Chapter 5: Current and Projected Condition of Mid-Elevation Sierra Nevada Forests

Malcolm P. North, Mark W. Schwartz, Brandon M. Collins, and John J. Keane[1]

Introduction

Most of the California spotted owl's (*Strix occidentalis occidentalis*) habitat is concentrated in mid-elevation forests of the Sierra Nevada (see chapter 9 for a discussion of southern California spotted owls and their habitat), which are made up primarily of ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), mixed-conifer, white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), and mixed-evergreen forest types. These forests have undergone substantial change since the arrival of Europeans and are projected to dynamically respond to ongoing factors affecting ecosystem conditions. In this chapter, we summarize some of the historical changes in mid-elevation forests that have most extensively altered ecosystem conditions. We also explore sources and spatial distribution of the more extant changes in forest condition. We then discuss likely trends in forest response to projected stressors, particularly climate change, drought, and fire. Finally, we examine recent research and resulting changes in management practices that might affect future forest conditions in an effort to increase ecosystem resilience.

Forest Management

Management practices, including fire suppression, over the past century have largely shaped current forest conditions in the Sierra Nevada. These conditions significantly vary with land ownership because owners have different incentives and constraints that influence their management practices. We examine the three main ownerships in the Sierra Nevada, their historical management practices, and current conditions of the different forests.

Ownerships

The U.S. Forest Service (USFS) is the largest steward of public lands in the Sierra Nevada. About 2.93 million ha (7.24 million ac; 47 percent of the 6.24 million ha

[1] **Malcolm P. North** is a research forest ecologist, **Mark W. Schwartz** is a professor in the Department of Environmental Science and Policy, University of California–Davis, Davis, CA 95616, **Brandon Collins** is a research scientist at the Center for Fire Research and Outreach, University of California Berkeley, Berkeley, CA 94598, **John J. Keane** is a research wildlife ecologist U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, 1731 Research Park Dr., Davis, CA 95618.

[15.42 million ac]) in the Sierra Nevada bioregion are under USFS management (Davis and Stoms 1996). The USFS has a broad mandate of managing national forests for multiple use and providing sustainable ecosystem services (NFMA 1976). About 2.34 million ha (5.77 million ac; about 37 percent) of conifer forests in the Sierra Nevada are in private ownership (Davis and Stoms 1996), and their management is governed by California's Forest Practice Regulations, which promotes "achiev[ing] a balance between growth and harvest over time consistent with the harvesting methods within the rules of the Board, maintain functional wildlife habitat…, retain or recruit late and diverse seral stage habitat components…, and maintain growing stock, genetic diversity, and soil productivity" (CA FPR Section 897, USDA FS 2012). There are five national parks in the Sierra Nevada and southern Cascades: Devil's Postpile, Lassen, Sequoia and Kings Canyon, and Yosemite covering 696 000 ha (1.73 million ac). The National Park Service (NPS) serves as steward of these parks and is under a mandate to provide recreational opportunities for people, and to protect and showcase natural resources without exploitation.

Historical Management Practices

Until about 1990, similar management objectives and silvicultural prescriptions were used on both public and private lands (McKelvey and Johnston 1992). Therefore, we combined our synopsis of management practices during this period for both ownerships.

Early logging prior to 1900 occurred mainly near mining operations and associated communities at low elevations in the southern and central Sierra Nevada, with most logging occurring below national forest lands. Logging extended to mid and high elevations in the northern Sierra Nevada to support mining at higher elevations and lands adjacent to the Southern Pacific Railroad line (McKelvey and Johnston 1992). The Lake Tahoe and Truckee River basins were exceptions to these general patterns as they were extensively logged to support the Comstock silver mines in western Nevada. Away from railroad lines, log removal was limited to wagons and short-haul skidding with animals and steam "donkeys." Because of these transportation limitations, most logging consisted of high-grading of large and valuable trees. With improvements in transportation, timber harvest increased steadily after 1900, although it declined for about a decade during the Great Depression. Timber harvest in the Sierra Nevada peaked in the post-World War II years, and then stabilized generally ranging between about 1.3 and 1.7 million board feet (mmbf) (1960 through 1990), with a short decline during the 1980s recession (McKelvey and Johnston 1992).

Prior to the 1980s, most silvicultural prescriptions were selection harvests of commercially valuable trees, leaving those with marginal value standing. Clearcutting prescriptions were incorporated in the 1970s, and clearcuts accounted for most of the volume in the mid-1980s. In the late 1980s, most volume was harvested using salvage prescriptions, following mortality principally from fire and insect events. In general, similar harvest prescriptions tended to be implemented on public and private lands prior to 1990 (McKelvey and Johnston 1992).

A legacy of management practices during this period is a reduction in "defect" trees (Bouldin 1999). Over many decades, stand improvement practices often involved removing "defects" such as trees with broken tops, missing limbs, rot and large cavities (see Walsh and North 2012 for examples). Such defect trees typically require many years to develop and thus decades of this practice have probably resulted in a significant decline in these structures often used by wildlife for nesting, resting, and roosting habitat (Bull et al. 1997, Carey 2002, Carey et al. 1997, Cockle et al. 2011, Hunter and Bond 2001, Wiebe 2011).

McKelvey and Johnston (1992) summarized four key changes in forest conditions that occurred from 1850 through 1992: (1) the loss of old, large-diameter trees and associated large downed logs; (2) a shift in species composition toward shade-tolerant, fire-sensitive tree species (i.e., from pines to fir and cedar); (3) increases in fuel loads associated with the mortality of small-diameter trees; and (4) the presence of fuel ladders that facilitate crown fire. Further, they indicated that management direction identified in land management plans (LMPs) for Sierra Nevada national forests current at that time would likely not alleviate these concerns and trends in forest dynamics. The LMPs projected that national forest lands in the western Sierra Nevada would be converted to even-age systems using clearcut, seed tree, and shelterwood prescriptions at a rate of 91 600 ha (226,350 ac) per decade and that selection logging would occur on 32 000 ha (80,000 ac) per decade. This management direction provided no guarantees that old, large trees and their derivatives (e.g., large snags and logs) would be maintained. Rather, it suggested large proportions of future forest would trend toward areas of even-aged plantations with stands of dense, smaller diameter trees (McKelvey and Johnston 1992).

Forest Management Since 1990

Concern for the conservation of California spotted owls began in the mid-1980s with awareness first raised over the status of the related subspecies, the northern spotted owl (*Strix occidentalis caurina*). With the adoption of the California spotted owl guidelines following the California spotted owl technical assessment (CASPO) in 1992 (Verner et al. 1992), national forest and private ownership management practices significantly diverged in the mid-1990s. Timber harvest dramatically decreased on USFS lands. Overall, 83.4 percent of the timber volume harvested between 1994 and 2013 was generated from private lands with public lands contributing from 10 to 24 percent (fig. 5-1).

National Forest System lands—

About 490 000 ha (1.2 million ac) of treatments occurred on National Forest System (NFS) lands in the Sierra Nevada between 1990 and 2014 (table 5-1; see appendix p. 155 for more detailed information). The number of treated acres has declined over time, from highs of 40 000 to 48 000 ha (100,000 to 120,000 ac) per year in 1990–1992 to a low of around 8000 ha (20,000 ac) per year in 2011–2013 (table 5-1, fig. 5-2). The highest proportion of total treated acres from 1990 through 2014 occurred on the Lassen (113 966 ha [284,916 ac]) and Plumas National Forests (101 764 ha [254,411 ac]), with intermediate amounts on the Stanislaus (63 802 ha [159,804 ac]), Tahoe (62 683 ha [156,708 ac]), and Eldorado (58 098 ha [145,244 ac]) National Forests (fig. 5-3).

Concurrent with a decline in the number of acres treated has been a change in the predominant silvicultural prescriptions used on NFS lands (table 5-1, fig. 5-2). From 1990 through 1994, the predominant silvicultural prescriptions were sanitation and salvage cuts, followed by lower amounts of clearcuts and overstory removal. Adoption of CASPO guidelines in 1993 led to an increase in commercial thinning following CASPO guidelines that maintained all trees >30 in diameter at breast height (d.b.h.), maintained overstory canopy cover >40 percent, and removed

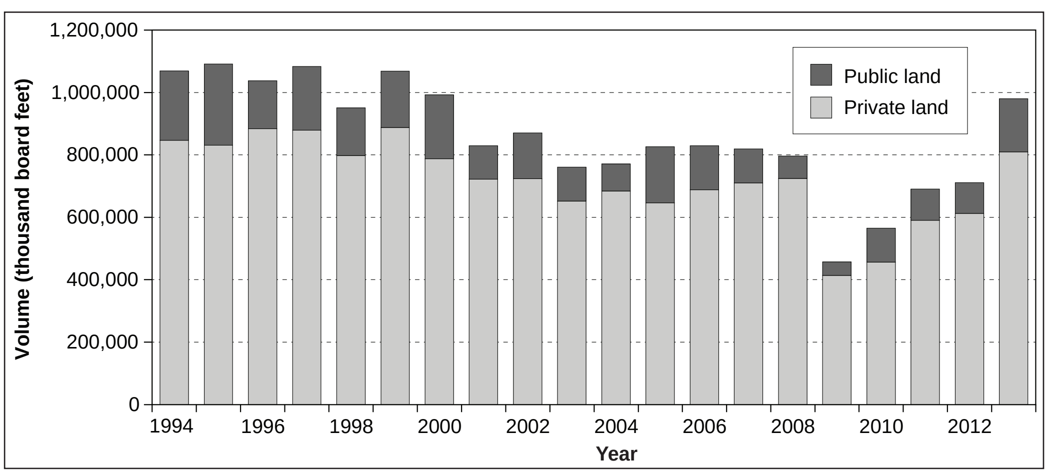

Figure 5-1—Annual timber volume harvested (thousand board feet [mmbf]) by year on public and private lands from counties in the Sierra Nevada 1994–2013. See text for further details. Source: Timber Yield Tax program, California State Board of Equalization.

Table 5-1— Treatment hectares accomplished on National Forest in the Sierra Nevada by silvicultural prescription and year during 1990–2014

Year	Clear-cut	Commercial thin	Group selection	Other	Overstory removal cut	Salvage cut	Sanita-tion cut	Seed tree cut	Single tree selection cut	Total
1990	5,390	4,113	109	134	4,023	16,052	15,017	970	1,625	47,434
1991	4,000	1,746	49	67	2,086	14,004	14,831	999	1,951	39,733
1992	3,770	855	33	16	2,121	11,714	29,853	341	929	49,631
1993	2,425	2,052	90	22	1,799	10,620	18,011	552	686	36,258
1994	2,063	1,131	36	56	1,506	3,278	8,924	283	152	17,428
1995	3,120	5,082	33	143	2,636	674	7,529	369	437	20,022
1996	7,389	7,905	148	79	582	576	9,764	511	2,581	29,535
1997	2,191	10,199	120	207	135	688	9,815	146	658	24,159
1998	689	12,573	25	142	559	2,335	12,910	74	620	29,927
1999	346	13,617	449	423	45	2,622	948	21	826	19,296
2000	31	10,940	93	929	0	183	1,728	17	248	14,169
2001	28	7,635	280	632	8	126	1,094	48	96	9,947
2002	263	8,864	400	370	43	160	256	0	330	10,686
2003	2,451	7,611	100	186	0	9,893	1,010	0	91	21,342
2004	13	13,611	273	0	0	1,414	1,402	5	1,250	17,969
2005	3	5,067	108	0	1	669	1,245	2	900	7,995
2006	1,714	12,755	410	29	3	3,648	617	0	454	19,630
2007	201	11,331	252	17	0	297	388	0	31	12,516
2008	0	3,743	26	8	0	1,047	315	0	77	5,216
2009	103	6,668	157	42	0	6,632	452	0	307	14,360
2010	238	7,546	399	97	0	1,046	176	0	320	9,822
2011	110	6,086	184	96	0	690	24	0	1	7,190
2012	0	7,924	203	44	0	216	0	0	90	8,477
2013	0	5,046	39	1	0	3,675	84	0	0	8,844
2014	194	1,931	45	3	0	5,437	0	0	91	7,701
Total	36,732	176,028	4,063	3,740	15,547	97,694	136,393	4,339	14,751	489,287

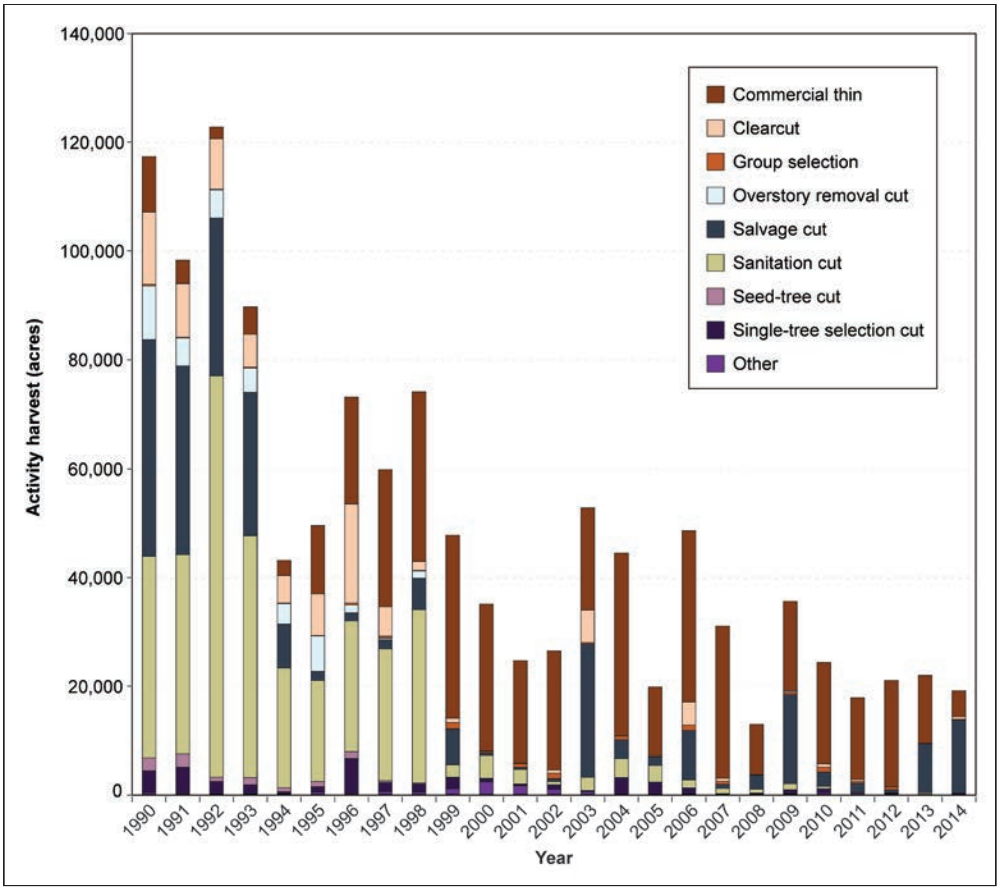

Figure 5-2—Treatment acres accomplished on national forests in the Sierra Nevada by silvicultural prescription and year, 1990–2014. Source: Taken from USFS Forest Activities Tracking System courtesy of Joe Sherlock (Pacific Southwest Research Region silviculturist).

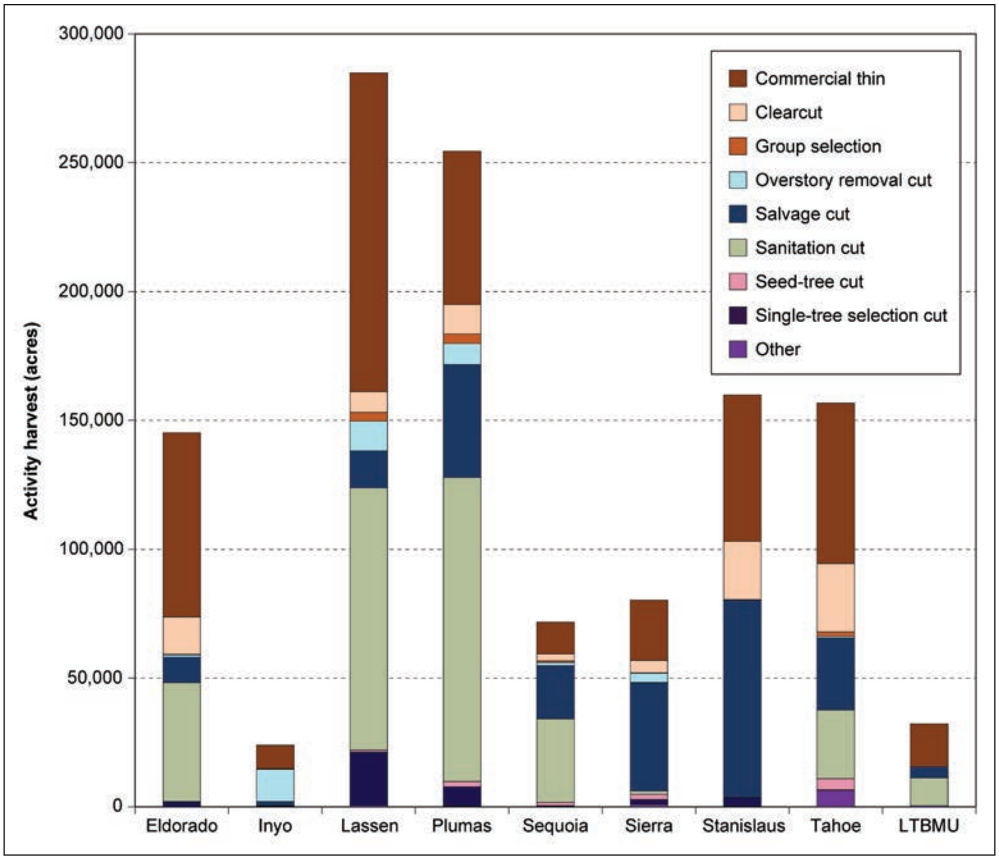

Figure 5-3—Treatment acres accomplished on national forests in the Sierra Nevada by silvicultural prescription and national forest, 1990–2014. LTBMU = Lake Tahoe Basin Management Unit. Source: Taken from USFS Forest Activities Tracking System courtesy of Joe Sherlock (Pacific Southwest Research Region silviculturist).

small trees to an upper diameter limit. The proportion of sanitation cuts dropped in 1999 as existing contracts established before CASPO were completed and CASPO prescriptions became the predominant silvicultural prescription. Commercial thinning associated with CASPO guidelines, a focus on forest thinning to meet fuels reduction objectives, and postfire salvage logging have been the dominant prescriptions on NFS lands in the Sierra Nevada between 1999 and 2014 (fig. 5-2).

About 255 143 ha [665,357 ac] of silvicultural treatments occurred within the range of the California spotted owl in the Sierra Nevada (see appendix p. 155 for details), of which about 199 600 ha (299,000 ac; 45 percent) were treated from 2002 through 2014 when NFS spatial data on treatments was complete (table 5-2, fig. 5-4). Sanitation cuts were the predominant silvicultural prescription used during 1990–1994. Commercial thin was the predominant prescription used during 1996–2013, followed by episodic salvage events and smaller amounts of clearcutting (table 5.2).

Table 5-2—Treatment hectares accomplished on National Forest lands within the range of the California spotted owl in the Sierra Nevada by silvicultural prescription and year during 1990–2014

Year	Clear Cut	Commercial Thin	Group Selection	Other	Overstory Removal Cut	Salvage Cut	Sanitation Cut	Seed-tree Cut	Single Tree Selection Cut	Total
1990	4,123	1,246	109	121	3,213	1,467	10,348	899	1,577	23,104
1991	3,262	1,486	49	67	1,864	5,803	6,599	986	1,936	22,051
1992	3,488	609	11	16	973	4,888	7,257	305	834	18,380
1993	1,790	1,228	89	22	417	4,268	733	424	641	9,613
1994	1,746	851	36	56	551	166	4,164	253	152	7,975
1995	1,341	2,852	11	143	559	24	747	348	137	6,163
1996	742	3,655	78	40	89	0	5,316	139	286	10,344
1997	2,146	4,396	66	180	135	47	746	146	459	8,322
1998	418	8,151	20	90	29	2,298	1,088	57	83	12,235
1999	250	8,555	282	286	45	2,469	670	6	811	13,375
2000	4	6,837	43	470	0	168	1,340	17	214	9,094
2001	28	5,925	127	632	8	111	439	48	94	7,412
2002	263	4,847	28	370	43	93	79	0	329	6,051
2003	2,450	5,032	3	186	0	9,480	814	0	45	18,010
2004	13	9,255	94	0	0	824	1,284	5	643	12,119
2005	3	4,359	50	0	1	576	650	2	209	5,850
2006	1,724	11,012	408	29	3	2,187	568	0	454	16,386
2007	115	9,448	231	7	0	281	319	0	28	10,428
2008	0	3,203	8	8	0	1,041	313	0	66	4,639
2009	4	5,295	96	38	0	6,343	411	0	43	12,230
2010	7	7,971	343	89	0	1,045	174	0	314	9,942
2011	12	5,431	172	77	0	201	20	0	0	5,912
2012	0	6,734	183	37	0	75	0	0	90	7,119
2013	0	5,632	14	1	0	3,574	82	0	0	9,303
2014	0	760	14	0	0	2,339	0	0	91	3,203
Total	23,928	124,768	2,565	2,964	7,929	49,770	44,162	3,637	9,538	269,261

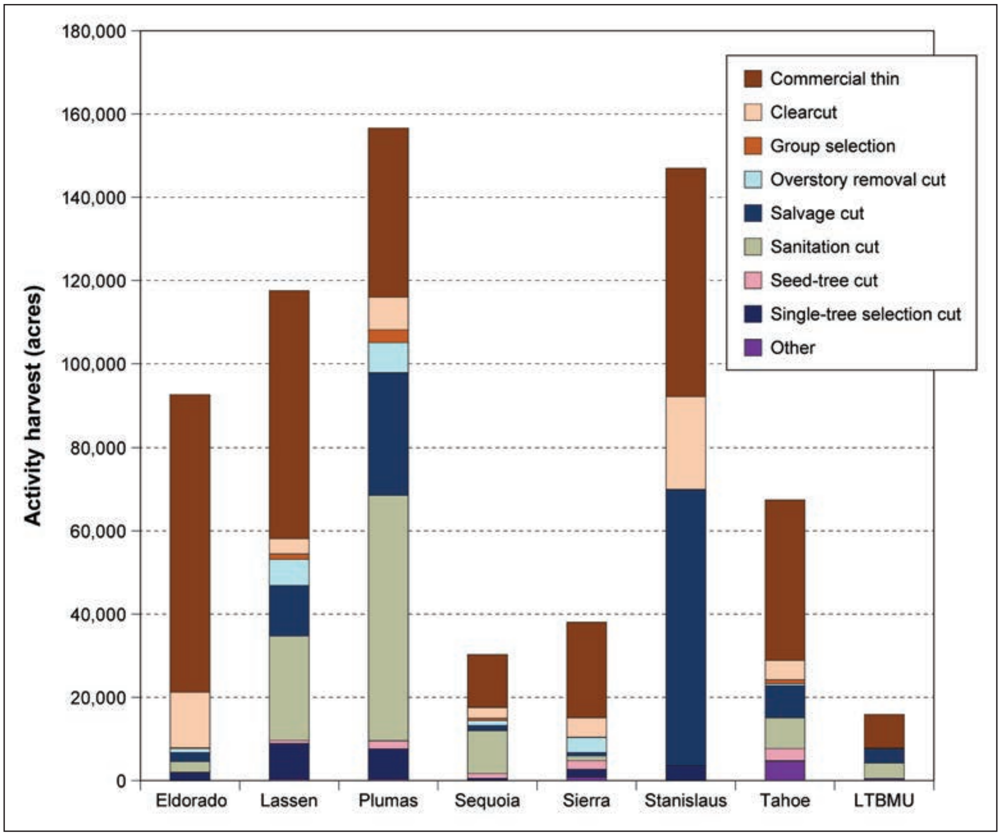

Figure 5-4—Treatment acres accomplished on national forest lands within the range of the California spotted owl in the Sierra Nevada by silvicultural prescription and national forest, 1990–2014. Sources: Taken from USFS Forest Activities Tracking System courtesy of Joe Sherlock (Pacific Southwest Research Region silviculturist); owl range from California Department of Fish and Wildlife.

Current USFS practices often focus on two metrics when implementing management treatments; maximum tree diameter removed (“diameter limits”) and residual canopy cover. Although trees up to 75 cm (30 in) d.b.h can be marked for removal, in many forests that have been previously thinned, the maximum diameter limit is set to a lower size because removing larger trees would drop the residual canopy cover below the target. Canopy cover is usually indirectly estimated using the Forest Vegetation Simulator or FVS model based upon the number, size, and species of the leave trees. As an indirect estimate, FVS assumes a certain amount of crown overlap (Crookston and Stage 1999) and does not account for spatial variability in tree locations (Christopher and Goodburn 2008). Nor does the FVS-generated canopy cover target consider canopy closure patterns or distinguish between clumped or regular distributions, differences that appear to be important functional and structural attributes of fire-adapted forests (Churchill et al. 2013, Larson and Churchill 2012, Lydersen et al. 2013).

Canopy cover targets are a featured objective in recent management guidance documents (e.g., Sierra Nevada Forest Plan Amendments of 2001 and 2004; USDA FS 2001, 2004) and are set to be no lower than an average of 40 percent in the larger “home range core area” (HRCA), and no lower than an average of 50 percent in the “protected activity center” (PAC). Treatment in owl PACs is intended to be limited (see the 2004 Sierra Nevada Forest Plan Amendment; USDA FS 2004), but canopy cover targets are still widely used when fuels reduction treatments are implemented within the HRCA on NFS lands. The cumulative area of PACs and HRCAs affects a fairly large proportion of a landscape. See chapter 3 for more details on these management designations and their detailed definitions.

On national forests, some aspects of spotted owl habitat have likely improved since the 1992 release of the CASPO guidelines. Average tree diameter in many forests has increased because of growth and the removal of smaller trees in treated stands while retaining all trees ≥75 cm (30 in) d.b.h. In general, the amount of forest dominated by large trees is probably gradually increasing, although some studies suggest climate change or drought mortality may be disproportionately higher in larger than smaller trees (Lutz et al. 2009, van Mantgem et al. 2009). Likewise, forest growth increases canopy cover and, even in treated stands, cover is retained at 40 percent or greater.

However, in three of the four owl demographic areas, populations are declining. It is uncertain to what degree some of this decline is due to legacy effects (e.g., loss of large tree and defect structure removal and reduction in canopy cover) before CASPO guidelines took hold after 1992. Compounding the uncertainty is the increased role of high-severity wildfire in changing forest conditions. More owl habitat is now affected by wildfire than by mechanical treatment each year (North et al. 2012), and its effects on habitat conditions likely vary with severity and patch size effects of fire behavior.

Private industrial forest lands—
About 1.2 million ha (2.9 million ac) of silvicultural treatments were approved or completed on private industrial forestlands between 1990 and 2013 (table 5-3; see appendix p. 155 for detailed information). Of the majority of acres attributed with a specific silvicultural prescription, the predominant treatments were selection cuts (322 652 ha [806,630 ac]), shelterwood cuts (201 622 ha [504,054 ac]), commercial thins (114 460 ha [286,152 ac]), clearcuts (105 493 ha [263,733 ac]), and sanitation salvage cuts (82 541 ha [206,352 ac]) across the 1990-2013 assessment period (table 5-3). The highest numbers of treated acres were recorded for Shasta, Lassen, Plumas and Tehama Counties (table 5-4). At least 403 876 ha (998,000 ac) of treatment are recorded to have occurred within the range of the California spotted owl in the Sierra Nevada during 1997–2013 (table 5-5).

On average, forests on private land are younger (71 years) than those on public land (104 to 115 years) (Stewart et al. 2016) and often lack the stand structural features associated with old forests such as "defect" trees and large snags and logs. Most commercial harvest is concentrated on the large ownerships predominantly in the southern Cascades and northern Sierra Nevada. Almost 60 percent of commercial harvest on private lands comes from five northern California counties (Humboldt, Shasta, Siskiyou, Mendocino, and Plumas), and collectively, private ownership forests produce about 85 percent of California forests' lumber, pulp and bioenergy products (Morgan et al. 2012).

National parks—
The NPS maintenance of mid-elevation forest conditions faces three challenges. A primary constraint to NPS resource management is that much of these parks is within federally designated wilderness areas, and mechanical manipulation is restricted in these areas. The NPS does not generally mechanically manipulate vegetation but will for human safety or park infrastructure. Hence, managing tree density can only be accomplished with fire; both prescribed fire and managed wildland fire. The NPS is further constrained by a limited capacity to deploy prescribed fire. Limited staffing and air quality restrictions generally result in a relatively small fraction of the national parks being treated with prescribed fire (North et al. 2012). The prescribed fire that has been deployed is typically limited to areas of high

Table 5-3—Treatment hectares completed or approved in timber harvest plans on private industrial forest lands in the Sierra Nevada by silvicultural prescription and year during 1990–2013

Year	Clear cut	Commercial thin	Conversion	Damaged timber-land	Fuel break	Group selection	Other	Rehabilitation	Sanitation salvage	Seed-tree cut	Selection cut	Shelter-wood cut	Transition	Total
1990	1,595	1,265	0	0	0	0	21,271	679	1,811	2,024	6,643	8,673	8,026	51,986
1991	3,852	6,909	0	0	0	0	11,041	229	2,827	1,171	6,206	12,483	9,014	53,732
1992	478	4,677	0	0	0	0	9,180	568	1,382	795	7,620	4,622	9,909	39,231
1993	956	1,945	6	0	0	0	10,056	826	1,813	1,061	12,410	7,303	7,129	43,504
1994	998	3,826	16	0	0	0	8,105	634	2,044	1,659	7,708	3,498	9,750	38,236
1995	1,075	4,689	8	0	0	0	9,710	623	2,446	1,205	12,767	5,354	8,864	46,741
1996	1,132	6,789	177	0	0	5	12,130	585	4,567	1,395	13,860	7,073	10,374	58,087
1997	711	5,887	0	0	0	96	7,513	773	3,309	833	20,285	3,369	5,021	47,798
1998	735	5,763	220	0	0	18	6,869	1,957	11,368	419	10,976	3,809	4,024	46,159
1999	1,977	6,480	2	0	32	407	6,384	1,635	2,781	864	27,269	5,931	4,041	57,803
2000	2,099	7,733	600	0	193	1,404	1,412	559	4,214	1,162	19,386	10,665	713	50,139
2001	4,274	9,477	27	2	274	1,316	2,819	1,252	3,272	2,519	11,973	12,582	348	50,136
2002	2,618	6,679	164	17	380	2,463	307	806	2,691	1,019	14,041	10,079	91	41,356
2003	4,325	4,920	334	75	254	1,239	876	1,581	4,991	2,951	17,619	10,361	465	49,990
2004	4,730	6,692	149	104	902	1,294	344	743	4,015	921	14,841	11,297	459	46,490
2005	5,990	3,997	436	107	243	2,053	22	688	2,852	1,861	11,713	14,582	166	44,709
2006	4,901	1,663	382	15	160	2,110	45	933	1,120	1,076	12,160	5,095	59	29,719
2007	4,066	4,460	576	0	203	3,086	47	703	3,639	2,965	9,821	14,357	170	44,092
2008	6,768	4,813	107	3	1,049	3,185	133	402	2,649	1,013	16,403	8,625	90	45,240
2009	4,963	1,591	85	192	181	1,962	0	327	767	454	13,150	10,238	49	33,959
2010	3,674	328	46	239	179	2,772	0	122	1,049	196	4,614	4,210	0	17,428
2011	3,755	881	193	405	1,068	4,509	0	158	639	187	7,093	3,122	27	22,038
2012	7,678	2,311	139	377	433	2,566	0	104	4,443	347	9,625	5,567	93	33,684
2013	6,552	1,124	238	1,390	408	10,996	0	153	436	194	5,058	4,317	186	31,053
Approved (no completion date)	26,829	10,900	794	1,803	2,505	42,383	1,716	1,181	12,384	2,091	33,189	16,773	817	153,366
Total	106,729	115,801	4,699	4,730	8,464	83,866	109,980	18,221	83,508	30,380	326,432	203,983	79,884	1,176,677

Table 5-4—Treatment hectares completed or approved in timber harvest plans on private industrial forest lands in the Sierra Nevada by silvicultural prescription and county during 1990–2013

County	Clear-cut	Commercial thin	Conversion	Damaged timber-land	Fuel break	Group selection	Other	Rehabilitation	Sanitation salvage	Seed-tree cut	Selection cut	Shelter-wood cut	Transition	Total
Alpine	55	44	8	0	0	0	246	0	181	25	92	38	0	688
Amador	2,262	449	341	0	873	630	5,021	737	312	1,535	5,684	3,575	3,209	24,629
Butte	9,384	6,254	35	656	258	2,362	7,598	3,909	2,724	2,910	13,516	13,282	2,056	64,945
Calaveras	5,788	3,391	84	0	1,192	1,721	9,709	1,237	747	2,071	16,909	3,991	4,659	51,500
El Dorado	12,968	6,170	237	0	619	1,321	13,020	1,732	7,975	5,962	9,651	19,576	6,937	86,169
Fresno	117	490	326	0	0	457	1,153	157	214	116	20,655	15	352	24,053
Kern	0	12	0	0	0	0	258	97	652	0	3,383	0	0	4,403
Lassen	12,610	20,689	37	55	457	15,660	10,313	1,891	12,162	167	66,747	18,695	8,200	167,682
Madera	0	0	0	0	0	0	165	81	77	7	905	151	466	1,852
Mariposa	38	81	0	0	124	343	682	642	208	330	3,766	399	812	7,426
Nevada	2,897	2,545	1,021	5	321	8,107	3,017	704	5,377	1,386	11,709	13,926	3,886	54,902
Placer	2,831	2,688	1,023	0	203	3,815	10,000	1,325	8,454	1,729	10,019	17,115	4,075	63,276
Plumas	6,003	13,664	1,267	2,266	1,438	15,356	3,780	732	5,761	2,021	45,186	19,998	12,413	129,887
Shasta	34,379	40,782	283	1,410	1,785	19,590	29,586	2,434	28,402	4,692	63,212	47,773	15,069	289,396
Sierra	1,406	4,902	2	0	42	6,794	1,522	202	3,498	719	7,546	6,886	1,067	34,586
Tehama	9,674	8,639	7	297	565	3,994	9,255	873	3,910	3,259	22,219	29,974	8,559	101,224
Tulare	0	91	0	0	122	2	334	42	453	40	2,111	0	164	3,359
Tuolumne	4,222	3,860	7	26	404	133	1,932	551	1,368	2,709	20,139	1,479	6,663	43,493
Yuba	2,095	1,050	21	15	61	3,583	2,391	874	1,031	703	2,983	7,108	1,294	23,209
Total	106,729	115,801	4,699	4,730	8,464	83,866	109,980	18,221	83,508	30,380	326,432	203,983	79,884	1,176,677

Table 5-5—Treatment hectares completed in timber harvest plans on private industrial forest lands within the range of the California spotted owl in the Sierra Nevada by silvicultural prescription during 1997–2013

Silvicultural Prescription	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	Totals
Clearcut	14	55	449	896	1,873	1,901	3,321	3,467	4,642	2,762	2,748	5,195	3,240	2,252	1,768	4,423	3,112	42,118
Commercial thin	42	22	295	416	1,115	1,606	2,770	2,646	2,360	696	1,066	3,478	1,418	265	805	1,221	1,020	21,241
Conversion	0	1	2	31	1	17	201	93	29	188	388	89	59	46	177	119	236	1,679
Fuelbreak/defensible space	0	0	32	173	165	304	250	889	141	143	203	583	181	105	954	387	408	4,917
Group selection	7	8	394	601	514	1,895	1,096	1,167	2,038	2,022	2,886	3,101	1,724	1,381	4,381	1,861	8,900	33,976
Rehabilitation—understocking	113	30	32	124	215	165	912	354	450	286	382	197	233	74	11	46	116	3,742
Sanitation salvage	44	21	313	163	594	1,235	1,715	2,296	2,402	821	2,089	2,431	640	622	246	2,633	318	18,582
Seed tree removal cut	157	21	212	694	173	300	958	136	264	437	489	97	28	40	55	172	92	4,326
Seed tree seed cut	30	3	22	36	157	297	603	467	681	483	2,023	430	426	147	132	159	94	6,189
Selection	127	547	3,047	7,253	2,647	6,415	12,218	9,617	8,749	7,029	7,321	13,049	11,803	4,133	3,257	4,401	3,741	105,355
Shelterwood prep cut	0	0	0	0	0	94	71	1	752	58	53	0	0	13	0	5	0	1,047
Shelterwood rem/commercial thin	0	93	0	0	5	239	0	358	829	32	935	874	505	668	60	1,091	0	5,688
Shelterwood removal cut	123	461	1,316	1,595	2,718	2,688	5,425	5,885	7,184	3,498	6,230	5,826	4,523	1,598	1,641	2,769	2,342	55,821
Substantially damaged timberland	0	0	0	0	0	17	31	103	0	15	0	3	192	1	405	135	511	1,413
Transition	87	48	38	225	50	45	83	87	58	59	101	86	49	0	26	52	44	1,137
Totals	743	1,311	6,152	12,207	10,228	17,219	29,653	27,566	30,581	18,528	26,914	35,440	25,020	11,345	13,917	19,474	20,934	307,230

human use (e.g., sequoia groves, Yosemite Valley). Consequently, the NPS uses wildfire to the extent possible to accomplish forest management objectives (van Wagtendonk and Lutz 2007). Managing to retain and restore resilient forest ecosystems has been more aggressive on NPS lands than other areas because NPS policy enables wildfires in appropriate locations to run their course when feasible.

There has been a recent untethering of the NPS resource stewardship from directives of striving for historical representation (USDI NPS 2012) with growing recognition that this is an unattainable and undesirable goal (USDI NPS 2012). In response, NPS has taken on management planning to build ecosystem resilience for coping with changing climates. “National Park Natural Condition Assessments” are designed to identify key indicators of natural condition (http://www.nature.nps.gov/water/nrca/). “Resource Stewardship Strategies” (http://www.nature.nps.gov/water/planning/resourcestewardshipstrategies.cfm) are attempts to plan for future management, including climate change (http://www.nps.gov/subjects/climatechange/response.htm). Parks units are now compelled to consider climate change adaptation and how to manage for climate-resilient forests. This new management directive is likely to include incentives to foster forested ecosystems that are resilient to a range of future stressors.

Current Status of Forests With Potential California Spotted Owl Habitat

Focusing solely on lands included within the California wildlife habitat relations (CWHR)-defined California spotted owl range map for the Sierra Nevada, existing vegetation classification and mapping (EVEG) estimates that there are about 1.98 million ha (4.9 million ac) of CWHR class 4M or greater habitat (4M, 4D, 5M, 5D, 6) (>30 cm [12in] d.b.h., >40 percent canopy cover), with approximately 75, 7, and 18 percent occurring on NFS, NPS, and private/other government (POG) lands, respectively (table 5-6). About 53 percent of the 4M and greater classes are classified as Sierra Nevada mixed conifer (SMC), with the majority of SMC occurring on NFS lands. About 1.2 million ha (2.9 million ac) of 4D and greater classes (4D, 5D, 6) (>30 cm [12 in] d.b.h., >60 percent canopy cover) are estimated to be present, with 73, 9, and 18 percent distributed across NFS, NPS, and POG lands, respectively (table 5-7). The 4D and greater class habitat is predominantly classified as white fir (53 percent). For CWHR class 5M and above (5M, 5D, 6) (>60 cm [24 in] d.b.h., >40 percent canopy cover), about 607 029 ha (1.5 million ac) are estimated with 80, 10, and 10 percent distributed on NFS, NPS, and POG lands, respectively (table 5-8). The 5M and above class is classified primarily as SMC (63 percent).

Table 5-6—Distribution (hectares) of California wildlife habitat relationships class 4M or greater (4M, 4D, 5M, 5D, 6) by vegetation type and land ownership within the range of the California spotted owl in the Sierra Nevada

Agency	Blue oak woodland	Douglas-fir	Eastside pine	Jeffrey pine	Lodgepole pine	Montane hardwood/conifer	Montane hardwood	Ponderosa pine	Red fir	Sierran mixed/conifer	White fir	Other	Total
National forest:													
Eldorado	76	0	3	1,147	1,371	5,361	6,836	12,492	31,045	110,115	9,708	545	178,698
Inyo	0	0	128	130	1,760	14	108	0	581	1,943	134	1,648	6,448
Lassen	61	0	2,443	3,053	3,014	2,101	1,628	7,276	9,733	109,522	26,607	395	165,834
LTBMU	0	0	14	5,985	2,041	8	0	0	3,928	13,671	2,083	925	28,655
Plumas	63	10,817	5,736	277	218	19,422	7,505	6,450	7,725	189,595	40,043	172	288,024
Sequoia	3,014	0	918	8,259	3,818	11,508	20,575	9,744	27,643	71,187	950	3,282	160,898
Sierra	1,317	0	0	2,311	8,833	21,699	26,239	25,945	36,208	89,457	934	21,977	234,921
Stanislaus	798	0	0	2,475	1,263	14,242	17,660	21,667	25,933	108,734	6,086	569	199,427
Tahoe	262	20,594	1,004	446	2,689	11,322	7,865	5,411	22,943	130,935	17,331	488	221,291
National park:													
Lassen Volcanic National Park	0	0	0	45	91	83	0	0	1,475	5,772	1,464	162	9,092
Sequoia and Kings Canyon National Parks	0	0	0	4,605	1,299	3	707	1	10,185	29,100	3,876	3,204	52,980
Yosemite National Park	51	124	0	6,379	3,577	151	2,560	2,594	15,873	51,009	2,055	2,417	86,791
Private/other government	7,676	32,841	5,453	266	433	50,934	44,103	58,134	357	146,686	2,961	3,090	352,934
Total	13,319	64,376	15,699	35,381	30,407	136,851	135,786	149,714	193,629	1,057,726	114,232	38,873	1,985,993

Table 5-7—Distribution (hectares) of California wildlife habitat relationships class 4D or greater (4D, 5D, 6) by vegetation type and land ownership within the range of the California spotted owl in the Sierra Nevada

Agency	Blue oak woodland	Douglas-fir	Eastside pine	Jeffrey pine	Lodgepole pine	Montane hardwood/conifer	Montane hardwood	Ponderosa pine	Red fir	Sierran mixed/conifer	White fir	Other	Total
National forest:													
Eldorado	43	0	1	276	279	4,559	6,733	171	10,348	7,191	78,489	4,506	112,597
Inyo	0	0	9	16	552	10	72	210	0	98	285	14	1,267
Lassen	18	0	367	1,221	801	1,558	1,335	188	3,262	4,568	41,096	10,967	65,381
LTBMU	0	0	0	255	184	6	0	22	0	95	540	289	1,391
Plumas	22	9,520	943	56	4	15,993	5,160	164	4,164	2,038	82,963	12,766	133,793
Sequoia	728	0	27	1,347	573	9,011	19,073	1,105	7,657	8,982	40,907	249	89,658
Sierra	796	0	0	719	5,505	14,989	23,848	9,956	23,198	21,436	65,164	707	166,318
Stanislaus	461	0	0	329	320	12,539	16,982	155	16,148	10,150	80,245	4,823	142,153
Tahoe	127	16,881	239	189	1,644	7,434	5,426	182	2,704	8,826	97,631	8,270	149,552
National park:													
Lassen Volcanic National Park	0	0	0	41	2	50	0	21	0	124	1,920	282	2,440
Sequoia and Kings Canyon National Parks	0	0	0	1,590	517	3	615	848	1	8,150	24,686	3,106	39,516
Yosemite National Park	4	118	0	1,386	3,178	59	1,557	1,794	623	13,786	41,346	1,990	65,841
Private/other government	3,274	25,002	881	41	228	41,461	31,520	1,612	39,692	112	74,863	1,148	219,833
Total	5,473	51,520	2,468	7,465	13,786	107,674	112,320	16,426	107,797	85,555	630,137	49,117	1,189,740

Table 5-8—Distribution (hectares) of California wildlife habitat relationships class 5M or greater (5M, 5D, 6) by vegetation type and land ownership within the range of the California spotted owl in the Sierra Nevada

Agency	Blue oak woodland	Douglas-fir	Eastside pine	Jeffrey pine	Lodgepole pine	Montane hardwood/conifer	Montane hardwood	Ponderosa pine	Red fir	Sierran mixed/conifer	White fir	Other	Total
National forest:													
Eldorado	0	0	1	113	247	53	10	356	3 494	19 255	1 862	45	25 436
Inyo	0	0	1	4	170	2	0	0	156	74	10	81	497
Lassen	12	0	39	196	96	468	56	978	1 754	26 684	5 423	96	35 803
LTBMU	0	0	0	931	467	2	0	0	774	1 865	46	260	4 346
Plumas	18	7 349	767	57	25	7 588	1 332	2 915	2 823	86 033	14 335	3	123 244
Sequoia	186	0	211	4 563	798	1 575	1 880	2 033	6 064	25 406	329	565	43 611
Sierra	188	0	0	41	1 953	2 902	1 607	9 108	12 169	40 341	490	2 542	71 340
Stanislaus	12	0	0	352	273	1 035	64	1 155	7 298	26 106	2 738	86	39 120
Tahoe	89	15 545	378	279	1 661	6 383	757	1 661	16 704	95 909	12 072	290	151 729
National park:													
Lassen Volcanic National Park	0	0	0	36	18	6	0	0	478	2 997	445	104	4 085
Sequoia and Kings Canyon National Parks	0	0	0	0	0	2	0	0	1	4 397	0	0	4 400
Yosemite National Park	7	46	0	1 389	1 686	20	82	1 074	10 750	33 991	1 708	984	51 737
Private/other government	456	15 202	416	16	5	4 505	1 115	13 596	67	27 765	349	355	63 847
Total	968	38 141	1 813	7 978	7 402	24 542	6 902	32 876	62 532	390 825	39 807	5 412	619 196

LTBMU = Lake Tahoe Basin Management Unit.

Most acres of important California spotted owl habitat classes occur on NFS lands. Between about 133 547 and 166 326 ha (330,000 and 411,000 ac) of 4D and greater habitat is estimated to occur on the Sierra, Tahoe, Stanislaus, and Plumas National Forests, while between 65 155 and 112 503 ha (161,000 and 278,000 ac) are estimated to occur on the Eldorado, Sequoia, and Lassen National Forests (tables 5-6 to 5-8). The Inyo National Forest and Lake Tahoe Basin Management Unit support fewer habitat acres, as the Inyo overlaps minimally with the range of the California spotted owl in the Sierra Nevada, while habitat is generally limited to the western half of the Lake Tahoe basin. Although inferences about amounts and distributional patterns of California spotted owl habitat may be tempered given the uncertainty regarding the accuracy and consistency of the base vegetation maps, results highlight the importance of NFS lands for providing spotted owl habitat in the Sierra Nevada. About 73 to 80 percent of the CWHR habitat classes most often used by owls are estimated to currently occur on NFS lands.

Historical Fire Effects on Mid-Elevation Forests

Fire is a critical ecosystem process throughout Sierra Nevada mid-elevation forests. This is particularly the case for yellow and Jeffrey pine (*P. jeffreyi* Balf.) and mixed-conifer forest types within the Sierra Nevada, where fire historically (i.e., pre-Euro-American settlement) occurred frequently, with generally low- to moderate-severity effects (Skinner and Taylor 2006, van Wagtendonk and Fites-Kaufman 2006). Numerous studies demonstrate that this fire frequency (5 to 15 years) maintained low-density stands across much of the landscape, composed of primarily large, fire-resistant trees. Reconstructed conifer densities (trees >15 cm [6 in] d.b.h.) in these forest types ranged from 60 to 82 trees/ha (24 to 41 trees/ac) (Collins et al. 2011; Scholl and Taylor 2010; Taylor 2004, 2010). Collins et al. (2011) estimated the average canopy cover for historical forest conditions was 22 percent, with a range of 8 to 37 percent. Interestingly, these canopy cover estimates are similar to those measured in a contemporary Jeffrey pine-mixed-conifer forest that has a more intact disturbance regime (i.e., no timber harvesting and limited fire suppression) in the Sierra San Pedro Martir, Baja, California (Stephens and Gill 2005). However, stand density, structure, and composition likely varied depending on topographic and edaphic conditions, as well as a result of the stochastic patchiness of fire effects.

The preponderance of evidence in the scientific literature currently supports the notion that contemporary forests that have not been subject to recent forest management (i.e., tree removal) are generally considerably denser than forests found prior to 100+ years of fire exclusion and selective logging. However, a few recent studies conducted in the Sierra Nevada challenge the prevailing understanding of historical forest structure and fire patterns (see Baker 2014, Odion et al. 2014). They indicate that stand-replacing fire effects were a greater component of historical fire regimes than the predominant body of research suggests, and that resulting tree densities were greater than those reported in previous studies (e.g., Ansley and Battles 1998, Bouldin 1999, Collins et al. 2011, Knapp et al. 2013, McKelvey and Johnson 1992, North et al. 2007, Parson and Debenedetti 1979, Scholl and Taylor 2010, Taylor 2004, Taylor et al. 2014, Vankat and Major 1978). Odion et al. (2014) used stand age estimates from Forest Inventory and Analysis data to infer past proportions of stand-replacing fire. From this they concluded that current "reference" conditions underrepresent early successional plant communities created by stand-replacing fire. Baker (2014) used historical tree data from land survey markers to reconstruct historical proportions and patch sizes of stand-replacing fire across large landscapes. He concluded that historical forests in the Sierra Nevada were generally much denser, hence supported much greater amounts of stand-replacing fire than other historical forest reconstructions have reported. The significance of his conclusions, and their applicability to restoration of mixed-conifer forests in the Sierra Nevada, merit careful consideration and vetting through the scientific community to reconcile the foundation of the discrepancies with existing published literature. Concerns about the source of the observed discrepancies include:

- Potential bias in plot/tree selections. Baker used General Land Office survey witness trees that have been shown to be biased toward trees that were less likely to be harvested—smaller trees or less commercially valuable species, hence higher likelihood that the trees would persist as markers for locating survey points (see Bouldin 2008, Manies and Mladenoff 2000). Odion et al. 2014 only included plot data from wilderness areas and national parks, which in the Sierra Nevada tend to be in higher elevations, hence a greater proportion of upper montane forest types. Upper montane forests are associated with longer intervals between fire and greater proportions of high-severity relative to the pine-mixed-conifer forests in the lower montane zone (Van de Water and Safford 2011). This limits the applicability of the study across the pine-mixed-conifer zone.
- Limited density of tree samples. Baker (2014) relied on sampling densities that are less than 1 tree per (80 ac) 32.3 ha.
- Misinterpretation of tree data. Odion et al. (2014) used composite stand-age estimates as evidence of postfire cohort initiation dates. These composite estimates have a high degree of error in capturing actual tree initiation dates, and as a result, are a poor representation of the time since last stand-replacing disturbance (Stevens et al. 2016).

These limitations and others (see Fulé et al. 2013) call into question the robustness of these studies and their applicability toward forest restoration efforts.

Several studies have demonstrated a high degree of spatial complexity across historical landscapes, which consisted of early seral vegetation (e.g., dense conifer regeneration, shrubs) and denser mature forest stands (e.g., Beaty and Taylor 2001, Collins et al. 2015, Nagel and Taylor 2005, Stephens et al. 2015, Taylor 2000), within a matrix of generally low-density stands. This complexity was likely a product of differential fire effects and timing, including some stand-replacing fire, driven by variability in multiple factors: vegetation/fuels, topography, site productivity/moisture availability, and climate. Estimates of historical stand-replacing fire in mixed-conifer and yellow pine forests range from 5 to 10 percent of the area within a burn at any given time (Mallek et al. 2013), which was likely aggregated in small patches (usually <2 ha [5 ac]) distributed across the landscape (Collins and Stephens 2010, Show and Kotok 1924). Drainage bottoms associated with larger perennial streams may have experienced less frequent fire than more upslope locations and thus were able to sustain more consistently dense and multilayered canopies (Collins and Skinner 2014). Another attribute associated with frequent fire in these forests is a complex spatial pattern of trees, consisting of isolated individuals, multiple tree clumps, and openings (Churchill et al. 2013, Fry et al. 2014, Knapp et al. 2012, Lydersen et al. 2013). This complexity was also most likely driven by fine-scale patchiness in fire effects and was yet another source of heterogeneity in historical forest conditions (Show and Kotok 1924).

Drivers of Forest Change

Current and Projected Fire Effects

Irrespective of any uncertainty about the historical role of fire in the Sierra Nevada, contemporary fire patterns in the Sierra Nevada differ from those that occurred historically. The differences are in both overall proportion and patch sizes of stand-replacing fire, which are in many cases greater for contemporary fires (Mallek et al. 2013; Stephens et al. 2013, 2014). The proportion of stand-replacing fires and burn patch sizes also have been increasing in the Sierra Nevada from 1984 through 2010 (Miller et al. 2009, Miller and Safford 2012, Steel et al. 2015). These changes in fire characteristics are driven by (1) fire suppression, which tends to constrain fire occurrence to burning primarily under the most extreme fire weather conditions because these are the conditions when a small minority of fires escape initial suppression efforts (Finney et al. 2011), allows an increase in surface and ladder fuels to accumulate, and fosters increased connectivity and homogeneity of vegetation patterns (Collins et al. 2011, Hessburg et al. 2005, Parsons and Debenedetti 1979, Taylor et al. 2014); and (2) climate change, which has and will increase the length of the dry season, which increases both the risk and scale for high-severity fires (Collins 2014, Westerling et al. 2011). Further, projected increases in temperature and decreases in snowpack for the Sierra Nevada (Safford et al. 2012) are likely to result not only in a continued increasing trend in both patch size and proportion of landscape with stand-replacing fire but also an increasing potential for repeated stand-replacing fire, which can lead to vegetation type conversion (Stephens et al. 2013). Current trajectories of fire size and impact, along with predicted doubling of predicted future fire likelihoods, suggest a future in which proportions of stand-replacing fire in the Sierra Nevada exceed levels interpreted from historical data, regardless of sources.

Postfire Forest Management

A recent assessment of land cover change in California demonstrated that fire now accounts for a greater proportion of live tree mortality or "loss" than any other activity (e.g., timber harvesting, development) (Sleeter et al. 2011). Recent research has also demonstrated an increasing proportion of stand-replacing fires and fire patch sizes since 1984 (Miller and Safford 2012, Miller et al. 2009), which has raised concerns about what type of forest, if any, will be reestablished following stand-replacing fire. Recent studies from the northern Sierra Nevada and southern Cascade Range found very low natural conifer regeneration in areas affected by stand-replacing fire up to 11 years following the burn (Collins and Roller 2013, Crotteau et al. 2013). The low conifer regeneration has been attributed mainly to the lack of direct mechanisms for seed persistence or dispersal into large stand-replacing patches (Barton 2002, Goforth and Minnich 2008, Keeley 2012). This suggests that frequent fire intervals for high-intensity fires result in slow and uncertain reforestation of conifer forests, which is particularly evident for pine species (Collins and Roller 2013). If the desired condition for mixed-conifer forests affected by stand-replacing fire is to have mixed-conifer forests return within several decades, then some management intervention may be necessary, particularly by planting pine species, to ensure greater future fire resilience.

Harvest of fire-killed trees (salvage) commonly accompanies reforestation efforts in burned areas. Salvage can have a range of ecological effects depending on the extent of burn area harvested and the removal method (i.e., whole tree harvest, cut to length, etc.). Management objectives for salvage-harvesting include recovering economic value of timber (Sessions et al. 2004), increasing personnel safety for reforestation efforts, and reducing large woody surface fuel accumulation (Peterson et al. 2015, Ritchie et al. 2013, Zhang et al. 2008), which can increase fire resilience. Although salvage-harvesting generally achieves these objectives, it has fewer short-term (<10 years) ecological benefits (Long et al. 2014, Peterson et al. 2009, Ritchie and Knapp 2014). In particular, there can be negative impacts on habitat, including the removal of snags, which also ultimately reduces coarse wood on the forest floor (Swanson et al. 2011). Over the long term (>30 years), the tradeoffs of salvage harvesting versus leaving fire-killed stands unaltered are less clear. A salvage-harvested and reforested area may return to mature conifer forest more quickly than an unaltered burned area, but could lead to a loss of habitat diversity over the landscape if large areas are planted using conventional techniques (i.e., equal spacing among planted trees).

However, stand-replacing fire facilitates development of alternate vegetation types (e.g., montane chaparral or California black oak forests [*Quercus kelloggii* Newberry]), which may be underrepresented in many contemporary landscapes (Cocking et al. 2012, 2014; Nagel and Taylor 2005). Spatial scale is a critical consideration when balancing these tradeoffs. For example, if patches of stand-replacing fire are large (e.g., >200 ha [500 ac]) and left unaltered, the potential for colonization by montane chaparral across the entire patch is high (Collins and Roller 2013, Conrad and Radosevich 1982, Crotteau et al. 2013, Goforth and Minnich 2008), resulting in a homogenization of landscape vegetation rather than increasing vegetation diversity.

Climate Change

General climate change model projections for the Sierra Nevada have temperatures increasing 3 to 6 °C (5.2 to 10.4 °F) during the 21st century (Cayan et al. 2013). Precipitation models differ, with some predicting increases and others decreases in net precipitation (Cayan et al. 2013). These models, however, mask consistent predictions of decreased winter snowpack and increased ecosystem moisture stress (Cayan et al. 2013), accompanied by an increase in the frequency of extreme climatic events (droughts as well as flooding) (Gershunov et al. 2013). These climate change models consistently suggest that by the late 21st century, the Sierra Nevada will experience (1) a decreasing fraction of its annual precipitation as snow, and hence loss of snowpack; (2) increasing temperatures that will increase dry season soil moisture stress (climate water deficit [CWD]); (3) a higher fraction of annual precipitation in fewer storm events; (4) an increased frequency of drought, and (5) a lengthening of the fire season because of earlier onset and later ending of warm, dry conditions.

There are several ways to project the potential consequences of a changing climate on the distribution of Sierra Nevada vegetation types. One approach to projecting future ecosystem composition as a consequence of climate change is to project the future distribution of forest types. Ecosystem models, such as the MC1 model (Lenihan et al. 2008), are used to project the distribution of ecosystems into the future. The results of these models suggest upward shifts in most vegetation types, loss of subalpine forests, and massive forest conversion from types that now dominate to those characteristic of warmer and drier environments.

Another approach uses simple climatic envelope modeling to identify locations where current forest cover is projected to fall outside historical climatic parameters for that forest type (Schwartz unpublished data). These models also predict significant reorganization of forested ecosystems during the next century as warmer and drier conditions prevail, driving upslope expansion of grassland, savannah, and shrub-dominated ecosystems. This approach identifies the climatic attributes that describe present occurrences of each ecosystem type, and then overlays climate projections for different periods into the future (e.g., 2040–2070) onto sites to identify when and where instances of an ecosystem type are projected to no longer be within a suitable climate space for that ecosystem.

Changing climates are relevant to risk of fire, and all projections of future fire conditions that consider climate models predict a near doubling of fire likelihoods (e.g., Westerling et al. 2011). With fire extent, severity, and frequency already increasing in many places (Miller and Safford 2012, Miller et al. 2009), fire is likely to influence changes in forest cover types. Site type change from repeated high-severity fire is already occurring (Stephens et al. 2013). Future changes may be driven by voluntary recruitment or as an active adaptation strategy by planting different species in an effort to create more resilient forests. Vegetation models suggest that many portions of the mid-elevation conifer zone will be vulnerable to such changes.

Upper montane forests will likely also undergo significant changes (North et al. 2016). Modeling of predicted conditions in the Lake Tahoe basin suggests that forested areas that would not have benefited greatly from fuels treatments in the 20th century owing to low fire activity may need significant fuels treatments by the end of this century because of projected increase in fire activity (Loudermilk et al. 2013, 2014). Modeling also suggests that fire activity will increase significantly because of longer fire seasons that will allow more widespread fire ignitions from lightning (Yang et al. 2015). An analysis of trends in the upper elevation of burn areas over the past several decades suggests that wildfires may already be increasing in frequency in upper montane forests (Schwartz et al. 2015).

Increasing frequency and intensity of drought result in increased tree stress and have been implicated in widespread increases in large tree mortality (Dolanc et al. 2014, McIntyre et al. 2015, van Mantgem et al. 2009). Climate change projections of forest ecosystems (Lenihan et al. 2003), forest communities (Schwartz, unpublished report) and tree species (McKenzie 2010) all suggest that existing mid-elevation coniferous forests are poised for conversion to other forest types over much of their current distribution, given drivers such as wildfire. Pests and pathogens as drivers of forest change may also be increasing (e.g., Smith et al. 2005). Collectively, these trends strongly suggest that, under current management practices, all mid-elevation coniferous forests are threatened with conversion to vegetation characteristic of warmer, drier, and more frequently burned types such as montane chaparral, mixed-hardwood forests, and even grasslands (Lenihan et al. 2008).

Putting these predictions into context, however, requires understanding of the spatial resolution of climate projections. Projecting future climate is done using one or more "general circulation models" (GCMs) (IPCC 2013). Although the list of GCMs continues to grow (>15), each GCM is a complex multivariate simulation of future climate on a global scale (IPCC 2013). The global nature of these models is such that they might not capture local processes well, even after downscaling (Gershunov et al. 2013). Although multiple models provide the opportunity to estimate variance in outcomes, they are likely to underestimate the true uncertainty with respect to climate futures. The variation among interrelated and nonindependent global models does not allow capturing the range of variability that might be expected in future climates. Further, microscale variation projections (e.g., cold air drainages) are locally downscaled under the general assumption that current patterns of local variation will be the same in the future. Hence, cold air drainages remain cold air drainages. Finally, we have a relatively poor understanding of forest soils in the Sierra Nevada and an equally poor understanding of the way that soils modify the extent to which changing climate will be expressed by changing forest composition, structure, and function. The consequence of this fine-scale uncertainty is that despite strong predictions of major forest changes in response to climate predictions at large spatial scales, there are likely to be refugia where cooler, moister forest types may persist. Identifying and conserving forests in these refugia might help provide long-term owl habitat even under accelerating changes in climate conditions.

Projections of forest change suggest that under warmer and drier future climate scenarios, all Sierra Nevada forest types are at risk of conversion to some other plant community over the majority of their current distributions. This includes the mid-elevation coniferous forests upon which California spotted owls currently depend. Many currently forested regions of the Sierra Nevada are predicted to be shrub or grassland dominated in the future. Models of late 21st century climate also suggest a future replete with unique combinations of species pools, climate, and disturbance regimes on the complex mixes of Sierra Nevada geologic substrates. The result is many regions may experience conditions that have no strict analogs in the past. This reduces our capacity to predict how they may respond. Forests in some geographic locations (e.g., drainage bottoms) may persist; others (e.g., south-facing slopes) may undergo pronounced shifts in environmental conditions and thus be more likely to change in structure and composition.

The forests of the Sierra Nevada are complex in composition, structure, and function. This complexity reflects wide variation in environmental conditions at both local and regional scales, rich floristic diversity, and a highly varied history of natural and human disturbances (Franklin and Fites-Kaufmann 1996). The role of geological and climatic diversity in creating this complex mosaic of vegetation is prominent. It is this very complexity that may provide an opportunity to ameliorate the potential for total conversion through forest management.

Future Management of Mid-Elevation Forests

> If owl habitat has improved as a result of fire suppression, such improvement may well be illusory and short-lived. Fire is inevitable in these forests, and the probability of catastrophic fire—certainly one of the greatest threats to owl habitat—increases as surface fuels and ladder fuels continue to accumulate. Overly dense stands are subject to extensive mortality from drought and insects, including loss of the most desirable large, old trees—a management policy characterized as 'hands-off plus fire exclusion' (allow forest succession to proceed uninterrupted by periodic natural disturbances) would likely lead to degraded and depauperate, rather than healthy and biologically diverse, ecosystems (Weatherspoon et al. 1992: 253).

Currently, mid-elevation forests in the Sierra Nevada are prone to high-severity fire, drought stress and loss of large trees, and climatically driven vegetation changes. Hands-off management is likely to perpetuate the compromised resilience of mid-elevation forests. Active management that decreases fuel loads and stand density can help reduce wildfire severity, water competition, and slow vegetation change. These active management choices may also affect forest conditions, particularly in dense stands with high canopy cover, that have been associated with preferred spotted owl habitat. New management practices are needed that can accommodate the multitude of management objectives that include fuels reduction, forest resilience, and some high canopy cover forest conditions (McKelvey and Weatherspoon 1992). Some studies have suggested this can be accomplished by increasing structural heterogeneity associated with ecosystem resilience in fire-dependent forests (Churchill et al. 2013, Lydersen and North 2012, North et al. 2009, Stephens and Gill 2005, Stephens et al. 2007). New management practices now attempt to realign forest conditions with their historical variability using existing stand structure ("what you've got to work with") and topography to structure management actions. Topography is used because it is closely tied to two key processes that seem to strongly influence forest conditions: local productivity (associated with water availability) and fire regime.

Creating Forest Heterogeneity

Forest heterogeneity at the landscape level in the Sierra Nevada is strongly influenced by water availability (Tague et al. 2009) as measured by CWD (the difference between potential and actual plant evapotranspiration). Stephenson (1998) first proposed that topographic differences in plant water availability (actual evaporative transpiration [AET) and CWD determined forest type and productivity. Subsequent modeling found general agreement between predicted and actual forest conditions in the southern Sierra Nevada using just AET and CWD (Miller and Urban 1999a, 1999b). For example, fir-dominated forests are usually most abundant where water availability is high (such as on deep soils with their high water-holding capacities); whereas pine-dominated forests are most abundant where water availability is low (such as on shallow soils or in rain shadows) (Fites-Kaufman et al. 2007, Meyer et al. 2007, Stephenson 1998). Slope steepness and slope position (e.g., ridgetop, midslope, valley bottom) are also important factors, as they affect the reception and retention of both meteoric waters and water flowing above, within, and beneath the soil. Recent large-scale analysis of forests in Yosemite National Park using light detection and ranging found CWD to be the best predictor of forest conditions, including canopy cover (Kane et al. 2013, 2014, 2015a, 2015b).

Although overstory forest patterns seem to be associated with CWD, understory conditions are strongly shaped by fire. Lydersen and North (2012) assessed a wide topographic distribution of forests with restored fire regimes. They found that fire history had the strongest influence on understory stand structure. Small-tree density decreased and shrub cover increased with the increased fire severity and frequency that tend to occur on upper slope and ridgetop locations (Lydersen and North 2012). Consistent with other studies, they found that overstory forest conditions were associated with topographic differences in CWD (Lutz et al. 2010). The greatest densities of large, overstory trees, high total basal area and canopy cover, and an abundance of large snags and logs were in more mesic, productive sites such as lower slopes and riparian areas, which have lower CWD. This high biomass forest structure existed in these topographic positions regardless of fire history. These findings suggest that CWD and fire intensity strongly influence forest overstory and understory conditions, respectively. Topography's influence on these two factors appears to produce the heterogeneity characteristic of montane forest landscapes (Lydersen and North 2012, Taylor and Skinner 2003). It also provides a means to estimate which areas in the landscape had the high stem density and canopy closure conditions that might support species associated with these conditions (Taylor and Skinner 1998). Underwood et al. (2010) tested this idea using fisher and California spotted owl radiotelemetry locations. They found higher than expected use of topographic areas associated with higher productivity, forest biomass, and canopy cover such as found in canyon bottoms, lower slopes, and northeast aspect positions.

Heterogeneity within frequent-fire forest types across the Western United States has recently been examined using a meta-analysis of historical forest structure (Larson and Churchill 2012). The within-stand structure has been characterized as containing three main conditions: individual trees, clumps of trees, and openings or gaps (ICO) (Abella and Denton 2009, Churchill et al. 2013, Larson and Churchill 2012, Larson et al. 2012, Sánchez Meador et al. 2011). In this pattern, openings may inhibit crown-fire spread under most (less than severe) weather conditions (Agee et al. 2000, Agee and Skinner 2005, Stephens and Moghaddas 2005) and may be as effective as fuel breaks with regularly spaced trees with wide crown separations (Kennedy and Johnson 2014, Ritchie et al. 2007). The variable microclimate and vegetation conditions between the three conditions may also provide greater habitat diversity for both plants and animals (Roberts et al. 2015). Recent work in the Sierra Nevada using a rare stem map from 1929 has quantified an ICO pattern in mixed-conifer forest (Lydersen et al. 2013). This work provides measures of the relative proportions, sizes, and compositions of each of the three conditions, individual trees, clumps of trees, and openings within active-fire forests. Because stand conditions with an active fire regime vary with topography (Lydersen and North 2012) and different forest types, this single study with a small sample size might be used with caution until more research has been completed.

The openings in an ICO pattern may also increase forest drought resilience. Models suggest that openings could increase soil moisture (Bales et al. 2011) because more snow reaches the forest floor, melting into the soil instead of being intercepted in tree crowns where some of the snow directly sublimates back into the atmosphere (Molotch et al. 2007). Although montane forests are adapted to annual drought stress characteristic of Mediterranean climates, periods of multiple, consecutive dry years can have major impacts (e.g., Guarin and Taylor 2005). For example, there was substantial mortality of conifer trees in the San Bernardino Mountains after the drought of the late 1990s and early 2000s. In the absence of frequent fire, increases in forest density result in greater competition for scarce water (Dolph et al. 1995, Innes 1992). A major concern is potential increases in older tree mortality because large trees are often more prone to drought-induced mortality (Allen et al. 2010). Some studies have found higher than expected mortality rates in large trees (Dolph et al. 1995, Lutz et al. 2009, Ritchie et al. 2008 Smith et al. 2005). Research has not yet been conducted about whether ICOs reduce drought stress in adjacent tree groups. However, current large-tree mortality rates (van Mantgem and Stephenson 2007, van Mantgem et al. 2009) suggest that a ”leave-it-alone” forest management policy that does not reduce stand density could contribute to the loss of old-growth trees (Fettig et al. 2008, 2010a, 2010b; Ritchie et al. 2008).

There are many areas in the Sierra Nevada where mechanical treatment is currently infeasible (e.g., steep slopes, wilderness, roadless areas, etc.) (North et al. 2015). An alternative is the use of managed fire, which is one of the most effective and efficient means of promoting forest resilience (Collins et al. 2009, North et al. 2012). Although first-entry burns may actually increase fire hazard because of tree mortality and vigorous shrub regrowth (Schmidt et al. 2008, Skinner 2005), subsequent low-intensity burns can often produce greater heterogeneity and are more effective at reducing surface fuels than mechanical treatments. However, using fire in forests that have imbedded human development has significant risks. These risks include potential impacts to people and property from smoke production, reduced recreation opportunities, inadequate personnel to conduct and monitor fires, liability for fire escapes, and risk-adverse policies and institutions. Many of the issues relating to fuel treatment intensity and fire use are inherently social in nature (McCaffrey and Olsen 2012). In the future, managed fire may be more widely used but will probably be relegated to more remote areas where potential effects on rural communities are greatly reduced.

Chapter Summary

The processes that influence the distribution and dynamics of forests in the Sierra Nevada occur across large landscapes and multiple land ownerships. Yet, public land agencies struggle to coordinate management strategies and actions across management units, as well as ownership boundaries. A regional strategy to manage for the long-term viability of mid-elevation coniferous forests that accounts for climate change and fire-resilient forest ecosystems would be an important and valuable step toward these desired outcomes.

The mid-elevation coniferous forests of the Sierra Nevada, in their entirety, are highly threatened with conversion to warmer, drier adapted vegetation types. The drivers of this forecasted change are the synergy of warming and drying climate, unsustainable and unprecedented densities of trees, ensuing drought-induced stress, and increasingly severe wildfires. Large fires such as the Rim Fire in 2013 (100 000 ha [250,000 ac]) and the King Fire in 2014 (40 000 ha [100,000 ac]) have resulted in dead tree swaths (i.e., at or close to 100 percent mortality) of unprecedented size in the mid-elevation zone in just the past two years. Even larger fires have occurred in the Western United States in recent decades and are plausible for the Sierra Nevada. With climate change models predicting significant increases in fire probabilities (as much as double current probabilities) during this century, and increasing fuel loads, the prospect of large-scale, stand-replacing fire effects that affect significant portions of the lower and middle elevations of the Sierra Nevada over the next few decades is an increasing possibility. These conditions pose significant challenges to land managers because efforts to maintain current forest conditions are likely to fail. This represents a severe threat to sustaining old-growth habitat conditions associated with the spotted owl.

Our survey of forest change from historical to current conditions, and discussion of drivers of change, suggest there are significant management challenges in maintaining a well-connected network of closed-canopy mid-elevation conifer stands. We focus on five fundamental conclusions regarding the response of mid-elevation coniferous forests to contemporary and anticipated future drivers of change in the Sierra Nevada. First, based on our collective knowledge of pre-European forest structure and composition, the heterogeneity of historical forests likely provided a variety of conditions, including patches of forest vegetation that were suitable for species requiring high densities of large trees. However, the size and connectivity of high-density patches of medium to larger trees (i.e., 27 to 60 cm [11 to 24 in] d.b.h and >60 cm [24 in] d.b.h) in the Sierra Nevada under an active fire regime was likely much smaller than it is currently. These largely second-growth trees have grown and expanded on the landscape after most of the very large trees (i.e., >100 cm [40 in] d.b.h) were removed and fire suppression reduced young tree mortality.

Second, changing climate and increasing severity of wildfires threaten to decrease the current extent and connectivity of mature, dense stands. Third, management decisions predicated on reducing proximate threats to ecosystems (e.g., large-scale stand-replacing fire) by reducing fuels and tree density will result in some decreases in the concentration of high-density, mature-tree patches. Current management on NFS lands predominantly consists of protecting remaining high-priority pockets of suitable habitat while reducing fuels over broader landscapes. These fuel treatments have been applied on only small portions of the landscape (North et al. 2012, 2015) and have been inadequate in preventing large patches of stand-replacing fire. In contrast, the strategic and careful reduction of continuous high fuel loads in portions of high-density, mature forests by mechanical thinning and prescribed fire may reduce the risk of stand-replacing fire and forest type conversion. This fuel reduction effort would sustain larger forested landscapes that include suitable nesting, roosting, and foraging habitat. Ecosystem response models to changing climates suggest that stand-replacing fire will result in conversion of significant amounts of mid-elevation mixed-conifer forests to hardwood, scrub, and grassland vegetation. Based on modeling, conservation strategies for the fisher (*Martes pennanti*), another threatened species in the southern Sierra Nevada, project similar habitat loss because of climate- and disturbance-driven changes in forest conditions (Scheller et al. 2011, Spencer et al. 2010, Syphard et al. 2011). A calculated response to restore resiliency at a landscape scale is necessary to maintain a network of mature, closed-canopy coniferous forests in the Sierra Nevada.

Fourth, owing to different management priorities on private lands and constraints on mechanical thinning in national parks, the opportunities for meaningful long-term ecosystem management experiments may be largely limited to lands managed by the USFS. Evaluation of forest-restoration approaches will depend upon actually using adaptive management strategies and incorporating scientific support needed to monitor management effectiveness and inform changes to improve success (Gutiérrez et al. 2015). Further, all federal land managers are faced with demanding management objectives (e.g., clean air, water provisioning to lowlands, minimizing human risk, maintaining species diversity and ecosystem integrity) such that ecosystem-driven objectives that reduce specific attention to any individual species are favored.

Fifth, there is inadequate understanding of the degree to which California spotted owls would be affected by the predominant ecosystem-based approaches to managing for fire and adapting to climate change. A silvicultural strategy that creates a mosaic of different density patches (e.g., North et al. 2009) is currently viewed by some as the best opportunity to preserve some intact old-growth, legacy forests in the Sierra Nevada. An ecosystem-based forest restoration strategy that prioritizes resilience to fire and changing climates appears to offer a defensible approach to the dilemma that western coniferous forests face in the coming decades.

Literature Cited

Abella, S.R.; Denton, C.W. 2009. Spatial variation in reference conditions: historical tree density and pattern on a *Pinus ponderosa* landscape. Canadian Journal of Forest Research. 39: 2391–2403.

Agee, J.K.; Bahro, B.; Finney, M.A.; Omi, P.N.; Sapsis, D.B.; Skinner, C.N.; van Wagtendonk, J.W.; Weatherspoon, C.P. 2000. The use of shaded fuel breaks in landscape fire management. Forest Ecology and Management. 127: 55–66.

Agee, J.K.; Skinner, C.N. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management. 211: 83–96.

Allen, C.D.; Alison, K.; Macalady, H.C.; Bachelet, D.; McDowell, N.G.; Vennetier, M.; Kitzberger, T.; Rigling, A.; Breshears, D.D.; Hogg, E.H.; Gonzalez, P.; Fensham, R.J.; Zhang, Z.; Castro, J.; Demidova, N.; Lim, J.H.; Gillian, A.; Running, S.W.; Semerci, A.; Cobb, N. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management. 259: 660–684.

Ansley, J.A.S.; Battles, J.J. 1998. Forest composition, structure, and change in an old-growth mixed conifer forest in the northern Sierra Nevada. Journal of the Torrey Botanical Society. 125: 297–308.

Baker, W.L. 2014. Historical forest structure and fire in Sierran mixed-conifer forests reconstructed from General Land Office survey data. Ecosphere. 5(7): 79.

Bales, R.C.; Hopmans, J.W.; O'Green, A.T.; Meadows, M.; Hartsough, P.C.; Kirchner, P.; Hunsaker, C.T.; Beaudette, D. 2011. Soil moisture response to snowmelt and rainfall in a Sierra Nevada mixed-conifer forest. Vadose Zone Journal. 10: 786–799.

Barton, A.M. 2002. Intense wildfire in southeastern Arizona: transformation of a Madrean oak-pine forest to oak woodland. Forest Ecology and Management. 165: 205–212.

Beaty, R.M.; Taylor, A.H. 2001. Spatial and temporal variation of fire regimes in a mixed conifer forest landscape, southern Cascades, California, USA. Journal of Biogeography. 28: 955–966.

Bouldin, J. 1999. Twentieth-century changes in forests of the Sierra Nevada, California. Davis, CA: University of California–Davis. 222 p. Ph.D. dissertation.

Bouldin, J. 2008. Some problems and solutions in density estimation from bearing tree data: a review and synthesis. Journal of Biogeography. 35: 2000–2011.

Bull, E.L.; Parks, C.G.; Torgersen, T.R. 1997. Trees and logs important to wildlife in the interior Columbia River basin. Gen. Tech. Rep. PNW-GTR-391. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 55 p.

California Forestry Practices Regulations. [CA-FPR]. 2012.

Carey, A.B. 2002. Response of northern flying squirrels to supplementary dens. Wildlife Society Bulletin. 30: 547–556.

Carey, A.B.; Wilson, T.M.; Maguire, C.C.; Biswell, B.L. 1997. Dens of northern flying squirrels in the Pacific Northwest. Journal of Wildlife Management. 61: 684–699.

Cayan, D.R.; Tyree, M.; Kunkel, K.E.; Castro, C.; Gershunov, A.; Barsugli, J.; Ray, A.J.; Overpeck, J.; Anderson, M.; Russell, J.; Rajagopalan, B.; Rangwala, I.; Duffy, P. 2013. Future climate: projected average. In: Garfin, G.; Jardine, A.; Merideth, R.; Black, M.; LeRoy, S., eds. Assessment of climate change in the Southwest United States: a report prepared for the national climate assessment. A report by the Southwest Climate Alliance. Washington, DC: Island Press: 101–125.

Christopher, T.A.; Goodburn, J.M. 2008. The effects of spatial patterns on the accuracy of Forest Vegetation Simulator (FVS) estimates of forest canopy cover. Western Journal of Applied Forestry. 23: 5–11.

Churchill, D.J.; Larson, A.J.; Dahlgreen, M.C.; Franklin, J.F.; Hessburg, P.F.; Lutz, J.A. 2013. Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management. 291: 442–457.

Cocking, M.I.; Varner, J.M.; Knapp, E.E. 2014. Long-term effects of fire severity on oak–conifer dynamics in the southern Cascades. Ecological Applications. 24: 94–107.

Cocking, M.I.; Varner, J.M.; Sherriff, R.L. 2012. California black oak responses to fire severity and native conifer encroachment in the Klamath Mountains. Forest Ecology and Management. 270: 25–34.

Cockle, K.L.; Martin, K.; Wesolowski, T. 2011. Woodpeckers, decay, and the future of cavity-nesting vertebrate communities worldwide. Frontiers in Ecology and the Environment. 9: 377–382.

Collins, B.M. 2014. Fire weather and large fire potential in the northern Sierra Nevada. Agricultural and Forest Meteorology. 189–190: 30–35.

Collins, B.M.; Everett, R.G.; Stephens, S.L. 2011. Impacts of fire exclusion and managed fire on forest structure in an old growth Sierra Nevada mixed-conifer forest. Ecosphere. 2: 51.

Collins, B.M.; Lydersen, J.M.; Everett, R.G.; Fry, D.L.; Stephens, S.L. 2015. Novel characterization of landscape-level variability in historical vegetation structure. Ecological Applications. 25: 1167–1174.

Collins, B.M.; Miller, J.D.; Thode, A.E.; Kelly, M.; van Wagtendonk, J.W.; Stephens, S.L. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems. 12: 114–128.

Collins, B.M.; Roller, G.B. 2013. Early forest dynamics in stand-replacing fire patches in the northern Sierra Nevada, California, USA. Landscape Ecology. 28: 1801–1813.

Collins, B.M.; Skinner, C. 2014. Fire and fuels. In: Long, J.; Skinner, C.; Quinn-Davidson, L., eds. Science synthesis to promote resilience of social-ecological systems in the Sierra Nevada and southern Cascade Range. Gen. Tech. Rep. PSW-GTR-247. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 723 p. Chapter 4.1.

Collins, B.M.; Stephens, S.L. 2010. Stand-replacing patches within a 'mixed severity' fire regime: quantitative characterization using recent fires in a long-established natural fire area. Landscape Ecology. 25: 927–939.

Crookston, N.L.; Stage, A.R. 1999. Percent canopy cover and stand structure statistics from the Forest Vegetation Simulator. Gen. Tech. Rep. RMRS-GTR-24. Fort. Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 11 p.

Crotteau, J.S.; Varner, J.M.; Ritchie, M.W. 2013. Post-fire regeneration across a fire severity gradient in the southern Cascades. Forest Ecology and Management. 287: 103–112.

Davis, F.W.; Stoms, D.M. 1996. Sierran vegetation: a gap analysis. In: Sierra Nevada Ecosystem Project: final report to Congress. Vol. II. Assessments and scientific basis for management options. Wildland Resources Center Report No. 37. Davis, CA: University of California, Centers for Water and Wildlands Resources: 671–690.

Dolanc, C.R.; Safford, H.D.; Dobrowski, S.Z.; Thorne, J.H. 2014. Twentieth century shifts in abundance and composition of vegetation types of the Sierra Nevada, CA, US. Applied Vegetation Science. 17: 442–455.

Dolph, K.L.; Mori, S.R.; Oliver, W.W. 1995. Long-term response of old-growth stands to varying levels of partial cutting in the eastside pine type. Western Journal of Applied Forestry. 10(3): 101–108.

Fettig, C.J.; Borys, R.; Dabney, C. 2010a. Effects of fire and fire surrogate treatments on bark beetle-caused tree mortality in the Southern Cascades, California. Forest Science. 56: 60–73.

Fettig, C.J.; Borys, R.R.; McKelvey, S.R.; Dabney, C.P. 2008. Blacks Mountain Experimental Forest: bark beetle responses to differences in forest structure and the application of prescribed fire in interior ponderosa pine. Canadian Journal of Forest Research. 38: 924–935.

Fettig, C.J.; McKelvey, S.R. 2010. Bark beetle responses to stand structure and prescribed fire at Blacks Mountain Experimental Forest, California: 5-year data. Fire Ecology. 6(2): 26–42.

Fettig, C.J.; McKelvey, S.R.; Cluck, D.R.; Smith, S.L. 2010b. Effects of prescribed fire and season of burn on direct and indirect levels of tree mortality in ponderosa and Jeffrey pine forests in California. Forest Ecology and Management. 260: 207–218.

Finney, M.A.; Grenfell, I.C.; McHugh, C.W.; Seli, R.C.; Tretheway, D.; Stratton, R.D.; Brittain, S. 2011. A method for ensemble wildland fire simulation. Environmental Modeling and Assessment. 16: 153–167.

Fites-Kaufman, J.A.; Rundel, P.; Stephenson, N.L.; Weixelman D.A. 2007. Montane and subalpine vegetation of the Sierra Nevada and Cascade Ranges. In: Barbour, M.G.; Keeler-Wolf, T.; Schoenherr, A.A., eds. Terrestrial vegetation of California: Los Angeles, CA: University of California Press: 456–501.

Franklin, J.F.; Fites-Kaufmann, J.A. 1996. Assessment of late-successional forests of the Sierra Nevada: In: Sierra Nevada Ecosystem Project: final report to Congress. Volume II. Assessments and scientific basis for management options. Wildland Resources Center Report No. 37. Davis, CA: University of California–Davis, Centers for Water and Wildland Resources. 627–662. Vol. 2. Chapter 21.

Fry, D.L.; Stephens, S.L.; Collins, B.M.; North, M.P.; Franco-Vizcaino, E.; Gill, S.J. 2014. Contrasting spatial patterns in active-fire and fire-suppressed Mediterranean climate old-growth, mixed conifer forests. PLoS ONE. doi:10.1371/journal.pone.0088985.

Fulé, P.Z.; Swetnam, T.W.; Brown, P.W.; Falk, D.A.; Peterson, D.L.; Allen, C.D.; Aplet, G.A.; Battaglia, M.A.; Binkley, D.; Farris, C.; Keane, R.E.; Margolis, E.Q.; Grissino-Mayer, H.; Miller, C.; Sieg, C.H.; Skinner, C.; Stephens, S.L.; Taylor, A. 2013. Unsupported inferences of high-severity fire in historical dry forests of the western United States: response to Williams and Baker. Global Ecology and Biogeography. 23(7): 825–830.

Gershunov, A.; Rajagopalan, B.; Overpeck, J.; Guirguis, K.; Cayan, D.; Hughes, M.; Dettinger, M.; Castro, C.; Schwartz, R.E.; Anderson, M.; Ray, A.J.; Barsugli, J.; Cavazos, T.; Alexander, M. 2013. Future climate: projected extremes. In: Garfin, G.; Jardine, A.; Merideth, R.; Black, M.; LeRoy, S., eds. Assessment of climate change in the United States: a report prepared for the national climate assessment. Washington, DC: Island Press: 126–147.

Goforth, B.R.; Minnich, R.A. 2008. Densification, stand-replacement wildfire, and extirpation of mixed conifer forest in Cuyamaca Rancho State Park, southern California. Forest Ecology and Management. 256: 36–45.

Guarin, A.; Taylor, A.H. 2005. Drought triggered tree mortality in mixed conifer forests in Yosemite National Park, California. Forest Ecology and Management. 218: 229–244.

Gutiérrez, R.J.; Cheng, A.S.; Becker, D.R.; Cashen, S.; Ganz, D.; Gunn, J.; Liquori, M.; Merrill, A.; Saah, D.S.; Price, W. 2015. Legislated collaboration in a conservation conflict: a case study of the Quincy Library Group, California: In: Redpath, S.; Gutiérrez, R.J.; Wood, K.; Young, J.C., eds. Cambridge, United Kingdom: Cambridge University Press: 271–283.

Hessburg, P.F.; Agee, J.K.; Franklin, J.F. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management. 211: 117–139.

Hunter, J.E.; Bond, M.L. 2001. Residual trees: wildlife associations and recommendations. Wildlife Society Bulletin. 29: 995–999.

Innes, J.L. 1992. Forest decline. Progress in Physical Geography. 16: 1–64.

Intergovernmental Panel on Climate Change [IPCC]. 2013. Climate change 2013: the physical science basis. Contribution of working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change: In: Stocker, T.F.; Qin, D.; Plattner, G.-K.; Tignor, M.; Allen, S.K.; Boschung, J.; Nauels, A.; Xia, Y.; Bex, V.; Midgley, P.M., eds. Cambridge, United Kingdom and New York: Cambridge University Press. 1535 p.

Kane, V.R.; Cansler, C.A.; Povak, N.A.; Kane, J.T.; McGaughey, R.J.; Lutz, J.A.; Churchill, D.J.; North, M.P. 2015a. Mixed severity fire effects within the Rim Fire: relative importance of local climate, fire weather, topography and forest structure. Forest Ecology and Management. 358: 62–79.

Kane, V.R.; Lutz, J.A.; Cansler, C.A.; Povak, N.A.; Churchill, D.; Smith, D.F.; Kane, J.T.; North, M.P. 2015b. Water balance and topography predict fire and forest structure patterns. Forest Ecology and Management. 338: 1–13.

Kane, V.R.; Lutz, J.A.; Roberts, S.L.; Smith, D.F.; McGaughey, R.J.; Povak, N.A.; Brooks, M.L. 2013. Landscape-scale effects of fire severity on mixed-conifer and red fir forest structure in Yosemite National Park. Forest Ecology and Management. 287: 17–31.

Kane, V.R.; North, M.; Lutz, J.A.; Churchill, D.; Roberts, S.L.; Smith, D.F.; McGaughey, R.J.; Kane, J.T.; Brooks, M.L. 2014. Assessing fire-mediated change to forest spatial structure using a fusion of Landsat and airborne LiDAR data in Yosemite National Park. Remote Sensing of Environment. 151: 89–101.

Keeley, J.E. 2012. Ecology and evolution of pine life histories. Annual Review of Forest Science. 69: 445–453.

Kennedy, M.C.; Johnson, M.C. 2014. Fuel treatment prescriptions alter spatial patterns of fire severity around the wildland-urban interface during the Wallow Fire, Arizona. Forest Ecology and Management. 318: 122–132.

Knapp, E.; North, M.; Benech, M.; Estes, B. 2012. The variable-density thinning study at Stanislaus-Tuolumne Experimental Forest. In: North, M., ed. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 127–140.

Knapp, E.E.; Skinner, C.N.; North, M.P.; Estes, B.L. 2013. Long-term overstory and understory change following logging and fire exclusion in a Sierra Nevada mixed-conifer forest. Forest Ecology and Management. 310: 903–914.

Larson, A.J.; Churchill, D. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management. 267: 74–92.

Larson, A.J.; Stover, K.C.; Keyes, C.R. 2012. Effects of restoration thinning on spatial heterogeneity in mixed-conifer forest. Canadian Journal of Forest Research. 42: 1505–1517.

Lenihan, J.M.; Bachelet, D.; Neilson, R.P.; Drapek, R. 2008. Response of vegetation distribution, ecosystem productivity, and fire to climate change scenarios for California. Climatic Change. 87: S215–S230.

Lenihan, J.M.; Drapek, R.; Bachelet, D.; Neilson, R.P. 2003. Climate change effects on vegetation distribution, carbon, and fire in California. Ecological Applications. 13: 1667–1681.

Long, J.; Skinner, C.; Charnley, S.; Hubbert, K.R.; Quinn-Davidson, L.; Meyer, M.D. 2014. Post-wildfire management. In: Long, J.W.; Quinn-Davidson, L.; Skinner, C. N., eds. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. Gen. Tech. Rep. PSW-GTR-247. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 187–220. Chapter 4.3.

Loudermilk, E.L.; Scheller, R.M.; Weisberg, P.J.; Yang, J.; Dilts, T.E.; Karam, S.; Skinner, C.N. 2013. Carbon dynamics in the future forest: the importance of long-term successional legacy and climate-fire interactions. Global Change Biology. 19: 3502–3515.

Loudermilk, E.L.; Stanton, A.; Scheller, R.M.; Dilts, T.E.; Weisberg, P.J.; Skinner, C.; Yang, J. 2014. Effectiveness of fuel treatments for mitigating wildfire risk and sequestering forest carbon: a case study in the Lake Tahoe Basin. Forest Ecology and Management. 323: 114–125.

Lutz, J.A.; van Wagtendonk, J.A.; Franklin, J.F. 2009. Twentieth-century decline of large-diameter trees in Yosemite National Park, California. Forest Ecology and Management. 257: 2296–2307.

Lutz, J.A.; van Wagtendonk, J.A.; Franklin, J.F. 2010. Climatic water deficit, tree species ranges, and climate change in Yosemite National Park. Journal of Biogeography. 37: 936–950.

Lydersen, J.; North, M. 2012. Topographic variation in active-fire forest structure under current climate conditions. Ecosystems. 15: 1134–1146.

Lydersen, J.M.; North, M.P.; Knapp, E.E.; Collins, B.M. 2013. Quantifying spatial patterns of tree groups and gaps in mixed-conifer forests: reference conditions and long-term changes following fire suppression and logging. Forest Ecology and Management. 304: 370–382.

Mallek, C.; Safford, H.; Viers, J.; Miller, J. 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California. Ecosphere. 4: 153.

Manies, K.L.; Mladenoff, D.J. 2000. Testing methods to produce landscape-scale presettlement vegetation maps from the U.S. public land survey records. Landscape Ecology. 15: 741–754.

McCaffrey, S.M.; Olsen, C.S. 2012. Research perspectives on the public and fire management: a synthesis of current social science on eight essential questions. Gen. Tech. Rep. NRS-GTR-104. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 40 p.

McIntyre, P.J.; Thorne, J.H.; Dolanc, C.R.; Flint, A.L.; Flint, L.E.; Kelly, M.; Ackerly, D.D. 2015. Twentieth-century shifts in forest structure in California: denser forests, smaller trees, and increased dominance of oaks. Proceedings of the National Academy of Sciences of the United States of America. doi: 10.1073/pnas.1410186112.

McKelvey, K.S.; Johnston, J.D. 1992. Historical perspectives on forest of the Sierra Nevada and the Transverse Range of southern California: forest conditions at the turn of the century. In: Verner, J.; McKelvey, K.S.; Noon, B.R.; Gutierrez, R.J.; Gould, G.I., Jr.; Beck, T.W., eds. The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 225–246. Chapter 11.

McKenzie, D. 2010. The effects of climatic change and wildland fires on air quality in national parks and wilderness areas. Fire Management Today. 70: 26–28.

Meyer, M.; North, M.; Gray, A.; Zald, H. 2007. Influence of soil thickness on stand characteristics in a Sierra Nevada mixed-conifer forest. Plant and Soil. 294: 113–123.

Miller, C.; Urban, D.L. 1999a. Forest pattern, fire, and climatic change in the Sierra Nevada. Ecosystems. 2: 76–87.

Miller, C.; Urban, D.L. 1999b. A model of surface fire, climate and forest pattern in the Sierra Nevada, California. Ecological Modelling. 114: 113–135.

Miller, J.D.; Safford, H.D. 2012. Trends in wildfire severity 1984–2010 in the Sierra Nevada, Modoc Plateau and southern Cascades, California. Fire Ecology. 8: 41–57.

Miller, J.D.; Safford, H.D.; Crimmins, M.; Thode, A.E. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada. Ecosystems. 12: 16–32.

Molotch, N.P.; Blanken, P.D.; Williams, M.W. Turnipseed, A.A.; Monson, R.K.; Margulis, S.A. 2007. Estimating sublimation of intercepted and sub-canopy snow using eddy covariance systems. Hydrological Processes. 21: 1567–1575.

Morgan, T.A.; Brandt, J.P.; Songster, K.E.; Keegan, C.E., III; Christensen, G.A. 2012. California's forest products industry and timber harvest, 2006. Gen. Tech. Rep. PNW-GTR-866. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 48 p.

Nagel, T.A.; Taylor A.H. 2005. Fire and persistence of montane chaparral in mixed conifer forest landscapes in the northern Sierra Nevada, Lake Tahoe Basin, California. Journal of the Torrey Botanical Society. 132: 442–457.

National Forest Management Act [NFMA]. 1976. 16 U.S.C. 1600 (note).

North, M.; Brough, A.; Long, J.; Collins, B.; Bowden, P.; Yasuda, D.; Miller, J.; Suighara, N. 2015. Constraints on mechanized treatment significantly limit mechanical fuels reduction extent in the Sierra Nevada. Journal of Forestry. 113: 40–48.

North, M.; Collins, B.; Safford, H.; Stephenson, S. 2016. Montane Forests. In: Mooney, H.; Zavelta, E., eds. Ecosystems of California. Berkeley, CA: University of California Press: 553–578.

North, M.; Collins, B.M.; Stephens, S. 2012. Using fire to increase the scale, benefits, and future maintenance of fuels treatments. Journal of Forestry. 110: 392–401.

North, M.; Innes, J.; Zald, H. 2007. Comparison of thinning and prescribed fire restoration treatments to Sierran mixed-conifer historic conditions. Canadian Journal of Forest Research. 37: 331–342.

North, M.; Stine, P.; O'Hara, K.; Zielinski, W.; Stephens, S. 2009. An ecosystem management strategy for Sierran mixed-conifer forests. Gen. Tech. Rep. PSW-GTR-220. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 184 p.

Odion, D.C.; Hanson, C.T.; Arsenault, A.; Baker, W.L.; DellaSala, D.A.; Hutto, R.L.; Klenner, W.; Moritz, M.A.; Sherriff, R.L.; Veblen, T.T.; Williams, M.A. 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS ONE. 9:e87852.

Parsons, D.J.; Debenedetti, S.H. 1979. Impact of fire suppression on a mixed-conifer forest. Forest Ecology and Management. 2: 21–33.

Peterson, D.L.; Agee, J.K.; Aplet, G.H.; Dykstra, D.P.; Graham, R.T.; Lehmkuhl, J.F.; Pilliod, D.S.; Potts, D.F.; Powers, R.F.; Stuart, J.D. 2009. Effects of timber harvest following wildfire in western North America. Gen. Tech. Rep. PNW-GTR-776. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 51 p.

Peterson, D.W.; Dodson, E.K.; Harrod, RJ. 2015. Post-fire logging reduces surface woody fuels up to four decades following wildfire. Forest Ecology and Management. 338: 84–91.

Ritchie, M.W.; Knapp, E.E. 2014. Establishment of a long-term fire salvage study in an interior ponderosa pine forest. Journal of Forestry. 112: 395–400.

Ritchie, M.W.; Knapp, E.E.; Skinner, C.N. 2013. Snag longevity and surface fuel accumulation following post-fire logging in a ponderosa pine dominated forest. Forest Ecology and Management. 287: 113–122.

Ritchie, M.W.; Skinner, C.N.; Hamilton, T.A. 2007. Probability of wildfire-induced tree mortality in an interior pine forest: effects of thinning and prescribed fire. Forest Ecology and Management. 247: 200–208.

Ritchie, M.W.; Wing, B.M.; Hamilton, T.A. 2008. Stability of the large tree component in treated and untreated late-seral interior ponderosa pine stands. Canadian Journal of Forest Research. 38: 919–923.

Roberts, S.L.; Kelt, D.A.; van Wagtendonk, J.W.; Miles, A.K.; Meyer, M.D. 2015. Effects of fire on small mammal communities in frequent-fire forests in California. Journal of Mammalogy. 96: 107–119.

Safford, H.D.; North, M.; Meyer, M.D. 2012. Climate change and the relevance of historical forest conditions. In: North, M., ed. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 23–46. Chapter 3.

Scheller, R.M.; Spencer, W.D.; Rustigian-Romsos, H.; Syphard, A.D. 2011. Using stochastic simulation to evaluate competing risks of wildfires and fuels management on an isolated forest carnivore. Landscape Ecology. 26: 1491–1504.

Schmidt, D.A.; Taylor, A.H.; Skinner, C.N. 2008. The influence of fuels treatment and landscape arrangement on simulated fire behavior, southern Cascade Range, California. Forest Ecology and Management. 255: 3170–3184.

Scholl, A.E.; Taylor, A.H. 2010. Fire regimes, forest change, and self-organization in an old-growth mixed-conifer forest, Yosemite National Park. Ecological Applications. 20: 362–380.

Schwartz, M.W.; Butt, N.; Dolanc, C.R.; Holguin, A.; Moritz, M.A.; North, M.P.; Safford, H.D.; Stephenson, N.L.; Thorne, J.T.; van Mantgem, P.J. 2015. Increasing elevation of fire in the Sierra Nevada and implications for forest change. Ecosphere. 6(7): art121.

Sessions, J.; Bettinger, P.; Buckman, R.; Newton, M.; Hamann, A.J. 2004. Hastening the return of complex forests following fire: the consequences of delay. Journal of Forestry. 102: 38–45.

Show, S.B.; Kotok, E.I. 1924. The role of fire in the California pine forests. Bulletin No. 1294. Washington, DC: U.S. Department of Agriculture, Government Printing Office. 80 p.

Skinner, C.N. 2005. Reintroducing fire into the Blacks Mountain Research Natural Area: effects on fire hazard. In: Ritchie, M.W.; Maguire, D.A.; Youngblood, A., tech. coords. Proceedings of the symposium on ponderosa pine: issues, trends, and management. Gen. Tech. Rep. PSW-GTR-198. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 245–257.

Skinner, C.N.; Taylor, A.H. 2006. Southern Cascade bioregion. In: Sugihara, N.G.; van Wagtendonk, W.; Fites-Kaufman, J.W.J.; Shaffer, K.E.; Thode, A.E., eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 195–224.

Sleeter, B.M.; Wilson, T.S.; Soulard, C.E.; Liu, J. 2011. Estimation of late twentieth century land-cover change in California. Environmental Monitoring and Assessment. 173: 251–266.

Smith, T.F.; Rizzo, D.M.; North, M. 2005. Patterns of mortality in an old-growth mixed-conifer forest of the southern Sierra Nevada, California. Forest Science. 51: 266–275.

Spencer, W.D.; Rustigian-Romsos, H.; Strittholt, J.; Scheller, R.; Zielinski, W.J.; Truex, R.L. 2010. Using occupancy and population models to assess habitat conservation opportunities for an isolated carnivore population. Biological Conservation. 144: 788–803.

Steel, Z.L.; Safford, H.D.; Viers, J.H. 2015. The fire frequency-severity relationship and the legacy of fire suppression in California forests. Ecosphere. 6: 8.

Stephens, S.L.; Agee, J.K.; Fulé, P.Z.; North, M.P.; Romme, W.H.; Swetnam, T.W.; Turner, M.G. 2013. Managing forests and fire in changing climates. Science. 342: 41–42.

Stephens, S.L.; Bigelow, S.W.; Burnett, R.D.; Collins, B.M.; Gallagher, C.V.; Keane, J.; Kelt, D.A.; North, M.P.; Roberts, S.L.; Stine, P.; Van Vuren, D.H. 2014. California spotted owl, songbird, and small mammal responses to landscape-scale fuel treatments. Bioscience. 64: 893–906.

Stephens, S.L.; Gill, S.J. 2005. Forest structure and mortality in an old-growth Jeffrey pine-mixed conifer forest in north-western Mexico. Forest Ecology and Management. 205: 15–28.

Stephens, S.L.; Lydersen, J.M.; Collins, B.M.; Fry, D.L.; Meyer, M.D. 2015. Historical and current landscape-scale ponderosa pine and mixed conifer forest structure in the Southern Sierra Nevada. Ecosphere. 6: art79.

Stephens, S.L.; Martin, R.E.; Clinton, N.E. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands. Forest Ecology and Management. 251: 205–216.

Stephens, S.L.; Moghaddas, J.J. 2005. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. Forest Ecology and Management. 215: 21–36.

Stephenson, N.L. 1998. Actual evapotranspiration and deficit: biologically meaningful correlates of vegetation distribution across spatial scales. Journal of Biogeography. 25: 855–870.

Stevens, J.T.; Safford, H.D.; North, M.P.; Fried, J.S.; Gray, A.N.; Brown, P.M.; Dolanc, C.R.; Dobrowski, S.Z.; Falk, D.A.; Farris, C.A.; Franklin, J.F.; Fulé, P.Z.; Hagmann, R.K.; Knapp, E.E.; Miller, J.A.; Smith, D.F.; Swetnam, T.W.; Taylor, A.H. 2016. Average stand age from forest inventory plots does not describe historical fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS One. 11(5): e0147688.

Stewart, W.; Sharma, B.; York, R.; Diller, L.; Hamey, N.; Powell, R.; Swiers, R. 2016. Forestry. In: Mooney, H.; Zavaleta, E., eds. Ecosystems of California. Oakland, CA: University of California: 817–834.

Swanson, M.E.; Franklin, J.F.; Beschta, R.L.; Crisafulli, C.M.; Dellasala, D.; Hutto, R.L.; Lindenmayer, R.B.; Swanson, F.J. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment. 9: 117–125.

Syphard, A.D.; Scheller, R.M.; Ward, B.C.; Spencer, W.D. 2011. Simulating landscape-scale effects of fuels treatments in the Sierra Nevada, California. International Journal of Wildland Fire. 20: 364–383.

Tague, C.; Heyn, K.; Christensen, L. 2009. Topographic controls on spatial patterns of conifer transpiration and net primary productivity under climate warming in mountain ecosystems. Ecohydrology. 2: 541–554.

Taylor, A.H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California. Journal of Biogeography. 27: 87–104.

Taylor, A.H. 2004. Identifying forest reference conditions on early cut-over lands, Lake Tahoe Basin. Ecological Applications. 14: 1903–1920.

Taylor, A.H. 2010. Fire disturbance and forest structure in an old-growth *Pinus ponderosa* forest, southern Cascades. Journal of Vegetation Science. 21: 561–572.

Taylor, A.H.; Skinner, C.N. 1998. Fire history and landscape dynamics in a late-successional reserve in the Klamath Mountains, California. Forest Ecology and Management. 111: 285–301.

Taylor, A.H.; Skinner, C.N. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications. 13: 704–719.

Taylor, A.H.; Vandervlugt, A.M.; Maxwell, R.S.; Beaty, R.M.; Airey, C.; Skinner, C.N. 2014. Changes in forest structure, fuels and potential fire behaviour since 1873 in the Lake Tahoe Basin. Applied Vegetation Science. 17: 17–31.

Underwood, E.C.; Viers, J.H.; Quinn, J.F.; North, M. 2010. Using topography to meet wildlife and fuels treatment objectives in fire-suppressed landscapes. Journal of Environmental Management. 46: 809–819.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2001. Sierra Nevada forest plan amendment: final environmental impact statement. Vallejo, CA: Pacific Southwest Region. http://www.sierraforestlegacy.org/Resources/Conservation/LawsPoliciesRegulation/KeyForestServicePolicy/SierraNevadaFramework/Framework-FSROD01.pdf.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2004. Sierra Nevada forest plan amendment: final supplemental environmental impact statement. Vallejo, CA: Pacific Southwest Region. Final programmatic environmental impact statement: National Forest System land management planning. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev3_046095.pdf USFS. 2012.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2012. National forest system land management planning. Federal Register. 77. No. 68: 21162–21276.

U.S. Department of the Interiors, National Park Service [USDI NPS]. 2012. National Park System Advisory Board Science Committee. Revisiting Leopold: resource stewardship in the National Park System. Washington, DC: National Park System Advisory Board Science Committee 23 p. http://www.nps.gov/calltoaction/pdf/LeopoldReport_2012.pdf.

Vankat, J.L.; Major, J. 1978. Vegetation changes in Sequoia National Park, California. Journal of Biogeography. 5: 377–402.

Van de Water, K.M.; Safford, H.D. 2011. A summary of fire frequency estimates for California vegetation before Euro-American settlement. Fire Ecology. 7: 26–58.

van Mantgem, P.J.; Stephenson, N.L. 2007. Apparent climatically induced increase of tree mortality rates in a temperate forest. Ecology Letters. 10: 909–916.

van Mantgem, P.J.; Stephenson, N.L.; Byrne, J.C.; Daniels, L.D.; Franklin, J.F.; Fule, P.Z.; Harmon, M.E.; Larson, A.J.; Smith, J.M.; Taylor, A.H.; Veblen, T.T. 2009. Widespread increase of tree mortality rates in the Western United States. Science. 323: 521–524.

van Wagtendonk, J.W.; Fites-Kaufman, J.A. 2006. Sierra Nevada bioregion. In: Sugihara, N.G.; van Wagtendonk, J.W.; Shaffer, K.E.; Fites-Kaufman J.A.; Thode, A.E., eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 264–294.

van Wagtendonk, J.W.; Lutz, J.A. 2007. Fire regime attributes of wildland fires in Yosemite National Park. Fire Ecology. 3: 34–52.

Verner, J.; McKelvey, K.S.; Noon, B.R.; Gutiérrez, R.J.; Gould, G.I., Jr.; Beck, T.W., tech. coords. 1992. The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 285 p.

Walsh, D.; North, M. 2012. Appendix: examples of forest structures that may provide wildlife habitat. In: North, M., ed. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 177–184.

Weatherspoon, C.P.; Husari, S.J.; van Wagtenkonk, J.W. 1992. Fire and fuels management in relation to owl habitat in forests of the Sierra Nevada and southern California: In: Verner, J.; McKelvey, K.S.; Noon, B.R.; Gutiérrez, R.J.; Gould, G.I., Jr.; Beck, T.W., tech. coords. 1992. The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 2470–260.

Westerling, A.; Bryant, B.; Preisler, H.; Holmes, T.; Hidalgo, H.; Das, T.; Shrestha, S. 2011. Climate change and growth scenarios for California wildfire. Climatic Change. 109: 445–463.

Wiebe, K.L. 2011. Nest sites as limiting resources for cavity-nesting birds in mature forest ecosystems: a review of the evidence. Journal of Field Ornithology. 82: 239–248.

Yang, J.; Weisberg, P.J.; Dilts, T.E.; Loudermilk, E.L.; Scheller, R.M.; Stanton, A.; Skinner, C. 2015. Predicting wildfire occurrence distribution with spatial point process models and its uncertainty assessment: a case study in the Lake Tahoe Basin. International Journal of Wildland Fire. 24: 380–390.

Zhang, J.; Webster, J.; Powers, R.F.; Mills, J. 2008. Reforestation after the Fountain Fire in northern California: an untold success story. Journal of Forestry. 106: 425–430.

Appendix 5-1: Information on Source and Data Quality Issues for Timber Harvest Volume, Silvicultural Prescriptions, and Habitat Data

Introduction

Assessing past trends and current status of timber harvest volume, number of treated acres, and predominant silvicultural prescriptions used, and the distribution and amounts of important California spotted owl (*Strix occidentalis occidentalis*) habitat types across the Sierra Nevada are a fundamental component for evaluating conservation status. This appendix identifies the data sources for the information summarized in chapter 5 for the above metrics. As described in the following sections, each data set has strengths and limitations as to completeness or accuracy of the data that must be considered when drawing inferences. Nevertheless, these data provide the sole sources of currently available data that provide valuable insight and information on trends and status of forest management treatments and habitat status.

Trends in Timber Volume Harvested From the Sierra Nevada: 1994–2013

Annual summaries of timber volume harvested from public and private lands by county in California are available from the California State Board of Equalization, Timber Tax Program, 2014. Annual summaries are available for 1994–2013, consisting of nonspatial, tabular data reporting annual timber volume harvested in thousands of board feet by county. Counties were filtered to include only those that intersect any portion of the California spotted owl range in the Sierra Nevada as determined using the species distribution map maintained by the California Wildlife Habitat Relationships Program, California Department of Fish and Wildlife. Because the timber volume data are nonspatial, some portion of the volume was harvested from county areas outside of the range of the California spotted owl.

Patterns in Silvicultural Prescriptions on National Forest Lands: 1990–2014

Prior to 2002, forest management treatments on national forest lands were tracked using the Stand Record (SRF) system. The SRF was a nonspatial, tabular database that recorded acres treated by silvicultural prescription by U.S. Department of Agriculture Forest Service (USFS) management unit (national forest and ranger district). Beginning in 2002, the USFS switched to use of the Forest Activity C Tracking System (FACTS) system for recording forest management treatments and activities. The FACTS is a spatial database that records the footprint of timber management activities as well as acres treated by silvicultural prescription by management unit. Efforts have been made to generate and incorporate spatial data for forest treatments conducted prior to 2002, but not all projects have been entered into the database, and some unquantified proportion of the total area treated prior to 2002 is not spatially mapped. Thus, available information on USFS treatments consists of a complete tabular, nonspatial summary of activities from 1990 through 2014 by national forest and ranger district (though October 2014). Spatial data on treatment type, amount, and location are complete from 2002 through 2014. Spatial data are incomplete for 1990–2001 and include some unquantified proportion of the actual activities.[1] The nonspatial data provides insight into the acres treated by silvicultural prescription and trends in the use of different silvicultural prescriptions over time during 1990–2014 on national forests that intersect any portion of the range of the California spotted owl in the Sierra Nevada (Sequoia, Sierra, Stanislaus, Inyo, Eldorado, Tahoe, Plumas, and Lassen National Forests, and the Lake Tahoe Basin Management Unit). Treated acres on some national forests are located outside of the range of the spotted owl so not all treatments occurred within the range of the owl. Numbers reported consist of the number of acres accomplished. The spatial data provide opportunity to assess treatment acres and silvicultural prescriptions used within the range of the California spotted owl in the Sierra Nevada as determined by using the species distribution map maintained by the California Wildlife Habitat Relationships Program, California Department of Fish and Wildlife. However, the spatial data are incomplete for 1990–2002, and thus summaries based on the spatial data do not include all acres treated during the 1990–2002 period.

Patterns in Silvicultural Prescriptions on Private Industrial Forest Lands: 1990–2013

Information on acres treated by silvicultural method on private industrial timberland and nonindustrial private lands is available in the CALFIRE Forest Practice Database managed by the California Department of Forestry and Fire Protection. Nonspatial, tabular data are available to assess acres by silvicultural prescription by county for the 1990–2013 time period. Counties were filtered to include only those that intersect any portion of the California spotted owl range in the Sierra Nevada as determined using the species distribution map maintained by the California

[1] **Sherlock, J. 2015.** Personal communication. regional silviculturalist, USDA Forest Service, Pacific Southwest Research Region.

Wildlife Habitat Relationships Program, California Department of Fish and Wildlife. Because the timber volume data are nonspatial, some portion of the volume was harvested from county areas outside of the range of the California spotted owl. Spatial data on private industrial forest land silvicultural treatments are available for the 1997–2013 period. This database includes all acres approved and completed for treatment under all timber harvest plans (THPs) approved beginning in 1997 and extending through 2013. However, this database does not include acres that were approved in THPs prior to 1997, yet the actual on-the-ground projects were conducted after 1997. A review of the 1997–2013 database indicates that most treatments are completed 4 to 6 years after approval, but that many acres are not reported as completed until 6 to 12 years after approval. Thus, the spatial data include all acres approved/or completed for 1997–2013 THPs, but more acres were actually treated then are shown because of pre-1997 THP acres not being included in the database.

Status and Trends in California Spotted Owl Habitat in the Sierra Nevada

The only source of information on the current distribution and abundance of California spotted owl habitat across the owl's range in the Sierra Nevada is provided by the existing vegetation classification and mapping (EVEG) map maintained by the Remote Sensing Laboratory, Pacific Southwest Region, USFS. The EVEG map stiches together map products developed using different imagery and methods at the national forest and national park scale to provide a bioregional-scale map product of habitat across the Sierra Nevada. No formal accuracy assessments have been conducted to validate the map across the bioregion or to resolve differences in habitat classifications resulting from different mapping approaches using different imagery at different spatial and temporal scales. Thus, inferences about habitat amounts and distributions should be tempered until formal accuracy assessments are completed to validate map accuracy and consistency across the Sierra Nevada. Nevertheless, these data provide the sole source of information on current amounts of California spotted owl habitat across the Sierra Nevada.