.50	1.54	0.654	0.162	80.73
	KANF	U	60	28.93	6.39	114.50	25.28	0.275	0.033	52.45
		C1	62	19.36	5.89	25.00	7.60	0.340	0.042	58.29
		B	129	21.29	3.80	84.50	15.09	0.605	0.034	77.76
		C2	99	13.11	3.22	17.00	4.18	0.604	0.039	77.65
HAWO	ASNF	U	16	6.27	1.70	15.67	4.25	0.776	0.172	87.78
		C1	17	6.66	1.88	6.67	1.89	0.755	0.160	86.72
		B	88	34.13	4.36	84.00	10.73	0.341	0.033	58.32
		C2	51	19.71	3.86	19.67	3.85	0.373	0.050	60.98
	CONF	U	16	4.69	1.42	19.33	5.87	0.645	0.131	80.21
		C1	21	6.19	1.61	25.00	6.51	0.600	0.101	76.92
		B	57	16.11	3.09	65.33	12.54	0.340	0.041	58.26
		C2	49	14.14	2.91	57.00	11.73	0.366	0.050	60.45
	GINF	U	32	11.21	2.88	29.33	7.52	0.519	0.089	72.05
		C1	19	6.05	1.62	14.67	3.92	0.679	0.159	81.70
		B	69	31.58	6.29	82.67	16.47	0.331	0.037	57.50
		C2	23	10.99	3.35	26.67	8.13	0.350	0.070	59.12
	KANF	U	21	4.88	1.50	19.00	5.84	0.539	0.104	73.19
		C1	12	2.21	0.77	3.00	1.05	0.550	0.141	73.97
		B	46	10.32	2.02	40.67	7.94	0.482	0.067	69.20
		C2	40	7.16	1.52	9.00	1.91	0.476	0.071	68.81

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[1] See Table 1 for total effort in each stratum. [2] See Table 1 for total area of each stratum.

Appendix 2.F. Models of the response of 14 inference species to habitat covariates on four sites in Arizona and New Mexico, 2002-2005. Only those models with Akaike weights (w_i) ≥ 0.02, plus global (fully parameterized) and intercept-only models, computed from all possible subsets (n = 256) are shown. Species are arranged from most to least abundant and models for each species are arranged from lowest to highest ΔAICc value. Species codes are defined in Appendix 2.A. "*K*" = total number of model parameters.

Species code	Model	*K*	AICc	ΔAICc	w_i
DEJU	PatchRich	4	623.77	0.00	0.033
	PatchRich + FAC_1	5	623.98	0.21	0.030
	PatchRich + FAC_2	5	624.39	0.62	0.024
	PatchRich + FAC_1 + FAC_2	6	624.52	0.75	0.023
	Intercept only	3	625.39	1.62	0.015
	Global	11	632.59	8.82	0.000
PYNU	Treatment + Elev + PatchType + FAC_2	7	422.90	0.00	0.068
	Treatment + Elev + PatchType + PatchRich + FAC_2	8	424.55	1.66	0.030
	Treatment + Elev + PatchType + FAC_2 + FAC_3	8	424.58	1.68	0.030
	Treatment + Elev + PatchType + FAC_1 + FAC_2	8	424.71	1.81	0.028
	Treatment + Elev + Slope + PatchType + FAC_2	8	424.79	1.90	0.027
	Treatment + PatchType + FAC_2	6	425.02	2.12	0.024
	Treatment + Slope + PatchType + FAC_2	7	425.06	2.17	0.023
	Treatment + Slope + PatchType + FAC_1 + FAC_2	8	425.22	2.32	0.021
	Global	11	429.44	6.54	0.003
	Intercept only	3	432.50	9.60	0.001
WEBL	Treatment + Elev + FAC_3	6	314.69	0.00	0.035
	Treatment + FAC_3	5	315.18	0.50	0.027
	Treatment + Elev	5	315.20	0.51	0.027
	Elev + FAC_3	5	315.36	0.67	0.025
	FAC_3	4	315.60	0.91	0.022
	Intercept only	3	316.55	1.86	0.014
	Global	11	323.67	8.98	0.000

Appendix 2.F. Continued.

Species code	Model	*K*	AICc	ΔAICc	w_i
GRWA	Elev + FAC_1	5	174.12	0.00	0.050
	Elev + FAC_1 + FAC_2	6	174.47	0.35	0.042
	Elev + Slope	5	174.74	0.62	0.036
	Elev	4	174.97	0.85	0.032
	Elev + Slope + FAC_1	6	175.17	1.05	0.029
	Elev + Slope + FAC_1 + 1 FAC_2	7	175.54	1.42	0.025
	Elev + Slope + FAC_2	6	175.61	1.49	0.024
	Elev + PatchType + FAC_1	6	175.76	1.64	0.022
	Elev + PatchRich + FAC_1	6	175.88	1.76	0.021
	Treatment + Elev + FAC_1	6	175.91	1.79	0.020
	Global	11	182.37	8.25	0.001
	Intercept only	3	185.73	11.61	0.000
MOCH	Elev + PatchType + FAC_1 + FAC_2	7	416.75	0.00	0.080
	Treatment + Elev + PatchType + FAC_1 + FAC_2	8	417.51	0.77	0.055
	PatchType + FAC_1 + FAC_2	6	417.52	0.77	0.054
	Treatment + PatchType + FAC_1 + FAC_2	7	418.24	1.50	0.038
	Slope +PatchType + FAC_1 + FAC_2	7	418.39	1.64	0.035
	Elev + PatchType + FAC_1 + FAC_2 + FAC_3	8	418.54	1.79	0.033
	Elev + PatchType + PatchRich + FAC_1 + FAC_2	8	418.64	1.90	0.031
	Elev + Slope + PatchType + FAC_1 + FAC_2	8	418.82	2.07	0.029
	Treatment + Slope + PatchType + FAC_1 + FAC_2	8	418.89	2.14	0.028
	PatchType + FAC_1 + FAC_2 + FAC_3	7	419.03	2.28	0.026
	Treatment + Elev + PatchType + FAC_1 + FAC_2 + FAC_3	9	419.43	2.68	0.021
	Treatment + Elev + PatchType + PatchRich + FAC_1 + FAC_2	9	419.44	2.69	0.021
	Global	11	423.39	6.64	0.003
	Intercept only	3	429.75	13.00	0.000

Appendix 2.F. Continued.

Species code	Model	K	AICc	ΔAICc	w_i
WETA	Elev + PatchType + FAC_1	6	-218.84	0.00	0.034
	Elev + Slope + PatchType + FAC_1	7	-218.57	0.27	0.029
	Elev + PatchType	5	-218.43	0.42	0.027
	Elev + FAC_3	5	-218.32	0.52	0.026
	Global	11	-211.82	7.02	0.001
	Intercept only	3	-211.50	7.34	0.001
YRWA	Elev + Slope + FAC_1 + FAC_2	7	382.27	0.00	0.038
	Elev + Slope + FAC_1 + FAC_2 + FAC_3	8	382.48	0.21	0.034
	Elev + Slope + FAC_1 + FAC_3	7	382.87	0.60	0.028
	Elev + FAC_1 + FAC_2	6	383.02	0.75	0.026
	Elev + Slope + FAC_1	6	383.14	0.87	0.025
	Elev + FAC_1 + FAC_2 + FAC_3	7	383.22	0.95	0.024
	Elev + Slope + PatchRich + FAC_1 + FAC_2	8	383.35	1.08	0.022
	Elev + Slope + FAC_3	6	383.38	1.11	0.022
	Global	11	387.87	5.50	0.002
	Intercept only	3	392.42	10.15	0.000
WBNU	Elev + FAC_2	5	278.78	0.00	0.052
	FAC_2	4	279.96	1.18	0.029
	Slope + FAC_2	5	280.08	1.30	0.027
	Treatment + Elev + FAC_2	6	280.20	1.42	0.025
	Elev + PatchType + FAC_2	6	280.31	1.53	0.024
	Elev + Slope + FAC_2	6	280.50	1.72	0.022
	Elev + FAC_2 + FAC_3	6	280.65	1.87	0.020
	Intercept only	3	283.68	4.90	0.004
	Global	11	289.25	10.47	0.000

Appendix 2.F. Continued.

Species code	Model	*K*	AICc	ΔAICc	w_i
PLVI	Elev Slope + FAC_2	6	31.87	0.00	0.060
	Treatment + Elev + Slope + FAC_2	7	33.43	1.56	0.027
	Elev + Slope + FAC_2 + FAC_3	7	33.44	1.57	0.027
	Elev + Slope + PatchRich + FAC_2	7	33.55	1.68	0.026
	Elev Slope + FAC_1 + FAC_2	7	33.66	1.79	0.024
	Elev + Slope + PatchType + FAC_2	7	33.72	1.85	0.024
	Elev + Slope	8	34.00	2.13	0.021
	PatchType + FAC_2 + FAC_3	5	34.02	2.15	0.020
	Intercept only	3	36.63	4.76	0.005
	Global	11	39.70	7.83	0.001
STJA	Treatment + Elev + FAC_3	6	43.43	0.00	0.022
	Global	11	47.51	4.08	0.004
	Intercept	3	48.68	5.25	0.003
AMRO	Treatment + FAC_2 + FAC_3	6	1196.69	0.00	0.051
	Treatment + FAC_3	5	1197.57	0.88	0.033
	Treatment + FAC_2	5	1197.67	0.98	0.031
	Treatment	4	1197.80	1.10	0.029
	Treatment + Slope + FAC_2 + FAC_3	7	1197.84	1.15	0.029
	Treatment + Slope + FAC_3	6	1198.04	1.35	0.026
	Treatment + PatchRich + FAC_2 +FAC_3	7	1198.40	1.71	0.022
	Treatment + Slope	5	1198.42	1.73	0.022
	Treatment + FAC_1 + FAC_2 + FAC_3	7	1198.44	1.75	0.022
	Global	11	1204.08	7.39	0.001
	Intercept	3	1209.78	13.09	0.000

Appendix 2.F. Continued.

Species code	Model	*K*	AICc	ΔAICc	w_i
CHSP	Elev + PatchType + FAC_1 + FAC_2 + FAC_3	8	1542.21	0.00	0.116
	Elev + Slope + PatchType + FAC_2 + FAC_3	8	1542.82	0.61	0.086
	Treatment + Elev + PatchType + FAC_1 + FAC_2 + FAC_3	9	1543.43	1.22	0.063
	Treatment + Elev + Slope + PatchType + FAC_2 + FAC_3	9	1543.48	1.27	0.062
	Elev + PatchType + FAC_2 + FAC_3	7	1543.78	1.57	0.053
	Elev + Slope + PatchType + FAC_1 + FAC_2 + FAC_3	9	1543.80	1.59	0.053
	Elev + PatchType + PatchRich + FAC_1 + FAC_2 + FAC_3	9	1544.14	1.93	0.044
	Elev + Slope + PatchType + PatchRich + FAC_2 + FAC_3	9	1544.58	2.38	0.036
	Treatment + Elev + Slope + PatchType + FAC_1 + FAC_2 + FAC_3	10	1544.65	2.44	0.035
	Treatment + Elev + PatchType + FAC_2 + FAC_3	8	1545.23	3.02	0.026
	Global	11	1545.30	3.09	0.025
	Treatment + Elev + PatchType + PatchRich + FAC_1 + FAC_2 + FAC_3	10	1545.45	3.24	0.023
	Elev + Slope + PatchType + PatchRich + FAC_1 + FAC_2 + FAC_3	10	1545.45	3.24	0.023
	Intercept	3	1572.79	30.58	0.000
WEWP	Elev + PatchType + PatchRich	6	1111.65	0.00	0.041
	PatchType + PatchRich	5	1111.71	0.06	0.039
	Elev + Slope + PatchType + PatchRich	7	1112.11	0.46	0.032
	Elev + PatchType	5	1113.06	1.41	0.020
	Intercept	3	1114.78	3.13	0.008
	Global	11	1119.85	8.23	0.001

Appendix 2.F. Continued.

Species code	Model	K	AICc	ΔAICc	w_i
HAWO	Treatment + Slope + PatchType + FAC_1	7	-185.97	0.00	0.065
	Treatment + PatchType + FAC_1	6	-185.67	0.30	0.055
	Treatment + PatchType + PatchRich + FAC_1	7	-185.31	0.66	0.046
	Treatment + Slope + PatchType + PatchRich + FAC_1	8	-185.15	0.82	0.043
	Treatment + Slope + FAC_1	6	-184.31	1.66	0.028
	Treatment + Elev + Slope + PatchType + FAC_1	8	-184.14	1.83	0.026
	Treatment + Slope + PatchType + FAC_1 + FAC_3	8	-183.99	1.98	0.024
	Treatment + Slope + PatchType + FAC_1 + FAC_2	8	-183.87	2.10	0.023
	Treatment + Elev + PatchType + FAC_1	7	-183.75	2.23	0.021
	Treatment + PatchType + FAC_1 + FAC_3	7	-183.69	2.28	0.021
	Treatment + PatchType + FAC_1 + FAC_2	7	-183.68	2.29	0.021
	Global	11	-179.14	6.83	0.002
	Intercept	3	-169.22	16.75	0.000

Appendix 2.G. Stratum-level estimates for density ($\hat{D}$; individuals/100 ha), abundance ($\hat{N}$), detection probability ($\hat{p}$), and effective detection radius (EDR) by cluster on the four study sites in Arizona and New Mexico. Estimates are for pre-treatment years (2002, 2003) on the unburned treatment (U) and control (C1) units and separately for post-treatment years (2004, 2005) on the burned treatment (B) and control (C2) units. See Appendix 2.B for cluster definitions and Table 1 for site definitions.

Cluster #	Site	Stratum level[1]	Total # of detections	$\hat{D}$	SE	$\hat{N}$[2]	SE	$\hat{p}$	SE	EDR
1	ASNF	U	224	200.72	27.34	493.76	67.25	0.305	0.017	55.26
		C1	228	202.46	19.27	200.44	19.10	0.307	0.017	55.36
		B	625	291.47	28.71	717.03	70.56	0.292	0.010	54.07
		C2	552	258.56	21.77	255.99	21.55	0.291	0.010	53.92
	CONF	U	296	219.55	25.36	889.20	102.70	0.267	0.013	51.63
		C1	268	199.54	22.79	806.13	92.06	0.265	0.014	51.49
		B	900	321.50	29.26	1302.10	118.49	0.277	0.008	52.59
		C2	783	275.71	22.99	1113.90	92.90	0.280	0.009	52.93
	GINF	U	355	256.25	26.57	671.36	69.62	0.248	0.012	49.78
		C1	321	229.48	28.96	562.21	71.01	0.251	0.012	50.10
		B	569	291.52	35.83	763.79	93.87	0.308	0.011	55.48
		C2	408	208.58	26.64	511.02	65.26	0.309	0.013	55.56
	KANF	U	459	261.64	28.41	1036.10	112.52	0.231	0.010	48.06
		C1	384	178.35	18.62	226.50	23.65	0.227	0.010	47.64
		B	779	240.51	22.18	952.43	87.81	0.320	0.009	56.53
		C2	649	158.89	14.92	201.78	18.95	0.323	0.010	56.81
2	ASNF	U	130	75.02	9.68	184.54	23.84	0.472	0.031	68.67
		C1	69	41.16	6.84	40.73	6.77	0.455	0.042	67.43
		B	388	178.44	15.31	438.95	37.66	0.296	0.013	54.38
		C2	377	172.20	18.55	170.49	18.36	0.298	0.013	54.61
	CONF	U	379	203.89	19.86	825.74	80.43	0.367	0.015	60.58
		C1	340	188.23	23.15	760.46	93.54	0.358	0.016	59.80
		B	1387	424.92	30.04	1720.90	121.50	0.322	0.007	56.74
		C2	1442	430.13	40.99	1737.80	165.61	0.330	0.007	57.48
	GINF	U	424	221.31	20.23	579.83	53.00	0.345	0.015	58.78
		C1	234	124.43	17.03	304.86	41.74	0.338	0.020	58.15
		B	521	178.03	20.79	466.43	54.48	0.461	0.015	67.92
		C2	289	98.34	14.84	240.93	36.33	0.466	0.020	68.30
	KANF	U	366	204.28	23.51	808.95	93.11	0.236	0.012	48.62
		C1	631	275.26	25.71	349.57	32.65	0.242	0.009	49.17
		B	571	134.66	16.12	533.27	63.89	0.417	0.013	64.59
		C2	898	167.10	14.17	212.21	18.00	0.424	0.011	65.11

Appendix 2.G. Continued.

Cluster #	Site	Stratum level[1]	Total # of detections	$\hat{D}$	SE	$\hat{N}$[2]	SE	$\hat{p}$	SE	EDR
3	ASNF	U	113	88.69	11.30	218.19	27.80	0.348	0.027	58.99
		C1	122	94.64	14.38	93.69	14.24	0.352	0.026	59.36
		B	302	151.83	15.38	373.49	37.83	0.272	0.014	52.12
		C2	281	140.53	15.68	139.12	15.54	0.273	0.014	52.29
	CONF	U	84	72.32	12.77	292.90	51.76	0.233	0.023	48.31
		C1	85	73.57	16.54	297.22	66.82	0.232	0.022	48.19
		B	309	143.76	17.54	582.24	71.09	0.214	0.011	46.28
		C2	279	126.67	12.77	511.76	51.59	0.218	0.012	46.73
	GINF	U	171	128.48	13.43	336.60	35.17	0.240	0.017	49.01
		C1	167	126.01	16.66	308.73	40.81	0.239	0.017	48.87
		B	329	192.51	20.35	504.35	53.31	0.272	0.013	52.11
		C2	254	143.68	17.05	352.02	41.78	0.280	0.015	52.89
	KANF	U	366	247.14	33.81	978.64	133.88	0.196	0.009	44.30
		C1	446	234.25	29.80	297.49	37.87	0.202	0.009	44.94
		B	431	160.49	16.16	635.52	64.00	0.266	0.011	51.54
		C2	612	180.94	16.30	229.79	20.70	0.269	0.009	51.85

[1] See Table 1 for total effort in each stratum.

[2] See Table 1 for total area of each stratum.

Appendix 2.H. Models of avian cluster response ($\hat{\Delta}^*_j$) to habitat covariates on four sites in Arizona and New Mexico, 2002-2005. Only those models with Akaike weights (w_i) ≥ 0.02, plus global (fully parameterized) and intercept-only models, computed from all possible subsets ($n = 256$) are shown. Models within clusters are arranged from lowest to highest ΔAICc value. "*K*" = total number of model parameters. See Appendix 2.B for cluster definitions.

Cluster #	Model	*K*	AICc	ΔAICc	w_i
1	Elev + PatchType + FAC_1 + FAC_2	7	1045.46	0.00	0.047
	Elev + FAC_1 + FAC_2	6	1045.80	0.34	0.040
	Treatment + Elev + PatchType + FAC_1 + FAC_2	8	1045.97	0.51	0.037
	Treatment + Elev + FAC_1 + FAC_2	7	1046.17	0.71	0.033
	Elev + PatchType + FAC_2	6	1046.89	1.43	0.023
	Treatment + Elev + PatchType + FAC_2	7	1046.97	1.51	0.022
	Global	11	1052.30	6.84	0.002
	Intercept	3	1053.15	7.69	0.001
2	Elev + FAC_1 + FAC_2	6	984.44	0.00	0.042
	Elev + Slope + FAC_1 + FAC_2	7	985.17	0.73	0.030
	Elev + FAC_1	5	985.24	0.80	0.028
	Elev + FAC_1 + FAC_2 + FAC_3	7	985.57	1.13	0.024
	Elev + PatchRich + FAC_1 + FAC_2	7	985.82	1.38	0.021
	Elev + Slope + FAC_2	6	985.85	1.41	0.021
	Global	11	990.11	5.67	0.003
	Intercept	3	999.61	15.17	0.000
3	Elev + PatchType + FAC_1 + FAC_2	7	899.71	0.00	0.113
	Elev + PatchType + FAC_1 + FAC_2 + FAC_3	8	900.76	1.06	0.067
	Treatment + Elev + PatchType + FAC_1 + FAC_2	8	901.01	1.31	0.059
	Elev + PatchType + PatchRich + FAC_1 + FAC_2	8	901.23	1.52	0.053
	Elev + PatchType + FAC_2	6	901.65	1.95	0.043
	Elev + Slope + PatchType + FAC_1 + FAC_2	8	901.66	1.95	0.043
	Elev + PatchType + FAC_2 + FAC_3	7	901.71	2.00	0.042
	Treatment + Elev + PatchType + FAC_1 + FAC_2 + FAC_3	9	902.16	2.46	0.033
	Elev + PatchType + PatchRich + FAC_1 + FAC_2 + FAC_3	9	902.31	2.60	0.031
	Elev + Slope + PatchType + FAC_2	7	902.53	2.83	0.028
	Treatment + Elev + PatchType + PatchRich + FAC_1 + FAC_2	9	902.61	2.90	0.027
	Elev + Slope + PatchType + FAC_1 + FAC_2 + FAC_3	9	902.78	3.08	0.025
	Treatment + Elev + PatchType + FAC_2	7	902.85	3.15	0.023
	Elev + Slope + PatchType + FAC_2 + FAC_3	8	902.96	3.25	0.022
	Treatment + Elev + PatchType + FAC_2 + FAC_3	8	903.04	3.33	0.021
	Treatment + Elev + Slope + PatchType + FAC_1 + FAC_2	9	903.05	3.34	0.021
	Elev + Slope + PatchType + PatchRich + FAC_1 + FAC_2	9	903.14	3.44	0.020
	Global	11	905.85	6.14	0.005
	Intercept	3	913.90	14.18	0.000