34. ENVIRONMENTAL PROTECTION INFORMATION CENTER'S INDEPENDENT REPORT (CONSENT)

Today's Item **Information** ☒ **Action** ☐

Receive Environmental Protection Information Center's (EPIC) independent report on the petition to list northern spotted owl as a threatened or endangered species under the California Endangered Species Act.

Summary of Previous/Future Actions

- Publish notice of receipt of petition — Oct 5, 2012
- DFW's initial evaluation — Feb 7, 2013
- Candidacy decision — Aug 7, 2013
- **Today's receipt of independent report — Aug 5, 2015**
- DFW's status report — Dec 9-10, 2015; San Diego
- Take action to determine if listing is warranted — Feb 2016

Background

On June 2, 2015, EPIC submitted a status report of the northern spotted owl petition, pursuant to Subsection 670.1(h), which allows interested parties to submit a detailed written scientific report to FGC on the petitioned action.

Significant Public Comments

FGC received a letter from EPIC regarding the status of DFW's status report for the northern spotted owl.

Recommendation (N/A)

Exhibits

1. EPIC independent status report, dated May 7, 2015
2. Peer review comments on independent report, dated Jul 3, 2015
3. Letter from EPIC, received May 11, 2015

Motion/Direction

Moved by _____ and seconded by _____that the Commission adopts the Consent Calendar, items 33-37.

The Northern Spotted Owl in California: Current Status and Threats

Dan L. Hansen[1]

May 7, 2015

Photograph courtesy of Peter Carlson.

[1]danhansen03@gmail.com

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Introduction

This Independent Status Report was prepared pursuant to the California Endangered Species Act's (CESA) implementing regulations, specifically Cal. Code Regs., tit. 14, § 670.1 (h), which allows "interested parties . . . to submit a detailed written scientific report to the commission on the petitioned action." This same regulation explains that parties "may seek independent and competent peer review of this report prior to submission", and the author did so (see acknowledgements section at end of each chapter). Furthermore, to comply with the Fish and Game Code, section 2074.6, this report must be "based upon the best scientific information available."

This report was prepared by Wildlife Ecologist, Dan L. Hansen. His CV is attached, and a brief description of his qualifications is included herein (see *Project Author and Funding*).

This Independent Status Report was commissioned by the Environmental Protection Information Center (EPIC). However, its contents, conclusions, and management recommendations were exclusively developed by the author.

Executive Summary

This synthesis is organized into two parts. Part I consists of a single chapter: Status and Trends in California (Ch. 1). Part II covers four primary potential threats to northern spotted owls (*Strix occidentalis caurina*; NSOs) in California: Timber Harvesting (Ch. 2), Wildfires (Ch. 3), Barred Owls (Ch. 4), and Outdoor Marijuana Cultivation (Ch. 5). Following these chapters, a brief list of management recommendations is provided based on the information reviewed herein.

Chapter 1 is a review of the current status and trends of NSOs in California. Multiple types of information are available for evaluating the subspecies' status and trends in the state, including potential changes in its range, distribution, population densities, occupancy rates, demographic rates, metapopulation dynamics, and genetics. However, the most compelling information comes from long-term demographic studies in northwestern California (Forsman et al. 2011, Higley and Mendia 2013, Franklin et al. 2014, Green Diamond Resource Company 2014). These studies indicate that NSOs are declining in that portion of the state and that the rate of decline is accelerating. Competitive pressure from the congeneric barred owl (*S. varia*) appears to be the primary cause of increasing rates of population decline in the three demographic study areas. Occupancy data further support conclusions that NSOs in California's three demographic study areas are declining at an increasing rate and that the decline is largely driven by negative effects of barred owls (Franklin et al. 2014, Green Diamond Resource Company 2014). Less rigorous information is available for describing the NSO's current status and trends in California outside the state's demographic study areas. Only one published paper described occupancy in the eastern portion of the NSO's range in California (eastern Klamath and southern Cascades) (Farber and Kroll 2012). That paper described a substantial decline in occupancy by NSOs, which was likely associated with intensive timber harvesting and possibly, wildfires. Recent reports from demographic studies in southern Oregon further suggest that NSOs may be declining in relatively nearby and ecologically similar areas in California (eastern Klamath and southern Cascades) (Davis et al. 2013b, Dugger et al. 2014). Most of the other information for describing the subspecies' status and trends in California comes from monitoring reports by National Parks and industrial timber companies. NSOs appear to have been mostly displaced by barred owls in the Redwood National and State Parks (Schmidt 2013). In contrast, few barred owls have invaded National Park land in Marin County and occupancy by NSOs appears to be relatively stable in the area (Ellis et al. 2013). Industrial timber companies in California have uniformly concluded that NSO populations are stable on their lands (Calforests 2014). However, the available information for those ownerships does not support strong conclusions about the NSO's status or trends and some of the information actually appears to indicate at least gradual declines in occupancy.

Chapter 2 is a review of timber harvesting as a potential threat to NSOs in California. Habitat loss to timber harvesting was a primary impetus for listing the NSO under the federal Endangered Species Act (USFWS 1990). An estimated 60-88% of old forest was harvested within the NSO's range during the 19th and 20th centuries (USFWS 1990, Strittholt et al. 2006). Following federal listing of the NSO and adoption of the Northwest Forest Plan in the early 1990s, timber harvesting was dramatically curtailed on federal lands (Healey et al. 2008, Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012). However, there is currently considerable interest among some ecologists, land managers, and agencies in use of widespread

forest thinning to reduce the risk of large severe wildfires on public lands (USFS and BLM 1994, USFWS 2008, 2011, 2012a, Franklin and Johnson 2012). The term thinning can encompass a wide array of silvicultural practices and prescriptions but the limited available evidence suggests that NSOs and their primary prey in California generally respond negatively to thinning and partial harvesting (see Ch. 2). Timber harvesting is still responsible for most habitat loss and degradation for NSOs on private lands (Davis and Dugger 2011). Habitat loss and degradation on private lands does not appear to be offset by habitat recruitment; even in California, which has more stringent habitat protection measures on private lands than do Oregon and Washington (Davis and Dugger 2011, Moeur et al. 2011). Although timber harvesting is generally accepted to have been the primary cause of the NSO's initial decline and federal listing, its effects on the subspecies are poorly known. Several rigorous studies in the southern part of the NSO's range, including in northwestern California, have found that the NSO's fitness (a function of survival and reproduction) is typically highest in landscapes with both a core concentration of mature and old forest and some degree of habitat heterogeneity (e.g., a moderate amount of habitat edge due to convoluted shapes of older forest patches) (Franklin et al. 2000, Olson et al. 2004, Dugger et al. 2005, Schilling et al. 2013). Timber harvesting that substantially reduces either of these habitat attributes could negatively affect NSOs (USFWS 2009). The U.S. Fish and Wildlife Service (2009) concluded that current habitat retention guidelines for NSOs on industrial timberlands in interior California (CAL FIRE 2014) are inadequate and are not based on a current understanding of the subspecies' ecology.

Chapter 3 is a review of wildfire as a potential threat to NSOs in California. Several studies have investigated responses of NSOs to wildfires but their inferences are limited due to small sample sizes, short time frames, confounding effects of post-fire salvage logging, or pooling of data from all three subspecies. This information is supplemented in Chapter 3 with reviews of studies of effects of fire on California spotted owls (*S. o. occidentalis*) and Mexican spotted owls (*S. o. lucida*). Inferences from those studies are similarly limited by small sample sizes, short time frames, confounding effects of post-fire salvage logging, or pooling of different kinds of fire (prescribed fires, wildfires, and wildfires allowed to burn under specified conditions). Currently available information indicates that spotted owls respond in variable and complex ways to fire. The species appears to be generally resilient to low-, moderate-, and mixed-severity or patchy fires (Ch. 3: Table 3.1). It is possible that fires such as these sometimes benefit spotted owls by temporarily increasing access to prey that respond positively to fire (Ream 1981, Zwolak and Foresman 2007, Zwolak 2009, Bond et al. 2013). In contrast, spotted owls appear to generally respond negatively to extensive severe wildfires (Ch. 3: Table 3.1). While some spotted owls may preferentially forage in or near severely burned areas, they rarely nest and roost in such areas and may generally avoid foraging deep within them (Ch. 3: Table 3.1). The limited available information suggests that post-fire salvage logging negatively affects spotted owls (Clark 2007 and Clark et al. 2013, Lee et al. 2012, 2013). Salvage logging could reduce prey availability after fires by removing important structures, such as snags, logs, and shrubs. Habitat suitability modeling projected that wildfires caused substantial loss, degradation, and fragmentation of nesting and roosting habitat for NSOs on federal lands during the first 15 years of the Northwest Forest Plan (Davis and Dugger 2011). Most of these habitat changes were caused by a small number of extensive severe fires in southern Oregon and northern California. There is scientific debate regarding recent versus historical frequencies of high severity fire in southern Oregon and northern California (e.g., Hanson et al. 2009, 2010 vs. Spies et al. 2010).

Nonetheless, it is clear that large wildfires are now relatively common within the NSO's range in California (CAL FIRE 2008, Davis et al. 2011), and that some recent wildfires have severely burned very large areas (e.g., 2002 Biscuit Fire). Climate change research generally projects that large wildfires will become more common in California (Westerling et al. 2006, Lenihan et al. 2008, Westerling and Bryant 2008, Littell et al. 2009, Moritz et al. 2012, Stavros et al. 2014). It is reasonable to assume that some of these large wildfires will include extensive areas of high-severity fire and will therefore continue to be a source of habitat loss for NSOs.

Chapter 4 contains an evaluation of the barred owl as a potential threat to NSOs in California. The available information suggests that barred owls are currently the primary threat to NSOs throughout their range, including in California. Information from long-term demographic studies indicates that barred owls have contributed to the NSO's population declines in multiple study areas (Forsman et al. 2011) and that the barred owl's presence and negative impacts on NSOs are continuing to increase (Davis et al. 2013b, Higley and Mendia 2013, Franklin et al. 2014, Dugger et al. 2014). A large body of research conducted across much of the NSO's range has also shown that barred owls are associated with declines in occupancy rates by NSOs (Kelly 2001, Kelly et al. 2003, Pearson and Livezey 2003, Gremel 2005, Olson et al. 2005, Kroll et al. 2010, Dugger et al. 2011, Higley and Mendia 2013, Franklin et al. 2014, Green Diamond Resource Company 2014). Barred owls negatively affect NSOs by competing for space, habitat, and food (Gutiérrez et al. 2004, 2007, USFWS 2013, Wiens et al. 2014). The barred owl appears to be a superior competitor to the NSO due to its larger size, more aggressive behavior, higher reproductive potential, higher population densities, and broader ecological niche (e.g., USFWS 2013, Wiens et al. 2014). The available information suggests that lethal control of barred owls is a viable management option for some areas, although there is ethical and emotional resistance to this within some segments of society (Diller et al. 2013, Higley 2014). The negative effects of habitat loss and fragmentation on NSOs can be exacerbated by the presence of barred owls (Dugger et al. 2011). Thus, the barred owl threat magnifies the importance of habitat conservation for NSOs, rather than reducing it.

Chapter 5 is a review of outdoor marijuana cultivation as an emerging potential threat to NSOs in California. Marijuana is one of California's largest cash crops (Gettman 2006) but little is known about the environmental effects of its cultivation. Recent research in northwestern California has shown that both fishers (*Pekania pennanti*) and barred owls are regularly exposed to anti-coagulant rodenticides used to protect marijuana plants from rodents (Gabriel et al. 2012, 2014). Multiple fishers are known to have died due to poisoning from anti-coagulant rodenticides (Gabriel et al. 2012, 2013). Fishers and barred owls have overlapping distributions, habitat associations, and diets with NSOs so it is likely that many NSOs in California are likewise exposed to these toxicants. This was supported by recovery of a dead NSO in Mendocino County, which tested positive for exposure to anti-coagulant rodenticides (Calforests 2014). Marijuana cultivation could also negatively affect NSOs through habitat changes caused by illegal and poorly planned logging, road construction, pollution, and water diversion (Gabriel et al. 2013, Bauer et al. 2015). Marijuana cultivation can particularly impact riparian areas. These impacts could indirectly affect NSOs, which often show a preference for nesting, roosting, and foraging in riparian areas (e.g., Hamer et al. 2007, Irwin et al. 2012). Safety concerns about encounters with armed marijuana growers are resulting in reduced conservation research and

monitoring effort and efficiency for NSOs and other sensitive wildlife species in California (Keane et al. 2011, Gabriel et al. 2013, D. Hansen pers. obs.).

Overall, this synthesis supports conclusions that NSOs in California are declining at an increasing rate (Ch. 1) and that they face an array of threats to their persistence (Chs. 2-5). Barred owls appear to pose the greatest current threat to NSOs (Ch. 4). If conservation of NSOs is to remain a priority then widespread barred owl removal programs may be necessary. Habitat protection also remains an important aspect of NSO conservation. Listing under the California Endangered Species Act, substantial changes to the California Forest Practice Rules habitat retention guidelines, and greater involvement by knowledgeable spotted owl biologists in the Timber Harvest Plan review process may be necessary to adequately protect habitat for NSOs on private lands in the state. The available information suggests that large severe wildfires pose a threat to NSOs on federal lands in California (Ch. 3). Some ecologists and land management agencies have proposed widespread use of forest thinning and prescribed fire to reduce this risk. However, the available information also suggests that spotted owls often respond negatively to thinning (Ch. 2). It is important for land managers to consider potential tradeoffs in costs and benefits of thinning in landscapes occupied by NSOs. Thinning could potentially be focused in areas that generally receive the least use by spotted owls and that have the highest fire risk, such as upper and southwesterly slopes (see Ch. 3). The limited information currently available suggests that post-fire salvage logging negatively affects NSOs and their prey (Ch. 2). Further research of this topic is needed but this practice does not appear to be generally concordant with conservation of NSOs in California. Marijuana cultivation appears to pose a substantial emerging threat to NSOs in California; particularly trespass operations on federal lands (Ch. 5). Increased research, law enforcement, and site cleanup and restoration efforts are likely needed to protect NSOs from negative effects of outdoor marijuana cultivation in California.

Methods

The northern spotted owl (*Strix occidentalis caurina*; NSO) has been a focus of conservation concern and research for more than three decades (see reviews in Courtney et al. 2004 and USFWS 2011a). Although substantial habitat protection measures exist for NSOs on federal lands, the subspecies has continued to decline across much of its range (Forsman et al. 2011). Indeed, the U.S. Fish and Wildlife Service (2015) recently concluded that uplisting the NSO from 'threatened' to 'endangered' under the federal Endangered Species Act may be warranted. The California Fish and Game Commission will soon decide whether or not to list the NSO under the California Endangered Species Act. The large body of research and monitoring information concerning the NSO can be challenging for natural resource agencies and policymakers to evaluate. In order to inform the California Endangered Species Act listing decision and other policy and management actions, I have endeavored to synthesize much of the available scientific information concerning the NSO's current status, trends, and threats in the state.

While writing this synthesis, I reviewed information from a variety of sources but generally gave greater weight to peer-reviewed publications, particularly those based on more rigorous field and analytical methods. For example, in Chapter 1, I attempted to carefully consider all available information about the NSO's current status and trends in California but gave greatest weight to results of long-term demographic studies. Some topics related to the NSO's ecology and conservation are scientifically and politically contentious; for example, whether wildfire constitutes a threat to the subspecies and, if so, whether or not it should be addressed through active management approaches, such as forest thinning (see Ch. 3). In these situations, or when published information was limited (e.g., regarding effects of forest thinning on NSOs: see Ch. 2), I treated research results as 'case studies' and described each study's methods and findings in more detail than is common in these kinds of reviews. Although this approach could have underweighted peer-reviewed publications, it allowed me to thoroughly search for patterns among numerous studies, draw tentative conclusions based on those patterns, and highlight gaps in available information about the topic. I also felt that it was important to carefully consider all available sources of information, rather than peer-reviewed publications alone, due to the tremendous variation in ecology and management history that exists within the NSO's range in California. For instance, I felt that it was especially important to evaluate timber industry and National Park monitoring data for portions of California outside the area that includes the state's three demographic studies (see Ch. 1: Figure 1.22 or USFWS 2011a Appendix C for California ecoregional boundaries). In all cases, I was transparent about my approach and the strengths and weaknesses of the available information.

Project Author and Funding

Dan Hansen is a freelance wildlife ecologist (see attached CV). He has an M.S. in animal behavior (conservation biology track) from the University of California at Davis and a B.A. in biology (ecology, evolution, and behavior emphasis) from the University of California at Santa Cruz. He has worked as a contractor for the U.S. Fish and Wildlife Service (USFWS) and U.S. Forest Service (USFS) through the Humboldt State University Sponsored Programs Foundation for most of the last decade. During this period, he primarily coordinated, wrote, and edited scientific syntheses aimed at informing public policy and land management decisions. Documents that Dan contributed to include the USFWS revised recovery plan for the northern spotted owl (critical habitat modeling and effects of forest thinning), the USFS conservation assessment for the northern goshawk in California (a peer-reviewed book to be published in 2015), a USFWS peer-reviewed white paper supporting changes to the California Forest Practice Rules habitat retention guidelines for northern spotted owls in interior northern California, and a USFWS 12-month finding on a petition to list two salamander species under the Endangered Species Act. In addition to working as a technical coordinator, writer, and editor, Dan has worked in the field researching or monitoring spotted owls, golden eagles, bald eagles, peregrine falcons, northern goshawks, carnivores, small mammals, marine mammals, and other wildlife. Prior to working at Humboldt State University, Dan co-managed a large study of California spotted owls in the northern Sierra Nevada.

This document was funded by the Environmental Protection Information Center (EPIC), a non-profit organization focused on environmental issues in northwestern California. EPIC filed a petition with the California Fish and Game Commission in September 2012 to list the northern spotted owl as 'threatened' or 'endangered' pursuant to the California Endangered Species Act. Although EPIC funded the following synthesis, they provided minimal feedback during its writing and were supportive of Dan's goal of producing an unbiased, peer-reviewed synthesis concerning the northern spotted owl's current status and threats in California. All conclusions and management recommendations in this synthesis are the author's alone and do not necessarily reflect those of EPIC or any of the document's reviewers.

Part I, Ch. 1: Status and Trends in California

Introduction

Several lines of evidence are available for evaluating the northern spotted owl's (*Strix occidentalis caurina*; NSO) status and trends in California. These include changes in the subspecies' geographic range and distribution, density and abundance, occupancy and demographic rates, meta-population dynamics, and genetics. The most reliable information available for examining the NSO's status and trends is provided by long-term demographic studies. Data from these studies are periodically analyzed together in meta-analyses for describing larger demographic patterns within ecoregions and rangewide. A new demographic meta-analysis is expected to be released later this year. This document will provide the best available information for determining the NSO's current status and trends. However, only a small portion of the NSO's range in California occurs within demographic study areas and those study areas all occur in relatively productive forests in the northwestern part of the state. It is therefore important to consider other sources of information, such as data collected for monitoring NSOs in National Parks and on industrial timber company lands and results of demographic studies in areas that ecologically resemble portions of interior northern California. The NSO's status and trends likely vary among regions, forest types, and ownerships and could be influenced by a host of factors, such as differences in forest ecology, management history, and stressors such as competition with invasive barred owls (*Strix varia*). It is also important to remember that available sources of information for evaluating the NSO's status and trends in California vary substantially in terms of their purpose and scientific rigor.

Range

The current range of the NSO includes southwestern British Columbia and the Cascade Mountains, coastal ranges, and intervening forests of Washington, Oregon, and California (USFWS 2011a). In California, the NSO's range extends from the Oregon border through the Northern Coast Ranges to Marin County, across the Klamath Mountains, and down the southern Cascades to the vicinity of the Pit River, where it contacts the range of the California spotted owl (*S. o. occidentalis*) (Figure 1.1).

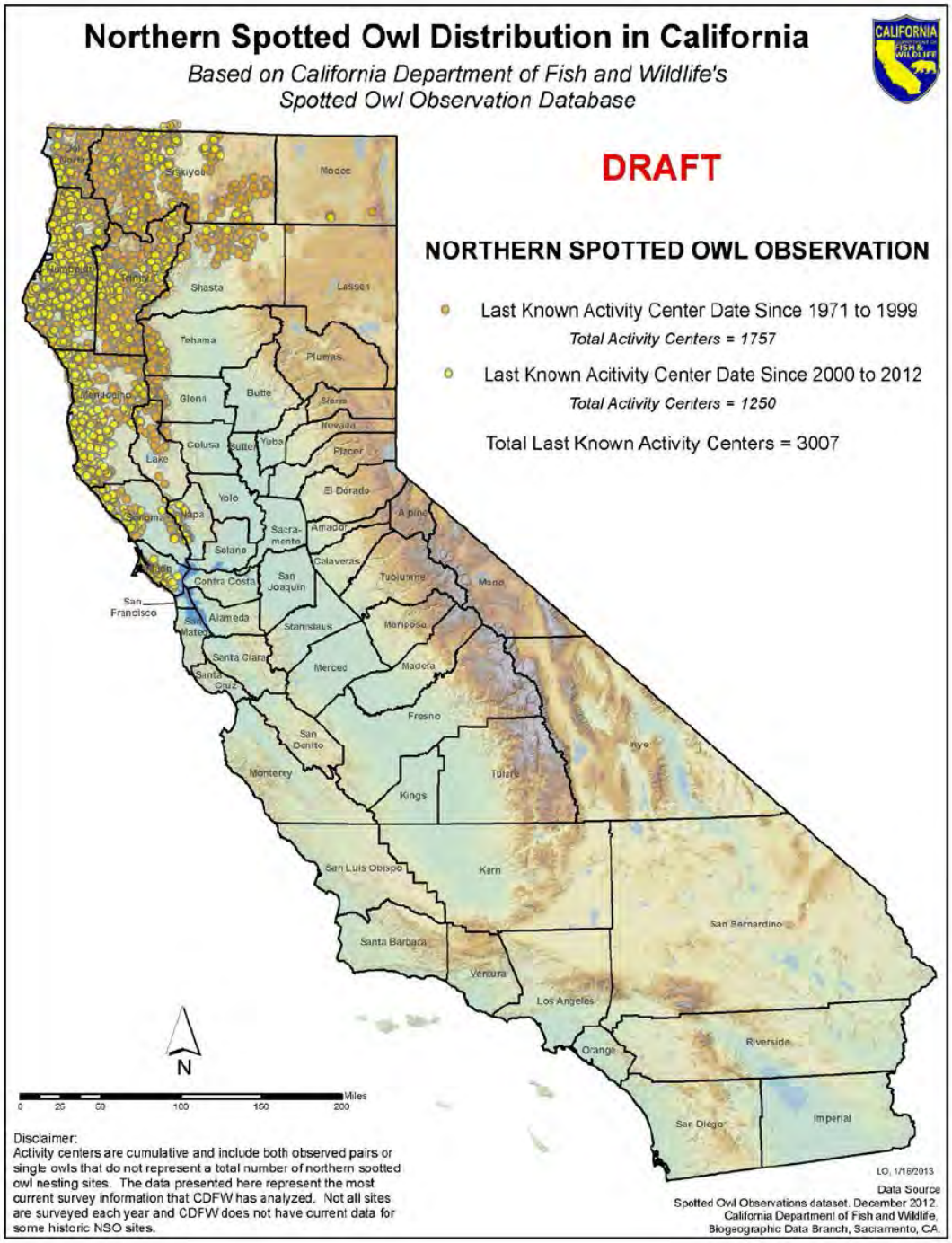
The precise historical range of the NSO is unknown. Thus, despite substantial loss, degradation, and fragmentation of NSO habitat (see Ch. 2 and 3), there is no evidence that the subspecies' range has contracted since Euro-American settlement (Thomas et al. 1990). However, British Columbia's NSO population has declined to very low numbers and is highly vulnerable to extirpation (Chutter et al. 2004). NSO populations in the Cascades and Olympic Peninsula of Washington and the Northern Coast Range of Oregon are also rapidly declining and may become vulnerable to extirpation (Forsman et al. 2011; see *Demography*, below). Loss of NSO populations could cause substantial contraction of the subspecies' range. For example, extirpation of NSOs from British Columbia alone would reduce the subspecies' range by approximately 8% (Cooper 2006).

Distribution

NSOs are thought to have been well distributed throughout most coniferous forests in the Pacific Northwest and northwestern California prior to Euro-American settlement (USFWS 2011a). The abundance and distribution of NSOs have likely declined due to removal of most (ca. 60-88%) old forest within its range (USFWS 1990; see Ch. 2). For example, the Puget Trough in Washington and the Willamette Valley in Oregon no longer support NSOs due to land conversion and timber harvesting and very few NSOs remain in British Columbia (Thomas et al. 1990, Courtney et al. 2004). The NSO's distribution has decreased in other areas of Washington and Oregon as well, due primarily to negative effects of timber harvesting, wildfires, and competition with barred owls (*Strix varia*) (Thomas et al. 1990, USFWS 2011a; see Ch. 2-4).

It is unknown if the NSO's distribution has changed in California. A difference is evident in the distribution of known historically (1971-1999) and recently (2000-2012) occupied activity centers (ACs) in the Eastern Klamath, Interior Northern Coast Ranges, and Southern Cascades regions of the state (Figure 1.1; see Figure 1.22 [left side] and USFWS 2011a Appendix C for ecoregional boundaries generally followed in this synthesis). It is unclear from these data, however, whether the distribution of NSOs has in fact decreased in these areas or if the apparent decline in distribution is due to some other factor such as decreased survey effort or reporting of detections. It is also possible that this difference is due to the greater number of years included in the historical period than in the recent period (29 vs. 13 yrs). However, the two periods are similar in length relative to federal listing of the NSO (10 vs. 13 yrs) when survey effort presumably became more intensive and widespread. Some portions of the Klamath, Interior Northern Coast Ranges, and Southern Cascades have experienced widespread intensive timber harvesting or large wildfires, which could have reduced the NSO's distribution (see Ch. 2 and 3). These forms of disturbance, along with competition with invasive barred owls, have likely contributed to declining occupancy by NSOs in some areas of California (see *Occupancy*, below). Nonetheless, the Klamath and Interior Northern Coast Ranges (but not the Southern Cascades) still appear to contain relatively large amounts of well connected suitable habitat and may function as crucial population sources for NSOs (Schumaker et al. 2014; see *Source-Sink Dynamics*, below). It is also possible that the distribution of NSOs has expanded at local or sub-regional levels in some portions of California due to increased distribution or density of suitable forest habitat in the absence of fire (Skinner 1995, Spies et al. 2006).

Figure 1.1: Distribution of northern spotted owl activity centers in the California Department of Fish and Wildlife's Spotted Owl Observation Database (from California Department of Fish and Wildlife 2013).



Density and Abundance

Species are rarely uniformly distributed across their range. Knowledge of variation in the density and abundance of NSOs is of potential conservation value because it can help identify areas where limited conservation resources should be focused. For example, while declines in low-abundance areas may be more likely to cause contraction of a species' range or distribution (see *Range* and *Distribution*, above), declines in high-abundance areas may disproportionately impact the species' probability of long-term persistence; particularly when high-abundance areas function as population sources (Pulliam 1988, Rodríguez 2002, Schumaker et al. 2014; see *Source-Sink Dynamics*, below).

Several studies have estimated either crude densities (owls or occupied territories per unit area) or ecological densities (owls per unit area of specified habitat class[es]) of NSOs in California (Blakesley et al. 2004; Table 1.1). These estimates are interesting in that they appear to reflect geographic variation in the ecology of NSOs (see below). However, they have limited utility for evaluating the NSO's status or trends in California. Available density estimates for the state are largely restricted to relatively mesic areas of northwestern California, which differ ecologically from drier interior forests (e.g., in terms of climate, forest productivity, and prey communities). Inferences from most density estimates are also limited because they are based on empirical counts of unmarked NSOs, which can bias estimates (Franklin et al. 1990, Diller and Thome 1999). Many of the currently available density and occupancy estimates for NSOs in California were provided by timber companies (Tables 1.1 and 1.2). While potentially useful for evaluating effects of management activities on timber company lands, these estimates do not describe population trends. Rigorous evaluation of NSO population trends require long-term statistically valid sampling designs from which estimates of abundance, or population growth rate with confidence intervals, can be repeatedly obtained within the same study area. In contrast, timber companies generally shift their NSO survey areas over time as timber harvest projects are completed in some areas and begun in others.

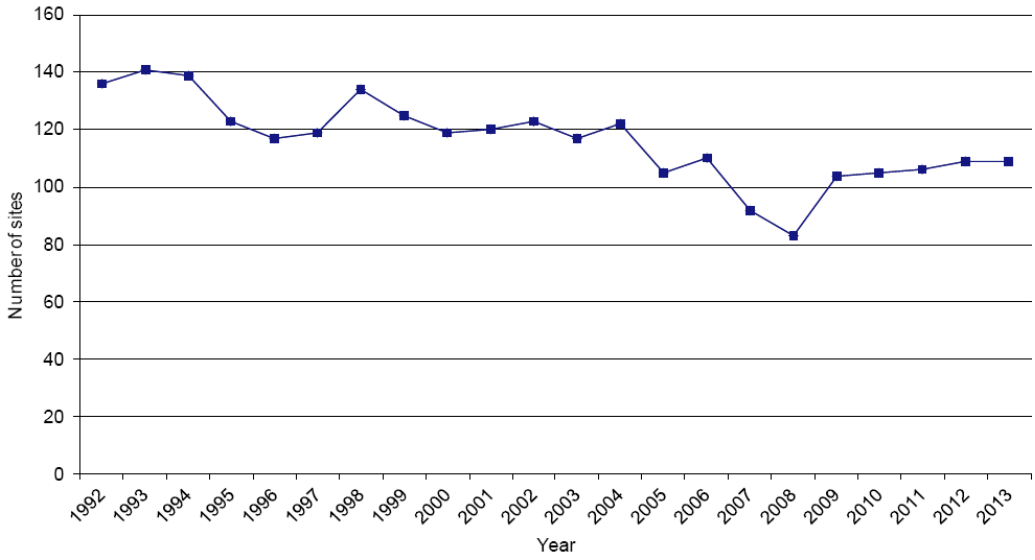
Based on limited information, both crude and ecological densities of NSOs appear to be substantially higher in northwestern California than in the Oregon Coast Ranges (Blakesley et al. 2004; Table 1.1). Lower densities in the Oregon Coast Ranges could be partially related to widespread intensive timber harvesting, which apparently contributed to a major decline in densities during the early 1990s (Thrailkill et al. 1998). Some areas of northwestern California have also experienced widespread intensive timber harvesting (see Ch. 2) but its effect on NSOs might have differed from that in the Oregon Coast Ranges. In general, NSOs in California primarily subsist on dusky-footed woodrats (*Neotoma fuscipes*) (in terms of biomass contribution to diets). NSOs that primarily subsist on dusky-footed woodrat often have smaller home ranges, and apparently occur at higher densities, than those that primarily rely on smaller-bodied prey (Carey et al. 1992, Zabel et al. 1995). Furthermore, in contrast with other primary prey species, such as northern flying squirrels (*Glaucomys sabrinus*) and tree voles (*Arborimus* spp.), dusky-footed woodrats seem to respond positively, albeit temporarily, to some forms of intensive timber harvesting (see Ch. 2).

Densities of NSOs are also thought to be higher in northwestern California than in the state's interior (Calforests 2014). However, there are apparently only two density estimates currently

available for interior northern California and both of these were for crude densities of occupied territories, rather than for individual owls as estimated by most studies in northwestern California (Table 1.1). These crude territory densities are substantially lower than those found on two timber companies' lands in the Redwood Region (Table 1.1). NSO densities may be relatively low in the Southern Cascades of California due to the prevalence of drier, less productive forests, a history of widespread intensive harvesting, and effects of recent large wildfires (see Ch. 2 and 3). Additional density estimates are needed for the Eastern Klamath of California. It is uncertain whether Sierra Pacific Industries' (2013) estimates are representative of densities across the region as a whole. Most ACs included in Sierra Pacific's density estimates were located near the margins of the company's lands or on adjacent ownerships, rather than within the interiors of the company's holdings (see Maps 2-5 in Sierra Pacific Industries 2013, which are copyrighted and cannot be reproduced without permission). This pattern suggests that densities could be higher on neighboring lands, such as the Shasta-Trinity National Forest, which have generally experienced less intensive management.

The California Forestry Association cited annual density estimates in timber company monitoring reports as evidence of stable or increasing NSO populations on private timberlands in the state (Calforests 2014). Reported crude densities on Humboldt Redwood Company, Mendocino Redwood Company, and The Conservation Fund lands in the Redwood Region were indeed relatively similar among years (Calforests 2014). However, it is unclear how changing survey methods and survey areas, as well as changing detectability of NSOs, influenced these companies' estimates over time (see Franklin et al. 1990). For example, recent adoption of survey protocols requiring more survey passes and use of electronic callers likely increased detection rates, and thus density estimates, on some of these lands. Estimates of crude densities of NSOs and numbers of ACs on Green Diamond Resource Company lands in the Redwood Region suggest that NSO densities have declined on that ownership (Figure 1.2; Table 1.1). The number of NSO ACs on Green Diamond lands briefly increased in 1998, apparently due to the company's acquisition of 70,000 acres of timberland that year (Green Diamond Resource Company 2014). Following a substantial decline during 2004-2008, the number of ACs began to gradually increase in 2009 (Figure 1.2). This increase appears to have been due to the company's adoption of a more rigorous survey protocol and implementation of a barred owl removal experiment during that same year (Green Diamond Resource Company 2014; see Ch. 4). Sierra Pacific Industries' (2013) density trends for its ownerships in the Eastern Klamath are difficult to evaluate and are therefore not included here. Sierra Pacific Industries' (2013) estimates are empirical, potentially influenced by changing survey effort and areas, mostly descriptive of ACs at the margin of or outside the company's ownership, and were compared among blocks of years, rather than annually.

Figure 1.2: Number of NSO activity centers ("sites") on Green Diamond Resource Co. lands during 1992-2013 (from Green Diamond Resource Co. 2014).



Rigorous ecological density estimates can be used to estimate population sizes for ecologically similar areas (Franklin et al. 1990). However, there are currently insufficient data for producing such an estimate for California or any of its regions. The California Forestry Association estimated that as many as 6,000 NSO territories currently exist in the state (Calforests 2014). This figure was based on an estimated statewide crude density of 0.28 territories per mile^2. This density estimate was, in turn, based on the cumulative number of known NSO ACs in California (see *Distribution*, above) and the proportion of "potential" habitat in the state that has been surveyed. The number of ACs known to have been recently occupied is substantially lower than the cumulative number that have been identified since the early 1970s (USFWS 2011a; see *Distribution*, above). This could be due to multiple factors, including declining occupancy rates (see *Occupancy*, below) and NSOs' use of different ACs over time. Thus, the timber industry's estimate provides little or no insight into the current number of NSOs or occupied ACs in the state. Furthermore, while reasonable projections of suitable habitat exist for NSOs in California (Davis and Dugger 2011, USFWS 2011a Appendix C, Schumaker et al. 2014), the California Forestry Association did not cite these data and it is unclear how it estimated the total and surveyed areas of suitable habitat in the state.

Table 1.1: Density estimates for NSOs in California and Oregon.

Study	Region	Owner	Method	Crude Density (owls/ mi²)*	Crude Density (occupied territories/ mi²)*	Ecological Density (owls/ mi²)**†
Diller and Thome 1999	N Redwood	Green Diamond	Mark-Recapture	0.54 (0.24-0.91)		0.97-2.72
Green Diamond Resource Co. 2014	N Redwood	Green Diamond	Empirical (marked)	0.34 (0.12-0.53)		
Tanner and Gutierrez 1995 cited in Diller and Thome 1999	N Redwood	Redwood National Park	Empirical	0.57		
Humboldt Redwood Co. 2013	N Redwood	Humboldt Redwood	Empirical	0.53-1.01	0.36-0.50	
Mendocino Redwood Co. 2014	N Redwood	Mendocino Redwood	Empirical	0.47-.077		
The Conservation Fund unpubl. data in Calforests 2014	N Redwood	The Conservation Fund	Empirical		0.29-.036	
Chow 2001	S Redwood	Public (Various)	Empirical	0.97		2.09
Franklin et al. 1990	W Klamath	Six Rivers National Forest, Other	Mark-Recapture	0.61		1.41-1.71
Sierra Pacific Industries 2013	E Klamath	Sierra Pacific Industries	Empirical		0.17-0.18	
Woodbridge and Cheyne 1995	So Cascades	Klamath National Forest	Empirical		0.05-0.20	
Thrailkill et al. 1998	OR Coast Ranges	Bureau of Land Management	Empirical	0.07-0.25		0.57-0.90
Anthony et al. 2000 cited in Blakesley et al. 2004	OR Coast Ranges	Oregon Dept. of Forestry	Empirical	0.13-0.27		

*Ranges = low-high survey areas (Woodbridge and Cheyn 1995, Thraillkill et al. 1998, Diller and Thome 1999, Anthony et al. 2000, Green Diamond Resource Co. 2014), low-high survey years (Humboldt Redwood Co. 2013, Mendocino Redwood Co. 2014). **Habitat definitions used to calculate ecological densities: Franklin et al. 1990: all conifer cover classes weighted by NSO use (based on telemetry), but mostly ≥20.6 in DBH; Diller and Thome 1999: all forest classes weighted by NSO use (based on nest locations) but mostly >40 yrs; Chow 2001: all forested area; Thrailkill et al. 1998: old, mature, old over young, mature over young. †Ranges of ecological densities: Franklin et al. 1990: with two different habitat definitions; Diller and Thome 1999: low-high survey areas.

Occupancy

NSO population trends are most directly evaluated with demographic data (see *Demography*, below). However, occupancy data are often more logistically and economically feasible to collect than demographic data and, with proper accounting of detection probability, can provide a useful index of spotted owl population rates (MacKenzie 2005, Olson et al. 2005, Mackenzie et al. 2012, Tempel 2014). Occupancy data that inadequately incorporate detection probabilities for spotted owls must be interpreted carefully since they can be strongly influenced by survey effort, habitat attributes, social and reproductive status of NSOs, presence of barred owls, and other factors (Mackenzie 2005, Olson et al. 2005). Recent research in NSO demographic study

areas suggests that competition with barred owls is driving NSOs to move large distances (several miles) between different territories within the same season (Davis et al. 2013, Higley and Mendia 2013). Higley and Mendia (2013) warned that occupancy estimates for unmarked populations may therefore be inflated (i.e., the same individual could appear to occupy multiple territories within the same season) and suggested using the presence of pairs, rather than individuals, to determine occupancy.

There is limited information available for describing occupancy trends for NSOs in California. Much of the available information is from annual monitoring reports provided by industrial timber companies (Table 1.2). These data show trends in annual proportions of known, surveyed, or previous year's ACs found to be occupied (see Table 1.2 footnote). It is important to acknowledge that much of the data presented in Table 1.2 provide only crude indices of occupancy in California and that most of them cannot be compared among ownerships due to differences in monitoring and analytical methods. Future efforts to evaluate the status of NSOs in California would benefit from greater consistency in occupancy monitoring and from reporting of modeled occupancy rates, which account for detectability of NSOs and other factors that can obscure occupancy trends (e.g., Figure 1.5).

Recent occupancy estimates are unavailable for the Redwood National and State Parks in the northern portion of the Redwood Region. The National Park Service has discontinued surveying most historical territories in these parks due to apparent widespread displacement of NSOs by barred owls (Schmidt 2013; see Ch. 4). In contrast with an apparently strong decline in occupancy in the Redwood National and State Parks, NSO occupancy rates on National Park Service lands in the southern portion of the Redwood Region have fluctuated annually but suggest a stable trend over time (Ellis et al. 2013; Table 1.2; Figure 1.3). Perhaps due to the area's geographic isolation, barred owls are still relatively uncommon the southern Redwood Region (Ellis et al. 2013; see Ch. 4). Occupancy by NSOs appears to be gradually declining on industrial timberlands in the northern Redwood Region (Table 1.2; Figures 1.2, 1.4, 1.5, and 1.7). Given the substantial and increasing presence of barred owls in NSO territories on these lands (see Ch. 4), it is surprising that more dramatic declines in NSO occupancy are not evident (e.g., see Table 1.2 for occupancy rates in Washington and Oregon). It is possible that NSOs respond differently to barred owls on these lands than elsewhere within their range. It is also possible that a more rapid decline is currently occurring than is indicated by the crude data presented in these companies' reports. Yet another possibility is that a more rapid decline will occur on these lands after a post-colonization lag period has elapsed or a critical threshold level of barred owl presence is reached (USFWS 2013).

NSO occupancy in the Northwestern California demographic study in the Western Klamath Region has declined dramatically in recent years (Franklin et al. 2013, 2014; Table 1.2). This decline has coincided with increasing barred owl presence in the study area, suggesting that NSOs are being displaced by barred owls (see Ch. 4). The recently increased rate of declining occupancy by NSOs in this study area appears to support the hypothesis that barred owls can have lag or threshold effects on NSO populations. Recent declines in occupancy in the Northwestern California study area may also be related to effects of multiple consecutive years of poor weather conditions on demographic rates (see *Demography*, below). Recent annual reports from the Hoopa demographic study did not include analyses of occupancy data for NSOs

(Higley and Mendia 2012, 2013). However, unmodeled occupancy rates in 2012 and 2013 were low (0.40 and 0.35, respectively). Low occupancy rates on the Hoopa Reservation may be related to substantial declines in numbers of NSOs, likely due to decreasing demographic rates (see *Demography*, below) and increasing numbers of NSO territories with barred owl detections (see Ch. 4). Greater declines in numbers of NSOs and increases in NSO territories with barred owl detections beginning in 2005 provide additional support for the hypothesis that barred owls have lag or threshold effects on NSOs.

There is currently no clear pattern in occupancy data available for the Eastern Klamath and Southern Cascades of California. Timber companies in those regions have reported evidence of stable occupancy rates (Sierra Pacific Industries 2013, Michigan-California Timber Company 2014; Figure 1.6; note: Sierra Pacific's estimates are not provided in Table 1.2 for reasons discussed in *Density and Abundance*, above). However, more rigorous, published research conducted primarily on industrial timberlands in the Eastern Klamath and Southern Cascades found substantial declines in both simple (total) and pair occupancy (Farber and Kroll 2012; Figure 1.7). The barred owl invasion appears to still be in the early colonization phase in the Eastern Klamath, where this study was primarily conducted (Farber and Kroll 2012; see Ch. 4). Thus, declining occupancy during the study was likely caused by some other factor, such as timber harvesting on the industrial timberlands that comprised much of the study area or wildfires on neighboring public lands (see Ch. 2 and 3). Research in other areas of the NSO's range indicates that occupancy can be negatively affected by habitat loss and fragmentation (Dugger et al. 2011, Sovern et al. 2014).

Table 1.2: Estimates and indices of occupancy by northern spotted owls in California, Oregon, and Washington.

Study	Region	Owner	Years	Number of Sites	Proportion Occupied (Total)	Proportion Occupied (Pairs)	Modeled Occupancy (Total)	Modeled Occupancy (Pairs)	Apparent Trend
Mendocino Redwood Company 2013	N Redwood	Private	2001-2013 (proportion); 2001-2008 (modeled)		0.75 - 0.69*		0.88 - 0.78*†		Declining (weak)
Humboldt Redwood Company 2014	N Redwood	Private	2003-2013		0.81 - 0.63*†				Declining (weak)
Green Diamond Resource Company 2014	N Redwood	Private	1999-2013 (no. sites); 2009-2013 (occupancy)	135 - 108*†	0.88 - 0.83*‡				Declining (weak)
Ellis et al. 2013	S Redwood	NPS	1999-2012		0.86 - 0.94*	0.72 - 0.87*			Stable
Franklin et al. 2002, 2003, 2010-2014	W Klamath	USFS, BLM, Private	2001-2013		0.67 - 0.37*	0.59 - 0.28*			Declining (strong)
Farber and Kroll 2012	E Klamath, S Cascades	Private, USFS	1995-2009				0.81 - 0.50*	0.75 - 0.46*	Declining (strong)
Michigan-California Timber Company 2014	E Klamath, S Cascades	Private	2000-2013		0.35 - 0.52 (2001-2013: 0.66 - 0.52)*†				Stable
Davis et al. 2013a	OR Klamath	BLM, State, Private	2001-2013		0.86 - 0.49*	0.62 - 0.30*			Declining (strong)
Dugger et al. 2011	OR S Cascades	USFS	1991/1992-2006				w/o barred owls: 0.86 - 0.71*†; w/ barred owls: 0.87 - 0.11*†		Declining (strong)
Kroll et al. 2010	WA E Cascades	NPS, USFS, Private	1990/1-2003				w/o barred owls: 0.83 - 0.64*†; w/ barred owls: 0.73 - 0.30*†	w/o barred owls: 074 - 0.36*†	Declining (strong)

*Start and end values. †Estimated from graph. ‡Occupancy of previous year's sites.

Figure 1.3: Occupancy status at monitored northern spotted activity centers on National Park Service lands in Marin County, California during 1999-2012 (from Ellis et al. 2013).

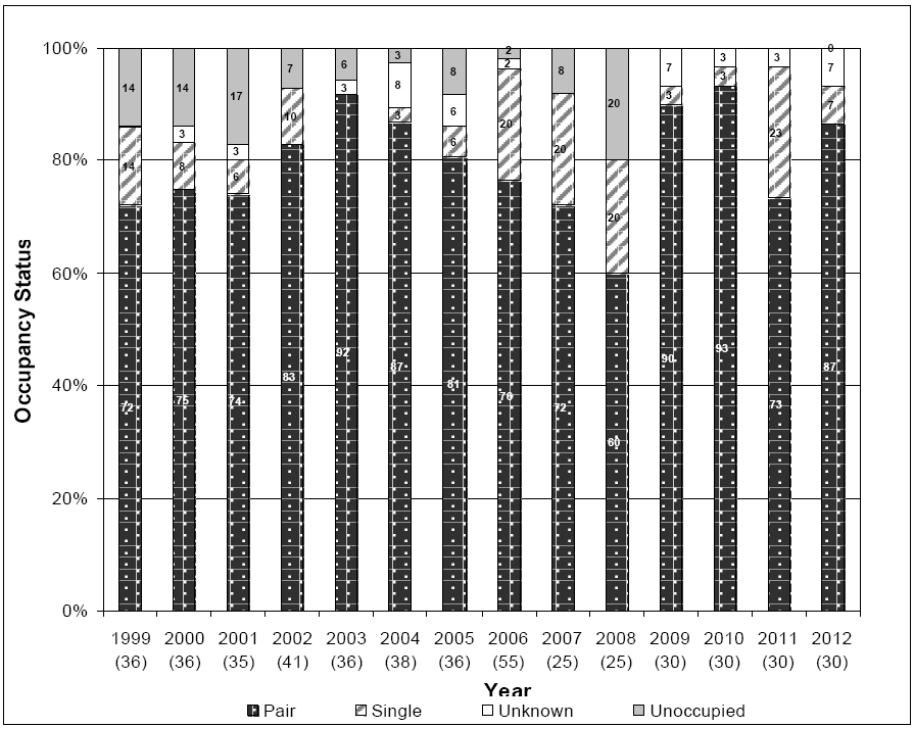


Figure 1.4: Annual numbers of known and occupied northern spotted owl activity sites (activity centers) on Humboldt Redwood Company lands during 2003-2013 (from Humboldt Redwood Company 2014).

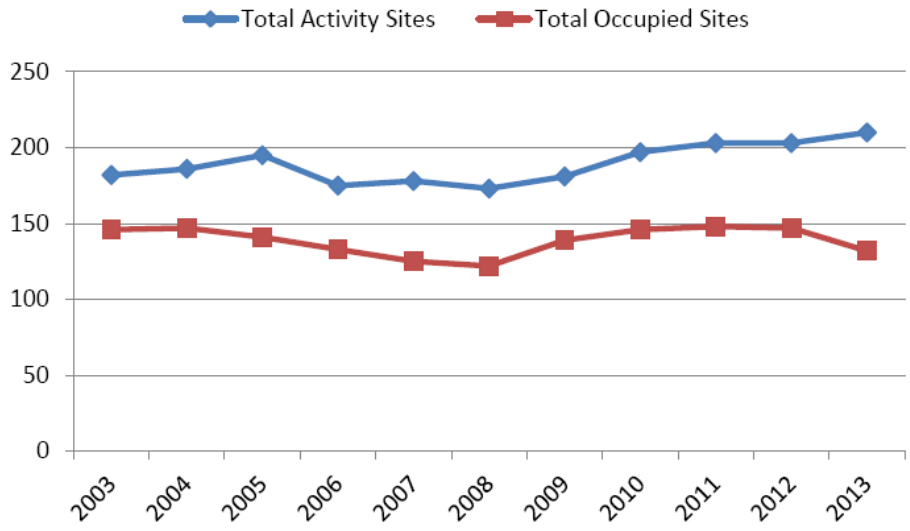


Figure 1.5: Annual proportion of northern spotted owl activity centers occupied (blue line) and modeled occupancy probability (red line) on Mendocino Redwood Company lands (from Mendocino Redwood Company 2013). Note the apparent decline in modeled occupancy compared with the lack of a clear trend in unmodeled occupancy.

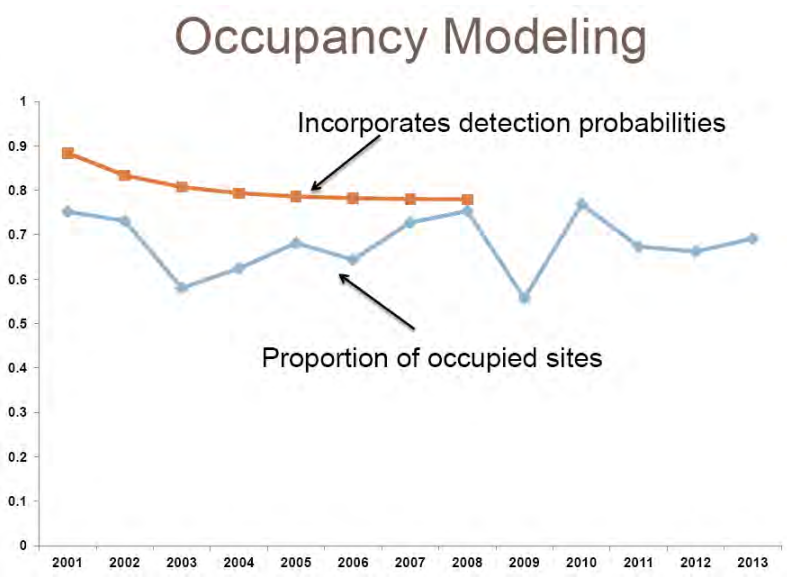


Figure 1.6: Percent of surveyed northern spotted owl sites occupied on Michigan-California Timber Company lands during 2000-2013 (from Michigan-California Timber Company 2014).

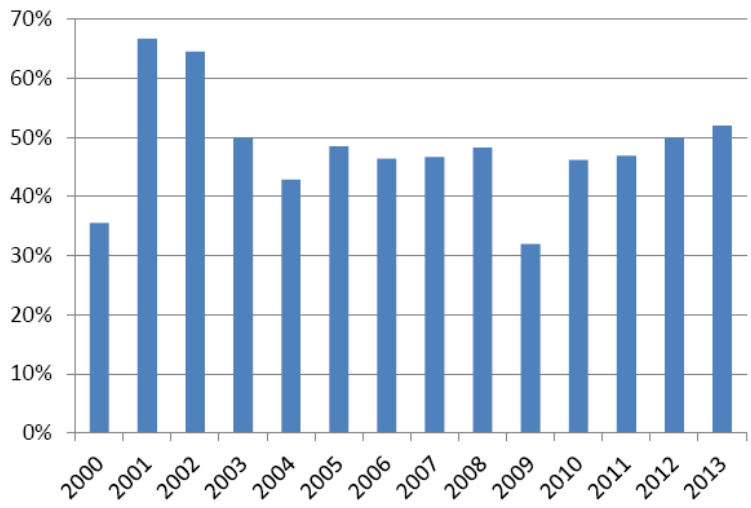
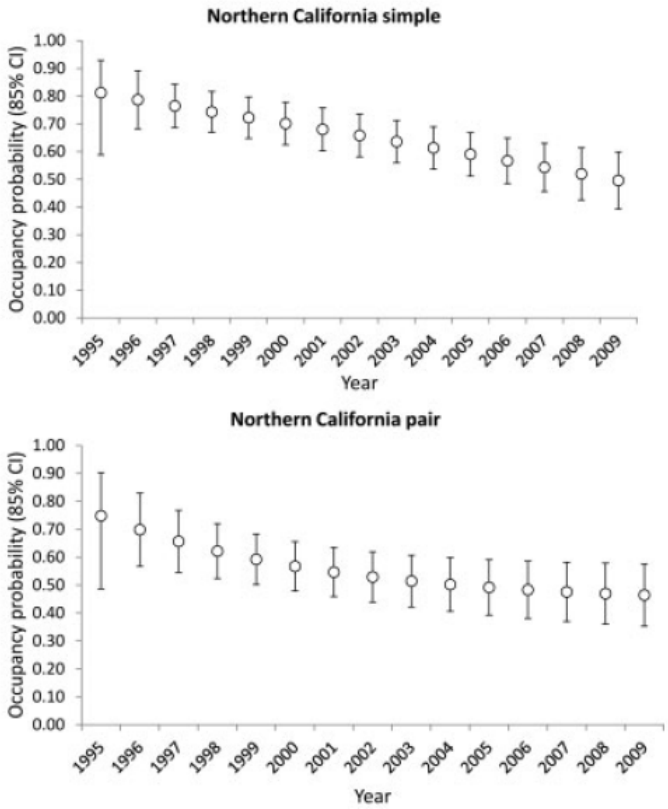


Figure 1.7: Estimated annual simple (total) and pair occupancy probabilities (with 85% confidence intervals) for northern spotted owls in the Eastern Klamath Region of California during 1995-2009 (from Farber and Kroll 2012).



Demography

Reproduction

Reproductive data are commonly collected as part of monitoring efforts for NSOs (e.g., Calforests 2014). They are easier and more cost-effective to obtain than those required for estimating survival or population trends. NSOs exhibit considerable annual fluctuations in reproduction (Forsman et al. 2011, Calforests 2014). Given often large annual fluctuations in reproduction, evaluation of trends in reproduction could require longer-term datasets than are available for many monitoring areas.

The 2011 demographic meta-analysis reported that fecundity of NSOs (number of female fledglings per female) significantly declined during 1985-2008 in four of 11 density study areas, may have declined in three other areas, and was stable in four areas (Forsman et al. 2011). Two of the four study areas with significant declines in fecundity were located in California (Northwestern California in the Western Klamath Region and Green Diamond in the Redwood Region). Two others were located in portions of southwestern Oregon (Klamath and South Cascades) that are nearby and ecologically similar to the Eastern Klamath and Southern Cascades of California (see USFWS 2011a Appendix C, USFWS 2012a, and Figure 1.22 [left side] for regions). Also, the one area in California with stable fecundity (Hoopa) had low fecundity estimates compared to other areas. Together these data, which represent the most

reliable evidence currently available, indicate that NSO reproduction could be declining across much of California and southwestern Oregon.

Annual fluctuations in fecundity were evident in all three demographic studies in California and were remarkably synchronous (Forsman et al. 2011; Figure 1.8). Forsman et al. (2011) found that variation in fecundity was associated with a variety of variables, including the age of breeding females, whether the year was even or odd, weather or climate (e.g., early nesting season temperature or precipitation), percent cover of suitable habitat, and the presence of barred owls. Franklin et al. (2013) noted a pattern of "good" and "bad" reproductive years in the Northwestern California demographic study area, which is likely associated with annual variation in weather during the early nesting season (also see Franklin et al. 2000). Franklin et al. (2013) also observed that particularly poor reproductive years have occurred in their study area at four-year intervals, suggesting that "some other extrinsic factor may be operating, such as seed production governing small mammal populations." Forsman et al. (2011) reported that barred owl presence was in the top models explaining fecundity in the Green Diamond study area, suggesting that competition with barred owls contributed to declining reproduction on that ownership. Reports from the Klamath and South Cascades demographic studies in southern Oregon noted negative associations between reproduction and rainfall during the early nesting season (Davis et al. 2013b, Dugger et al. 2014). Declining reproduction in these study areas also appears to be related to increasing presence of barred owls.

Following publication of the 2011 meta-analysis, California's demographic studies reported three consecutive years (2011-2013) of very low reproduction (Franklin et al. 2013, Higley and Mendia 2013, Green Diamond Resource Company 2014). This dip in reproduction might have been partially driven by high rainfall during the early nesting season during 2010-2012 (see below). Those three consecutive years of low reproduction exacerbated the negative long-term trend that was already occurring on Green Diamond lands (Green Diamond Resource Company 2014; Figure 1.9). Negative trends in reproduction also occurred in the Klamath and South Cascades demographic study areas subsequent to the end of the 2011 meta-analysis study period (Davis et al. 2013b, Dugger et al. 2014; Figures 1.10 and 1.11). Davis et al. (2013b) concluded that particularly poor reproduction during recent years "…may indicate potentially serious problems with maintaining a stable population. This is even more alarming since these results are following a long term downward trend."

Figure 1.8: Mean annual fecundity in California's three northern spotted owl demographic studies during 1985-2008 (from Forsman et al. 2011).

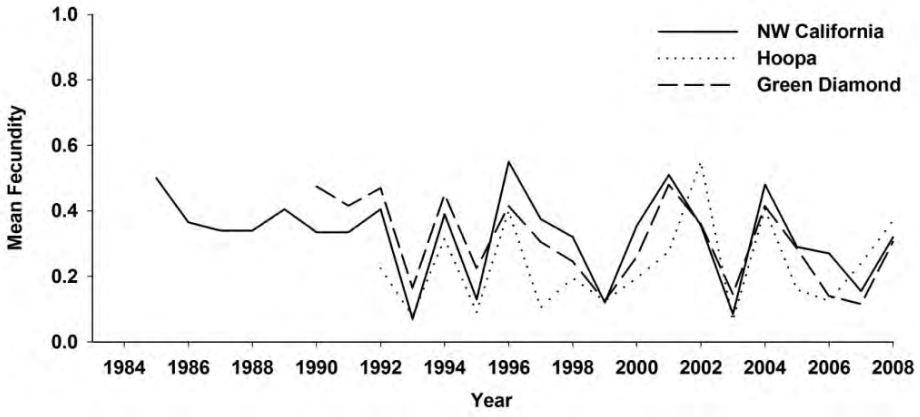


Figure 1.9: Number of fledglings produced per monitored pair of northern spotted owls on Green Diamond Resource Company lands during 1992-2013 (from Green Diamond Resource Company 2014).

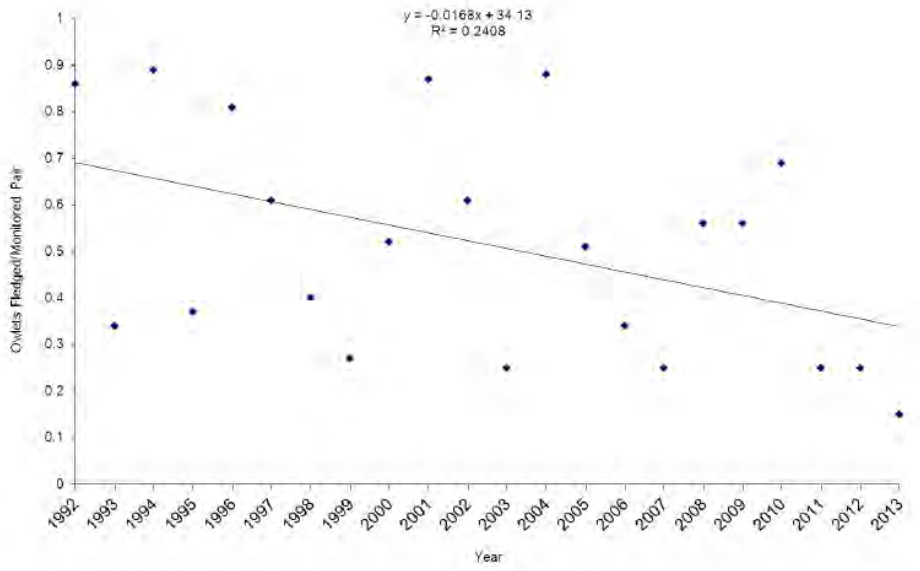


Figure 1.10: Fecundity of northern spotted owls ("STOC") in the Klamath demographic study area during 1990-2013 (from Davis et al. 2013a). Dashed line is a polynomial trend line ($r^2 = 0.419$). Vertical line represents the first year in which barred owls ("STVA") were detected in more than 10% of spotted owl territories.

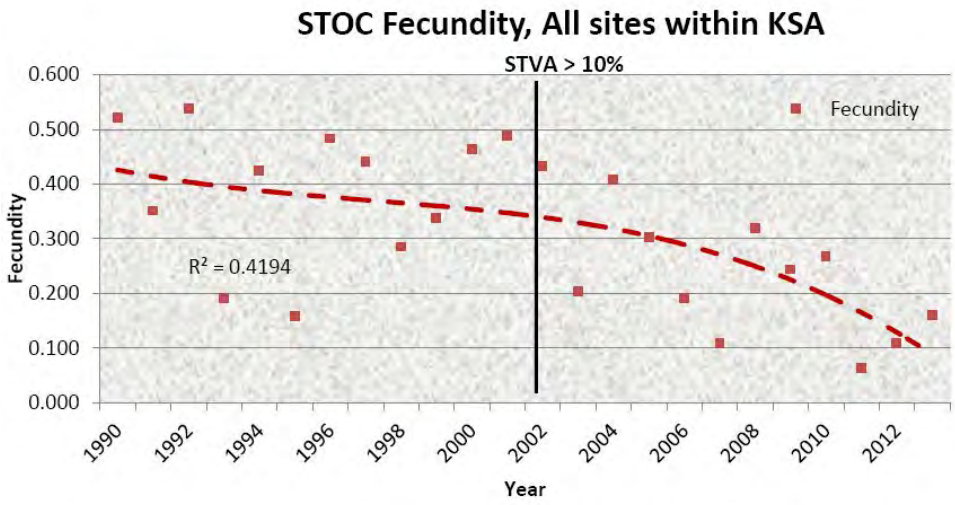
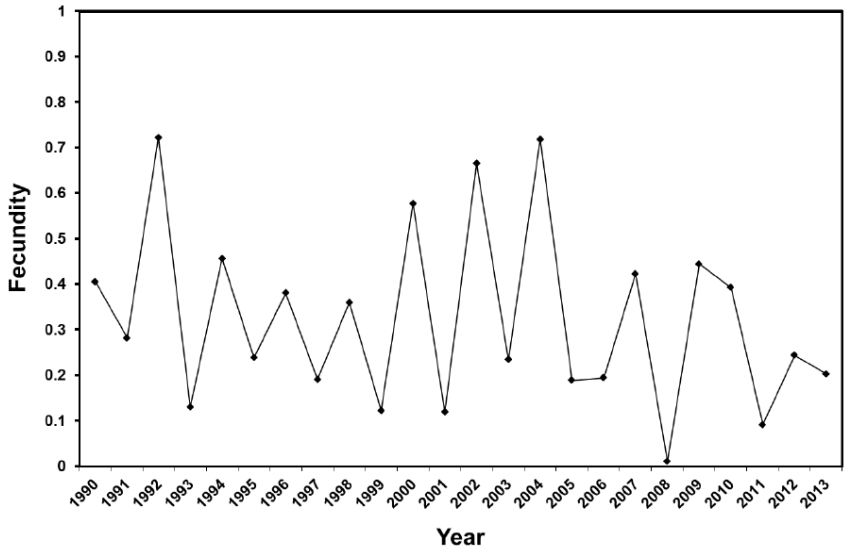


Figure 1.11: Fecundity of northern spotted owls in the South Cascades demographic study area during 1990-2013 (from Dugger et al. 2014).



Information is also available for describing recent trends in NSO reproduction in portions of California outside of demographic study areas. Ellis et al. (2013) found below average fecundity during 2007 and 2010-2012 on National Park Service lands in the southern Redwood Region (Figure 1.12). Humboldt Redwood Company (2013) and Mendocino Redwood Company (2014) likewise reported low reproduction during those years (Figures 1.13 and 1.14). These observations, along with those from demographic studies in California and southern Oregon described above, suggest that low reproduction during recent years was primarily driven by a factor that acted at a very large spatial scale, rather than at the scale of individual ownerships or ecological regions. As noted above, high rainfall during the early

nesting season was likely a primary cause of low reproduction during recent years. This apparent relationship is illustrated by the negative association between NSO reproductive success and early season rainfall observed on Mendocino Redwood Company lands (Figure 1.13).

Monitoring results suggest a stable long-term trend in reproduction on National Park Service lands in the southern Redwood Region (Figure 1.12). In the northern Redwood Region, Humboldt Redwood Company (2013) data likewise suggest little or no trend, although the period covered could be too short to capture long-term trends in reproduction (Figure 1.14). Mendocino Redwood Company (2014) provided a longer-term data set that suggests that a shallow decline in reproduction has occurred on their lands, primarily due to below average reproduction during seven of eight years during 2006-2013 (Figure 1.13). Data provided by the Fruit Grower's Supply Company (2014) suggests that a decline in reproduction occurred on their lands in the Eastern Klamath and Southern Cascades regions of California during 1990-2005 (Figure 1.15). It is important to note, however, that these are only descriptions of apparent trends based on patterns in relatively crude data. A more rigorous analysis of the data is needed to support strong conclusions about reproductive trends on industrial timberlands in California.

Figure 1.12: Fecundity of northern spotted owls on National Park Service lands in Marin County during 1999-2005 and 2007-2012 (from Ellis et al. 2013). The solid line indicates mean fecundity during these periods combined, the dashed lines are one standard deviation from the mean, error bars indicate ±1 standard error, and n is the total number of spotted owl territories.

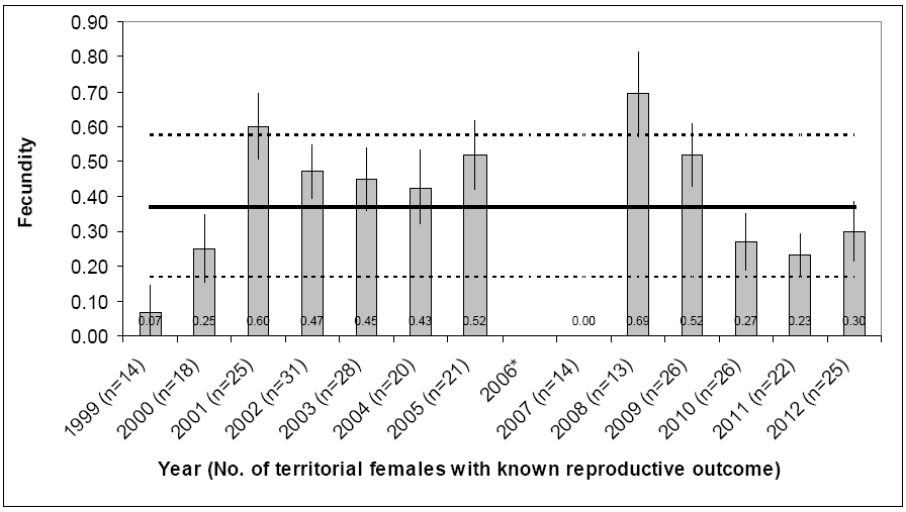


Figure 1.13: Reproductive success (average number of fledglings/pair) of northern spotted owls and amounts of rainfall during the early nesting season on Mendocino Redwood Company lands during 1989-2013 (from Mendocino Redwood Company 2014).

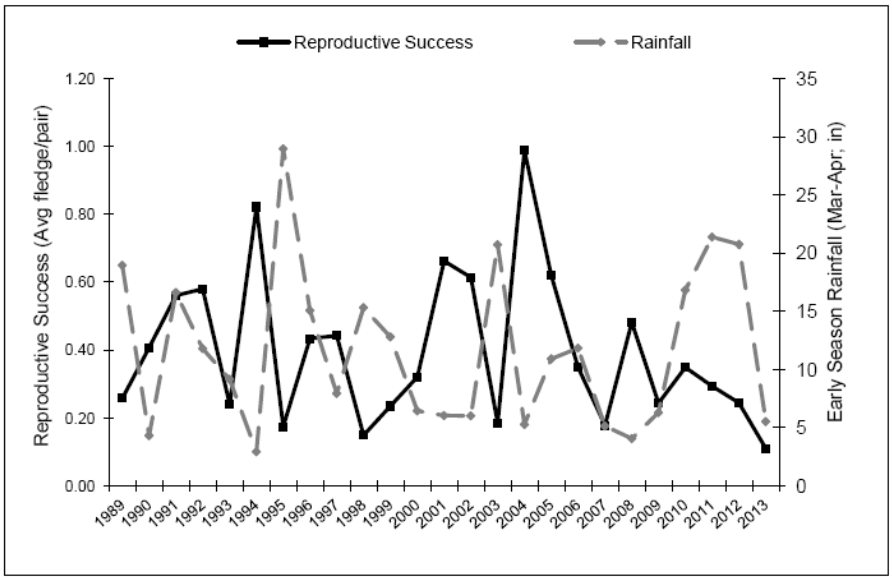


Figure 1.14: Reproductive rate and numbers of nesting pairs and juveniles on Humboldt Redwood Company lands during 2003-2012 (from Humboldt Redwood Company 2013).

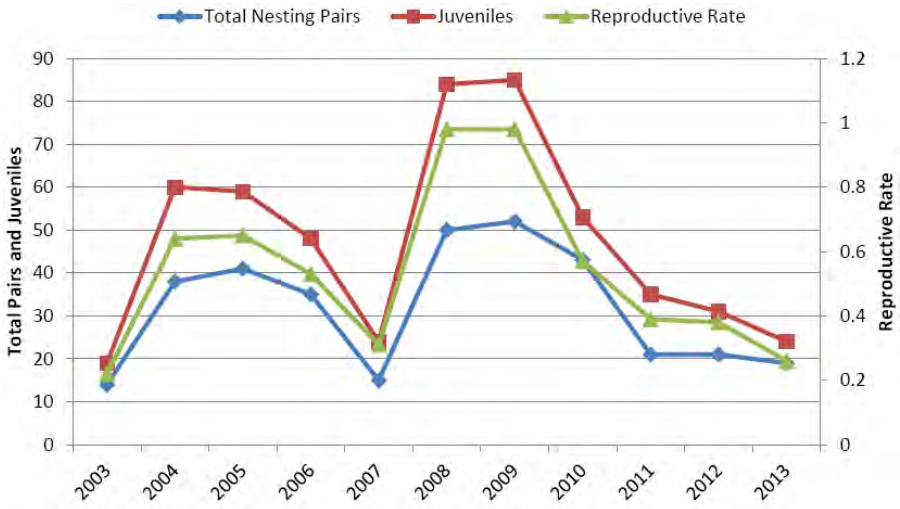
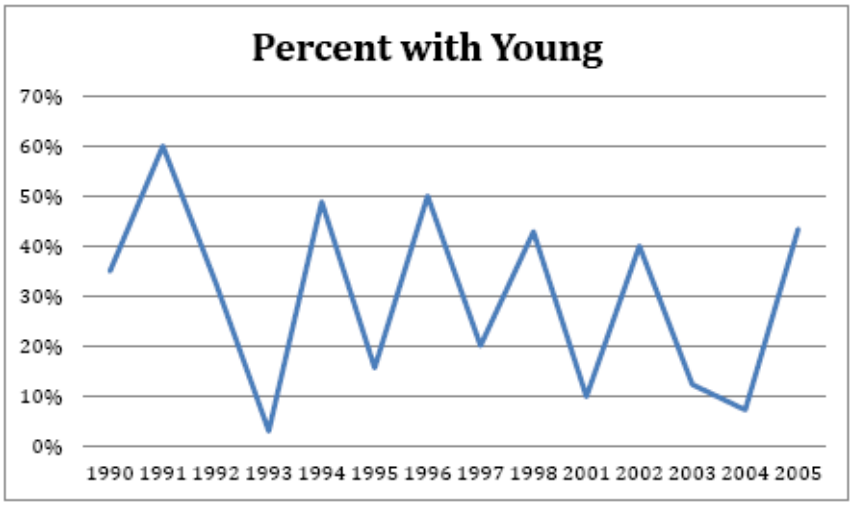


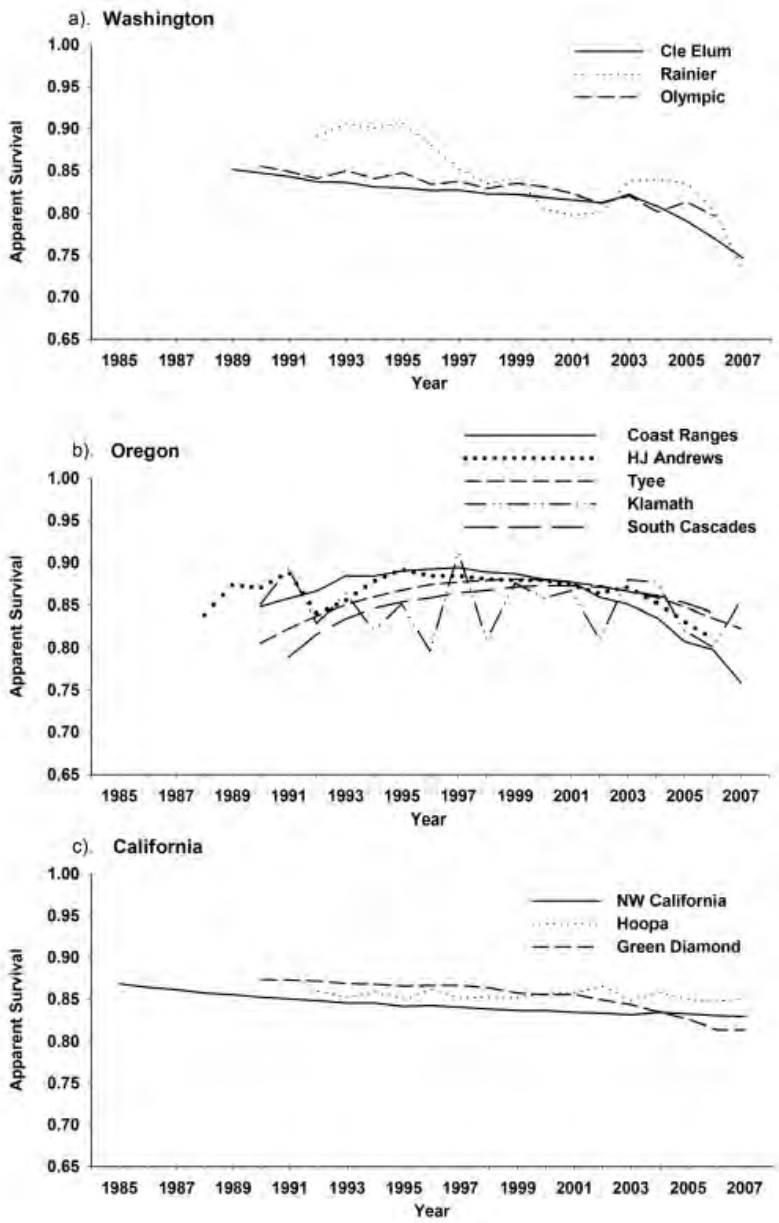
Figure 1.15: Fecundity of northern spotted owls on Fruit Growers Supply Company land during 1990-2005 (from Fruit Growers Supply Company 2014).



Survival

Available information concerning recent survival rates of NSOs is mostly limited to that provided in the 2011 demographic meta-analysis (Forsman et al. 2011). Survival data are not collected by timber companies other than Green Diamond Resource Company, which submits its data for analysis and reporting in the demographic meta-analyses. Forsman et al. (2011) reported statistically significant declines in apparent survival for 10 of 11 NSO demographic study areas, including all three study areas in California (Figure 1.16). Declines in many study areas were most precipitous during the last five years of the study period (i.e., 2003-2007 for survival; Figure 1.16). The Klamath in southern Oregon was the only study area that did not have a significantly declining survival rate through 2007. Forsman et al. (2011) stated that "collectively, the declines in apparent survival of Northern Spotted Owls across much of the subspecies' range are cause for concern because Spotted Owl populations are most sensitive to changes in adult survival rates (Noon and Biles 1990, Lande 1991)."

Figure 1.16: Model averaged estimates of apparent survival of adult female northern spotted owls three study areas in Washington (a), five study areas in Oregon (b), and three study areas in California (c) during 1985-2007 (from Forsman et al. 2011).



NSO demographic studies have largely deferred reporting of more recent survival data to the forthcoming meta-analysis, which is expected to be released in 2015. The limited information available prior to release of that meta-analysis suggests that survival has continued to decline since the 2011 meta-analysis study period. Davis et al. (2013b) reported that subsequent "…data regarding occupancy (in the Klamath study area) has shown a rapid decline, which suggests the stability of the survival rate may no longer be valid." Franklin et al. (2013) reported an alarming drop in apparent survival in 2011 on the Northwestern California demographic study area (Figure 1.17). Their subsequent annual report deferred reporting of 2012-2013 survival data to the forthcoming meta-analysis (Franklin et al. 2014). Higley and Mendia (2013) reported a statistically non-significant decline in survival of NSOs on the Hoopa Valley Indian Reservation in the Western Klamath (Figure 1.18). Their best model explaining survival of NSOs suggested that the decline was at least partially related to increasing numbers of barred owls in the study area.

Figure 1.17: Annual estimates (solid dots with 95% confidence intervals) of, and trend in (solid line), apparent survival for subadult and adult northern spotted owls in northwestern California during 1985-2012 (from Franklin et al. 2013).

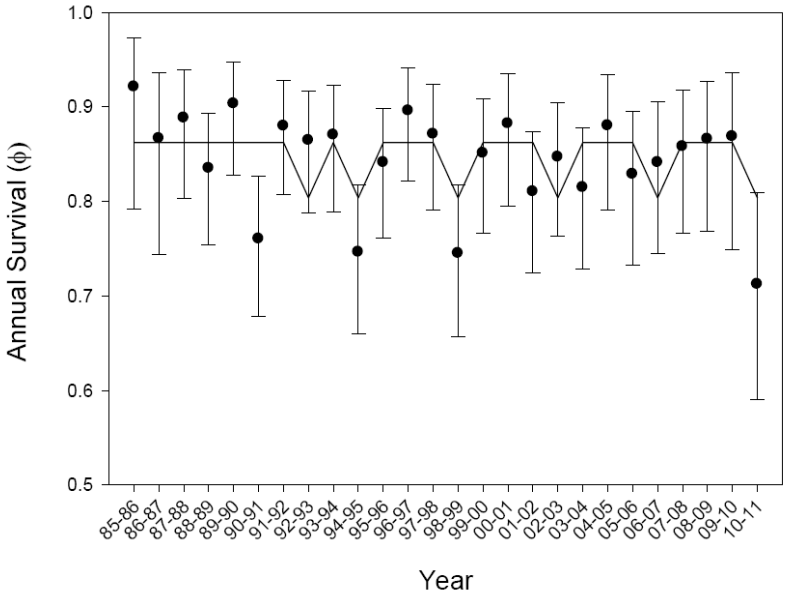
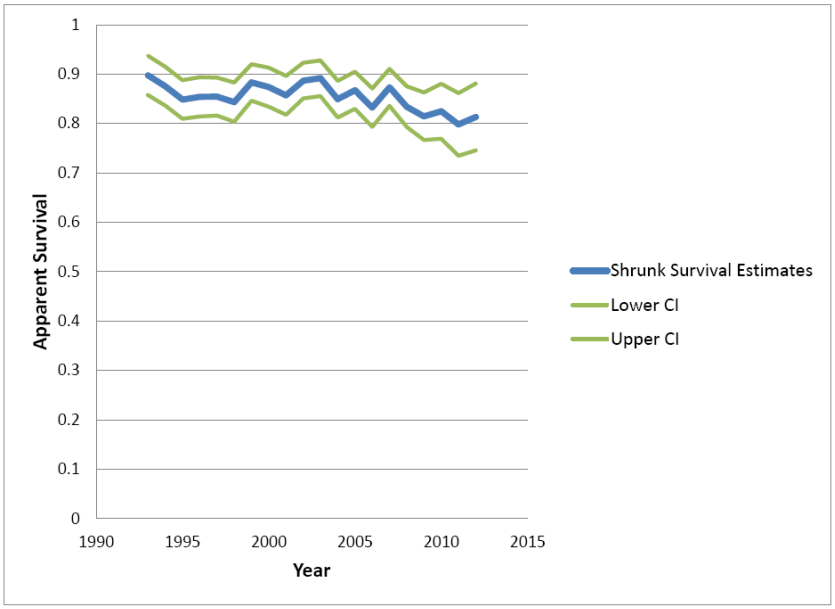


Figure 1.18: Estimates and 95% confidence intervals for NSO apparent survival on the Hoopa Valley Indian Reservation, Humboldt County, California during 1994-2012 (from Higley and Mendia 2013).



Forsman et al. (2011) reported that the presence of barred owls was included in the best model structures for several study areas, including the Green Diamond and Klamath, and was in a competitive model for Northwestern California (Forsman et al. 2011). Given evidence that barred owl presence continued to increase after the study period covered by Forsman et al. (2011) (see Ch. 4), it is likely that the forthcoming meta-analysis will report continued declines in apparent survival for many, if not all, demographic study areas. Franklin et al. (2013) noted that apparent survival in the Northwestern California study area, like reproduction, is influenced by annual variation in weather during the early spring. Thus, recent consecutive years with poor weather during the early spring further suggest that survival has likely continued to decline since the period analyzed by Forsman et al. (2011).

Population Change

A new demographic meta-analysis is expected to be released in 2015. Until then, the 2011 meta-analysis (Forsman et al. 2011) provides the most current available estimates of population change for NSOs across their range. Except for the Green Diamond Resource Company, which submits its data for analysis and reporting in periodic meta-analyses, timber companies do not estimate population change for NSOs. Forsman et al. (2011) stated that their results likely "…reflected conditions on federal lands and areas of mixed federal and private lands within the range of the Northern Spotted Owl because the study areas were (1) large, covering ≈ 9% of the range of the subspecies, (2) distributed across a broad geographic region and within most of the geographic provinces occupied by the owl, and (3) the percent cover of owl habitat was similar between our study areas and the surrounding landscapes." Only one of the study areas included in the meta-analysis was entirely located on private lands (Green Diamond). Thus, it is unclear whether results from the 2011 meta-analysis reflect demographic trends on private lands across the range of the NSO. Given weaker habitat conservation measures for NSOs on many private ownerships compared with federal lands, Forsman et al. (2011) stated that, "if anything, our results depict an optimistic view of the overall population status of the Northern Spotted Owl."

Forsman et al. (2011) reported estimates of the annual finite rate of population change (λ) for 11 study areas located across the NSO's range. Estimates of λ ranged from 0.929 to 0.996 (i.e., declines of 0.4 to 7.1% per year) for these study areas during the period of 1990-2006. There was strong evidence of population declines on seven of the study areas, including the Northwestern California (-1.7% per year) and Green Diamond (-2.8% per year) study areas in California. Negative population trends were also found on the Hoopa study area in California (-1.1% per year) and on the Klamath and South Cascades in southern Oregon (-1.0% and -1.8% per year, respectively) but they were not statistically significant. The weighted mean estimate of λ for all study areas combined was 0.971, indicating an average population decline of 2.9% per year during the study. Variables included in the best model in the meta-analysis of λ indicated effects of ecoregion (geographic location and major forest type) and the proportion of NSO territories with barred owl detections.

In addition to estimates of annual rate of population change, Forsman et al. (2011) provided estimates of realized population change, which describes population change over the study period (Figure 1.19). NSO populations in Washington and northern Oregon declined by approximately 40-60% during 1990-2006. Populations on the Northwestern California and Green Diamond study areas declined by 20-30% during the study period, although the 95% confidence intervals for these estimates slightly overlapped zero (Figure 1.19). Declines of 5-15% were evident on the Hoopa, Klamath and South Cascades study areas but these trends were not statistically significant (Figure 1.19).

Figure 1.19: Estimates of realized population change with 95% confidence intervals for northern spotted owls in California and southern Oregon (from Forsman et al. 2011).

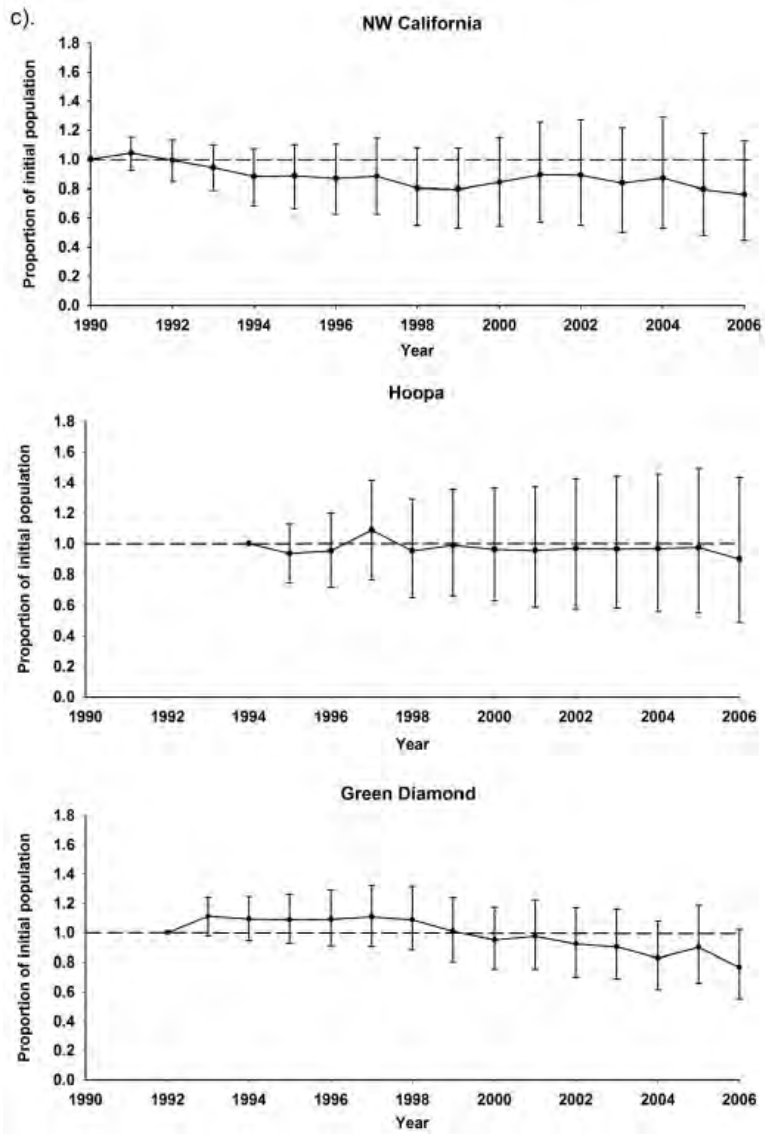
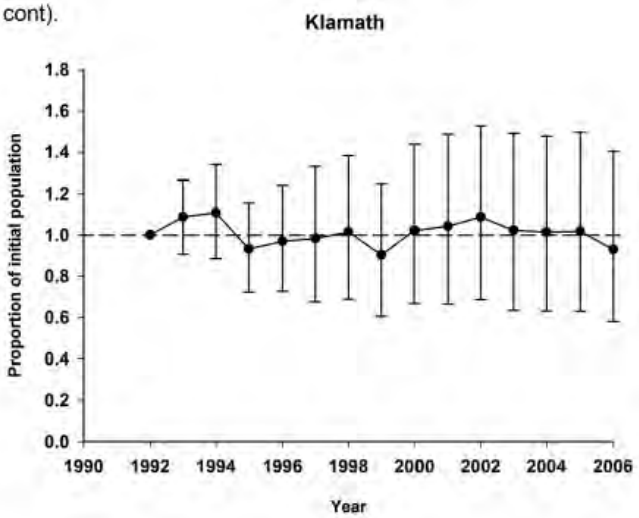
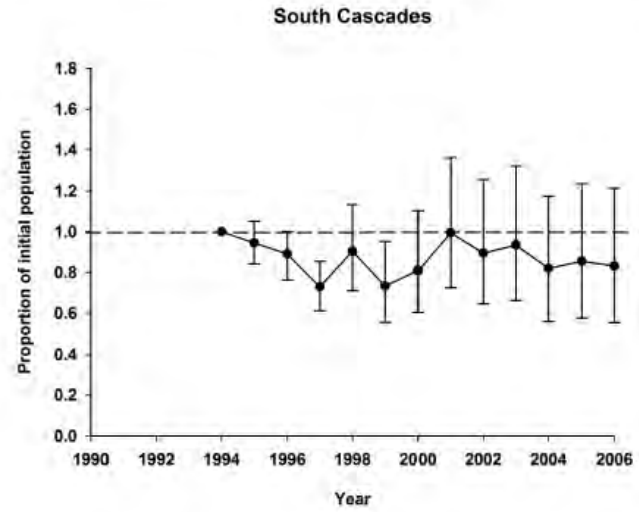


Figure 1.19 (cont.).

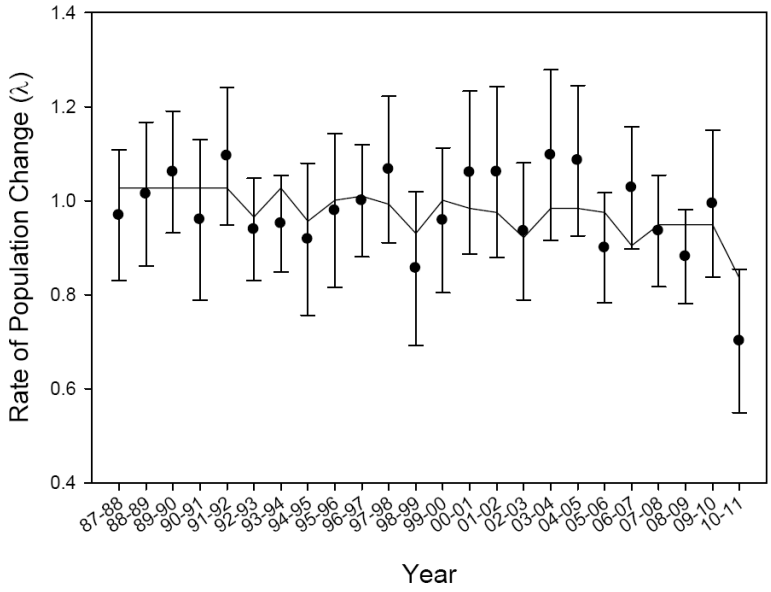
b cont).





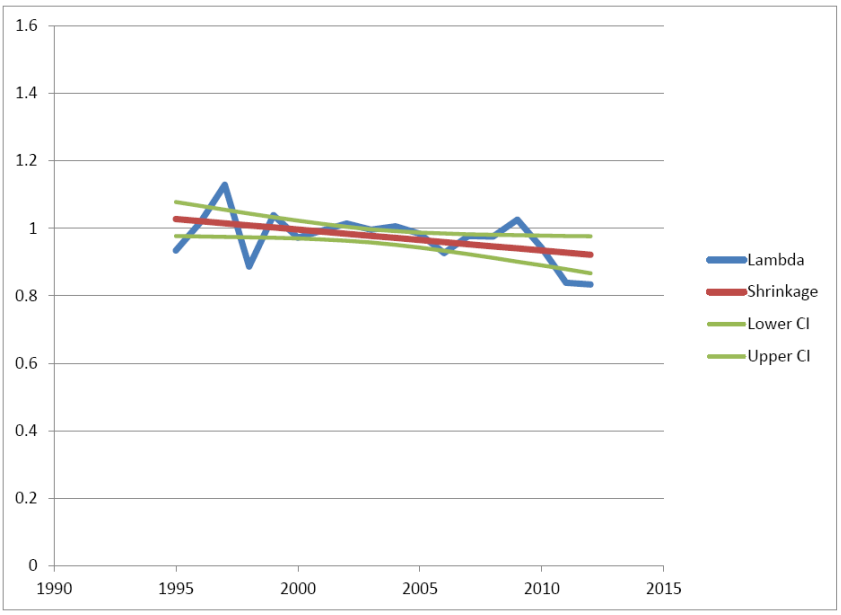
Following the 2011 meta-analysis study period (i.e., >2007), NSOs in the Northwestern California study area experienced a further decline in λ (mean = 0.978 or -2.2% per year) (Figure 1.20). The last year included in this analysis (2011) had the lowest annual estimate of λ found during the 24-year analysis period (Figure 1.20). The forthcoming meta-analysis should reveal whether the substantial drops in apparent survival and λ in the Northwestern California study area in 2011 were anomalous or indicative of an increased rate of population decline in the study area. Franklin et al. (2013) found that fecundity, apparent survival, and λ in the study area fluctuated during "good" and "bad" years, which was likely at least partially related to weather (see above). Annual rate of population change was also apparently negatively affected by increasing presence of barred owls. Given continued increases in barred owls (see Ch. 4), poor weather during the early spring during 2010-2012, and poor reproduction by NSOs during 2011-2013 (see above), it is likely that λ continued to decline on this study area and probably others in California and southern Oregon.

Figure 1.20: Annual estimates of (dots with 95% confidence intervals) and trend in (solid line) rate of population change in the Northwestern California study area (from Franklin et al. 2013).



Higley and Mendia (2013) reported that the estimate of λRJS (Jolly-Seber Capture-Recapture model) for the Hoopa demographic study during 1995-2012 was 0.977, indicating a mean annual population decline of 2.3%. The decline was statistically significant in 2011 and 2012 (point estimates of λRJS not included in the 95% CI; Figure 1.20). Higley and Mendia (2013) noted that "the recent decline in survival, the point estimate of λRJS and the actual number of birds detected this past season all point to a population that is in fact, declining. This apparent decline in spotted owls corresponds with an increase in total annual barred owl detections and proportion of spotted owl territories with barred owl detections." They further noted that the forthcoming meta-analysis will show that it is "...very clear that northern spotted owls are in decline across all 11 study areas and that in many cases the decline is accelerating."

Figure 1.21: Trend in rate of population change on the Hoopa Valley Indian Reservation, Humboldt County, California during 1994-2012 (from Higley and Mendia 2013).



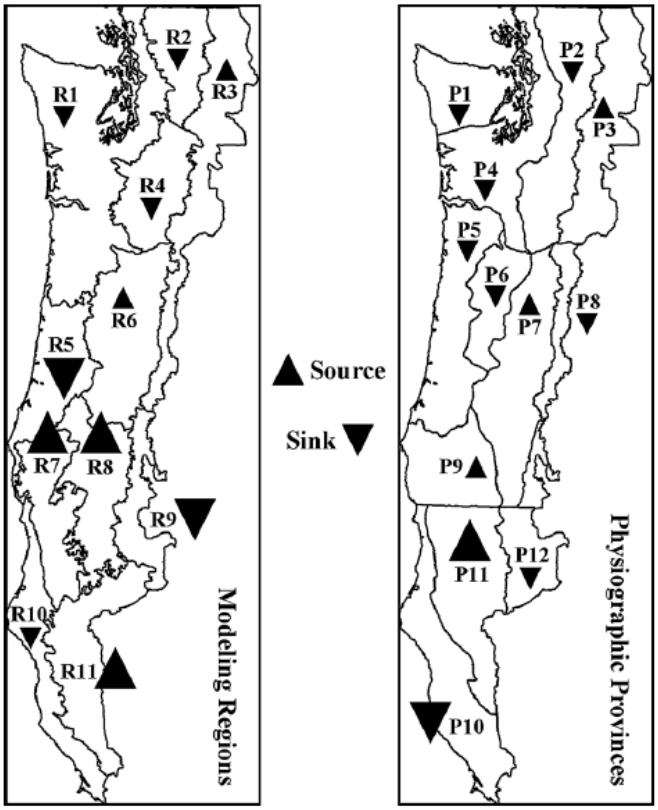
Source-Sink Dynamics

As described by Gutiérrez and Harrison (1996), source-sink dynamics exist for species "…that occupy both high-quality habitats (sources) where populations grow and produce emigrants, and low-quality habitats (sinks) where populations cannot sustain themselves in the absence of immigration." Population sinks potentially function as reservoirs for repopulation of sources that go extinct but may also reduce population growth rates (Pulliam 1988, Gutiérrez and Harrison 1996). Identifying source and sink areas is therefore, an important component of conservation research and planning. For example, identification of population sinks might be useful for determining where to focus habitat restoration or barred owl removal efforts. Empirical studies of relationships between NSO fitness and habitat attributes (Habitat Fitness Potential *sensu* Franklin et al. 2000) provide a rigorous measure of sources and sinks but only at the territory scale and within a given study area, rather than at population or regional levels (see Ch. 2 and 3 for further discussion of Habitat Fitness Potential). In the absence of direct empirical measures of large-scale source-sink dynamics, it may be useful to evaluate the results of source-sink simulation modeling based on empirical data.

Schumaker et al. (2014) recently published a rangewide study of source-sink dynamics for NSOs at the spatial scales of ecological regions and physiographic provinces. Their source-sink simulation modeling incorporated an array of regional data for NSO movement distances and rates, life history attributes, habitat suitability and connectivity, encounter rates with barred owls, and environmental stochasticity. Source-sink dynamics in this study emerged from simulated interactions between individual NSOs and landscapes, rather than being predefined based on habitat suitability as was done in previous studies. The simulation models by Schumaker et al. (2014) predicted that most ecological regions and physiographic provinces currently function as population sinks for NSOs (Figures 1.22 and 1.23). The study's results projected that the Klamath Provinces of California and Oregon and the Interior Northern Coast Ranges of California are the subspecies' strongest population sources (Figure 1.23). The Klamath

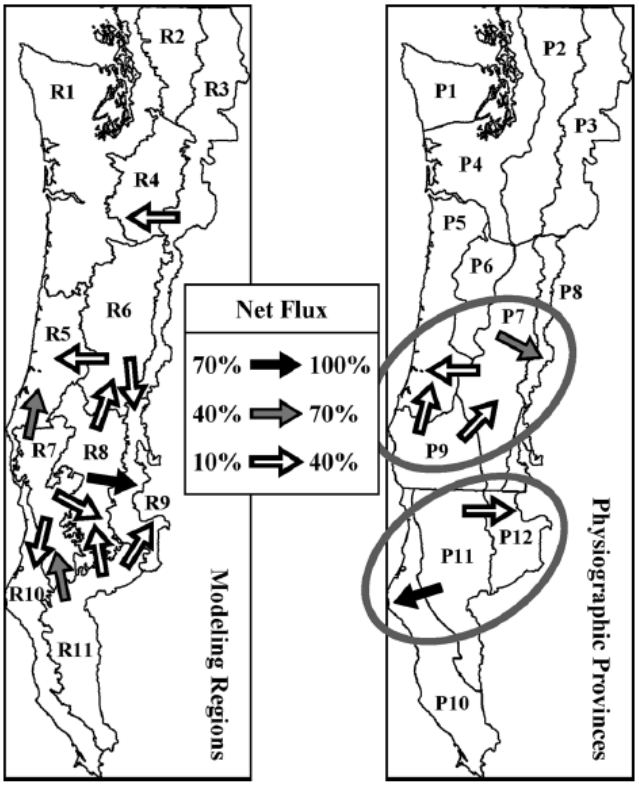
Provinces may be particularly important for maintaining NSO population stability due not only to being net population sources but to their high levels of population connectivity with multiple surrounding regions (Schumaker et al. 2014; Figure 1.23). The Redwood and Southern Cascades regions in California were both classified as moderate population sinks. Schumaker et al. (2014) identified the Klamath Provinces and California Cascades as areas in which it could be particularly important to focus habitat protection and restoration efforts, respectively.

Figure 1.22: Relative source and sink values in northern spotted owl modeling regions and physiographic provinces (from Schumaker et al. 2014). The sizes of symbols denote major versus minor or moderate sources and sinks.



R7: Klamath West, R8: Klamath East, R9: Eastern Cascades South, R10: Redwood Coast, R11: Inner California Coast Ranges, P10: California Coast Range, P11: California Klamath, P12: California Cascades. See Schumaker et al. (2014) for other modeling regions and physiographic provinces.

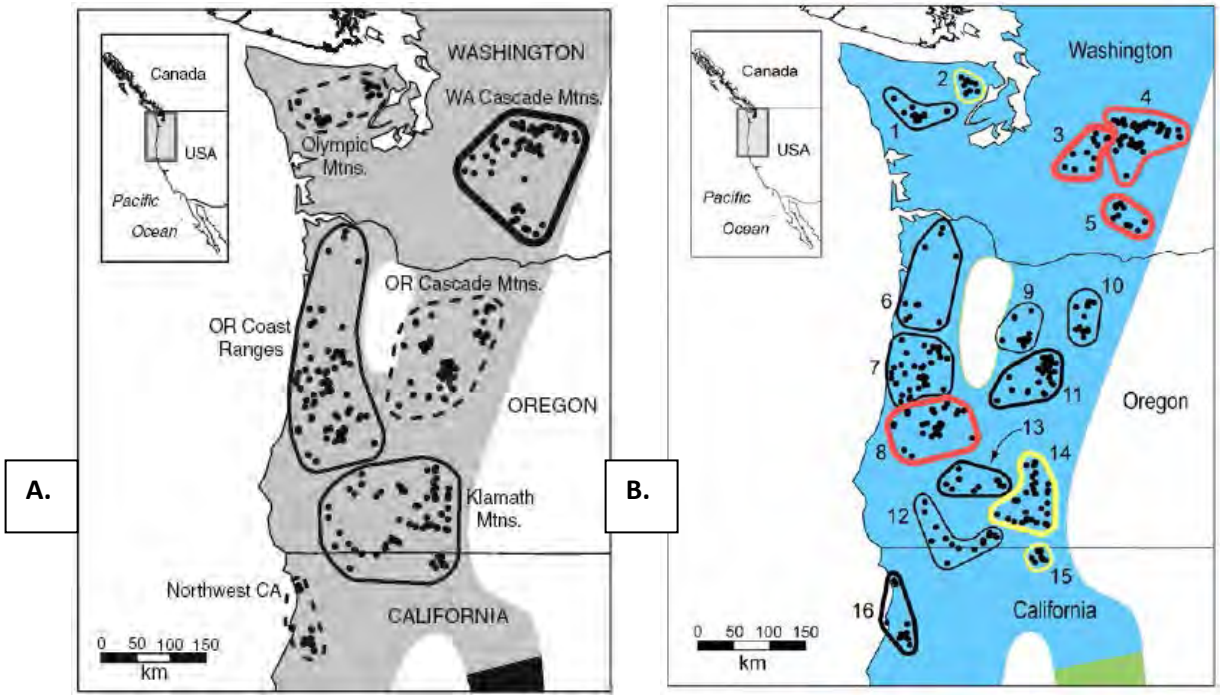
Figure 1.23: Graphical representation of net movement ("Net Flux") of individual (simulated) northern spotted owls from one modeling region or physiographic province to another (from Schumaker et al. 2014). The largest Net Flux values are shown in black, intermediate values in gray, and smallest values in white. Gray ovals highlight two areas with strong patterns of Net Flux.



Genetics

Funk et al. (2010) found statistically significant evidence that NSOs have experienced genetic bottlenecks during recent decades in the Washington Cascades, Oregon Coast Ranges, and "Klamath Mountains" of Oregon and California (Figure 1.24). An earlier report on this study indicates that evidence of a bottleneck in the Klamath Mountains analysis area was primarily driven by data from the southern Cascades of Oregon and California, rather than from the Klamath Provinces (Funk et al. 2008; Figure 1.24). Evidence of recent genetic bottlenecks in the Washington Cascades, Oregon Coast Ranges, and southern Cascades are concordant with recent demographic declines in these regions (Forsman et al. 2011; see *Demography*, above). Surprisingly, Funk et al. (2010) did not find evidence of a genetic bottleneck in the Olympic Mountains of Washington, where NSOs have recently experienced dramatic population declines (Forsman et al. 2011). However, they noted that their small sample size for this region limited their power to detect a genetic bottleneck if one occurred. Funk et al. (2010) did not find statistically significant evidence of a recent genetic bottleneck in northwestern California (Western Klamath and Redwood regions). They suggested that this could likewise have been due to low statistical power or to the relatively gradual population declines reported for that area at the time (see *Demography*, above). The analyses of Funk et al. (2010) did not address whether genetic bottlenecks were solely a result of population declines or were also contributing to them. Genetic declines can contribute to reduced demographic rates through effects of inbreeding depression and loss of adaptive genetic variation (reviewed in Funk et al. 2010).

Figure 1.24: Recent population bottlenecks in NSOs. Points represent 352 individual owls included in the analysis which are grouped into six (A) and 16 (B) regions. Statistically significant bottlenecks are represented by solid lines (A) or yellow (p = 0.05) and red (p = 0.01) lines (B). (A) represents significant bottlenecks under 5, 10, and 15% multi-step mutation models as solid bold lines and under 10 and 15% multi-step mutation models as finer solid lines (see Funk et al. 2010). (B) indicates greater magnitude bottlenecks with bolder lines. From Funk et al. 2010 (A) and 2008 (B).



Summary of Current Status and Trends

Rigorous long-term research has indicated that NSO populations are dramatically declining in Washington and northern Oregon and more gradually declining in southern Oregon and California (Forsman et al. 2011). Yet, while less precipitous than those in the northern portion of the NSO's range, the rapidity of population declines in southern Oregon and California are cause for grave concern regarding the subspecies' status and trends. A new demographic meta-analysis, which is due for public release during 2015, will replace the 2011 meta-analysis and provide the most reliable information for evaluating the NSO's current status and trends. Based on information available in annual research reports, it is clear that the forthcoming meta-analysis will show that populations in southern Oregon and California are declining more rapidly than was evident in the 2011 meta-analysis (Davis et al. 2013a, Franklin et al. 2013, Higley and Mendia 2013, Dugger et al. 2014, Green Diamond Resource Company 2014).

The NSO's status and trends are less clear in portions of California outside the state's three demographic study areas. Much of the information for these areas is provided by industrial timber companies, which have uniformly concluded that NSO populations on their lands are stable (Calforests 2014). However, the data provided by these companies are insufficient for drawing strong conclusions about the NSO's status and trends, and may in fact indicate declines in occupancy and reproduction on some ownerships (see *Occupancy* and *Demography*, above). Forsman et al. (2011) suggested that, due to weaker habitat

protection, NSO demographic trends could generally be worse on non-federal lands than on the federal and mixed federal/non-federal lands on which most demographic studies are conducted. This appears to be true in California, where NSOs experienced greater declines on Green Diamond Resource Company lands than on nearby tribal and Forest Service lands (Forsman et al. 2011; see *Occupancy* and *Demography*, above). However, the degree to which these differences were due to variation in land management, effects of competition with barred owls, or other factors is unclear.

It is likewise unclear if demographic trends in California's three demographic study areas accurately represent those in drier, less productive forests in the state's interior. An occupancy study in California's Eastern Klamath and Southern Cascades (Farber and Kroll 2012) and demographic studies in ecologically similar areas of southern Oregon (Davis et al. 2013a, Dugger et al. 2014) could provide the most reliable information currently available for evaluating NSO's status and trends in interior California (see *Occupancy* and *Demography*, above). These studies indicate that NSOs are currently declining in at least some portions of the Eastern Klamath and Southern Cascades regions (note: these regions cover portions of both California and southern Oregon as they are ecologically rather than politically defined; see USFWS 2011a Appendix C and Figure 1.22 [left side]). Evidence of population declines in the Klamath regions (Forsman et al. 2011, Farber and Kroll 2012, Davis et al. 2013a, and Franklin et al. 2013) are particularly concerning in light of the critical contributions these areas may provide to the NSO's long-term persistence (Schumaker et al. 2014; see *Source-Sink Dynamics*, above).

Although the Redwood Region is projected to currently function as a population sink, it still retains high densities and abundances of NSOs and is therefore important to the subspecies' conservation (Schumaker et al. 2014; see *Density and Abundance* and *Source-Sink Dynamics*, above). There is limited information available for evaluating the NSO's status and trends in portions of the Redwood Region outside of Green Diamond's lands. Monitoring on National Park Service lands and adjacent ownerships suggest that the population in Marin County is stable while NSOs in the Redwood National and State Parks have substantially declined. These differences appear to be largely due to negative effects of high barred owl densities in the Redwood National and State Parks and the relatively slow rate of the barred owl invasion in Marin County (see *Occupancy*, above). In contrast with the Green Diamond Resource Company, other timber companies in the northern portion of the Redwood Region have concluded that their NSO populations are stable (Calforests 2014). It is possible that NSOs have indeed fared better on these ownerships than on Green Diamond lands; for example, due to less intensive timber harvesting or more recent colonization by barred owls. However, the data provided by these companies are insufficient for drawing firm conclusions about the NSO's status and trends on these lands, and actually appear to indicate at least gradual declines in some areas. More consistent and rigorous monitoring (e.g., consistent survey areas and protocols; reporting of modeled occupancy rates) would assist future evaluations of the NSO's status and trends on industrial timberlands in California.

Acknowledgements

I am extremely grateful to Lowell Diller for his helpful review of this chapter and to Jeff Dunk for his comments on the *Source-Sink Dynamics* section.

Part II: Threats to Northern Spotted Owls in California

Forsman et al. (2011) concluded that habitat loss was partly responsible for declines in NSO fecundity, apparent survival, and/or populations observed in most demographic study areas. Due to a lack of a suitable habitat map at the time, they did not include a habitat variable in models for California. However, a substantial body of research has shown that stand- and landscape-level habitat attributes influence habitat selection, densities, occupancy, reproduction, survival, and metapopulation dynamics of NSOs in California and southern Oregon (e.g., Carey et al. 1992, Gutiérrez et al. 1998, Hershey et al. 1998, Thome et al. 1999, Franklin et al. 2000, Zabel et al. 2003, Carroll and Johnson 2008, Schumaker et al. 2014). Loss of approximately 60-88% of all old forest within the NSOs range during the 19th and 20th centuries was a primary reason for the subspecies' federal listing (USFWS 1990, Strittholt et al. 2006). Despite greater habitat protection following federal listing and implementation of the Northwest Forest Plan, intensive timber harvesting and large wildfires have continued to cause a downward trend in suitable habitat for NSOs and thus, continue to threaten the subspecies (Davis and Dugger 2011, USFW 2011). Yet, NSOs in California and southern Oregon may have complex relationships with these disturbances. For example, low-to-moderate or mixed severity wildfires could sometimes benefit NSOs in these areas by contributing to prey diversity and abundance, provided they do not excessively remove nesting and roosting habitat. In-depth reviews of these topics are provided in Chapters 2 and 3 of this volume.

Demographic analyses indicate that worsening NSO population declines in California and southern Oregon have been driven to a large degree by increasing competitive pressure from invasive barred owls (Forsman et al. 2011, Davis et al. 2013a, Franklin et al. 2013, Higley and Mendia 2013, Dugger et al. 2014, Green Diamond Resource Company 2014). A large body of quantitative and anecdotal information indicates that barred owls negatively affect NSOs in a variety of ways and that they currently pose one of the primary threats to the NSO's long-term persistence (USFWS 2013). These topics, with particular emphasis on information from California, are reviewed in Chapter 4.

Outdoor marijuana cultivation has dramatically increased in recent years and has emerged as a serious potential threat to NSOs in California (Gabriel et al. 2013, Calforests 2014). There is little quantitative information concerning impacts of outdoor marijuana cultivation on NSOs. However, published research of fishers (*Pekania pennanti*), which have overlapping home ranges, habitat associations, and diets with NSOs, suggests that anti-coagulant rodenticides and other pesticides used in outdoor marijuana cultivation currently pose a widespread risk to NSOs in California (Gabriel et al. 2012, 2013, Thompson et al. 2014). In addition to potential behavioral and demographic effects of pesticides on NSOs, outdoor marijuana cultivation could impact the subspecies through suppression of prey populations; ecological changes due to water diversion, clearcutting, and pollution; or habitat loss to wildfires ignited by growers (Gabriel et al. 2013, Bauer et al. 2015). Marijuana cultivation could also impact conservation of NSOs by reducing the ability of biologists to safely and efficiently conduct conservation research and monitoring (Gabriel et al. 2013). These topics are further discussed in Chapter 5.

The apparent effects of weather and climate variables on NSO demographic rates suggest that anthropogenic climate change could pose a major threat to the subspecies (Glenn et al. 2010). This hypothesis is further supported by projections of increased numbers of large wildfires in California under plausible climate change scenarios (see Ch. 3). Climate change could also impact NSOs in California through other climate or weather effects (e.g., increased frequency of droughts), outbreaks of insects and pathogens, large-scale redistribution of major vegetation types, and unpredictable effects on

prey communities (reviewed in USFWS 2012b). Due to limited time and funding, and the complex and ever-increasing body of science covering these topics, a synthesis of this information is not included in this document. State and federal agencies should thoroughly evaluate climate change as a potential threat to NSOs and other species prior to determining their conservation status.

Although not reviewed herein, the stressors described above and in the remainder of this document could have cumulative and interactive impacts on NSOs. For example, Dugger et al. (2011) found that barred owls and habitat fragmentation had an additive negative effect on NSO occupancy rates in southern Oregon. This finding suggests that habitat loss and fragmentation due to timber harvesting or severe wildfires can increase competitive pressure from barred owls. Decreasing population sizes, apparently due primarily to habitat loss and competition with barred owls, can increase risks posed to NSOs by other factors. For example, small NSO populations may become vulnerable to extinction due to chance events such as epidemics or extreme weather or climate events (Franklin et al. 2000). Decreasing population sizes may also have negative genetic effects on NSOs. For example, genetic bottlenecks could further reduce demographic rates through inbreeding depression and loss of adaptive variation (Funk et al. 2010). Also, hybridization between NSOs and barred owls could become more frequent in the future as NSOs become less able to find conspecific mates (Gutiérrez et al. 2007). Policymakers and land managers should acknowledge that, despite limited research of the topic, multiple past and current stressors for NSOs could have important cumulative and interactive impacts on the subspecies.

Ch. 2: Timber Harvesting

Introduction

Timber harvesting was a primary impetus for federal listing of the NSO and is still regarded as one of the major threats to the subspecies (Thomas et al. 1990, USFWS 1990, Courtney et al. 2004, USFWS 2011a). Timber harvesting can directly impact NSO populations by removing, degrading, or fragmenting habitat for them or their prey (reviewed below). Harvesting might also indirectly affect NSOs by increasing effects of other stressors, such as competitive pressure from barred owls (*Strix varia*) (Forsman et al. 2011, USFWS 2011a; see Ch. 4). However, timber harvesting likely has complex effects on NSOs in the southern part of their range due to divergent effects of habitat conditions on survival versus reproduction (Franklin et al. 2000, Olson et al. 2004, Diller et al. 2010). The information reviewed herein suggests that some forms and amounts of harvesting may be sustainable in northern California but that large-scale removal or fragmentation of habitat around activity centers can have strong negative impacts on NSOs (reviewed below and in USFWS 2009).

The kinds of habitat concentrations associated with high survival and fitness of NSOs may be limited in some parts of the subspecies' range due to removal of the majority of old forest during the 19th and 20th centuries (USFWS 1990, Strittholt et al. 2006). Harvesting has been substantially curtailed on federal lands since implementation of the Northwest Forest Plan (Healey et al. 2008, Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012). However, removal of suitable NSO habitat continues on federal lands and is occurring at higher rates on non-federal lands (Davis and Dugger 2011). On non-federal lands, habitat loss to logging is only partially offset by recruitment of new habitat (Davis and Dugger 2011, Moeur et al. 2011; reviewed below). This is cause for concern since non-federal lands contain a considerable portion of remaining suitable breeding habitat for the subspecies (e.g., >30% of older forest in the Pacific Northwest and northwestern California currently exists on non-federal lands: Moeur et al. 2011) and because recovery of the NSO could partially depend on voluntary conservation efforts on these lands (USFWS 2011a). The timber industry has cited relatively strict harvest regulations in California as evidence that listing of the NSO under the California Endangered Species Act is unnecessary (Calforests 2014). Yet, contemporary harvesting has still resulted in a net loss of suitable breeding habitat for NSOs on non-federal lands in California (reviewed below). Furthermore, the U.S. Fish and Wildlife Service (2009) recently concluded that California's regulations for avoiding "take" of NSOs inadequately protect the subspecies and do not reflect the best available science. Inconsistent or poor implementation of existing regulations could further weaken protections for NSOs on private timberlands in California (reviewed below).

Responses of NSOs to Timber Harvesting

<u>Interior of Northern California and Southern Oregon</u>

NSOs in interior northern California show a strong general preference for relatively old, structurally complex forest. This is illustrated by studies describing both small-scale plots around NSO locations (Solis and Gutiérrez 1990, Rissler 1995, White 1996, Hershey et al. 1998; but see Irwin et al. 2013) and landscape-scale analysis areas around activity centers (Chávez-León 1989, Solis and Gutiérrez 1990, Hunter et al. 1995, Gutiérrez et al. 1998). This body of research can be used to inform conservation measures for NSOs in interior northern California (e.g., for evaluating appropriate habitat definitions in take-avoidance guidelines: USFWS 2009). However, the following review is focused on studies of associations between landscape-scale habitat attributes and NSO demography in interior forests

(Franklin et al. 2000, Olson et al. 2004, Dugger et al. 2005, Schilling et al. 2013). These studies are based on rigorous demographic data and provide the best available insight into potential effects of timber harvesting on NSO populations (USFWS 2009). This review is supplemented with information from studies of associations between landscape-level habitat characteristics in southern interior forests and NSO home range sizes and probability of occurrence (Carey et al. 1992, Carey and Peeler 1995, Zabel et al. 2003, Carroll and Johnson 2008, Schilling et al. 2013).

In the California Klamath, Franklin et al. (2000) found that NSO survival was highest when estimated breeding core areas (390 ac) contained large amounts of both interior (>326 ft from edge) older forest (conifer or mixed forest with conifer QMD ≥21 in and canopy cover ≥70%) and edge with other vegetation classes. In contrast, reproduction was typically highest with lower amounts of interior older forest and greater amounts of edge. Estimated breeding core areas supporting high fitness for NSOs (a function of both survival and reproduction) contained both a large concentration of interior older forest and considerable habitat edge provided by a mosaic of other vegetation patches with convoluted shapes. Franklin et al. (2000) emphasized the difference between total area of older forest versus area of interior older forest. For example, they noted that large amounts of older forest edge cannot occur with low total amounts of older forest. This study did not directly address effects of timber harvesting on NSOs. Vegetation other than older forest was combined into a single class and edges occurred wherever that class and older forest met. Franklin et al. (2000) noted, however, that the dominant silvicultural system in their study area at that time was large-scale clearcutting, which they concluded was unlikely to contribute to the kinds of habitat mosaics found in territories supporting high fitness.

In an unpublished report, Matthews et al. (2008) evaluated the demography of NSOs on the Hoopa Valley Indian Reservation in the California Klamath. Their best performing model explaining NSO survival indicated that survival increased with greater amounts of interior mature or old forest (>80 yrs with "heavy" canopy cover, ≥328 ft from edge) up to about half of a 200-acre analysis area around activity centers and then slightly declined with higher proportions. Survival also increased with increasing amounts of brushy pole-timber forest (conifer stands 10-20 yrs with a "heavy brush component", meant to represent dusky-footed woodrat [*Neotoma fuscipes*] habitat) within estimated territories (917 ac) up to about 16% of the area and then leveled off. Survival was negatively associated with pre-commercial thinning (prescription not described) of brushy pole-timber forest, which Matthews et al. (2008) attributed to a negative long-term effect of thinning on dusky-footed woodrat populations. The best performing model explaining patterns of NSO reproduction indicated that the influence of woodrat habitat on reproduction depended on whether it was a high or low reproduction year and on amounts of mature and old forest. During years with high reproduction, productivity was highest at sites with moderate amounts (19%) of woodrat habitat in a larger core analysis area around activity centers (517 ac), whereas woodrat habitat had little influence on NSO reproduction during low reproduction years. Furthermore, high amounts of mature and old forest apparently offset negative effects of low amounts of woodrat habitat on reproduction; possibly by providing access to alternative prey (e.g., northern flying squirrels) or greater protection from predators or inclement weather (Matthews et al. 2008).

In the interior of the Oregon Coast Range, Olson et al. (2004) found that NSO survival was positively associated with greater amounts of both "mid-seral" (9.5-31.5 in DBH) and "late-seral" (>31.5 in DBH) forest in landscapes around activity centers (≤4,921 ft) and lower amounts of early-seral forest and non-forest (<9.5 in DBH). Reproduction, in contrast, was negatively associated with area of mid- and late-seral forest and positively associated with edge between early-seral and non-forest and other vegetation classes. Olson et al. (2004) encountered technical difficulties with the habitat fitness potential portion of

their modeling but noted that territories supporting high fitness must contain attributes associated with both high survival and high reproduction. This was supported by diagrams made from aerial photos of landscapes around activity centers, which showed remarkably similar habitat mosaics to those presented by Franklin et al. (2000).

Dugger et al. (2005) found a positive association between NSO survival in the Oregon Klamath Province and greater amounts of mature and old forest (≥100 yrs) within estimated breeding core areas (413 ac) and a moderate amount of non-habitat (non-forest, early-seral vegetation, and older forest with harvest entries ≥40% basal area) in the landscape beyond the core area (3,430-ac ring). The specific contribution of timber harvesting (and of different harvest types and intensities) to the non-habitat class and thus, its effects on NSO fitness, were not reported. This study's findings differed from others in that reproduction was positively, rather than negatively, associated with greater amounts of older forest within estimated core areas. These findings suggest that widespread harvesting of older forest within NSO core areas would negatively affect both survival and reproduction in this area but that some level of harvesting might be sustainable in the broader landscape (to the degree that it contributes to "optimal" amounts of non-habitat in the 3,430-ac ring surrounding estimated core areas).

Schilling et al. (2013) found additional evidence of a positive influence of both older forest and habitat heterogeneity on NSO survival in the Oregon Klamath. Their best performing model indicated that monthly survival probabilities for NSOs were highest when home ranges (based on radio-telemetry) contained more patches of mature and old forest (≥20 in DBH and ≥40% canopy cover). The second best performing model indicated a positive association between survival and clustering of (i.e., close distances between) older forest patches. Unlike other studies, they did not find an association between survival and total amount of older forest. They noted that this could have occurred due to their small sample size or because most NSO home ranges in their study had amounts of mature and old forest (mean = 72%) that exceeded threshold amounts required for survival. A third competitive model suggested that survival was also positively associated with a moderate amount of edge between forest (mean DBH ≥5 in) and other cover classes; thus providing additional support for the value of habitat heterogeneity for NSOs in southern interior forests.

Zabel et al. (2003) modeled probability of NSO occurrence (i.e., habitat suitability) across interior northern California based on habitat conditions at an estimated core area scale (500 ac). The best performing model in their study indicated that the probability of NSOs occurring in a given location was highest with large amounts of suitable nesting-roosting habitat (generally ≥17 in DBH and canopy cover ≥60%) and intermediate amounts of foraging habitat (≥10 in DBH and canopy cover ≥40%) at the core area scale. The second and third best performing models at the core area scale included habitat edge. The results of this modeling study provide further support for conclusions that a combination of both a large concentration of suitable habitat and some form of habitat heterogeneity is important to NSOs in interior northern California.

Carroll and Johnson (2008) also modeled probability of NSO occurrence in interior northern California. Based on their best model, predicted abundance of NSOs in the area was highest when most of the landscape (5,930-ac areas) consisted of mature and old forest (>50 yrs). However, predicted abundance slightly declined when area of mature and old forest increased beyond about 80% of the landscape. This study therefore, provides evidence of at least a slight positive effect of other vegetation classes on probability of NSOs occurring in a given area. These results contrasted with the study's findings for more northern parts of the NSO's range, where the probability of occurrence continued to increase (albeit diminishingly) with greater amounts of older forest.

Studies of home range sizes provide another line of evidence concerning habitat and harvesting influences on NSOs in interior southern forests. Home range studies in the Oregon Klamath found that home range size increased with habitat fragmentation (Carey et al. 1992, Schilling et al. 2013). NSOs in the area are known to use regenerating harvest units for foraging, particularly when closer to the activity center or outside the breeding season (Carey and Peeler 1995, Irwin et al. 2013). However, Carey and Peeler (1995) concluded that the energetic benefit of increased access to dusky-footed woodrats in heavily fragmented forest is often outweighed by the energetic cost of increased travel.

In summary, studies in interior northern California have found that NSOs in the region benefit from both large amounts of older forest concentrated around activity centers and some form of habitat heterogeneity (Franklin et al. 2000, Zabel et al. 2003, Carroll and Johnson 2008, Matthews et al. 2008). Similar results have been found in the Klamath (Dugger et al. 2005, Schilling et al. 2013) and interior Coast Range of Oregon (Olson et al. 2004). These findings suggest that NSO populations in southern interior forests can tolerate some level of timber harvesting provided suitable breeding habitat is retained in sufficiently large concentrations around activity centers (USFWS 2009). However, whether and how timber harvesting contributes to beneficial habitat heterogeneity in interior southern forests is unclear. Available studies differed in their findings of types, amounts, and locations of beneficial heterogeneity and did not directly evaluate whether timber harvesting contributed to it. In contrast, it is clear from research of associations between landscape-level habitat attributes and the demography, presence, and home range sizes of NSOs that harvesting within core concentrations of suitable habitat has the potential to negatively affect populations in southern interior forests (USFWS 2009).

Redwood Province

Most of the literature concerning NSOs in the Redwood Province pertains to research on intensively harvested lands owned by the Green Diamond Resource Company. Studies on these lands found a preference among NSOs for landscapes with greater amounts of intermediate-age or older forest than expected based on general availability of those forest classes (Thome et al. 1999, Folliard et al. 2000, Keithley and Motroni 2000, Gonzales 2005, Diller et al. 2010). However, site fidelity and reproduction on these lands were positively associated with presence of younger forest classes and measures of habitat heterogeneity (e.g., edge) (Thome et al. 1999, 2000, Diller et al. 2010). Studies of the habitat associations of dusky-footed woodrats on these lands appear to provide additional support for the value of younger forest and habitat heterogeneity to NSOs in the area (Hamm et al. 2007, Hamm and Diller 2009). Unpublished but relatively rigorous modeling of associations between landscape-level habitat attributes and NSO fitness and population growth rate has confirmed that NSOs on Green Diamond lands have complex relationships with timber harvesting (Diller et al. 2010). NSOs on Green Diamond lands indeed appear to benefit from some level of habitat heterogeneity, which is currently maintained through “small-patch” ($\leq$20 ac) clearcutting (Diller et al. 2010). Yet, habitat quality on these lands (measured as habitat fitness potential, *sensu* Franklin et al. 2000) is positively associated with protection of suitable breeding habitat, and both habitat quality and population growth rate are negatively associated with harvesting of suitable habitat (Diller et al. 2010). Thus, appropriate management of NSOs on Green Diamond lands appears to include avoiding take, setting aside suitable habitat from harvesting, and focusing economically-driven harvest requirements in relatively homogeneous blocks of unsuitable forest created by past large-block clearcutting. Diller et al. (2010) did not describe habitat conditions associated with habitat fitness potential $\geq$1 (i.e., conditions associated with NSOs replacing themselves or contributing to a population surplus). Peer reviewed reporting of these conditions could

be used to refine current take-avoidance guidelines for the Redwood Province (see USFWS 2011b, CAL FIRE 2014).

There does not appear to be any published information concerning the ecology and appropriate management of NSOs on other ownerships within the Redwood Province. Habitat conditions available to and selected by NSOs appear to differ among public and private ownerships (Keithley and Motroni 2000), industrial timber company ownerships (Appendix 2.1), and industrial versus non-industrial timberland ownerships (K. Hoffman, pers. comm.). This variability could reflect differences among forest types (e.g., redwood vs. mixed-evergreen), management regimes (e.g., intensive even-age, intensive uneven-age, and low-intensity uneven-age), and natural disturbance regimes (e.g., pre-settlement fire return intervals in northern vs. southern forests) (see Stuart and Stephens 2006, Sawyer 2007).

The USFWS (2011a, 2012a) recently conducted habitat suitability modeling based on attributes of landscapes (494 ac) surrounding 392 activity centers distributed across much of the Redwood Province. The model selected for the region included a suite of habitat variables and performed well in terms of its ability to discriminate between areas around NSO activity centers and random sites. The resulting map of relative habitat suitability was incorporated into the USFWS (2012a) process for designating critical habitat for NSOs but has limited utility for characterizing habitat selection by the subspecies. However, "deconstruction" of the habitat suitability modeling outputs (cf. Dunk and Hawley 2009, Woodbridge et al. 2012, Zielinski et al. 2012) allows evaluation of associations between habitat suitability and the full range of candidate variables, including ones not included in the best performing model. Deconstruction of the habitat modeling output for the Redwood Province shows that the probability of NSOs occurring in a given area in the region increases with larger amounts of forest with relatively dense canopy cover and large diameter trees (Appendix 2.2). Compared with those in the lowest suitability class, landscapes in the highest suitability class contained an average of 1.8 times more nesting-roosting habitat; 2.4 times higher basal area of conifers $\geq$20 inches DBH; 2.3 times higher basal area of live trees $\geq$30 inches DBH; and 2.0, 1.8 and 1.9 times higher densities of conifers $\geq$20, 30, and 39 inches DBH, respectively. There was a high degree of variability (standard deviation) in terms of structural attributes within habitat suitability classes, particularly for rare habitat elements such as very large diameter trees. As noted above, this variability likely reflects the high diversity of forest types, management histories, and natural disturbance regimes in the region (see Stuart and Stephens 2006, Sawyer 2007). Nonetheless, consistent patterns of association between habitat suitability and mean amounts of these variables are evident. In addition, variability in amounts of many of these habitat attributes (coefficient of variation) declined with increasing habitat suitability, further indicating that they are often important to NSOs in the province. These results suggest that timber harvesting that reduces availability of these structural attributes would generally reduce the probability of NSOs occurring in a given area within the Redwood Province. Changes in availability of these structural attributes can occur with a variety of silvicultural approaches and are not solely caused by even-age harvesting.

Effects of Uneven-Age Harvesting and Thinning

Some private landowners in northern California currently emphasize uneven-age regeneration or management, which typically cause less visually dramatic changes to forests than does even-age harvesting. These forms of harvesting, particularly intensive uneven-age regeneration, nonetheless have the potential to cause substantial changes to forest structure or composition. For example, intensive selective logging of Douglas-fir (*Pseudotsuga menziesii*) has resulted in extensive conversion of mixed-evergreen forest to hardwood-dominated forest in parts of the Redwood Province (Sawyer 2006).

Relatively little harvesting has occurred on federal lands within the NSO's range since adoption of the Northwest Forest Plan (Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012; see below). However, federal agencies have expressed support for widespread thinning to reduce wildfire risk, restore wildlife habitat, and meet economic objectives in the Plan area (USFS and BLM 1994, USFWS 2011a, 2012a).

Effects of contemporary uneven-age harvesting and thinning on NSOs are difficult to evaluate due to the paucity of rigorous research on the topic. Most of the available information about NSO responses to these silvicultural systems is based on the behavior of very small numbers of telemetered owls and was gathered in an opportunistic fashion during studies of other topics (reviewed in Hansen and Mazurek 2010, USFWS 2011a; see below). Evaluation of this topic is further complicated by poor descriptions of harvest methods, locations and intensities and, perhaps more importantly, post-harvest habitat conditions. The terms uneven-age harvesting and thinning encompass a tremendous variety of harvest types, objectives, and effects (e.g., Graham et al. 1999). Harvesting described in relation to NSO telemetry consisted of a variety of commercial thinning or partial harvesting (leaving residual trees) prescriptions, including understory thinning of various intensities, removal of most trees up to a relatively large diameter class, and shelterwood harvests prior to or without removal of residual trees (see Hansen and Mazurek 2010). The effects of thinning and uneven-age harvesting on NSOs may also be influenced by the condition of the landscape surrounding the harvest unit (e.g., amount, contiguity, and location of suitable NSO habitat), which could be affected by climate, soils, natural disturbance regimes, and past harvesting.

In a synthesis prepared for the 2011 revised recovery plan for the NSO (USFWS 2011a), Hansen and Mazurek (2010) provided detailed summaries of data concerning responses of both NSOs and California spotted owls (*S. o. occidentalis*; CSOs) to uneven-age harvesting, partial harvesting, and thinning. This information was gleaned from both peer-reviewed and gray literature and was based on small sample sizes. The authors therefore, opted to review each data source as a "case study" so that relatively detailed descriptions of harvesting and post-harvest conditions could be provided and so that the methodological strengths and weaknesses of studies could be evaluated. Their review is summarized below, with the addition of thee citations: Matthews et al. 2008, Gallagher 2010, and Tempel et al. *in press*.

All of the reviewed studies that described habitat use patterns by NSOs or CSOs documented at least some use of areas harvested with uneven-age harvesting, partial harvesting, or thinning. At least four of the studies found owls nesting in harvest areas (Forsman et al. 1984, Zabel et al. 1992, King 1993, and Buchanan et al. 1995) and at least five recorded roosting in them (Solis 1983, Sisco 1990, King 1993, Hicks et al. 1999, and Meiman et al. 2003). It is important to note, however, that older-forest structural attributes had been retained or regenerated in most of the harvest areas used for nesting or roosting. Three of the four studies that documented nesting in harvest areas described the nest stands as mature or old forest or an equivalent classification (USFS Region 5 "suitable habitat"; "understory reinitiation phase…of stand development"). The other study did not describe the harvest area used for nesting (King 1993). Harvest areas used for roosting in three studies likewise were either classified as mature or old forest (Solis 1983) or contained some older-forest structural characteristics, such as relatively high basal area or dense canopy cover (King 1993, Meiman et al. 2003). Two studies observed roosting in harvested stands that appeared to differ from this pattern; but one of the authors thought that the deaths of three spotted owls that roosted in them were due to higher predation risk in the more open stands (Sisco 1990, Hicks et al. 1999).

Most of the reviewed studies found that spotted owls foraged to some degree in uneven-age harvested, partially harvested, or thinned areas. Irwin et al. (2005, 2008) stated that some NSOs in their study areas selectively used certain harvest units but not others. However, they did not provide quantitative comparisons of prescriptions, post-harvest conditions, or proximities of harvest units to activity centers. Two other studies found that spotted owls generally avoided foraging in areas that recently experienced moderate to intensive partial harvesting or thinning, whereas use of lightly harvested areas varied among individuals (Anthony and Wagner 1999, Gallagher 2010). Anthony and Wagner (1999) found that NSOs (n = 15) in southern Oregon foraged in heavy and moderate partial-cuts less than expected (old stands with ≥30-40% of the original basal area removed and ≥"moderate" canopy cover reduction [not described]). Light partial-cuts (old forest with ≤20% of the original basal area removed and "small" reductions of "crown cover" [not described]) were used more than expected by two owls, as often as expected by five, and less than expected by eight. In the northern Sierra Nevada, Gallagher (2010) found that CSOs (n = 9) used heavily thinned "defensible fuel profile zones" (canopy cover reduced to 40%, removal of trees <30 in DBH, reduction of tree density and ladder and surface fuels) less than expected based on availability. She also reported a near-significant tendency ($p = 0.08$, n = 5) for avoidance of areas recently treated with understory thinning. Use and availability of harvest areas varied among individuals. Most individuals exhibited avoidance of defensible fuel profile zones and areas treated with understory thinning but one male showed strong selection for thinned areas (primarily understory thinning). It is possible that understory thinning improved prey availability or otherwise benefited this male. However, Gallagher (2010) noted that thinning treatments were located unusually close to this male's activity center, which potentially increased his likelihood of using them due to central place foraging. She also noted that an unusually large proportion of understory thinning units in the male's home range were also treated with prescribed fire, which could have temporarily increased abundances of deer mice (*Peromyscus* spp.) or other prey that tend to respond positively to fire (see Ch. 3).

The limited available information suggests that thinning and uneven-age harvesting causes some spotted owls to increase their home range sizes, which could impose energetic costs on individuals (Meiman et al. 2003, Gallagher 2010). Meiman et al. (2003) reported that a male NSO's breeding season home range in the Oregon Coast Range was slightly larger before commercial thinning than afterward but that its nonbreeding season home range was 2.3 times larger afterward. The individual also appeared to shift its breeding season core area to include less of the thinned area. In the northern Sierra Nevada, Gallagher (2010) found that the home range sizes of CSOs (n = 9) significantly increased with greater total area of fuels treatments (defensible fuel profile zones and understory thinning). She also reported near-significant trends of increasing home range size with greater area of defensible fuel profile zone ($p = 0.08$) and group selection harvesting ($p = 0.06$).

Four studies reported that thinning or partial harvesting near nests or roosts displaced spotted owls from those areas (Forsman et al. 1984, King 1993, Hicks et al. 1999, Meiman et al. 2003; also J. Reid, pers. comm.). The only study to describe this effect for more than two NSOs suggested that pairs' responses to harvesting near their nests depended on the intensity of the harvest, whether or not habitat in the nest area was excluded from harvesting, and whether or not suitable alternative habitat was available within the home range (Forsman et al. 1984).

At least two studies have evaluated potential relationships between spotted owl demographic rates and forest thinning. On the Hoopa Indian Reservation in the Western Klamath region of California, Matthews et al. (2008) found a negative association between survival of NSOs and pre-commercial thinning (prescription not described) of brushy-poletimber forest (conifer forest 10-20 yrs with a dense

brush layer). The researchers attributed this finding to a long-term negative effect of thinning on dusky-footed woodrat populations in the area (see below regarding timber harvest effects on spotted owl prey). Tempel et al. (*in press*) examined associations between CSO demographic rates at 70 territories in the central Sierra Nevada and area of "medium-intensity" harvesting (generally, retention of trees ≥30 in DBH and 40% mean DBH, reduction of fuels). Their best performing model explaining reproduction included a negative effect of medium-intensity harvesting, although evidence for this was statistically weak (95% CI of the beta coefficient broadly overlapped zero).

A recent study modeled recruitment of habitat for NSOs under a particular wildfire and forest thinning scenario in the Klamath and "dry Cascades" regions and concluded that negative effects of thinning on NSOs will outweigh potential benefits to the subspecies due to reduced risk of severe wildfire (Odion et al. 2014). Some of this study's assumptions do not appear to reflect the current scientific understanding of spotted owl-habitat relationships and wildfire and thinning effects on the species. For example, recruitment of NSO habitat was broadly defined in the study (recruitment of forest with basal area ≥120 ft^2/ac) and does not reflect the subspecies' relationships with other structural attributes, such as canopy cover, canopy layering, and large diameter trees. This study was also based on an assumption that commercially thinned and severely burned areas are always unsuitable for NSOs. NSOs are known to nest, roost, and forage in thinned areas (see above) and patchy severe fire appears to benefit NSOs in some areas, provided it does not result in extensive loss or degradation of nesting and roosting habitat (see above and Ch. 3). This study was further based on an assumption that federal agencies will blindly apply thinning to landscapes, including substantial areas of NSO habitat, rather than strategically locating treatments in areas more likely to burn at high severity and less likely to be used by NSOs (e.g., upper slopes, southwesterly aspects, densely-canopied young forest: Skinner and Taylor 2006, Skinner et al. 2006, Irwin et al. 2012). Modeling simulations have suggested that thinning can be strategically applied to relatively small portions of landscapes to reduce fire risk while minimizing negative short-term effects on spotted owls (Ager et al. 2007, Lehmkuhl et al. 2007, Prather et al. 2008).

Federal agencies have expressed support for widespread thinning to reduce the risk of severe wildfire in dry forests within the NSO's range (USFS and BLM 1994, USFWS 2011a, 2012a). The review provided in Chapter 3 suggests that spotted owls are often resilient to wildfire, and may benefit from some amount of low-to-moderate severity, mixed severity, or patchy fire in interior forests in southern Oregon and California, but that extensive severe fire can negatively affect the species by reducing amounts and contiguity of nesting and roosting habitat. This conclusion might appear to support widespread thinning to reduce the risk of large severe fires in NSO home ranges. However, preliminary findings of negative effects of thinning on spotted owls and the overall lack of reliable information on the topic suggest that rigorous research is needed to determine how best to balance tradeoffs for habitat conservation and fuels reduction objectives. If thinning is applied prior to conducting rigorous research of its effects on NSOs, research of the subspecies' habitat and prey relationships suggests that it should generally be located well away from activity centers and focused in young, closed-canopy stands with poorly developed brush layers. Thinning in these stands has the potential to increase habitat heterogeneity and accelerate development of complex, older-forest structure for NSOs and their prey (Carey 2006; but see below regarding effects of thinning on primary prey species). Planning of treatments should also integrate regional or local information about relationships between wildfires and topography (see Ch. 3), the composition of NSO diets or prey communities, and other ecological factors that could influence how thinning affects wildfires and NSOs.

Timber Harvest Effects on Prey

The primary prey for NSOs in California are dusky-footed woodrats, northern flying squirrels (*Glaucomys sabrinus*), and tree voles (*Arborimus* spp.) (Zabel et al. 1995, White 1996, Ward et al. 1998, Farber and Whitaker 2005, Diller et al. 2010, Klamath National Forest, unpubl. data). Other important prey in the state (either in terms of frequency or biomass contributions to diets) include other voles (*Myodes californicus*, *Phenacomys* spp., and *Microtus* spp.), deer mice (*Peromyscus* spp.), pocket gophers (*Thomomys* spp.), broad-footed moles (*Scapanus latimanus*), and juvenile brush rabbits (*Sylvilagus bachmani*) and snowshoe hares (*Lepus americanus*). These species have a broad array of habitat associations and thus, likely respond quite differently to timber harvesting and other forest disturbances. The review below focuses solely on timber harvest effects on the three primary prey species for NSOs in California. It is important to acknowledge, however, that NSOs typically have broad diets (see diet studies cited above) and that other prey species may also influence spotted owl demographic rates (Ward and Block 1995, Rosenberg et al. 2003).

Dusky-footed woodrats can occur in relatively high abundances in old forest, particularly in riparian areas and other locations with a well developed understory or brush layer (Carey et al. 1992, 1999). However, they generally reach their highest abundances in stands of brushy poletimber that develop following severe disturbances (Carey et al. 1992, 1999, Sakai and Noon 1993, Anthony et al. 2003, Hamm et al. 2007). Thus, intensive harvesting of intermediate-age stands could potentially result in temporary increases in abundance of dusky-footed woodrats. There is little information regarding effects of less intensive harvesting on dusky-footed woodrats. Hamm and Diller (2009) rarely found dusky-footed woodrats in thinned stands on Green Diamond Resource Company lands in the Redwood Region. They suggested that thinning without prescribed burning was insufficient for promoting growth of the disturbance-adapted shrubs locally favored by the species (see Ch. 3 regarding short-term effects of fire on prey). Matthews et al. (2008) did not directly evaluate effects of thinning on dusky-footed woodrats in the California Klamath. However, they suggested that the negative association between NSO survival and pre-commercial thinning of brushy-poletimber forest in their study was likely due to long-term declines in woodrats following thinning.

Densities and demographic rates of northern flying squirrels are positively associated with habitat elements found in forests (e.g., arboreal lichens, truffles, and snags: Rosenberg and Anthony 1992, Carey 1995, Waters and Zabel 1995, Gomez et al. 2005, Meyer et al. 2005, Lehmkuhl et al. 2006). Thus, they are likely to respond negatively to intensive forms of timber harvesting (e.g., Waters and Zabel 1995). Northern flying squirrels are also generally sensitive to habitat fragmentation caused by intensive harvesting (Smith 2007). For example, Rosenberg and Raphael (1986) found that densities of northern flying squirrels in the California Klamath Province were substantially lower in the smallest and most insular habitat patches (due to surrounding clearcut harvesting) than in the largest and best connected patches.

Research concerning the effects of thinning and other lower-intensity forms of harvesting on northern flying squirrels has generated inconsistent results (e.g., Carey 2000, Ransome and Sullivan 2002, Ransome et al. 2004, Gomez et al. 2005, Manning et al. 2012). Some of the inconsistency appears to be due to whether treated young stands are compared with structurally simple young stands (e.g., Gomez et al. 2005), structurally complex young stands (e.g., Carey 2000), or stands that have not recently experienced harvesting (Holloway and Smith 2011). The available research suggests that treated stands are more likely to contain relatively low abundances of northern flying squirrels when compared with structurally complex or mature and old stands, whereas they may exhibit similar or even higher

abundances when compared with structurally simple young stands. Harvest intensity and levels of retention appear to be another major determinant of thinning effects on northern flying squirrels, with higher intensity thinning (lower retention levels) having stronger negative effects (Meyer et al. 2007, Holloway and Smith 2011, Manning et al. 2012; but see Ransome et al. 2004). Whether thinning is patchy or uniform (in terms of location and intensity) might also be important. For example, thinning can reduce the availability of truffles, the northern flying squirrel's primary food, for more than 10-20 years; but variable-density thinning appears to be less harmful than commercial thinning (Waters et al. 1994, Colgan et al. 1999, Luoma et al. 2003, Meyer et al. 2005).

Tree voles generally occur at higher densities in old forests than in young forests (reviewed in Sztukowski and Courtney 2004, USFWS 2011c) and selectively use forests containing higher concentrations of habitat elements typically found in older stands (e.g., older stand age, larger diameter downed wood, greater basal area: Dunk and Hawley 2009). Tree voles are thought to be highly vulnerable to logging and other disturbances that reduce the extent and contiguity of old forests (Carey 1991, Huff et al. 1992, Hayes 1996, Adam and Hayes 1998, USFWS 2011c). Some tree vole populations occur in intensively managed landscapes with little or no old forest (e.g., Thompson and Diller 2002). However, the U.S. Fish and Wildlife Service (2011b) noted that "the limited evidence available suggests that tree vole occupation of younger forest stands may be relatively short-lived (Diller 2010, pers. comm.) or intermittent (Hopkins 2010, pers. comm.)." Based on the natural histories of these species, reducing or fragmenting older forest could negatively affect them; but retention of older Douglas-fir (*Pseudotsuga menziesii*) trees and patches of well-connected canopy might ameliorate those impacts (Hayes 1996, Adam and Hayes 1998, USFWS 20011b). Clear-cutting and other severe disturbances should have the strongest effects on tree voles, due to the species' diet, nesting habitat associations, arboreal mode of travel, and apparently poor mobility (USFWS 2011c). However, for these same reasons, thinning could also negatively affect tree voles (Wilson and Forsman 2013).

Habitat Lost to Past Timber Harvesting (1800s to 1994)

Rigorous research has shown that the fitness of NSOs in the southern part of their range is highest in landscapes with large concentrations of suitable breeding habitat (reviewed above). The following review shows that the current availability of suitable breeding habitat was strongly affected by past timber harvesting, which removed or modified the majority of old forest that existed historically.

There do not appear to be any existing estimates of the amount of suitable NSO habitat that existed at the time of Euro-American settlement (early to mid-1800s). Nesting-roosting habitat for NSOs generally occurs in relatively old, structurally complex conifer forest (Blakesley 2004). It is therefore, reasonable to evaluate historical trends in old conifer forest as a rough proxy for changes in amounts of suitable NSO habitat (USFWS 1990). Estimates reviewed for the NSO's federal listing determination indicated that approximately 18-24 million acres of old forest existed in western Oregon and Washington and northwestern California during the early to mid-1800s (USFWS 1990). These estimates did not include all regions or potentially suitable forest types within the subspecies' range. After including all regions and conifer forests, Strittholt et al. (2006) estimated that about 40 million acres of old conifer forest (>150 yrs) existed at the time of Euro-American settlement (Table 2.1). This is a crude approximation, as it is based on incomplete historical information and an assumption that nearly all pre-settlement conifer forest was old (i.e., had not experienced severe disturbance within the previous 150 years). However, Strittholt et al. (2006) noted that their regional estimates closely matched previous estimates for similar regions and forest types, suggesting that they provide a reasonable baseline for comparison with contemporary forest conditions.

Using satellite imagery, Strittholt et al. (2006) estimated that 11.5 million acres of old conifer forest existed in 2000 (Table 2.1). Thus, an estimated 72% of old conifer forest was lost in the Pacific Northwest during the 19th and 20th centuries (Table 2.1). This estimated post-settlement loss of old conifer forest is similar to earlier estimates of 60-88% reviewed in the NSO's federal listing determination (USFWS 1990). Strittholt et al. (2006) did not provide estimates of old forest declines for California alone. Old conifer forests declined by 62% in the Klamath provinces ("Klamath-Siskiyou Forests") and 79% in the eastern Cascades provinces ("Eastern Cascades Forests"), both of which substantially overlap with the NSO's range in northern California. Other than in two small regions surrounding major population centers in Washington and Oregon, declines in old conifer forest were primarily caused by widespread intensive logging (Strittholt et al. 2006). Mountainous terrain in the Klamath and eastern Cascades limited timber harvesting compared with more accessible areas but major losses of old conifer forest nonetheless occurred in those areas (Strittholt et al. 2006). Strittholt et al. (2006) did not evaluate trends in amounts of old forest for the Redwood Province but other sources estimated that 85-96% of old redwood forest was lost to intensive timber harvesting during the post-settlement period (USFWS 1992).

Table 2.1: Area (ha) of Pacific Northwest ecoregions and estimated historical (early to mid-1800s) and contemporary (2000) extents of old (>150 yrs) and mature (50-150 yrs) conifer forest within them (from Strittholt et al. 2006).

	Cascade Mountain Leeward Forests	*North Cascades Forests*	*Puget Lowland Forests*	*Central Pacific Coastal Forests*	*Central & Southern Cascades Forests*	*Eastern Cascades Forests*	*Klamath-Siskiyou Forests*	*Willamette Valley Forests*
Ecoregion area	1,600,444	1,278,056	1,719,726	4,267,988	4,481,279	5,398,139	5,033,181	1,487,769
Historical old conifer area	960,279	766,844	1,289,794	3,627,790	3,361,005	3,238,883	2,516,625	371,942
Percentage of ecoregion	60	60	75	85	75	60	50	25
Current old conifer area	420,394	511,396	67,353	668,281	1,328,796	687,836	946,394	34,993
Percentage of historical old conifer	44	67	5	18	39	18	38	9
Percent change	−56	−33	−95	−82	−61	−79	−62	−91
Mature conifer area	285,441	203,994	229,634	683,098	899,266	947,215	1,239,586	270,362
Percentage of historical old conifer	30	27	18	19	27	25	49	73
Combined area	705,835	715,391	296,987	1,351,379	2,228,063	1,635,051	2,185,980	305,355
Percentage of historical old conifer	74	93	23	37	66	43	87	82

Strittholt et al. (2006) reported that the majority of old (78%) and mature (50%) conifer forest in 2000 existed on public lands. Nearly all of the remaining old redwood forest likewise occurs on public lands, such as California state parks. Much of the current difference among ownerships in amounts of older forest and suitable breeding habitat is due to past timber harvest rates. For example, loss of forest to harvesting during the 1970s through early 1990s occurred at substantially higher rates on private timberlands than on federal lands (e.g., >2 times faster in western Oregon) (Cohen et al. 2002, Staus et al. 2002, Healey et al. 2008). Nonetheless, an estimated 32% of suitable breeding habitat for NSOs occurred on non-federal lands at the time of the Northwest Forest Plan's implementation (1994), so conservation efforts for NSOs on non-federal lands remain important.

Although timber harvesting was substantially curtailed on federal lands following implementation of the Northwest Forest Plan (Healey et al. 2008, Kennedy et al. 2012; reviewed below), biologists noted the possibility that NSOs would continue to decline for many years due to lag effects of past harvesting (Courtney et al. 2004). The NSO has a relatively low reproductive rate and might therefore be unable to

immediately recover following removal or reduction of threats (Noon and Biles 1990). Furthermore, substantial recruitment of old forest and suitable nesting-roosting habitat could take multiple decades in areas that formerly experienced widespread intensive harvesting (Moeur et al. 2011; see below). Past harvesting could therefore be among the causes of continuing poor demographic performance of some NSO populations (Courtney et al. 2004). Forsman et al. (2011) noted, however, that some populations are declining on lands not previously subjected to widespread intensive timber harvesting (e.g., some National Parks). Based on this observation, they concluded that lag effects of past timber harvesting poorly explain continuing population declines. Yet, it is possible that historical timber harvesting does continue to contribute to population declines but that this effect is obscured by that of other stressors, such as competition with invasive barred owls (see Ch. 4). Regardless of potential lag effects of historical harvesting on NSOs, timber harvesting continues to occur at high rates on private lands and is one of the primary sources of habitat loss for the subspecies (Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012; reviewed below).

Habitat Lost to Contemporary Timber Harvesting (1994-2007)

Davis and Dugger (2011) estimated NSO habitat trends following implementation of the Northwest Forest Plan. Their analyses were mostly limited to federal lands within the Plan area but they also estimated habitat trends on non-federal lands, as reported in the 2011 revised NSO recovery plan (USFWS 2011a). In addition to these analyses, the following review includes results of research by Moeur et al. (2011), which provide additional insight into recent habitat trends for NSOs on non-federal lands. This review does not include habitat trend estimates based on federal ESA Section 7 consultation records (Bigley and Franklin 2004, USFWS 2012b). These records provide a less consistent and complete data source than those used by Davis and Dugger (2011) (see Bigley and Franklin 2004). They may also overestimate habitat changes since they evaluate effects of planned projects, which may be greater than what is actually implemented (Bigley and Franklin 2004).

Davis and Dugger (2011) used remotely sensed (satellite imagery) and forest inventory plot vegetation data to model changes in habitat suitability for NSOs during the first 15 years of the Northwest Forest Plan (1994-2007 in California and 1996-2006 in Oregon and Washington). They modeled habitat suitability based on habitat attributes surrounding thousands of NSO pair locations. Suitable breeding habitat was defined as having both a probability of owl presence greater than expected based on random chance and environmental conditions typical of those found around nesting and roosting pairs. Habitat loss was defined as a change in suitability rank from suitable or highly suitable to marginal or unsuitable due to vegetation changes caused by forest disturbances. Davis and Dugger (2011) did not estimate recruitment of, or net changes, in breeding habitat. They felt that their remotely sensed data poorly captured the kinds of slow and subtle habitat changes that occur during development of intermediate-aged and older stands. However, Moeur et al. (2011) estimated trends in mature and old forests during the same time period, which could provide insight into net changes in breeding habitat for NSOs.

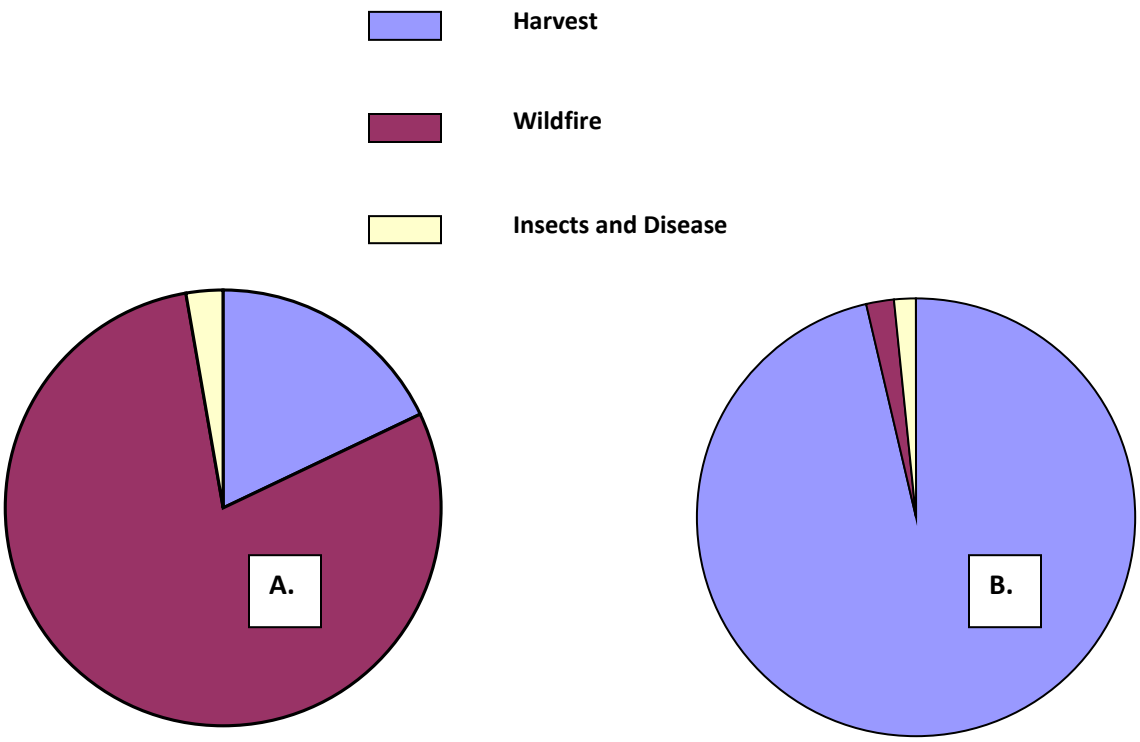
Table 2.2 shows estimated gross losses of suitable NSO breeding habitat to timber harvesting on federal and non-federal lands during the first 15 years of the Northwest Forest Plan (Davis and Dugger 2011, USFW 2011a). Timber harvesting was responsible for a gross loss of about 54,000 acres (0.6%) of suitable breeding habitat on federal lands. This loss likely had little rangewide effect on NSOs but could have impacted the subspecies at local or regional scales. For example, harvesting resulted in a 3% gross loss of suitable breeding habitat on federal lands in the California Cascades, where habitat was already relatively limited. Approximately 92% of total suitable breeding habitat lost to timber harvesting occurred on non-federal lands. In contrast with federal lands, nearly all estimated gross habitat loss on

non-federal lands was due to timber harvesting rather than natural disturbances (Figure 2.1; see Ch. 3). In just 11-13 years, timber harvesting caused an estimated rangewide gross loss of 625,600 acres (15%) of suitable breeding habitat on non-federal lands. The largest losses on non-federal lands occurred in Oregon (301,200 ac, 22%) and Washington (234,200 ac, 19%). Non-federal lands in California experienced lower gross losses of suitable breeding habitat to harvesting (90,200 acres, 6%). Nonetheless, losses in all three states were substantial given the short time frame during which they occurred and the likelihood that little of the loss was offset by recruitment of suitable breeding habitat during that period (see below).

Table 2.2: Estimated gross losses of suitable NSO breeding habitat on federal and non-federal lands due to timber harvesting during 1994-2007 in California and 1996-2006 in Oregon and Washington (adapted from Davis and Dugger 2011, USFWS 2011a).

State	Ownership	Province	1994/1996 Ac	Harvest Ac	Harvest %
California	**Federal**	CA Cascades	213,200	6,500	3.0%
		CA Klamath	1,489,800	4,400	0.3%
		CA Coast	145,400	300	0.2%
		CA Federal Total	**1,848,400**	**11,200**	**0.6%**
	Non-Federal		**1,556,700**	**90,200**	**5.8%**
	Combined		**3,405,100**	**101,400**	**3.0%**
Oregon	**Federal**	OR Coast Range	611,200	3,300	0.5%
		Western OR Cascades	2,258,700	13,900	0.6%
		Eastern OR Cascades	402,900	5,800	1.4%
		Willamette Valley	3,400	100	2.9%
		OR Klamath	985,000	6,800	0.7%
		OR Federal Total	**4,261,200**	**29,900**	**0.7%**
	Non-Federal		**1,382,400**	**301,200**	**21.8%**
	Combined		**5,643,600**	**331,100**	**5.9%**
Washington	**Federal**	Olympic Peninsula	763,100	500	0.1%
		Eastern WA Cascades	673,600	8,100	1.2%
		Western WA Cascades	1,283,000	3,700	0.3%
		Western WA Lowlands	24,700	400	1.6%
		WA Federal Total	**2,744,400**	**12,700**	**0.5%**
	Non-Federal		**1,258,900**	**234,200**	**18.6%**
	Combined		**4,003,300**	**246,900**	**6.2%**
Rangewide	**Federal**		**8,853,800**	**53,800**	**0.6%**
	Non-Federal		**4,198,000**	**625,600**	**14.9%**
	Combined		**13,051,800**	**679,400**	**5.2%**

Figure 2.1: Proportions of suitable breeding habitat loss attributed to harvesting, wildfire, and insects and diseases on (A) federal lands and (B) non-federal lands during 1994-2007 (adapted from Davis and Dugger 2011 and USFWS 2011a).



The USFWS (2011a) and Davis and Dugger (2011) did not describe regional habitat trends for non-federal lands. However, insight into regional habitat trends on non-federal lands can be obtained from trends in mature and old forest during the same time period. Moeur et al. (2011) reported substantial gross losses of mature and old forest (mean DBH ≥20 in) on non-federal lands during the first 15 years of the Northwest Forest Plan. The largest gross losses, in terms of acreage, occurred in the Western Washington Lowlands (387,200 ac, 49%), Oregon Coast Range (362,500 ac, 50%), and California Coast Range (259,000 ac, 35%). All provinces and states within the NSO's range experienced large proportional losses, ranging from 31% in the Eastern Washington Cascades to 48% and 52% in the Klamath and Eastern Cascades of Oregon. Confirming the results of Davis and Dugger (2011; Figure 2.1), Moeur et al. (2011) found that gross losses of mature and old forest on non-federal lands were almost entirely due to timber harvesting (also see Kennedy et al. 2012).

Moeur et al. (2011) reported that gross loss of mature and old forest was substantially offset by recruitment into that habitat class. They noted, however, that given the short length of the monitoring period (10-14 yrs), recruitment was "likely due to incremental stand growth over the 20-in diameter threshold, or from understory disturbances that removed smaller diameter trees and raised the average stand diameter above the threshold, rather than from an increase in forests of much larger and older trees." Thus, it is unlikely that there was substantial recruitment of suitable and highly suitable breeding habitat for NSOs during this time period. This conclusion is supported by Davis and Dugger (2011), who found that most of the detectable habitat recruitment during their monitoring period occurred in the marginal suitability class, which more closely resembled their definition for dispersal habitat than for breeding habitat. Even if all mature and old forest recruited during the first 15 years of the Plan provided suitable breeding habitat for NSOs, non-federal lands in California still experienced a net decline in area of mature and old forest during that period, and those in Washington and Oregon fared substantially worse (Moeur et al. 2011).

Future Harvesting in California

It is impossible to provide reliable projections of future timber harvesting or its effects on NSOs in California. Federal agencies have expressed support for widespread thinning to address wildfire risk on public lands in the state but there do not appear to be any projections of future harvest volume or effects on NSOs from these activities (USFS and BLM 1994, USFWS 2011a, 2012a). Documents associated with Habitat Conservation Plans for private timberlands in California project substantial impacts of harvesting on some ownerships and relatively low impacts on others. Many landowners, in the state, including some large industrial timber companies, conduct timber harvesting outside of Habitat Conservation Plans. The state requires evaluation of potential environmental impacts of all Timber Harvest Plans but both landowners and responsible agencies have used inconsistent methods for conducting these evaluations. For example, some entities have strictly adhered to the state's Forest Practice Rules (CAL FIRE 2014), others have relied on poorly described and vetted variants of those rules (e.g., "option g+"), and still others have opted to follow the Yreka or Arcata U.S. Fish and Wildlife Service Offices' (2009, 2011b) recommendations. Based on an in-depth review of research concerning the NSO's habitat and spatial relationships, the U.S. Fish and Wildlife Service Yreka Office (2009) recommended sweeping changes to NSO habitat retention guidelines in the Forest Practice Rules for California's northern interior. These recommendations are more scientifically supportable than are habitat retention guidelines in the Forest Practice Rules (CAL FIRE 2014), as they incorporate the large body of research of NSO-habitat relationships conducted since 1992 when guidelines in the Forest Practice Rules were created. In addition they were designed to enable CAL FIRE personnel lacking expertise with NSO-habitat relationships to properly determine if take would occur. However, the state

has not officially adopted these recommendations or any other changes that incorporate the tremendous body of research and biological expertise concerning NSO-habitat relationships developed since 1992 (USFW 2009). Furthermore, since 2008, when the US Fish and Wildlife Service largely ceased providing technical assistance with timber harvest reviews in northern California, relatively few Timber Harvest Plans have been reviewed by personnel with sufficient biological expertise to evaluate whether or not take will occur. Lacking reliable harvest and take projections, and barring a major change in the legal or regulatory framework protecting NSOs, there is currently no reason to conclude that timber harvest effects on NSOs in California will substantially decline in the near future.

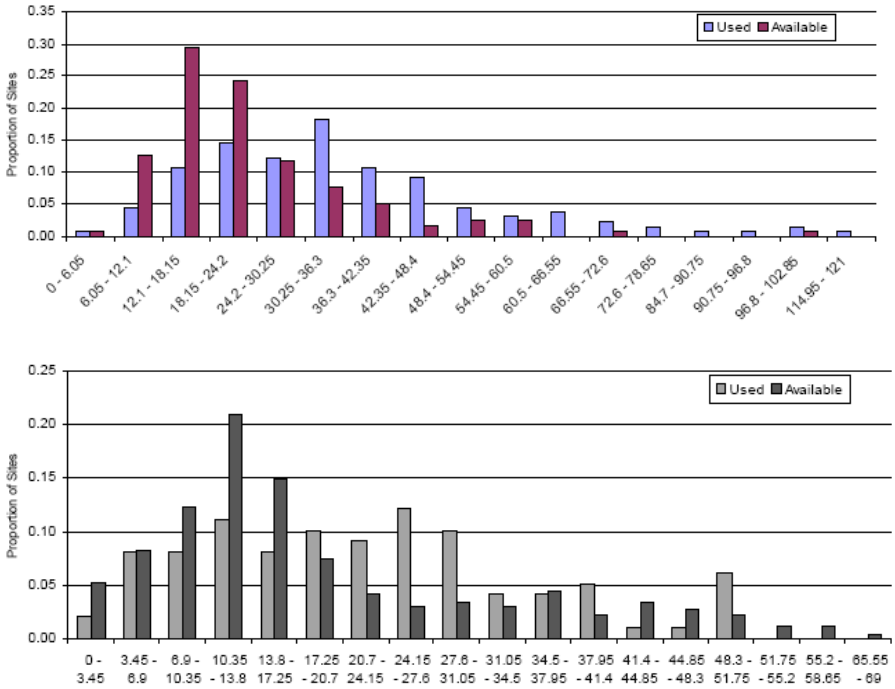
Acknowledgements

I am grateful to Ray Davis and Ken Hoffman for their insightful reviews of this chapter.

Appendix 2.1

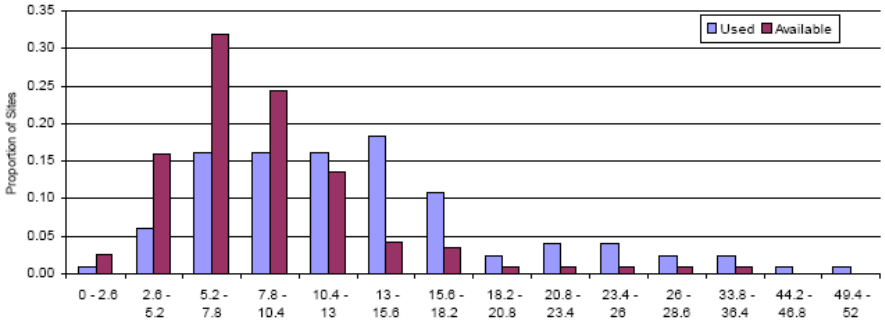
USFWS GNN (see USFWS 2011a) histograms showing use by NSOs versus availability of select habitat attributes at a landscape-scale (200 ha, 494 ac) on Mendocino Redwood Co. (left column) and Green Diamond Resource Co. lands (right column).

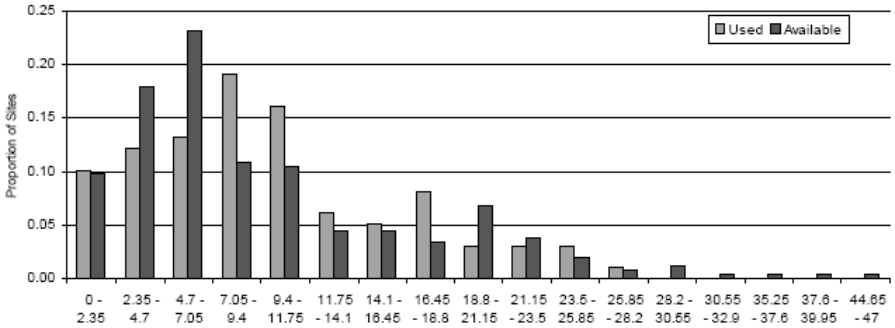
Live conifer trees per hectare ≥20 inches DBH:



Appendix 2.1 (continued)

Basal area of live conifers ≥20 inches DBH:

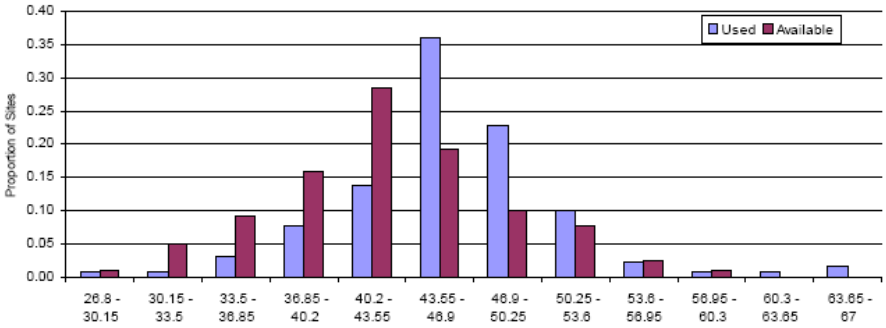


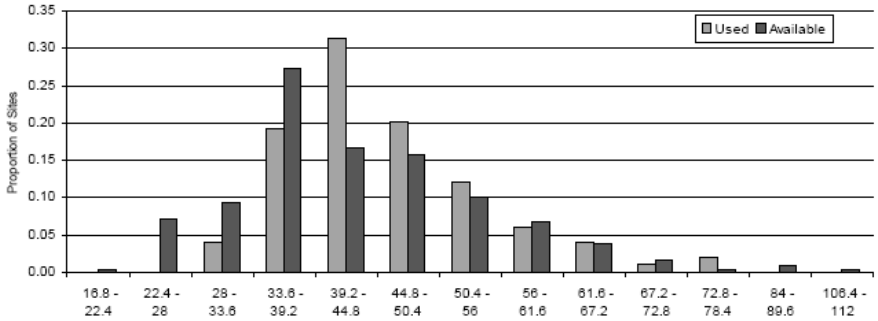


Appendix 2.1 (continued)

USFWS GNN (see USFWS 2011a) histograms showing use by NSOs versus availability of select habitat attributes at a landscape-scale (200 ha, 494 ac) on Mendocino Redwood Co. (left column) and Green Diamond Resource Co. lands (right column).

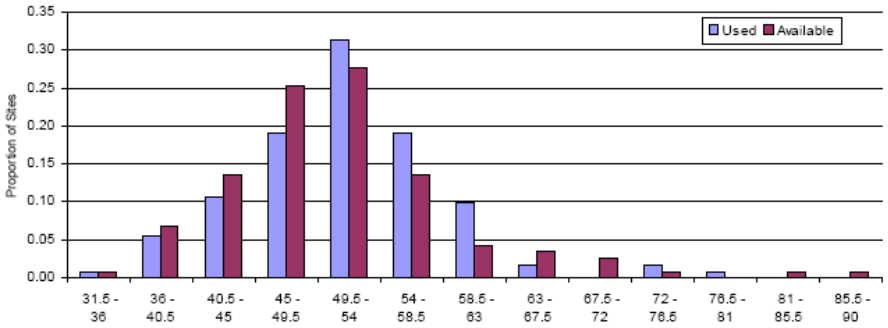
Basal area weighted mean diameter of all live conifers:

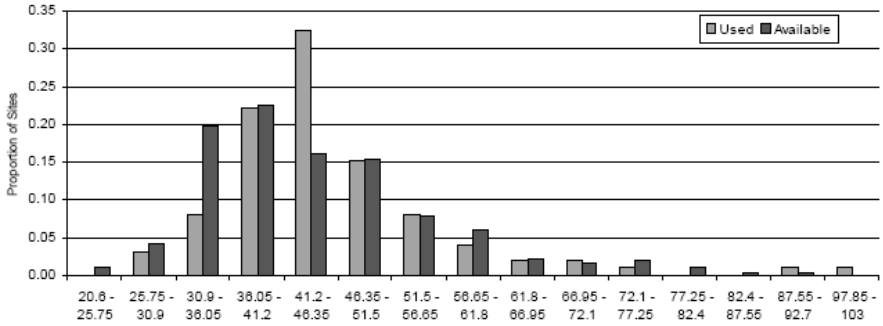




Appendix 2.1 (continued)

Basal area weighted stand age:

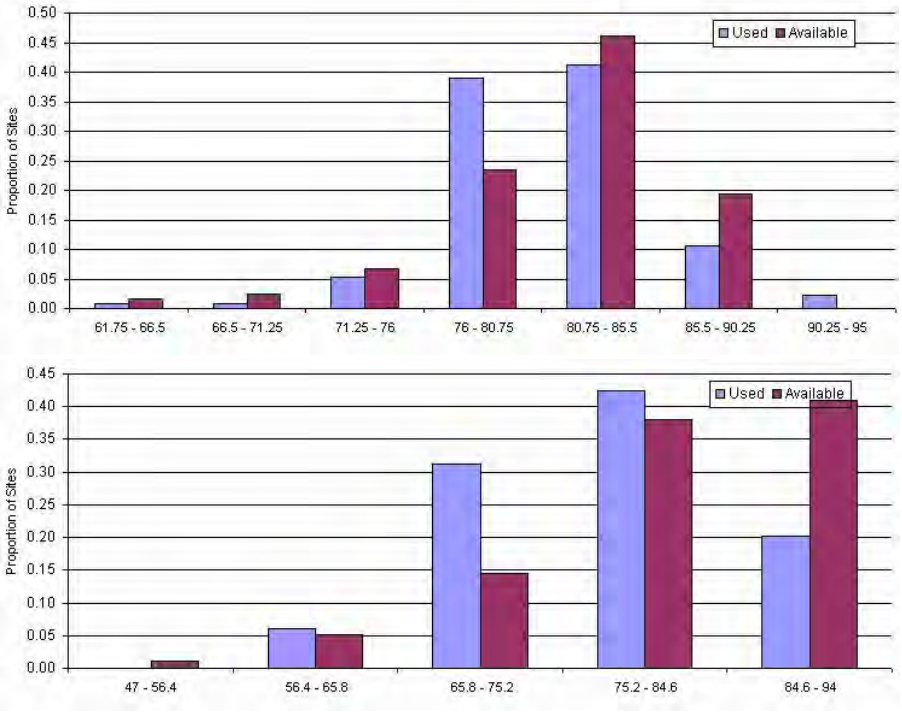




Appendix 2.1 (continued)

USFWS GNN (see USFWS 2011a) histograms showing use by NSOs versus availability of select habitat attributes at a landscape-scale (200 ha, 494 ac) on Mendocino Redwood Co. (left column) and Green Diamond Resource Co. lands (right column).

Canopy cover:



Appendix 2.2

"Deconstructed" habitat suitability modeling (see text) for NSOs in the Redwood Province showing mean (SD, CV) values of select habitat structural attributes at the 200 ha (494 ac) scale by relative habitat suitability rank (USFWS 2011a, unpubl. data).

Variable	Relative Habitat Suitability					
	Very Low	**Low**	**Med. Low**	**Med.**	**Med. High**	**High**
Strength of Selection*	-9.2	-2.9	-1.3	1.6	3.2	8.6
Percent of Region	16.3	24.5	26.2	22.4	9.7	0.9
Nesting-Roosting (ha)	57.9 (43.2, 75)	69.4 (40.8, 59)	79.9 (37.3, 47)	87.6 (33.6, 38)	94.3 (31.1, 33)	105.5 (30.8, 29)
Canopy Cover All Trees (%)	71.2 (23.2, 33)	75.2 (20.7, 28)	78.9 (18.1, 23)	81.0 (16.2, 20)	82.1 (15.5, 19)	82.9 (15.7, 19)
Canopy Cover Conifer (%)	43.9 (31.8, 72)	48.8 (30.0, 61)	53.4 (28.3, 53)	57.4 (27.2, 47)	61.4 (26.5, 43)	64.6 (26.4, 41)
BA Conifers ≥50 cm (m^2/ha)	10.6 (20.7, 195)	12.5 (23.0, 184)	14.1 (24.8, 176)	15.2 (25.6, 168)	17.6 (30.1, 171)	25.2 (45.9, 182)
BA Live Trees ≥75cm (m^2/ha)	7.3 (17.6, 241)	8.5 (20.1, 236)	9.3 (21.8, 234)	9.3 (22.4, 241)	10.4 (27.1, 261)	17.0 (44.0, 259)
Density Trees ≥50 cm (no./ha)	32.3 (37.3, 115)	36.1 (38.4, 106)	39.8 (40.0, 101)	42.5 (42.1, 99)	45.4 (44.5, 98)	50.0 (46.1, 92)
Density Trees ≥75 cm (no./ha)	8.2 (14.8, 180)	9.2 (15.6, 170)	9.9 (16.0, 162)	10.0 (15.5, 155)	10.4 (15.3, 147)	12.8 (16.8, 131)
Density Conifers ≥50 cm (no./ha)	22.1 (34.7, 157)	25.5 (36.4, 143)	28.9 (38.3, 133)	32.6 (40.7, 125)	37.3 (43.6, 117)	43.1 (45.6, 106)
Density Conifers ≥ 75 cm (no./ha)	6.6 (14.5, 220)	7.6 (15.4, 203)	8.4 (15.7, 187)	8.7 (15.2, 175)	9.4 (15.1, 161)	12.0 (16.7, 139)
Density Conifers ≥100 cm (no./ha)	2.5 (7.7, 308)	2.9 (8.4, 290)	3.2 (8.8, 275)	3.1 (8.3, 268)	3.2 (8.5, 266)	4.7 (10.4, 221)
Mean DBH Conifers by BA (cm)	42.8 (34.5, 81)	48.5 (35.5, 73)	51.7 (35.4, 68)	52.1 (34.0, 65)	52.9 (36.1, 68)	60.8 (51.1, 84)
QMD Dominant/Codominant Conifers (cm)	35.7 (29.1, 82)	40.2 (29.9, 74)	42.7 (29.7, 70)	42.6 (28.3, 66)	42.7 (29.7, 70)	48.1 (41.5, 86)

*Calculated as the proportion of activity centers in a habitat suitability class divided by the proportion of the modeling region in that class.

Ch. 3: Wildfire and Salvage Logging

Introduction

Recent status reviews have identified wildfire as a primary threat to the recovery of the NSO (Courtney et al. 2004, Davis et al. 2011, USFWS 2011a). Much of this concern was based on recent loss of suitable breeding habitat to wildfires and to the risk of extensive severe fires occurring in the future (Davis and Dugger 2011, USFWS 2011a). Other researchers and stakeholders have questioned the scientific basis of claims that wildfires pose a threat to NSOs and have expressed distrust of agency recommendations for widespread use of forest thinning to reduce fire risk (e.g., Hanson et al. 2009, Heiken 2010, DellaSala et al. 2013).

There is currently limited information with which to evaluate responses of spotted owls to wildfires and post-fire salvage logging. This research suggests that wildfires have variable and complex effects on the species (Table 3.1; reviewed below). This is unsurprising given differences in wildfires, research methods, study areas, and spotted owl subspecies and populations. Nonetheless, patterns are evident in the literature concerning spotted owl responses to wildfires and salvage logging and these can be evaluated in light of the species' habitat and prey relationships. Currently available research suggests that low-to-moderate, mixed-severity, or patchy wildfires have limited effects on spotted owls (Table 3.1). In fact, such fires may benefit NSOs in the southern portion of their range by contributing to landscape-level habitat heterogeneity associated with high fitness (Franklin et al. 2000). In contrast, large-scale severe (stand-replacing) wildfires appear to have strong negative effects on spotted owls (Table 3.1; reviewed below). This likely occurs when fires excessively modify, reduce, or fragment concentrations of suitable nesting, roosting, and foraging habitat needed for survival and reproduction (Franklin et al. 2000, Dugger et al. 2005, Schilling et al. 2013; reviewed below and in Ch. 2). Negative effects of extensive severe wildfires appear to be exacerbated by post-fire salvage logging, which structurally simplifies burned areas and removes important habitat legacies for prey (Clark 2007, Lee et al. 2012, 2013, Clark et al. 2013, Comfort 2013; reviewed below).

Regardless of scientific uncertainty concerning spotted owl responses to wildfire, it is clear that recent large wildfires have caused tremendous loss, degradation, and fragmentation of suitable breeding habitat for NSOs on federal lands (Davis and Dugger 2011, USFWS 2011a; also see Healey et al. 2008, Moeur et al. 2011). This is cause for concern because recovery of the subspecies largely relies on habitat protection on federal lands (USFWS 2011a). Furthermore, much of the climate change research indicates that wildfires will be an increasing source of large-scale habitat change in California and other western states during coming decades (Westerling et al. 2006, Lenihan et al. 2008, Westerling and Bryant 2008, Littell et al. 2009, Moritz et al. 2012, Stavros et al. 2014). These concerns have prompted ecologists and federal agencies to advocate widespread forest thinning and other forms of active management to reduce wildfire risk within the range of the NSO (USFS and BLM 1994, USFWS 2008, 2011a, 2012a, Franklin and Johnson 2012). However, the limited information currently available suggests that spotted owls often respond negatively to forest thinning in the short-term (reviewed in Hansen and Mazurek 2010, USFWS 2011a; see Ch. 2); and possibly in the long-term as well (Matthews et al. 2008, Tempel et al. *in press*). Further research is needed to determine how best to balance

potential tradeoffs in objectives for NSO conservation and fuels reduction at multiple spatial and temporal scales. Currently available information suggests that spotted owls tolerate and possibly benefit from low severity, mixed severity, or patchy fire, suggesting that prescribed fire and allowing wildfires to burn under favorable conditions is compatible with conservation objectives for the species (Bond et al. 2002, 2009, Roberts and van Wagtendonk 2006, Roberts et al. 2011, Keane et al. 2011, 2012, Lee et al. 2012, 2013, Comfort 2013, Eyes 2014).

Wildfire Effects on Spotted Owls

Indirect Evidence

Research of associations between landscape-scale habitat attributes and the demography and presence of NSOs has consistently found that the subspecies benefits from some form of habitat heterogeneity in the southern portion of its range (e.g., ecotones or edges between different vegetation classes) (Franklin et al. 2000, Zabel et al. 2003, Olson et al. 2004, Dugger et al. 2005, Carroll and Johnson 2008, Diller et al. 2010, Schilling et al. 2013; reviewed in Ch. 2, including studies' definitions of spotted owl habitat). Yet, these same studies have also strongly demonstrated the importance of large concentrations of suitable breeding habitat around activity centers (reviewed in Ch. 2). Based on this research, wildfires likely have positive effects on NSOs in California when they contribute to beneficial forms of habitat heterogeneity and negative effects when they substantially reduce or degrade suitable habitat around breeding season activity centers. Extensive stand-replacing wildfires have the potential to remove or fragment core concentrations of suitable breeding habitat across multiple NSO territories. These fires, therefore, have the greatest likelihood of substantially impacting NSO populations. Smaller, less severe, or patchy wildfires may impact fewer NSO territories and have weaker negative effects on populations, burn in a manner that contributes to beneficial forms of habitat heterogeneity, or have variable effects among territories.

Direct Evidence

Several studies have investigated responses of NSOs to wildfires (Table 3.1). These studies provide crucial information for evaluating wildfire as a potential threat to the subspecies. However, their inferences are limited due to small sample sizes and short time frames in all cases, the confounding effects of post-fire salvage logging in one case, and pooling of data from all three spotted owl subspecies in another case (Table 3.1; see below). In order to supplement these studies, research of wildfire effects on California spotted owls (*S. o. occidentalis*; CSOs) and Mexican spotted owls (*S. o. lucida*; MSOs) is also included in the following review (Table 3.1). Because inferences from these studies are likewise limited, and given differences among fires, spotted owl subspecies and populations, and research methods, each project is reviewed as a "case study". Relatively thorough descriptions of these studies allow identification of patterns in the literature, which could provide insights into general effects of wildfires on the species.

Table 3.1: Apparent effects of wildfires on spotted owls. See text for additional descriptions of study methods and findings.

Response Metric	Study†	Subspecies	Location*	Apparent Effect**	Notes/Caveats
Survival	*Bond et al. 2002*	NSO, CSO, MSO	CA, AZ, NM	0	Only one post-fire survey season
	Clark et al. 2011	NSO	OR KLA	-	Likely cumulative effect of timber harvesting, severe fire, and salvage logging
Productivity	Gaines et al. 1997	NSO	E WA CAS	0	Apparently no decline but possibly obscured by low reproduction year across population; Possibly lower total reproduction in burned landscapes due to lower pair occupancy; Only one post-fire season
	Bond et al. 2002	NSO, CSO, MSO	CA, AZ, NM	0	Only one post-fire season
	Clark 2007	NSO	OR KLA	0	Apparently no decline but low statistical power; Possibly lower total reproduction in burned landscapes due to lower pair occupancy
	Roberts and van Wagtendonk 2006	CSO	CA SIERRA	+	Apparent higher productivity by four pairs nesting in low-to-moderate severity burns than by 18 pairs in unburned areas
Site Fidelity	*Bond et al. 2002*	NSO, CSO, MSO	CA, AZ, NM	0	Site fidelity similar to other studies
Occupancy	Elliot 1985	CSO	CA COAST	-	Apparent abandonment by two pairs
	Gaines et al. 1997	NSO	E WA CAS	-	Post-fire occupancy was lowest found during five-year study; Only one post-fire season
	Jenness et al. 2004	MSO	AZ, NM	-	Near-significant negative trend; Pooled all fire types and severities
	Keane et al. 2011, 2012	CSO	CA SIERRA/CAS	-/0	Extensive high severity fire apparently had a strong negative effect; Extensive low severity fire apparently had a neutral or weak negative effect; Possibly confounded by salvage logging
	Roberts et al. 2011	CSO	CA SIERRA	-	Modeled-occupancy lower in burned areas but not statistically analyzed; Pooled all fire types and severities
	Lee et al. 2012	CSO	CA SIERRA	0	Similar occupancy at burned vs. unburned
	Lee et al. 2013	CSO	SO CA	-/0	Similar occupancy at burned vs. unburned; Significant reduction with extensive high severity fire in core area
	Clark et al. 2013	NSO	OR KLA	-	Cumulative effect of timber harvesting, severe fire, and salvage logging
	Tempel et al. In Press	CSO	CA SIERRA	-	Site colonization probability negatively associated with area of wildfire; Relatively large sample size and long time frame
Home Range	Clark 2007	NSO	OR KLA	-	Larger home ranges post-fire
	Bond et al. 2013	CSO	CA SIERRA	0	Similar home range sizes to unburned areas
Roosting	King et al. 1997	NSO	E WA CAS	-	Apparent avoidance of moderate and severe burns
	Bond et al. 2009	NSO	CA SIERRA	-	Significant avoidance of moderate and severe burns during breeding season
Foraging	Clark 2007	NSO	OR KLA	+	Apparent weak selection of moderately burned suitable habitat; Possible weak selection for severely burned suitable habitat; Very low use and availability of both moderately and severely burned suitable habitat
	Bond et al. 2009	NSO	CA SIERRA	+	Significant selection of severe burns
	Eyes 2014	CSO	CA SIERRA	-/+	Preference for edges created by fire (particularly high contrast); Avoidance of severely burned areas
Roosting and Foraging	Comfort 2013	NSO	OR KLA	-/+	Preference for small patches of severely burned/salvage logged and avoidance of larger patches; Weak preference for low contrast edges created by fire

†Peer-reviewed publications shown in italics. * Locations: California Klamath (CA KLA); Eastern Washington Cascades (E WA CAS); Oregon Klamath (OR KLA); California, Arizona and New Mexico (CA, AZ, NM); California Central Coast Range (CA COAST); California at margin of northern Sierra Nevada and southern Cascades (CA SIERRA/CAS); California Sierra Nevada (CA SIERRA); southern California San Bernardino and San Jacinto Mountains (SO CA). **Apparent Effect: negative (-), positive (+), neutral (0), varied with fire severity and/or scale (/)—see Notes column and text for further explanations.

Survival

Wildfires may influence spotted owl survival in both the short- and long-term. For example, spotted owls, like other wildlife, could be injured or killed by smoke during fires (Singer and Schullery 1989, Smith 2000). Due to their poor mobility, young spotted owls with undeveloped flight feathers may be at particular risk of mortality during wildfires (Smith 2000). In addition to potential immediate effects, extensive moderate or severe wildfires might influence spotted owl survival over the longer-term by modifying habitat for roosting, foraging, or prey (see below).

Only two studies are currently available for evaluating effects of wildfires on spotted owl survival rates (Bond et al. 2002, Clark et al. 2011; Table 3.1). Bond et al. (2002) reported that 18 of 21 (86%) marked spotted owls were resighted one year after wildfires occurred in California, Arizona, and New Mexico. This minimum survival rate was similar to survival estimates found by long-term studies of the three spotted owl subspecies in unburned landscapes (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data). Of the eight territories for which fire severity was mapped, two experienced severe fire within 50-88% of their areas, two experienced 36-50% severe fire, and the remaining four experienced <36% severe fire. Thus, mixed severity wildfires did not appear to have a substantial effect on spotted owl survival in this study one year post-fire.

Clark et al. (2011) found evidence of a negative effect of wildfires and/or post-fire salvage logging on survival of 23 NSOs in the Oregon Klamath Province. Severe fire and/or post-fire salvage logging occurred in 30% and 41% of suitable NSO habitat in the two study areas (suitability score >25: Davis and Lint 2005 [essentially nesting-roosting-foraging habitat; QMD generally > ca. 12 in]). Estimated mean annual survival rates for NSOs located inside fire perimeters (0.69) and apparently displaced by fires and post-fire salvage logging (0.66) were lower than in areas just outside the fire perimeters (0.85) and in an unburned reference study area in the neighboring southern Cascades (0.85: Anthony et al. 2006). The degree to which post-fire salvage logging in the study areas influenced NSO survival rates is unknown. The study's occupancy analyses indicated that pre-fire timber harvesting, high severity wildfire, and post-fire salvage logging cumulatively impacted NSOs through reductions of suitable breeding habitat (Clark et al. 2013; see below).

The findings of Bond et al. (2002) and Clark et al. (2011) regarding effects of moderate-to-extensive amounts of severe wildfire (>36% of the area in half of the territories: Bond et al. 2002; 30-41% of the study area: Clark et al. 2011) appear to be contradictory. Several factors may explain this apparent inconsistency. The most obvious difference between the studies is that the areas studied by Clark et al. (2011) experienced post-fire salvage logging while those studied by Bond et al. (2002) did not. The limited available information suggests that salvage logging negatively affects spotted owls (reviewed below). Additionally, the populations studied by Clark et al. (2011) may have been particularly sensitive to habitat loss to wildfires due to intensive pre-fire timber harvesting across a checkerboard ownership. It should also be noted that Bond et al. (2002) only examined wildfire effects one year post-fire. Fire injuries and post-fire outbreaks of insects and pathogens can continue to result in tree mortality for up to several years after a wildfire (Ryan and Amman 1996, Gaines et al. 1997, Hood et al. 2007).

Reproduction

The spotted owl is a relatively long-lived species (up to 12-17+ yrs in the wild: Gutiérrez et al. 1995) that exhibits a bet-hedging life history strategy (Noon and Biles 1990, Franklin et al. 2000). This means that individuals often forego breeding during poor environmental conditions in order to maximize their chance of surviving and reproducing in the future. Given the species' life history strategy, spotted owl reproductive rates are likely sensitive to environmental changes, including those brought about by wildfires. However, annual fluctuations in spotted owl reproduction caused by variation in weather, prey populations, or breeding condition could obscure effects of wildfires or other factors on reproduction (e.g., Franklin et al. 2000).

At least four studies have examined potential effects of wildfires on spotted owl reproduction (Gaines et al. 1997, Clark 2007, Bond et al. 2002, Roberts and van Wagtendonk 2006 [also Roberts 2008]; Table 3.1). None of these studies found substantive evidence of a wildfire-induced decline in reproduction by the species and one indicated a potentially positive effect. In the eastern Washington Cascades, Gaines et al. (1997) found little or no difference in productivity (number of young per pair) at burned (0.2; n = 5 or 6/ yr) and unburned sites (0.3; n = 13-17/yr) one year after a predominantly moderate to severe wildfire. However, the post-fire survey season clearly occurred during a poor reproduction year, potentially making it difficult to detect a difference between burned and unburned sites. Clark (2007) found no significant differences in productivity in burned areas in the Oregon Klamath Province (n = 31 territories) and an unburned study area in the neighboring southern Cascades (Anthony et al. 2006). He noted, however, that his study likely lacked the statistical power to detect a difference if one occurred. Bond et al. (2002) found that seven pairs of spotted owls produced an average of 1.0 offspring during a single breeding season following wildfires in California, Arizona, and New Mexico. This was similar to productivity rates found in unburned areas during long-term studies of the three spotted owl subspecies (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data). In the Sierra Nevada of California, Roberts and van Wagtendonk (2006) reported that four CSO pairs in areas that experienced extensive low-to-moderate severity fire produced eight fledglings, compared with 17 fledglings produced at 18 nests in unburned areas (i.e., burned = 18% of pairs and 32% of fledglings). The authors did not statistically analyze the apparent positive effect of low-to-moderate severity fire on productivity (note: it is possible that Roberts [2008] statistically analyzed this effect but I was unable to obtain a copy of her dissertation for inclusion in this synthesis).

Currently available studies suggest that wildfires generally have minimal short-term effects on spotted owl reproduction and that primarily low-to-moderate severity fire could positively affect reproduction (Gaines et al. 1997, Bond et al. 2002, Roberts and van Wagtendonk 2006, Clark 2007). As noted above, it might be difficult to capture fire effects on spotted owl reproduction (whether positive or negative) during short-term studies, particularly with only a single year of post-fire data (Franklin et al. 2000). In addition, it is possible that solely comparing productivity (e.g., offspring per pair) in burned and unburned areas could obscure a change in total reproduction in burned areas. Studies in Washington and Oregon reported post-fire declines in occupancy by pairs, suggesting that extensive severe fires can reduce reproductive opportunities for spotted owls (Gaines et al. 1997, Clark 2007; see below).

Occupancy

Potential wildfire effects on NSO population rates are most directly evaluated with measures of survival and reproduction. However, occupancy data are often more logistically and economically feasible to collect than are demographic data and could provide an early indication of population trends (MacKenzie 2005, Olson et al. 2005). Spotted owl occupancy is sensitive to environmental factors (Blakesley et al. 2005, Olson et al. 2005) so it is a potentially valuable measure of wildfire effects on the species. Nonetheless, occupancy data must be interpreted carefully since they can be strongly influenced by survey effort, analytical methods, and the presence of barred owls (*Strix varia*) (Olson et al. 2005).

I evaluated 10 studies of wildfire effects on spotted owl occupancy (Table 3.1). As summarized below and in Table 3.1, eight of these provided evidence of a negative effect of either severe wildfires or wildfires in general.

Two studies indicated potentially negative effects of wildfires on spotted owl occupancy but included few territories (Elliot 1985, Gaines et al. 1997). In Monterey County, California, informal yearly surveys suggested that two pairs of CSOs abandoned their territories for at least four years following a wildfire (Elliot 1985). The author did not describe the fire other than noting that it was extensive and caused substantial damage to understories and oaks in the previously occupied areas. In the eastern Washington Cascades, Gaines et al. (1997) found that two of six NSO sites were occupied one year after a predominantly moderate to severe wildfire. This was the lowest occupancy rate found during the five-year study period.

Two studies found statistically weak evidence of a negative effect of fire on spotted owl occupancy, but their methods may have precluded detection of stronger effects (Jenness et al. 2004, Roberts et al. 2011). Jenness et al. (2004) found a statistically insignificant tendency (p = 0.11) for higher occupancy rank (in ascending order: no owls, singles, pairs, reproductive pairs) in unburned sites than in paired burned sites in Arizona and New Mexico (paired sites were close to each other and had similar habitat and topography). Of the 29 paired-site comparisons, 14 (48%) had a higher occupancy rank in unburned sites, 6 (21%) had a higher rank in burned sites, and 9 (31%) were tied. In the Sierra Nevada of California, Roberts et al. (2011) found lower occupancy rates for CSOs in burned areas than in unburned areas, but the difference was not statistically analyzed (modeled-occupancy = 0.46 in burned and 0.72 in unburned). Modeling by both studies indicated that spotted owl occupancy was more strongly influenced by habitat composition or structure than by whether or not fire had recently occurred in territories. However, both studies may have underestimated the impacts of severe wildfires due to pooling of diverse fire types and severities for analysis (including prescribed fires, wildfires, and wildfires allowed to burn under prescribed conditions).

Another study found stronger evidence of a negative effect of wildfires on occupancy by CSOs. Tempel et al. (*in press*) collected occupancy data at 74 CSO territories during long-term (1993/1997-2012) density and regional studies in the central Sierra Nevada. Twelve (16%) territories experienced wildfire during the studies, including nine (12%) that were affected by a mostly-severe wildfire in 2001. The best performing model explaining site colonization during the studies included area of wildfire within estimated territories (988 ac). In this model, wildfire

had a strong negative effect on the probability of colonization, even though relatively few territories were affected by fire. Only three site colonization events were observed in burned territories during six post-fire years. However, CSOs exhibited variable responses to wildfire. For example, five of the territories affected by a largely severe wildfire in 2001 were occupied every year post-fire.

Two studies found evidence of strong declines in occupancy in areas recently burned by extensive severe wildfire but both may have been confounded by post-fire salvage logging (Keane et al. 2011, 2012, Clark et al. 2013). In southwestern Oregon, Clark et al. (2013) examined how extensive wildfires and subsequent salvage logging affected occupancy dynamics of NSO pairs. In their first analysis, the authors compared pre- and post-fire occupancy dynamics in a burned study area in the Oregon Klamath Province (n = 22) to those in an unburned area in the nearby southern Cascades (Anthony et al. 2006). Combined, high severity fire and salvage logging removed or modified 26% of suitable habitat (suitability score >25: Davis and Lint 2005 [essentially nesting, roosting, and foraging habitat; generally QMD > ca. 12 in DBH in study area] in landscapes surrounding NSO activity centers in this area. The burned and salvage-logged study area experienced a 64% reduction in site occupancy during the post-fire period, compared with a 25% reduction in the unburned study area (difference not statistically analyzed). In the second analysis, the authors examined possible effects of severe wildfire and salvage logging on occupancy dynamics in 40 territories located in three burned study areas in the Oregon Klamath Province. In these areas, 19-26% of suitable habitat was burned at high severity and/or salvage logged. During the study's three-year post-fire period, site extinction probabilities were as high as 72% in two combined study areas and 92% in the third area. Site extinction probabilities in the burned study areas were best explained by a model that included extents of high severity fire, salvage logging, and early seral forest. Models that included these variables separately were not competitive with the model containing all three variables, suggesting that NSO occupancy declined due to cumulative habitat loss from severe fire and pre- and post-fire timber harvesting (see Table 6 in Clark et al. 2013). The relative influence of these factors on occupancy is unknown, but the role of severe wildfire cannot be dismissed. For example, the highest extinction probability (92%) occurred in a study area with little salvage logging (<2%) of previously suitable habitat.

Keane et al. (2011, 2012) estimated occupancy of CSOs in two recently burned study areas near the margin of the Sierra Nevada and southern Cascades of California. One wildfire complex, and an unreported amount of post-fire salvage logging, resulted in an almost complete loss of potentially suitable CSO habitat in the area (forest with mean canopy cover $\geq$40% and mean DBH $\geq$11 comprised 70% of the area pre-fire vs. 6% post-fire). Pre-fire occupancy in this study area was unknown but the Forest Service identified 23 CSO activity centers in the area prior to the fires. Rigorous landscape survey coverage by Keane et al. (2011, 2012) confirmed occupancy in only one territory within the fire perimeter during each of two post-fire years, whereas approximately seven to nine territories were found post-fire in a surrounding one-mile survey buffer (total survey area and buffer survey area sizes not reported). The other area studied by Keane et al. (2011, 2012) primarily burned at low severity (ca. 60% of the area). Pre-fire occupancy was likewise unknown in this area but Forest Service pre-project surveys indicated the presence of about 10 territories. Surveyors confirmed occupancy of six territories in this area during the first and second years post-fire. However, the number of occupied

territories in this study area could have been higher during the second post-fire season as survey effort was hindered by safety concerns associated with extensive illegal marijuana cultivation (see Ch. 5 for further discussion of this topic). While the study's findings are preliminary and may have been influenced by post-fire salvage logging, they suggest that effects of large wildfires on CSOs are dependent on the extent of high severity wildfire.

Another study provided further evidence that effects of wildfires on spotted owl occupancy depend on the extent and location of high severity wildfire. Lee et al. (2013) compared occupancy dynamics of CSOs in 71 recently burned sites and 97 unburned sites in the San Bernardino Mountains and San Jacinto Mountains of southern California. An average of 23% of the forest within burned "core areas" (500 ac around activity centers) experienced high severity fire (this percent is based on an assumption that the amount of pre-fire forest in burned core areas was the same as that reported for burned and unburned core areas combined). Mean annual probability of occupancy was 0.48 in unburned sites and 0.31 in burned sites. This difference was not statistically significant. However, Lee et al. (2013) did detect a statistically significant negative effect on occupancy when high severity fire burned more than 125 acres of forest within estimated core areas.

Two studies found that wildfires had little or no effect on spotted owl occupancy (Bond et al. 2002, Lee et al. 2012). Bond et al. (2002) calculated site fidelity for spotted owls in 11 territories burned by wildfires in California, Arizona, and New Mexico. The fires burned most of the area within each estimated territory (territory size = half the nearest neighbor distance in each study area, based on previous studies). Half of the eight territories for which fire severity was mapped primarily burned at low to moderate severity (<36% high severity) and the other half experienced moderate to extensive amounts of severe fire (36-88%). Of 21 color-banded owls in the study, 18 (86%) were resighted the year after the fires and 16 (89%) of these were located in their pre-fire territory. Site fidelity in this study was comparable to that in long-term studies of the three subspecies in unburned areas (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data). In the Sierra Nevada, Lee et al. (2012) compared post-fire occupancy in 41 recently burned and 145 unburned historical CSO territories. An average of 32% of forest in burned territories experienced high severity fire. The authors found no significant association between CSO occupancy and whether or not territories had recently experienced wildfire within a 494-acre circle around activity centers (mean occupancy was 0.76 at unburned sites and 0.80 at burned sites).

The studies reviewed above are not directly comparable due to differences in methods, spotted owl subspecies and populations, fire extents and severities, and the presence or absence of post-fire salvage logging. The preponderance of evidence suggests that spotted owls in dry, fire-prone forests are generally resilient to wildfires (Bond et al. 2002, Jenness et al. 2004, Roberts and van Wagtendonk 2006, Keane et al. 2011, 2012, Roberts et al. 2011, Lee et al. 2012, 2013). However, wildfires that severely burn large areas of potentially suitable habitat can substantially impact spotted owl occupancy, particularly when they occur in breeding-season core areas (Elliot 1985, Gaines et al. 1997, Clark et al. 2013, Keane et al. 2011, 2012, Lee et al. 2013, Tempel et al. *in press*). Post-fire salvage logging appears to increase the negative effects of extensive severe wildfires on spotted owl occupancy, most likely by reducing suitability of burned areas for prey and foraging (Lee et al. 2012, 2013, Clark et al. 2013, Comfort 2013; reviewed below).

Home Range Size and Habitat Use

Changes in the behavior of individual spotted owls may provide insight into the mechanisms by which wildfires affect populations. For example, post-fire changes in home range size may reflect wildfire effects on spotted owl energy budgets through changes in travel distances and prey availability. Changes in energy intake and output could, in turn, influence survival, reproduction, and occupancy of spotted owls. Patterns of habitat use may also be informative. For example, selection or avoidance of burned areas may reflect changes in availability of prey or roosting habitat, which could, in turn, influence occupancy, reproduction, or survival of spotted owls.

To my knowledge, only two studies have evaluated spotted owl home range sizes in relation to wildfires (Clark 2007, Bond et al. 2013; Table 3.1). Clark (2007) found that annual home range sizes of NSOs inside two fire perimeters in the Oregon Klamath Province were larger after wildfires than before them (n = 14 owls pre-fire and 20 post-fire). He attributed this difference to owls expanding their home ranges in response to habitat fragmentation caused by severe wildfire and post-fire salvage logging. This hypothesis is supported by other research in the region, which found that NSOs had larger home ranges in fragmented forests than in areas with larger, more intact patches of habitat (Carey et al. 1992, Schilling et al. 2013). Another study in the region suggested that the energetic cost of increased travel in fragmented forest was greater than the energetic benefit of increased access to prey associated with early-successional habitats (Carey and Peeler 1995).

Bond et al. (2013) compared the breeding season home ranges of seven CSOs (from four territories) during a single post-fire year in the Sierra Nevada of California with those in other studies during the same year in other parts of the subspecies range (D. Call, T. Munton, and G. Zimmerman unpubl. data). An average of 23% of forest burned at moderate severity and 9% at high severity within a 1.2 mile radius of the four nests. Pre-fire home range sizes were unknown but CSOs in the four territories did not appear to have unusually large home ranges following predominantly low to moderate severity wildfire.

At least five studies have described patterns of habitat use by spotted owls in burned areas (King et al. 1997, Clark 2007, Bond et al. 2009, Comfort 2013, Eyes 2014; Table 3.1).

King et al. (1997; also Bevis et al. 1997) described initial effects of wildfires on NSOs in two territories in the eastern Washington Cascades. One territory primarily experienced low to moderate severity fire and the other mostly burned at high severity. Both territories experienced an unreported amount of salvage logging in “unsuitable” or severely burned habitat. Most NSO locations (84% and 89%) in the two territories were daytime roosts. In the territory primarily burned at low to moderate severity, 80% of the pair’s post-fire locations were in unburned habitat, 16% were in low severity burns, and 4% were in moderate severity burns. The pair did not appear to roost in severely burned areas. The second territory studied by King et al. (1997) was occupied by a single male. After the wildfire, the male shifted his activity to an unburned area two to three miles away but continued to occasionally use areas near his former activity center. Of those locations, 74% were in unburned habitat, 17% were in low severity burns, 5% were in moderate severity burns, and 4% were in high severity burns. Maps of burn severity

classes and NSO locations indicate that owls in these two territories strongly selected unburned areas for roosting.

Clark (2007) evaluated habitat selection by 12 NSOs (7 territories) inside a wildfire perimeter in the Oregon Klamath Province. NSO locations were primarily nocturnal and may therefore, have largely represented foraging activity. Individuals in this area used all habitat classes, including moderate and severe burns and areas that had been salvage logged. However, when the data from individuals were pooled for analysis, the owls exhibited a strong preference for nesting-roosting habitat (suitability score >50: Davis and Lint 2005 [QMD generally > ca. 27 in DBH in study area]) that was unburned or burned at low severity (unburned and low severity were combined into a single class for analyses). NSOs in the study also selectively used moderately burned, previously-suitable habitat; although both use and availability of this habitat class were low compared with unburned or lightly burned habitat. Owls' use of burned areas was concentrated closer to activity centers, which was expected, given that spotted owls are central place foragers during the breeding season (see Rosenberg and McKelvey 1999).

In the Sierra Nevada, California, Bond et al. (2009) described the habitat associations of seven CSOs from four territories during a single post-fire season. Of the four nests found during the study, one was approximately 0.3 mile outside the fire perimeter, one was in forest burned at low severity, and two were in forest burned at moderate severity. One of the two nest trees found in moderate severity burns was apparently killed by the fire and one produced the only fledgling detected during the study. It is unclear from the paper whether these events occurred at the same nest or different nests. The four pairs roosted in all burn severity classes but exhibited statistically significant selection of low severity burns and avoidance of moderate and high severity burns. Only one of 60 roost sites was located in a high severity burn. Burned roost sites generally resembled unburned roost sites (>60% canopy cover and large-diameter trees). Bond et al. (2009) also evaluated CSO selection of foraging habitat in the area. Probability of use for foraging was highest when sites were burned and within 0.6 mile of nests or roosts. Probability of use was also positively associated with presence of edge between burn severity classes. Five of the owls foraged in high severity burns within 0.9 mile of nests or roosts more often than in other burn severity classes. Bond et al. (2009) suggested that CSOs in these four territories selectively foraged in high severity burns in order to access abundant prey in those areas. This hypothesis was supported by their finding that high severity burns had the highest herb and shrub cover and highest basal area of snags of any burn severity class, including unburned. These features are key resources for spotted owl prey communities (Carraway and Verts 1991, Carey et al. 1999, Holloway and Smith 2011).

Comfort (2013) evaluated habitat selection (roost and foraging locations combined) by 23 NSOs in a burned area in the Oregon Klamath Province. Her best performing model for explaining habitat selection included habitat suitability, disturbance severity, high contrast edge, and low contrast edge. Habitat selection varied with spatial scale but NSOs exhibited a strong preference for higher habitat suitability and avoidance of patches affected by higher severity disturbance (high severity wildfire and/or salvage logging). NSOs showed a preference for high contrast edge at small spatial scales (2-8 ac) and avoidance at medium and large scales (32-2,049 ac). NSOs also exhibited a weak preference for low contrast edge. Comfort (2013) concluded that patchy, mixed severity fire (small patches of high severity fire within a matrix of unburned and

low-to-moderate severity fire) created conditions favored by NSOs in her study, whereas large patches created by high severity fire and subsequent salvage logging were strongly avoided. Salvage logging apparently contributed to conditions avoided by NSOs by structurally homogenizing burned areas, which increased the sizes of high severity patches and amounts of high contrast edge (Comfort 2013). However, the relative influence of high severity wildfire and post-fire salvage logging on habitat selection by NSOs in this study is unknown.

Eyes (2014) evaluated foraging habitat selection by 13 CSOs (8 territories) during three breeding seasons in a recently burned landscape (1-15 yrs prior) in Yosemite National Park in the Sierra Nevada. On average, 25% of the home range (minimum convex polygon) had recently experienced low severity fire, 16% moderate severity fire, and 4% high severity fire. Three of Eye's (2014) four best performing models explaining habitat selection by foraging CSOs included a fire severity metric (Fire Severity Index). These models indicated that the probability of an area's use by foraging CSOs decreased with increasing fire severity. However, foraging CSOs were more likely to use edge sites than non-edge sites and exhibited a tendency for greater use of high contrast edges created by severe fire than for lower contrast edges created by low/moderate severity fire or other disturbances. Eye's (2014) findings that CSOs avoided the interiors of high severity burns and favored high contrast edges created by severe fire is consistent with Comfort's (2013) findings at smaller spatial scales around NSO locations.

The limited available information concerning spotted owl habitat use following wildfires indicates that the species avoids roosting in moderate and high severity burns (King et al. 1997, Clark 2007, Bond et al. 2009). This finding is concordant with the spotted owl's close association with densely-canopied older forest for roosting (Blakesley 2004). Little is known about the effects of wildfire on selection of nest sites. Bond et al. (2009) found three CSO nests in forest recently burned at low and moderate severity, and young fledged from one nest in a moderate severity burn. Moderate severity fire killed one of the four CSO nest trees in their study. Eyes (2014) found a CSO nest adjacent to a high severity burn but the nest failed during her study. Based on the species' nesting habitat requirements (Blakesley 2004), long-term use of severely burned areas for nesting is likely uncommon. Three studies specifically examined selection of foraging habitat by spotted owls. All three found use of all burn severity classes, but Clark (2007) and Eyes (2014) found a preference for foraging in unburned to moderately burned areas (also see Comfort 2013, which combined foraging and roost locations in analyses) while Bond et al. (2009) found a preference for severe burns. It is unclear if this difference was due to differences in the studies' methods, effects of fire and timber harvesting on vegetation, or the composition of prey communities and spotted owl diets. Findings by Comfort (2013) and Eyes (2014) suggest that foraging spotted owls avoid large patches recently burned by high severity fire but benefit from some amount of high contrast habitat edge created by patchy high severity fire.

Wildfire Effects on Prey

In New Mexico, Ganey et al. (2014) found that species richness, relative abundance, and biomass of small mammals were greater in four MSOs' burned wintering areas than in their nest core areas. Abundances of deer mice, pocket gophers (*Thomomys* spp.), and other "pioneer" or "early-successional" prey often increase following fires (Ream 1981, Zwolak and Foresman

2007, Zwolak 2009, Bond et al. 2013). Dusky-footed woodrats appear to initially respond negatively to severe fires (Schwilk and Keeley 1998, Smith 2000) but not to patchy low severity fire, although loss of nest houses might have a brief negative effect on reproduction (Lee and Tietje 2005). However, it is possible that severe fire benefits dusky-footed woodrats over longer time periods (e.g., ≥5-20 yrs) through creation of brushy habitat. Stand-replacing fires should negatively affect abundances of prey associated with well-canopied forest, such as northern flying squirrels and tree voles. These taxa, along with dusky-footed woodrats, are the primary prey for NSOs in California (reviewed in Ch. 2). Low severity fires could also have negative effects on northern flying squirrels and other prey associated with closed canopy forests by reducing dead woody materials, fire-intolerant understory plants, and truffles (Lehmkuhl et al. 2006, Meyer et al. 2007). Thus, wildfires likely have complex effects on NSO prey communities, depending on local or regional differences in prey community composition; wildfire size, severity, and configuration; and the length of time vegetation has had to regenerate following fire.

Post-Fire Salvage Logging

While salvage logging might be judiciously used to meet certain conservation objectives (e.g., generating downed wood to minimize erosion or create wildlife habitat), it is generally conducted to meet economic goals or remove hazard trees (Peterson et al. 2009). Intensive or poorly planned salvage logging can have a variety of negative effects on ecosystems, such as soil compaction, increased erosion, and impacts on insectivorous and cavity-nesting and -denning animals (reviewed in McIver and Starr 2000, Noss et al. 2006, Peterson et al. 2009).

At least three studies have directly evaluated effects of post-fire salvage logging on spotted owls (Clark 2007 and Clark et al. 2013, Lee et al. 2012, 2013). Clark (2007) conducted a radio-telemetry study in areas recently burned by wildfires in the Oregon Klamath Province. He recorded limited use of salvage logged areas; presumably for foraging since locations were primarily nocturnal. Use of salvage logged areas was slightly lower than expected based on its abundance in territories (not statistically analyzed), indicating weak avoidance of salvage logged areas by NSOs. However, avoidance might have been stronger since some of the study's telemetry locations were potentially recorded prior to the occurrence of salvage logging. Most (60%) NSO locations in salvage logged areas occurred in riparian buffers, thinned areas, and patches of wildlife leave trees, rather than intensively salvaged areas. During the same study, Clark et al. (2013) found that post-wildfire declines in NSO occupancy were best explained by a model that included extents of pre-fire timber harvesting, severe wildfire, and post-fire salvage logging. Models that included these factors separately were not competitive with this model, indicating that severe fire and pre- and post-fire harvesting collectively contributed to declines in NSO occupancy; most likely through cumulative habitat loss or degradation.

In the Sierra Nevada, Lee et al. (2012) recorded occupancy for eight CSO territories that experienced wildfire and post-fire salvage logging. Seven of the territories were occupied during the two-year period between the occurrence of wildfire and post-fire salvage logging, whereas none of the territories were occupied following salvage logging.

Lee et al. (2013) evaluated effects of salvage logging on CSOs in the San Bernardino and San Jacinto Mountains of Southern California. They noted that salvage logging in their study area was modest compared with commercial salvage logging typically employed in the Pacific Northwest and Sierra Nevada (salvage logging in their study area mostly consisted of firewood cutting on private in-holdings and hazard tree removal along Forest Service roads). Lee et al. (2013) did not find a statistically significant effect of post-fire salvage logging on CSO occupancy dynamics. However, site extinction probability was slightly higher, and mean annual probability of occupancy was slightly lower, in salvage logged areas than in other burned areas. Weak negative effects of light salvage logging were apparent during all eight post-fire study years.

The limited available evidence suggests that salvage logging decreases the probability that spotted owls will use burned areas (Comfort 2013) and increases the probability that they will abandon their territories following wildfires (Clark 2007, Lee et al. 2012, 2013, Clark et al. 2013). This could occur because salvage logging reduces suitability of burned areas for foraging spotted owls and their prey. Stands recently burned by moderate or severe fire often contain high biodiversity due to the presence of both early-successional conditions and key biological legacies in the form of snags, logs, and residual live trees (Noss et al. 2006). Due to fire suppression and salvage logging, stands with these conditions are currently rare in many fire-prone forests within the spotted owl's range (Noss et al. 2006). Selective use of burned areas for foraging is likely due to spotted owls exploiting short-term increases in prey associated with both early-successional vegetation (e.g., shrubs) and legacy habitat elements (e.g., large diameter snags, logs, and live trees) (Ream 1981, Zwolak 2009, Bond et al. 2013). Salvage logging removes legacy elements, while associated use of herbicides reduces shrubs and grasses important to many prey species (Bond et al. 2013, Comfort 2013). In the longer-term, spotted owls can continue to benefit from the contributions of legacy habitat elements to regenerating stands. For example, large legacy snags, trees, and logs can provide valuable habitat elements for northern flying squirrels and other prey (Holloway and Smith 2011). Removal of these elements through salvage logging could therefore reduce the value of subsequent regenerating stands as prey habitat. Harvesting of legacy snags and live trees could also directly affect spotted owls by reducing availability of foraging perches in the short-term and suitable nest trees during later successional stages.

Summary of Direct Evidence Concerning Wildfire and Salvage Logging Effects

Inferences from studies of direct effects of wildfires and salvage logging on spotted owls are limited due to inclusion of only a small number of spotted owls or territories. The preponderance of currently available evidence indicates that spotted owls are often resilient to low-, moderate-, or mixed-severity wildfires but can be strongly impacted by extensive severe wildfires. Following wildfire, many spotted owls may remain in their territories, exploit short-term increases in prey in burned areas, and continue to reproduce at reasonably high rates. However, wildfires that result in substantial loss or fragmentation of suitable habitat, particularly within breeding core areas, can cause spotted owls to increase their home range sizes, abandon their territories, and possibly, emigrate from burned landscapes or die of starvation or disease. Negative effects of severe wildfires appear to be greatest when suitable habitat is already limited

(e.g., due to widespread intensive timber harvesting) and when post-fire salvage logging reduces suitability of burned areas for foraging and prey.

Wildfire Effects on Recent Habitat Trends

Past and continuing habitat loss was a primary reason for listing the NSO under the federal Endangered Species Act (USFWS 1990). At the time of listing, the Fish and Wildlife Service estimated that 60-88% of the subspecies' habitat had already been lost (USFWS 1990; also see Ch. 2). They attributed most of this loss to widespread intensive timber harvesting. Since listing of the NSO and subsequent adoption of the Northwest Forest Plan, timber harvesting has declined and wildfire has been identified as the primary source of forest disturbance and habitat loss on federal lands (Courtney et al. 2004, Healey et al. 2008, Davis and Dugger 2011, Moeur et al. 2011, USFWS 2011a, Kennedy et al. 2012). Nonetheless, timber harvesting continues to be the primary source of habitat loss on non-federal lands (Healey et al. 2008, Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012; see below).

Estimates of recent trends in amounts of NSO habitat, and of older forest in general, have been produced as part of monitoring efforts for the Northwest Forest Plan; and are therefore, largely restricted to the Plan's area and time span (Davis and Lint 2005, Moeur et al. 2005, 2011, Healey et al. 2008, Davis and Dugger 2011). I have focused on estimates by Davis and Dugger (2011) because they replaced those of Davis and Lint (2005) and are more specific to NSO habitat than those of Moeur et al. (2005, 2011) and Healey et al. (2008). I did not review habitat trend estimates based on federal ESA Section 7 consultation records (Bigley and Franklin 2004, USFWS 2012b) due to greater scientific uncertainty and methodological bias associated with those data (see Bigley and Franklin 2004). Trends described by Bigley and Franklin (2004), Moeur et al. (2005, 2011), Healey et al. (2008), and the US Fish and Wildlife Service (2012b) quantitatively differ from those of Davis and Dugger (2011) but similarly indicate that wildfires have been the primary source of recent habitat loss on federal lands.

Davis and Dugger (2011) used remotely sensed vegetation data (satellite imagery) to model changes in habitat suitability across the NSO's range during the first 15 years of the Northwest Forest Plan (1994-2007 in California and 1996-2006 in Oregon and Washington). Habitat suitability was based on characteristics surrounding thousands of NSO pair locations across the Plan area. Suitable breeding habitat was defined as having both a probability of owl presence greater than expected based on random chance and environmental conditions typical of those found around nesting and roosting pairs. Estimated habitat trends included gross loss of both suitable breeding ("nesting/roosting") habitat and interior ("core") suitable breeding habitat (>330 ft from edge). Davis and Dugger (2011) considered habitat loss to have occurred when an area classified as suitable at the beginning of Northwest Forest Plan was later downgraded to a lower suitability rank (unsuitable or marginal) due to vegetation changes caused by forest disturbances. Davis and Dugger (2011) did not estimate recruitment of, or net changes in, breeding habitat because their remotely sensed data was incapable of accurately capturing relatively slow and subtle habitat changes during development of intermediate-aged and older stands. They did, however, estimate net trends in NSO dispersal habitat, which they defined as forest with a mean conifer canopy cover of at least 40% and a mean conifer DBH of at least 11 inches. Recruitment of dispersal habitat was more detectable than that of breeding habitat due to

more rapid and measurable growth in younger forest (some recruitment also occurred due to degradation of suitable breeding habitat by forest disturbances).

Estimated gross losses of suitable breeding habitat on federal and non-federal lands are presented in Tables 3.2 and 3.3, respectively. During the first 15 years of the Northwest Forest Plan, wildfires were responsible for an estimated gross loss of 236,700 acres (2.7%) of suitable breeding habitat on federal lands rangewide and 13,100 acres (0.3%) on non-federal lands (1.9% of federal and non-federal lands combined). Estimated habitat loss on federal lands was similar to that expected at the time of the Northwest Forest Plan's implementation; however, relatively high rates of habitat loss in relatively dry, fire-prone regions have been a source of conservation concern for NSOs in those areas (Davis and Dugger 2011). In California, wildfires removed an estimated 75,500 acres (4.1%) of suitable breeding habitat on federal lands and 5,600 acres (0.4%) on non-federal lands (2.4% combined). Approximately 70% of habitat loss to wildfires on federal lands occurred within the Oregon and California Klamath Provinces (Table 3.2). Most of this habitat loss was caused by the 1999 Megram Fire and 2002 Biscuit Fire (Table 3.4). Fires in the Eastern Cascades Provinces of Washington, Oregon, and California contributed less to total habitat loss than did fires in the Klamath Provinces, but were often more destructive in terms of proportion of suitable habitat lost within individual fire perimeters. In contrast with federal lands, wildfires were responsible for very little habitat loss on non-federal lands; rather, timber harvesting accounted for most losses in these areas (Figure 3.1).

Table 3.2: Estimated gross losses of suitable NSO breeding habitat on federal lands due to wildfires during 1994-2007 in California and 1996-2006 in Oregon and Washington (adapted from Davis and Dugger 2011, USFWS 2011a).

State	Province	Initial Acres	Acres Lost	Percent Lost
California	CA Cascades	213,200	1,800	0.8%
	CA Klamath	1,489,800	71,600	4.8%
	CA Coast	145,400	2,100	1.4%
Oregon	OR Coast Range	611,200	0	0.0%
	Western OR Cascades	2,258,700	28,900	1.3%
	Eastern OR Cascades	402,900	17,800	4.4%
	Willamette Valley	3,400	0	0.0%
	OR Klamath	985,000	93,600	9.5%
Washington	Olympic Peninsula	763,100	200	0.0%
	Eastern WA Cascades	673,600	20,000	3.0%
	Western WA Cascades	1,283,000	700	0.1%
	Western WA Lowlands	24,700	0	0.0%
Rangewide Total		8,854,000	236,700	2.7%

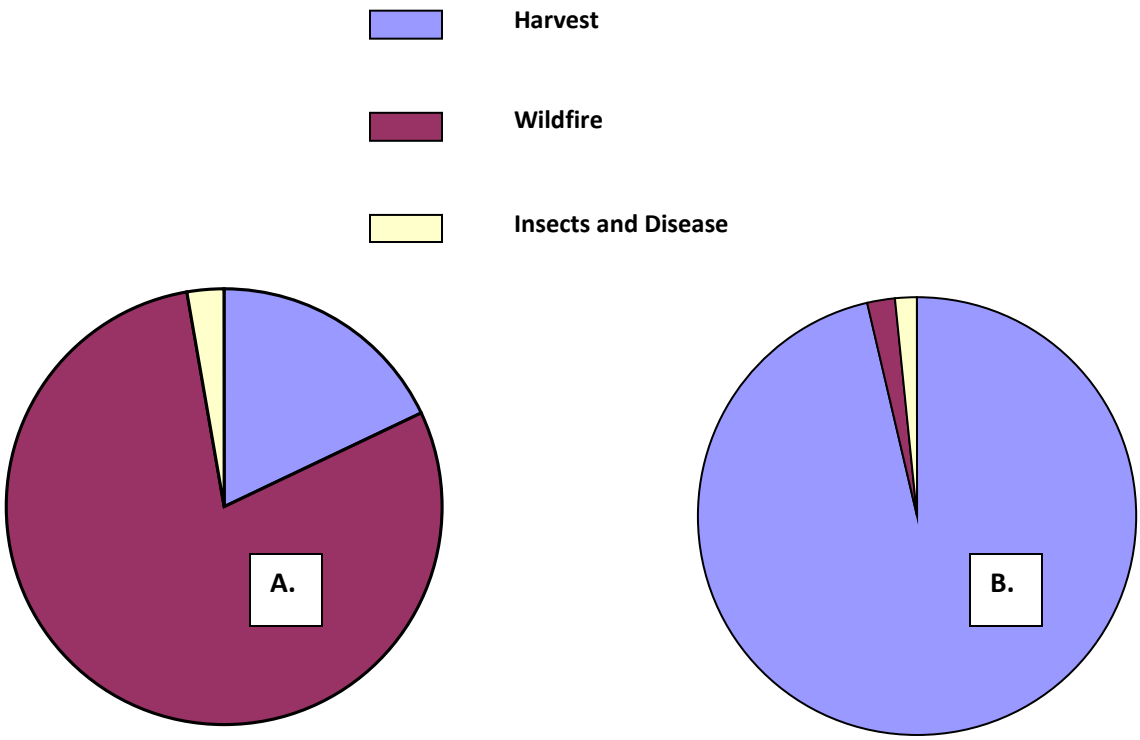
Table 3.3: Estimated gross losses of suitable NSO breeding habitat on non-federal lands due to wildfires during 1994-2007 in California and 1996-2006 in Oregon and Washington (adapted from Davis and Dugger 2011, USFWS 2011a).

State	Initial Acres	Acres Lost	Percent Lost
California	1,556,700	5,600	0.4%
Oregon	1,382,400	5,100	0.4%
Washington	1,258,900	2,400	0.2%
Total	4,198,000	13,100	0.3%

Table 3.4: Estimated gross losses of suitable NSO breeding habitat to individual wildfires during 1994-2007 (note: "habitat degraded" describes areas downgraded from highly suitable to suitable; from Davis and Dugger 2011).

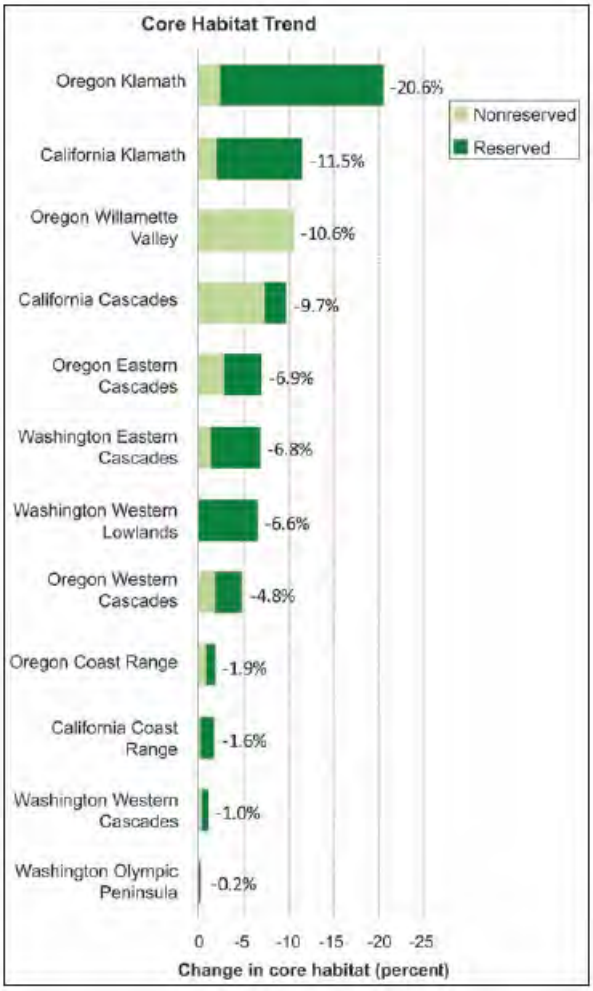
Fire name	Year	Broad physiographic province	Habitat in wildfire perimeter	Habitat lost		Habitat degraded	
			Acres	*Acres*	*%*	*Acres*	*%*
Biscuit Fire	2002	Klamath Mountains	226,230	93,730	41	12,019	5
Megram Fire	1999	Klamath Mountains	76,337	27,520	36	4,589	6
B&B Complex	2003	East Cascades	26,269	16,403	62	907	3
Bake-oven Fire	2006	Klamath Mountains	23,946	8,873	37	581	2
Boulder Fire	2002	West Cascades	34,059	8,460	25	2,074	6
Davis Fire	2003	East Cascades	8,050	6,943	86	5	0
Pigeon Fire	2006	Klamath Mountains	13,896	5,634	41	327	2
Rex Complex	2001	East Cascades	8,548	4,750	56	278	3
Timbered Rock	2002	West Cascades	10,216	4,539	44	569	6
Spring Fire	1996	West Cascades	13,504	3,931	29	858	6
Deep Harbor Fire	2004	East Cascades	5,761	3,930	68	64	1
Hancock Fire	2006	Klamath Mountains	12,712	3,132	25	336	3
Apple Fire 2	2002	West Cascades	12,227	2,810	23	928	8
Fischer Fire	2004	East Cascades	4,479	2,340	52	34	1
Fork Fire	1996	Klamath Mountains	2,962	2,113	71	14	0
Needles Fire	2003	East Cascades	1,946	874	45	1	0
Trough Fire	2001	Klamath Mountains	1,851	798	43	4	0
Hunter Fire	2006	Klamath Mountains	2,236	789	35	40	2
Deer Point Fire	2002	East Cascades	505	380	75	0	0
Tatoosh Complex	2006	East Cascades	666	378	57	0	0

Figure 3.1: Proportions of suitable breeding habitat loss attributed to harvesting, wildfire, and insects and diseases on (A) federal lands and (B) non-federal lands during 1994-2007 (adapted from Davis and Dugger 2011 and USFWS 2011a).



Davis and Dugger (2011) reported substantial losses of interior (>330 ft from edge) breeding habitat on federal lands during the first 15 years of the Northwest Forest Plan (Figure 3.2). These losses primarily occurred in reserved areas. Changes in ratios of interior and edge habitat classes indicated that increased fragmentation of suitable breeding habitat on federal lands was greatest in the Oregon and California Klamath Provinces and California Cascades Province (see Table 3-3 in Davis and Dugger 2011). Increased fragmentation in these regions was primarily due to wildfires.

Figure 3.2: Gross losses of interior suitable breeding habitat on reserved and non-reserved federal lands during 1994-2007 in California and 1996-2006 in Oregon and Washington (from Davis and Dugger 2011).



Davis and Dugger (2011) estimated a 5.2% net gain in NSO dispersal habitat. Much of this gain was due to succession in young forests in non-reserved lands at the margins of federal forests. However, accounting for forest connectivity and NSO dispersal distances, Davis and Dugger (2011) reported a 1% net loss of "dispersal-capable landscape". Much of the loss of dispersal-capable landscape in the Klamath and Eastern Cascades Provinces was due to large wildfires,

whereas timber harvesting on non-federal lands was responsible for much of the loss in other regions.

Loss and fragmentation of breeding habitat to wildfires was likely at least partially offset by recruitment of new habitat through succession of mature and old forest. However, Davis and Dugger (2011) found that detectable recruitment of breeding habitat primarily occurred in the marginal suitability class. This finding was supported by Moeur et al. (2005), who found that about 90% of recruitment of older forest (mature and old-growth combined) during the first 10 years of the Northwest Forest Plan was at the lower end of the class' diameter range (i.e., mean DBH 20-30 in). In their subsequent report, Moeur et al. (2011) noted that given the short length of the monitoring period (10-14 yrs), recruitment was "likely due to incremental stand growth over the 20-in diameter threshold, or from understory disturbances that removed smaller diameter trees and raised the average stand diameter above the threshold, rather than from an increase in forests of much larger and older trees." It is likely that some newly recruited mature forest provides suitable habitat for NSOs but much if it could lack the canopy layering, large diameter snags and logs, and other structural attributes typical of nesting and roosting habitat (Blakesley et al. 2004).

Loss and fragmentation of suitable breeding habitat does not necessarily equate to negative impacts on NSOs (Franklin and Gutiérrez 2002; e.g., Bond et al. 2002, 2009, Lee et al. 2012). Studies in southern Oregon and northern California found that the presence and fitness of NSOs are generally highest in landscapes with a mix of both large amounts of suitable breeding habitat and other habitat classes, such as foraging habitat or "non-habitat" (Franklin et al. 2000, Zabel et al. 2003, Olson et al. 2004, Dugger et al. 2005; reviewed above and in Ch. 2 [see Ch. 2 for studies' habitat definitions]). Fitness is also generally highest when suitable breeding habitat occurs in large or clustered patches with large amounts of ecotone or edge between vegetation classes (Franklin et al. 2000, Dugger et al. 2005). This combination of conditions is important because it provides NSOs with a balance of resources needed for both survival and reproduction (Franklin et al. 2000). Active fire regimes in dry forests within the NSO's range in California (e.g., mixed-conifer and interior mixed-evergreen) historically contributed to these conditions by generally sparing older forest and maintaining some form of habitat heterogeneity at both stand and landscape scales (Skinner and Taylor 2006, Skinner et al. 2006, Stuart and Stephens 2006). Some contemporary wildfires may still burn in this manner and thereby continue to perform an important ecosystem function in these forests. However, large severe wildfires have contributed, along with fire suppression and timber harvesting, to homogenization of some forests in interior northern California (Skinner et al. 2006). Thus, large severe wildfires may impact NSOs in California through loss of habitat heterogeneity, as well as reduced amounts and connectivity of suitable breeding habitat.

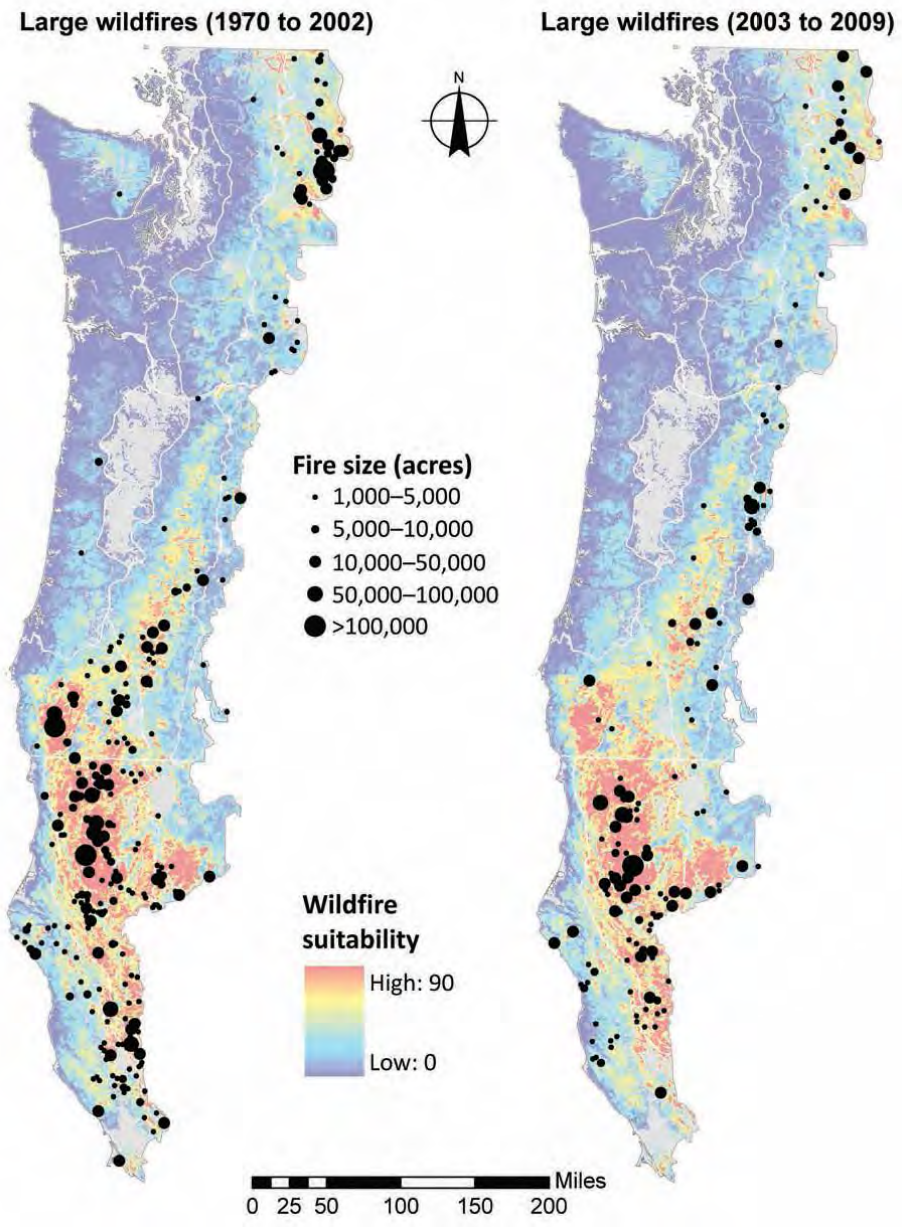
Fire Risk in California

Prior to Euro-American settlement, dry forests in California generally experienced relatively frequent, low-to-moderate or mixed severity fire regimes (Skinner and Taylor 2006, Skinner et al. 2006, Stuart and Stephens 2006, Van de Water and Safford 2011). Mean pre-settlement fire return intervals in California were 11 years in yellow pine forests (e.g., *Pinus ponderosa*, *P. jeffreyi*), 11-16 years in mixed-conifer forests, and 29 years in mixed-evergreen forests (Van de

Water and Safford 2011). Mean fire return intervals in redwood forests south of Del Norte County were also relatively frequent (6-44 yrs: Stuart and Stephens 2006). Frequent fire during the pre-settlement period generally maintained forests with less dense and more clumped tree distributions, higher proportions of fire-resistant trees (i.e., larger size classes and more fire-tolerant species), and lighter and less continuous fuel beds than occur today (Skinner and Taylor 2006, Skinner et al. 2006, Stuart and Stephens 2006). In northern California, this characteristic fine-scale structural heterogeneity was often overlaid with coarser variability created by effects of elevation, terrain, soils, and other physiographic factors on fire and vegetation patterns (Skinner and Taylor 2006, Skinner et al. 2006, Sawyer 2007). For example, in areas of deeply incised terrain in the Klamath Mountains and southern Cascades of California, upper slopes and south and west facing aspects typically experienced more frequent and severe fire than did other areas (Skinner and Taylor 2006, Skinner et al. 2006). Due to fire suppression, early-successional vegetation communities formerly maintained by frequent, small-scale severe wildfire have greatly declined in some areas of California (Skinner 1995, Nagel and Taylor 2005). Overall, research indicates that fire suppression and other human activities have led to decreased forest heterogeneity at both stand and landscape scales and have contributed to substantial changes in fire regimes in California's dry forests (Skinner and Taylor 2006, Skinner et al. 2006, Stuart and Stephens 2006).

Due to fire suppression, fire-free periods have dramatically increased in California's interior forests (Skinner and Taylor 2006, Skinner et al. 2006). For example, fire rotation near Hayfork in the Klamath Mountains increased more than 10-fold (from 19 to 238 yrs) during the post-settlement period (Taylor and Skinner 2003). Abnormally long fire-free periods have facilitated increased accumulation and continuity of fuels in dry, fire-prone forests (Skinner and Taylor 2006, Skinner et al. 2006, Stuart and Stephens 2006). When wildfires do occur there is often an increased risk of them becoming very large and for suppression forces to be overwhelmed by their size and number (e.g., CAL FIRE 2008). As in other dry forests across the western U.S., the mean and maximum sizes of wildfires, and the total annual area burned, significantly increased in California's dry montane forests during the 20th and early 21st Centuries (Miller et al. 2009, 2012). CAL FIRE (2008) noted that more than half of the 26 largest fires recorded in California during 1932-2008 occurred during the last eight years of that period. Based on recent (1970-2002) fire behavior, most of northern California's interior can be classified as highly prone to large (>1,000 ac) wildfires (Davis et al. 2011; Figure 3.3).

Figure 3.3: Relative probability of large (>1,000 ac) wildfires ("wildfire suitability") across the NSO's range (from Davis et al. 2011). Modeling was based on landscape and climatic characteristics of locations at which large wildfires occurred during 1970-2002 (left) and was compared with subsequent (2003-2009) locations (right).



Studies of recent trends in extents of high severity wildfire in California have found conflicting results. For example, Miller et al. (2009) and Miller and Safford (2012) reported a substantial increase in the extent of high severity wildfire in the Sierra Nevada and southern Cascades of California during 1984-2010; while Hanson and Odion (2014) did not find an increase in the Sierra Nevada during the same period. Similarly conflicting results have been found for the California Klamath Province. Hanson et al. (2009) found a significant increase in high severity wildfire in the California and Oregon Klamath Provinces during 1984-2005; whereas Miller et al. (2012) did not find an increase in the California Klamath Province during a similar period (1987-2008). Scientific debate ensued regarding the appropriateness of methods used in various studies to determine trends in high severity fire (e.g., Hanson et al. 2009, 2010 vs. Spies et al. 2010).

Differences in findings regarding trends in high severity wildfire are related to variation in studies' temporal and spatial scales of analysis, as well as in methods for determining fire severity (Courtney et al. 2008, Miller et al. 2012, Hanson and Odion 2014). For example, Miller et al. (2012) noted that both their own study and those of Odion et al. (2004, 2010) may have underestimated trends in high severity wildfire in the California Klamath Province due to inclusion of unusual fire years. Studies by Odion et al. (2004, 2010) were based on fire effects during a single year (1987), which Miller et al. (2012) described as unusual. Large areas burned at below-average severity during 1987 due to abnormally strong inversions, and the fact that some of the wildfires burned well into fall when conditions often favor lower severity fire. Miller et al. (2012) also noted that their own ability to detect a trend in wildfire severity could have been compromised by inclusion of both this year and 2008, which likewise experienced unusually large, low severity wildfires. Differences in the area analyzed could also affect evaluations of trends in high severity wildfire. For example, the Biscuit Fire, which was predominantly located in Oregon, included extensive areas of high severity fire and therefore could have influenced results of trends analyses for the Oregon and California Klamath Provinces combined versus the California Klamath alone (Miller et al. 2012).

Regardless of whether or not the extent of high severity wildfire increased in California during the last two decades, large severe wildfires have recently occurred in these areas, and they were responsible for most loss, degradation, and fragmentation of suitable breeding habitat for NSOs on federal lands (Davis and Dugger 2011, Davis et al. 2011; see above). These data cannot be used to project how fires in the future will affect NSOs since they do not necessarily represent past or future fire conditions or effects. However, it is highly unlikely that wildfires will cease to be a major source of habitat loss for NSOs in the future. Rather, climate change research has generally projected a continued increase in the number and sizes of wildfires and the annual area burned in California and other western states during coming decades (Westerling et al. 2006, Lenihan et al. 2008, Westerling and Bryant 2008, Littell et al. 2009, Moritz et al. 2012, Stavros et al. 2014). There is scientific uncertainty regarding recent and future trends in the extent of high severity wildfire in California. Nonetheless, large severe wildfires will at least occasionally occur in the future and will continue to be a source of habitat loss and modification for NSOs in the state.

Increases in the number and sizes of wildfires, and effects of wildfires on NSO habitat trends, have led to calls for widespread use of thinning and other forms of active management in dry, fire-prone forests within the Northwest Forest Plan area (USFS and BLM 1994, USFWS 2008,

2011, 2012a, Franklin and Johnson 2012). Some researchers and stakeholders, however, have expressed doubts regarding estimates of wildfire risk and effects on NSOs, concerns about potential effects of thinning on NSOs, and distrust of federal agency intentions (Hanson et al. 2009, Heiken 2010, DellaSala et al. 2013, Odion et al. 2014). As discussed in Chapter 2 (also Hansen and Mazurek 2010, USFWS 2011a), there is currently little known about the effects of forest thinning on spotted owls but the preponderance of evidence indicates that commercial thinning can have negative short-term effects on the species (also see Tempel et al. *in press* regarding potential long-term effects). Federal agencies should carefully consider this information, as well as apparent effects of wildfires on NSOs, when formulating land management policies and prescriptions aimed at reducing wildfire risk in landscapes occupied by the subspecies. Land managers should also consider greater use of prescribed fire and allowing wildfires to burn under favorable conditions; particularly at lower elevations in the California Klamath Province, where summertime inversions often minimize fire severity (Miller et al. 2012).

Acknowledgements

I am grateful to Ray Davis and Derek Lee for their helpful comments on earlier drafts of this chapter. I recognize that many of the topics reviewed in this chapter are scientifically or politically contentious. The approach and conclusions in this review are my own and do not necessarily reflect those of the reviewers.

Ch. 4: Barred Owls

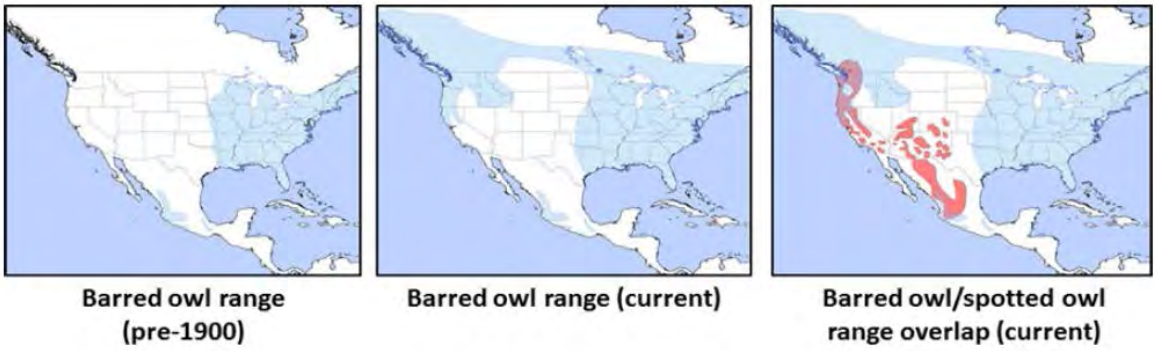
Introduction

At the time of the NSO's federal listing, the barred owl was recognized as a potential threat to the subspecies (USFWS 1990, Thomas et al. 1990). Since then, barred owls have continued to invade the range of the NSO and are apparently increasing in numbers (USFWS 2013). As reviewed herein, a large body of correlational and anecdotal evidence, supplemented by preliminary findings from barred owl removal experiments, indicates that barred owls are negatively impacting NSO populations across their range and that this is due to competition between the two species for space, habitat, and food. Research reviewed below indicates that the barred owl is a superior competitor to the NSO due to its larger size, more aggressive behavior, higher reproductive potential, higher population densities, and broader ecological niche (e.g., USFWS 2013, Wiens et al. 2014). The USFWS (2011a) recently listed the barred owl invasion as one of three main threats currently faced by NSOs (along with timber harvesting and wildfires). The USFWS (2011a) described this threat as "extremely pressing and complex" and "requiring immediate consideration." Information reviewed below suggests that lethal control of barred owls is a viable management option for some areas, although it may be difficult to overcome emotional and ethical resistance to killing one charismatic species to save another (Diller et al. 2013, Higley 2014). Habitat conservation for NSOs appears to be of increasing importance because the negative effects of habitat loss and fragmentation on NSOs can be exacerbated by the presence of barred owls (Dugger et al. 2011).

The Barred Owl's Expansion

Prior to the mid-1900s, the barred owl's range was confined to southeastern Canada, the eastern U.S., and portions of Mexico (Gutiérrez et al. 2004, 2007; Figure 4.1). By the mid-20th century the barred owl's range began expanding westward across North America and currently includes the southern boreal forest zone and British Columbia in Canada and the northern Rocky Mountains, Pacific Northwest, and northern California in the U.S. (Livezey 2009, USFWS 2013; Figure 4.1). The barred owl's range now completely overlaps that of the NSO and partially overlaps that of the California spotted owl (*S. o. occidentalis*) (Figure 4.1). It is unclear whether the barred owl's westward range expansion occurred via the Great Plains or Canada's boreal forests (USFWS 2013). It is also uncertain whether this range expansion was facilitated by natural factors, human activities, or a combination of the two. Hypotheses concerning the cause of the barred owl's range expansion include increased adaptation by the species to coniferous forests; natural climate change; environmental changes associated with widespread intensive timber harvesting; and conversion of open areas to forest due to fire suppression, planting parks and woodlands, removal of keystone species, or other human activities (Gutiérrez et al. 2004, USFWS 2013).

Figure 4.1: Historical and current range of the barred owl and comparison of the current ranges of the barred owl and spotted owl (from USFWS 2013).



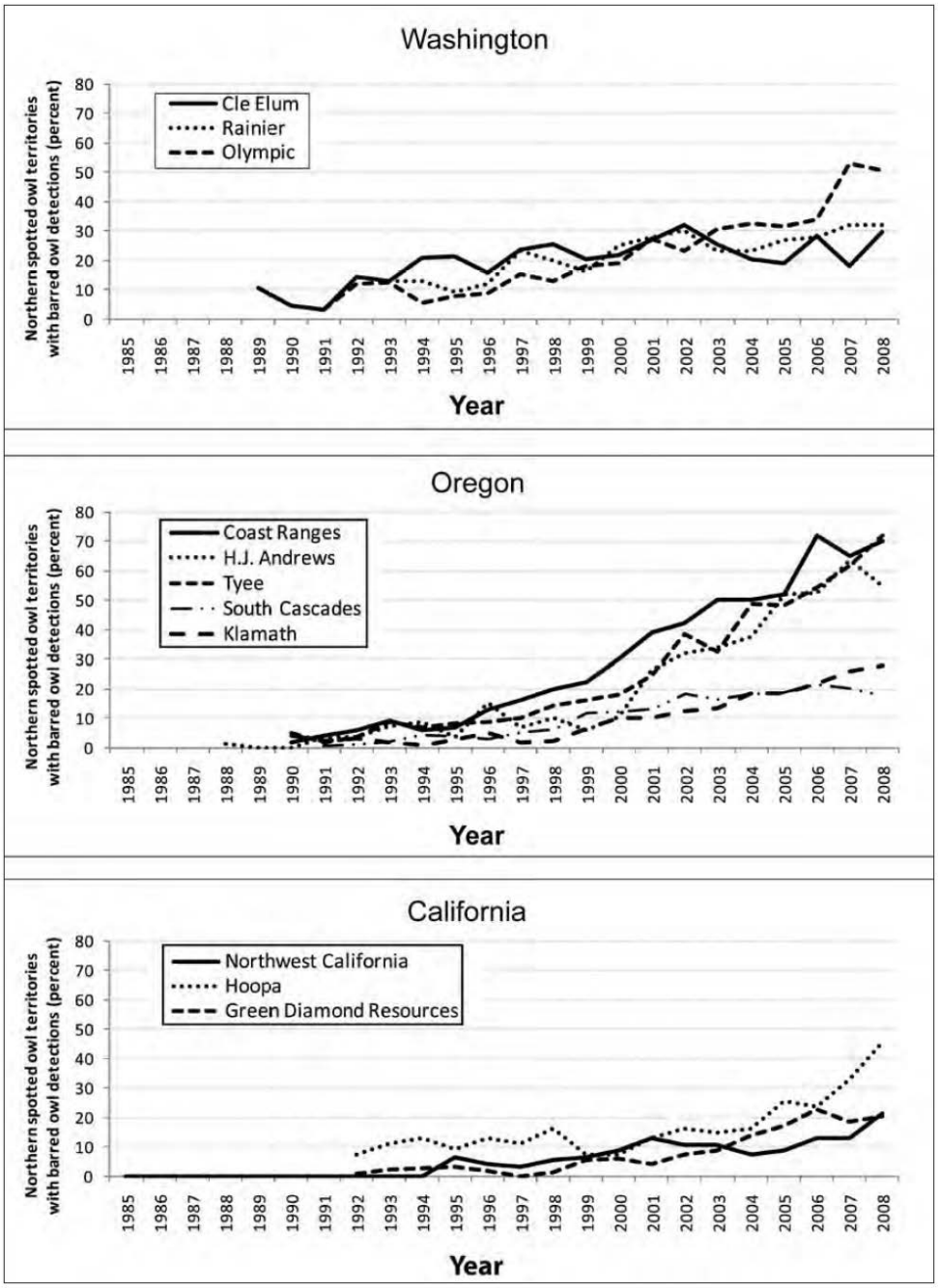
Reliable data concerning current barred owl densities and population trends are unavailable. Most information about barred owl trends in the Pacific Northwest and California is based on incidental detections during surveys for spotted owls (USFWS 2013). Researchers have estimated that only 1/2 to 2/3 of barred owls present are detected during NSO surveys (Gutiérrez et al. 2004, Bailey et al. 2009, Wiens et al. 2011). Researchers have only recently begun to systematically survey for barred owls within the range of the NSO and only in limited areas. In California, the Northwest California, Green Diamond, and Hoopa NSO demographic studies initiated systematic surveys for barred owls in 2009 (Franklin et al. 2014, Green Diamond Resource Company 2014).

Some early studies of barred owls within the NSO's range likely overestimated barred owl numbers by focusing on cumulative and nighttime detections (Gutiérrez et al. 2004). However, barred owl trends are now more frequently evaluated in terms of the number of spotted owl territories in which barred owls have been detected (USFWS 2013; e.g., Forsman et al. 2011, Calforests 2014). Reports of the number of spotted owl territories with barred owl detections probably underestimate barred owl densities and population trends (USFWS 2013). Barred owls often have substantially smaller home ranges than spotted owls so it is possible for a single spotted owl home range to encompass multiple barred owl home ranges (Singleton et al. 2010, Wiens 2012; see below). Without color-banding or follow-up surveys to determine the identity and occupancy status of barred owls, it is often unclear if multiple detections within spotted owl territories represent one or multiple barred owls. Furthermore, a lack of barred owl-specific surveys in many areas has likely led to underestimates of barred owl presence. For example, in 2013, NSO-specific surveys on the Hoopa reservation in northwestern California indicated that barred owls were present in 43% of NSO territories on the reservation, whereas barred owl-specific surveys revealed that barred owls were present in 75% of NSO territories (Higley and Mendia 2013).

Despite uncertainty regarding barred owl densities and population trends, incidental detections clearly indicate that the species rapidly expanded its range into that of the NSO (USFWS 2013). For example, barred owls expanded their range from western Washington to northern California in less than 10 years (Gutiérrez et al. 2004). Incidental detections also suggest that barred owl densities are continuing to increase within the range of the NSO. From 1985-2008, the

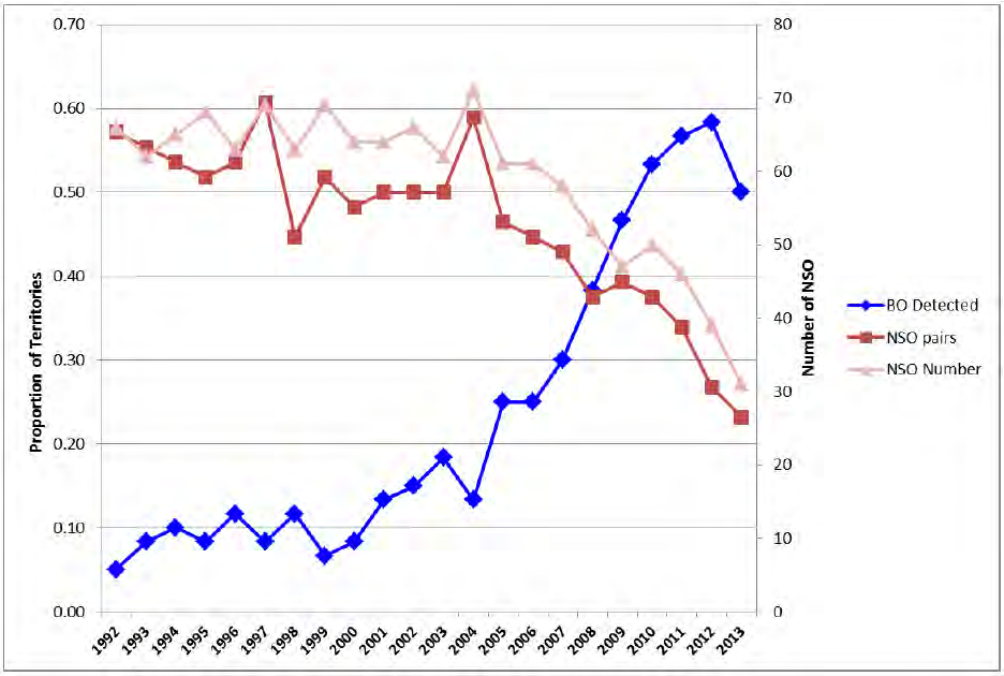
proportion of NSO territories with known barred owl presence increased in demographic study areas throughout the NSO's range, suggesting increasing barred owl populations (Forsman et al. 2011; Figure 4.2). Until the mid-2000s, barred owl increases in Washington and Oregon were steeper than those in California (Forsman et al. 2011; Figure 4.2), which is consistent with the species' later colonization of California (USFWS 2013). However, barred owls are currently increasing at an accelerated pace in at least some portions of northwestern California (see below).

Figure 4.2: Annual proportion of northern spotted owl territories with barred owl detections (≤0.62 mi from spotted owl activity center) on demographic study areas in Washington, Oregon, and California (from Dugger and Davis 2011, adapted from Forsman et al. 2011).



Recent reports from California's demographic studies indicate that barred owls continued to increase in numbers and to invade additional NSO territories during 2009-2013 (Higley and Mendia 2013, Franklin et al. 2014, Green Diamond 2014). As noted above, California's demographic studies initiated barred owl-specific surveys in 2009. The Green Diamond Resource Company also began a barred owl removal experiment in a portion of their demographic study area during the same year (see below). Detection and occupancy data reported here for 2009-2013 are not directly comparable to previously collected data, except for from the Hoopa demographic study, which separately reported data from NSO- and barred owl-specific surveys. In the Northwest California demographic study area and nearby Regional Study Area, the number of NSO territories with barred owl detections increased by 76% (from 21 to 37) during 2009-2013 and the estimated number of barred owl sites increased by 170% (10 to 27) (Franklin et al. 2014). On Green Diamond Resource Company lands, the total estimated number of barred owl sites increased by 57% (numbers not reported) during 2011-2013 (Green Diamond 2014). In the Hoopa demographic study area, NSO-specific surveys indicated that the percent of NSO territories with barred owl presence increased from 47% to 58% during 2009-2012 and dropped to 50% in 2013 for unreported reasons (Higley and Mendia 2013; Figure 4.3).

Figure 4.3: Total number of northern spotted owls detected (NSO Number), percentage of territories occupied by spotted owl pairs (NSO Pairs) and percent of historical spotted owl territories with at least one barred owl detection (BO Detected) received during spotted owl surveys, annually within the Hoopa Valley Indian Reservation demographic study area during 1992-2013 (from Higley and Mendia 2013).



Cumulative detections in the California Natural Diversity Database (CNDDB) from 1978 through 1990, 1998, 2006, and 2013 provide a crude picture of the barred owl's expansion in California (Figure 4.4). These detection data suggest that the species expanded its range into the state along the northern coast and southern Cascades and more rapidly increased in wetter regions (Figures 4.4 and 4.5). Earlier and more rapid colonization of relatively mesic forests by barred owls is consistent with observations from studies in Washington (Gutiérrez et al. 2007). While cumulative detections cannot be used to evaluate the abundance or densities of barred owls, they suggest that the species is relatively abundant along the coast and in the adjacent Six Rivers National Forest and vicinity (including the Northwest California and Hoopa demographic study areas), whereas they appear to still be in the process of colonizing the Mendocino, Klamath, and Shasta-Trinity National Forests and other interior areas of northern California. However, it is possible that this pattern is biased by the fact that most barred owl detections occur during surveys for spotted owls, which may be more densely concentrated or better surveyed in northwestern California. Concentrations of barred owl detections in northwestern California may be partly associated with intensive survey effort for spotted owls in the region's three demographic study areas. Furthermore, the Six Rivers National Forest conducted forest-wide surveys for NSOs in 2010 and subsequently continued to survey large areas associated with forest projects. Barred owls have a broader ecological range than do spotted owls, so they may be more widely distributed than is shown in Figure 4.4 (USFWS 2013).

Figure 4.4: Cumulative northern California barred owl detections in the California Natural Diversity Database from 1978 through A) 1990, B) 1998, C) 2006, and D) 2013 (note: dark red symbols denote higher concentrations of detections).

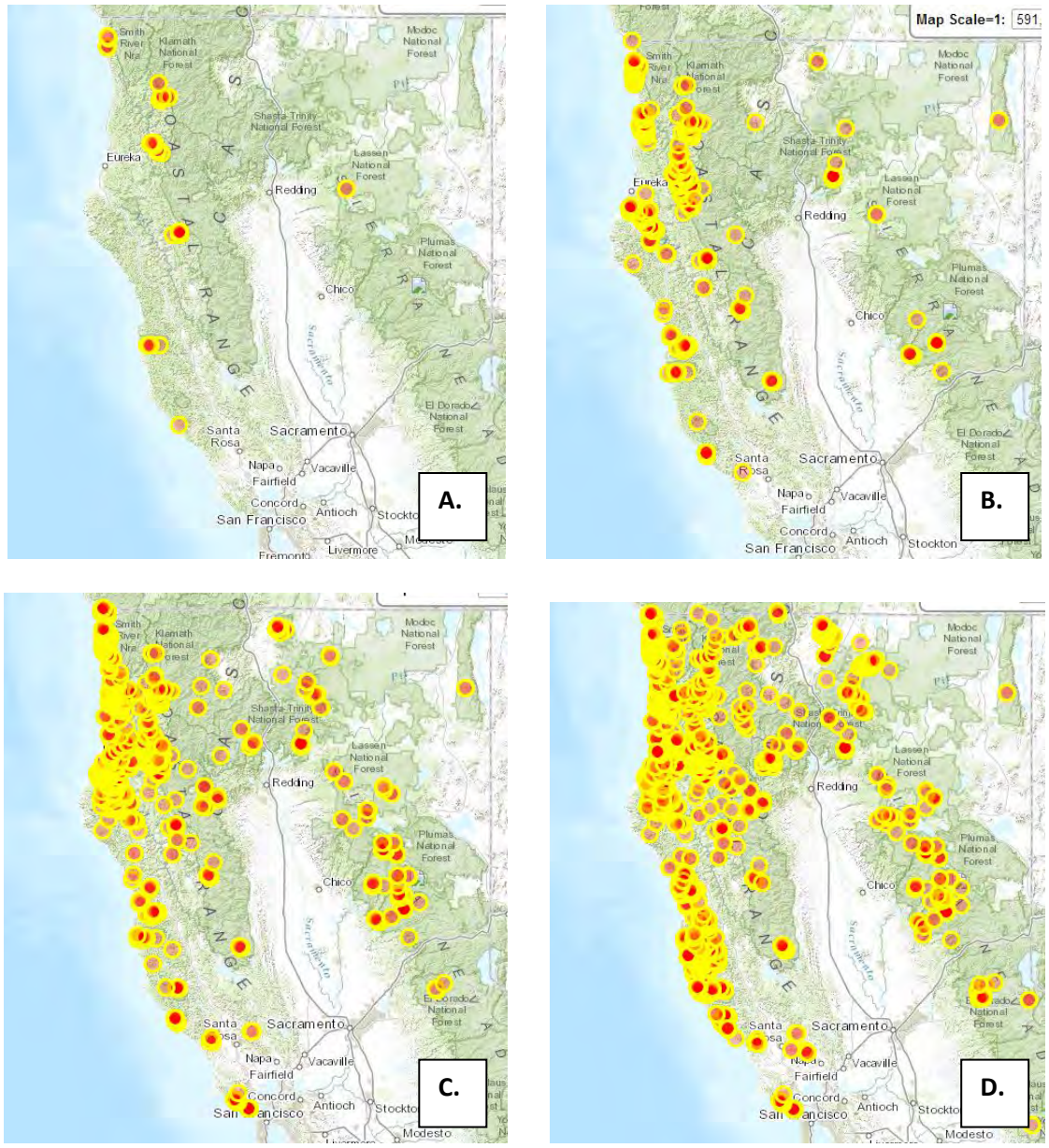
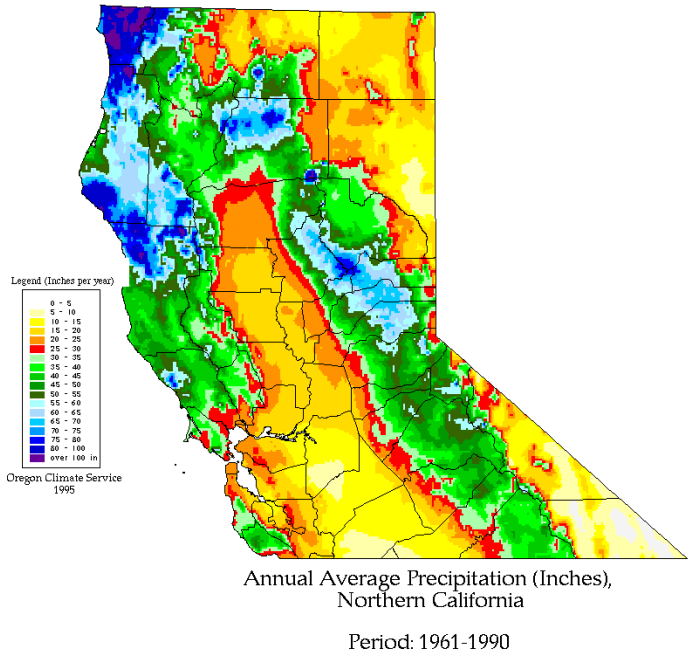


Figure 4.5: PRISM precipitation map for northern California (1961-1990) (Western Regional Climate Center).



Barred owls have increased dramatically in California's Redwood National and State Parks (Gutiérrez et al. 2004). In 2012 alone, NSO surveyors detected at least 17 barred owls at 10 different sites (Schmidt 2013). In contrast, NSOs were detected at only four historical territories in 2012. It is possible that more NSO territories were active during that year as relatively little of the landscape was surveyed. However, Schmidt (2013) noted that the Redwood National and State Parks have discontinued surveying many areas due to what appears to be almost complete displacement of NSOs by barred owls. Based on clusters of barred owl detections ($\leq$1 mi apart) during 1993-2012, Schmidt (2013) estimated that the Redwood National and State Parks contain a total of 58 barred owl territories. Barred owls have only recently invaded National Park lands in Marin County, California and have been slow to increase in numbers thus far (Ellis et al. 2013). Barred owls were first detected in Marin County in 2002 and were detected at only four (13%) NSO monitoring sites in the area in 2012 (Ellis et al. 2013). Ellis et al. (2013) hypothesized that the barred owl's expansion into Marin County has been limited due to extensive agricultural and urban lands surrounding the area. They also stated however, that barred owl numbers would likely continue to increase in the area over time and could eventually pose a substantial conservation problem for this small and relatively isolated population.

Most information about barred owls on private timberlands in California is from ownerships in the Redwood Province. Data provided by the Green Diamond Resource Company are described above, along with those from other NSO demographic studies in California. Reports from both the Humboldt Redwood Company (Humboldt County) and Mendocino Redwood Company (Mendocino and Sonoma Counties) suggest that barred owls are currently rapidly increasing on those lands. Despite a marked decline in annual numbers of nighttime NSO surveys, Humboldt

Redwood Company (2013) found an overall increase in barred owl detections within 0.5 mile of NSO activity centers during 2003-2013 (Figure 4.6). However, some of the increase in barred owl detections between 2010 and 2011 could have been due to greater survey effort associated with adoption of the USFWS (2012c) revised survey protocol for NSOs. Mendocino Redwood Company (2014) data suggest that barred owls increased substantially on their lands during 2005-2013 (Figure 4.7). Barred owls were detected within one mile of 47 (45%) NSO activity centers on these lands in 2013. The number of NSO territories with barred owl detections increased by 113% during 2010-2013. Mendocino Redwood Company (2014) noted that barred owls have been detected within one mile of 71 NSO activity centers on their lands since 2005. The Conservation Fund (in Calforests 2014), which manages lands in Mendocino and Sonoma Counties, stated that barred owls have been detected "across their ownership" since 2009. They currently have four sites at which barred owls are "regularly detected" and another area in which they are "occasionally detected".

Figure 4.6: Number of barred owl detections within 0.5 mile of northern spotted owl activity centers on Humboldt Redwood Company lands during 2003-2013 (from Humboldt Redwood Company 2013).

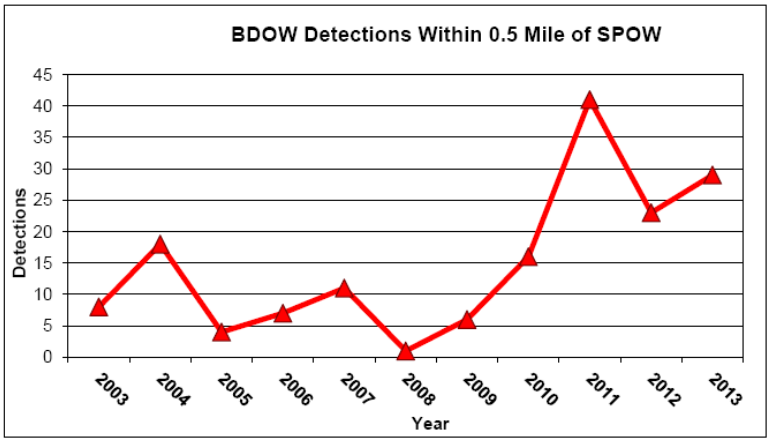
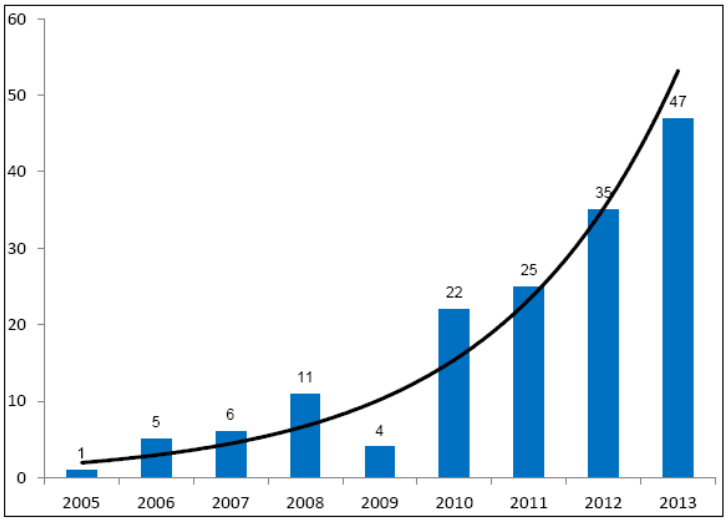


Figure 4.7: Percent (y-axis) and number of northern spotted owl territories (1 mi radius around activity centers) on Mendocino Redwood Company lands with barred owl detections during 2005-2013 (note: the apparent decline in NSO territories with barred owl detections in 2009 coincided with a substantial dip in NSO survey effort) (from Mendocino Redwood Company 2014).



There is relatively little information concerning barred owl numbers on private timberlands in California's Eastern Klamath and Southern Cascade regions. In 2013, Sierra Pacific Industries confirmed barred owl occupancy at 14 of 28 known/historical barred owl sites within their Redding and Weaverville Districts (eastern Klamath and Southern Cascades regions) and located two additional sites (SPI 2014). I was unable to locate any reports of barred owl presence or trends within NSO territories on Sierra Pacific lands. Fruit Growers Supply Company (in Calforests 2014) reported barred owl detections within or "nearby" (distance not reported) five NSO activity centers in Siskiyou County. It was unclear whether these numbers were cumulative or from 2013 alone. Michigan-California Timber Company (in Calforests 2014) reported that barred owls were detected at one site on their lands in Siskiyou County in 2010 and 2012 and two sites in 2009 and 2013. Crane Mills (in Calforests 2014) has detected only one barred owl on their lands in Shasta County. Roseburg Resource Company (in Calforests 2014) has never detected a barred owl during their NSO surveys on lands in Siskiyou and Shasta Counties. The low numbers of barred owl detections on some private timberlands in California's Eastern Klamath and Southern Cascades regions is surprising given the numbers of detections on the Shasta-Trinity and Klamath National Forests and on Sierra Pacific Industries lands in those regions (Figure 4.4), as well as in the Klamath and Southern Cascades demographic study areas in southern Oregon (Davis et al. 2013, Dugger et al. 2014). It is unclear if this is due to differences in survey effort, ecological conditions, management histories, or other factors.

In summary, the barred owl's range now completely overlaps the NSO's range and partially overlaps the California spotted owl's range. Barred owls rapidly expanded their range southward from British Columbia, through Washington and Oregon, and into northern California. The species' range expansion into California appears to have occurred first and most rapidly in northern coastal forests and near the margin of the southern Cascades and northern Sierra

Nevada. The available information suggests that there is currently a high potential for interactions between NSOs and barred owls throughout much of NSO's range in California. The barred owl invasion does not appear to have peaked in the state. Rather, the information reviewed above suggests that the species is continuing to expand into drier, interior portions of California and that their presence in NSO territories is increasing at accelerating rates in the Western Klamath and Redwood regions.

Effects on NSOs

Scientific Uncertainty

There is currently little reliable information concerning trends in barred owl densities or population numbers. Most of what is known about the potential for barred owl interactions with NSOs is based on incidental detections of barred owls during NSO surveys (USFWS 2013). Reliance on NSO surveys limits many studies' inferences concerning barred owl numbers. Gutiérrez et al. (2004) noted that some studies found large numbers of historical NSO territories apparently vacant of both NSOs and barred owls. It is unclear if these vacancies were primarily due to inadequate survey effort for one or both species or if they were caused by some other factor (Gutiérrez et al. 2004). Some of this uncertainty has been reduced due to increased survey effort required by revised survey protocols for NSOs (e.g., USFWS 2012c) and implementation of barred owl-specific surveys in some areas. Inferences from studies of barred owl effects on NSOs are further limited by the observational and retrospective nature of most research of this topic (Livezey and Fleming 2007, USFWS 2013). Most studies of barred owl effects on NSOs have examined correlations between changes in NSO occupancy or demography and barred owl presence near NSO activity centers (see below). These studies do not definitively prove that barred owl presence causes changes in NSO occupancy or demography. However, preliminary results of barred owl removal experiments more directly support conclusions that barred owl presence negatively affects NSOs in a variety of ways (see below).

Hybridization

At the time of the NSO's listing there was concern among some researchers that hybridization between spotted owls and barred owls would lead to the loss of the spotted owl as a distinct species (Gutiérrez et al. 2004). Despite genetic, morphological, ecological, and behavioral differences between spotted owls and barred owls, there appear to be few strong isolating mechanisms to prevent them from interbreeding (Gutiérrez et al. 2007). Yet, hybridization between the two species appears to be relatively rare. For example, an extensive review of NSO survey and banding records from 1970-1999 in Oregon and Washington found reports of only 47 NSO-barred owl hybrids (Kelly and Forsman 2004). Hybridization therefore appears to pose little threat to NSOs compared with the effects of competition (Kelly and Forsman 2004; see below). However, hybridization could become a more serious issue in the future as NSOs continue to decline and become less able to locate conspecific mates (Gutiérrez et al. 2007).

Demography

Forsman et al. (2011) evaluated demographic trends for NSOs in 11 demographic study areas during 1985-2008. They found that NSO reproduction (fecundity) was negatively associated with the presence of barred owls (≤0.62 mi from NSO activity centers) in four study areas, including Green Diamond's in California. Inclusion of the barred owl covariate in one of the best performing models in the meta-analysis of reproduction across all study areas provided weak support for an effect of barred owl presence on reproduction throughout the NSO's range (Forsman et al. 2011). The negative association between reproduction and barred owl presence was likely underestimated since researchers often cannot relocate NSOs displaced by barred owls, and many displaced NSOs may be unable to find new territories and reproduce (Forsman et al. 2011). Apparent survival of NSOs was negatively associated with the presence of barred owls in six of the study areas, including Northwest California and Green Diamond (95% confidence intervals did not overlap zero or only slightly overlapped zero). The meta-analysis for all study areas showed a negative association between barred owl presence and apparent survival and recruitment, although the evidence for an effect on recruitment was statistically weak. Populations in seven study areas, including Northwest California and Green Diamond, declined during the latter portion of the study period (95% confidence intervals did not overlap zero or only slightly overlapped zero for these areas). Model selection results for the meta-analysis of population change indicated support for models that included the barred owl covariate. Forsman et al. (2011) noted that, of the various factors evaluated for potential effects on NSO vital rates, negative associations with the presence of barred owls were the strongest and most consistent among study areas. Forsman et al. (2011) also noted that they likely underestimated these negative associations by applying the barred owl covariate at the population scale rather than the territory scale. Studies of associations between NSO occupancy rates and barred owl presence suggest that the territory is a more appropriate spatial scale for detecting effects of barred owls on NSOs (see below).

Annual reports from NSO demographic studies in the southern portion of the subspecies' range indicate that negative effects of barred owls on NSO demographic and occupancy rates continued to increase following the study period covered by Forsman et al. (2011) (Davis et al. 2013b, Higley and Mendia 2013, Dugger et al. 2014, Franklin et al. 2014, Green Diamond Resource Company 2014; see Ch. 1). The Northwest California and Hoopa study areas experienced dramatic declines in demographic rates subsequent to the end of the 2011 meta-analysis study period and the declines appeared to be largely driven by increasing competition from barred owls (Higley and Mendia 2013, Franklin et al. 2014). Strong negative effects of barred owls on NSO demography will likely be evident for most or all demographic study areas in the forthcoming meta-analysis, which is due for release in 2015.

Occupancy

Numerous studies distributed across the NSO's range have found evidence of a negative effect of barred owl presence on occupancy by the subspecies (Kelly 2001, Kelly et al. 2003, Pearson and Livezey 2003, Gremel 2005, Olson et al. 2005, Kroll et al. 2010, Dugger et al. 2011, Higley and Mendia 2013, Franklin et al. 2014, Green Diamond Resource Company 2014). These findings

suggest that barred owls are causing large-scale displacement of NSOs and that negative effects of barred owls are largely due to interference competition between the two species (see below).

Gremel (2005) found that occupancy by pairs of NSOs in the Olympic National Park in Washington declined significantly at sites with barred owl presence, whereas pair occupancy remained stable at sites without barred owl detections. During 1992-2003 in this study area, the number of barred owl detections in NSO sites per number of survey days increased by 15% per year. During the same period, mean occupancy by NSOs declined from 61% to 42% in sites with barred owl detections.

In the southwestern Washington Cascades, Pearson and Livezey (2003) found a 9% annual increase in the number of barred owl detections relative to the number of NSO detections during 1982-2003. Unoccupied historical NSO core areas (500 ac) had significantly more barred owl activity centers at three spatial scales (0.5, 1.0, 1.8 mi) than did occupied core areas.

Kelly et al. (2003) evaluated potential effects of barred owl presence on NSO occupancy in five demographic study areas in Washington and Oregon. Their analyses indicated that barred owls had a stronger negative effect on NSO occupancy when located closer to activity centers than when farther away. Occupancy by NSOs exhibited a highly significant decline when barred owls were detected within 0.5 mile of activity centers ($P = 0.001$), compared with a lower tendency to decline when barred owls were detected farther away ($P = 0.06$).

Olson et al. (2005) examined associations between NSO occupancy dynamics and the presence of barred owls near activity centers (distance not reported) in three study areas in western Oregon. Barred owl presence was low (detected near <10% of NSO sites each year) in all three study areas during the first eight years of the study (1990-1997), increased substantially in one of the study areas thereafter (maximum of 28% of sites in 2001), and more gradually in the other two (ca. 10% of sites in 2001). Despite relatively low barred owl presence (e.g., compared with Forsman et al. 2011, Figure 4.2), site occupancy probabilities for NSO pairs decreased by 5-15% with increasing barred owl presence.

In one of the western Oregon demographic study areas evaluated by Olson et al. (2005) (Tyee), Bailey et al. (2009) found no evidence of an effect of barred owl presence on NSO occupancy during 2002-2003. They cautioned however, that their study's inferences were weak since barred owl presence was relatively low at the time of the study (detections in <12% of NSO territories based on Olson et al. 2005). Just five years after their study period ended, barred owls were detected in 70% of NSO territories (0.62 mi around activity centers) in the study area (Forsman et al. 2011; Figure 4.2).

In the eastern Cascades of Washington, Kroll et al. (2010) found that the mean probability of occupancy by NSOs was significantly lower for sites with barred owl presence (0.50) than without it (0.76) (presence not spatially defined in the paper). Although the percent of NSO sites with barred owl detections was moderate compared with some areas (max. ca. 33% [estimated from graph] vs. Figure 4.2), site occupancy probabilities for NSOs declined by about 50% during 1990-2003.

In the southwestern Cascades of Oregon, Dugger et al. (2011) found significant differences in mean annual occupancy for NSO pairs with nearby barred owl detections (presence not spatially defined) than in those without. During the latter 13 years of the study, mean annual site occupancy was approximately 10-15% at sites with barred owl presence compared with about 58-78% at sites without barred owl presence (estimated from graph).

Both the Northwest California and Hoopa demographic studies reported dramatic recent declines in NSO occupancy coincident with rapid increases in the percent of NSO territories with barred owl presence (Higley and Mendia 2013, Franklin et al. 2014; e.g., Figure 4.3). Both studies reported sharp declines in NSO numbers and occupancy in the mid-2000s subsequent to a longer period of gradual decline (e.g., Figure 4.3). These observations suggest that the barred owl expansion and its effects on NSOs in the Western Klamath rapidly changed after a post-colonization lag period elapsed or when a crucial threshold in barred owl density was reached.

Green Diamond Resource Company is currently conducting an experiment to evaluate the effects of lethally removing barred owls from NSO territories in California's Redwood Province. Preliminary results from this study suggest that removal of barred owls results in rapid recolonization of sites by NSOs. Green Diamond Resource Company (2014) reported a 43% increase in the number of sites occupied by NSOs in their treatment (barred owl removal) area during the first year (2008 to 2009) and an additional 9% increase the following year (2009 to 2010). In contrast, the number of sites occupied by NSOs in an adjacent control portion of the study area (no barred owl removal) was virtually unchanged from 2008 to 2009 and declined by 23% from 2009 to 2010. However, the degree to which barred owl removal positively affected NSO occupancy in this study is difficult to evaluate. The beginning of Green Diamond's removal study roughly coincided with implementation of a new survey protocol which likely resulted in greater detections of both NSOs and barred owls. Future analyses from this study should provide clearer insight into the effects of barred owl presence and removal on NSO occupancy.

Habitat Use

Dugger and Davis (2011) stated that "the relationship between spotted owl fitness and habitat characteristics may have become disconnected through interspecific competition with barred owls." They based this hypothesis on their finding that mean habitat suitability at NSO pair locations within the Northwest Forest Plan area decreased by approximately 9% between 1994/1996 and 2006/2007 (Davis and Dugger 2011). This decline in mean habitat suitability at NSO pair locations did not appear to be solely due to a loss of suitable breeding habitat, which declined by about 3% in this area during the study period. This hypothesis is also supported by findings that NSOs in the northern portion of their range are often displaced by barred owls into steeper and higher elevation areas (Pearson and Livezey 2003, Gremel 2005, Hamer et al. 2007). Hamer et al. (2007) suggested that displacement of NSOs to higher elevation forests could result in reduced survival or reproduction during years with severe winters. Wiens et al. (2014) found a high degree of overlap in the habitat associations of NSOs and barred owls in western Oregon (e.g., strong selection of old conifer forest). Their best model of habitat use indicated that NSOs were less likely to use locations within or in close proximity to the core-use area of a barred owl.

This finding provides further evidence that barred owls displace NSOs from their preferred habitat.

Territorial Behavior

Barred owl presence and calling is associated with reduced responsiveness of spotted owls to conspecific calls, including survey broadcasts (Olson et al. 2005, Crozier et al. 2006, Bailey et al. 2009, Kroll et al. 2010). This is partially of interest to researchers and land managers because it influences the field and analytical methods required for measuring occupancy by NSOs (Olson et al. 2005, USFWS 2012c). Reduced vocalizing by NSOs in the presence of barred owls is also of concern because NSOs rely on vocalizations to defend their territories, locate vacant territories and potential mates, form pair bonds, and announce prey deliveries (Gutiérrez et al. 1995). Widespread disruption of these activities could impact NSO demographic rates.

Interspecific Competition

Gutiérrez et al. (2004) described the ecological and morphological separation that exists among sympatric owls worldwide. Their review found that species within the same genus are generally segregated by geographic range or habitat associations. It also showed that sympatric congeneric owls are usually strongly divergent in size, which varies with diet and possibly, hunting mode. Spotted owls and barred owls are both members of the genus *Strix*. Gutiérrez et al. (2004) noted that the two species only differ in body mass by 18%, which is likely too little to allow coexistence. A building body of evidence indicates that barred owls indeed negatively affect spotted owls (reviewed above), and that this occurs through both direct (interference) and indirect (exploitative) competition (Gutiérrez et al. 2004, 2007, USFWS 2013; see below). Wiens (2012) stated that "when viewed collectively, the behavioral and life history traits exhibited by barred owls may give them a significant advantage over spotted owls when competing for critical resources such as space, habitat, and food."

Surveyors have observed barred owls attacking spotted owls and have themselves been attacked while imitating spotted owl calls (Gutiérrez et al. 2004). There is also limited evidence of barred owl predation of NSOs. Leskiw and Gutiérrez (1998) provided strong circumstantial evidence that a barred owl killed and partially consumed an NSO in Redwood National Park, California. Johnston (2002 cited in Gutiérrez et al. 2004) found circumstantial evidence of barred owl predation of a juvenile NSO in the southern Oregon Cascades. However, most cases of barred owl aggression toward NSOs appear to be a related to territorial defense, rather than predation (USFWS 2013).

Studies in Washington found that barred owl home ranges are relatively small and tend to have little overlap, which is consistent with aggressive territorial behavior (Hamer et al. 2007, Singleton et al. 2010). In contrast, neighboring NSO home ranges often broadly overlap, particularly during winter (Hamer et al. 2007). There is limited anecdotal evidence of spotted owls aggressively interacting with barred owls (Gutiérrez et al. 2004), and spotted owls appear to reduce detection probabilities for barred owls (Bailey et al. 2009). Nonetheless, barred owls generally exhibit higher levels of vocal and physical aggression than do NSOs and are typically dominant during interactions between the two species (Van Lanen et al. 2011). Significantly

reduced detection probabilities for NSOs in the presence of barred owls provides further evidence of the larger, more aggressive barred owl's behavioral dominance over NSOs (see above).

Barred owls are dietary generalists compared with NSOs (USFWS 2013, Wiens et al. 2014). As dietary generalists, barred owls may be better able to colonize a wider variety of habitats than NSOs and may be more resilient to fluctuations in prey populations (USFWS 2013). The barred owl's generalist diet is likely a primary reason for the species' relatively small home ranges and associated ability to occur at high densities (see below). Furthermore, because barred owl diets overlap with those of NSOs, it is possible that they negatively affect NSOs by depressing populations of key prey, such as northern flying squirrels and woodrats (Gutiérrez et al. 2007, USFWS 2013).

The barred owl's habitat associations in the Pacific Northwest and California are poorly understood. As discussed earlier, most barred owl detections are incidental to spotted owl surveys so relatively little is known about the ecology of barred owls outside of areas occupied by spotted owls. Early studies of the barred owl's habitat associations in the Pacific Northwest suggested that the species is more associated with younger forest types than are NSOs (Gutiérrez et al. 2007). However, subsequent research has found that barred owls use a variety of habitats and, like NSOs, often show a preference for old forest (Gutiérrez et al. 2007, USFWS 2011a, Wiens et al. 2014). Thus, barred owls appear to be capable of occupying a broader variety of habitat types than NSOs but the two species likely compete for access to mature and old forest. The two species may also compete for nest sites since they both rely on the same kinds of pre-existing nest structures (Gutiérrez et al. 2007). Together, overlapping habitat associations with NSOs, use of a broader range of habitat types, and the ability to occur at relatively high densities allows barred owls to form large source populations in close proximity to NSOs (USFWS 2013).

Perhaps due to their generalist diet, barred owls often have substantially smaller home ranges than do NSOs (Gutiérrez et al. 2007, USFWS 2013). Estimates of barred owl home ranges in Washington were three to nine times smaller than those of NSOs in the state (Hamer et al. 2007, Singleton et al. 2010). Barred owl home ranges in the Oregon Coast Ranges were two to four times smaller than those of NSOs (Wiens et al. 2014). There does not appear to be any existing research comparing the home range sizes of barred owls and NSOs in California (Gutiérrez et al. 2007). Annual home range sizes for NSOs (100% minimum convex polygon) in the California and Oregon Klamath Provinces varied among studies and with forest types and contiguity but were similar to or somewhat larger (0-60% larger) than those for barred owls in the Oregon Coast Ranges (Sisco 1990, Carey et al. 1992, Zabel et al. 1993, Irwin et al. 2006). The NSO's smaller home ranges in California suggest that they may have lower encounter rates with barred owls than occurs in Oregon and Washington. However, it is possible that barred owls also have smaller home ranges in California. Estimates of barred owl home range sizes in California are needed.

Wiens et al. (2014) found that barred owls in the Oregon Coast Ranges had a slightly higher annual survival probability than sympatric NSOs (0.92 vs. 0.81; not statistically significant) and that pairs produced an average of 4.4 times as many young. Barred owls have a wider range of clutch sizes than NSOs (1-5 vs. 1-3), are capable of laying additional clutches within a season if

the first is lost, and appear to exhibit lower annual fluctuations in reproduction (USFWS 2013). The USFWS (2013) noted that "the ability of barred owls to forage on a wider diversity of prey species and in a wider diversity of habitats may explain their reproductive success in comparison with spotted owls." There is a need for further research of the barred owl's natural history and ecology within the range of the NSO (Gutiérrez et al. 2007, Livezey and Fleming 2007). However, the currently available information indicates that the demographic performance of barred owls is superior to that of NSOs.

Overall, barred owls appear to primarily impact NSOs through interference competition for space, habitat, and food, although they may also indirectly affect NSOs by depressing prey populations (Gutiérrez et al. 2007, Hamer et al. 2007, Van Lanen et al. 2011, USFWS 2013, Wiens et al. 2014). Barred owls are generally superior competitors to NSOs in terms of size, aggression, demographic performance, and ability to exploit a wider array of habitats and prey (Gutiérrez et al. 2007, USFWS 2013, Wiens et al. 2014).

Barred Owl Management

The information reviewed above indicates that barred owls pose a serious and increasing threat to NSOs throughout their range, including in California. Barred owl presence in NSO territories has continued to increase across the NSO's range and appears to have strongly contributed to declining NSO demographic and occupancy rates. Barred owl presence can also reduce the ability of biologists and land managers to effectively locate and conserve NSOs. Given the negative effects of barred owls on NSOs, it is clear that policymakers and land managers must address the barred owl threat if successful recovery of the NSO is to remain a conservation priority. Current proposals for addressing the barred owl threat include barred owl removal and habitat conservation.

Some researchers have expressed concern that barred owl removal experiments would be costly, ineffective, and distracting (Livezey 2010, Rosenberg et al. 2012). However, preliminary results from barred owl removal experiments indicate that lethal removal of barred owls is effective, relatively inexpensive, and conforms to animal welfare standards (Diller 2013, Diller et al. 2013, Higley 2014). Other objections to barred owl removal are primarily ethical or emotional (Gutiérrez et al. 2007, Diller 2013). There is substantial emotional resistance to lethal removal of barred owls, even among scientists and land managers involved with barred owl removal experiments (e.g., Diller 2013). Relocation of barred owls to zoos or their native forests in the eastern U.S. is logistically and politically unfeasible, so killing barred owls appears to be the only viable removal option (USFWS 2013). Primary ethical concerns regarding barred owl removal include whether or not it is appropriate to remove or control a native species or to intervene in its potentially natural range expansion (USFWS 2013). The USFWS (2013) reviewed scientific literature regarding the barred owl's status as a native or nonnative species and whether its range expansion was natural or human caused. It found that the literature was inconclusive regarding both issues. However, it concluded that humans are responsible for intervening in the barred owl's expansion because the NSO's vulnerability to barred owl competition and other stressors is due to timber harvesting and other past and continuing human activities. Regardless, the currently available evidence suggests that NSOs will continue to

decline, and could ultimately become extinct, without widespread or strategic barred owl control measures.

The barred owl's increasing impact on NSOs in Late Successional Reserves, National Parks, and other reserved lands demonstrates that habitat protection alone is insufficient for addressing the barred owl threat (Gutiérrez et al. 2004). Nonetheless, habitat conservation remains crucial to the NSO's conservation (USFWS 2011a). The importance of retaining suitable breeding habitat for NSOs has been well demonstrated at individual, territory, and population scales (e.g., Solis and Gutiérrez 1990, Franklin et al. 2000, Forsman et al. 2011; see Ch. 2). Habitat conservation might also be important for minimizing barred owl impacts on NSOs (USFWS 2011a). Habitat loss and fragmentation due to timber harvesting or other disturbances could intensify competition between NSOs and barred owls by bringing them into closer proximity (Gutiérrez et al. 2007, USFWS 2011a). Dugger et al. (2011) found some support for this hypothesis in their study of NSO occupancy in southern Oregon. Their results indicated that barred owl presence and landscape-level habitat characteristics have additive effects on NSO occupancy rates. Specifically, the presence of barred owls appeared to exacerbate the negative effects of habitat loss and fragmentation on NSO occupancy. The USFWS (2011a) suggested that retaining and restoring habitat may provide displaced or recruited NSOs with refugia from negative interactions with barred owls (USFWS 2011a). Dugger et al. (2011) did not find direct support for this hypothesis. In their study, higher amounts and lower fragmentation of older forest did not reduce the negative effects of barred owls on NSO occupancy. However, they noted that some NSOs in their study continued to survive and successfully reproduce in areas with barred owl presence, possibly indicating that there are ecological conditions under which the two species can coexist. Additional and more direct research of the potential value of habitat refugia for NSOs is needed.

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I am extremely grateful to Lowell Diller for his constructive review of this chapter.

Ch. 5: Outdoor Marijuana Cultivation

Introduction

Although marijuana is perhaps the largest cash crop in California (Gettman 2006), little is known about environmental effects associated with its cultivation. Recent research has indicated that outdoor marijuana cultivation is currently having widespread and profound environmental impacts in California (Gabriel et al. 2012, 2013, Thompson et al. 2014, Bauer et al. 2015). Negative impacts of outdoor marijuana cultivation include wildlife deaths caused by pesticide exposure and poaching; habitat degradation caused by logging, road construction, pollution, and water diversion; and heightened safety concerns for research and resource personnel (Gabriel et al. 2012, 2013, Thompson et al. 2014, Bauer et al. 2015). The specific effects of outdoor marijuana cultivation on NSOs are unknown. Recent findings of widespread pesticide exposure among fishers (*Pekania pennanti*) and barred owls in northwestern California suggest that NSOs within the state are likewise exposed and could be experiencing the same effects seen in fishers (Gabriel et al. 2012, 2014). NSOs could also be directly affected by environmental degradation from outdoor marijuana cultivation via habitat modification (e.g., clearing or logging) or suppression of rodent populations (poisoning), or indirectly affected through ecological changes caused by reduced streamflows or pollution (e.g., impacts on vegetation or prey from reduced water availability). Safety concerns associated with illegal marijuana cultivation may also be impacting NSOs and other wildlife through reduced research and survey efficiency and effort (Gabriel et al. 2013).

Pesticides

Pesticide application is usually intended to suppress populations of rodents, insects, mollusks, and other agricultural and urban pests, but can have inadvertent negative impacts on humans, pets, and other non-target animals (Erickson and Urban 2004, Albert et al. 2010, Mnif et al. 2011, Gabriel et al. 2012). Widespread secondary exposure to pesticides has been reported for raptors, carnivores, and other wildlife that consume poisoned rodents around farms and human dwellings (Albert et al. 2010, Murray 2013). Researchers have generally assumed that pesticides pose little threat to wildlife outside of agricultural and urban areas (Gabriel et al. 2013). However, a recent publication reported that 79% of fishers tested in two study areas on federal and tribal forest lands in California had been exposed to anticoagulant rodenticides (ARs), including four that died from lethal toxicosis (Gabriel et al. 2012; note: at least two more fishers in California died from AR poisoning following publication of this study: Gabriel et al. 2013). Most fishers in the study had been exposed to multiple AR compounds (range = 1-4, mean = 1.6). These findings not only raised concern for the West Coast fisher population, which the U.S. Fish and Wildlife Service recently proposed to list as threatened under the federal Endangered Species Act, but for the NSO, which overlaps the fisher in terms of distribution, habitat associations, and diet (Gabriel et al. 2013, Calforests 2014, USFWS 2014). Subsequently, ARs have been detected in a dead NSO recovered in Mendocino County (Calforests 2014) and 34 of 84 (40%) barred owls tested for exposure in Humboldt County (Gabriel et al. 2014). Although barred owls were tested as a proxy for NSOs, NSOs may be more widely exposed to ARs given their greater dietary specialization on rodents (see USFWS 2013). Strong circumstantial evidence implicates pervasive illegal outdoor marijuana cultivation

as the primary source of pesticide exposure for forest predators in California (Gabriel et al. 2012, 2013, Thompson et al. 2014).

ARs detected in fishers in northwestern California include brodifacoum, bromadiolone, chlorophacinone, diphacinone, and warfarin (Gabriel et al. 2012). Brodifacoum and bromadiolone are classified as second-generation anticoagulant rodenticides (SGARs). SGARs were introduced in the 1970s due to widespread development of resistance among rodents to first-generation anticoagulant rodenticides (FGARs), such as warfarin, chlorophacinone, and diphacinone (Buckle et al. 1994). SGARs are more acutely toxic than FGARs and generally require only a single dose to kill rodents (Erickson and Urban 2004). However, rodents usually survive 5-10 days after consuming a lethal dose, during which time they may continue to consume additional rodenticide and remain available to predators (Cox and Smith 1992, Erickson and Urban 2004). SGARs are more persistent in animal tissues than FGARs and insecticides, which are more rapidly metabolized and excreted (Erickson and Urban 2004). Thus, exposure to FGARs and other non-SGAR pesticides is more difficult to detect than for SGARs and exposure to them could be underestimated (Albert et al. 2010, Thompson et al. 2014). Rodents, such as dusky-footed woodrats (*Neotoma fuscipes*) and deer mice (*Peromyscus* spp.), may be the primary source of AR exposure for NSOs because they are targeted by AR application and because they generally comprise most of the biomass in NSO diets (Forsman et al. 2004). Insects may be an additional source of AR exposure for NSOs and other wildlife. In terms of frequency of consumption, insects can substantially contribute to NSO diets regionally, locally, or seasonally (e.g., 1-14% of prey items in various regions of Oregon: Forsman et al. 2004). Insects are not killed by ARs and may therefore continue to accumulate them in their tissues, essentially becoming small “packets” of AR (Gabriel et al. 2014).

Large quantities of ARs, particularly SGARs, are often spread across large areas in and around illegal outdoor marijuana grow sites (Gabriel et al. 2012, Thompson et al. 2014; Figure 5.1). Gabriel et al. (2012) noted that thousands of pounds of pesticides were found at illegal outdoor marijuana grow sites in California in 2008 and that 150 pounds of pesticide were found during a single three-week eradication operation on the Mendocino National Forest in 2011. Three sites raided in Humboldt County in 2013 contained a total of at least 17 pounds of SGAR bait, which researchers estimated was sufficient to kill 2,753 woodrats, 14 fishers, or five spotted owls (Humboldt County Sheriff’s Office press release). Other pesticides, such as organochlorine, organophosphate, and carbamate insecticides, some of which are banned in the U.S., are also frequently found at illegal outdoor marijuana grow sites (HSVTC 2012, Thompson et al. 2014). Pesticides are often applied along with large quantities of fertilizer at the base of marijuana plants grown outdoors (Thompson et al. 2014; Figure 5.1), suggesting that marijuana and surrounding plants may be taking up pesticidal compounds from the soil. If this occurs, then rodents and insects may accumulate pesticides through consumption of plants as well as pesticidal bait. Investigation of pathways of pesticide exposure for NSOs, as well as levels of exposure and potential physiological, behavioral, and population impacts, is needed.

Figure 5.1: (A) Rodenticide and other pesticides found at a trespass outdoor marijuana grow site in Humboldt County (photo: Humboldt County Sheriff's Office) and (B) rodenticide bait and dry fertilizer strewn together below approximately 2,000 marijuana plants at a trespass outdoor grow site in Humboldt County (from Gabriel et al. 2012).

ARs are vitamin K antagonists, which cause impairment of the blood's ability to clot (Murray 2013). A lethal dose of AR causes animals to die from hemorrhage (Erickson and Urban 2004). Animals may also exhibit weakness prior to death or with a sublethal dose (Erickson and Urban 2004). Rodents exposed to ARs show altered behavior, such as spending more time in the open, freezing rather than bolting when threatened, and staggering (Cox and Smith 1992). These behaviors may increase predation risk for affected rodents and the opportunity for secondary exposure of predators to ARs (Cox and Smith 1992). Owls and other raptors with sublethal secondary exposure to ARs may often have reduced blood-clotting activity and can die from minor wounds such as those commonly inflicted by prey (Erickson and Urban 2004, Murray 2013, Thompson et al. 2014). Non-AR pesticides have a variety of physiological effects, such as disrupting endocrine function or damaging the central nervous system (Grue et al. 1997, Mnif et al. 2011). Chronic or sublethal exposure to carbamate or organophosphate pesticides has been shown to reduce immune response, cause neurological disorders, reduce thermoregulatory control, and impair anti-predator behavior in wildlife (Grue et al. 1997; reviewed in Thompson et al. 2014). Pesticides can also have additive or synergistic effects on animals (Larsen et al. 2003 cited in Mnif et al. 2011, Relyea 2009). This is a source of additional concern for NSOs and

other forest predators active near outdoor marijuana cultivation sites, where multiple types of pesticide are often present (HSVTC 2012, Thompson et al. 2014). Mortalities associated with exposure to pesticides are likely underestimated because carcasses are often predated or scavenged before biological samples can be obtained and because sublethal exposure can predispose wildlife to death from other causes (Albert et al. 2010).

There is no information available concerning population-level impacts of secondary pesticide exposure for NSOs in California. As reviewed above, secondary exposure to pesticides can kill raptors and other wildlife both through lethal toxicosis and by increasing the risk of mortality due to other factors such as predation, hypothermia, disease, parasites, or injury. NSO population rates are particularly sensitive to changes in adult survival (Noon and Biles 1990) so it is possible that direct and indirect mortalities associated with pesticide exposure could contribute to population declines (Sibly et al. 2000, Thompson et al. 2014). Pesticide exposure could also negatively impact NSO reproduction. Gabriel et al. (2012) noted that fisher mortalities caused by AR poisoning occurred between mid-April and mid-May. The timing of these deaths coincided with the planting phase of outdoor marijuana cultivation, when seedlings are most vulnerable to rodent pests and AR use is likely highest (Gabriel et al. 2012). This time of year is also when NSOs incubate and brood young (Gutiérrez et al. 1995). Reduced parental care during this phase, for example due to compromised behavior or death of one or both parents, could result in death of offspring due to exposure, undernourishment, or predation (Grue et al. 1997). NSO populations could also be negatively impacted by pesticide suppression of prey populations or changes in community ecology caused by reductions of insects, small mammals, carnivores, raptors, amphibians, and aquatic animals. For example, pesticide impacts on plants, herbivores, or predators could cause wider ecological effects through trophic cascades (Relyea and Diecks 2008).

Other Environmental Effects

Activities related to outdoor marijuana cultivation can have a variety of environmental impacts beyond exposure of wildlife to pesticides, including negative effects of illegal and poorly planned water diversion, logging, and road construction; pollution of water and soils; poaching of wildlife; and ignition of wildfires (reviewed below). Negative impacts occur on both public and private lands (Gabriel et al. 2013, Bauer et al. 2015). However, the potential environmental impacts of marijuana cultivation could vary considerably, depending on the operation's location, scale, and practices. For example, some marijuana industry organizations and growers advocate growing practices aimed at minimizing environmental damage from outdoor cultivation (e.g., http://emeraldgrowers.org/). There does not appear to be any information available at this time regarding effects of ecological degradation from marijuana cultivation on NSOs. Negative effects on NSOs are possible given the subspecies' sensitivity to habitat modification (see Ch. 2 and 3) and close association with riparian areas (e.g., Hamer et al. 2007, Irwin et al. 2012), where the impacts of marijuana cultivation are often concentrated. Furthermore, safety concerns associated with marijuana cultivation can substantially impact the ability of land managers to effectively survey and manage spotted owls and other sensitive wildlife (Keane et al. 2011, Gabriel et al. 2013, D. Hansen pers. obs.; see below).

Personnel with the California Department of Fish and Wildlife recently estimated hydrologic impacts of marijuana cultivation in northwestern California using high-resolution aerial imagery in Google Earth (Bauer et al. 2015; e.g., Figure 5.2) and marijuana industry estimates of marijuana plant water requirements. Using these methods, they estimated that more than 112,000 marijuana plants were cultivated in 2011/2012 in just four watersheds in Humboldt and Mendocino Counties (Table 5.1). Based on estimated numbers of marijuana plants and assumed water usage of 6 gallons per day per plant, they calculated that marijuana cultivation uses between 2% and 173% of the water that flows in the Redwood Creek and Salmon Creek watersheds per day during periods of minimum streamflow (Table 5.2). Although based on several assumptions (marijuana cultivation water sources and usage, complete visibility of cultivation sites in aerial imagery, complete usage of greenhouses), these estimates have raised considerable concern about potential negative impacts of marijuana cultivation on watershed health and aquatic animals. NSOs often exhibit a preference for nesting, roosting, and foraging in and near riparian areas (e.g., Hamer et al. 2007, Irwin et al. 2007). Thus, it is plausible that ecological changes caused by widespread water diversion for marijuana cultivation have negative indirect effects on NSOs.

Figure 5.2: Outdoor (A) and greenhouse (B) marijuana cultivation sites identified with Google Earth (from S. Bauer, unpubl.).

Table 5.1: Estimated numbers of outdoor marijuana plants, marijuana greenhouses, marijuana plants in greenhouses, total number of marijuana plants, and water use per day for marijuana cultivation in four watersheds in Humboldt and Mendocino Counties (from Bauer et al. 2015).

Watershed	Outdoor Plants	Green-houses (counted)	Total area, m^2 (Green-houses)	Estimated Plants in Green-houses	Estimated Total Plants in Watershed	Estimated Water Use per Day (Liters)
Upper Redwood Creek	4,434	220	20749.4	18,612	23,046	523,144
Salmon Creek	11,697	302	20557.5	18,440	30,137	684,110
Redwood Creek South	10,475	324	18703.9	16,777	27,252	618,620
Outlet Creek	15,165	266	18651.1	16,730	31,895	724,016

Table 5.2: Estimated percent of low stream flow used for marijuana cultivation in four watersheds in Humboldt and Mendocino Counties (from Bauer et al. 2015).

Watershed	Area (km^2)	Plants per km^2	Demand as percent of seven-day low flow	
			Percent of low flow maximum	Percent of low flow minimum
Upper Redwood Creek	175.3	131.6	2%	23%
Salmon Creek	95.1	316.9	36%	173%
Redwood Creek South	64.7	421.2	34%	165%
Outlet Creek	419.1	76.1	17%	>100%*

* The seven-day low flow minimum was measured as 0.0 CMS at the gage.

Marijuana growers on both public and private lands often illegally clearcut vegetation in order to create growing space for marijuana plants and room for artificial ponds and other structures (Gabriel et al. 2013, Bauer et al. 2015; Figure 5.3). Illegal cutting, along with creation of roads to access grow sites, can increase sedimentation in streams and creeks and thereby degrade habitat quality for aquatic and amphibious animals (Bauer et al. 2015). The effect of illegal vegetation clearing on NSOs is unknown. Given the close association of both marijuana cultivation and NSOs with riparian areas and surrounding uplands (e.g., Hamer et al. 2007, Irwin et al. 2007), it is plausible that widespread marijuana cultivation results in habitat loss or fragmentation for NSOs.

Figure 5.3: Areas cleared for outdoor marijuana cultivation on private and public lands, respectively.

Unidentified source.

C. Thompson.

Widespread outdoor marijuana cultivation can further damage watershed health by polluting water and soils. In addition to pesticides, tremendous quantities of fertilizer are often applied at marijuana grow sites (HSVTC 2012, Thompson et al. 2014). Fertilizers, along with low flows caused by drought conditions and water diversion, might contribute to algae blooms and reduced oxygen levels in creeks and rivers. Other pollution, including human waste, trash, and spilled diesel fuel from generators, is also frequently observed in and around streams at raided outdoor marijuana grow sites (HSVTC 2012, C. Thompson, pers. comm.). Like water diversion and increased stream sedimentation, pollution from outdoor marijuana cultivation is primarily a source of concern for aquatic and amphibious animals but could have indirect ecological effects on NSOs and other wildlife.

Multiple recent wildfires in California have been attributed to marijuana growers. For example, in 2014, a marijuana grower was indicted on charges of starting the Nicolls Fire that burned

nearly 1,700 acres in the Sequoia National Forest (SacBee 2014b). This grower allegedly set multiple fires in an attempt to avoid capture by other growers that he claimed were trying to kill him. Another marijuana grower was recently arrested for igniting the 2014 Bully Fire that burned nearly 13,000 acres in Shasta County (SacBee 2014a). This fire was apparently started when the grower's rental truck ignited dry grass when driven off-road in order to deliver soil amendments to a grow site. In 2009, a marijuana grower's camp stove ignited the 90,000-acre La Brea Fire in the Los Padres National Forest in southern California (inciweb). Large wildfires (e.g., thousands of acres) can burn through multiple NSO territories and can negatively affect NSOs in a variety of ways, particularly when they burn large areas at moderate to high severity (see Ch. 3).

Illegal marijuana growers are often heavily armed in order to protect their crops and for poaching wildlife for food or to prevent wildlife damage to plants, equipment, or food caches (Gabriel et al. 2013, Boehm 2014; Figure 5.4). Many biologists and other field personnel working in California's forests have been interrogated, pursued, or shot at by marijuana growers (Gabriel et al. 2013, D. Hansen pers. obs.). Safety concerns associated with widespread illegal marijuana cultivation can substantially curtail the ability of researchers and land managers to effectively locate, study, and manage spotted owls and other wildlife. For example, Six Rivers National Forest biologists were repeatedly excluded from entire pre-project NSO survey units in 2013 due to evidence of trespass outdoor marijuana cultivation (D. Hansen pers. obs.). Similarly, Keane et al. (2011) stated that their California spotted owl (*S. o. occidentalis*) survey crew was excluded from large portions of a study area in the Sierra Nevada in 2010 due to extensive illegal marijuana cultivation operations. Gabriel et al. (2013; M. Gabriel unpubl. data) estimated that safety concerns due to trespass outdoor marijuana cultivation resulted in exclusion of researchers from 15-25% of one fisher study area in California and a projected additional cost of $500,000-750,000 for the life of the combined budgets of two of California's fisher research projects. Wildlife surveyors who were able to work alone in the past must now frequently work in pairs for safety reasons, reducing survey efficiency and increasing project costs (Gabriel et al. 2013, D. Hansen pers. obs.). Exclusion from study areas can also compromise the ability of researchers to properly design and complete research investigating important conservation issues, such as effects of pesticides on fishers (Gabriel et al. 2013) and wildfires on spotted owls (Keane et al. 2011).

Figure 5.4: Armed marijuana growers posing in front of a poached deer (from Gabriel et al. 2013).

Magnitude and Location of Threat

Estimates of marijuana production and value are generally based on either federal marijuana seizure data (e.g., assuming that seizures represent 10 or 15% of the total amount produced) or marijuana consumption surveys and estimates of plant yields and market value (Gettman 2006) (reviewed in PBS 2014). Estimates from both of these methods indicate that California is, by far, the primary marijuana-producing state in the U.S. and that most of this production is from outdoor cultivation (NDIC 2011, Gettman 2006). Gettman (2006) estimated that, in 2006 alone, California produced an estimated 8.6 million pounds of marijuana with a value of more than 13.8 billion dollars. An estimated 89% of this product and value was from outdoor cultivation. If correct, marijuana is the largest cash crop in California (Gettman 2006), which is remarkable given that California is the most productive agricultural state in the U.S. (USDA 2014).

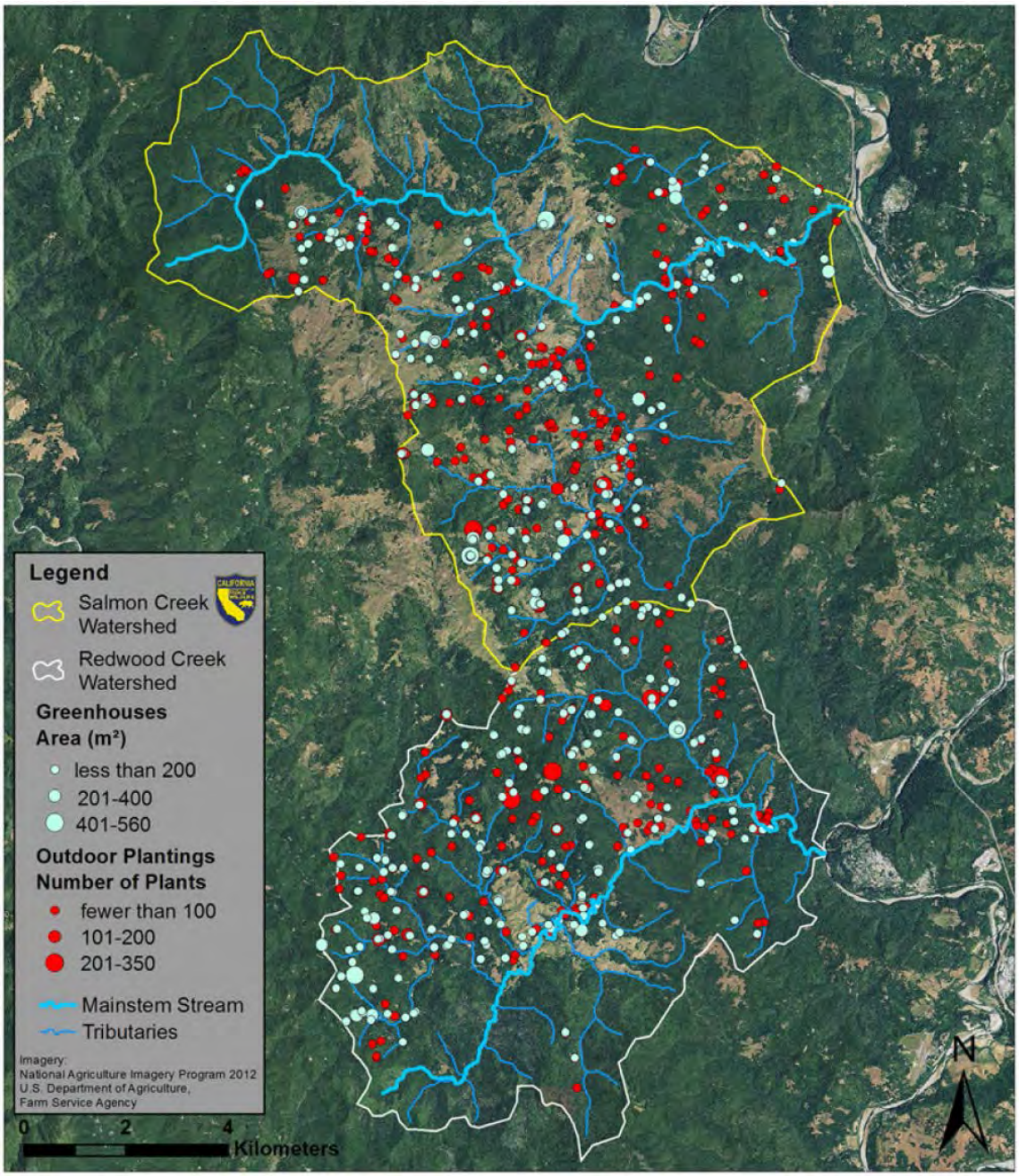
Outdoor marijuana cultivation in California has increased dramatically in recent years, including on both public and private lands (NDIC 2011, S. Bauer, unpubl. data). This rapid growth was due to increased demand for domestically grown marijuana; possibly driven by state legalization of marijuana for medical use, changes in public perception of health or legal risk associated with marijuana use, or reduced imports from other countries due to tighter border control measures implemented after 9/11 (NDIC 2007, 2011, SacBee 2012). However, rapid growth of marijuana production in California apparently outstripped consumer demand in the last few years, particularly following federal crackdowns on medical marijuana dispensaries in the state (SacBee 2012). Desire among growers to maintain high profits in the face of increasing supply and decreasing prices could be a factor driving recent increases in the size and intensity (e.g., use of pesticides, fertilizer, and water) of many outdoor marijuana cultivation operations.

The number of outdoor marijuana plants eradicated in the U.S. increased by 250% between 2005 and 2010 (NDIC 2011). Federal eradication data suggest that trespass outdoor marijuana cultivation is increasing particularly rapidly on National Forests in California (NDIC 2011).

Between 2005 and 2013, over 16 million marijuana plants were eradicated at approximately 3,356 sites on National Forests in California (Boehm 2014). Federal agencies have largely attributed increased numbers of trespass outdoor marijuana cultivation sites to expansion of operations by international drug trafficking organizations (e.g., Mexican drug cartels) into remote mountainous areas (particularly in northern California) in order to avoid detection by law enforcement personnel (NDIC 2007, Boehm 2014). However, the degree to which increases in amounts of eradicated or seized marijuana reflect increased production versus increased drug enforcement effort is unclear, as is the scale of international drug trafficking organizations' role in outdoor marijuana cultivation in California (NDIC 2010).

Outdoor marijuana production in California is also growing rapidly on private lands (NDIC 2007, S. Bauer, unpubl. data). Personnel with the California Department of Fish and Wildlife (S. Bauer, unpubl. data) used aerial imagery in Google Earth to estimate changes in the number and sizes of marijuana cultivation operations in four watersheds in Humboldt and Mendocino Counties during 2009-2012. In 2011 and 2012 they identified nearly 1,300 outdoor grow sites and more than 1,100 greenhouses likely used for marijuana cultivation in these watersheds (Table 5.1, S. Bauer, unpubl. data; e.g., Figure 5.5). The number and size of marijuana cultivation operations identified increased in all four watersheds by 68-104% between 2009 and 2012 (S. Bauer, unpubl. data). The total number of greenhouses and the number of greenhouses greater than 1,000 ft^2 increased by 69% and 87%, respectively. Continued use of aerial imagery and flyovers will shed greater light on the number, size, and location of outdoor marijuana operations on both public and private lands. For example, the California Department of Fish and Wildlife (2012) noted that law enforcement officers spotted more than 200 new marijuana grow operations in the Mattole Watershed in Humboldt County during a single flyover.

Figure 5.5: Locations and sizes of marijuana cultivation operations identified in the Salmon Creek and Redwood Creek South Watersheds using aerial imagery in Google Earth (from Bauer et al. 2015).



Summary and Management Implications

There is currently little direct information regarding potential impacts of illegal outdoor marijuana cultivation on NSOs. However, widespread application of ARs and other toxicants at outdoor grow sites are negatively impacting fishers, which have overlapping home ranges and

diets with NSOs in northwestern California (Gabriel et al. 2012, 2013, Thompson et al. 2014). There is also evidence of widespread exposure to toxicants among barred owls in Humboldt County, and an NSO from Mendocino County recently tested positive for ARs (Calforests 2014, Gabriel et al. 2014). Thus, it is likely that NSOs in California are widely exposed to toxicants applied at illegal outdoor marijuana grow sites. ARs and other pesticides can directly kill owls and other raptors or increase their vulnerability to other sources of mortality such as predation, disease, parasites, hypothermia, or injury (reviewed above). Furthermore, illegal outdoor marijuana cultivation is apparently causing widespread environmental degradation through toxicant exposure in other animals, reduced streamflows, pollution, poorly planned logging and road construction, and wildlife poaching (Gabriel et al. 2013, Bauer et al. 2015). Safety concerns associated with the widespread presence of heavily armed marijuana growers may also be impacting conservation of spotted owls and other wildlife by reducing the efficiency and effectiveness of research and survey efforts (Keane et al. 2011, Gabriel et al. 2013, D. Hansen pers. obs.).

Increased funding and effort are needed for evaluation of effects of outdoor marijuana cultivation on NSOs, other wildlife, and ecosystems. Greater funding and coordination are also needed for interdiction, clean-up, and restoration at illegal outdoor grow sites. These efforts require a substantial, multi-agency law enforcement presence, experts capable of identifying and properly disposing of toxicants, personnel and equipment for removing large amounts of trash and other material, and natural resource specialists for rehabilitating or restoring sites (Gabriel et al. 2013, Boehm 2014). Even if marijuana is legalized in California, tremendous resources, effort, and coordination may still be needed to regulate the industry and to continue to locate, clean up, and restore abandoned or interdicted illegal grow sites. Only a small portion of interdicted outdoor grow sites in California have been cleaned up thus far and even less have been restored. Many of these sites may continue to pose an environmental threat long after they are abandoned by growers (Gabriel et al. 2013). For example, water-resistant packaging can keep ARs and other toxicants viable for years, which bears can eventually find and open, allowing further poisoning and exposure of wildlife even after growing operations have ceased at the site (HSVTC 2012, M. Gabriel, pers. comm.).

The Environmental Protection Agency (EPA) and California Department of Pesticide Regulation (CDPR) have recently taken steps to reduce threats from SGARs to wildlife, children, and pets. Since 2011, EPA regulations have prohibited the sale of SGARs to the general consumer. Under EPA regulations, SGARs may only be purchased at agricultural stores, in bait station form rather than as loose pellets, and in relatively small quantities (Bradbury 2008). Most AR manufacturers quickly complied with these regulations and brought replacement products to market containing FGARs or neurotoxins, rather than SGARs (CDPR 2013, Murray 2013). However, the EPA only recently reached an agreement with Reckitt Benckiser to end distribution of their popular d-Con® products containing SGARs and sold without a protective bait station by March 31, 2015 (EPA 2014). On July 1, 2014, the CDPR further limited access to SGAR products in California by classifying them as restricted materials (California Department of Consumer Affairs 2014). In California, products containing SGARs can only be purchased from CDPR-licensed pest control dealers by certified applicators (California Department of Consumer Affairs 2014). Increased restrictions on public access to products containing SGARs should help to reduce exposure of wildlife to these compounds. However, considering that banned pesticides are

commonly found at trespass outdoor marijuana cultivation sites (HSVTC 2012, Gabriel et al. 2013, Thompson et al. 2014) growers will likely continue to widely apply SGARs in forests occupied by NSOs, fishers, and other sensitive wildlife. Furthermore, reduced availability of SGARs could simply contribute to greater application of other pesticides, including newly emerging toxicants or large amounts of legal FGARs.

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I am deeply grateful to Mourad Gabriel and Craig Thompson for their valuable insights and assistance with this chapter.

Management Recommendations

Below I provide a brief list of management recommendations and research needs. This list is solely based on information reviewed in this report. Additional management and research needs may exist for northern spotted owls (*Strix occidentalis caurina*; NSOs). Furthermore, land managers, land agencies, and policymakers may need to consider other management information and objectives alongside those for NSOs. These recommendations are mine alone and do not necessarily reflect the positions of the Environmental Protection Information Center (EPIC) or any of the document's reviewers.

1. The NSO is rapidly declining across its range. The subspecies' rate of decline has recently accelerated in California. The NSO faces an array of threats to its persistence in California and elsewhere within its range. Therefore, I recommend that:
    a. The California Fish and Game Commission list the NSO as threatened or endangered under the California Endangered Species Act.
    b. The U.S. Fish and Wildlife Service uplist the NSO from threatened to endangered under the federal Endangered Species Act.

2. Habitat retention guidelines for NSOs in the California Forest Practice Rules (CAL FIRE 2014) should be revised. The U.S. Fish and Wildlife Service's (2009) recommended guidelines for portions of California outside the redwood zone should be adopted as soon as possible. The U.S. Fish and Wildlife Service's (2011b) recommendations for the redwood zone are based on less extensive and rigorous scientific information but also appear to reflect a more accurate and current understanding of the NSO's ecology than do the California Forest Practice Rules. Recommendations for the redwood zone could potentially be revised based on modeling of Habitat Fitness Potential (Diller et al. 2010), as were those for the state's northern interior (USFWS 2009).

3. The California Department of Fish and Wildlife should continue to hire biologists with expertise in NSO-habitat relationships to assist with reviews of Timber Harvest Plans. I am under the impression that few CAL FIRE biologists have specialized knowledge of raptor ecology and conservation. Qualified wildlife biologists with the U.S. Fish and Wildlife Service formerly consulted on Timber Harvest Plans but the agency has not been regularly involved in the review process since 2008.

4. Industrial timber companies required to monitor NSOs (e.g., as part of Habitat Conservation Plans) should, whenever possible, provide modeled occupancy rates that account for detectability of NSOs and other factors that can obscure occupancy trends (e.g., see Ch. 1, Figure 1.5). Unmodeled occupancy rates are frequently cited as evidence of stable NSO populations on industrial timberlands in California (Calforests 2014). Claims of population stability or increase on industrial timberlands conflict with evidence from more rigorous research projects that have found declines in occupancy and population vital rates on federal, tribal, and private lands in northwestern California (Forsman et al. 2011, Franklin et al. 2013, 2014, Higley and Mendia 2013, Green Diamond Resource Company 2014); interior northern California (Farber and Kroll 2012);

and the Oregon Klamath and southern Cascades (Forsman et al. 2011, Davis et al. 2013, 2014, Dugger et al. 2014).

5. Barred owl removal experiments should be continued and more widespread removal programs should be planned for both public and private lands. Without barred owl removal programs, the NSO is likely to continue to spiral toward extinction, regardless of habitat protection measures.

6. Rigorous studies of effects of forest thinning and partial harvesting on NSOs and their key prey (especially dusky-footed woodrats) are needed (e.g., using a before-after-control-impact study design and an adequate sampling framework). Ideally, the silvicultural prescriptions would resemble those proposed for widespread use on federal lands. Meanwhile, land managers should assume that commercial thinning and partial harvesting negatively affect NSOs and their primary prey in California; as the currently available information generally supports this assumption. If land managers or agencies deem that thinning is necessary to address wildfire risk or meet other objectives, it should be focused outside of core patches of mature and old forest (i.e., those surrounding NSO activity centers). Thinning and other fuels reduction activities could potentially be focused in portions of the landscape that are least likely to receive use by NSOs and that are most likely to experience fire (e.g., upper and southwesterly slopes).

7. Additional research is needed to evaluate effects of severe wildfires on NSOs. This issue is scientifically and politically contentious; although there is fairly broad consensus that extensive severe fires pose a threat to NSOs in dry, fire-prone forests, such as those that occur within much of northern California outside the redwood zone (USFWS 2011a). Land agencies could better support research of wildfire effects on spotted owls by avoiding or postponing post-fire salvage logging in burned study areas. The confounding effects of salvage logging are often cited as reason to ignore research indicating that extensive severe wildfires negatively affect spotted owls. Both territory and population/landscape level and multi-year studies of severe fire effects would be useful.

8. Additional research is needed to investigate effects of post-fire salvage logging on NSOs. However, the limited available evidence suggests that salvage logging negatively affects both spotted owls and their prey. Salvage logging does not appear to be generally concordant with conservation of NSOs, as it removes important biological legacies and structurally simplifies burned areas.

9. Prescribed fire appears to have neutral or positive effects on spotted owls and therefore, appears to be consistent with the species' conservation. Allowing wildfires to burn under favorable conditions could also hold promise for reducing understory densities and reducing risk of severe fire, fostering growth of fire-adapted vegetation favored by NSO prey communities, and maintaining or restoring habitat heterogeneity in landscapes homogenized by fire suppression and timber harvesting. Favorable conditions could often exist early or late in the season or in areas where deeply incised topography creates inversions that trap smoke and minimize risk of severe fire.

10. Studies evaluating effects of marijuana cultivation on NSOs are needed. Potential research topics include investigating exposure to anti-coagulant rodenticides and other toxicants, determining effects of rodenticides on prey populations around grow sites, and examining whether or not marijuana growing on private lands potentially affects NSOs (e.g., proximity to activity centers, potential negative effects of illegal water diversion and logging on riparian areas and watersheds used by NSOs).

11. Increased financial and logistical support is likely needed for interdiction, clean-up, and restoration at trespass marijuana grow sites on public lands. Increased law enforcement could also potentially alleviate financial strains and safety concerns for NSO research and monitoring projects.

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Tom Wheeler <tom@wildcalifornia.org>

NSO in CA Reviews

Dan Hansen <danhansen03@gmail.com> Fri, Jul 3, 2015 at 4:14 PM
To: Tom Wheeler <tom@wildcalifornia.org>

Tom,

I'm also pasting email contents in some emails as they also contain comments from the reviewers.

Dan

Hi Dan,

I did not have much to comment on this one. Again, you did a very thorough review of the information out there, and your summary of the work I've been involved with (on both timber and fire) was good. I added a few minor clarifications here and there. I suggested one more paper by Wilson and Forsman, this link:

http://www.fs.fed.us/pnw/pubs/pnw_gtr880/pnw_gtr880_009.pdf

On both papers, I recommend you be careful in the use of the general term "habitat" as it can mean many different things, and tends to confuse the issue if used without providing good definitions.

Good luck on this and best regards,

Ray

Raymond J. Davis

Monitoring Lead

Older forests & spotted owls

USFS - Forestry Sciences Lab

3200 SW Jefferson Way

Corvallis, OR 97331

541-750-7179

rjdavis@fs.fed.us

Hi Dan,

Attached is my review and comments of your wildfire writeup. I'm still reviewing the timber writeup and will aim to get that to you by this afternoon.

Overall, a very thorough review, but there are a few more articles that I would add. In particular a paper by Tempel et al (in press). Effects of forest management on California Spotted Owls:

implications for reducing wildfire risk in fire-prone forests. Ecological Applications. Other papers that might provide good info:

1. Dennison, P. E., S. C. Brewer, J. D. Arnold, and M. A. Moritz (2014), Large wildfire trends in the western United States, 1984–2011, Geophys. Res. Lett., 41, 2928–2933

2. Moritz, M. A., M.-A. Parisien, E. Batllori, M. A. Krawchuk, J. Van Dorn, D. J. Ganz, and K. Hayhoe. 2012. Climate change and disruptions to global fire activity. Ecosphere 3(6):49.

3. Stavros et al. 2014. Regional projections of the likelihood of very large wildland fires under a changing climate in the contiguous Western United States. Climatic Change

4. Mallek, C., H. Safford, J. Viers, and J. Miller. 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. Ecosphere 4(12):15

5. Ganey et al. 2014. Relative abundance of small mammals in nest core areas and burned wintering areas of Mexican spotted owls in the Sacramento Mountains, New Mexico. The Wilson Journal of Ornithology 126(1):47–52

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Ray

Raymond J. Davis

Monitoring Lead

Older forests & spotted owls

USFS - Forestry Sciences Lab

3200 SW Jefferson Way

Corvallis, OR 97331

541-750-7179

rjdavis@fs.fed.us

Tom Wheeler <tom@wildcalifornia.org>

NSO in CA Reviews

Dan Hansen <danhansen03@gmail.com> Fri, Jul 3, 2015 at 4:19 PM
To: Tom Wheeler <tom@wildcalifornia.org>

Diller emails pasted below....

Hi Dan,

I completed the review of the barred owl chapter and I hope to complete the timber section by Saturday. Overall, I thought you did a great job with a thorough and comprehensive review of the barred owl issues in the NW. I have some specific comments on the attached copy, but they mostly related to the emphasis on certain publications. In particular, Dave Wiens study in the Oregon coast range is the best data available on habitat use, competitive interactions, food habits and reproduction. It is superior to any other study to date, because he had radio transmitters on both species simultaneously using the same landscape meaning we know they had equal opportunity to select habitat, prey and etc. When you wrote this, you only had it available in the more cumbersome dissertation, but you could now use the recently published monograph.

Good luck with this.

Lowell

I agree that the old meta-analysis (Forsman et al. 2011) is very dated at this point. I would rely on the annual reports from all the various study areas to provide the best current status.

Lowell

Hi Lowell,

I have a quick question about the site density estimates in Green Diamond's 2014 report. Would you call them empirical or mark-recapture estimates? Although most of the owls were likely banded, I didn't see any mention of mark-recapture methods for estimating density as was done in Diller and Thome 1999.

Thanks!

Hi Dan,

Yes, they were empirical counts based on marked birds. So the marking prevents double counting birds, which could happen frequently for non-nesting birds that move around a lot, but the empirical counts don't account for missed birds due to less than perfect detection probabilities. So these empirical counts of marked owls is equivalent to what used to be called "minimum number alive." However, spotted owls have such high detection probabilities that the estimate from using mark-recapture techniques would only provide a minor inflation of the empirical counts. The trends from spotted owls on Green Diamond will be available soon from the most recent 2015 meta-analysis.

Lowell

Hi Dan,

I reviewed the status and trends chapter, and like the barred owl chapter, I thought it was very thorough and well written. I inserted some comments for you to consider, but I didn't have any major concerns. Probably my most substantial comment is that I think modeling exercises are primarily useful for developing testable hypotheses, and although I haven't actually reviewed it in detail, I don't put a lot of credence in the source-sink model you cited. Obviously, you could really benefit from the new meta-analysis, but you pretty much guessed what it is going to say.

Lowell

Tom Wheeler <tom@wildcalifornia.org>

NSO in CA Reviews

Dan Hansen <danhansen03@gmail.com> Fri, Jul 3, 2015 at 4:33 PM
To: Tom Wheeler <tom@wildcalifornia.org>

Hi Rob,

I've marked this doc up in track changes and with comments. I assume you're trying to say fire is a real threat as part of your listing petition, but you should reconsider that position.

I read most of the section you sent me, but was irritated by the overall bias and anti-fire tone throughout, and gave up with my careful review about 1/3 of the way through. It didn't read like an objective review of existing owl and fire data. It seemed the author assumed fire could only be bad for owls. It also failed to properly weigh studies according to sample size and whether or not they were peer-review publications. There were too many instances where stats in a paper said something (or said there was no effect there), then the author used an anecdote to refute the stats and advance a 'fire is bad' position. Also, author speculation in discussion sections shouldn't be reported as results.

I feel it also mischaracterized the risk of severe fire as a forgone conclusion with some minor uncertainty, while completely ignoring the threats posed by logging in the name of fire risk reduction. If the threat from logging is expounded upon in other sections, it should also reverberate in the wildfire and salvage section. Furthermore, the author fails to establish whether fuels thinning projects have any effect on fire severity during the extreme fire weather that accompanies the vast majority of big fires (typically they don't).

I think continuing in the current anti-fire tone might alienate potential allies of your petition in the environmental and scientific communities. I suggest a focus on logging as the main threat of the past, and the continued threat of the present even though it is now sold as 'fuels thinning'.

Best,
-Derek Lee

Tom Wheeler <tom@wildcalifornia.org>

NSO in CA Reviews

Dan Hansen <danhansen03@gmail.com> Fri, Jul 3, 2015 at 4:34 PM
To: Tom Wheeler <tom@wildcalifornia.org>

Dan,

Attached is the draft that you sent a while back. With a few edits. Not many, I think you did a pretty good job of outlining the potential risks given the lack of information available. I couldn't really comment much on the environmental degradation issues, only pesticide exposure. I tried to fill in a number of your citations and made a few changes to the text. I do have a number of photos as well, if you want to highlight anything.

Craig

Craig Thompson

Research Wildlife Ecologist

USDA Forest Service

Pacific Southwest Research Station

2081 E Sierra Av, Fresno CA 93710

(559) 868-6296 - *office*

(559) 916-6223 – *cell*

cthompson@fs.fed.us

Barred Owls

Introduction

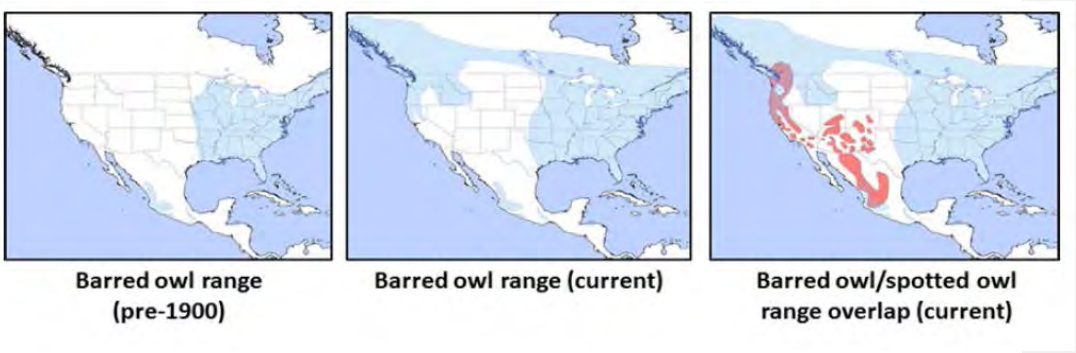
At the time of the northern spotted owl's (*Strix occidentalis caurina*; NSO) federal listing, the barred owl was recognized as a potential threat to the subspecies (USFWS 1990, Thomas et al. 1990). Since then, barred owls have continued to invade the range of the NSO and are apparently increasing in numbers (USFWS 2013). As reviewed herein, a large body of correlational and anecdotal evidence, supplemented by preliminary findings from barred owl removal experiments, indicates that barred owls are negatively impacting NSO populations across their range and that this is due to competition between the two species for space, habitat, and food. Research reviewed below indicates that that the barred owl is a superior competitor to the NSO due to its larger size, more aggressive behavior, higher reproductive potential, and broader ecological niche. The USFWS (2011) recently listed the barred owl invasion as one of three main threats currently faced by NSOs (along with timber harvesting and wildfire). The USFWS (2011) described this threat as "extremely pressing and complex" and "requiring immediate consideration." Information reviewed below suggests that lethal control of barred owls is a viable management option for some areas, although it may be difficult to overcome emotional and ethical resistance to killing one charismatic species to save another. Habitat conservation for NSOs appears to be of increasing importance because the negative effects of habitat loss and fragmentation on NSOs can be exacerbated by the presence of barred owls (Dugger et al. 2011).

Comment [LVD1]: It may not have been available when you were writing this, but the Wiens et al. 2014 monograph would be a good citation here.

The Barred Owl's Expansion

Prior to the mid-1900s, the barred owl's range was confined to southeastern Canada, the eastern U.S., and portions of Mexico (Gutiérrez et al. 2004, 2007; Figure 1). By the mid-20th century the barred owl's range began expanding westward across North America and currently includes the southern boreal forest zone and British Columbia in Canada and the northern Rocky Mountains, Pacific Northwest, and northern California in the U.S. (Livezey 2009, USFWS 2013; Figure 1). The barred owl's range now completely overlaps that of the NSO and partially overlaps that of the California spotted owl (*S. o. occidentalis*) (Figure 1). It is unclear whether the barred owl's westward range expansion occurred via the Great Plains or Canada's boreal forests (USFWS 2013). It is also uncertain whether this range expansion was facilitated by natural factors, human activities, or a combination of the two. Hypotheses concerning the cause of the barred owl's range expansion include increased adaptation by the species to coniferous forests; natural climate change; environmental changes associated with widespread intensive timber harvesting; and conversion of open areas to forest due to fire suppression, planting parks and woodlands, removal of keystone species, or other human activities (Gutiérrez et al. 2004, USFWS 2013).

Figure 1: Historical and current range of the barred owl and comparison of the current ranges of the barred owl and spotted owl (from USFWS 2013).



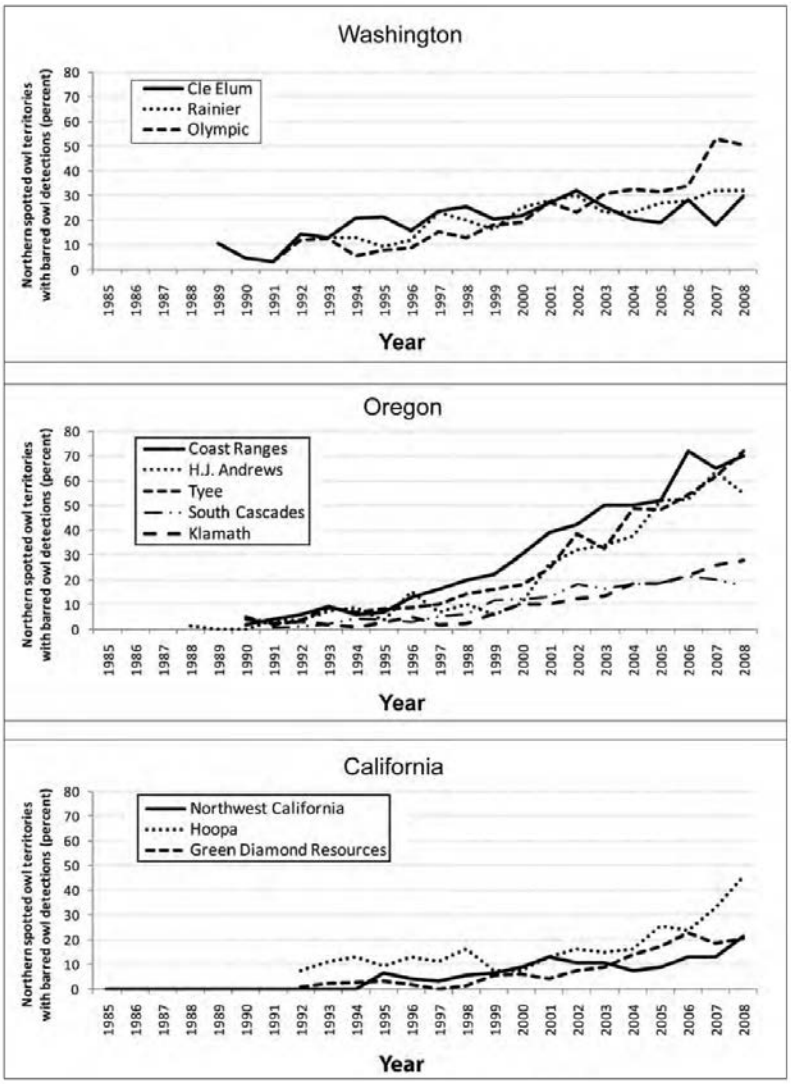
Reliable data concerning current barred owl densities and population trends are unavailable. Most information about barred owl trends in the Pacific Northwest and California is based on incidental detections during surveys for spotted owls (USFWS 2013). Researchers have estimated that only 1/2 to 2/3 of barred owls present are detected during NSO surveys (Gutiérrez et al. 2004, Bailey et al. 2009, Wiens et al. 2011). Researchers have only recently begun to systematically survey for barred owls within the range of the NSO and only in limited areas. In California, the Northwest California, Green Diamond, and Hoopa NSO density studies initiated systematic surveys for barred owls in 2009 (Franklin et al. 2014, Green Diamond Resource Company 2014, *Hoopa citation*).

Some early studies of barred owls within the NSO's range likely overestimated the species' numbers by focusing on cumulative and nighttime detections (Gutiérrez et al. 2004). However, barred owl trends are now more frequently evaluated in terms of the number of spotted owl territories in which barred owls have been detected (USFWS 2013; e.g., Forsman et al. 2011, Calforests 2014). Reports of the number of spotted owl territories with barred owl detections

probably underestimate barred owl densities and population trends (USFWS 2013). Barred owls often have substantially smaller home ranges than spotted owls so it is possible for a single spotted owl home range to encompass multiple barred owl home ranges (Singleton et al. 2010, Wiens 2012; see below). Without color-banding or follow-up surveys to determine the identity and occupancy status of barred owls, it is often unclear if multiple detections within spotted owl territories represent one or multiple barred owls.

Despite uncertainty regarding barred owl densities and population trends, incidental detections clearly indicate that the species rapidly expanded its range into that of the NSO (USFWS 2013). For example, barred owls expanded their range from western Washington to northern California in less than 10 years (Gutiérrez et al. 2004). Incidental detections also suggest that barred owl densities are continuing to increase within the range of the NSO. From 1985-2008, the proportion of NSO territories with known barred owl presence increased in density study areas throughout the NSO's range, suggesting increasing barred owl populations (Forsman et al. 2011; Figure 2). Barred owl increases in Washington and Oregon have been steeper than those in California (Forsman et al. 2011; Figure 2), which is consistent with the species' later colonization of California (USFWS 2013). More recent information suggests that barred owl increases are also currently accelerating in northwestern California (see below).

Figure 2: Annual proportion of northern spotted owl territories with barred owl detections (≤0.62 mi from spotted owl activity center) on density study areas in Washington, Oregon, and California (from Dugger and Davis 2011, adapted from Forsman et al. 2011).



Recent reports from California's density studies suggest that barred owls continued to increase in numbers and to invade additional NSO territories during 2009-2013 (Franklin et al. 2014, Green Diamond 2014, *Hoopa citation*). As noted above, California's density studies initiated barred owl-specific surveys in 2009. The Green Diamond Resource Company *(and Hoopa?)* also began a barred owl removal experiment in a portion of their density study area during the same year (see below). Thus, detection and occupancy data from 2009-2013 are not directly comparable to previously collected data. In the Northwest California density study area and nearby Regional Study Area, the number of NSO territories with barred owl detections increased by 76% (from 21 to 37) during 2009-2013 and the estimated number of barred owl sites increased by 170% (10 to 27) (Franklin et al. 2014). On Green Diamond Resource Company lands, the total estimated number of barred owl sites increased by 57% (numbers not reported) during 2011-2013 (Green Diamond 2014). *Hoopa 2009-2013 information…* The degree to which estimated increases in barred owl sites and NSO territories with barred owl detections reflect growing barred owl populations as opposed to increased and cumulative survey effort is unclear. However, the available data indicate that barred owls are continuing to invade NSO territories in California's density study areas and that this is occurring at an increasing rate.

Comment [LVD2]: Hoopa's study was initiated in September 2013 following signing of the ROD for the barred owl removal experiment. Their study is one of four planned for the FWS removal experiments, but lack of funding has delayed the implementation of the other study areas until this year (2015).

Comment [LVD3]: There are still comparable surveys that are being done using the original protocol and those data are kept separate from new barred owl-specific surveys.

Cumulative detections in the California Natural Diversity Database (CNDDB) from 1978 through 1990, 1998, 2006, and 2013 provide a crude picture of the species' expansion in California (Figure 3). These detection data suggest that barred owls expanded their range into the state along the northern coast and southern Cascades and more rapidly increased in wetter regions (Figures 3 and 4). Earlier and more rapid colonization of relatively mesic forests by barred owls is consistent with observations from studies in Washington (Gutiérrez et al. 2007).

While cumulative detections cannot be used to evaluate the barred owl's densities or population trends, they suggest that barred owls are relatively abundant along the coast and in the adjacent Six Rivers National Forest, whereas they appear to still be in the process of colonizing the Mendocino, Klamath, and Shasta-Trinity National Forests and other interior areas of northern California. However, it is possible that this pattern is ~~due to~~biased by the fact that most barred owl detections occur during surveys for spotted owls, which may be more densely concentrated or better surveyed in northwestern California. Concentrations of barred owl detections in northwestern California may be partly associated with intensive survey effort for spotted owls in the Green Diamond, Northwest California, and Hoopa density study areas. Furthermore, the Six Rivers National Forest conducted forest-wide surveys for NSOs in 2010 and 2011 and subsequently continued to survey large areas associated with forest projects (cite). Barred owls have a broader ecological range than do spotted owls, so they may be more widely distributed than is shown in Figure 3 (USFWS 2013).

Comment [LVD4]: The detections can be used to establish general trends, but they don't allow for estimates of population density or abundance.

Comment [LVD5]: THP surveys throughout the region are also reporting barred owl detections.

Figure 3: Cumulative northern California barred owl detections in the California Natural Diversity Database from 1978 through A) 1990, B) 1998, C) 2006, and D) 2013 (note: dark red symbols denote higher concentrations of detections).

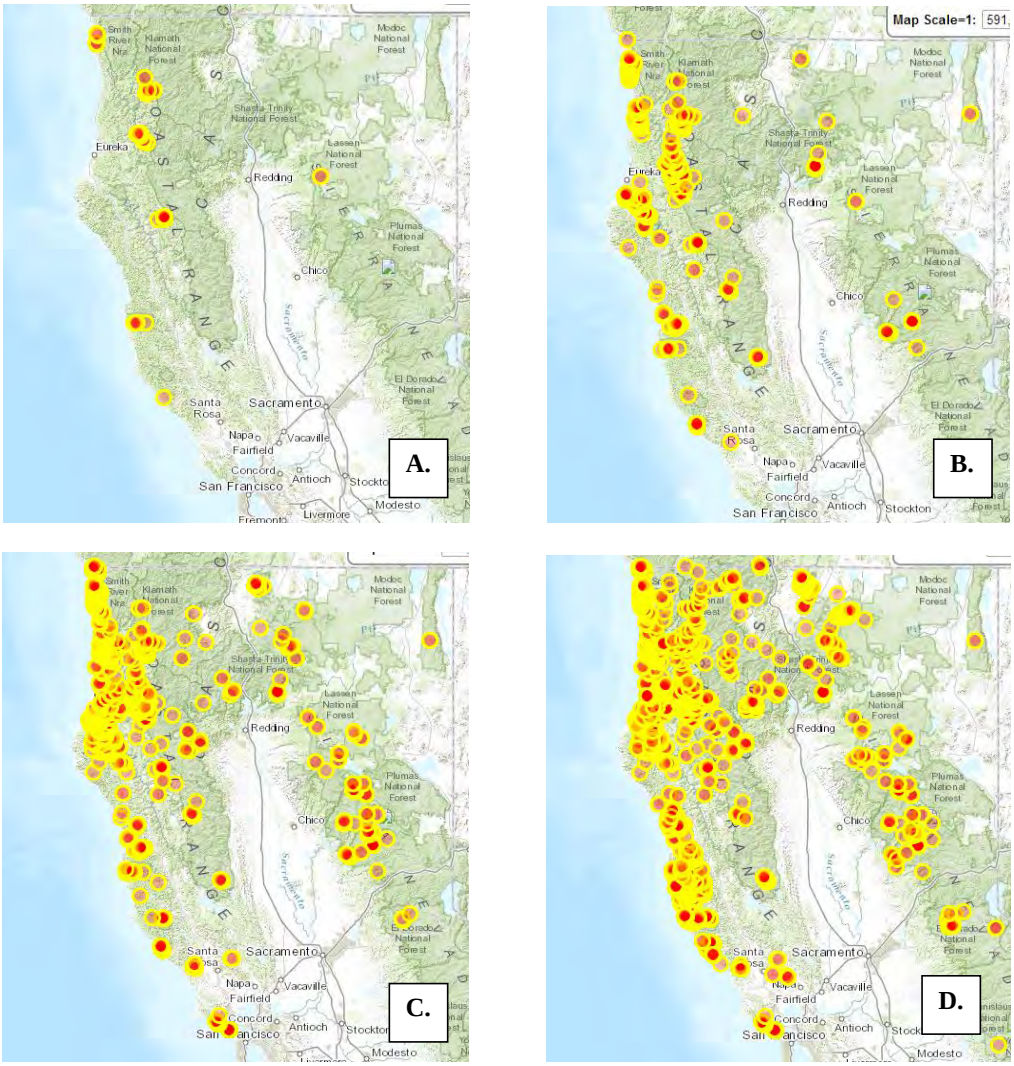
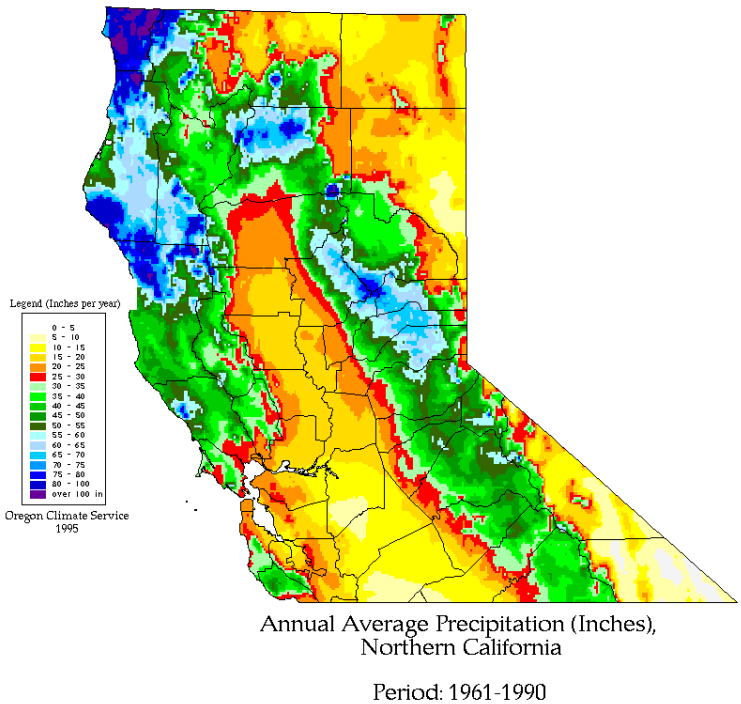


Figure 4: PRISM precipitation map for northern California (1961-1990) (Western Regional Climate Center).



Barred owls have increased dramatically in California's Redwood National and State Parks (Gutiérrez et al. 2004). In 2012 alone, NSO surveyors detected at least 17 barred owls at 10 different sites (Schmidt 2013). In contrast, NSOs were detected at only four historical territories in 2012. It is possible that more NSO territories were active during that year as relatively little of the landscape was surveyed. However, Schmidt (2013) noted that the Redwood National and State Parks have discontinued surveying many areas due to what appears to be almost complete displacement of NSOs by barred owls. Based on clusters of barred owl detections ($\leq$1 mi apart) during 1993-2012, Schmidt (2013) estimated that the Redwood National and State Parks contain a total of 58 barred owl territories. Barred owls have only recently invaded National Park lands in Marin County, California and have been slow to increase in numbers thus far (Ellis et al. 2013). Barred owls were first detected in Marin County in 2002 and were detected at only four (13%) NSO monitoring sites in the area in 2012 (Ellis et al. 2013). Ellis et al. (2013) hypothesized that the barred owl's expansion into Marin County has been limited due to extensive agricultural and urban lands surrounding the area. They also stated however, that barred owl numbers would likely continue to increase in the area over time and could eventually pose a substantial conservation problem for this small and relatively isolated population.

Most information about barred owls on private timberlands in California is from ownerships in the Redwood Province. Data provided by the Green Diamond Resource Company are described above, along with those from other NSO density studies in California. Reports from both the Humboldt Redwood Company (Humboldt County) and Mendocino Redwood Company (Mendocino and Sonoma Counties) suggest that barred owls are currently rapidly increasing on those lands. Despite a marked decline in annual numbers of nighttime surveys, Humboldt

Redwood Company (2013) found an overall increase in barred owl detections within 0.5 mile of NSO activity centers during 2003-2013 (Figure 5). However, some of the increase in barred owl detections between 2010 and 2011 was likely due to greater survey effort associated with adoption of the USFWS (2012) revised survey protocol for NSOs. Mendocino Redwood Company (2014) data suggest that barred owls increased dramatically on their lands during 2005-2013 (Figure 6). Barred owls were detected within one mile of 47 (45%) NSO activity centers on these lands in 2013. The number of NSO territories with barred owl detections increased by 113% during 2010-2013. Mendocino Redwood Company (2014) noted that barred owls have been detected within one mile of 71 NSO activity centers on their lands since 2005. The Conservation Fund (in Calforests 2014), which manages lands in Mendocino and Sonoma Counties, stated that barred owls have been detected "across their ownership" since 2009. They currently have four sites at which barred owls are "regularly detected" and another area in which they are "occasionally detected".

Figure 5: Number of barred owl detections within 0.5 mile of northern spotted owl activity centers on Humboldt Redwood Company lands during 2003-2013 (from Humboldt Redwood Company 2013).

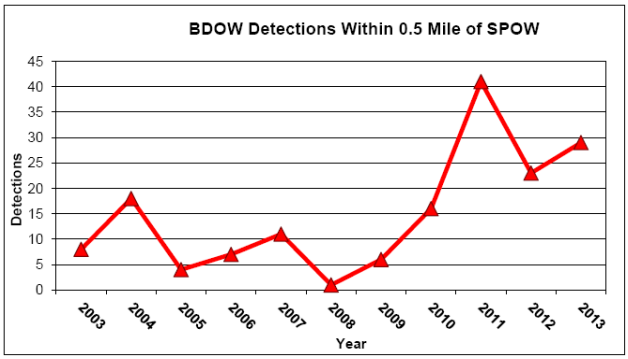
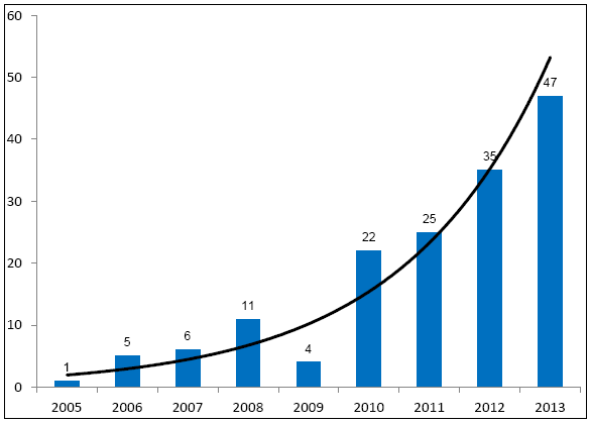


Figure 6: Percent (y-axis) and number of northern spotted owl territories (1 mi radius around activity centers) on Mendocino Redwood Company lands with barred owl detections during 2005-2013 (note: the apparent decline in NSO territories with barred owl detections in 2009 coincided with a substantial dip in NSO survey effort) (from Mendocino Redwood Company 2014).



There is relatively little information concerning barred owl numbers on private timberlands in California's eastern Klamath and Southern Cascade regions. In 2013, Sierra Pacific Industries confirmed barred owl occupancy at 14 of 28 known/historical barred owl sites within their Redding and Weaverville Districts (eastern Klamath and Southern Cascades regions) and located two additional sites (SPI 2014). We are unaware of any reports of barred owl trends or presence in NSO territories on Sierra Pacific lands. Fruit Growers Supply Company (in Calforests 2014) reported barred owl detections within or "nearby" (distance not reported) five NSO activity centers in Siskiyou County. It was unclear whether these numbers were cumulative or were from 2013 alone. Michigan-California Timber Company (in Calforests 2014) reported that barred owls were detected at one site on their lands in Siskiyou County in 2010 and 2012 and two sites in 2009 and 2013. Crane Mills (in Calforests 2014) has detected only one barred owl on their lands in Shasta County. Roseburg Resource Company (in Calforests 2014) has never detected a

barred owl during their NSO surveys on lands in Siskiyou and Shasta Counties. The low numbers of barred owl detections on some private timberlands in California's eastern Klamath and Southern Cascades regions is somewhat surprising given the numbers of detections on the Shasta-Trinity and Klamath National Forests and on Sierra Pacific Industries lands (Figure 3). It is unclear if this is due to a difference in survey effort, ecological conditions, management history, or some other factor.

In summary, the barred owl's range now completely overlaps the NSO's range and partially overlaps the California spotted owl's range. Barred owls rapidly expanded their range southward from British Columbia, through Washington and Oregon, and into northern California. The species' range expansion into California appears to have occurred first and most rapidly in northern coastal forests and near the margin of the southern Cascades and northern Sierra Nevada. However, it is unclear if these patterns actually reflect the species' expansion or are merely an artifact of higher survey effort for spotted owls in these areas. Regardless, the available information suggests that there is currently a high potential for interactions between NSOs and barred owls throughout much of NSO's range in California. The barred owl invasion does not appear to have peaked in the state. Rather, the information reviewed above suggests that the species is continuing to expand into drier, interior portions of the Klamath and Southern Cascades Provinces and that their presence in NSO territories is increasing at accelerating rates in the Redwood Province and western Klamath.

Comment [LVD6]: I don't think there is any doubt that it is a reflection of the species' expansion, because it mimics what happened in WA and OR as well. That said, I agree that the magnitude of the expansion is likely somewhat biased by the greater survey effort on the coast.

Effects on NSOs

Scientific Uncertainty

As discussed above, there is currently little reliable information concerning trends in barred owl densities or population numbers. Most of what is known about the potential for barred owl interactions with NSOs is based on incidental detections of barred owls during NSO surveys (Gutiérrez et al. 2004). Reliance on NSO surveys limits many studies' inferences concerning barred owl numbers. Gutiérrez et al. (2004) noted that some studies have found large numbers of historical NSO territories apparently vacant of both NSOs and barred owls. It is unclear if these vacancies were primarily a reflection of inadequate survey effort for one or both species or if they are caused by some other factor (Gutiérrez et al. 2004). Some of this uncertainty has been reduced due to increased survey effort required by revised survey protocols for NSOs (e.g., USFWS 2012) and implementation of barred owl-specific surveys in some areas. Inferences from studies of barred owl effects on NSOs are further limited by the observational and retrospective nature of most research of this topic (Livezey and Fleming 2007). Most studies of barred owl effects on NSOs have examined correlations between changes in NSO occupancy or demography and barred owl presence near NSO activity centers (see below). These studies do not definitively prove that barred owl presence causes changes in NSO occupancy or demography. However, preliminary results of barred owl removal experiments more directly support conclusions that barred owl presence negatively affects NSOs in a variety of ways (see below).

Comment [LVD7]: That assessment is over 10 years out of date, and as rapidly as the barred owl science is advancing, I would not consider it appropriate to cite for scientific uncertainty. You should be using the barred owl EIS, Wiens et al. 2014 and other more recent studies.

Comment [LVD8]: This is way out of date and irrelevant at this point.

Comment [LVD9]: But the Wiens study does – you could cite his dissertation or the 2014 monograph.

Hybridization

At the time of the NSO's listing there was concern among some researchers that hybridization between spotted owls and barred owls would lead to the loss of the spotted owl as a distinct species (Gutiérrez et al. 2004). Despite genetic, morphological, ecological, and behavioral differences between spotted owls and barred owls, there appear to be few strong isolating mechanisms to prevent them from interbreeding (Gutiérrez et al. 2007). Yet, hybridization between the two species appears to be relatively rare. For example, an extensive review of NSO survey and banding records from 1970-1999 in Oregon and Washington found reports of only 47 NSO-barred owl hybrids (Kelly and Forsman 2004). Hybridization therefore appears to pose little threat to NSOs compared with the effects of competition (Kelly and Forsman 2004; see below). However, hybridization could become a more serious issue in the future as NSOs continue to decline and become less able to locate conspecific mates (Gutiérrez et al. 2007).

Demography

Forsman et al. (2011) evaluated demographic trends for NSOs in 11 density study areas during 1985-2008. They found that NSO reproduction (fecundity) was negatively associated with the presence of barred owls ($\leq$0.62 mi from NSO activity centers) in four study areas, including Green Diamond's in California. Inclusion of the barred owl covariate in one of the best performing models in the meta-analysis of reproduction across all study areas provided weak support for an effect of barred owl presence on reproduction throughout the NSO's range (Forsman et al. 2011). The negative association between reproduction and barred owl presence

was likely underestimated since researchers often cannot relocate NSOs displaced by barred owls, and many displaced NSOs may be unable to find new territories and reproduce (Forsman et al. 2011). Apparent survival of NSOs was negatively associated with the presence of barred owls in six of the study areas, including Northwest California and Green Diamond (95% confidence intervals did not overlap zero or only slightly overlapped zero). The meta-analysis for all study areas showed a negative association between barred owl presence and apparent survival and recruitment, although the evidence for an effect on recruitment was statistically weak. Populations in seven study areas, including Northwest California and Green Diamond, declined during the latter portion of the study period (95% confidence intervals did not overlap zero or only slightly overlapped zero for these areas). Model selection results for the meta-analysis of population change indicated support for models that included the barred owl covariate. Forsman et al. (2011) noted that, of the various factors evaluated for potential effects on NSO vital rates, negative associations with the presence of barred owls were the strongest and most consistent among study areas. Forsman et al. (2011) also noted that they likely underestimated these negative associations by applying the barred owl covariate at the population scale rather than the territory scale. Studies of associations between NSO occupancy rates and barred owl presence suggest that the territory is a more appropriate spatial scale for detecting effects of barred owls on NSOs (see below).

Occupancy

Several studies distributed across the NSO's range have found evidence of a negative effect of barred owl presence on occupancy by the subspecies (Kelly 2001, Kelly et al. 2003, Pearson and

Livezey 2003, Gremel 2005, Olson et al. 2005, Kroll et al. 2010, Dugger et al. 2011, Green Diamond Resource Company 2014, *Hoopa citation*). These findings suggest that barred owls are causing large-scale displacement of NSOs and that negative effects of barred owls are largely due to interference competition between the two species (see below).

Gremel (2005) found that occupancy by pairs of NSOs in the Olympic National Park in Washington declined significantly at sites with barred owl presence, whereas pair occupancy remained stable at sites without barred owl detections. During 1992-2003 in this study area, the number of barred owl detections in NSO sites per number of survey days increased by 15% per year. During the same period, mean occupancy by NSOs declined from 61% to 42% in sites with barred owl detections.

In the southwestern Washington Cascades, Pearson and Livezey (2003) found a 9% annual increase in the number of barred owl detections relative to the number of NSO detections during 1982-2003. Unoccupied historical NSO core areas (500 ac) had significantly more barred owl activity centers at three spatial scales (0.5, 1.0, 1.8 mi) than did occupied core areas.

Kelly et al. (2003) evaluated potential effects of barred owl presence on NSO occupancy in five density study areas in Washington and Oregon. Their analyses indicated that barred owls had a stronger negative effect on NSO occupancy when located closer to activity centers than when farther away. Occupancy by NSOs exhibited a highly significant decline when barred owls were detected within 0.5 mile of activity centers ($P = 0.001$), compared with a lower tendency to decline when barred owls were detected farther away ($P = 0.06$).

Olson et al. (2005) examined associations between NSO occupancy dynamics and the presence of barred owls near activity centers (distance not reported) in three study areas in western Oregon. Barred owl presence was low (detected near <10% of NSO sites each year) in all three study areas during the first eight years of the study (1990-1997), increased substantially in one of the study areas thereafter (maximum of 28% of sites in 2001), and more gradually in the other two (ca. 10% of sites in 2001). Despite relatively low barred owl presence (e.g., compared with Forsman et al. 2011, Figure 2), site occupancy probabilities for NSO pairs decreased by 5-15% with increasing barred owl presence.

In one of the western Oregon density study areas evaluated by Olson et al. (2005) (Tyee), Bailey et al. (2009) found no evidence of an effect of barred owl presence on NSO occupancy during 2002-2003. They cautioned however, that their study's inferences were weak since barred owl presence was relatively low at the time of the study (detections in <12% of NSO territories based on Olson et al. 2005). Just five years after their study period ended, barred owls were detected in 70% of NSO territories (0.62 mi around activity centers) (Forsman et al. 2011; Figure 2).

In the eastern Cascades of Washington, Kroll et al. (2010) found that the mean probability of occupancy by NSOs was significantly lower for sites with barred owl presence (0.50) than without it (0.76) (presence not spatially defined). Although the percent of NSO sites with barred owl detections was moderate compared with some areas (max. ca. 33% [estimated from graph] vs. Figure 2), site occupancy probabilities for NSOs declined by about 50% during 1990-2003.

In the southwestern Cascades of Oregon, Dugger et al. (2011) found significant differences in mean annual occupancy for NSO pairs with nearby barred owl detections (presence not spatially defined) than in those without. During the latter 13 years of the study, mean annual site occupancy was approximately 10-15% at sites with barred owl presence compared with about 58-78% at sites without barred owl presence (estimated from graph).

Green Diamond Resource Company (2014) is currently conducting an experiment to evaluate the effects of lethally removing barred owls from NSO territories in California's Redwood Province. Preliminary results from this study suggest that removal of barred owls results in rapid recolonization of sites by NSOs. Green Diamond reported a 43% increase in the number of sites occupied by NSOs in their treatment (barred owl removal) area during the first year (2008 to 2009) and an additional 9% increase the following year (2009 to 2010). In contrast, the number of sites occupied by NSOs in an adjacent control portion of the study area (no barred owl removal) was virtually unchanged from 2008 to 2009 and declined by 23% from 2009 to 2010. However, the degree to which barred owl removal positively affected NSO occupancy in this study is difficult to evaluate. The beginning of Green Diamond's removal study roughly coincided with implementation of a new survey protocol which likely resulted in greater detections of both NSOs and barred owls. Future analyses from this study should provide clearer insight into the effects of barred owl presence and removal on NSO occupancy.

Add Hoopa barred owl removal preliminary results if obtainable…

Habitat Use

Comment [LVD10]: You should include Wiens dissertation or the recent monograph in the section on habitat use.

Dugger and Davis (2011) stated that "the relationship between spotted owl fitness and habitat characteristics may have become disconnected through interspecific competition with barred owls." They based this hypothesis on their finding that mean habitat suitability at NSO pair locations within the Northwest Forest Plan area decreased by approximately 9% between 1994/1996 and 2006/2007 (Davis and Dugger 2011). This decline in mean habitat suitability at NSO pair locations did not appear to be solely due to a loss of suitable breeding habitat, which declined by about 3% in this area during the study period. This hypothesis is also supported by findings that NSOs in the northern portion of their range are often displaced by barred owls into steeper and higher elevation areas (Pearson and Livezey 2003, Gremel 2005, Hamer et al. 2007). Hamer et al. (2007) suggested that displacement of NSOs to higher elevation forests could result in reduced survival or reproduction during years with severe winters.

Territorial Behavior

Barred owl presence and calling is associated with reduced responsiveness of spotted owls to conspecific calls, including survey broadcasts (Olson et al. 2005, Crozier et al. 2006, Bailey et al. 2009, Kroll et al. 2010). This is partially of interest to researchers and land managers because it influences the field and analytical methods required for measuring occupancy by NSOs (Olson et al. 2005, MacKenzie et al. 2006, USFWS 2012). Accurate assessments of occupancy are needed for evaluating effects of barred owls and other potential stressors on NSOs, and for avoiding inappropriate management activities, such as timber harvesting near nests in occupied territories.

Comment [LVD11]: This is true, but reduced detection probabilities don't necessarily mean reduced ability to detect NSO. The number of surveys required have been increased so that overall detection probabilities are >95%.

Reduced vocalizing by NSOs in the presence of barred owls is also of concern because NSOs rely on vocalizations to defend their territories, locate vacant territories and potential mates, form pair bonds, and announce prey deliveries (Gutiérrez et al. 1995). Widespread disruption of these activities could impact NSO demographic rates.

Interspecific Competition

Gutiérrez et al. (2004) described the ecological and morphological separation that exists among sympatric owls worldwide. Their review found that species within the same genus are generally segregated by geographic range or habitat associations. It also showed that sympatric congeneric owls are usually strongly divergent in size, which varies with diet and possibly, hunting mode. Spotted owls and barred owls are both members of the genus *Strix*. Gutiérrez et al. (2004) noted that the two species only differ in body mass by 18%, which is likely too little to allow coexistence. A building body of evidence indicates that barred owls indeed negatively affect spotted owls (reviewed above), and that this occurs through both direct (interference) and indirect (exploitative) competition (Gutiérrez et al. 2004, 2007, USFWS 2013; see below). Wiens (2012) stated that "when viewed collectively, the behavioral and life history traits exhibited by barred owls may give them a significant advantage over spotted owls when competing for critical resources such as space, habitat, and food."

Surveyors have observed barred owls attacking spotted owls and have themselves been attacked while imitating spotted owl calls (Gutiérrez et al. 2004). There is also limited evidence of barred owl predation of NSOs. Leskiw and Gutiérrez (1998) provided strong circumstantial evidence

that a barred owl killed and partially consumed an NSO in Redwood National Park, California. Johnston (2002 cited in Gutiérrez et al. 2004) found circumstantial evidence of barred owl predation of a juvenile NSO in the southern Oregon Cascades. However, most cases of barred owl aggression toward NSOs appear to be a related to territorial defense, rather than predation (USFWS 2013).

Studies in Washington found that barred owl home ranges are relatively small and tend to have little overlap, which is consistent with aggressive territorial behavior (Hamer et al. 2007, Singleton et al. 2010). In contrast, neighboring NSO home ranges often broadly overlap, particularly during winter (Hamer et al. 2007). There is limited anecdotal evidence of spotted owls aggressively interacting with barred owls (Gutiérrez et al. 2004), and spotted owls appear to reduce detection probabilities for barred owls (Bailey et al. 2009). Nonetheless, barred owls generally exhibit higher levels of vocal and physical aggression than do NSOs and are typically dominant during interactions between the two species (Van Lanen et al. 2011). Significantly reduced detection probabilities for NSOs in the presence of barred owls provides further evidence of the larger, more aggressive barred owl's behavioral dominance over NSOs (see above).

Barred owls are dietary generalists compared with NSOs (USFWS 2013). As dietary generalists, barred owls may be better able to colonize a wider variety of habitats than NSOs and may be more resilient to fluctuations in prey populations (USFWS 2013). The barred owl's generalist diet is likely a primary reason for the species' relatively small home ranges and associated ability to occur at high densities (see below). Furthermore, because barred owl diets overlap with those

of NSOs, it is possible that they negatively affect NSOs by depressing populations of key prey, such as northern flying squirrels and woodrats (Gutiérrez et al. 2007, USFWS 2013).

The barred owl's habitat associations in the Pacific Northwest and California are poorly understood. As discussed earlier, most barred owl detections are incidental to spotted owl surveys so relatively little is known about the ecology of barred owls outside of areas occupied by spotted owls. Early studies of the barred owl's habitat associations in the Pacific Northwest suggested that the species is more associated with younger forest types than are NSOs (Gutiérrez et al. 2007). However, subsequent research has found that barred owls use a variety of habitats and that some individuals prefer densely canopied mature and old forest (Gutiérrez et al. 2007, USFWS 2011). That is, barred owls appear to be capable of occupying a broader variety of habitat types than NSOs but the two species likely compete for access to mature and old forest. The two species may also compete for nest sites since they both rely on the same kinds of pre-existing nest structures (Gutiérrez et al. 2007). Together, overlapping habitat associations with NSOs, use of a broader range of habitat types, and the ability to occur at relatively high densities allows barred owls to form large source populations in close proximity to NSOs (USFWS 2013).

Comment [LVD12]: The only real definitive data on habitat use of NSO and BO is the Wiens study, because he was radio tracking both species simultaneously on the same landscape. This study showed that the two species have almost identical selection for different aged stands with the only difference being that BO tend to use riparian areas more than NSO.

Perhaps due to their generalist diet, barred owls often have substantially smaller home ranges than do NSOs (Gutiérrez et al. 2007, USFWS 2013). Estimates of barred owl home ranges in Washington were three to nine times smaller than those of NSOs in the state (Hamer et al. 2007, Singleton et al. 2010). Barred owl home ranges in the Oregon Coast Ranges were two to four times smaller than those of NSOs (Wiens 2012). There does not appear to be any existing research comparing the home range sizes of barred owls and NSOs in California (Gutiérrez et al.

2007). Annual home range sizes for NSOs (100% minimum convex polygon) in the California and Oregon Klamath Provinces varied among studies and with forest types and contiguity but were similar to or somewhat larger (0-60% larger) than those for barred owls in the Oregon Coast Ranges (Sisco 1990, Carey et al. 1992, Zabel et al. 1993, Irwin et al. 2006). The NSO's smaller home ranges in California suggest that they may have lower encounter rates with barred owls than occurs in Oregon and Washington. However, it is possible that barred owls also have smaller home ranges in California. Estimates of barred owl home range sizes in California are needed.

Wiens (2012) found that barred owls in the Oregon Coast Ranges had a higher annual survival probability (0.92 vs. 0.81) and produced over six times as many young as sympatric NSOs. Barred owls have a wider range of clutch sizes than NSOs (1-5 vs. 1-3), are capable of laying additional clutches within a season if the first is lost, and appear to exhibit lower annual fluctuations in reproduction (USFWS 2013; but see Mazur and James 2000). The USFWS (2013) noted that "the ability of barred owls to forage on a wider diversity of prey species and in a wider diversity of habitats may explain their reproductive success in comparison with spotted owls." There is a need for further research of the barred owl's natural history and ecology within the range of the NSO (Gutiérrez et al. 2007, Livezey and Fleming 2007). However, the currently available information indicates that the demographic performance of barred owls is superior to that of NSOs.

Overall, barred owls appear to primarily impact NSOs through interference competition for space, habitat, and food, although they may also indirectly affect NSOs by depressing prey

populations (Gutiérrez et al. 2007, Hamer et al. 2007, Van Lanen et al. 2011, Wiens 2012, USFWS 2013). Barred owls are generally superior competitors to NSOs in terms of size, aggression, demographic performance, and ability to exploit a wider array of habitats and prey (Gutiérrez et al. 2007, USFWS 2013).

Barred Owl Management

The information reviewed above indicates that barred owls pose a serious and increasing threat to NSOs throughout their range, including in California. Barred owl presence in NSO territories has continued to increase across the NSO's range and appears to be partially responsible for declining NSO demographic and occupancy rates. Barred owl presence can also reduce the ability of biologists and land managers to effectively locate and conserve NSOs. Given the negative effects of barred owls on NSOs, it is clear that policymakers and land managers must address the barred owl threat if successful recovery of the NSO is to remain a conservation priority. Current proposals for addressing the barred owl threat include barred owl removal and habitat conservation.

Some researchers have expressed concern that barred owl removal experiments would be costly, ineffective, and distracting (Livezey 2010, Rosenberg et al. 2012). However, preliminary results from barred owl removal experiments indicate that lethal removal of barred owls is effective, relatively inexpensive, and conforms to animal welfare standards (Diller 2013, Diller et al. 2013, *Hoopa/other experiment area citations*). Other objections to barred owl removal are primarily ethical or emotional (Gutiérrez et al. 2007, Diller 2013). There is substantial emotional

resistance to lethal removal of barred owls, even among scientists and land managers involved with barred owl removal experiments (e.g., Diller 2013). Relocation of barred owls to zoos or their native forests in the eastern U.S. is logistically and politically unfeasible, so killing barred owls appears to be the only viable removal option (USFWS 2013). Primary ethical concerns regarding barred owl removal include whether or not it is appropriate to remove or control a native species or to intervene in its potentially natural range expansion (USFWS 2013). The USFWS (2013) reviewed scientific literature regarding the barred owl's status as a native or nonnative species and whether its range expansion was natural or human caused. It found that the literature was inconclusive regarding both issues. However, it concluded that humans are responsible for intervening in the barred owl's expansion because the NSO's vulnerability to barred owl competition and other stressors is due to timber harvesting and other past and continuing human activities. Regardless, the currently available evidence suggests that NSOs will continue to decline, and could ultimately become extinct, without widespread or strategic barred owl control measures.

The barred owl's increasing impact on NSOs in Late Successional Reserves, National Parks, and other reserved lands demonstrates that habitat protection alone is insufficient for addressing the barred owl threat (Gutiérrez et al. 2004). Nonetheless, habitat conservation remains crucial to the NSO's conservation (USFWS 2011). The importance of retaining suitable breeding habitat for NSOs has been well demonstrated at individual, territory, and population scales (e.g., Solis and Gutiérrez 1990, Franklin et al. 2000, Forsman et al. 2011; see *Threats: Timber Harvesting*). Habitat conservation might also be important for minimizing barred owl impacts on NSOs (USFWS 2011). Habitat loss and fragmentation due to timber harvesting or other disturbances

could intensify competition between NSOs and barred owls by bringing them into closer proximity (Gutiérrez et al. 2007, USFWS 2011). Dugger et al. (2011) found some support for this hypothesis in their study of NSO occupancy in southern Oregon. Their results indicated that barred owl presence and landscape-level habitat characteristics have additive effects on NSO occupancy rates. Specifically, the presence of barred owls appeared to exacerbate the negative effects of habitat loss and fragmentation on NSO occupancy. The USFWS (2011) suggested that retaining and restoring habitat may provide displaced or recruited NSOs with refugia from negative interactions with barred owls (USFWS 2011). Dugger et al. (2011) did not find direct support for this hypothesis. In their study, higher amounts and lower fragmentation of older forest did not reduce the negative effects of barred owls on NSO occupancy. However, they noted that some NSOs in their study continued to survive and successfully reproduce in areas with barred owl presence, possibly indicating that there are ecological conditions under which the two species can coexist. Additional and more direct research of the potential value of habitat refugia for NSOs is needed.

Acknowledgements

Xxxx

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Outdoor Marijuana Cultivation

Introduction

Although marijuana is perhaps the largest cash crop in California (Gettman 2006), its cultivation is largely unregulated and little is known about ~~its~~ environmental effects associated with its cultivation ~~environmental effects~~. Recent research has indicated that outdoor marijuana cultivation is currently having widespread and profound environmental impacts in California (Bauer cite, Gabriel et al. 2012, 2013, Thompson et al. 2014). Negative impacts of outdoor marijuana cultivation include wildlife deaths caused by pesticide exposure and poaching; habitat degradation caused by logging, road construction, pollution, and water diversion; and heightened safety concerns for research and resource personnel (Bauer cite, Gabriel et al. 2012, 2013, Thompson et al. 2014). The specific effects of outdoor marijuana cultivation on northern spotted owls (*Strix occidentalis caurina*; NSOs) are unknown. Recent findings of widespread pesticide exposure among fishers (~~*Martes*~~ *Pekania pennanti*) and barred owls (*Strix varia*) in northwestern California suggest that NSOs ~~in~~ within the state are likewise exposed and could be ~~negatively affected at both territory and population levels~~experiencing the same effects seen in fishers (~~Gabriel cite, Gabriel et al. 2012; reviewed below~~). NSOs could also be directly affected by environmental degradation from outdoor marijuana cultivation, ~~such as through~~via habitat modification ~~or suppression of rodent prey populations~~, or indirectly affected through ecological changes caused by reduced streamflows or pollution. Safety concerns associated with illegal marijuana cultivation may also be impacting NSOs and other wildlife through reduced research and survey efficiency and effort (Gabriel et al. 2013).

Comment [MWG1]: The plant itself is not hazardous, it's the activities associated with its cultivation.

Comment [MWG2]: New genus for the species.

Comment [MWG3]: Anticoagulant Rodenticide Exposure in Barred Owls, a Proxy for the Northern Spotted Owl in California Managed Forests: Implications of Food Web Contamination

Gabriel, M.W.1*, L. Diller2, J. Dumbacher3, J.M. Higley4, G.M. Wengert1, S. Mendia4, D. Lamphear2, D. Early2,K. Hamm2, and R. Poppenga5.
1*

The exposure of wildlife to anticoagulant rodenticides (AR) has been well documented in non-forest settings. Until recently with the discovery of rampant use of ARs within trespass marijuana sites, exposure to ARs in remote forest wildlife was unknown and unexpected. We investigated the landscape-wide AR contamination in 84 Barred Owls, a proxy species for the Northern Spotted Owl (NSO), within NSO suitable habitat on northwestern California managed timberlands. Additionally, we investigated whether owl prey sampled in owl habitat were exposed to ARs. A total of 40% of all tested owls were exposed to ARs while 37 rodents sampled within NSO habitat were negative. However, all invertebrate samples (100%) from trespass grows were positive for ARs. Due to the inherent and swift lethality of ARs, negative rodent data was anticipated; however this is the first report of AR exposure to field collected invertebrates. These results demonstrate that AR contamination within NSO populations in NW California is likely and that food web contamination for these owls and numerous forest wildlife species is concerning.

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Comment [MWG4]: Will still need to cite it as unpublished presented data.

Comment [MWG5]: Mentioned above

Comment [MWG6]: How would NSO be affected by reduced wateflows? They do not use inverts >10% of their diet. Barred owls, possible, but NSO impact's due to streamflow's would need to have some substantial premise to make this point.

Pesticides

Pesticide application is usually intended to suppress populations of rodents, insects, mollusks, and other agricultural and urban pests, but can have inadvertent negative impacts on humans, pets, and other non-target animals (Erickson and Urban 2004, Albert et al. 2010, Mnif et al. 2011, Gabriel et al. 2012). Widespread secondary exposure to pesticides has been reported for raptors, carnivores, and other wildlife that consume poisoned rodents around farms and human dwellings (Albert et al. 2010, Murray 2013, ~~Thompson et al. 2014).~~ Researchers have generally assumed that pesticides pose little threat to wildlife outside of agricultural and urban areas (Gabriel et al. 2013). However, a recent publication reported that 79% of fishers tested in two study areas on federal and tribal forest lands in California had been exposed to anticoagulant rodenticides (ARs), including four that ~~apparently~~ died from lethal toxicosis (Gabriel et al. 2012; note: at least two more fishers in California died from AR poisoning following publication of this study: Gabriel et al. 2013). Most fishers in the study had been exposed to multiple AR compounds (range = 1-4, mean = 1.6). These findings not only raised concern for the West Coast fisher population, which the U.S. Fish and Wildlife Service recently proposed to list as threatened under the federal Endangered Species Act, but for the NSO, which has an overlapping distribution and diet with the fisher (Gabriel et al. 2013, Calforests 2014, USFWS 2014). Subsequently, ARs have been detected in a dead NSO recovered in Mendocino County (Calforests 2014) and xx of xx (xx%) barred owls tested for exposure in Humboldt County (Gabriel cite). Although barred owls are being tested as a proxy for NSOs (Gabriel cite), NSOs may be more widely exposed to ARs given their greater dietary specialization on rodents (see

Comment [MWG7]: Cite primary literature, Gabriel and Thompson are primary for fisher but not for other species.

Comment [MWG8]: More of habitat characteristics and some diet overlap.

Comment [MWG9]: See above presentation

USFWS 2013). Strong circumstantial evidence implicates pervasive illegal outdoor marijuana cultivation as the primary source of pesticide exposure for forest predators in California (Gabriel et al. 2012, 2013, Thompson et al. 2014).

ARs detected in forest predators in northern California include brodifacoum, bromadiolone, difethialone, chlorophacinone, diphacinone, warfarin, and coumachlor (Gabriel cite, Gabriel et al. 2012). Brodifacoum, bromadiolone, difethialone are classified as second-generation anticoagulant rodenticides (SGARs). SGARs were introduced in the 1970s due to widespread development of resistance among rodents to first-generation anticoagulant rodenticides (FGARs), such as warfarin, chlorophacinone, and diphacinone (Buckle et al. 1994). SGARs are more acutely toxic than FGARs and generally require only a single dose to kill rodents (Erickson and Urban 2004). However, rodents usually survive 5-10 days after consuming a lethal dose, during which time they may continue to consume additional rodenticide and remain available to predators (Cox and Smith 1992, Erickson and Urban 2004). SGARs are more persistent in animal tissues than FGARs and insecticides, which are more rapidly metabolized and excreted (Erickson and Urban 2004). Thus, exposure to FGARs and other non-SGAR pesticides is more difficult to detect than for SGARs and exposure to them could be underestimated (Albert et al. 2010, Thompson et al. 2014). Rodents such as dusky-footed woodrats (*Neotoma fuscipes*) and deer mice (*Peromyscus* spp.) are likely the primary source of AR exposure for NSOs because they are targeted by AR application and because they generally comprise most of the biomass in NSO diets (Forsman et al. 2004). Insects may be an additional source of AR exposure for NSOs and other wildlife. In terms of frequency of consumption, insects can substantially contribute to NSO diets regionally, locally, or seasonally (Forsman et al. 2004). Insects are not killed by ARs

Comment [MWG10]: Not all were detected, see 2012 Plos paper

Comment [MWG11]: Are they? I would rewrite with a general blanket statement that rodents are the primary source, however the main rodents most likely affected are…

Comment [MWG12]: Please see John Weins 2012 dissertation and publication https://ir.library.oregonstate.edu/xmlui/bitstream/handle/1957/28475/WiensJohnD2012.pdf?sequence=2

Comment [MWG13]: See weins study of minimal insect use by NSO

and may therefore continue to accumulate them in their tissues, essentially becoming small "packets" of AR (~~C. Thompson, pers. comm~~This would be primary data from Gabriel un published.).

Large quantities of ARs, particularly SGARs, are often spread across large areas in and around illegal outdoor marijuana grow sites (Gabriel et al. 2012, Thompson et al. 2014; Figure 1). Gabriel et al. (2012) noted that thousands of pounds of pesticides were found at illegal outdoor marijuana grow sites in California in 2008 and that 150 pounds of pesticide were found during a single three-week eradication operation on the Mendocino National Forest in 2011. Three sites raided in Humboldt County in 2013 contained a total of at least 17 pounds of SGAR bait, which researchers estimated was sufficient to kill 2,753 woodrats, 14 fishers, or five spotted owls (Humboldt Sentinel August 2, 2013). Other pesticides, such as organochlorine, organophosphate, and carbamate insecticides, some of which are banned in the U.S., are also frequently found at illegal outdoor marijuana grow sites (HSVTC 2012, Thompson et al. 2014). Pesticides are often applied along with large quantities of fertilizer at the base of marijuana plants grown outdoors (Thompson et al. 2014; Figure 1), suggesting that marijuana and surrounding plants may be taking up pesticidal compounds from the soil. If this occurs, then rodents and insects may accumulate pesticides through consumption of plants as well as pesticidal bait. Investigation of pathways of pesticide exposure for NSOs, as well as levels of exposure and potential physiological, behavioral, and population impacts, is needed.

Comment [MWG14]: Primary source was Humboldt county sheriffs office which we provided them the data for the press release. Cite HCSO

Figure 1: (A) Rodenticide and other pesticides found at a trespass outdoor marijuana grow site in Humboldt County (photo: Humboldt County Sheriff's Office) and (B) rodenticide bait and dry fertilizer strewn together below approximately 2,000 marijuana plants at a trespass outdoor grow site in Humboldt County (from Gabriel et al. 2012).

ARs are vitamin K antagonists, which cause impairment of the blood's ability to clot (Murray 2013). A lethal dose of AR causes animals to die from hemorrhage (Erickson and Urban 2004). Animals may also exhibit weakness prior to death or with a sublethal dose (Erickson and Urban 2004). Rodents exposed to ARs show altered behavior, such as spending more time in the open, freezing rather than bolting when threatened, and staggering (Cox and Smith 1992). These behaviors may increase predation risk for affected rodents and the opportunity for secondary exposure of predators to ARs (Cox and Smith 1992). Owls and other raptors with sublethal secondary exposure to ARs may often have reduced blood-clotting activity and can die from

minor wounds such as those commonly inflicted by prey (Erickson and Urban 2004, Murray 2013, Thompson et al. 2014). Non-AR pesticides have a variety of physiological effects, such as disrupting endocrine function or damaging the central nervous system (Grue et al. 1997, Mnif et al. 2011). Chronic or sublethal exposure to carbamate or organophosphate pesticides has been shown to reduce immune response, cause neurological disorders, reduce thermoregulatory control, and impair anti-predator behavior in wildlife (Grue et al. 1997; reviewed in Thompson et al. 2014). Pesticides can also have additive or synergistic effects on animals (Larsen et al. 2003 cited in Mnif et al. 2011, Relyea 2009). This is a source of additional concern for NSOs and other forest predators active near outdoor marijuana cultivation sites, where multiple types of pesticide are often present (HSVTC 2012, Thompson et al. 2014). Mortalities associated with exposure to pesticides are likely underestimated because carcasses are often predated or scavenged before biological samples can be obtained and because sublethal exposure can predispose wildlife to death from other causes (Albert et al. 2010).

There is no information available concerning population-level impacts of secondary pesticide exposure for NSOs in California. As reviewed above, secondary exposure to pesticides can kill raptors and other wildlife both through lethal toxicosis and by increasing the risk of mortality due to other factors such as predation, hypothermia, disease, parasites, or injury. NSO population rates are particularly sensitive to changes in adult survival (Noon and Biles 1990) so it is possible that direct and indirect mortalities associated with pesticide exposure could contribute to population declines (Sibly et al. 2000, Thompson et al. 2014). Pesticide exposure could also negatively impact NSO reproduction. Gabriel et al. (2012) noted that fisher mortalities caused by AR poisoning occurred between mid-April and mid-May. The timing of

these deaths coincided with the planting phase of outdoor marijuana cultivation, when seedlings are most vulnerable to rodent pests and AR use is likely highest (Gabriel et al. 2012). This time of year is also when NSOs incubate and brood young (Gutiérrez et al. 1995). Reduced parental care during this phase, for example due to compromised behavior or death of one or both parents, could result in death of offspring due to ~~chilling~~exposure, undernourishment, or predation (Grue et al. 1997). NSO populations could also be negatively impacted by pesticide suppression of prey populations or changes in community ecology caused by reductions of insects, small mammals, carnivores, raptors, amphibians, and aquatic animals (Relyea and Diecks 2008).

Comment [MWG15]: Why would NSO populations decrease if a carnivore population decreases, or for the other examples mentioned. It is plausible for the small mammals due to them being prey items. Explain for the reader

Other Environmental Effects

Activities related to outdoor marijuana cultivation can have a variety of environmental impacts beyond exposure of wildlife to pesticides, including negative effects of illegal and poorly planned water diversion, logging, and road construction; pollution of water and soils; poaching of wildlife; and ignition of wildfires (reviewed below). Negative impacts occur on both public and private lands (Bauer cite, Gabriel et al. 2013). However, the potential environmental impacts of marijuana cultivation could vary considerably, depending on the operation's location, scale, and practices. For example, some marijuana industry organizations and growers advocate growing practices aimed at minimizing environmental damage from outdoor cultivation (e.g., http://emeraldgrowers.org/). ~~No~~ there does not appear to be any information available at this time regarding effects of ecological degradation from marijuana cultivation on NSOs. Negative effects on NSOs are possible given the subspecies' sensitivity to habitat modification (see *Timber Harvesting* and *Wildfires*, this volume) and close association with riparian areas (e.g.,

Hamer et al. 2007, Irwin et al. 2012), where the impacts of marijuana cultivation are often concentrated. Furthermore, safety concerns associated with marijuana cultivation can substantially impact the ability of land managers to effectively survey and manage spotted owls and other sensitive wildlife (Keane et al. 2011, Gabriel et al. 2013, D. Hansen pers. obs.; see below).

Personnel with the California Department of Fish and Wildlife (Bauer cite) recently estimated hydrologic impacts of marijuana cultivation in northwestern California using high-resolution aerial imagery in Google Earth™ (e.g., Figure 2) and marijuana industry estimates of marijuana plant water requirements. Using these methods, they estimated that more than 82,000 marijuana plants were cultivated in 2012 in just four watersheds in Humboldt and Mendocino Counties (Table 1). Based on estimated numbers of marijuana plants and assumed water usage of 6 gallons per day per plant, they calculated that marijuana cultivation uses approximately 29% and 21% of the water that flows in the Redwood Creek and Salmon Creek watersheds per day during periods of minimum streamflow (Table 2). Although based on several assumptions (marijuana cultivation water sources and usage, complete visibility of cultivation sites in aerial imagery, complete usage of greenhouses), these estimates raised considerable concern about potential negative impacts of marijuana cultivation on watershed health and aquatic animals (cite). NSOs often exhibit a preference for nesting, roosting, and foraging in and near riparian areas (e.g., Hamer et al. 2007, Irwin et al. 2007). Thus, it is plausible that ecological changes caused by widespread water diversion for marijuana cultivation have negative indirect effects on NSOs.

Comment [MWG16]: This sentence fits the concern that I had above about water usage. But this should definitely be mentioned above

Figure 2: Outdoor (A) and greenhouse (B) marijuana cultivation sites identified with Google Earth™ (from Bauer cite). *Add Google Earth image(s) of trespass grow site(s).*

Table 1: Estimated numbers of outdoor marijuana cultivation sites, outdoor marijuana plants, marijuana greenhouses, marijuana plants in greenhouses, and total number of marijuana plants in four watersheds in Humboldt and Mendocino Counties (from Bauer cite).

	Redwood Creek (67.3 km²)	Salmon Creek (95.6 km²)	Redwood Creek North (730 km²)	Outlet Creek (419 km²)
Total outdoor cultivation sites	231	306	120	633
Total outdoor plants	10,475	11,697	5,984	16,843
Plant : Land area $\left(\frac{1}{km^2}\right)$	173.7	122.4	8.2	40.2
Total Greenhouses	324	302	273	320
Estimated Total Plants in Greenhouses	8,053	8,851	11,029	9,180
Greenhouse : Land Area $\left(\frac{1}{km^2}\right)$	4.8	3.1	0.4	0.8
Total Plants per Watershed*	**18,528**	**20,548**	**17,013**	**26,023**

**Field data indicate these totals may be underestimates.*

Table 2: Estimated daily water use for marijuana cultivation per day and during an entire growing season compared with minimum streamflow in four watersheds in Humboldt and Mendocino Counties (from Bauer cite).

	Redwood Creek	Salmon Creek	Redwood Creek North	Outlet Creek
Daily Water Use (gpd)	111,167	123,289	102,078	156,138
Minimum Streamflow (gpd)	387,763	581,645	---	---
Percent Use	**28.7%**	**21.2%**	---	---
Water Use per 150 Day Season (gal)	16,675,308	18,493,344	15,311,700	23,420,700

Marijuana growers on both public and private lands often illegally clearcut vegetation in order to create growing space for marijuana plants and room for artificial ponds and other structures (Bauer cite, Gabriel et al. 2013; Figure 3). Illegal cutting, along with creation of roads to access grow sites, can increase sedimentation in streams and creeks and thereby degrade habitat quality for aquatic and amphibious animals (Bauer cite). The effect of illegal vegetation clearing on NSOs is unknown. Given the close association of both marijuana cultivation and NSOs with riparian areas and surrounding uplands (e.g., Hamer et al. 2007, Irwin et al. 2007), it is plausible that widespread marijuana cultivation results in habitat loss or fragmentation for NSOs.

Figure 3: Areas cleared for outdoor marijuana cultivation.

From cite.

Clearing of a riparian area for marijuana cultivation at a trespass grow site.
Courtesy of Craig Thompson.

Widespread outdoor marijuana cultivation can further damage watershed health by polluting water and soils. In addition to pesticides, tremendous quantities of fertilizer are often applied at marijuana grow sites (HSVTC 2012, Thompson et al. 2014). Fertilizers, along with low flows caused by drought conditions and water diversion, may contribute to algae blooms and reduced oxygen levels in creeks and rivers (cite). Other pollution, including human waste, trash, and spilled diesel fuel from generators, is also frequently observed in and around streams at raided

outdoor marijuana grow sites (HSVTC 2012, C. Thompson, pers. comm.). Like water diversion and increased stream sedimentation, pollution from outdoor marijuana cultivation is primarily a source of concern for aquatic and amphibious animals but could have indirect ecological effects on NSOs and other wildlife.

Multiple recent wildfires in California have been attributed to marijuana growers. For example, in 2014, a marijuana grower was indicted on charges of starting the Nicolls Fire that burned nearly 1,700 acres in the Sequoia National Forest (SacBee 2014b). This grower allegedly set multiple fires in an attempt to avoid capture by other growers that he claimed were trying to kill him. Another marijuana grower was recently arrested for igniting the 2014 Bully Fire that burned nearly 13,000 acres in Shasta County (SacBee 2014a). This fire was apparently started when the grower's rental truck ignited dry grass when driven off-road in order to deliver soil amendments to a grow site. In 2009, a marijuana grower's camp stove ignited the 90,000-acre La Brea Fire in the Los Padres National Forest in southern California (inciweb). Large wildfires (e.g., thousands of acres) can burn through multiple NSO territories and can negatively affect NSOs in a variety of ways, particularly when they burn large areas at moderate to high severity (reviewed in *Threats: Wildfires*).

Illegal marijuana growers are often heavily armed in order to protect their crops and for poaching wildlife for food or to prevent wildlife damage to plants, equipment, or food caches (Gabriel et al. 2013, Boehm 2014; Figure 4). Many biologists and other field personnel working in California's forests have been interrogated, pursued, or shot at by marijuana growers (Gabriel et al. 2013, D. Hansen pers. obs.). Safety concerns associated with widespread illegal marijuana

cultivation can substantially curtail the ability of researchers and land managers to effectively locate, study, and manage spotted owls and other wildlife. For example, Six Rivers National Forest biologists were repeatedly excluded from entire pre-project NSO survey units in 2013 due to evidence of trespass outdoor marijuana cultivation (D. Hansen pers. obs.). Similarly, Keane et al. (2011) stated that their California spotted owl (*S. o. occidentalis*) survey crew was excluded from large portions of a study area in the Sierra Nevada in 2010 due to extensive illegal marijuana cultivation operations. Gabriel et al. (2013; Gabriel cite) estimated that safety concerns due to trespass outdoor marijuana cultivation resulted in exclusion of researchers from 15-25% of one fisher study area in California and a projected additional cost of $500,000-750,000 for the life of the combined budgets of two of California's fisher research projects. Wildlife surveyors who were able to work alone in the past must now frequently work in pairs for safety reasons, reducing survey efficiency and increasing project costs (Gabriel et al. 2013, D. Hansen pers. obs.). Exclusion from study areas can also compromise the ability of researchers to properly design and complete research investigating important conservation issues, such as effects of pesticides on fishers (Gabriel et al. 2013) and wildfires on spotted owls (Keane et al. 2011).

Figure 4: Armed marijuana growers posing in front of a poached deer (from Gabriel et al. 2013).

Magnitude and Location of Threat

Estimates of marijuana production and value are generally based on either federal marijuana seizure data (e.g., assuming that seizures represent 10 or 15% of the total amount produced) or marijuana consumption surveys and estimates of plant yields and market value (Gettman 2006) (reviewed in PBS 2014). Estimates from both of these methods indicate that California is, by far, the primary marijuana-producing state in the U.S. and that most of this production is from outdoor cultivation (NDIC 2011, Gettman 2006). Gettman (2006) estimated that, in 2006 alone, California produced an estimated 8.6 million pounds of marijuana with a value of more than 13.8 billion dollars. An estimated 89% of this product and value was from outdoor cultivation. If correct, marijuana is the largest cash crop in California (Gettman 2006), which is remarkable given that California is the most productive agricultural state in the U.S. (USDA 2014).

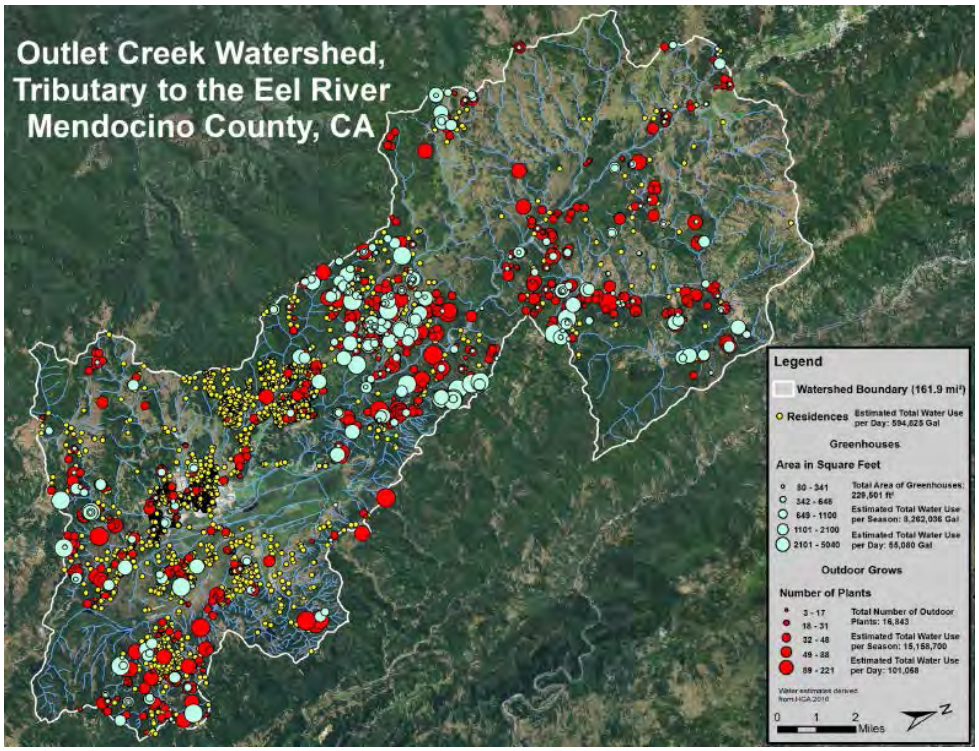
Outdoor marijuana cultivation in California has increased dramatically in recent years, including on both public and private lands (Bauer cite, NDIC 2011). This rapid growth was due to increased demand for domestically grown marijuana; possibly driven by state legalization of marijuana for medical use, changes in public perception of health or legal risk associated with marijuana use, or reduced imports from other countries due to tighter border control measures implemented after 9/11 (NDIC 2007, 2011, SacBee 2012). However, rapid growth of marijuana production in California apparently outstripped consumer demand in the last few years, particularly following federal crackdowns on medical marijuana dispensaries in the state (SacBee 2012). Desire among growers to maintain high profits in the face of increasing supply and decreasing prices could be a factor driving recent increases in the size and intensity (e.g., use of pesticides, fertilizer, and water) of many outdoor marijuana cultivation operations (cite, Bauer cite).

The number of outdoor marijuana plants eradicated in the U.S. increased by 250% between 2005 and 2010 (NDIC 2011). Federal eradication data suggest that trespass outdoor marijuana cultivation is increasing particularly rapidly on National Forests in California (NDIC 2011). Between 2005 and 2013, over 16 million marijuana plants were eradicated at approximately 3,356 sites on National Forests in California (Boehm 2014). Federal agencies have largely attributed increased numbers of trespass outdoor marijuana cultivation sites to expansion of operations by international drug trafficking organizations (e.g., Mexican drug cartels) into remote mountainous areas (particularly in northern California) in order to avoid detection by law enforcement personnel (NDIC 2007, Boehm 2014). However, the degree to which increases in amounts of eradicated or seized marijuana reflect increased production versus increased drug

enforcement effort is unclear, as is the scale of international drug trafficking organizations' role in outdoor marijuana cultivation in California (cite, NDIC 2010).

Outdoor marijuana production in California is also growing rapidly on private lands (NDIC 2007, Bauer cite). Personnel with the California Department of Fish and Wildlife (Bauer cite) used aerial imagery in Google Earth™ to estimate changes in the number and sizes of marijuana cultivation operations in four watersheds in Humboldt and Mendocino Counties during 2009-2012. In 2012 they identified nearly 1,300 outdoor grow sites and more than 1,200 greenhouses likely used for marijuana cultivation in these watersheds (Table 1; e.g., Figure 5). The number and size of marijuana cultivation operations identified increased in all four watersheds by 68-104% between 2009 and 2012. The total number of greenhouses and the number of greenhouses greater than 1,000 ft^2 increased by 69% and 87%, respectively. Continued use of aerial imagery and flyovers will shed greater light on the number, size, and location of outdoor marijuana operations on both public and private lands. For example, the California Department of Fish and Wildlife (2012) noted that law enforcement officers spotted more than 200 new marijuana grow operations in the Mattole Watershed in Humboldt County during a single flyover. *Describe proportion of marijuana cultivation sites identified by Bauer which were located on properties owned by private citizens vs. trespass grows on federal and/or timber co. lands. Incorporate other law enforcement information (e.g., any additional FOIA information from USFS law enforcement).*

Figure 5: Locations and sizes of marijuana cultivation operations identified in the Outlet Creek Watershed in Mendocino County using aerial imagery in Google Earth™ (from Bauer cite).



Summary and Management Implications

There is currently little direct information regarding potential impacts of illegal outdoor marijuana cultivation on NSOs. However, widespread application of ARs and other toxicants at outdoor grow sites are negatively impacting fishers, which have overlapping home ranges and diets with NSOs in northwestern California (Gabriel et al. 2012, 2013, Thompson et al. 2014). There is also evidence of widespread exposure to toxicants among barred owls in Humboldt County, and an NSO from Mendocino County recently tested positive for ARs (Gabriel cite, Calforests 2014). Thus, it is likely that NSOs in California are widely exposed to toxicants

applied at illegal outdoor marijuana grow sites. ARs and other pesticides can directly kill owls and other raptors or increase their vulnerability to other sources of mortality such as predation, disease, parasites, hypothermia, or injury (reviewed above). Furthermore, illegal outdoor marijuana cultivation is apparently causing widespread environmental degradation through toxicant exposure in other animals, reduced streamflows, pollution, poorly planned logging and road construction, and wildlife poaching (Bauer cite, Gabriel et al. 2013). Safety concerns associated with the widespread presence of heavily armed marijuana growers may also be impacting conservation of spotted owls and other wildlife by reducing the efficiency and effectiveness of research and survey efforts (Keane et al. 2011, Gabriel et al. 2013, D. Hansen pers. obs.).

Increased funding and effort are needed for evaluation of effects of outdoor marijuana cultivation on NSOs, other wildlife, and ecosystems (cite). Greater funding and coordination are also needed for interdiction, clean-up, and restoration at illegal outdoor grow sites (cite). These efforts require a substantial, multi-agency law enforcement presence, experts capable of identifying and properly disposing of toxicants, personnel and equipment for removing large amounts of trash and other material, and natural resource specialists for rehabilitating or restoring sites (Gabriel et al. 2013, Boehm 2014). Even if marijuana is legalized in California, tremendous resources, effort, and coordination may still be needed to regulate the industry and to continue to locate, clean up, and restore abandoned or interdicted illegal grow sites. Only a small portion of interdicted outdoor grow sites in California have been cleaned up thus far and even less have been restored (cite). Many of these sites may continue to pose an environmental threat long after they are abandoned by growers (Gabriel et al. 2013). For example, trespass

outdoor growers often cache ARs and other toxicants in water-proof containers, which bears can eventually find and open, allowing further poisoning and exposure of wildlife even after growing operations have ceased at the site (cite, HSVTC 2012).

Comment [MWG17]: They don't really cache these toxicants, the toxicants are manufactured in child proof or waterproof/resistance containers that leave the material still viable for years. This is citable by percom from many folks, including myself and craig.

The Environmental Protection Agency (EPA) and California Department of Pesticide Regulation (CDPR) have recently taken steps to reduce threats from SGARs to wildlife, children, and pets. Since 2011, EPA regulations have prohibited the sale of SGARs to the general consumer. Under EPA regulations, SGARs may only be purchased at agricultural stores, in bait station form rather than as loose pellets, and in relatively small quantities (Bradbury 2008). Most AR manufacturers quickly complied with these regulations and brought replacement products to market containing FGARs or neurotoxins, rather than SGARs (CDPR 2013, Murray 2013). However, the EPA only recently reached an agreement with Reckitt Benckiser to end distribution of their popular d-Con® products containing SGARs and sold without a protective bait station by March 31, 2015 (EPA 2014). On July 1, 2014, the CDPR further limited access to SGAR products in California by classifying them as restricted materials (California Department of Consumer Affairs 2014). In California, products containing SGARs can only be purchased from CDPR-licensed pest control dealers by certified applicators (California Department of Consumer Affairs 2014). Increased restrictions on public access to products containing SGARs should help to reduce exposure of wildlife to these compounds. However, considering that banned pesticides are commonly found at trespass outdoor marijuana cultivation sites (HSVTC 2012, Thompson et al. 2014) growers will likely continue to widely apply SGARs in forests occupied by NSOs, fishers, and other sensitive wildlife.

Comment [MWG18]: Cite Gabriel 2013 for banned pesiticides too. Showing a 3 years of publications documenting these toxicants.

Comment [MWG19]: Or other new emerging toxicants, or massive amounts of legal , high poundage FGARs.

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Outdoor Marijuana Cultivation

Introduction

Although marijuana is perhaps the largest cash crop in California (Gettman 2006), its cultivation is largely unregulated and little is known about its environmental effects. Recent research has indicated that outdoor marijuana cultivation is currently having widespread and profound environmental impacts in California (Bauer cite, Gabriel et al. 2012, 2013, Thompson et al. 2014). Negative impacts of outdoor marijuana cultivation include wildlife deaths caused by pesticide exposure and poaching; habitat degradation caused by logging, road construction, pollution, and water diversion; and heightened safety concerns for research and resource personnel (Bauer cite, Gabriel et al. 2012, 2013, Thompson et al. 2014). The specific effects of outdoor marijuana cultivation on northern spotted owls (*Strix occidentalis caurina*; NSOs) are unknown. Recent findings of widespread pesticide exposure among fishers (*Martes pennanti*) and barred owls (*Strix varia*) in northwestern California suggest that NSOs in the state are likewise exposed and could be negatively affected at both territory and population levels (Gabriel cite, Gabriel et al. 2012; reviewed below). NSOs could also be directly affected by environmental degradation from outdoor marijuana cultivation, such as through habitat modification or suppression of rodent prey populations, or indirectly affected through ecological changes caused by reduced streamflows or pollution. Safety concerns associated with illegal marijuana cultivation may also be impacting NSOs and other wildlife through reduced research and survey efficiency and effort (Gabriel et al. 2013).

Pesticides

Pesticide application is usually intended to suppress populations of rodents, insects, mollusks, and other agricultural and urban pests, but can have inadvertent negative impacts on humans, pets, and other non-target animals (Erickson and Urban 2004, Albert et al. 2010, Mnif et al. 2011, Gabriel et al. 2012). Widespread secondary exposure to pesticides has been reported for raptors, carnivores, and other wildlife that consume poisoned rodents around farms and human dwellings (Albert et al. 2010, Murray 2013, Thompson et al. 2014). Researchers have generally assumed that pesticides pose little threat to wildlife outside of agricultural and urban areas (Gabriel et al. 2013). However, a recent publication reported that 79% of fishers tested in two study areas on federal and tribal forest lands in California had been exposed to anticoagulant rodenticides (ARs), including four that apparently died from lethal toxicosis (Gabriel et al. 2012; note: at least two more fishers in California died from AR poisoning following publication of this study: Gabriel et al. 2013). Most fishers in the study had been exposed to multiple AR compounds (range = 1-4, mean = 1.6). These findings not only raised concern for the West Coast fisher population, which the U.S. Fish and Wildlife Service recently proposed to list as threatened under the federal Endangered Species Act, but for the NSO, which has an overlapping distribution and diet with the fisher (Gabriel et al. 2013, Calforests 2014, USFWS 2014). Subsequently, ARs have been detected in a dead NSO recovered in Mendocino County (Calforests 2014) and xx of xx (xx%) barred owls tested for exposure in Humboldt County (Gabriel cite). Although barred owls are being tested as a proxy for NSOs (Gabriel cite), NSOs may be more widely exposed to ARs given their greater dietary specialization on rodents (see USFWS 2013). Strong circumstantial evidence implicates pervasive illegal outdoor marijuana

cultivation as the primary source of pesticide exposure for forest predators in California (Gabriel et al. 2012, 2013, Thompson et al. 2014).

ARs detected in forest predators in northern California include brodifacoum, bromadiolone, difethialone, chlorophacinone, diphacinone, warfarin, and coumachlor (Gabriel cite, Gabriel et al. 2012). Brodifacoum, bromadiolone, difethialone are classified as second-generation anticoagulant rodenticides (SGARs). SGARs were introduced in the 1970s due to widespread development of resistance among rodents to first-generation anticoagulant rodenticides (FGARs), such as warfarin, chlorophacinone, and diphacinone (Buckle et al. 1994). SGARs are more acutely toxic than FGARs and generally require only a single dose to kill rodents (Erickson and Urban 2004). However, rodents usually survive 5-10 days after consuming a lethal dose, during which time they may continue to consume additional rodenticide and remain available to predators (Cox and Smith 1992, Erickson and Urban 2004). SGARs are more persistent in animal tissues than FGARs and insecticides, which are more rapidly metabolized and excreted (Erickson and Urban 2004). Thus, exposure to FGARs and other non-SGAR pesticides is more difficult to detect than for SGARs and exposure to them could be underestimated (Albert et al. 2010, Thompson et al. 2014). Pesticides are often applied along with large quantities of fertilizer at the base of marijuana plants grown outdoors, suggesting that marijuana plants may be taking up pesticidal compounds from the soil (Thompson et al. 2014; Figure 1). ~~If this occurs, NSOs and other wildlife could be exposed to pesticides through consumption of insects and rodents that eat marijuana plants, as well as by eating rodents that ingest AR bait (cite).~~

Comment [UFS1]: Can't cite me here, I did not talk about uptake in the paper. As far as I know, this is strongly suspected but hasn't been proven.

Large quantities of ARs, particularly SGARs, are often spread across large areas in and around illegal outdoor marijuana grow sites (Gabriel et al. 2012, Thompson et al. 2014; Figure 1). Gabriel et al. (2012) noted that thousands of pounds of pesticides were found at illegal outdoor marijuana grow sites in California in 2008 and that 150 pounds of pesticide were found during a single three-week eradication operation on the Mendocino National Forest in 2011. Three sites raided in Humboldt County in 2013 contained a total of at least 17 pounds of SGAR bait, which researchers estimated was sufficient to kill 2,753 woodrats, 14 fishers, or five spotted owls (Humboldt Sentinel August 2, 2013). Other pesticides, such as organochlorine, organophosphate, and carbamate insecticides, some of which are banned in the U.S., are also frequently found at illegal outdoor marijuana grow sites (~~cite~~HSVTC 1012, Thompson et al. 2014).

Figure 1: (A) Rodenticide and other pesticides found at a trespass outdoor marijuana grow site in Humboldt County (photo: Humboldt County Sheriff's Office) and (B) rodenticide bait and dry fertilizer strewn together below approximately 2,000 marijuana plants at a trespass outdoor grow site in Humboldt County (from Gabriel et al. 2012).

ARs are vitamin K antagonists, which cause impairment of the blood's ability to clot (Murray 2013). A lethal dose of AR causes animals to die from hemorrhage (Erickson and Urban 2004). Animals may also exhibit weakness prior to death or with a sublethal dose (Erickson and Urban 2004). Rodents exposed to ARs show altered behavior, such as spending more time in the open, freezing rather than bolting when threatened, and staggering (Cox and Smith 1992). These behaviors may increase predation risk for affected rodents and the opportunity for secondary exposure of predators to ARs (Cox and Smith 1992). Owls and other raptors with sublethal secondary exposure to ARs may often have reduced blood-clotting activity and can die from

minor wounds such as those commonly inflicted by prey (Erickson and Urban 2004, Murray 2013, Thompson et al. 2014). It is also worth noting that invertebrates respond differently and are not negatively impacted by the uptake of vitamin K antagonists. They are therefore capable of accumulating both FGAR and SGAR compounds either through direct consumption or potentially or the consumption of vegetation where uptake of pesticidal compounds has occurred. NSOs and other wildlife may therefore be exposed to pesticides through consumption of live or dead insects in the vicinity of grow sites.

Non-AR pesticides have a variety of physiological effects, such as disrupting endocrine function or damaging the central nervous system (Grue et al. 1997, Mnif et al. 2011). Chronic or sublethal exposure to carbamate or organophosphate pesticides has been shown to reduce immune response, cause neurological disorders, reduce thermoregulatory control, and impair anti-predator behavior in wildlife (Grue et al. 1997; reviewed in Thompson et al. 2014). Pesticides can also have additive or synergistic effects on animals (Larsen et al. 2003 cited in Mnif et al. 2011). This is a source of additional concern for NSOs and other forest predators active near outdoor marijuana cultivation sites, where multiple types of pesticide are often present (HSVTC 2012, Thompson et al. 2014). Mortalities associated with exposure to pesticides are likely underestimated because carcasses are often predated or scavenged before biological samples can be obtained and because sublethal exposure can predispose wildlife to death from other causes (cite, Albert et al. 2010).

There is no information available concerning population-level impacts of secondary pesticide exposure for NSOs in California. As reviewed above, secondary exposure to pesticides can kill

raptors and other wildlife both through lethal toxicosis and by increasing the risk of mortality due to other factors such as predation, hypothermia, disease, parasites, or injury. NSO population rates are particularly sensitive to changes in adult survival (Noon and Biles 1990) so it is possible that direct and indirect mortalities associated with pesticide exposure could contribute to population declines (Sibley et al. 2000~~cite~~, Thompson et al. 2014). Pesticide exposure could also negatively impact NSO reproduction. Gabriel et al. (2012) noted that fisher mortalities caused by AR poisoning occurred between mid-April and mid-May. The timing of these deaths coincided with the planting phase of outdoor marijuana cultivation, when seedlings are most vulnerable to rodent pests and AR use is likely highest (Gabriel et al. 2012). This time of year is also when NSOs incubate and brood young (cite). Reduced parental care during this phase, for example due to compromised behavior or death of one or both parents, could result in death of offspring due to chilling, undernourishment, or predation (Grue et al. 1997~~cite~~). NSO populations could also be negatively impacted by pesticide suppression of prey populations or changes in community ecology caused by reductions of insects, small mammals, carnivores, raptors, amphibians, and aquatic animals (Relyea and Diecks 2008~~cite~~).

Other Environmental Effects

Activities related to outdoor marijuana cultivation can have a variety of environmental impacts beyond exposure of wildlife to pesticides, including negative effects of illegal and poorly planned water diversion, logging, and road construction; pollution; poaching; and ignition of wildfires (reviewed below). Negative impacts occur on both public and private lands (cite, Bauer cite, Gabriel et al. 2013). The potential environmental impacts of marijuana cultivation

could vary considerably, depending on the operation's location, scale, and practices. We are unaware of any information regarding effects of ecological degradation from marijuana cultivation on NSOs. Negative effects on NSOs are possible given the subspecies' sensitivity to habitat modification (cite) and close association with riparian areas (cite), where the impacts of marijuana cultivation are often concentrated. Furthermore, safety concerns associated with marijuana cultivation can substantially impact the ability of land managers to effectively survey and manage spotted owls and other sensitive wildlife (Keane et al. 2011, Gabriel et al. 2013, D. Hansen pers. obs.).

Personnel with the California Department of Fish and Wildlife (Bauer cite) recently estimated hydrologic impacts of marijuana cultivation in northwestern California using high-resolution aerial imagery in Google Earth™ (e.g., Figure 2) and marijuana industry estimates of marijuana plant water requirements. Using these methods, they estimated that more than 82,000 marijuana plants were cultivated in 2012 in just four watersheds in Humboldt and Mendocino Counties (Table 1). Based on estimated numbers of marijuana plants and assumed water usage of 6 gallons per day per plant, they calculated that marijuana cultivation uses approximately 29% and 21% of the water that flows in the Redwood Creek and Salmon Creek watersheds per day during periods of minimum streamflow (Table 2). Although based on several assumptions (marijuana cultivation water sources and usage, complete visibility of cultivation sites in aerial imagery, complete usage of greenhouses), these estimates raised considerable concern about potential negative impacts of marijuana cultivation on watershed health and aquatic animals (cite). This concern is currently heightened given California's ongoing severe drought (cite). NSOs often exhibit a preference for nesting, roosting, and foraging in and near riparian areas (cite). Thus, it

is plausible that ecological changes caused by widespread water diversion for marijuana cultivation have negative indirect effects on NSOs.

Figure 2: Outdoor (A) and greenhouse (B) marijuana cultivation sites identified with Google Earth™ (from Bauer cite). *Add Google Earth image(s) of trespass grow site(s).*

Table 1: Estimated numbers of outdoor marijuana cultivation sites, outdoor marijuana plants, marijuana greenhouses, marijuana plants in greenhouses, and total number of marijuana plants in four watersheds in Humboldt and Mendocino Counties (from Bauer cite).

	Redwood Creek (67.3 km²)	Salmon Creek (95.6 km²)	Redwood Creek North (730 km²)	Outlet Creek (419 km²)
Total outdoor cultivation sites	231	306	120	633
Total outdoor plants	10,475	11,697	5,984	16,843
Plant : Land area $\left(\frac{1}{km^2}\right)$	173.7	122.4	8.2	40.2
Total Greenhouses	324	302	273	320
Estimated Total Plants in Greenhouses	8,053	8,851	11,029	9,180
Greenhouse : Land Area $\left(\frac{1}{km^2}\right)$	4.8	3.1	0.4	0.8
Total Plants per Watershed*	**18,528**	**20,548**	**17,013**	**26,023**

**Field data indicate these totals may be underestimates.*

Table 2: Estimated daily water use for marijuana cultivation per day and during an entire growing season compared with minimum streamflow in four watersheds in Humboldt and Mendocino Counties (from Bauer cite).

	Redwood Creek	Salmon Creek	Redwood Creek North	Outlet Creek
Daily Water Use (gpd)	111,167	123,289	102,078	156,138
Minimum Streamflow (gpd)	387,763	581,645	---	---
Percent Use	**28.7%**	**21.2%**	---	---
Water Use per 150 Day Season (gal)	16,675,308	18,493,344	15,311,700	23,420,700

Marijuana growers on both public and private lands often illegally clearcut vegetation in order to create growing space for marijuana plants and room for artificial ponds and other structures (cite, Gabriel et al. 2013; Figure 3). Illegal cutting, along with creation of roads to access grow sites, can increase sedimentation in streams and creeks and thereby degrade habitat quality for aquatic and amphibious animals (cite). The effect of illegal vegetation clearing on NSOs is unknown. Given the close association of both marijuana cultivation and NSOs with riparian areas and surrounding uplands, it is plausible that widespread marijuana cultivation results in habitat loss or fragmentation for NSOs.

Figure 3: Area cleared for outdoor marijuana cultivation (from cite). *Find and add photo from trespass grow site adjacent to creek.*

Widespread outdoor marijuana cultivation can further damage watershed health by polluting water and soils. In addition to pesticides, tremendous quantities of fertilizer are often applied at marijuana grow sites (~~cite~~HSVTC 2012, Thompson et al. 2014). Fertilizers, along with low flows caused by drought conditions and water diversion, can contribute to algae blooms and reduced oxygen levels in creeks and rivers (cite). Other pollution, including human waste, trash, and spilled diesel fuel from generators, is also frequently observed in and around streams at raided outdoor marijuana grow sites (~~cite~~HSVTC 2012, C. Thompson pers com). Like water diversion and increased stream sedimentation, pollution from outdoor marijuana cultivation is primarily a source of concern for aquatic and amphibious animals but could have indirect ecological effects on NSOs and other wildlife.

Multiple recent wildfires in California have been attributed to marijuana growers. For example, in 2014, a marijuana grower was indicted on charges of starting the Nicolls Fire that burned nearly 1,700 acres in the Sequoia National Forest (SacBee 2014b). This grower allegedly set

multiple fires in an attempt to avoid capture by other growers that he claimed were trying to kill him. Another marijuana grower was recently arrested for igniting the 2014 Bully Fire that burned nearly 13,000 acres in Shasta County (SacBee 2014a). This fire was apparently started when the grower's rental truck ignited dry grass when driven off-road in order to deliver soil amendments to a grow site. In 2009, a marijuana grower's camp stove ignited the 90,000-acre La Brea Fire in the Los Padres National Forest in southern California (inciweb). Large wildfires (e.g., thousands of acres) can burn through multiple NSO territories and can negatively affect NSOs in a variety of ways, particularly when they burn large areas at moderate to high severity (reviewed in *Threats: Wildfires*).

Illegal marijuana growers are often heavily armed in order to protect their crops and for poaching wildlife for food or to prevent damage to plants, equipment, or food caches (Boehm cite, Gabriel et al. 2013; Figure 4). Many biologists and other field personnel working in California's forests have been interrogated, pursued, or shot at by marijuana growers (cite, Gabriel et al. 2013, D. Hansen pers. obs.). Safety concerns associated with widespread illegal marijuana cultivation can substantially curtail the ability of researchers and land managers to effectively locate, study, and manage spotted owls and other wildlife. For example, Six Rivers National Forest biologists were repeatedly excluded from entire pre-project NSO survey units in 2013 due to evidence of trespass outdoor marijuana cultivation (D. Hansen pers. obs.). Similarly, Keane et al. (2011) stated that their California spotted owl (*S. o. occidentalis*) survey crew was excluded from large portions of a study area in the Sierra Nevada in 2010 due to extensive illegal marijuana cultivation operations. Gabriel et al. (2013; Gabriel cite) estimated that safety concerns due to trespass outdoor marijuana cultivation resulted in exclusion of researchers from 15-25% of one

fisher study area in California and a projected additional cost of $500,000-750,000 for the life of the combined budgets of two of California's fisher research projects. Wildlife surveyors who were able to work alone in the past must now frequently work in pairs for safety reasons, reducing survey efficiency and increasing project costs (Gabriel et al. 2013, D. Hansen pers. obs.). Exclusion from study areas can also compromise the ability of researchers to properly design and complete research investigating important conservation issues, such as effects of pesticides on fishers (Gabriel et al. 2013) and wildfires on spotted owls (Keane et al. 2011).

Figure 4: Armed marijuana growers posing in front of a poached deer (from Gabriel et al. 2013).

Magnitude and Location of Threat

Estimates of marijuana production and value are generally based on either federal marijuana seizure data (e.g., assuming that seizures represent 10% of the total amount produced) or marijuana consumption surveys and estimates of plant yields and market value (Gettman 2006) (reviewed in PBS 2014). Estimates from both of these methods indicate that California is, by

far, the primary marijuana-producing state in the U.S. and that most of this production is from outdoor cultivation (cite, NDIC 2011, Gettman 2006). Gettman (2006) estimated that, in 2006 alone, California produced an estimated 8.6 million pounds of marijuana with a value of more than 13.8 billion dollars. An estimated 89% of this product and value was from outdoor cultivation. If correct, marijuana is the largest cash crop in California (Gettman 2006), which is remarkable given that California is the most productive agricultural state in the U.S. (USDA 2014).

Outdoor marijuana cultivation in California has increased dramatically in recent years, including on both public and private lands (cite, Bauer cite, NDIC 2011). This rapid growth was due to increased demand for domestically grown marijuana; possibly driven by state legalization of marijuana for medical use, changes in public perception of health or legal risk associated marijuana use, or reduced imports from other countries due to tighter border control measures implemented after 9/11 (cite, NDIC 2007, 2011, SacBee 2012). However, rapid growth of marijuana production in California apparently outstripped consumer demand in the last few years, particularly following federal crackdowns on medical marijuana dispensaries in the state (SacBee 2012). Desire among growers to maintain high profits in the face of increasing supply and decreasing prices could be a factor driving recent increases in the size and intensity (e.g., use of pesticides, fertilizer, and water) of many outdoor marijuana cultivation operations (cite, Bauer cite).

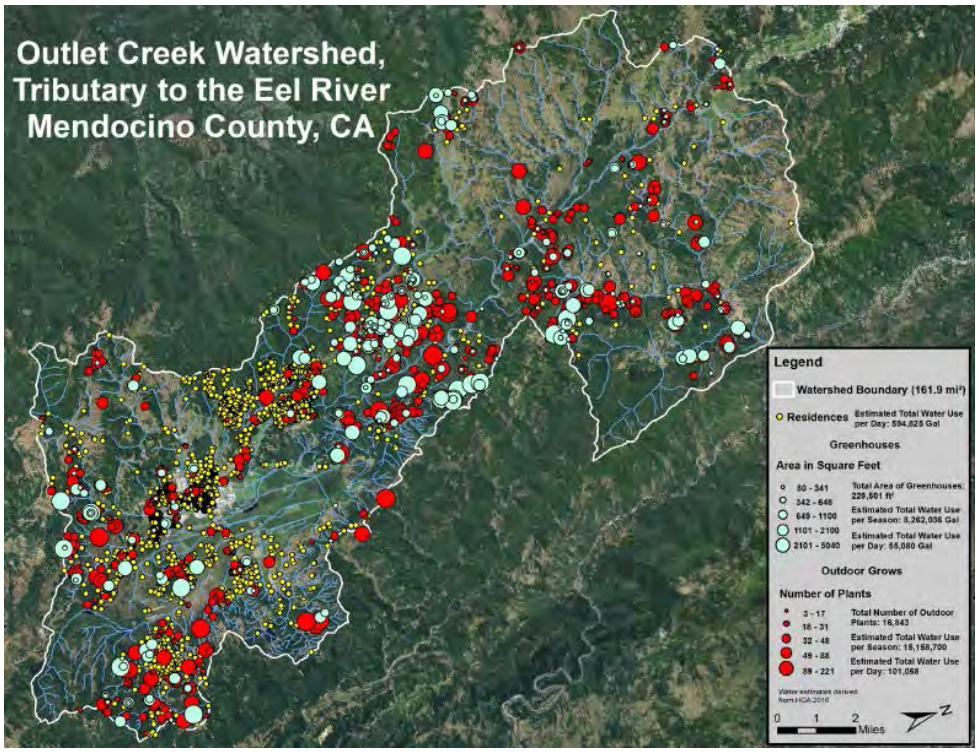
The number of outdoor marijuana plants eradicated in the U.S. increased by 250% between 2005 and 2010 (NDIC 2011). Federal eradication data suggest that trespass outdoor marijuana

cultivation is increasing particularly rapidly on National Forests in California (NDIC 2011). Between 2005 and 2013, over 16 million marijuana plants were eradicated at approximately 3,356 sites on National Forests in California (Boehm 2014). Federal agencies have largely attributed increased numbers of trespass outdoor marijuana cultivation sites to expansion of operations by international drug trafficking organizations (e.g., Mexican drug cartels) into remote mountainous areas (particularly in northern California) in order to avoid detection by law enforcement (NDIC 2007, Boehm cite). However, the degree to which increases in amounts of eradicated or seized marijuana reflect increased production versus increased drug enforcement effort is unclear, as is the scale of international drug trafficking organizations' role in outdoor marijuana cultivation in California (cite, NDIC 2010).

Outdoor marijuana production in California is also growing rapidly on private lands (NDIC 2007, Bauer cite). Personnel with the California Department of Fish and Wildlife (Bauer cite) used aerial imagery in Google Earth™ to estimate changes in the number and sizes of marijuana cultivation operations in four watersheds in Humboldt and Mendocino Counties during 2009-2012. In 2012 they identified nearly 1,300 outdoor grow sites and more than 1,200 greenhouses likely used for marijuana cultivation in these watersheds (Table 1; e.g., Figure 5). The number and size of marijuana cultivation operations identified increased in all four watersheds by 68-104% between 2009 and 2012. The total number of greenhouses and the number of greenhouses greater than 1,000 ft^2 increased by 69% and 87%, respectively. Continued use of aerial imagery and flyovers will shed greater light on the number, size, and location of outdoor marijuana operations on both public and private lands. For example, the California Department of Fish and Wildlife (2012) noted that law enforcement officers spotted more than 200 new marijuana grow

operations in the Mattole Watershed in Humboldt County during a single flyover. *Describe proportion of marijuana cultivation sites identified by Bauer which were located on properties owned by private citizens vs. trespass grows on federal and/or timber co. lands. Incorporate other law enforcement information (e.g., FOIA information from USFS law enforcement).*

Figure 5: Locations and sizes of marijuana cultivation operations identified in the Outlet Creek Watershed in Mendocino County using aerial imagery in Google Earth™ (from Bauer cite).



Summary and Management Implications

There is currently little direct information regarding potential impacts of illegal outdoor marijuana cultivation on NSOs. However, widespread application of ARs and other toxicants at

outdoor grow sites are negatively impacting fishers, which have overlapping home ranges and diets with NSOs in northwestern California (Gabriel et al. 2012, 2013, Thompson et al. 2014). There is also evidence of widespread exposure to toxicants among barred owls in Humboldt County, and an NSO from Mendocino County recently tested positive for ARs (Gabriel cite, Calforests 2014). Thus, it is likely that NSOs in California are widely exposed to toxicants applied at illegal outdoor marijuana grow sites. ARs and other pesticides can directly kill owls and other raptors or increase their vulnerability to other sources of mortality such as predation, disease, parasites, hypothermia, or injury (reviewed above). Furthermore, illegal outdoor marijuana cultivation is apparently causing widespread environmental degradation through toxicant exposure in other animals, reduced streamflows, pollution, poorly planned logging and road construction, and wildlife poaching (cite, Bauer cite, Gabriel et al. 2013). Safety concerns associated with the widespread presence of heavily armed marijuana growers may also be impacting conservation of spotted owls and other wildlife by reducing the efficiency and effectiveness of research and survey efforts (Keane et al. 2011, Gabriel et al. 2013, D. Hansen pers. obs.).

Increased funding and effort are needed for evaluation of effects of outdoor marijuana cultivation on NSOs, other wildlife, and ecosystems (cite). Greater funding and coordination are also needed for interdiction, clean-up, and restoration at illegal outdoor grow sites (cite). These efforts require a substantial, multi-agency law enforcement presence, experts capable of identifying and properly disposing of toxicants, personnel and equipment for removing large amounts of trash and other material, and natural resource specialists for rehabilitating or restoring sites (Gabriel et al. 2013, Boehm 2014). Even if marijuana is legalized in California,

tremendous resources, effort, and coordination may still be needed to regulate the industry and to continue to locate, clean up, and restore abandoned or interdicted illegal grow sites. Only a small portion of interdicted outdoor grow sites in California have been cleaned up thus far and even less have been restored (cite). Many of these sites may continue to pose an environmental threat long after they are abandoned by growers (Gabriel et al. 2013). For example, trespass outdoor growers often cache ARs and other toxicants in water-proof containers, which bears can eventually find and open, allowing further poisoning and exposure of wildlife even after growing operations have ceased at the site (cite).

The Environmental Protection Agency (EPA) and California Department of Pesticide Regulation (CDPR) have recently taken important steps to reduce threats from SGARs to wildlife, children, and pets. Since 2011, EPA regulations have prohibited the sale of SGARs to the general consumer. Under EPA regulations, SGARs may only be purchased at agricultural stores, in bait station form rather than as loose pellets, and in relatively small quantities (Bradbury 2008). Most AR manufacturers quickly complied with these regulations and brought replacement products to market containing FGARs or neurotoxins, rather than SGARs (CDPR 2013, Murray 2013). However, the EPA only recently reached an agreement with Reckitt Benckiser to end distribution of their popular d-Con® products containing SGARs and sold without a protective bait station by March 31, 2015 (EPA 2014). On July 1, 2014, the CDPR further limited access to SGAR products in California by classifying them as restricted materials (California Department of Consumer Affairs 2014). In California, products containing SGARs can only be purchased from CDPR-licensed pest control dealers by certified applicators (California Department of Consumer Affairs 2014). Increased restrictions on public access to products containing SGARs

should help to reduce exposure of wildlife to these compounds. However, considering that banned pesticides are commonly found at trespass outdoor marijuana cultivation sites (HSVTC 2012~~cite~~, Thompson et al. 2014), growers will likely continue to apply SGARs in forests occupied by NSOs, fishers, and other sensitive wildlife.

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Ch. 1: Status and Trends in California

Introduction

Several lines of evidence are available for evaluating the northern spotted owl's (*Strix occidentalis caurina*; NSO) status and trends in California. These include changes in the subspecies' geographic range and distribution, density and abundance, occupancy and demographic rates, meta-population dynamics, and genetics. The most reliable information available for examining the NSO's status and trends is provided by long-term demographic studies. Data from these studies are periodically analyzed together in meta-analyses for describing larger demographic patterns within ecoregions and states. A new demographic meta-analysis is expected to be released in mid-2015. This document will provide the best available information for determining the NSO's current status and trends. However, only a small portion of the NSO's range in California occurs within demographic study areas and those study areas all occur in relatively productive forests in the northwestern part of the state. It is therefore important to consider other sources of information, such as data collected for monitoring NSOs in National Parks and on industrial timber company lands located in different regions of northern California. The NSO's status and trends likely vary among regions, forest types, and ownerships and could be influenced by a host of factors, such as differences in forest ecology, management history, and stressors such as competition with invasive barred owls (*Strix varia*). It is also important to remember that available sources of information for evaluating the NSO's status and trends in California vary substantially in terms of their purpose and scientific rigor.

Range

The current range of the NSO includes southwestern British Columbia and the Cascade Mountains, coastal ranges, and intervening forests of Washington, Oregon, and California (USFWS 2011a). In California, the NSO's range extends from the Oregon border through the Northern Coast Ranges to Marin County, across the Klamath Mountains, and down the southern Cascades to the vicinity of the Pit River, where it contacts the range of the California spotted owl (*S. o. occidentalis*) (Figure 1.1).

The precise historical range of the NSO is unknown. Thus, despite substantial loss, degradation, and fragmentation of NSO habitat (see Chapters 2 and 3 of this volume), there is no evidence that the subspecies' range has contracted since Euro-American settlement (Thomas et al. 1990). However, British Columbia's NSO population has declined to very low numbers and is highly vulnerable to extirpation (Chutter et al. 2004). NSO populations in the Cascades and Olympic Peninsula of Washington and the Northern Coast Range of Oregon are also rapidly declining and may become vulnerable to extirpation (Forsman et al. 2011; see *Demography*, below). Loss of NSO populations could cause substantial contraction of the subspecies' range. For example, extirpation of NSOs from British Columbia alone would reduce the subspecies' range by approximately 8% (Cooper 2006).

Comment [LVD1]: Yes, and we also can't rule out that there may have been some localized expansions of the species range. Historically, prairies were much more extensive in coastal CA (see Redwood National Parks prairie management plan). In fact, the coastal prairies that occur primarily on the ridges and south-facing slopes in the redwood region are part of the California coast grassland that was ranked as one of the most endangered ecosystems in America (Noss and Peters 1995 report on endangered ecosystems). Currently, there are NSO living in prairie intrusion forests throughout coastal CA that would not have been forest lands at all 100+ years ago.

Distribution

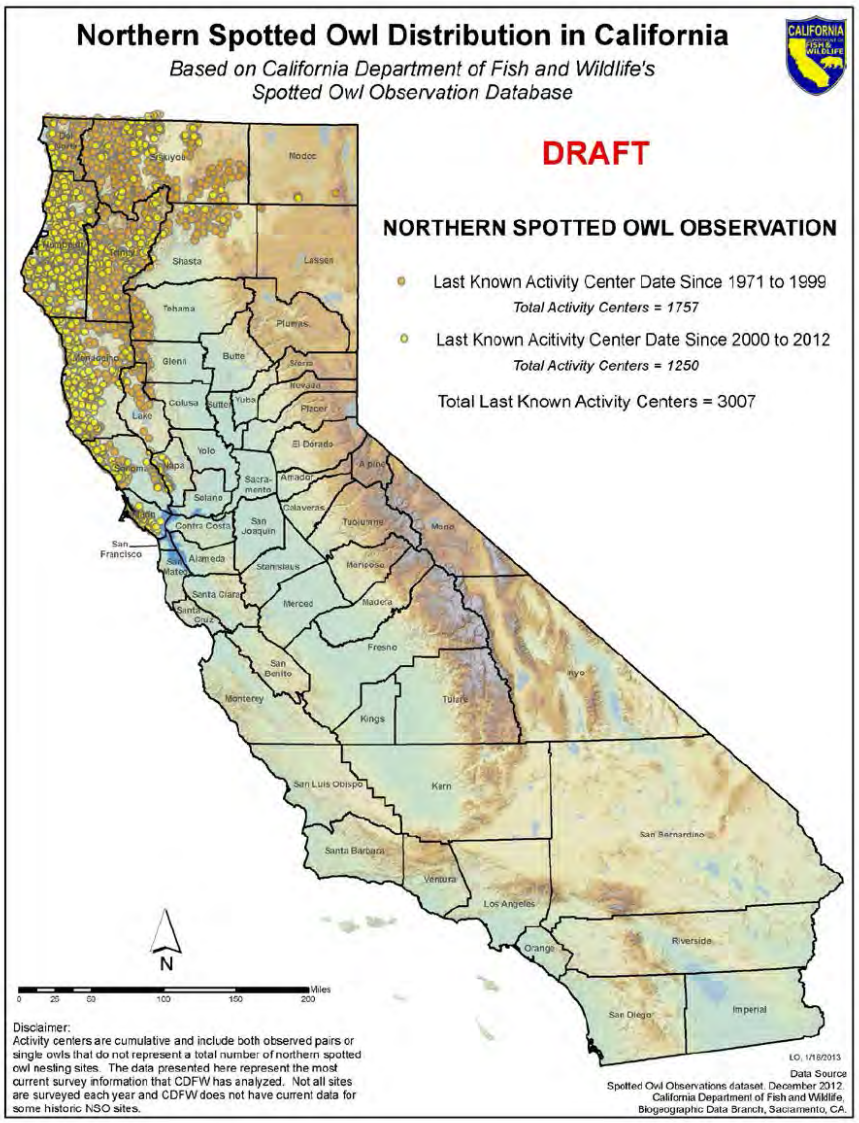
NSOs are thought to have been well distributed throughout most coniferous forests in the Pacific Northwest and northwestern California prior to Euro-American settlement (USFWS 2011a). The abundance and distribution of NSOs have likely declined due to removal of most (ca. 60-88%) old forest within its range (USFWS 1990; see Chapter 2 of this volume). For example, the Puget Trough in Washington and the Willamette Valley in Oregon no longer support NSOs due to land conversion and timber harvesting and very few NSOs remain in British Columbia (Thomas et al. 1990). The NSO's distribution has decreased in other areas of Washington and Oregon as well, due primarily to negative effects of timber harvesting, wildfires, and competition with barred owls (*Strix varia*) (Thomas et al. 1990).

Comment [LVD2]: I think the statement is accurate, but it doesn't seem like a 1990 publication would be the best source to support the conclusion. I would recommend using several publications including the status review (Courtney et al. 2004), Revised NSO recovery plan and possibly Forsman et al. 2011.

It is unknown if the NSO's distribution has changed in California. A difference is evident in the distribution of known historically (1971-1999) and recently (2000-2012) occupied activity centers (ACs) in the Eastern Klamath, Interior Northern Coast Ranges, and Southern Cascades regions of the state (Figure 1.1; see USFWS 2011a Appendix C for ecoregional boundaries generally followed in this synthesis). It is unclear from these data, however, whether the distribution of NSOs has in fact decreased in these areas or if the apparent decline in distribution is due to some other factor such as decreased survey effort or reporting of detections. It is also possible that this difference is due to the greater number of years included in the historical period than in the recent period (29 vs. 13 yrs). However, the two periods are similar in length relative to federal listing of the NSO (10 vs. 13 yrs) when survey effort presumably became more intensive and widespread. Some portions of the Klamath, Interior Northern Coast Ranges, and Southern Cascades have experienced widespread intensive timber harvesting or large wildfires, which could have reduced the NSO's distribution (see Chapters 2 and 3 of this volume). These forms of disturbance, along with competition with invasive barred owls, have likely contributed to declining occupancy by NSOs in some areas of California (see *Occupancy*, below). Nonetheless, the Klamath and Interior Northern Coast Ranges (but not the Southern Cascades) still appear to contain relatively large amounts of well connected suitable habitat and likely function as crucial population sources for NSOs (Schumaker et al. 2014; see *Source-Sink Dynamics*, below).

Comment [LVD3]: In balance, I think it should be mentioned that some level of disturbance in portion of the NSO range in CA contributes to increased habitat heterogeneity that actually improves habitat fitness for NSO (Franklin et al. 2000). This same phenomenon has been demonstrated by Olson et al. 2004, Hoopa and Green Diamond (10-year status review).

Figure 1.1: Distribution of northern spotted owl activity centers in the California Department of Fish and Wildlife's Spotted Owl Observation Database (from California Department of Fish and Wildlife 2013).



Density and Abundance

Species are rarely uniformly distributed across their range. Knowledge of variation in the density and abundance of NSOs is of potential conservation value because it can help identify areas where limited conservation resources should be focused. For example, while declines in low-abundance areas may be more likely to cause contraction of a species' range or distribution (see *Range* and *Distribution*, above), declines in high-abundance areas may disproportionately impact the species' probability of long-term persistence; particularly when high-abundance areas function as population sources (Pulliam 1988, Rodríguez 2002, Schumaker et al. 2014; see *Source-Sink Dynamics*, below).

Several studies have estimated either crude densities (owls or occupied territories per unit area) or ecological densities (owls per unit area of specified habitat class[es]) of NSOs in California (Blakesley et al. 2004; Table 1.1). These estimates are interesting in that they appear to reflect geographic variation in the ecology of NSOs (see below). However, they have limited utility for evaluating the NSO's status or trends in California. Available density estimates for the state are largely restricted to relatively mesic areas of northwestern California, which differ ecologically from drier interior forests (e.g., in terms of climate, forest productivity, and prey communities). Inferences from most density estimates are also limited because they are based on empirical counts of unmarked NSOs, which can bias estimates (Franklin et al. 1990, Diller and Thome 1999). Many of the currently available density and occupancy estimates for NSOs in California were provided by timber companies (Tables 1.1 and 1.2). While potentially useful for evaluating effects of management activities on timber company lands, these estimates do not describe population trends. Rigorous evaluation of NSO population trends require~~s~~ ~~adequate~~ long term statistically valid sampling designs ~~effort~~ from which with estimates of abundance or population lambda with confidence intervals can be repeatedly obtained within the same study area ~~year after year~~. In contrast, timber companies generally shift their NSO survey areas over time as timber harvest projects are completed in some areas and begun in others.

Based on limited information, both crude and ecological densities of NSOs appear to be substantially higher in northwestern California than in the Oregon Coast Ranges (Blakesley et al. 2004; Table 1.1). Lower densities in the Oregon Coast Ranges could be partially related to widespread intensive timber harvesting, which apparently contributed to a major decline in densities during the early 1990s (Thrailkill et al. 1998). Some areas of northwestern California have also experienced widespread intensive timber harvesting (see Chapter 2 of this volume) but its effect on NSOs might have differed from that in the Oregon Coast Ranges. In general, NSOs in California primarily subsist on dusky-footed woodrats (*Neotoma fuscipes*) (in terms of biomass contribution to diets). NSOs that primarily subsist on dusky-footed woodrat often have smaller home ranges, and apparently occur at higher densities, than those that primarily rely on smaller-bodied prey (Carey et al. 1992, Zabel et al. 1995). Furthermore, in contrast with other primary prey species, such as northern flying squirrels (*Glaucomys sabrinus*) and tree voles (*Arborimus* spp.), dusky-footed woodrats seem to respond positively, albeit temporarily, to some forms of intensive timber harvesting (see Chapter 2 of this volume).

Comment [LVD4]: Again this is where the issue of habitat heterogeneity (Franklin et al. 2000) should be mentioned in terms of the differential NSO response in NW CA versus areas where NSO feed primarily on flying squirrels. The mixture of older stands for roosting and nesting and young stands for woodrats provides the best habitat. As you noted, this woodrat response is only temporary, which makes the best NSO habitat highly dynamic in this region.

Densities of NSOs are also thought to be higher in northwestern California than in the state's interior (Calforests 2014). However, there are apparently only two density estimates currently

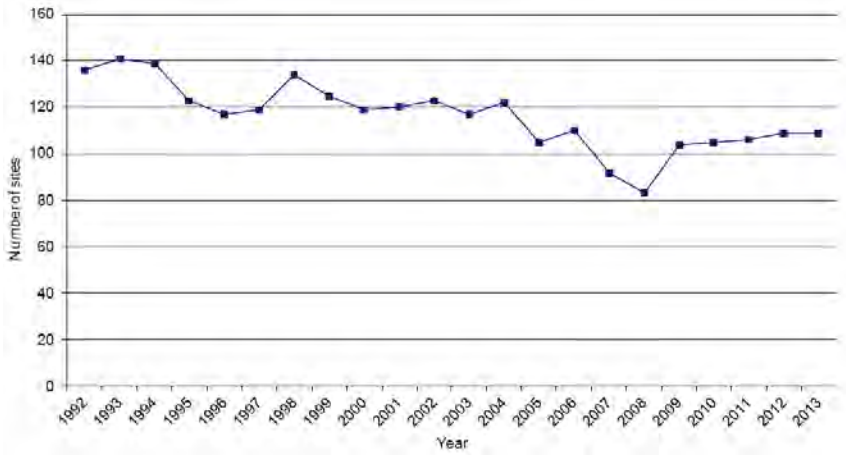
available for interior northern California and both of these were for crude densities of occupied territories, rather than for individual owls as estimated by most studies in northwestern California (Table 1.1). These crude territory densities are substantially lower than those found on two timber companies' lands in the Redwood Region (Table 1.1). NSO densities may be relatively low in the Southern Cascades of California due to the prevalence of drier, less productive forests, a history of widespread intensive harvesting, and effects of recent large wildfires (see Chapters 2 and 3 of this volume). Additional density estimates are needed for the Eastern Klamath of California. It is uncertain whether Sierra Pacific Industries' (2013) estimates are representative of densities across the region as a whole. Most NSO activity centers (ACs) included in Sierra Pacific's density estimates were located near the margins of the company's lands or on adjacent ownerships, rather than within the interiors of the company's holdings (see Maps 2-5 in Sierra Pacific Industries 2013, which are copyrighted and cannot be reproduced without permission). This pattern suggests that densities could be higher on neighboring lands such as the Shasta-Trinity National Forest.

Comment [LVD5]: This may also be a case where the adjacent FS lands provide the roosting and nesting habitat while SPI is producing the woodrats.

The California Forestry Association cited annual density estimates in timber company monitoring reports as evidence of stable or increasing NSO populations on private timberlands in the state (Calforests 2014). Reported crude densities on Humboldt Redwood Company, Mendocino Redwood Company, and The Conservation Fund lands in the Redwood Region were indeed relatively similar among years (Calforests 2014). However, it is unclear how changing survey methods and survey areas, as well as changing detectability of NSOs, influenced these companies' estimates over time (see Franklin et al. 1990). For example, recent adoption of survey protocols requiring more survey passes and use of electronic callers likely increased detection rates, and thus density estimates, on some of these lands. Estimates of crude densities of NSOs and numbers of ACs on Green Diamond Resource Company lands in the Redwood Region suggest that NSO densities have declined on that ownership (Figure 1.2; Table 1.1). The number of NSO ACs on Green Diamond lands briefly increased in 1998, apparently due to the company's acquisition of 70,000 acres of timberland that year (Green Diamond Resource Company 2014). Following a substantial decline during 2004-2008, the number of ACs began to gradually increase in 2009 (Figure 1.2). This increase appears to have been due to the company's adoption of a more rigorous survey protocol and implementation of a barred owl removal experiment during that same year (Green Diamond Resource Company 2014; see Chapter 4 of this volume). Sierra Pacific Industries' (2013) density trends for its ownerships in the Eastern Klamath are difficult to evaluate and are therefore not included here. Sierra Pacific Industries' (2013) estimates are empirical, potentially influenced by changing survey effort and areas, mostly descriptive of ACs at the margin of or outside the company's ownership, and were compared among blocks of years, rather than annually.

Comment [LVD6]: It would be useful to know which estimates are empirical counts with no statistical estimates of variance (i.e., no confidence intervals and therefore no way to assess the probability that the estimate represents the true value of the parameter).

Figure 1.2: Number of NSO activity centers ("sites") on Green Diamond Resource Co. lands during 1992-2013 (from Green Diamond Resource Co. 2014).



Rigorous ecological density estimates can be used to estimate population sizes for ecologically similar areas (Franklin et al. 1990). However, there are currently insufficient data for producing such an estimate for California or any of its regions. The California Forestry Association estimated that as many as 6,000 NSO territories currently exist in the state (Calforests 2014). This figure was based on an estimated statewide crude density of 0.28 territories per mile². This density estimate was, in turn, based on the cumulative number of known NSO ACs in California (see *Distribution*, above) and the proportion of "potential" habitat in the state that has been surveyed. The number of ACs known to have been recently occupied is substantially lower than the cumulative number that have been identified since the early 1970s (USFWS 2011a; see *Distribution*, above). This could be due to multiple factors, including declining occupancy rates (see *Occupancy*, below) and NSOs' use of different ACs over time. Thus, the timber industry's estimate provides little or no insight into the current number of NSOs or occupied ACs in the state. Furthermore, while reasonable projections of suitable habitat exist for NSOs in California (Davis and Dugger 2011, USFWS 2011a, Schumaker et al. 2014), the California Forestry Association did not cite these data and it is unclear how it estimated the total and surveyed areas of suitable habitat in the state.

Table 1.1: Density estimates for NSOs in California and Oregon.

Study	Region	Owner	Method	Crude Density (owls/ mi²)*	Crude Density (occupied territories/ mi²)*	Ecological Density (owls/ mi²)**†
Diller and Thome 1999	N Redwood	Green Diamond	Mark-Recapture	0.54 (0.24-0.91)		0.97-2.72
Green Diamond Resource Co. 2014	N Redwood	Green Diamond	Empirical (marked)	0.34 (0.12-0.53)		
Tanner and Gutierrez 1995 cited in Diller and Thome 1999	N Redwood	Redwood National Park	Empirical	0.57		
Humboldt Redwood Co. 2013	N Redwood	Humboldt Redwood	Empirical	0.53-1.01	0.36-0.50	
Mendocino Redwood Co. 2014	N Redwood	Mendocino Redwood	Empirical	0.47-.077		
The Conservation Fund unpubl. data in Calforests 2014	N Redwood	The Conservation Fund	Empirical		0.29-.036	
Chow 2001	S Redwood	Public (Various)	Empirical	0.97		2.09
Franklin et al. 1990	W Klamath	Six Rivers National Forest, Other	Mark-Recapture	0.61		1.41-1.71
Sierra Pacific Industries 2013	E Klamath	Sierra Pacific Industries	Empirical		0.17-0.18	
Woodbridge and Cheyne 1995	So Cascades	Klamath National Forest	Empirical		0.05-0.20	
Thrailkill et al. 1998	OR Coast Ranges	Bureau of Land Management	Empirical	0.07-0.25		0.57-0.90
Anthony et al. 2000 cited in Blakesley et al. 2004	OR Coast Ranges	Oregon Dept. of Forestry	Empirical	0.13-0.27		

*Ranges = low-high survey areas (Woodbridge and Cheyn 1995, Thraillkill et al. 1998, Diller and Thome 1999, Anthony et al. 2000, Green Diamond Resource Co. 2014), low-high survey years (Humboldt Redwood Co. 2013, Mendocino Redwood Co. 2014). **Habitat definitions used to calculate ecological densities: Franklin et al. 1990: all conifer cover classes weighted by NSO use (based on telemetry), but mostly ≥20.6 in DBH; Diller and Thome 1999: all forest classes weighted by NSO use (based on nest locations) but mostly >40 yrs; Chow 2001: all forested area; Thrailkill et al. 1998: old, mature, old over young, mature over young. †Ranges of ecological densities: Franklin et al. 1990: with two different habitat definitions; Diller and Thome 1999: low-high survey areas.

Occupancy

NSO population trends are most directly evaluated with demographic data (see *Demography*, below). However, occupancy data are often more logistically and economically feasible to collect than demographic data and, with proper accounting of detection probability, can provide a useful index of spotted owl population rates (MacKenzie 2005, Olson et al. 2005, Mackenzie et al. 2012, Tempel 2014). Occupancy data that inadequately incorporate detection probabilities for spotted owls must be interpreted carefully since they can be strongly influenced by survey effort, habitat attributes, social and reproductive status of NSOs, presence of barred owls, and other factors (Mackenzie 2005, Olson et al. 2005). Recent research in NSO demographic study

areas suggests that competition with barred owls is driving NSOs to move large distances (several miles) between different territories within the same season (Davis et al. 2013, Higley and Mendia 2013). Higley and Mendia (2013) warned that occupancy estimates for unmarked populations may therefore be inflated (i.e., the same individual could appear to occupy multiple territories within the same season) and suggested using the presence of pairs, rather than individuals, to determine occupancy.

There is limited information available for describing occupancy trends for NSOs in California. Much of the available information is from annual monitoring reports provided by industrial timber companies (Table 1.2). These data show trends in annual proportions of known, surveyed, or previous year's ACs found to be occupied (see Table 1.2 footnote). It is important to acknowledge that much of the data presented in Table 1.2 provide only crude indices of occupancy in California and that most of them cannot be compared among ownerships due to differences in monitoring and analytical methods. Future efforts to evaluate the status of NSOs in California would benefit from greater consistency in occupancy monitoring and from reporting of modeled occupancy rates, which account for detectability of NSOs and other factors that can obscure occupancy trends (e.g., Figure 1.5).

Recent occupancy estimates are unavailable for the Redwood National and State Parks in the northern portion of the Redwood Region. The National Park Service has discontinued surveying most historical territories in these parks due to apparent widespread displacement of NSOs by barred owls (Schmidt 2013; see Chapter 4 of this volume). In contrast with an apparently strong decline in occupancy in the Redwood National and State Parks, NSO occupancy rates on National Park Service lands in the southern portion of the Redwood Region have fluctuated annually but suggest a stable trend over time (Ellis et al. 2013; Table 1.2; Figure 1.3). Perhaps due to the area's geographic isolation, barred owls are still relatively uncommon the southern Redwood Region (Ellis et al. 2013; see Chapter 4 of this volume). Occupancy by NSOs appears to be gradually declining on industrial timberlands in the northern Redwood Region (Table 1.2; Figures 1.2, 1.4, 1.5, and 1.7). Given the substantial and increasing presence of barred owls in NSO territories on these lands (see Chapter 4 of this volume), it is surprising that more dramatic declines in NSO occupancy are not evident (e.g., see Table 1.2 for occupancy rates in Washington and Oregon). It is possible that NSOs respond differently to barred owls on these lands than elsewhere within their range. It is also possible that a more rapid decline is currently occurring than is indicated by the crude data presented in these companies' reports. Yet another possibility is that a more rapid decline will occur on these lands after a post-colonization lag period has elapsed or a critical threshold level of barred owl presence is reached (USFWS 2013).

NSO occupancy in the Northwestern California demographic study in the Western Klamath Region has declined dramatically in recent years (Franklin et al. 2013, 2014; Table 1.2). This decline has coincided with increasing barred owl presence in the study area, suggesting that NSOs are being displaced by barred owls (see Chapter 4 of this volume). The recently increased rate of declining occupancy by NSOs in this study area appears to support the hypothesis that barred owls can have lag or threshold effects on NSO populations. Recent declines in occupancy in the Northwestern California study area may also be related to effects of multiple consecutive years of poor weather conditions on demographic rates (see *Demography*, below). Recent annual reports from the Hoopa demographic study did not include analyses of occupancy data for

NSOs (Higley and Mendia 2012, 2013). However, unmodeled occupancy rates in 2012 and 2013 were low (0.40 and 0.35, respectively). Low occupancy rates on the Hoopa Reservation may be related to substantial declines in numbers of NSOs, likely due to decreasing demographic rates (see *Demography*, below) and increasing numbers of NSO territories with barred owl detections (see Chapter 4 of this volume). Greater declines in numbers of NSOs and increases NSO territories with barred owl detections beginning in 2005 provide additional support for the hypothesis that barred owls have lag or threshold effects on NSOs.

There is currently no clear pattern in occupancy data available for the Eastern Klamath and Southern Cascades of California. Timber companies in those regions have reported evidence of stable occupancy rates (Sierra Pacific Industries 2013, Michigan-California Timber Company 2014; Figure 1.6; note: Sierra Pacific's estimates are not provided in Table 1.2 for reasons discussed in *Density and Abundance*, above). However, more rigorous, published research conducted primarily on industrial timberlands in the Eastern Klamath and Southern Cascades found substantial declines in both simple (total) and pair occupancy (Farber and Kroll 2012; Figure 1.7). The barred owl invasion appears to still be in the early colonization phase in the Eastern Klamath, where this study was primarily conducted (Farber and Kroll 2012; see Chapter 4 of this volume). Thus, declining occupancy during the study was likely caused by some other factor, such as timber harvesting on the industrial timberlands that comprised much of the study area or wildfires on neighboring public lands (see Chapters 2 and 3 of this volume). Research in other areas of the NSO's range indicates that occupancy is negatively affected by habitat loss and fragmentation (Dugger et al. 2011, Sovern et al. 2014).

Table 1.2: Estimates and indices of occupancy by northern spotted owls in California, Oregon, and Washington.

Study	Region	Owner	Years	Number of Sites	Proportion Occupied (Total)	Proportion Occupied (Pairs)	Modeled Occupancy (Total)	Modeled Occupancy (Pairs)	Apparent Trend
Mendocino Redwood Company 2013	N Redwood	Private	2001-2013 (proportion); 2001-2008 (modeled)		0.75 - 0.69*		0.88 - 0.78*†		Declining (weak)
Humboldt Redwood Company 2014	N Redwood	Private	2003-2013		0.81 - 0.63*†				Declining (weak)
Green Diamond Resource Company 2014	N Redwood	Private	1999-2013 (no. sites); 2009-2013 (occupancy)	135 - 108*†	0.88 - 0.83*‡				Declining (weak)
Ellis et al. 2013	S Redwood	NPS	1999-2012		0.86 - 0.94*	0.72 - 0.87*			Stable
Franklin et al. 2002, 2003, 2010-2014	W Klamath	USFS, BLM, Private	2001-2013		0.67 - 0.37*	0.59 - 0.28*			Declining (strong)
Farber and Kroll 2012	E Klamath, S Cascades	Private, USFS	1995-2009				0.81 - 0.50*	0.75 - 0.46*	Declining (strong)
Michigan-California Timber Company 2014	E Klamath, S Cascades	Private	2000-2013		0.35 - 0.52 (2001-2013: 0.66 - 0.52)*†				Stable
Davis et al. 2013a	OR Klamath	BLM, State, Private	2001-2013		0.86 - 0.49*	0.62 - 0.30*			Declining (strong)
Dugger et al. 2011	OR S Cascades	USFS	1991/1992-2006				w/o barred owls: 0.86 - 0.71*†; w/ barred owls: 0.87 - 0.11*†		Declining (strong)
Kroll et al. 2010	WA E Cascades	NPS, USFS, Private	1990/1-2003				w/o barred owls: 0.83 - 0.64*†; w/ barred owls: 0.73 - 0.30*†	w/o barred owls: 074 - 0.36*†	Declining (strong)

*Start and end values. †Estimated from graph. ‡Occupancy of previous year's sites.

Figure 1.3: Occupancy status at monitored northern spotted activity centers on National Park Service lands in Marin County, California during 1999-2012 (from Ellis et al. 2013).

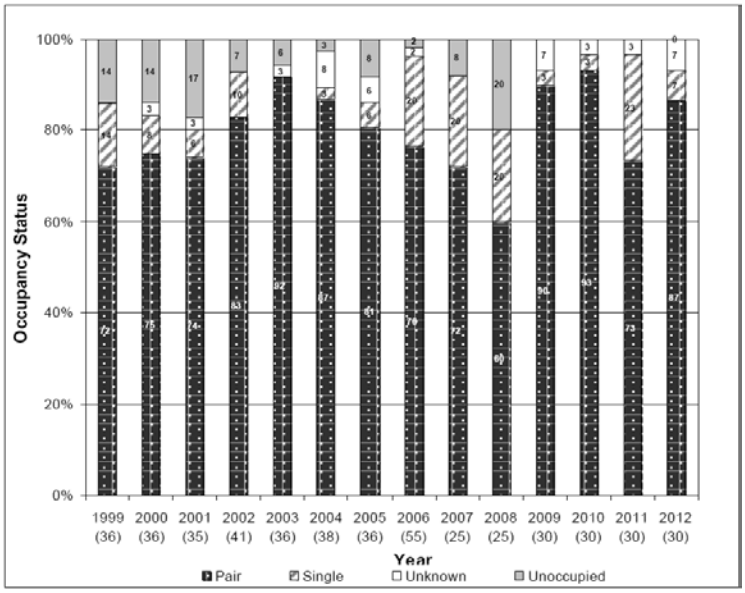


Figure 1.4: Annual numbers of known and occupied northern spotted owl activity sites (activity centers) on Humboldt Redwood Company lands during 2003-2013 (from Humboldt Redwood Company 2014).

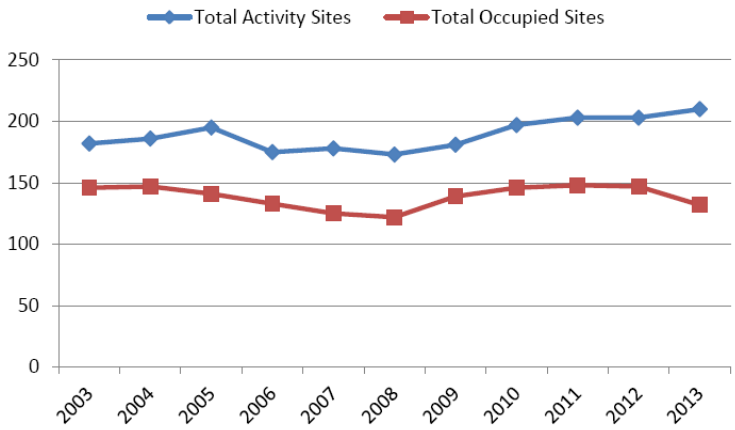


Figure 1.5: Annual proportion of northern spotted owl activity centers occupied (blue line) and modeled occupancy probability (red line) on Mendocino Redwood Company lands (from Mendocino Redwood Company 2013).

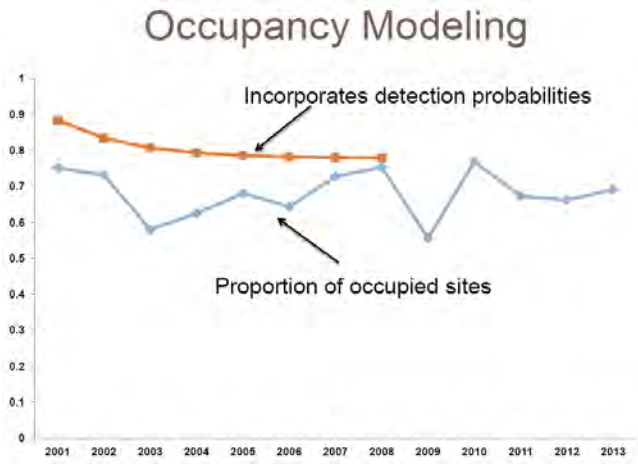


Figure 1.6: Percent of surveyed northern spotted owl sites occupied on Michigan-California Timber Company lands during 2000-2013 (from Michigan-California Timber Company 2014).

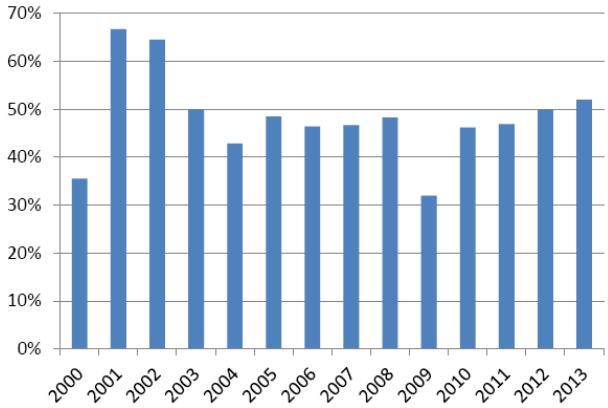
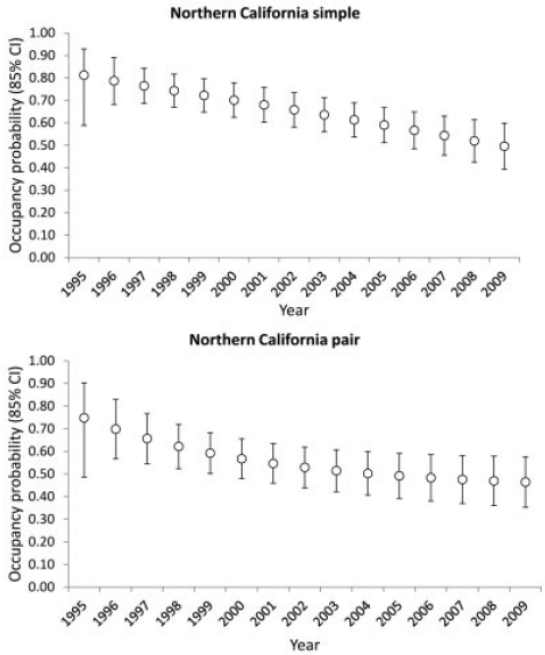


Figure 1.7: Estimated annual simple (total) and pair occupancy probabilities (with 85% confidence intervals) for northern spotted owls in the Eastern Klamath Region of California during 1995-2009 (from Farber and Kroll 2012).



Demography

Reproduction

Reproductive data are commonly collected as part of monitoring efforts for NSOs (e.g., Calforests 2014). They are easier and more cost-effective to obtain than those required for estimating survival or population trends. NSOs exhibit considerable annual fluctuations in reproduction (Forsman et al. 2011, Calforests 2014). Given often large annual fluctuations in reproduction, evaluation of trends in reproduction could require longer-term datasets than are available for many monitoring areas.

The 2011 demographic meta-analysis reported that fecundity of NSOs (number of female fledglings per female) significantly declined during 1985-2008 in four of 11 density study areas, may have declined in three other areas, and was stable in four areas (Forsman et al. 2011). Two of the four study areas with significant declines in fecundity were located in California (Northwestern California in the Western Klamath Region and Green Diamond in the Redwood Region). Two others were located in portions of southwestern Oregon (Klamath and South Cascades) that are nearby and ecologically similar to the Eastern Klamath and Southern Cascades of California (see USFWS 2011a and 2012a for regions). Also, the one area in

California with stable fecundity (Hoopa) had low fecundity estimates compared to other areas. Together these data, which represent the most reliable evidence currently available, indicate that NSO reproduction could be declining across much of California and southwestern Oregon.

Annual fluctuations in fecundity were evident in all three demographic studies in California and were remarkably synchronous (Forsman et al. 2011; Figure 1.8). Forsman et al. (2011) found that variation in fecundity was associated with a variety of variables, including the age of breeding females, whether the year was even or odd, weather or climate (e.g., early nesting season temperature or precipitation), percent cover of suitable habitat, and the presence of barred owls. Franklin et al. (2013) noted a pattern of "good" and "bad" reproductive years in the Northwestern California demographic study area, which is likely associated with annual variation in weather during the early nesting season (also see Franklin et al. 2000). Franklin et al. (2013) also observed that particularly poor reproductive years have occurred in their study area at four-year intervals, suggesting that "some other extrinsic factor may be operating, such as seed production governing small mammal populations." Forsman et al. (2011) reported that barred owl presence was in the top models explaining fecundity in the Green Diamond study area, suggesting that competition with barred owls contributed to declining reproduction on that ownership. Reports from the Klamath and South Cascades demographic studies in southern Oregon noted negative associations between reproduction and rainfall during the early nesting season (Davis et al. 2013b, Dugger et al. 2014). Declining reproduction in these study areas also appears to be related to increasing presence of barred owls.

Following publication of the 2011 meta-analysis, California's demographic studies reported three consecutive years (2011-2013) of very low reproduction (Franklin et al. 2013, Higley and Mendia 2013, Green Diamond Resource Company 2014). This dip in reproduction might have been partially driven by high rainfall during the early nesting season during 2010-2012 (see below). Those three consecutive years of low reproduction exacerbated the negative long-term trend that was already occurring on Green Diamond lands (Green Diamond Resource Company 2014; Figure 1.9). Negative trends in reproduction also occurred in the Klamath and South Cascades demographic study areas subsequent to the end of the 2011 meta-analysis study period (Davis et al. 2013b, Dugger et al. 2014; Figures 1.10 and 1.11). Davis et al. (2013b) concluded that particularly poor reproduction during recent years "…may indicate potentially serious problems with maintaining a stable population. This is even more alarming since these results are following a long term downward trend."

Figure 1.8: Mean annual fecundity in California's three northern spotted owl demographic studies during 1985-2008 (from Forsman et al. 2011).

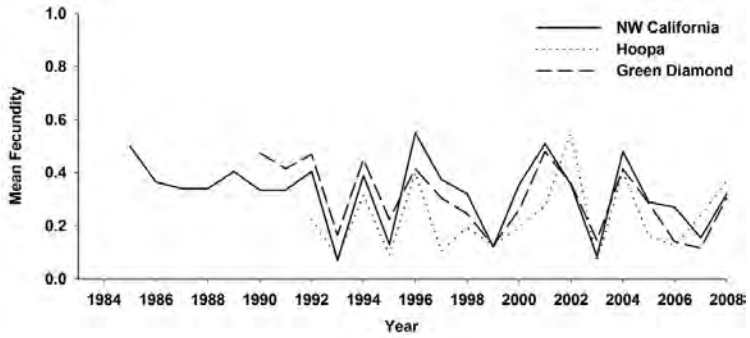


Figure 1.9: Number of fledglings produced per monitored pair of northern spotted owls on Green Diamond Resource Company lands during 1992-2013 (from Green Diamond Resource Company 2014).

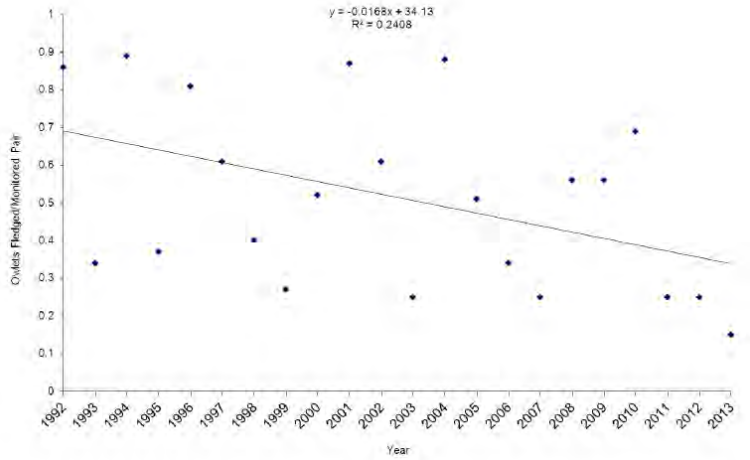


Figure 1.10: Fecundity of northern spotted owls ("STOC") in the Klamath demographic study area during 1990-2013 (from Davis et al. 2013a). Dashed line is a polynomial trend line (r^2 = 0.419). Vertical line represents the first year in which barred owls ("STVA") were detected in more than 10% of spotted owl territories.

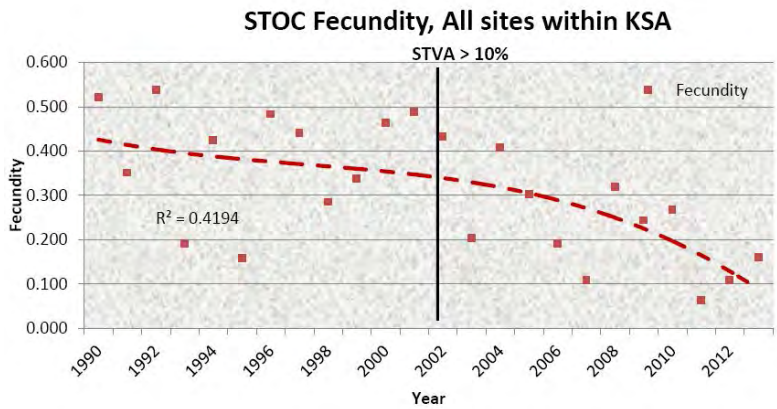
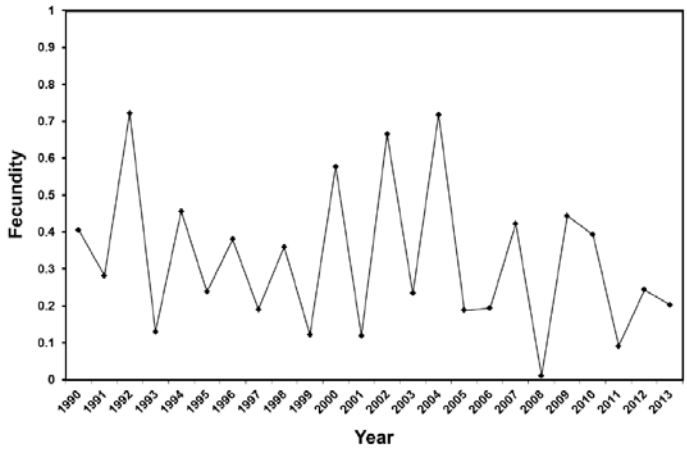


Figure 1.11: Fecundity of northern spotted owls in the South Cascades demographic study area during 1990-2013 (from Dugger et al. 2014).



Information is also available for describing recent trends in NSO reproduction in portions of California outside of demographic study areas. Ellis et al. (2013) found below average fecundity during 2007 and 2010-2012 on National Park Service lands in the southern Redwood Region (Figure 1.12). Humboldt Redwood Company (2013) and Mendocino Redwood Company (2014) likewise reported low reproduction during those years (Figures 1.13 and 1.14). These observations, along with those from demographic studies in California and southern Oregon described above, suggest that low reproduction during recent years was primarily driven by a

factor that acted at a very large spatial scale, rather than at the scale of individual ownerships or ecological regions. As noted above, high rainfall during the early nesting season was likely a primary cause of low reproduction during recent years. This apparent relationship is illustrated by the negative association between NSO reproductive success and early season rainfall observed on Mendocino Redwood Company lands (Figure 1.13).

Monitoring results suggest a stable long-term trend in reproduction on National Park Service lands in the southern Redwood Region (Figure 1.12). In the northern Redwood Region, Humboldt Redwood Company (2013) data likewise suggest little or no trend, although the period covered could be too short to capture long-term trends in reproduction (Figure 1.14). Mendocino Redwood Company (2014) provided a longer-term data set that suggests that a shallow decline in reproduction has occurred on their lands, primarily due to below average reproduction during seven of eight years during 2006-2013 (Figure 1.13). Data provided by the Fruit Grower's Supply Company (2014) suggests that a decline in reproduction occurred on their lands in the Eastern Klamath and Southern Cascades regions of California during 1990-2005 (Figure 1.15). It is important to note, however, that these are only descriptions of apparent trends based on patterns in relatively crude data. A more rigorous analysis of the data is needed to support strong conclusions about reproductive trends on industrial timberlands in the Redwood Region.

Figure 1.12: Fecundity of northern spotted owls on National Park Service lands in Marin County during 1999-2005 and 2007-2012 (from Ellis et al. 2013). The solid line indicates mean fecundity during these periods combined, the dashed lines are one standard deviation from the mean, error bars indicate ±1 standard error, and n is the total number of spotted owl territories.

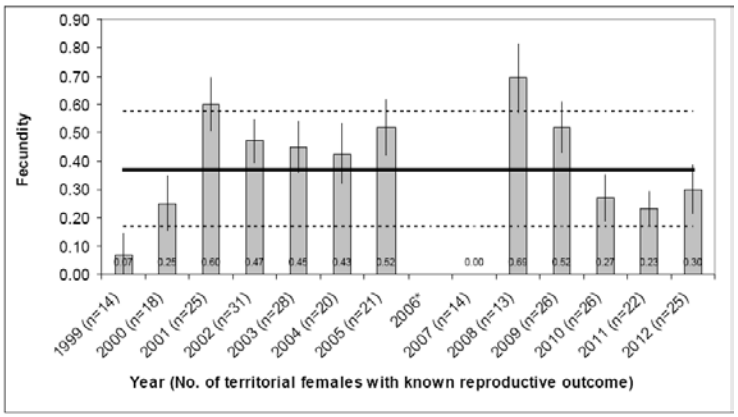


Figure 1.13: Reproductive success (average number of fledglings/pair) of northern spotted owls and amounts of rainfall during the early nesting season on Mendocino Redwood Company lands during 1989-2013 (from Mendocino Redwood Company 2014).

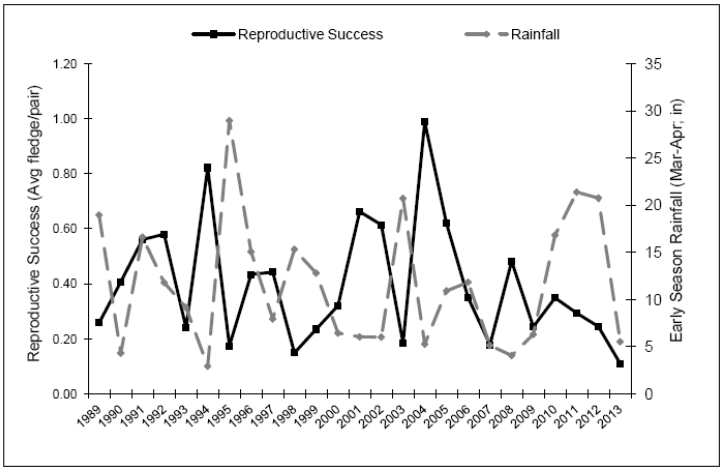


Figure 1.14: Reproductive rate and numbers of nesting pairs and juveniles on Humboldt Redwood Company lands during 2003-2012 (from Humboldt Redwood Company 2013).

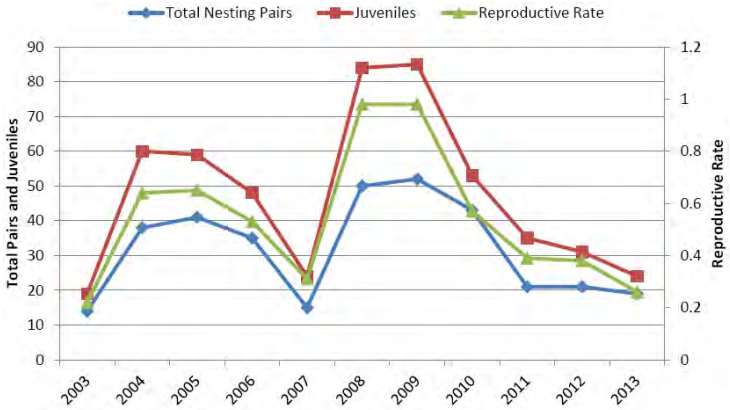
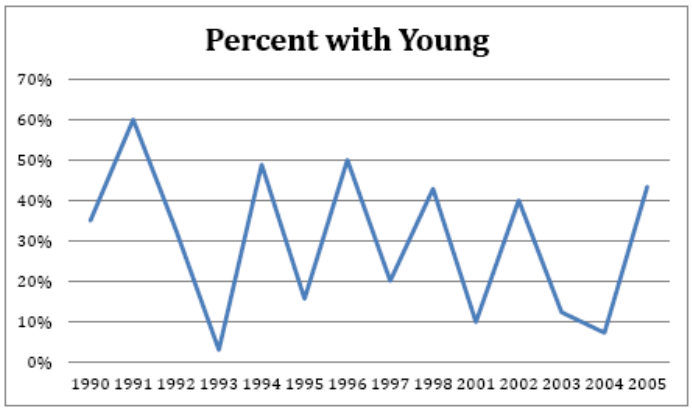


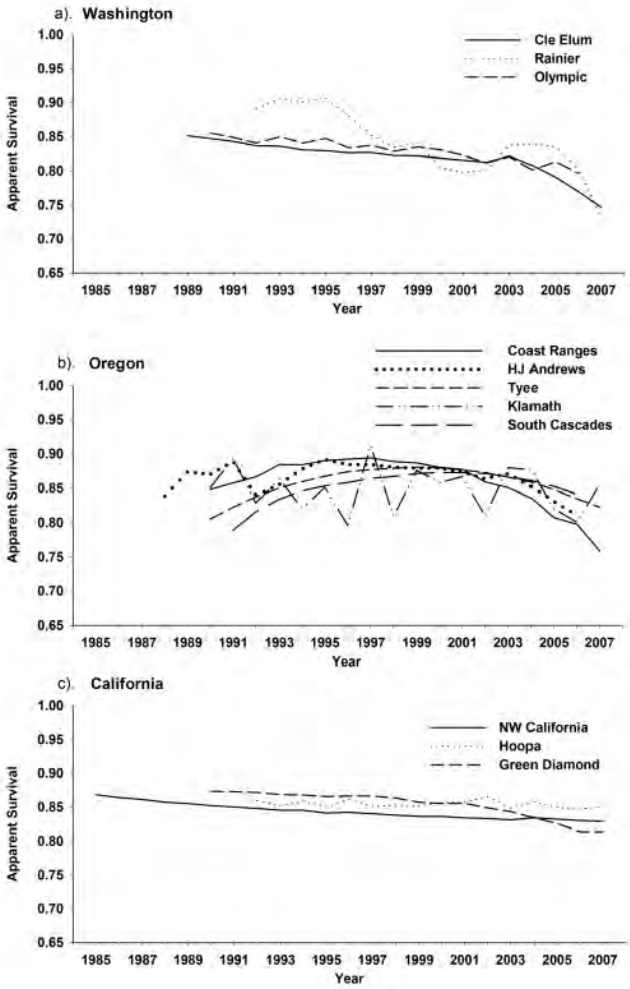
Figure 1.15: Fecundity of northern spotted owls on Fruit Growers Supply Company land during 1990-2005 (from Fruit Growers Supply Company 2014).



Survival

Available information concerning recent survival rates of NSOs is mostly limited to that provided in the 2011 demographic meta-analysis (Forsman et al. 2011). Survival data are not collected by timber companies other than Green Diamond Resource Company, which submits its data for analysis and reporting in the demographic meta-analyses. Forsman et al. (2011) reported statistically significant declines in apparent survival for 10 of 11 NSO demographic study areas, including all three study areas in California (Figure 1.16). Declines in many study areas were most precipitous during the last five years of the study period (i.e., 2003-2007 for survival; Figure 1.16). The Klamath in southern Oregon was the only study area that did not have a significantly declining survival rate through 2007. Forsman et al. (2011) stated that "collectively, the declines in apparent survival of Northern Spotted Owls across much of the subspecies' range are cause for concern because Spotted Owl populations are most sensitive to changes in adult survival rates (Noon and Biles 1990, Lande 1991)."

Figure 1.16: Model averaged estimates of apparent survival of adult female northern spotted owls three study areas in Washington (a), five study areas in Oregon (b), and three study areas in California (c) during 1985-2007 (from Forsman et al. 2011).



NSO demographic studies have largely deferred reporting of more recent survival data to the forthcoming meta-analysis, which is expected to be released in 2015. The limited information available prior to release of that meta-analysis suggests that survival has continued to decline since the 2011 meta-analysis study period. Davis et al. (2013b) reported that subsequent "…data regarding occupancy (in the Klamath study area) has shown a rapid decline, which suggests the stability of the survival rate may no longer be valid." Franklin et al. (2013) reported an alarming drop in apparent survival in 2011 on the Northwestern California demographic study area (Figure 1.17). Their subsequent annual report deferred reporting of 2012-2013 survival data to the forthcoming meta-analysis (Franklin et al. 2014). Higley and Mendia (2013) reported a statistically non-significant decline in survival of NSOs on the Hoopa Valley Indian Reservation in the Western Klamath (Figure 1.18). Their best model explaining survival of NSOs suggested that the decline was at least partially related to increasing numbers of barred owls in the study area.

Figure 1.17: Annual estimates (solid dots with 95% confidence intervals) of, and trend in (solid line), apparent survival for subadult and adult northern spotted owls in northwestern California during 1985-2012 (from Franklin et al. 2013).

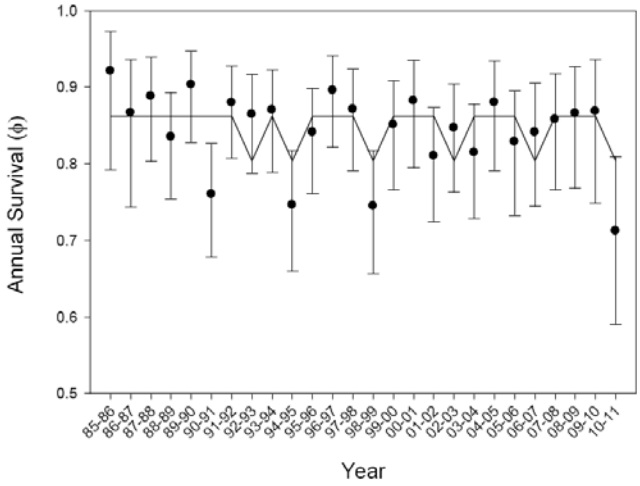
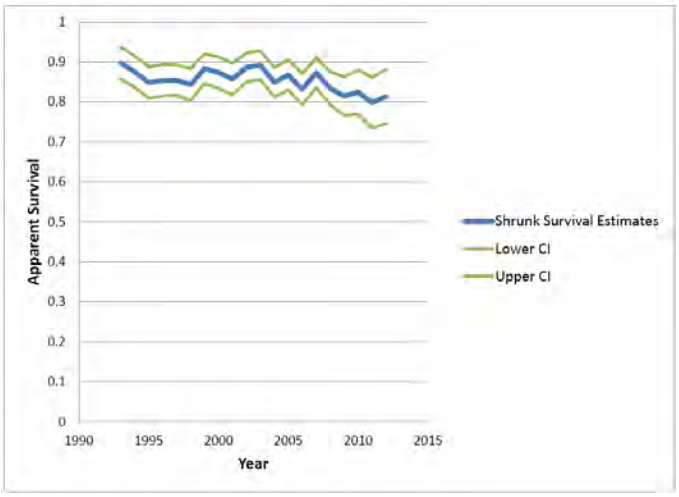


Figure 1.18: Estimates and 95% confidence intervals for NSO apparent survival on the Hoopa Valley Indian Reservation, Humboldt County, California during 1994-2012 (from Higley and Mendia 2013).



Forsman et al. (2011) reported that the presence of barred owls was included in the best model structures for several study areas, including the Green Diamond and Klamath, and was in a competitive model for Northwestern California (Forsman et al. 2011). Given evidence that barred owl presence continued to increase after the study period covered by Forsman et al. (2011) (see Chapter 4 of this volume), it is likely that the forthcoming meta-analysis will report continued declines in apparent survival for many, if not all, demographic study areas. Franklin et al. (2013) noted that apparent survival in the Northwestern California study area, like reproduction, is influenced by annual variation in weather during the early spring. Thus, recent consecutive years with poor weather during the early spring further suggest that survival has likely continued to decline since the period analyzed by Forsman et al. (2011).

Population Change

A new demographic meta-analysis is expected to be released in 2015. Until then, the 2011 meta-analysis (Forsman et al. 2011) provides the most current available estimates of population change for NSOs across their range. Except for the Green Diamond Resource Company, which submits its data for analysis and reporting in periodic meta-analyses, timber companies do not estimate population change for NSOs. Forsman et al. (2011) stated that their results likely "…reflected conditions on federal lands and areas of mixed federal and private lands within the range of the Northern Spotted Owl because the study areas were (1) large, covering ≈ 9% of the range of the subspecies, (2) distributed across a broad geographic region and within most of the geographic provinces occupied by the owl, and (3) the percent cover of owl habitat was similar between our study areas and the surrounding landscapes." Only one of the study areas included in the meta-analysis was entirely located on private lands (Green Diamond). Thus, it is unclear whether results from the 2011 meta-analysis reflect demographic trends on private lands across

the range of the NSO. Given weaker habitat conservation measures for NSOs on many private ownerships compared with federal lands, Forsman et al. (2011) stated that, "if anything, our results depict an optimistic view of the overall population status of the Northern Spotted Owl."

Forsman et al. (2011) reported estimates of the annual finite rate of population change (λ) for 11 study areas located across the NSO's range. Estimates of λ ranged from 0.929 to 0.996 (i.e., declines of 0.4 to 7.1% per year) for these study areas during the period of 1990-2006. There was strong evidence of population declines on seven of the study areas, including the Northwestern California (-1.7% per year) and Green Diamond (-2.8% per year) study areas in California. Negative population trends were also found on the Hoopa study area in California (-1.1% per year) and on the Klamath and South Cascades in southern Oregon (-1.0% and -1.8% per year, respectively) but they were not statistically significant. The weighted mean estimate of λ for all study areas combined was 0.971, indicating an average population decline of 2.9% per year during the study. Variables included in the best model in the meta-analysis of λ indicated effects of ecoregion (geographic location and major forest type) and the proportion of NSO territories with barred owl detections.

In addition to estimates of annual rate of population change, Forsman et al. (2011) provided estimates of realized population change, which describes population change over the study period (Figure 1.19). NSO populations in Washington and northern Oregon declined by approximately 40-60% during 1990-2006. Populations on the Northwestern California and Green Diamond study areas declined by 20-30% during the study period, although the 95% confidence intervals for these estimates slightly overlapped zero (Figure 1.19). Declines of 5-15% were evident on the Hoopa, Klamath and South Cascades study areas but these trends were not statistically significant (Figure 1.19).

Figure 1.19: Estimates of realized population change with 95% confidence intervals for northern spotted owls in California and southern Oregon (from Forsman et al. 2011).

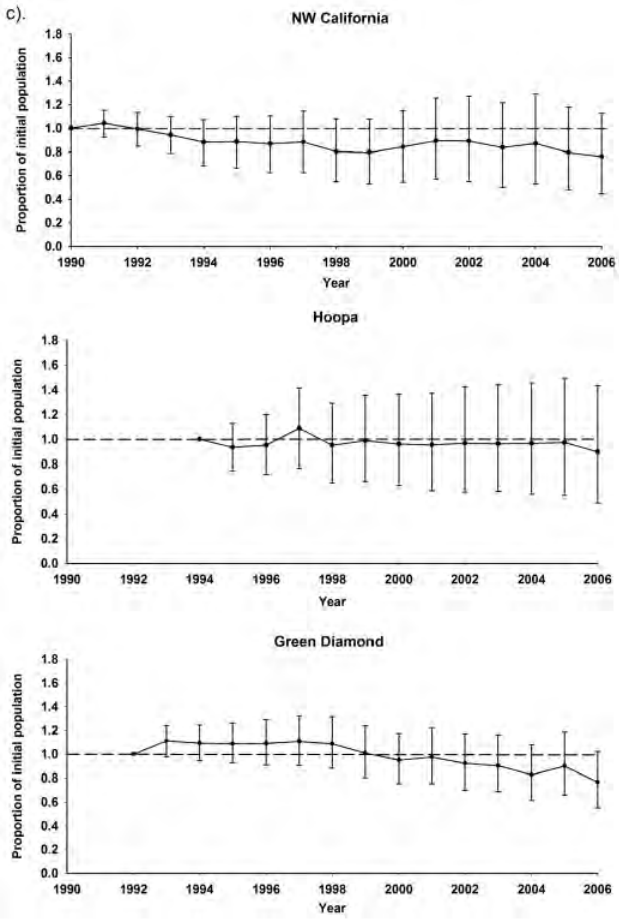
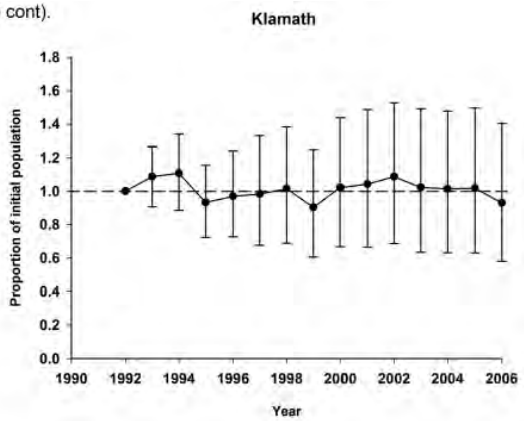
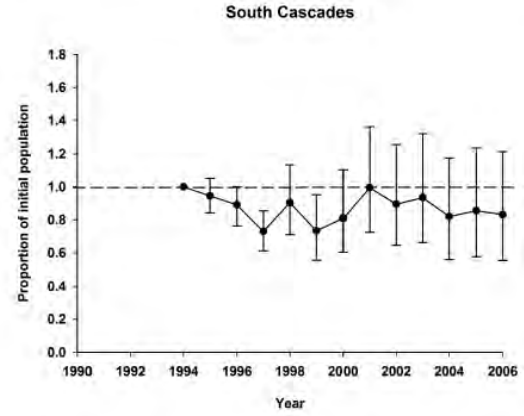


Figure 1.19 (cont.).

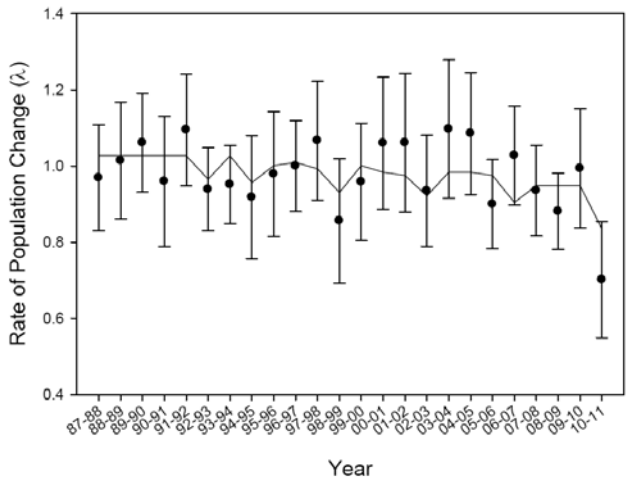
b cont).





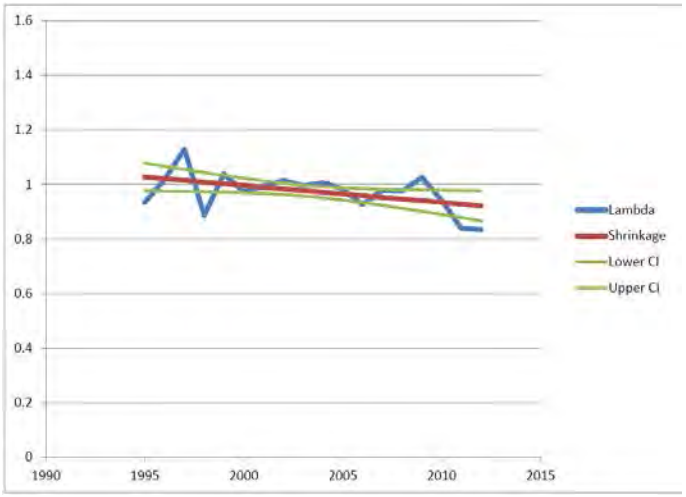
Following the 2011 meta-analysis study period (i.e., >2007), NSOs in the Northwestern California study area experienced a further decline in λ (mean = 0.978 or -2.2% per year) (Figure 1.20). The last year included in this analysis (2011) had the lowest annual estimate of λ found during the 24-year analysis period (Figure 1.20). The forthcoming meta-analysis should reveal whether the substantial drops in apparent survival and λ in the Northwestern California study area in 2011 were anomalous or indicative of an increased rate of population decline in the study area. Franklin et al. (2013) found that fecundity, apparent survival, and λ in the study area fluctuated during "good" and "bad" years, which was likely at least partially related to weather (see above). Annual rate of population change was also apparently negatively affected by increasing presence of barred owls. Given continued increases in barred owls (see Chapter 4 of this volume), poor weather during the early spring during 2010-2012, and poor reproduction by NSOs during 2011-2013 (see above), it is likely that λ continued to decline on this study area and probably others in California and southern Oregon.

Figure 1.20: Annual estimates of (dots with 95% confidence intervals) and trend in (solid line) rate of population change in the Northwestern California study area (from Franklin et al. 2013).



Higley and Mendia (2013) reported that the estimate of λRJS (Jolly-Seber Capture-Recapture model) for the Hoopa demographic study during 1995-2012 was 0.977, indicating a mean annual population decline of 2.3%. The decline was statistically significant in 2011 and 2012 (point estimates of λRJS not included in the 95% CI; Figure 1.20). Higley and Mendia (2013) noted that "the recent decline in survival, the point estimate of λRJS and the actual number of birds detected this past season all point to a population that is in fact, declining. This apparent decline in spotted owls corresponds with an increase in total annual barred owl detections and proportion of spotted owl territories with barred owl detections." They further noted that the forthcoming meta-analysis will show that it is "...very clear that northern spotted owls are in decline across all 11 study areas and that in many cases the decline is accelerating."

Figure 1.21: Trend in rate of population change on the Hoopa Valley Indian Reservation, Humboldt County, California during 1994-2012 (from Higley and Mendia 2013).



Source-Sink Dynamics

As described by Gutiérrez and Harrison (1996), source-sink dynamics exist for species "…that occupy both high-quality habitats (sources) where populations grow and produce emigrants, and low-quality habitats (sinks) where populations cannot sustain themselves in the absence of immigration." Population sinks potentially function as reservoirs for repopulation of sources that go extinct but may also reduce population growth rates (Pulliam 1988, Gutiérrez and Harrison 1996). Identifying source and sink areas is therefore, an important component of conservation research and planning.

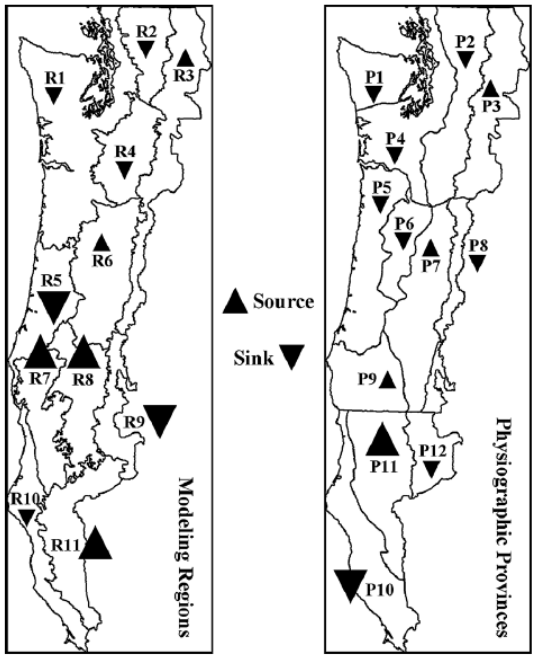
Schumaker et al. (2014) recently published a rangewide study of source-sink dynamics for NSOs. Their source-sink simulation modeling incorporated an array of regional data for NSO occupancy rates, movement distances and rates, life history attributes, habitat suitability and connectivity, encounter rates with barred owls, and environmental stochasticity. Source-sink dynamics in this study emerged from simulated interactions between individual NSOs and landscapes, rather than being predefined based on habitat suitability as was done in previous studies. The simulation models by Schumaker et al. (2014) ~~found~~ predicted that most ecological regions and physiographic provinces currently function as population sinks for NSOs (Figures 1.22 and 1.23). The study's results ~~indicated~~ projected that the Klamath Provinces of California and Oregon and the Interior Northern Coast Ranges of California are the subspecies' strongest population sources (Figure 1.23). The Klamath Provinces appear to be particularly important for maintaining NSO population stability due not only to being net population sources but to their high levels of population connectivity with multiple surrounding regions (Schumaker et al. 2014; Figure 1.23). The Redwood and Southern Cascades regions in California were both classified as moderate population sinks. Schumaker et al. (2014) identified the Klamath Provinces and

Comment [LVD7]: I think this section should also include a discussion of habitat fitness (Franklin et al. 2000), which is another method to quantify source versus sink habitats.

Comment [LVD8]: I personally don't think a simulation model should be used to draw management conclusions. In my opinion, they require so many assumptions to generate the necessary model parameters that they should only be used to develop testable hypotheses. The data we have accumulated over the years on population recruitment through immigration of subadults indicates that most of the birds are coming from within the redwood region. We don't have the data, but my guess is that the redwood region has contributed more recruits to the Klamath region than the reverse.

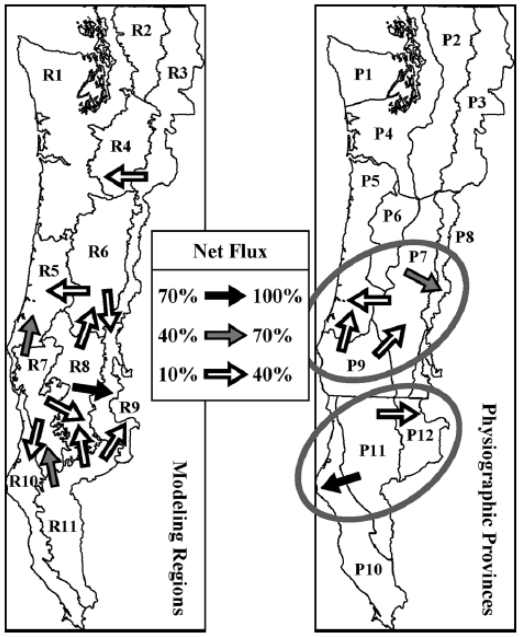
California Cascades as areas in which it could be particularly important to focus habitat protection and restoration efforts, respectively.

Figure 1.22: Relative source and sink values in northern spotted owl modeling regions and physiographic provinces (from Schumaker et al. 2014). The sizes of symbols denote major versus minor or moderate sources and sinks.



R7: Klamath West, R8: Klamath East, R9: Eastern Cascades South, R10: Redwood Coast, R11: Inner California Coast Ranges, P10: California Coast Range, P11: California Klamath, P12: California Cascades. See Schumaker et al. (2014) for other modeling regions and physiographic provinces.

Figure 1.23: Graphical representation of net movement ("Net Flux") of individual (simulated) northern spotted owls from one modeling region or physiographic province to another (from Schumaker et al. 2014). The largest Net Flux values are shown in black, intermediate values in gray, and smallest values in white. Gray ovals highlight two areas with strong patterns of Net Flux.

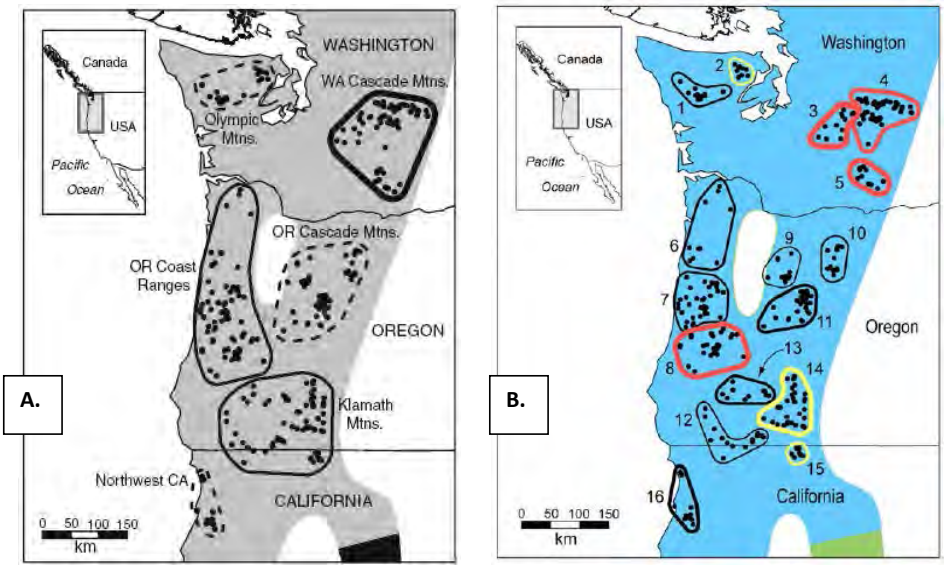


Genetics

Funk et al. (2010) found statistically significant evidence that NSOs have experienced genetic bottlenecks during recent decades in the Washington Cascades, Oregon Coast Ranges, and "Klamath Mountains" of Oregon and California (Figure 1.24). An earlier report on this study indicates that evidence of a bottleneck in the Klamath Mountains analysis area was primarily driven by data from the southern Cascades of Oregon and California, rather than from the Klamath Provinces (Funk et al. 2008; Figure 1.24). Evidence of recent genetic bottlenecks in the Washington Cascades, Oregon Coast Ranges, and southern Cascades are concordant with recent demographic declines in these regions (Forsman et al. 2011; see *Demography*, above). Surprisingly, Funk et al. (2010) did not find evidence of a genetic bottleneck in the Olympic Mountains of Washington, where NSOs have recently experienced dramatic population declines (Forsman et al. 2011). However, they noted that their small sample size for this region limited their power to detect a genetic bottleneck if one occurred. Funk et al. (2010) did not find statistically significant evidence of a recent genetic bottleneck in northwestern California (Western Klamath and Redwood regions). They suggested that this could likewise have been due to low statistical power or to the relatively gradual population declines reported for that area at the time (see *Demography*, above). The analyses of Funk et al. (2010) did not address whether

genetic bottlenecks were solely a result of population declines or were also contributing to them. Genetic declines can contribute to reduced demographic rates through effects of inbreeding depression and loss of adaptive genetic variation (reviewed in Funk et al. 2010).

Figure 1.24: Recent population bottlenecks in NSOs. Points represent 352 individual owls included in the analysis which are grouped into six (A) and 16 (B) regions. Statistically significant bottlenecks are represented by solid lines (A) or yellow ($p = 0.05$) and red ($p = 0.01$) lines (B). (A) represents significant bottlenecks under 5, 10, and 15% multi-step mutation models as solid bold lines and under 10 and 15% multi-step mutation models as finer solid lines (see Funk et al. 2010). (B) indicates greater magnitude bottlenecks with bolder lines. From Funk et al. 2010 (A) and 2008 (B).



Summary of Current Status and Trends

Rigorous long-term research has indicated that NSO populations are dramatically declining in Washington and northern Oregon and more gradually declining in southern Oregon and California (Forsman et al. 2011). Yet, while less precipitous than those in the northern portion of the NSO's range, the rapidity of population declines in southern Oregon and California are cause for grave concern regarding the subspecies' status and trends. A new demographic meta-analysis, which is due for public release during 2015, will replace the 2011 meta-analysis and provide the most reliable information for evaluating the NSO's current status and trends. Based on information available in annual research reports, it is clear that the forthcoming meta-analysis will show that populations in southern Oregon and California are declining more rapidly than was evident in the 2011 meta-analysis (Davis et al. 2013a, Franklin et al. 2013, Higley and Mendia 2013, Dugger et al. 2014, Green Diamond Resource Company 2014).

The NSO's status and trends are less clear in portions of California outside the state's three demographic study areas. Much of the information for these areas is provided by industrial timber companies, which have uniformly concluded that NSO populations on their lands are stable (Calforests 2014). However, the data provided by these companies are insufficient for drawing strong conclusions about the NSO's status and trends, and may in fact indicate gradual declines in occupancy and reproduction on some ownerships (see *Occupancy* and *Demography*, above). Forsman et al. (2011) suggested that, due to weaker habitat protection, NSO demographic trends could generally be worse on non-federal lands than on the federal and mixed federal/non-federal lands on which most demographic studies are conducted. This appears to be true in California, where NSOs experienced greater declines on Green Diamond Resource Company lands than on nearby tribal and Forest Service lands (Forsman et al. 2011; see *Occupancy* and *Demography*, above). However, the degree to which these differences were due to variation in land management, effects of competition with barred owls, or other factors is unclear.

It is likewise unclear if demographic trends in California's three demographic study areas accurately represent those in drier, less productive forests in the state's interior. An occupancy study in California's Eastern Klamath and Southern Cascades (Farber and Kroll 2012) and demographic studies in ecologically similar areas of southern Oregon (Davis et al. 2013a, Dugger et al. 2014) could provide the most reliable information currently available for evaluating NSO's status and trends in interior California (see *Occupancy* and *Demography*, above). These studies indicate that NSOs are currently declining in at least some portions of the Eastern Klamath and Southern Cascades regions (note: these regions cover portions of both California and southern Oregon as they are ecologically rather than politically defined; see USFWS 2011a Appendix C). Evidence of population declines in the Klamath regions (Forsman et al. 2011, Farber and Kroll 2012, Davis et al. 2013a, and Franklin et al. 2013) are particularly concerning in light of the critical contributions these areas likely provide to the NSO's long-term persistence (Schumaker et al. 2014; see *Source-Sink Dynamics*, above).

Although the Redwood Region ~~appears to currently~~is projected to function as a population sink, it still retains high densities and abundances of NSOs and is therefore important to the subspecies' conservation (Schumaker et al. 2014; see *Density and Abundance* and *Source-Sink Dynamics*, above). There is limited information available for evaluating the NSO's status and trends in portions of the Redwood Region outside of Green Diamond's lands. Monitoring on National Park Service lands and adjacent ownerships suggest that the population in Marin County is stable while NSOs in the Redwood National and State Parks have substantially declined. These differences appear to be largely due to negative effects of high barred owl densities in the Redwood National and State Parks and the relatively slow rate of the barred owl invasion in Marin County (see *Occupancy*, above). In contrast with the Green Diamond Resource Company, other timber companies in the northern portion of the Redwood Region have concluded that their NSO populations are stable (Calforests 2014). It is possible that NSOs have indeed fared better on these ownerships than on Green Diamond lands; for example, due to less intensive timber harvesting or more recent colonization by barred owls. However, the data provided by these companies are insufficient for drawing firm conclusions about the NSO's status and trends on these lands, and actually appear to indicate gradual declines in some areas. More consistent and rigorous monitoring (e.g., consistent survey areas and protocols; reporting

Comment [LVD9]: There are no data to support this assertion so I think simulation model projections need to be reported with the uncertainty they deserve. As the great statistician George E. P. Box stated: "All models are wrong, but some are useful."

of modeled occupancy rates) would assist future evaluations of the NSO's status and trends on industrial timberlands in California.

Acknowledgements

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Comment [JRD1]: They also may help identify areas where restoration of habitat, or management of barred owls, could be warranted.

Comment [JRD2]: I think Lowell's comment is reasonable here – maybe the point is that source-sink evaluations, and subsequently identifying sources and sinks is absolutely scale dependent. Thus, the sources and sinks we found in the Schumaker et al. paper, because the scale was so big (regions/provinces were millions of acres each) means that they were comprised of both sources and sinks. If we were to have looked at a smaller scale we would have more nuanced descriptions of sources/sinks – it is an important point.

Comment [JRD3]: I might add to that sentence "…for NSO at the (large) spatial scale of modeling regions and ecogeographic provinces.

Comment [JRD4]: We didn't really use "occupancy rates" but we did include habitat suitability.

Comment [JRD5]: See previous comment on spatial scale.

Comment [JRD6]: Lowell's comment on not drawing management conclusions from simulations/models. I see his point, but none of the demographic studies is near big enough to do what he's saying – we did a range-wide analyses (not a tiny portion of the range, like individual DSAs are…and even cumulatively they're <12% of the range). Thus, I kind of think that saying the source/sink stuff should "only be used to develop testable hypotheses" – really means that it will never happen at the range-wide scale. We either go with the higher uncertainty simulation data, or simply leave it a "blank spot on the map."

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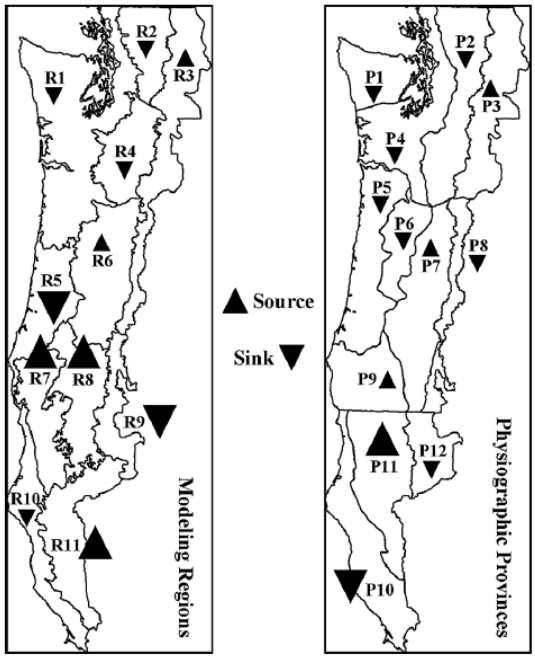

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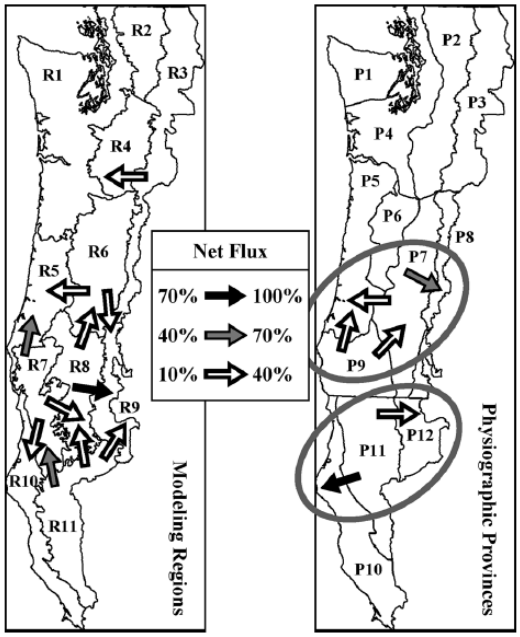


Timber Harvesting

Timber harvesting was a primary impetus for federal listing of the NSO and is still regarded as one of the major threats to the subspecies (Thomas et al. 1990, USFWS 1990, Courtney et al. 2004, USFWS 2011a). Timber harvesting can directly impact NSO populations by removing, degrading, or fragmenting habitat for them or their prey (reviewed below). Harvesting might also indirectly affect NSOs by increasing effects of other stressors, such as competitive pressure from barred owls (*Strix varia*) (Dugger et al. 2011, Forsman et al. 2011; see *Threats: Cumulative and Interactive Effects*). However, timber harvesting likely has complex effects on NSOs in the southern part of their range due to divergent effects of habitat conditions on survival versus reproduction (Franklin et al. 2000, Olson et al. 2004, Diller et al. 2010). The information reviewed herein suggests that some forms and amounts of harvesting are sustainable in northern California but that large-scale removal or fragmentation of habitat around activity centers can strongly impact NSOs (reviewed below and in USFWS 2009).

Comment [UFS1]: As in fire review… should define habitat in more specific terms when warranted.

The kinds of habitat concentrations associated with high survival and fitness of NSOs may be limited in some parts of the subspecies' range due to removal of the majority of old forest during the 19th and 20th centuries (USFWS 1990, Strittholt et al. 2006). Harvesting has been substantially reduced on federal lands since implementation of the Northwest Forest Plan (Healey et al. 2008, Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012). However, removal of suitable NSO habitat continues on federal lands and is occurring at higher rates on non-federal lands (Davis and Dugger 2011). Habitat loss to logging is only partially offset by recruitment of new habitat ~~on non-federal lands~~ (Davis and Dugger 2011, Moeur et al. 2011;

Comment [UFS2]: Not just federal

reviewed below). This is cause for concern since non-federal lands contain a considerable portion of remaining suitable breeding habitat for the subspecies and because recovery of the NSO partially depends on voluntary conservation efforts on these lands (USFWS 2011a; see *Legal and Regulatory Framework*). The timber industry has cited relatively strict harvest regulations in California as evidence that listing of the NSO under the California Endangered Species Act is unnecessary (California Forestry Association 2014). Yet, contemporary harvesting has still resulted in a net loss of suitable breeding habitat for NSOs on non-federal lands in California (Davis and Dugger 2011, Moeur et al. 2011; reviewed below). Furthermore, the U.S. Fish and Wildlife Service (2009) recently concluded that California's regulations for avoiding take inadequately protect the subspecies and do not reflect the best available science (reviewed below; also see *Threats: Inadequacy of Regulatory Mechanisms*).

Comment [UFS3]: How much is "considerable"? Perhaps state as a percentage of total.

Responses of NSOs to Timber Harvesting

Interior Northern California

NSOs in interior northern California show a strong general preference for relatively old, structurally complex forest. This is illustrated by studies describing both plots around NSO locations (Solis and Gutierrez 1990, Rissler 1995, White 1996, Hershey et al. 1998; but see Irwin et al. 2013) and landscape-scale analysis areas around activity centers (Chávez-León 1989, Solis and Gutierrez 1990, Hunter et al. 1995, Gutierrez et al. 1998). This body of research can be used to inform conservation measures for NSOs in interior northern California (e.g., for evaluating appropriate habitat definitions in take-avoidance guidelines: USFWS 2009). However, the

following review is focused on studies of associations between landscape-scale habitat attributes and NSO demography in interior forests (Franklin et al. 2000, Olson et al. 2004, Dugger et al. 2005, Schilling et al. 2013). These studies are based on rigorous demographic data and provide the best available insight into potential effects of timber harvesting on NSO populations (USFWS 2009). We supplement this review with information from studies of associations between landscape-level habitat characteristics in southern interior forests and the NSO's home range sizes (Carey et al. 1992, Carey and Peeler 1995, Schilling et al. 2013) and probability of occurrence (Zabel et al. 2003, Carroll and Johnson 2008).

Comment [UFS4]: Tempel et al in press might also be good here.

In the California Klamath, Franklin et al. (2000) found that NSO survival was highest when estimated breeding core areas (390 ac) contained large amounts of both "core" (>326 ft from edge) older forest (conifer or mixed forest with conifer QMD ≥21 in and canopy cover ≥70%) and edge with other vegetation classes. In contrast, reproduction was typically highest with lower amounts of core older forest and greater amounts of edge. Estimated core areas supporting high fitness for NSOs (a function of both survival and reproduction) contained both a large concentration of core older forest and considerable habitat edge provided by a mosaic of other vegetation patches with convoluted shapes. Franklin et al. (2000) emphasized the difference between total area of older forest versus area of core older forest. For example, they noted that large amounts of older forest edge cannot occur with low total amounts of older forest. This study did not directly address effects of timber harvesting on NSOs. Vegetation other than older forest was combined into a single class and edges occurred wherever that class and older forest met. Franklin et al. (2000) noted, however, that the dominant silvicultural system in their study

area at that time was large-scale clearcutting, which they concluded was unlikely to contribute to the kinds of habitat mosaics found in territories supporting high fitness.

In the interior of the Oregon Coast Range, Olson et al. (2004) found that NSO survival was positively associated with greater amounts of both "mid-seral" (9.5-31.5 in DBH) and "late-seral" (>31.5 in DBH) forest in landscapes around activity centers (≤4,921 ft) and lower amounts of early-seral forest and non-forest (<9.5 in DBH). Reproduction, in contrast, was negatively associated with area of mid- and late-seral forest and positively associated with edge between early-seral and non-forest and other vegetation classes. Olson et al. (2004) encountered technical difficulties with the habitat fitness potential portion of their modeling but noted that territories supporting high fitness must contain attributes associated with both high survival and high reproduction. This was supported by diagrams made from aerial photos of landscapes around activity centers, which showed remarkably similar habitat mosaics to those presented by Franklin et al. (2000).

Dugger et al. (2005) found a positive association between NSO survival in the Oregon Klamath Province and greater amounts of mature and old forest (≥100 yrs) within estimated core areas (413 ac) and a moderate amount of non-habitat (non-forest, early-seral vegetation, and older forest with harvest entries ≥40% basal area) in the landscape beyond the core area (3,430-ac ring). The specific contribution of timber harvesting (and of different harvest types and intensities) to the non-habitat class and thus, its effects on NSO fitness, were not reported. This study's findings differed from others in that reproduction was positively, rather than negatively, associated with greater amounts of older forest within estimated core areas. These findings

Comment [UFS5]: Differentiate between "core area" and "core habitat".

suggest that widespread harvesting of older forest within NSO core areas would negatively effect both survival and reproduction in this area but that some level of harvesting might be sustainable in the broader landscape (to the degree that it contributes to "optimal" amounts of non-habitat in the 3,430-ac ring surrounding estimated core areas).

Schilling et al. (2013) found additional evidence of a positive influence of both older forest and habitat heterogeneity on NSO survival in the Oregon Klamath. Their best performing model indicated that monthly survival probabilities for NSOs were highest when home ranges (based on radio-telemetry) contained more patches of mature and old forest (≥20 in DBH and ≥40% canopy cover). The second best performing model indicated a positive association between survival and clustering of (i.e., close distances between) older forest patches. Unlike other studies, they did not find an association between survival and total amount of older forest. They noted that this could have occurred due to their small sample size or because most NSO home ranges in their study had amounts of mature and old forest (mean = 72%) that likely exceeded threshold amounts required for survival. A third competitive model suggested that survival was also positively associated with a moderate amount of edge between forest (mean DBH ≥5 in) and other cover classes; thus providing additional support for the value of habitat heterogeneity for NSOs in southern interior forests.

Zabel et al. (2003) modeled probability of NSO occurrence (i.e., habitat suitability) across interior northern California based on habitat conditions at an estimated core area scale (500 ac). The best performing model in their study indicated that the probability of NSOs occurring in a given location was highest with large amounts of suitable nesting-roosting habitat (generally ≥17

in DBH and canopy cover ≥60%) and intermediate amounts of foraging habitat (≥10 in DBH and canopy cover ≥40%) at the core area scale. The second and third best performing models at the core area scale included habitat edge. The results of this modeling study provide further support for conclusions that a combination of both a large concentration of suitable habitat and some form of habitat heterogeneity is important to NSOs in interior northern California.

Carroll and Johnson (2008) also modeled probability of NSO occurrence in interior northern California. Based on their best model, predicted abundance of NSOs in the area was highest when most of the landscape (5,930-ac areas) consisted of mature and old forest (>50 yrs). However, predicted abundance slightly declined when area of mature and old forest increased beyond about 80% of the landscape. This study therefore, provides evidence of at least a slight positive effect of other vegetation classes on probability of NSOs occurring in a given area. These results contrasted with the study's findings for more northern parts of the NSO's range, where the probability of occurrence continued to increase (albeit diminishingly) with greater amounts of older forest.

Studies of home range sizes provide another line of evidence concerning habitat and harvesting influences on NSOs in interior southern forests. Home range studies in the Oregon Klamath found that home range size increased with habitat fragmentation (Carey et al. 1992, Schilling et al. 2013). NSOs in the area are known to use regenerating harvest units for foraging, particularly when closer to the activity center or outside the breeding season (Carey and Peeler 1995, Irwin et al. 2013). However, Carey and Peeler (1995) concluded that the energetic benefit of increased

access to dusky-footed woodrats (*Neotoma fuscipes*) in heavily fragmented forest is often outweighed by the energetic cost of increased travel.

In summary, studies in interior northern California have found that NSOs in the region benefit from both large amounts of older forest concentrated around activity centers and some form of habitat heterogeneity (Franklin et al. 2000, Zabel et al. 2003, Carroll and Johnson 2008). Similar results have been found in the Klamath (Dugger et al. 2005, Schilling et al. 2013) and interior Coast Range of Oregon (Olson et al. 2004). These findings suggest that timber harvesting is sustainable in southern interior forests, provided that suitable breeding habitat is retained in sufficiently large concentrations around NSO activity centers (USFWS 2009). However, whether and how timber harvesting contributes to beneficial habitat heterogeneity in interior southern forests is unclear. Available studies differed in their findings of types, amounts, and locations of beneficial heterogeneity and did not directly evaluate whether timber harvesting contributed to it. In contrast, it is clear from research of associations between landscape-level habitat attributes and NSO demography, presence, and home range size that harvesting within core concentrations of suitable habitat has the potential to strongly impact populations in southern interior forests (USFWS 2009). Despite the volume, rigor, and applicability of research showing associations between landscape-level habitat attributes and NSO fitness, California has yet to integrate it into take-avoidance regulations for interior timberlands (see *Threats: Inadequacy of Regulatory Mechanisms*).

Comment [UFS6]: Is this the proper word? It implies even flow through time. I think you mean compatible or something along those lines.

Comment [UFS7]: Positively or negatively? Be clear

Redwood Province

Most of what is known about NSOs in the Redwood Province is based on research on intensively harvested lands owned by the Green Diamond Resource Co. Studies on these lands found a preference among NSOs for landscapes with greater amounts of intermediate-age or older forest than expected based on general availability of those forest classes (Thome et al. 1999, Folliard et al. 2000, Keithley and Motroni 2000, Gonzales 2005, Diller et al. 2010). However, site fidelity and reproduction on these lands were positively associated with presence of younger forest classes and measures of habitat heterogeneity (e.g., edge) (Thome et al. 1999, 2000, Diller et al. 2010). Studies of the habitat associations of dusky-footed woodrats on these lands appear to provide additional support for the value of younger forest and habitat heterogeneity to NSOs in the area (Hamm et al. 2007, Hamm and Diller 2009). Unpublished but relatively rigorous modeling of associations between landscape-level habitat attributes and NSO fitness and population growth rate has confirmed that NSOs on Green Diamond lands have complex relationships with timber harvesting (Diller et al. 2010). NSOs in this area indeed appear to benefit from some level of habitat heterogeneity, which is currently maintained on Green Diamond lands through small-patch clearcutting (Diller et al. 2010). Yet, habitat quality on these lands (measured as habitat fitness potential *sensu* Franklin et al. 2000) is positively associated with protection of suitable breeding habitat and both habitat quality and population growth rate are negatively associated with harvesting of suitable habitat (i.e., take) (Diller et al. 2010). Thus, appropriate management of NSOs on Green Diamond lands appears to include avoiding take, setting aside suitable habitat from harvesting, and focusing economically-driven harvest requirements in relatively homogeneous blocks of unsuitable forest. Diller et al. (2010)

did not describe habitat conditions associated with habitat fitness potential ≥1 (i.e., conditions associated with NSOs replacing themselves or contributing to a population surplus). Peer reviewed reporting of these conditions is needed in order to identify appropriate take-avoidance guidelines for the northern part of the Redwood Province (see *Threats: Inadequacy of Regulatory Mechanisms*).

Less is known about the ecology and appropriate management of NSOs on other ownerships within the Redwood Province. Habitat selection by NSOs appears to vary among ownerships in the region (Keithley and Motroni 2000, see Appendix 1). The USFWS (2011a, 2012a) recently conducted habitat suitability modeling based on attributes of landscapes (494 ac) surrounding 392 activity centers distributed across much of the province. The model selected for the region included a suite of habitat variables and performed well in terms of its ability to discriminate between areas around NSO activity centers and random sites. The resulting map of relative habitat suitability was incorporated into the USFWS (2012a) process for designating critical habitat for NSOs but has limited utility for characterizing habitat selection by the subspecies. However, "deconstruction" of the habitat suitability modeling outputs (cf. Dunk and Hawley 2009, Woodbridge et al. 2012, Zielinski et al. 2012) allows evaluation of associations between habitat suitability and the full range of candidate variables, including ones not included in the best performing model. Deconstruction of the habitat modeling output for the Redwood Province shows that the probability of NSOs occurring in a given area in the region increases with larger amounts of forest with relatively dense canopy cover and large diameter trees (Appendix 2). Compared with those in the lowest suitability class, landscapes in the highest suitability class contained an average of 1.8 times more nesting-roosting habitat; 2.4 times higher

basal area of conifers ≥20 inches DBH; 2.3 times higher basal area of live trees ≥30 inches DBH; and 2.0, 1.8 and 1.9 times higher densities of conifers ≥20, 30, and 39 inches DBH, respectively. There was a high degree of variability (standard deviation) in terms of structural attributes within habitat suitability classes, particularly for rare habitat elements such as very large diameter trees. This variability likely reflects the high diversity of forest types, management histories, and natural disturbance regimes in the region (see Sawyer 2006, 2007, Stuart and Stephens 2006). Nonetheless, consistent patterns of association between habitat suitability and mean amounts of these variables are evident. In addition, variability in amounts of many of these habitat attributes (coefficient of variation) declined with increasing habitat suitability, further indicating that they are often important to NSOs in the province. These results suggest that timber harvesting that reduces availability of these structural attributes would generally reduce the probability of NSOs occurring in a given area within the Redwood Province. Changes in availability of these structural attributes can occur with a variety of silvicultural approaches and are not solely caused by even-age harvesting.

Effects of Uneven-Age Harvesting and Thinning

Comment [UFS8]: There is a recent pub (GTR) that has a section on this. See Wilson and Forsman. 2013. Thinning Effects on Spotted Owl Prey and Other Forest-Dwelling Small Mammals, in:

Anderson, Paul D.; Ronnenberg, Kathryn L., eds. 2013. Density Management in the 21st Century: West Side Story. Gen. Tech. Rep. PNW-GTR-880. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 249 p.

Some private timberlands in northern California currently emphasize uneven-age regeneration, which typically causes less visually dramatic changes to forests than does even-age harvesting. This form of harvesting nonetheless has the potential to cause substantial changes to forest structure or composition. For example, intensive selective logging of Douglas-fir (*Pseudotsuga mensiesii*) has resulted in extensive conversion of mixed-evergreen forest to hardwood-dominated forest in parts of the Redwood Province (Sawyer 2006). Relatively little harvesting

has occurred on federal lands within the NSO's range since adoption of the Northwest Forest Plan (Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012; see below). However, federal agencies have recently expressed support for widespread thinning to reduce wildfire risk in interior forests in the Plan area (USFWS 2011a, 2012a).

Effects of contemporary uneven-age harvesting and thinning on NSOs are difficult to evaluate due to the paucity of rigorous research on the topic. Most of the available information about NSO responses to these silvicultural systems is based on the behavior of very small numbers of telemetered owls and was gathered in an opportunistic fashion during studies of other topics (reviewed in Hansen and Mazurek 2010, USFWS 2011a; see below). Evaluation of this topic is further complicated by poor descriptions of harvest methods, locations and intensities and, perhaps more importantly, post-harvest habitat conditions. The terms uneven-age harvesting and thinning encompass a tremendous variety of harvest types, objectives, and effects (Smith 1986). Harvesting described in relation to NSO telemetry consisted of a variety of commercial thinning or partial harvesting (leaving residual trees) prescriptions, including understory thinning of various intensities, removal of most trees up to a relatively large diameter class, and shelterwood harvests prior to or without removal of residual trees (see Hansen and Mazurek 2010).

In a synthesis prepared for the 2011 revised recovery plan for the NSO (USFWS 2011a), Hansen and Mazurek (2010) provided detailed summaries of data concerning responses of both NSOs and California spotted owls (CSOs) to uneven-age harvesting, partial harvesting, and thinning. This information was gleaned from both peer-reviewed and gray literature and was based on small sample sizes. The authors therefore, opted to review each data source as a "case study" so

that relatively detailed descriptions of harvesting and post-harvest conditions could be provided and so that the methodological strengths and weaknesses of studies could be evaluated. We summarize their review below, with the addition of one subsequent citation (Gallagher 2010).

Each of the 12 studies that we reviewed documented at least some use by NSOs or CSOs of areas harvested with uneven-age harvesting, partial harvesting, or thinning. At least four of the studies found owls nesting in harvest areas (Forsman et al. 1984, Zabel et al. 1992, King 1993, and Buchanan et al. 1995) and at least five recorded roosting in them (Solis 1983, Sisco 1990, King 1993, Hicks et al. 1999, and Meiman et al. 2003). It is important to note, however, that older forest structural attributes had been retained or regenerated in most of the harvest areas used for nesting or roosting. Three of the four studies that documented nesting in harvest areas described the nest stands as mature or old forest or an equivalent classification (USFS Region 5 "suitable habitat"; "understory reinitiation phase…of stand development"). The other study did not describe the harvest area used for nesting (King 1993). Harvest areas used for roosting in three studies likewise were either classified as mature or old forest (Solis 1983) or contained some older-forest structural characteristics, such as relatively high basal area or dense canopy cover (King 1993, Meiman et al. 2003). Two studies observed roosting in harvested stands that appeared to differ from this pattern; but one of the authors thought that the deaths of three birds that roosted in them were due to higher predation risk in the more open stands (Sisco 1990, Hicks et al. 1999).

Most of the reviewed studies found that spotted owls foraged to some degree in uneven-age harvested, partially harvested, or thinned areas. Irwin et al. (2005, 2008) stated that some NSOs

in their study areas selectively used certain harvest units but not others. However, they did not provide quantitative comparisons of prescriptions, post-harvest conditions, or proximities of harvest units to activity centers. Two other studies found that spotted owls generally avoided foraging in areas that recently experienced moderate to intensive partial harvesting or thinning, whereas use of lightly harvested areas varied among individuals (Anthony and Wagner 1999, Gallagher 2010). Anthony and Wagner (1999) found that NSOs (n = 15) in southern Oregon foraged in heavy and moderate partial-cuts less than expected (old stands with ≥30-40% of the original basal area removed and ≥"moderate" canopy cover reduction). Light partial-cuts (old forest with ≤20% of the original basal area removed and "small" reductions of "crown cover" [not described]) were used more than expected by two owls, as often as expected by five, and less than expected by eight. In the northern Sierra Nevada, Gallagher (2010) found that CSOs (n = 9) used heavily thinned "defensible fuel profile zones" (canopy cover reduced to 40%, removal of trees <30 in DBH, reduction of tree density and ladder and surface fuels) less than expected based on availability. She also reported a near-significant tendency ($p = 0.08$, n = 5) for avoidance of areas recently treated with understory thinning. Use and availability of harvest areas varied among individuals. Most individuals exhibited avoidance of defensible fuel profile zones and understory thins but one male showed strong selection for thinned areas (primarily understory thins). It is possible that thinning improved prey availability or otherwise benefited this male. However, Gallagher (2010) noted that thinning treatments were located unusually close to this male's activity center, which potentially increased his likelihood of using them due to central place foraging. She also noted that an unusually large proportion of understory thin units in the male's home range were also treated with prescribed fire, which could have temporarily increased abundances of deer mice (*Peromyscus* spp.) or other prey.

The limited available information suggests that thinning and uneven-age harvesting causes some spotted owls to increase their home range sizes, which could impose energetic costs on individuals (Meiman et al. 2003, Gallagher 2010). Meiman et al. (2003) reported that a male NSO's breeding season home range in the Oregon Coast Range was slightly larger before commercial thinning than afterward but that its nonbreeding season home range was 2.3 times larger. The individual appeared to shift its breeding season core area to include less of the thinned area and its nonbreeding season core area was more than twice as large following thinning as it was prior to thinning. In the northern Sierra Nevada, Gallagher (2010) found that the home range sizes of CSOs (n = 9) significantly increased with greater total area of fuels treatments (defensible fuel profile zones and understory thinning). She also reported near-significant trends of increasing home range size with greater area of defensible fuel profile zone ($p = 0.08$) and group selection harvesting ($p = 0.06$).

Four studies reported that thinning or partial harvesting near nests or roosts displaced spotted owls from those areas (Forsman et al. 1984, King 1993, Hicks et al. 1999, Meiman et al. 2003; also J. Reid, pers. comm.). The only study to describe this effect for more than two NSOs suggested that pairs' responses to harvesting near their nests depended on the intensity of the harvest, whether or not habitat in the nest area was excluded from harvesting, and whether or not suitable alternative habitat was available within the home range (Forsman et al. 1984).

The US Fish and Wildlife Service has expressed support for widespread thinning to reduce the risk of severe wildfire in dry forests within the NSO's range (USFWS 2011a, 2012a). Our

review suggests that spotted owls are often resilient to wildfire (and may benefit from low severity or patchy fire in southern forests) but that extensive severe fire can negatively affect the species by reducing amounts and contiguity of nesting and roosting habitat (see *Threats: Wildfire*). This conclusion might appear to support widespread thinning to reduce the risk of large severe fires. However, preliminary findings of negative effects of thinning on the species and the overall lack of reliable information on the topic suggest that more research is needed before thinning is employed at broad scales within the NSO's range. If widespread thinning is applied prior to rigorous study of its effects, our review suggests that it should be conducted well away from NSO activity centers and focused in young, homogeneous stands that are less suitable for NSOs and where thinning might increase habitat heterogeneity or accelerate development of complex, older-forest structure for prey (Carey 2006; but see below regarding effects of thinning on primary prey species). Planning of treatments should also integrate regional or local information about relationships between wildfires and topography (see *Threats: Wildfire*), the composition of NSO diets or prey communities, and other ecological factors that could influence how thinning affects wildfires and NSOs.

Timber Harvest Effects on Prey

The primary prey for NSOs in California are dusky-footed woodrats (*Neotoma fuscipes*), northern flying squirrels (*Glaucomys sabrinus*), and tree voles (*Arborimus* spp.) (Zabel et al. 1995, White 1996, Ward et al. 1998, Farber and Whitaker 2005, Diller et al. 2010, Klamath National Forest, unpubl. data). Other important prey in the state (either in terms of frequency or biomass contributions to diets) include other voles (*Myodes californicus*, *Phenacomys* spp., and

Microtus spp.), deer mice (*Peromyscus* spp.), pocket gophers (*Thomomys* spp.), broad-footed moles (*Scapanus latimanus*), and juvenile brush rabbits (*Sylvilagus bachmani*) and snowshoe hares (*Lepus americanus*). These species have a broad array of habitat associations and thus, respond quite differently to timber harvesting and other forest disturbances (Zeiner et al. 1990). Below we focus solely on timber harvest effects on the three primary prey species for NSOs in California. It is important to acknowledge, however, that NSOs typically have broad diets (see diet studies cited above) and that other prey species may also influence spotted owl demographic rates (Ward and Block 1995, Rosenberg et al. 2003).

Dusky-footed woodrats can occur in relatively high abundances in old forest, particularly in riparian areas and other locations with a well developed understory or brush layer (Carey et al. 1992, 1999). However, they generally reach their highest abundances in stands of brushy pole-timber that develop following severe disturbances (Carey et al. 1992, 1999, Sakai and Noon 1993, Anthony et al. 2003, Hamm et al. 2007). Thus, intensive harvesting of intermediate-age stands can result in temporary increases in abundance of dusky-footed woodrats. There is little information regarding effects of less intensive harvesting on dusky-footed woodrats. Hamm and Diller (2009) rarely found dusky-footed woodrats in thinned stands on private timberlands in the Redwood Region. They suggested that thinning without prescribed burning was insufficient for promoting growth of the disturbance-adapted shrubs locally favored by the species (see *Threats: Wildfire* regarding short-term effects of fire on prey).

Densities and demographic rates of northern flying squirrels are positively associated with habitat elements found in forests (e.g., arboreal lichens, truffles, and snags: Rosenberg and

Anthony 1992, Carey 1995, Waters and Zabel 1995, Gomez et al. 2005, Meyer et al. 2005, Lehmkuhl et al. 2006). Thus, they are likely to respond negatively to intensive forms of timber harvesting (e.g., Waters and Zabel 1995). Northern flying squirrels are also generally sensitive to habitat fragmentation caused by intensive harvesting (Smith 2007). For example, Rosenberg and Raphael (1986) found that densities of northern flying squirrels in the California Klamath Province were substantially lower in the smallest and most insular habitat patches (due to surrounding clearcut harvesting) than in the largest and best connected patches.

Research concerning the effects of thinning and other lower-intensity forms of harvesting on northern flying squirrels has generated inconsistent results (e.g., Carey 2000, Ransome and Sullivan 2002, Ransome et al. 2004, Gomez et al. 2005, Manning et al. 2012). Some of the inconsistency appears to be due to whether treated young stands are compared with structurally simple young stands (e.g., Gomez et al. 2005), structurally complex young stands (e.g., Carey 2000), or stands that have not recently experienced harvesting (Holloway and Smith 2011). The available research suggests that treated stands are more likely to contain relatively low abundances of northern flying squirrels when compared with structurally complex or mature and old stands, whereas they may exhibit similar or even higher abundances when compared with structurally simple young stands. Harvest intensity and levels of retention appear to be another major determinant of thinning effects on northern flying squirrels, with higher intensity thinning (lower retention levels) having stronger negative effects (Meyer et al. 2007, Holloway and Smith 2011, Manning et al. 2012; but see Ransome et al. 2004). Whether thinning is patchy or uniform (in terms of location and intensity) might also be important. For example, thinning can reduce the availability of truffles, the northern flying squirrel's primary food, for more than 10-20 years;

but variable-density thinning appears to be less harmful than commercial thinning (Waters et al. 1994, Colgan et al. 1999, Luoma et al. 2003, Meyer et al. 2005).

Tree voles generally occur at higher densities in old forests than in young forests (reviewed in Sztukowski and Courtney 2004, USFWS 2011b) and selectively use forests containing higher concentrations of habitat elements typically found in older stands (e.g., older stand age, larger diameter downed wood, greater basal area: Dunk and Hawley 2009). Tree voles are thought to be highly vulnerable to logging and other disturbances that reduce the extent and contiguity of old forests (Carey 1991, Huff et al. 1992, Hayes 1996, Adam and Hayes 1998, USFWS 2011b). Some tree vole populations occur in intensively managed landscapes with little or no old forest (e.g., Thompson and Diller 2002). However, the US Fish and Wildlife Service (2011b) noted that "the limited evidence available suggests that tree vole occupation of younger forest stands may be relatively short-lived (Diller 2010, pers. comm.) or intermittent (Hopkins 2010, pers. comm.)." Based on the natural histories of these species, reducing or fragmenting older forest could negatively affect them; but retention of older Douglas-fir (*Pseudotsuga menziesii*) trees and patches of well-connected canopy might ameliorate those impacts (Hayes 1996, Adam and Hayes 1998, USFWS 20011b). Clear-cutting and other severe disturbances should have the strongest effects on tree voles, due to the species' diet, nesting habitat associations, arboreal mode of travel, and apparently poor mobility (USFWS 2011b).

Habitat Lost to Past Timber Harvesting (1800s to 1994)

Rigorous research has shown that the fitness of NSOs in the southern part of their range is highest in landscapes with large concentrations of suitable breeding habitat (reviewed above). Herein we review information showing that the current availability of suitable breeding habitat was strongly affected by past timber harvesting, which removed or modified the majority of old forest that existed historically.

We are unaware of any estimates of the amount of suitable NSO habitat that existed at the time of Euro-American settlement (early to mid-1800s). Nesting-roosting habitat for NSOs generally occurs in relatively old, structurally complex conifer forest (Blakesley 2004). It is therefore, reasonable to evaluate historical trends in old conifer forest as a rough proxy for changes in amounts of suitable NSO habitat (USFWS 1990). Estimates reviewed for the NSO's federal listing determination indicated that approximately 18-24 million acres of old forest existed in western Oregon and Washington and northwestern California during the early to mid-1800s (USFWS 1990). These estimates did not include all regions or potentially suitable forest types within the subspecies' range. After including all regions and conifer forests, Strittholt et al. (2006) estimated that about 40 million acres of old conifer forest (>150 yrs) existed at the time of Euro-American settlement (Table 2). This is a crude approximation, as it is based on incomplete historical information and an assumption that nearly all pre-settlement conifer forest was old (i.e., had not experienced severe disturbance within the previous 150 years). However, Strittholt et al. (2006) noted that their regional estimates closely matched previous estimates for similar

regions and forest types, suggesting that they provide a reasonable baseline for comparison with contemporary forest conditions.

Using satellite imagery, Strittholt et al. (2006) estimated that 11.5 million acres of old conifer forest existed in 2000 (Table 2). Thus, approximately 72% of old conifer forest was lost in the Pacific Northwest during the 19th and 20th centuries (Table 2). This estimated post-settlement loss of old conifer forest is similar to earlier estimates of 60-88% reviewed in the NSO's federal listing determination (USFWS 1990). Strittholt et al. (2006) did not provide estimates of old forest declines by political boundaries so we cannot report their estimates for California alone. Old conifer forests declined by 62% in the Klamath provinces ("Klamath-Siskiyou Forests") and 79% in the eastern Cascades provinces ("Eastern Cascades Forests"), both of which substantially overlap with the NSO's range in northern California. Other than in two small regions surrounding major population centers in Washington and Oregon, declines in old conifer forest were primarily caused by widespread intensive logging (Strittholt et al. 2006). Mountainous terrain in the Klamath and eastern Cascades limited timber harvesting compared with more accessible areas but major losses of old conifer forest nonetheless occurred in those areas (Strittholt et al. 2006). Strittholt et al. (2006) did not evaluate trends in amounts of old forest for the Redwood Province but other sources estimated that 85-96% of old redwood forest was lost to intensive timber harvesting during the post-settlement period (USFWS 1992).

Table 2: Area (ha) of Pacific Northwest ecoregions and estimated historical (early to mid-1800s) and contemporary (2000) extents of old (>150 yrs) and mature (50-150 yrs) conifer forest within them (from Strittholt et al. 2006).

	Cascade Mountain Leeward Forests	*North Cascades Forests*	*Puget Lowland Forests*	*Central Pacific Coastal Forests*	*Central & Southern Cascades Forests*	*Eastern Cascades Forests*	*Klamath-Siskiyou Forests*	*Willamette Valley Forests*
Ecoregion area	1,600,444	1,278,056	1,719,726	4,267,988	4,481,279	5,398,139	5,033,181	1,487,769
Historical old conifer area	960,279	766,844	1,289,794	3,627,790	3,361,005	3,238,883	2,516,625	371,942
Percentage of ecoregion	60	60	75	85	75	60	50	25
Current old conifer area	420,394	511,396	67,353	668,281	1,328,796	687,836	946,394	34,993
Percentage of historical old conifer	44	67	5	18	39	18	38	9
Percent change	−56	−33	−95	−82	−61	−79	−62	−91
Mature conifer area	285,441	203,994	229,634	683,098	899,266	947,215	1,239,586	270,362
Percentage of historical old conifer	30	27	18	19	27	25	49	73
Combined area	705,835	715,391	296,987	1,351,379	2,228,063	1,635,051	2,185,980	305,355
Percentage of historical old conifer	74	93	23	37	66	43	87	82

Strittholt et al. (2006) reported that the majority of old (78%) and mature (50%) conifer forest in 2000 existed on public lands. Nearly all of the remaining old redwood forest likewise occurs on public lands (cite *[http://www.nps.gov/redw/faqs.htm]*). Much of the current difference among ownerships in amounts of older forest and suitable breeding habitat is due to past timber harvest rates. For example, loss of forest to harvesting during the 1970s through early 1990s occurred at substantially higher rates on private timberlands than on federal lands (e.g., >2 times faster in western Oregon) (Cohen et al. 2002, Staus et al. 2002, Healey et al. 2008). Nonetheless, an estimated 32% of suitable breeding habitat for NSOs occurred on non-federal lands at the time of the Northwest Forest Plan's implementation (1994), so conservation efforts for NSOs on non-federal lands remain important.

Although timber harvesting was substantially curtailed on federal lands following implementation of the Northwest Forest Plan (Healey et al. 2008, Kennedy et al. 2012; reviewed below), biologists noted the possibility that NSOs would continue to decline for many years due to lag effects of past harvesting (Courtney et al. 2004). The NSO has a relatively low

reproductive rate and might therefore be unable to immediately recover following removal or reduction of threats (Noon and Biles 1990). Furthermore, substantial recruitment of old forest and suitable nesting-roosting habitat could take multiple decades in areas that formerly experienced widespread intensive harvesting (Moeur et al. 2011; see below). Past harvesting could therefore be among the causes of continuing poor demographic performance of some NSO populations (Courtney et al. 2004). Forsman et al. (2011) noted, however, that some populations are declining on lands not previously subjected to widespread intensive timber harvesting (e.g., some National Parks). Based on this observation, they concluded that lag effects of past timber harvesting poorly explain continuing population declines. Yet, it is possible that lag effects of past timber harvesting do contribute to some population declines but that these effects are obscured by those of other stressors, such as competition with barred owls or large severe wildfires. Lag effects from past harvesting might be similarly obscured on private timberlands by impacts from barred owls and continuing timber harvesting. Timber harvesting continues to occur at high rates on private lands and is still the primary source of habitat loss for NSOs in those areas (Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012).

Comment [UFS9]: Define "high" rate

Habitat Lost to Contemporary Timber Harvesting (1994-2007)

Davis and Dugger (2011) estimated NSO habitat trends following implementation of the Northwest Forest Plan. Their analyses were mostly limited to federal lands within the Plan area but they also estimated habitat trends on non-federal lands, as reported in the current NSO recovery plan (USFWS 2011a). In addition to these analyses, we review results presented by Moeur et al. (2011) because they provide some additional insight into recent habitat trends for

NSOs on non-federal lands. We do not review habitat trend estimates based on federal ESA Section 7 consultation records (Bigley and Franklin 2004, USFWS 2012b). These records provide a less consistent and complete data source than those used by Davis and Dugger (2011) (see Bigley and Franklin 2004). They may also overestimate habitat changes since they evaluate effects of planned projects, which may be greater than what is actually implemented (Bigley and Franklin 2004).

Davis and Dugger (2011) used remotely sensed (satellite imagery) vegetation data to model changes in habitat suitability for NSOs during the first 15 years of the Northwest Forest Plan (1994-2007 in California and 1996-2006 in Oregon and Washington). They modeled habitat suitability based on habitat attributes surrounding thousands of NSO pair locations. Suitable breeding habitat was defined as having both a probability of owl presence greater than expected based on random chance and environmental conditions typical of those found around nesting and roosting pairs. Habitat loss was defined as a change in suitability rank from suitable or highly suitable to marginal or unsuitable due to vegetation changes caused by forest disturbances. Davis and Dugger (2011) did not estimate recruitment of, or net changes, in breeding habitat. They felt that their remotely sensed data poorly captured the kinds of slow and subtle habitat changes that occur during development of intermediate-aged and older stands. However, Moeur et al. (2011) estimated trends in mature and old forests during the same time period, which could provide insight into net changes in breeding habitat for NSOs.

Comment [UFS10]: And forest inventory plot data

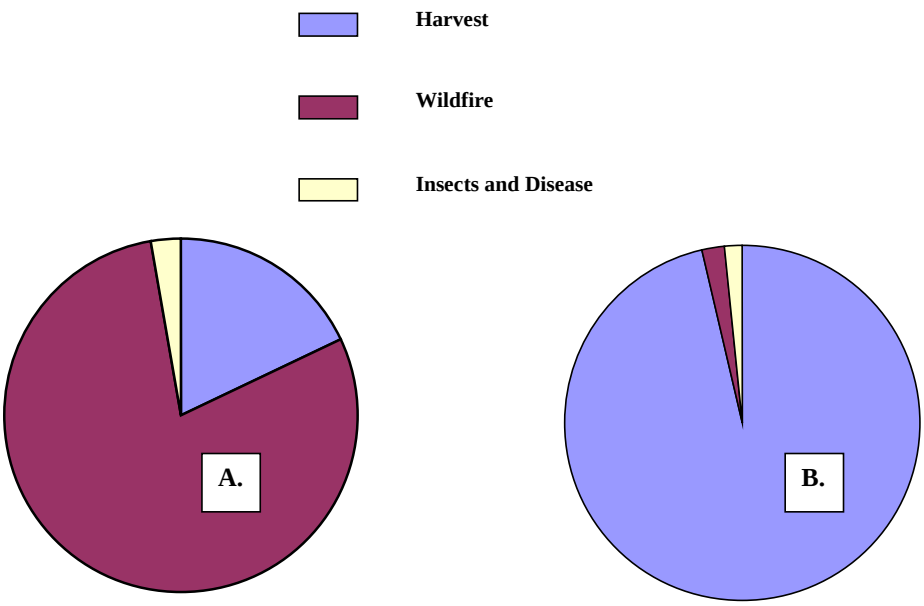
Table 3 shows estimated gross losses of suitable NSO breeding habitat to timber harvesting on federal and non-federal lands during the first 15 years of the Northwest Forest Plan (Davis and

Dugger 2011, USFW 2011a). Timber harvesting was responsible for a gross loss of about 54,000 acres (0.6%) of suitable breeding habitat on federal lands. This loss likely had little rangewide effect on NSOs but could have impacted the subspecies at local or regional scales. For example, harvesting resulted in a 3% gross loss of suitable breeding habitat on federal lands in the California Cascades, where habitat was already relatively limited. Approximately 92% of total suitable breeding habitat lost to timber harvesting occurred on non-federal lands. In contrast with federal lands, nearly all estimated gross habitat loss on non-federal lands was due to timber harvesting rather than natural disturbances (Figure 1; see *Threats: Wildfires*). In just 11-13 years, timber harvesting caused an estimated rangewide gross loss of 625,600 acres (15%) of suitable breeding habitat on non-federal lands. The largest losses on non-federal lands occurred in Oregon (301,200 ac, 22%) and Washington (234,200 ac, 19%). Non-federal lands in California experienced lower gross losses of suitable breeding habitat to harvesting (90,200 acres, 6%). Nonetheless, losses in all three states were substantial given the short time frame during which they occurred and the likelihood that little of the loss was offset by recruitment of suitable breeding habitat during that period (see below).

Table 3: Estimated gross losses of suitable NSO breeding habitat on federal and non-federal lands due to timber harvesting during 1994-2007 in California and 1996-2006 in Oregon and Washington (adapted from Davis and Dugger 2011, USFWS 2011a).

State	Ownership	Province	1994/1996 Ac	Harvest Ac	Harvest %
California	**Federal**	CA Cascades	213,200	6,500	3.0%
		CA Klamath	1,489,800	4,400	0.3%
		CA Coast	145,400	300	0.2%
		CA Federal Total	**1,848,400**	**11,200**	**0.6%**
	Non-Federal		**1,556,700**	**90,200**	**5.8%**
	Combined		**3,405,100**	**101,400**	**3.0%**
Oregon	**Federal**	OR Coast Range	611,200	3,300	0.5%
		Western OR Cascades	2,258,700	13,900	0.6%
		Eastern OR Cascades	402,900	5,800	1.4%
		Willamette Valley	3,400	100	2.9%
		OR Klamath	985,000	6,800	0.7%
		OR Federal Total	**4,261,200**	**29,900**	**0.7%**
	Non-Federal		**1,382,400**	**301,200**	**21.8%**
	Combined		**5,643,600**	**331,100**	**5.9%**
Washington	**Federal**	Olympic Peninsula	763,100	500	0.1%
		Eastern WA Cascades	673,600	8,100	1.2%
		Western WA Cascades	1,283,000	3,700	0.3%
		Western WA Lowlands	24,700	400	1.6%
		WA Federal Total	**2,744,400**	**12,700**	**0.5%**
	Non-Federal		**1,258,900**	**234,200**	**18.6%**
	Combined		**4,003,300**	**246,900**	**6.2%**
Rangewide	**Federal**		**8,853,800**	**53,800**	**0.6%**
	Non-Federal		**4,198,000**	**625,600**	**14.9%**
	Combined		**13,051,800**	**679,400**	**5.2%**

Figure 1: Proportions of suitable breeding habitat loss attributed to harvesting, wildfire, and insects and diseases on (A) federal lands and (B) non-federal lands during 1994-2007 (adapted from Davis and Dugger 2011 and USFWS 2011a).



The USFWS (2011a) and Davis and Dugger (2011) did not describe regional habitat trends for non-federal lands. However, insight into regional habitat trends on non-federal lands can be obtained from trends in mature and old forest during the same time period. Moeur et al. (2011) reported substantial gross losses of mature and old forest (mean DBH ≥20 in) on non-federal lands during the first 15 years of the Northwest Forest Plan. The largest gross losses, in terms of acreage, occurred in the Western Washington Lowlands (387,200 ac, 49%), Oregon Coast Range (362,500 ac, 50%), and California Coast Range (259,000 ac, 35%). All provinces and states within the NSO's range experienced large proportional losses, ranging from 31% in the Eastern Washington Cascades to 48% and 52% in the Klamath and Eastern Cascades of Oregon. Confirming the results of Davis and Dugger (2011; Figure 1), Moeur et al. (2011) found that gross losses of mature and old forest on non-federal lands were almost entirely due to timber harvesting (also see Kennedy et al. 2012).

Moeur et al. (2011) reported that gross loss of mature and old forest was substantially offset by recruitment into that habitat class. They noted, however, that given the short length of the monitoring period (10-14 yrs), recruitment was "likely due to incremental stand growth over the 20-in diameter threshold, or from understory disturbances that removed smaller diameter trees and raised the average stand diameter above the threshold, rather than from an increase in forests of much larger and older trees." Thus, it is unlikely that there was substantial recruitment of suitable and highly suitable breeding habitat for NSOs during this time period. This conclusion is supported by Davis and Dugger (2011), who found that most of the detectable habitat recruitment during their monitoring period occurred in the marginal suitability class, which more closely resembled their definition for dispersal habitat than for breeding habitat. Even if all

mature and old forest recruited during the first 15 years of the Plan provided suitable breeding habitat for NSOs, non-federal lands in California still experienced a net decline in area of mature and old forest during that period, and those in Washington and Oregon fared substantially worse (Moeur et al. 2011).

Future Harvesting in California

It is impossible to provide reliable projections of future timber harvesting or its effects on NSOs in California. Federal agencies have expressed support for widespread thinning to address wildfire risk on public lands in the state but we are unaware of any projections for harvest volume or effects on NSOs from these activities (USFWS 2011a, 2012a). Documents associated with Habitat Conservation Plans for private timberlands in California project substantial impacts of harvesting on some ownerships and relatively low impacts on others (see *Legal and Regulatory Framework* and *Threats: Inadequacy of Regulatory Mechanisms*). However, many landowners, in the state, including some large industrial timber companies, conduct timber harvesting outside of Habitat Conservation Plans. The state requires evaluation of potential environmental impacts of all Timber Harvest Plans but both landowners and responsible agencies have used inconsistent methods for conducting these evaluations. For example, some entities have strictly adhered to the state's Forest Practice Rules (CAL FIRE 2013), others have relied on poorly described and vetted variants of those rules (e.g., option "g+"), and still others have opted to follow US Fish and Wildlife Service (2009) recommendations. Based on an in-depth review of research concerning the NSO's habitat and spatial relationships, the US Fish and Wildlife Service (2009) recommended sweeping changes to NSO habitat retention guidelines in

the Forest Practice Rules for California's northern interior (CAL FIRE 2013). These recommendations are more scientifically supportable than are habitat retention guidelines in the Forest Practice Rules (CAL FIRE 2013), as they incorporate the large body of research of NSO-habitat relationships conducted since 1992 when guidelines in the Forest Practice Rules were created. In addition they were designed to enable CAL FIRE personnel lacking expertise with NSO-habitat relationships to properly determine if take would occur. However, the state has not officially adopted these recommendations or any other changes that incorporate the tremendous body of information about NSO-habitat relationships produced since 1992 (USFW 2009; see *Threats: Inadequacy of Regulatory Mechanisms*). Furthermore, since 2008, when the US Fish and Wildlife Service largely ceased providing technical assistance with timber harvest reviews in northern California, relatively few Timber Harvest Plans have been reviewed by personnel with sufficient biological expertise to evaluate whether or not take will occur (see *Threats: Inadequacy of Regulatory Mechanisms*). Lacking reliable harvest and take projections, and barring a major change in the legal or regulatory framework protecting NSOs, there is currently no reason to conclude that timber harvest effects on NSOs in California will substantially decline in the near future.

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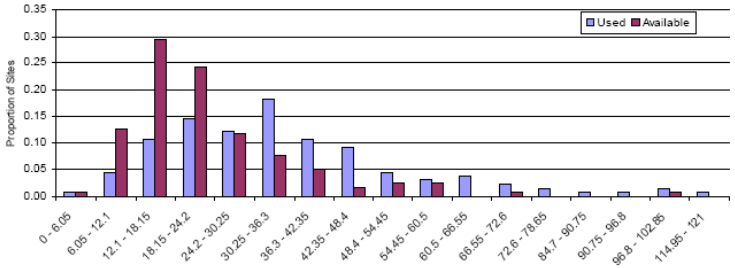
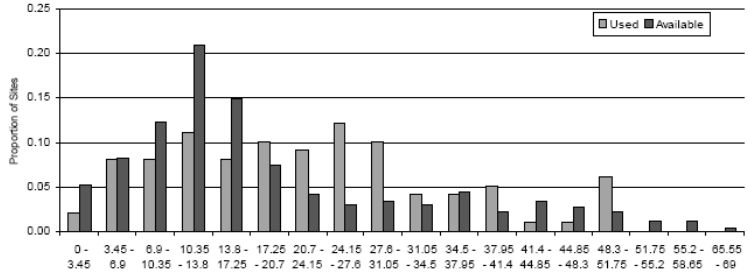
Appendix 1

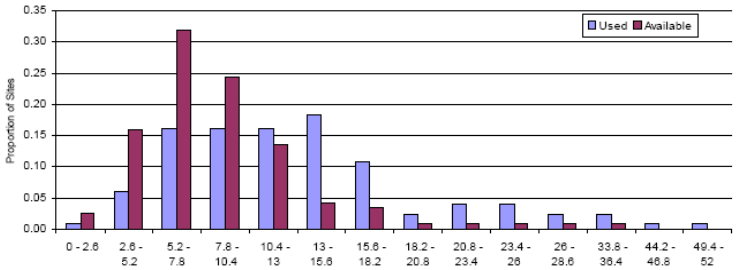
USFWS GNN (see USFWS 2011) histograms showing use by NSOs versus availability of select habitat attributes at a landscape-scale (200 ha, 494 ac) on Mendocino Redwood Co. (left column) and Green Diamond Resource Co. lands (right column).

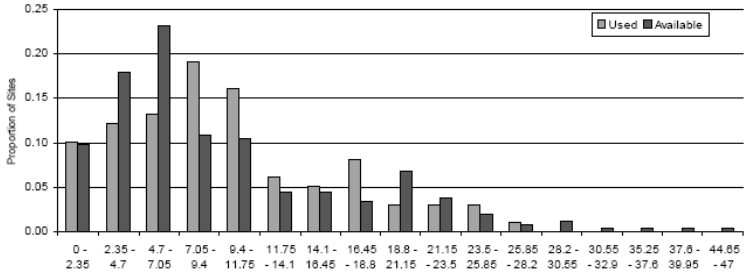
Live conifer trees per hectare ≥20 inches DBH:





Basal area of live conifers ≥20 inches DBH:

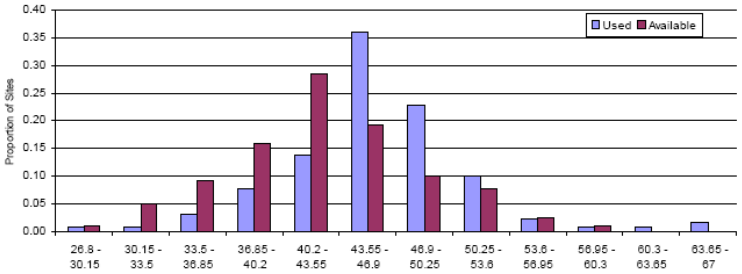


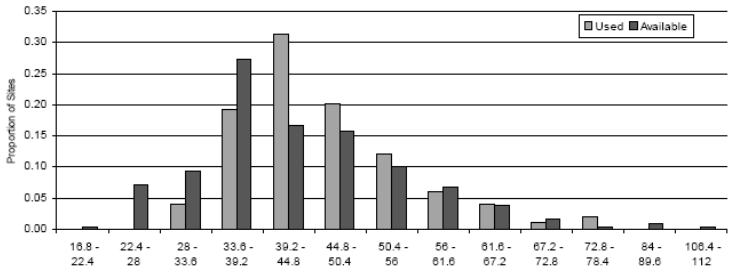


Appendix 1 (continued)

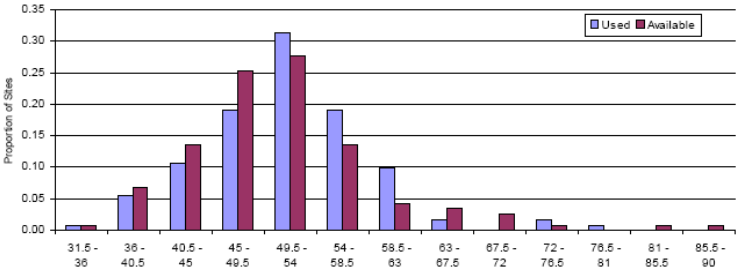
USFWS GNN (see USFWS 2011) histograms showing use by NSOs versus availability of select habitat attributes at a landscape-scale (200 ha, 494 ac) on Mendocino Redwood Co. (left column) and Green Diamond Resource Co. lands (right column).

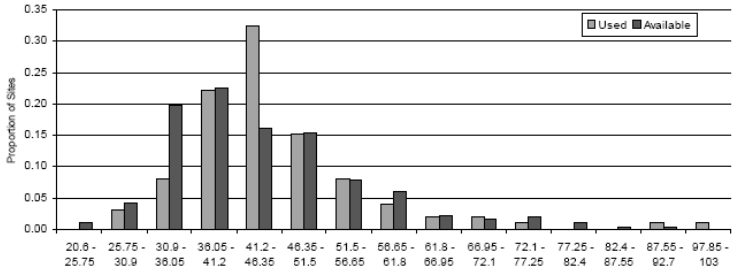
Basal area weighted mean diameter of all live conifers:





Basal area weighted stand age:

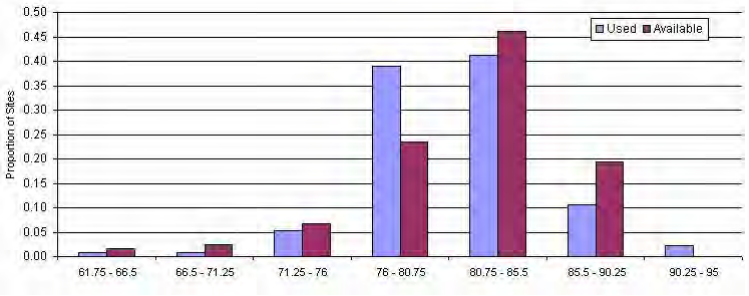


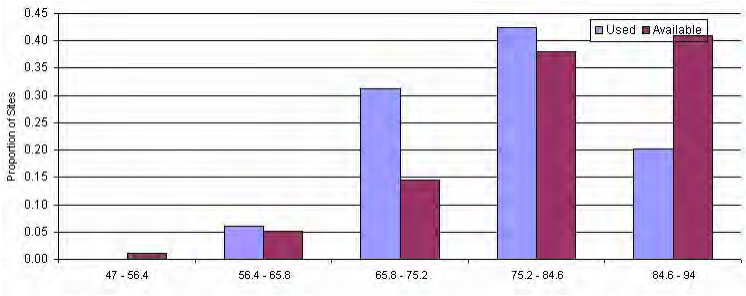


Appendix 1 (continued)

USFWS GNN (see USFWS 2011) histograms showing use by NSOs versus availability of select habitat attributes at a landscape-scale (200 ha, 494 ac) on Mendocino Redwood Co. (left column) and Green Diamond Resource Co. lands (right column).

Canopy cover:





Appendix 2

"Deconstructed" habitat suitability modeling (see text) for NSOs in the Redwood Province showing mean (SD, CV) values of select habitat structural attributes at the 200 ha (494 ac) scale by relative habitat suitability rank (USFWS 2011, unpubl. data).

Variable	Relative Habitat Suitability					
	Very Low	**Low**	**Med. Low**	**Med.**	**Med. High**	**High**
Strength of Selection*	-9.2	-2.9	-1.3	1.6	3.2	8.6
Percent of Region	16.3	24.5	26.2	22.4	9.7	0.9
Nesting-Roosting (ha)	57.9 (43.2, 75)	69.4 (40.8, 59)	79.9 (37.3, 47)	87.6 (33.6, 38)	94.3 (31.1, 33)	105.5 (30.8, 29)
Canopy Cover All Trees (%)	71.2 (23.2, 33)	75.2 (20.7, 28)	78.9 (18.1, 23)	81.0 (16.2, 20)	82.1 (15.5, 19)	82.9 (15.7, 19)
Canopy Cover Conifer (%)	43.9 (31.8, 72)	48.8 (30.0, 61)	53.4 (28.3, 53)	57.4 (27.2, 47)	61.4 (26.5, 43)	64.6 (26.4, 41)
BA Conifers ≥50 cm (m^2/ha)	10.6 (20.7, 195)	12.5 (23.0, 184)	14.1 (24.8, 176)	15.2 (25.6, 168)	17.6 (30.1, 171)	25.2 (45.9, 182)
BA Live Trees ≥75cm (m^2/ha)	7.3 (17.6, 241)	8.5 (20.1, 236)	9.3 (21.8, 234)	9.3 (22.4, 241)	10.4 (27.1, 261)	17.0 (44.0, 259)
Density Trees ≥50 cm (no./ha)	32.3 (37.3, 115)	36.1 (38.4, 106)	39.8 (40.0, 101)	42.5 (42.1, 99)	45.4 (44.5, 98)	50.0 (46.1, 92)
Density Trees ≥75 cm (no./ha)	8.2 (14.8, 180)	9.2 (15.6, 170)	9.9 (16.0, 162)	10.0 (15.5, 155)	10.4 (15.3, 147)	12.8 (16.8, 131)
Density Conifers ≥50 cm (no./ha)	22.1 (34.7, 157)	25.5 (36.4, 143)	28.9 (38.3, 133)	32.6 (40.7, 125)	37.3 (43.6, 117)	43.1 (45.6, 106)
Density Conifers ≥ 75 cm (no./ha)	6.6 (14.5, 220)	7.6 (15.4, 203)	8.4 (15.7, 187)	8.7 (15.2, 175)	9.4 (15.1, 161)	12.0 (16.7, 139)
Density Conifers ≥100 cm (no./ha)	2.5 (7.7, 308)	2.9 (8.4, 290)	3.2 (8.8, 275)	3.1 (8.3, 268)	3.2 (8.5, 266)	4.7 (10.4, 221)
Mean DBH Conifers by BA (cm)	42.8 (34.5, 81)	48.5 (35.5, 73)	51.7 (35.4, 68)	52.1 (34.0, 65)	52.9 (36.1, 68)	60.8 (51.1, 84)
QMD Dominant/Codominant Conifers (cm)	35.7 (29.1, 82)	40.2 (29.9, 74)	42.7 (29.7, 70)	42.6 (28.3, 66)	42.7 (29.7, 70)	48.1 (41.5, 86)

*Strength of selection calculated as the proportion of activity centers in a habitat suitability class divided by the proportion of the modeling region in that class.

September 5, 2014

Ken Hoffman's Peer Review of Timber Harvesting threat section:

Pg. 2

Non-federal lands contain the majority of the remaining breeding pairs of NSO. Not just the breeding habitat. There are more NSO in the Redwood zone on private land in Cali9fornia than the rest of the species range combined. Redwood NSOs are the last source population.

Pg. 7

"Most of the published research" or "a large amount of research" should replace "most of what is known"

'Diller et, al. 2010' is an unpublished document. Non-peer reviewed annual report.

Pg. 8

"Small patch clear-cutting"? WTF?

"Harvesting of suitable habitat (i.e. take)" should say "occupied" habitat.

Pg. 9

"Less is known"? A great deal is known. Less is published.

Pg. 10

Private timberland is divided almost equally between industrial and non-industrial. Most non-industrial timberland is uneven-aged managed – not uneven-aged regenerated.

Pg. 10-14

Thinning on National Forests occurs in even-aged stands surrounded by a sea of suitable habitat. Thinning on private land occurs either in long-term NTMPs or on industrial timberland which is a sea of unsuitable habitat.

Pg. 27

Market conditions drive harvest on non-industrial timberland. Industrial timberland is harvested on a schedule to keep mills in material.

-Thinning on Public land is insignificant for two reasons:

#1 Most NSO are on private land is in the Redwood zone

#2 Most Forest Service thinning occurs in a sea of suitable habitat.

HCPs all result in substantial impacts to NSO habitat and individuals through permitted take.

No mention of the USFWS guidance for the coast?

USFWS Technical Assistance almost immediately increased NSO habitat protections starting in 1999. Coastal guidance also far exceeds FPRs.

-Technical Assistance continued through 2008.

-Paper fails to recognize difference between industrial and non-industrial timber management.

Three Major Points

1. Redwood zone NSO is the last source population.
2. Redwood zone timberland is almost 50/50 industrial/non-industrial.
3. Inadequate regulatory mechanism and process for private land has been obvious since at least 1999.

Wildfire and Salvage Logging

Introduction

Recent status reviews have identified wildfire as a primary threat to the recovery of the northern spotted owl (NSO) (Courtney et al. 2004, USFWS 2011). Much of this concern was based on recent loss of suitable breeding habitat to wildfires and to the risk of extensive severe fires occurring in the future (Davis and Dugger 2011, USFWS 2011). Other researchers and stakeholders have questioned the scientific basis of claims that wildfires pose a threat to NSOs and have expressed distrust of agency recommendations for widespread use of forest thinning to reduce fire risk (e.g., Hanson et al. 2009, Heiken 2010, DellaSala et al. 2013).

There is currently limited information with which to evaluate responses of spotted owls to wildfires and post-fire salvage logging. This research suggests that wildfires have variable and complex effects on the species (Table 1). This is unsurprising given differences in wildfires, research methods, study areas, and spotted owl subspecies and populations. Nonetheless, patterns are evident in the literature concerning spotted owl responses to wildfires and salvage logging and these can be evaluated in light of the species' habitat and prey relationships. Currently available research suggests that low-to-moderate or mixed-severity wildfires have limited effects on spotted owls (Table 1). In fact, such fires could benefit NSOs in southern forests by contributing to landscape-level habitat heterogeneity associated with high fitness (Franklin et al. 2000). In contrast, large-scale severe (stand-replacing) fire can have strong negative effects on spotted owls (Table 1). This likely occurs when fires excessively modify,

reduce, or fragment concentrations of suitable nesting and roosting habitat needed for reproduction and survival (Franklin et al. 2000, Dugger et al. 2005, Schilling et al. 2013). Negative effects of extensive severe wildfires appear to be exacerbated by post-fire salvage logging, which structurally simplifies burned areas, removes important habitat legacies for prey, and creates high contrast habitat edges that educe spotted owls' use of burned areas (Clark 2007, Lee et al. 2012, 2013, Clark et al. 2013, Comfort 2013).

Regardless of scientific uncertainty regarding spotted owl responses to wildfire, it is clear that recent large wildfires have caused tremendous loss, degradation, and fragmentation of suitable habitat for NSOs on federal lands (Davis and Dugger 2011, USFWS 2011; also see Healey et al. 2008, Moeur et al. 2011). This is cause for concern since recovery of the subspecies largely relies on habitat protection on federal lands (USFWS 2011). Furthermore, much of the climate change research indicates that wildfires will continue to be a source of large-scale habitat change during coming decades (Westerling et al. 2006, Lenihan et al. 2008, Westerling and Bryant 2008, Littell et al. 2009). These concerns have prompted ecologists and federal agencies to advocate widespread forest thinning to reduce wildfire risk within the range of the NSO (USFWS 2008, 2011, 2012a, Franklin and Johnson 2012). However, the limited information currently available suggests that spotted owls often respond negatively to forest thinning (reviewed in Hansen and Mazurek 2010, USFWS 2011; see *Threats: Timber Harvesting*). Further research is needed to determine whether and how widespread thinning should be used in forests occupied by NSOs. Currently available information suggests that spotted owls tolerate, and possibly benefit from, low severity or patchy fire (Bond et al. 2002, 2009, Roberts 2008, Roberts et al. 2011, Keane et al. 2011, 2012, Lee et al. 2012, 2013, Comfort 2013). Thus, the current body of research

Comment [UFS1]: Not just continue, but become an increasing source…See Moritz et al. 2012, Dennison et al. 2014, Stavros et al. 2014 as additional references.

Comment [UFS2]: Perhaps couch NSO effects in terms of duration/time (e.g., short-term vs long-term). Much of the problem you write about here will be on how to balance the two.

Comment [UFS3]: Timber harvesting, which has mostly shifted from regeneration to thinning, was and is planned to occur under the NWFP (owl's range). Thinning is not always proposed by the land management agencies to reduce risks of large wildfires and future fire loss of habitat. It is also designed to accelerate development of old forest structure and species composition. Also, it is important to remember that it is also proposed to provide economic inputs to the local economies, another NWFP objective.

So, this statement should read that **research is needed to help land managers understand the effects of thinning on NSO**. Not to determine whether the agencies should thin. The NWFP calls for it.

supports use of prescribed fire and allowing wildfires to burn under prescribed conditions in dry forests within the NSO's range, provided sufficient concentrations of suitable habitat are retained.

Comment [UFS4]: This conclusion seems out of place. You have not made the argument yet.

Wildfire Effects on Spotted Owls

Indirect Evidence

Research of associations between landscape-scale habitat attributes and the demography and presence of NSOs has consistently found that the subspecies benefits from some form of habitat heterogeneity in the southern portion of its range (e.g., ecotones or edges between different vegetation classes) (Franklin et al. 2000, Zabel et al. 2003, Olson et al. 2004, Dugger et al. 2005, Carroll and Johnson 2008, Diller et al. 2010, Schilling et al. 2013; reviewed in *Threats: Timber Harvesting*). Yet, these same studies have also strongly demonstrated the importance of large concentrations of suitable breeding habitat around activity centers (reviewed in *Threats: Timber Harvesting*). Based on this research, wildfires likely have positive effects on NSOs in California when they contribute to beneficial forms of habitat heterogeneity and negative effects when they substantially reduce or degrade suitable habitat around activity centers. Extensive severe wildfires have the potential to remove or fragment core concentrations of suitable breeding habitat across multiple NSO territories. These fires, therefore, have the greatest likelihood of substantially impacting NSO populations. ~~Other~~ Smaller, less severe wildfires may impact fewer territories and have weaker negative effects on populations, burn in a manner that contributes to

Comment [UFS5]: I think it is more "fire-prone" portions rather than southern. The east side of the Cascades in Washington as example.

Comment [UFS6]: Delete?

Comment [UFS7]: Might want to define what you mean by "severe". Studies have used different definitions, which sometimes causes confusion.

Comment [UFS8]: Be careful on use of the term "habitat", should also define this too… nesting/roosting?

beneficial forms of habitat heterogeneity in some territories, or have a combination of these effects.

Direct Evidence

Several studies have investigated responses of NSOs to wildfires (Table 1). These studies provide crucial information for evaluating ~~fire~~ wildfire as a potential threat to the subspecies. However, their inferences are limited due to small sample sizes in all cases, the confounding effects of post-fire salvage logging in one case, and pooling of data from all three spotted owl subspecies in another case (Table 1; see below). In order to supplement these studies, we also reviewed research of fire effects on California spotted owls (*S. o. occidentalis*; CSOs) and Mexican spotted owls (*S. o. lucida*; MSOs) (Table 1). Because inferences from these studies are also limited, and given differences among fires, spotted owl populations, and research methods, we reviewed each project as a "case study". Relatively thorough descriptions of these studies allow identification of patterns in the literature, which could provide insights into general effects of wildfires on the species.

Table 1: Apparent effects of wildfires on spotted owls. See text for additional descriptions of study methods and findings.

Response Metric	Study†	Subspecies	Location*	Apparent Effect**	Notes
Survival	*Bond et al. 2002*	NSO, CSO, MSO	CA, AZ, NM	0	Only one post-fire survey season
	Clark et al. 2011	NSO	OR KLA	-	Likely cumulative effect of timber harvesting, severe fire, and salvage logging
Productivity	Gaines et al. 1997	NSO	E WA CAS	0 (-?)	Apparently no decline but possibly obscured by low reproduction year across population; Possibly lower total reproduction in burned landscapes due to lower pair occupancy; Only one post-fire season
	Bond et al. 2002	NSO, CSO, MSO	CA, AZ, NM	0 (+?)	Slightly higher than in other studies; Only one post-fire season
	Clark 2007	NSO	OR KLA	0 (-?)	Apparently no decline but low statistical power; Possibly lower total reproduction in burned landscapes due to lower pair occupancy
	Roberts 2008	CSO	CA SIERRA		
Site Fidelity	*Bond et al. 2002*	NSO, CSO, MSO	CA, AZ, NM	0	Site fidelity similar to other studies
Occupancy	Elliot 1985	CSO	CA COAST	-	Apparent abandonment by two pairs
	Gaines et al. 1997	NSO	E WA CAS	-	Post-fire occupancy was lowest found during five-year study; Only one post-fire season
	Jenness et al. 2004	MSO	AZ, NM	-	Statistically insignificant effect; Pooled all fire types and severities
	Keane et al. 2011, 2012	CSO	CA SIERRA/CAS	-/0	Extensive high severity fire apparently had a strong negative effect; Extensive low severity fire apparently had a neutral or weak negative effect; Possibly influenced by salvage logging
	Roberts et al. 2011	CSO	CA SIERRA	-	Authors concluded weak effect but pooled all fire types and severities
	Lee et al. 2012	CSO	CA SIERRA	0 (-?)	Similar occupancy at burned vs. unburned; Possibly lower occupancy at severely burned vs. other burned
	Lee et al. 2013	CSO	SO CA	-/0	Similar occupancy at burned vs. unburned; Significant reduction with extensive high severity fire in core area
	Clark et al. 2013	NSO	OR KLA	-	Cumulative effect of timber harvesting, severe fire, and salvage logging
Home Range	Clark 2007	NSO	OR KLA	-	Larger home ranges post-fire
	Bond et al. 2013	CSO	CA SIERRA	0(?)	Similar home range sizes to unburned areas
Roosting	King et al. 1997	NSO	E WA CAS	-	Apparent avoidance of moderate and severe burns
	Bond et al. 2009	NSO	CA SIERRA	-	Significant avoidance of moderate and severe burns during breeding season
Foraging	Clark 2007	NSO	OR KLA	+(?)	Apparent weak selection of moderately burned suitable habitat; Possible weak selection for severely burned suitable habitat; Very low use and availability of both moderately and severely burned suitable habitat
	Bond et al. 2009	NSO	CA SIERRA	+	Significant selection of severe burns
	Eyes 2014	CSO	CA SIERRA		
Roosting and Foraging	Comfort 2013	NSO	OR KLA	-/+ (?)	Preference for small patches of severely burned/salvage logged and avoidance of larger patches; Weak preference for low contrast edges (ecotones) created by fire

†Peer-reviewed publications shown in italics. * Locations: California Klamath (CA KLA); Eastern Washington Cascades (E WA CAS); Oregon Klamath (OR KLA); California, Arizona and New Mexico (CA, AZ, NM); California Central Coast Range (CA COAST); California at margin of northern Sierra Nevada and southern Cascades (CA SIERRA/CAS); California Sierra Nevada (CA SIERRA); southern California San Bernardino and San Jacinto Mountains (SO CA). **Apparent Effect: negative (-), positive (+), neutral (0), varied with fire severity and/or scale (/)—see Notes column and text for further explanations.

Comment [UFS9]: Not statistically, I'd keep the "0" here and delete the "+"

Survival

Wildfires may influence spotted owl survival in both the short- and long-term. For example, spotted owls, like other wildlife, could be injured or killed by smoke during fires (Singer and Schullery 1989, Smith 2000). Due to their poor mobility, young spotted owls with undeveloped flight feathers may be at particular risk of mortality during wildfires (Smith 2000). In addition to potential immediate effects, extensive moderate or severe wildfire might influence spotted owl survival over the longer-term by modifying habitat for roosting, foraging, or prey (see below).

Only two studies are currently available for evaluating effects of wildfires on spotted owl survival rates (Bond et al. 2002, Clark et al. 2011; Table 1). Bond et al. (2002) reported that 18 of 21 (86%) marked spotted owls were resighted one year after wildfires occurred in California, Arizona, and New Mexico. This minimum survival rate was similar to survival estimates found by long-term studies of the three spotted owl subspecies in unburned landscapes (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data). Extensive severe fire (36-88%) occurred in four of the eight territories for which fire severity was mapped and the other half primarily burned at low to moderate severity. Thus, even extensive severe fire did not appear to have a large effect on spotted owl survival one year post-fire.

Clark et al. (2011) found evidence of a negative effect of wildfires and/or salvage logging on survival of 23 NSOs in the Oregon Klamath Province. Severe fire and/or post-fire salvage logging occurred in 30% and 41% of suitable NSO habitat in the two study areas. Estimated mean annual survival rates for NSOs located inside fire perimeters (0.69) and apparently

Comment [UFS10]: You should include "in press" results from Tempel et al.

High severity fire had negative effect on survival.

Their results suggest that reductions in the area of high-canopy forest (>70%) resulting from either logging or high-severity wildfire could reduce the viability of California spotted owl populations and may be contributing to ongoing declines in abundance and territory occupancy.

They recommended fuels Rx in dense stands while trying to avoid decreasing canopy too much.

This is good one for short vs long term effects.

Comment [UFS11]: Page 1026 of the paper states this, "Most (6 of 8) territories burned >50% at low to moderate severity."

So, two of the four that had 36-80% were <50%.

Comment [UFS12]: I disagree with this conclusion. Extensive, yes… severe, no.

They concluded that "spotted owls may have the ability to withstand the immediate, short-term (1-year) effects of fire **occurring at primarily low to moderate severities** within their territory.

displaced by fires and post-fire salvage logging (0.66) were lower than in areas just outside the fire perimeters (0.85) and in an unburned reference study area in the neighboring southern Cascades (0.85: Anthony et al. 2006). The degree to which post-fire salvage logging in the study areas influenced NSO survival rates is unknown. The study's occupancy analyses indicated that pre-fire timber harvesting, high severity wildfire~~s~~, and post-fire salvage logging cumulatively impacted NSOs through reductions of suitable nesting/roosting habitat (Clark et al. 2013; see below).

Apparently contradictory findings by Bond et al. (2002) and Clark et al. (2011) may be due to several factors. The most obvious difference between the studies is that the areas studied by Clark et al. (2011) experienced post-fire salvage logging while those studied by Bond et al. (2002) did not. The limited information currently available indicates that salvage logging negatively affects spotted owls (reviewed below). Additionally, the populations studied by Clark et al. (2011) may have been particularly sensitive to habitat loss to wildfires due to intensive pre-fire timber harvesting across a checkerboard ownership. It should also be noted that Bond et al. (2002) only examined wildfire effects one year post-fire. Fire injuries and post-fire outbreaks of insects and pathogens can continue to result in tree mortality for up to several years after a wildfire (Ryan and Amman 1996, Gaines et al. 1997, Hood et al. 2007).

Reproduction

The spotted owl is a relatively long-lived species that exhibits a bet-hedging life history strategy (Noon and Biles 1990, Franklin et al. 2000). This means that individuals often forego breeding

Comment [UFS13]: Based on this, their recommendation was to:
1) Increase protection of older forest in dry forest ecosystems to prevent future habitat loss to timber harvest and
2) Mitigate potential losses of older forest to stand-replacing fire and subsequent salvage logging

I assume that recommendation #1 refers to "nesting/roosting" habitat (a.k.a. older forest)

Comment [UFS14]: I hear and read this a lot. Upon careful reading of Bond et al. (2002), these two papers actually don't contradict each other. Bond et al. (2002) make no conclusion on high-severity wildfire. They only summarized effect (or lack thereof) on low-moderate severity wildfire.

In fact, Clark and Bond actually corroborate on low-moderate wildfire.

Please end this misconception here!

Comment [UFS15]: This is very misleading. I suggest deleting this sentence.

Clark et al. clearly present that high severity wildfire is 1 of 3 causes of loss of NSO N/R habitat.

Bond et al. show no effect from low-mod severity fire.

Comment [UFS16]: One reason why mitigation to reduce future wildfire losses is more important now.

If we had landscapes similar to what they were historically, prior to extensive harvesting, then large wildfires would not be so worrisome.

Comment [UFS17]: Good point. Another potential effect (from anecdotal observations in some of the demographic study areas) could be that sometimes a pair will stick around for a year or two after a wildfire removes extensive amounts of nesting/roosting habitat in their territory. Then eventually, the move on or disappear. We have observed this in a few cases. But again, that was due to high severity, not low-moderate.

Comment [UFS18]: Maybe put years here? It might give some needed context to "short-term" vs "long-term" effects.

during poor environmental conditions in order to maximize their chance of surviving and reproducing in the future. Given the species' life history strategy, spotted owl reproductive rates are likely sensitive to environmental changes, including those brought about by wildfires. However, annual fluctuations in spotted owl reproduction caused by variation in weather, prey populations, or breeding condition could obscure effects of wildfires or other factors on reproduction (e.g., Franklin et al. 2000).

We are aware of four studies that examined potential effects of wildfires on spotted owl reproduction (Gaines et al. 1997, Clark 2007, Bond et al. 2002, Roberts 2008; Table 1). None of these studies found substantive evidence of a wildfire-induced decline in reproduction by the species and one indicated a potentially positive effect. In the eastern Washington Cascades, Gaines et al. (1997) found little difference in productivity (number of young per pair) between burned (0.2; n = 5 or 6/ yr) and unburned sites (0.3; n = 13-17/yr) one year after a predominantly moderate to severe wildfire. However, the post-fire survey season clearly occurred during a poor reproduction year, potentially making it difficult to detect a difference between burned and unburned sites. Clark (2007) found no significant differences in productivity in burned areas in the Oregon Klamath Province (n = 31 territories) and an unburned study area in the neighboring southern Cascades (Anthony et al. 2006). He noted, however, that his study likely lacked the statistical power to detect a difference if one occurred. Bond et al. (2002) found that seven pairs of spotted owls produced an average of 1.0 offspring during a single breeding season following wildfires in California, Arizona, and New Mexico. This was higher than productivity rates found in unburned areas during long-term studies of the three spotted owl subspecies (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data). *Add discussion of Roberts (2008)…*

Comment [UFS19]: Not significantly higher, and they cautioned against drawing conclusions from this given high annual variation in reproduction as you state above.

You need to clarify this as it implies that burned areas resulted in higher reproduction. Or delete it and stick to findings that have some statistical backing, otherwise you chance confusing the readers.

Currently available studies suggest that wildfires generally have minimal short-term effects on spotted owl reproduction (Gaines et al. 1997, Clark 2007, Bond et al. 2002) and that primarily low-to-moderate severity fire could positively affect reproduction (Roberts 2008). However, it might be difficult to capture fire effects on spotted owl reproduction (whether positive or negative) during short-term studies, particularly with only a single year of post-fire data (Franklin et al. 2000). In addition, it is possible that solely comparing productivity (e.g., offspring per pair) in burned and unburned areas could obscure a change in total reproduction in burned areas. Studies in Washington and Oregon reported post-fire declines in occupancy by pairs, suggesting that extensive severe fire can reduce reproductive opportunities for spotted owls (Gaines et al. 1997, Clark 2007; see below).

Comment [UFS20]: Apparently no effect.

Comment [UFS21]: Repetitious, already stated

The limited research investigating spotted owl-prey relationships has found positive associations between spotted owl reproduction and abundances or consumption of dusky-footed woodrats (*Neotoma fuscipes*) (White 1996), deer mice (*Peromyscus* spp.) (Ward 2001 cited in Ward and Block 1995, Rosenberg et al. 2003), or a suite of prey with diverse habitat associations (Ward and Block 1995). Abundances of deer mice, pocket gophers (*Thomomys* spp.), and other "pioneer" or "early-successional" prey often increase following fires (Ream 1981, Zwolak and Foresman 2007, Zwolak 2009, Bond et al. 2013). Dusky-footed woodrats appear to initially respond negatively to severe fires (Schwilk and Keeley 1998, Smith 2000) but not to patchy low severity fire, although loss of nest houses might have a brief negative effect on reproduction (Lee and Tietje 2005). However, it is possible that severe fire benefits dusky-footed woodrats over longer time periods (e.g., ≥5-20 yrs) through creation of brushy habitat. Crown fires should

Comment [UFS22]: There's a new paper by Ganey et al. 2014. Relative abundance of small mammals in nest core Areas and burned wintering areas of Mexican spotted Owls in the Sacramento Mountains New Mexico

Species richness and relative abundance of small mammals were greater in the burned wintering areas than in the associated unburned nest core areas

Comment [UFS23]: You jump straight from reproduction to prey… might want to add new section or make connection to reproduction as it also affects other demographic parameters.

negatively affect abundances of prey associated with well-canopied forest, such as northern flying squirrels (*Glaucomys sabrinus*) and tree voles (*Arborimus* spp.). These, along with dusky-footed woodrats, are the primary prey for NSOs in California (reviewed in *Threats: Timber Harvesting*). Low severity fires could also negatively affect northern flying squirrels and other prey associated with closed canopy forests by reducing dead woody materials, fire-intolerant understory plants, and truffles (Lehmkuhl et al. 2006, Meyer et al. 2007). Thus, wildfires likely have complex effects on NSO prey communities, depending on local or regional differences in prey community composition; wildfire size, severity, and configuration; and the length of time vegetation has had to regenerate following fire.

Occupancy

Potential wildfire effects on NSO population rates are most directly evaluated with measures of survival and reproduction. However, occupancy data are often more logistically and economically feasible to collect than are demographic data and could provide an early indication of population trends (MacKenzie 2005, Olson et al. 2005). Spotted owl occupancy is sensitive to environmental factors (Blakesley et al. 2005, Olson et al. 2005) so it is a potentially valuable measure of wildfire effects on the species. Nonetheless, occupancy data must be interpreted carefully since they can be strongly influenced by survey effort, analytical methods, and the presence of barred owls (Olson et al. 2005).

We evaluated nine studies of wildfire effects on spotted owl occupancy (Table 1). As described below and in Table 1, seven of these provided evidence of a negative effect of either severe fire or fire in general.

Two studies indicated potentially negative effects of wildfires on spotted owl occupancy but included few territories (Elliot 1985, Gaines et al. 1997). In Monterey County, California, informal yearly surveys suggested that two pairs of CSOs abandoned their territories for at least four years following a wildfire (Elliot 1985). The author did not describe the fire other than noting that it was extensive and caused substantial damage to understories and oaks in the previously occupied areas. In the eastern Washington Cascades, Gaines et al. (1997) found that two of six NSO sites were occupied one year after a predominantly moderate to severe wildfire. This was the lowest occupancy rate found during the five-year study period.

Two studies found statistically weak evidence of a negative effect of fire on spotted owl occupancy, but their methods may have precluded detection of stronger effects (Jenness et al. 2004, Roberts et al. 2011). Jenness et al. (2004) found a statistically insignificant tendency (p = 0.11) for higher occupancy rank (in ascending order: no owls, singles, pairs, reproductive pairs) in unburned sites than in paired burned sites in Arizona and New Mexico (paired sites were close to each other and had similar habitat and topography). Of the 29 paired-site comparisons, 14 (48%) had a higher occupancy rank in unburned sites, 6 (21%) had a higher rank in burned sites, and 9 (31%) were tied. In the Sierra Nevada of California, Roberts et al. (2011) found lower occupancy rates for CSOs in burned areas than in unburned areas, but the difference was not statistically analyzed (unmodeled occupancy = 0.50 in burned and 0.69 in unburned; modeled

occupancy = 0.46 in burned and 0.72 in unburned). Modeling by both studies indicated that spotted owl occupancy was more strongly influenced by habitat composition or structure than by whether or not fire had recently occurred in territories. However, both studies may have underestimated the impacts of severe fire due to pooling of diverse fire types and severities for analysis (including prescribed fires, wildfires, and wildfires allowed to burn under prescribed conditions).

Two studies found evidence of strong declines in occupancy in areas recently burned by extensive severe wildfire but both may have been confounded by post-fire salvage logging (Keane et al. 2011, 2012, Clark et al. 2013). In southwestern Oregon, Clark et al. (2013) examined how extensive wildfires and subsequent salvage logging affected occupancy dynamics of NSO pairs. In their first analysis, the authors compared pre- and post-fire occupancy dynamics in a burned study area in the Oregon Klamath Province (n = 22) to those in an unburned area in the nearby southern Cascades (Anthony et al. 2006). Combined, high severity fire and salvage logging removed or modified 26% of suitable nesting/roosting habitat in landscapes surrounding NSO activity centers in this area. The burned and salvage-logged study area experienced a 64% reduction in site occupancy during the post-fire period, compared with a 25% reduction in the unburned study area (difference not statistically analyzed). In the second analysis, the authors examined possible effects of severe fire and salvage logging on occupancy dynamics in 40 territories located in three burned study areas in the Oregon Klamath Province. In these areas, 19-26% of suitable habitat was burned at high severity and/or salvage logged. During the study's three-year post-fire period, site extinction probabilities were as high as 72% in two combined study areas and 92% in the third area. Site extinction probabilities in the

Comment [UFS24]: Always state what kind of habitat. Nesting/roosting, foraging, or dispersal?

burned study areas were best explained by a model that included extents of high severity fire, salvage logging, and early seral forest. Models that included these variables separately were not competitive with the model containing all three variables, suggesting that NSO occupancy declined due to cumulative habitat loss from severe fire and pre- and post-fire timber harvesting (see Clark et al. 2013: Table 6). The relative influence of these factors on occupancy is unknown, but the role of severe fire cannot be dismissed. For example, the highest extinction probability (92%) occurred in a study area with little salvage logging (<2%) of previously suitable habitat.

Keane et al. (2011, 2012) estimated occupancy of CSOs in two recently burned study areas near the margin of the Sierra Nevada and southern Cascades of California. One wildfire complex, and an unreported amount of post-fire salvage logging, resulted in an almost complete loss of potentially suitable CSO habitat in the area (70% of the area pre-fire vs. 6% post-fire consisted of mean canopy cover ≥40% and mean DBH ≥11 in). Pre-fire occupancy in this study area was unknown but the Forest Service identified 23 CSO activity centers in the area prior to the fires. Rigorous landscape survey coverage by Keane et al. (2011, 2012) confirmed occupancy in only one territory within the fire perimeter during each of two post-fire years, whereas approximately seven to nine territories were found post-fire in a surrounding one-mile survey buffer. The other area studied by Keane et al. (2011, 2012) primarily burned at low severity (ca. 60% of the area). Pre-fire occupancy was likewise unknown in this area but Forest Service pre-project surveys indicated the presence of about 10 territories. Surveyors confirmed occupancy of six territories in this area during the first and second years post-fire. While the study's findings are

preliminary and may have been influenced by post-fire salvage logging, they suggest that effects of large wildfires on CSOs are dependent on the extent of high severity fire.

Another study provided further evidence that effects of wildfires on spotted owl occupancy depend on the extent and location of high severity fire. Lee et al. (2013) compared occupancy dynamics of CSOs in 71 recently burned sites and 97 unburned sites in the San Bernardino Mountains and San Jacinto Mountains of southern California. An average of 23% of forest within burned "core areas" (500 ac around activity centers) experienced high severity fire (this percent is based on an assumption that the amount of pre-fire forest in burned core areas was the same as that reported for burned and unburned core areas combined). Mean annual probability of occupancy was 0.48 in unburned sites and 0.31 in burned sites. This difference was not statistically significant. However, Lee et al. (2013) did detect a statistically significant negative effect on occupancy when high severity fire burned more than 125 acres of forest within estimated core areas.

Two studies found that wildfires had neutral or positive effects on spotted owl occupancy (Bond et al. 2002, Lee et al. 2012). Bond et al. (2002) calculated site fidelity for spotted owls in 11 territories burned by wildfires in California, Arizona, and New Mexico. The fires burned most of the area within each estimated territory (territory size = ½ the nearest neighbor distance in each study area, based on previous studies). Half of the eight territories for which fire severity was mapped primarily burned at low to moderate severity and the other half experienced extensive severe fire (36-88%). Of 21 color-banded owls in the study, 18 (86%) were resighted the year after the fires and 16 (89%) of these were located in their pre-fire territory. Site fidelity

in this study was comparable to that in long-term studies of the three subspecies in unburned areas (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data). In the Sierra Nevada, Lee et al. (2012) compared post-fire occupancy in 41 recently burned and 145 unburned historical CSO territories. An average of 32% of forest in burned territories experienced high severity fire. The authors found no significant association between CSO occupancy and whether or not territories had recently experienced wildfire within a 494-acre circle around activity centers (mean occupancy was 0.76 at unburned sites and 0.80 at burned sites).

The studies reviewed above are not directly comparable due to differences in methods, spotted owl subspecies and populations, fire extents and severities, and the presence or absence of post-fire salvage logging. The preponderance of evidence suggests that spotted owls in fire-prone forests are generally resilient to wildfires (Bond et al. 2002, Jenness et al. 2004, Keane et al. 2011, 2012, Roberts et al. 2011, Lee et al. 2012, 2013). However, wildfires that severely burn large areas of suitable habitat can substantially impact spotted owl occupancy, particularly when it occurs in breeding-season core areas (Elliot 1985, Gaines et al. 1997, Clark et al. 2013, Keane et al. 2011, 2012, Lee et al. 2013). Post-fire salvage logging appears to increase the negative effects of extensive severe fire on spotted owl occupancy; most likely by reducing suitability of burned areas for prey and foraging (Lee et al. 2012, 2013, Clark et al. 2013; reviewed below).

Home Range Size and Habitat Use

Changes in the behavior of individual spotted owls may provide insight into the mechanisms by which wildfires affect populations. For example, post-fire changes in home range size may

reflect fire effects on spotted owl energy budgets through changes in travel distances and prey availability. Changes in energy intake and output could, in turn, influence survival, reproduction, and occupancy of spotted owls. Patterns of habitat use may also be informative. For example, selection or avoidance of burned areas may reflect changes in availability of prey or roosting habitat, which could, in turn, influence occupancy, reproduction, or survival of spotted owls.

To our knowledge, only two studies have evaluated spotted owl home range sizes in relation to wildfires (Clark 2007, Bond et al. 2013; Table 1). Clark (2007) found that annual home range sizes of NSOs inside two fire perimeters in the Oregon Klamath Province were larger after wildfires than before them (n = 14 owls pre-fire and 20 post-fire). He attributed this difference to owls expanding their home ranges in response to habitat fragmentation caused by severe fire and post-fire salvage logging. This hypothesis was supported by other research in the region, which found that NSOs had larger home ranges in fragmented forests than in areas with larger, more intact patches of habitat (Carey et al. 1992, Schilling et al. 2013). Another study in the region suggested that the energetic cost of increased travel in fragmented forest was greater than the energetic benefit of increased access to prey associated with early-successional habitats (Carey and Peeler 1995).

Bond et al. (2013) compared the breeding season home ranges of seven CSOs (from four territories) during a single post-fire year in the Sierra Nevada of California with those in other studies during the same year in other parts of the subspecies range (D. Call, T. Munton, and G. Zimmerman unpubl. data). An average of 23% of forest burned at moderate severity and 9% at

high severity within a 1.2 mile radius of the four nests. Pre-fire home range sizes were unknown but CSOs in the four territories did not appear to have unusually large home ranges following predominantly low to moderate severity wildfire.

Five studies have described patterns of habitat use by spotted owls in burned areas (King et al. 1997, Clark 2007, Bond et al. 2009, Comfort 2013, Eyes 2014; Table 1). King et al. (1997; also Bevis et al. 1997) described initial effects of wildfires on NSOs in two territories in the eastern Washington Cascades. One territory primarily experienced low to moderate severity fire and the other mostly burned at high severity. Both territories experienced an unreported amount of salvage logging in unsuitable or severely burned habitat. Most NSO locations (84% and 89%) in the two territories were daytime roosts. In the territory primarily burned at low to moderate severity, 80% of the pair's post-fire locations were in unburned habitat, 16% were in low severity burns, and 4% were in moderate severity burns. The pair did not appear to roost in severely burned areas. The second territory studied by King et al. (1997) was occupied by a single male. After the fire, the male shifted his activity to an unburned area two to three miles away but continued to occasionally use areas near his former activity center. Of those locations, 74% were in unburned habitat, 17% were in low severity burns, 5% were in moderate severity burns, and 4% were in high severity burns. Maps of burn severity classes and NSO locations indicate that owls in these two territories strongly selected unburned areas for roosting.

Clark (2007) evaluated habitat selection by 12 NSOs (7 territories) inside a wildfire perimeter in the Oregon Klamath Province. NSO locations were primarily nocturnal and may therefore, have largely represented foraging activity. Individuals in this area used all habitat classes, including

moderate and severe burns and areas that had been salvage logged. However, when the data from individuals were pooled for analysis, the owls exhibited a strong preference for nesting-roosting habitat that was unburned or burned at low severity (unburned and low severity were combined into a single class). NSOs in the study also selectively used moderately burned, previously-suitable habitat; although both use and availability of this habitat class were low compared with unburned or lightly burned habitat. Owls' use of burned areas was concentrated closer to activity centers, which was expected, given that spotted owls are central place foragers during the breeding season (Rosenberg and McKelvey 1999).

In the Sierra Nevada, California, Bond et al. (2009) described the habitat associations of seven CSOs from four territories during a single post-fire season. Of the four nests found during the study, one was approximately 0.3 mile outside the fire perimeter, one was in forest burned at low severity, and two were in forest burned at moderate severity. One of the two nest trees found in a moderate severity burn was apparently killed by the fire and one produced the only fledgling detected during the study. It is unclear from the paper whether these events occurred at the same nest or different nests. The four pairs roosted in all burn severity classes but exhibited statistically significant selection of low severity burns and avoidance of moderate and high severity burns. Only one of 60 roost sites was located in a high severity burn. Burned roost sites generally resembled unburned roost sites (>60% canopy cover and large-diameter trees). Bond et al. (2009) also evaluated CSO selection of foraging habitat in the area. Probability of use for foraging was highest when sites were burned and within 0.6 mile of nests or roosts. Probability of use was also positively associated with presence of edge between burn severity classes. Five of the owls foraged in high severity burns within 0.9 mile of nests or roosts more often than in

other burn severity classes. Bond et al. (2009) suggested that CSOs in these four territories selectively foraged in high severity burns in order to access abundant prey in those areas. This hypothesis was supported by their finding that high severity burns had the highest herb and shrub cover and highest basal area of snags of any burn severity class, including unburned. These features are key resources for spotted owl prey communities (Carraway and Verts 1991, Carey et al. 1999, Holloway and Smith 2011).

Comfort (2013) evaluated habitat selection (roost and foraging locations combined) by 23 NSOs in a burned area in the Oregon Klamath Province. Her best performing model for explaining habitat selection included habitat suitability, disturbance severity, high contrast edge, and low contrast edge. Habitat selection varied with spatial scale but NSOs exhibited a strong preference for higher habitat suitability and avoidance of higher severity disturbance (high severity fire and/or salvage logging). NSOs showed a preference for high contrast edge at small spatial scales (2-8 ac) and avoidance at medium and large scales (32-2,049 ac). NSOs also exhibited a weak preference for low contrast edge. Comfort (2013) concluded that patchy, mixed severity fire (small patches of high severity fire within a matrix of unburned and low-to-moderate severity fire) created conditions favored by NSOs in her study, whereas large patches created by high severity fire and salvage logging were strongly avoided. Salvage logging apparently contributed to conditions avoided by NSOs by structurally homogenizing burned areas, which increased the sizes of high severity patches and amounts of high contrast edge. However, the relative influence of high severity fire and salvage logging on habitat selection by NSOs in this study is unknown.

Add discussion of Eyes (2014) (selection of low severity burned edges and proportional use of high severity burned edges)…

The limited available information concerning spotted owl habitat use following wildfires indicates that the species avoids roosting in moderate and high severity burns (King et al. 1997, Clark 2007, Bond et al. 2009). This finding is concordant with the spotted owl's close association with densely-canopied older forest for roosting (Blakesley 2004). Little is known about the effects of wildfire on selection of nest sites. Bond et al. (2009) found three CSO nests in forest recently burned at low and moderate severity, and young fledged from one nest in a moderate severity burn. Moderate severity fire killed one of the four CSO nest trees in their study. We are unaware of any reports of spotted owls nesting in severely burned areas. Based on the species' nesting habitat requirements (Blakesley 2004), long-term use of severely burned areas for nesting is likely uncommon. Two studies specifically examined selection of foraging habitat by spotted owls *(adding Eyes 2014 will make three studies)*. Both found use of all burn severity classes, but Clark (2007) found a preference for foraging in unburned to moderately burned older forest while Bond et al. (2009) found a preference for severe burns. It is unclear if this difference was due to differences in the studies' methods, spotted owl diets, or effects of fire and timber harvesting (including post-fire salvage logging) on vegetation. Comfort's (2013) research suggested that NSOs respond positively to the presence of severe burns when they occur in small patches within a matrix of unburned or low-to-moderate severity burns. However, she combined roost and foraging locations in her analysis, which might have obscured differences in NSO use of burn severity classes for different functions. Furthermore, hers and Clark's (2007) studies were confounded by post-fire salvage logging, which appears to negatively affect spotted

owls (reviewed below). *Discussion of use of fire-created edges in Comfort 2013 and Eyes 2014…*

Salvage Logging

While salvage logging might be judiciously used to meet certain conservation objectives (e.g., generating downed wood to minimize erosion or create wildlife habitat), it is generally conducted to meet financial goals or remove hazard trees (Peterson et al. 2009). Intensive or poorly planned salvage logging can have a variety of negative effects on ecosystems, such as soil compaction, increased erosion, and impacts on insectivorous and cavity-nesting and -denning animals (reviewed in McIver and Starr 2000, Noss et al. 2006, Peterson et al. 2009).

Comment [UFS25]: Economic?

We know of three studies that have directly evaluated effects of post-fire salvage logging on spotted owls (Clark 2007 and Clark et al. 2013, Lee et al. 2012, 2013). Clark (2007) conducted a radio-telemetry study in areas recently burned by wildfires in the Oregon Klamath Province. He recorded limited use of salvage logged areas; presumably for foraging since locations were primarily nocturnal. Use of salvage logged areas was slightly lower than expected based on its abundance in territories (not statistically analyzed), indicating weak avoidance of salvage logged areas by NSOs. However, avoidance might have been stronger since some of the study's telemetry locations were potentially recorded prior to the occurrence of salvage logging. Most (60%) NSO locations in salvage logged areas occurred in riparian buffers, thinned areas, and patches of wildlife leave trees, rather than intensively salvaged areas. During the same study, Clark et al. (2013) found that post-wildfire declines in NSO occupancy were best explained by a

model that included extents of pre-fire timber harvesting, severe fire, and post-fire salvage logging. Models that included these factors separately were not competitive with this model, indicating that severe fire and pre- and post-fire harvesting collectively contributed to declines in NSO occupancy; most likely through cumulative habitat loss or degradation.

In the Sierra Nevada, Lee et al. (2012) recorded occupancy for eight CSO territories that experienced wildfire and post-fire salvage logging. Seven of the territories were occupied during the two-year period between the occurrence of wildfire and salvage logging, whereas none of the territories were occupied following salvage logging.

Lee et al. (2013) evaluated effects of salvage logging on CSOs in the San Bernardino and San Jacinto Mountains of Southern California. They noted that salvage logging in their study area was modest compared with commercial salvage logging typically employed in the Pacific Northwest and Sierra Nevada (salvage logging in their study area mostly consisted of firewood cutting on private in-holdings and hazard tree removal along Forest Service roads). Lee et al. (2013) did not find a statistically significant effect of post-fire salvage logging on CSO occupancy dynamics. However, site extinction probability was slightly higher, and mean annual probability of occupancy was slightly lower, in salvage logged areas than in other burned areas. Weak negative effects of light salvage logging were evident during all eight post-fire study years.

The limited available evidence suggests that salvage logging decreases the probability that spotted owls will use burned areas (Comfort 2013) and increases the probability that they will

abandon their territories following wildfires (Clark 2007, Lee et al. 2012, 2013, Clark et al. 2013). This likely occurs because salvage logging reduces suitability of burned areas for foraging spotted owls and their prey. Stands recently burned by moderate or severe fire often contain high biodiversity due to the presence of both early-successional conditions and key biological legacies in the form of snags, logs, and live trees (Noss et al. 2006). Due to fire suppression and salvage logging, stands with these conditions are currently rare in many fire-prone forests within the spotted owl's range (Noss et al. 2006). Selective use of moderate or high severity burns for foraging is likely due to spotted owls exploiting short-term increases in prey associated with both early-successional vegetation (e.g., shrubs) and legacy habitat elements (e.g., large diameter snags, logs, and live trees) (Ream 1981, Zwolak 2009, Bond et al. 2013). Salvage logging removes legacy elements, while associated use of herbicides reduces shrubs and grasses important to many prey species (Bond et al. 2013, Comfort 2013). In the longer-term, spotted owls can continue to benefit from the contributions of legacy habitat elements to regenerating stands. For example, large legacy snags, trees, and logs can provide valuable habitat elements for northern flying squirrels and other prey (Holloway and Smith 2011). Removal of these elements through salvage logging could therefore reduce the value of subsequent regenerating stands as prey habitat. Harvesting of legacy snags and live trees could also directly affect spotted owls by reducing availability of foraging perches in the short-term and suitable nest trees during later successional stages. Large-scale salvage logging could also reduce NSOs' use of burned areas by extensively replacing low contrast (diffuse) edges with less favorable high contrast (hard) edges (Comfort 2013).

Summary

Inferences from studies of direct effects of wildfires and salvage logging on spotted owls are limited due to inclusion of only a small number of spotted owls or territories. Nonetheless, more information is available concerning this topic than is generally acknowledged. The preponderance of currently available evidence indicates that spotted owls are often resilient to wildfires but can be strongly impacted by extensive severe fire. Following wildfire, many spotted owls may remain in their territories, exploit short-term increases in prey in burned areas, and continue to reproduce at reasonably high rates. However, wildfires that result in substantial loss or fragmentation of suitable habitat, particularly within breeding core areas, can cause spotted owls to increase their home range sizes, abandon their territories, and possibly, emigrate from burned landscapes or die of starvation or disease. Negative effects of severe fire appear to be greatest when suitable habitat is already limited (e.g., due to widespread intensive timber harvesting) and when post-fire salvage logging reduces suitability of burned areas for foraging and prey.

Comment [UFS26]: Seems to contradict previous sentence. Is the unacknowledged info in this paper? Suggest deleting this sentence.

Wildfire Effects on Recent Habitat Trends

Past and continuing habitat loss was a primary reason for listing the NSO under the federal Endangered Species Act (USFWS 1990). At the time of listing, the Fish and Wildlife Service estimated that 60-88% of the subspecies' habitat had already been lost (USFWS 1990; also see *Threats: Timber Harvesting*). They attributed most of this loss to widespread intensive timber harvesting. Since listing of the NSO and subsequent adoption of the Northwest Forest Plan,

timber harvesting has declined and wildfire has been identified as the primary source of forest disturbance and habitat loss on federal lands (Courtney et al. 2004, Healey et al. 2008, USFWS 2011, Kennedy et al. 2012). Nonetheless, timber harvesting continues to be the primary source of habitat loss on non-federal lands (Healey et al. 2008, Davis and Dugger 2011, Moeur et al. 2011, Kennedy et al. 2012).

Comment [UFS27]: This and Kennedy focused on LSOG. Not all late-successional/old-growth forest is NSO habitat (e.g., ponderosa pine, subalpine forests, etc.)

You should put in Lint 2005 and Davis et al. 2011 as they focused on NSO habitat status and trends.

Comment [UFS28]: Or include "older forests and" before "habitat loss".

Estimates of recent trends in amounts of NSO habitat, and of older forest in general, have been produced as part of monitoring efforts for the Northwest Forest Plan; and are therefore, largely restricted to the Plan's area and time span (Davis and Lint 2005, Moeur et al. 2005, 2011, Healey et al. 2008, Davis and Dugger 2011). We focus on estimates by Davis and Dugger (2011) because they replaced those of Davis and Lint (2005) and are more specific to NSO habitat than those of Moeur et al. (2005, 2011) and Healey et al. (2008). We do not review habitat trend estimates based on federal ESA Section 7 consultation records (Bigley and Franklin 2004, USFWS 2012) due to greater scientific uncertainty and methodological bias associated with those data (see Bigley and Franklin 2004). Trends described by Bigley and Franklin (2004), Moeur et al. (2005, 2011), Healey et al. (2008), and the US Fish and Wildlife Service (2012) quantitatively differ from those of Davis and Dugger (2011) but similarly indicate that wildfires have been the primary source of recent habitat loss on federal lands.

Comment [UFS29]: Like this is stated

Comment [UFS30]: Good! Finally, this is clarified. Thank you.

Davis and Dugger (2011) used remotely sensed (satellite imagery) vegetation data to model changes in habitat suitability across the NSO's range during the first 15 years of the Northwest Forest Plan (1994-2007 in California and 1996-2006 in Oregon and Washington). Habitat suitability was based on characteristics surrounding thousands of NSO pair locations across the

Plan area. Suitable breeding habitat was defined as having both a probability of owl presence greater than expected based on random chance and environmental conditions typical of those found around nesting and roosting pairs. Estimated habitat trends included gross loss of both suitable breeding habitat and "core" suitable breeding habitat (>330 ft from edge). Davis and Dugger (2011) considered habitat loss to have occurred when an area classified as suitable at the beginning of Northwest Forest Plan was later downgraded to a lower suitability rank (unsuitable or marginal) due to vegetation changes caused by forest disturbances. Davis and Dugger (2011) did not estimate recruitment of, or net changes in, breeding habitat because their remotely sensed data was incapable of accurately capturing relatively slow and subtle habitat changes during development of intermediate-aged and older stands. They did, however, estimate net trends in NSO dispersal habitat, which they defined as forest with a mean ~~canopy~~conifer cover of at least 40% and a mean conifer DBH of at least 11 inches. Recruitment of dispersal habitat was more detectable than that of breeding habitat due to more rapid and measurable growth in younger forest (some recruitment also occurred due to degradation of suitable breeding habitat by forest disturbances).

Comment [UFS31]: Nesting/roosting

Estimated gross losses of suitable breeding habitat on federal and non-federal lands are presented in Tables 2 and 3, respectively. During the first 15 years of the Northwest Forest Plan, wildfires were responsible for an estimated gross loss of 236,700 acres (2.7%) of suitable breeding habitat on federal lands rangewide and 13,100 acres (0.3%) on non-federal lands (1.9% combined). In California, wildfires removed an estimated 75,500 acres (4.1%) of suitable breeding habitat on federal lands and 5,600 acres (0.4%) on non-federal lands (2.4% combined). Approximately 70% of habitat loss to wildfire on federal lands occurred within the Oregon and California

Klamath Provinces (Table 2). Most of this habitat loss was caused by the 1999 Megram Fire and 2002 Biscuit Fire (Table 4). Fires in the Eastern Cascades Provinces of Washington, Oregon, and California contributed less to total habitat loss than did fires in the Klamath, but were often more destructive in terms of proportion of suitable habitat lost within individual fire perimeters. In contrast with federal lands, wildfires were responsible for very little habitat loss on non-federal lands; rather, timber harvesting accounted for most losses in these areas (Figure 1).

Table 2: Estimated gross losses of suitable NSO breeding habitat on federal lands due to wildfires during 1994-2007 in California and 1996-2006 in Oregon and Washington (adapted from Davis and Dugger 2011, USFWS 2011).

State	Province	Initial Acres	Acres Lost	Percent Lost
California	CA Cascades	213,200	1,800	0.8%
	CA Klamath	1,489,800	71,600	4.8%
	CA Coast	145,400	2,100	1.4%
Oregon	OR Coast Range	611,200	0	0.0%
	Western OR Cascades	2,258,700	28,900	1.3%
	Eastern OR Cascades	402,900	17,800	4.4%
	Willamette Valley	3,400	0	0.0%
	OR Klamath	985,000	93,600	9.5%
Washington	Olympic Peninsula	763,100	200	0.0%
	Eastern WA Cascades	673,600	20,000	3.0%
	Western WA Cascades	1,283,000	700	0.1%
	Western WA Lowlands	24,700	0	0.0%
Rangewide Total		8,854,000	236,700	2.7%

Comment [UFS32]: Initial assumption at implementation of the NWFP was that about 2.5% per decade would be lost to wildfire. We are seeing this exceeded in some provinces, but the reserve network was designed to be large enough to function with losses from large wildfires.

There was not "trigger" to review this in the NWFP but the large amounts lost in some areas is concerning. Most if this loss is also occurring in the reserved land use allocations. This may be a focus of future review after the 20yr monitoring reports are finished (2015).

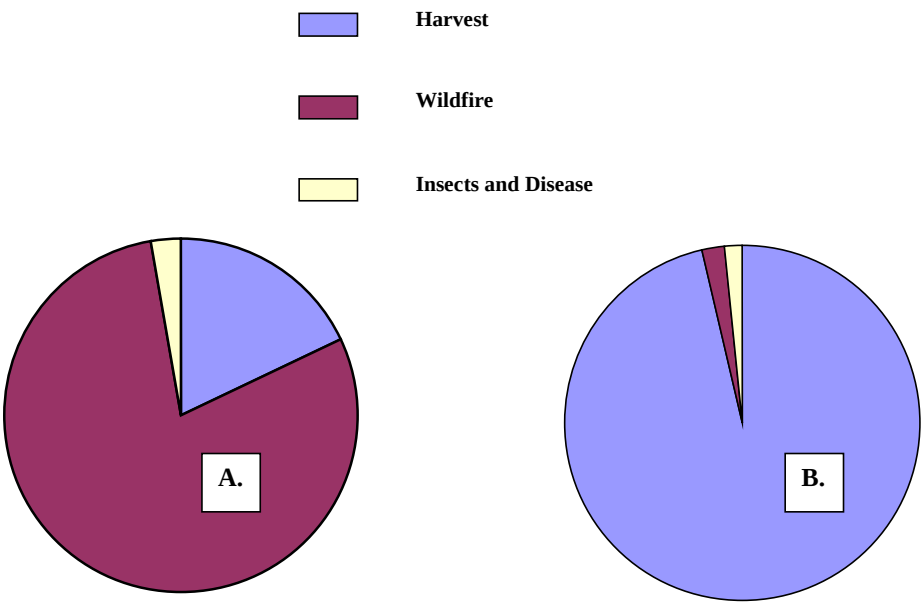
Table 3: Estimated gross losses of suitable NSO breeding habitat on non-federal lands due to wildfires during 1994-2007 in California and 1996-2006 in Oregon and Washington (adapted from Davis and Dugger 2011, USFWS 2011).

State	Initial Acres	Acres Lost	Percent Lost
California	1,556,700	5,600	0.4%
Oregon	1,382,400	5,100	0.4%
Washington	1,258,900	2,400	0.2%
Total	4,198,000	13,100	0.3%

Table 4: Estimated gross losses of suitable NSO breeding habitat to individual wildfires during 1994-2007 (note: "habitat degraded" describes areas downgraded from highly suitable to suitable; from Davis and Dugger 2011).

Fire name	Year	Broad physiographic province	Habitat in wildfire perimeter	Habitat lost		Habitat degraded	
			Acres	*Acres*	*%*	*Acres*	*%*
Biscuit Fire	2002	Klamath Mountains	226,230	93,730	41	12,019	5
Megram Fire	1999	Klamath Mountains	76,337	27,520	36	4,589	6
B&B Complex	2003	East Cascades	26,269	16,403	62	907	3
Bake-oven Fire	2006	Klamath Mountains	23,946	8,873	37	581	2
Boulder Fire	2002	West Cascades	34,059	8,460	25	2,074	6
Davis Fire	2003	East Cascades	8,050	6,943	86	5	0
Pigeon Fire	2006	Klamath Mountains	13,896	5,634	41	327	2
Rex Complex	2001	East Cascades	8,548	4,750	56	278	3
Timbered Rock	2002	West Cascades	10,216	4,539	44	569	6
Spring Fire	1996	West Cascades	13,504	3,931	29	858	6
Deep Harbor Fire	2004	East Cascades	5,761	3,930	68	64	1
Hancock Fire	2006	Klamath Mountains	12,712	3,132	25	336	3
Apple Fire 2	2002	West Cascades	12,227	2,810	23	928	8
Fischer Fire	2004	East Cascades	4,479	2,340	52	34	1
Fork Fire	1996	Klamath Mountains	2,962	2,113	71	14	0
Needles Fire	2003	East Cascades	1,946	874	45	1	0
Trough Fire	2001	Klamath Mountains	1,851	798	43	4	0
Hunter Fire	2006	Klamath Mountains	2,236	789	35	40	2
Deer Point Fire	2002	East Cascades	505	380	75	0	0
Tatoosh Complex	2006	East Cascades	666	378	57	0	0

Figure 1: Proportions of suitable breeding habitat loss attributed to harvesting, wildfire, and insects and diseases on (A) federal lands and (B) non-federal lands during 1994-2007 (adapted from Davis and Dugger 2011 and USFWS 2011).

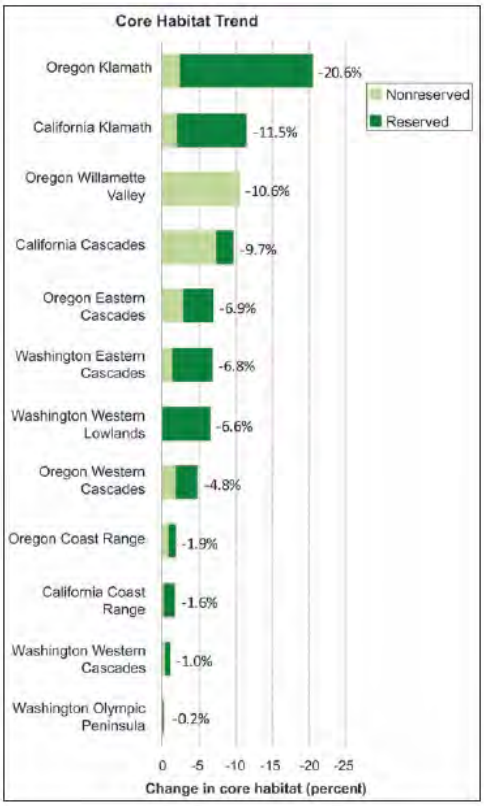


Davis and Dugger (2011) reported substantial losses of core breeding habitat on federal lands during the first 15 years of the Northwest Forest Plan (Figure 2). These losses primarily occurred in reserved areas. Changes in ratios of core and edge habitat classes indicated that increased fragmentation of suitable breeding habitat on federal lands was greatest in the Oregon and California Klamath Provinces and California Cascades Province (see Table 3-3 in Davis and Dugger 2011). Increased fragmentation in these regions was primarily due to wildfires.

Comment [UFS33]: Should define this as there are numerous uses of the term "core" in the literature.

Describe how it differs from the other habitat (e.g., nesting./roosting). Make it clear that the previous numbers above relate to N/R and not "core".

Figure 2: Gross losses of "core" suitable breeding habitat on reserved and non-reserved federal lands during 1994-2007 in California and 1996-2006 in Oregon and Washington (from Davis and Dugger 2011).



Davis and Dugger (2011) estimated a 5.2% net gain in NSO dispersal habitat. Much of this gain was due to succession in young forests in non-reserved lands at the margins of federal forests. However, accounting for forest connectivity and NSO dispersal distances, Davis and Dugger (2011) reported a 1% net loss of "dispersal-capable landscape". Much of the loss of dispersal-capable landscape in the Klamath and Eastern Cascades Provinces was due to large wildfires, whereas timber harvesting on non-federal lands was responsible for much of the loss in other regions.

Loss and fragmentation of breeding habitat to wildfires was likely at least partially offset by recruitment of new habitat through succession of mature and old forest. However, Davis and Dugger (2011) found that detectable recruitment of breeding habitat primarily occurred in the marginal suitability class. This finding was supported by Moeur et al. (2005), who found that about 90% of recruitment of older forest (mature and old-growth combined) during the first 10 years of the Northwest Forest Plan was at the lower end of the class' diameter range (i.e., mean DBH 20-30 in). In fact, Moeur et al. (2011) noted that given the short length of the monitoring period (10-14 yrs), recruitment was "likely due to incremental stand growth over the 20-in diameter threshold, or from understory disturbances that removed smaller diameter trees and raised the average stand diameter above the threshold, rather than from an increase in forests of much larger and older trees." It is likely that some newly recruited mature forest provides suitable habitat for NSOs but much if it could lack the canopy layering, large diameter snags and logs, and other structural attributes typical of nesting and roosting habitat (Blakesley et al. 2004).

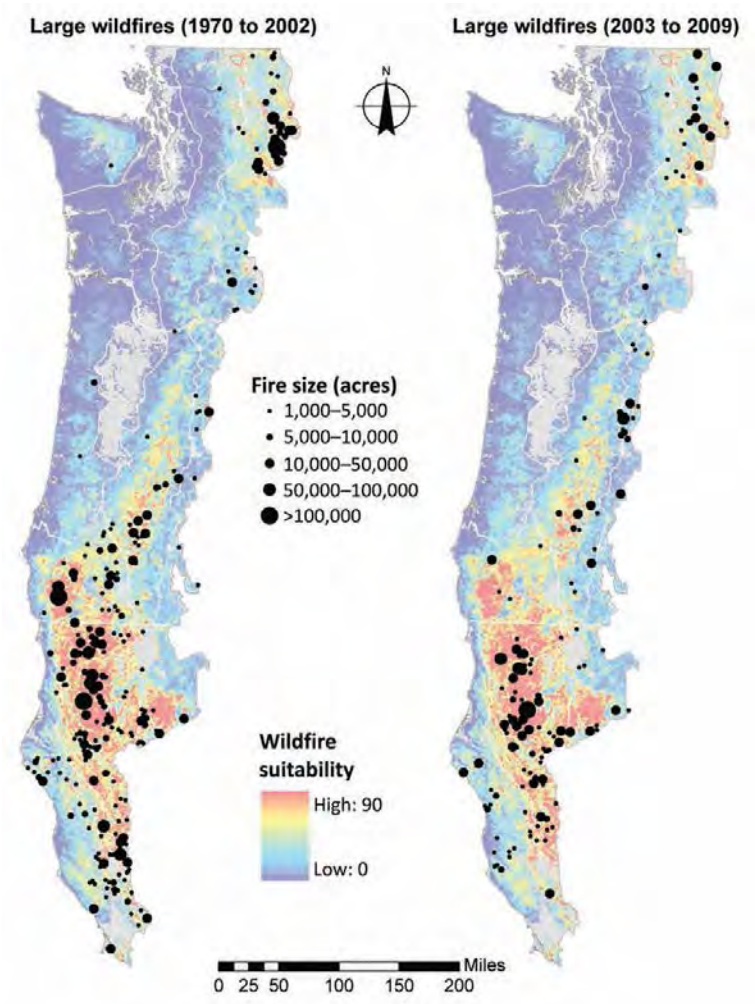
Loss and fragmentation of suitable breeding habitat does not necessarily equate to negative impacts on NSOs (Franklin and Gutiérrez 2002; e.g., Bond et al. 2002, 2009, Lee et al. 2012). Studies in southern Oregon and northern California found that the presence and fitness of NSOs are generally highest in landscapes with a mix of both large amounts of suitable breeding habitat and other habitat classes, such as foraging habitat or "non-habitat" (Franklin et al. 2000, Zabel et al. 2003, Olson et al. 2004, Dugger et al. 2005; reviewed above and in *Threats: Timber Harvesting*). Fitness is also generally highest when suitable breeding habitat occurs in large or clustered patches with large amounts of ecotone or edge between vegetation classes (Franklin et al. 2000, Dugger et al. 2005). This combination of conditions is important because it provides NSOs with a balance of resources needed for both survival and reproduction (Franklin et al. 2000). Active fire regimes in dry forests within the NSO's range in California (e.g., mixed-conifer and interior mixed-evergreen) historically contributed to these conditions by generally sparing older forest and maintaining some form of habitat heterogeneity at both stand and landscape scales (Sugihara et al. 2006). Some contemporary wildfires may still burn in this manner and thereby continue to perform an important ecosystem function in these forests. However, large severe wildfires have contributed, along with fire suppression and timber harvesting, to homogenization of some forests in interior northern California (Skinner et al. 2006). Thus, large severe fires may impact NSOs in California through loss of habitat heterogeneity, as well as reduced amounts and connectivity of suitable breeding habitat.

Fire Risk in California

Prior to Euro-American settlement, dry forests in the western U.S. generally experienced relatively frequent, low-to-moderate or mixed severity fire regimes (Agee 1993, Sugihara et al. 2006, Van de Water and Safford 2011). Mean pre-settlement fire return intervals in California were 11 years in yellow pine forests (e.g., *Pinus ponderosa*, *P. jeffreyi*), 11-16 years in mixed-conifer forests, and 29 years in mixed-evergreen forests (Van de Water and Safford 2011). Mean fire return intervals in redwood forests south of Del Norte County were also relatively frequent (6-44 yrs: Stuart and Stephens 2006). Frequent fire during the pre-settlement period generally maintained forests with less dense and more clumped tree distributions, higher proportions of fire-resistant trees (i.e., larger size classes and more fire-tolerant species), and lighter and less continuous fuel beds than occur today (Sugihara et al. 2006). In northern California, this characteristic fine-scale structural heterogeneity was often overlaid with coarser variability created by effects of elevation, terrain, soils, and other physiographic factors on fire and vegetation patterns (Skinner and Taylor 2006, Skinner et al. 2006, Sawyer 2007). For example, in areas of deeply incised terrain in the Klamath Mountains and southern Cascades of California, upper slopes and south and west facing aspects typically experienced more frequent and severe fire than did other areas (Skinner and Taylor 2006, Skinner et al. 2006). Due to fire suppression, early-successional vegetation communities formerly maintained by frequent, small-scale severe fire have greatly declined in some areas of California (Skinner 1995, Nagel and Taylor 2005). Overall, research indicates that fire suppression and other human activities have led to decreased forest heterogeneity at both stand and landscape scales and have contributed to substantial changes in fire regimes in California's dry forests (Sugihara et al. 2006).

Due to fire suppression, fire-free periods have dramatically increased in California's interior forests (Sugihara et al. 2006). For example, fire rotation near Hayfork in the Klamath Mountains increased more than 10-fold (from 19 to 238 yrs) during the post-settlement period (Taylor and Skinner 2003). Abnormally long fire-free periods have facilitated increased accumulation and continuity of fuels in dry, fire-prone forests (Sugihara et al. 2006). When wildfires do occur there is often an increased risk of them becoming very large and for suppression forces to be overwhelmed by their size and number (e.g., CAL FIRE 2008). As in other dry forests across the western U.S., the mean and maximum sizes of wildfires, and the total annual area burned, significantly increased in California's dry montane forests during the 20th and early 21st Centuries (Miller et al. 2009, 2012). CAL FIRE (2008) noted that more than half of the 26 largest fires recorded in California during 1932-2008 occurred during the last eight years of that period. Based on recent (1970-2002) fire behavior, most of northern California's interior can be classified as highly prone to large (>1,000 ac) wildfires (Davis et al. 2011; Figure 3).

Figure 3: Relative probability of large (>1,000 ac) wildfires ("wildfire suitability") across the NSO's range (from Davis et al. 2011). Modeling was based on landscape and climatic characteristics of locations at which large wildfires occurred during 1970-2002 (left) and was compared with subsequent (2003-2009) locations (right).



Studies of recent trends in extents of high severity fire in California have found conflicting results. For example, Miller et al. (2009) and Miller and Safford (2012) reported a substantial increase in the extent of high severity fire in the Sierra Nevada and southern Cascades of California during 1984-2010; while Hanson and Odion (2014) did not find an increase in the Sierra Nevada during the same period. Similarly conflicting results have been found for the California Klamath Province. Hanson et al. (2009) found a significant increase in high severity fire in the California and Oregon Klamath Provinces during 1984-2005; whereas Miller et al. (2012) did not find an increase in the California Klamath Province during a similar period (1987-2008). Scientific debate ensued regarding the appropriateness of methods used in various studies to determine trends in high severity fire (e.g., Hanson et al. 2009, 2010 vs. Spies et al. 2010).

Differences in findings regarding trends in high severity fire are related to variation in studies' temporal and spatial scales of analysis, as well as in methods for determining fire severity (Courtney et al. 2008, Miller et al. 2012, Hanson and Odion 2014). For example, Miller et al. (2012) noted that both their own study and those of Odion et al. (2004, 2010) may have underestimated trends in high severity fire in the California Klamath Province due to inclusion of unusual fire years. Studies by Odion et al. (2004, 2010) were based on fire effects during a single year (1987), which Miller et al. (2012) described as unusual. Large areas burned at below-average severity during 1987 due to abnormally strong inversions, and the fact that some of the fires burned well into fall (when conditions often favor lower severity fire). Miller et al. (2012) also noted that their own ability to detect a trend in fire severity could have been compromised by inclusion of both this year and 2008, which likewise experienced unusually large, low severity fires. Differences in the area analyzed could also affect evaluations of trends

in high severity fire. For example, the Biscuit Fire, which was predominantly located in Oregon, included extensive areas of high severity fire and therefore could have influenced results of trends analyses for the Oregon and California Klamath Provinces combined versus the California Klamath alone (Miller et al. 2012).

Regardless of whether or not the extent of high severity fire increased in California during the last two decades, large severe fires have recently occurred in these areas, and they were responsible for most loss, degradation, and fragmentation of suitable breeding habitat for NSOs on federal lands (Davis and Dugger 2011, Davis et al. 2011; see above). These data cannot be used to project how fires in the future will affect NSOs since they do not necessarily represent past or future fire conditions or effects. However, it is highly unlikely that wildfires will cease to be a major source of habitat loss for NSOs in the future. Rather, climate change research has generally projected a continued increase in the number and sizes of wildfires and the annual area burned during coming decades (Westerling et al. 2006, Lenihan et al. 2008, Westerling and Bryant 2008, Littell et al. 2009). There is scientific uncertainty regarding recent and future trends in the extent of high severity fire in California. Nonetheless, large severe fires will at least occasionally occur in the future and will continue to be a source of habitat loss and modification for NSOs in the state.

Increases in the number and sizes of wildfires, and effects of fires on NSO habitat, have led to calls for widespread use of thinning and other forms of active management in dry, fire-prone forests (USFWS 2008, 2011, 2012a, Franklin and Johnson 2012). Some researchers and stakeholders, however, have expressed doubts regarding estimates of fire risk and effects on

Comment [UFS34]: The trend in large wildfires has increased, and seems to be continuing (Happy Camp, July Complex…etc). With increased area burned, you get increased area burned in high severity.

Comment [UFS35]: Moritz et al. 2012, Stavros et al. 2014, Dennison et al. 2013

Comment [UFS36]: Perhaps it would be informative to include what was originally planned in the NWFP? The way this reads, and in general, the perception is that this is something newly proposed in response to recent large wildfires.

The NWFP (1994) actually called for widespread thinning in the fire prone portions of the owl's range to reduce risks of stand-replacing wildfires. And this was not just for NSO, it also focused on other late-successional species.

In fact, the agencies developed a specific Land Use Allocation called the "Managed Late Successional Area". Managed Late-Successional Areas are similar to Late- Successional Reserves (LSR) but were identified for certain owl locations in the drier provinces where regular and frequent fire is a natural part of the ecosystem. Certain silvicultural treatments and fire hazard reduction treatments are allowed to help prevent complete stand destruction from large catastrophic events such as high intensity, high severity fires; or disease or insect epidemics.

Even in regular LSRs in the Eastern Cascades or Klamath Provinces, silviculture aimed at reducing the risk of stand-replacing fires may be appropriate. Treatments may include thinning and underburning. Such activities in older stands in westside provinces may be warranted when levels of fire risk are high. (Record of Decision, B-7 to B-8).

Many of these treatments may reduce the quality of habitat for late-successional organisms. Thus, managers need to seek a balanced approach that reduces risk of fire while protecting large areas of fire-prone late-successional forest (B-8).

NSOs, concerns about potential effects of thinning on NSOs, and distrust of federal agency intentions (Hanson et al. 2009, Heiken 2010, DellaSala et al. 2013). As discussed in *Threats: Timber Harvesting* (also Hansen and Mazurek 2010, USFWS 2011), there is currently little known about the effects of forest thinning on spotted owls but the preponderance of evidence indicates that commercial thinning can have negative short-term effects on the species. Federal agencies should carefully consider this information when formulating land management policies and prescriptions aimed at reducing wildfire risk. Land managers should also consider greater use of prescribed fire and allowing wildfires to burn under favorable conditions; particularly at lower elevations in the California Klamath Province, where summertime inversions often minimize fire severity (Miller et al. 2012).

Comment [UFS37]: But how is this balanced with potential long-term beneficial effects? This is what needs to be determined in my opinion.

Again, from NWFP ROD - Many of these treatments may reduce the quality of habitat for late-successional organisms. Thus, managers need to seek a balanced approach that reduces risk of fire while protecting large areas of fire-prone late-successional forest (B-8).

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Wildfire and Salvage Logging

Comment [M1]: The entire document is written within a framework that assumes fire is bad. This biased treatment is unfortunate. I assume you've made fire a threat to the species as reason to list, but this synthesis of fire and owls is biased and poorly done. I stopped reading closely on page 15 because I was tired of correcting the errors and biased statements.

Recent status reviews have identified wildfire as a primary threat to the recovery of the northern spotted owl (NSO) (Courtney et al. 2004, USFWS 2011). Much of this concern is based on recent loss of suitable breeding habitat to wildfires and to the risk of extensive severe fires occurring in the future. Other researchers and stakeholders have questioned the scientific basis of claims that wildfires pose a threat to NSOs and have expressed distrust of agency recommendations for widespread use of forest thinning to reduce fire risk (Hanson et al. 2009, Heiken 2010, DellaSala et al. 2013). Surprisingly, existing reviews supporting both sides of this debate have only considered a portion of the available information concerning spotted owl responses to wildfires, and mostly in regard to their limitations and inconsistent findings. Our review confirms these limitations and suggests that wildfires have variable and complex effects on spotted owls. Fire is a crucial ecosystem process in dry forests within the species' range, and some spotted owl populations are known to benefit from a mix of habitat conditions resembling those historically maintained by active fire regimes (e.g., Franklin et al. 2000). However, the preponderance of currently available evidence indicates that large severe wildfires can have strong negative effects on spotted owls. Wildfires may also negatively affect spotted owls through cumulative or interactive effects with other environmental stressors, such as timber harvesting, salvage logging, and competition with barred owls (see *Potential Threats: Cumulative and Interactive Effects*).

Comment [M2]: unclear

Comment [M3]: wording ?

Comment [M4]: hmmmm.

Wildfire Effects on Spotted Owls

Several studies have investigated responses of NSOs to wildfires (Table A). These studies provide crucial information for evaluating fire as a potential threat to the subspecies. However, their inferences are limited due to small sample sizes in all cases, the potentially confounding effects of post-fire salvage logging in one case, and pooling of data from all three spotted owl subspecies in another case (Table A; see below). In order to supplement these studies, we also reviewed research of fire effects on California spotted owls (CSOs) and Mexican spotted owls (MSOs) (Table A). Because inferences from these studies are also limited, and given differences among fires, spotted owl populations, and research methods, we review each project as a "case study". Relatively thorough descriptions of these studies allow identification of patterns in the literature, which could provide insights into general effects of wildfires on the species.

Comment [M5]: ??

Table A: Apparent effects of wildfires on spotted owls. See text for additional descriptions of study methods and findings.

Response Metric	Study	Subspecies	Location*	Apparent Effect**	Notes
Mortalities	~~Paton et al. 1991~~	~~NSO~~	~~CA KLA~~	~~(?)~~	~~Likely due to heavy radio tags~~
	King et al. 1997	NSO	E WA CAS	-(?)	Only one individual; Possibly due to wildfire also logging
	Clark 2007	NSO	OR KLA	-(?)	Possibly due to wildfire and logging (unpublished grey lit.; analysis has problems)
Survival	Bond et al. 2002	NSO, CSO, MSO	CA, AZ, NM	0	Only one post-fire survey season
	Clark et al. 2011	NSO	OR KLA	-	~~Possible cumulative~~ Cumulative effect of timber harvesting, severe fire, and salvage logging
Productivity	~~Paton et al. 1991~~	~~NSO~~	~~CA KLA~~	~~(?)~~	~~Possible decline; Likely due to heavy radio tags~~
	Gaines et al. 1997	NSO	E WA CAS	0	Apparently no decline but possibly obscured by low reproduction year across population; Possibly lower total reproduction in burned landscapes due to lower pair occupancy; Only one post-fire season Anecdotal, no statistical power!
	Bond et al. 2002	NSO, CSO, MSO	CA, AZ, NM	0 (+?)	Slightly higher than in other studies; Only one post-fire season
	Clark 2007	NSO	OR KLA	0 ?	Apparently no decline but low statistical power; Possibly lower total reproduction in burned and logged landscapes due to lower pair occupancy
	Roberts 2008	CSO	CA SIERRA	+	Repro higher in burned.
Site Fidelity	Bond et al. 2002	NSO, CSO, MSO	CA, AZ, NM	0	Site fidelity similar to other studies
Occupancy	Elliot 1985	CSO	CA COAST	-?	Apparent abandonment by two pairs (this is anecdotal)
	Gaines et al. 1997	NSO	E WA CAS	-?	Post-fire occupancy was lowest found during five-year study; Only one post-fire season (anecdotal, no stats)
	Jenness et al. 2004	MSO	AZ, NM	0	Statistically ~~weak but pooled all fire types and severities~~no difference
	Keane et al. 2011, 2012	CSO	CA SIERRA/CAS	-?	Extensive high severity fire apparently had a strong negative effect; Extensive low severity fire apparently had a neutral or weak negative effect; Result influenced by extensive salvage logging
	Roberts et al. 2011	CSO	CA SIERRA	0	~~Authors concluded weak effect but pooled all fire types and severities~~No difference between burned and unburned
	Lee et al. 2012	CSO	CA SIERRA	0 (+?)	Similar occupancy at burned vs. unburned; Possibly ~~lower~~ higher occupancy at ~~severely~~ burned vs. ~~other~~ unburned
	Lee et al. 2013	CSO	SO CA	0/-	Similar occupancy at burned vs. unburned; Significant reduction with extensive high severity fire in core area
	Clark et al. 2013	NSO	OR KLA	-?	Cumulative effects of timber harvesting, severe fire, and salvage logging
Home Range	Clark 2007	NSO	OR KLA	-?	Larger home ranges post-fire and salvage logging
	Bond et al. 2013	CSO	CA SIERRA	0	Similar home range sizes to unburned areas
Roosting	King et al. 1997	NSO	E WA CAS	-	Apparent avoidance of moderate and severe burns
	Bond et al. 2009	NSO	CA SIERRA	-	Significant avoidance of moderate and severe burns
Foraging	Clark 2007	NSO	OR KLA	+(?)	Apparent weak selection of moderately burned suitable habitat; Possible weak selection for severely burned suitable habitat; Very low use and availability of both moderately and severely burned suitable habitat

Comment [M6]: This study says nothing about fire effects, should be omitted.

Comment [M7]: Always confounded fire and salvage logging. Don't assume or report that the effects were from fire!

Comment [M8]: This study says nothing about fire effects, should be omitted.

Comment [M9]: Always confounded fire and salvage logging. Don't assume or report that the effects were from fire!

Comment [M10]: Leave this to occupancy

Comment [M11]: Always confounded fire and salvage logging. Don't assume or report that the effects were from fire!

Comment [M12]: Always confounded fire and salvage logging. Don't assume or report that the effects were from fire!

	Bond et al. 2009	NSO	CA SIERRA	+	Significant selection of severe burns

* Locations: California Klamath (CA KLA); Eastern Washington Cascades (E WA CAS); Oregon Klamath (OR KLA); California, Arizona and New Mexico (CA, AZ, NM); California Central Coast Range (CA COAST); California at margin of northern Sierra Nevada and southern Cascades (CA SIERRA/CAS); California Sierra Nevada (CA SIERRA); southern California San Bernardino and San Jacinto Mountains (SO CA). **Apparent Effect: negative (-), positive (+), neutral (0), varied with fire severity (/)—see Notes column and text for further explanations.

Survival

Although adult spotted owls are capable of rapid and sustained flight, it is possible that they are occasionally killed by large or fast moving fires. Young owls with undeveloped flight feathers may be at particular risk of mortality during wildfires due to poor mobility (Smith 2000). Even in unburned areas, spotted owls could be injured or killed by smoke (Singer and Schullery 1989, Smith 2000). In addition to these potential immediate effects, extensive moderate or severe wildfire could influence spotted owl survival over the longer-term by ~~removing or~~ modifying habitat for roosting, foraging, or prey (see below).

Comment [M13]: Data-free statement, recommend deletion

Comment [M14]: Could be positive or negative modification, author should avoid judgmental language when speculating.

Only two studies are currently available for evaluating effects of wildfires on spotted owl survival rates (Table A). Bond et al. (2002) reported that 18 of 21 (86%) marked spotted owls were resighted one year after wildfires occurred in California, Arizona, and New Mexico. This minimum survival rate was similar to survival estimates in unburned areas found by other, longer-term studies of the three spotted owl subspecies (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data). Extensive severe fire (36-88%) occurred in four of the eight territories for which fire severity was mapped and the other half primarily burned at low to moderate severity. Thus, even extensive severe fire did not appear to have a large effect on spotted owl survival one year post-fire. ~~In contrast,~~ Clark et al. (2011) found evidence of a negative effect of combined wildfires and salvage logging on survival of 23 NSOs in the Oregon Klamath Province. Severe fire and salvage logging occurred in 30% and 41% of suitable NSO habitat in the two study areas. Estimated mean annual survival rates for NSOs located inside fire perimeters (0.69) and apparently displaced by fires and post-fire salvage logging (0.66) were

lower than in areas just outside the fire perimeters (0.85) and in an unburned reference study area in the neighboring southern Cascades (0.85: Anthony et al. 2006). The degree to which post-fire salvage logging in the study areas influenced NSO survival rates is unknown, but we know salvage logging has a documented negative effect on occupancy (Lee et al. 2013).~~.~~ The study's occupancy analyses indicated that pre-fire timber harvesting, wildfires, and post-fire salvage logging cumulatively impacted NSOs through reductions of suitable habitat (Clark et al. 2013; see below).

Apparently contradictory findings by Bond et al. (2002) and Clark et al. (2011) may be due to several factors. The most obvious difference between the studies is that the areas studied by Clark et al. (2011) experienced post-fire salvage logging while those studied by Bond et al. (2002) did not. Additionally, the populations studied by Clark et al. (2011) may have been particularly sensitive to habitat loss to wildfires due to intensive pre-fire timber harvesting across a checkerboard ownership. ~~It is also possible that Bond et al. (2002) failed to detect a negative effect of wildfires on spotted owls due to their reliance on data collected one year after fires occurred.~~It should be noted that Bond et al (2002) only examined effects 1 year post fire. Fire injuries and post-fire outbreaks of insects and pathogens can continue to result in tree mortality for up to several years after a wildfire (Ryan and Amman 1996, Gaines et al. 1997, Hood et al. 2007).

Comment [M15]: Bad language

Comment [M16]: Why assume tree death can only negatively affect SPOW? What about small mammal prey increases post fire?

There is relatively little information about the causes of spotted owl mortalities in recently burned areas. Not all spotted owl projects include radio-telemetry, which enables researchers to

recover dead birds. Even with radio-telemetry, only a portion of owl carcasses are recovered before they are too scavenged or decomposed to conduct a necropsy.

We reviewed three studies that described the condition of dead spotted owls found in areas recently burned by wildfire (Table A). In the Western Klamath Region of California, Paton et al. (1991) reported that one telemetered NSO died during a wildfire and that four more died within seven weeks. There was no evidence that any of the birds died from smoke inhalation or burns; rather, survival in the study area appeared to have declined due to the energetic costs of radio packs that were quite heavy by current standards. However, the authors noted that the telemetered owls in their study were exposed to high levels of carbon monoxide and total suspended particulates when an inversion trapped a dense layer of smoke near the ground for more than three weeks. In the eastern Washington Cascades, King et al. (1997; also Bevis et al. 1997) described a female NSO's behavior and death following a wildfire. The female and her fledgling survived when a low to moderate severity fire burned through the nest grove and produced thick smoke. Shortly after her offspring dispersed, the female moved to a new location outside the burned area and then died less than a month later. Her carcass was emaciated, indicating that she died from starvation or illness. King et al. (1997) speculated that this spotted owl died due to a post-fire decline in prey availability, but no prey data were presented. Clark (2007) reported that a total of eight NSOs died during his telemetry study in the Oregon Klamath Province. Six of the owls were recovered, all of which were emaciated. He likewise suggested that the deaths were related to a ~~post-fire~~ decline in prey availability due to combined effects of fire and salvage logging, but no prey data were presented. ~~This hypothesis was indirectly supported by the study's finding that NSOs' annual home ranges increased after wildfires~~

Comment [M17]: Without presenting examples direct mortality detected in telemetry studies of unburned areas, these anecdotes are biased and misleading.

Comment [M18]: Delete, this paper does not provide any data to judge fire effects on SPOW

Comment [M19]: Data-free anecdotes are dangerous.

Comment [M20]: But like King he had no data to support this speculation.

~~occurred (see below).~~ It is also possible that a factor other than fire caused a decline in prey availability or that the wildfires affected NSO survival in other ways. For example, loss or degradation of suitable roosting habitat could stress NSOs by limiting their ability to thermoregulate (Barrows 1981, Ganey et al. 1993, Ting 1998, Weathers et al. 2001).

Reproduction

The spotted owl is a relatively long-lived species that exhibits a bet-hedging life history strategy (Noon and Biles 1990, Franklin et al. 2000). This means that individuals often forego breeding during poor environmental conditions in order to maximize their chance of surviving and reproducing in the future. Given the species' life history strategy, spotted owl reproductive rates are likely sensitive to environmental changes, including those brought about by wildfires. However, annual fluctuations in spotted owl reproduction caused by variation in weather, prey populations, or breeding condition could obscure effects of wildfires on reproduction (e.g., Franklin et al. 2000).

We are aware of four studies that examined potential effects of wildfires on spotted owl reproduction (Table A). None of these studies found substantive evidence of a wildfire-induced decline in reproduction by the species. In the Western Klamath Region of California, ~~Paton et al. (1991) noted a possible difference in reproduction by NSOs in two areas burned by the same wildfire. Reproduction and fire effects were poorly described for burned versus unburned areas (e.g., number of eggs vs. number of fledglings; inversion-trapped smoke vs. understory burning in different areas of the fire), but demographic rates in the study appeared to be influenced by whether or not owls were fitted with heavy radio tags, rather than by variation in fire effects~~. In

the eastern Washington Cascades, Gaines et al. (1997) found little difference in productivity (number of young per pair) between burned (0.2; n = 5 or 6/ yr) and unburned sites (0.3; n = 13-17/yr) one year after a predominantly moderate to severe wildfire. However, it is possible that coincidentally low reproduction across the population during the post-fire year made it difficult to detect a difference between burned and unburned sites; particularly with such a small sample size. Clark (2007) found no significant differences in productivity in burned areas in the Oregon Klamath Province (n = 31 territories) and an unburned study area in the neighboring southern Cascades (Anthony et al. 2006). He noted, however, that his study likely lacked the statistical power to detect a difference if one occurred. Bond et al. (2002) found that seven pairs of spotted owls produced an average of 1.0 offspring during a single breeding season following wildfires in California, Arizona, and New Mexico. This was higher than productivity rates ~~found~~ found in unburned areas during long~~er~~-term studies of the three spotted owl subspecies (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data).

Comment [M21]: No statistics were presented.

Comment [M22]: Why speculate that a difference ought to be there?

Currently available studies suggest that wildfires generally have little or no short-term effects on spotted owl reproduction. However, it might be difficult to capture fire effects on spotted owl reproduction during short-term studies, particularly those with only a single year of post-fire data. In addition, it is possible that solely comparing productivity (e.g., offspring per pair) in burned and unburned areas could obscure a decline in total reproduction in burned areas. Studies in Washington and Oregon reported post-fire declines in occupancy by pairs, suggesting that wildfires may reduce reproductive opportunities for spotted owls (Gaines et al. 1997, ~~Clark 2007;~~ see below). Alternatively, it is possible that wildfires sometimes contribute to higher reproduction by spotted owls. For example some pairs or populations may experience higher

reproduction following wildfires due to short-term increases in availability of deer mice (*Peromyscus* spp.), pocket gophers (*Thomomys* spp.), and other prey (Ream 1981, Zwolak 2009, Bond et al. 2013).

Comment [M23]: Finally, the potential positive effects are mentioned.

Occupancy

Potential wildfire effects on NSO population rates are most directly evaluated with measures of survival and reproduction. However, occupancy data are often more logistically and economically feasible to collect than are demographic data and could provide an early indication of population trends (MacKenzie 2005, Olson et al. 2005). Spotted owl occupancy is sensitive to environmental factors (Blakesley et al. 2005, Olson et al. 2005) so it is a potentially valuable measure of wildfire effects on the species. Nonetheless, occupancy data must be interpreted carefully since they can be strongly influenced by survey effort, analytical methods, and the presence of barred owls (Olson et al. 2005).

We evaluated nine studies of wildfire effects on spotted owl occupancy (Table A). As described below and in Table A, all but one of these provided at least weak evidence of a negative effect of either severe fire or fire in general.

Two studies indicated potentially ~~strong declines~~ negative effects in spotted owl occupancy but included very few territories (Elliot 1985, Gaines et al. 1997). In Monterey County, California, informal yearly surveys suggested that two pairs of CSOs abandoned their territories for at least four years following a wildfire (Elliot 1985). The author did not describe the fire other than

noting substantial damage to understories and oaks in the previously occupied areas. In the eastern Washington Cascades, Gaines et al. (1997) found that ~~only~~ two of six NSO sites were occupied one year after a predominantly moderate to severe wildfire. This was the lowest occupancy rate found during the five-year study period.

Two studies found ~~only weak~~no evidence of ~~a negative~~any effect of fire on spotted owl occupancy, but their methods may have precluded detection of stronger effects (Jenness et al. 2004, Roberts et al. 2011). Jenness et al. (2004) found a statistically insignificant tendency for higher occupancy rank (no owls, singles, pairs, reproductive pairs) in unburned sites than in paired burned sites in Arizona and New Mexico (paired sites were close to each other and had similar habitat and topography). Of the 29 paired-site comparisons, 14 (48%) had a higher rank in unburned sites, 6 (21%) had a higher occupancy rank in burned sites, and 9 (31%) were tied. In the Sierra Nevada of California, Roberts et al. (2011) found ~~lower occupancy estimates for CSOs in burned areas (0.46) than in unburned areas (0.72); but the difference was not statistically analyzed~~no difference in occupancy between burned and unburned sites. Modeling by both studies indicated that spotted owl occupancy was more strongly influenced by habitat composition or structure than by whether or not fire had recently occurred in territories. However, both studies may have underestimated the impacts of severe fire due to pooling of diverse fire types and severities for analysis (including prescribed fires, wildfires, and wildfires allowed to burn under prescribed conditions). Roberts et al. (2011) may also have underestimated shorter-term effects of wildfires due to inclusion of data collected up to 15 years post-fire.

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Comment [M24]: Exactly corrrect

Comment [M25]: Why are you reaching for reasons that bad fire effects were absent rather than presenting the data in an unbiased voice?

Two studies found evidence of strong declines in occupancy in areas recently burned by extensive severe wildfire but both ~~may have been~~were seriously confounded by post-fire salvage logging (Keane et al. 2011, 2012, Clark et al. 2013). In southwestern Oregon, Clark et al. (2013) examined how extensive wildfires and subsequent salvage logging affected occupancy dynamics of NSO pairs. In their first analysis, the authors compared pre- and post-fire occupancy dynamics in a burned study area in the Oregon Klamath Province (n = 22) to those in an unburned area in the nearby southern Cascades (Anthony et al. 2006). High severity fire and salvage logging removed or modified 26% of suitable habitat in landscapes surrounding NSO sites in this area. The burned and salvage logged study area experienced a 64% reduction in site occupancy during the post-fire period, compared with a 25% reduction in the unburned study area (difference not statistically analyzed). In the second analysis, the authors examined possible effects of severe fire and salvage logging on occupancy dynamics in 40 territories located in three burned study areas in the Oregon Klamath Province. In these areas, 19-26% of suitable habitat was burned at high severity and~~/or~~ salvage logged. During the study's three-year post-fire period, site extinction probabilities were as high as 72% in two combined study areas and 92% in the third area. Site extinction probabilities in the burned and logged study areas were best explained by a model that included extents of high severity fire, salvage logging, and early seral forest. Models that included these variables separately were not competitive with the model containing all three variables, suggesting that NSO occupancy declined due to cumulative habitat loss from severe fire and pre- and post-fire timber harvesting. ~~The relative influence of these factors on occupancy is unknown, but the role of severe fire cannot be dismissed. For example, the highest extinction probability (92%) occurred in a study area with little salvage logging (<2%) of previously suitable habitat.~~

Comment [M26]: Incorrect. Their model structures were not organized to differentiate among these effects.

Comment [M27]: Using anecdotal examples as refutation of statistics is wrong and should be avoided.

Keane et al. (2011, 2012) estimated occupancy of CSOs in two recently burned study areas near the margin of the Sierra Nevada and southern Cascades of California. One wildfire complex, and an unreported amount of post-fire salvage logging, resulted in an almost complete loss of potentially suitable CSO habitat in the area (70% of the area pre-fire vs. 6% post-fire consisted of mean canopy cover ≥40% and mean DBH ≥11 in). Pre-fire occupancy in this study area was unknown but the Forest Service identified 23 CSO activity centers in the area prior to the fires. Rigorous landscape survey coverage by Keane et al. (2011, 2012) confirmed occupancy in only one territory within the fire perimeter during each of two post-fire years, whereas approximately seven to nine territories were found post-fire in a surrounding one-mile survey buffer. The other area studied by Keane et al. (2011, 2012) primarily burned at low severity (ca. 60% of the area). Pre-fire occupancy was likewise unknown in this area but Forest Service pre-project surveys indicated the presence of about 10 territories. Surveyors confirmed occupancy of six territories in this area during the first and second years post-fire. While the study's findings are preliminary and may have been influenced by post-fire salvage logging, they suggest that effects of large wildfires on CSOs are strongly dependent on the extent of high severity fire.

Comment [M28]: This study is weak, unpublished, and uninterpretable due to the unquantified salvage logging. Use with care, or not at all.

~~Another study provided further~~ The only published study that accounted for effects of logging and fire separately documented evidence that effects of wildfires on spotted owl occupancy depend on the extent of high severity fire. Lee et al. (2013) compared occupancy dynamics of CSOs in 71 recently burned sites and 97 unburned sites in the San Bernardino Mountains and San Jacinto Mountains of southern California. An average of 23% of forest within burned "core areas" (500 ac around activity centers) experienced high severity fire (this percent is based on an

assumption that the amount of pre-fire forest in burned core areas was the same as that reported for burned and unburned core areas combined). Mean annual probability of occupancy was 0.48 in unburned sites and 0.31 in burned sites. This difference was not statistically significant. ~~However, probability of occupancy was consistently lower in burned sites during all eight post-fire years, suggesting that wildfire had a biologically meaningful effect on CSO occupancy. Furthermore, a~~A statistically significant negative effect on occupancy was detected~~, particularly for pairs,~~ when more than 125 acres of forest within core areas burned at high severity.

~~Finally, t~~Two studies found that wildfires had ~~little~~ positive or no effect on spotted owl occupancy (Bond et al. 2002, Lee et al. 2012). Bond et al. (2002) calculated site fidelity for spotted owls in 11 territories burned by wildfires in California, Arizona, and New Mexico. The fires burned most of the area within each estimated territory (territory size = ½ the nearest neighbor distance in each study area, based on previous studies). Half of the eight territories for which fire severity was mapped primarily burned at low to moderate severity and the other half experienced extensive severe fire (36-88%). Of 21 color-banded owls in the study, 18 (86%) were resighted the year after the fires and 16 (89%) of these were located in their pre-fire territory. Site fidelity in this study was comparable to that in other, longer-term studies of the three subspecies (Seamans et al. 1999, Franklin et al. 2000, W.S. LaHaye unpubl. data).

In the Sierra Nevada, Lee et al. (2012) compared post-fire occupancy in 41 recently burned and 145 unburned historical CSO territories. An average of 32% of forest in burned territories experienced high severity fire. The authors found no significant association between CSO occupancy and whether or not territories had recently experienced wildfire within a 494-acre

circle around activity centers (but mean occupancy was higher in burned sites: 0.76 at unburned sites and 0.80 at burned sites). ~~Of the nine sites in which at least 50% of forest was severely burned within larger 988-acre areas around activity centers, eight were surveyed post-fire and CSOs were detected in five of those (63%). This level of occupancy appears to have been lower than that of burned sites as a whole (not statistically analyzed), but it also indicates that CSOs can persist in areas with extensive severe fire.~~

The studies reviewed above are not directly comparable due to differences in methods, spotted owl subspecies and populations, fire extents and severities, and the presence or absence of post-fire salvage logging. ~~Nonetheless, the weight of currently available evidence indicates that, while spotted owls can persist in burned landscapes, wildfires may often reduce occupancy.~~Current data (weighted by sample sizes, statistical power, and confounding effects of salvage logging) indicate that wildfire does not significantly affect occupancy except rarely, in the most extreme situations, when most suitable habitat in the nest core area is severely burned. ~~The studies further suggest that the magnitude of wildfire effects on occupancy depends on the extent of severe fire. However, even mixed severity wildfires may substantially reduce occupancy when suitable habitat is already limited (e.g., due to intensive timber harvesting: Clark et al. 2013)~~. Post-fire salvage logging can increase the negative effects of extensive severe fire on spotted owl occupancy; most likely by reducing suitability of burned areas for prey and foraging (see below).

Home Range Size and Habitat Use

Changes in the behavior of individual spotted owls may provide insight into the mechanisms by which wildfires affect populations. For example, post-fire changes in home range size may reflect fire effects on spotted owl energy budgets through changes in travel distances and prey availability. Changes in energy intake and output could, in turn, influence survival, reproduction, and occupancy of spotted owls. Patterns of habitat use may also be informative. For example, selection or avoidance of burned areas may reflect changes to availabilities of prey or roosting habitat, which could, in turn, influence occupancy, reproduction, or survival of spotted owls.

To our knowledge, only two studies have evaluated spotted owl home range sizes in relation to wildfires (Clark 2007, Bond et al. 2013). Clark (2007) found that annual home range sizes of NSOs inside two fire perimeters in the Oregon Klamath Province were larger after the fires than before them (n = 14 owls pre-fire and 20 post-fire). He attributed this difference to owls expanding their home ranges in response to habitat fragmentation caused by severe fire and salvage logging. This hypothesis was supported by other research in the region, which found that NSOs had larger home ranges in fragmented forests than in areas with larger, more intact patches of habitat (Carey et al. 1992, Schilling et al. 2013). Another study in the region suggested that the energetic cost of increased travel in fragmented forest was greater than the energetic benefit of increased access to prey associated with early-successional habitats (Carey and Peeler 1995).

Comment [M29]: Whenever you mention this Clark 2007study you must call any affects due to"fire and salvage logging" because they did not discriminate between these thing, but lumped all together.

Bond et al. (2013) compared the breeding season home ranges of seven CSOs (from four territories) during a single post-fire year in the Sierra Nevada of California with those in other studies during the same year in other parts of the subspecies range (D. Call, T. Munton, and G. Zimmerman unpubl. data). An average of 23% of forest burned at moderate severity and 9% at high severity within a 1.2 mile radius of the four nests. Pre-fire home range sizes were unknown but CSOs in the four territories did not appear to have unusually large home ranges following predominantly low to moderate severity wildfire.

Three studies have described patterns of habitat use by spotted owls in burned areas (Table A). King et al. (1997; also Bevis et al. 1997) described initial effects of wildfires on NSOs in two territories in the eastern Washington Cascades. One territory primarily experienced low to moderate severity fire and the other mostly burned at high severity. Both territories experienced an unreported amount of salvage logging in unsuitable or severely burned habitat. Most NSO locations (84% and 89%) in the two territories were daytime roosts. In the territory primarily burned at low to moderate severity, 80% of the pair's post-fire locations were in unburned habitat, 16% were in low severity burns, and 4% were in moderate severity burns. The pair did not appear to use severely burned areas. The second territory studied by King et al. (1997) was occupied by a single male. After the fire, the male shifted his activity to an unburned area two to three miles away but continued to occasionally use areas near his former activity center. Of those locations, 74% were in unburned habitat, 17% were in low severity burns, 5% were in moderate severity burns, and 4% were in high severity burns. Maps of burn severity classes and NSO locations indicate that owls in these two territories strongly selected unburned areas for roosting.

Clark (2007) evaluated habitat selection by 12 NSOs (7 territories) inside a wildfire perimeter in the Oregon Klamath Province. NSO locations were primarily nocturnal and may therefore, have largely represented foraging activity. Individuals in this area used all habitat classes, including moderate and severe burns and areas that had been salvage logged. However, when the data from individuals were pooled for analysis, the owls exhibited a strong preference for nesting-roosting habitat that was unburned or burned at low severity (unburned and low severity were combined into a single class). NSOs in the study also selectively used moderately burned, previously-suitable habitat; although both use and availability of this habitat class were low compared with unburned or lightly burned habitat. Clark (2007) speculated that selection of moderately burned habitat was related to increased prey availability following fire in those areas. Owls' use of burned areas was concentrated closer to activity centers, which was expected, given that spotted owls are central place foragers during the breeding season (Rosenberg and McKelvey 1999).

In the Sierra Nevada, California, Bond et al. (2009) described the habitat associations of seven CSOs from four territories during a single post-fire season. Of the four nests found during the study, one was approximately 0.3 mile outside the fire perimeter, one was in forest burned at low severity, and two were in forest burned at moderate severity. One of the two nest trees found in a moderate severity burn was apparently killed by the fire and one produced the only fledgling detected during the study. It is unclear from the paper whether these events occurred at the same nest or different nests. The four pairs roosted in all burn severity classes but exhibited statistically significant selection of low severity burns and avoidance of moderate and high

severity burns. Only one of 60 roost sites was located in a high severity burn. Burned roost sites generally resembled unburned roost sites (>60% canopy cover and large-diameter trees). Bond et al. (2009) also evaluated CSO selection of foraging habitat in the area. Probability of use for foraging was highest when sites were burned and within 0.6 mile of nests or roosts. Probability of use was also positively associated with presence of edge between burn severity classes. Five of the owls foraged in high severity burns within 0.9 mile of nests or roosts more often than in other burn severity classes. Bond et al. (2009) suggested that CSOs in these four territories selectively foraged in high severity burns in order to access abundant prey in those areas. This hypothesis was supported by their finding that high severity burns had the highest herb and shrub cover and highest basal area of snags of any burn severity class, including unburned. These features are key resources for spotted owl prey communities (Carraway and Verts 1991, Carey et al. 1999, Holloway and Smith 2011).

The limited available information concerning spotted owl habitat use following wildfires indicates that the species ~~strongly~~ avoids roosting in moderate and high severity burns. This is unsurprising, given the spotted owl's close association with densely-canopied older forest for roosting (*section_xxx*). Little is known about the effects of wildfire on selection of nest sites. Bond et al. (2009) found three CSO nests in forest recently burned at low and moderate severity, and young fledged from one nest in a moderate severity burn. Moderate severity fire killed one of the four CSO nest trees in their study. We are unaware of any reports of spotted owls nesting in severely burned areas. Because of the species' nesting habitat requirements (*section_xxx*), it is likely uncommon. Two studies examined selection of foraging habitat by spotted owls. Both found use of all burn severity classes, but Clark (2007) found a preference for foraging in

Comment [M30]: Always confounded fire and salvage logging. Don't assume or report that the effects were from fire! Maybe move all mention of Clark 2007 to salvage logging section.

unburned to moderately burned older forest while Bond et al. (2009) found a preference for severe burns. It is unclear if this difference was due to differences in the studies' methods, spotted owl diets, or effects of fire and timber harvesting on vegetation.

Salvage Logging

Salvage logging further modifies recently burned forests and could exacerbate negative effects of severe fire on spotted owls. While salvage logging might be judiciously used to meet certain conservation objectives (e.g., generating downed wood to minimize erosion or create wildlife habitat), it is generally conducted to meet financial goals or remove hazard trees (Peterson et al. 2009). Intensive or poorly planned salvage logging can have a variety of negative effects on ecosystems, such as soil compaction, increased erosion, and impacts on insectivorous and cavity-nesting and -denning animals (reviewed in McIver and Starr 2000, Noss et al. 2006, Peterson et al. 2009).

We know of three studies that have directly evaluated effects of post-fire salvage logging on spotted owls (Clark 2007 and Clark et al. 2013, Lee et al. 2012, 2013). Clark (2007) conducted a radio-telemetry study in areas recently burned by wildfires in the Oregon Klamath Province. He recorded limited use of salvage logged areas; presumably for foraging since locations were primarily nocturnal. Use of salvage logged areas was slightly lower than expected based on its abundance in territories (not statistically analyzed), indicating weak avoidance of salvage logged areas by NSOs. However, avoidance might have been stronger since some of the study's telemetry locations were potentially recorded prior to the occurrence of salvage logging. Most

Comment [M31]: These papers confound fire and salvage.

(60%) NSO locations in salvage logged areas occurred in riparian buffers, thinned areas, and patches of wildlife leave trees, rather than intensively salvaged areas. During the same study, Clark et al. (2013) found that post-wildfire declines in NSO occupancy were best explained by a model that included extents of pre-fire timber harvesting, severe fire, and post-fire salvage logging. Models that included these factors separately were not competitive with this model, indicating that severe fire and pre- and post-fire harvesting collectively contributed to declines in NSO occupancy; most likely through cumulative habitat loss or degradation.

In the Sierra Nevada, Lee et al. (2012) recorded occupancy for eight CSO territories that experienced wildfire and post-fire salvage logging. Seven of the territories were occupied during the two-year period between the occurrence of wildfire and salvage logging, whereas none of the territories were occupied following salvage logging.

Lee et al. (2013) evaluated effects of salvage logging on CSOs in the San Bernardino and San Jacinto Mountains of Southern California. They noted that salvage logging in their study area was modest compared with commercial salvage logging typically employed in the Pacific Northwest and Sierra Nevada (salvage logging in their study area mostly consisted of firewood cutting on private in-holdings and hazard tree removal along Forest Service roads). Lee et al. (2013) did not find a statistically significant effect of post-fire salvage logging on CSO occupancy dynamics. However, site extinction probability was slightly higher, and mean annual probability of occupancy was slightly lower, in salvage logged areas than in other burned areas. Weak negative effects of light salvage logging were evident during all eight post-fire study years, suggesting that they were biologically, if not statistically, significant.

The limited available evidence suggests that salvage logging increases the probability that spotted owls will abandon their territories following wildfires. This likely occurs because salvage logging reduces suitability of burned areas for foraging spotted owls and their prey. Stands recently burned by moderate or severe fire often contain high biodiversity due to the presence of both early-successional conditions and key biological legacies in the form of snags, logs, and live trees (Noss et al. 2006). Due to fire suppression and salvage logging, stands with these conditions are currently rare in many fire-prone forests within the spotted owl's range (Noss et al. 2006). Selective use of moderate or high severity burns for foraging is likely due to spotted owls exploiting short-term increases in prey associated with both early-successional vegetation (e.g., shrubs) and legacy habitat elements (e.g., large diameter snags, logs, and live trees) (Ream 1981, Zwolak 2009, Bond et al. 2013). Salvage logging removes legacy elements, while associated use of herbicides reduces shrubs and grasses important to many prey species (Bond et al. 2013). In the longer-term, spotted owls can continue to benefit from the contributions of legacy habitat elements to regenerating stands. For example, large legacy snags, trees, and logs can provide valuable habitat elements for northern flying squirrels and other prey (Holloway and Smith 2011). Removal of these elements through salvage logging could therefore reduce the value of subsequent regenerating stands as prey habitat. Harvesting of legacy snags and live trees could also directly affect spotted owls by reducing availability of foraging perches in the short-term and suitable nest trees during later successional stages.

<u>Summary</u>

Research of direct effects of wildfires and salvage logging on spotted owls have been opportunistic and have therefore, lacked the ability to compare pre- and post-fire data. Most studies' inferences are also limited due to inclusion of only a small number of spotted owls or territories. Nonetheless, more information is available concerning this topic than is generally acknowledged. The preponderance of currently available evidence indicates that spotted owls are often resilient to wildfires but can be strongly impacted by extensive severe fire. Following wildfire, many spotted owls may remain in their territories, exploit short-term increases in prey in burned areas, and continue to reproduce at reasonably high rates. However, wildfires that result in substantial loss or fragmentation of habitat can cause spotted owls to increase their home range sizes, abandon their territories, and possibly, die of starvation or disease. Negative effects of severe fire appear to be greatest when suitable habitat is already limited (e.g., due to widespread intensive timber harvesting) and when post-fire salvage logging reduces suitability of burned areas for foraging and prey.

Comment [M32]: Lee et al 2012 and 2013 do look pre- and post-fire!

Wildfire Effects on Recent Habitat Trends

Past and continuing habitat loss was a primary reason for listing the NSO under the federal Endangered Species Act (USFWS 1990). At the time of listing, the Fish and Wildlife Service estimated that 60-88% of the subspecies' habitat had already been lost (USFWS 1990; also see *Potential Threats: Timber Harvesting*). They attributed most of this loss to timber harvesting and land conversion but also acknowledged the roles of wildfire and other natural disturbances. Since listing of the NSO and subsequent adoption of the Northwest Forest Plan, timber harvesting has declined and wildfire has been identified as the primary source of forest

Comment [M33]: How is habitat lost? Define how suitable habitat is defined, and how it is lost.

disturbance and habitat loss on federal lands (Courtney et al. 2004, USFWS 2011, Kennedy et al. 2012).

Estimates of recent trends in amounts of NSO habitat, and of older forest in general, have been produced as part of monitoring efforts for the Northwest Forest Plan; and are therefore, largely restricted to the Plan's area and time span (Davis and Lint 2005, Davis and Dugger 2011, Moeur et al. 2005, 2011). We focus on estimates by Davis and Dugger (2011) because they replaced those of Davis and Lint (2005) and are more specific to NSO habitat than those of Moeur et al. (2005, 2011). We do not review habitat trend estimates based on federal ESA Section 7 consultation records (Bigley and Franklin 2004, USFWS 2012) due to greater scientific uncertainty and methodological bias associated with those data (see Bigley and Franklin 2004). Trends described by Bigley and Franklin (2004), Moeur et al. (2005, 2011), and USFWS (2012) quantitatively differ from those of Davis and Dugger (2011) but similarly indicate that wildfires have been the primary source of recent habitat loss on federal lands (see below).

Comment [M34]: There is a problem of bias in how suitable habitat is defined, and the criteria for 'loss' or 'degraded'.

Davis and Dugger (2011) used remotely sensed (satellite imagery) vegetation data to model changes in habitat suitability across the NSO's range during the first 15 years of the Northwest Forest Plan (1994-2007 in California and 1996-2006 in Oregon and Washington). Habitat suitability was based on characteristics surrounding thousands of NSO pair locations across the Plan area. Suitable breeding habitat was defined as having both a probability of owl presence greater than expected based on random chance and environmental conditions typical of those found around nesting and roosting pairs. Estimated habitat trends included gross loss of both suitable breeding habitat and "core" suitable breeding habitat (>330 ft from edge). Davis and

Comment [M35]: But rarely include burned sites, so definition of 'suitable habitat' is biased against burned forest.

Dugger (2011) considered habitat loss to have occurred when an area classified as suitable at the beginning of Northwest Forest Plan was later downgraded to a lower suitability rank (unsuitable or marginal) due to vegetation changes caused by forest disturbances. Davis and Dugger (2011) did not estimate recruitment of, or net changes in, breeding habitat because their remotely sensed data was incapable of accurately capturing relatively slow and subtle habitat changes during development of intermediate-aged and older stands. They did, however, estimate net trends in NSO dispersal habitat, which they defined as forest with a mean canopy cover of at least 40% and a mean DBH of at least 11 inches. Recruitment of dispersal habitat was more detectable than that of breeding habitat due to more rapid and measurable growth in younger forest (some recruitment also occurred due to degradation of suitable breeding habitat brought about by forest disturbances).

Estimated gross losses of suitable breeding habitat on federal and non-federal lands are presented in Tables B and C, respectively. During the first 15 years of the Northwest Forest Plan, wildfires were responsible for an estimated gross loss of 236,700 acres (2.7%) of suitable breeding habitat on federal lands rangewide and 13,100 acres (0.3%) on non-federal lands (1.9% combined). In California, wildfires removed an estimated 75,500 acres (4.1%) of suitable breeding habitat on federal lands and 5,600 acres (0.4%) on non-federal lands (2.4% combined). Approximately 70% of habitat loss to wildfire on federal lands occurred within the Oregon and California Klamath Provinces (Table B). Most of this habitat loss was caused by the1999 Megram Fire and 2002 Biscuit Fire (Table D). Fires in the Eastern Cascades Provinces of Washington, Oregon, and California contributed less to total habitat loss than did fires in the Klamath, but were often more destructive in terms of proportion of suitable habitat lost within individual fire perimeters.

In contrast with federal lands, wildfires were responsible for very little habitat loss on non-federal lands; rather, timber harvesting accounted for most losses in these areas (Figure A).

Table B: Estimated gross losses of suitable NSO breeding habitat on federal lands due to wildfires during 1994-2007 in California and 1996-2006 in Oregon and Washington (adapted from Davis and Dugger 2011, USFWS 2011).

State	Province	Initial Acres	Acres Lost	Percent Lost
California	CA Cascades	213,200	1,800	0.8%
	CA Klamath	1,489,800	71,600	4.8%
	CA Coast	145,400	2,100	1.4%
Oregon	OR Coast Range	611,200	0	0.0%
	Western OR Cascades	2,258,700	28,900	1.3%
	Eastern OR Cascades	402,900	17,800	4.4%
	Willamette Valley	3,400	0	0.0%
	OR Klamath	985,000	93,600	9.5%
Washington	Olympic Peninsula	763,100	200	0.0%
	Eastern WA Cascades	673,600	20,000	3.0%
	Western WA Cascades	1,283,000	700	0.1%
	Western WA Lowlands	24,700	0	0.0%
Rangewide Total		8,854,000	236,700	2.7%

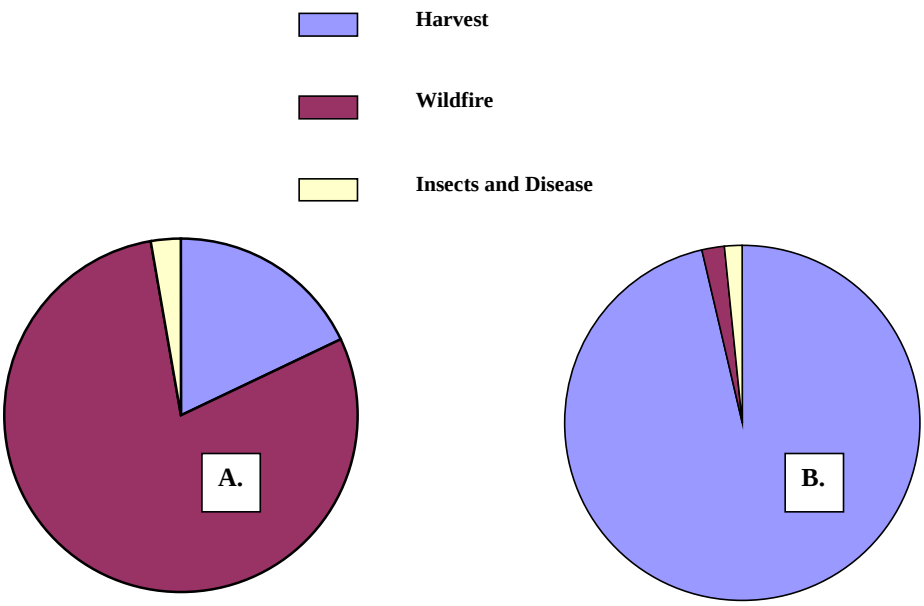
Table C: Estimated gross losses of suitable NSO breeding habitat on non-federal lands due to wildfires during 1994-2007 in California and 1996-2006 in Oregon and Washington (adapted from Davis and Dugger 2011, USFWS 2011).

State	Initial Acres	Acres Lost	Percent Lost
California	1,556,700	5,600	0.4%
Oregon	1,382,400	5,100	0.4%
Washington	1,258,900	2,400	0.2%
Total	4,198,000	13,100	0.3%

Table D: Estimated gross losses of suitable NSO breeding habitat to individual wildfires during 1994-2007 (note: "habitat degraded" describes areas downgraded from highly suitable to suitable; from Davis and Dugger 2011).

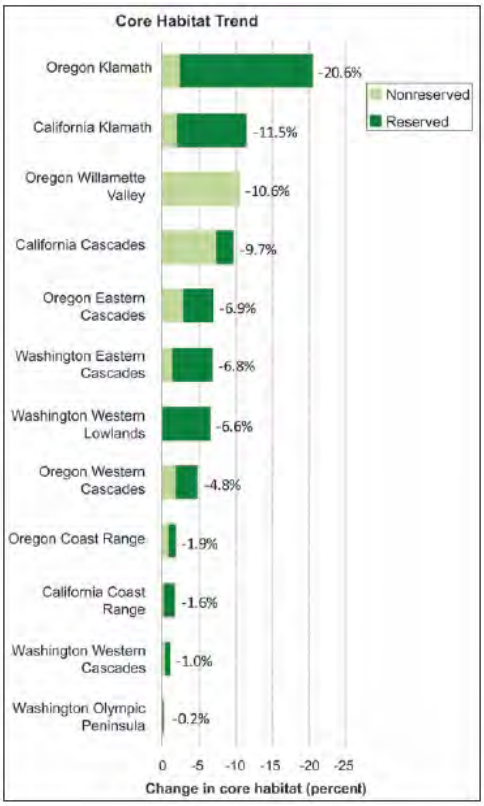
Fire name	Year	Broad physiographic province	Habitat in wildfire perimeter	Habitat lost		Habitat degraded	
			Acres	*Acres*	*%*	*Acres*	*%*
Biscuit Fire	2002	Klamath Mountains	226,230	93,730	41	12,019	5
Megram Fire	1999	Klamath Mountains	76,337	27,520	36	4,589	6
B&B Complex	2003	East Cascades	26,269	16,403	62	907	3
Bake-oven Fire	2006	Klamath Mountains	23,946	8,873	37	581	2
Boulder Fire	2002	West Cascades	34,059	8,460	25	2,074	6
Davis Fire	2003	East Cascades	8,050	6,943	86	5	0
Pigeon Fire	2006	Klamath Mountains	13,896	5,634	41	327	2
Rex Complex	2001	East Cascades	8,548	4,750	56	278	3
Timbered Rock	2002	West Cascades	10,216	4,539	44	569	6
Spring Fire	1996	West Cascades	13,504	3,931	29	858	6
Deep Harbor Fire	2004	East Cascades	5,761	3,930	68	64	1
Hancock Fire	2006	Klamath Mountains	12,712	3,132	25	336	3
Apple Fire 2	2002	West Cascades	12,227	2,810	23	928	8
Fischer Fire	2004	East Cascades	4,479	2,340	52	34	1
Fork Fire	1996	Klamath Mountains	2,962	2,113	71	14	0
Needles Fire	2003	East Cascades	1,946	874	45	1	0
Trough Fire	2001	Klamath Mountains	1,851	798	43	4	0
Hunter Fire	2006	Klamath Mountains	2,236	789	35	40	2
Deer Point Fire	2002	East Cascades	505	380	75	0	0
Tatoosh Complex	2006	East Cascades	666	378	57	0	0

Figure A: Proportions of suitable breeding habitat loss attributed to harvesting, wildfire, and insects and diseases on (A) federal lands and (B) non-federal lands during 1994-2007 (adapted from Davis and Dugger 2011, USFWS 2011).



Davis and Dugger (2011) reported substantial losses of core breeding habitat on federal lands during the first 15 years of the Northwest Forest Plan (Figure B). These losses primarily occurred in reserved areas. Changes in ratios of core and edge habitat classes indicated that increased fragmentation of suitable breeding habitat on federal lands was greatest in the Oregon and California Klamath Provinces and California Cascades Province (see Table 3-3 in Davis and Dugger 2011). Increased fragmentation in these regions was primarily due to wildfires.

Figure B: Gross losses of "core" suitable breeding habitat on reserved and non-reserved federal lands during 1994-2007 in California and 1996-2006 in Oregon and Washington (from Davis and Dugger 2011).



Davis and Dugger (2011) estimated a 5.2% net gain in NSO dispersal habitat. Much of this gain was due to succession in young forests in non-reserved lands at the margins of federal forests. However, accounting for forest connectivity and NSO dispersal distances, Davis and Dugger (2011) reported a 1% net loss of "dispersal-capable landscape". Much of the loss of dispersal-capable landscape in the Klamath and Eastern Cascades Provinces was due to wildfires, whereas timber harvesting on non-federal lands was responsible for much of the loss in other regions.

Loss and fragmentation of breeding habitat to wildfires was likely at least partially offset by recruitment of new habitat through succession of mature and old forest. However, Davis and Dugger (2011) found that detectable recruitment of breeding habitat primarily occurred in the marginal suitability class. This finding was supported by Moeur et al. (2005), who found that about 90% of recruitment of older forest (mature and old-growth combined) during the first 10 years of the Northwest Forest Plan was at the lower end of the class' diameter range (i.e., mean DBH 20-30 in). It is likely that some newly recruited mature forest provides suitable habitat for NSOs but much if it could lack the canopy layering, large diameter snags and logs, and other structural attributes typical of nesting and roosting habitat (*section_xxx*).

Loss and fragmentation of suitable breeding habitat does not necessarily equate to negative impacts on NSOs (Franklin and Gutiérrez 2002; e.g., Bond et al. 2002, 2009, Lee et al. 2012). Studies in southern Oregon and northern California found that the presence and fitness of NSOs are generally highest in landscapes with a mix of both large amounts of suitable breeding habitat and other habitat classes, such as foraging habitat or "non-habitat" (Franklin et al. 2000, Zabel et al. 2003, Olson et al. 2004, Dugger et al. 2005; reviewed in *section_xxx*). Fitness is also

generally highest when suitable breeding habitat occurs in large or clustered patches with large amounts of ecotone or edge (Franklin et al. 2000, Dugger et al. 2005). This combination of conditions is likely important because it provides NSOs with a balance of resources needed for survival and reproduction (Franklin et al. 2000). Active fire regimes in dry forests within the NSO's range in California historically contributed to these conditions by generally sparing older forest and maintaining some form of habitat heterogeneity at both stand and landscape scales (Sugihara et al. 2006). Some contemporary wildfires may still burn in this manner and thereby continue to perform an important ecosystem function in these forests. However, large severe wildfires have contributed, along with fire suppression and timber harvesting, to homogenization of some dry forests within the NSO's range (Skinner et al. 2006). Thus, large severe fires may impact NSOs through loss of habitat heterogeneity, as well as reduced amounts and connectivity of suitable breeding habitat.

Fire Risk in California

Prior to Euro-American settlement, dry forests in the western U.S. generally experienced relatively frequent, low-to-moderate or mixed severity fire regimes (Agee 1993, Sugihara et al. 2006, Van de Water and Safford 2011). Mean pre-settlement fire return intervals in California were 11 years in yellow pine forests (e.g., *Pinus ponderosa*, *P. jeffreyi*), 11-16 years in mixed-conifer forests, and 29 years in mixed-evergreen forests (Van de Water and Safford 2011). Mean fire return intervals in redwood forests south of Del Norte County were also relatively frequent (6-44 yrs: Stuart and Stephens 2006). Frequent fire during the pre-settlement period generally maintained forests with less dense and more clumped tree distributions, higher proportions of

fire-resistant trees (i.e., larger size classes and more fire-tolerant species), and lighter and less continuous fuel beds than occur today (Sugihara et al. 2006). This characteristic fine-scale structural heterogeneity was often overlaid with coarser variability created by effects of elevation, terrain, soils, and other physiographic factors on fire and vegetation patterns (Sawyer 2007). For example, in areas of deeply incised terrain in the Klamath Mountains and southern Cascades of California, upper slopes and south and west facing aspects typically experienced more frequent and severe fire than did other areas (reviewed in Skinner and Taylor 2006, Skinner et al. 2006). Due to fire suppression, early-successional vegetation communities formerly maintained by frequent, small-scale severe fire have greatly declined in some areas of California (Skinner 1995, Nagel and Taylor 2005). Overall, research indicates that fire suppression and other human activities have led to decreased forest heterogeneity at both stand and landscape scales and have contributed to substantial changes in fire regimes in California's dry forests (Sugihara et al. 2006).

Due to fire suppression, fire-free periods have dramatically increased in California's dry forests (Sugihara et al. 2006). For example, fire rotation near Hayfork in the Klamath Mountains increased more than 10-fold (from 19 to 238 yrs) during the post-settlement period (Taylor and Skinner 2003). Abnormally long fire-free periods have facilitated increased accumulation and continuity of fuels in dry forests (Sugihara et al. 2006). When wildfires do occur there is often an increased risk of them becoming very large and for suppression forces to be overwhelmed by their size and number (e.g., CAL FIRE 2008). As in other dry forests across the western U.S., the mean and maximum sizes of wildfires, and the total annual area burned, significantly increased in California's dry montane forests during the 20th and early 21st Centuries (Miller et

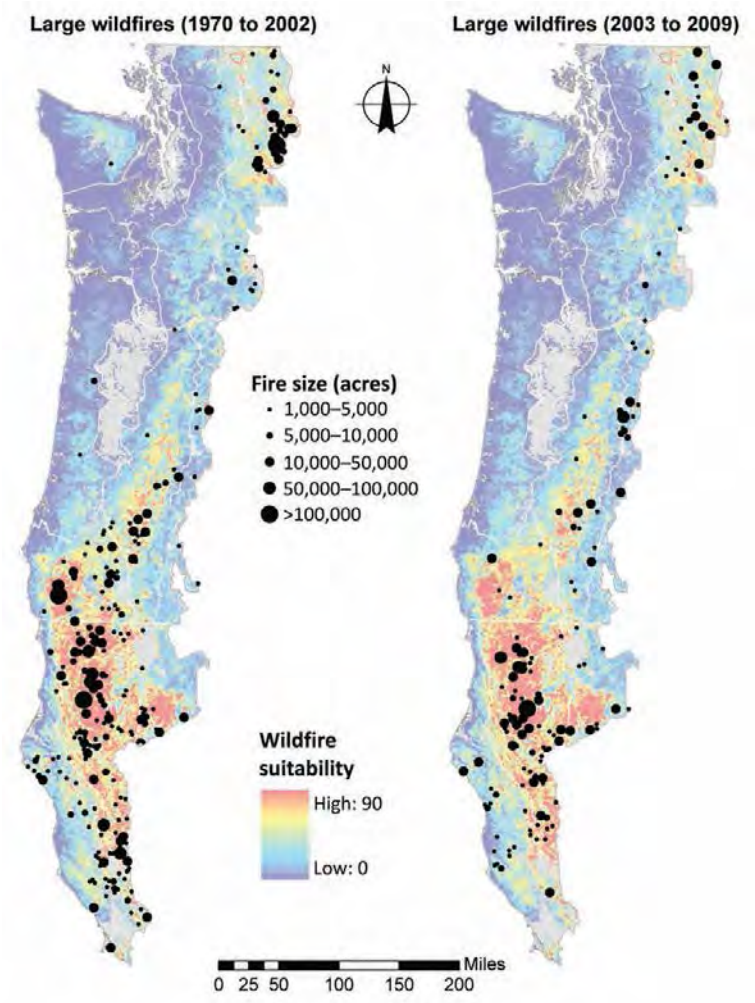
al. 2009, 2012). CAL FIRE (2008) noted that more than half of the 26 largest fires recorded in California during 1932-2008 occurred during the last eight years of that period. Based on recent (1970-2002) fire behavior, the majority of northern California's interior can be classified as highly prone to large (>1,000 ac) wildfires (Davis et al. 2011; Figure C).

Studies of recent trends in extents of high severity fire in California have found conflicting results. For example, Miller et al. (2009) reported a substantial increase in the extent of high severity fire in the Sierra Nevada and southern Cascades of California during 1984-2006; while Hanson and Odion (2014) did not find an increase in the Sierra Nevada during nearly the same period (1984-2010). Similarly conflicting results have been found for the California Klamath Province. Hanson et al. (2009) found a significant increase in high severity fire in the California and Oregon Klamath Provinces during 1984-2005; whereas Miller et al. (2012) did not find an increase in the California Klamath Province during a similar period (1987-2008). Scientific debate ensued regarding the appropriateness of methods used to determine fire severity (e.g., Hanson et al. 2009, 2010 vs. Spies et al. 2010).

Differences in findings regarding trends in high severity fire are related to variation in studies' temporal and spatial scales of analysis, as well as in methods for determining fire severity (Courtney et al. 2008, Miller et al. 2012, Hanson and Odion 2014). For example, Miller et al. (2012) noted that both their own study and those of Odion et al. (2004, 2010) may have underestimated trends in high severity fire in the California Klamath Province due to inclusion of unusual fire years. Studies by Odion et al. (2004, 2010) were based on fire effects during a single year (1987), which Miller et al. (2012) described as unusual. Large areas burned at

below-average severity during 1987 due to abnormally strong inversions, and the fact that some of the fires burned well into fall (when conditions often favor lower severity fire). Miller et al. (2012) also noted that their own ability to detect a trend in fire severity could have been compromised by inclusion of both this year and 2008, which likewise experienced unusually large, low severity fires. Differences in the area analyzed could also affect evaluations of trends in high severity fire. For example, the Biscuit Fire, which was predominantly located in Oregon, included extensive areas of high severity fire and therefore could have influenced results of trends analyses for the Oregon and California Klamath Provinces combined versus the California Klamath alone (Miller et al. 2012).

Figure C: Relative probability of large (>1,000 ac) wildfires ("wildfire suitability") across the NSO's range (from Davis et al. 2011). Modeling was based on landscape and climatic characteristics of locations at which large wildfires occurred during 1970-2002 (left) and was compared with subsequent (2003-2009) locations (right).



Regardless of whether or not the extent of high severity fire increased in California during the last two decades, large severe fires have recently occurred in these areas, and they were responsible for most loss, degradation, and fragmentation of suitable breeding habitat for NSOs on federal lands (Davis and Dugger 2011, Davis et al. 2011; see above). These data cannot be used to project how fires in the future will affect NSOs since they do not necessarily represent past or future fire conditions or effects. However, it is highly unlikely that wildfires will cease to be a major source of habitat loss for NSOs in the future. Rather, climate change research has generally projected a continued increase in the number and sizes of wildfires and the annual area burned during coming decades (see *Potential Threats: Climate Change*). There is scientific uncertainty regarding recent and future trends in the extent of high severity fire in California. Nonetheless, large severe fires will at least occasionally occur in the future and will therefore continue to be a source of habitat loss and modification for NSOs in the state.

Comment [M36]: These aren't peer reviewed. More accurately USFS biologists are responsible for loss and degradation when they redraw PACS and re-designate habitat post fire.

Increases in the number and sizes of wildfires, and effects of fires on NSO habitat, have led to calls for widespread use of thinning and other forms of active management in dry forests (USFWS 2008, 2011, 2012a, Franklin and Johnson 2012). Some researchers and stakeholders, however, have expressed doubts regarding estimates of fire risk and effects on NSOs; concerns about potential effects of thinning on NSOs; and distrust of federal agency intentions (Hanson et al. 2009, Heiken 2010, DellaSala et al. 2013). As discussed in *Potential Threats: Timber Harvesting* (also Hansen and Mazurek 2010, USFWS 2011), there is currently little known about the effects of forest thinning on spotted owls. The available information suggests that low intensity thinning and prescribed fire could be judiciously used to reduce fire risk or restore habitat for NSOs. In contrast, poorly planned thinning could have unintended consequences for

Comment [M37]: This is wrong and biased. Where's Seamans and Gutierrez 2007, Williams et al. 2011. Whre is your citation for how thinning reduced fire risk and 'restores' habitat?

the subspecies. If thinning is used in landscaped occupied by NSOs, it should generally be limited to strategic locations outside of owl core areas and its effects should be monitored within an adaptive management framework. Land managers should also consider allowing more wildfires to burn under prescribed conditions; particularly at lower elevations in the California Klamath Province, where summertime inversions often minimize fire severity (Miller et al. 2012).

Comment [M38]: Where is your evidence that thinning affects fire behavior? This is an enormous leap in logic

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Keeping Northwest California wild since 1977

Sent to: fgc@fgc.ca.gov on date shown below

May 11, 2015

Mr. Jack Bayless, President
California Fish and Game Commission
1416 Ninth Street, room 1320
Sacramento, CA 95814

RE: California Department of Fish and Wildlife Status Report for the Northern Spotted Owl (*Strix occidentalis caurina*)

Dear President Bayless and Commissioners:

This letter is submitted on behalf of the Environmental Protection Information Center (EPIC), regarding the status of the California Department of Fish and Wildlife's (CDFW or Department) Status Report for the Northern Spotted Owl (*Strix occidentalis caurina*) (NSO).

In 2012, EPIC, concerned with the long-term viability of the NSO in California, submitted a listing petition under the California Endangered Species Act to the California Fish and Game Commission (Commission). On August 7, 2013 the Commission voted to advance the NSO to candidacy status. The Commission adopted findings for this decision on December 11, 2013, thereby beginning the Department's obligation to conduct a Status Report within 12 months to aid the Commission in making its final determination. On October 9, 2014, the Commission granted the Department a six month extension to complete the Status Report pursuant to Cal. Fish & Game Code § 2074.6(a).

It has come to our attention that the Department does not intend to submit its Status Report for the NSO on June 26, 2015, as was agreed by the Commission. In a personal communication with Ms. Carie Battistone at the Department's Wildlife Branch dated May 11, 2015, it was relayed to us that the Department does not intend to submit its Status Report for the NSO at the time agreed pursuant to the six-month extension granted by the Commission. Ms. Battistone imparted to us that the Department was facing "circumstances" that have caused delays in the production of its Status Report for the NSO. Ms. Battistone indicates that there is now no date-certain for submittal of the Department's Status Report for the NSO to the Commission. Further, Ms. Battisone stated that the Department hopes to submit the Report to the Commission by the end of 2015.

We wish to remind the Commission that CESA allows for a six-month extension on submittal of Status Reports. The Department has already applied for, and received a six-month extension on the production of its Status Report. We see no statutory or regulatory authority which would allow the Department to push-out the date for submittal of its Status Report for the

Environmental Protection Information Center
145 G Street Suite A Arcata, CA 95521
(707) 822-7711
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NSO indefinitely, or even beyond the six-month extension for submittal of the Report. As provided in Fish and Game Code § 2074.6(a), "The commission may grant an extension of *up to* six months *if the director determines an extension is necessary to complete independent peer review of the report*." (Emphasis added). From EPIC's understanding, CDFW is not even at the independent peer review stage, thus questioning the legality of the original deadline extension.

While we understand and appreciate that the Department is faced with various "circumstances" that may complicate its production of the NSO Status Report, there simply does not appear to be any authority to allow for further delays.

EPIC therefore requests that the Commission inquire with the Department as to the reasons for the indefinite delay of its submittal of the NSO Status Report. We request a written response from the Commission which details the Commission's findings in this regard.

Please do not hesitate to contact me at the number provided below should there be any questions regarding this matter. We look forward to the Commission's response.

Sincerely,

Rob DiPerna
California Forest and Wildlife Advocate

Environmental Protection Information Center
145 G Street, Suite A
Arcata, California 95521
Office: (707) 822-7711
Email: rob@wildcalifornia.org

Cc: Mr. Sonke Mastrup, Executive Director, California Fish and Game Commission
Mr. Charlton H. Bohnam, Director, California Department of Fish and Wildlife
Ms. Lacy Bauer, California Department of Fish and Wildlife