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Effects of Fire and Commercial Thinning on Future Habitat of the Northern Spotted Owl

Dennis C. Odion[1,*], Chad T. Hanson[2], Dominick. A. DellaSala[3], William L. Baker[4] and Monica L. Bond[5]

[1]*Earth Research Institute, University of California, Santa Barbara, California. 93106. Environmental Studies Department, Southern Oregon University, Ashland, Oregon, 97520*

[2]*Earth Island Institute, 2150 Allston Way, Suite #460, Berkeley, California, 94704*

[3]*Geos Institute, Ashland, Oregon, 97520*

[4]*Program in Ecology and Department of Geography, University of Wyoming, Laramie, WY, 80271*

[5]*Wild Nature Institute, PO Box 165, Hanover, NH 03755*

Abstract: The Northern Spotted Owl (*Strix occidentalis caurina*) is an emblematic, threatened raptor associated with dense, late-successional forests in the Pacific Northwest, USA. Concerns over high-severity fire and reduced timber harvesting have led to programs to commercially thin forests, and this may occur within habitat designated as "critical" for spotted owls. However, thinning is only allowed under the U.S. Government spotted owl guidelines if the long-term benefits clearly outweigh adverse impacts. This possibility remains uncertain. Adverse impacts from commercial thinning may be caused by removal of key habitat elements and creation of forests that are more open than those likely to be occupied by spotted owls. Benefits of thinning may accrue through reduction in high-severity fire, yet whether the fire-reduction benefits accrue faster than the adverse impacts of reduced late-successional habitat from thinning remains an untested hypothesis. We found that rotations of severe fire (the time required for high-severity fire to burn an area equal to the area of interest once) in spotted owl habitat since 1996, the earliest date we could use, were 362 and 913 years for the two regions of interest: the Klamath and dry Cascades. Using empirical data, we calculated the future amount of spotted owl habitat that may be maintained with these rates of high-severity fire and ongoing forest regrowth rates with and without commercial thinning. Over 40 years, habitat loss would be far greater than with no thinning because, under a "best case" scenario, thinning reduced 3.4 and 6.0 times more dense, late-successional forest than it prevented from burning in high-severity fire in the Klamath and dry Cascades, respectively. Even if rates of fire increase substantially, the requirement that the long-term benefits of commercial thinning clearly outweigh adverse impacts is not attainable with commercial thinning in spotted owl habitat. It is also becoming increasingly recognized that exclusion of high-severity fire may not benefit spotted owls in areas where owls evolved with reoccurring fires in the landscape.

Keywords: Fire rotation, forest regrowth rate, forest thinning, future habitat, habitat loss, late-successional forest, policy implications, severe fire, spotted owl.

INTRODUCTION

Conservation of the emblematic Northern Spotted Owl (*Strix occidentalis* ssp. *caurina*) in the Pacific Northwest of North America has become a global example of balancing conflicting land management goals (DellaSala and Williams 2006). Concern over degradation of the owl's dense, late-successional forest habitat led to the 1994 Northwest Forest Plan (NWFP). The NWFP shifted management on ~100,000 km^2 of federal USA forestlands from an emphasis on resource extraction to embrace ecosystem management and biodiversity conservation goals. Under the NWFP, ~30% of federal lands traditionally managed for timber production were placed in late-successional reserves that emphasized conservation goals and limited timber harvesting (USFS/USDI 1994).

Over the last decade, managers and policy makers have become increasingly concerned about high-severity fire and reduced timber harvesting in NWFP dry forests (e.g., Spies *et al.* 2006, Power 2006, Thomas *et al.* 2006, Ager *et al.* 2007, USFWS 2011). Forest thinning has been viewed as a solution for controlling fires in dry forests throughout western North America (Agee and Skinner 2005, Stephens and Ruth 2005) and commercial criteria have been included to pursue timber harvest goals (Johnson and Franklin 2009, Franklin and Johnson 2012). Commercial thinning prescriptions currently being implemented under these

*Address correspondence to this author at the Earth Research Institute, University of California, Santa Barbara, California. 93106. Environmental Studies Department, Southern Oregon University, Ashland, Oregon, 97520; Tel/Fax: 541 821-0738; E-mail: dennis@odion.name

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criteria may remove up to one-half of forest basal area, and may also include patch cutting or small clear cuts (USDI 2011). Commercial thinning is now proceeding rapidly without a full understanding of the long-term risks.

For spotted owls, thinning and associated activities often remove or reduce key habitat features in direct proportion to the intensity of the commercial prescription. Key spotted owl habitat features that may be reduced or removed directly or indirectly include high tree density and canopy cover (King 1993, Pidgeon 1995), recently killed pines (*Pinus* spp.) and abundant snags (Pidgeon 1995), multiple tree layers, with abundant medium and small white fir (*Abies concolor*) or Douglas-fir (*Pseudotsuga menziesii*) (King 1993, Pidgeon 1995, Everett *et al.* 1997, Irwin *et al.* 2012), large volume of mature-sized down logs (Pidgeon 1995), shrubs (King 1993, Pidgeon 1995, Irwin *et al.* 2012) and trees with heavy mistletoe infections (Hessburg *et al.* 2008), which are essential for spotted owl nesting (USFWS 2011). Thinning or contemporary harvest near the nest or activity center has been shown to displace Northern Spotted Owls (Forsman *et al.* 1984, King 1993, Hicks *et al.* 1999, Meiman *et al.* 2003). Telemetry studies on California Spotted Owls (*Strix occidentalis* ssp. *occidentalis*) in the Sierra Nevada found that owls avoided Defensible Fuel Profile Zones (an intensive thinning treatment) (USFS 2010). Unoccupied California Spotted Owl territories had a lower probability of re-occupancy after timber harvest, even when habitat alterations comprised <5% of a territory (Seamans and Gutiérrez 2007). In addition, Barred Owls (*S. varia*), which out-compete spotted owls (Dugger *et al.* 2011), use younger and more open forests compared to Northern Spotted Owls (Wiens 2012).

Studies also have found negative impacts of thinning to northern flying squirrels (*Glaucomys sabrinus*), the primary prey of Northern Spotted Owls in most of its range (Waters and Zabel 1995, Waters *et al.* 2000, Carey 2001, Ransome and Sullivan 2002, Gomez *et al.* 2005, Ransome *et al.* 2004, Bull *et al.* 2004, Meyer *et al.* 2007, Wilson 2008, Holloway and Smith 2011, Manning *et al.* 2012). Negative effects may persist for 15 years or longer (Wilson 2008). In addition, openings between trees from thinning may create barriers, due to predator avoidance, for flying squirrels to cross using its gliding locomotion (Manning *et al.* 2012). Thinning has also been found to have negative effects on the abundance of other main prey species for Northern Spotted Owls such as red-backed voles (*Myodes californicus*) (Suzuki and Hayes 2003) and woodrats (*Neotoma cinerea, N. fuscipes*) (Lehmkuhl *et al.* 2006).

Because of the many conflicts between thinning and spotted owl conservation, some authors have recommended that treatments aimed at controlling fire avoid spotted owl habitat and instead treat vegetation elsewhere that is the most flammable and strategic for accomplishing fuel treatment goals (Gaines *et al.* 2010). The 2011 Recovery Plan for the Northern Spotted Owl, the blueprint for management of this species on federal lands in the region (USFWS 2011), contains the proviso that long-term benefits to spotted owls of forest thinning treatments must clearly outweigh adverse impacts (USFWS 2011). The U.S. Fish and Wildlife agency that developed the plan suggested that benefits over time might accrue from a net increase in habitat because fire disturbances would be reduced (USFWS 2011). But whether the benefits would outweigh the impacts remains uncertain due to limitations of previous assessments.

Previous assessments of the efficacy of thinning treatments in reducing fire disturbances in spotted owl habitat (Wilson and Baker 1998, Lee and Irwin 2005, Roloff *et al.* 2005, 2012, Calkin *et al.* 2005, Hummel and Calkin 2005, Ager *et al.* 2007, Lehmkuhl *et al.* 2007) have not incorporated the probability of high-severity fires occurring during the treatment lifespan. The effect of this is to overestimate treatment efficacy in potentially controlling fire or fire behavior (Rhodes and Baker 2008). Nor have the effects of recruitment of dense, late-successional forest that act to offset loss from fire been included in prior assessments. In addition, impacts of the kind of commercial thinning treatments being implemented to address dry forest concerns have not been fully considered for the owl or its prey (e.g., Ager *et al.* 2007, Lehmkuhl *et al.* 2007, Roloff *et al.* 2012). Current commercial thinning prescriptions being implemented in dry forests specifically identify desired future conditions to be maintained (e.g. Johnson and Franklin 2009) that have basal area and other structural targets mostly well below the minimum levels that have been found in spotted owl nesting, roosting and foraging habitat (NRF) in dry forests. For example, basal area targets in a project in southwest Oregon designed to demonstrate the thinning prescriptions in dry forest spotted owl habitat were 13.75-27.5 m^2/ha (USDI 2011), while stands < 23 m^2/ha very rarely support spotted owl nesting territories (Buchanan and Irwin 1995). In addition, the Recovery Plan (USFWS 2011) permits thinning in core areas, but emphasizes treating areas outside of core areas, so there is a need for assessment of impacts outside core areas as well. Areas outside cores may be essential for foraging and be part of the breeding season home range. Furthermore, owls often move outside core areas (USFWS 2011). Lastly, available habitat outside existing cores may become important to owl recovery, particularly if spotted owls are displaced from higher quality habitat by Barred Owls (Dugger *et al.* 2011).

To assess whether benefits of commercial thinning outweigh adverse impacts to spotted owls in dry forests (USFWS 2011), quantitative assessments are needed that allow for direct assessment of the amounts of any dense, mature or late-successional habitat that would be reduced by both commercial prescriptions and severe fire. Accordingly, we calculated these amounts by projecting them over 40 years and incorporated into our calculations the effects of forest regrowth. For our calculations, we used empirical data on fire and forest regrowth from the potential habitat within the two dry forest regions where spotted owls occur, the Klamath and dry Cascades of California, Oregon, and Washington, that are subject to thinning. We analyzed each region separately using region-wide data. Conservation planning for spotted owls commonly occurs at the scale of these regions. For our thinning treatment, we chose a “best” scenario for minimizing the amount of dense, late-successional forest to be treated (Lehmkuhl *et al.* 2007); while we used an optimistic scenario for treatment efficacy, assuming that a 50% reduction in high-severity fire would occur (Ager *et al.* 2007). We also illustrate the effects of varying treatment amount and efficacy. To calculate rotations of severe fire in the forests of the study area, we used available fire data from a time period, 1996-2011, which includes exceptionally large, rare fire events. Our approach may be useful to managers interested in maintaining habitat for other species that rely on dense forests in fire-prone regions (Odion and Hanson 2013).

METHODS

Study Area

We analyzed fire and forest recruitment trends in 19,000 km^2 of dry forests in the Klamath and 18,400 km^2 in the Cascades provinces. As in Hanson *et al.* (2009), we analyzed only late-successional, or "older" forests present in 1995, as mapped by Moeur *et al.* (2005). This is a small fraction of the dry forest regions. Our analysis was further restricted to federal lands. Mapping by Moeur *et al.* (2005) corresponds to mid-montane forest zones where Northern Spotted Owls occur. These montane forest zones include forests dominated mainly by true firs (*A. grandis, A. concolor*), Douglas-fir (*Pseudotsuga menziesii*), and Ponderosa pine (*P. ponderosa*): Other conifers found in the central and northern Cascades in dry forests frequented by spotted owls are western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis*), and limited amounts of western red cedar (*Thuja plicata*) and Engelmann spruce (*Picea engelmannii*). Forests in the Klamath are noted for high conifer diversity, with species such as incense cedar (*Calocedrus decurrens*) commonly found in the range of spotted owls. A variety of broad-leaved evergreen trees, such as madrone (*Arbutus menziesii*) and tanoak (*Lithocarpus densiflorus*) are also characteristic of these forests (Whittaker 1960).

Quantifying Future Habitat

We determined existing rates of dry-forest redevelopment following stand initiation in the forests of the study regions as delineated by Mouer *et al.* (2005) using the extensive U.S. Forest Service Forest Inventory and Analysis (FIA) forest monitoring data (http://www.fia.fs.fed.us/tools-data/). FIA is a monitoring system based on one permanent, random plot per ~2400 ha across forested lands. We excluded plots from forests not used by spotted owls (e.g. lodgepole pine, oak forest) and from non-conifer vegetation and non-federal lands. Most of these plots were already excluded by the mapping by Mouer *et al.* (2005) that delineated the study area.

An FIA plot consists of a 1-ha area. For tree measurements, this area is sub-sampled with four circular subplots that are 0.1 ha for large-tree sampling and 0.017 ha for smaller-tree sampling (defined by region). The diameter-at breast-height (dbh) and crown position of each tree and the ring count from two cores from dominant/codominant trees are measured in each subplot (USFS 2010). Stand age for an FIA plot is determined from the average of all ring counts from sub-plot samples, weighted by cover of sampled trees, and 8 years are added for estimated time to grow to breast height (1.4 m). We used live-tree dbh data to prepare regressions with stand age.

FIA data were available from 2001-2009, comprising 90% of the plots available within our study area. A total of 581 plots from the Klamath and 441 from the dry Cascades were considered, representing 13,944 and 10,680 km^2 in each region, respectively. The number would be higher, but we eliminated 139 plots in the Klamath and 141 in the Cascades that had different stand-initiation dates from different subplots of the main FIA plot. This situation occurs throughout the study area due to the patchy nature of mixed-severity fire. Including all the subplots as individual plots creates a larger sample size, but we chose not to do this because some individual locations would be overrepresented. Most importantly, both approaches lead to the same results.

We analyzed fire severity from 1996-2011 in late-successional, or "older" forests mapped by Moeur *et al.* (2005). For 1996-2008, we used the Monitoring Trends in Burn Severity (MTBS) (http://www.mtbs.gov/) data. We used the ordinal classification from MTBS, as MTBS analysts determine for each fire where significant thresholds exist in digital prefire and postfire images, supplemented with plot data and analyst experience with fire effects. In plot data, a composite burn index that sums mortality by vegetation stratum is used to identify high fire severity (see http://www.mtbs.gov/). For 2009-2011, we obtained U.S. Forest Service digital data (http://www.fs.fed.us/postfire-vegcondition) and classified these data following Miller and Thode (2007). We could not use pre-1996 MTBS fire severity data because the pre-burn map of spotted owl forest habitat is from 1995 (Moeur *et al.* 2005). From severity data we calculated high-severity fire rotation (FR^{hs}), the expected time to severely burn an area equivalent to the area of interest once, or the landscape mean interval for severe fire (Baker 2009).

We calculated annual high-severity fire and forest regrowth rates to future proportions for early-, mid- and mature or late-successional forests, denoted herein by "E," "M," and "L," respectively, using annual time steps. We defined late-successional forests by selecting a value, 27.5 m^2/ha. This amount corresponds with the maximum basal area that would be left according to currently implemented thinning prescriptions (USDI 2011). This is somewhat higher than the minimum basal area where spotted owls have been found to nest in dry forests. For example, the mean value minus one standard deviation in all the dry forest stands studied by Buchanan *et al.* (1995) was 23 m^2/ha. However, we did not want to identify the rate of regrowth to the very minimum basal area that constitutes habitat, but regrowth to a basal area more likely to function as habitat. Mid- and early-successional forests were defined as 13.5-27.5 and <13.5 m^2/ha tree basal area, respectively. We separated mid-successional from early-successional forest because, mid-successional forests may be included in thinning treatments, but early-successional forests may not. Thinned forest ("T") was our fourth vegetation state. The forest states are diagramed in Fig. (**1**). The proportion of each state in the landscape at time *t*, defined a vector (P_t^E, P_t^M, P_t^T, P_t^L). Transition probabilities ϕ_t^{rs} equaled the probability that any portion of state *r* at time *t* transitions to

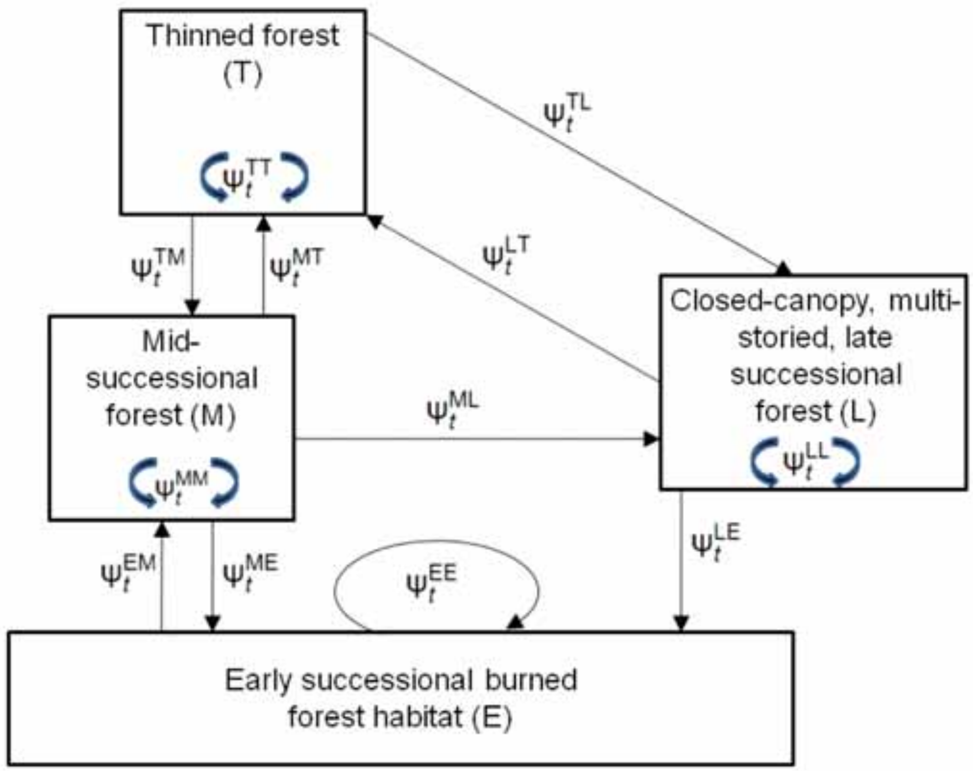

Fig. (1). State (boxes) and transition (arrows) model for dry Pacific Northwest Forest vegetation with fire disturbances and thinning. Variables are the transition rates between states indicated by the associated arrow.

state *s* at time *t* + 1, allowing calculation of future amounts of each forest type using the following equation:

$$\begin{bmatrix} \phi_t^{EE} \phi_t^{ME} \phi_t^{TE} \phi_t^{LE} \\ \phi_t^{EM} \phi_t^{MM} \phi_t^{TM} \phi_t^{LM} \\ \phi_t^{ET} \phi_t^{MT} \phi_t^{TT} \phi_t^{LT} \\ \phi_t^{EL} \phi_t^{ML} \phi_t^{TL} \phi_t^{LL} \end{bmatrix} \begin{bmatrix} P_t^E \\ P_t^M \\ P_t^T \\ P_t^L \end{bmatrix} = \begin{bmatrix} P_{t+1}^E \\ P_{t+1}^M \\ P_{t+1}^T \\ P_{t+1}^L \end{bmatrix} \quad (1)$$

The initial proportions, $P_{t=0}^{E\text{-}L}$ of the three natural-forest states were from the FIA basal-area analyses, with thinned forests considered zero for simplicity and because of lack of data. The annual transition from mid- and late- to early-successional forest from high-severity fire (ϕ_t^{LE}, ϕ_t^{ME}) was $1/FR^{hs}$. Early-successional forests also burned at this rate (ϕ_t^{EE}). Annual rates of forest redevelopment were from the inverse of the growth period ($1/G^{EM}$) to reach 13.5 m^2/ha live-tree basal area, or to grow from 13.5 to 27.5 m^2/ha live-tree basal area ($1/G^{ML}$), calculated from the regression of live basal area on age (see results). Lower-severity fire can reduce basal area from >27.5 m^2/ha basal area to <27.5 m^2/ha. However, this transition is already considered in the regrowth rate, which also incorporates the effects of lower-severity fires that have occurred on rates of forest redevelopment. Because natural disturbances that may temporarily lower basal area are captured in the transitions from early- to late-successional forest, the transition from late to mid-successional forest was set to zero. Transition rates to thinned forest were based on treatment within 20 years, beginning in year *t* + 1, of the mid- and late-successional forests present at *t* = 0 (see Table **1** for annual rate). Based upon the empirical FIA and MTBS data described above, we used these transitions (Table **1**) and Eq. 1 to project forward 40 years (see sample calculation in the Supplementary Materials). We chose this time interval because it represents one cycle of thinning and forest recovery.

Next, we calculated the effects of varying levels of thinning, and treatment efficacy (in terms of the effect on high-severity fire rotation intervals), over the study period. According to an analysis of a spotted owl landscape by Lehmkuhl *et al.* (2007), a "best" scenario for minimizing the short-term adverse impacts of thinning while reducing fire frequency and severity was one that treated only 22% of the landscape, and limited thinning in nesting, roosting, and foraging habitat to 21% of the area of this habitat. We used this prescription in our calculations to illustrate the effects under a best-case scenario. In our calculations, the amount of mid-successional forest thinning differed between the two regions because amounts of both mid- and late-successional forests were not the same. We also considered the effects of treating from 0 to 45% of forests, holding constant the proportions of treatments that were in late-successional vs. mid-successional forests.

We assumed that there would be no high-severity fire in treated forests over the treatment lifespan. We additionally assumed that thinning 22% of the landscape would lower the amount of high-severity fire in the unthinned landscape by half. This is based on the findings of Ager *et al.* (2007) who simulated the effects of wildfire ignitions following strategic

Table 1. Annual transition probabilities used in transition matrices for each scenario analyzed for dry provinces within the range of the Northern Spotted Owl. FR^{hs} is the high-severity fire rotation. G is the time required for stands to grow from early to mid- (EM) or mid- to late-successional (ML) forest (see Table 2). K = Klamath, C = Cascades. R is the amount that high severity fire is reduced by thinning (50% reduction at 22 percent of late-successional forest thinned).

Transition Probabilities	No Treat	Treat 22% Maintain	Treat 22% Recover
ϕ_t^{LE}	$1/FR^{hs}$	$(1/FR^{hs}-R)$	$(1/FR^{hs}-R)$
ϕ_t^{EM}	$1/G^{EM}$	$1/G^{EM}$	$1/G^{EM}$
ϕ_t^{ET}	0	0	0
ϕ_t^{EL}	0	0	0
ϕ_t^{ME}	$2/FR^{hs}$	$2/FR^{hs}$	$2/FR^{hs}$
ϕ_t^{ML}	$1/G^{ML}$	$1/G^{ML}$	$1/G^{ML}$
ϕ_t^{EE}	$1-1/G^{EM}$	$1-1/G^{EM}$	$1-1/G^{EM}$
ϕ_t^{MM}	$1-1/G^{ML}-(1/FR^{hs})$	$1-1/G^{ML}-(1/FR^{hs}-R)-\phi_t^{MT*}$	$1-1/G^{ML}-(1/FR^{hs}-R)-\phi_t^{MT*}$
ϕ_t^{MT*}	0	K = 0.033 C = 0.018	K = 0.033 C = 0.018
$\phi_t^{TM\dagger}$	0	0	K = 0.033 C = 0.018
ϕ_t^{TE}	0	0	0
$\phi_t^{TT\dagger}$	0	0	$1-\phi_t^{TL}-\phi_t^{TM\dagger}$
$\phi_t^{TL\dagger}$	0	0	K = 0.0114 C = 0.0105
ϕ_t^{LM}	0	0	0
ϕ_t^{LT*}	0	K = 0.0114 C = 0.0105	K = 0.0114 C = 0.0105
ϕ_t^{LL}	$1-1/FR^{hs}$	$1-1/FR^{hs}-R-\phi_t^{LT}$	$1-1/FR^{hs}-R-\phi_t^{LT}$

*Only in effect for the first 20 years.
†Does not take effect until after 20 years.

thinning treatments in a spotted owl landscape. When <22% of the landscape was affected at any given time (such as any time prior to year 20 when the full treatment would be incomplete, or after one-time treatments began to recover, or for scenarios with <22% of the landscape treated) the same ratio of area treated to reduction in high-severity fire (22% treat: 50% reduction in fire) was used to reduce the area burned at high severity (see Supplementary Material for an illustration). Thus, the amount that fire was reduced by thinning increased with each year as a function of the total area thinned (all other variables were constant). Ager *et al.* (2007) found little additional effect of treatments in reducing wildfires as treatment level increased beyond 20%, so we did not calculate greater reductions in fire as treatment levels went from 22-45%. However, we additionally calculated future habitat amounts as a function of fire rotation to evaluate the effects of varying treatment efficacy, in which case we did calculate the reduced amount of habitat burned severely. This amount is the dependent variable in our summary figures. Treatment lifespan was assumed to be 20 years (Rhodes and Baker 2008) for "one-time thinning," or maintained in perpetuity over the 40 years for "maintained." A sample calculation using the model (equation 1) is presented in the Supplementary Material.

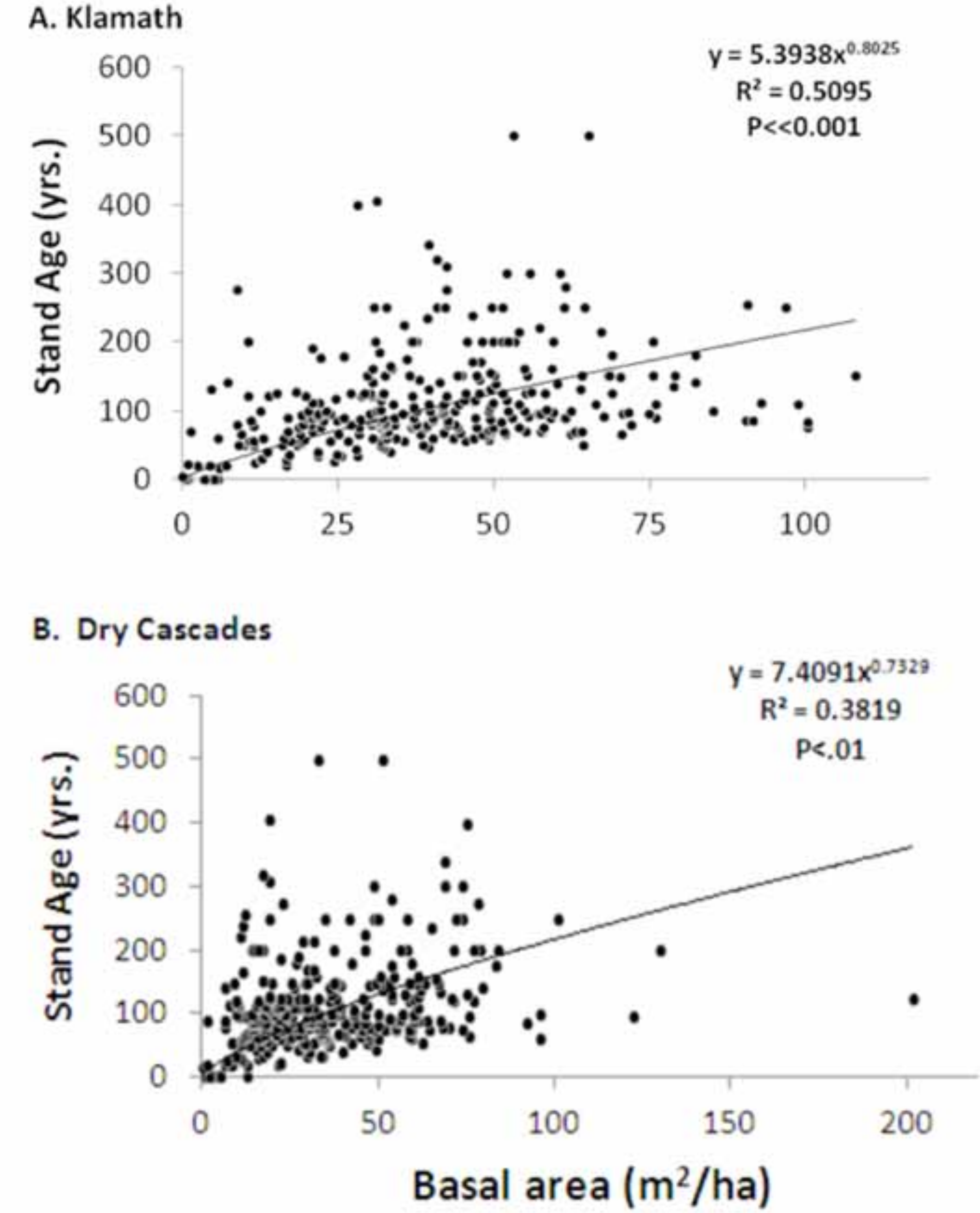

Fig. (2a-b). Scatterplots of live-tree basal area per hectare and stand age from US Forest Service FIA data for the A. Klamath region and B. dry Cascades region.

The only owl habitat we considered for impacts from thinning was suitable nesting, roosting, and foraging (so called NRF habitat). Because treatments aimed at demonstrating the type of thinning to be implemented in spotted owl habitat reduce basal area down to 13.75-27.5 m^2/ha, mostly well-below the minimum amounts for NRF habitat (Pidgeon 1995, Buchanan and Irwin 1998, LeHaye and Gutiérrez 1999), and because treated forests also have reduced amounts of key habitat features like multi-canopy structure, down wood, small firs and mistletoe infections, the area affected by these treatments will largely correspond to the amount of habitat lost. Thinning may also render adjacent, unthinned forest unsuitable or less suitable (Seamans and Gutiérrez 2007), but we did not account for this effect. The lifespan for thinning treatments that we used was 20 years for one-time thinning (Rhodes and Baker 2008), and 40 years for maintained treatments. Transition from late- to early-successional vegetation due to high-severity fire also was considered habitat loss. This may overestimate the impacts of fire on Northern Spotted Owl foraging habitat (Bond *et al.* 2009, USFWS 2011), but the assumption is largely irrelevant due to the low rates of high-severity fire in both study regions in relation to forest regrowth, as described next.

RESULTS

We found a highly significant relationship between live-tree basal area and stand age in both regions (Figs. **2a-b**, Klamath n = 442, dry Cascades n = 304). Much of the variance in the plot data was caused by a modest number of relatively old stands that had much lower basal area for their

Table 2. Forest Inventory and Analysis (FIA) plot parameters for the Klamath and dry Cascades provinces, California, Oregon, and Washington, based on most recent survey data from 2001-2009. Also shown are the amounts of time after fire that is takes forest to regrow to the specified live basal area (BA) thresholds using the regression equations shown in Figs. (2a-b).
[a]These plots have 2 or more stand ages associated with them due to different disturbance histories within the main FIA plot.

Entity	Klamath	Dry Cascades
Number of plots (total)	581	445
Number of plots excluded from analysis†	139	141
Initial (P^{E}_{t+0}) early-successional forest (%)	9	14.5
Initial (P^{M}_{t+0}) mid-successional forest (%)	14.4	26.9
Initial (P^{L}_{t+0}) late-successional forest (%)	76.6	55.6
Regrowth period, 0-13.5 m^2/ha live BA (yrs)	44	53
Regrowth period, 13.5-27.5 m^2/ha live BA (yrs)	32	36
Regrowth period, 0-27.5 m^2/ha live BA (yrs)	76	89
High-severity fire rotation	362	913

†These plots have 2 or more stand ages associated with them due to different-aged sub-plots within the main FIA plot.

age than did other plots. The amount of time following disturbance needed for regenerating forests to reach live-tree basal area >27.5 m^2/ha was 77 and 90 years, respectively, for the Klamath and dry Cascades (Table **2**).

Using the MTBS data, the rotation for high-severity fire from 1996-2011 was 362 to 913 years in the Klamath and dry Cascades, respectively (Table **2**). At these rates, a total of 1,221 and 325 km^2 of high-severity fire would occur in Klamath and dry Cascades late-successional forests, respectively, in 40 years. With annual regrowth rates of late-successional forests that were 4.5 to >10 times greater than the rates of fire disturbances (i.e. (1/77)/(1/362) for the Klamath and (1/89)/(1/913) for the dry Cascades, and no disturbances other than fire, late-successional forests would eventually come to occupy 83% of the potential forested area in the Klamath and 91% in the Cascades. Thus, over 40 years, late-successional forests in the Klamath increased slightly over their current amount of 77% of the forested landscape FIA plots to 81% or from about 10,668 km^2 to 11,335 km^2 (Fig. **3a**). In the dry Cascades, where late-successional forests were 59% of the forested landscape FIA plots, they increased relatively rapidly to 77% of the forested landscape, or from 6,253 km^2 to 8,234 km^2 in 40 years (Fig. **4a**).

Simulated thinning of 21% of dense, late-successional forest of the Klamath landscape meant that a total of 2,225 km^2 would be reduced, while treatments in mid-successional forests would cover 840 km^2 to reach a treatment level of 22% of the whole landscape. After the one-time thinning, late-successional forests returned to slightly lower amounts than occurred without thinning after 40 years (Fig. **3a**). The net effect of the one-time thinning was to reduce late-successional habitat by 10.7% over the 40-year period, or from an average of 11,086 km^2 to 9,996 km^2 over 40 years (i.e., 1,090 km^2 less each year on average, Fig **3b**). The amount of dense, late-successional forest that was prevented from burning at high severity was 16 km^2/year, resulting in 320 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, in this scenario, thinning reduced 3.4 times more late-successional forest than it increased. The maintained treatment reduced habitat by 15.3%, from 11,086 km^2 on average over 40 years to 9,396 km^2 (i.e., 1,690 km^2 less each year on average, Fig. **3c**). In both cases, 13% of the habitat loss was from thinning in mid-successional forest that prevented or slowed these forests from developing into dense, late-successional forest. The amount of dense, late-successional forest that was prevented from burning at high severity was 20 km^2/year, resulting in 400 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, the combination of thinning and maintenance reduced 4.2 times more late-successional forest than it increased.

In the Cascades, to treat 22% of the landscape, the thinning scenario targeted 1,313 km^2 of dense, late-successional forest, and 1,036 km^2 of mid-successional forest. After the one-time thinning, late-successional forests again returned to slightly lower amounts than occurred without thinning after 40 years (Fig. **4a**). The net effect of the one-time thinning treatment over 40 years was to reduce dense, late-successional forest by an average level of 11.1% (836 km^2 less each year on average, Fig. **4b**). The amount of dense, late-successional forest that was prevented from burning at high severity from the one time treatment was 3.5 km^2/year, resulting in 140 km^2 of dense, late-successional forest, which would otherwise have been transformed into

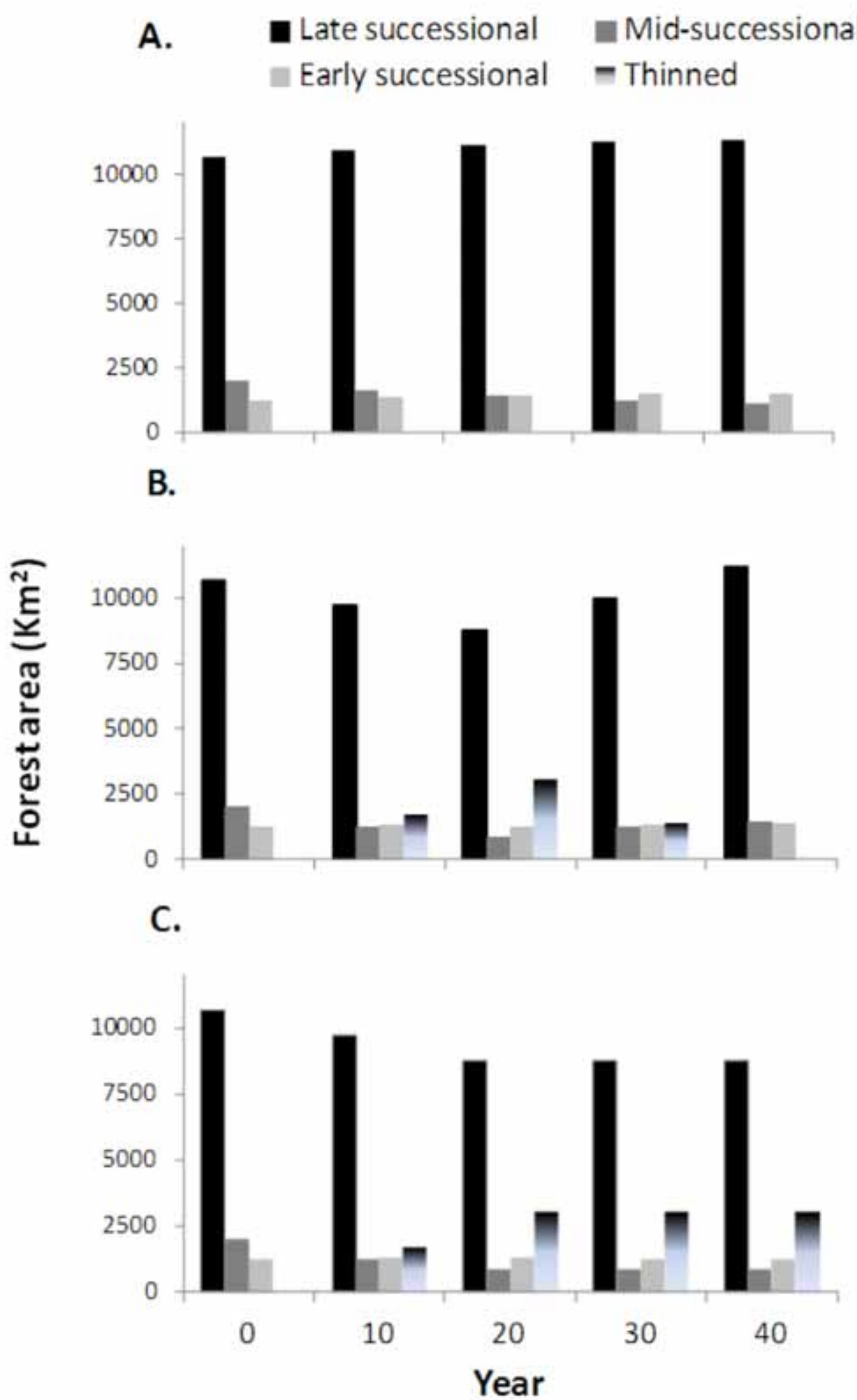

Fig. (3a-c). Amounts of the four forest types (early-, mid-, late-successional, and thinned) in the landscape over a 40-year period based on the states shown in (Fig. **1**) and transition rates (Table **2**) for the Klamath province, California, and Oregon, and the following scenarios: A) no treatment; B) one-time treatment of 21% of late-successional forests (>27.5 m^2/ha live-tree basal area) and 42% of mid-successional forests (= total of 22% of landscape treated) followed by recovery in 20 years to late-successional forest; C) treatment of 21% of late-successional forests (>27.5 m^2/ha live-tree basal area) and 42% of mid-successional (= total of 22% of landscape treated) forests with future maintenance. We converted proportions of forest types from modeling output to km^2 using the area estimate from FIA for the Klamath study region.

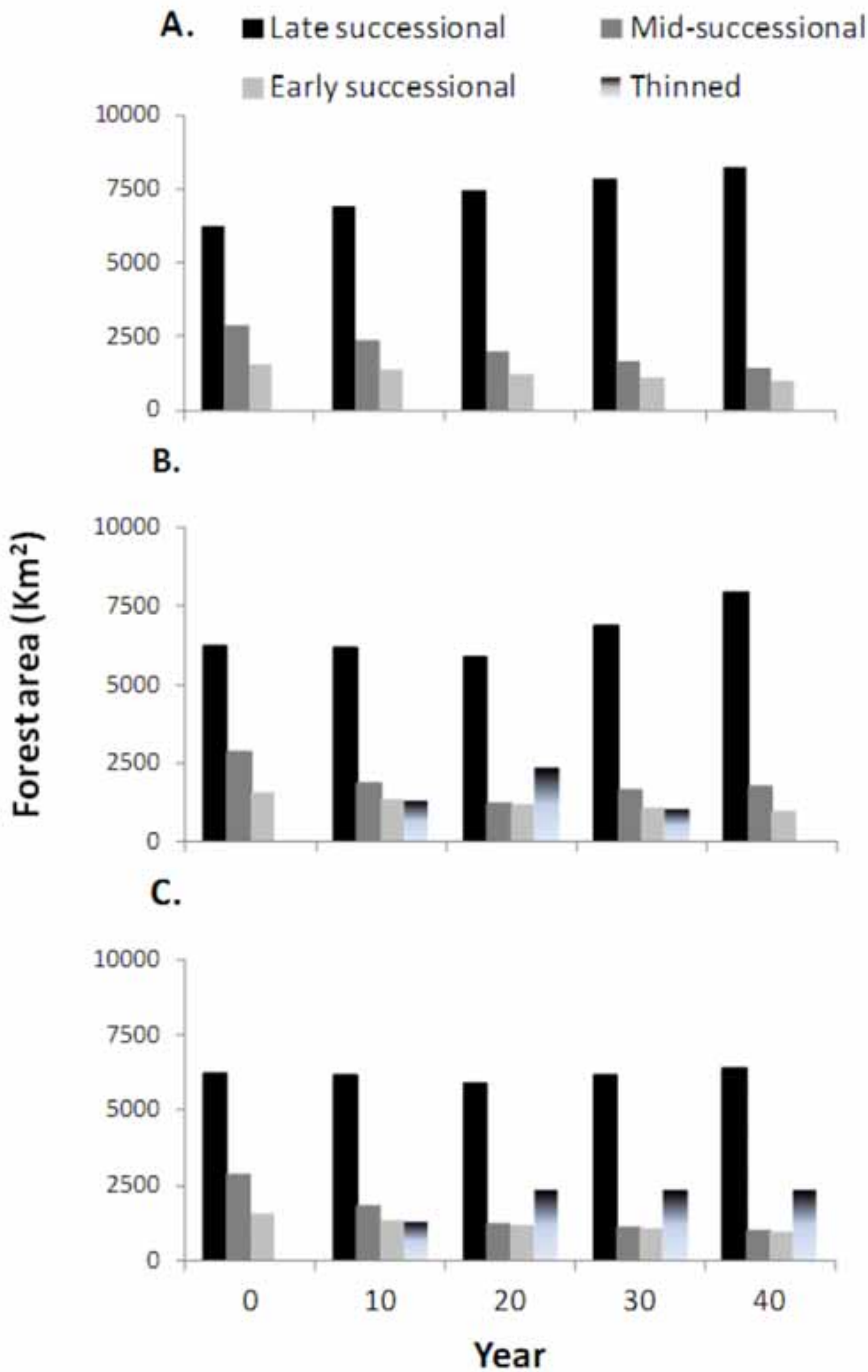

Fig. (4a-c). Amounts of the four forest types (early-, mid-, late-successional, and thinned) in the landscape over a 40-year period based on the states in (Fig. **1**) and transition rates (Table **2**) for the dry Cascades province, California, Oregon, and Washington and the following scenarios: **A**) no treatment; **B**) one time treatment of 21% of late-successional forests (>27.5 m^2/ha live tree basal area) and 36% of mid-successional forests (=22% of landscape treated) followed by recovery in 20 years to late-successional forest; **C**) treatment of 21% of late-successional forests (>27.5 m^2/ha live tree basal area) and 36% of mid-successional forests (=22% of landscape treated) in perpetuity. We converted proportions of forest types from modeling output to km^2 using the area estimate from FIA for the dry Cascades study region.

early-successional forest, in each year on average over the 40-year period. Therefore, thinning reduced 6.0 times more late-successional forest than it increased. The maintained treatment reduced dense, late-successional forest by an average of 16.4% (1,212 km^2less each year on average, Fig. **4c**). Of this reduction, 30% was from the indirect effect of thinning in mid-successional forests, more of which were treated in the Cascades scenario. The amount of dense, late-

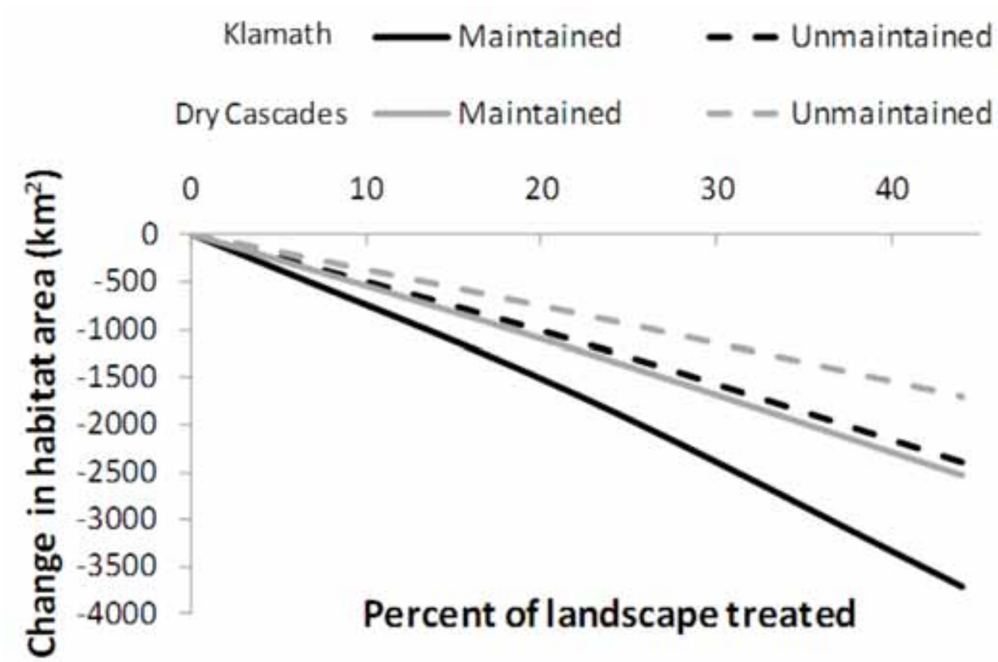

Fig. (5). Net amount of habitat lost over 40 years compared to the no-treatment scenario as a function of treatment of 0-45% of the landscape. The amount of late-successional forest treated was held constant at 21% of the area of this forest, except at very low levels of treatment. The amount of mid-successional forest treated varied from zero at very low treatment levels, to a large proportion of the mid-successional forests when 45% of the landscape was treated, particularly in the Klamath region.

successional forest that was prevented from burning at high severity from the maintained treatment scenario was 4.5 km^2/year, resulting in 180 km^2 of dense, late-successional forest, which would otherwise have been transformed into early-successional forest, in each year on average over the 40-year period. Therefore, the combination of thinning and maintenance reduced 6.7 times more late-successional forest than it increased.

As treatment level increased from 11 to 22%, habitat loss doubled (Fig. **5**). With 22% of the landscape treated, the effect of reducing fire by 50% in the rest of the landscape was reached, and there was no further reduction in fire with increasing treatment amount. With less fire prevented per km^2 treated, the rate of habitat loss increased as treatment went from 22 to 45% of the landscape.

We also assessed the effect of holding treatment level constant and varying the efficacy of treatments. Even if treatment efficacy was considerably greater than we assumed and rotations of high-severity fire substantially longer than twice their current length, the amount of dense, late-successional forest habitat that would be reduced due to thinning would only be slightly lower (Figs. **6a-b**). With complete elimination of fire over 40 years as a result of treatments, the amount of dense, late-successional forest would be 9-10% less than with no treatment. This becomes a large amount of habitat loss over time.

DISCUSSION

We found that the habitat recruitment rate exceeded the rate of severe fire by a factor of 4.5 in the Klamath and 10 in the dry Cascades, leading to a deterministic increase in dense forest habitat over time, assuming no other disturbance events. In contrast, previous published assessments of fire on spotted owls have not explicitly considered fire and forest regrowth rates (Wilson and Baker 1998, Lee and Irwin 2005, Roloff *et al.* 2005, 2012, Calkin *et al.* 2005, Hummel and Calkin 2005, Ager *et al.* 2007, Lehmkuhl *et al.* 2007). Not including the probability of high-severity fire, which is low, leads to highly inflated projections of the effects of thinning versus not thinning on high-severity fire (Rhodes and Baker 2008, Campbell *et al.* 2012).

Our calculations of thinning effects included rates of forest regrowth along with high-severity fire. The calculations illustrate how the requirement that the long-term benefits of thinning clearly outweigh adverse impacts (USFWS 2011) is not attainable as long as treatments have adverse impacts on spotted owl habitat. This is because the amount of dense, late-successional forest that might be prevented from burning severely would be a fraction of the area that would be thinned. Under our "best case" scenario, thinning reduced dense, late-successional forest by 3.4 and 6.0 times more than it prevented such forest from experiencing high-severity fire in the Klamath and dry Cascades, respectively, similar to findings in a recent unpublished report by U.S. Forest Service scientists from the Pacific Northwest Research Station (Raphael *et al.* 2013). This would not be a concern if thinning effects were neutral, but the commercial thinning prescriptions being implemented call for forests with basal area reduced by nearly half to 13.5-27.5 m^2/ha, which is mostly well below the minimum level known to function as nesting and roosting habitat (ca. 23 m^2/ha) (Buchanan *et al.* 1995, 1998). Thus, if dense forests are subjected to these treatments, much of the impacted area would no longer have minimum basal area needed to function as nesting and roosting habitat. Even an immediate doubling of fire rates due to climate change or

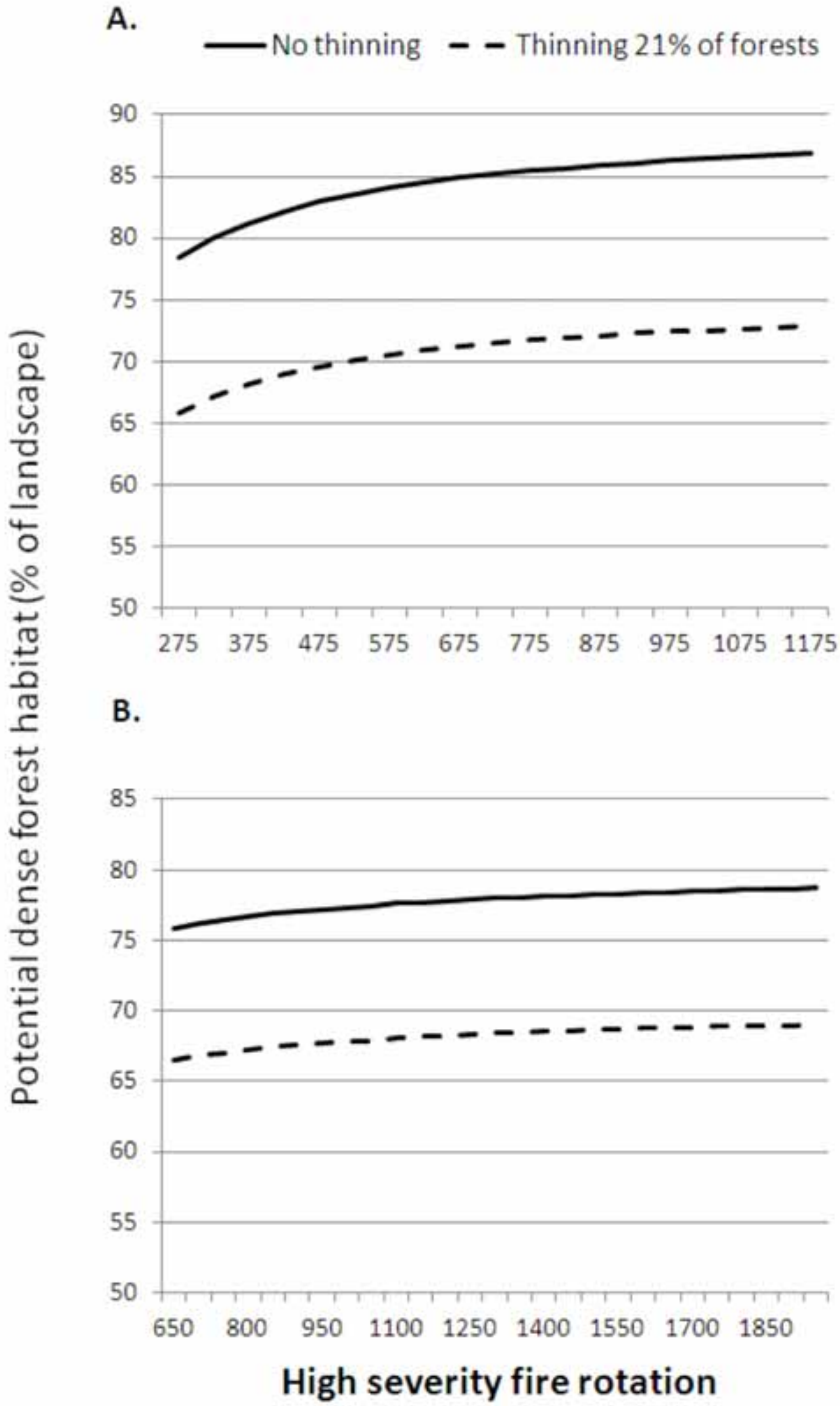

Fig. (6a-b). Amount of forest habitat in the range of the Northern Spotted Owl in the A. Klamath, and B. dry Cascades 40 years in the future as a function of the average high severity rotation over that time period, and longer rotations.

other factors would result in far less habitat affected by high-severity fire than thinning. In addition, much of the high-severity fire might occur regardless of thinning, especially if the efficacy of thinning in reducing high-severity fire is reduced as fire becomes more controlled by climate and weather (Cruz and Alexander 2010). Clearly, the strategy of trying to maintain more dense, late-successional forest habitat by reducing fire does not work if the method for reducing fire adversely affects far more of this forest habitat than would high-severity fire, and the high-severity fire might occur anyway because it is largely controlled by climate and weather.

There may be silvicultural treatments that can be done in spotted owl habitat that may reduce adverse impacts. For example, thinning that maintains at least 23-27.5 m^2 ha basal area. However, given that key habitat elements such as small trees, down wood, and likely some intermediate-sized trees are going to be targeted in any forest fuel reduction treatment, it appears unlikely that any conventional fuels reduction treatment in spotted owl habitat would not have at least some adverse impacts. This is supported by research on thinning that was often less intensive than commercial thinning prescriptions. This research showed negative impacts on spotted owls or their prey, as summarized in our introduction (Waters and Zabel 1995, Waters *et al.* 2000, Carey 2001, Ransome and Sullivan 2002, Gomez *et al.* 2003, Suzuki and Hayes 2003, Ransome *et al.* 2004, Bull *et al.* 2004, Lehmkuhl *et al.* 2006, Meyer *et al.* 2007, Wilson 2010, Holloway and Smith 2011, Manning *et al.* 2012), and how spotted owls have been displaced by even very limited amounts of thinning or contemporary harvest near the nest or activity center (Forsman *et al.* 1984, King 1993, Hicks *et al.* 1999, Meiman *et al.* 2003, Seamans and Gutiérrez 2007). Even if adverse impacts were quite modest, the amount of dense, late-successional forest that might be prevented from experiencing high-severity fire is so much smaller than the area that would be treated in an effort to accomplish this reduction in fire, that the net impact of the thinning would still be much greater. In addition, it is becoming increasingly less clear whether a reduction in high-severity fire below current rates would necessarily be beneficial to spotted owls. The dry forests in which spotted owls are found were historically characterized by mixed-severity fires (see Hessburg *et al.* (2007), Baker (2012), and Odion *et al.* (2014) for historic fire in the dry Cascades of Washington and Oregon, Beaty and Taylor (2001) and Bekker and Taylor (2001, 2010) for the California Cascades, and Wills and Stuart (1994), Taylor and Skinner (1998, 2003), and Odion *et al.* (2014) for the Klamath). Recent research suggests that this historic fire may have neutral and beneficial effects to spotted owls.

Studies on the effects of fire on spotted owls are few and often focused on other owl subspecies and some studies are confounded by post-fire logging effects (Clark *et al.* 2013). Nonetheless, it has long been known that fire in woody vegetation causes an increase in small rodent populations and consequently raptor populations (Lawrence 1966), and studies on spotted owls and fire where no logging occurred suggest that high-severity fire at current rates may confer benefits or be neutral. Bond *et al.* (2009) found that California Spotted Owls in the Sierra Nevada preferentially foraged in severely burned forests more than unburned forests within about 1.5 km of a core-use area. The percentage of high-severity fire in burned Mexican Spotted Owl (*Strix occidentalis* ssp. *lucida*) sites had no significant influence (Jenness *et al.* 2004). Roberts *et al.* (2011) found no support for an occupancy model for California Spotted Owls that distinguished between burned and unburned sites in unmanaged forests; the mean “owl survey area” that burned at high-severity was 12%, with one survey area experiencing up to 52% high-severity fire, which is almost three times the current amount of severe fire in owl habitat, according to the MTBS data. In a longer-term (1997-2007) study of California Spotted Owl site-occupancy dynamics throughout the Sierra Nevada, high-severity fire that burned on average 32% of forested vegetation around nests and core roosts had no significant effect on extinction or colonization probabilities, and overall occupancy probabilities were slightly higher in mixed-severity burned areas than in unburned forest (Lee *et al.* 2012), while other research found no significant difference in home range size between mixed-severity fire areas and unburned forest (Bond *et al.* 2013). Studies on reproduction in occupied sites of all three spotted owl subspecies indicated no difference between unburned sites and mixed-severity burned sites (excluding burn out areas created by fire suppression operations) (Jenness *et al.* 2004), or in some cases reproduction may have been greater in burned sites (Bond *et al.* 2002, Roberts 2008). The longer-term value of fire disturbances is in the creation of landscape heterogeneity with inclusions of young stands, improving habitat at the landscape scale. Fire also plays a vital role in creating snags, large down logs, and other key elements of the highest quality spotted owl habitat at the territory scale (Franklin *et al.* 2000). No assessments of fire and thinning effects on spotted owls, including this one, have accounted for any potential beneficial effects of mixed-severity fire, nor the potential negative effects of lack of mixed-severity fire in treated areas.

While much of the concern about fire and thinning in dry forests of the Pacific Northwest has focused on spotted owls, it may also apply to other biota associated with dense, old forests, including species of conservation concern, such as Pacific fisher (*Martes pennanti pacifica*), which research indicates may benefit from mixed-severity fire (Hanson 2013), the Northern Goshawk (*Accipiter gentilis*), and, following fire, the Black-backed Woodpecker (*Picoides arcticus*), which depends upon higher-severity fire in dense, older forest (Odion and Hanson 2013). Like the spotted owl, studies have documented that this woodpecker is also negatively affected by thinning (Hutto 2008). Also, like the spotted owl, the Back-backed Woodpecker, Pacific Fisher and Northern Goshawk occur in forests where the historic fire regime was not low-severity. Modeling for the fisher, similar to modeling for the spotted owl, has not used the actual rates of high-severity fire and forest regrowth to assess possible impacts of fire, and has assumed that fire represents a loss of fisher habitat (Scheller *et al.* 2011), contrary to more recent empirical findings (Hanson 2013). Not including the actual probability of fire leads to considerably inflated projections of the effects of thinning vs. not thinning in reducing high-severity fire (Rhodes and Baker 2008, Campbell *et al.* 2012). Our findings highlight the need to be cautious about conclusions that thinning treatments are needed for species found in dense forest and that they will not have unintended consequences (e.g., Stephens *et al.* 2012) until long-term, cumulative impacts are better understood. As we found with spotted owls, long-term and unintended consequences may be substantial for species that rely on dense, late-successional forests, especially when these species are sensitive to small amounts of thinning in their territory.

CONCLUSION

We used a quantitative approach that, unlike others, accounted for rates of high-severity fire and forest recruitment, allowing assessment of future amounts of spotted owl habitat at current rates of fire, with and without thinning. We found that the long-term benefits of commercial thinning would not clearly outweigh adverse impacts, even if much more fire occurs in the future. This conclusion applies even if adverse impacts of treatments are quite modest because of the vastly larger area that would need to be treated compared to area of high-severity fire that might be reduced by thinning. Moreover, our results indicate that, even if a longer time interval is analyzed (e.g., 100 years), the declines in dense, late-successional habitat due to thinning would not flatten, as long as thinning is reoccurring. Thus, where spotted owl management goals take precedence, the best strategy for maintaining habitat will be to avoid thinning treatments that have adverse impacts in spotted owl habitat or potential habitat (Gaines *et al.* 2010). There is ample area outside of existing or potential spotted owl habitat where managers wishing to suppress fire behavior or extent may focus their efforts without directly impacting spotted owls (Gaines *et al.* 2010), such as in areas adjacent to homes or in dense conifer plantations with high fuel hazards (Odion *et al.* 2004). In addition, there are management approaches that may be more effective than thinning in helping accomplish these fire prevention goals, such as controlling human-caused fire ignitions (Cary *et al.* 2009). Lastly, emerging research suggests that fire is not the threat it has been assumed to be for spotted owls, suggesting that, rather than management that focuses on suppressing fire behavior, other, no regrets active management may be more appropriate (Hanson *et al.* 2010). Research is needed to determine if these findings might apply to other species that are characteristic of dense forests, particularly given the widespread and growing emphasis on thinning as a management tool for suppressing wildland fires.

CONFLICT OF INTEREST

The authors confirm that this article content has no conflict of interest.

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SUPPORTIVE/SUPPLEMENTARY MATERIAL

Supplementary material is available on the publishers Web site along with the published article.

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