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Real versus perceived conflicts between restoration of ponderosa pine forests and conservation of the Mexican spotted owl

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Abstract

Progress in implementing ecosystem approaches to conservation and restoration is slowed by legitimate concerns about the effects of such approaches on individual imperiled species. The perceived conflict between the restoration of fire-excluded forests and concomitant reduction of dense fuels and high-severity wildfire, versus the recovery of endangered species, has led to a policy ambiguity that has slowed on-the-ground action at a time when active management is urgently needed, both for ecosystem restoration and species conservation. The Mexican spotted owl (*Strix occidentalis lucida*) in the southwestern U.S.A. is emblematic of this perceived conflict, with numerous appeals and lawsuits focused on the species and vast acres of forest managed with habitat quality for this species in mind. We use spatial analysis across large landscapes in Arizona to examine potential conflicts between the desire to reduce the likelihood of uncharacteristically severe wildfire and restore native fire regimes, and the concurrent desire and legal mandate to manage forests for the recovery of the owl. Our spatially explicit analysis indicates that real conflicts between these management objectives exist, but that locations where conflicts might inhibit active forest management represent less than 1/3 of the 811,000 ha study region. Furthermore, within the areas where conflicts might be expected, the majority of the forest could be managed in ways that would reduce fire hazard without eliminating owl habitat. Finally, management treatments that emphasize ecosystem restoration might improve the suitability of large areas of forest habitat in the southwest that is currently unsuitable for owls. These results demonstrate that even where policy conflicts exist, their magnitude has been overstated. Active restoration of dry forests from which fire has been excluded is compatible in many areas with conservation and recovery of the owl. Identifying and prioritizing areas to meet the dual goals of ecosystem restoration and imperiled species conservation require a broad spatial approach that is analytically feasible but currently underutilized. Working together, conservation biologists, restoration ecologists, and forest managers can employ landscape-level spatial analysis to identify appropriate areas for management attention, identify suitable management practices, and explore the predicted consequences of alternative management scenarios on forests, fire ecology, and the fate of sensitive species of conservation concern.

Keywords: Environmental policy; Forest management; *Strix occidentalis lucida*; Restoration ecology; Spatial analysis; Imperiled species

1. Introduction

The evolution of wildlife conservation has proceeded from protection of favored species, recreation areas, and hunting and fishing grounds, to conservation and restoration of ecosystems (Noss and Cooperrider, 1994; Redford et al., 2003). Restoration of ecosystem function and natural fire regimes has emerged as the primary objective of private, state, and federal forest managers in the extensive ponderosa pine (*Pinus ponderosa*) forests of the arid southwestern USA (Friederici, 2003). After a century of fire exclusion and heavy livestock grazing, more of the southwestern landscape is characterized by stands of ponderosa pine that are unusually dense and fire prone (Covington and Moore, 1994; Covington et al., 1994; Allen et al., 2002; Brown et al., 2004; Noss et al., 2006a). The resulting increase in the frequency and severity of large, stand-replacing crown fires has placed human communities and

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infrastructure at risk, while degrading forest ecosystems and wildlife habitat (Covington, 2003). Ecosystem restoration, typically involving aggressive thinning of dense stands of small trees, followed by prescribed fire, is intended to return ponderosa pine forests to a more open structure, with fewer, larger trees (Mast et al., 1999; Covington et al., 1997; Moore et al., 1999). Historically this structure was maintained by frequent surface fires, favoring larger trees and more productive herbaceous understories, whereas crown fires tended to be uncommon and localized (Covington and Moore, 1992, 1994). Restoration and fuels-reduction treatments are encouraged by the Healthy Forests Restoration Act (HFRA) of 2003, but HFRA promotes a relatively narrow definition of restoration that focuses almost exclusively on fuels (DellaSala et al., 2004; Schoennagel et al., 2004; Noss et al., 2006b).

Progress in carrying out actual forest restoration treatments has been slow, in part because restoration treatments, especially those involving mechanical thinning and prescribed burning, are controversial (Tiedemann et al., 2000; Wagner et al., 2000; Allen et al., 2002) and may conflict with actions to benefit particular threatened and endangered species, as required by law (Noss et al., 1997). In addition, exurban development and the expansion of the wildland–urban interface (Marzluff and Bradley, 2003; Radeloff et al., 2005) in the southwest has exacerbated the need for fuels reduction and increased the conflict between protection of human communities and habitat management for wildlife. This has resulted in the suggestion that a state of "analysis paralysis" (Bosworth, 2002) exists, which prevents the implementation of beneficial forest management.

No species is more symbolic of the conflict between forest management and species conservation than the spotted owl (*Strix occidentalis*). In ponderosa pine forests of the western USA, the northern spotted owl (*S. o. caurina*) and the Mexican spotted owl (*S. o. lucida*), both of which are listed as threatened under the U.S. Endangered Species Act, as well as California spotted owls (*S. o. occidentalis*) east of the Sierra Nevada crest, tend to inhabit areas of denser forest within generally open landscapes historically maintained by frequent, low-severity fires (Beier and Maschinski, 2003). In many cases fire exclusion may have increased the amount of suitable habitat for spotted owls in the short-term (Huff et al., 2005). Stands suitable to the owls generally have high canopy closure and high basal area with large trees and multiple canopy layers. These stands are often found on north-facing slopes and in steep canyons, which burn less frequently under a natural fire regime than much of the rest of the landscape (Beier and Maschinski, 2003). Under fire exclusion policies of the past 100 years, some areas that once supported more open stands now more closely resemble owl habitat. Nevertheless, the increased area of dense forest that has developed since fire exclusion is more vulnerable to large, stand-replacing fires that will likely reduce the area of this partially anthropogenic forest type and also eliminate historical owl habitat (Huff et al., 2005; Noss et al., 2006a). Indeed, the Rodeo-Chediski Fire in Arizona affected 55 Mexican Spotted Owl Protected Activity Centers, many of which are no longer occupied by owls (United States Forest Service, Region 3 Staff, pers. comm.), and owl populations are likely to be compromised if large portions of the landscape continue to experience high-intensity crown fires (Jenness et al., 2004).

A widespread perception is that efforts to restore ponderosa pine landscapes and reduce the risk of unnatural high-severity fire conflict with efforts to protect spotted owls and other species associated with dense stands (Allen et al., 2002; Beier and Maschinski, 2003). However, many endangered species conflicts are more a matter of perception than reality (Noss et al., 1997). In the vast ponderosa pine ecosystem, in particular, the relatively small area occupied by imperiled species is unlikely to preclude ecosystem restoration efforts in general (Beier and Maschinski, 2003), albeit it might constrain particular forest treatments in specific locations. Taking a landscape-level perspective (Sisk et al., 2005), we use the dry ponderosa pine forests and Mexican spotted owl population of the western Mogollon Plateau of northern Arizona (USA) as a case study in the conflict between habitat management for an endangered species and restoration to reduce fire threat. We show that although some degree of conflict between ecosystem restoration and species conservation is real, a considerable amount of restoration and fuels-reduction treatments can be accomplished without negative impacts on owls.

In this study we examine the degree of spatial concordance between areas identified as high priority for aggressive thinning and prescribed fire, and areas supporting occupied or potential habitat for Mexican spotted owls. We model the likely impacts of forest restoration treatments across an 811,000-ha study area, explore the extent and location of potential conflicts between restoration activities and owl habitat, and assess tradeoffs between owl habitat management and abatement of fire threat. In addition to identifying potential conflicts, we explore the possible compatibility of forest treatments and owl conservation, focusing on the hypothesis that well-designed forest restoration treatments may conserve, and in some cases enhance, the quality of owl habitat.

2. Methods

2.1. Study area

Our study area covers just over 800,000 ha of primarily forested land on the western Mogollon Plateau, Arizona, beginning just north and west of Flagstaff and extending southeast to the edge of the 180,000 ha Rodeo-Chediski burn of 2002 (Fig. 1a). Elevation varies from approximately 1500 m in canyons along the edge of the plateau, to over 3600 m on the San Francisco Peaks. The dominant vegetation is stands of nearly pure ponderosa pine and pine mixed with Gambel oak (*Quercus gambelii*). Scattered areas of Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), and white pine (*Pinus strobiformis*) forest, pinyon–juniper (*Pinus edulis* and *Juniperus* spp.) woodlands, aspen (*Populus tremuloides*) groves, and open grasslands are also present (Fig. 1a). The primary land manager in the region is the USDA Forest Service, with parts of four national forests (Coconino, Kaibab, Tonto, and Apache-Sitgreaves) comprising over 75% of the study area. The area includes the cities of

Flagstaff and Williams, AZ, as well as many smaller communities.

2.2. Spatial data layers

We developed a suite of spatial data layers for our analyses. All spatial data were converted to 90-m (0.81-ha) resolution raster formats compatible with ArcGIS 9.x (ESRI Corp., Redlands, CA) geographic information system (GIS) software. Analyses, primarily spatial overlays, were performed within the ArcGIS Spatial Analyst platform, through use of the raster calculator and other standard tools and methodologies.

2.2.1. Vegetation layers

In 2004 the Forest Ecosystem Restoration Analysis (ForestERA) project produced a set of spatially explicit maps of vegetation composition and structure across the analysis area. Layers representing dominant overstory vegetation, basal area (m^2/ha), and tree density (stems/ha) were developed using Enhanced Thematic Mapper (ETM; year 2000; 30-m

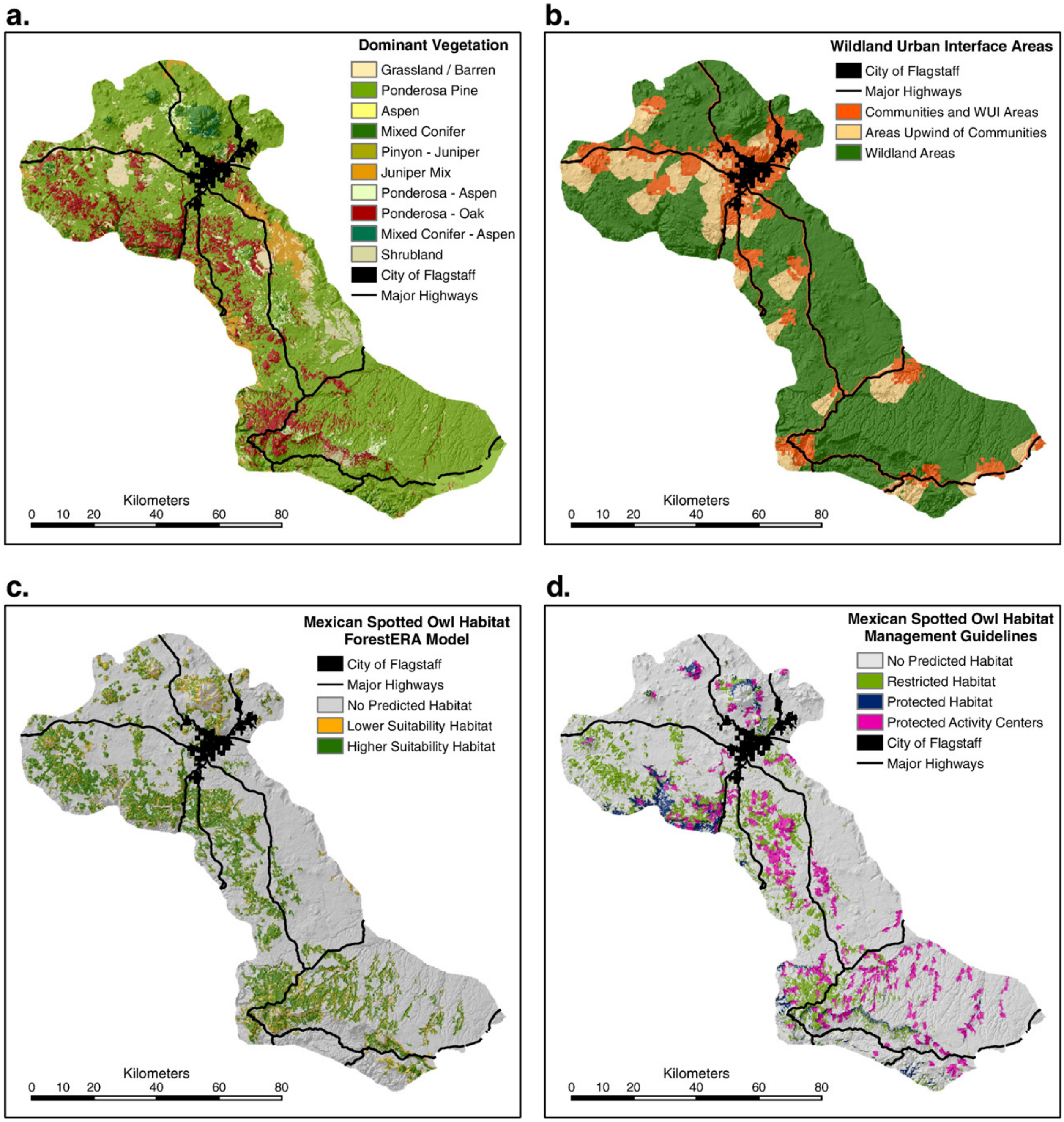

Fig. 1. a) Vegetation types across the 811,000-ha ForestERA study area; b) wildland–urban interface (WUI) areas, areas upwind of communities, and wildland areas; c) predicted Mexican spotted owl habitat, as per the ForestERA model; d) areas predicted to be under the purview of the Mexican spotted owl management guidelines.

resolution) imagery coupled with data collected at more than 500 ground plots (Hampton et al., 2003; Sisk et al., 2006). These analyses were carried out using a classification tree (vegetation composition) or regression tree (basal area and tree density) methodology (Breiman et al., 1984) in See5 and Cubist software (Rulequest Research, St. Ives, NSW, Australia). A canopy cover layer was developed using Digital Orthophoto Quads (DOQs) and a form of advanced exploratory data analysis (Xu et al., 2006). In this methodology each 1-m pixel in the DOQ is classified as canopy foliage, shadow, or ground vegetation based on its spectral signature. Percent canopy cover is derived by determining the number of 1-m pixels identified as canopy foliage within a larger window. To the best of our knowledge these are the most accurate spatial data layers of vegetation characteristics for the analysis area.

2.2.2. Physiographic layers

A 30-m resolution digital elevation model (DEM) from the United States Geological Survey (USGS) was used to derive spatial data layers representing slope and aspect. Owing to its circular nature ($0^0 = 360^0$), for some analyses the aspect layer was converted to radians, and the sine and cosine were taken (Beers et al., 1966). This converts the aspect layer from a circular variable to two layers that have continuous values between 1 and -1. The sine function divides aspect into a north–south component, while the cosine function divides aspect into an east–west component.

2.2.3. Communities and wildland–urban interface layers

We developed a spatially-explicit data layer identifying communities within the analysis area based on a layer representing land ownership obtained from the Arizona Land Resource Information System (ALRIS). From the land ownership layer we identified private property areas that were either inside or contiguous with incorporated communities. The community layer was further modified by local experts during two planning workshops held by the ForestERA project (Sisk et al., 2006). During these workshops the experts added additional private land parcels that contained developments that were either listed on the national registry of communities at risk from wildfire or had their own fire protection district. Based on this community layer, we then developed two additional spatial data layers that explicitly identify areas that are likely to be among the highest priorities for fuels-reduction treatments in the immediate future: a wildland–urban interface (WUI) layer and a layer representing areas upwind of communities. The WUI layer was developed using our interpretation of guidelines for identification of the WUI found in the Healthy Forest Restoration Act of 2003. We considered areas to be part of the WUI if they were 1) within 0.8 km of a community (as identified above), 2) within 2.4 km of a community and were identified as areas with "high fire hazard" (see below), or 3) were within 0.4 km of a major highway (Fig. 1b). In general, we only eliminated areas within 2.4 km of a community if they did not contain forest vegetation; our interpretation of the WUI is therefore conservative. Areas were considered to be upwind of communities if they fell within 10 km of a community (as identified above) and were upwind of the community in the direction in which winds typically blow in the analysis area during the fire season (SSW or 210–240^0; NOAA 2004; Fig. 1b).

2.2.4. Mexican spotted owl habitat layers

We used three different layers representing Mexican spotted owl (hereafter MSO) habitat in our analyses. First, we created a MSO nesting and roosting habitat layer developed by the ForestERA project based on a literature review and input from local MSO experts, including members of the MSO recovery team (Fig. 1c). The ForestERA MSO habitat layer identifies the spatial extent of potential MSO nesting and roosting habitat as those areas of pine–oak or mixed-conifer habitat with basal area >17 m^2/ha. In addition, areas of pure ponderosa pine on slopes $>8°$ ($\sim 14\%$) and with basal area >17 m^2/ha are considered habitat. Potential habitat predicted by the model captures approximately 85% of the 132 georeferenced MSO nest sites known to exist within the analysis area. Because specific microsite characteristics (e.g., presence of very large oaks or conifers), which could not be mapped effectively across a large landscape, appear to influence MSO nest site selection (May et al., 2004), this model overestimates the habitat used by MSO.

This model was refined using the Mahalanobis distance statistic (Clark et al., 1993; Farber and Kadmon, 2003; hereafter M-distance) and vegetation and physiographic characteristics at 132 georeferenced MSO nest sites. For this analysis basal area, tree stem density, canopy cover, slope, and sine and cosine of aspect were determined at each nest site by extracting those data from the raster pixel corresponding to the nest site. M-distance values were then computed for all nest sites, and all other 90-m pixels across the analysis area, using the mean and covariance values from the nest site dataset. This was accomplished using an extension for ArcView 3.x (Jenness Enterprises, Flagstaff, Arizona), which computes M-distance values for any landscape based on a set of up to 8 input raster layers. The M-distance values were then placed into four categories that represent the range of values in which 50%, 75%, 90%, and 100% of the nest sites could be found. The areas covered by the four categories of M-distance values were considered to be areas of high, moderate, low, and very low suitability for owls. The final spatial layer representing categorical M-distance values was cut to the extent of potential MSO nesting habitat (ForestERA, unpublished).

To create our second layer we obtained a shapefile identifying MSO Protected Activity Centers (PACs) from GIS staff at the USDA Forest Service (Coconino, Kaibab, and Apache-Sitgreaves National Forests). Under the MSO recovery plan (USFWS, 1995) the Forest Service is required to identify approximately 600 acres (~ 240 ha) of the "highest quality" MSO habitat around areas either currently or historically used as nesting or roosting sites by owls. Each of these areas is designated as a PAC. The shapefile was converted to a raster coverage using standard techniques.

For our third layer we created a spatial representation of areas that would fall under the purview of the MSO recovery plan (USFWS, 1995). This layer was developed from ForestERA vegetation data layers, in collaboration with members of the MSO recovery team, and represents a spatial interpretation of habitat as identified under the recovery plan (Fig. 1d). The plan places MSO habitat into two categories, protected habitat and restricted habitat, and recommends constraints on the types of management that may be undertaken in each of these categories. Protected habitat comprises areas identified as PACs (see above) as well as all areas of pine–oak or mixed-conifer habitat that fall within protected areas (e.g., wilderness, designated roadless areas, national parks, or national monuments) or fall on slopes >40%. These areas may only be managed using low-intensity prescribed fire and wildland fire use (allowing previously ignited fires to burn, when they meet management objectives), except in special circumstances (USFWS, 1995). Restricted habitat is any other area identified as containing pine–oak or mixed-conifer habitat. These areas may be managed with considerable latitude, provided that management treatments retain key owl habitat components, such as large trees, downed logs, and potential nest sites (USFWS, 1995).

2.2.5. Fire hazard layers

Fire hazards are defined as the types and amounts of fuels available for a fire to consume (Sampson et al., 2000). While ground and ladder fuels influence the severity of understory fires and their likelihood of climbing into the forest canopy, these elements of fire hazard are highly variable and difficult to assess over large areas (Sampson et al., 2000). At the landscape level, one of the most important components of fire hazard models is the amount and distribution of canopy fuels (Cruz et al., 2003), often described in terms of crown bulk density (hereafter CBD). As CBD increases, so does the likelihood of sustaining an active crown fire (one that actively moves through forest canopy as opposed to passive crown fire, in which some trees may burn, but the fire does not spread through the canopy). Active crown fires are difficult to control or suppress. The value 0.1 kg/m^3 CBD has been identified by fire experts as a threshold below which active crown fire is unlikely to occur (Agee, 1996; USDA Forest Service Fire Sciences Laboratory staff, pers. comm.), and this value has been supported by some empirical studies (Scott and Reinhardt, 2002). At values of 0.15 kg/m^3 CBD or more, any crown fire is likely to actively spread through the canopy (Agee, 1996).

The ForestERA project developed a spatial data layer representing CBD (kg/m^3) for the entire analysis area. For the mixed-conifer (including mixed-conifer and aspen mixtures) and ponderosa pine (including pine–oak and pine–aspen mixtures) vegetation types, CBD was estimated using the ForestERA basal area and tree density spatial data layers along with allometric equations developed by Cruz et al. (2003). The resulting values for CBD in the analysis area ranged from near zero to about 0.5 kg/m^3 in these vegetation types. These values are similar to values for CBD obtained from empirical studies in the same vegetation types (Brown, 1978; Fulé et al., 2001a,b; Scott and Reinhardt 2002). The CBD was set at 0.01 kg/m^3 for pure aspen stands. In pinyon–juniper CBD was calculated as 0.001 kg/m^3 per 1% of canopy cover. These values were modifications of estimated CBD for aspen stands in the Pacific Northwest (Pollard, 1972) and for pinyon–juniper woodlands on the Gila National Forest (Keane et al., 2000). Although this methodology was less defensible than that used for mixed-conifer and ponderosa pine dominated areas, it appears to provide reasonable predictions of fire behavior in these habitat types, which cover a very small portion (<1%) of the total study area.

For analysis purposes we classified fire hazard into three categories based on CBD values: 1) low-moderate = 0–0.1 kg/m^3 CBD, 2) high = 0.1–0.15 kg/m^3 CBD, and 3) very high = >0.15 kg/m^3 CBD. We assumed that areas with CBD below 0.1 kg/m^3 would generally be low priority areas for restoration and fuels-reduction treatments, whereas areas above 0.15 kg/m^3 CBD would be of highest priority.

2.3. Treatment effects

Through literature review and expert input, the ForestERA project has developed predictions of the potential effects of various restoration and fuels-reduction treatments on forest structural attributes (ForestERA, unpublished). Initially, the mean effects of different treatments were described using empirical data from before and after treatment on ponderosa pine and dry mixed-conifer stands in Arizona (Fulé et al., 2001a,b, 2002), Colorado (Lynch et al., 2000), and Montana (Scott, 1998). Forestry experts from Colorado State University and Northern Arizona University were then asked to describe the likely range of variation around the mean. These data were synthesized to create descriptions of a final set of treatment types (Table 1).

We determined whether a treatment would affect MSO habitat by applying the maximum predicted basal area reduction of a specific treatment on potential MSO nesting habitat and determining whether the treatment would reduce the basal area below the threshold value (17.0 m^2/ha) for potential nesting habitat. For example, areas with basal area values above 18.9 m^2/ha could receive maintenance burning without being reduced below the threshold. The maximum basal area reduction expected from this type of treatment is 10%; thus, in locations with basal area of 18.9 m^2/ha, this would reduce basal area to just over 17.0 m^2/ha.

Table 1
Predicted effects of different treatment types used in our analyses, based on percent reduction of basal area, tree stem density, and canopy cover

Type	Mean (range) percent reduction		
	Basal area	Tree stem density	Canopy cover
Maintenance burn [a]	5 (0–10)	5 (0–20)	5 (0–10)
Very low-intensity thin [b]	10 (5–20)	40 (30–50)	20 (5–20)
Low-intensity thin [c]	20 (10–30)	50 (40–60)	20 (10–30)
Intermediate-intensity thin [d]	40 (30–50)	65 (55–75)	30 (15–45)
High-intensity thin [e]	60 (50–70)	80 (70–90)	40 (25–55)

[a] Representative of a light “broadcast” burn, or fuels maintenance treatment.
[b] Very light thinning, followed by a prescribed burn. Representative of a “very light” thinning or fuels maintenance treatment targeting primarily ladder fuels.
[c] Light thinning, followed by a prescribed burn. Representative of a “light” restoration or fuels maintenance treatment.
[d] Moderate-heavy thinning, followed by a prescribed burn. Representative of a “moderate” to “full restoration, or moderate fuels reduction.
[e] Heavy thinning, followed by a prescribed burn. Representative of a “full” restoration, heavy fuels reduction, or multi-age group selection.

Table 2
Area of low (<0.10 kg/m^3) intermediate (0.10–0.15 kg/m^3) and high (>0.15 kg/m^3) crown bulk density (CBD) within the western Mogollon Plateau analysis area, and within predicted Mexican spotted owl (MSO) habitat based on a) the ForestERA model, b) areas under the purview of the MSO management guidelines, and c) within MSO Protected Activity Centers (PACs)

Analysis area	Total hectares	Hectares of CBD >0.15 kg/m^3	Hectares of CBD 0.10–0.15 kg/m^3	Hectares of CBD <0.10 kg/m^3
Entire study area	811,100	265,900 (32.8%)	270,100 (33.3%)	275,100 (39.5%)
ForestERA model	216,400	139,900 (64.7%)	69,600 (32.2%)	6,900 (3.2%)
MSO guidelines	169,300	87,700 (51.8%)	52,300 (30.9%)	29,300 (17.3%)
MSO PACs	64,500	38,500 (59.7%)	17,200 (26.7%)	8,800 (13.6%)

We also determined whether different treatment types would affect the suitability of owl habitat, as predicted using the M-distance statistic. We applied the mean reduction values for each treatment type to the basal area, tree density, and canopy cover layers. The M-distance extension was then rerun using the same mean and covariance matrices created using the original nest site dataset, the original slope and aspect layers, and the modified vegetation structure layers. The resulting M-distance layer was then compared with the original layer. Areas in the new layer with values >1 unit of distance higher than those in the original layer were considered decreased in suitability, areas with values that did not change by >1 unit of distance were considered to have remained unchanged in suitability, and areas with values >1 unit of distance lower than the original layer were considered to have improved in suitability.

2.4. Example management scenarios

To examine options for reducing fire hazard around communities we created three scenarios that ranged from complete protection or enhancement of areas identified as MSO habitat, to fire hazard reduction with minimal consideration for MSO habitat. In each scenario we treated areas contained within the WUI designated under HFRA and/or the 10-km upwind vector from communities at risk (hereafter priority areas). These areas, encompassing a total of 245,100 ha, are likely to be priorities for treatment given current political and social values. Within these areas we set a goal of reducing CBD values to below 0.1 kg/m^3. We used CBD as our measure of fire hazard because it does not change with differing weather conditions.

In all three scenarios we did not apply treatments to private land, to areas with slope >40%, to specially designated areas (wilderness and roadless areas), or to vegetation types other than ponderosa pine and dry mixed-conifer (e.g., pinyon–juniper, aspen). These constraints are representative of those in real-life scenarios and reduced the treatable area to 180,000 ha.

In the first scenario, “Maximum Fuels Reduction,” we used the minimum intensity of treatment necessary to reduce CBD to below 0.1 kg/m^3 in each location (pixel) on the landscape. If CBD was already below this threshold, no treatment was applied. In locations where CBD was so high that none of the treatment models would reduce the value below the 0.1 k kg/m^3 threshold, the treatment resulting in maximum reduction of CBD, high-intensity thinning, was applied. The only consideration for MSO habitat in this scenario was that no treatments were applied in PACs.

In the second scenario, “Maximum Owl Habitat Protection,” we applied treatments as outlined in scenario 1 outside of areas identified as MSO habitat (using the ForestERA model). Within MSO habitat we applied treatments only if crown bulk density was above 0.1 kg/m^3 and if those treatments would either improve, or not significantly alter, MSO habitat. These rules applied even within MSO PAC areas. A treatment that would change the M-distance value of the MSO habitat by more than one unit was considered to have a significant impact. This value is somewhat arbitrary, but a lesser change is unlikely to move an area of habitat from one designation of quality to another.

In the final scenario, “Owl Habitat Mitigation,” we applied treatments as in scenario 2 within “high-quality” MSO habitat, and as in scenario 1 outside of MSO habitat and in “low-quality” MSO habitat. “High-quality” MSO habitat was considered as those areas that had M-distance values within the range covered by the top 50% of MSO nest sites (Fig. 1c).

Table 3
Area of landscape treated (ha and percent by treatment type) within priority areas (wildland–urban interface areas and areas up to 10 km upwind of communities) across the western Mogollon Plateau, under three different forest management scenarios covering 180,000 ha

Treatment type	Scenario: Maximum fuels reduction	Scenario: Maximum owl habitat protection	Scenario: Owl habitat mitigation
High-intensity thin	16,800 (9.4%)	6900 (3.7%)	12,100 (6.7%)
Intermediate-intensity thin	7600 (9.8%)	8200 (4.7%)	10,900 (6.1%)
Low-intensity thin	25,000 (13.9%)	14,000 (7.7%)	17,300 (9.6%)
Very low-intensity thin	54,500 (30.4%)	38,400 (21.3%)	49,500 (27.5%)
Maintenance burn	30,800 (17.2%)	53,300 (29.7%)	45,900 (25.5%)
No treatment	35,300 (19.2%)	59,200 (32.9%)	44,300 (24.6%)

Table 4
Mean crown bulk density values (kg/m^3), before and after modeled forest treatments, within priority areas (wildland–urban interface areas and areas up to 10 km upwind of communities) under three forest management scenarios

Scenario	WUI areas	Areas upwind of communities	All priority areas	Within treatable areas
Pre-treatment	0.119	0.133	0.126	0.149
Maximum fuels reduction	0.086	0.089	0.087	0.096
Maximum owl habitat protection	0.099	0.109	0.104	0.119
Owl habitat mitigation	0.093	0.010	0.096	0.109

3. Results

3.1. Overlap of MSO habitat with priority areas for fuels reduction

The western Mogollon Plateau study area covers 811,000 ha, of which 216,400 ha (26.7%) qualified as potential MSO nesting habitat by the ForestERA habitat model, 169,300 ha (20.9%) qualified for status as either protected (85,700 ha) or restricted (83,600 ha) habitat, and 64,500 ha (8.0%) fell within PACs. Although the ForestERA habitat model identified a greater proportion of the study region as potential MSO habitat, it did not include all areas identified as protected or restricted habitat by our spatial interpretation of guidelines in the MSO recovery plan (USFWS, 1995). A total of 275,600 ha (34.0%) was considered potential habitat by at least one model.

Crown bulk density was generally high in areas of MSO habitat. Values for CBD averaged 0.18 kg/m^3 (range 0.011–

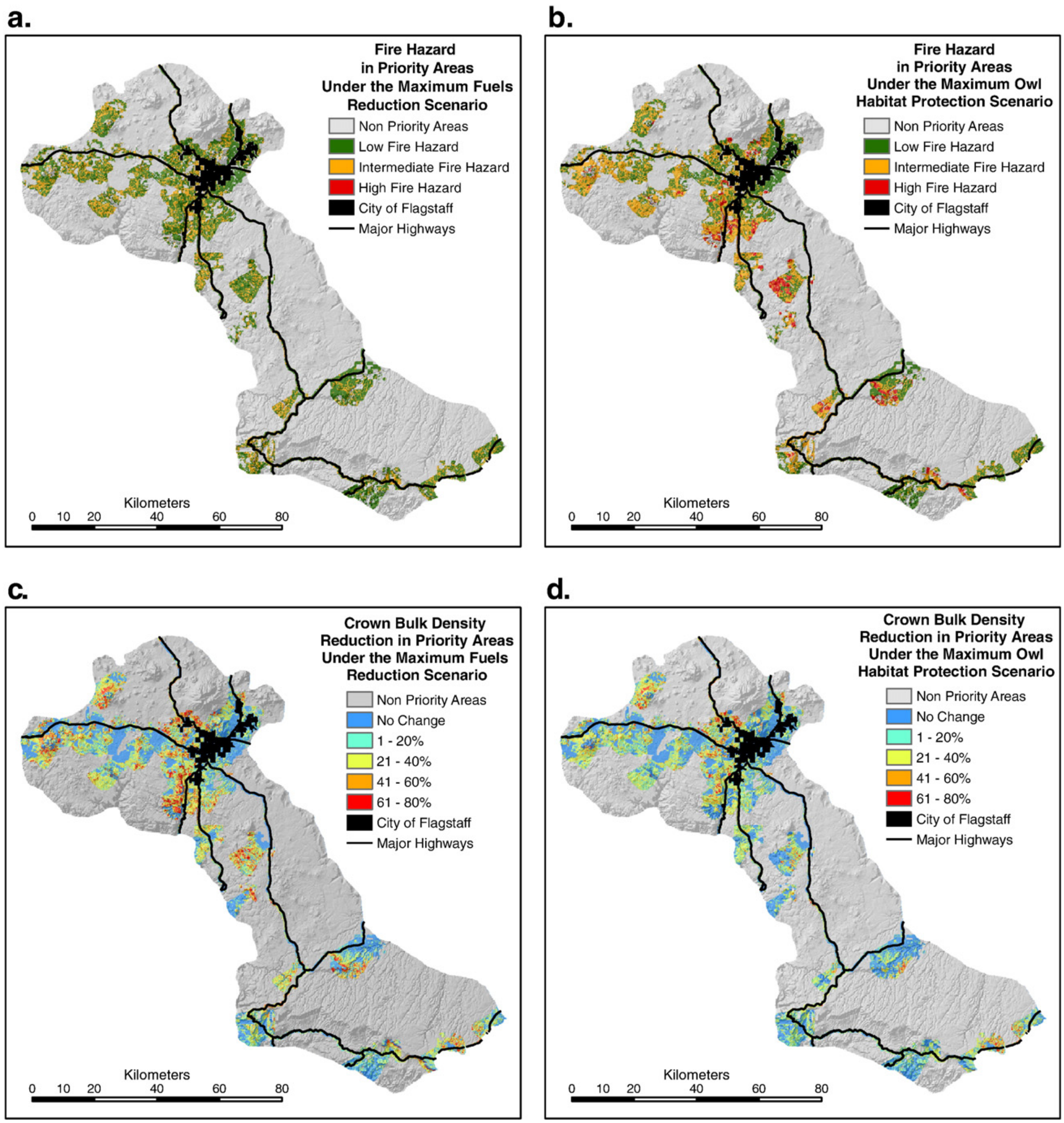

Fig. 2. a) Percent reduction in crown bulk density in priority areas under the “maximum fire hazard reduction” management scenario; b) percent reduction in crown bulk density in priority areas under the “maximum owl habitat protection” management scenario; c) proportion of the landscape having low, intermediate, and high fire hazard after “maximum fire hazard reduction” management scenario; d) proportion of the landscape having low, intermediate, and high fire hazard after “maximum owl habitat protection” management scenario.

Table 5
Percent of western Mogollon Plateau landscape within priority areas (wildland–urban interface areas and areas up to 10 km upwind of communities) that are under the 0.1 kg/m^3 crown bulk density threshold before treatment and after treatment under three management scenarios

Scenario	WUI areas	Areas upwind of communities	All priority areas	Within treatable areas
Pre-treatment	36.7%	28.7%	32.6%	19.5%
Maximum fuels reduction	62.8%	62.5%	62.6%	60.0%
Maximum owl habitat protection	53.4%	48.1%	50.5%	43.6%
Owl habitat mitigation	58.8%	56.2%	57.4%	52.9%

0.399) inside of MSO habitat and 0.107 outside of MSO habitat (range 0–0.502), as identified by the ForestERA potential nesting habitat model. Within areas identified as PACs, CBD averaged 0.168 (range 0–0.443). As would be expected, a greater proportion of the landscape within MSO habitat was in the very high fire hazard (CBD >0.15 kg/m^3) category (Table 2).

We identified 146,700 ha (18.1% of the study area) as lying within HFRA WUI areas and 154,100 ha (19.0% of the study area) as lying upwind of communities. In total, 30.2% of the study area (245,000 ha) was identified as lying within the spatial extent of these priority areas. Within priority areas, 64,400 ha (26.3%) qualified as potential MSO nesting habitat, 49,200 ha (20.1%) qualified for status as either protected (20,800 ha) or restricted (28,400 ha) habitat, and 16,500 ha (6.7%) fell within PACs, with overlap among these categories. A total of 80,000 ha (32.7%) was considered potential habitat by at least one model, suggesting potential for conflict between hazardous fuels reduction and conservation of the MSO.

3.2. Changes in MSO habitat after treatment

The analysis of treatment types that could be used within MSO habitat, based on the ForestERA potential nesting habitat model, revealed that only 39,500 ha (18.3%) of potential habitat could not be treated at some level, without removing it from prediction as potential habitat. Of the remaining habitat 12,500 ha (5.8%) could be treated with intermediate-intensity thinning, 83,700 ha (38.7%) could be treated with low-intensity thinning, 46,100 ha (21.3%) could be treated with very low-intensity thinning, and 34,600 ha (16.0%) could be treated with maintenance burning.

Application of the maximum treatment intensity would reduce MSO habitat suitability in nearly all MSO potential nesting habitat. In some areas, however, treatments of somewhat lower intensity could be used to improve MSO habitat quality. In total, 28,500 ha (13.2%) of potential MSO nesting habitat could be improved using treatments. This includes 7,900 ha of low-intensity thinning, 8,000 ha of very low-intensity thinning, and 12,600 ha of maintenance burning. Another 95,500 ha (44.1%) of habitat could be treated with maintenance burning without substantially altering its suitability.

3.3. Scenario results

After removal of constraint areas from consideration, 180,000 ha were identified as treatable under each of the three scenarios. Of this area 35,300 ha (19.6%) had CBD values below 0.1 kg/m^3 before treatment. These areas were therefore identified as not in need of treatment to reduce canopy fuels, although treatments to reduce ladder fuels and litter, as could be accomplished with maintenance burning, might be appropriate. Taking MSO habitat into consideration reduced the amount of total treatable area as well as the intensity with which many areas could be treated (Table 3).

Each of the scenarios resulted in a substantial reduction in CBD values in priority areas (Table 4). The maximum fuels reduction scenario reduced CBD by approximately 31% in priority areas, whereas the scenario maximizing protection of MSO habitat reduced CBD by approximately 18% (Table 4; Fig. 2a,b). However, we were not only interested in the percent reduction in CBD, but also the proportion of the landscape in which our targets for fire hazard reduction were met. Before treatment only 33% of the landscape had CBD values below 0.1 kg/m^3. In the maximum treatment scenario almost 63% of the landscape met this criterion, whereas almost 51% of the landscape met this criterion in the scenario maximizing protection of MSO habitat (Table 5; Fig. 2c,d).

4. Discussion

When formerly extensive ecosystems are reduced in area, fragmented, and altered in terms of natural processes and structure, the remaining area is often difficult to manage in a way that maintains the ecological integrity of the system as a whole while meeting the needs of all native species (Noss et al., 1995, 1997). In some cases not only do the needs of individual imperiled species conflict with one another, but restoration and management goals for the ecosystem do not coincide with species-specific management objectives. For example, management of the Colorado River to simulate natural floods through dam releases and restore beach habitat conflicts with objectives under the U.S. Endangered Species Act of 1973 to recover populations of the endangered Kanab ambersnail (*Oxyloma haydeni kanabensis*) and the endangered southwestern willow flycatcher (*Empidonax traillii extimus*), both of which require riparian vegetation that has increased in extent in some areas since completion of the dam (Stevens et al., 2001). In the Everglades of South Florida, not only do the water-depth requirements of the endangered snail kite (*Rostrhamus sociabilis*) and wood stork (*Mycteria americana*) conflict with each other, there is little assurance that the multi-billion dollar Everglades restoration plan will succeed in saving either of these or many other imperiled species in the region (Adams, 2006). For such reasons conservationists depend on the “safety net" of the Endangered Species Act and emphasize the habitat requirements of individual species, but in so doing they may delay or prevent restoration and management programs that would benefit the broader ecosystem (Noss et al., 1997).

Our study of ponderosa pine forests in northern Arizona indicates that even where ecosystem restoration and species conservation objectives are in conflict, spatial analysis and modeling approaches can be useful for designing strategies to minimize or eliminate the negative effects of restoration activities on imperiled species habitat. In some cases carefully designed thinning and burning treatments may actually enhance habitat quality for Mexican spotted owls, while aggressive treating of adjacent, forest stands can reduce fire hazard across the landscape, thus better protecting both human communities and owl habitat. Thus, far from being mutually exclusive objectives, forest restoration and the conservation of Mexican spotted owls can be compatible given careful planning on a broad scale, as was suggested by Beier and Maschinski (2003) and Noss et al. (2006a). Although aggressive thinning may reduce owl habitat quality in a specific location, when examined from a landscape perspective, well-placed thinning and prescribed fire can enhance overall owl habitat quality, both through improving the quality of marginal stands and reducing the likelihood of large losses of habitat from stand-replacing fire.

Mexican spotted owls occupy relatively dense ponderosa pine, pine–oak and mixed-conifer forests that, after decades of fire exclusion, tend to have an elevated likelihood of uncharacteristically severe crown fire. There is an inherent conflict in management objectives: protection of owl habitat appears not to coincide with goals for ecosystem restoration that involve reducing tree density by two-thirds or more of the present density. On the other hand, not all dense forests constitute owl habitat, and the reduction of tree densities in and near owl habitat should result in minor impacts on owls if properly located. Approximately 34% of our study area constitutes potential habitat for Mexican spotted owls. A conservation strategy, designed to minimize management in MSO habitat, might prohibit tree cutting and/or prescribed fire in these areas, leaving 56% (534,000 ha) of the current priority areas accessible to intensive management. Even applying treatments to only the 125,000 ha of area outside of MSO habitat and with CBD values above 0.15 kg/m^3 (see Table 2) would leave open management options that might take 15 years or more to apply, given the current capacity of regional forest managers (United States Forest Service, Region 3 Staff, pers. comm.). Hence, protection of spotted owl habitat and cautious implementation of fuels-reduction treatments over much of the landscape show surprisingly little potential for conflict in the near term.

Nevertheless, greater conflict arises near communities, where the demand for forest thinning and prescribed fire is more urgent, given the potential losses to stand-replacement fire. Our analysis indicates that approximately 33% (80,000 ha) of forested area falling within a common definition of the WUI can be considered potential habitat for MSO based on the models used in these analyses. Restriction of intensive forest management, such as mechanical thinning, in these areas could severely constrain management and compromise the protection of human communities. Our analyses show, however, that even without application of treatments that would seriously affect MSO habitat, managers could achieve approximately 60% of the fuels reduction that would be achieved if there were no restrictions on treatments. With reasonable tradeoffs considered in planning, such as largely treating in lower suitability owl habitat, this figure would rise to over 80% (Tables 4 and 5). Thus, there is considerable opportunity to move rapidly forward on ecosystem restoration objectives in the most critical areas from a human perspective, with minimal threat to owls.

Even if treatments were applied only outside of MSO habitat, substantial progress could be made on fuels reduction. Our analysis suggests that treatments in areas outside of MSO habitat increase the area unlikely to be at high risk for active crown fire from 80,000 ha (33%) to 124,000 ha (63%, Table 5). In addition, with careful planning of the timing, intensity, and placement of treatments, treated areas might slow the rate of fire spread, allowing greater opportunity for effective fire suppression, or for precipitation events that might extinguish an uncharacteristic crown fire (Agee et al., 2000; Finney, 2001). Nearby communities and MSO habitat could potentially benefit from these efforts.

Arguments that treatments within owl habitat are generally unnecessary are short-sighted. Treatments within MSO habitat could result in considerable benefits, both for communities and MSO habitat, albeit potentially more costly and difficult to implement than treatments outside of MSO habitat. The MSO management guidelines (USFWS, 1995) specifically allow for low-intensity thinning and prescribed burning, even inside PACs, provided nest sites and their immediate vicinity are excluded and disturbance to owls is minimized. Our analysis suggests that over 80% of potential MSO habitat within priority treatment areas could be treated without removing it from potential use by owls, and over 50% of the habitat could be treated, primarily with the sorts of low-intensity treatment recommended in the guidelines, without reducing its suitability for owls. In addition, the "restricted habitat" designation from the guidelines is meant to include areas that could be managed for "future owl habitat" (USFWS, 1995). Such areas may undergo very intense treatments if those treatments would result in improved owl habitat over time.

Treatments such as prescribed burning and light understory thinning would seem to have little effect on fire threat. However, in the Rodeo-Chediski fire area, even low-intensity prescribed burning appears to have reduced crown fire severity and tree mortality (Wilmes et al., 2002; Finney et al., 2005). Analysis of post-fire burn severity reveals a clear pattern of less severe wildfire where fuels-reduction treatments had occurred during the preceding 10 years. Furthermore, recent research suggests that owls can persist and successfully reproduce, at least over the short-term, in areas that have been burned at low to moderate severity (Bond et al., 2002; Jenness et al., 2004). If large portions of the landscape burn at high severity, however, owl populations are unlikely to persist (Jenness et al., 2004). Thus, well-designed treatments that reduce the extent and severity of wildfires while maintaining suitable owl habitat are likely to be beneficial in the long-term.

In addition to the direct benefit of reducing the likelihood of uncharacteristically severe fire, some treated areas can be managed for future owl habitat. Mexican spotted owls require large diameter trees and snags for nesting and roosting sites, and many areas currently suitable for owls are dominated by smaller trees. Low-intensity thinning of such areas would release water and nutrients that would allow retained trees to grow more rapidly (Mast, 2003). Over time, and with proper management, such areas should become more similar to the historical forests of the southwestern U.S. (Covington, 2003). These areas would have lower tree density, but would likely maintain high basal area and canopy cover as a result of the larger trees dominating the overstory. As these conditions are presumably those under which the MSO evolved, such areas would be likely to have forest structural conditions favorable to owls. In addition, they are likely to benefit key prey species, including deer mice (*Peromyscus spp.*), Mexican wood rat (*Neotoma mexicana*), and Mexican vole (*Mirotus mexicanus*) (Block et al., 2005).

Arguments over whether treatments designed to reduce fire hazard and restore forests to historical conditions can be undertaken while still providing protection to MSO populations have been pursued in an abstract, aspatial, and largely theoretical context. This has provided ample opportunities for all sides of the controversy to advance claims and counterclaims with little reference to on-the-ground reality. Whereas some accuse the USDA Forest Service of ignoring its own rules for the protection of threatened and endangered species (e.g., Center for Biological Diversity, 2003), the Forest Service and other agencies blame litigation and an overly bureaucratic approach to public lands management for the impasse in implementing restoration plans (Bosworth, 2002). Spatial analysis puts the perceived conflicts “on the map” and allows all parties to see where and how much imperiled species’ habitat is placed at risk by planned restoration actions. In our study, and we expect in many similar cases that have yet to be assessed, the area of conflict is significant and real, but it occupies a smaller extent of the landscape than one might expect based on the sweeping statements and long-term controversy that characterize the management debate. Instead of slowing on-the-ground management actions, spatial analysis including the modeling of management actions and their likely effects on forest structure and fire behavior, can provide managers with a sharp focus on management objectives and the tools needed for better coordinated and more effective planning (Sisk et al., 2006). Analysis of perceived conflicts and exploration of the compatibility of active forest management and imperiled species conservation allows managers to identify areas where conflict truly exists and design actions that are compatible with multiple management objectives. When conservation and restoration planning is scaled-up from a stand to landscape scale, many apparent conflicts disappear as management actions are spatially partitioned and prioritized (Noss et al., 2006a). Importantly, as shown in our analyses, this approach can identify large portions of the landscape where no conflict exists, and where relatively aggressive approaches to ecosystem restoration can be pursued without placing sensitive species at significant risk.

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