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Comparative Hazard Assessment for Protected Species in a Fire-prone Landscape

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ABSTRACT

We conducted a comparative hazard assessment for 325,000 ha in a fire-prone area of southwest Oregon, USA. The landscape contains a variety of land ownerships, fire regimes, and management strategies. Our comparative hazard assessment evaluated the effects of two management strategies on crown fire potential and northern spotted owl (*Strix occidentalis* caurina) conservation: 1) no action, and 2) active manipulation of hazardous fuels. Model simulations indicated that active management of sites with high fire hazard was more favorable to spotted owl conservation over the long term (75 years) than no management, given our modeling assumptions. Early in the model simulation, young seral stages were mostly responsible for high fire hazard, and active management in young stands tended to perpetuate that hazard. Later in the simulation, older seral stages accounted for most of the high fire hazard and active management could be used to ameliorate that hazard. At any given time period, ≤8% of the landscape was identified for treatment. Fire hazard fluctuated over time depending on vegetation regeneration, maturation, and response to treatments. Active management resulted in greater numbers of potential spotted owl territories in lower fire hazard conditions, particularly during later years of our simulation. Our results support the contention that short term risks to protected species from active management can be less than longer term risk of no management in fire-prone landscapes. Thus, a short term, risk averse strategy for protected species in fire-prone landscapes may not be the best long term alternative for conservation. We caution that this finding warrants landscape-level field evaluation and structured adaptive management and monitoring prior to broad scale adoption as environmental policy.

Key Words: comparative hazard assessment, risk analysis, spotted owls, fire, Oregon, hazardous fuels management

1. Introduction

Decades of grazing, fire exclusion and logging in dry forest landscapes of the Pacific Northwest, USA resulted in vegetation communities that, in many cases, currently contain uncharacteristic fuel conditions (Agee, 1993; Morgan et al., 2001; US General Accounting Office, 2003; Wright and Agee, 2004). Many of these dry forest landscapes currently provide habitat for protected species, including northern spotted owls (*Strix occidentalis* caurina) and several salmonids (Rieman and Clayton, 1997; Rieman et al., 2003; Courtney et al., 2004). Protected species habitat loss and alteration from wildfires in these dry forest landscapes is well documented (Courtney et al., 2004; Lint, 2005; Spies et al., 2006) and partly responsible for Federal legislation and policy that encourages hazardous fuels reduction (e.g., Williams and Hogarth, 2002; HFRA 2003).

Hazardous fuel reductions through active management on federal lands in the United States (US), particularly those associated with protected species habitats, are influenced by a complex interaction of environmental laws, regulatory agency interpretations, court decisions, and land management policy. Decisions on whether to allow active management are often based on precaution, particularly when compliance with the US Endangered Species Act (ESA) is involved (Mealey et al., 2005). The precautionary principle limits management action that could change the environment unless there is certainty that no immediate harm to protected species will result (Mealey et al., 2005). This implementation framework results in a short term, risk averse resource management strategy that, when combined with the dynamic tendencies of fire-prone landscapes, may put the resources that ESA was intended to protect at increased longer term risk (Irwin and Thomas, 2002; Mealey and Thomas, 2002; Rochelle, 2002; Mealey et al., 2005). Yaffee (1997) noted that this approach to implementing environmental policy results in poor long term direction and piecemeal solutions to complex problems.

Recent environmental laws codified in support of the U.S. National Fire Plan (http://www.forestsandrangelands.gov/NFP/index.shtml) recognize the temporal dimension of risk. Some laws and policy call for consideration of short and long term risks during ESA consultation on hazardous fuels reduction projects (e.g., HFRA 2003; Sec 106[c][3]; US Fish and Wildlife Service 2011). At the national level, evidence suggests that National Fire Plan implementation has not been hindered by regulatory constraints related to ESA (Hayes et al., 2008), but this trend is likely to change as land managers shift their focus to the wildlands, where much of the protected species habitat occurs (e.g., Ager et al., 2007). Explicit recognition that risk has a temporal dimension coupled with a need for tools to aid in implementation of the National Fire Plan brought comparative assessments to the forefront of a nation-wide effort to quantify fire hazards and risks on public lands. Without hazard and risk based assessments land management agencies cannot defend fuel reduction projects or make fully informed decisions about which effects and project alternatives are more desirable (GAO, 2004; Fairbrother and Turnley, 2005).

Comparative hazard assessment is defined as “an analysis and evalution of the physical, chemical and biological properties of the hazard” (Society for Risk Analysis, 2012). Comparative hazard assessment is recognized as a useful process for fulfilling the legislative requirements of the National Environmental Policy Act and ESA Section 7 consultation regulations issued by the U.S. Fish and Wildlife Service and National Oceanic and Atmospheric Administration-Fisheries Service (US-FWS and NOAA, 2003).

Several methodologies for conducting comparative hazard and risk assessments on fire and protected species habitats have been published (Hummel and Calkin, 2005; O’Laughlin, 2005; Roloff et al., 2005a,b; Ager et al., 2007). Comparative assessments for hazardous fuels projects involve complex data and models and thus, uncertainty with the outputs is generally high. In uncertain situations, resource managers and decision-makers have historically favored precaution and hence inaction (e.g., Ruhl, 2004; Prato, 2005; Schultz, 2008), even though vigorous trial and error is likely the best way to proceed (Wildavsky, 2000). Indicators of high fire hazard in dry western forests such as uncharacteristic fuel conditions (Graham et al., 2004), a prevalence of insect and disease infestations (Filip et al., 2007, Jenkins et al., 2008), wildfires of greater intensity and extent (Graham et al., 2004), and a warming and drying climate (Westerling et al., 2006; Allen et al. 2010) suggest that the potential for large-scale habitat alteration is increasing. Hence, decisions on acceptable levels of short and long term risks are warranted.

In this paper we present results from a comparative hazard assessment between no management and active fuels management in a fire-prone landscape of western North America. The fire management goal was to reduce hazard where fire risk was high while conserving protected species. Our objectives were to: 1) identify those forest types and seral stages in highest hazard conditions, 2) quantify the short and long term effects of active management and no management to northern spotted owls, and 3) portray our results in the context of current land management policies. Our approach provides a strategic evaluation in that it is coarse, occurs over substantial temporal and spatial scales, and relies on indices of ecosystem responses to management alternatives. Results from our model simulations should be used only as relative indices to evaluate trends in resource conditions.

2. Materials and Methods

The data, prescriptions, and processes used for our comparative hazard assessment have been described elsewhere (Roloff et al., 2005a,b; Mealey and Roloff, 2010). Our previous publications described model and data linkages, helped identify quantifiable hazard metrics, revealed some ecological characteristics of our landscape that warranted further scrutiny, and offered preliminary insights into hazards associated with three different management scenarios (Roloff et al., 2005a,b). Here we present an abbreviated study area description, synopsis of the modeling process, and modifications that were unique to the current model simulation.

2.1 Study Area

The Southwest Oregon Hazard Demonstration Project Area (SOHDPA) encompasses 336,000 ha, with its southwest boundary located approximately 19 km northeast of Medford, Oregon, USA (Fig. 1). The SOHDPA boundary is based on drainage units (Roloff et al. 2005a) and is located at the southern edge of the Western Cascades ecoregion (McNab and Avers, 1994). Elevations range from 300 to 2,200 m above sea level. Precipitation varies depending on elevation and topography. Average annual precipitation near the center of the project area is 107 cm (received mostly during October to June) with average annual temperatures ranging from lows of 2° to highs of 19°C (Western Regional Climate Center, Prospect, Oregon, http://www.wrcc.dri.edu/index.html). Fire is an important disturbance agent in the SOHDPA, with the landscape dominated (59%) by mixed-fire regime plant association groups (PAGS, *sensu* Atzet et al., 1996, Table 1). Frequent-fire regime PAGS (19% of the landscape) occur on lower to mid elevations. Evidence suggests that Native Americans frequently ignited these types to enhance forage production (South Cascades Late Successional Reserve Assessment, 1998). Moist forests or long-fire-regime PAGS (20% of the landscape) tend to occur at the higher elevations where lightning was and continues to be the primary fire ignition source (South Cascades Late Successional Reserve Assessment, 1998). Records of organized fire suppression in the SOHDPA date to 1902 and, coupled with lack of prescribed fire, has allowed the development of conditions suitable for spotted owl occupancy, insect and disease infestations, and large-scale, high intensity wildfires (Campbell and Liegel, 1996; South Cascades Late Successional Reserve Assessment, 1998). Statistics from 16 years (1987-2002) of fires that occurred in our study landscape indicated that ignition probability ranged from 0.03 to1.51 ignitions/100 ha (Roloff et al. 2005a). We documented 45 large (>2,500 ha) fires between 1992 and 2002. In the late 1990s land ownership included 74% federal, 17% private industrial and 9% other. Approximately 97% of the landscape is forested, with the majority (53%) federally reserved or subjected to management restrictions because of northern spotted owls; not all owls are centered on federal lands (Roloff et al., 2005a). Approximately 22% of the forested area is being managed for industrial timber production.

2.2 Comparative Hazard Model

Our comparative hazard model was based on projecting and managing vegetation states. Each vegetation state contained information on vegetation structure and composition (collectively called vegetation attributes; Roloff et al. 2005a). The vegetation attributes were then used as criteria for implementing management prescriptions and modeling fire and spotted owl responses (Roloff et al. 2005a,b). We developed an ecological land classification that portrayed different vegetation states. A vegetation state was defined by existing vegetation conditions (i.e., dominant tree species, density, and canopy structure as derived from 4 independent vegetation classifications of satellite imagery) and PAG. Map accuracy was >85% based on field sampling a subset of vegetation states (Roloff et al. 2005a). The resulting classification defined >900 potential vegetation states for mapping (mean patch size = 91 ha, min = 0.09 ha, max = 8,796 ha) in our study landscape; at any given time period about 400 states actually occurred. We compiled geo-referenced tree inventory plots (n=810) to quantify vegetative structure and composition of different vegetation states. The number of inventory plots per state ranged from 0 to 4. For those states without an inventory plot, we used the Forest Vegetation Simulator (FVS, West cascades Variant; Keyser 2008) to simulate vegetation dynamics for a plot that occurred in the same PAG. We simulated plot dynamics until the state-specific vegetation criteria were met. The simulated tree list was then assigned to the state. For those states with multiple plots, average vegetation condition (as portrayed by a tree list) was calculated and this average subsequently assigned to a state. State-based tree lists were then used in FVS to implement management prescriptions and project vegetation conditions 75 years into the future at 5-year time intervals. The FVS simulated natural seedling establishment (parameterized from field plots) and tree growth and mortality. The simulator produced an average tree (both live and dead) inventory for each time step and was programmed to assign a corresponding vegetation state from the diameter distribution of live trees.

In our original work (i.e., Roloff 2010a, b) we relied on the US Forest Service's strategic forest planning model (ForPlan; Iverson and Alston, 1986). Our previous results using ForPlan were based on optimizing net present value of timber while reducing fire hazard and protecting spotted owl habitat (Roloff et al., 2005a,b). Using this objective function we found that economic and regulatory constraints on hazardous fuels treatments resulted in an ineffective ForPlan solution for reducing fire hazard (Roloff et al., 2005a), similar to results observed by Hummel and Calkin (2005). In our current model the objective function specifically emphasized fire hazard reduction without economic or regulatory constraints. Thus, we were willing to sacrifice economic return and potentially some spotted owl territories to provide a less hazardous forest landscape. This rationale is consistent with recommended management direction for fire-prone ecosystems (Irwin and Thomas, 2002). In our revised model we allocated and implemented management prescriptions in ArcGIS 9.2 (Environmental Systems Research Institute, Redlands, California) and not in ForPlan. As vegetation states entered a hazard condition that triggered management, we assigned the appropriate prescription using queries and lookup tables. As a result, ArcGIS 9.2 allowed us to more tightly control the timing and spatial placement of prescriptions, an activity we found critical to producing a working solution (also see Ager et al., 2007, 2010; Finney et al., 2007).

We characterized fire hazard by using the US Forest Service’s FlamMap model (Finney, 2006). FlamMap output lends itself to landscape comparisons (e.g., pre- and post-treatment). FlamMap requires data on weather and wind, fuel characteristics for different vegetation states, and topography to predict areas of potential crown fire (Finney, 2006).

FlamMap inputs were generated from tree lists assigned to each vegetation state using existing FVS extensions (e.g., COVER; Moeur, 1985) and some additional programming code. FlamMap inputs included height to base of live tree crown, canopy bulk density, canopy closure and canopy height. Fuel models (13-class; Anderson, 1982) were assigned by conducting field visits to representative states and subsequently extrapolating the field data to unvisited states (Roloff et al., 2005 a). This process resulted in fuel characteristics that were mapped (by state) as FlamMap input landscapes. We created FlamMap landscapes immediately following implementation of the active management prescriptions. We assumed that logging debris and understory vegetation were managed to reduce hazard.

We conducted FlamMap simulations using preconditioned fuel moistures and extreme weather and wind conditions compiled from 10 years (1992-2002) of large-fire history data in Oregon. Initial fuel moisture conditions (weight of water/dry weight of fuel) were 5%, 8%, and 12% for 1, 10, and 100 hour fuel moistures, respectively; and 30% and 70% for duff and live vegetation, respectively. Weather was portrayed from August 19-24, with daily temperature and relative humidity ranging between 19 to 37°C and 53 to 16%, respectively, at average elevation. Wind speeds at 6 m height were modeled at 37 kph from the northwest (300°).

We verified pre- and post-treatment fuel conditions for each vegetation state by conducting field visits (described in Roloff et al. 2005a) and visually inspecting tree inventory data in Stand Visualization Software (USDA Forest Service, Pacific Northwest Research Station, Portland, OR; see Roloff et al. 2005b:214). We used the map of potential crown fire activity from FlamMap to identify those portions of the study area with surface or crown fire potential (Scott and Reinhardt, 2001). We were specifically interested in the hazard resulting from the occurrence of crown fire and not the mechanism for fire reaching the tree canopy. Thus we combined passive and active crown fire types into a single crown fire category.

In our current model, fuel reduction activities occurred only on frequent-fire PAGs with the potential for crown fire. Large contiguous areas of frequent-fire PAGS tended to occur at lower elevations in our landscape (Fig. 1). At the mid-elevation interface of frequent-, mixed-, and long-fire return interval PAGS, topographic aspect exerted a strong influence on PAG distribution. Frequent-fire PAGs tended to occur on southerly and westerly aspects at the mid elevations, whereas mixed- and long-fire PAGS occurred on northerly and easterly aspects. Our maps of potential spotted owl territory cores (i.e., the 40-80 ha area likely to contain a nest tree) at lower and mid elevations indicated a consistent positive association with the mixed- and long-fire PAGS on northerly and easterly aspects (Fig. 1). Hence, we hypothesized that hazardous fuels on the frequent-fire PAGS associated with lower and mid elevation spotted owl territories (i.e., the >1134 ha area that contains a core) could be treated and result in negligible negative effects on spotted owl habitat potential.

Vegetation states subjected to fuel reduction activities fell into two categories 1) older, multilayered forests with abundant surface and ladder fuels, and 2) young, dense regenerating forests. Under a typical multilayered forest management scenario, vegetation states were treated using a q-ratio prescription (Bailey and Covington, 2002), with repeated entries every 30 years. A typical prescription in our model was to sustain 10 to 20 m^2/ha basal area with thinning based on a q-ratio of 1.15 (i.e., 15% more trees in each successively smaller diameter class) over the size distribution ≤91 cm diameter, retaining fire tolerant species. Trees >91 cm diameter were fully retained. Simulations and field data indicate that this type of prescription can result in forest structures resistant to crown fire (Fulé et al., 2001; Stephens et al., 2009) and may positively contribute to wood fiber markets (Ince et al., 2008). The same q-ratio was applied to regenerating forests but no residual basal area target was identified.

We evaluated hazard to spotted owls by comparing potential crown fire activity to the location of modeled spotted owl territories. Spotted owl territories were mapped by combining a nesting habitat regression model that was developed for northern California (Zabel et al., 2003) with information on foraging habitat use from central and southern Oregon (Zabel et al., 1995; Franklin et al., 2000; Irwin et al., 2000). Nesting and foraging habitats were modeled into viable nesting cores using the process described by Roloff and Haufler (2001). Each nesting core was buffered by 1.9 km to delineate spotted owl territories. Size of these territories approximated the areas around spotted owl site centers subjected to ESA restrictions on forest management. In this restricted area we implemented fuel reduction prescriptions only if the spotted owl territory was in a high hazard condition (as defined below). We did not manage owl habitat with the objective of retaining habitat structure; a strategy that previously failed in our modeling framework (Roloff et al., 2005a). Instead, we focused treatments on reducing fire hazard, accepting the fact that some spotted owl territories may be lost or displaced as a result of management.

Our metric for hazard evaluation was the potential number of spotted owl territories in the frequent-fire portion of the landscape. The number of spotted owls impacted by a management action, not the amount of habitat impacted, is often an important component of judicial decisions (e.g., Oregon Natural Resources v. Allen, 2007). Our model compares the hazards or benefits of management to the hazards or benefits of no management at a particular time step:

$$\text{Management: (Total Provided}_{\mathit{Time}\ x} - \text{Total in High Hazard}_{\mathit{Time}\ x}\text{)}$$
$$- \text{No Management: (Total Provided}_{\mathit{Time}\ x} - \text{Total in High Hazard}_{\mathit{Time}\ x}\text{)}$$
$$= \text{Net Hazard or Benefit of Action}_{\mathit{Time}\ x}$$

where *Total Provided* refers to the total number of spotted owl territories located in our management area of interest. Here, our management area of interest is defined as those territories with >50% frequent fire PAG. *High Hazard* in our model is defined as those spotted owl territories with substantial crown fire potential (here defined as those territories containing >50% crown fire potential). In our model we used the amount of a spotted owl territory with crown fire potential (>50%) as an index to fire spread potential though more sophisticated modeling approaches exist (e.g., Ager et al., 2007). We focused our definition of high hazard on crown fire because spotted owls have been documented using habitats burned by low to moderate severity fires (reviewed by Bond et al., 2002). Our hazard model assumes that crown fire in >50% of a spotted owl territory will result in loss of that territory.

3. Results

3.1 Forest Types in Hazardous Conditions

Vegetation states on frequent-fire PAGS subjected to no management followed an expected trajectory of fire hazard. Young seral stages (classified as seedling-sapling in our analysis; Table 2) exhibited high crown fire hazard regardless of tree density due to low canopy heights and low heights to live crown. The majority of seedling-sapling seral stages on frequent-fire PAGS transitioned into a lower hazard designation 15 years into the simulation (at year 2018; Table 2), consistent with the relationship between plantation age and fire canopy damage observed by Thompson et al. (2011). As younger seral stages matured into single-storied, closed canopied, taller vegetation states (denoted as Small tree in our analysis) the potential for crown fire from a ground source ignition decreased because height to live crown increased. On some sites, these seral stages again entered a hazardous condition as they entered the Medium tree category in year 2038 (Table 2), likely 40 to 60 years after plantation establishment. This increase in hazard was associated with tree regeneration in the understories according to our FVS model. This hazardous condition persisted as Medium and Large tree vegetation states for the duration of our model simulation (Table 2). Medium and Large tree vegetation states with high fire hazard were multilayered (through canopy gaps or proliferation of shade-tolerant species) and densely stocked and accumulated abundant ladder fuels over time.

Early in the model simulation active management occurred mostly on seedling-sapling seral stage because these stands were hazardous and occurred on frequent-fire PAGS. In contrast to the no management vegetation trajectories, active management on seedling-saplings perpetuated fire hazard (as multi-storied small trees) into 2018 (Table 3). The amount of active management in Medium and Large tree vegetation states consistently increased over time (Table 3) as a result of two factors: 1) vegetation states maturing to the stage at which ladder fuels develop under tree canopies, and 2) spotted owl territories exceeded the fire hazard threshold and thus, older vegetation states in those territories were designated for management. In any given time period, ≤8% of the landscape was identified for active management (Table 3).

For the time steps we evaluated, crown fire potential ranged from 11% (Active Management, Year 2018) to 32% (No Management, Year 2078) of the landscape (Fig. 2). Crown fire potential in Year 2003 was mostly influenced by an abundance (27% of the landscape) of seedling-sapling seral stages. Although our management prescription reduced the stocking density of these young forests, they remained susceptible to crown fire (Table 3). At the landscape scale, actively managed young forests matured into single-storied, taller, closed canopy forests, and canopy fire hazard decreased (Year 2018), even though some of the managed younger forests on frequent-fire PAGS remained hazardous (see Small Tree, Table 3). As forests in the landscape continued to mature, crown fire potential increased from 2018 to 2078, the exception being for active management in 2078 (Fig. 2). For the entire landscape, crown fire potential for no management was higher than active management in all time steps, with differences more pronounced later in the model simulation as treatment of older forests dominated management activities (Fig. 2).

The majority (>58%) of crown fire occurred on frequent-fire PAGS regardless of management scenario (i.e., no management or active management), the exception being in Year 2003 during which >51% of the total crown fire occurred on mixed fire PAGS (Fig. 3). Thus, our decision to focus active management on frequent-fire PAGS was supported by the tendency for crown fire hazard to disproportionately increase on frequent-fire PAGS over time (Fig. 3). Crown fire persisted on frequent-fire PAGs under the active management scenario because fuels in those spotted owl territories designated as low hazard (i.e., ≤50% of the territory on frequent fire PAGS and ≤50% crown fire potential) were not being treated.

3.2 Fire Hazard to Spotted Owls

The number of modeled spotted owl territories encompassing >50% frequent-fire PAGs during our 75-year simulation ranged from 21 (No Management, Year 2038) to 7 (No Management, Year 2078) (Fig. 4). During a time period, these territories accounted for <18% of the total spotted owl territories modeled for our entire study area (Fig. 1). Active management occurred within spotted owl territories in 2003 (n=3 territories), 2018 (n=1), and 2078 (n=2). Spotted owl territories averaged 2,218 ha in size, and the amount of area managed within a spotted owl territory ranged from 731 ha (Year 2003) to 1,372 ha (Year 2078). When owl territories were identified as high hazard, active management was used to treat 33-62% of the territory on average.

We observed a peak in spotted owl territory numbers in 2038, followed by a steady decline (Fig. 4). This declining trend in spotted owl territories during later simulation years seems counter-intuitive in that larger, homogenous areas of older forests are often presumed to provide high quality spotted owl habitat (Forsman et al., 1984; Noon and Blakesly, 2006; Seamans and Gutiérrez, 2007). The decline in spotted owl habitat potential was caused by a reduction of suitable foraging habitat as portrayed by our habitat model. Our foraging habitat model ranked riparian zones and edges as important to spotted owl fitness; a pattern consistent with results from field studies conducted in comparable landscapes (Zabel et al., 1995; Franklin et al., 2000). According to our vegetation state-transition model and our spotted owl habitat model, no management resulted in a more homogenous forest landscape that lacked edges, whereas active management resulted in greater heterogeneity. Heterogeneity in dry forest landscapes of the Pacific Northwest is common (Spies et al., 2006; Kennedy and Wimberly, 2009) and, according to our owl habitat model, increases forage habitat potential.

Our model simulations suggest that active management helped reduce fire hazard without compromising spotted owl habitat potential (Fig. 4). The active management scenario resulted in more low hazard territories in 4 of the 5 simulation years; the exception being in 2038 when both management scenarios resulted in the same number of low hazard territories (Fig. 4). The benefits of active management were most pronounced during later simulation years (Fig. 4), as the cumulative effect of the management regime focused on fire-prone older forest types that also tended to support owls (Table 3).

4.0 Discussion

Active management at appropriate scales can effectively reduce crown fire hazard and not compromise northern spotted owl habitat potential if that management emphasizes fuel reduction and ecosystem restoration (as opposed to financial return) and focuses on those portions of the landscape at greatest hazard to crown fire (also see Gaines et al. 2010). Disagreement exists over the effects of fire on spotted owl population persistence, with some arguing that fires create elements of suitable habitat (Hanson et al., 2009a,b). Our results support the contention that small-scale heterogeneity caused by a patchy distribution of fire intensity (or, in our case, active management) is favorable to spotted owls in disturbance-prone landscapes; consistent with field observations of spotted owls using burned patches for foraging (Bond et al., 2009). However, conclusions from our comparative hazard analysis are based on a different premise and scale, i.e., the potential for large-scale habitat loss (i.e., >50% of a spotted owl territory) caused by extensive crown fire. Our premise is based on the observation that spotted owls will rarely use large areas that burn at high severity (Weatherspoon et al., 1992; MacCracken et al., 1996; Gaines et al., 1997; Bond et al., 2002). Thus, loss of habitat from large-scale crown fire is a primary conservation concern (Courtney et al., 2004).

Young conifer forests are susceptible to high levels of canopy damage from wildfires (Thompson et al. 2011). We contend that a thinning treatment of these younger seral stages actually prolongs the period of crown fire susceptibility because the canopies remain more open thereby encouraging retention of lower branches and the development of herbaceous and shrubby understories. Hence thinning programs should also include understory vegetation control and appropriate slash management. Our simulation results suggest that early seral stages should be encouraged to rapidly develop into closed-canopy forests to reduce understories and raise height to live crown (self-pruning of lower branches). As such, no management and lighter thinning treatments in denser stands appears to be the best option for younger seral stages.

Active management in older forests was effective at reducing crown fire potential, but we caution that logging debris and surface fuels must be managed for this prescription to be effective (e.g., piled and burned or broadcast prescribed fire; Stephens et al., 2009). Hazardous, older forest vegetation conditions are often associated with spotted owl habitat, particularly at lower elevations in fire-prone forests of the western US (Courtney et al., 2004; Ager et al., 2007). Spatial discontinuity of surface, ladder and crown fuels are recommended.

The percentage of landscape treated and positioning of treatments in the landscape are crucial management considerations. In our simulation, active management was implemented on ≤8% of our study area in any given 15 to 20 year time period. We reiterate that our approach focused management only on high hazard areas and did not attempt to explicitly influence fire spread or intensity by managing adjacent harvest units, topographic connectivity, and other vegetation states. Simulation modeling suggests that

>20% of a fire-prone landscape must be treated to begin altering fire behavior and help reduce the chances of spotted owl habitat loss (Ager et al., 2007). Our results suggest that effective and sustained fire hazard management and spotted owl conservation are compatible, though effective control of fire spread likely requires more tactical treatment.

Fire hazard to spotted owls fluctuates due to changes in fuel structure as vegetation regenerates, matures, and responds to management and natural disturbances. Vegetation dynamics in dry western forests are strongly influenced by disturbance agents like insects and disease (US Fish and Wildlife Service, 2011:III-7) in conjunction with fire (US Fish and Wildlife Service, 2011:III-6; Simard et al. 2011). Although our current results do not incorporate the likelihood for stochastic disturbance agents at different time steps, those capabilities exist (e.g., Roloff et al., 2005b). Based solely on fuel dynamics as vegetation states matured, our model indicated that lower elevation forests in the planning landscape were particularly hazardous in 2003 and 2078 and that hazard was absent in 2018 (Fig. 4). These results underscore the importance of long term assessments with periodic evaluations of hazard when deciding on a management trajectory for large landscapes (Fairbrother and Turnely, 2005; US Fish and Wildlife Service 2011:III-14). Given the assumptions of our simulation, basing a decision on a short term analysis (i.e., the next 15-years) would lead to the conclusion that no management is the best option for reducing fire hazard to northern spotted owls in SOHDPA. However, a decision based on a longer term analysis (i.e., 75 years) leads to the conclusion that active management is the best option. A hazard profile like that portrayed in Figure 4 improves the quality of management decisions because it permits a simultaneous evaluation of short, long, and periodic hazard.

We recommend that hazard profiles (e.g., Fig. 4) in dry forest types of the Pacific Northwest include a hazard calculation at least every 20 years and span sufficient time to include at least one forest successional cycle. Based on such a hazard profile, decision-makers can decide whether to subject protected species to no management periods of high potential volatility (e.g., time periods with high hazard conditions; Fig. 4) or to subject those species to management disturbances that result in less volatile conditions over the same time period. Our results confirm that impacts resulting from short term decisions compound and manifest themselves over long time periods with potentially profound consequences on protected species conservation.

4.1 Model limitations

Our findings are based on models that assume vegetation states can be accurately described and mapped, that states are defined at sufficient resolution to assume vegetative homogeneity, and that all areas of a particular state simultaneously transition into a new state (Ravindran et al., 1987). Additionally, we assumed that FVS accurately portrayed vegetation dynamics and that other major disturbances (like wildfire) did not occur. These simplifying assumptions have led some to question the utility of models for portraying vegetation dynamics (Olson et al., 1985). Models like those used in our study have a long history of utility in strategic forest planning and as such are useful for identifying broad vegetation categories for management (Iverson and Alston, 1986). Implementation of our model solution requires scaling down to site level decisions with management activities spread over multiple years.

Outcomes from our model were strongly influenced by our definition of high hazard; >50% of an owl territory occurring on frequent fire PAGS and >50% of the territory in vegetation conditions conducive to crown fire. This definition of high hazard may be conservative in light of recent publications noting increased vulnerability of western forests to uncharacteristic fire because of an increasingly warm and dry climate (Allen et al., 2010; Liu et al., 2010; US Fish and Wildlife Service 2011:III-6) and high incidence of insects and disease outbreaks (Campbell and Liegel, 1996; US Fish and Wildlife Service 2011:III-7). Additionally, surface fires may result in loss of spotted owl habitat, depending on fire intensity (Stephens and Finney, 2002; Schwilk et al., 2006). Our comparative hazard model permits future evaluation of alternative hazard definitions that might be more appropriate under changing landscape conditions. For example, if a warmer and drier climate increases the prevalence of insects and diseases, a lower hazard threshold may be warranted. In a different model simulation we demonstrated that the case for active management was even more compelling under a lower hazard threshold (i.e., 40% of an owl territory in crown fire potential; Mealey and Roloff, 2010).

We acknowledge that our model contains uncertainty and untested assumptions. Perhaps most importantly, we did not model vegetation heterogeneity within states (i.e., we assumed a single tree list represented average conditions across the landscape), resulting in a generalized portrayal of hazard and habitat covariates. We also did not include elements of unpredictable environmental stochasticities (e.g., fire, insect outbreaks). Thus, focus should remain on the relative comparisons and not the absolutes generated by our model. Habitat amount and quality thresholds used to portray spotted owl territories remain untested although findings from field studies were compiled to develop our habitat model. Also, we assumed that high hazard was likely to result in habitat loss; an outcome dependent on highly variable weather, climate, and fire factors.

4.2 Model application

Some have questioned the use of predictive models for natural resource planning and management (reviewed by Starfield, 1997); however, modeling is often the only alternative for informing decision-makers on long term impacts (Roloff et al., 2001). Whereas experimentation is recognized as the best approach for understanding the complexities of protected species conservation and fire risk management (Hanson et al, 2009a,b), proliferation of the precautionary principle has limited actual experimentation (Wildavsky, 2000). We emphasize the importance of continuously improving these models for use in natural resources decision-making through critical evaluation of model assumptions, inputs, outputs, and linkages. Additionally, strategic models (like the one presented herein) should be periodically (5-10 year intervals) implemented to incorporate landscapes changes that were not initially accounted for (e.g., large areas of tree mortality from insect outbreaks).

5.0 Conclusion

Our analysis of the interaction between management regime and northern spotted owl habitat conservation in a dry forest landscape of the Pacific Northwest suggested that active management reduces fire hazard and provides better habitat conditions for spotted owls over the long term. This finding provides specific hypotheses for field testing prior to broad scale implementation, with such testing focused on spotted owl responses to levels of management and fire within territories. A positive association between spotted owl dispersal and habitat alteration has been documented, though questions remain as to population-level impacts (Bond et al., 2002; Seamans and Gutiérrez, 2007; US Fish and Wildlife Service 2011:III-11). A testable hypothesis is that active management of fuels (i.e., using a q-ratio for thinning), if conducted tactfully, can occur on frequent-fire PAGS without compromising the quality of the spotted owl territory core. Our model simulations suggested that the locations of habitats suitable for spotted owl nesting cores remained relatively stationary over time, but that active management caused spatial shifts in suitable foraging resources within territories. Strategically, this active management strategy for fuels reduction and spotted owl habitat conservation appears to be a better alternative than no management.

Spotted owl habitat in many dry forest landscapes often exists over a mosaic of public and private ownerships as well as vegetation communities and fuel profiles. Ignoring fire hazard is not a socially or economically acceptable option in these mixed ownership landscapes. For example, some industrial forest landowners have questioned the long term value of owning timber assets in high-risk landscapes and, in some instances, these risk perceptions have factored into divestiture decisions. Our results should not be used as an argument for abandoning late successional reserves for spotted owl conservation in mixed ownership, dry forest landscapes. Rather, our results suggest that high risk areas in reserves can be tactfully managed to perpetuate their functionality as spotted owl habitat.

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Table 1. Fire regime and component plant association groups (PAGS, *sensu* Atzet et al. 1996) used for the Southwest Oregon Risk Demonstration Project.

Fire Regime	Plant Association Group (PAG)
Frequent	Warm, Dry Douglas-fir
	Warm, Dry White Fir-Grand Fir
Mixed	Warm, Moist Douglas-fir
	Warm, Moist White Fir-Grand Fir
	Cool White Fir-Grand Fir
	Shasta Red Fir
Long	Pacific Silver Fir
	Western Hemlock
	Mountain Hemlock

Table 2. Vegetation states on frequent-fire plant association groups (PAGs) with crown fire potential resulting from no management by time period. Table values represent ha (% of total landscape).

	Simulation Year				
Vegetation State[1]	2003	2018	2038	2058	2078
Seedling-sapling	24,313 (7)	22 (<1)	.[2]	.	.
Single and Multi-storied					
Small tree	236 (<1)	1 (<1)	.	10 (<1)	.
Medium tree	3,219 (1)	5,827 (2)	34,226 (10)	35,412 (11)	6,831 ((2)
Large tree	115 (<1)	640 (<1)	4,851 (1)	13,261 (4)	43,447 (13)
X-large tree	65 (<1)	.	.	.	.
Multi-storied					
Old Growth	.	.	.	.	.

[1] Seedling-sapling = average quadratic mean diameter (QMD) 1.3 – 12.7 cm diameter breast height (dbh); Small tree = 12.8 – 38.1 cm QMD; Medium tree = 38.2 – 50.8 cm QMD; Large tree = 50.9 – 76.2 cm QMD; X-large tree = 51.0 – 127.0 cm QMD; Old Growth = X-large tree size criteria plus trees >127.0 cm dbh with snags, cull trees, and abundant downed wood.

[2] No area identified.

Table 3. Vegetation states on frequent-fire plant association groups (PAGs) with crown fire potential identified for Active Management by time period. Table values represent ha (% of total landscape) subjected to management in each time period.

	Simulation Year				
Vegetation Structure[1]	2003	2018	2038	2058	2078
Seedling-sapling	16,396 (5)	30 (<1)	13 (<1)	1 (<1)	.[2]
Single and Multi-storied					
Small tree	163 (<1)	14,438 (4)	.	.	.
Medium tree	1,623 (<1)	505 (<1)	4,428 (1)	6,388 (2)	15,287 (5)
Large tree	55 (<1)	65 (<1)	1,715 (1)	3,943 (1)	11,642 (3)
X-large tree	15 (<1)	54 (<1)	.	.	.
Multi-storied					
Old Growth	.	1 (<1)	.	.	.

[1] Seedling-sapling = average quadratic mean diameter (QMD) 1.3 – 12.7 cm diameter breast height (dbh); Small tree = 12.8 – 38.1 cm QMD; Medium tree = 38.2 – 50.8 cm QMD; Large tree = 50.9 – 76.2 cm QMD; X-large tree = 51.0 – 127.0 cm QMD; Old Growth = X-large tree size criteria plus trees >127.0 cm dbh with snags, cull trees, and abundant downed wood.

[2] No area identified.

Figure Captions

Figure 1. Study area location, major bodies of water, fire regime (*sensu* Atzet et al. 1996), and northern spotted owl territory centers (2003) for the Southwest Oregon Hazard Demonstration Project.

Figure 2. Crown fire potential (modeled via FlamMap; Finney, 2006) for the Southwest Oregon Hazard Demonstration Project landscape by simulation year for active management and no management scenarios.

Figure 3. Association between crown fire potential (modeled via FlamMap; Finney, 2006) and fire regime by simulation year for active management and no management scenarios in the Southwest Oregon Hazard Demonstration Project.

Figure 4. Modeled northern spotted owl territories and corresponding hazard ranking by simulation year for active management and no management scenarios in the Southwest Oregon Hazard Demonstration Project. Numbers above each management bar denote the net benefit or loss of territories resulting from management.