From: Dominick DellaSala <dominick@geosinstitute.org>
Sent: Friday, April 25, 2014 4:22 PM
To: Wildlife Management
Subject: submission of comments on NSO status review
Attachments: geosNSOstatelistingcomments.pdf

Attached are comments I would like to submit for the record regarding the state of California's status review of the Northern Spotted Owl. My comments provide extensive documentation of the status of the owl and need for listing under the California Endangered Species Act given the precarious status of the species rangewide and in California and the numerous listing factors documented herein.

Feel free to contact me should you have any questions on our submission.

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"Those who have the privilege to know, have the duty to act." Albert Einstein

May 1, 2014

California Department of Fish and Wildlife
Nongame Wildlife Program
Attn: Neil Clipperton
1812 9th Street
Sacramento, California 95811
Submitted via: *wildlifemgt@wildlife.ca.gov*

Re: Comments Regarding CDFW Status Review for the Northern Spotted Owl (*Strix occidentalis caurina*)

Dear Mr. Clipperton:

As a member of the U.S. Fish & Wildlife Service recovery team for the Northern Spotted Owl (*Strix occidentialis caurina*) from 2006-2008, I am considered an expert on habitat needs and population status of this imperiled species. Thus, pursuant to the state's status review and potential listing of this species under the California Endangered Species Act (CESA), please consider these comments in your review. Specifically, the Northern Spotted Owl (NSO) warrants listing under the CESA because, like the federal listing, it also meets several listing criteria, including: 1) past, present, and threatened habitat destruction, modification or curtailment; 2) competition from invasive species; 3) inadequate regulatory mechanisms; and 4) climate change threats.

In study areas not managed under the Northwest Forest Plan, such as nonfederal lands in California, owl declines are about twice as great (Anthony et al. 2006) due primarily to higher rates of logging and inadequate regulatory mechanisms. Moerover, a recently published large-scale demographic study (Forsman et al. 2011) found that the species has been declining on seven of eleven active demographic study areas, including California, at about 3% annually range-wide from 1985-2008. Funk et al. (2010) provides evidence for recent genetic bottlenecks in NSO that increase the species' vulnerability to range-wide extinction.

Areas that have little federal land support few or no owls, and Forsman et al. (2011) state that as a result too few NSO exist in four regions (southwestern Washington, the Coast Range of northwest Oregon, the California Cascades, and much of Washington's Olympic Peninsula) to conduct a demographic study with their methods. Further, the literature suggests these declines are not likely to lessen even with the recent federal owl recovery plan in place due to the un-quantified and unmitigated risks from active management and post-fire logging and high rates of logging on nonfederal lands. Thus, review of recent demographic rates, competitive interactions with Barred Owls (*Strix varia*), inadequate state regulations, climate change threats, and other recent threats discussed herein, provide sufficient justification for a determination by the state of California that the species warrants state-listing and the protections afforded it under the CESA.

PRESENT OR THREATENED DESTRUCTION, MODIFICATION, OR CURTAILMENT OF THE OWL'S HABITAT OR RANGE

The NSO is threatened by historic and ongoing loss and adverse modification of habitat due especially to logging. Over a century of logging has removed much of the owls' habitat. In 1990, habitat loss was estimated at 60-88% since the early part of the 19th century (USFWS 1990a, b, also see Strittholt et al. 2006 for similar estimates). Since the owl was federally listed in 1990, habitat loss has continued range-wide, most notably on nonfederal lands (Stauss et al. 2002, Courtney et al. 2004, Anthony et al. 2006, USFWS 2008, 2009, 2010a) and is likely to continue from post-disturbance logging, thinning, and logging of old forests on nonfederal and federal (to a lesser degree) lands. Additionally, it appears that the effects of past logging still are occurring on both federal and nonfederal lands as increased fragmentation and habitat loss propagate through the range of the owl (see FEMAT 1993, Courtney et al. 2004 for further discussion of lag effects) combining synergistically with Barred Owl extirpations of NSO territories (Dugger et al. 2011, Wiens 2012).

Important components of functional old-forest habitat for NSO and their prey such as standing dead trees, large down wood, multi-layered canopies, and other features have been lost throughout much of the owls' range mainly due to logging. In many places, it will take centuries for forests to recover their former productivity even with the Northwest Forest Plan, recovery plan, critical habitat determination, and other measures on federal lands. In particular, the Northwest Forest Plan assumed a period of decades would be necessary before habitat in many of the late-successional reserves (LSRs) became suitable for owls; only about 36% of the reserves currently are functioning as old-growth forests >150 years with about 59% in late-seral condition (Strittholt et al. 2006). Thus, it cannot be assumed that the LSR network and critical habitat is sufficient to recover the owl, particularly under increased threats on nonfederal lands (see below). Additionally, other human actions, including post-disturbance logging and extensive fuel treatments, and urban development have contributed to past and continue to contribute to present cumulative losses and degradation of NSO habitat and their prey.

CURRENT AND HISTORIC DISTRIBUTION

Historically, NSO was found from British Columbia south through western Washington, western Oregon, and northwestern California from Siskiyou County south to Marin County (American

Ornithological Union 1957, Forsman et al. 1984, Gutiérrez et al. 1995). The ranges of the NSO and California Spotted Owl meet at the southern end of the Cascade Range in northern California (Thomas et al. 1990, USFWS 1992, Barrowclough et al. 1999, Haig et al. 2001).

The owls' range includes three states and generally is divided as: Washington (four physiographic provinces), Oregon (five provinces), and California (three provinces) (Thomas et al. 1993). Long-term monitoring sites have been established in all three states, with 3 in Washington, 5 in Oregon, and 3 in California. In California, populations are declining in two of three long-term monitoring sites (see Table 3 below). It is clear that NSO status and distribution have declined since the subspecies originally was listed in 1994, and that the NSO is at risk of extinction throughout a significant portion, it not all, of its range.

HABITAT REQUIREMENTS

Large areas of older, structurally complex forests provide the habitat necessary to support viable populations of NSO. Extensive studies have supported the strong association of NSO and older forests (Table 1), particularly to adult survival. NSO select older forests for nesting (Hershey et al. 1998, Swindle et al. 1999), roosting, and foraging (Forsman et al. 1984, Bart and Forsman 1992, Thomas et al. 1990, Herter et al. 2002, Glenn et al. 2004, Forsman et al. 2005). Nest occupancy is related to the presence of mature and old-growth forests although the nature of this relationship varies regionally (Carroll and Johnson 2008). On private lands in northwestern California, NSOs usually occur in the oldest forests available (Diller and Thome 1999).

Table 1. Studies documenting the association between NSO and older forest habitats.

Variable	Effect	Association	Reference
"demographic parameters on some study areas"	+	% cover suitable habitat	Forsman et al. 2011
recruitment	+	% habitat	Forsman et al. 2011
recruitment	+	federal lands (contained highest proportion of habitat)	Forsman et al. 2011
nesting	+	older forests	Hershey et al. 1998, Swindle et al. 1999
roosting and foraging	+	older forests	Forsman et al. 1984, Bart and Forsman 1992, Thomas et al. 1990, Herter et al. 2002, Glenn et al. 2004, Forsman et al. 2005
occurrence	+	oldest forests available on managed forests on private lands in northwestern California	Diller and Thome (1999) Thome (1997)

nesting, roosting, and foraging	+	strong associations with older forests	LaHaye and Gutiérrez 1999
understory structure important for spotted owls and their prey	+	older forests	Carey et al. 1992, Rosenberg and Anthony 1992 Buchanan et al. 1995, LaHaye and Gutiérrez 1999, Lehmkuhl et al. 2006
apparent survival	+	amount of old forest habitat surrounding nesting territories	Franklin et al. 2000, Dugger et al. 2005, Olson et al. 2004
fecundity	+	amount of old forest habitat surrounding nesting territories- northern California	Franklin et al. 2000, southern Oregon - Dugger et al. 2005, Olson et al. 2004
reproductive rate	+	proportion of old-growth forest within a 730-m-radius circle around annual activity centers - in southern Oregon	Dugger et al. 2005
colonization rate	+	territories with more mature conifer forest - California Spotted Owls, Sierra Nevada of California	Seamans and Gutiérrez (2007)
extinction rate	-	territories with more mature conifer forest - California Spotted Owls, Sierra Nevada of California	Seamans and Gutiérrez (2007)
occupancy	-	additive and negative effect of barred owls and decreased amounts of habitat – nesting territory scale	Dugger et al. 2011, Carroll and Johnson 2008
colonization	-	additive and negative effect of barred owls and decreased amounts of habitat – nesting territory scale	Dugger et al. 2011
extinction	+	additive and negative effect of barred owls and decreased	Dugger et al. 2011

	amounts of habitat – nesting territory scale	

Recruitment is positively related to the proportion of older forest habitat in owl territories and higher levels of recruitment have been witnessed on federal lands with high proportions of old forest habitat (Forsman et al. 2011). Other studies have documented lower reproduction in areas with less suitable habitat. For example pairs produced fewer fledglings in areas with < 20% habitat (average = 0.33 fledglings/pair) than in areas with > 60% habitat (average = 0.93 fledglings/pair) (Bart and Forsman 1992). Understory structure is important for owl prey (Carey et al. 1992, Rosenberg and Anthony 1992, Buchanan et al. 1995, LaHaye and Gutiérrez 1999, Lehmkuhl et al. 2006). Survival and fecundity are positively associated with the proportion of old forest surrounding nesting territories (Franklin et al. 2000, Dugger et al. 2005, Olson et al. 2004). In southern Oregon reproduction increased as the proportion of old forest within 730 m of activity centers increased (Dugger et al. 2005). Habitat may partially mitigate the effects of the invasive Barred Owl as NSO had lower extirpation rates in territories with high levels of suitable habitat (Dugger et al. 2011, Wiens 2012).

PRESENT OR THREATENED DESTRUCTION, CURTAILMENT, OR ADVERSE MODIFICATION OF HABITAT OR RANGE

Impacts of historic habitat destruction were particularly severe at lower elevations, in the Coast Range of Oregon and California and in southwest Washington, where substantial owl habitat was high-graded by logging the biggest trees first (USFWS 1990b). The few federal lands present in these regions are the backbone for owl recovery because of heavy logging in surrounding non-federal lands provided they are managed with protection of owl nesting, roosting, and foraging habitat in mind.

According to conservative estimates provided by the U.S. Fish & Wildlife Service NSO habitat losses continue across ownerships, but are of particular concern on nonfederal lands (Table 2).

Table 2. NSO habitat losses across ownerships, 1994 to 2004.

Area (acres)	Time	Ownership	Cause	Description	Citation
16,900	1994 to 2003	Federal	Clearcutting	older forest	Moeur et al. 2005
141,300	1994 to 2004	Federal and non-Federal	Stand replacing fire[1]	owl habitat	Raphael 2006
155,999	1994 to	Federal	Management	owl habitat	Courtney et

[1] We note that the evidence for fire impacts to owls is currently being debated in the scientific literature (see Hanson et al. 2009 for a summary of the issues)

	2003		including partial harvest		al. 2004
583,500	1994 to 2004	Non-Federal	Clear cut	owl habitat	Courtney et al. 2004

In excess of 96% of California's old-growth redwood forests are gone (Noss 2000). Important components of functional old-forest habitat for owls and their prey such as standing dead trees, large down wood, multi-layered canopies, and other features have been lost throughout much of the owls' range and are in short supply particularly on nonfederal lands mainly because of lax forest practices. In many places, it will take centuries for forests to recover their former productivity even with the Northwest Forest Plan, and other measures in place due to the extensive ecological debt in late-seral habitat (see Strittholt et al. 2006).

In addition to the above losses, competitive pressure from Barred Owls appears to be limiting NSO use of the LSRs (see Pearson and Livezey 2003, 2007). Thus, many of the LSRs may lose their functionality as a result of exclusion by Barred Owls requiring stepped up habitat conservation. These losses combined with ongoing post-disturbance logging, forest thinning for fuels reduction that may be more harmful to owls than forest fires (see Hanson et al. 2009, 2010, Odion et al. in press – Appendix A), and logging on nonfederal lands all demonstrate increasing risk factors. In sum, there is ample evidence for state listing of the owl as the combination of range contraction, population declines (throughout most of the range – Anthony et al. 2006, Forsman et al. 2011), ongoing habitat losses (range-wide), increasing threats from multiple interacting factors, and inadequate regulations, particularly on nonfederal lands, likely will result in the owls' eventual extinction absent stepped up habitat protections and improved regulations.

DISEASE OR PREDATION

The NSO is subject to disease and predation pressures that have increased substantially since listing. West Nile Virus has killed wild birds since its introduction in 1999 and subsequent spread across North America (McLean et al. 2001, Caffrey 2003, Marra et al. 2004, Blakesley et al. 2004), and owls are known to be susceptible (Fitzgerald et al. 2003, Gancz et al. 2004). In addition, recent examination of the rates of infection by blood parasites indicates that the NSO has a high rate of infection by blood parasites (Ishak et al. 2008). Changes in habitat that result in more open areas (e.g., from forest thinning) and increased fragmentation of older forests likely cause an increase in predation by Great Horned Owls (*Bubo virginianus*), Northern Goshawks (*Accipiter gentilis*), and Red-tailed Hawks (*Buteo jamaicensis*) that either increase mortality on adult spotted owls or dispersing juveniles. In addition, Leskiw and Gutiérrez (1998) present evidence of predation on NSO by Barred Owls, a risk that is growing with increasing overlap in distribution of these co-generic owls.

INADEQUACY OF REGULATORY MECHANISMS

The status of NSO and its old-forest habitat is subject to adverse modification due to the inadequacy of existing regulations. Existing regulations have failed to protect habitat on nonfederal lands. This failure is evidenced by the continued loss and degradation of habitat range-wide, particularly on nonfederal lands (e.g., Stauss et al. 2002, Courtney et al. 2004, Anthony et al. 2006, USFWS 2008, 2010a), the failure of habitat degraded by past management practices to be fully restored (e.g., Courtney et al. 2004), and by a demonstrated failure to reverse the decline of the NSO over the last two decades (e.g., Forsman et al. 2011). Inadequacies generally fall into the following categories: variable level of protection given to owls and habitat depending on the presence or absence of special designation (e.g., activity center, nest site); lack of landscape-scale planning on nonfederal lands; use of survey protocols and other standards that fail to incorporate current relevant science; prevalence of discretionary guidelines and/or unclear or unsuitable direction; failure to consistently require involvement of personnel with biological expertise in evaluating/assessing ecological information (discussed below).

One review by USFWS examined 75 verified NSO territories on private timberlands in two counties in California; 77% had declined to either "no response" or a "territorial single owl." Of the sites on Forest Service-administered lands, only 20% of the pair sites changed status during the same time period (USFWS 2010). Such a strong difference between relative success on federal and private lands "supports the contention that management on private timberlands is creating habitat conditions that do not support sustained occupancy by northern spotted owl" (USFWS 2010).

In California, since the 1992 adoption of the Forest Practice Rule provisions related to the NSO, further research has been conducted that has caused concern over the adequacy and continued relevance of the Rules. USFWS has expressly indicated that the use of measures contained in 14 CCR § 919.9(g) may not always ensure NSO take avoidance. According to several emails, USFWS staff believes that the application of the Rule "*typically does not* avoid or reduce the likelihood of take of northern spotted owl" because the habitat definitions and retention standards in the Rules "represent minimum values that are *below* the habitat parameters associated with reasonable levels of territory occupancy survival, and reproduction by northern spotted owl" (see: Jan 24, 2008 email from USFWS's Brian Woodbridge to CAL FIRE's Chris Browder; April 3, 2009 Email from USFWS' Ken Hoffman to CAL FIRE's Chris Browder; and April 22, 2009 Email from USFWS' Brian Woodbridge to CAL FIRE's Chris Browder).

The USFWS has stated that the use of California Wildlife Habitat Relationships [WHR] habitat definitions in the Rules is "unlikely to avoid take" (according to the emails identified above). This is because the WHR types are considered to be NSO habitat (i.e., 4M & 4D) are widely variable, and, at the lower end of size class/density, typically are poor habitat or non-habitat.

Harvest within 4D and 4M stands typically further reduces habitat quality significantly, sometimes to the point where take is likely, even when the post-harvest structure still meets 4M or 4D criteria." In fact, the standards for habitat typing and retention developed in 1992 are known to be inadequate to prevent owl take. CAL FIRE has accepted USFWS' arguments openly, and as a result has requested that timber harvest plan submitters provide substantial evidence in the plan record that NSO take has been avoided, and recommends that the plan proponent use habitat descriptions contained in the USFWS Habitat Descriptions 2 when addressing NSO take avoidance using guidelines and §919.9(e) and (g) (PRC §21081(a), 14 CCR §§15065(a)(1), 15091(a)(1) and (b); CAL FIRE's Use of 14 CCR §919.9(g) [939.9(g)] in Making Northern Spotted Owl Take Avoidance Determinations; CAL FIRE (2008)).

One blatant failure of the Rules is its inability to incorporate the relevant science that has been developed since the original provisions related to NSO were adopted in 1992. The USFWS provided guidelines for habitat typing and protection in 2008 that provide substantially more protection for high-quality owl habitat than do California's Rules. While CAL FIRE may prefer that logging proponents follow the USFWS Guidelines, the agency clearly lacks authority to require protections in excess of those provided in its Rules. In addition, many studies have been published on the topic since February 2008 and should be considered and incorporated into updated take evaluation guidelines. Unfortunately, in their essential provisions with respect to owl habitat typing, California's Rules have not changed since 1992. Today, they fall far short of the best available science as embodied in the USFWS's 2008 Guidelines.

The Rules do not consider the concept of habitat fitness potential (HFP), wherein evaluation of habitat parameters influencing survival and reproduction rates provides a more rigorous measure of "significant impairment of essential behavioral patterns such as breeding, feeding, or sheltering" that is readily incorporated into review of timber harvest plans. The evaluation of predicted effects of habitat modification on northern spotted owl affected by a project would be more robust by the incorporation of HFP. Section 919.9(g)(3) ignores the well-documented fact that NSO territories require a combination of habitat types to provide habitat for breeding, feeding, and sheltering, be functional, and retain occupancy (Hoffman, April 3, 2009 email, citing Zabel et al. 2001, 2003). USFWS staff asserted in the April 3, 2009 email that, as written, §919.9(g)(3) allows harvest of virtually the entire core area down to unsuitable conditions. Section 919.9(g)(4) includes the same definitions that allow poor quality habitat. Along with the Rules' general lack of grounding in the best Barred Owls may be present, as current survey protocols readily yield false negative results due to changes in NSO behavior when Barred Owls are present. The new rules continue to incorporate the 1992 survey protocol, which does not reflect the best available science. This is particularly relevant given the new rules adoption of "unoccupied" status in §895.1(a)(4).

More recent data, including modeling efforts, have raised concerns about the efficacy and accuracy of the 1992 protocol. Of particular concern is how well the 1992 protocol works in areas experiencing the recent invasion of the Barred Owl, which has had a suppression effect on NSO response rates, and may be affecting occupancy dynamics of spotted owls in the landscape (Olson et al. 2005, Crozier et al. 2006). According to one report, "estimates of annual colonization rates and the summary of empirical data, indicated that three years of surveys were not sufficient to conclude that a site historically occupied by NSO, but then unoccupied (or at least a spotted owl is not detected), will never be occupied in the future" (Dugger et al. 2009). Dugger et al. (2009) further goes on to state that for historically occupied sites, it's probably not ever appropriate to consider a site incapable of being occupied if there have been no habitat changes. Conversely, allowance of habitat modifications likely will cause the site to become permanently 'extinct.' In closing, Dugger et al. (2009) state that the current protocol is a prescription for continued habitat loss and declines of spotted owl breeding populations.

Direction Is Unsuitable, Unclear, or Discretionary

In California, habitat definitions in §895.1 describe habitats that typically are considered unsuitable, or at best represent the bare minimum conditions. For example, discussions in the USFWS emails identified previously point out that while functional "nesting habitat" is defined essentially as 4M/D or greater, virtually all NSO research describes nesting habitat as consisting of stands of much larger trees, with nest sites associated with very dense clumps.

The definition of an "active nest site or pair activity center" in §919.9(g)(1) is vague, and exclusionary. It fails to include all the sites entitled to protection under the Endangered Species Act. In addition, definitions for "timber operations" (refers to all activities that are involved in a logging operation, up to and including the removal of trees) and "Nesting habitat" are inappropriately or ill defined. It is virtually impossible to say exactly what characteristics of the habitat within 500 feet of an activity center the owls are keyed in on when selecting the nest site. This further renders it impossible for a registered professional forester (RPF), CDFG, the Director of California Department of Forestry (CAL FIRE), or USFWS to determine what measures are appropriate to adopt to protect nesting habitat, other than to prohibit tree removal. Inappropriate standards may allow adverse modification, even within critical nesting core areas that are likely to result in take. "Habitat" definitions allow for practices that result in poor quality habitat; the rules allow harvest of virtually entire core areas down to unsuitable conditions.

Finally, CAL FIRE lacks both the biological expertise and the regulatory authority to adequately evaluate take avoidance. Given that the 2008 USFWS Guidelines were written to provide a functional mechanism for translating the best available scientific information into effective habitat protections when employed by non-experts in owl habitats and biology, it is far from clear that CAL FIRE can reliably determine when a departure from the Guidelines will not result in an owl take. It is clear that CAL FIRE may not require THP proponents to follow the

Guidelines, which do not have the force of a regulation absent promulgation by the Board of Forestry.

Involvement of Qualified Personnel Is Inconsistent

Ensuring the participation of qualified, independent, biological experts is critical to reducing the risk of inadvertent harm. Unfortunately, state regulatory mechanisms in California do not provide such assurances, and both state and federal wildlife agencies have ceased to review timber harvest plans for owl impacts. Rule changes adopted in 2009 and 2010 minimize impartial scientific input, vesting responsibility key functions in evaluating and conserving owl habitat in private parties likely to have a financial interest in minimizing protections for owls and their habitat.

The prior rules required a "state-employed" biologist to participate in take avoidance determinations and mitigation of habitat impacts. A "Spotted Owl Expert" [SOE] is given such responsibility in the new rules. CAL FIRE designates SOE's, who need demonstrate only a limited level of expertise, and who usually are persons employed by timber companies. The new rules also removed requirements for CDFG review previously found in §919.9(g)(1), (3) and (4). Section 919.9(g)(2) allows a non-biologist, the Registered Professional Forester [RPF], to determine what is sufficient in terms of functional characteristics to be provided post-harvest, without requiring approval from CDFG or the Director of CAL FIRE. The rule further does not take into consideration 1,000-foot circles that may be shared by adjoining landowners. The review process does not incorporate information among landowners, so it is possible to have two unqualified RPFs making independent determinations regarding what is sufficient to retain within a single roost zone. Furthermore, without a requirement that the 1,000-foot circle contain even a minimum amount of habitat described in the rules as roosting prior to proposing operations within the circle, this rule easily could result in the only actual roosting habitat contiguous with the nest tree being reduced to some RPFs idea of minimum functionality without benefit of review by an independent state-employed biologist, DFG, or even the Director of CAL FIRE. USFWS staff emails indicate that it is highly possible that the removal of habitat necessary to provide sheltering would occur.

In addition, during the past four years, the USFWS has ceased to offer informal consultation on California THPs, and the California Department of Fish and Game has ceased to review THPs, including field reviews, expect for anadromous fisheries impacts. Thus, neither the state nor federal wildlife agencies are engaging in expert evaluation of proposed logging plans in the way that has proven critical, if inadequate, in the past. There is no evidence to suggest that it is reasonable to expect such a half-dismantled system of self-administered guidelines to effectively prevent the continued loss of owl habitat and owl take.

Landscape-level Planning Is Lacking

There is a significant lack of comprehensive planning for NSO on nonfederal lands, especially at a landscape-scale. However, these owls are associated strongly with particular landscape features, such as lower slopes and stream courses. Further, they are sensitive to landscape-scale spatial relationships between nesting, foraging, and roosting habitats. A failure to understand current scientific findings regarding these relationships *and* incorporate these understandings into landscape-scale planning mechanisms is a significant failure of state regulatory schemes.

Notably, discretionary guidelines in California focus solely on individual NSO territories. They fail to incorporate issues such as connectivity and dispersal habitat, wintering habitat, or longer-term habitat disturbance patterns. The state's rules fail to address habitat quality and quantity at scales relevant to territorial occupancy and fitness. The rules do not require any consideration of the spatial distribution of retained habitat. As a result, the rules enable harvest operations to occur in preferred areas where effects to NSO are relatively greater. Finally, the timber harvest plan review process is conducted on an individual case-by-case basis. This approach preempts a systematic region- or ownership-wide assessment of habitat conditions and owl status, and therefore makes the owl and its habitat particularly vulnerable to a magnification of effects arising from multiple separate harvest plans. In fact, the USFWS (undated) has noted problems with the cumulative effects of repeated entries within many NSO home ranges that have reduced habitat quality to such a degree that it causes reduced occupancy rates and frequent site abandonment.

The USFWS has asserted that under California's rules, there is strong evidence that habitat modification within critical nesting core areas is likely to result in a take. US Fish and Wildlife Service emails attribute this partially to the fact that the Forest Practice Rules allow low habitat quality, but also recognize that the actual habitat features selected by a given pair of NSO are unknown (although likely associated with features such as dense clumps, deformed trees, shading, aspect, water, and others that in combination result in a suitable nest site). Timber harvest typically disrupts, modifies, and removes these elements. These same USFWS emails assert that studies of NSO territory occupancy and fitness relative to habitat quality and quantity strongly indicate that in the Interior zone, NSO rely on functional (= high quality) habitat ***at much larger scales*** than described in the rules [emphasis added]. The small patches of habitat within 500 to 1000' buffers (even if maintained well above the minimum "suitable habitat" definition) are much less than the 200 to 300-acre core areas associated with continued occupancy and reproduction by NSO. Further, NSO nesting core areas often consist of multiple nest sites within a cluster of stands, not just one. All of these factors can be dealt with only as part of a landscape-scale planning effort.

CURRENT RELEVANT SCIENCE IS IGNORED IN TIMBER HARVEST PLANS

The NSO is threatened by continued increase in Barred Owl populations. In addition, ongoing human-caused climate change will magnify the threats the NSO already faces as fecundity levels have been shown to be determined, in part, by weather extremes (Anthony et al. 2006, Forsman et al. 2011). These detrimental impacts may be interacting with habitat loss and fragmentation to accelerate the decline of NSO populations (Anthony et al. 2006, Dugger et al. 2011, Wiens 2012), particularly on nonfederal lands as noted. Barred Owls compete with NSOs and are considered a major threat. Collapse of NSO populations has followed the north to south invasion of the Barred Owl and areas that recently have been invaded by this owl, such as in northern California, are beginning to show signs of population declines. Additionally, climate change is an emerging threat. Projected climatic changes in the Pacific Northwest likely to negatively affect spotted owls include increases in spring precipitation (a condition associated with decreased NSO reproductive success), increases in weather extremes, and changes that will affect prey availability and abundance. Projected changes in precipitation and temperature also likely will increase stress on NSO habitat, magnify the detrimental impacts of past and ongoing habitat modifications, and may impair habitat recovery rates.

Table 3. Changes in NSO demographic parameters up to 24 years (USFWS 2010).

Study Area	**Fecundity**	**Apparent Survival**[1]	**Population Change**[2]
Cle Elum (WA)	Declining	Declining	Declining
Ranier (WA)	Increasing	Declining	Declining
Olympic (WA)	Stable	Declining	Declining
Coast Range (OR)	Increasing	Declining since 1998	Declining
HJ Andrews (OR)	Increasing	Declining since 1997	Declining
Tyee (OR)	Stable	Declining since 2000	Stationary
Klamath (OR)	Declining	Stable	Stationary
Southern Cascades (OR)	Declining	Declining since 2000	Stationary
NW California (CA)	Declining	Declining	Declining
Hoopa (CA)	Stable	Declining since 2004	Stationary
Green Diamond (CA)	Declining	Declining	Declining

[1]Apparent survival calculations are based on model average.
[2] Population trends are based on estimates of realized population change.

Forsman et al. (2011) clearly demonstrate that NSO is on a downward trajectory with an estimated 2.9% decline per year from 1985 to 2006. The authors concluded that fecundity, apparent survival, and/or populations were declining on most study areas, and that there was evidence that increasing numbers of Barred Owls and loss of habitat were at least partly the cause for these declines. Concerns about habitat loss are attributable to extensive historic destruction and degradation of habitat, ongoing habitat loss, increasing risks from extensive thinning in owl habitat (Odion et al. in press, Appendix A), threats posed by climate change, and the lack of significant provisions to protect owl habitat on nonfederal lands.

Areas that have little federal land support few or no owls and Forsman et al. (2011) state that too few NSOs exist in these regions (i.e., southwestern Washington, the Coast Range of northwest Oregon, the California Cascades, and much of Washington's Olympic Peninsula) even to conduct a demographic study with their methods. It is likely that these declines will continue on both federal and especially on non-federal lands unless significant changes are made.

CLIMATE CHANGE

The USGCRP (2009) reported that in the Pacific Northwest, annual average temperature rose about 1.5°F over the past century, with some areas experiencing increases up to 4°F. Further, the region's average temperature is projected to rise another 3° to 10°F later this century, with higher emissions scenarios resulting in warming in the upper end of this range. USGCRP (2009) also reports that many climate models project further increases and decreases in winter and in summer precipitation, respectively, for the Northwest. They conclude that impacts related to changes and snowpack, streamflows, sea level, forest composition and other factors are already underway, with more severe impacts expected in the coming decades in response to continued warming. Researchers from the Pacific Northwest also report that current projections for future climatic conditions include year-round warming, wetter winters, and hotter, drier summers (Mote and Salathé 2009, Salathé 2006).

These changes in climate are an important direct threat to conservation and recovery of NSO. Researchers have documented the association of weather and climate patterns and NSO demography (Wagner et al. 1996, Franklin et al. 2000, Olson et al. 2004, Glenn 2009). The demographic study found that associations between fecundity, apparent survival, or recruitment, and weather covariates varied among study areas (Forsman et al. 2010). While past weather may not explain much of the decline in NSO populations over recent decades, weather conditions caused by a climate change may add to the existing problems faced by NSO. Glenn et al. (2010) found that projected climate changes: "…have the potential to negatively affect annual survival, recruitment, and consequently population growth rates for northern spotted owls."

On four of six areas studied, λ (or population growth rate) was positively associated with growing season that likely affects prey populations and negatively associated with cold, wet winters and nesting seasons and the number of hot summer days (Glenn et al. 2010). Interestingly, annual survival was more closely related to regional climate conditions, while recruitment was often associated with local weather. There also are important indirect impacts associated with climate change of concern. The International Panel on Climate Change (2001,

2007) noted that synergisms between the effects of climate change and other stressors pose the greatest threat to the world's biodiversity. Not surprisingly, climate change presents a serious threat to the continued persistence of NSO, especially when coupled with the already occurring impacts resulting from past and ongoing habitat loss, disease, predation, the invasion of the Barred Owl, and the inadequacy of state and federal regulatory mechanisms.

CONCLUSIONS

Since the NSO was listed in 1990, owl populations have continued a downward spiral, including in northern California. Many populations have been extirpated, others are been reduced dramatically, and threats are escalating region-wide as well as in California. The scientific evidence is clear that the owl is in danger of extinction throughout all or a significant portion of its range. Further, the road to recovery will not be easy. The NSO is facing a number of serious threats, especially on nonfederal lands where rates of logging are much higher than federal lands and regulatory mechanisms inadequate to reverse population declines. Responding to multiple threats is complicated given that threats may act individually, synergistically, and cumulatively. Thus, the result is that the NSO is facing increasing risks throughout its range and warrants listing under the CESA and this information, particularly the extensive and new studies cited herein, should be included in the states' status review and determination for listing under CESA.

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Appendix A (in press, Open J of Ecology)

EFFECTS OF FIRE AND COMMERCIAL THINNING ON FUTURE HABITAT OF THE NORTHERN SPOTTED OWL

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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 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. 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.

KEY WORDS: Forest thinning; habitat loss; fire rotation, forest regrowth rate, future habitat, 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 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.* 2003, 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 to 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. 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*), with mixed forests of Douglas-fir and white fir. 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/postfirevegcondition) 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 state *s* at time *t* + 1, allowing calculation of future amounts of each forest type using the following equation:

$$\begin{bmatrix} \phi_t^{EE} & \phi_t^{ME} & \phi_t^{TE} & \phi_t^{LE} \\ \phi_t^{EM} & \phi_t^{MM} & \phi_t^{TM} & \phi_t^{LM} \\ \phi_t^{ET} & \phi_t^{MT} & \phi_t^{TT} & \phi_t^{LT} \\ \phi_t^{EL} & \phi_t^{ML} & \phi_t^{TL} & \phi_t^{LL} \end{bmatrix} \begin{bmatrix} P_t^E \\ P_t^M \\ P_t^T \\ P_t^L \end{bmatrix} = \begin{bmatrix} P_{t+1}^E \\ P_{t+1}^M \\ P_{t+1}^T \\ P_{t+1}^L \end{bmatrix} \begin{bmatrix} P_{t+1}^E \\ P_{t+1}^M \\ P_{t+1}^T \\ P_{t+1}^L \end{bmatrix} = \begin{bmatrix} \Psi_t^{EE} & \Psi_t^{ME} & \Psi_t^{TE} & \Psi_t^{LE} \\ \Psi_t^{EM} & \Psi_t^{MM} & \Psi_t^{TM} & \Psi_t^{LM} \\ \Psi_t^{ET} & \Psi_t^{MT} & \Psi_t^{TT} & \Psi_t^{LT} \\ \Psi_t^{EL} & \Psi_t^{ML} & \Psi_t^{TL} & \Psi_t^{LL} \end{bmatrix} \begin{bmatrix} P_t^E \\ P_t^M \\ P_t^T \\ P_t^L \end{bmatrix} \qquad \text{Eq. 1}$$

The initial proportions, $P_{t=0}^{E-L}$ 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 transitions from late to mid-successional forest and mid- to early-successional forest were 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). We used these transitions (Table 1) and Eq. 1 to project forward 40 years. We chose this time interval because it represents one cycle of thinning and forest recovery.

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 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. 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."

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 (Figures 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 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 percent of the whole landscape (which included some early-successional forest). 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 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^2 less each year on average, Figs. 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-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 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. 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.* 2003). 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 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) and Baker (2012) 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), and Taylor and Skinner (1998, 2003) 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 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, as well its role in creating snags, large down logs, shrub regeneration 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.

CONCLUSIONS

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.

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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. P is the proportion of the landscape in E = early successional forest, M = mid-successional forest, and L = late-successional forest. K = Klamath, C = Cascades.

Transition probabilities	No treat	Treat 20% maintain	Treat 20% recover
ϕ_t^{LE}	$1/FR^{hs}$	$1/FR^{hs}$	$1/FR^{hs}$
ϕ_t^{EM}	$1/G^{EM}$	$1/G^{EM}$	$1/G^{EM}$
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^{MT*}	0	K = 0.0315 C = 0.0273	K = $0.0315(P^{M}_{1,20})$ C = $0.0273(P^{M}_{1,20})$
$\phi_t^{TM\dagger}$	0	0	K = $0.0315(P^{M}_{1,20})$ C = $0.0273(P^{M}_{1,20})$
ϕ_t^{TE}	0	0	0
$\phi_t^{TL\dagger}$	0	0	K = $0.0114(\ P^{L}_{1,20})$ C = $0.0105(\ P^{L}_{1,20})$
ϕ_t^{LT*}	0	K = 0.0114 C = 0.0105	K = $0.0114(\ +P^{L}_{1,20})$ C = $0.0105(\ +P^{L}_{1,20})$

[a]Only in effect for the first 20 years.
[b]Does not take effect until after 20 years.

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 Figures 2a-b.

[a]These plots have 2 or more stand ages associated with them due to different disturbance histories within the main FIA plot.

[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 subplotswithin the main FIA plot.

FIGURE LEGENDS

Figure 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.

Figure 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.

Figure 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.

Figure 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.

Figure 5. Net amount of habitat lost over 40 years compared to the no-treatment scenario as a function of treatment of 0-44% 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 44% of the landscape was treated, particularly in the Klamath region.

Figure 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.

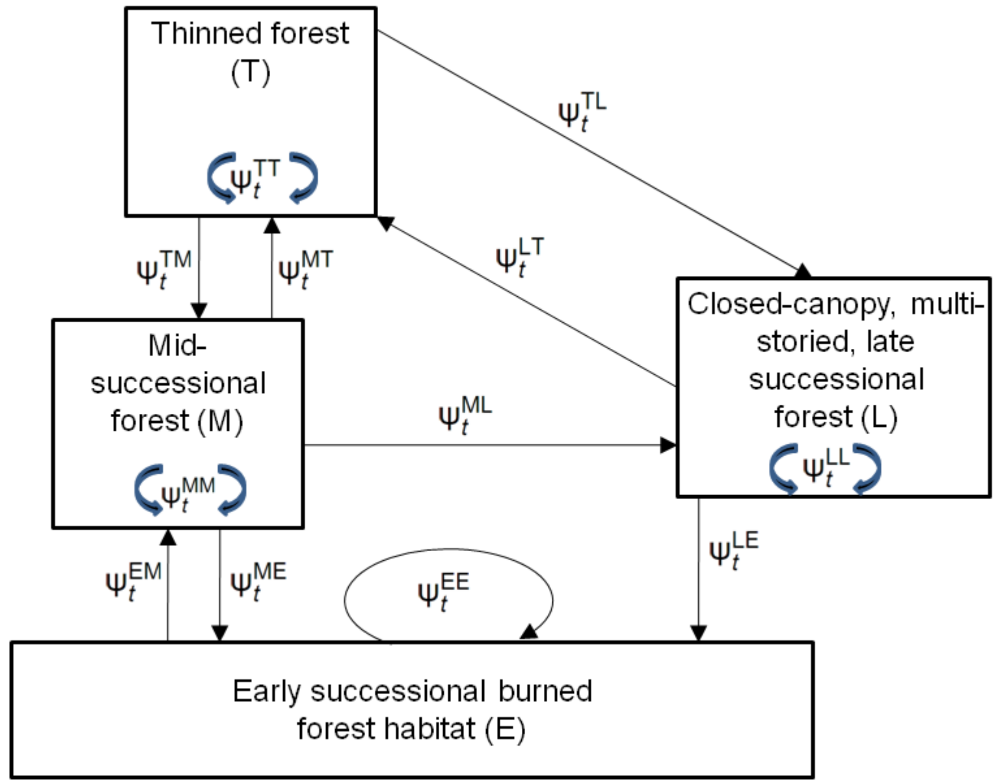

Thinned forest (T)
Ψ_t^TT
Ψ_t^TL
Ψ_t^LT
Ψ_t^TM
Ψ_t^MT
Closed-canopy, multi-storied, late successional forest (L)
Ψ_t^LL
Mid-successional forest (M)
Ψ_t^MM
Ψ_t^ML
Ψ_t^LE
Ψ_t^EM
Ψ_t^ME
Ψ_t^EE
Early successional burned forest habitat (E)

Figure 2

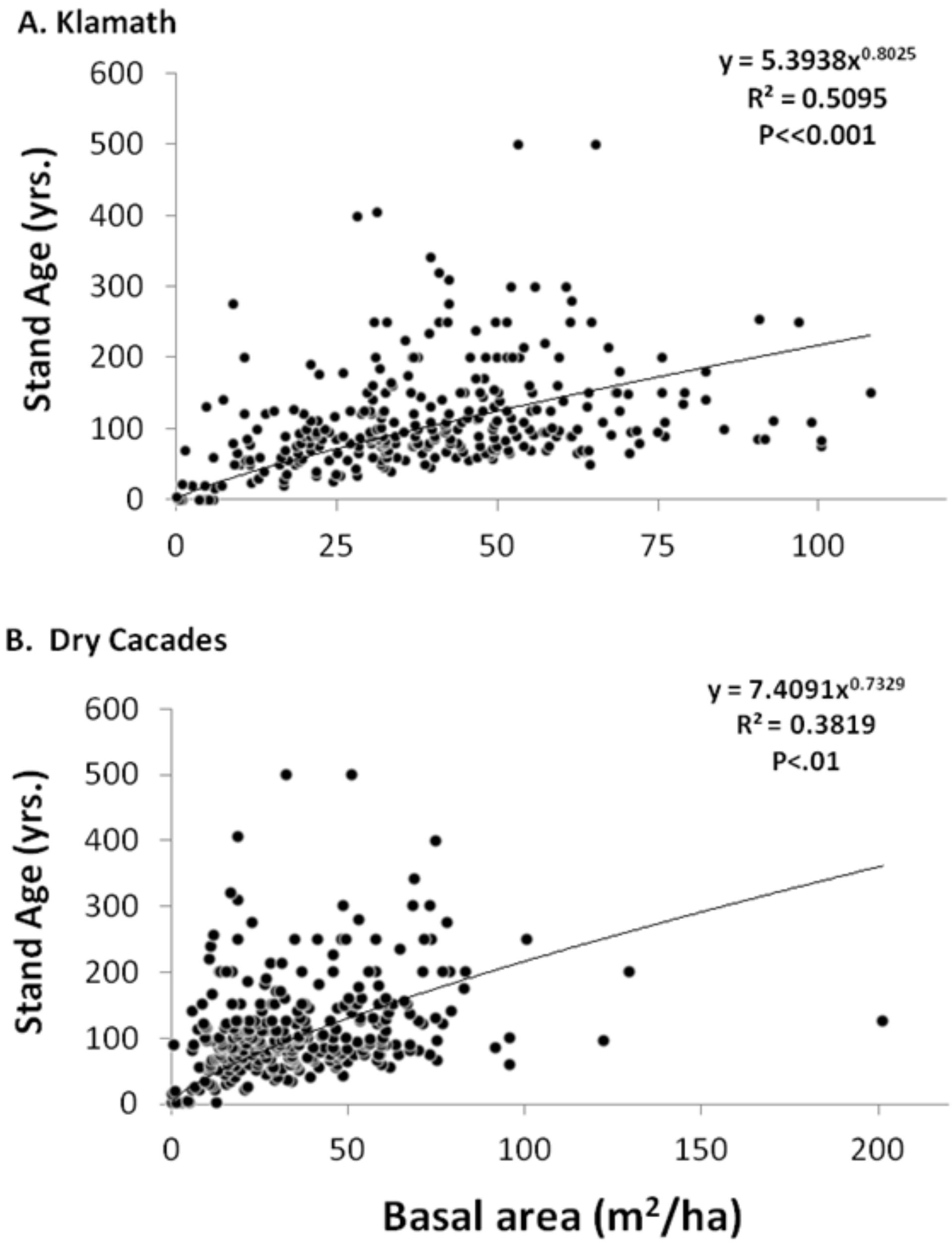

A. Klamath
y = 5.3938x^0.8025
R² = 0.5095
P<<0.001
Stand Age (yrs.)
600
500
400
300
200
100
0
0
25
50
75
100
B. Dry Cacades
y = 7.4091x^0.7329
R² = 0.3819
P<.01
Stand Age (yrs.)
600
500
400
300
200
100
0
0
50
100
150
200
Basal area (m²/ha)

Figure 3

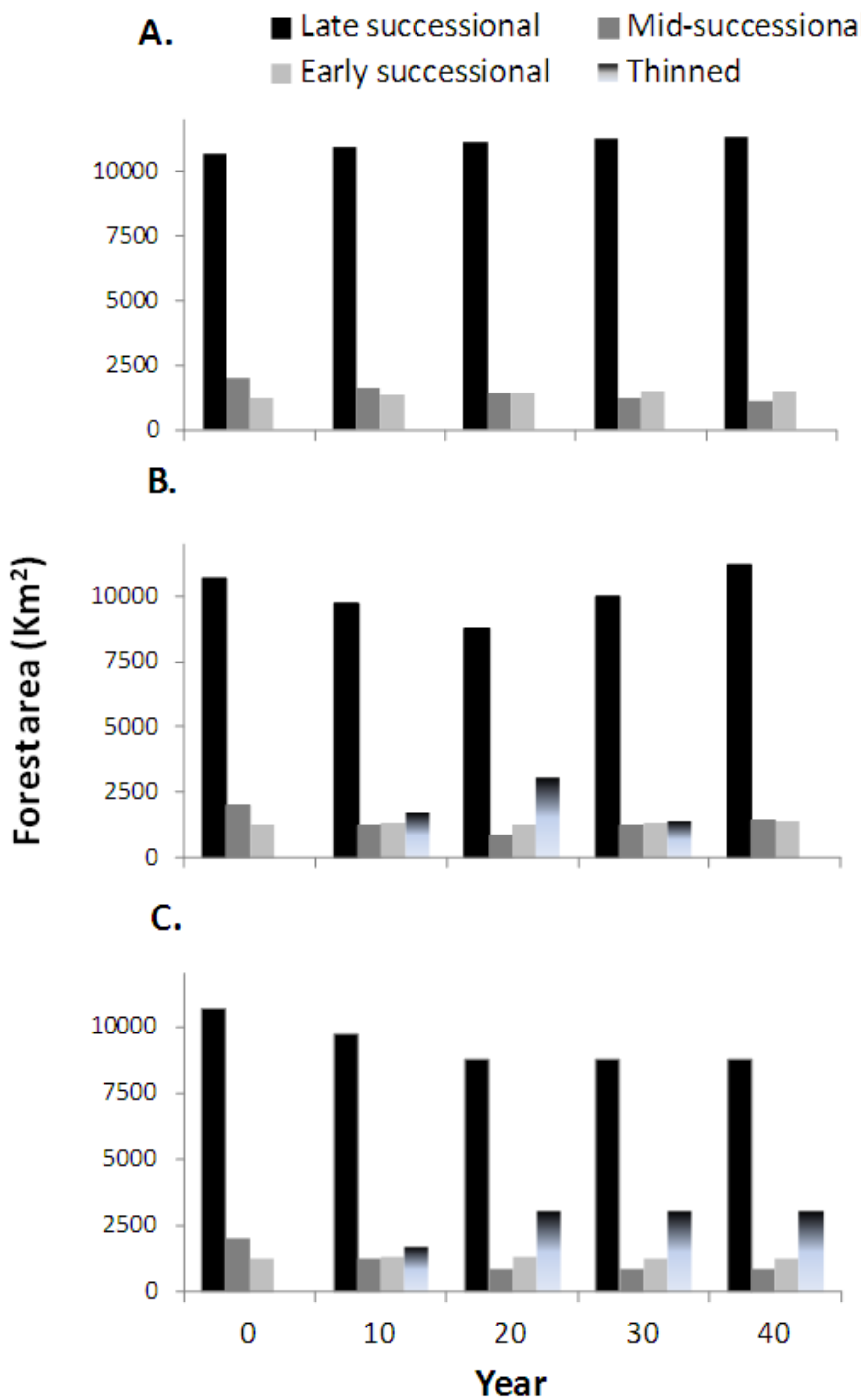
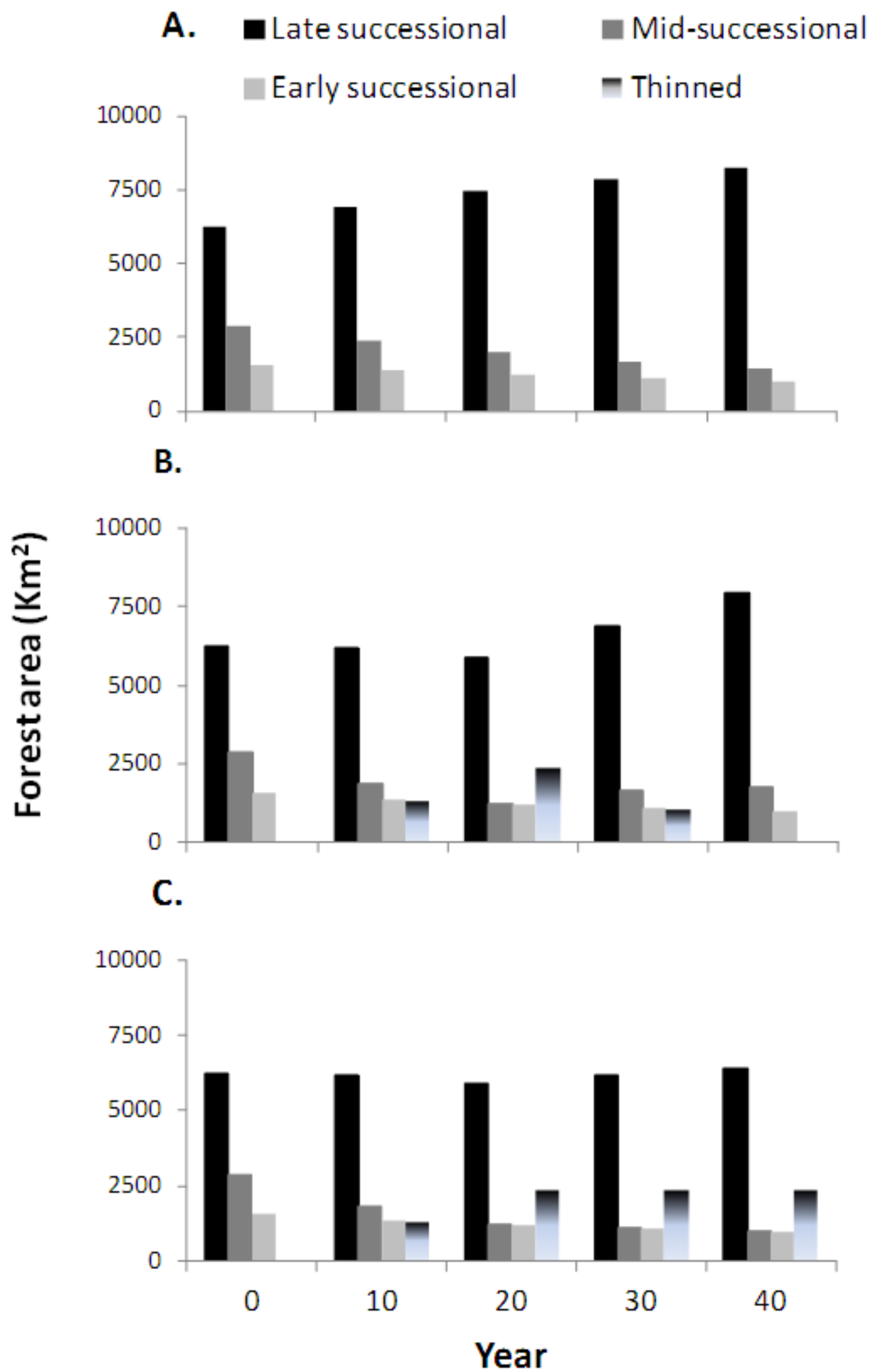
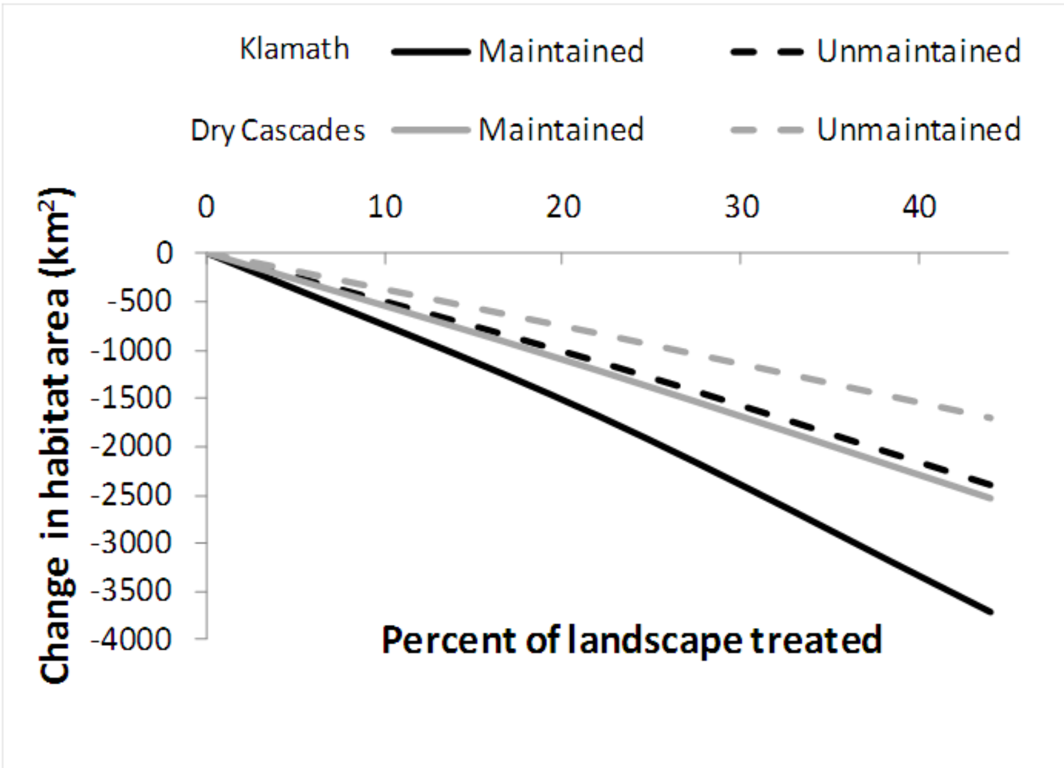
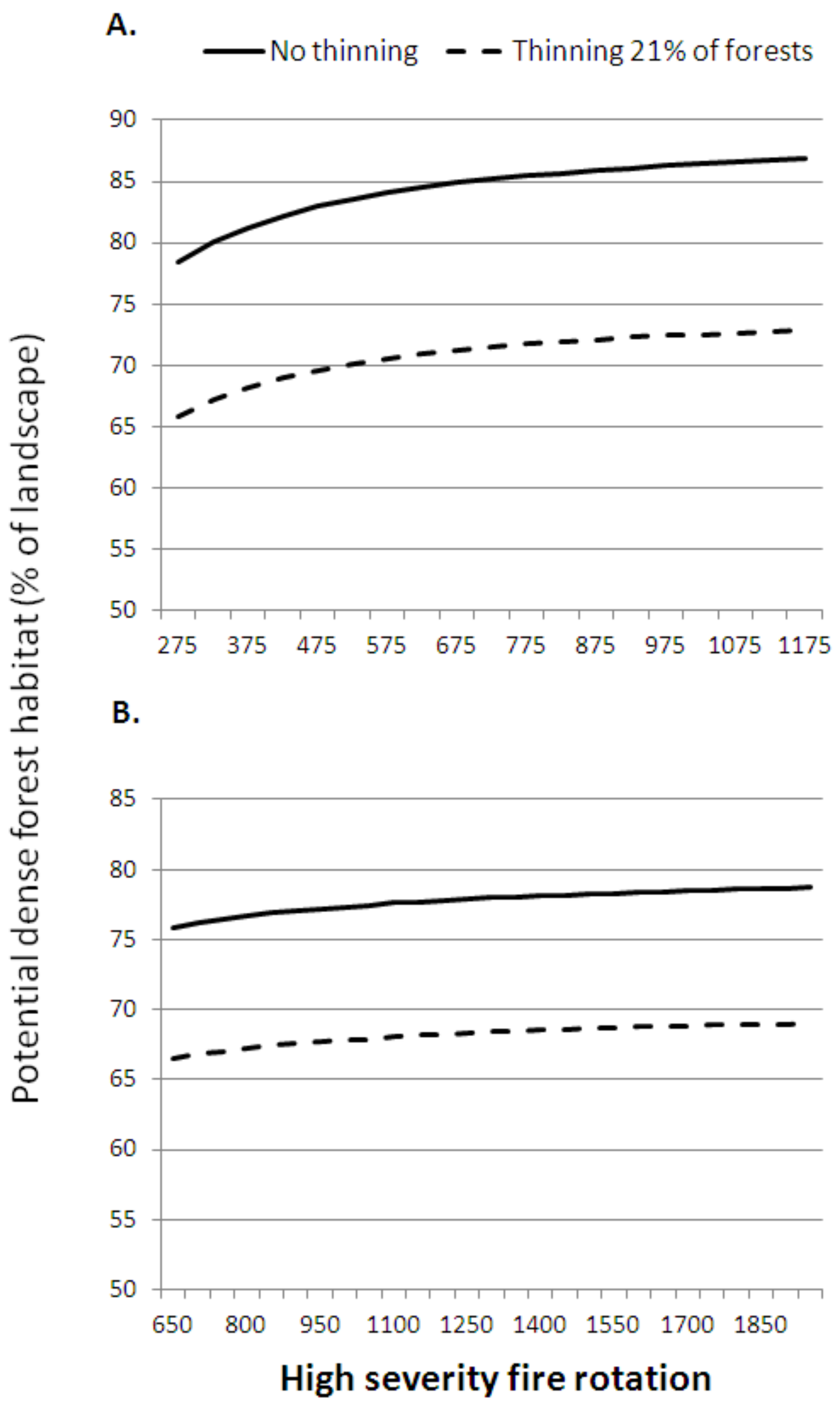

A.
Late successional
Mid-successional
Early successional
Thinned
B.
C.
Forest area (Km²)
10000
7500
5000
2500
0
0
10
20
30
40
Year

Figure 4



A.
Late successional
Mid-successional
Early successional
Thinned
B.
C.
10000
7500
5000
2500
0
Forest area (Km²)
0
10
20
30
40
Year


Klamath
Maintained
Unmaintained
Dry Cascades
Maintained
Unmaintained
0
10
20
30
40
0
-500
-1000
-1500
-2000
-2500
-3000
-3500
-4000
Change in habitat area (km2)
Percent of landscape treated


A.
No thinning
Thinning 21% of forests
90
85
80
75
70
65
60
55
50
275
375
475
575
675
775
875
975
1075
1175
B.
85
80
75
70
65
60
55
50
650
800
950
1100
1250
1400
1550
1700
1850
Potential dense forest habitat (% of landscape)
High severity fire rotation