Spatial application of a predictive wildlife occurrence model to assess alternative forest management scenarios in northern Arizona

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ABSTRACT

Wildlife species of conservation concern can present forest managers with a particular challenge when habitat needs appear to be in contrast with other management objectives, particularly fuel reduction to reduce wildfire risk. Proposed actions can be opposed by stakeholders, delaying management activities until a resolution is met. In the southwestern USA, the primary goal of forest management is to reduce the risk of severe wildfire through forest restoration treatments. The USDA Forest Service has designated the northern goshawk (Accipiter gentilis) a management indicator species in this region. However, it has been difficult to achieve a common understanding of goshawk habitat needs among forest stakeholders. We combined two separate and complimentary modeling approaches – a statistically based occurrence model and alternative forest treatment models – to yield predictions about forest management effects on the goshawk in ponderosa pine-dominated forests (Pinus ponderosa) on the Kaibab Plateau, Arizona. Forest treatment models were developed based on USDA Forest Service recommendations for goshawk habitat and post-treatment data from ecological restoration experiments, both of which were also components of forest treatment guidance from the Kaibab Forest Health Focus collaborative planning effort. All treatment alternatives resulted in a 22–26% reduction in estimated goshawk occurrence, but the declines were not uniform across the study area, varied by forest type, and were not as large as the effects of recent and severe wildfire (44% reduction in occurrence). Considering the controversial history of forest management with respect to the goshawk, it is prudent to interpret results from this study in the context of tradeoffs between wildfire risk reduction and wildlife habitat quality that can be effectively evaluated through science-based collaborative assessment and planning. While developed for a specific, high-profile species in the southwestern USA, the approach is applicable to many other species whose occurrence has been monitored over multiple years.

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1. Introduction

The northern goshawk (Accipiter gentilis; hereafter goshawk) is a large and widespread forest raptor that typically nests in dense stands of large trees (Kennedy, 2003). The goshawk is designated a sensitive species and a Management Indicator Species (MIS) by the USDA Forest Service (USFS; Foster et al., 2010), and has been unsuccessfully petitioned for listing under the Endangered Species Act (Goad, 2005). The species consequently has a history of influencing forest policy through species-level planning and subsequent appeals and litigation of these efforts (Peck, 2000). Concerns about declining habitat quality (Crocker-Bedford, 1990, 1995) prompted the USFS to develop guidelines for goshawk habitat management (Management Recommendations for the Northern Goshawk in the Southwestern United States; hereafter, MRNG; Reynolds et al., 1992). These guidelines provide recommendations for ponderosa pine (Pinus ponderosa), mixed conifer, and spruce-fir forests (Abies concolor, Abies lasiocarpa, Picea engelmannii, Picea pungens, Pinus contorta, Pseudotsuga menziesii). They describe a set of conditions in which relatively dense tree stands with lifted, interlocking crowns are maintained in a 12-ha “nest area” and the surrounding 170-ha “post-fledging area” (PFA), with a more open forest in the remainder of the 2430-ha home range, or “foraging area.” The MRNG were incorporated into the USFS Southwestern

Abbreviations: KHFH, Kaibab Forest Health Focus; KNFL&I, Kaibab National Forest Interpretation and Implementation Guide; MRNG, Management Recommendations for the Northern Goshawk in the Southwest United States.

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Region’s forest plans in 1996, an action that spurred multiple re-
responses from researchers and others who found fault with some
of the assumptions and habitat provisions adopted by the USFS
(e.g., Beier and Drennan, 1997; Greenwald et al., 2005). The debate
surrounding forest management in the Southwest has centered on
the unknown effects of tree thinning activities in goshawk territo-
ries, according to the MRNG specifications or otherwise, specific-
ically considering whether the potential for diminished habitat
quality is offset by reductions in the risk of severe wildfire.

Contemporary southwestern ponderosa pine forests have dra-
matically different structural characteristics than those typical
prior to Euro-American settlement (c. 1883; Fulé et al., 1997).
Under “pre-settlement” conditions, the fire regime in this region
consisted of frequent (fire-return interval of two to three years)
low-intensity surface fires that rarely killed entire stands of mature
trees, prevented the buildup of fine fuels, and maintained resis-
tance to active crown fires (Covington et al., 1997). After more than
a century of vigilant fire suppression and industrial timber harvest
practices, however, ponderosa pine-dominated forests in the re-
gion have become more susceptible to stand-replacing crown fire
events, due to higher densities of small trees with interlocking
crowns. Large, high-intensity fires threaten the ecological integrity
of these forests, alter wildlife habitat, and threaten public safety and
property (Sisk et al., 2006). In this context, the restoration of
pre-settlement structural conditions in ponderosa pine forests is
a priority for forest managers and human communities (Omri and
Joyce, 2003; Western Governors’ Association, 2011). Restoration
experiments have provided managers with suitable methods for
determining pre-settlement conditions at particular locations
(Fulé et al., 1997) and information on the relative effects of treat-
ment intensities on forest structure (Fulé et al., 2002a), but none
have addressed the effects of proposed forest restoration activities
on goshawks.

The MRNG propose that prey is the limiting resource for gos-
hawks in the region and that an interspersed mosaic of forest
structural stages should be maintained to provide habitat for key
prey species. Much debate has occurred over the appropriateness
of these provisions (Beier et al., 2008), causing extended delays
in the implementation of the MRNG and thereby preventing expli-
cit evaluation of their impacts on goshawk populations. Mean-
while, the risk of destructive wildfire continues to increase in
areas where fuel reduction (see Agee and Skinner, 2005) or resto-
ration-type forest treatments have not taken place, as illustrated by
the size and severity of recent fires – most notably the
217,740-ha Wallow Fire in 2011, the largest in Arizona history.
The impact of these fires on goshawk habitat is pronounced and
potentially more devastating than the forest treatments that have
recently been proposed in the region (USDA, 2011). Thus, quantifi-
ing the relative effects of such forest treatments, and comparing
them to the effects of severe wildfires that are likely to occur under
a ‘no-action alternative’ constitutes a plausible first step in evalu-
ating forest management trade-offs and advancing goshawk con-
servation within forest management.

For this study, we sought to model changes in forest structure
under alternative management scenarios (i.e., mechanical thinning
treatments that alter forest structure), and estimate the effects of
these treatments on the likelihood of goshawk territory occurrence
in a ponderosa pine-dominated forest of northern Arizona. Our
objectives were to: (1) integrate models of forest structure derived
using remote sensing techniques into a statistical model of gos-
hawk territory occurrence; (2) manipulate these forest structure
variables in a spatially explicit environment to approximate the
impacts of alternative forest treatments (i.e., mechanical thinning);
and (3) implement a spatial model of goshawk territory occurrence
across a ‘digitally treated’ landscape, including a forest area re-
cently burned in a severe wildfire, to estimate and evaluate the
effects of these forest treatments, as compared to the effects of
habitat degradation due to wildfire. By presenting results from this
effort to model actual forest treatment proposals, we directly ad-
dress the tradeoffs among these treatment scenarios, focusing on
the potential effects of both active management and wildfire on
habitat conservation. This approach can be refined with relative
ease and applied over large areas, providing new tools for assessing
post-treatment conditions important to fuel reduction and wildlife
values.

2. Material and methods

2.1. Study area

Our research focused on the North Kaibab Ranger District of the
Kaibab National Forest in northern Arizona (Fig. 1). The ponderosa
pine forest on the western part of the District was identified as a
management priority prior to a recent collaborative assessment and
planning initiative, the Kaibab Forest Health Focus (KFHF, Sisk
et al., 2009). The KFHF, an innovative venture of the Kaibab Na-
tional Forest in partnership with Northern Arizona University,
was tasked with prioritizing areas for treatment and developing
general treatment guidelines. While it did not explicitly address
goshawk habitat needs, participants did recommend that particu-
lar areas of the forest be treated according to the MRNG. Our
73,740-ha analysis area was focused around the ponderosa pine-
dominated forests identified by the KFHF. To assess wildfire effects
similar to what is expected of future fires in the region, we in-
cluded a 12,830-ha adjacent area that was burned primarily at a
high severity in the 2006 Warm Fire (Wimberly et al., 2009). We
buffered the extent of the study area by 1900 m, the radius of an
assumed circular goshawk territory as defined in our territory
occurrence model (see below; Reynolds et al., 2005), in order to
avoid introducing biases within our analysis area. The elevation
range in our study area was 1890–2710 m. The lower elevations
are dominated by piñon-juniper (Pinus edulis-Juniperus spp.) wood-
lands. Ponderosa pine occurs between 2075 and 2500-m elevation,
occasionally interspersed with Gambel oak (Quercus gambelii).
Above 2500 m, aspen (Populus tremuloides), mixed conifer and
spruce-fir forests are most prevalent.

2.2. Forest structure and environmental variables

We developed forest structure and environmental data layers to
use as variables in competing goshawk territory occurrence models
(see Section 2.3). The variables were generated through statistical
imputation techniques that combine forest inventory and remotely
sensed data (more extensive detail on these methods are presented
in Hampton et al., 2011 and Dickson et al., 2014). We used 2006
leaf-on and leaf-off 30-m resolution Landsat Thematic Mapper
imagery and data from 781 USFS Forest Inventory and Analysis
(FIA) plots classified as forest and 579 non-forest points in north-
ern Arizona for the same spatial extent as the goshawk territory
model to develop models of forest structure in a regression tree
analysis. Landsat bands and spectral vegetation indices such as the
normalized difference vegetation index (NDVI) were used as
predictor variables of forest structural attributes summarized for
FIA plots using the Forest Vegetation Simulator (FVS) Central Rock-
ies variant (Dixon, 2002). Predictor variables also included eleva-
tion and terrain information derived from a 30-m digital elevation
model (DEM). We identified relationships among the training data
(FIA plots) and predictor variables using the Random Forest regression tree (Breiman, 2001) package in R statistical
software.
2.3 Goshawk territory occurrence model

The territory center occurrence model used in this analysis estimated the relative likelihood that the territory center for a breeding pair of goshawks is present at a given point (i.e., pixel), as a function of remotely-sensed variables describing forest structural attributes associated with habitat conditions (and fire behavior). We obtained nest location data from multiple agency partners from forested areas across northern Arizona (Apache-Sitgreaves, Coconino, Kaibab, and Tonto National Forests) for the analysis period 1991–2007 (\(n = 895\)). After Reynolds et al. (2005), we calculated territory centroids as the geographic mean of all nests belonging to each nesting pair during the period of analysis. This process yielded 272 estimated goshawk territory center locations. Because the nest locations represented data only on presence, we generated 272 pseudo-absence locations restricted to a forested habitat envelope model (Pearce and Boyce, 2006; Zarnetske et al., 2007).

Our modeling approach leveraged a collaboratively derived multiple logistic regression model of goshawk territory occurrence (Dickson et al., 2014). Through an expert-based process facilitated by the Lab of Landscape Ecology and Conservation Biology at Northern Arizona University, we identified a set of hypotheses used to guide model development. In a series of workshops, four researchers with expertise on goshawks in the Southwest developed competing territory occurrence models, drawing from a list of available environmental variables, and arrived at a single ‘consensus’ model.

Within an information-theoretic framework, we used multimodel inference (all possible model subsets) to compute model-averaged parameter estimates (\(\beta\)) for each expert-defined variable in the consensus model of territory occurrence (Burnham and Anderson, 2002). We used Akaike’s Information Criterion (AIC) to compare this model to an intercept-only (i.e., null) model, and considered a difference (\(\Delta\text{AIC}\)) value <10.0 to suggest a good approximation of the data (Anderson, 2008). To evaluate model fit, we...
calculated the Hosmer-Lemeshow goodness-of-fit statistic (Hosmer and Lemeshow, 2000; test statistic $= \chi^2, \chi = 0.05$). We computed the area under the receiver operating characteristic (ROC) curve to evaluate model classification accuracy and considered ROC values $>0.70$ as indicative of acceptable discrimination (Hosmer and Lemeshow, 2000). Dickson et al. (2014) provide detailed methods used to derive and evaluate the regional model.

The consensus model that emerged from the expert-based process included 11 topographic, forest structure, and spatial trend variables. Because our study area was smaller than the extent of the expert-based model, and was entirely located on the Kaibab Plateau, we did not incorporate a five-parameter trend surface for this analysis. We also excluded a binary variable used to account for the differences in survey efforts and management history between the Kaibab Plateau and woodland vegetation south of the Grand Canyon. Thus, our final occurrence model included five environmental predictor variables: tree canopy bulk density (CBD; in kg/m$^3$), canopy base height (CBH; in linear and quadratic forms, in meters), trees per hectare (TPH), and aspect (i.e., northeastness) (Table 1). We used a geographic information system (GIS; Esri, 2011) to compute descriptive statistics (e.g., mean, standard deviation) for all variables within a circular moving window of area equal to the average goshawk territory size in northern Arizona (1134 ha; Reynolds et al., 2005). We then used the GIS to implement our final occurrence model and derived a spatially explicit response surface constrained to our study area (Fig. 1).

### 2.4. Synthetic home range boundaries

We deemed that modeling the MRNG as an alternative treatment scenario was necessary because of the importance the guidelines have in the region and the historical management struggles that have ensued since the MRNG were adopted. Our aim was to present a method for implementing a spatially explicit MRNG scenario as a starting point for comparison with other plausible alternatives. Because the USFS management guidelines for goshawk habitat are given at the home range scale—approximately twice the size (2150–2450 ha) of a territory (Reynolds et al., 2005), some representation of home range boundaries was necessary in our MRNG treatment scenario. We therefore created synthetic goshawk home ranges within the study area to provide a basis for treatment and a standard unit for comparative analyses. Given the estimate of 146 goshawk breeding pairs on the Kaibab Plateau, Arizona, to make our MRNG treatment scenario a realistic model for the MRNG treatment scenario, we applied the occurrence model for each treatment scenario. We also implemented the occurrence model for areas where the KFHF recommended they be implemented, and we extracted pixels from our restoration scenario using ArcGIS (Fig. 2).

### 2.5. Forest treatment scenarios

We developed forest treatment scenarios for three management alternatives: ecological restoration, MRNG guidelines, and the KFHF recommendations. We designed all scenarios to incorporate changes in the CBD, CBH, and TPH forest structure variables described above. To establish target values for treatment results, we drew on literature that included post-treatment measurements of the variables whenever possible, but also used regression-based estimates from other forest structure datasets when necessary (see Appendix A). All scenarios involved reclassifying the pretreatment forest structure data into four data classes using natural breaks (Coulson, 1987) and implementing percent changes to each data class to simulate how treatments are planned on the ground (i.e., the data classes represent a range of current conditions and trees to thin would be marked according to the management goals) (Table 2). We restricted the scenarios to pixels defined as ponderosa pine forest by the Landfire Project ‘rapid-refresh’ existing vegetation type data (Landfire, 2008).

Our ecological restoration treatment scenario was based directly on post-treatment experimental data (Fulé et al., 2002a; Waltz et al., 2003; Hurteau et al., 2008) and estimates of pre-settlement forest structure (Covington et al., 1997; Fulé et al., 2002b). For our MRNG treatment, we implemented percent changes on Reineke’s stand density index (SDI) for ponderosa pine and quadratic mean diameter (Dq) to meet targets given in the Kaibab National Forest’s Implementation and Interpretation of Management Recommendations for the Northern Goshawk, Version 2.1 (KNFI&I; Higgins, 2005). We then calculated the CBD, CBH, and TPH variables from SDI and Dq using allometric relationships. We computed the mean and standard deviation of each variable by stand to assess how well the MRNG scenario achieved the desired future conditions (DFC) in the KNFI&I. To model the treatment categories identified by the Kaibab Forest Health Focus group, we extracted forest structure pixels for each variable from our MRNG scenario inputs for areas where the KFHF recommended they be implemented, and we extracted pixels from our restoration scenario where the KFHF recommended strategic treatment optimization modeling for fuel reduction (Finnie, 2007).

Using the parameter estimates generated by the pretreatment goshawk territory occurrence model, we applied the occurrence model for each treatment scenario. We also implemented the occurrence model within the boundary of the 2006 Warm Fire to assess effects of severe wildfire on goshawk habitat.

### 2.6. Analysis

We placed 218 random sample points per synthetic home range ($n = 5014$), which provided a large sample size for a complete
representation of the entire study area. We proportionately placed 1329 sample points within the Warm Fire area so that the sample point density was equal in both areas. All sample points were placed a minimum 90 m apart – three pixels wide or the average diameter of a “group” of trees as defined in the KNFI&I. We also classified each final spatial occurrence model – pretreatment (PRET), restoration (REST), MRNG, KFHF – into quartiles. We extracted occurrence and forest structure values from an additional 1250 random sample points in each quartile ($n = 5000$).

We computed descriptive statistics for each home range, quartile, and forest type. To test for broad differences among treatment scenarios, we conducted a one-way analysis of variance (ANOVA) on the predicted occurrence values under each treatment. We conducted a separate ANOVA to test for statistically significant differences in the forest structure values associated with each predicted occurrence quartile (for both ANOVAs, test statistic = F, $z = 0.05$). We also performed post-hoc Tukey-Kramer HSD tests on all possible pairs of treatment scenarios results to determine statistically significant ($z = 0.05$) differences among all pairs of the predicted estimates of occurrence and forest structure variables associated with the highest predicted estimates.

3. Results

3.1. Goshawk territory occurrence model

The AIC value of our occurrence model was 194 units lower than the null model, indicating an excellent approximation of the data, and model fit was acceptable ($\chi^2 = 5.32$, $df = 8$, $P = 0.72$). Our final model also had good classification accuracy (ROC $= 0.83$).

3.2. Forest treatment scenarios

All treatment scenarios reduced the overall predicted goshawk territory occurrence (hereafter ‘occurrence’) across the study area.
The mean estimate of occurrence before treatment was 0.60 (SD = 0.26). The mean estimate was reduced to 0.44 (26.8%) under the restoration and MRNG treatment scenarios, but the restoration landscape was slightly more variable than the MRNG (SD = 0.24 for restoration, SD = 0.22 for MRNG). The KFHF treatment model reduced the mean estimate of occurrence to 0.47 (22.3%, SD = 0.25). The mean estimate in the Warm Fire area (mean = 0.34, SD = 0.26) was 43% lower than in the pretreatment scenario. The Warm Fire had a lower mean predicted occurrence than at least 70% of the synthetic goshawk home ranges under pretreatment conditions or any treatments. Under pretreatment conditions, 78% of the synthetic home ranges had an overall predicted occurrence of at least 0.50. This percentage was 44% under the REST and MRNG treatments, and 57% under the KFHF treatment. The differences in the estimates among pretreatment conditions and the treatment scenarios were statistically significant (F = 509.8, P < 0.001), but Tukey-Kramer HSD test results indicated that the MRNG and REST scenarios were not significantly different from each other. Treatment did not universally result in reduced occurrence; each treatment scenario resulted in some proportion of the study area in which the model predicted zero or positive change in predicted occurrence from pretreatment conditions (Table 3).

In comparison with pretreatment conditions, changes in the distribution of occurrence estimates were pronounced for all treatment scenarios, especially for the Warm Fire area (Fig. 3). Under pretreatment conditions, most of the study area had predicted occurrence above 0.50, and nearly 40% of the sample points with an estimate > 0.75. After treatment, the majority of sample points had more moderate (0.25–0.75) predicted occurrence. The top two quartiles in the MRNG, REST, and KFHF treatments had a 7–28% reduction in average predicted occurrence, compared to pretreatment conditions. In addition, the area encompassed by the top two quartiles (50th percentile) of predicted occurrence shifted from predominantly ponderosa pine forests to mixed conifer and spruce-fir forest areas, for all three treatments.

The distributions of TPH standard deviation and mean CBD values were relatively consistent across pretreatment and all treatment scenarios. Tukey-Kramer HSD test results for differences between forest structure inputs for each treatment by quartile showed that the forest structure variables in the highest quartile did not differ among the three scenarios. In the lower quartiles, at the most two treatment scenarios differed, and in nearly half (44%) of the comparisons, the forest structure variables in each of the treatments were all significantly different (P < 0.05).

4. Discussion

4.1. Goshawk territory occurrence model results

Implementing our goshawk territory occurrence model in conjunction with the alternative forest treatment scenarios allowed us to better understand how sensitive wildlife populations may respond to ongoing forest management actions at a landscape-scale. Useful insight from this applications-focused research emerged from comparisons showing the landscape-scale effects of each forest treatment alternative and the effects of wildfire, rather than

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**Table 2**

Changes in forest structure variables, as modeled for ecological restoration treatment effects on trees per hectare (TPH), canopy bulk density (CBD, kg/m³), and canopy base height (CBH; m) in ponderosa pine forests in northern Arizona. Data classes were based on natural breaks in the pretreatment forest structure data. Post-treatment values were calculated for each class, based on the intensity of forest treatment needed to approach the target values.

<table>
<thead>
<tr>
<th>Variable (Target value)</th>
<th>Data class</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
</tr>
</thead>
<tbody>
<tr>
<td>TPH (47)</td>
<td>Pre-treatment</td>
<td>0–38</td>
<td>383–682</td>
<td>683–1417</td>
<td>1418–2880</td>
</tr>
<tr>
<td></td>
<td>Post-treatment</td>
<td>0–38</td>
<td>14–25</td>
<td>12–26</td>
<td>13–25</td>
</tr>
<tr>
<td>CBD (0.03)</td>
<td>Pre-treatment</td>
<td>0–0.03</td>
<td>0.03–0.05</td>
<td>0.05–0.07</td>
<td>0.07–0.13</td>
</tr>
<tr>
<td></td>
<td>Post-treatment</td>
<td>0–0.03</td>
<td>0.02–0.04</td>
<td>0.02–0.03</td>
<td>0.02–0.04</td>
</tr>
<tr>
<td>CBH (6.5)</td>
<td>Pre-treatment</td>
<td>0–2.3</td>
<td>2.3–3.7</td>
<td>3.7–5.0</td>
<td>5.0–7.5</td>
</tr>
<tr>
<td></td>
<td>Post-treatment</td>
<td>0–2.3</td>
<td>4.6–7.5</td>
<td>5.5–7.3</td>
<td>5.2–7.7</td>
</tr>
</tbody>
</table>

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**Table 3**

Percent of sample points and home ranges, and percent change in predicted likelihood of goshawk territory occurrence from pretreatment conditions as a result of three forest treatment scenarios (MRNG = Management Recommendations for the Northern Goshawk in the Southwestern United States, US Forest Service GTR RM-217; REST = ecological restoration, KFHF = Kaibab Forest Health Focus) on the Kaibab Plateau, AZ. The percent of random sample points and synthetic goshawk territories that had negative, positive, and no change in predicted likelihood of occurrence are given, also the corresponding percent changes in each area.

<table>
<thead>
<tr>
<th>Change in likelihood</th>
<th>Sample points (n = 5014)</th>
<th>Home ranges (n = 23)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>MRNG</td>
<td>REST</td>
</tr>
<tr>
<td>Positive percent</td>
<td>20.7</td>
<td>26.0</td>
</tr>
<tr>
<td>Positive mean</td>
<td>1.17</td>
<td>0.90</td>
</tr>
<tr>
<td>None</td>
<td>3.7</td>
<td>7.8</td>
</tr>
<tr>
<td>Negative percent</td>
<td>75.5</td>
<td>66.2</td>
</tr>
<tr>
<td>Negative mean</td>
<td>0.34</td>
<td>0.40</td>
</tr>
</tbody>
</table>
focusing on absolute values of occurrence. Currently, the Kaibab Plateau is densely occupied by goshawks (Reynolds et al., 2005). Because our occurrence model was conditioned on contemporary forest structure attributes, it is reasonable to expect that any broad-scale deviation in forest structure, such as the changes due to our treatment scenarios, could result in a reduction in average predicted territory occurrence. Restoration of ponderosa pine forests has been the subject of more research than restoration of other forest types (Sisk et al., 2009), so all of our treatments were restricted to ponderosa pine. We therefore expected relatively large changes in predicted occurrence in home ranges that were predominantly ponderosa pine, and relatively high occurrence rates in the post-treatment landscape occurred predominantly in untreated mixed conifer and spruce-fir forests.

The most dramatic changes in predicted territory occurrence from our treatment scenarios indicated a reduction in the proportion of the study area in the highest predicted occurrence quartile (>0.75, pretreatment). In contrast, the proportion of the study area with a predicted occurrence between 0.50 and 0.75 remained essentially unchanged from pretreatment conditions. Treatment effects on goshawk territory occurrence were relatively minor in comparison with the post-Warm Fire landscape, in which there was a substantial increase in the area with a predicted occurrence between zero and 0.25. Detrimental effects of forest treatment were not as severe as habitat loss due to stand-replacing wildfire, a likely outcome in high-density ponderosa pine forests in the arid Southwest (Winemiller et al., 2009). Real management decisions require stakeholders and decision-makers to consider the restored conditions and reduced fire risk associated with treatment as a trade-off with the risk of wildlife habitat degradation (see Allen et al., 2002; Prather et al., 2007). However, until recently, few practical examples have illustrated robust, quantitative approaches for informing these decisions. Our approach can be adapted to different, possibly more complex scenarios; in this case the three treatment scenarios were likely to be less detrimental to goshawks in the short term than a possible stand-replacing wildfire where fuel hazard is high. These results demonstrate how modeling plausible management scenarios can fill key information gaps and permit an informed approach for integrating forest restoration and biodiversity conservation (Noss et al., 2006).

The changes to forest structural conditions as a result of on-the-ground silvicultural treatments are less likely to be so uniform as in our modeling analysis. Our study area was a mosaic of treated and untreated stands, and forest structure data was summarized (i.e., “smoothed”) at the scale of a goshawk territory using a moving window function. Therefore, slight deviations from our reference data are unlikely to impact our overall results. Adjusting the degree to which the forest structure values in our treatment scenarios were changed would likewise alter the absolute territory occurrence estimates, but the adjustments would have to be substantial to influence our assessment of the relative effects of treatments and wildfire.

4.2. Forest treatment scenarios

Our REST, MRNG, and KFHF treatment scenarios all simulated the entire study area being treated in a single action. While our study area has been designated as a priority for active management, it is larger than a single treatment project. Therefore, we expect that the effects modeled here would realistically be diluted through time as projects to reduce high fire risk are implemented in sequence.

Restoration treatments in the southwestern USA have not previously been defined in a manner that allows comparison of possible trade-offs and resulting forest conditions that may impact wildlife, fire behavior, and other public values. General concordance between results of the ecological restoration and MRNG scenarios indicated that these types of treatments can be effectively simulated across large areas of heterogeneous forest, and that our approach is adaptable to variation in the degree of change necessary to meet management objectives. The process we developed can be used early in the planning process to assess management effects on fuels and wildlife habitat. To help guide management decisions, our models and treatment scenarios should be used together with other wildfire and fire models to best identify locations where public values are at the greatest risk of change.

Our ecological restoration scenario was relatively easy to parameterize and implement, due largely to the availability of post-treatment data from restoration-based experiments and ongoing forest management activities across northern Arizona. Using 2006 satellite imagery and custom-derived forest structure data allowed us to implement the restoration treatment scenario as a function of contemporary conditions. We were able to make detailed adjustments to forest structure attributes, ensuring accurate post-treatment conditions across the study area while maintaining heterogeneity. The restoration scenario, therefore, reasonably approximates how a treatment might be planned and implemented by forest managers (Fulé et al., 1997), and can be altered to reflect existing landscape conditions and desired future conditions.

The MRNG have proven difficult to implement on the ground, largely due to opposition by stakeholders and legal barriers (Goad, 2005). The recommendations do not lend themselves to broad-scale spatial modeling because of the high degree of specificity regarding the age class composition of relatively small (compared to a 30-meter pixel) groups or clumps of trees (Higgins, 2005). We attempted to meet the intent of the MRNG by implementing treatments according to the DFC while maintaining variability in forest structure. The DFC in the KNFI&$^*$I all fell within one standard deviation of our treatment scenario mean, and we considered this variability to be acceptable. It is impossible to assess our MRNG treatment in terms of small (<0.1 ha) groups of trees, but the scenario does result in a mosaic of forest structure classes, despite the substantial changes to overall conditions. The MRNG scenario’s similarity to the restoration treatment is consistent with the perspective that on-the-ground forest treatments according to the MRNG will accomplish many restoration goals (Long and Smith, 2000; Reynolds et al., 2006). The reduction in wildfire risk as a result of restoration has been assessed (Fulé et al., 2001; Pollet and Omi, 2002); conducting restoration or MRNG treatments will mitigate the risk of the severe habitat degradation as seen in the Warm Fire scenario.

Predictions of climate change impacts on forest health indicate that background tree mortality may increase as a result of future warming and increasingly severe drought conditions (Allen et al., 2010), compounded by a longer fire season and heightened fire risk in the Southwest (Liu et al., 2013). This anticipated state of greatly stressed and fire-prone forests presents a more urgent need to make use of tools such as those we describe here to plan and balance forest management activities with concern for goshawk habitat.

5. Conclusions

The arguments that have characterized forest management with respect to the goshawk reflect a polarized debate that tends to obscure the actual nature of the tradeoffs that must be assessed to advance active forest management. The information generated through our management-oriented application of spatial modeling and analysis should provide managers with critical components of a comprehensive and more measured approach. With regular use...
of these tools, forest stakeholders could be better informed when locating fuel reduction treatments, thus maintaining wildlife habi-

tad as a complementary forest management objective. Because for-
est treatments are not likely to severely degrade habitat, enough flexi-
bility can be built into forest restoration projects to ensure wildlife needs are met, including goshawks. With information about the relative effects of forest treatment alternatives on gos-
hawks, managers will be better able to ensure that the species will persist while actions designed to restore the ecological integrity of southwestern ponderosa pine forests proceed.

Our results indicate that one key benefit to goshawks from res-
toration or MRNG-based forest treatment activities will not neces-
sarily be immediate, in the sense of higher quality nesting or for-
gaging habitat. Instead, forest restoration is likely to benefit gos-
hawks in the longer term by ensuring that suitable habitat will per-
sist through time, reducing the risk of catastrophic wildfire that might otherwise severely degrade goshawk habitat. While our treatment scenarios reduced the overall predicted goshawk terri-

tory occurrence, the effects very rarely approached the level of habitat degradation associated with severe wildfire. Long-term survival of this species in the region may depend on forest stake-
holders’ willingness to accept a modest reduction in goshawk habi-
tat quality, but only in specific areas associated with forest restoration treatments. These treatments to restore pre-settlement forest structures and fire regimes may be necessary in order to avoid the more pronounced and widespread degradation or loss of current habitat.

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Appendix A

Detailed description of an approach for modeling forest struc-
ture changes under three management alternatives: (1) ecological restoration, (2) Management Recommendations for the Northern Goshawk in the Southwestern United States (MRNG; Reynolds et al., 1992), and (3) the Kaibab Forest Health Focus collaborative planning group. We also explain the rationale for assessing habitat change within a recent wildfire boundary instead of modeling the changes directly in our study area. The objective of the treatment scenarios was to output altered forest structure data layers that could subsequently be used as covariates in a predictive model of northern goshawk territory occurrence.

A.1. Ecological restoration treatment scenario

We reclassified all pretreatment forest structure spatial data layers into four categories according to natural breaks in the data distribution (Coulson, 1987). We established a range of target for-
est structure values from a review of post-treatment experimental data (Covington et al., 1997; Heinlein et al., 2000; Fulé et al., 2001, 2002a; Waltz et al., 2003; Hurteau et al., 2008) and estimated pre-settlement structure data (Covington et al., 1997; Fulé et al., 2002b) in the Southwest. We implemented percent changes in each forest structure data class using conditional statements in the ArcGIS Spatial Analyst extension raster calculator (Table 2). The four resulting layers were merged to produce a final layer rep-

resenting forest structure conditions after treatment.

A.2. MRNG treatment scenario

We were unable to establish target values for an MRNG treat-

ment based on actual post-treatment measurements because such data do not exist. Additionally, recommendations for goshawk habi-
tat in the MRNG and subsequent interpretations are not explicitly given in terms of CBD or CBH. We therefore relied on the minimum desired future conditions (DFC) given in the Kaibab National Forest Implementation and Interpretation of Management Recommenda-
tions for the Northern Goshawk, Version 2.1 (KNFI&I; Higgins, 2005). DFC in the KNFI&I are given at group (0.2–1.6 ha) and site (i.e., stand) scales for PFAs and foraging areas in ponderosa pine, mixed conifer, and spruce-fir forests. We modeled the conditions for ponderosa pine foraging areas because foraging areas account for the greatest portion of a goshawk home range and our analysis is focused on ponderosa pine-dominated habitat. The KNFI&I doc-

ument provides DFC for the average values by stand of TPH, Dq and SDI. Because we altered SDI values to predict the treated CBD and CBH, we also used SDI and Dq to calculate a treated TPH instead of using the actual DFC given for TPH. This ensured that the treated

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Fig. A1. Flow chart of a forest treatment simulation process to meet the conditions in the Management Recommendations for the Northern Goshawk in the Southwestern United States (MRNG). In step 1, current forest structure data from ponderosa pine forests on the Kaibab Plateau in northern Arizona were used to derive regression equations that predict canopy bulk density (CBD) and canopy base height (CBH) from stand density index (SDI). In step 2, SDI and quadratic mean diameter (Dq) were adjusted using percent changes to approximate the desired future conditions prescribed in the Kaibab National Forest Implementation and Interpretation of Management Recommendations for the Northern Goshawk, Version 2.1 (KNFI&I). Next, the equations derived in step 1 were used to compute treated CBD and CBH from the treated SDI. Finally, the treated SDI and Dq data layers were used to compute values for treated stem density (trees per hectare, TPH). The final treated forest structure datasets, in the gray box, were used as predictor variables in a spatial goshawk territory occurrence model to estimate the effects of the MRNG on goshawk habitat.
SDI and TPH values were not unrealistically decoupled in the treatment scenario (Fig. A1). Using data from 2000 random sample points placed only in ponderosa pine pixels in the 23 home ranges, we computed nonlinear regression equations to predict CBD and CBH from SDI. The use of SDI was not only convenient in that specific DFC were given, but it has also been demonstrated to be a useful silvicultural tool with respect to goshawk and other wildlife habitat management (Lilieholm et al., 1993, 1994). We obtained the highest R-squared values by log-transforming the CBD data, inverse-transforming the CBH data, and fitting a logistic curve to both datasets. R-squared results from the regression equations were 0.74 for CBD and 0.61 for CBH. We implemented percent changes in SDI and Dq to closely match the DFC (Table A1), then used the regression equations to predict post-treatment CBD and CBH.

A.3. Kaibab Forest Health Focus treatment scenario

The Kaibab Forest Health Focus (KFHF) group identified a variety of treatment alternatives as priorities for implementation. To develop a KHF scenario from these categories, we extracted forest structure pixels for each variable from our MRNG treatment scenario inputs for areas where the KFHF recommended they be implemented, and we extracted pixels from our restoration scenario where the KHFH recommended strategic treatment optimization modeling for fuel reduction (Finney, 2007) or reduction of fire risk from predicted active crown to surface fire. We left all other pixels in their current, untreated condition.

A.4. Wildfire alternative scenario

The Warm Fire was ignited by lightning in June 2006 and was actively suppressed by the USFS after growing past 7690 ha. Over 15,780 ha just east of our study area on the Kaibab Plateau burned in the fire. Approximately 60% of the burned area was classified as moderate or high burn severity (USDA, 2009). The fire therefore represented an appropriate case study for examining a possible "no-action" treatment alternative followed by fire. We implemented our goshawk occurrence model in the Warm Fire suppression area to assess the effects of severe wildfire on habitat (9 territories in our database were within the Warm Fire boundary and were excluded from the occurrence model).

References
