Past, Current, and Future Conditions

Notes:

1. Lots of this information is summarized quite well in other locations, so I've tried to restrict the summary here to variables/traits that are directly tied to Owl needs/risks (ie canopy cover, large trees, heterogeneity snags, coarse woody debris, fuels, fire), and to the most condensed summary possible
2. Lots of the below is plagiarized and needs some editing
3. Currently, this largely only includes yellow pine and mixed conifer veg types - Any info anyone wants to add relevant to other owl forest types would be much appreciated
4. The owl sections are highly lacking right now, and we're in the process of getting input!
5. The Table summarizing numbers at the end needs a lot more values, in some cases we may not have values and might need to delete rows- input on that would be helpful

Environmental Context

Mid-Elevation Sierra Nevada forests are highly productive and have some of the highest biomass values for temperate forests worldwide. Their component tree species are long-lived, achieving a size that produces tall, complex canopy structures that significantly influence understory microclimate and habitat conditions. Additionally, Sierra Nevada forests are fire-dependent ecosystems. Historically, frequent (generally < 20 years), low-moderate intensity (generally surface with localized higher-severity patches) fire reduced stem density and moisture stress in these drier environments while increasing spatial, habitat and microclimate heterogeneity. Early surveys noted that fire produced variable but generally low-density forest conditions with one source (Lieberg 1902) lamenting Sierra Nevada forests were only 30% of their carrying capacity for timber production. Over the last few decades, research has demonstrated that more stable, resilient forests depend on lower density more open conditions than the traditional concept of full stocking for maximum timber production [Collins et al. 2011, 2015, Stephens et al. 2015].

The Sierra Nevada forests are unique relative to their closest neighbors (the moist forests of the Pacific Northwest and the drier forests of the Southwest), partially a function of the combination of these two characteristics in one ecosystem. Like the Pacific Northwest, overstory forest conditions are shaped by the number, size, and composition of large, long-lived trees. These conditions are influenced by local levels of productivity. In particular, areas with higher soil moisture availability support more large structures (live and dead), denser canopy cover and greater biomass. Understory conditions, however, like the Southwest, are strongly influenced by local fire regime, with tree regeneration dynamics, stand density, shrub cover and microclimate conditions affect by the frequency, intensity and spatial extent of burn patterns (Knapp et al. 2013, Collins et al. 2015). Because fire is a frequent and keystone process shaping the system in the Sierra Nevada, old forests there generally exhibited spatial segregation of different canopy strata (Stephens and Gill 2005), which reduces crown fire potential. This is very different from the stand structural measures many consider indicative of old-growth forests elsewhere (like the Pacific Northwest), such as multi-layer canopies. The Sierra Nevada ecosystem condition cannot be assessed by the abundance and size of forest structures alone, but needs to strongly consider fire history including severity, frequency and patch structure (Collins and Stephens 2010).

Stand and landscape patterns of forest conditions appear to be generally influenced by local rates of actual evapotranspiration (AET) and climatic water deficit (CWD). AET is a measure of how much water actually transpired and consequently potential tree growth and size. Climatic water deficit is a measure of the difference between potential and actual evapotranspiration and consequently an indirect measure of a site's moisture stress (i.e., how 'dried out'). AET has been significantly correlated with the abundance of large tree structures (live trees, snags and large logs), canopy cover and biomass. CWD has been generally correlated with fuel moisture conditions and therefore indirectly with local fire regimes, although the association is not as strong as AET's correlation with large tree biomass. Local fire regimes most directly influences understory conditions such as shrub cover and composition, small tree density, and surface soil substrates (i.e., litter and bare ground conditions that influence germination success of different species). In general AET may be an improvement over topographic categories in predicting historic forest condition associated with large trees because it is one step closer to a more mechanistic understanding of ecosystem processes. Areas of low productivity can be generally identified through their association with low AET. Frequent fires locations may be roughly associated with high CWD, but physiographic characteristics (slope position, aspect, steepness, etc.) may be a more direct measure of factors that affect fire occurrence and intensity. However, historic forest conditions were also influenced by other factors such as tree killing insects, disease, and wind-throw, as well as the spatial and temporal variability inherent in disturbance events.

Current Conditions Relative to Historic Conditions

Driving Forces

In general the two strongest influences on current forest conditions in the Sierra Nevada are logging and fire suppression over the last 100 years. Logging often removed the largest trees and preferentially selected pines over fir and cedar (Laudenslayer and Darr 1990, Stephens 2000). Forest management also removed 'defect' trees (i.e., broken tops, multiple leader, mistletoe-infested, etc.), which had characteristics associated with preferred habitat for some sensitive species such as the California spotted owl. In general logging and forest management practices reduced stand structures (large trees, snags and logs, and defects) associated with old-forest conditions.

The effects of fire suppression are well documented and are influenced by the highly productive conditions of the Sierra Nevada. Small trees rapidly in-filled understory conditions and with enough time (generally > 40 years) grew to intermediate and then co-dominant size in many stands (Parsons and Debenedetti 1979). This often eliminated the spatial heterogeneity (i.e., ICO pattern), reduced species diversity (as the number of fire-sensitive, shade-tolerant stems increased), structural diversity (variability in tree size and canopy position) and understory variability in microclimate, substrate and habitat conditions. In general fire suppression has homogenized forest structure in the Sierra Nevada.

McKelvey and Johnston (1992) highlight four key resulting changes in forests since 1850: 1) loss of old, large-diameter trees and associated large downed logs; 2) shift in species composition towards shade-tolerant; 3) increase in fuel associated with mortality of smaller trees; and 4) presence of ladder fuels that facilitate crown fire. Similarly, Franklin and Johnson (2012) outline four significant changes seen in forests over the last century: (1) many fewer old trees of fire-resistant species, (2) denser forests with multiple canopy layers, (3) more densely forested landscapes with continuous high fuel levels, and, consequently, (4) more stands and landscapes highly susceptible to stand-replacement wildfire and insect epidemics.

These changes generally make current forests less resilient to the two most common disturbances in the Sierra Nevada, fire and drought. Fuel load, ladder fuels and crown connectivity increase the likelihood of high-intensity crown fire occurrence and extent with surface and ladder fuels the most hazardous and in need of treatments (Agee and Skinner 2005, Stephens et al. 2009). Stands with uniformly distributed high tree density (i.e., without gaps), particularly in areas with low soil moisture holding capacity (for example shallow soils) are susceptible to drought stress that increases the likelihood of pest and pathogen damage and mortality, particularly from bark beetles. This decrease in resilience is likely most significant in locations where low productivity and/or frequent fire historically kept forests generally at a low density and with a higher percentage of drought and fire-resistant pines. Below, changes in forest structure and composition, and essential disturbance processes are described. A summary of information on California Spotted Owl populations relative to past conditions is also included.

Climate

Over the last 7000 to 8000 years, dry climate periods have occurred on average every 80 to 260 years, with durations of droughts lasting 20 to 100 years on many occasion (Safford and Stevens 2015). While the 19th and 20th centuries have been anomalously wet (Haston and Michaelsen 1997, Hughes and Brown 1992, Safford and Stevens 2015)), climate-related forest mortality is on the rise (Allen et al 2010), suggesting that changes to forest structure, composition, and function over the last century place these forests, and particularly larger and older trees on which many wildlife species depend, at high risk for drought stress and mortality (van Mantgem et al. 2013).

Forest Conditions

A defining characteristic of historic Sierra Nevada forests was **heterogeneity** in distribution, density and species composition (North et al. 2009, Collins et al. 2015). Recent studies have quantified the distributional heterogeneity of frequent-fire forests as characterized by a pattern of individual trees, clumps of trees and openings (ICO) (Fry et al. 2014). The proportion of area in each of these conditions, the tree basal area and size of opening likely varied with local differences in productivity and fire regime often associated with topography. Drier conditions associated with upper slope, ridge top and southwest aspects likely had smaller trees clumps, larger openings, lower basal area and a higher percentage of pine species. In contrast, more mesic locations such as lower slope and valley bottom sites more often supported large tree clumps, higher canopy closure, smaller openings and a higher percentage of fire-sensitive, shade-tolerant species such as fir and cedar.

This heterogeneity was likely linked to forest resistance and resilience to disturbance. For example, gaps in ICO dominated forests under moderate fire weather may limit the patch size of high-intensity crown fires. Spatial variability may also have made forests more drought resilient because trees in clump had adjacent openings and areas of low density individual trees that reduced moisture competition and subsequent susceptibility to bark-beetle damage and mortality. Likewise the extent and severity of some pathogen and pest damage can be limited by variability in species and spatial composition because some of pests and pathogens are host-specific and influenced by overall stand density.

Forest stands at fine (stand and sub-stand) scales are more homogeneous today than historically, with less patchy patterns of tree size and density (Agee 1993, Barbour et al. 1993, 2007; SNEP 1996, Sugihara et al. 2006), increased tree clump size (Lydersen et al. 2013), and decreased proportion in canopy gaps (Lydersen et al. 2013). Forest structure has also been 'simplified', including declines in large trees, snags, woody debris of large diameter, canopies of multiple heights and closures, and complex spatial mosaics of vegetation (SNEP 1996, Safford and Stevens 2015). This has likely dramatically decreased forest resistance and resilience to disturbance.

Reconstructed **tree densities** from presettlement conditions range from 60 to 328 trees/ha (24 – 132 trees/ac), with an average of 159 trees/ha (64 trees/ac; Safford and Stevens 2015, Taylor 2004, Scholl and Taylor 2010, Collins et al. 2011, 2015, Stephens et al. 2015). Contemporary mean tree density is 397 trees/ha (160 trees/ac), with densities ranging from 238 to 755 trees/ha in stands for which presettlement reconstructions exist. Increases in forest density range from 80% to 600% (Safford and Stevens 2015). Most of this increase is in trees <60 cm dbh (Safford and Stevens 2015). Historically, the yellow pine and mixed-conifer forest types were characterized by higher densities of large trees and lower densities of small trees than today, with about the same overall basal area (Dolanc et al. 2014) although some studies report lower basal areas in historical periods in ponderosa pine forests (Stephens et al. 2015). Trees 24-36 in dbh, and especially trees >36 in dbh, have declined in abundance, and trees <24 in dbh have increased (Verner et al. 1992, North et al. 2007, Fellows and Goulden 2008, Lutz et al. 2009, Scholl and Taylor 2011, Dolanc et al. 2014, McIntyre et al. 2015, Stephens et al. 2015).

Average and maximum **tree sizes** have declined relative to historic conditions. The lack of fire and moderated microclimate have decoupled the mortality and regeneration processes from burn and climatic conditions, changing the age and size structure from one with a more even distribution to one weighted toward young, smaller trees. In terms of diameter, this changes the distribution (size [x axis] plotted against frequency) from a fairly flat slope to one more closely approximating a reverse-J. In silvicultural terms this change broadly indicates a forest shift from a diversified structure largely controlled by disturbance (fire) to a forest approaching maximum carrying capacity and controlled by resource limitation (competition for water and/or light and insect mortality). Taylor et al. (2014) found that average tree size in current forests was only about 60% of average tree size in 1873, and Lydersen et al. (2013) found that average tree size on the Stanislaus National Forest had declined 26% since 1929. Modern FIA data indicates an average tree diameter of 26 cm, and a quadratic mean diameter of 32 cm. Although the exact size threshold above which larger trees are in deficit varies among places, trees >36 in dbh are in deficit throughout the Sierra Nevada-Cascade Region (Dolanc et al. 2014) and trees between 24 and 36 inches dbh are more common today than historically in some areas (Stephens et al. 2015). While timber harvest explains some of these declines, similar patterns also occur in unlogged forests, suggesting that other factors are at play. These might include insects, pathogens, and drought stress, likely exacerbated by the much higher stand densities in modern forests (Safford and Stevens 2015).

Recent studies have documented high **mortality rates** of trees throughout the Sierra Nevada (van Mantgem et al. 2009), including higher than expected and accelerating rates of loss of the largest size classes (e.g., >36 in dbh, Smith et al. 2005, Lutz et al. 2009, Fellows and Goulden 2012, McIntyre et al. 2015). Historically, mortality was primarily driven by fire, which selects for smaller tree sizes and fire-sensitive species. As a recurring event, small trees only survive and grow large enough to escape this mortality cycle through the stochastic nature of fire frequency and extent (Stephens et al. 2008). Some areas are randomly missed by fire and other microsites are less likely to burn due to mesic conditions or fuel barriers (i.e., streams and rocks). In general this produced forests characterized by a low-density of large trees because while few individuals escape the cycle of fire-driven mortality, those that do may thrive in conditions with reduced water and light competition, producing large, long-lived trees. The increasing mortality of large trees is suspected to reflect effects of climate change, drought, and water stress (Fellows and Goulden 2008, Lutz et al. 2009, McIntyre et al. 2015) in interaction with multiple other factors, including pathogens, insects, and air pollution (Guarin and Taylor 2005, Smith et al. 2005, Das et al. 2011, McIntyre et al. 2015). In particular, there has been a recent dramatic increase in loss of large trees due to bark beetles, which are currently considered one of the principal agents of tree mortality in the Sierra Nevada (Fettig 2012). Tree stand densities have a strong relationship to bark beetle-induced mortality. Higher density stands suffer increased competition for resources (especially water and light) and reduced tree vigor, which makes individual trees less able to withstand insect attack (Safford and Stevens 2015). Lower density stands are much less susceptible to bark beetle attack and subsequent mortality (Fettig et al. 2007). The few data available on tree mortality rates suggest that background rates today are higher than historically. Background mortality rates (averaged over multiple years) in the Sierra-Cascade forests are between about 0.25% and 1.4% for fire-excluded forests but less than 0.5% for contemporary reference forests with a largely intact fire regime (In Safford and Stevens 2015: Ansley and Battles 1998, Maloney and Rizzo 2002, Stephens and Gill 2005).

Average **canopy cover** for historical forests has been estimated at anywhere from 17% to 49% (Safford and Stevens 2015, Collins et al. 2011, Lydersen and North 2012, Collins et al. 2015, Stephens et al. 2015) with many studies reporting canopy cover below 35%. Current conditions represent an increase in average canopy cover of around 25% (Safford and Stevens 2015) to about 46%. Models predict that most of the landscape was historically in open conditions of

This seems low. I think I've seen data that suggests >55% but of course this varies with site and forest type.

less than 40% canopy cover, especially in the yellow pine and dry mixed conifer types (Safford and Stevens 2015) and that dense, older stands occupied around 5% of the landscape in the yellow pine and dry mixed conifer types, and around 20% of the moist mixed conifer type (Safford and Stevens 2015). CURRENT COVERAGE?

Basal area estimates generally ranged from 21 m2/ha to 54 m2/ha depending on site productivity, with a mean of 35 m2/ha (Safford and Stevens 2015). Current FIA data suggest that mean basal area has not changed significantly over the last century (Safford and Stevens 2015) a result to two countervailing trends, increasing tree density and decreasing average tree size.

Sierra-Cascade Forests have seen **increases in snag density, coarse woody debris, litter and duff depth, and surface fuel volume and continuity** (Safford and Stevens 2015). Current trends in snag dynamics suggest that snags are more abundant but significantly smaller than historical conditions (Knapp 2015). Current snag densities (>15 cm dbh) average about 20 – 50 snags/ha (~8- 20 snags/ac) (Safford and Stevens 2015, Stephens et al. 2007, Younglood et al. 2004, Dunbar-Irwin and Safford in Review), while historic average densities likely ranged from 4 to 12 snags/ha (~1.6 – 5 snags/ac) (Stephens 2004, Stephens et al. 2007, Dunbar-Irwin and Safford in review). Agee (2002) suggested that a Fire Regime I forest should support around 5 snags/ha (~2 snags/ac), with the average snag size about 75 cm dbh (30 in).

Forest composition has shifted from historic conditions, with declines in abundance of shade-intolerant pines and increases in shade-tolerant species like firs and cedars (Barbour et al. 2002, Guarin and Taylor 2005, Dolanc et al. 2014, McIntyre et al. 2015, Stephens et al. 2015). Reduced understory light and thick litter layers favor regeneration of fire-sensitive, shade-tolerant species. Relative proportions of shade intolerant to shade tolerant species changed from 60:40 to 35:55 between 1930s and 2000s (Safford and Stevens 2015). In other words, the component of shade intolerant species like yellow pines has dropped from about 2/3 of the mature forest stand to about 1/3 of mature forests over the last century (Safford and Stevens 2015). In some areas, pine forests have been replaced by mixed-conifer forests. Dolanc et al. (2014), found that 19.7% of 1930s plots were classified as ponderosa pine, versus just 8.9% of the plots from the 2000s; 27.4% of plots were classified as mixed conifer in the 1930s dataset, versus 37.1% in the 2000s; and both eastside and westside Jeffrey pine also declined as a proportion of the sampled vegetation between the 1930s and 2000s. Current Forest Service vegetation maps show 17% of the region in yellow pine and 30% in mixed conifer forests, compared to 33.7% in yellow pine and 19.8% in mixed conifer in Show and Kotok's (1929) summary of 1920s conditions (Safford and Stevens 2015). It may be more useful to separate mixed conifer forests into pine dominated mixed conifer (Collins et al. 2011, 2015, Stephens et al. 2015) and fir dominated mixed conifer forests (Stephens and Collins unpublished data from the El Dorado National Forest). The pine dominated mixed conifer forests had lower tree densities, canopy cover, and were dominated by shade intolerant species versus the fir dominated areas that had higher tree densities, tree basal area, and were dominated by fir.

Fire and Ecological Function

Fire, a key ecological process in the Sierra-Cascade region, has changed significantly over the last century. Yellow Pine/Mixed Conifer forests historically supported **fire regimes** characterized by frequent, low to moderate (or "mixed") severity fires (from Safford and Stevens 2015: Agee 1993, Arno 2000, Barbour et al. 2007, Barbour et al. 1993, Skinner and Taylor 2006, van Wagtendonk and Fites-Kaufman 2006). Mean Fire Return Intervals (FRIs) for yellow pine and mixed conifer forests across California ranged from 11 to 16 years (Stephens et al. 2007, Van de Water and Safford 2011, Safford and Stevens 2015). Historic fire frequencies were highest in the drier, lower elevation forest types (yellow pine and dry mixed conifer) and lower in moister and higher elevation stands (In Safford and Stevens 2015: Caprio and 1106 Swetnam 1995, Fites-Kaufman et al. 2007, Gill and Taylor 2009, Sugihara et al. 2006, Taylor 2000). Today, most pine/conifer forests in the central and northern portions of Sierra-Cascade range are more than 85% departed from historic fire return intervals (i.e. have seen zero to one fire in the last century; Safford and Van de Water 2014) and most in the eastern and southern portion of the region are at least 67% departed from historic FRI (ie three or fewer fires over the last century; Safford and Van de Water 2014). Historic fire rotation for the pine/conifer forests ranged from 22 to 31 years (Mallek et al. 2013), while current fire rotation on USFS managed pine/conifer forests averages 258 to 280 years (range 95 – 516; Miller et al. 2012b), and about 55 years in Yosemite National Park. In other words, fire rotations are about 10 times longer than historically on Forest Service lands, but only about twice as long in Yosemite (Miller et al. 2012b, Safford and Stevens 2015).

Historically, **high severity** likely represented a very small proportion of area burned. Mallek et al. (2013) indicate that 5-10% of any burn at any given time would have been high severity. Stephens et al. (2015) suggest and even lower proportion of high severity fire in the southern Sierra Nevada (1-3% in mixed conifer and 4-6% in Ponderosa Pine forest). These high severity areas were likely aggregated in small patches (usually <5 acres) distributed across the landscape (Show and Kotok 1924, Collins and Stephens 2010; North et al Assessment).

Recent decades have seen **increases in both overall proportion and patch sizes of stand-replacing fire** compared to historic conditions (Mallek et al. 2013, Stephens et al. 2013, Stephens et al. 2014, North et al. Assessment). The proportion of stand-replacing fires and burn patch sizes also have been increasing in the Sierra Nevada from 1984–2010 (Miller et al. 2009, Miller and Safford 2012, Steel et al. 2015), with the average fire in modern mixed-conifer and yellow pine forests on USFS lands supporting 5 to 7 times more area of stand-replacing fire than fires before Euro-American settlement (29-35% high severity; Miller et al. 2009; Miller and Safford 2012; Mallek et al. 2013; Safford 2013). Recent fires in the Sierra Nevada have included some huge patches of stand-replacing fire, extending for thousands or even tens-of-thousands of acres. This is in direct contrast to the size of stand-replacing patches from active fire regime forests in reference landscapes of the Sierra Nevada (areas where the fire regime is minimally influenced by humans), where mean stand-replacing patch size is <4 ha (10 ac) and maximum patch size generally is ≤100 ha (250 ac) (Collins and Stephens 2010; Miller et al. 2012; Safford 2013). Large, contiguous areas of severe fire can result in the long-term replacement of conifer forest by shrubs, which are maintained by subsequent fires (Willken 1967; Biswell 1974; Bock and Bock 1977). Recent studies also suggest that high severity re-burns are likely in these areas of initial high severity fire. Under fire weather conditions, the high densities of snags, down woody debris, and shrubs that result from initial high severity burns are driving factors in high severity re-burns (Coppoletta et al in press).

Average fire size in California mixed-conifer forests before Euro-American settlement has been estimated at <300 ha (750 ac), while the average over the last 25 years is closer to 1,500 ha (3,750 ac); and recent fires on USFS lands in California are much larger than that (Show and Kotok 1923; Taylor and Skinner 1998; Minnich et al. 2000; Taylor 2000; Beaty and Taylor 2001; Taylor and Solem 2001; Collins and Stephens 2007; Miller et al. 2012; Safford and Stevens 2015; A. Taylor, Pennsylvania State University, unpublished data).

something more about **pests and pathogens**? Warming temperatures have triggered population increases in many insect species which have served as catalyst for widespread outbreaks. (Millar and Stephenson 2015)It would be good to integrate this back into the above sections about beetles and mortality rates. I've kept an eye out for several years looking for an info on historic levels of these in the Sierra without much luck. I do think we might take a higher level approach to this and point out that mortality is a strong influence on forest dynamics and stand structure and that fundamentally its shifted from mostly fire caused (selecting by small size and species) to beetle (selected by density and large size)

California Spotted Owl Populations

Genetic diversity is lower in the California Spotted Owl than the other subspecies of Spotted Owl. Tempel et al (Assessment Ch 4.) note three potential explanations for this: persistently small populations, population bottlenecks, and recent colonization followed by population expansion (Barrowclough et al. 1999, 2005). The heterozygosity level reported in Funk et al 2008a (0.685) is typical of wild populations and not different than expected, and thus does not seem symptomatic of a population bottleneck. Therefore, two main explanations for the lower genetic diversity seem more likely: persistently small populations or recent colonization followed by population expansion. Current information cannot discern between the two possible explanations. *The first would suggest that CSO populations in California were historically small. The current population may represent an increase from this historically small population (potentially due to recent habitat changes described above), status quo compared to historic population numbers (?), or further minor reductions (ie not bottleneck) from a historically small population (this seems unlikely, right?). Any evidence for the second explanation? Recent colonization as habitat changed relative to historic conditions?*

Evidence suggests that the current California Spotted Owl population is experiencing population declines (Tempel et a. 2014; other refs in Assessment). How these declines compare to historic population sizes/densities remains largely unknown. *Do we have an overall estimated range/size of the total CSO population that we can include here?*

Future Conditions

Needs further development.

needs help from Zach – derived from info summarized in Conservation Assessment

Predictions of future forest conditions should always be viewed with caution because of large uncertainties in how complex ecosystems may respond to potentially novel climatic and disturbance conditions and their interaction. While all the climate models agree upon increasing temperatures, predictions about changes in precipitation distribution and amount are highly variable. What all the models do agree upon is that a larger percentage of precipitation will occur as rain rather than snow and that year-to-year variability will increase. Because of these trends, models consistently suggest the frequency and strength of drought events will increase, likely making them a stronger influence on forest dynamics. This change in precipitation variability and form, coupled with increasing temperatures is why all models also suggest an increase in fire frequency, size and severity.

If current forest conditions (i.e., often high density, fuel-loaded stands) continue into the future, coupled with increasing disturbance frequency and severity, some general patterns in future forest conditions are suggested. One study simulated these changes in disturbance and compared historic (low density, pine dominated) and current (high density, fir dominated) forest response. As disturbance frequency and severity increased current forest conditions became unstable and in a large portion of the simulations shifted toward a high density, small tree size condition. Historic forest conditions were much more stable, generally perpetuating a low-density, large-pine dominated condition in most scenarios unless severe disturbances occurred consecutively. What influences these outcomes result from how forest growth models (in this case the Forest Vegetation Simulator [FVS]) simulate tree regeneration and mortality dynamics. High density, fuel loaded conditions tend to increase mortality and eventually reduce large trees and their associated structures (large snags and logs) abundance. Regeneration dynamics are harder to predict because disturbance timing, severity, climate and seed dynamics all interact. Generally, however, under more unstable conditions, species with the largest, most consistent seed production (i.e., white fir and incense cedar) tend to be favored. Millar and Stephenson (2015) found that interactions from increasing temperature, drought, native insects and pathogens, and uncharacteristically severe wildfires are resulting in forest mortality beyond the levels of 20th-century experience. Large areas of the southern Cascades and Sierra Nevada forests are likely to experience uncharacteristic stand-replacement fires without active fuel treatments and prescribed burn programs, with the resulting loss of critical watershed and habitat for California spotted owl and other endangered wildlife. Substantial restoration efforts will be needed to protect them (Agee and Franklin 2003, North et al. 2012).

Elevated surface fuels created by high severity burned areas can constitute a significant risk to the succeeding stand (Agee and Skinner, 2005), and van Wagtendonk et al (2012) found that high severity burn patches were perpetuated by subsequent fires. At the landscape scale, this may drive an increase in chaparral as high severity patches are converted from their initial vegetation type providing few opportunities to recreate late seral forest habitat for core nesting and roosting area due to lost habitat.

Although controversial some researchers have suggested fire-suppressed forests are an example of hysteresis, where an ecosystem enters an alternative stable state and requires a significant, guided perturbation to return to its historic range of variability. Regardless of whether Sierra Nevada forests strictly meet this definition, the conceptual model is useful for considering potential future forests conditions. Models and some empirical evidence suggest the potential for high-severity fire to perpetuate high severity effects in future fires. Likewise if increasing drought severity and frequency reduces large tree abundance, forests stands may cycle through a more truncated succession where high-density, small size dominated conditions are perpetuated. These cycles may be difficult to break out without either an approximation of historic disturbance patterns or management manipulation of forest conditions.

References

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

Collins, B.M., and S.L. Stephens. 2010. Stand-replacing patches within a mixed severity fire regime: quantitative characterization using recent fires in a long-established natural fire area. Landscape Ecology 25:927939.

Collins, B.M., R.G. Everett, and S.L. Stephens. 2011. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. Ecosphere 2(4):art51.

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.

Fry, D.L., S.L. Stephens, B.M. Collins, M.P. North, E. Franco-Vizcaino, and S.J. Gill. 2014. Contrasting Spatial Patterns in Active-Fire and Fire-Suppressed Mediterranean Climate Old-Growth Mixed Conifer Forests. PLOS ONE 9(2): e88985.

Knapp, E. E., C. N. Skinner, M. P. North, and B. L. Estes. 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.

Laudenslayer, W.F., Darr, H.H., 1990. Historical effects of logging on forests of the Cascade and Sierra Nevada Ranges of California. Trans. West. Sect. Wildl. Soc. 26: 12–23.

North, M. P. Stine, K. O'Hara, W. Zielinski, and S. Stephens. 2009. An Ecosystems Management Strategy for Sierra Mixed-Conifer Forests. US Dept. Agriculture Forest Service Pacific Southwest Research Station. General Technical Report PSW-GTR-220 w/ addendum. 52 pages.

North, M.P., B.M. Collins, and S.L. Stephens. 2012. Using fire to increase the scale, benefits and future maintenance of fuels treatments. Journal of Forestry 110(7):392-401.

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

Stephens, S.L. 2000. Mixed conifer and upper montane forest structure and uses in 1899 from the Central and Northern Sierra Nevada, CA. Madrono 47:43-52.

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

Stephens S.L, Fry D, Franco-Vizcano E. 2008. Wildfire and forests in Northwestern Mexico: the United States wishes it had similar fire 'problems'. Ecology and Society. 13(2): 10

Stephens, S.L., J.J. Moghaddas, C. Ediminster, C.E. Fiedler, S. Hasse, M. Harrington, J.E. Keeley, J.D. McIver, K. Metlen, C.N. Skinner, and A. Youngblood. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecological Applications 19: 305-320.

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(5) art 79.

Ecosystem or population attribute	Metric	Within NRV?*	NRV	Current	Difference	Notes
Composition	Proportion of shade tolerant vs. shade intolerant spp.	No	40:60	55:35		Major shift from dominance of shade intolerant species to dominance of shade tolerant species
Composition	Area of yellow pine vs. mixed conifer forest	No	34% yellow pine, 20% mixed conifer	17% yellow pine, 30% mixed conifer		
Structure	Overstory Density: Number of trees per unit area	No	Mean: 159 trees/ha (64 trees/ac Range: 60 to 328 trees/ha (24 – 132 trees/ac)	397 trees/ha (160 trees/ac) Range: 238 to 755 trees/ha	80% - 600%	Current density higher on average than presettlement
Structure	Overstory Density: Number of large trees per unit area	No	55-75 trees/ha (interpreted from fig 15, need numbers)	25-35 trees/ha		Large tree density is lower in modern forests
Structure	Tree size class distribution	No				Major increases in small size classes, and general decreases in large size classes. Change in distribution shape from +/- flat, hump-shaped, or weakly J-shaped in average presettlement forest to strongly J-shaped in average modern forest
Structure	Average tree size (Mean dbh or quadratic mean diameter)	No		average tree diameter of 26 cm quadratic mean diameter of 32 cm	Current 26-60% of NRV	Average conifer tree in modern YPMC forests about 1/2 the diameter of the average tree in presettlement forests
Structure	Percent Canopy Cover	No	17-49%		25-45% increase (maybe ref for >55% increase)	Modern mean canopy cover is above presettlement
Structure	Area covered by dense (high canopy cover forest)	No	5%-20%			
Structure	Pieces of Coarse woody debris (CWD) per unit area	No				Density of CWD is higher in contemporary forests
Structure	Mass of Coarse woody debris (CWD) per unit area	No				Average tons/ha of CWD is higher in contemporary forests
Structure	Forest Fuels (Tons/ha)	No				On average, contemporary YPMC forests support much higher fuel loadings than presettlement forests, in both fine-fuel and coarse-fuel classes
Structure	Proportion early/middle/late seral forest	No				Current lack of old forest successional stages, perhaps some localized lack of early stages
Structure	Gap size	No	0.003 ha -1.17 ha (0.007 - 2.89 ac)			Gap sizes are generally decreasing (in undisturbed forests), but also increasing in disturbed forests due to more severe

Ecosystem or population attribute	Metric	Within NRV?*	NRV	Current	Difference	Notes
						disturbance
Structure	Shrub Cover (percent cover)	Maybe	16.9 – 28.6%			Difficult to assess, little presettlement data. Overall shrub cover on landscape not much changed over time; cover within forest stands may be lower due to fire suppression
Structure	Number of snags per unit area	No				Snag density is higher in contemporary forests
Structure	Tree Basal area	Yes	Mean: 35 m2/ha Range: 21 – 54 m2/ha		No difference	Basal area similar in modern forests; major difference is distribution of more biomass in small and medium trees in contemporary forest than in presettlement forest This is a result to two countervailing trends, increasing tree density and decreasing average tree size.
Function	Tree mortality	No				Higher background mortality and large tree mortality than NRV
Function	Fire regime	No	I	III and IV		Shift from Fire Regime I to Fire Regimes III and IV
Function	Fire frequency (FRI)	No	Mean: 11-16 years Range: 5 – 80 years			Current frequency far below presettlement but rising
Function	Fire severity (proportion of fire in high severity)	No	1-10%	29-35%		Current severity higher than presettlement and rising
Function	Fire size (excluding immediately suppressed)	No	<750 ac	3750 ac		Current mean and mean max fire sizes larger than presettlement mean (when excluding immediately suppressed fires)
Function	High severity fire patch size	No	< 10 acres			Current high severity patch sizes higher than presettlement mean and rising
Function	Fire rotation	No	Mean: 22-31 years Range: 11-70	Mean: 258-280 Range: 95-516	10 times longer	Fire rotations much longer today than presettlement
Function	Fire season	No				Fire season is becoming longer but general seasonal patterns are similar
Function	Annual area burned	No				Current mean annual area burned is much lower than all estimates of presettlement area
Population	Owl population size					
Population	Owl population distribution					