United States Department of Agriculture

Rim Fire Reforestation (45612)

Vegetation Report

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1. Introduction

1.01 OVERVIEW

The Stanislaus National Forest is preparing an environmental impact statement for the proposed Rim Fire Reforestation (Rim Reforestation project). This project proposes activities within the Rim Fire perimeter in the Stanislaus National Forest on portions of the Mi-Wok and Groveland Ranger Districts in Mariposa and Tuolumne Counties. The project boundary includes only National Forest System (NFS) lands within the fire perimeter. Proposed activities would reestablish forested conditions in severely burned areas and use prescribed fire and thinning in existing plantations to improve forest health and reduce fuels. The total Rim Reforestation project area is about 41,933 acres and includes up to 26,009 acres of reforestation activities, 15,924 acres of prescribed burning and thinning operations, and up to 5,915 acres of noxious weed treatments.

The Rim Fire started in a remote section of the Stanislaus National Forest on August 17, 2013 and burned 257,314 acres, or 400 square miles, including 154,530 acres of NFS lands, becoming the largest conifer forest fire in California's recorded history. The fire burned with high vegetation burn severity across 98,049 acres (38% of the burned area) consuming nearly all woody materials located on the ground (USDA 2014e).

Organization of this Document

This report discloses the existing conditions and any direct, indirect, and cumulative environmental effects that would result from the proposed action and alternatives. The document is organized into the following parts:

- Chapter 1 provides a brief overview of the project and summary of the document organization.
- Chapter 2 provides a description of the agency's proposed action and alternatives.
- Chapter 3 describes the pertinent laws, regulations, and direction; the environmental setting, existing conditions, methodology and metrics used to conduct the analysis; and, the environmental effects of implementing the proposed action and other alternatives. The effects of the no action alternative are described to provide a baseline for evaluation and comparison of the other alternatives.
- References provide the bibliographical information of any literature cited and personal communications used to write this report.
- Appendix A provides more detailed summaries of tree characteristics to support the analyses presented in the report.
- Appendix B provides rationale for using Rapid Assessment of Vegetation Condition (RAVG) analysis mapping instead of Monitoring Trends in Burn Severity Mapping.

Supporting Documentation

All supporting documentation is available in the project record. Supporting documentation includes plot data, field notes, GIS data, growth model outputs, and data analyses and summaries. Original growth model inputs and outputs are archived in their original format (e.g., Comma Separated Values, MS Access, and MS Excel).

2. The Alternatives

2.01 ALTERNATIVES CONSIDERED IN DETAIL

Alternative 1 (Proposed Action)

A complete description of the alternative is found in the Rim Reforestation EIS, 2.02 Alternatives Considered in Detail.

Alternative 2 (No Action)

A complete description of the alternative is found in the Rim Reforestation EIS, 2.02 Alternatives Considered in Detail.

Alternative 3

A complete description of the alternative is found in the Rim Reforestation EIS, 2.02 Alternatives Considered in Detail.

Alternative 4

A complete description of the alternative is found in the Rim Reforestation EIS, 2.02 Alternatives Considered in Detail.

Alternative 5

A complete description of the alternative is found in the Rim Reforestation EIS, 2.02 Alternatives Considered in Detail.

3. Affected Environment and Environmental Consequences

3.01 VEGETATION

The **National Forest Management Act** of 1976, Sec. 4. (d)(1), states that "It is the policy of Congress that all forested lands in the National Forest System shall be maintained in appropriate forest cover with species of trees, degree of stocking, rate of growth and conditions of stands designed to secure the maximum benefits of multiple use sustained yield management in accordance with land management plans." Furthermore, the Act requires that "All national forest lands treated from year to year shall be examined after the first and third growing seasons and certified … as to stocking rate, growth rate… Any lands not certified as satisfactory shall be returned to the backlog and scheduled for prompt treatment" (NFMA 1976).

The **Forest Service Land Management Planning Handbook** (FSH 1909.12) shows the vision for ecosystems is to have ecological integrity and adaptive capacity. The handbook states: "Ecosystems have integrity when their composition, structure, function, and connectivity are operating normally over multiple spatial and temporal scales" (USDA 2015b, p. 58). The handbook provides the following definitions, which are provided here for reference and clarification (USDA 2015c).

- **Adaptive capacity**: The ability of ecosystems to respond, cope, or adapt to disturbances and stressors, including environmental change, to maintain options for future generations. As applied to ecological systems, adaptive capacity is determined by genetic diversity of species, biodiversity within a particular ecosystem, and heterogeneous ecosystem mosaics as applied to specific landscapes or biome regions.
- **Ecological integrity**: The quality or condition of an ecosystem when its dominant ecological characteristics (for example, composition, structure, function, connectivity and species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence.

Forest Service Region 5 developed guidelines for ecological restoration to help retain and restore forest resilience and provide a sustainable ecosystem from a broad range of services to humans and other organisms (USDA 2011a). Within Region 5, three major drivers of change have been identified: climate change, shifting hydrologic patterns and increasingly dense unhealthy forests coupled with rapidly growing human populations. Ecological restoration will be the core objective used to promote an all lands approach to restoration. Vegetation and fire management will support the development of biodiversity and ecological processes before and after fire disturbance, and ensure the retention of forest resources over the long-term. This approach will promote activities that include the following:

- Reforesting after wildfire and implementing suitable stand maintenance activities that meet project goals and site conditions.
- Forest thinning and prescribed fire to decrease fuel loading and increase forest heterogeneity.
- Providing wildlife and plant habitat for threatened and endangered species.

Forest Plan Direction (USDA 2010a) provides broad management goals and strategies that address problems related to old forest ecosystems and associated species (p. 11). These goals and strategies include:

- Increasing the frequency of large trees, increasing structural diversity and improving the continuity and distribution of old forests across the landscape.

- Restoring forest species composition and structure following large scale, stand-replacing disturbance events.
- Restoring ecosystems across all land allocations following large-scale catastrophic disturbance events.

Forest Plan Direction outlines management practices that are actions that achieve the goals and objectives of the Forest Plan (p. 17-32). Actions applicable to the Rim Reforestation project include:

- All activities necessary to reestablish desirable tree species by artificial methods on deforested areas. Minimum standards for reestablishment are contained in the Regional Standards. Activities included in this practice are: preparation of the seed bed or planting site; planting seedlings or direct seeding; saving natural regeneration; animal, insect and disease control when necessary; and examinations, evaluation, certification and monitoring of stands to achieve the reestablishment objectives.
- Activities necessary to reduce the effect of competing vegetation, animals, insects and disease on the growth and development of desired tree species.
- Removal of surplus trees in areas with excess stocking by cutting, mowing or herbicide injection to favor potential crop tree growth and development. Excess trees thinned do not have a commercial value because of tree size, species comparison or access to available markets.
- Removal of trees in stands of less than rotation age to periodically reduce the stocking level to a point where the stand will grow back to 90% of normal stocking as indicated in-yield tables within a specified time period. There are no minimum or maximum treatment area sizes. This cutting method applies to stands on all forest types which carry stocking in excess of desired amounts.

Forest Plan Direction outlines forestwide standards and guidelines that provide management direction applying to all Forest lands (p. 33-64). Specific standards and guidelines that are applicable to the Rim Reforestation project include:

- Maintain the species composition of the major forest types existing where projects occur.
- Promote shade intolerant pines (sugar and ponderosa) and hardwoods.
- Where possible, create openings around existing California black oak and canyon live oak to stimulate natural regeneration.
- Retain the mix of mast-producing species where they exist within a stand.

Forest Plan Direction outlines direction and standards and guidelines that apply to specific management areas (p. 65-164). Management areas applicable to the Rim Reforestation project include: Wildlife (p. 123-127) and General Forest (p. 161-164). Direction and standard and guidelines for these management areas are the same:

- Reforest all openings in available, capable and suitable lands for timber production created by timber harvest, wind, fire, or insect and disease pests.
- Preparation of sites for artificial or natural stand reestablishment will be completed sufficiently in advance of planting or natural seeding to provide control of competing vegetation. Normally control of competing vegetation will be designed to ensure prescribed first year survival of planted or natural seedlings. This will often involve more than one treatment on more than one competing species prior to planting. It may involve a variety of techniques including fire, mechanical bunching and shredding, discing and pesticides. Pest management will be considered as necessary.
- Natural seeding or planting will be done with tree species, seed zones and elevations determined to be appropriate through a silvicultural examination and prescription.
- Reduce the effect of competing vegetation on the growth and development of desired species on lands available, capable and suitable for timber production.

- Release efforts will only be done after appropriate stand examination and prescription. The objective will be to treat stands before brush or undesired hardwood densities reach 10,000 cubic feet per acre. Ideally competing vegetation will be treated while seedlings and sprouts from this vegetation are small and easily treated by a variety of techniques. Often this will be within two or three years after site preparation. On plantations five years or older where conifers are established over most of the site, the objective will be to treat competing vegetation based on the actual growth and development of the tree stand. Brush and undesired vegetation will be treated where the conifer stand is not meeting site objectives for growth and it is apparent that competing vegetation is the cause. A variety of techniques including mechanical piling, shredding, hand grubbing and herbicides may be used.
- Remove surplus trees on available, capable and suitable lands with excess stocking. This will be done to favor growth and development of potential crop trees, prevent disease and insect outbreaks, or meet other resource objectives.
- Pre-commercial thinning is a tool that will be used to maintain diversity by improving species composition in many stands. A variety of techniques may be used including crushing, piling, shredding, hand cutting and pesticides.
- Design cutting methods to obtain specific management objectives for late successional Management Indicator Species (MIS habitat).

Effects Analysis Methodology

Assumptions Specific to Vegetation

- Plot data, field observations and severity mapping are representative of the project area. Severity mapping measures the fire severity present at about 30 days after fire containment; however, it does not measure the effects of drought and insects that can occur after images are processed.
- Region 5 CALVEG cover types conifer (CON) and mixed hardwood-conifer (MIX) represent suitable and productive forestland.
- Post-fire conditions assume the Rim Recovery and Rim HT projects are fully completed or will be completed prior to Rim Reforestation treatment activities.
- Examples of post-fire reforestation and secondary succession are representative of the Rim Reforestation project area. Most of the examples are taken from neighboring projects or other areas of the Sierra Nevada; therefore, this is a reasonable assumption.
- Current and recent climate conditions provide a reasonable baseline for this analysis time frame (about 60 years). Temperatures in the Sierra Nevada have become incrementally warmer since the 1960s (Thorne et al. 2008) and are expected to continue warming (Safford et al. 2012). This analysis assumes this trend will continue, but not to an extent that loss of habitat suitability over the next 50 to 60 years would be significant. In other words, no dramatic shifts in vegetation would occur beyond what has or is currently being observed. Tree competition-related thresholds of stand density are still applicable as they relate to insect, disease and drought.

Data Sources

- Survival exams from the Mi-Wok and Groveland Ranger Districts covering 4,966 acres of plantations established in the 1980s and 1990s.
- Field observations and plot data. Two datasets were developed to assess post-fire conditions. From 2014 to 2015 field crews conducted walkthroughs of each proposed unit noting operability; need for reforestation, thinning and fuels reduction; and, post-fire response of shrubs and other herbaceous vegetation. Fixed area circular plots (1/50th-acre), distributed on a systematic random grid (656 feet by 656 feet) were used to collect information about: natural regeneration, competing vegetation cover, distance to potential seed source and whether or not the plot had been impacted by post-fire mechanical operations. Complete details about the plot sampling protocol are available in the project record.

- Common stand exam (CSE) data collected in the Rim Fire perimeter. Data downloaded from the Natural Resources Management Natural Resource Information System (NRM NRIS) Field Sampled Vegetation Database (FSVeg). A total of 843 CSE plots were collected between 2005 and 2013 (prior to the 2013 Rim Fire).
- California Wildlife Habitat Relationships (CWHR, CDFW 2014b) and CALVEG cover types.
- GIS data including: Rapid Assessment of Vegetation Condition (RAVG) analysis mapping (refer to Appendix B for rationale), Worldview Imagery, Soil Survey Geographic (SSURGO) database for the Stanislaus, land allocations, project unit boundaries and road treatments.

Vegetation Indicators

The following indicators provide a relative measure of the direct and indirect effects to changes in forest vegetation due to project alternatives. They are intended to respond to the agency vision of promoting ecological integrity and adaptive capacity (USDA 2015b; USDA 2015c).

TREE SIZE AND SPECIES COMPOSITION

- Average tree diameter at breast height (dbh) at years 10, 20 and 60.
- Average tree height at years 10, 20 and 60.
- Percent trees per acre by species, in particular shade-intolerant pine (sugar and ponderosa) and hardwoods.
- Relative change in abundance of understory vegetation (shrubs, grasses and forbs).

Tree size influences many different aspects of forest ecosystems. This is especially true for large trees, which contribute disproportionately to reproduction (van Wagtendonk and Moore 2010), are critical wildlife habitat features (North et al. 2000; Zielinski et al. 2004) and contribute to fire resiliency (Agee and Skinner 2005; Brown et al. 2004; van Mantgem et al. 2011). The number of trees per acre is used to describe the recovery of tree density, forest structure and habitat. It provides a basis for comparing the direct and indirect effects to species composition, which is important in assessing forest health and resiliency. Species composition is also an important component of wildlife habitat and strongly influences ecosystem processes and functions (Tilman et al. 1997; CREP 2008).

STAND DENSITY INDEX

- Years until tree densities reach or exceed the stand density index (SDI) zone of increased bark beetle-related mortality (SDI of about 230 or about 45% of maximum SDI) and the bark beetle induced maximum SDI of 365.

Tree density influences the dynamics of vegetation competition, tree growth and forest health. SDI is an age and site productivity-independent measure of density. SDI is used to evaluate forest health and vigor in terms of resiliency of trees to drought, insects and disease. In general, higher stand densities predispose trees to damage or mortality. Bark beetles and disease agents are often more damaging at high densities and also limit diameter growth. Densities below 55 to 60% of the maximum SDI level provide for reduced density-related mortality and relatively high vigor (Oliver 1995; Long and Shaw 2012). According to Oliver (1995) and Oliver and Uzoh (1997) ponderosa pine stands, especially plantations, start to show increased bark beetle-related mortality at SDIs above 230 (about 45% of maximum SDI). They also suggest the possibility of a bark beetle induced maximum SDI of 365, though they state that outbreaks often reduce stand density to levels well below that SDI.

FUTURE MANAGEMENT FEASIBILITY

- Years until stand densities near or exceed the threshold of imminent competition-related mortality (SDI of about 260 or 50% of maximum; Long and Shaw 2012) and can produce at least 2,000 board feet per acre if thinned to SDI 170 (about 35% of maximum).

An obvious effect of regenerating conifer forests is the future ingrowth of small trees and accumulation of surface fuels (e.g., tree litter and shrub cover). Even in the absence of coniferous

trees, vegetative biomass will accumulate over the next several decades and contribute to fuel loading in the absence of fire. This is evident in historical records, which indicate that fire played an essential role in moderating shrubs, tree regeneration and tree development even with low overstory tree densities (Collins et al. 2015; Show and Kotok 1924; Sudworth 1900). Furthermore, shrub cover is a good predictor of fire behavior (Lydersen and North 2012; van Wagtendonk et al. 2012), and can contribute to high-severity fire (Harris and Taylor 2015; Lydersen et al. 2014). Today, prescribed burning and managed wildfire are faced with numerous operational and social constraints that limit their use in effectively reducing hazardous fuels and maintaining natural processes (Quinn-Davidson and Vaner 2012). An important alternative to fire has been the use of mechanical thinning to reduce ladder and canopy fuels through the removal of trees and other vegetation (Graham et al. 1999; 2004). To create resilient stands, however, mechanical thinning is more effective when accompanied by reduction in surface fuels either by prescribed surface fire or piling and burning (Agee and Skinner 2005; Reinhardt et al. 2008). Therefore, we are faced with the reality that effective suppression and containment of wildfire is more likely with the complementary use of both mechanical thinning and prescribed fire (North et al. 2015). Although the focus is often set on the costly removal of small-diameter trees and surface fuels, such costs can be offset through the necessary removal of some merchantable intermediate sized trees to maintain low canopy bulk density and open stand conditions, as well as accelerate development of large trees (Moghaddas and Craggs 2007; North et al. 2009). It stands to reason then that management feasibility would improve if increasing fuel loads and stand densities are balanced with larger trees that can at least partially offset operational costs when thinned.

CHANGE IN FOREST STRUCTURE

- Change in the proportion of forest successional classes.

The Rim Fire changed vegetation conditions across the project area. The fire killed and injured trees and shrubs, which changed forest structure and habitat. Treatments have the potential to change post-fire forest structure and resulting habitat. This analysis uses the Region 5 CALVEG classification system to evaluate changes in forest successional stages.

The CALVEG system conforms to the upper levels of the National Vegetation Classification standards hierarchy. This analysis used the CALVEG vegetation cover type to identify a general distinction between broad vegetation types: conifer (including mixed hardwood and conifer), hardwood, shrub, grass, barren and other life form classifications. The primary focus of this analysis is the conifer and mixed hardwood-conifer cover types. Within these two conifer-dominated cover types, CWHR vegetation type, size class and density were used to describe forest structure. Table 3.01-1 shows size classes 0 to 2 are usually indicative of young trees, habitat best described as dominated by seedlings and saplings. Larger size classes are often interpreted as “old” or “mature” forest. Large size classes can take 40 to 150 years to develop. Multilayered forest provides a variable tree structure; both large and medium trees are present with saplings or small trees. These multistory forests develop in a variety of ways; however, the large tree component takes the longest to develop.

Table 3.01-1 California Wildlife Habitat Relationship (CWHR) size classes

CWHR Size	Description	Diameter at Breast Height
1	Seedling	Less than 1 inch
2	Sapling	1 to 6 inches
3	Pole	6 to 11 inches
4	Small tree	11 to 24 inches
5	Medium/Large Tree	Greater than 24 inches
6	Multilayered	Size 5 over size 4 or 3; total tree crown closure greater than 60%

Table 3.01-2 shows density of forest cover is expressed in terms of tree canopy cover. Dense forest cover can develop in as little as 10 years when dominated by small trees; or take hundreds of years when dominated by large trees.

Table 3.01-2 California Wildlife Habitat Relationship (CWHR) density classes

CWHR Density	Description	Canopy Cover
blank or X	No cover, definite forest habitat	Less than 10%
S	Sparse cover	10% to 24%
P	Open cover	25% to 39%
M	Moderate cover	40% to 59%
D	Dense cover	Greater than 60%

In a bioregional assessment for natural range of variability, Safford (2013) provides an estimate of historic landscape variability based on LANDFIRE biophysical settings (BpS) state and transition models. The BpS represent the vegetation that may have been dominant on the landscape prior to Euro-American settlement and is based on the current biophysical environment and an approximation of the historical disturbance regime, but with the current climatic conditions (LANDFIRE 2009). Reference landscape forest structure is described using five successional classes (seral stage) for yellow pine and mixed conifer (YPMC) forests in the Sierra Nevada: early successional, mid successional (open and closed canopy) and late successional (open and closed canopy). Early successional is described as vegetation not dominated by trees greater than 4 inches dbh, which includes areas dominated by herbaceous plants, shrubs, seedlings and saplings. Open canopy for mid and late successional stages is defined as less than 40 to 50% canopy cover and closed canopy is greater than 40 to 50% canopy cover. Mid successional stages are defined by trees 5 to 21 inches dbh and late successional stages are defined by trees greater than 21 inches dbh. These definitions are quite similar to the CWHR size and density classes; therefore, this analysis adapted the CWHR size and density classes shown in Table 3.01-3 to these successional classes for comparison to reference forest landscape structure.

Table 3.01-3 Adaption of CALVEG and CWHR to Forest Successional Classes

Forest Successional Class[1]	CALVEG Cover Type	CWHR Size	CWHR Density	Historic Forest Landscape
Early	Conifer and mixed hardwood-conifer	2 or less	ALL	15 to 20%
Mid seral open canopy (mid-open)	Conifer and mixed hardwood-conifer	3 and 4	S, P	20 to 30%
Mid seral closed canopy (mid-closed)	Conifer and mixed hardwood-conifer	3 and 4	M, D	5 to 15%
Late seral open canopy (late-open)	Conifer and mixed hardwood-conifer	5 and 6	S, P	25 to 45%
Late seral closed canopy (late-closed)	Conifer and mixed hardwood-conifer	5 and 6	M, D	5 to 20%

[1] As defined by Safford (2013)

Vegetation Methodology by Action

The methodology for evaluating effects on vegetation is based upon assessing changes to vegetation following the Rim Fire and the associated timber salvage, hazard tree abatement and fuels reduction work. The analysis evaluates cumulative changes in vegetation composition, growth and structure caused by the project activities as well as the Rim Reforestation project. Changes in vegetation are assessed post treatment and over a 60-year time frame. This temporal scope was selected because the impacts to seedling survival and growth, species composition and forest structure at a given location can accumulate over time from different activities or events. The analysis compares effects found in scientific literature to the effects experienced with similar treatments on the Stanislaus.

DIRECT AND INDIRECT EFFECTS

Unless otherwise specified, the analysis area used to analyze the direct and indirect effects on forest vegetation is 41,933 acres and includes only NFS lands within the Rim Fire perimeter. The analysis area is based on the Rim Reforestation project treatment unit area where project activities would impact forest vegetation. It includes 26,009 acres of reforestation activities (including natural regeneration and deer habitat enhancement with reforestation), 13,934 acres of prescribed fire with thinning and 1,990 acres of prescribed fire only for deer habitat enhancement. The analysis encompasses hardwood forest, chaparral, grasslands and riparian vegetation; however, given the administrative and ecological setting of the affected environment, as well as the purpose and need of the project, the analysis primarily focuses on coniferous forest.

CUMULATIVE EFFECTS

Cumulative effects on vegetation are analyzed at the project scale. Vegetative cumulative effects are additive. That is they are the total of changes of proposed treatments to vegetative structure. The project scale analysis allows for comparison of changes that are occurring as a result of the past, present and reasonably foreseeable projects across several large watersheds. Changes in coniferous forest have the greatest potential for cumulative effects on conifer establishment, species composition and forest structure. Although the cumulative effects analysis focuses on the project scale, the effects of the project on landscape forest structure are sometimes discussed inside of the larger Rim Fire landscape (257,314 acres); however, evaluation of vegetation change beyond the project scale is limited and outside the scope of this analysis.

ANALYSIS

Natural regeneration is proposed in some treatment units. Natural regeneration in these units would be monitored to assess the need to release, alter species composition, or if necessary, conduct site preparation and planting. Although field assessments of these units suggest that natural regeneration is likely and reforestation treatments will not be needed, this analysis assumes all reforestation actions would be completed in natural regeneration units; therefore, analysis of all natural regeneration units are included with the analysis of other reforestation units to address all potential effects. No further distinctions between natural regeneration units and reforestation units are made in this analysis. Also, no special distinction is made for reforestation activities proposed for deer habitat improvement other than differences in planting density.

Assessment of Existing Conditions

Given the large number of treatment units and limited available resources, it was impractical to collect enough plot-level data within each unit to accurately assess conifer regeneration and vegetation composition on a per unit basis. Instead, this analysis first vetted the data using methods similar to Crotteau et al. (2013), stratifying plots across a gradient of four fire burn severities. Fire severity was determined remotely using Relative differenced Normalized Burn Ratio (RdNBR) derived from pre- and post-fire LANDSAT Thematic Mapper images, which are subsequently transformed to four nominal Composite Burn Index (CBI) categories: unchanged (minimal or no visible effect of fire), low-severity, medium-severity and high-severity. RdNBR has been tested in similar conifer dominated vegetation types and proven to produce fire severity classifications with similar accuracy as other fire mapping processes (Miller et al. 2009a).

Reforestation

The analysis of conifer forest establishment and growth required a synthesis of local reforestation records, scientific literature and plot data. This synthesis helped parameterize two forest growth models used to evaluate the short- and long-term changes in vegetation: CONIFERS in R (RCONIFERS) and the Forest Vegetation Simulator (FVS).

Growth Models

RCONIFERS was used to predict growth and development of trees and competing vegetation in the project area to age 20. RCONIFERS is a model for young stand growth developed by the Forest Service Pacific Southwest Research Station (Ritchie and Hamann 2015). In general, most forest growth and yield simulators are ineffective at simulating the growth of very young stands, especially any stand in which non-tree vegetation contributes significantly to the level of competitive stress to which trees are exposed (Ibid). The Southwest Oregon (SWO) variant of RCONIFERS was developed using data from mixed-conifer stands heavy to Douglas-fir in southern Oregon and heavy to ponderosa pine in northern California. The SWO variant is based on a limited list of conifer, hardwood and shrub species. The list includes all major conifer tree species found in the project area and the majority of the dominate hardwood and shrub species. With the exception of bearclover (*Chamaebatia foliolosa*), the SWO variant is the most appropriate model available to forecast the dynamic effect of competing vegetation on young forest growth in the project area. Although bearclover is not included in RCONIFERS, its effects on young conifer establishment and growth have been studied for several decades (McDonald et al. 2004). To account for the effect of bearclover competition, conifer growth in RCONIFERS was projected in the absence of competing vegetation (free to grow). Findings from scientific literature were then used to adjust conifer height and diameter growth accordingly.

The FVS Western Sierra Nevada Variant was used to simulate tree growth and summarize forest structure beyond age 20. The FVS projections use the default growth rates, which are calibrated to reflect inter-tree competition of established stands that have developed to a point where non-tree vegetation plays a minimal role in tree growth. This distinction between FVS and RCONIFERS is important to note for this analysis because growth projections are transitioned between the two models at age 20 despite tree size. This could affect long-term growth projections by essentially eliminating the variable of competitive stress caused by non-tree vegetation from the growth models at year 20. The effect is likely most pronounced in alternatives where early conifer growth has been reduced the most, resulting in the smallest tree heights and diameters at year 20. The aboveground photosynthetic potential of young trees is balanced by belowground root development and the ability to effectively compete for limited water and nutrients (Grossnicle 2005); therefore, these smaller trees are still aggressively competing with other vegetation and growing at a reduced potential compared to trees that have benefited from about two decades of greater root development and height growth. Removing the effect of non-tree competition at year 20 artificially provides these smaller trees a window of relatively free to grow conditions until they reach a size where inter-tree competition becomes a factor. During this period, trees that were larger at age 20 would experience a potentially smaller window of free to grow conditions before the onset of inter-tree competition.

While computer models attempt to display the complex reality of vegetation; modeling results fall short of a precise (perfect) depiction of the variability of real forest vegetation. This short-coming is due to the variability associated with measuring vegetation, the variability in locating plots, the errors associated with drawing boundaries around vegetation and the ability of algorithms used in the computer models to effectively emulate natural variability. Despite these short-comings, computer models do provide a means for relative comparison.

Model Parameterization

Given the large scale of this project, a variety of model states, or scenarios, representing common conditions were identified and analyzed rather than simulating forest growth of every treatment unit for each alternative. This approach was used partly because plot data was limited on a per unit basis. While a robust dataset was created by collecting data from numerous plots across the project area, there was only a subset of units that had more than a few plots. Having a small sample within a unit increased the likelihood of outlier plots having undue effects on describing existing conditions. For example, a unit might have abundant conifer regeneration and little competing vegetation. If only one

plot was completed in that unit and it happened to land in a patch of shrubs with no conifer regeneration, then the data would misrepresent the broader existing conditions across the unit.

The modeling scenarios were based on the five alternatives and their associated planting patterns, which are determined by slope position (EIS, 2.02 Alternatives Considered in Detail). Like FVS, RCONIFERS is a semi-distance-independent individual tree growth model. In general, this means it does not readily account for the distance between individual trees and how the proximity and arrangement of neighboring trees might influence individual tree development. Rather, tree growth and development is determined by stand-level conditions (e.g., trees per acre), thereby, inferring distance to neighboring trees given that conditions in a stand are consistent. Basing stand-level conditions on plot-level data, however, does provide some ability to model localized competition and site variables within a stand (Dixon 2002). Attempting to model the fine-scale nuances of the various proposed planting patterns would have resulted in a tenuous and overwhelmingly complex analysis with numerous scenarios (greater than 100) after factoring in biophysical variables. Instead, a simplified approach was used that based the modeling scenarios on slope position and planting density, assuming this would provide an adequate representation of the predominant conditions and vegetation dynamics germane to this analysis. Effects of fine-scale planting patterns on individual trees are left to qualitative discussion. Both RCONIFERS and FVS structure input data by stands and plots within stands. Therefore, each scenario was treated as a stand. To account for the range of biophysical factors, mock plots were developed within each scenario's stand (i.e., topographic slope position) based on environmental factors, density of hardwoods, competing vegetation and natural conifer regeneration as summarized from plot data.

Environmental Factors

The following environmental factors were incorporated into RCONIFERS: elevation, aspect, percent slope, water holding capacity, annual precipitation and growing season precipitation. Water holding capacity was determined using the available water supply (AWS) from the Soil Survey Geographic database (USDA 2008a). The available water supply is defined as the total volume of water available to plants when soil is at field capacity (USDA 1999). Mean annual precipitation (36.8 inches) and mean growing season precipitation (3.2 inches) were calculated based on precipitation records from the last 15 years (WRCC 2015).

Competing Vegetation

Variability within each slope position was addressed by identifying dominant shrub types based on plot data. After categorizing plots into dominate shrub types, the proportion of plots within each slope position by shrub type were calculated. Overall, bearclover and deerbrush (*Ceanothus integerrimus*) were disproportionately the dominate shrub species on the most plots (greater than 72% of all plots). Manzanita (*Arctostaphylos* spp.) was the third most common dominant shrub in the project area, but on a much smaller proportion of the plots (8.5%). Given the large number of plots dominated by deerbrush and bearclover, these shrub types were used to represent competing vegetation in RCONIFERS. That is, within each slope position all the deerbrush plots were averaged to estimate the species composition, cover and height of deerbrush and associated vegetation (i.e., other shrubs, grasses and forbs). This approach accounted for both the abundance of deerbrush and other shrub species, grasses and forbs. As mentioned previously, RCONIFERS does not include bearclover, so conifer growth adjustments were made based on scientific literature. Details on methodology and growth adjustments associated with bearclover are discussed in the following sections.

Natural Conifer and Hardwood Regeneration

Natural conifer regeneration was only used to model Alternative 2 and areas in Alternative 4 that would not undergo artificial reforestation. For the action alternatives and in areas proposed for reforestation under Alternative 4, planting or control of natural conifer regeneration was assumed to only occur in areas where natural regeneration was not meeting desired conditions; therefore, desired

conifer stocking and species composition would always occur under these scenarios whether it be natural or artificial.

Density of natural conifer and hardwood regeneration was estimated using plot data stratified across slope positions. Salvage and fuels reduction operations can reduce survival of naturally regenerating conifer seedlings through soil disturbance and physically burying seedlings in woody material (Donato et al. 2006). Although each plot was not sampled twice to explicitly assess pre- and post-logging conditions, data was collected in recently logged areas and in areas scheduled for salvage operations, but not yet completed. Assuming natural regeneration in these areas was relatively similar prior to mechanical operations, natural regeneration in logged and unlogged areas was compared. While both hardwood and conifer regeneration was lower in areas that had been logged, hardwood densities were impacted less. The majority of the oak and hardwood regeneration occurred in the form of stump sprouts, which are less vulnerable to damage by mechanical operations than small seedlings germinated from seed. For example, plots with top-killed oaks were likely less impacted by mechanical equipment given presence of standing dead oak and not conifers. Conifer regeneration, however, is more vulnerable to mechanical operations because it only regenerates from seed within close proximity to species typically targeted for salvage. Plot data suggests that salvage and fuels reduction operations reduced conifer regeneration density by 72% and oak regeneration by about 26%; however, the proportion of null plots were minimally impacted. When comparing logged to unlogged plots, there was less than a 4% difference in the proportion of plots without conifer regeneration. Because plot data was collected in some areas prior to the scheduled salvage and fuels reduction activities, results from these plots were adjusted to account for the estimated 72% reduction in seedling density that might result from mechanical operations.

Survival of Natural Regeneration

Aside from effects of salvage and hazard tree abatement operations, all hardwood regeneration is expected to initially survive and only experience mortality related to SDI thresholds built into default model settings. As already noted, the majority of hardwood regeneration is from root and stump sprouts, which afford these species ready-access to water and nutrients. Conifers, however, are expected to experience some levels of mortality as they must expend resources toward root development before they can invest in aboveground growth (Grossnicle 2005). Saigo (1969) found that pine would initially establish in high numbers, but quickly decline due to various factors such as predation, competition and poor microsite conditions. These studies suggest that conifer seedling survival is probably highly diverse and influenced by numerous factors including the species, competing vegetation, site conditions and climate.

While some research has taken a “snap-shot” of conifer seedling density several years to decades after wildfire (e.g., Collins and Roller 2013; Crotteau et al. 2013; Shatford et al. 2007), no studies were found that specifically tracked the establishment and survival of naturally established conifer seedlings immediately following high-severity fire. Saigo (1969) tracked establishment and survival in a relatively similar forest type as the project area, but not following moderate- or high-severity fire. Given the potential for high variability in natural conifer regeneration and the scant research on this this topic, this analysis based survival of naturally regenerating conifers on research conducted on planted conifers. This approach was taken because it helped control for the two very distinct shrub types being modeled for this analysis (deerbrush dominated and bearclover dominated), which would provide some diversity of results that are likely to occur. McDonald and Abbott (1997) reported between 99 and 81% survival of planted conifers over the course of an 18-year study on the effects of competing vegetation and young conifer growth. The study included a variety of shrub, grass and forb species common in Sierra Nevada mixed conifer forests, but did not include bearclover. Conifer survival rates were lowest in the high shrub cover plots and highest in the no shrub cover plots. Average survival rates of these low and high shrub plots (90%) was assumed to reflect the variation of potential natural seedling survival in areas dominated by deerbrush.

High initial bearclover cover (20 to 40%) can substantially decrease initial conifer survival by as much as 80% or more (Tappeiner and Radosevich 1982; McDonald and Fiddler 1999). Tappeiner and Radosevich (1982) reported only 13% conifer survival after 19 years in study plots with 20 to 40% bearclover cover. Reducing initial bearclover cover increased conifer survival to 71% (Tappeiner and Radosevich 1982). Other studies reported similar conifer survival (69 to 73%) in study plots that had 20 to 30% bearclover cover present in the first few growing seasons (McDonald and Everest 1996; McDonald and Fiddler 1999). Therefore, it was assumed that 71% of the natural conifer regeneration would survive in bearclover-dominated shrub types that had less than 30% bearclover cover and 13% of the natural conifer regeneration would survive in bearclover-dominated shrub types that had greater than 30% bearclover cover.

Research investigating effects of bearclover on growth of young ponderosa pine suggests that the majority of mortality occurs in the first few years after seedling establishment (Tappeiner and Radosevich 1982); therefore, natural regeneration mortality was incorporated into all model scenarios during the first projected growing season. Any additional mortality during the first 20 years was accounted for using default settings in RCONIFERS for SDI-related mortality. Default FVS mortality settings were used for projections beyond year 20.

Future Conifer Dispersal and Establishment

Conifer dispersal was not modeled in this analysis. The CWHR definition of shrub-dominated vegetation types require at least 10% shrub cover and less than10% cover of tree species. The definition for the mixed hardwood-conifer forest type requires at least 25% cover of conifer species when there is greater than 50% cover of hardwood species. Therefore, at least 10 to 25% conifer cover is required to be classified as a conifer vegetation type. Post-fire conifer dispersal, establishment and growth are influenced by shrub and hardwood vegetation. Findings from Shatford et al. (2007) indicate that conifer seedling establishment declines over time with an initial pulse in conifer establishment post-fire. This pulse of regeneration is likely the result of both conifer establishment near surviving mature conifers and dispersal and establishment of conifer seedlings that occurs during a short period of time when shrub and hardwood species are still responding to the fire; and therefore, have not yet dominated the site. Once all available growing space is occupied, conifer establishment steadily declines. Despite this increase in presence of established conifer seedlings, it does not necessarily suggest conifer dominance. Similar to plot data in the Rim Fire, high burn severity areas studied by Shatford et al. (2007) had much lower densities of pine regeneration compared to Douglas-fir or true firs, which comprised greater than 50 to 80% of the conifer regeneration. Shatford et al. (2007) found that distance to seed trees did not appear to be a major limiting factor in conifer seed dispersal; however, no indication is made as to how far specific conifer species were dispersing. Given the disproportionate abundance of Douglas-fir and true firs, these species likely accounted for the majority of longer dispersal distances. Despite gradual increases in establishment, Shatford et al. (2007) notes that even after 19 years “conifer seedlings were frequently overtopped by shrubs and hardwoods” (p. 144). Other studies have also observed dominance of non-pine regeneration in areas of high burn severity that is also commonly overtopped by other vegetation (Crotteau et al. 2013; Nagel and Taylor 2005; SNRC 2012). Research on the Tahoe National Forest concluded that high-severity patches created in four different fires required 30 to 50 years for conifers to establish (Russell et al. 1998). Additionally, other studies have demonstrated that large areas dominated by shrubs can considerably slow or inhibit the development of dry conifer forests (Barton 2002; Goforth and Minnich 2008; Roccaforte et al. 2012; Collins and Roller 2013).

Despite the presence of some conifers, their contribution to vegetative cover may not warrant reclassification from a CWHR shrub or hardwood dominated vegetation type to a conifer vegetation type. Although Douglas-fir and true firs are able to slowly disperse into large severely burned areas and persist among dense shrubs and hardwood vegetation even when overtopped for extended periods of time, their growth is greatly diminished (Conard and Radosevich 1982a; Conard and Radosevich

1982b; Nagel and Taylor 2005; Shatford et al. 2007). Shatford et al. (2007) noted that in addition to the sporadic nature of dispersal, conifer growth would be delayed for about 20 years until shrub growth slows, but beyond that "successional development cannot be precisely predicted for specific locations" (p. 145). Nagel and Taylor (2005) observed even longer periods of significantly stunted growth in white fir, noting that on average it took about 30 years for a white fir seedling to grow one foot in height and about 120 years of fire suppression for white fir to establish and overtake chaparral vegetation.

The long time frame required for recovery of conifer forest cover is also attributable to the time required for conifers to produce viable cone crops that allow for increased expansion. For example, immature white fir and ponderosa pine can produce seed crops, but their performance is more erratic than that of mature trees, which typically do not produce dependable cone crops until 40 to 60 years of age (Laacke 1990; Oliver and Ryker 1990). In California, ponderosa pine trees more than 25 inches dbh were the best producers (Oliver and Ryker 1990). The best seed producing size for white fir is between 12 and 30 inches dbh. Similarly, Douglas-fir seed production increases with age. Old-growth Douglas-fir may produce 20 to 30 times the number of cones than younger trees that are 50 to 100 years old (Hermann and Lavender 1990).

Based on the very slow dispersal and growth of conifers among shrubs and hardwoods, conifer expansion beyond the proportion of the analysis area already regenerating with conifers was not modeled during the analysis time frame of 60 years. During the analysis time frame, it is unlikely that significant acreage would experience an increase in conifer cover of more than the 10 to 25% required to qualify as a CWHR conifer forest type or mixed hardwood-conifer type. Nor would seedlings that might establish post-fire grow to an age or size that would result in significant additional seed fall within the next 60 years. Therefore, the proportion of plots with and without conifer regeneration were used to determine the acres that would reliably regenerate as conifer forest during the analysis time frame. All other acres were treated as chaparral or hardwood vegetation types.

Conifer Planting Density and Survival

Conifer planting was based on densities identified for each alternative and topographic position (EIS, 2.02 Alternatives Considered in Detail). For each scenario, the 25-foot planting buffer on 5 oaks per acre were used to adjust the number of conifers planted per acre. Conifer planting densities were also adjusted to account for potential initial mortality that commonly occurs shortly after being planted. Local reforestation records from the 1990s covering 4,966 acres on the Mi-Wok and Groveland Ranger Districts were evaluated to determine the effects of different site preparation and release treatments on planted conifer seedling survival. Records were categorized into different site preparation and release categories according to the different action alternatives proposed under the Rim Reforestation project. Table 3.01-4 shows third-year survival exam results which were used to calculate an average percent survival, weighted based on plantation acres. Survival exam records distinguished between naturally regenerating conifers and planted conifers. The results presented here lump natural and artificial regeneration; thereby, incorporating potential natural regeneration that may occur within the Rim Reforestation project area.

Table 3.01-4 Third-Year Seedling Survival: Weighted Average Estimates Based on Local Records

Site Preparation and Release Treatments	**Planted in 1990s** (acres)	**Initial Planted** (per acre)	**Survived** (per acre)	**Percent Survival**[1]
Site preparation with herbicides or deep tilling with forest cultivation followed by herbicide release treatments	1,767	453	330	73
Deep till site preparation followed by manual releases	2,304	420	201	48
No site preparation followed by manual releases	895	390	110	28

[1] Seedling survival assumptions are based on these values rounded up to the nearest multiple of 5.

Of the 4,966 acres evaluated, 1,767 acres were treated with herbicides as shown in Table 3.01-4. While herbicides were used for release treatments on all of these acres, site preparation treatments included different combinations of herbicides and deep tilling with forest cultivation. Effects on seedling survival of site preparation with either deep tilling or herbicides were assumed to be similar; and therefore, lumped together. This assumption was made for two reasons. First, records for a large portion of the acres (732 acres) in one of the timber compartments evaluated (Walton Cabin) were incomplete; however, multiple Forest employees (both retired and still working for the Forest Service) that assisted with reforestation efforts in the 1990s confirmed that both deep tilling and herbicides were widely used in this compartment. Second, records from the other 1,035 acres indicate that when followed by herbicide release treatments, site preparation with deep tilling versus herbicides resulted in similar survival rates. On average, 73% of seedlings survived when prepped with herbicides versus 67% when prepped with deep tilling. Similarly, areas that were not site prepped, but were released using herbicides were included in this category based on the assumption that no site preparation was deemed necessary at the time. While all units under Alternatives 1, 4 and 5 are prescribed either herbicides or deep tilling for site preparation, as described in Chapter 2 these treatments would only be used if still deemed necessary at the time of implementation.

The remaining 3,199 acres were reforested without herbicides. Treatment and survival details for these acres are described in USDA (1995). About 2,300 acres were site prepped with deep tilling and released using manual grubbing. The other 895 acres were planted the first spring after the 1987 Stanislaus Complex Fire before competing vegetation had begun to respond.

Conifer Growth and Competing Vegetation

The Forest Service in Region 5 has extensive experience and a large body of research including long-term studies that clearly establish the effect of competing vegetation on the growth of young conifer trees. For example, in a 31 yearlong study McDonald and Abbott (1997) evaluated the effect of multiple shrub densities on the growth of ponderosa pine seedlings. Four shrub densities ranging from none to heavy were maintained for the first few years and pine growth was documented periodically for 26 additional years. At the end of the study, tree heights ranged from about 9 feet in the heavy shrub plots to just over 30 feet in plots with no shrubs. RCONIFERS was built using extensive study plots documenting the dynamics of young conifers and shrubs (Ritchie and Hamonn 2015); therefore, this analysis uses RCONIFERS to simulate the effect of competing vegetation on young conifers. As mentioned earlier, however, RCONIFERS does not include bearclover, so the effect of bearclover was analyzed based on relevant research.

Herbicide Control of Bearclover – Alternatives 1, 4 and 5

Multiple studies have shown that in the first few growing seasons, bearclover reduces conifer height growth by about 50% (Tappeiner and Radosevitch 1982; McDonald and Everest 1996; McDonald and Fiddler 1999). During these first years of growth, the ability of confer seedlings to grow in height and diameter is greatly dependent on root development and the ability of the seedling to access soil moisture (Grossnicle 2005). Lack of water stresses conifer seedlings by causing decreased root expansion, which reduces resource collection and causes losses in growth that are seldom made up (McDonald and Fiddler 2010). It has been well substantiated that suppressing competing vegetation during the first few years of tree development has the greatest impact on the survival and growth of conifer seedlings (Balandier et al. 2006; McDonald and Fiddler 2010). In a study near Mount Shasta, white fir seedlings were released with herbicides every year for the first 3 years and for the first 6 years after planting; however, treating for the first 6 years provided no significant gain over treating for the first 3 years (McDonald and Fiddler 2001). Conversely, when competing vegetation was not controlled during the first 3 years, but each year 4 to 6 years after planting, seedlings had statistically smaller average diameters than if released each year for only the first 3 years. In another study, 5-year old Douglas-fir seedlings grown with and without competition showed that the root biomass of seedlings in a free to grow environment was 9 to 22 times larger than those grown amongst sprouting

shrubs whereas seedlings exposed to competition produced virtually no new root biomass (McDonald and Fiddler 2010). This critical period of early competition control has also been demonstrated in a study of northern conifers, which showed that the vegetation community and growth patterns established during the first 3 to 5 years after planting were relatively consistent through the subsequent 5 years (Wagner and Robinson 2006).

Although herbicide release treatments would not permanently eliminate bearclover or other competing vegetation (Tappeiner and Radosevitch 1982; Tesch and Hobbs 1989; McDonald and Fiddler 2010), herbicide release treatments would provide several years of free to grow conditions that would allow conifer seedlings to develop robust root systems that can effectively compete with other vegetation as it reestablishes following cessation of herbicide applications. Tappeiner and Radosevitch (1982) found that bearclover aggressively recovered after a single herbicide application, but this short-term setback still resulted in increased conifer growth. Multiple herbicide applications following planting will likely cause greater dieback of bearclover roots and rhizomes, which will increase the time necessary for bearclover to recover and afford conifer seedlings a longer period of reduced competition. After multiple applications of herbicides, it will likely take several years for most other competing vegetation to regain densities similar to current levels. By that time, however, conifers will be able to effectively compete for soil resources as well as slow shrub expansion with shade. Based on this critical period, herbicide release treatments in bearclover shrub types were modeled using free to grow conditions for the first 5 years. After 5 years, other competing vegetation was introduced back into the model to simulate some level of competition after release treatments stop.

No Control of Bearclover – Alternatives 2 and 4

While initial lower bearclover cover substantially increases conifer seedling survival; competition from bearclover can greatly reduce young conifer growth. After 3 growing seasons, plots that had lower initial bearclover cover versus plots with higher initial bearclover cover experienced 50% and 56% lower tree heights compared to a free to grow plot (Tappeiner and Radosevich 1982). After 19 years, the effect of bearclover on tree heights amounted to 67% and 72%, respectively. Although McDonald and Fiddler (1999) reported taller average tree heights at age 11 than that reported by Tappeiner and Radosvich (1982), they did not have a free to grow study plot to compare the effect of bearclover on conifer growth; therefore, this study could not be used to deduce the percent reduction in conifer growth that might have resulted from bearclover competition during the first several years of conifer development. The taller tree heights may have been a result of site conditions or other environmental factors. After modeling the natural conifer regeneration in a free to grow scenario, tree heights were adjusted according to bearclover abundance to reflect the findings of Tappeiner and Radosevich (1982). That is, all conifers in bearclover shrub types were modeled in RCONIFER without competing vegetation. Tree heights were then adjusted according to the abundance of bearclover. In bearclover shrub types with greater than 30% bearclover cover, free to grow tree heights were reduced by 72% after 20 years. In bearclover shrub types with less than 30% bearclover cover, tree heights were reduced by 67% after 20 years. Tree diameters were adjusted using species specific regressions of tree height-diameter relationships based on model outputs.

Partial Control of Bearclover – Alternative 3

While hand grubbing will kill the above ground portions of bearclover, belowground rhizomes will readily sprout and conifer seedlings will still experience some level of competition. Bearclover has been documented to aggressively recover following a single application of a single herbicide application (Tappeiner and Radosevich 1982). This is likely because a single herbicide application may reduce live root and rhizome biomass, but not completely kill the plant. Prescribed fire has resulted in similar effects by killing the aboveground portions of the plant, but the following spring it vigorously resprouts (McDonald and Everest 1996; McDonald et al. 2004). The same would be true for hand-grubbing bearclover, but manual treatment would likely be even less effective. To

effectively control bearclover, both the below-ground and above-ground portions of the plant must be repeatedly disrupted (McDonald et al. 2004). For example, repeated herbicide applications reduce both the photosynthetic tissues and root biomass. Hand grubbing, even multiple times would only kill the top portion of the plant.

Although manual treatments would not directly kill bearclover roots, it would reduce root vigor as the plant would repeatedly experience reduced photosynthetic function and need to invest belowground resources into growing aboveground portions of the plant. Therefore, hand grubbing bearclover does provide some relief from soil moisture depletion that gives conifer seedlings an opportunity to make gains in root and shoot growth (Tappeiner and Radosevich 1982). Because the growth model RCONIFERS does not include bearclover, the effect of manual release treatments were based on the findings of Tappeiner and Radosevich (1982) for a single herbicide treatment. This study observed a small increase in conifer growth over the control study plots. These effects were compounded 5 times to account for 5 years of reduced competition from bearclover. A single herbicide treatment increased conifer mean annual height growth 0.056 feet between years 3 and 19 compared to the control (Tappeiner and Radosevich 1982). Assuming each additional year of release would increase mean annual height growth the same amount, 5 years of release would result in conifer heights of about 10 feet at age 19 as opposed to 6.2 feet with a single release. If free to grow conifers grew to 18.7 feet by age 19, then the effect of bearclover even with 5 releases that kill only the aboveground portions of the plant would result in a 46.5% reduction in height growth. Diameter growth was adjusted using the same height-diameter growth relationship described previously.

Prescribed Fire in New Plantations

Prescribed fire would only be introduced to new plantations after 10 years if fuels conditions meet specific criteria (EIS, 3.05 Fire and Fuels). As a result, it is likely that some new plantations would not meet these criteria. Where prescribed fire is introduced, conifer mortality would likely vary considerably, but burning would not proceed if conifer mortality is expected to exceed 20%. Given the potential for some plantations to not meet suitable burning criteria and the potential for minimal conifer mortality, conifer densities were not adjusted for prescribed fire; therefore, projections of stand density are representative of higher potential conifer densities.

Stand Attributes

Species composition, trees per acre and volume were calculated based on either RCONIFERS outputs or using FVS default equations. Pre-Rim Fire CWHR type, size and density within the project area were determined using CWHR spatial data (CDFW 2014b). The Rim Fire caused significant changes to CWHR classifications, which have not been remapped. Expected changes described in Table 4 of the Rim Recovery Vegetation Report provided the basis for the Rim Recovery project analysis (USDA 2014). For consistency, this analysis is based on the same expected changes. During this analysis, however, some errors in the CWHR pre-fire data were discovered. The errors occurred primarily in the plantations created in the 1990s after the 1987 Stanislaus Complex Fire. Many of the plantations were incorrectly coded as grass or shrub vegetation types because they were either in the process of being prepared for planting or were recently planted when the imagery used for creating the CWHR dataset was created. The differences in the acreages between this analysis and the Rim Recovery analyses are based on corrections by the Forest GIS staff on the CWHR classification in areas with recently discovered errors. Post-fire CWHR vegetation types were assumed to remain unchanged unless active reforestation or natural regeneration occurs. If converted to conifer forest, then it is assumed to be either ponderosa pine (PPN) or Sierran mixed conifer (SMC). CWHR size and density was projected for all analyses in FVS using an AddFile developed by FVS staff (Rebain 2005).

Changes in SDI are evaluated for existing plantations, new plantations and in areas with natural conifer regeneration. SDI was calculated using the equation developed by Reineke (1933) for trees

greater than or equal to 1 inch dbh. Long and Shaw (2012) recommend this SDI calculation for mixed conifer stands that are not compositionally "pure" (i.e., greater than 80% of basal area composed of either true firs and Douglas-fir or ponderosa pine and Jeffrey pine). Depending on stand attributes and age, other methods of calculating SDI may be more appropriate. Values of SDI produced using the Reineke (1933) equation are essentially equal to other equations for younger even age-aged stands, but increasingly diverges with increasing skewness of the diameter distribution (Long and Shaw 2012). Therefore, the Reineke (1933) equation is appropriate for calculating SDI for this analysis because plantations and natural regeneration is generally going to be even-aged during the analysis period and basal area rarely composed of a single species.

Thin Existing Plantations

Common stand exam (CSE) data for this analysis was downloaded from the Natural Resources Management Natural Resource Information System (NRM NRIS) Field Sampled Vegetation Database (FSVeg). All data was collected between 2005 and 2013 (prior to the 2013 Rim Fire). Only plots representing conifer or mixed hardwood-conifer types were used to simulate effects of thinning and growth in existing plantations. All other CWHR vegetation types were assumed unchanged throughout the analysis time frame. CWHR size and density in non-conifer vegetation types are expected to increase over time, but proposed treatments in these types will primarily focus on fuels reduction and not significantly alter tree structure if trees are present. A total of 843 plots located within the Rim Fire perimeter were processed using the Western Sierras variant of FVS (Dixon 2002). Each plot was categorized by CWHR vegetation type based on the relative proportion of conifer and hardwood canopy cover (CDFG 2005). FVS was used to classify each plot according to size and density (Rebain 2005). Plots were then processed together as stands depending on CWHR size and density to represent CWHR classifications that occur in proposed thinning units.

Thinning was simulated in FVS based on the ICO thinning guidelines (EIS, 2.01 How the Alternatives Were Developed) and CWHR size and density classifications. For size classes 3 and less the ICO thinning guidelines would likely result in an average residual tree density of about 128 trees per acre. Residual tree densities in larger CWHR size classes would be about 105 trees per acre. These distinctions were made based on the assumed current spacing relative to size. For example, the majority of the larger size classes are located in the Granite plantations, which are older than the 1987 Stanislaus Complex plantations. Many of the Granite plantations were thinned prior to the Rim Fire. Target tree spacing was typically about 18 to 20 feet between trees. Conversely, the younger Complex plantations have experienced only minimal pre-commercial thinning and some have never been thinned. Stand exams prior to the Rim Fire showed that tree densities ranged between 200 and 350 trees per acre in these plantations, suggesting that tree spacing was between about 7 by 14 feet to 14 by 14 feet. The ICO thinning guidelines call for creating 6 small clumps and 2 large clumps per acre with individual trees spaced about 25 feet apart. Prescribing these tree spacing guidelines to plantations that have different distances between trees would result in a range of clump footprint sizes despite the clumps having about the same number of trees. For example, if trees are about 20 feet apart, then a clump of 6 trees would have a footprint of about 2,600 square feet. If the trees were closer, then a smaller area would be required to achieve a clump with the same number of trees. If clumps tend to be small in area, then more space would be available for individual trees and result in higher residual tree densities. Based on these guidelines then, smaller CWHR size classes were estimated to have more residual trees per acre after thinning. While prescribed fire is proposed within existing plantations to maintain low fuel levels prior to mechanical operations, burning would not occur unless conditions would allow for target tree densities to be achieved. Follow-up mechanical treatments would likely be necessary to further reduce tree densities and to remove small standing dead trees; therefore, thinning to target residual tree densities was simulated in all existing plantations one year after prescribed fire.

Affected Environment

The project area encompasses 41,933 acres within the Rim Fire perimeter with elevation ranges from about 3,000 to 7,000 feet. Predominant topographic features include steep river canyons, intermittent tributary drainages and broad sloping benches. Elevational differences in river canyons can range from 800 to 1,300 feet in less than half a mile, but such topographic features occur primarily outside project treatment units. Prior to the Rim Fire, forests within the project area consisted of Sierran mixed-conifer, mixed hardwood-conifer and conifer plantations. Other vegetation included annual and perennial grasslands, chaparral and hardwood types.

Historic Forest Structure

Historic forest structure provides a baseline that can be used to help inform forest management decisions regarding reforestation, density management and how forest structure is expected to change over time. While variables such as climate change introduce uncertainty, the past can provide some insight into how forest structure, function and processes might respond to changing environmental conditions. Several historical accounts and reconstructions of historic structure and composition of Sierran mixed-conifer forests exist. These sources describe a wide range of conditions. This variation is evident in the accounts of Sudworth (1900). Sudworth described distinct forest structures within the low to middle elevations on the Stanislaus National Forest. Ponderosa pine was the most abundant conifer in the middle elevations comprising 45 to 55% of most stands and up to 90% of stands at lower elevations. Sugar pine was less common, but still comprised 5 to 25% of the species composition. Incense cedar and white fir typically comprised 20 to 30% and 30 to 40%, respectively. White fir densities were typically highest at elevations between 4,000 to 5,000 feet and much lower and inconsistent at lower elevations. California black oak comprised a much smaller amount in most stands, with the highest proportions (5 to 10% of stands) occurring at low to mid elevations. Sudworth (1900) noted that there were some areas dominated by open conditions with very large (greater than 30 inches dbh), old-growth pine. In contrast, he also noted abundant dense patches of pine regeneration and dense stands of yellow pine 25 to 50 years old; however, frequent fire was noted as continually thinning such stands and often killing most of the regeneration. Lydersen et al. (2013) reconstructed reference forest conditions on the Stanislaus-Tuolumne Experimental Forest. Fire records indicated that the mean fire return interval was about 5 years; however, the last recorded fire occurred 40 years prior to the collection of the 1929 forest mapping data, so fire exclusion may have already become a factor. Lydersen et al. (2013) estimated that 40 years without fire resulted in an increase of tree densities from about 54 to 133 trees per acre. This ingrowth of smaller trees was possible given the presence of mature trees as seed sources and the absence of fire; however, Lydersen et al. (2013) caution that historic small tree densities were likely underestimated due to limitations related to their dataset.

In contemporary mixed-conifer forests with relatively intact fire regimes, Lydersen and North (2012) found high levels of conifer seedlings and saplings (486 to 4,087 per acre). While seedling and sapling density was very high, densities of larger trees ranged from 45 to 134 trees per acre depending on topographic slope position. Collins et al. (2015) evaluated timber transects conducted in 1911 that were located within the Rim Fire perimeter (in the vicinity of Evergreen Road and Peach Growers). These records suggest areas with even lower tree densities (11 to 32 trees per acre) existed. Seedlings and saplings (trees less than 6 inches dbh) were also recorded along these transects, and similar to the findings of Lydersen and North (2012), occurred at significantly higher densities than larger trees (369 to 637 seedlings/saplings per acre, personal communication, Brandon Collins). The large difference in densities of conifer regeneration and established trees suggests that fire, other disturbances and competition significantly regulated the number of small trees that grew to maturity. In a Jeffery pine mixed-conifer forest with a relatively intact fire regime, Minnich et al. (2000) found that 30 to 60% of pole sized trees survived wildfire. Another study found that about 50% of trees less than 8 inches dbh survived fire (Stephens et al. 2008).

While frequent fire likely killed large portions of conifer regeneration, historic accounts of dense 20 to 50 year old pine stands suggest that a fair amount of conifer regeneration escaped wildfire during early stand development (Sudworth 1900). In contrast to studies that attribute increases in small and intermediate sized trees to the introduction of fire exclusion (Collins et al. 2015; Lydersen and North 2012; Lydersen et al. 2013; Minnich et al. 2000), others argue that a dominance of young- and intermediate-aged stands prior to fire exclusion were actually common because they would regenerate following periodic disturbances that caused significant tree mortality (Odion et al. 2014). Baker (2014) asserts that while there were open, park like stands, Sierran mixed-conifer forests were generally dense and prone to high-severity fire. His results suggest that forest openings and areas with relatively low tree density accounted for about 17 to 30% of the area. He contends that these sparsely forested areas are evidence of high-severity fire that occurred because of dense forest conditions. Although his estimate of mean historic tree density (about 119 trees per acre) is within the range suggested by Lydersen and North (2012), Baker (2014) notes that about 16% of the forested area exceeded 162 trees per acre and about 3% had 405 to 3,642 trees per acres, which suggests that there were areas of relatively dense mixed-conifer forest.

In terms of landscape forest structure, there is no clear answer to the question of how much fire-initiated early seral vegetation is desirable or what proportion of the landscape should be in the early seral stage and how large the patches should be (Coppoletta et al. 2015). Some studies have concluded that the effects of long-term fire exclusion, railroad logging and other past management actions are now contributing to an increasing trend in high-severity fire and area burned in forests of the western United States (Miller et al. 2009; Miller and Safford 2012). As a result, it has been suggested that management needs to focus on replacing high-severity fire acres by increasing the acres of low- and moderate-severity fire in low and middle elevation forests like yellow pine and mixed-conifer (Mallek et al. 2013). Additionally, mounting evidence is showing that climate change exacerbates the negative correlation between increasing acres of high-severity fire and decreasing acres of low- and moderate-severity fire (Westerling et al. 2006; Lenihan et al. 2008; Westerling and Bryant 2008; Littell et al. 2009; Lutz et al. 2009). Under the current and predicted climate scenarios, early seral conditions have been predicted to increase on the landscape (Lenihan et al. 2008). Despite this body of evidence, some assert that there is still a large deficit of high-severity fire in Sierran mixed-conifer forests and because of fire suppression, “even doubling of fire would still be far lower than the historic levels” (Odion and Hanson 2013, p. 20). This school of thought views the lack of high-severity fire as a threat to Sierra Nevada forests (Baker 2014; Hanson and Odion 2014; William and Baker 2012). As a result of this deficit, they argue that post-fire vegetation, not impacted by activities such as salvage logging and reforestation, has become increasingly rare and should be protected (DellaSala et al. 2014; Odion and Hanson 2013).

Some methodologies used to reconstruct historic forest structure and how they relate to fire severity have been called into question. Baker (2014) used similar data and methodology as William and Baker (2012) to determine historical forest structure and fire severity extent. Fulé et al. (2014) identified important errors in basic assumptions and methods in William and Baker (2012): the use of tree size distributions to reconstruct past fire severity and extent is not supported by empirical age-size relationships or by studies that directly quantified disturbance history in dry western forests. Fulé et al. (2014) also note that while William and Baker (2012) asserted surprising levels of heterogeneity in their reconstructions of stand density and species composition, their data are not substantially different from many previous studies which reached very different conclusions about subsequent forest structure and fire behavior changes. In response to Fulé et al. (2014), William and Baker (2014) defend their methodologies and findings, entrenching the scientific disconnect in conclusions regarding historic forest structure and high-severity fire in western dry forests.

As noted by Fulé et al. (2014), despite reaching very different conclusions about fire behavior and subsequent forest structure, the results of William and Baker (2012) are not substantially different

from other studies with regard to forest structure. The same can be said with regard to results of Safford (2013) and Baker (2014). LANDFIRE BpS models provide an estimate of course landscape forest structure that may have occurred historically and is based on both biophysical conditions and current climatic conditions (LANDFIRE 2009). Safford (2013) used the BpS models to estimate landscape forest structure for Sierran mixed-conifer forests. He estimated that 5 to 15% of the landscape was in a mid-seral closed canopy structural class as shown in Table 3.01-3. This range is similar to the estimate made by Baker (2014) for relatively dense conditions (16%) – assuming values presented by Baker (2014) would classify as mid seral. Safford (2013) also estimated 15 to 20% of the landscape was in an early seral condition, which overlaps with the Baker (2014) estimate of 17 to 30% for forest openings and scattered trees. This difference in the early seral successional class is likely a result of methodologies and interpretations regarding the historic occurrence of high-severity fire. The LANDFIRE BpS models used by Safford (2013) provide a general range of conditions that are within range or close to estimates made in other studies of reference conditions and historical accounts (Franklin and Fites-Kaufman 1996; Bonnicksen and Stone 1982; Show and Kotok 1924).

Pre-Fire Vegetation

LANDSCAPE FOREST STRUCTURE

Table 3.01-5 shows the following CALVEG cover types comprised the majority of the vegetation impacted by the Rim Fire: conifer and mixed hardwood-conifer. Other vegetation cover types included hardwood, shrub, herbaceous, barren and other non-vegetation types.

Table 3.01-5 CALVEG Cover Types within Rim Fire Perimeter, Stanislaus and Project Area

CALVEG Cover Type	Rim Fire Perimeter	Stanislaus National Forest	Rim Reforestation
CON - conifer	138,249	75,042	29,179
MIX - mixed hardwood-conifer	44,078	27,576	7,466
HDW - hardwood	30,756	22,055	1,473
SHB - shrub	27,448	22,724	3,260
HEB - herbaceous	14,267	6,141	533
OTHER - barren, urban, water	2,515	991	22
Total (acres)	**257,314**	**154,530**	**41,933**

Table 3.01-5 shows hardwood, shrub and herbaceous types accounted for about 30% of vegetation depending on the scale. Barren and non-vegetation types accounted for less than 1% of the total Rim Fire footprint. Although the Rim Fire impacted non-conifer cover types, they are expected to maintain relatively the same CWHR type despite fire severity and changes that might occur in CWHR size and density.

CALVEG cover types provide a general classification of what the dominate vegetation is. Within each cover type, some portions of non-conifer CWHR vegetation types may exist. For example, a conifer cover type might encompass both CWHR conifer types as well as CWHR shrub and hardwood types. As noted by Safford (2013), shrub cover occurring within the conifer biophysical setting is considered an early seral structure. Therefore, pre-fire landscape forest structure was estimated for CALVEG conifer and mixed hardwood-conifer cover types because they most closely resemble the LANDFIRE biophysical settings used by Safford (2013) to classify landscape structure of Sierran mixed-conifer forests. Within these two cover types, CWHR vegetation type, size and density was used to classify seral structural classes in Table 3.01-3. Compared to Safford's (2013) estimates of landscape forest structure, Figure 3.01-1 shows a deficit of the following structural classes prior to the Rim Fire, regardless of land ownership: early, mid seral open canopy and late seral open canopy. As a result, there was an excess of mid and late seral closed canopy. The majority of the excess closed canopy was in the mid-seral size class with more than 3 times the historic estimate across all lands and just over 2 times the historic estimate on NFS lands.

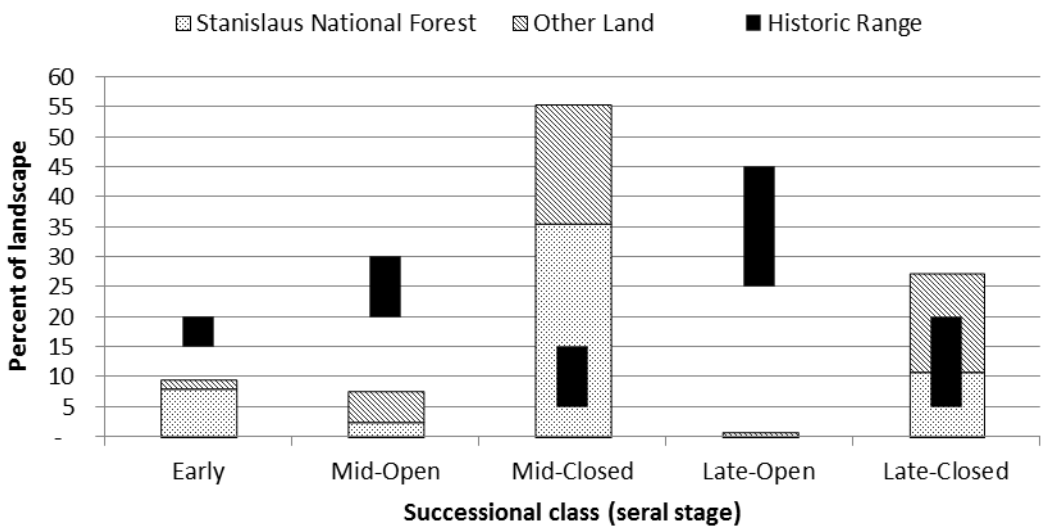


Figure 3.01-1 Average Landscape Forest Structure before Rim Fire Compared to Historic Conditions

CALIFORNIA WILDLIFE HABITAT RELATIONSHIPS

Vegetation within the Rim Reforestation project area was primarily conifer or mixed hardwood-conifer (36,645 acres). Other general cover types (e.g., shrub, hardwood and herbaceous) were omitted as to not overestimate the proportion of early seral conditions within the conifer and mixed-hardwood conifer cover types. Table 3.01-6 summarizes the CWHR vegetation type, size and density within these two conifer cover types within the project area. Tables 3.13-1 and 3.13-2 show the size and density class definitions. Ponderosa pine and Sierran mixed conifer were the dominant vegetation types, comprising just over 78% of the vegetation within the project area. The majority of the forest structure was also closed canopy (73%) with only 3.8% in a mid-seral open canopy condition. About 23% was dominated by early seral conditions: shrub (2,336 acres), grass (1,856 acres), and tree seedlings and saplings (4,333 acres). This proportion of early seral conditions slightly exceeded the estimated historic proportion for early seral conditions (15 to 20%, Safford 2013).

Table 3.01-6 Acres by Pre-Rim Fire CWHR Vegetation Type, Tree Size, Canopy and Seral Stage

Successional Class (Seral Stage)	**Size and Density**	**Shrub and Grass**	**MHW**[1]	**BOP**	**MHC**	**JPN**	**LPN**	**PPN**	**SMC**	**WFR**	**Total**
Early	1 and 2, all densities	4,192	52	3	118		5	3,607	548		8,525
Mid-Open	3S	3			8			91	48		150
Mid-Open	3P		3	2	43		4	413	431		898
Mid-Open	4S			1		2		20	31		53
Mid-Open	4P				6	2	6	113	188	5	320
Mid-Closed	3M			10	163		21	1,095	666		1,954
Mid-Closed	3D		70	3	155	8	41	1,130	328		1,735
Mid-Closed	4M			24	214		26	1,209	1,445	3	2,922
Mid-Closed	4D		61	35	2,193	3	33	4,788	9,172	11	16,296
Late-Closed	5M								15		15
Late-Closed	5D		53		217		129	313	3,065		3,777
	Total	**4,195**	**239**	**77**	**3,117**	**14**	**266**	**12,778**	**15,938**	**19**	**36,645**
	% of Total	**11.4**	**0.7**	**0.2**	**8.5**	**0.04**	**0.7**	**34.9**	**43.5**	**0.1**	**100.0**

[1] MHW=montane hardwood; BOP=blue oak-foothill pine; MHC=montane hardwood-conifer; JPN=Jeffrey pine; LPN=lodgepole pine; PPN=ponderosa pine; SMC=Sierran mixed conifer; WFR=white fir. All within CALVEG cover types conifer and mixed-hardwood conifer

Areas Proposed for Reforestation

At most, the total area proposed for reforestation is 26,009 acres (includes natural regeneration and deer habitat enhancement units). Dispersed throughout the proposed reforestation units were patches of the following general CALVEG cover types: hardwood (691 acres), shrub (1,722 acres), herbaceous (85 acres) and other non-vegetation cover types (19 acres) that add up to 2,517 acres. About 93% of the area proposed for reforestation was classified as CALVEG cover types conifer or mixed hardwood-conifer. Of the conifer and mixed hardwood-conifer cover types, 81% was classified as CWHR type ponderosa pine or Sierran mixed-conifer and 12% was classified as other conifer or montane hardwood-conifer types. Less than 1% was classified as montane hardwood and about 6% was classified as shrub or grass CWHR grass types. In total, there was 2,705 acres (12% of the conifer and mixed-hardwood CALVEG cover types) of early seral structure.

Existing Plantations Proposed for Thinning

Within the Rim Fire perimeter an estimated 17,773 acres of plantations were predominately planted after the 1973 Granite Fire, 1987 Stanislaus Complex Fire and 1996 Rogge Fire. All of these plantations are included in the Rim Reforestation project. A portion of these plantations were classified as hardwood (664 acres), shrub (1,880 acres), herbaceous (144 acres) and other non-vegetation (3 acres) CALVEG cover types. These non-conifer cover types are primarily small patches of distinct vegetation dispersed throughout the plantations. Of the 15,082 acres of conifer and mixed hardwood-conifer cover types, 134 acres were classified as late seral ponderosa pine and Sierran mixed-conifer forest (CWHR size class 5). These late seral conditions were also scattered throughout the existing plantations and were likely small remnant patches that escaped the previous fires.

Rim Fire

The Rim Fire caused extensive vegetative changes. High severity patches were uncharacteristically large and accounted for a larger proportion (35%) of the burned area than historically occurred (Miller et al. 2009). Other areas of low or very low severity continue to have intact forest tree cover, but are experiencing increasing levels of insect- and drought-related tree mortality due to sustained drought conditions (USDA 2015d).

Several factors contributed to the pattern of severity and fire effects: extreme drought, fire weather, years since last fire, elevation and topography, shrub cover, tree species composition and surface fuels (Harris and Taylor 2015; Lydersen et al. 2014; Kane et al. 2015). Historically, these same factors, with the exception of extreme drought, are believed to have resulted in a self-regulating system of interactions between vegetation characteristics and disturbance (North et al. 2009; Lydersen et al. 2012; Harris and Taylor 2015). However, the effects of fire exclusion and drought have altered forest structure and created burn conditions that resulted in historically novel fire effects during the Rim Fire. Both Harris and Taylor (2015) and Lydersen et al. (2014) found that study plots with a greater proportion of shade-tolerant tree species and higher densities of small and intermediate sized trees (conditions often associated with high fire severity), generally burned with lower fire severity. Lydersen et al. (2014) found that elevation and shrub cover contributed the most to high-severity fire. Their study plots with higher shrub cover were primarily those at lower elevations with lower tree densities, whereas the plots with less shrub cover were those at higher elevations with higher tree densities and more white fir. The plots at higher elevations were cooler and moister, which contributed to greater tree canopy cover. Harris and Taylor (2015) reported similar results, finding that fire severity was associated with shrub cover and topographic position. Shrub cover was associated with pine on dry upper slopes and ridges, whereas lower slopes and valley bottoms supported higher densities of trees with cooler temperatures and higher moisture levels. Given the relatively mild fire weather, Harris and Taylor (2015) conclude that cool, moist areas escaped high-severity fire despite higher densities of shade-tolerant species.

Kane et al. (2015) also conducted a study that evaluated biophysical factors of the Rim area. Unlike Lydersen et al. (2014) and Harris and Taylor (2015), this study found that areas associated with greater moisture and productivity actually contributed to the uncharacteristically large high severity burn patches of the Rim Fire. These contrasting findings are likely a result of scale. Kane et al. (2015) explain that larger scales represented by their biophysical predictors "may simply better match the scales at which fire behavior, the biophysical environment, and fuel variations correlate" (p. 70). Kane et al. (2015) compared the fire effects of the Rim Fire to previous fires in the study area using actual evapotranspiration, which is an indicator of productivity. Greater evapotranspiration infers greater productivity, which is associated with moist conditions. Compared to the Rim Fire, previous fires in the study area experienced significantly lower proportions of high severity fire. For pre-Rim fires, actual evapotranspiration was negatively correlated with burn severity. That is, moist and productive sites experienced lower burn severities. In the Rim Fire, however, this trend reversed and actual evapotranspiration correlated with higher burn severities. Kane et al. (2015) attribute this "flip" to the drought drying locations that usually had higher fuel moistures, and the Rim Fire then burned the fuel accumulations that had built up in these areas. An example of such an area is Corral Creek, which is a highly productive site that had a high proportion of mature Douglas-fir, incense cedar and white fir prior to the fire. The most productive areas of Corral Creek were the lower slopes and valley bottoms. Tree heights approached and exceeded 200 feet. Numerous trees exceeded 30 inches dbh and the largest trees exceeded 50 inches dbh. Both the 1987 Stanislaus Complex Fire and the 1996 Rogge Fire burned near this area. The cool moist conditions likely contributed to the area experiencing little disturbance during previous fires; however, Corral Creek experienced 100% tree mortality during the Rim Fire.

Existing Conditions

LANDSCAPE FOREST STRUCTURE

After the Rim Fire, RAVG analysis mapping was completed to assess burn severity and change in vegetation. CWHR habitat classifications were adjusted based on the RAVG burn severity mapping. Figure 3.01-2 summarizes the post-fire landscape forest structure for the entire Rim Fire footprint compared to historic reference conditions (Safford 2013). Both the mid and late seral open canopy successional classes slightly increased, but are still far less than historic ranges. Mid and late seral closed canopy successional classes experienced large decreases. Late seral closed canopy was reduced to just within the historic range across all lands. When considering only NFS lands, late seral closed canopy conditions are slightly greater than the historic minimum; however, there is still an excess of mid seral closed canopy conditions on NFS lands.

Across the entire Rim Fire landscape, almost twice as many acres of mid seral closed canopy conditions exist than the historic range. Given the uncharacteristically large proportion of the landscape that experienced high severity fire, a large portion of the closed canopy conditions were shifted to the early seral successional class. Across all lands, an excess of nearly twice as much early seral conditions, almost 40% or 71,552 acres, now exists compared to the historic maximum of 20%. The majority of the early seral successional structure is on NFS lands (44,975 acres) and consists mostly of mixed chaparral and montane chaparral (36,870 acres). Early seral structure on NFS lands (almost 25%) exceeds the historic range of 15 to 20%. Non-NFS lands are just within the historic range (about 15%).

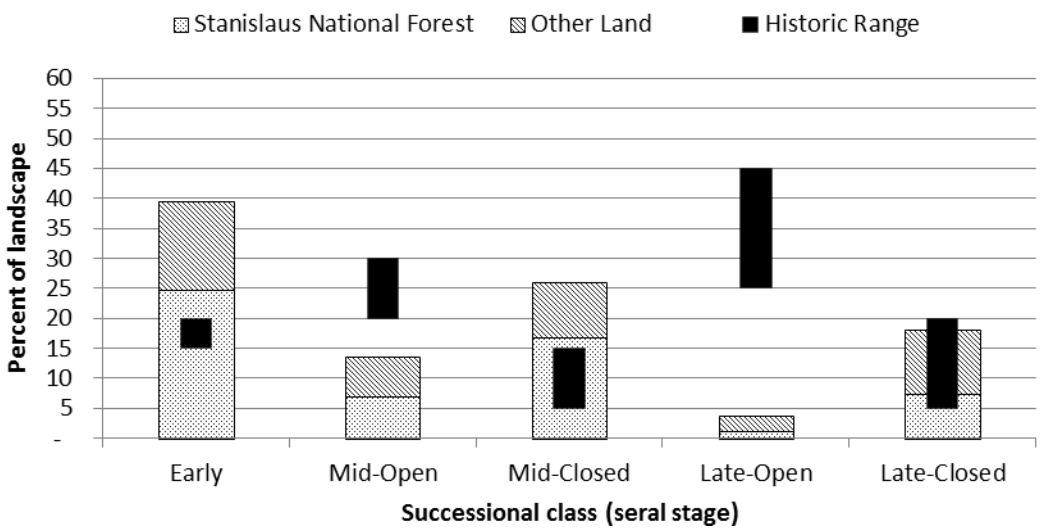


Figure 3.01-2 Average Landscape Forest Structure after Rim Fire Compared to Historic Conditions

COMPLEX EARLY SERAL FOREST

Immediately after the Rim Fire, timber salvage was conducted primarily in large patches of severely burned areas (USDA 2014). In many of these areas salvage operations have either been completed or are currently underway. Hazard tree removal has also been conducted along many forest roads and near structures. However, a considerable amount of acres were moderately or severely burned during the Rim Fire that have not been impacted by salvage or hazard tree operations. The term complex early seral has been defined by some as early seral conditions created by stand replacing disturbances that have not otherwise been altered by humans (DellaSala et al. 2014; Odion and Hanson 2014). That is, no standing dead trees are removed and post-fire vegetation is allowed to develop without human intervention. Stand-replacing events that kill all or most of the dominant trees therein, typically leave behind many biological legacies (e.g., standing dead trees and large down woody debris) that create distinct differences compared to pre-disturbance conditions (Swanson et al. 2011).

Figure 3.01-3 Retention of Standing Dead Trees during Salvage in Femmons Meadow Area

For this analysis, complex early seral forest was defined as all CWHR type, size and density classes within CALVEG conifer and mixed hardwood-conifer cover types that experienced at least 50% basal area mortality and were not impacted by the Rim Recovery project, Rim HT project, or were not located on private lands (i.e., assumed salvage operations conducted on all private lands). Once the Rim Recovery and Rim HT projects are fully completed, a minimum of 41,875 acres of complex early seral forest would remain within the Rim Fire perimeter, of which 19,971 acres is on NFS lands. Based on the definition of seral structural classes defined by Safford (2013), a portion of these complex early seral acres (7,376 acres; 2,808 acres on NFS lands) could be considered mid or late seral open canopy structure because they experienced moderate-severity fire and still have low densities of surviving trees; however, the vast majority of these acres (34,499 acres total; 17,163 acres on NFS lands) resulted from high-severity fire that killed virtually all trees and now make up 48% of the current early seral structure across the Rim Fire landscape (Figure 3.01-2). In other words, almost half of the 71,552 acres of early seral conifer and mixed hardwood-conifer structure within the Rim Fire landscape has not been altered by humans. Based on Safford's (2013) estimate of the proportion of early seral structure, the current amount of complex early seral forest (34,499 acres) nearly equals the historic maximum of 20% of the conifer and mixed-conifer forest within the Rim Fire landscape.

Although salvage and hazard tree operations removed standing dead trees, the Rim Fire Recovery project (USDA 2014) required that 4 to 6 of the largest trees per acre with a dbh of at least 24 inches be retained for wildlife purposes. When considering the diversity of historic tree density and the range of fire severities, an equivalent range of standing dead tree densities likely occurred historically. Dense forest conditions likely resulted in larger numbers of standing dead trees following high severity fire. Conversely, areas with lower tree densities may have resulted in lower numbers of dead standing trees. Collins et al. (2015) determined that large tracks of very low overstory tree densities (11 to 32 trees per acre) existed on the Stanislaus. With such low tree densities, it was likely that areas with only small patches of standing dead trees existed, especially following mixed severity fire conditions. Therefore, the areas that were salvage logged after the Rim Fire still provide some elements of structural complexity despite the effects of mechanical operations. Figure 3.01-3 shows retention of standing dead trees in an area recently salvaged near Femmons Meadow.

Table 3.01-7 Acres by Post-Rim Fire CWHR Vegetation Type, Tree Size, Canopy and Seral Stage

Successional Class (Seral Stage)	**Size and Density**	**Shrub and Grass**	**MHW**[1]	**BOP**	**MHC**	**JPN**	**LPN**	**PPN**	**SMC**	**WFR**	**Total**
Early	1 and 2, all densities	22,819	226	61	64			2,253	402		25,826
Mid-Open	3S	3	1	3	38		6	270	243		563
Mid-Open	3P		4	1	39	1	3	306	299		652
Mid-Open	4S			1	151		7	407	917	3	1,486
Mid-Open	4P		7	8	142		3	480	952	5	1,597
Mid-Closed	3M				18		6	299	192		516
Mid-Closed	3D		1		19	2	2	326	60		410
Mid-Closed	4M				42		2	511	465	2	1,023
Mid-Closed	4D		3	4	263			1,450	1,670	3	3,392
Late-Open	5S				14		9	18	237		277
Late-Open	5P				13		5	22	242		281
Late-Closed	5M								1		1
Late-Closed	5D				19		4	36	562		621
	Total	**22,822**	**241**	**77**	**822**	**3**	**48**	**6,375**	**6,242**	**13**	**36,645**
	% of Total	**62.3**	**0.7**	**0.2**	**2.2**	**0.01**	**0.1**	**17.4**	**17.0**	**0.04**	**100.0**

[1] MHW=montane hardwood; BOP=blue oak-foothill pine; MHC=montane hardwood-conifer; JPN=Jeffrey pine; LPN=lodgepole pine; PPN=ponderosa pine; SMC=Sierran mixed conifer; WFR=white fir. All within CALVEG cover types of conifer and mixed-hardwood conifer

CALIFORNIA WILDLIFE HABITAT RELATIONSHIPS

Table 3.01-7 summarizes the post-Rim Fire CWHR vegetation type, size and density classifications within CALVEG conifer and mixed hardwood-conifer cover types. Large wildfires that have high proportions of high severity fire can convert large areas of conifer forest to shrub vegetation (Collins and Roller 2013; Crotteau et al. 2013; Coppeletta et al. 2015; Nagel and Taylor 2005). The Rim Fire shifted a large portion of the project area into an early seral structure, increasing the abundance of grass and shrub vegetation types from about 11% to over 62% of the project area. Early seral conditions now comprise a significantly larger proportion of the project area compared to historic Sierran mixed-conifer landscapes (15 to 20%, Safford 2013). As a result, the proportion of ponderosa pine and Sierran mixed-conifer vegetation types was significantly reduced from 78% to less than 35% of the project area. Areas of low- to moderate-burn severity slightly increased the proportion of mid and late seral open canopy conditions.

Areas Proposed for Reforestation

The 2,517 acres of non-conifer CALVEG cover types did not significantly change despite high-severity effects during the Rim Fire. The hardwood types likely experienced high tree mortality in some areas, but hardwood species in the project area readily sprout after fire and quickly regain dominance over the site. Within CALVEG cover types conifer and mixed hardwood-conifer (23,492 acres), the Rim Fire significantly reduced the proportion of conifer-dominated CWHR vegetation types. Prior to the fire, conifer types comprised 93% of the vegetation, but now comprise about 20%. Much of the conifer vegetation was shifted to an early seral structure (18,479 acres), mostly consisting of mixed chaparral and montane chaparral (17,375 acres). Patches of mid and late seral conifer forest totaling 5,013 acres are distributed throughout the areas proposed for reforestation. Just over half of these acres (2,857) experienced moderate burn severity and are now in an open canopy condition with scattered conifers. The majority of these patches were incidentally included when drawing boundaries using GIS and account for small portions of units. They are also typically located within close proximity to the edge of high-severity burn patches. Stress caused by the Rim Fire (e.g., crown scorch and severe cambium heating) in addition to extended drought conditions and amplified bark beetle activity has increased levels of tree mortality reducing tree densities or in some cases resulting in 100% mortality. The majority of natural conifer regeneration, as discussed in the following sections, has been observed within close proximity of residual live trees and high-severity patch edges. These areas primarily account for the proposed natural regeneration units (EIS, 2.02 Alternatives Considered in Detail). For the purpose of this analysis, natural regeneration units are analyzed for active reforestation to account for the scenario of greatest potential impact. The 2,517 acres classified as non-conifer vegetation are areas that would be excluded from reforestation activities (e.g., dense oak patches, stringer meadows, lava caps and roads). These areas contribute to the diversity of vegetation and cover across the landscape, but are not included in calculations of landscape forest structure classes (seral stages) within conifer and mixed hardwood-conifer cover types. While discussions of effects may refer to all 26,009 acres proposed for reforestation, it is with the understanding that these non-conifer cover types would not be planted.

Existing Plantations Proposed for Thinning and Prescribed Fire

Overall, the Rim Fire burned with high burn severity across 98,049 acres (USDA 2014e). Despite the uncharacteristically large area of high burn severity, only a small portion of these acres can be attributed to plantations. Of the 17,773 acres of existing plantations, 9,919 acres (56%) experienced less than 50% basal area mortality. An additional 4,015 acres of plantations experienced a mix of burn severities, which left patches of dense plantations among patches of moderate to high burn severity patches. Together, these plantations compose the 13,934 acres (78% of existing plantations) that are proposed for thinning under the Rim Reforestation project.

Several factors likely contributed to the variation of burn severities in the existing plantations. Plantations are often considered to be vulnerable to wildfire because young conifers have thin bark and low branches that are sensitive to scorching (McGinnis 2010). Studies of the Rim Fire, however, have also shown that biophysical conditions and shrub cover seemed to play an important role in fire severity, more so than tree size and density (Lydersen et al. 2014; Harris and Taylor 2015). Similarly, an evaluation of the Granite plantations determined that the biggest risk to plantations are surface and ladder fuels; and therefore, plantations that have an overall continuity of surface fuels are more likely to support large severe fires as opposed to well established plantations with trees that have grown to heights providing distinct separation of surface litter fuels and the canopy, as well as successfully shading the forest floor and minimizing shrub cover continuity (Sapsis and Brandow 1997).

About one third of the plantations proposed for thinning were planted after the 1996 Rogge Fire. Tree sizes in these plantations currently range between 3 and 6 inches dbh with heights between 15 and 20 feet. These plantations were predominately planted at a density of 444 trees per acre. As previously noted, third year survival exams indicate that deep tilling and herbicide treatments result in about 25% mortality during the first few years of tree establishment. As a result, current tree density in these plantations averaged about 325 trees per acre prior to the fire; however, post-fire live tree density is somewhat lower in areas that experienced low- to moderate-severity fire (220 to 300 trees per acre). While some areas experienced higher severity fire, there are still patches with tree densities exceeding 300 trees per acre.

The next oldest plantations were established after the 1987 Stanislaus Complex Fire during the 1990s and also account for about one third of the plantations proposed for thinning. These plantations were planted using similar methods and planting densities as the Rogge plantations; therefore, post-fire tree densities are similar. Because these plantations are almost 10 years older, however, diameters and heights are larger on average. Tree diameters typically range from about 6 to 12 inches dbh with heights of 25 to 35 feet.

The oldest plantations were planted after the Granite fire in the 1970s. Many of these plantations had been thinned prior to the Rim Fire. On average, tree spacing ranges from 16 to 22 feet (about 135 trees per acre post-fire). Tree diameters are considerably larger than many of the Rogge or Complex plantations, ranging from 12 to 18 inches on average with heights of 50 to 70 feet.

Deer habitat enhancement using prescribed fire only is proposed on 1,990 acres. The Jawbone lava cap comprises the majority of these acres (50%). The remainder is a mix of Rogge plantations and hardwood vegetation that experienced a mix of low-, moderate-, and high-severity fire.

NATURAL REGENERATION

The correlation between natural conifer regeneration and the Composite Burn Index (CBI) within areas proposed for reforestation is similar to the findings of Crotteau et al. (2013). The highest conifer seedling densities are in the low- and moderate-severity burn patches. Seedling density in high-severity burn areas is the lowest and most inconsistent as shown in Table 3.01-8. The proportion of plots without conifer regeneration in high-severity burn areas is 71%, which is about twice as much as the low- and moderate-burn severities. Other studies of conifer regeneration and burn severity have observed a similar trend (Collins and Roller 2013). When extrapolated to acres, the total area within proposed reforestation units with and without natural conifer regeneration is estimated at 9,825 acres and 16,184 acres, respectively. These estimates are based on the proportion of null plots vs. stocked plots in the project area, making them rough approximations. While the actual stocked acres likely vary, the large number of plots completed provide a reasonable baseline for estimating the proportion of the project area with substantial conifer regeneration that has the potential to qualify as conifer cover according to CWHR standards (i.e., at least 10 to 25% conifer cover). The large number of plots also provide a statistically reliable estimate of conifer density and species composition to within at least six seedlings per acre based on an alpha level of 0.1. Although it is difficult to quantify and

describe the spatial distribution of natural regeneration based on plot data, the increasingly higher number of null plots in the more severely burned areas suggests a greater proportion of these areas have little to no natural conifer regeneration compared to areas that experienced lower burn severities. The overall average seedling density within each burn severity is shown in Table 3.01-8. Because natural conifer regeneration is typically non-uniform and occurs in clumps, the average seedling density calculated using only stocked plots is also shown. The average based on stocked plots is intended to provide a frame of reference in terms of conifer densities occurring in clumps as opposed to an overall average density that gives an inaccurate picture of uniformly distributed regeneration. Despite the majority of the areas proposed for reforestation experiencing high-severity fire, small patches of low-severity fire are distributed throughout treatment units. Many of these patches were included incidentally when delineating unit boundaries in ArcMap and would obviously not require reforestation.

White fir seedlings are more abundant than any other species; likely because it is a prolific seeder (Zald et al. 2008). It comprises about 50% of the natural conifer regeneration across most burn severities. Douglas-fir is the second most abundant and comprises about 25% of the conifer regeneration consistently across all burn severities. Pine species are the most abundant in unchanged areas and overall much lower in density compared to white fir and Douglas-fir. All pine species combined only account for about 13% of natural conifer regeneration. Ponderosa pine is the most common pine species, accounting for 11% while sugar pine contributes only about 1% and other pine species (grey pine, knobcone and Jeffrey pine) account for less than 1%. Some seedlings were unable to be identified, but only amounted to about 4% of all conifer regeneration.

Table 3.01-8 Natural Conifer Regeneration within Areas Proposed for Reforestation

Burn Severity (CBI) **and Condition**	**Total Acres**[1]	**Total Plots**	**Null Plots**	**Percent Null Plots**	**Average Seedlings Per Acre**		**WF** (%)	**IC** (%)	**DF** (%)	**PINE** (%)	**UNK** (%)
					All Plots	**Stocked Plots**[2]					
Unchanged: logged	30	2	0	0	2,925	2,925	40	34	14	12	0
Unchanged: unlogged	209	14	3	21	1,779	2,264	16	22	31	32	0
Unchanged: subtotal	**239**	**16**	**3**	**19**	**1,922**	**2,365**	**20**	**24**	**27**	**28**	**0**
Low: logged	96	4	0	0	2,575	2,575	52	0	0	48	0
Low: unlogged	1,275	53	9	17	4,967	5,983	52	6	18	11	14
Low: subtotal	**1,371**	**57**	**9**	**16**	**4,804**	**5,705**	**52**	**6**	**17**	**12**	**13**
Moderate: logged	1,496	83	35	42	881	1,524	8	23	44	23	3
Moderate: unlogged	4,193	237	89	38	3,134	5,019	59	4	27	8	2
Moderate: subtotal	**5,662**	**320**	**124**	**39**	**2,553**	**4,168**	**54**	**5**	**29**	**9**	**2**
High: logged	5,343	365	260	71	126	437	26	2	30	40	3
High: unlogged	13,393	915	645	71	304	1,031	51	1	24	19	4
High: subtotal	**18,736**	**1,280**	**905**	**71**	**255**	**870**	**48**	**1**	**25**	**22**	**4**
Total: logged	7,058	545	295	65	298	850	19	14	34	30	2
Total: unlogged	18,951	1,219	746	61	1,074	2,768	55	4	25	11	5
Grand Total	**26,009**	**1,673**	**1,041**	**62**	**865**	**2,291**	**52**	**5**	**26**	**13**	**5**

CBI=Composite Burn Index; WF=white fir; IC=incense cedar; DF=Douglas-fir; PINE=ponderosa pine, Jeffrey pine, sugar pine, grey pine and knobcone pine; UNK=unknown conifer

[1] Acres include deer reforestation units and natural regeneration units. Logged and unlogged acres calculated based on number of plots with evidence of logging activity.

[2] Calculated using only plots with natural conifer regeneration.

Although plots were not visited both pre- and post-salvage logging, a large portion of the plots were completed in areas that were expected to be salvaged. This plot data suggests that salvage and fuels reduction operations that have occurred in the project area have reduced conifer regeneration density by 72% and oak regeneration by 26%. While mechanical operations reduced conifer seedling density, it minimally impacted the proportion of null plots. When comparing logged to unlogged plots, less than a 4% difference exists in the proportion of plots with no conifer seedlings. Table 3.01-8 summarizes logged and unlogged plots together as well as separately, showing the existing conditions

of conifer regeneration at the time surveys were completed. Unlogged plots were adjusted accordingly in the growth modeling to account for scheduled salvage and fuels operations. Although oak and other hardwood regeneration has been impacted by mechanical operations, it is still quite abundant across all burn severities because of its ability to sprout from stumps and roots. Hardwood regeneration density across the areas proposed for reforestation averages about 70 seedlings or sprouts per acre, about 78% from stump and root sprouts. A disproportionate abundance of California black oak (80%) is regenerating compared to other hardwood species. Canyon live oak is the second most abundant (17%), while other hardwood species (interior live oak, blue oak, big leaf maple, dogwood, willow and alder) account for only about 3% of regenerating hardwoods.

Many factors, such as available moisture, soil insolation and rodent herbivory, influence post-fire seedling establishment (Crotteau et al. 2013; Nathan et al. 2002; Saigo 1969; Vander Wall et al. 2005), but the foremost requirement for natural conifer regeneration is seed source (Bonnet et al. 2005). Shatford et al. (2007) found that distance to seed source for Douglas-fir and white fir was not necessarily a limiting factor; however, pine species have heavier seed (Laacke 1990; Oliver and Ryker 1990), which likely decreases the rate of long-distance seed dispersal. While regeneration tends to be highest in low to moderate severity patches (Crotteau et al. 2013), Bonnet et al. (2005) found that seedling establishment was very successful in patches of high-severity that were within about 40 feet of unburned forest canopy, but decreased exponentially toward the center of burn patches. Distance to nearest potential seed source was recorded during surveys in the Rim Fire area.

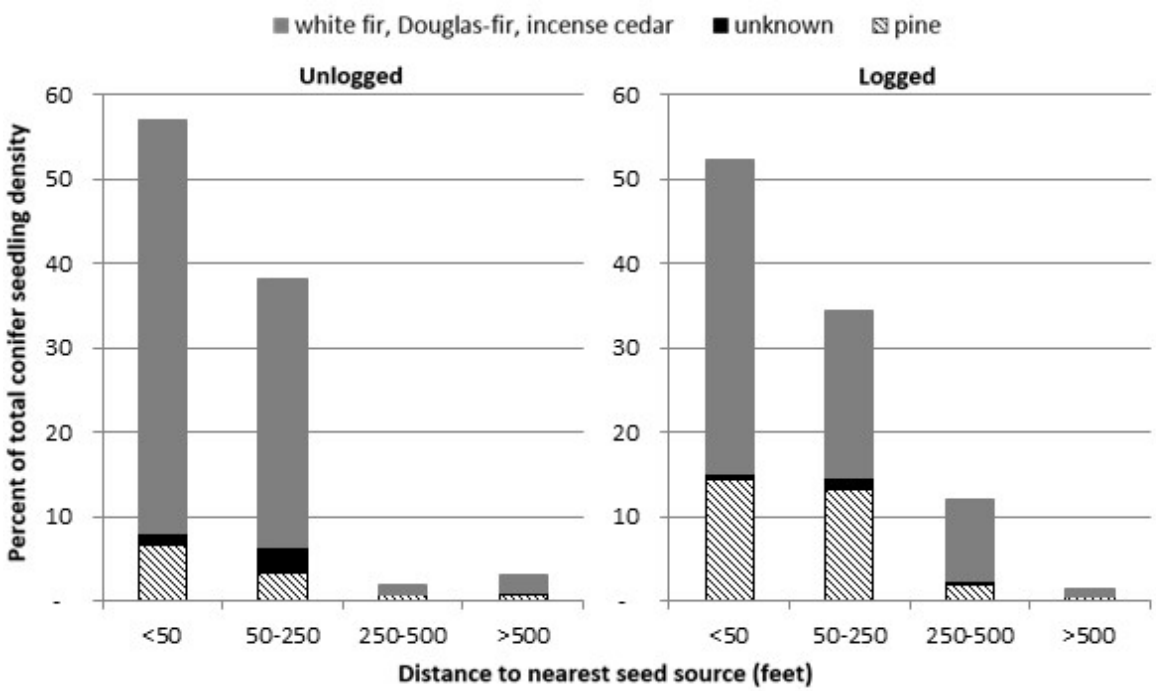


Figure 3.01-4 Distribution of Natural Conifer Regeneration within Areas Proposed for Reforestation

Figure 3.01-4 summarizes the density of natural conifer regeneration within units proposed for reforestation and shows the difference between logged and unlogged plots. The largest proportion of seedlings is within 50 feet of live mature conifer trees. The proportion of natural conifer regeneration sharply declines as the distance to live mature conifers increases. This trend is true for all conifer species. White fir, Douglas-fir and incense cedar comprise 71% and 68% of the conifer regeneration occurring at distances of 250 to 500 feet and greater than 500 feet from live trees, respectively.

UNDERSTORY VEGETATION

The relationship between shrub cover and burn severity followed general trends observed after other large wildfires (Collins and Roller 2013; Coppoletta et al. 2015; Crotteau et al. 2013). Overall cover of shrubs, grasses and forbs has a positive correlation with burn severity. Within unchanged, low-, moderate- and high-severity areas vegetation cover averages 4%, 23%, 50% and 61%, respectively. Grasses and forbs make up about 25% of the vegetative cover in both moderate- and high-burn severity. Bearclover and deerbrush are disproportionately more abundant than any other shrub species across all burn severities; however, moderate-severity has a more even distribution of species than high-severity. In moderate-severity, bearclover and deerbrush each comprise about one fifth of the vegetation (on average about 9% cover). In high-severity, bearclover accounts for about one third (20%) of the vegetative cover and deerbrush one sixth (10%). Bearclover has an extensive system of below-ground rhizomes and tap roots that aggressively respond to above ground disturbances such as fire (McDonald et al. 2004). In addition to effectively sprouting after disturbances, *Ceanothus* species (such as deerbrush, whitethorn, and buckbrush) commonly store more seed in the soil than other shrub species (Knapp et al. 2012), which has likely contributed to the high abundance of deerbrush in the project area. The majority of the plots that were dominated by deerbrush occurred in drainages, while mid-slopes, ridges and emergency travel routes were typically dominated by bearclover.

Manzanita species are the third most abundant shrub species. Knapp et al. (2012) found that after ceanothus species, manzanita was the next most abundant shrub seed stored in mixed-conifer forest soil. Numerous other shrub species exist throughout the project area including, but not limited to cherry, ribes, toyon and poison oak. Manzanita and these species individually account for less than 1% of the total vegetative cover, but where present species like manzanita and buckbrush can occur in large dense patches. Overall, other shrub species account for a much smaller portion of the total vegetative cover compared to bearclover or deerbrush.

Environmental Consequences

Alternative 1 (Proposed Action)

DIRECT AND INDIRECT EFFECTS

Tree Size and Species Composition

Reforestation

Reforestation efforts, including site preparation and release, plant seedlings and take advantage of natural regeneration resulting in predictable conifer survival and growth (Fiske 1981). Artificial regeneration allows for more control over competing vegetation and tree species composition. Ground disturbing activities (such as fire, tractor piling, hand cutting and shredding) during site preparation can stimulate sprouting of several shrub species (e.g., bearclover, *Ceanothus* species and some species of manzanita). Similarly, these activities can stimulate germination of shrub seeds stored in the soil, making control of competing vegetation challenging (Knapp et al. 2012).

Deep Tilling and Forest Cultivation

Areas site prepped with deep tilling and forest cultivation would have better survival than those areas not tilled. Deep tilling kills above-ground portions of shrubs and breaks-up below-ground roots and rhizomes (McDonald et al. 2004), which reduces initial competition for water during the first growing season after planting. As displayed in Table 3.01-4, initial conifer survival is estimated at about 75% in areas prepped with deep tilling and followed up with herbicide releases. Reduced competition would also increase conifer growth.

Herbicides

Herbicide site preparation and release treatments would effectively reduce shrub and grass competition for the first five years allowing establishment and development of conifers in a free to

grow environment. Initial herbicide release treatments would require the most widespread application because shrubs and other herbaceous vegetation would have relatively high cover. These first treatments would kill both above- and below-ground portions of shrubs and other understory vegetation, reducing their abundance. Each subsequent treatment would likely require less herbicide be dispensed to achieve desired levels of competing vegetation (i.e., less than 10,000 cubic feet per acre, USDA 2010a). Areas that are deep tilled would also require less herbicides be applied to effectively control competing vegetation.

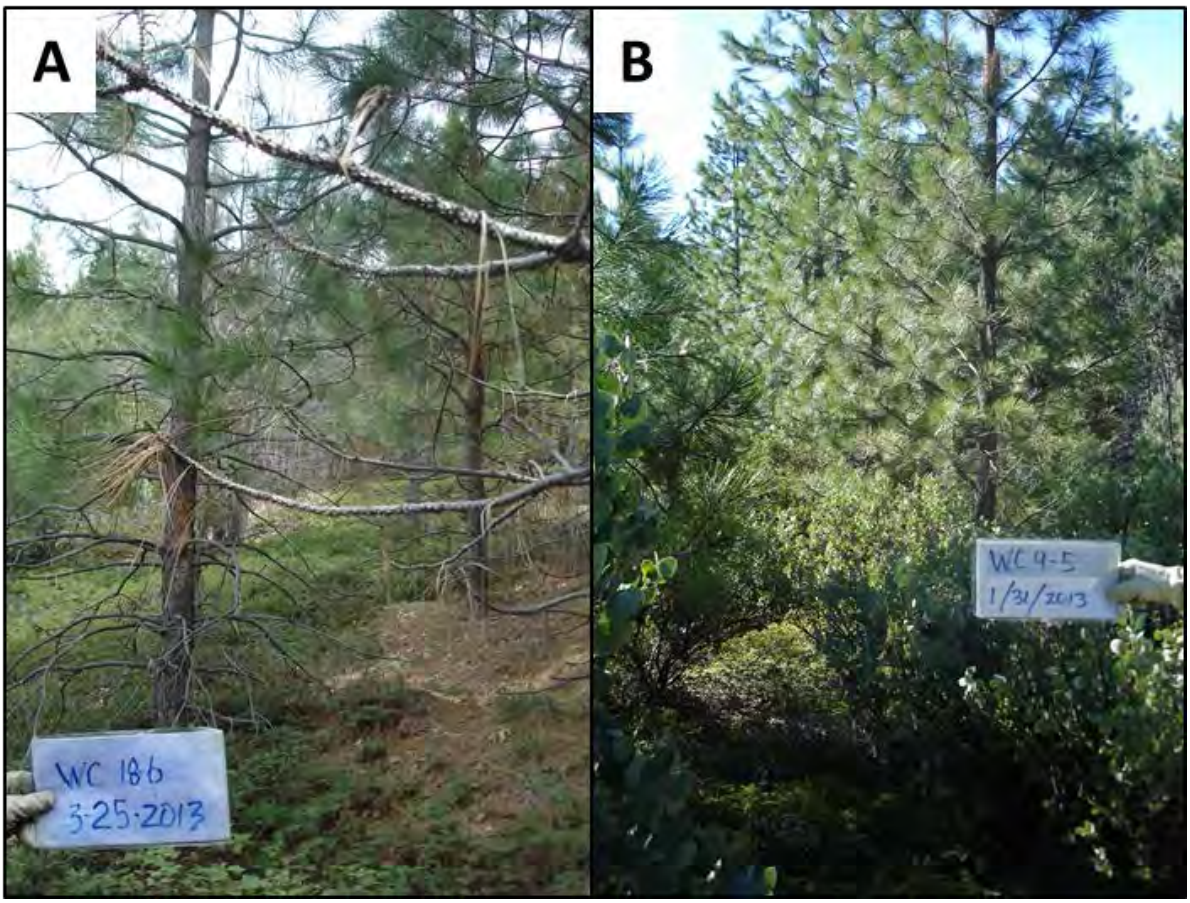


Figure 3.01-5 Understory Vegetation in 15-Year Old Walton Cabin Plantations Prior to the Rim Fire

The beneficial effects of herbicide release treatments on understory vegetation would likely last several years after release treatments stop, resulting in reduced abundance of shrubs, grasses and forbs. Experience on the Stanislaus National Forest as well as documented research, have shown understory vegetation does recover even after multiple herbicide applications (DiTomaso et al. 1997; McDonald and Fiddler 2010; Sapsis and Brandow 1997). Given the abundance of seed stored in the soil and numerous areas excluded from herbicide application (e.g., oak buffers, sensitive areas, riparian buffers, heritage exclusions), the diversity of understory plant communities are not at risk of being lost from the project area and can effectively recover and spread throughout treatment areas over time. Figure 3.01-5 shows bearclover and manzanita growing in 15-year old plantations where both deep tilling and herbicides were used after the 1996 Rogge Fire. Increases in herbaceous species richness have been observed following cessation of herbicide treatments (DiTomaso et al. 1997). The use of herbicides effectively reduces shrub cover, which increases solar radiation to the forest floor. As a result, grasses and forbs that would normally be outcompeted by the aggressive post-fire response of shrubs are able to persist in larger proportions (McDonald and Abbott 1997); however,

such increases may result from invasion by undesirable species such as cheat grass (McGinnis et al. 2010). Herbicide treatments for noxious weeds, as well as subsequent reforestation release treatments, would help to minimize spread and establishment of invasive exotic plant species.

Prescribed Fire

Introducing prescribed fire while plantations are still fairly young (about 10 years old) would also facilitate reestablishment of native shrubs species. Prescribed fire stimulates germination of *Ceanothus* species that require fire for seed scarification, promoting recovery of the herbaceous understory (Kauffman and Martin 1991). Prescribed fire, which would also slow development of large decadent shrubs after herbicide release treatments stop, would maintain small openings in shrub canopies and allow a greater abundance of grasses and forbs to persist within plantations for a longer period of time.

Tree Development and Densities

Tree crown characteristics are influenced by shrub competition and tree density. As trees grow and expand lateral branches, they must physically compete with neighboring trees and other vegetation for growing space as well as water and nutrients belowground. Higher stand densities reduce the amount of time before trees grow large enough to begin competing with each other for physical growing space. Like shrubs, higher tree densities can also reduce diameter and height growth, but depending on how closely trees are spaced this effect may not occur for 10 to 30 years. The effect of shrub competition is more pronounced in earlier years before inter-tree competition begins. Oliver (1979) evaluated the effect of different tree spacings with and without shrub removal. Trees spaced greater than 12 feet apart that were free of shrub competition did not experience reduced height or diameter growth during the 12 year study period; however, shrub competition significantly reduced diameter, height, crown width, and branch diameter. Trees spaced 6 feet apart did experience inter-tree competition for light and belowground resources. Similarly, trees grown close together such as in clumps are more likely to shed portions of their crown that are shaded by neighboring trees and vegetation than trees that are open grown. For the most part, however, tree spacing proposed under the Alternative 1 planting patterns is large enough that most trees would not experience significant declines in growth or self-pruning during the first 10 to 30 years.

Table 3.01-9 Alternative 1: Tree Density and Characteristics at Years 10, 20 and 60

Species	Year	PCR Mean	CBH Mean	DBH Mean	DBH Minimum	DBH Maximum	Height Mean	Height Minimum	Height Maximum	TPA Mean	TPA Minimum	TPA Maximum
Conifers	10	93	1	0.7	0.0	1.4	7.3	2.7	10.7	176	104	207
Conifers	20	70	7	4.3	0.4	6.8	23.2	6.7	34.0	175	104	207
Conifers	60	61	29	13.9	4.8	24.1	74.8	23.6	111.1	157	98	179
Hardwoods	10	78	2	1.0	0.0	3.0	9.6	4.6	22.5	50	38	63
Hardwoods	20	44	13	4.0	2.6	5.9	23.9	16.1	37.3	20	15	25
Hardwoods	60	29	27	6.1	4.0	10.6	38.3	23.0	53.8	16	10	22

PCR=percent crown ratio; CBH=canopy base height in feet; DBH=diameter at breast height in inches; TPA=trees per acre

Table 3.01-9 summarizes tree development and densities for Alternative 1. Growth simulations estimate that conifer diameters would average just over 4 inches at breast height at year 20 and almost 14 inches by year 60. Average conifer heights at years 20 and 60 would be about 23 feet and 75 feet, respectively similar to heights observed by McDonald and Abbott (1997) in study plots with low competing vegetation. As shown in Table 3.01-9, tree heights at year 10 in their study were slightly taller on average (10.6 feet and 8.7 feet in plots with low and moderate abundance of competing vegetation) compared to estimates made using RCONIFERS. Table 3.01-9 shows at year 18, McDonald and Abbott (1997) observed slightly shorter tree heights (14.9 to 20.9 feet) compared to 20-year estimates made using RCONIFERS. The RCONIFER estimates of dbh and height are also

comparable to other plantation growth models produced for northern California (Oliver and Powers 1978).

Tree density would vary by slope position because of the different planting patterns that are summarized in the EIS, Table 2.02-2. Lower tree densities would occur on fuelbreaks and primary ridges, while the highest densities would be planted in drainages where conditions are cooler, moister and better suited for denser forest. After accounting for initial mortality and oak buffers, tree densities would range from 104 to 207 trees per acre during the first 10 years. Overall, the average tree density at year 10 is estimated at 176 trees per acre. Assuming no major disturbances occur (e.g., stand replacing fire, mechanical thinning, insect outbreak), background or normal mortality rates would reduce conifer density to 157 trees per acre on average and range from about 98 to 179 trees per acre by year 60. Areas with lower tree density or larger spacing between clumps would likely develop higher shrub cover in the long-term. Lydersen and North et al. (2012) found that tree densities on ridges and mid-slopes were often associated with greater shrub cover.

Pine species would comprise a greater proportion of the plantations. Over the next 60 years, pine would comprise about 70% of the density, of which, 15 to 20% would be sugar pine. Oak and other hardwoods would initially comprise about 22% of overall tree density. Hardwood species generally never exceed heights of more than 60 to 80 feet, but conifer species commonly reach heights of 100 to 200 feet. Growth simulations predict that conifers would begin to overtop oak and other hardwoods after 15 to 20 years. As the conifers increase in size and can more effectively compete for water and light, hardwood density will decrease. Competition between hardwoods would also contribute to declining hardwood densities as more dominate individuals expand into the growing space of neighbors. By year 60, hardwoods are projected to comprise about 8-10% of the tree density, which is comparable to historic accounts of oak density at about the same elevation. On the Stanislaus, Sudworth (1900) estimated that the highest densities of California black oak occurred between 3,500 and 4,500 feet elevation, comprising 5 to 10% of most stands. The majority of the Rim Reforestation project area is less than 5,000 feet.

Various planting patterns are proposed based on slope position. The distribution of planting patterns reflects patterns observed by Lydersen and North (2012), with higher tree densities and larger tree-clumps located on lower slopes and lower tree densities and smaller tree-clumps located on upper slopes and ridges. Planting seedlings in clumps also reflects natural patterns of regeneration to an extent. For example, Sudworth (1900) described conifer regeneration as being patchy, but the patches of regeneration were growing among intact forest and not within large patches of severely burned forest. While planting patterns would promote a clumpy distribution of seedlings and saplings in the short-term, key structural components would be absent until the development of large trees, which with their presence, create conditions suitable for natural conifer regeneration. Based on an extensive review of research related to tree spatial patterns, Larson and Churchill (2012) describe the common driving mechanisms of tree spatial patterns in dry conifer forest of the western U.S. Key components include clumps of trees across a range of sizes and ages (e.g., patches of old large trees to patches of regeneration), widely spaced individual trees and openings. The process of frequent fire maintains this heterogenic structure by creating new openings, which then naturally regenerate from seed produced by nearby patches of mature trees. This vegetation-disturbance dynamic is reflected in other research in Sierran mixed-conifer forests. Lydersen and North (2012) note that areas burned at lower intensity promoted tree regeneration and areas burned at higher intensity created shrub habitat. Crotteau et al. (2013) observed greater conifer regeneration in areas with low to moderate burn severity, while areas that burned with high-severity had lower conifer regeneration and high shrub cover. Crotteau et al. (2013) attribute the lower shrub cover in areas that burned with low severity to high shade provided by residual mature trees. Lower shrub abundance and nearby seed sources then produce greater amounts of patchy conifer regeneration. The uncharacteristically large high-severity patches created by the Rim Fire left few if any remaining large trees and an abundance of shrub

cover; therefore, restoration of the vegetation-disturbance dynamic described by Larson and Churchill (2012) would not occur until a diversity of tree ages and sizes have developed. Planting patterns may result in a more patchy structure of seedlings and saplings in the short-term, but would not accelerate the development of large trees compared to simpler planting patterns (e.g. Alternative 5) as shown in Table 3.01-9 and Table 3.01-13. As the trees develop over the next 50 to 60 years, it is the future disturbances (e.g., fire, mechanical thinning, insects and diseases), future natural regeneration and variation in microsite productivity that will promote development of fine-scale heterogeneity in tree age, size, densities and spatial arrangements.

Nutrients and Water Availability to Trees

The effectiveness of site preparation and release treatments (especially herbicide treatments) strongly correlates with increased water availability to trees. Competition with dense shrubs slows the initial growth of conifer seedlings (Conard and Radosevich 1982a; Lanini and Radosevich 1986; McDonald and Fiddler 2001; Oliver 1990; Stuart et al. 1993). Although shrubs have been shown to improve nitrogen fixing and soil fertility (Conard et al. 1985; Busse 2000; Busse et al. 1996), water availability has been shown to override the beneficial effects of improved nutrient availability on tree growth. Powers and Ferrell (1996) concluded that on droughty sites, vegetation control plus fertilizer did not improve tree growth beyond vegetation control alone. Powers and Ferrell (1996) also found that on more productive sites, fertilizer application without vegetation control boosted shrub growth and blocked trees from the beneficial effects of increased nutrient availability. Similarly, Powers and Reynolds (1999) found that on sites that were both droughty and infertile, trees responded to increased water availability before increased nutrient availability. Others have affirmed the benefits to tree growth resulting from early shrub control and increased water availability (McDonald and Fiddler 2010; Stephenson 1990; Zhang et al. 2006). Reforestation treatments increase water availability to conifer seedlings; thereby, setting stands on a trajectory to conifer dominance and the creation of structures (e.g., large trees) more like those that existed pre-fire. Reforestation treatments also provide opportunities to restore forest structure consistent with historic conditions (Sensenig et al. 2013).

Climate Change Influences

Concern has been raised over whether or not it is appropriate to conduct reforestation throughout the project area given changes in site suitability and potential vegetation shifts resulting from warming climate trends. All tree species have a threshold for drought, beyond which no growth occurs (Hinckley and Scott 1971; Royce and Barbour 2001; Waring and Cleary 1967). With trends of increasing temperatures and decreasing snow packs in the Sierra Nevada (Lutz et al. 2010; Safford et al. 2012; Thorne et al. 2008), understanding the drought-related thresholds of tree species can help us understand the potential extent of vegetation change that may occur because of changing climate, and assess the sensitivity of individual species within a particular location (Hannah et al. 2002). Species that are currently close to their drought threshold or water-balance range limit may be affected by increasing water deficits (Breshears et al. 2009). Lutz et al. (2010) evaluated the effect of increasing summer water deficit on individual tree species found in Yosemite National Park. The metric used by Lutz et al. (2010) for measuring water balance was the predicted ratio between actual evapotranspiration and potential evapotranspiration as affected by warming climate trends. They compared the water balance where each tree species is found within Yosemite to the North American range of each species to determine which species might be close to their drought threshold. Of all the species evaluated, western white pine and mountain hemlock were the only two species with study plots clustered around the lower North American threshold for water balance; and therefore, are the most at risk. Lutz et al. (2010) concluded that most of the same species found in the Rim Reforestation project (i.e., California black oak, Douglas-fir, incense cedar, ponderosa pine, red fir, sugar pine and white fir) may occur in locations that will put them at increasing risk of water deficit-related mortality. That is, the study plots where these species were located were clustered toward the

arid end of their North American range; however, within the North American range each species was found on sites with a lower water balance than that observed in Yosemite. For example, Lutz et al. (2010) estimated the North American water balance range for ponderosa pine to be about 0.32 to 0.99, with a mean of about 0.6. The majority of the areas where ponderosa pine is located in Yosemite occurred between 0.42 and 0.63, with an average of about 0.53. While the average in Yosemite is lower than the average of the North American range, the majority of ponderosa pine sites are still well above the lower threshold. Therefore, some sites may reach their drought threshold for ponderosa pines as temperatures increase, but these would likely be areas with the lowest site productivity. Based on this information, reforestation in some areas under Alternative 1 may experience future water deficit-related mortality. As noted by Lutz et al. (2010), however, the scale used for modeling climate is still fairly coarse and underpinned by uncertainty. Topography in the project area is highly variable, and like most areas in the Sierra Nevada range, results in highly variable microclimatic conditions that drive vegetation structure and composition (Lydersen and North 2012); therefore, future climate related mortality is likely to reflect the fine-scale heterogeneity of the biophysical landscape. While conifer species may eventually die-off in isolated areas that are planted, many areas dispersed across the project area would likely remain habitable to conifers and persist despite warming temperatures. Furthermore, areas of poor site quality would likely be dropped during unit layout or skipped during site preparation and planting.

In light of potential future water deficit-related mortality caused by climate change, the concept of focusing reforestation efforts within climatic refugia, or envelopes, has been raised. Loarie et al. (2008) explain that climate trends in California will likely result in shifts in vegetation diversity and distribution. From a conservation perspective, species that are able to expand their ranges should garner the least concern; however, species with shrinking ranges will likely require the most attention and human assistance. As exemplified by Lutz et al. (2010), mountainous areas are expected to harbor species with shrinking ranges and may provide future refugia that act as a life boats for biodiversity into the next century (Loarie et al. 2008). Groves et al. (2012) argues for a broader approach to incorporating climate change into conservation planning because conserving climate refugia represents only a partial solution to climate change adaptation and relies largely on climate projections and all their associated uncertainties. Additionally, coarse-scale climate envelope models may overestimate or misrepresent the projected extinction rates for a given area because they often fail to capture topographic or microclimatic buffering (Groves et al. 2012; Willis and Bhagwat 2009). Millar et al. (2007) suggest that redundant planting across a range of environments may capture fine-scale microclimates, increase diversity and provide an ecological buffer that spreads risk rather than concentrates it. Furthermore, mixed-conifer refugia are often characterized as cool, moist sites that are relatively productive. Productive sites with infrequent fire will tend to burn at higher severities (Lutz et al. 2012). Kane et al. (2015) demonstrated that this was the case with the Rim Fire. Because fire responds rapidly to climate, effects of fire will likely overshadow the direct effects of climate on tree species distributions and migrations (Dale et al. 2001; Flannigan et al. 2000). Therefore, Alternative 1 would help spread the risk that fire poses to declining conifer species in lower elevations (especially sugar pine) across a greater proportion of the Rim Fire landscape; capturing a higher frequency of suitable microclimates that may otherwise remain dominated by shrubs and hardwoods for decades or centuries. It would also increase connectivity of conifer forest, which would enhance ecosystem integrity by increasing the ability of tree species to move and adjust to future climate conditions and disturbances (Groves et al. 2012).

Genetic Diversity

Some literature has raised concerns over genetic diversity being low in plantations due to planting commercial species and using nursery genomes (DellaSala et al. 2014). Region 5 currently does not use seed orchards, but instead depends solely on collection of wild seed grown in nurseries. Isozyme analyses comparing naturally established conifer seedlings and nursery-grown seedlings have found

no difference in genetic diversity (Shimizu and Adams 1993). Genetic variation would be reintroduced from seed collections within the local seed zone. This would be especially beneficial for promoting rust-resistant sugar pine within the project area. The trend of declining sugar pine has raised significant concern over maintaining genetic diversity in this species to ensure its long-term capacity to adapt and survive future natural disturbances and large scale threats to sustainability (Burns et al. 2008; Zeglan et al. 2010). Natural regeneration of sugar pine currently makes up only 1% of all natural conifer regeneration within the 26,009 acres proposed for reforestation. In addition to sugar pine, species composition would include a mix of planted conifers (Jeffrey pine, ponderosa pine, white fir, incense cedar, Douglas-fir and giant sequoia) and natural regeneration. The proportion of species planted would be tailored to fit the site. For example, white fir and Douglas-fir would be favored on northern aspects and drainages as opposed to a heavier mix of pine on southern aspects and ridges. Overall, however, a greater proportion of pine (about 50% ponderosa pine and 20% sugar pine) would be planted to reflect historic species composition (CREP 2008; Sudworth 1900) and because pine is less sensitive to drought than other conifer species (Ferrell et al. 1994; Hurteau et al. 2007).

Existing Plantations

Prescribed fire and mechanical thinning in existing plantations would maintain low fuel levels and remove small, excess standing dead and live trees to reduce stand densities and move plantations toward the desired ICO structure. Together these treatments would achieve the desired surface and ladder fuel loading, as well as reduce live tree density. If burn conditions do not permit safe use of prescribed fire, mechanical thinning would proceed to ensure inter-tree mortality does not occur. Tree species diversity would be promoted by removing ponderosa pine and leaving sugar pine, white fir and incense cedar and thinning around oak. On average, only about 300 board feet per acre would be removed because the majority of the merchantable size trees would be retained. Although the majority of these plantations have relatively small trees (less than 15 inches dbh), prescriptions would retain a portion of the largest standing dead trees to provide wildlife structures and future large down woody material. Prescribed fire and mechanical operations would initially kill aboveground portions of shrubs, grasses and forbs; however, reduced shading of the forest floor combined with prescribed fire and soil disturbance would stimulate sprouting and seed germination of understory vegetation. Treatments would also stimulate conifer regeneration (Zhang et al. 2013) and promote fine-scale structural complexity as new cohorts of conifers establish over time.

Thinning would improve residual tree health and vigor by removing mostly small, suppressed and intermediate sized trees that would otherwise compete for water and growing space. Reducing the number of smaller trees would initially increase the average tree diameter and height of the plantations and increase the growth of the remaining trees. Over the next 20 years, tree diameter would increase across all thinned plantations, with stand averages ranging from about 9 to 26 inches dbh in the youngest and oldest plantations. Average tree heights would range from about 40 to 105 feet, respectively. After 60 years, average stand diameters would range from 16 to 31 inches and height would range from 80 to 125 feet. As shown in Table 3.01-15, the density of large trees (greater than 24 inches dbh) across all thinned plantations would average about 21 trees per acre, which is about 15% more large trees than would occur if no action is taken.

Alternative 1 would treat about 1,990 acres, with prescribed burning only, to enhance deer habitat. Prescribed burning would cause low levels of tree mortality and potentially increase vigor of residual trees. Large areas with no conifers dominated by oak, shrubs and grasses would experience short-term effects such as aboveground mortality, but would readily sprout during subsequent growing seasons. Burning would break up the horizontal continuity of shrubs, providing more light, soil moisture and growing space for grasses and forbs. Reductions in shrub and oak cover would occur in the short-term. Over the next decade, however, sprouting shrubs and oaks, as well as new germinates stimulated during burning, would expand their crowns and reoccupy growing space.

Stand Density Index

Reforestation

Reforestation accelerates the establishment of conifer forest. Once trees have reached a size where shrub competition no longer significantly influences diameter and height growth, competition among trees for water, light and physical growing space becomes an important factor in forest development. Reforestation treatments decrease the time until inter-tree competition becomes a factor. SDI is used to evaluate the effect of inter-tree competition on forest health and vigor. Alternative 1 proposes higher planting densities based on topographic slope positions. The slope positions proposed for the highest tree densities are in drainages and on mid-slopes in areas with an old forest mosaic desired condition. These areas of higher planting densities (303 trees per acre) would require less time until the onset of inter-tree competition. Drainages and old forest mosaic mid-slopes would just reach a SDI of about 200 by year 40. By year 50 drainages and old forest mosaic mid-slopes would have a SDI of 275 and 267, respectively. Other mid-slope positions (open canopy mosaic outside SFMAs and old forest mosaic inside SFMAs) would also exceed a SDI of 230 by year 50 (SDI 244). In total, about 17,960 acres would exceed a SDI of 230 within 50 years. Therefore, it would take about 40 to 50 years before 70% of the project area would start to show increased bark beetle-related mortality (Oliver 1995; Oliver and Uzoh 1997). After 60 years, the remaining mid-slope positions (5,471 acres) exceed a SDI of 230; however, some slope positions that are planted with lower tree densities (fuelbreaks, primary ridges and emergency travel routes) would not reach a SDI of 230 within 60 years. Even in the absence of thinning or other major disturbances, none of the slope positions reach the bark beetle induced maximum SDI of 365 within the 60-year analysis timeframe.

Although primary ridges do not reach a SDI of 230 within the next 60 years, ridges typically have shallow soils, lower soil moisture availability, and are more exposed to solar radiation. As temperatures and summer water deficits increase with changing climate, these areas are the most vulnerable to drought-induced mortality. Having lower SDI levels would improve tree resilience during periods of extended drought. Lower SDI levels, however, suggest lower tree canopy cover and increased shrub cover would occur, which would increase the risk of high-severity fire in these areas during extreme drought (Harris and Taylor 2015; Lydersen et al. 2014).

Existing Plantations

After prescribed burning and thinning, a total of 2,868 acres of plantation (about 20% of thinned plantation acres) would still exceed a SDI of 230. A portion of these acres (105 acres) are small patches of late seral closed canopy forest that would exceed a SDI 365 and eventually reach a SDI of 528 during the 60 year analysis time frame. These patches of older remnant forest are relatively small, distributed throughout the existing plantations, and would experience minimal effects from treatments. The remaining acres that exceed a SDI of 230 are primarily the older Granite plantations that also have relatively larger trees, but cannot be thinned to a lower SDI because the Forest Plan Direction requires that thinning not reduce canopy cover by more than 30% (USDA 2010a, p. 36). Thinning is estimated to reduce canopy cover between 20 and 30% in the majority of these older plantations. While SDI would still exceed 230, it would be reduced below the insect-induced maximum SDI of 365. These plantations would remain below a SDI of 365 for about 20 years and reach a SDI of 419 during the 60 year analysis time frame. By year 20, an additional 916 acres (7% of thinned plantation acres) would increase in SDI and exceed the threshold of 230. In the absence of major disturbances, an additional 4,213 acres would not reach a SDI of 230 until sometime between year 20 and 60 years after thinning. The remaining plantations would remain in a relatively open canopy structure with low conifer densities over the next 60 years. By year 60, a range of stand densities, ages and structures would exist. Some openings created during prescribed fire would likely remain open and dominated by shrubs and hardwoods. Other openings would begin to regenerate with young conifers, promoting uneven-aged structures. Aside from the 105 acres of late seral closed

canopy forest, an additional 1,894 acres would exceed a SDI of 365, but no other thinned plantations would exceed the SDI threshold of 365 during the analysis period of 60 years.

During the first 10 to 20 years after thinning, residual trees would benefit from reduced stand densities, which would increase water and nutrient availability. Residual trees would increase in growth and vigor, which would accelerate development of larger trees during the analysis time frame. Reduced competition would ease water stress during dry summers and reduce tree mortality during periods of more severe drought (D'Amato et al. 2013). Although a large portion of these thinned plantations (about 6,000 acres) would not exceed a stand-level SDI of 230 during the analysis time frame, as the residual trees increase in size and occupy more growing space over the next 20 years, SDI levels would increase. Subsequently, inter-tree competition would slowly intensify and some stands would begin to show more evidence of insect-related mortality as they approach SDI levels of 230 (Oliver 1995; Oliver and Uzoh 1997). Mortality would be more severe during periods of drought when trees are water-stressed. If insects and drought are not a factor, SDI levels would continue to approach 260 to 280 (50 to 60% of maximum SDI) in some plantations. At these densities competition-related mortality would increase and result in “self-thinning” (Long and Shaw 2012). Mortality would typically occur in denser patches and on poorer sites where competition is more intense between trees, which will further promote structural heterogeneity and uneven-aged structures. As trees succumb to insect and competition, SDI levels would decrease and surviving trees would benefit from reduced live tree densities.

Future conifer regeneration was not included in these projections; therefore, SDI values are based on existing tree densities. Thinning and prescribed fire would likely stimulate some level of conifer regeneration that would eventually contribute to higher SDI levels. Future management decisions would need to assess the need for long-term management (e.g., mastication or additional prescribed fire) to address changing circumstances that are outside the scope of this analysis and project decision space.

Future Management Feasibility

Conifer forest that is stocked with high densities of small trees can result in high SDI levels. Conversely, high SDI levels can result from low densities of large trees because large trees require more growing space and resources for maintaining health and vigor. For this reason, conifer forests inevitably experience SDI increases, especially in the absence of significant disturbances. For this reason, thinning intermediate sized trees has been noted as sometimes being necessary to reduce competition, accelerate large tree development, and create desired fine-scale heterogeneity in Sierran mixed-conifer forests (North et al. 2009). Assuming prescribed burning or other disturbances over the next several decades do not significantly reduce tree density, plantations would eventually reach SDI levels that induce competition-related mortality. Ideally then, a scenario that balances the culmination of increasing SDI levels with the development of merchantable sized trees would provide future generations opportunities for offsetting management costs. Such a situation would avoid development of plantations with submerchantable trees that require thinning before undesirable SDI thresholds are exceeded.

Stand densities under Alternative 1 would begin to reach suitable SDI levels for requiring thinning at about the same time that merchantable sized trees have developed. Long and Shaw (2012) recommend thinning when SDI reaches 50 to 60% of maximum SDI. Assuming maximum SDI for ponderosa pine dominated mixed-conifer stands is 524, thinning should occur around the time they reach a SDI of 260. At this SDI, stands would show an increase in bark beetle-related mortality and also begin self-thinning; therefore, thinning would capture some of the potential timber volume while still maintaining desirable tree densities and sizes, as well as creating canopy openings that provide suitable conditions for pine regeneration. Drainages and old forest mosaic mid-slope plantations would be the first to reach a SDI of 260. This would occur by year 50 on 10,786 acres. At that time, thinning from below to an SDI of 170 (about 30% of maximum SDI) would remove 4,998 board feet

per acre and leave 125 to 130 conifers per acre. By year 60, an additional 12,644 acres of mid-slope plantations would meet thinning criteria; however, given the lower tree densities in these plantations, trees must grow larger to achieve a SDI of 260. Therefore, thinning to the same SDI target requires removing larger trees with more volume (8,664 board feet per acre) and thinning to a lower residual conifer density (89 to 93 conifers per acre). As noted previously, some plantations (2,578 acres of fuelbreaks and primary ridges) under Alternative 1 would not reach a SDI of 230 within the next 60 years; and therefore, would not meet thinning criteria. Maintaining these strategically placed fire management areas would not produce cost offsets from removal of merchantable timber because existing tree densities would be too low. Future maintenance of ladder fuels and shrub cover in these areas could be completed at cost using prescribed fire and mechanical operations.

Existing Plantations

Prescribed fire and thinning the existing plantations today would improve opportunities for future management. Reductions in tree density would increase growth and vigor of residual trees and result in 15% more large trees per acre greater than 24 inches dbh compared to no treatment as shown in Table 3.01-15. In general, a greater number of commercial size trees would exist in the future compared to Alternative 2. As stands continue to increase in density and accumulate ladder and surface fuels, removing some of the merchantable intermediate sized conifers would help offset costs associated with future fuels reduction operations and wildlife habitat improvement projects.

Change in Forest Structure

Reforestation

Reforestation treatments would accelerate the establishment of conifer forest in uncharacteristically large patches of high-severity fire within a matter of years as opposed to decades or potentially longer. Once conifers are established, mid and late seral successional structures would begin to develop. Alternative 1 would reforest 16,184 acres more than are currently regenerating naturally. In areas that are regenerating naturally (9,825 acres), treatments would help favor a desirable mix and density of conifer species. Of the 26,009 acres proposed for reforestation, 3,096 acres are currently considered complex early seral forest created by high-severity forest fire (not including areas of moderate burn severity) because they were severely burned and have not been altered by humans (DellaSala et al. 2014; Odion and Hanson 2014). Reforestation activities would reduce the “complexity” of these areas by removing standing dead trees and controlling understory vegetation, but would also accelerate the development of conifer forest. The other areas proposed for reforestation are primarily severely-burned forest that have been impacted by salvage logging or hazard tree removal under the Rim Recovery and Rim HT projects and would otherwise remain dominated by dense shrub thickets with scattered patches of dense white fir and Douglas-fir.

Over the next 30 years, all plantations would progress from early seral successional classes into mid seral successional classes as the average tree size increases. On average, canopy cover would range between 10% and 39%, which is still an open canopy condition. By year 40, most of the plantations (90%) would reach canopy covers of greater than 40%. Given the variation of planting densities and topographic positions, less than 3,000 acres would remain in a mid-seral open canopy condition with 25 to 39% canopy cover after 40 years. These open canopies would occur on the fuelbreaks, primary ridges and emergency travel routes. In the absence of major disturbance, all plantations would reach a mid-seral closed canopy condition by year 50 with canopy covers of 40 to 69%.

In the long-term, future management and disturbances would transition plantations into various successional classes. As exemplified by the Rim Fire, a portion (22%) of the existing plantations experienced high severity fire and were reset to an early successional stage; however, the majority of the plantations within the Rim Fire experienced a range of burn severities. About 22% experienced moderate burn severity, shifting them from a mid-seral closed canopy condition to an open canopy condition. About 56% experienced low burn severity. Although the canopy condition of these

plantations did not significantly change, low levels of mortality would increase the health and vigor of residual trees, accelerating development of late seral conditions. Had these plantations not existed prior to the Rim Fire, they would likely have been dominated by shrub and hardwood vegetation as a result of slow conifer succession following past fires; however, they are now contributing to structural diversity across the landscape.

Existing Plantations

Prescribed burning and thinning treatments in existing plantations would increase the diversity of forest structure across the landscape. The transition between different structural classes is not static; rather, it is a dynamic process driven by tree competition, development and disturbance. Thinning and prescribed fire would accelerate transition between structural classes. Some structural classes such as open canopy conditions would not occur in the absence of thinning, prescribed fire or some other disturbance. Treatments proposed in the younger Rogge plantations would reduce tree competition and accelerate the transition from an early seral structure (average dbh 1 to 6 inches) to a mid-seral structure (average dbh 6 to 11 inches). Similarly, thinning older plantations that are currently in a mid-seral closed canopy structure would transition them into an open canopy structure and accelerate the transition of some plantations into a late seral structure. After 20 years, a portion of the mid seral open canopy conditions would transition into a late seral open canopy condition, which is currently the rarest structural class on the Rim Fire landscape (Figure 3.01-2). Over the course of the next 60 years as trees continue to increase in size, all the plantations would eventually return to a closed canopy condition (greater than 40% canopy cover), but this would not occur until sometime between years 50 and 60. During the next 50 to 60 years, Alternative 1 would provide a greater range of both open and closed canopy conditions compared to Alternative 2.

A total of 2,187 acres of high-severity patches are distributed throughout existing plantations. Although relatively small, these patches would fit the definition of complex early seral forest (DellaSala et al. 2014; Odion and Hanson 2014). About 1,532 of these acres experienced high-severity fire, which occurred in small patches scattered throughout the 13,934 acres proposed for thinning. The remaining 655 acres resulted from moderate-severity fire and are also relatively small patches scattered throughout existing plantations. Trees in these patches are relatively small (less than 15 inches dbh), so thinning would remove some of the dead trees to reduce fuel loading; however, a portion of the largest dead trees would be retained.

To enhance deer habitat, an additional 1,990 acres are proposed for prescribed burning only. Areas that do not currently have trees would not contribute to changes in landscape forest structure. Prescribed fire may decrease tree canopy cover a small amount depending on fire-caused mortality. This effect would be minimal on a landscape scale and last for relatively short time (less than 20 years).

CUMULATIVE EFFECTS

The temporal boundary of this cumulative effects analysis is limited to the time frame in which future management activities are completed. Past management activities have shaped the existing condition and are not considered further. Activities that occur outside of the project area may affect vegetation, but would have no influence on vegetation growth, composition or structure within the project area. Therefore, all ongoing, present, and foreseeable activities that occur outside of the Rim Reforestation project area do not additively contribute to the direct and indirect effects discussed under Alternative 1 for the following indicators: tree size and composition, stand density index and future management feasibility.

The following list describes ongoing and present and reasonably foreseeable future actions within the project area that do not cumulatively affect vegetation.

- Ongoing livestock grazing may affect vegetation locally where cattle gather. Cattle do not typically eat shrubs or trees. Cattle can trample or crush young seedlings; however, management requirements are in place to reduce conflicts between range and reforestation activities. Range activities would have no detectable effect on shrubs or conifer growth and therefore no potential for cumulative effects.
- Present and future transportation-related treatments (maintenance, construction, rerouting, culvert replacement/repair, gate installations and designation changes to roads or trails) do not affect sufficient amounts of vegetation to create a detectable change from existing conditions.
- Ongoing, present and future recreation-related activities are primarily concentrated in areas that would not pose conflicts with reforestation and thinning activities. Any potential changes to vegetation caused by recreation would not be detectible at a significant scale and would not contribute to cumulative effects. Travel by recreationists on designated travel routes does not change vegetation; therefore, authorized motorized travel has no potential for a cumulative effect on vegetation.
- Present and future Rim Habitat and Rim Rehabilitation projects, including fence repair, great gray owl habitat improvement, installation of water troughs and guzzlers, botany restoration areas, weed treatments and meadow work could potentially impact a small number of shrubs and trees; however, the effect would not be detectible.

Present activities within the project area that do contribute to cumulative effects include the Rim Recovery and Rim HT projects which include various mechanical operations that would reduce natural conifer regeneration and understory vegetation. These effects are accounted for in the affected environment and direct and indirect effects under Alternative 1.

Change in Forest Structure

The present and foreseeable actions discussed here primarily occur outside of the Rim Reforestation project area (private lands, Yosemite National Park and other lands within the Rim Fire perimeter), but may contribute to changes in landscape forest structure. Present and future management activities not included here would not cumulatively contribute to detectible changes in landscape forest structure.

Present Actions

- The Rim Habitat project would remove encroaching conifers to improve meadow habitat. This project would reduce tree cover on 397 acres. This amounts to less than 1% of the entire Rim Reforestation project area and less than 0.2% of the Rim Fire landscape.
- The Groovy Stewardship project would primarily affect 1,319 acres of mid to late seral closed canopy forest. Treatments would decrease tree density and move stands into an open canopy forest structure.

Foreseeable Future Actions

- Reynolds Creek Ecological Restoration Aspen Release would reduce conifer density in aspen stands. Tree cover would remain and have minimal effects on landscape forest structure.
- Reynolds Creek Ecological Restoration Meadow Restoration would remove conifer tree cover from meadows on no more than 15 acres.
- The Reynolds Creek Fuels project would prescribe burn 2,288 acres in recently thinned stands. Prescribed fire would reduce shrub cover, conifer regeneration and small tree density, but likely have little effect on large tree density. By reducing conifer regeneration and small tree density, it would maintain 2,288 acres in an open canopy condition.
- Rim Rehabilitation project work including aspen release would reduce conifer density on 32 acres. Although the aspen release work would reduce conifer cover on a very small area, tree cover would remain.

- The Reynolds Creek and Funky Stewardship projects and the Soldier Creek, Campy, Looney and Thommy Timber Sales would affect 3,953 acres of mid to late seral closed canopy forest within the Rim Fire perimeter. Treatments would decrease tree density and move stands into an open canopy forest structure.

Most of the present and future management activities (e.g., aspen release) affect a small area and would not significantly contribute to cumulative changes in landscape forest structure. Projects that affect a larger area, such as green thinning and prescribed fire would contribute a more significant cumulative change to landscape forest structure. Currently, there is a deficit of mid and late seral open canopy structure (Figure 3.01-2). Green thinning projects would move 5,272 acres of closed canopy forest into an open canopy condition, which would increase the proportion of open canopy structure from about 17% to 20% within the entire Rim Fire landscape. These projects would move the landscape closer to a more natural range of variability by increasing forest structures that the landscape is lacking. Similarly, prescribed fire and treatments that reduce small tree density (e.g., prescribed fire in recently thinned stands) would maintain acres already in an open canopy structure for a longer period of time.

On private land, 15,479 acres have already been salvage logged, site prepped and planted. Present and future actions include herbicide release treatments to accelerate growth of conifers. Release treatments would reduce the cover of understory vegetation over the next few years. Because these acres have been salvage logged, they are not counted as complex early seral forest. These plantations would not impact the remaining 34,499 acres of complex early seral created by high-severity fire in the Rim Fire, but would accelerate the transition from early seral structure to mid and late seral structures. This would reduce the proportion of early seral structure within the Rim Fire landscape by about 8%. Within the next 20 to 40 years, plantations on private land would develop into mid seral forest. Transition into open or closed canopy conditions would be subject to land owner decisions and is outside the scope of this analysis.

In total, Alternative 1 would contribute to the following cumulative changes to landscape forest structure:

- On NFS lands, early-seral conifer forest would be reduced from 44,975 acres to 18,966 acres. The remaining early-seral forest makes up about 19% of the conifer forest on NFS lands (102,618 acres) within the Rim Fire perimeter, which is within the 15 to 20% historic range for early-seral structure (Safford 2013). The majority (66%) of these early-seral acres are comprised of complex early seral forest resulting from high-severity fire (not including areas of moderate burn severity). This project would reduce the 17,163 acres of high-severity complex early seral forest on NFS lands to 12,535 acres.
- On all lands within the Rim Fire perimeter, post-fire early-seral conifer forest would be reduced from 71,552 acres to 30,064 acres. This includes 15,479 acres of reforestation on private lands and the 26,009 acres of reforestation on NFS lands. The remaining early-seral forest makes up about 17% of the Rim Fire landscape (including private land, Yosemite National Park and other public lands summing to 182,327 acres of conifer forest), which is within the 15 to 20% historic range for early-seral structure (Safford 2013).
- On all lands within the Rim Fire perimeter, complex early seral forest created by high-severity fire (not including areas of moderate burn severity) would be reduced from 34,499 acres to 29,871 acres. These remaining 29,871 acres of severely burned conifer forest will not have been impacted by salvage, hazard tree or reforestation activities and are not expected to be impacted in the foreseeable future. These acres make up about 16% of the Rim Fire landscape (including private land, Yosemite National Park and other public lands summing to 182,327 acres of conifer forest), which is within the 15 to 20% historic range for early-seral structure (Safford 2013).
- Future mid seral closed canopy conditions would increase on up to 38,971 acres. This includes plantations on both private and NFS lands. Eventually these plantations would make up about

21% of the future Rim Fire landscape, which is 6% greater than the historic estimate of 5 to 15% (Safford 2013). This is assuming no thinning or stand replacing disturbance occurs within the next 60 years. Future thinning could transition these plantations into open canopy structures and accelerate the development of late seral open canopy forest. Historically, 1.5 to 2 times the amount of mid seral open forest existed than is found today. Late seral open canopy currently has the greatest deficit; historically, about 6 to 11 times more occurred across this landscape. Combined, the new plantations on private and NFS lands could someday make up just under half of the 45 to 75% of the Rim Fire landscape that should be in a mid to late seral open canopy condition (Figure 3.01-2).

- Mid and late seral open canopy structure would increase up to 19,206 acres resulting from thinning existing plantations and other green thinning. The cumulative effect of these treatments will vary considerably over the next 60 years because forests are dynamic and transition into various states of open canopy and seral structures depend on future climate conditions and disturbances. In general, however, thinning accelerates development of open canopy conditions, which are lacking across the Rim Fire landscape. Thinning could increase the open canopy structure across the Rim Fire landscape by up to about 11%.

Alternative 2 (No Action)

DIRECT AND INDIRECT EFFECTS

Tree Size and Species Composition

Reforestation

High- and moderate-severity patches are not reforested under Alternative 2. Immediately after the Rim Fire, vegetation (including shrubs, grasses, forbs and noxious weeds) rapidly responded and began increasing in abundance in moderate- and high-severity burn areas. During the first few years after a wildfire, turnover in species composition is considerable, as species that favor young burns give way to other species that favor older burns (Siegel et al. 2011). Although grasses, forbs and noxious weeds increase in abundance after wildfires, in a matter of years shrubs will overtop them and increase in dominance. For example, equations developed by Kie (1985) indicate that deerbrush and whitethorn shrub growth will increase 35 to 50% following reductions in canopy cover of 35 to 65%; similar to those found in moderate- and high-severity burn patches. During these first years after a wildfire, conifer regeneration also occurs; however, episodic events and conditions such as seed dispersal, bare soil, soil moisture and low shrub competition must coincide to result in continued conifer establishment and survival (Bonnet et al. 2005). As a result, conifer regeneration immediately after the Rim Fire was patchy, limited in extent (9,825 acres out of 26,009 acres), and mostly occurred within close proximity to mature conifers that survived the fire as displayed in Table 3.01-8 and Figure 3.01-4.

Additional reforestation would rely on secondary succession to reforest following the Rim Fire. Secondary succession is a plant-by-plant replacement process that occurs following a disturbance, where one plant species invades and replaces another (Horn 1974). Reinvasion of severely-disturbed forests often is very slow and unpredictable because of complex interactions among propagules as well as site and climatic conditions (Kozlowski 2002). While conifer seed dispersal is possible for longer distances (Shatford et al. 2007), the proportion of seed produced and dispersed is typically low at 1 to 5% (Nathan et al. 2002). Furthermore, seeds dispersed longer distances are less likely to successfully germinate and persist (Lesser and Jackson 2013; Nathan et al. 2002; Saigo 1969), which slows succession from shrub dominance to conifer forest. High and moderate severity areas would likely continue being dominated by shrubs similar to untreated control stands examined in research (McDonald and Fiddler 1995). In mixed-conifer forest types, Crotteau et al. (2013) found that shrubs continued to dominate high-severity patches 10 years after a wildfire, overtopping more than 60% of natural conifer regeneration in high-severity patches. McDonald and Fiddler (1995) found that forest

areas in the Sierra Nevada that were dominated by shrubs required treatment to return conifer dominance. McDonald and Abbott (1997) found that areas not treated to reduce ceanothus and manzanita experienced changes in dominance of shrub species through time, but shrubs continued to dominate and increase in dominance over the next 31 years.

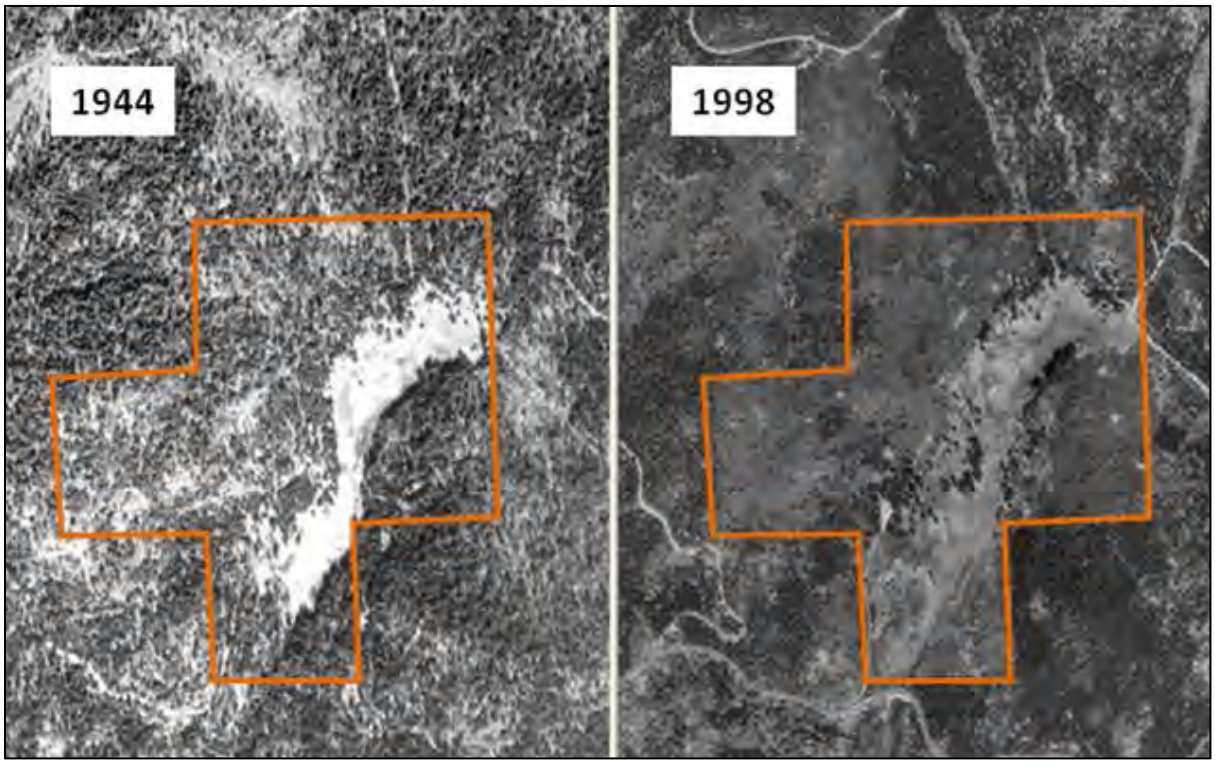


Figure 3.01-6 Forest Regeneration Comparison near Crane Meadow Following 1973 Granite Fire

Slow secondary succession from shrubs to conifer forest has been observed within the Rim Fire perimeter. Figure 3.01-6 displays aerial photographs of the Crane Meadow vicinity, which burned in the 1973 Granite Fire. The photographs were taken in 1944 (prior to the fire) and 1998 (25 years after the fire). Following the Granite Fire, the Forest Service artificially regenerated the areas surrounding the private parcel, outlined in the center of the photographs. As shown on the 1998 photo, the regenerated areas grew; however, the private land was never planted and exemplifies the lack of natural regeneration and the potential long-term impacts of high-severity fire in Sierran mixed-conifer forests.

Competition with dense shrubs has been shown to slow the initial growth of tree seedlings (Conard and Radosevich 1982a; Lanini and Radosevich 1986; McDonald and Fiddler 2001; Oliver 1990; Stuart et al. 1993). Therefore, natural conifer regeneration would likely experience the slowest growth of all the alternatives. Table 3.01-10 summarizes conifer and hardwood growth at years 10, 20 and 60. It also shows estimated tree densities throughout the 60-year analysis time frame. Compared to the other alternatives, the effects of no release treatments become evident by year 10. The average height of conifers is only 3.6 feet compared to 7.3 feet under Alternative 1. By year 20, average height under Alternative 2 is about half as much as Alternative 1. Competition also significantly slows tree diameter growth. The average conifer dbh under Alternative 2 at years 10 and 20 are about 3.5 times less than Alternative 1. The size of hardwoods would not be that different from Alternative 1. Height growth projected by RCONIFERS was similar to findings of McDonald and Abbott (1997) in study plots with medium to high levels of shrub cover. Slower conifer growth would also prolong

the time before conifers begin to overtop hardwoods. As a result, hardwoods would experience slightly lower levels of competition related mortality in the first 10 to 20 years; however, this effect would be small because inter-tree competition would still occur between hardwoods, which would also decrease hardwood densities over time.

Table 3.01-10 Alternative 2: Tree Density and Characteristics at Years 10, 20 and 60

Species	Year	PCR Mean	CBH Mean	DBH Mean	DBH Minimum	DBH Maximum	Height Mean	Height Minimum	Height Maximum	TPA[1] Mean	TPA[1] Minimum	TPA[1] Maximum
Conifers	10	87.8	0.4	0.2	0.0	0.8	3.6	1.4	7.9	571	279	6,506
Conifers	20	61.0	4.8	1.7	0.0	5.1	12.4	3.4	27.6	482	254	2,373
Conifers	60	48.0	29.5	9.0	3.3	25.5	56.7	17.6	110.2	344	236	814
Hardwoods	10	83.3	2.0	1.8	0.5	3.7	12.1	6.9	23.7	53	38	104
Hardwoods	20	58.6	10.5	4.2	2.1	6.0	25.5	17.7	32.0	20	14	37
Hardwoods	60	39.7	26.0	6.8	3.5	11.4	43.2	27.3	53.6	15	5	24

PCR=percent crown ratio; CBH=canopy base height in feet; DBH=diameter at breast height in inches; TPA=trees per acre
[1] Trees per acre is based on acres with natural conifer regeneration and does not include acres without conifer regeneration

By year 60, the difference between conifer tree size under Alternative 2 and the other alternatives tends to decrease. For example, dbh at year 60 under Alternative 2 is about 1.5 times smaller than Alternative 1, compared to about 3.5 times smaller than at year 20. This trend is likely caused by two factors: effects of tree competition and abrupt changes in competition derived from the modeling methodology. Although only a portion of the project area has conifer regeneration (9,825 acres out of 26,009 acres), where it does occur, it is patchy and typically very dense. On average there are over 500 seedlings per acre, but in many cases conifer seedling density exceeds 1,000 per acre. Table 3.01-10 displays high tree densities would cause intense competition between young trees and over time cause tree densities to sharply decline, but still remain relatively high. As tree densities decline, residual trees would experience increases in growth rate and the difference in dbh and heights compared to the other alternatives would slowly shrink with time. Beginning at year 20, FVS was used to model growth instead of RCONIFERS. Unlike RCONIFERS, FVS assumes trees are well-established and relatively free from competition-related growth effects from understory vegetation. Therefore, transferring to FVS would effectively emulate a release treatment at year 20. While all alternatives would experience some level of the same release, this effect is likely more exaggerated under Alternative 2 than any other alternative because the trees are smaller and the average tree height (about 12 feet) is small enough to suggest that trees are still competing with shrubs. For instance, Shatford et al. (2007) noted that during the first 10 to 20 years the majority of conifers they observed were overtopped by competing vegetation because shrub growth remains relatively vigorous during this time period. Although shrubs tend to slow in growth after year 20, they are fully developed and still sequestering a substantial amount of water and light resources to maintain respiration of living tissues. Despite this plateau in shrub growth rates, conifers must still compete for water and light, which results in sustained slow tree growth that is not accounted for in FVS.

Research in Sierran mixed-conifer forests suggests that, in addition to large patches of moderate- and high-severity fire being dominated by a mix of shrub species, tree species dominance typically shifts from ponderosa pine, sugar pine and Jeffrey pine to fir species (Collins and Roller 2013; Crotteau et al. 2013; Nagel and Taylor 2005). Some instances of relatively high densities of ponderosa pine regenerating after wildfires have been documented. After the Freds Fire on the El Dorado National Forest, Bohlman (2012) observed high levels of natural ponderosa pine regeneration; however, they note that the relatively high abundance of ponderosa pine was somewhat unique in their experience compared to regeneration inventories they had conducted on 15 other fires. Table 3.01-8, showing plot data in the Rim Reforestation project area, confirms that a trend of shrub and fir dominance is likely to occur. This table also shows the proportion of all pine species combined (ponderosa pine, sugar pine, knobcone pine, grey pine and Jeffrey pine), currently accounts for 13% of all conifer

regeneration within the project area and is not expected to naturally shift to pine dominance within the next 60 years. Areas of very high seedling densities tend to be dominated by white fir or Douglas-fir. During the first 10 years, these conifer thickets would experience intense competition and self-thin increasing the proportion of pine species from about 13% to 21% by year 10. This proportional shift to pine would significantly slow as trees increase in size and inter-tree competition begins. White fir and Douglas-fir can tolerate growing in denser stand conditions than pine, so the rate of pine mortality would likely increase as the rate of white fir and Douglas-fir mortality decreases. However, given the disproportionate densities of white fir and Douglas-fir compared to pine species, a greater number of white fir and Douglas-fir trees would still likely die despite the increasing rate of pine mortality. Therefore, the proportion of pine species would continue to increase between years 10 and 60, but only an additional 5%. By year 60, pine species would make up 26% of the species composition on average. This is much lower than historically occurred in most stands, with ponderosa pine alone accounting for greater than 50% of most stands in the lower to middle elevations on the Stanislaus (CREP 2008; Sudworth 1900).

Slower tree growth during the first 10 to 20 years would also prolong the time until dependable and abundant conifer seed production occurs. Conifers can produce seed as young as 7 to 10 years old; however, their performance is more erratic than mature trees (Laacke 1990; Oliver and Ryker 1990). White fir does not produce dependable cone crops until about age 40 (Laacke 1990), about 20 years before ponderosa pine (Oliver and Ryker 1990). The best seed producers for white fir are trees between 12 and 30 inches dbh, which is smaller than the best producers for ponderosa pine (Laacke 1990; Oliver and Ryker 1990). In California, the best seed producing size for ponderosa pine is greater than 25 inches dbh (Oliver and Ryker 1990). Based on tree age and growth model projections, white fir would likely reach necessary tree sizes for dependable seed production before ponderosa pine. Given the higher proportions of white fir and its ability to more dependably reproduce sooner than ponderosa pine, white fir would likely remain the dominant trees species.

Research suggests that low- and high-severity fire begets more of the same severity (van Wagtendonk et al. 2012). Shrub cover is a good predictor of fire behavior (Lydersen and North 2012; van Wagtendonk et al. 2012). Under low live fuel moisture conditions, shrub patches can burn at higher intensity than fuels in surrounding forest vegetation (Knapp et al. 2012; Skinner and Taylor 2006). High shrub cover and a shift toward fir dominance is likely to promote high-severity fire, especially when combined with current climate trends that are resulting in more frequent wildfire and extreme drought conditions (Coppoletta et al. 2015; Crotteau et al. 2013). The likelihood of future high-severity fire would further reduce conifer forest cover and result in expansion of shrub and chaparral vegetation. The Rim Fire is an indication that a shift in vegetation-disturbance dynamics is transpiring (Harris and Taylor 2015). This shift has precipitated as a result of forest densification (driven by fire suppression and other land management practices) and appears to be accelerated by drought conditions that are resulting from changing climate (Crotteau et al. 2013; Miller et al. 2012). If patterns of high-severity fire become entrenched, large portions of forests may become locked into cycles of repeat high-severity fires (Kane et al. 2015). Large fire-created openings in the project area are likely to remain filled with dense thickets of shrubs and scattered thickets of conifer regeneration. If a cycle of high-severity fire continues, these shrub-dominated patches would persist and expand.

Alternative 2 would provide no intervention in current departure of climate- and vegetation-disturbance dynamics compared to historic dynamics. Effects of fire will likely overshadow the direct effects of climate on tree species distributions and migrations because fire responds rapidly to climate (Dale et al. 2001; Flannigan et al. 2000). The Rim Fire has exemplified how fire might accelerate vegetation shifts by killing all or most conifer trees within uncharacteristically large high-severity patches (Harris and Taylor 2015; Miller et al. 2009). Although patches of conifer regeneration exist, they are dominated by white fir and Douglas-fir as opposed to pine. Pine is less sensitive to drought than other conifer species (Ferrell et al. 1994; Hurteau et al. 2007); therefore, these conifer patches

are more likely to be negatively affected by warmer and longer periods of summer drought. Lydersen et al. (2014) and Harris and Taylor (2015) found that cool, moist areas experienced lower fire severities during the Rim Fire. These areas may act as climate refugia that provide forest cover in light of changing climate; however, Kane et al. (2015) demonstrated that during periods of extreme drought, more productive sites that have higher densities of trees and surface fuels will likely contribute to the creation of large high-severity patches. In light of climate change, areas within the Rim Reforestation project area that are regenerating with high densities of white fir and Douglas-fir may be less affected by wildfire during years of normal precipitation, but prone to high-severity fire during extreme droughts. Therefore, the small proportion of the project area that does have natural conifer regeneration is at a greater risk of being lost if severe droughts increase in frequency. The risk of losing declining pine species in lower elevations (especially sugar pine) is greater in Alternative 2, due to the likelihood of high-severity wildfire removing the remaining conifer diversity. Microclimates that are well suited for conifers and are distributed throughout the severely burned project area, would not reforest naturally or be planted within the project time frame. Conifer forest connectivity would not be enhanced or accelerated, which would reduce the ability of pine (especially sugar pine) to move and adjust to future climate conditions and disturbances.

Existing Plantations

Without thinning, smaller suppressed and intermediate sized trees would continue to compete for water and growing space, slowing the growth of individual trees. Over the next 20 years, average tree diameters would range from about 6 to almost 12 inches dbh in the youngest and oldest plantations. Average tree heights would range from about 20 to 45 feet. After 60 years, diameters would range from 9 to 17 inches and height would range from 33 to 75 feet. After 60 years, large tree density (greater than 24 inches dbh) would average 18 trees per acre, which is 15% less than Alternative 1. As the trees continue to grow, canopy openings created by the Rim Fire would shrink, and in many cases, disappear as trees mature; therefore, fewer opportunities for conifer regeneration and development of understory vegetation would exist. As a result, horizontal and vertical forest structure would become more homogenized over time. Tree species composition would not significantly change in the absence of disturbance or thinning.

Stand Density Index

Reforestation

No reforestation activities would occur under Alternative 2, so discussion of SDI is limited to only portions of the project area that currently have natural conifer regeneration (9,825 out of 26,009 acres). While low levels of natural conifer regeneration outside these areas may occur, expansion of dry conifer forest into shrub dominated vegetation is slow (Barton 2002; Collins and Roller 2013; Goforth and Minnich 2008; Nagel and Taylor 2005; Roccaforte et al. 2012; Shatford et al. 2007), and would likely not qualify as conifer forest cover (greater than 10 to 25% conifer cover) according to CWHR criteria during the 60-year analysis time frame.

Overall, areas that are naturally regenerating within the project area increase in SDI rapidly because of the high densities of conifer seedlings. The highest densities are located in drainages and emergency travel routes. This is likely because these areas tend to be located in cooler, moister areas compared to mid-slopes and ridges; therefore, higher densities of white fir and Douglas-fir occurred in these areas prior to the Rim Fire. Within 20 to 30 years 385 acres of unsalvaged drainages and 31 acres of unsalvaged emergency travel routes would exceed a SDI of 300 and 365, respectively. After 40 years, a total of 2,940 acres would reach or exceed a SDI of 230 and 823 acres would exceed a SDI of 365. By year 60, all areas of natural regeneration would exceed a SDI of 230, of which, 38% would exceed a SDI of 365 and 23% would reach or exceed a SDI of 450.

The majority of the natural regeneration is white fir and Douglas-fir. These species tend to have a larger maximum SDI. Maximum SDI for ponderosa pine, white fir and Douglas-fir in Sierran mixed-

conifer forests is 446, 634 and 570, respectively (Dunning and Reineke 1933). When factored together, maximum SDI for mixed species stands generally range from 524 to 533 for pine dominated stands and 584 to 592 for fir dominated stands (Long and Shaw 2012). Lower maximum SDI levels induced by bark beetles (Oliver 1995) are typically associated with even-aged pine dominated stands. Therefore, patches of fir-dominated natural regeneration should be able to tolerate higher SDI levels. Increased competition-, insect- and drought-related mortality would become more prevalent as SDI levels reached and exceeded 365 to 400 (i.e., about 60% of maximum SDI, Long and Shaw 2012), which would occur between years 30 and 40. Although white fir and Douglas-fir would tolerate higher SDI levels, ponderosa pine would not. The maximum SDI for ponderosa pine is 446 (Dunning and Reineke 1933); therefore, it would experience increasing rates of mortality over time. Ponderosa pine mortality would likely begin as early as 20 to 30 years from now when SDI levels near and exceed 60% of its maximum SDI. As a result, white fir and Douglas-fir would maintain a greater proportion of the conifer forest within the project area over the next 60 years; further departing from desired species composition and ecological integrity.

Existing Plantations

Of the existing plantations proposed for thinning, 3,904 acres currently exceed a SDI of 230 and about half of these acres exceed a SDI of 365. Without thinning, the existing plantations would exceed SDI levels of 230, about 20 years faster than compared to the other alternatives. By year 20, an additional 2,821 acres would exceed a SDI of 230. By year 40, a total of 7,997 acres would exceed SDI levels of 230. Thinning under the other alternatives would prevent most of these plantations from exceeding a SDI of 365. Without thinning, however, 7,596 acres would exceed SDI of 365 by year 60. That is, almost four times as many acres would exceed a SDI of 365 by year 60 compared to the other alternatives.

Without thinning, trees would not benefit from reduced stand densities, which would decrease water and nutrient availability to individual trees. Trees would experience slowed growth and reduced vigor, which would slow development of larger trees during the analysis time frame. Water stress during dry summers would intensify and tree mortality during periods of more severe drought would increase. As the trees increase in size and occupy more growing space over the next 20 to 60 years, SDI levels would increase to levels well beyond 365; the insect-induced SDI maximum (Oliver 1995; Oliver and Uzoh 1997). Subsequently, inter-tree competition would severely intensify and stands would likely show widespread evidence of insect-related mortality. Mortality would be more severe during periods of drought when trees are water-stressed. If insects and drought were not a factor, SDI levels would still be well beyond the threshold for self-thinning (50 to 60% of maximum SDI). At these densities competition-related mortality would be inevitable and result in “self-thinning” (Long and Shaw 2012). Like the other alternatives, mortality would typically occur in denser patches and on poorer sites where competition is more intense between trees; however, given the homogenous structure of such high tree densities, patches of mortality would likely be larger. Large patches of mortality would promote homogeneity of species composition and structure.

Future Management Feasibility

Reforestation

Overall, Alternative 2 would provide the fewest acres that might someday provide merchantable wood products for offsetting future management costs. Like Alternative 1, thinning criteria would not be met until about 50 years from now. Within the 9,825 acres that are expected to successfully regenerate with conifers, 1,504 acres would satisfy thinning criteria by year 50. Thinning would remove 4,451 board feet per acre. By year 60, an additional 6,062 acres would satisfy thinning criteria and remove 5,840 board feet per acre. After thinning to a SDI of 170, residual conifer densities would range from 140 to 222 conifers per acre. During the first 60 years, 2,259 acres never satisfy the thinning criteria despite most of these areas exceeding SDI levels of 400. Therefore, thinning to the

target SDI of 170 would yield less than 2,000 board feet per acre, suggesting that small trees would still dominate these areas after 60 years. In total, Alternative 2 would result in 18,443 acres of suitable forest land that would not provide any cost offsets for future management.

Existing Plantations

As existing plantations increase in density, individual tree growth slows. Forgoing thinning would result in 15% fewer large trees (greater than 24 inches dbh) per acre. Similarly, fewer trees would reach merchantable sawtimber sizes during the next several decades and reduce potential opportunities to offset future costs associated with fuels reduction and wildlife habitat management.

Change in Forest Structure

Reforestation

Alternative 2 would rely on natural secondary succession to transition the excess of more than 30,000 early seral acres within the Rim Fire landscape into mid and late seral structures. Research suggests that high-severity fire creates vegetation conditions that promote more high-severity fire (van Wagtendonk et al. 2012). Patterns of shrub dominance and abundant white fir regeneration indicate a shift away from a pine-dominated, frequent lower-severity fire system (Coppoletta et al. 2015; Crotteau et al. 2013). Given the uncharacteristically large patches burned by high-severity fire during the Rim Fire (Miller et al. 2009), transition into forest cover would be unpredictable and slow (Crotteau et al. 2013; Kozlowski 2002; Nagel and Taylor 2005). Intervention is required to help restore both pine composition and frequent fire as an ecological process (Crotteau et al. 2013; Harris and Taylor 2015). Without reforestation, substantial expansion of conifer forest beyond areas already regenerating would likely not occur until trees reach a size or age that can produce dependable cone crops. For white fir, which is the most abundant natural regeneration in the project area, this would likely not occur until 40 to 60 years from now (Laake 1990); at which time, newly established trees would likely experience slow growth and remain in an early seral structure well beyond the 60-year analysis timeframe. Additionally, conifer regeneration would mostly occur near existing seed sources (Bonnet et al. 2005), but some small numbers of seedlings could establish farther away in severely burn areas (Shatford et al. 2007). Considering the slow growth of seedlings once established among dense shrubs, however, these trees would likely require several decades to overtop shrubs and establish conifer forest (Nagel and Taylor 2005; Shatford et al. 2007). As a result, about 58% of the 23,492 acres (26,009 acres minus 2,517 acres of non-conifer cover types) proposed for reforestation would remain in an early seral structure dominated by shrubs and hardwoods during the next 60 years. This is about three times the proportion of the project area that would have historically occurred in an early seral structure (Safford 2013).

The naturally regenerating areas would experience slow tree growth due to competition with shrubs or high densities of other trees. Development of mid-seral conditions in these patches would also be slow and not emerge until sometime between 40 and 60 years from now. In the absence of disturbance, tree densities in patches of regeneration would remain high, so no open canopy condition would occur during the analysis time frame.

Existing Plantations

No thinning or prescribed fire in existing plantations would result in lower diversity of landscape forest structure during the next 60 years. Without treatment, closed canopy conditions would persist in the absence of disturbances. Areas that currently have open canopy structure because of the Rim Fire would eventually return to closed canopy conditions. Within 10 years all but about 750 acres is estimated to return to closed canopy conditions. Within the next 30 to 40 years no mid- or late-seral open canopy structure would exist.

CUMULATIVE EFFECTS

The ongoing, present and reasonably foreseeable future actions within the analysis area that do not cumulatively affect vegetation are the same as discussed under Alternative 1. Present activities occurring within the project area that do contribute to cumulative effects are also the same as those discussed in Alternative 1 and have already been accounted for in the description of the affected environment and discussion of direct and indirect effects under Alternative 2.

Change in Forest Structure

The present and foreseeable actions that may contribute to a cumulative effect to landscape forest structure are the same as discussed under Alternative 1; therefore, the following list discusses only the cumulative effects resulting from Alternative 2 in the context of the Rim Fire landscape. In total, Alternative 2 would contribute to the following cumulative changes to landscape forest structure:

- On NFS lands, early-seral conifer forest would be reduced from 44,975 acres to 35,150 acres after excluding the estimated 9,825 acres naturally regenerating. The remaining early-seral forest makes up about 34% of the conifer forest on NFS lands (102,618 acres) within the Rim Fire perimeter, which is significantly greater than the 15 to 20% historic range for early-seral structure (Safford 2013). About half (17,163 acres) of these early-seral acres are comprised of complex early seral forest resulting from high-severity fire (not including areas of moderate burn severity).
- Post-Rim Fire early-seral conifer forest would be reduced from 71,552 acres to 46,248 acres. This includes 15,479 acres of reforestation on private lands and the 9,825 acres of natural conifer regeneration on NFS lands. The remaining early-seral forest makes up just over 25% of the Rim Fire landscape (including private land, Yosemite National Park and other public lands), which is greater than the 15 to 20% historic range for early-seral structure (Safford 2013).
- Complex early seral forest created by high-severity fire (not including areas of moderate burn severity) would remain at 34,499 acres, which makes up about 19% of the Rim Fire landscape (including private land, Yosemite National Park and other public lands). This is almost equal to the maximum of the 15 to 20% historic range for early-seral structure (Safford 2013).
- Future mid-seral closed canopy conditions would increase on up to 25,304 acres. This includes the plantations on private land and natural regeneration on NFS lands. Eventually these plantations and patches of conifer regeneration would make up about 14% of the future Rim Fire landscape, which is within the historic estimate of 5 to 15% (Safford 2013). This is assuming no thinning or stand replacing disturbance occurs within the next 60 years. Future thinning could transition these areas into open canopy structures and accelerate the development of late-seral open canopy forest. Historically, 1.5 to 2 times the amount of mid seral open forest existed than is present now. Late seral open canopy currently has the greatest deficit; historically, there were about 6 to 11 times more than currently exists. Without planting on NFS lands, development of future mid and late seral structures would be limited to the new plantations on private lands and natural regeneration on NFS lands. Assuming no stand-replacing events, these plantations and patches of forest would make up at most one third of the 45 to 75% of the Rim Fire landscape that should be in a mid to late seral open canopy condition (Figure 3.01-2).
- Mid and late seral open canopy structure would increase up to 5,272 acres resulting from other green thinning. Thinning accelerates development of open canopy conditions, which are lacking across the Rim Fire landscape. Thinning projects would increase the open canopy structure across the Rim Fire landscape by up to about 3%, which is less than Alternative 1.

Alternative 3

DIRECT AND INDIRECT EFFECTS

Tree Size and Species Composition

Reforestation

Alternative 3 would reforest the same number of acres as Alternative 1, but planting patterns and densities would differ (EIS, 2.02 Alternatives Considered in Detail). Although Alternative 3 has fewer planting patterns than Alternative 1, it has a larger range of conifer planting densities. Alternative 3 would plant more acres at higher initial densities than Alternative 1; however, no herbicides would be used. Only manual release treatments are proposed under this alternative. Despite a greater number of acres being site prepped with deep tilling and forest cultivation, planted conifers under Alternative 3 would likely experience higher rates of initial mortality. Based on survival exams from the Mi-Wok and Groveland Ranger Districts, areas site prepped with deep tilling would have better survival than areas that did not. Deep tilling kills above-ground portions of shrubs and breaks-up below-ground roots and rhizomes (McDonald et al. 2004), reducing initial competition for water during the first growing season after planting. Based on past experience on the Stanislaus, conifer survival is estimated at 50% of the initial planting density in areas prepped with deep tilling and cultivation when followed by manual releases as shown in Table 3.01-4. Experience on the Stanislaus has also shown that manual releases without deep tilling result in much lower initial survival rates of about 30% (Table 3.01-4). After accounting for initial seedling mortality and oak buffers, conifer density during the first decade would range between 41 and 172 trees per acre and decline to 38 to 162 trees per acre over the next 60 years as displayed in Table 3.01-11. Alternative 3 would have similar oak and hardwood sizes and densities as Alternative 1. The effect of planting pattern on the development of fine-scale heterogeneity is similar to Alternative 1. Planting patterns would create open stand structures with small clumps of seedlings and saplings in the short-term; however, variation in tree sizes, ages and spatial distribution would require several decades at a minimum to develop.

Table 3.01-11 Alternative 3: Tree Density and Characteristics at Years 10, 20 and 60

Species	Year	PCR Mean	CBH Mean	DBH Mean	DBH Minimum	DBH Maximum	Height Mean	Height Minimum	Height Maximum	TPA Mean	TPA Minimum	TPA Maximum
Conifers	10	92.3	0.4	0.4	0.0	0.8	5.3	1.7	7.5	120	41	172
Conifers	20	67.0	5.4	2.8	0.1	4.8	16.3	4.7	24.2	120	40	172
Conifers	60	63.8	24.5	13.6	5.1	26.8	67.6	25.0	102.2	112	38	162
Hardwoods	10	82.7	1.9	1.3	0.6	2.9	10.7	7.8	21.0	50	38	63
Hardwoods	20	64.9	9.1	4.4	3.0	6.2	26.0	19.1	32.9	19	14	25
Hardwoods	60	42.7	25.2	7.2	4.2	11.5	44.0	27.3	61.1	17	13	23

PCR=percent crown ratio; CBH=canopy base height in feet; DBH=diameter at breast height in inches; TPA=trees per acre

Deep tilling and forest cultivation (DTFC) would primarily be completed in the late summer and fall, so conifers planted in the following spring would benefit from reduced competition until shrubs, grasses, and forbs recover. Areas that are not suitable for deep tilling and forest cultivation would be manually released either during planting or shortly thereafter. Water availability to young trees would be effectively diminished in areas that are not deep tilled as compared to areas that are deep tilled because the available water would be more easily consumed by shrubs with intact below-ground roots and rhizomes. Although not as effective as herbicide releases, repeated annual manual releases have proven fairly effective on most shrubs, grasses and forbs. McDonald and Fiddler (1997) evaluated different manual release treatments and determined that grubbing a radius of less than 5 feet around conifer seedlings significantly decreased conifer growth; however, multiple manual releases that removed vegetation within a 5-foot radius around conifer seedlings increased tree heights by about 1.5 times compared to a control. Creating only a 2-foot radius had no effect on tree growth. Growth

projections made using RCONIFERS had similar results. Conifer heights at year 10 under Alternative 3 are about 1.5 times the heights estimated for Alternative 2. It should be noted, however, that tree heights in the McDonald and Fiddler (1997) study were about 14 feet after 10 years, which is considerably taller than the average height estimated for this analysis. This difference is likely a result of both site conditions and the large presence of bearclover within the Rim Reforestation project area. Bearclover was not present in the study conducted by McDonald and Fiddler (1997). Mechanical and manual releases have proven ineffective with bearclover (McDonald et al. 2004). Tappeiner and Radosevich (1982) found that trees grown in bearclover only achieved heights of about 6 feet after 19 years following a single herbicide application. While this was a slight increase over the control, multiple treatments are necessary to effectively control bearclover (McDonald et al. 2004). Tree heights estimated under Alternative 3, however, would average just over 16 feet after 20 years, which is similar to study plots with low shrub cover observed by McDonald and Abbott (1997). While not as tall as trees in Alternative 1, this is a large increase over Alternative 2 because hand-grubbing would provide some increase in water availability, albeit not as much as would occur with repeated herbicide applications. Taller average tree heights would also likely result from conifers growing in other shrub types where hand-grubbing would more effectively increase conifer growth compared to areas dominated by bearclover. Noxious weeds could rapidly establish in recently deep tilled or hand grubbed areas when mineral soil is exposed; therefore, conifer growth responses could be diminished if noxious weeds are present or if manual grubbing and pulling of known noxious weed treatments prove ineffective.

At year 20 when modeling is switched from RCONIFERS to FVS, tree growth rates artificially begin to match or even surpass rates in Alternative 1 because FVS does not account for shrub competition; however, shrubs would likely still impact tree growth beyond this age because shrubs would continue competing for water. Furthermore, by year 20 the average canopy base height under Alternative 1 is about 2 feet greater than Alternative 3. The average crown ratio is also less under Alternative 3. This suggests that trees under Alternative 1 are both taller and have larger crowns at year 20. Larger crowns indicate more needles (leaves) available for photosynthesis, which promotes faster height and diameter growth; yet, by year 60 the average dbh in Alternative 3 is almost the same as Alternative 1, suggesting that the sudden artificial release from shrub competition at year 20 (i.e., switching to FVS) contributed to sudden increases in height and diameter growth.

A hurdle related to Alternative 3 worth noting is the high cost and difficulty associated with effectively completing manual releases. The cost analysis is displayed in 3.10 Society, Culture and Economy. Manual release is difficult work and labor intensive, making it expensive, especially given the large number of acres that would require this treatment. Past experience on the Stanislaus has demonstrated the difficulties involved with manual release. Many contracts went into default during the 1990s and volunteer groups were only able to complete small areas, resulting in much of the work never being completed (USDA 1995).

Although Alternative 3 would result in lower conifer densities compared to Alternative 1, it would help increase conifer forest connectivity across a greater area than Alternatives 2 and 4. Alternative 3 would ensure that shade-intolerant species such as ponderosa pine and sugar pine make up a larger proportion of tree composition. Planting would increase the likelihood of suitable microsites across the project area being occupied by conifers rather than by indefinitely persisting shrubs and hardwoods. Establishing conifers, especially pine species, across a wider area, would increase the likelihood of more forest patches escaping future stand-replacing events, such as wildfire. As a result, conifers would be positioned to move and adapt to future changing climate conditions and disturbances as discussed under Alternative 1.

Effects of prescribed burning in young plantations would be the same as Alternative 1.

Existing Plantations

Same as Alternative 1.

Stand Density Index

Reforestation

Variable density planting is proposed on all mid-slopes and drainages outside SFMAs. Initial densities of variable density planting are higher than within SFMAs and features. Additionally, areas that are deep tilled are expected to have higher initial seedling survival. Although Alternative 3 would plant higher tree densities initially than Alternative 1, initial seedling mortality is expected to be much higher without the use of herbicides; therefore, tree densities on average would be lower than Alternative 1 and take longer to reach SDI levels of 230 and greater. Areas of higher planting densities and survival would require less time until the onset of inter-tree competition. These areas include drainages and mid-slopes that are deep tilled; however, no plantations would reach or exceed a SDI of 230 until sometime between years 50 and 60. After 60 years, 8,159 acres would exceed a SDI 230. The highest SDI at year 60 would be 272. Although the majority of the project area would not exceed a SDI of 230 within the next 40 to 50 years, low SDI levels suggest lower tree canopy cover and increased shrub cover. As temperatures and summer water deficits increase with changing climate, risk of insect-, competition-, and drought-related mortality would be slightly lower than Alternative 1 overall. Having lower SDI levels would improve tree resilience during periods of extended drought; however, increased shrub cover would increase the risk of high-severity fire during extreme drought (Harris and Taylor 2015; Lydersen et al. 2014).

Existing Plantations

Same as Alternative 1.

Future Management Feasibility

Reforestation

Low initial survival and initial planting densities would require longer periods of time for plantations to reach and exceed a SDI of 260. With lower tree densities, Alternative 3 would require larger trees to satisfy thinning criteria for SDI. While manual release treatments would improve tree growth compared to Alternative 2, they are not as effective as herbicide release treatments. Therefore, slower initial tree growth would further prolong the development of larger trees. Unlike the other alternatives, no plantations would satisfy the thinning criteria within the next 50 years. By year 60, a total of 8,136 acres would satisfy the thinning criteria. Thinning could remove 5,937 board feet per acre, which is more than could be removed under Alternative 2 and less than Alternative 1 in year 60. Residual conifer densities in these areas would range from 103 to 106 conifers per acre. Like Alternative 1, areas that satisfy the thinning criteria are those that initially have higher tree densities. These areas are the drainages and mid-slopes that are deep tilled during site preparation. The fuelbreaks, primary ridges, emergency travel routes and all areas that are not site prepped with deep tilling never satisfy the thinning criteria within the next 60 years because tree densities are too low. Alternative 3 would result in 17,873 acres of forest land that would not provide any cost offsets for future management.

Existing Plantations

Same as Alternative 1.

Change in Forest Structure

Reforestation

Alternative 3 would reforest the same number of acres as Alternative 1; however, the development of diverse forest structures would be dissimilar because of the different planting patterns, lower initial seedling survival and slower tree growth. Over the course of the next 30 years, not all plantations

would progress from early seral successional classes into mid seral classes. About 58% would qualify as mid seral open canopy, while 42% would remain in an early seral structure. By year 40, fuelbreaks and primary ridges (1,146 acres total) would remain in an early seral condition with average tree sizes of 1 to 6 inches dbh and low tree densities. The other plantations would have reached a mid-seral condition with 10 to 39% canopy cover. After 50 years all of the plantations would reach a mid-seral condition. About one third would develop closed canopies and the other two thirds would have open canopy conditions (less than 40% canopy cover). Between years 50 and 60 additional plantations would develop closed canopy conditions, but a small portion (2,702 acres) would retain open conditions with less than 40% canopy cover. Compared to Alternative 1, a greater proportion of the project area would maintain an open canopy condition throughout the next 60 years and have smaller tree sizes on average. As described under Alternative 1, future management and disturbances would transition plantations into various successional classes.

Existing Plantations

Same as Alternative 1.

CUMULATIVE EFFECTS

The cumulative effects to vegetation under Alternative 3 are the same as Alternative 1, with the following differences.

- Manual release treatments are proposed instead of herbicide applications and more acres of deep tilling with forest cultivation are proposed. These differences would not contribute to a cumulative effect to vegetation outside of the project area. Present activities occurring within the project area that do contribute to cumulative effects include the Rim Fire Recovery and Rim Hazard Tree projects. These projects include various types of mechanical operations that would reduce natural conifer regeneration and understory vegetation. These effects have already been accounted for in the description of the affected environment and discussion of direct and indirect effects under Alternative 3.
- Differences in planting density and conifer growth would result in slower development of mid-seral structures as well as closed canopy structures; however, proportional changes in early-seral and complex early seral structures would be the same as discussed in Alternative 1.

Alternative 4

DIRECT AND INDIRECT EFFECTS

Tree Size and Species Composition

Reforestation

In addition to the estimated 9,825 acres that are expected to naturally regenerate with conifers, Alternative 4 would reforest 2,954 acres using a founder stand planting design. Planting founder stands would allow control of competing vegetation and species composition. Trees planted in founder stands would grow faster than naturally regenerating conifers and increase the proportion of pine across the project area to 24%, 27% and 30% over the next 10, 20 and 60 years, respectively.. Founder stands would occur only on mid-slopes and consist of 2 to 10 acre patches of planted conifers. Founder stands would not be planted in areas with hardwoods. Similarly, planting would not occur if natural conifer regeneration is present. Table 3.01-12 summarizes tree characteristics and densities at years 10, 20 and 60. The use of herbicides would reduce competition in founder stands and result in similar average conifer dbh and heights as expected under Alternative 1. Outside of founder stands, however, competing vegetation would slow tree growth. When averaged together, the overall tree size of Alternative 4 would be considerably smaller than Alternative 1.

Table 3.01-12 Alternative 4: Tree Density and Characteristics at Years 10, 20 and 60

Species	Year	PCR Mean	CBH Mean	DBH Mean	DBH Minimum	DBH Maximum	Height Mean	Height Minimum	Height Maximum	TPA[1] Mean	TPA[1] Minimum	TPA[1] Maximum
Conifers	10	88.2	0.5	0.2	0.0	1.1	3.8	1.4	9.1	469	131	6,506
Conifers	20	61.8	5.0	1.9	0.0	7.0	13.1	3.4	32.6	401	131	2,373
Conifers	60	49.3	29.6	9.6	3.3	25.5	58.5	17.6	110.4	293	122	814
Hardwoods	10	83.3	2.0	1.8	0.5	3.7	12.1	6.9	23.7	53	38	104
Hardwoods	20	58.6	10.5	4.2	2.1	6.0	25.5	17.7	32.0	20	14	37
Hardwoods	60	39.7	26.0	6.8	3.5	11.4	43.2	27.3	53.6	15	5	24

PCR=percent crown ratio; CBH=canopy base height in feet; DBH=diameter at breast height in inches; TPA=trees per acre
[1] Trees per acre is based acres with natural conifer regeneration and founder stands, and does not include acres without regeneration.

Prescribed fire is proposed on a large scale under Alternative 4, within and outside of founder stands. At the earliest, prescribed fire would occur about 5 to 7 years after the Rim Fire. Although prescribed fire would reduce shrub cover, tree growth during these early years would have already been negatively impacted by competing vegetation. It has been well substantiated that suppressing competing vegetation during the first few years of tree development has the greatest impact on the survival and growth of conifer seedlings (Balandier et al. 2006; McDonald and Fiddler 2010). Growth lost during the first few years of seedling development will likely never be made up because seedlings do not respond as vigorously to delayed release treatments (McDonald and Fiddler 2001). Large-scale prescribed fire would, however, reduce surface fuels and tree densities where conifers are regenerating. Lower fuel loads and tree densities would help prolong the probability of another high-intensity fire, providing a longer window of opportunity for conifers to develop fire resilient characteristics such as thick bark and high canopy base heights. Trees growing in founder stands would develop these characteristics faster than trees outside founder stands. The early slow growth of trees outside founder stands would result in substantially smaller trees after 20 years than within founder stands. However, the area planted as founder stands is considerably smaller than the area naturally regenerating. Therefore, the overall averages for tree size and other characteristics increase very little compared to Alternative 2. As a result, the overall average canopy base height under Alternative 4 is still about 2 feet less than Alternative 1. The overall averages for dbh and height under Alternative 4 are also significantly less than Alternative 1.

Creation of founder stands throughout the project area would eventually lead to broader expansion of conifer forest throughout the large severely burned areas. Based on tree age and growth model projections, white fir would likely reach necessary tree sizes for dependable seed production before ponderosa pine. Within about 40 to 60 years, founder stands would likely have trees that are producing large, dependable cone crops that could then result in accelerated seed dispersal into adjacent areas. Although founder stands would have higher proportions of pine compared to naturally regenerating areas, white fir would still start producing more viable seed before any pine species; and therefore, would likely maintain dominance beyond the next 60 years.

Within founder stands, the effect of planting pattern on the development of fine-scale heterogeneity is similar to Alternative 1. Planting patterns would create open stand structures with small clumps of seedlings and saplings in the short-term; however, variation in tree sizes, ages and spatial distribution would require several decades at a minimum to develop. A key difference, however, is the small size of founder stands. The concept of a founder stand is to provide a seed source from which conifer forest can grow and expand. Founder stands are expected to require at least 40 to 60 years before substantial seed production begins and forest expansion is possible; therefore, Alternative 4 would likely require a longer period of time to establish conifer vegetation-disturbance dynamics, such as the ICO structures maintained by frequent fire, on a scale greater than 2 to 10 acres (i.e., the size of a founder stand).

Like Alternatives 1 and 3, the distribution of founder stands across the landscape would provide for a small level of ecological buffering that increases the probability of all conifer stands in the project area being lost to future disturbance (Millar et al. 2007). Planting small stands with ponderosa pine and sugar pine will provide a small level of planting redundancy that will disperse these species across a wider area than Alternative 2, and provide future seed sources. The founder stands would provide opportunities for these species to move and adapt to future changing climate conditions and disturbances, but to a lesser extent than Alternatives 1 and 3. The founder stands only increase the forested area by 2,954 acres compared to Alternative 2. Therefore, the founder stands would be dispersed throughout 13,230 acres of predominantly shrub and hardwood vegetation. Shrub cover is a good predictor of fire behavior (Lydersen and North 2012; van Wagtendonk et al. 2012). Under low live fuel moisture conditions, shrub patches can burn at higher intensity than fuels in surrounding forest vegetation (Knapp et al. 2012; Skinner and Taylor 2006). Today, prescribed burning and managed wildfire are faced with numerous operational and social constraints that limit their use in effectively reducing hazardous fuels and maintaining natural processes (Quinn-Davidson and Vaner 2012). If prescribed fire is not repeatedly used across the landscape to maintain low shrub levels, then the risk of high-severity fire killing founder stands would increase over time and limit their ability to function as sources for conifer dispersal in the long-term.

Existing Plantations

Same as Alternative 1.

Stand Density Index

Reforestation

Outside of founder stands, the direct and indirect effects related to SDI are the same as described for Alternative 2. Founder stands would only account for 2,954 acres of small plantations distributed across the project area. Planting density within founder stands would range between 100 and 200 trees per acre. At tree densities this low it would require almost 60 years to exceed a SDI of 230. As temperatures and summer water deficits increase with changing climate, risk of insect-, competition-, and drought-related mortality would be low within founder stands during the first 50 years. Having lower SDI levels would improve tree resilience during periods of extended drought; however, lower tree densities would increase the abundance of shrub cover within founder stands. Given the small size of founder stands, they would likely be adjacent to shrub dominated patches that have not been reforested. In the absence of frequent prescribed fire, shrubs would likely reach decadent structures after 10 to 20 years (Shatford et al. 2007). Saspis and Brandow (1997) concluded that one of the factors contributing to high-severity fire risk in the Granite plantations was the juxtaposition of different fuel types and ladder fuels. Therefore, increased shrub cover would increase the risk losing founder stands to high-severity fire during extreme drought (Harris and Taylor 2015; Lydersen et al. 2014).

Existing Plantations

Same as Alternative 1.

Future Management Feasibility

Reforestation

The effects of Alternative 4 are the same as Alternative 2. Although founder stands would increase the acres of reforested land by 2,954 acres, SDI levels would not reach a SDI of 260 within the next 60 years; therefore, no additional acres would satisfy the thinning criteria and no additional cost offsets would be realized for future management.

Existing Plantations

Same as Alternative 1.

Change in Forest Structure

Reforestation

With the exception of prescribed fire and creating founder stands, Alternative 4 would have similar effects on landscape forest structure as Alternative 2. Widespread use of prescribed fire could promote establishment of conifer regeneration in large areas that would otherwise remain dominated by shrubs and hardwoods. Prescribed fire would create openings in the shrub canopy where conifer seedlings could establish; however, reinvasion of conifers would still rely on complex interactions between seed supply, suitable medium for seed germination and favorable climatic conditions (Bonnet et al. 2005; Kozlowski 2002). A high abundance of white fir and Douglas-fir regeneration after the Rim Fire suggests that these species would have the highest probability of establishing after prescribe fires, so a desirable species composition would likely not be achieved within the next 60 years. Furthermore, high shade provided by residual mature conifers in a low- to moderate-severity fire system reduces shrub cover, limiting competition with conifer seedlings (Crotteau et al. 2013). While prescribed fire in large severely burned areas would create openings in shrub cover, most of these areas would have no conifer canopy to slow the response of shrub growth following prescribed fire. Consequently, shrub cover would quickly return and suppress any newly established conifers; thus, prolonging succession from shrub to conifer (Crotteau et al. 2013; Nagel and Taylor 2005; Shatford et al. 2007). While succession from early seral to mid-seral conditions would still likely occur on the order of multiple decades, frequent prescribed fire could accelerate transition from shrub cover to conifer cover quicker than Alternative 2, but the difference is difficult to quantify.

Founder stands would have no effect on the complex early seral forest remaining within the project area after Rim Recovery and Rim HT project work is fully completed. Herbicide use in founder stands would reduce competition with understory vegetation and promote faster conifer growth. Founder stands would reach a mid-seral open canopy condition by year 30, which would likely be the only mid seral conditions within the project area (2,954 acres). After 40 years an additional 7,087 acres that naturally regenerated would reach mid seral tree sizes. Given the low initial tree densities in founder stands, they would require about 50 years before trees reach a size that created a closed canopy (greater than 40% canopy cover). By year 60 all founder stands and all patches of white fir and Douglas-fir dominated natural regeneration would reach a mid-seral closed canopy condition. In total, 54% of the 23,492 acres (26,009 acres minus 2,517 acres of non-conifer cover types) proposed for reforestation under the other action alternatives would remain primarily dominated by shrubs and hardwoods after 60 years. This is more than twice the proportion of the project area that would have historically occurred in an early seral structure (Safford 2013).

Existing Plantations

Same as Alternative 1.

CUMULATIVE EFFECTS

The cumulative effects to vegetation caused by thinning existing plantations are the same as described under Alternative 1.The cumulative effects to vegetation caused by reforestation activities are the same as Alternative 2, with the following differences.

- Creating founder stands would increase the proportion of early seral forest that is transitioned to a mid-seral structure within the 60-year analysis time frame. In total, Alternative 4 would contribute to a cumulative reduction of early seral forest from 71,552 to 43,294 acres. This includes 15,479 acres of reforestation on private lands, 9,825 acres of natural conifer regeneration, and 2,954 acres of founder stands. The remaining early seral forest makes up about 24% of the Rim Fire landscape, which is greater than the 15 to 20% historic range for early seral structure (Safford 2013).

- Once founder stands develop into a mid-seral size, they would remain in an open canopy structure until year 50. Until that time, they would increase the proportion of open canopy structure across the Rim Fire landscape by 1.6%

Alternative 5

DIRECT AND INDIRECT EFFECTS

Tree Size and Species Composition

Reforestation

Alternative 5 would have similar tree sizes and species composition as Alternative 1 as shown in Table 3.01-13 and Table 3.01-9. Oliver (1979) found that seedlings spaced greater than 6 feet apart did not experience slowed growth during the first 12 years after planting. Although Alternative 5 has a higher initial planting density than Alternative 1, hand thinning would occur by year 7, reducing tree densities to about 220 to 240 trees per acre (about 14-foot spacing). Planting higher densities would initially ensure greater tree cover, which would increase the likelihood of trees being planted in favorable microsites (e.g., pockets of deep soil and high moisture availability), that might otherwise be passed over when planting seedlings farther apart.

Table 3.01-13 Alternative 5: Tree Density and Characteristics at Years 10, 20 and 60

Species	Year	PCR Mean	CBH Mean	DBH Mean	DBH Minimum	DBH Maximum	Height Mean	Height Minimum	Height Maximum	TPA Mean	TPA Minimum	TPA Maximum
Conifers	10	94.4	0.4	0.7	0.0	1.4	7.4	2.9	10.7	231	211	236
Conifers	20	70.5	7.0	4.3	0.1	7.2	23.6	5.1	38.1	232	211	236
Conifers	60	58.8	31.0	13.5	4.2	24.6	75.2	24.0	121.3	192	179	201
Hardwoods	10	79.8	2.1	1.2	0.1	2.5	10.2	5.5	20.3	50	38	63
Hardwoods	20	44.2	13.0	3.9	2.0	6.2	23.3	14.1	35.8	20	15	25
Hardwoods	60	29.3	26.9	6.0	3.4	10.2	38.1	20.9	53.8	12	10	18

PCR=percent crown ratio; CBH=canopy base height in feet; DBH=diameter at breast height in inches; TPA=trees per acre

Within the first several years, trees growing in more favorable conditions would have larger diameters and be taller. Thinning would favor the most vigorous trees, removing trees growing in less productive pockets. Thinning the slower growing trees would avoid a systematic thinning and begin moving the plantations toward a more random spatial distribution that is expressed through microsite productivity. Like all the other action alternatives, young plantations would still have a homogeneous vertical structure. Vertical structure would not begin to express itself until trees reach an age or size when natural regeneration begins and new cohorts are established. This would be the case for all alternatives, whether vegetation is dominated by shrubs or conifers. Alternative 5 would also have relatively the same effects as Alternative 1 in terms of future ecological resiliency, adaptation and diversity. Effects of herbicide applications for conifer release and noxious weed treatments would be the same as Alternative 1.

Existing Plantations

Same as Alternative 1.

Stand Density Index

Reforestation

Only one planting pattern is proposed under Alternative 5, but mortality and thinning, occurring at about age 7, would begin introducing horizontal complexity. After accounting for initial mortality, oak buffers and thinning, tree densities would average about 230 trees per acre. This is somewhat higher than Alternative 1, so SDI levels would increase faster. By year 40, the majority of the project area would near a SDI of 230, but not exceed it. By year 50, however, all plantations would exceed a SDI of 230. At this point, plantations would show an increase in insect-, competition- and drought-

related mortality. Assuming no thinning or other disturbances occur, all plantations would exceed a SDI of 300 by year 60, but not 365. The highest SDI achieved by year 60 would be 337 (about 64% of maximum SDI). Because all of the plantations would exceed a SDI of 230 by year 50, a greater number of acres would experience increased stress during periods of extreme drought compared to the other alternatives. As plantations approach and exceed SDI levels of 300, self-thinning would occur and SDI would decrease; however, higher tree densities would shade the forest floor and decrease shrub cover continuity. Lower shrub abundance would potentially result in more patches of conifers escaping high-severity fire, especially in cool and moist areas (Harris and Taylor 2015; Lydersen et al. 2014).

Existing Plantations

Same as Alternative 1.

Future Management Feasibility

Reforestation

Higher tree densities under Alternative 5 would result in all plantations meeting thinning criteria by year 50. Although tree densities are slightly higher than Alternative 1, both alternatives have about the same average tree sizes throughout the 60-year analysis time frame. Therefore, Alternative 5 would have more merchantable sized trees in year 50 and remove more board feet per acre sooner than all the alternatives. Residual conifer densities would still be somewhat higher than Alternative 1, ranging from 122 to 142 conifers per acre. A larger per acre volume is removed under Alternative 1 in year 60. This is partially a result of the 10-year cycles that FVS is based on. A large portion of the acres thinned under Alternative 1 in year 60 are close to a SDI of 260 in year 50 and would likely exceed this threshold well before 60 years. Because FVS works on 10-year cycles, these acres would continue to grow and accumulate volume over the course of several years despite exceeding a SDI of 260; therefore, more volume would be removed per acre in year 60. Similarly, waiting to thin until year 60 under Alternative 5 would result in larger per acre volume removed than Alternative 1, just as it did at year 50. Waiting until year 60 would also result in lower residual tree densities as it did with Alternative 1. While greater risk would be taken on if thinning were postponed 10 more years, SDI levels would still be well below the bark beetle induced maximum of 365 (Oliver 1995; Oliver and Uzoh 1997). As a result, Alternative 5 provides future generations more management options earlier across more acres.

Existing Plantations

Same as Alternative 1.

Change in Forest Structure

Reforestation

Alternative 5 would have similar effects as Alternative 1 on forest landscape structure over the next 60 years. Key differences would include less variation in canopy cover compared to Alternative 1. All plantations under both alternatives would reach mid seral structures within the next 30 years. Unlike Alternative 1, however, canopy cover would only vary between 40% and 59% as opposed to 20% to 59%. By year 50, canopy cover would increase in all plantations to greater than 60%. Under Alternative 1, the first, but not all, plantations would reach a canopy cover of greater than 60% by year 60. Canopy closure is accelerated under Alternative 5 because of higher initial planting densities. As described under Alternative 1, future management and disturbances would transition plantations into various successional classes.

Existing Plantations

Same as Alternative 1.

CUMULATIVE EFFECTS

The cumulative effects to vegetation caused by Alternative 5 are the same as Alternative 1, with the following difference. Higher planting density would result in faster development of closed canopy structures in some areas; however, proportional changes in early seral and complex early seral structures would be the same as Alternative 1.

Summary of Effects Analysis across All Alternatives

Tree Size and Species Composition

Alternatives 1 and 5 would have higher conifer survival and faster growth because herbicide applications would more effectively control competing vegetation compared to other alternatives. As a result, they would have the largest average conifer dbh after 20 years as well as throughout the next 60 years. Herbicides are proposed under Alternative 4 for use in founder stands, which would experience similar dbh and height growth as plantations in Alternatives 1 and 5; however, when averaged with tree growth outside founder stands, growth is less. Manual release treatments in Alternative 3 would not control competing vegetation as effectively as herbicides; therefore, Alternative 3 would experience higher initial mortality rates and slower conifer growth compared to Alternatives 1 and 5. Alternative 2 would not control competing vegetation and would experience the slowest conifer growth.

Alternatives 1, 3 and 5 would have more control over tree species composition. Table 3.01-14 and Table 3.01-15 show these alternatives would promote a higher proportion of ponderosa pine and sugar pine across the landscape. Alterative 2 would rely on natural regeneration; and therefore, white fir and Douglas-fir would comprise the largest proportion of future stands in the project area. Founder stands in Alternative 4 would increase the proportion of pine species across the project area, but the increase would be marginal given the small number of acres planted and the abundance of white fir and Douglas-fir outside founder stands.

Herbicide and manual release treatments would decrease the abundance of shrub cover throughout the project area. Reductions in shrub cover are expected to last for about 5 years at which time they would begin to recover and increase in abundance over the next 15 to 20 years until conifer canopies begin to close and shade the forest floor.

Thinning existing plantations would decrease tree density and increase health and vigor of residual trees. By year 60, thinned plantations would increase the number of large trees (trees greater than 24 inches dbh) by about 15%. Over the next 60 years, tree diameter would increase across all thinned plantations, with averages ranging from about 18 to almost 31 inches dbh in the youngest and oldest plantations. Average tree heights would range from about 60 to 105 feet. Alternative 2 would not thin existing plantations; therefore, higher tree densities would slow tree growth. After 60 years, average tree diameter would increase by about 50% less than if thinned. Average tree height heights would also increase by 30 to 50% less than if thinned.

Stand Density Index

Areas that initially have the highest tree densities would develop higher levels of SDI sooner than areas with lower initial tree densities. Although Alternatives 2 and 4 would have fewer acres of conifer forest, patches of unmanaged natural conifer regeneration would exceed SDI levels of 230 and 365 much faster than would occur under the other alternatives. About 30% of conifer forest under Alternatives 2 and 4 would exceed a SDI of 230 by year 40 and about 6 to 8% would exceed a SDI of 365 by year 40. No plantations under the other alternatives would exceed a SDI of 230 within 40 years. Therefore, conifer forest under Alternatives 2 and 4 would likely begin experiencing effects of insects, competition and drought before plantations in the other alternatives. Prescribed fire in Alternative 4 could slow the increase of SDI, but it is difficult to quantify the effect of fire given the high shrub cover expected under this alternative. Founder stands in Alternative 4 and plantations in

Alternatives 1, 3 and 5 never reach a SDI of 365 or more. Table 3.01-14 and Table 3.01-15 show, by year 60, most of the plantations in Alternative 1 and all of the plantations in Alternative 5 exceed a SDI of 230. Low initial survival and tree densities in Alternative 3 reduce the number of acres that reach a SDI of 230. By year 60 only 31% of Alternative 3 plantations exceed a SDI of 230.

Thinning existing plantations would reduce SDI levels and increase resiliency to insect-, drought-, and competition -related mortality. The plantations would benefit from reduced competition for a period of 20 to 40 years. During this time, residual trees would increase in size and SDI would slowly increase again. No thinning would occur under Alternative 2 and the existing plantations would experience higher levels of insect, drought, and competition related mortality sooner.

Future Management Feasibility

Alternatives 1 and 5 would provide the most potential for offsetting future management costs by harvesting intermediate sized trees. Table 3.01-14 shows reforestation activities under Alternatives 1 and 5 result in the most merchantable sized trees within the next 50 years, without first exceeding SDI levels that would result in wide spread insect, drought, and competition related mortality.

Slightly higher tree densities in Alternative 5 results in all plantations reaching a SDI that requires thinning by year 50, while some plantations in Alternative 1 would not reach as high of SDI levels in this time frame. Therefore, all plantations could be commercially thinned by year 50 under Alternative 5. Sometime between years 50 and 60, the remaining plantations in Alternative 1 would need to be thinned. Allowing the trees to accumulate volume for an additional 10 years would result in more volume being removed per acre. Low initial seedling survival and low planting densities in some areas would result in far fewer acres requiring thinning within the next 50 to 60 years; therefore, while trees would grow to a merchantable size, too few would exist to warrant harvesting and less cost offsets for future management. Very high tree densities in Alternative 2 would result in slow tree growth over the next 60 years. Some areas would eventually have trees that are of merchantable size; however, many trees would be too small to be commercially harvested even 60 years from now. Although founder stands in Alternative 4 would grow to commercial size, tree densities would be too low to commercially harvest and provide no cost offset for future fuels reduction and wildlife habitat management.

Table 3.01-14 Comparison of Alternatives: Summary of Vegetation Effects for Reforestation

Indicator	Alternative 1 (Proposed Action)	Alternative 2 (No Action)	Alternative 3	Alternative 4	Alternative 5
Average conifer dbh at year 20 (inches)	4.3	1.7	2.8	1.9	4.3
Average conifer height at year 20 (feet)	23.2	12.4	16.3	13.1	23.6
Percent conifer in pine	70	23	70	27	75
Percent conifer forest exceeding stand density index of 230 at year 60	90	100	31	100	100
Percent conifer forest exceeding stand density index of 365 at year 60	0	38	0	29	0
Future potential timber yield (mmbf)	163	42	48	42	160
Acres reforested	26,009	9,825	26,009	12,779	26,009
Acres not reforested	0	16,184	0	13,230	0

Table 3.01-15 shows thinning existing plantations would result in larger trees over the next 60 years compared to not thinning. Therefore, more merchantable intermediate-sized trees would exist in the future that could be harvested to offset future management costs.

Table 3.01-15 Comparison of Alternatives: Summary of Vegetation Effects for Existing Plantations

Indicator	Alternative 1 (Proposed Action)	Alternative 2 (No Action)	Alternative 3	Alternative 4	Alternative 5
Average conifer dbh at year 20	18-31 inches with 15% more large trees (greater than 24) than Alternative 2	9-17 inches with 15% fewer large trees (greater than 24) than Alternative 1	Same as 1	Same as 1	Same as 1
Average conifer height at year 20	60-105 feet	33-75 feet	Same as 1	Same as 1	Same as 1
Tree species composition	Increase in sugar pine, white fir, Douglas-fir and incense cedar	Unchanged	Same as 1	Same as 1	Same as 1
Stand density index	Majority of acres reduced well below 230 for 20 to 40 years	Majority of acres maintained high levels well over 230	Same as 1	Same as 1	Same as 1
Forest structure	Heterogeneous open canopy	Homogeneous closed canopy	Same as 1	Same as 1	Same as 1

Change in Forest Structure

Alternatives 1, 3 and 5 would reforest 26,009 acres. While not every acre or even 100% of each acre would be reforested (e.g., oak patches, rock outcrops, sensitive plant areas), these alternatives would provide a relatively broad distribution of tree regeneration across more acres than Alternatives 2 and 4. In the long-term, it would promote greater species diversity in large areas that currently have little or no natural conifer regeneration. Large patches of shrubs and hardwood vegetation slow establishment of conifers and can require decades to centuries to convert from shrub-dominance to conifer forest (Collins and Roller 2013; Conard and Radosevich 1982a; Nagel and Taylor 2005; Wilken 1967). Planting conifers in areas that lack natural conifer regeneration would accelerate the development of conifer canopy and promote structural diversity in areas that would otherwise remain dominated by shrubs or hardwoods in the long-term. While Alternative 4 would provide opportunities for planting desirable conifer species, the total acres planted are far less than under Alternatives 1, 3 and 5. Currently an excess of early seral structure exists across the Rim Fire landscape. Alternatives 1, 3 and 5 would accelerate the establishment of conifer forest on more acres, which would move a larger proportion of the Rim Fire landscape toward mid and late seral forest structures that are currently lacking.

Thinning existing plantations would increase the acres of mid seral open canopy forest within the Rim Fire landscape. Thinning will also promote fine-scale heterogeneity by creating a range of canopy openings, tree clumps and individual trees. Not thinning the existing plantations would maintain closed canopy conditions, which are currently in excess across the landscape. Alternative 2 would also maintain homogeneous structures.

References

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

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.

Balandier, P., C. Collet, J.H. Miller, P.E. Reynolds, and S.M. Zedaker. 2006. Designing forest vegetation management strategies based on the mechanisms and dynamics of crop tree competition by neighboring vegetation. Forestry 79(1):3-27.

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.

Bohlman, G. 2012. Inventory and Monitoring of Current Vegetation Conditions, Forest Stand Structure, and Regeneration of Conifers and Hardwoods within the Freds Fire Boundary, Annual Progress Report, 2012 Field Season. University of California-Davis.

Bonnet, V.H., A.W. Schoettle, and W.D. Shepperd. 2005. Postfire environmental conditions influence the spatial pattern of regeneration for *Pinus ponderosa*. Canadian Journal of Forest Research 35:37-47.

Bonnicksen, T.M. and E.C. Stone. 1982. Reconstruction of a presettlement giant sequoia-mixed conifer forest community using the aggregation approach. Ecology 63:1134-1148.

Breshears, D.D., O.B. Myers, C.W. Meyer, F.J. Barnes, C.B. Zou, C.D. Allen, N.G. McDowell, and W.T. Pockman. 2009. Tree die-off in response to global change-type drought: mortality insights from a decade of plant water-potential measurements. Frontiers in Ecology and the Environment 7(4):185-189.

Brown, R.T. and J.K. Agee, J.F. Franklin. 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18(4):903-912.

Burns, K.S., A.W. Schoettle, W.R. Jacobi, and M.F. Mahalovich. 2008. Options for the Management of White Pine Blister Rust in the Rocky Mountain Region. Gen. Tech. Rep. RMRS-GTR-206. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Busse, M.D. 2000. Ecological Significance of Nitrogen Fixation by Actinorhizal Shrubs in Interior Forests of California and Oregon. In: Forest Biology and Forest Management, (eds.) Powers, R.F., D.L. Hauxwell, G.M. Nakamura, pp. 23–41. Gen. Tech. Rep. PSW-GTR-178. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

Busse, M.D., P.H. Cochran, and J.W. Barrett. 1996. Changes in ponderosa pine site productivity following removal of understory vegetation. Soil Science Society of America Journal 60(6):1614-1621.

CDFG 2005. Habitat Classification Rules, California Wildlife Habitat Relationships System. California Department of Fish and Game, California Interagency Wildlife Task Group, April 2005.

CDFW 2014b. California Department of Fish & Wildlife California Natural Diversity Database (CNDDB). Current version and subscription. Biogeographic Data Branch, Sacramento, CA. Online: http://www.dfg.ca.gov/biogeodata/cnddb/

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

Collins, B.M., J.M. Lydersen, R.G. Everett, D.L. Fry, and S. Stephens. 2015. Novel characterization of landscape-level variability in historical vegetation structure. Ecological Applications 25(5):1167-1174.

Conard, S.G., A.E. Jaramillo, K. Cromack, Jr., and S.R. Compilers. 1985. The Role of the Genus *Ceanothus* in Western Forest Ecosystems. Gen. Tech. Rep. PNW-GTR-182. USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, OR.

Conard, S.G. and S.R. Radosevich. 1982a. Growth responses of white fir to decreased shading and root competition by montane chaparral shrubs. Forest Science 28:309-320.

Conard, S.G. and S.R. Radosevich. 1982b. Post-fire succession in white fir (*Abies concolor*) vegetation of the northern Sierra Nevada. Madroño 29:42-56.

Coppoletta, M., K.E. Merriam, and B.M. Collins. 2015. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecological Society of America preprint:1-50.

CREP 2008. Clavey River Watershed Assessment. Clavey River Ecosystem Project, Sonora, CA.

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

D'Amato, A.W., J.B. Bradford, S. Fraver, and B.J. Palik. 2013. Effects of thinning on drought vulnerability and climate response in north temperate forest ecosystems. Ecological Applications 23(8):1735-1742.

Dale, V.H., L.A. Joyce, S. McNulty, R.P. Neilson, M.P. Ayres, M.D. Flannigan, P.J. Hanson, L.C. Irland, A.E. Lugo, C.J. Peterson, D. Simberloff, F.J. Swanson, B.J. Stocks, and B.M. Wotton. 2001. Climate change and forest disturbances. BioScience 51(9):723-734.

DellaSala, D.A., M.L. Bond, C.T. Hanson, R.L. Hutto, and D.C. Odion. 2014. Complex early seral forests of the Sierra Nevada: What are they and how can they be managed for ecological integrity? Natural Areas Journal 34(3):310-324.

DiTomaso, J.M., D.B. Marcum, M.S. Rasmussen, E.A. Healy, and G.B. Kyser. 1997. Post-fire herbicide sprays enhance native plant diversity. California Agriculture 51(1):6-11.

Dixon, G.E. 2002. Essential FVS: A user's guide to the Forest Vegetation Simulator. Internal Rep. (Revised: February 2015). USDA Forest Service, Forest Management Service Center, Fort Collins, CO.

Donato, D.C., J.B. Fontaine, J.L. Campbell, W.D. Robinson, J.B. Kauffman, and B.E. Law. 2006. Post-wildfire logging hinders regeneration and increases fire risk. Science 311:351.

Dunning, D. and L.H. Reineke. 1933. Preliminary yield tables for second-growth stands in the California pine region. Technical Bulletin No. 354. USDA Forest Service, Washington D.C.

Eidenshink, J., B. Schwind, K. Brewer, Z.-L. Zhu, B. Quayle, and S. Howard. 2007. A project for monitoring trends in burn severity. Fire Ecology 3(1):3-21.

Ferrell, G.T., W.J. Otrosina, and C.J. Demars, Jr. 1994. Predicting susceptibility of white fir during a drought-associated outbreak of the fir engraver, *Scolytus ventralis*, in California. Canadian Journal of Forest Research 24:302-305.

Fiske, J.N. 1981. Evaluating the need for release from competition from woody plants to improve conifer growth rates. In: Proceedings, Third Annual Forest Vegetation Management Conference, November, 4-5, pp 24-44. Forest Vegetation Management Conference, Redding, CA.

Flannigan, M.D., B.J. Stocks, and B.M. Wotton. 2000. Climate change and forest fires. The Science of the Total Environment 262:221-229.

Franklin, J.F. and J. Fites-Kaufman. 1996. Assessment of late-successional forests of the Sierra Nevada. In: Sierra Nevada Ecosystem Project: Final Report to Congress, pp. 627-635. Center for Water and Wildland Resources, Davis, CA.

Fulé, P.Z., T.W. Swetnam, P.M. Brown, D.A. Falk, D.L. Peterson, C.D. Allen, G.H. Aplet, M.A. Battaglia, D. Binkley, C. Farris, R.E. Keane, E.Q. Margolis, H. Grissino-Mayer, C. Miller, C. H. Sieg, C. Skinner, S.L. Stephens, and A. Taylor . 2014. Unsupported inferences of high-severity fire in historical dry forests of the western United States: response to Williams and Baker. Global Ecology and Biogeography 23:825-830.

Goforth, B.R. and R.A. Minnich. 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.

Graham, R.T. S. McCaffrey, and T.B. Jain. 2004. Science Basis for Changing Forest Structure to Modify Wildfire Behavior and Severity. Gen. Tech. Rep. RMRS-GTR-120. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Graham, R.T., A.E. Harvey, T.B. Jain, and J.R. Tonn. 1999. The Effects of Thinning and Similar Stand Treatments on Fire Behavior in Western Forests. Gen. Tech. Rep. PNW-GTR-463. USDA Forest Service and USDI Bureau of Land Management, Pacific Northwest Research Station, Portland, OR.

Grossnicle, S.C. 2005. Importance of root growth in overcoming planting stress. New Forests 30:273-294.

Groves, C.R., E.T. Game, M.G. Anderson, M. Cross, C. Enquist, Z. Ferdaña, E. Girvetz, A. Gondor, K.R. Hall, J. Higgins, R. Marshall, K. Popper, S. Schill, and S.L. Shafer. 2012. Incorporating climate change into systematic conservation planning. Biodiversity Conservation 21:1651-1671.

Hannah, L., G.F. Midgley, T. Lovejoy, W.J. Bond, M. Bush, J.C. Lovett, D. Scott, and F.I. Woodward. 2002. Conservation of biodiversity in a changing climate. Conservation Biology 16(1):264-268.

Hanson, C.T. and D.C. Odion. 2014. Is fire severity increasing in the Sierra Nevada, California, USA? International Journal of Wildland Fire 23:1-8.

Harris, L. and A.H. Taylor. 2015. Topography, fuels, and fire exclusion drive fire severity of the Rim Fire in an old-growth mixed-conifer forest, Yosemite National Park, USA. Ecosystems 18:1192-1208.

Hermann, R.K., and D.P. Lavender. 1990. Pseudotsuga menziesii (mirb.) Franco Douglas-fir. In: Silvics of North America. Volume 1. Conifers, (eds.) Burns, R.M., and B.H. Honkala. Agric. Handb. USDA Forest Service, Washington, DC.

Hinckley, T.M. and D.R.M. Scott. 1971. Estimates of water loss and its relation to environmental parameters in Douglas-fir saplings. Ecology 52(3):520-524.

Horn, H.S. 1974. The ecology of secondary succession. Annual Review of Ecology and Systematics 5:25-37.

Hurteau, M., H. Zald, and M. North. 2007. Species-specific response to climate reconstruction in upper-elevation mixed-conifer forests of the western Sierra Nevada, California. Canadian Journal of Forest Research 37:1681-1691.

Kane, V.R., C.A. Cansler, N.A Povak, J.T. Kane, R.J. McGaughey, J.A. Lutz, D.J. Churchill, and M.P. North. 2015. 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.

Kauffman, J.B. and R.E. Martin. 1991. Factors influencing the scarification and germination of three montane Sierra Nevada shrubs. Northwest Science 65(4):180-187.

Kie, J.G. 1985. Production of deerbrush and mountain whitethorn related to shrub volume and overstory crown closure. Research Note PSW-377. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, CA.

Knapp, E.E., C.P. Weatherspoon, and C.N. Skinner. 2012. Shrub seed banks in mixed conifer forests of northern California and the role of fire in regulating abundance. Fire Ecology 8(1):32-48.

Kolden, C.A., A.M.S. Smith, and J.T. Abatzoglou. 2015. Limitations and utilization of Monitoring Trends in Burn Severity products for assessing wildfire severity in the USA. International Journal of Wildand Fire 24:1023-1028.

Kozlowski, T.T. 2002. Physiological ecology of natural regeneration of harvested and disturbed forest stands: implications for forest management. Forest Ecology and Management 158:195-221.

Laacke, R.J. 1990. Abies concolor (Gord. & Glend.) Lindl. ex Hildebr. white fir. In: Silvics of North America. Volume 1. Conifers, (eds.) Burns, R.M., and B.H. Honkala. Agric. Handb. USDA Forest Service, Washington, DC.

LANDFIRE 2009. Adapting LANDFIRE Vegetation Dynamics Models. Manual V. 1 – August 2009. LANDFIRE,The Nature Conservancy.

Lanini, W.T. and S.R. Radosevich. 1986. Response of three conifer species to site preparation and shrub control. Forest Science 32(1):61-77.

Lenihan, J.M. D. Bachelet, R.P. Neilson, and R. Drapek. 2008. Response of vegetation distribution, ecosystem productivity, and fire to climate change scenarios for California. Climatic Change 87(Suppl 1):S215-S230.

Lesser, M.R. and S.T. Jackson. 2013. Contributions of long-distance dispersal to population growth in colonizing *Pinus ponderosa* populations. Ecology Letters 16:380-389.

Littell, J.S., D. McKenzie, D.L. Peterson, and A.L. Westerling. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916-2003. Ecological Applications 19(4):1003-1021.

Loarie, S.R., B.E. Carter, K. Hayhoe, S. McMahon, R. Moe, C.A. Knight, and D.D. Ackerly. 2008. Climate change and the future of California's endemic flora. PLoS ONE 3(6):e2502.

Long, J.N. and J.D. Shaw. 2012. A density management diagram for even-aged Sierra Nevada mixed-conifer stands. Western Journal of Applied Forestry 27(4):187-195.

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

Lutz, J.A., J.W. van Wagtendonk, A.E. Thode, J.D. Miller, and J.F. Franklin. 2009. Climate, lightning ignitions, and fire severity in Yosemite National Park, California, USA. International Journal of Wildland Fire 18:765-774.

Lydersen, J. and M. North. 2012. Topographic variation in structure of mixed-conifer forests under an active-fire regime. Ecosystems 15:1134-1146.

Lydersen, J., M. North, E.E. Knapp, and B.M. Collins. 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.

Lydersen, J.M., M.P. North, and B.M. Collins. 2014. Severity of an uncharacteristically large wildfire, the Rim Fire, in forests with relatively restored frequent fire regimes. Forest Ecology and Management 328:326-334.

Mallek, C., H. Safford, J. Viers, and J. Miller. 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. Ecosphere 4(12):153.

McDonald, P.M. and C.S. Abbott. 1997. Vegetation trends in a 31-year-old ponderosa pine plantation: effect of different shrub densities. Research Paper PSW-RP-231. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

McDonald, P.M. and G.A. Everest. 1996. Response of young ponderosa pines, shrubs, and grasses to two release treatments. Research Note PSW-RN-419-Web. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

McDonald, P.M. and G.O. Fiddler. 1995. Development of a mixed shrub-ponderosa pine community in a natural and treated condition. Research Paper PSW-RP-224-Web. USDA Forest Service, Pacific Southwest Research Paper, Albany, CA.

McDonald, P.M. and G.O. Fiddler. 1997. Vegetation treatments in a young ponderosa pine plantation treated by manual release and mulching. Research Paper PSW-RP-234. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

McDonald, P.M. and G.O. Fiddler. 1999. Recovery of a bearclover (*Chamaebatia foliolosa*) plant community after site preparation and planting of ponderosa pine seedlings. Research Note PSW-RN-423. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

McDonald, P.M. and G.O. Fiddler. 2001. Timing and duration of release treatments affect Vegetation development in a young California white fir plantation. Research Paper PSW-RP-246. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

McDonald, P.M. and G.O. Fiddler. 2010. Twenty-five Years of Managing Vegetation in Conifer Plantations in Northern and Central California: Results, Application, Principles, and Challenges. Gen. Tech. Rep. PSW-GTR-231. USDA Forest Service, Pacific Southwest Research Stations, Albany, CA.

McDonald, P.M., G.O. Fiddler, and D.A. Potter. 2004. Ecology and Manipulation of Bearclover (*Chamaebatia foliolosa*) in Northern and Central California: The Status of Our Knowledge. Gen. Tech. Rep. PSW-GTR-190. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

McGinnis, T.W., J.E. Keeley, S.L. Stephens, and G.B. Roller. 2010. Fuel buildup and potential fire behavior after stand-replacing fires, logging fire-killed trees and herbicide shrub removal in Sierra Nevada forests. Forest Ecology and Management 260:22-35.

Millar, C.I., N.L. Stephenson, and S.L. Stephens. 2007. Climate change and forest of the future: managing in the face of uncertainty. Ecological Applications 17(8):2145-2151.

Miller, J.D. and A.E. Thode. 2007. Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). Remote Sensing of Environment 109:66-80.

Miller, J.D., and B. Quayle. 2015. Calibration and validation of immediate post-fire satellite derived data to three severity metrics. Fire Ecology 11(2):12-30.

Miller, J.D. and H. Safford. 2012. Trends in wildfire severity: 1984 to 2010 in the Sierra Nevada, Modoc Plateau, and Southern Cascades, California, USA. Fire Ecology 8(3):41-57.

Miller, J.D., E.E. Knapp, C.H. Key, C.N. Skinner, C.J. Isbell, R.M. Creasy, and J.W. Sherlock. 2009a. Calibration and validation of the relative differenced Normalized Burn Ratio (RdNBR) to three measures of fire severity in the Sierra Nevada and Klamath Mountains, California, USA. Remote Sensing of Environment 113:645-656.

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

Minnich, R.A., M.G. Barbour, J.H. Burk, and J. Sosa-Ramirez. 2000. Californian mixed-conifer forests under unmanaged fire regimes in the Sierra San Pedro Martir, Baja California, Mexico. Journal of Biogeography 27(1):105-129.

Moghaddas, J.J. and L. Craggs. 2007. A fuel treatment reduces fire severity and increases suppression efficiency in a mixed conifer forest. International Journal of Wildland Fire 16:673-678.

Nagel, T.A. and A.H. Taylor. 2005. Fire and persistence of montane chaparral in mixed conifer forest landscapes in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. Journal of the Torrey Botanical Society 132(3):442-457.

Nathan, R., G.G. Katul, H.S. Horn, S.M. Thomas, R. Oren, R. Avissar, S.W. Pacala, and S.A. Levin. 2002. Mechanisms of long-distance dispersal of seeds by wind. Nature 418:409-413.

North, M., A. Brough, J. Long, B. Collins, P. Bowden, D. Yasuda, J. Miller, and N. Sugihara. 2015. Constraints on mechanized treatment significantly limit mechanical fuels reduction extent in the Sierra Nevada. Journal of Forestry 113(1):40-48.

North, M., G. Steger; R. Denton, G. Eberlein, T. Munton, and K. Johnson. 2000. Association of weather and nest-site structure with reproductive success in California spotted owls. Journal of Wildlife Management. 64:797-807.

North, M. P. Stine, K. O'Hara, W. Zielinski, and S. Stephans. 2009. An Ecosystem Management Strategy for Sierran Mixed-Conifer Forests. Gen. Tech. Rep. PSW-GTR-220, 2nd printing, with addendum, February 2010. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

Odion, D.C. and C.T. Hanson. 2013. Projecting impacts of fire management on a biodiversity indicator in the Sierra Nevada and Cascades, USA: the black-backed woodpecker. The Open Forest Science Journal 6: doi: 10.2174/1874447820130508001

Odion, D.C., C.T. Hanson, A. Arsenault, W.L. Baker, D.A. DellaSala, R.L. Hutto, W. Klenner, M.A. Moritz, R.L. Sherriff, T.T. Veblen, and M.A. Williams. 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS ONE 9(2): e87852. doi:10.1371/journal.pone.0087852

Oliver, W.W. 1979. Early response of ponderosa pine to spacing and brush: observations on a 12-year-old plantation. Research Note PSW-341. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, CA.

Oliver, W.W. 1990. Spacing and shrub competition influence 20-year development of planted ponderosa pine. Western Journal of Applied Forestry 5(3):79-82.

Oliver, W.W. 1995. Is self-thinning in ponderosa pine ruled by Dendroctonus bark beetles? In: Proceedings of the 1995 National Silviculture Workshop, pp. 213-218. Gen. Tech. Rep. GTR-RM-267, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Oliver, W.W. 1997. Twenty-five-year growth and mortality of planted ponderosa pine repeatedly thinned to different stand densities in Northern California. Western Journal of Applied Forestry 12:122-130.

Oliver, W.W. and F.C.C. Uzoh. 1997. Maximum stand densities for ponderosa pine and red and white fir in northern California. In: Proceedings of the 18th Annual Forest Vegetation Management Conference, Sacramento, CA, January 14-16, 1997, pp. 57-65.

Oliver, W.W. and R.A. Ryker. 1990. Pinus ponderosa Dougl. Ex Laws. ponderosa pine. In: In: Silvics of North America. Volume 1. Conifers, (eds.) Burns, R.M., and B.H. Honkala. Agric. Handb. USDA Forest Service, Washington, DC.

Oliver, W.W. and R.F. Powers. 1978. Growth models for ponderosa pine: I. yield of unthinned plantations in Northern California. Research Paper PSW-133. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, CA.

Powers, R.F. and G.T. Ferrell. 1996. Moisture, nutrient, and insect constraints on plantation growth: the “Garden of Eden” study. New Zealand Journal of Forestry Science 26(1/2):126-144.

Powers, R.F. and P.E. Reynolds. 1999. Ten-year responses of ponderosa pine plantations to repeated vegetation and nutrient control along an environmental gradient. Journal of Canadian Forest Research 29:1027-1038.

Quinn-Davidson, L.N. and J.M. Varner. 2012. Impediments to prescribed fire across agency, landscape and manager: an example from northern California. International Journal of Wildland Fire 21:210-218.

Rebain, S. 2005. California Wildlife Habitat Relationship (CWHR) AddFile (for Region 5 Variants of FVS). USDA Forest Service, Fort Collins, CO. Online: http://www.fs.fed.us/fmsc/fvs/software/addfiles.php

Reineke, L.H. 1933. Perfecting a stand-density index for even-aged forests. Journal of Agricultural Research 46:627-638.

Reinhardt, E.D., R.E. Keane, D.E. Calkin, and J.D. Cohen. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. Forest Ecology and Management 256:1997-2006.

Ritchie, M.W. and J.D. Hamann. 2015. CONIFERS package in R: rconifers. USDA Forest Service, Pacific Southwest Research Station, Redding, CA.

Roccaforte, J.P., P.Z. Fulé, W.W. Chancellor, and D.C. Laughlin. 2012. Woody debris and tree regeneration dynamics following severe wildfires in Arizona ponderosa pine forests. Canadian Journal of Forest Research 42:593-604.

Royce, E.B. and M.G. Barbour. 2001. Mediterranean climate effects. I. conifer water use across a Sierra Nevada ecotone. American Journal of Botany 88(5):911-918.

Russell, W.H., J. McBride, and R. Rowntree. 1998. Revegetation after four stand-replacing fires in the Lake Tahoe Basin. Madroño 45(1):40-46.

Safford, H. 2013. Natural range of variation (NRV) for yellow pine and mixed conifer forests in the bioregional assessment area, including the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests. Unpublished report. USDA Forest Service, Pacific Southwest Region, Vallejo, CA.

Safford, H.D., J. Miller, D. Schmidt, B. Roath, and A. Parsons. 2008. BAER soil burn severity maps do not measure fire effects to vegetation: a comment on Odion and Hanson (2006). Ecosystems 11:1- 11.

Safford, H., M. North, and M.D. Meyer. 2012. Chapter 3: Climate change and the relevance of historical forest conditions. In: Managing Sierra Nevada Forests. Gen. Tech. Rep. PSW-GTR-237. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

Saigo, B.W. 1969. The Relationship of Non-recovered Rodent Caches to the Natural Regeneration of Ponderosa Pine. Masters of Arts Thesis. Oregon State University, Corvallis, OR.

Sapsis, D. and C. Brandow. 1997. Turning Plantations into Healthy, Fire Resistant Forests: Outlook for the Granite Burn. Fire and Resource Assessment Program, California Department of Fire Protection.

Sensenig, T., J.D. Bailey, and J.C. Tappeiner. 2013. Stand development, fire and growth of old-growth and young forests in southwestern Oregon, USA. Forest Ecology and Management 291:96-109.

Shatford, J.P.A., D.E. Hibbs, and K.J. Puettman. 2007. Conifer regeneration after forest fire in the Klamath-Siskiyous: how much, how soon? Journal of Forestry 105:139-146.

Shimizu, J.Y. and W.T. Adams. 1993. The effect of alternative silvicultural systems on genetic diversity in Douglas-fir. In: Proceedings, 22nd Southern Forest Tree Improvement Conference, June 14–17, Atlanta, GA, pp. 292–297. USDA Forest Service, Southern Region; Institute of Paper Science and Technology; Georgia Forestry Commission; and, University of Georgia Cooperative Extension Service.

Show, S.B. and E.I. Kotok. 1924. The Role of Fire in the California Pine Forests. Department Bulletin No. 1294. USDA Forest Service, Washington, D.C.

Siegel, R.B., M.W. Tingley, and R.L. Wilkerson. 2011. Black-backed Woodpecker MIS surveys on Sierra Nevada National Forests: 2010 annual report. Report to USFS Pacific Southwest Region. The Institute for Bird Populations, Point Reyes Station, California.

Skinner, C.N. and A.H. Taylor. 2006. Chapter 10: Southern Cascades Bioregion. In: Fire in California's ecosystems, (eds.) Sugihara, N.G., J.W. van Wagtendonk, K.E. Shaffer, J. Fites-Kaufman, and A.E. Thode. University of California Press, Berkeley, CA.

SNRC 2012. Plumas Lassen Study 2011 Annual Report. USDA Forest Service, Pacific Southwest Research Station, Sierra Nevada Research Center, Davis, CA.

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

Stephenson, N.L. 1990. Climatic control of vegetation distribution: the role of the water balance. American Naturalist 135(5):649-670.

Stuart, J.D., M.C. Grifantini, and L. Fox III. 1993. Early successional pathways following wildfire and subsequent silvicultural treatment in Douglas-fir/hardwood forests, NW California. Forest Science 39(3):561-572.

Sudworth, G.B. 1900. Stanislaus and Lake Tahoe Forest Reserves, California, and Adjacent Territory. In: Twenty-First Annual Report of the United States Geological Survey to the Secretary of the Interior, 1899-1900, (Director) C. D. Walcott, Part V - Forest Reserves, (Chief of Division) H. Gannett. Washington Government Printing Office. pp. 499-561.

Swanson, M.E., J.F. Franklin, R.L. Beschta, C.M. Crisafulli, D.A. DellaSala, R.L. Hutto, D.B. Lindenmayer, and F.J. Swanson. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9(2):117-125.

Tappeiner, J.C. II and S.R. Radosevich. 1982. Effect of bearmat (*Chamaebatia foliolosa*) on soils moisture and ponderosa pine (*Pinus ponderosa*) growth. Weed Science 30:98-101.

Tesch, S.D. and S.D. Hobbs. 1989. Impact of shrub sprout competition on Douglas-fir seedling development. Western Journal of Applied Forestry 4(3):89-92.

Thorne, J.H., B.J. Morgan, and J.A. Kennedy. 2008. Vegetation change over sixty years in the central Sierra Nevada, California, USA. Madroño 55(3):223-237.

Tilman, D., J. Knops, D. Wedin, P. Reich, M. Ritchie, and E. Siemann. 1997. The influence of functional diversity and composition on ecosystem processes. Science 277:1300-1302.

USDA 1995. Paper Reforestation Project Final Environmental Impact Statement (M0990-43). USDA Forest Service, Stanislaus National Forest, Sonora, CA, pages I-6 to I-10.

USDA 1999. Pacific Southwest Region Soil Interpretations. USDA Forest Service, Vallejo, CA.

USDA 2008a. Soil Survey Geographic (SSURGO) database for Stanislaus National Forest Area. USDA National Resources Conservation Services (NRCS). Online: http://www.csc.noaa.gov/digitalcoast/data/ssurgo

USDA 2010a. Stanislaus National Forest Plan Direction, April 21, 2010. Plus errata. USDA Forest Service, Stanislaus National Forest, Sonora, CA. Online: http://www.fs.fed.gov/Internet/FSE_DOCUMENTS/stelprdb5154788.pdf

USDA 2011a. Region 5 Ecological Restoration Leadership Intent. USDA Forest Service, Pacific Southwest Region, Vallejo, CA.

USDA 2014. Rim Fire Recovery Environmental Impact Statement. Forest Service, Stanislaus National Forest. Sonora, CA. August 2014.

USDA 2014e. Rim Fire Hazard Trees and Rim Fire Recovery Emergency Situation Determination Request Relevant Information. March 2014. Forest Service, Stanislaus National Forest, Sonora, CA.

USDA 2015a. FSH 1909.12 Land Management Planning Handbook, Chapter 20: Land Management Plan. WO Amendment 1909.12-2015-1. USDA Forest Service, Washington Office, Washington, D.C.

USDA 2015c. FSH 1909.12 Land Management Planning Handbook, Chapter Zero Code. WO Amendment 1909.12-2015-1. USDA Forest Service, Washington Office, Washington, D.C.

USDA 2015d. Forest Health Protection Aerial Detection Survey, North Sierra Highlands and Modoc Plateau/NF, July 22th-August 3rd, 2015. USDA Forest Service, Forest Health Protection.

USDA 2015g. Post-Fire Vegetation Severity Data for Pacific Southwest Region NEPA Planning. Memo signed by Jeanne Wade Evans on February 5, 2015 for Randy Moore, Regional Forester Pacific Southwest Region. USDA Forest Service, Pacific Southwest Region, Vallejo, CA.

Vander Wall, S.B., K.M. Kuhn, and M.J. Beck. 2005. Seed removal, seed predation, and secondary dispersal. Ecology 86(3):801-806.

van Mantgem, P.J., N.L. Stephenson, E. Knapp, J. Battles, and J.E. Keeley. 2011. Long-term effects of prescribed fire on mixed conifer forest structure in the Sierra Nevada, California. Forest Ecology and Management 261:989-994.

van Wagtendonk, J.W. and P.E. Moore.2010. Fuel deposition rates of montane and subalpine conifers in the central Sierra Nevada, California, USA. Forest Ecology and Management 259: 2122–2132.

Wagner, R.G. and A.P. Robinson. 2006. Critical period of interspecific competition for four northern conifers: 10-year growth response and associated vegetation dynamics. Canadian Journal of Forest Research 36:2474-2485.

Waring, R.H. and B.D. Clearly. 1967. Plant moisture stress: evaluation by pressure bomb. Science 155(3767):1248-1254.

Westerling, A.L. and B.P. Bryant. 2008. Climate change and wildfire in California. Climatic Change 87(Suppl 1):S231-S249.

Westerling, A.L. H.G. Hidalgo, D.R. Cayan, and T.W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313:940-943.

Williams, M.A. and W.L. Baker. 2012. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Global Ecology and Biogeography 21(10):1042-1052.

Williams, M.A. and W.L. Baker. 2014. High-severity fire corroborated in historical dry forests of the western United States: response to Fulé et al. Global Ecology and Biogeography 23(7):831-835.

Willis, K.J. and S.A. Bhagwat. 2009. Biodiversity and climate change. Science 326:806-807.

Wilken, G.C. 1967. History and fire record of a timberland brush field in the Sierra Nevada of California. Ecology 48(2):302-304.

Whittier, T.R. and A.N. Gray. 2016. Tree mortality based fire severity classification for forest inventories: a Pacific Northwest national forests example. Forest Ecology and Management 359:199-209.

WRCC 2015. RAWS USA Climate Archive, Mount Elizabeth California. Western Regional Climate Center. Online: http://www.raws.dri.edu/cgi-bin/rawMAIN.pl?caCELI

Zald, H.S.J., A.N. Gray, M. North, and R.A. Kern. 2008. Initial tree regeneration responses to fire and thinning treatments in a Sierra Nevada mixed-conifer forest, USA. Forest Ecology and Management 256:168-179.

Zeglen, S., J. Pronos, and H. Merler. 2010. Silvicultural management of white pines in western North America. Forest Pathology 40:347-368.

Zhang, J., W.W. Oliver, and M.D. Busse. 2006. Growth and development of ponderosa pine on sites of contrasting productivities: relative importance of stand density and shrub competition effects. Journal of Canadian Forest Research 36:2426-2438.

Zhang, J., W.W. Oliver, M.W. Ritchie, and D.L. Neal. 2013. Overstory and understory dynamics in a ponderosa pine plantation vary with stand density in Sierra Nevada: 40-year results. Forest Science 59(6):670-680.

Zielinski, W.J., R.L. Truex, G.A. Schmidt, F.V. Schlexer, K.N. Schmidt, and R.H. Barrett. 2004. Resting habitat selection by fishers in California. Journal of Wildlife Management 68(3):475-492.

Personal Communication

Collins, B.M. 2015. USDA Forest Service, Pacific Southwest Research Station, Davis, CA; Center for Fire Research and Outreach, University of California, Berkeley, CA.

A. Summary of Growth Model Outputs

The following acronyms are used throughout Appendix A:

BO = California black oak

BM = bigleaf maple

CBH = canopy base height (feet)

DBH = diameter at breast height (inches)

DF = Douglas-fir

DTFC = deep till with forest cultivation

HT = height (feet)

IC = incense cedar

JP = Jeffrey pine

OCM = open canopy mosaic

OFM = old forest mosaic

OH = other hardwoods

PCR = percent crown ratio

PP = ponderosa pine

SFMA = strategic fire management area

SP = sugar pine

TPA = trees per acre

WF = white fir

A.01 LANDSCAPE POSITIONS

The following tables summarize tree characteristics and density by type (conifers and hardwoods), landscape position and alternative.

Table A.01-1 Alternative 1: Tree Characteristics by Landscape Position for Conifers and Hardwoods

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
Emergency Travel Routes (866 acres)	Conifers	10	0.7	7.5	91.2	0.7	104
		20	4.4	23.6	71.8	6.6	104
		60	15.5	77.6	66.5	26.0	98
	Hardwoods	10	1.0	9.7	78.5	2.1	48
		20	3.7	22.8	43.8	12.8	19
		60	5.8	37.3	28.8	26.5	17
Fuelbreaks and Primary Ridges within SFMA (825 acres)	Conifers	10	0.7	7.0	92.4	0.5	120
		20	4.3	23.3	71.8	6.6	120
		60	13.7	66.3	63.6	24.1	114
	Hardwoods	10	0.4	6.8	85.5	1.0	63
		20	3.9	23.5	49.5	11.9	25
		60	6.0	34.6	32.5	23.3	22
Primary Ridges outside SFMA (308 acres)	Conifers	10	0.7	7.0	92.4	0.5	132
		20	4.3	23.3	71.6	6.6	132
		60	13.5	66.0	62.7	24.6	125
	Hardwoods	10	0.4	6.8	85.5	1.0	63
		20	3.9	23.5	49.5	11.9	25
		60	5.9	34.4	32.5	23.2	22
Mid-slope OCM within SFMA (4,259 acres)	Conifers	10	0.7	7.1	93.4	0.5	143
		20	4.3	23.2	70.0	7.0	143
		60	14.5	76.1	62.6	28.4	135

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
	Hardwoods	10	1.2	10.1	81.6	1.9	51
		20	4.2	24.8	46.0	13.4	21
		60	6.3	39.5	30.1	27.6	17.9
Mid-slope OCM outside SFMA and OFM within SFMA (5,594 acres)	Conifers	10	0.6	7.1	93.4	0.5	175
		20	4.3	23.7	69.3	7.3	174
		60	14.1	75.8	60.6	29.8	160
	Hardwoods	10	1.0	9.2	73.5	2.4	51
		20	3.8	22.7	40.1	13.6	21
		60	5.9	37.4	26.3	27.5	17
Mid-slope OFM outside SFMA (5,587 acres)	Conifers	10	0.6	7.3	92.8	0.5	207
		20	4.1	22.5	69.7	6.8	207
		60	13.5	74.0	59.5	29.9	179
	Hardwoods	10	1.0	9.4	74.5	2.4	51
		20	4.2	24.4	45.4	13.3	21
		60	6.4	39.4	29.9	27.6	15
Drainages (3,858 acres)	Conifers	10	0.7	7.6	91.2	0.7	207
		20	4.4	23.6	70.2	7.1	206.8
		60	13.7	75.0	59.6	30.3	177
	Hardwoods	10	1.3	11.3	81.3	2.1	38
		20	4.1	24.2	45.1	13.3	15
		60	6.1	39.4	29.7	27.7	10
Deer Thermal (471 acres)	Conifers	10	0.6	7.3	92.8	0.5	207
		20	4.1	22.5	69.7	6.8	207
		60	13.5	74.0	59.5	29.9	179
	Hardwoods	10	1.0	9.4	74.5	2.4	51
		20	4.2	24.4	45.4	13.3	21
		60	6.4	39.4	29.9	27.6	15
Deer Hiding (175 acres)	Conifers	10	0.7	7.1	93.4	0.5	143
		20	4.3	23.2	70.0	7.0	143
		60	14.5	76.1	62.6	28.4	135
	Hardwoods	10	1.2	10.1	81.6	1.9	51
		20	4.2	24.8	46.0	13.4	21
		60	6.3	39.5	30.1	27.6	18

Table A.01-2 Alternative 2: Tree Characteristics by Landscape Position for Conifers and Hardwoods

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
Emergency Travel Routes - salvaged (407 acres)	Conifers	10	0.0	2.2	83.6	0.4	1,757
		20	1.0	9.4	59.8	3.8	1,408
		60	7.0	49.1	43.2	27.9	733
	Hardwoods	10	1.0	9.6	76.0	2.3	53
		20	3.8	23.3	53.2	10.9	21
		60	6.5	39.9	35.7	25.7	9
Emergency Travel Routes - not salvaged (31 acres)	Conifers	10	0.0	2.2	80.0	0.4	6,506
		20	1.3	11.0	53.1	5.2	2,373
		60	6.7	51.3	42.7	29.4	806
	Hardwoods	10	1.1	10.0	78.9	2.1	78
		20	3.4	22.3	35.5	14.4	21
		60	6.3	39.9	25.9	29.6	6
Fuelbreaks and Primary Ridges - salvaged (478 acres)	Conifers	10	0.0	2.4	89.9	0.2	511
		20	1.0	8.3	69.2	2.5	372
		60	8.4	44.0	51.8	21.2	346
	Hardwoods	10	0.5	7.3	87.3	0.9	63
		20	3.1	19.8	75.3	4.9	25
		60	5.7	36.8	49.9	18.4	24
Fuelbreaks and Primary Ridges - not salvaged (113 acres)	Conifers	10	0.0	2.4	88.7	0.3	1,891

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
		20	0.9	8.2	67.0	2.7	1,214
		60	6.5	40.2	45.9	21.7	814
	Hardwoods	10	0.5	7.3	87.3	0.9	104
		20	2.5	18.9	68.5	5.9	37
		60	5.3	40.9	45.5	22.3	20
Mid-slope - salvaged (5,584 acres)	Conifers	10	0.2	4.2	89.0	0.5	279
		20	1.9	13.2	62.2	5.0	254
		60	10.4	60.8	51.5	29.5	236
	Hardwoods	10	1.7	12.7	83.3	2.1	49
		20	4.5	26.5	59.8	10.7	19
		60	7.0	44.0	39.7	26.5	17
Mid-slope - not salvaged (1,323 acres)	Conifers	10	0.2	4.2	88.4	0.5	1,035
		20	1.8	13.2	60.1	5.3	867
		60	8.0	55.6	44.8	30.7	556
	Hardwoods	10	1.7	12.8	84.2	2.0	77
		20	4.2	25.9	53.3	12.1	28
		60	6.6	43.7	35.3	28.3	13
Drainages - salvaged (1,504 acres)	Conifers	10	0.1	3.8	89.0	0.4	447
		20	2.0	13.0	61.5	5.0	446
		60	9.4	59.6	47.4	31.3	347
	Hardwoods	10	3.6	12.2	81.9	2.2	38
		20	4.3	25.3	58.0	10.6	14
		60	6.8	43.3	38.3	26.7	9
Drainages - not salvaged (385 acres)	Conifers	10	0.1	3.8	88.7	0.4	1,654
		20	1.8	13.6	56.9	5.8	1,388
		60	7.7	56.0	43.7	31.5	594
	Hardwoods	10	1.5	12.0	82.7	2.1	59
		20	4.1	24.9	46.9	13.2	22
		60	6.4	43.2	31.3	29.7	5

Table A.01-3 Alternative 3: Tree Characteristics by Landscape Position for Conifers and Hardwoods

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
Emergency Travel Routes - DTFC (408 acres)	Conifers	10	0.5	5.6	94.3	0.3	69
		20	3.1	17.4	68.5	5.5	69
		60	15.3	70.9	67.9	22.7	65
	Hardwoods	10	1.0	9.2	0.8	9.1	54
		20	3.9	23.4	53.8	10.8	22
		60	6.3	38.4	35.6	24.7	19
Emergency Travel Routes - no DTFC (611 acres)	Conifers	10	0.5	5.6	94.3	0.3	41
		20	3.1	17.4	68.7	5.5	41
		60	16.3	71.3	69.4	21.8	39
	Hardwoods	10	1.0	9.2	0.8	9.1	54
		20	3.9	23.4	53.8	10.8	22
		60	6.3	38.5	35.6	24.8	19
Fuelbreaks and Primary Ridges within SFMA - DTFC (210 acres)	Conifers	10	0.2	4.2	89.5	0.4	68
		20	2.1	12.6	69.3	3.9	67
		60	12.9	53.2	64.4	18.9	63
	Hardwoods	10	0.6	7.8	0.9	7.8	63
		20	3.3	20.8	74.0	5.4	25
		60	6.0	36.1	48.8	18.5	23
Fuelbreaks and Primary Ridges within SFMA - no DTFC (690 acres)	Conifers	10	0.2	4.2	89.5	0.4	41
		20	2.1	12.7	69.3	3.9	40
		60	13.5	52.8	65.0	18.5	38
	Hardwoods	10	0.6	7.8	0.9	7.8	63
		20	3.3	20.8	74.0	5.4	25

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
		60	6.0	36.2	48.8	18.5	23
Primary Ridges outside SFMA - DTFC (138 acres)	Conifers	10	0.2	4.2	89.5	0.4	88
		20	2.0	12.6	69.2	3.9	87
		60	12.4	52.9	63.7	19.2	82
	Hardwoods	10	0.6	7.8	0.9	7.8	63
		20	3.3	20.8	74.0	5.4	25
		60	6.0	36.4	48.8	18.6	23
Primary Ridges outside SFMA - no DTFC (246 acres)	Conifers	10	0.2	4.2	89.5	0.4	53
		20	2.1	12.7	69.2	3.9	52
		60	13.2	53.1	64.8	18.7	49
	Hardwoods	10	0.6	7.8	0.9	7.8	63
		20	3.3	20.8	74.0	5.4	25
		60	6.1	36.5	48.8	18.7	23
Mid-slope - DTFC (6,787 acres)	All Conifers	10	0.4	5.2	92.5	0.4	172
		20	2.8	16.2	66.7	5.4	172
		60	13.0	67.4	61.4	26.0	162
	Hardwoods	10	1.3	10.9	0.8	10.8	51
		20	4.6	26.5	64.6	9.4	19
		60	7.3	44.7	42.5	25.7	17
Mid-slope - no DTFC (11,750 acres)	Conifers	10	0.4	5.2	92.6	0.4	103
		20	2.8	16.3	66.6	5.4	103
		60	13.9	68.1	66.1	23.0	97
	Hardwoods	10	1.3	10.9	0.8	10.8	51
		20	4.6	26.6	64.7	9.4	19
		60	7.4	44.7	42.5	25.7	17
Drainages - DTFC (1,349 acres)	Conifers	10	0.5	5.6	91.5	0.5	172
		20	3.0	16.6	67.6	5.4	172
		60	13.2	67.9	60.0	27.2	161
	Hardwoods	10	1.1	11.5	0.9	11.4	38
		20	4.5	26.7	65.0	9.4	14
		60	7.3	45.5	43.1	25.9	13
Drainages - no DTFC (3,138 acres)	Conifers	10	0.4	5.6	91.5	0.5	103
		20	3.0	16.6	68.1	5.3	103
		60	14.3	68.9	64.4	24.5	97
	Hardwoods	10	1.5	11.5	0.9	11.4	38
		20	4.6	26.8	65.5	9.2	14
		60	7.4	45.7	43.3	25.9	13
Deer Thermal - DTFC (23 acres)	Conifers	10	0.4	5.2	92.6	0.4	138
		20	2.8	16.3	66.4	5.5	138
		60	13.4	67.8	63.9	24.5	129
	Hardwoods	10	1.4	11.1	0.8	11.0	51
		20	4.6	26.9	65.3	9.3	19
		60	7.5	45.4	43.1	25.8	17
Deer Thermal - no DTFC (156 acres)	Conifers	10	0.4	5.2	92.6	0.4	83
		20	2.8	16.3	66.7	5.4	83
		60	14.4	68.7	67.3	22.4	78
	Hardwoods	10	1.3	10.9	0.8	10.8	51
		20	4.6	26.6	64.7	9.4	19
		60	7.3	44.6	42.5	25.6	17
Deer Hiding - DTFC (102 acres)	Conifers	10	0.4	5.2	92.6	0.4	96
		20	2.8	16.3	66.6	5.4	96
		60	14.1	68.4	66.7	22.8	91
	Hardwoods	10	1.3	10.9	0.8	10.8	51
		20	4.6	26.6	64.7	9.4	19
		60	7.3	44.5	42.5	25.6	17
Deer Hiding - no DTFC (398 acres)	Conifers	10	0.4	5.2	92.6	0.4	58
		20	2.8	16.3	66.9	5.4	58

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
		60	15.1	69.2	68.3	22.0	55
	Hardwoods	10	1.3	10.9	0.8	10.8	51
		20	4.6	26.6	64.7	9.4	19
		60	7.4	44.7	42.5	25.7	17

Table A.01-4 Alternative 4: Tree Characteristics by Landscape Position for Conifers and Hardwoods

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
Mid-slope Founder Stands (2,954 acres – Remaining slope positions are the same as shown in Table A-2)	Conifers	10	0.5	5.6	94.3	0.3	69
		20	3.1	17.4	68.5	5.5	69
		60	15.3	70.9	67.9	22.7	65
	Hardwoods	10	0	0	0	0	0
		20	0	0	0	0	0
		60	0	0	0	0	0

Table A.01-5 Alternative 5: Tree Characteristics by Landscape Position for Conifers and Hardwoods

Landscape Position	Type	Age	DBH	HT	PCR	CBH	TPA
Emergency Travel Routes (1,013 acres)	Conifers	10	0.8	7.8	94.9	0.4	230
		20	4.5	23.8	70.2	7.1	231
		60	13.6	75.4	59.1	30.8	189
	Hardwoods	10	1.0	10.1	81.1	1.9	48
		20	4.1	24.9	46.7	13.3	19
		60	6.3	40.1	30.6	27.8	12
Fuelbreaks and Primary Ridges (1,566 acres)	Conifers	10	0.8	7.6	94.9	0.4	231
		20	4.6	24.6	73.2	6.6	232
		60	12.6	66.9	59.0	27.5	201
	Hardwoods	10	0.6	7.7	84.2	1.2	63
		20	3.1	20.0	39.5	12.1	25
		60	5.0	30.6	26.0	22.6	18
Mid-slope (18,985 acres)	Conifers	10	0.7	7.3	94.1	0.4	236
		20	4.1	23.1	69.9	6.9	237
		60	13.4	75.2	58.6	31.2	194
	Hardwoods	10	1.2	10.2	78.3	2.2	51
		20	3.9	23.4	44.1	13.1	21
		60	6.1	38.5	29.3	27.2	13
Drainages (4,455 acres)	Conifers	10	0.9	8.1	95.3	0.4	211
		20	4.9	25.8	72.4	7.1	212
		60	13.8	78.3	59.8	31.5	179
	Hardwoods	10	1.5	12.0	85.3	1.8	38
		20	4.1	24.7	46.9	13.1	15
		60	6.3	40.4	31.1	27.8	10

A.02 INDIVIDUAL TREE SPECIES

The following tables summarize tree characteristics and density by individual species, landscape position and alternative.

Table A.02-1 Alternative 1: Tree Characteristics by Landscape Position for Individual Tree Species

Landscape Position	Year	Species	DBH	HT	PCR	CBH	TPA
Drainages	10	BO	1.4	11.6	86.8	1.5	32.6
Drainages	10	DF	0.2	5.8	96.3	0.2	20.7
Drainages	10	IC	0.0	3.8	96.6	0.1	20.7
Drainages	10	OH	2.0	14.9	84.0	2.4	0.9

Landscape Position	Year	Species	DBH	HT	PCR	CBH	TPA
Drainages	10	OH	0.8	8.5	35.1	5.5	4.0
Drainages	10	PP	1.4	10.7	89.7	1.1	103.4
Drainages	10	SP	0.0	4.6	88.1	0.5	41.4
Drainages	10	WF	0.0	3.5	93.7	0.2	20.7
Drainages	20	BO	4.0	23.9	43.8	13.5	13.0
Drainages	20	DF	4.0	24.6	63.8	8.9	20.7
Drainages	20	IC	2.6	16.3	47.4	8.6	20.7
Drainages	20	OH	4.7	25.6	53.8	11.8	2.0
Drainages	20	PP	6.2	29.7	83.5	4.9	103.4
Drainages	20	SP	2.6	17.9	61.6	6.9	41.4
Drainages	20	WF	1.5	11.3	49.6	5.7	20.7
Drainages	60	BO	6.0	40.1	28.8	28.6	8.9
Drainages	60	DF	13.8	84.0	62.0	31.9	17.4
Drainages	60	IC	8.6	43.8	41.1	25.8	16.6
Drainages	60	OH	7.2	35.0	35.5	22.6	1.4
Drainages	60	PP	15.6	84.6	64.1	30.3	92.1
Drainages	60	SP	14.3	68.6	64.9	24.1	34.1
Drainages	60	WF	7.2	57.2	39.6	34.5	16.8
Emergency Travel Routes	10	BM	2.5	17.5	82.8	3.0	0.8
Emergency Travel Routes	10	BO	1.2	10.8	89.0	1.2	38.0
Emergency Travel Routes	10	DF	0.2	5.7	96.2	0.2	10.4
Emergency Travel Routes	10	IC	0.0	3.7	96.6	0.1	10.4
Emergency Travel Routes	10	OH	0.0	4.6	33.9	3.0	9.0
Emergency Travel Routes	10	PP	1.4	10.6	89.8	1.1	51.9
Emergency Travel Routes	10	SP	0.0	4.5	88.1	0.5	20.7
Emergency Travel Routes	10	WF	0.0	3.5	93.8	0.2	10.4
Emergency Travel Routes	20	BM	4.9	29.1	56.8	12.6	0.3
Emergency Travel Routes	20	BO	3.8	23.7	45.3	13.0	15.2
Emergency Travel Routes	20	DF	4.0	24.5	66.5	8.2	10.4
Emergency Travel Routes	20	IC	2.6	16.2	50.5	8.0	10.4
Emergency Travel Routes	20	OH	3.2	18.7	36.5	11.9	3.6
Emergency Travel Routes	20	PP	6.2	29.6	84.5	4.6	51.9
Emergency Travel Routes	20	SP	2.6	17.9	63.7	6.5	20.7
Emergency Travel Routes	20	WF	1.5	11.2	51.3	5.4	10.4
Emergency Travel Routes	60	BM	7.4	36.3	37.3	22.8	0.3
Emergency Travel Routes	60	BO	5.9	39.8	29.8	28.0	13.3
Emergency Travel Routes	60	DF	16.9	82.4	69.4	25.2	9.8
Emergency Travel Routes	60	IC	10.3	47.2	48.0	24.5	9.6
Emergency Travel Routes	60	OH	5.2	26.6	24.0	20.2	3.1
Emergency Travel Routes	60	PP	17.2	87.6	70.5	25.9	49.8
Emergency Travel Routes	60	SP	16.1	70.7	72.4	19.5	19.1
Emergency Travel Routes	60	WF	9.2	64.6	49.8	32.5	9.6
Mid-slope OCM within SFMA and deer hiding	10	BM	3.0	22.5	77.9	5.0	0.1
Mid-slope OCM within SFMA and deer hiding	10	BO	1.3	10.7	89.5	1.1	43.2
Mid-slope OCM within SFMA and deer hiding	10	DF	0.2	5.6	95.5	0.3	14.3
Mid-slope OCM within SFMA and deer hiding	10	IC	0.0	4.0	94.6	0.2	14.3
Mid-slope OCM within SFMA and deer hiding	10	OH	1.5	11.7	78.5	2.5	0.0
Mid-slope OCM within SFMA and deer hiding	10	OH	0.4	6.8	39.1	4.1	8.0
Mid-slope OCM within SFMA and deer hiding	10	PP	1.2	9.8	95.1	0.5	71.7
Mid-slope OCM within SFMA and deer hiding	10	SP	0.2	4.7	87.3	0.6	28.7
Mid-slope OCM within SFMA and deer hiding	10	WF	0.0	3.6	94.3	0.2	14.3
Mid-slope OCM within SFMA and deer hiding	20	BM	5.2	33.6	60.5	13.3	0.0
Mid-slope OCM within SFMA and deer hiding	20	BO	4.4	25.5	47.8	13.3	17.3
Mid-slope OCM within SFMA and deer hiding	20	DF	4.1	24.5	63.6	8.9	14.3
Mid-slope OCM within SFMA and deer hiding	20	IC	2.3	13.6	44.8	7.5	14.3
Mid-slope OCM within SFMA and deer hiding	20	OH	3.6	21.2	36.6	13.4	3.2
Mid-slope OCM within SFMA and deer hiding	20	PP	5.8	28.2	83.4	4.7	71.7

Landscape Position	Year	Species	DBH	HT	PCR	CBH	TPA
Mid-slope OCM within SFMA and deer hiding	20	SP	2.8	20.1	62.8	7.5	28.7
Mid-slope OCM within SFMA and deer hiding	20	WF	1.8	12.6	48.8	6.4	14.3
Mid-slope OCM within SFMA and deer hiding	60	BM	7.9	42.1	39.8	25.4	0.0
Mid-slope OCM within SFMA and deer hiding	60	BO	6.4	41.6	31.3	28.6	15.1
Mid-slope OCM within SFMA and deer hiding	60	DF	15.7	83.5	66.1	28.3	13.5
Mid-slope OCM within SFMA and deer hiding	60	IC	9.1	42.4	43.3	24.0	13.2
Mid-slope OCM within SFMA and deer hiding	60	OH	5.6	28.1	23.8	21.4	2.8
Mid-slope OCM within SFMA and deer hiding	60	PP	15.9	84.3	65.6	29.0	68.6
Mid-slope OCM within SFMA and deer hiding	60	SP	15.9	73.5	70.3	21.8	26.5
Mid-slope OCM within SFMA and deer hiding	60	WF	8.6	64.5	47.2	34.1	13.2
Mid-slope OCM outside SFMA	10	BM	2.8	21.5	73.7	5.7	0.1
Mid-slope OCM outside SFMA	10	BO	1.1	9.7	80.9	1.9	43.2
Mid-slope OCM outside SFMA	10	DF	0.3	5.6	94.5	0.3	17.5
Mid-slope OCM outside SFMA	10	IC	0.1	3.8	92.8	0.3	17.5
Mid-slope OCM outside SFMA	10	OH	1.6	12.3	80.7	2.4	0.0
Mid-slope OCM outside SFMA	10	OH	0.4	6.4	33.4	4.2	8.0
Mid-slope OCM outside SFMA	10	PP	1.1	9.6	94.5	0.5	87.7
Mid-slope OCM outside SFMA	10	SP	0.3	5.3	90.8	0.5	35.1
Mid-slope OCM outside SFMA	10	WF	0.0	3.4	92.6	0.3	17.5
Mid-slope OCM outside SFMA	20	BM	5.2	33.8	61.0	13.2	0.0
Mid-slope OCM outside SFMA	20	BO	3.8	22.8	40.0	13.7	17.3
Mid-slope OCM outside SFMA	20	DF	4.2	25.5	65.3	8.9	17.5
Mid-slope OCM outside SFMA	20	IC	2.4	14.7	44.9	8.1	17.5
Mid-slope OCM outside SFMA	20	OH	3.8	21.9	40.3	13.1	3.2
Mid-slope OCM outside SFMA	20	PP	5.8	28.4	83.1	4.8	87.7
Mid-slope OCM outside SFMA	20	SP	2.8	21.0	62.1	8.0	35.1
Mid-slope OCM outside SFMA	20	WF	1.7	12.1	40.9	7.2	16.0
Mid-slope OCM outside SFMA	60	BM	7.7	41.3	40.5	24.6	0.0
Mid-slope OCM outside SFMA	60	BO	5.9	38.8	26.3	28.6	14.1
Mid-slope OCM outside SFMA	60	DF	15.3	85.7	64.5	30.4	16.1
Mid-slope OCM outside SFMA	60	IC	8.9	42.7	42.4	24.6	15.5
Mid-slope OCM outside SFMA	60	OH	6.0	29.6	26.5	21.7	2.6
Mid-slope OCM outside SFMA	60	PP	15.5	83.8	64.0	30.2	82.2
Mid-slope OCM outside SFMA	60	SP	15.4	73.4	67.4	23.9	31.5
Mid-slope OCM outside SFMA	60	WF	7.8	59.3	41.8	34.5	14.3
Mid-slope OFM outside SFMA and deer thermal	10	BM	2.7	21.0	73.9	5.5	0.1
Mid-slope OFM outside SFMA and deer thermal	10	BO	1.1	10.0	81.4	1.9	43.2
Mid-slope OFM outside SFMA and deer thermal	10	DF	0.2	5.4	94.8	0.3	20.7
Mid-slope OFM outside SFMA and deer thermal	10	IC	0.1	4.1	96.7	0.1	20.7
Mid-slope OFM outside SFMA and deer thermal	10	OH	1.6	12.2	82.4	2.2	0.0
Mid-slope OFM outside SFMA and deer thermal	10	OH	0.4	6.5	36.8	4.1	8.0
Mid-slope OFM outside SFMA and deer thermal	10	PP	1.2	10.0	94.2	0.6	103.4
Mid-slope OFM outside SFMA and deer thermal	10	SP	0.2	4.8	85.7	0.7	41.4
Mid-slope OFM outside SFMA and deer thermal	10	WF	0.0	3.4	94.3	0.2	20.7
Mid-slope OFM outside SFMA and deer thermal	20	BM	5.0	32.8	57.8	13.8	0.0
Mid-slope OFM outside SFMA and deer thermal	20	BO	4.2	24.7	46.0	13.3	17.3
Mid-slope OFM outside SFMA and deer thermal	20	DF	4.0	25.0	63.0	9.2	20.7
Mid-slope OFM outside SFMA and deer thermal	20	IC	2.3	14.1	46.1	7.6	20.7
Mid-slope OFM outside SFMA and deer thermal	20	OH	4.0	23.1	42.6	13.3	4.0
Mid-slope OFM outside SFMA and deer thermal	20	PP	5.7	27.1	83.4	4.5	103.4
Mid-slope OFM outside SFMA and deer thermal	20	SP	2.6	19.0	61.1	7.4	41.4
Mid-slope OFM outside SFMA and deer thermal	20	WF	1.6	12.0	48.8	6.1	20.7
Mid-slope OFM outside SFMA and deer thermal	60	BM	7.7	41.3	38.3	25.5	0.0
Mid-slope OFM outside SFMA and deer thermal	60	BO	6.4	41.4	30.3	28.8	12.2
Mid-slope OFM outside SFMA and deer thermal	60	DF	14.3	84.4	63.5	30.8	18.0
Mid-slope OFM outside SFMA and deer thermal	60	IC	8.1	40.4	40.5	24.1	16.8
Mid-slope OFM outside SFMA and deer thermal	60	OH	6.2	30.8	28.1	22.2	2.8

Landscape Position	Year	Species	DBH	HT	PCR	CBH	TPA
Mid-slope OFM outside SFMA and deer thermal	60	PP	15.1	82.5	63.1	30.5	92.5
Mid-slope OFM outside SFMA and deer thermal	60	SP	14.3	69.5	65.4	24.0	34.6
Mid-slope OFM outside SFMA and deer thermal	60	WF	7.5	58.9	42.9	33.6	17.0
Fuelbreaks and Primary Ridges within SFMA	10	BO	0.4	6.8	85.5	1.0	63.4
Fuelbreaks and Primary Ridges within SFMA	10	DF	0.2	5.4	92.6	0.4	12.0
Fuelbreaks and Primary Ridges within SFMA	10	IC	0.0	3.6	94.9	0.2	12.0
Fuelbreaks and Primary Ridges within SFMA	10	PP	1.3	9.7	91.6	0.8	60.1
Fuelbreaks and Primary Ridges within SFMA	10	SP	0.1	4.9	94.7	0.3	24.0
Fuelbreaks and Primary Ridges within SFMA	10	WF	0.0	2.7	89.7	0.3	12.0
Fuelbreaks and Primary Ridges within SFMA	20	BO	3.9	23.5	49.5	11.9	25.3
Fuelbreaks and Primary Ridges within SFMA	20	DF	3.7	22.3	66.0	7.6	12.0
Fuelbreaks and Primary Ridges within SFMA	20	IC	2.2	14.0	47.6	7.3	12.0
Fuelbreaks and Primary Ridges within SFMA	20	PP	6.0	29.0	83.4	4.8	60.1
Fuelbreaks and Primary Ridges within SFMA	20	SP	2.8	20.5	68.7	6.4	24.0
Fuelbreaks and Primary Ridges within SFMA	20	WF	1.6	11.0	49.3	5.6	12.0
Fuelbreaks and Primary Ridges within SFMA	60	BO	6.0	34.6	32.5	23.3	22.3
Fuelbreaks and Primary Ridges within SFMA	60	DF	14.9	66.4	66.6	22.2	11.3
Fuelbreaks and Primary Ridges within SFMA	60	IC	8.3	37.1	40.3	22.1	11.1
Fuelbreaks and Primary Ridges within SFMA	60	PP	15.0	74.6	67.3	24.4	57.7
Fuelbreaks and Primary Ridges within SFMA	60	SP	14.9	64.5	72.3	17.8	22.3
Fuelbreaks and Primary Ridges within SFMA	60	WF	8.7	56.1	47.2	29.6	11.1
Primary Ridges outside SFMA	10	BO	0.4	6.8	85.5	1.0	63.4
Primary Ridges outside SFMA	10	DF	0.2	5.4	92.6	0.4	13.2
Primary Ridges outside SFMA	10	IC	0.0	3.6	94.9	0.2	13.2
Primary Ridges outside SFMA	10	PP	1.3	9.7	91.6	0.8	66.2
Primary Ridges outside SFMA	10	SP	0.1	4.9	94.7	0.3	26.5
Primary Ridges outside SFMA	10	WF	0.0	2.7	89.7	0.3	13.2
Primary Ridges outside SFMA	20	BO	3.9	23.5	49.5	11.9	25.3
Primary Ridges outside SFMA	20	DF	3.7	22.3	66.0	7.6	13.2
Primary Ridges outside SFMA	20	IC	2.2	14.0	47.2	7.4	13.2
Primary Ridges outside SFMA	20	PP	6.0	29.0	83.4	4.8	66.2
Primary Ridges outside SFMA	20	SP	2.8	20.4	68.4	6.4	26.5
Primary Ridges outside SFMA	20	WF	1.6	11.0	49.0	5.6	13.2
Primary Ridges outside SFMA	60	BO	5.9	34.4	32.5	23.2	22.3
Primary Ridges outside SFMA	60	DF	14.6	67.5	65.9	23.0	12.5
Primary Ridges outside SFMA	60	IC	8.1	36.6	39.8	22.0	12.2
Primary Ridges outside SFMA	60	PP	14.8	74.4	66.4	25.0	63.5
Primary Ridges outside SFMA	60	SP	14.6	63.9	71.1	18.5	24.6
Primary Ridges outside SFMA	60	WF	8.3	54.7	45.8	29.6	12.3

Table A.02-2 Alternative 2: Tree Characteristics by Landscape Position for Individual Tree Species

Landscape Position	Salvaged	Year	Species	DBH	HT	PCR	CBH	TPA
Drainages	No	10	BO	1.4	11.6	84.1	1.8	53.4
Drainages	No	10	DF	0.1	4.4	91.3	0.4	411.6
Drainages	No	10	IC	0.0	3.3	94.4	0.2	68.7
Drainages	No	10	OH	2.6	15.8	70.5	4.7	6.0
Drainages	No	10	PP	0.7	7.9	90.7	0.7	253.0
Drainages	No	10	SP	0.0	3.5	69.1	1.1	27.6
Drainages	No	10	WF	0.0	2.5	87.2	0.3	893.3
Drainages	No	20	BO	4.0	24.7	46.5	13.2	20.6
Drainages	No	20	DF	3.2	20.2	60.5	8.0	387.0
Drainages	No	20	IC	0.3	9.1	47.6	4.8	64.6
Drainages	No	20	OH	4.8	27.6	53.5	12.8	1.1
Drainages	No	20	PP	3.5	18.6	61.4	7.2	237.9
Drainages	No	20	SP	1.3	12.6	61.0	4.9	26.0

Landscape Position	Salvaged	Year	Species	DBH	HT	PCR	CBH	TPA
Drainages	No	20	WF	0.5	8.5	54.0	3.9	672.0
Drainages	No	60	BO	6.4	43.4	31.0	30.0	4.7
Drainages	No	60	DF	9.7	71.2	51.7	34.4	205.5
Drainages	No	60	IC	3.9	26.0	31.6	17.8	18.7
Drainages	No	60	OH	7.6	39.9	35.8	25.6	0.4
Drainages	No	60	PP	10.6	68.3	46.4	36.6	125.3
Drainages	No	60	SP	9.6	53.7	47.7	28.1	9.7
Drainages	No	60	WF	4.5	38.5	36.0	24.6	234.3
Drainages	Yes	10	BO	3.7	11.6	84.1	1.8	32.6
Drainages	Yes	10	DF	0.1	4.4	91.5	0.4	111.1
Drainages	Yes	10	IC	0.0	3.3	95.2	0.2	18.6
Drainages	Yes	10	OH	2.7	16.3	68.6	5.1	5.4
Drainages	Yes	10	PP	0.7	7.9	90.9	0.7	68.3
Drainages	Yes	10	SP	0.0	3.6	72.6	1.0	7.5
Drainages	Yes	10	WF	0.0	2.5	87.3	0.3	241.2
Drainages	Yes	20	BO	4.2	25.0	56.8	10.8	13.0
Drainages	Yes	20	DF	3.3	20.6	64.6	7.3	111.0
Drainages	Yes	20	IC	1.1	9.4	56.3	4.1	18.5
Drainages	Yes	20	OH	5.2	28.1	73.3	7.5	1.1
Drainages	Yes	20	PP	4.1	19.6	74.4	5.0	68.2
Drainages	Yes	20	SP	2.0	14.2	67.4	4.6	7.5
Drainages	Yes	20	WF	0.8	7.9	56.6	3.4	240.9
Drainages	Yes	60	BO	6.7	43.5	37.3	27.3	8.4
Drainages	Yes	60	DF	13.6	77.9	58.8	32.1	93.6
Drainages	Yes	60	IC	6.4	33.7	38.4	20.8	13.5
Drainages	Yes	60	OH	8.4	40.7	48.8	20.9	0.7
Drainages	Yes	60	PP	13.2	73.4	55.9	32.4	56.6
Drainages	Yes	60	SP	13.7	64.0	59.8	25.7	5.9
Drainages	Yes	60	WF	6.0	47.4	39.1	28.9	177.7
Emergency Travel Routes	No	10	BM	2.2	16.2	81.7	3.0	1.2
Emergency Travel Routes	No	10	BO	1.2	10.5	85.2	1.6	62.3
Emergency Travel Routes	No	10	DF	0.0	2.6	88.7	0.3	2403.0
Emergency Travel Routes	No	10	IC	0.0	2.3	88.9	0.3	1095.3
Emergency Travel Routes	No	10	OH	0.5	6.9	50.9	3.4	14.0
Emergency Travel Routes	No	10	PP	0.1	4.8	82.1	0.9	344.8
Emergency Travel Routes	No	10	SP	0.0	2.5	67.9	0.8	76.4
Emergency Travel Routes	No	10	WF	0.0	1.4	68.2	0.5	2586.3
Emergency Travel Routes	No	20	BM	4.9	29.9	40.3	17.9	0.3
Emergency Travel Routes	No	20	BO	3.1	21.0	34.8	13.7	17.0
Emergency Travel Routes	No	20	DF	1.6	12.4	56.1	5.4	1641.3
Emergency Travel Routes	No	20	IC	0.2	6.6	48.3	3.4	448.8
Emergency Travel Routes	No	20	OH	4.7	27.3	38.5	16.8	3.9
Emergency Travel Routes	No	20	PP	1.5	11.2	44.2	6.3	235.5
Emergency Travel Routes	No	20	SP	0.5	6.9	40.2	4.2	47.0
Emergency Travel Routes	No	60	BM	7.2	40.3	28.8	28.7	0.3
Emergency Travel Routes	No	60	BO	5.4	41.1	23.0	31.6	2.6
Emergency Travel Routes	No	60	DF	6.8	53.7	44.3	29.9	655.0
Emergency Travel Routes	No	60	IC	3.9	23.3	32.3	15.8	74.0
Emergency Travel Routes	No	60	OH	7.1	38.9	28.1	27.9	3.0
Emergency Travel Routes	No	60	PP	9.3	59.7	39.8	36.0	67.6
Emergency Travel Routes	No	60	SP	7.7	42.3	39.9	25.4	9.2
Emergency Travel Routes	Yes	10	BM	2.2	16.2	81.7	3.0	0.7
Emergency Travel Routes	Yes	10	BO	1.2	10.5	85.2	1.6	38.0
Emergency Travel Routes	Yes	10	DF	0.0	2.6	90.9	0.2	648.8
Emergency Travel Routes	Yes	10	IC	0.0	2.4	93.0	0.2	295.7
Emergency Travel Routes	Yes	10	OH	0.5	6.9	50.9	3.4	14.0
Emergency Travel Routes	Yes	10	PP	0.2	4.9	88.7	0.6	93.1

Landscape Position	Salvaged	Year	Species	DBH	HT	PCR	CBH	TPA
Emergency Travel Routes	Yes	10	SP	0.0	2.5	79.0	0.5	20.6
Emergency Travel Routes	Yes	10	WF	0.0	1.4	72.2	0.4	698.3
Emergency Travel Routes	Yes	20	BM	5.0	29.9	61.0	11.7	0.3
Emergency Travel Routes	Yes	20	BO	3.4	21.7	51.5	10.5	15.2
Emergency Travel Routes	Yes	20	DF	1.7	12.7	63.3	4.7	649.2
Emergency Travel Routes	Yes	20	IC	0.3	6.3	56.4	2.7	295.9
Emergency Travel Routes	Yes	20	OH	4.9	27.3	57.3	11.7	5.6
Emergency Travel Routes	Yes	20	PP	2.0	12.5	62.0	4.8	93.2
Emergency Travel Routes	Yes	20	SP	0.8	8.1	62.1	3.1	20.6
Emergency Travel Routes	Yes	20	WF	0.1	5.1	55.6	2.2	349.3
Emergency Travel Routes	Yes	60	BM	7.8	40.6	40.7	24.1	0.2
Emergency Travel Routes	Yes	60	BO	5.7	40.6	33.8	26.9	4.9
Emergency Travel Routes	Yes	60	DF	8.1	58.7	47.0	31.1	407.4
Emergency Travel Routes	Yes	60	IC	4.9	26.9	37.6	16.8	113.6
Emergency Travel Routes	Yes	60	OH	7.5	38.9	37.8	24.2	3.7
Emergency Travel Routes	Yes	60	PP	9.5	59.2	42.9	33.8	58.3
Emergency Travel Routes	Yes	60	SP	9.8	50.8	46.2	27.3	11.7
Emergency Travel Routes	Yes	60	WF	4.5	34.8	36.8	22.0	142.0
Mid-slopes	No	10	BM	3.2	23.7	76.8	5.5	0.1
Mid-slopes	No	10	BO	1.7	12.8	86.3	1.8	70.9
Mid-slopes	No	10	DF	0.2	4.3	93.9	0.3	275.8
Mid-slopes	No	10	IC	0.0	3.3	93.0	0.2	52.1
Mid-slopes	No	10	JP	0.8	7.6	89.3	0.8	2.6
Mid-slopes	No	10	OH	1.7	11.8	58.9	4.8	6.1
Mid-slopes	No	10	PP	0.8	7.5	89.4	0.8	229.1
Mid-slopes	No	10	SP	0.2	4.0	82.2	0.7	24.5
Mid-slopes	No	10	WF	0.0	2.5	84.3	0.4	450.5
Mid-slopes	No	20	BM	4.5	30.5	58.5	12.7	0.0
Mid-slopes	No	20	BO	4.1	25.8	52.8	12.2	27.3
Mid-slopes	No	20	DF	2.4	17.2	58.2	7.2	253.2
Mid-slopes	No	20	IC	0.5	9.3	52.1	4.5	47.8
Mid-slopes	No	20	JP	3.4	18.1	74.3	4.6	2.4
Mid-slopes	No	20	OH	5.0	28.3	67.3	9.3	1.1
Mid-slopes	No	20	PP	3.7	18.9	66.4	6.3	210.3
Mid-slopes	No	20	SP	0.9	10.1	61.1	3.9	22.5
Mid-slopes	No	20	WF	0.4	7.3	58.5	3.0	330.9
Mid-slopes	No	60	BM	7.0	38.9	38.6	23.9	0.0
Mid-slopes	No	60	BO	6.5	43.9	34.8	28.6	12.5
Mid-slopes	No	60	DF	8.7	63.9	48.4	33.0	164.5
Mid-slopes	No	60	IC	4.8	29.0	34.3	19.1	25.8
Mid-slopes	No	60	JP	11.2	63.4	63.0	23.5	1.7
Mid-slopes	No	60	OH	8.0	40.5	44.9	22.3	0.6
Mid-slopes	No	60	PP	11.6	71.0	49.2	36.1	161.6
Mid-slopes	No	60	SP	8.9	49.3	47.3	26.0	12.9
Mid-slopes	No	60	WF	4.8	39.4	39.0	24.0	190.0
Mid-slopes	Yes	10	BM	3.2	23.7	76.8	5.5	0.1
Mid-slopes	Yes	10	BO	1.7	12.8	86.3	1.8	43.2
Mid-slopes	Yes	10	DF	0.2	4.3	94.5	0.2	74.5
Mid-slopes	Yes	10	IC	0.0	3.4	93.6	0.2	14.1
Mid-slopes	Yes	10	JP	0.8	7.7	93.2	0.5	0.7
Mid-slopes	Yes	10	OH	1.7	11.8	61.5	4.5	6.1
Mid-slopes	Yes	10	PP	0.8	7.6	91.0	0.7	61.9
Mid-slopes	Yes	10	SP	0.2	4.1	84.2	0.6	6.6
Mid-slopes	Yes	10	WF	0.0	2.5	84.3	0.4	121.6
Mid-slopes	Yes	20	BM	4.6	30.5	62.0	11.6	0.0
Mid-slopes	Yes	20	BO	4.4	26.4	58.8	10.9	17.3
Mid-slopes	Yes	20	DF	2.4	17.4	59.9	7.0	74.2

Landscape Position	Salvaged	Year	Species	DBH	HT	PCR	CBH	TPA
Mid-slopes	Yes	20	IC	0.9	8.8	55.8	3.9	14.0
Mid-slopes	Yes	20	JP	3.8	18.9	77.1	4.3	0.7
Mid-slopes	Yes	20	OH	5.3	28.5	74.6	7.2	1.2
Mid-slopes	Yes	20	PP	3.9	19.4	70.6	5.7	61.7
Mid-slopes	Yes	20	SP	1.1	10.0	67.7	3.2	6.6
Mid-slopes	Yes	20	WF	0.5	6.9	59.0	2.8	97.0
Mid-slopes	Yes	60	BM	7.3	39.2	40.8	23.2	0.0
Mid-slopes	Yes	60	BO	6.9	44.3	39.0	27.0	15.4
Mid-slopes	Yes	60	DF	12.7	69.4	58.7	28.6	69.4
Mid-slopes	Yes	60	IC	6.7	34.0	38.6	20.9	12.7
Mid-slopes	Yes	60	JP	17.6	79.7	74.3	20.5	0.7
Mid-slopes	Yes	60	OH	8.5	40.4	49.6	20.4	1.1
Mid-slopes	Yes	60	PP	14.2	75.8	60.7	29.8	58.4
Mid-slopes	Yes	60	SP	12.1	56.5	62.1	21.4	6.0
Mid-slopes	Yes	60	WF	6.5	48.3	40.7	28.6	88.5
Fuelbreaks and Primary Ridges	No	10	BO	0.5	7.3	87.3	0.9	103.9
Fuelbreaks and Primary Ridges	No	10	DF	0.0	3.1	92.6	0.2	295.8
Fuelbreaks and Primary Ridges	No	10	IC	0.0	1.8	93.0	0.1	147.2
Fuelbreaks and Primary Ridges	No	10	PP	0.0	3.8	86.0	0.5	579.3
Fuelbreaks and Primary Ridges	No	10	SP	0.0	2.4	82.2	0.4	14.7
Fuelbreaks and Primary Ridges	No	10	WF	0.0	1.4	88.6	0.2	853.8
Fuelbreaks and Primary Ridges	No	20	BO	2.5	18.9	68.5	5.9	36.6
Fuelbreaks and Primary Ridges	No	20	DF	1.8	12.3	70.7	3.6	263.4
Fuelbreaks and Primary Ridges	No	20	IC	0.2	5.2	55.9	2.3	118.0
Fuelbreaks and Primary Ridges	No	20	PP	1.1	9.1	68.8	2.9	515.9
Fuelbreaks and Primary Ridges	No	20	SP	0.7	7.0	68.4	2.2	13.1
Fuelbreaks and Primary Ridges	No	20	WF	0.1	4.4	65.0	1.5	304.1
Fuelbreaks and Primary Ridges	No	60	BO	5.3	40.9	45.5	22.3	20.0
Fuelbreaks and Primary Ridges	No	60	DF	8.7	52.0	51.2	25.4	232.3
Fuelbreaks and Primary Ridges	No	60	IC	4.5	22.2	37.1	13.9	67.3
Fuelbreaks and Primary Ridges	No	60	PP	6.5	40.1	45.6	21.8	320.3
Fuelbreaks and Primary Ridges	No	60	SP	9.0	42.4	50.1	21.2	9.7
Fuelbreaks and Primary Ridges	No	60	WF	4.6	31.7	42.8	18.1	184.1
Fuelbreaks and Primary Ridges	Yes	10	BO	0.5	7.3	87.3	0.9	63.4
Fuelbreaks and Primary Ridges	Yes	10	DF	0.0	3.1	92.7	0.2	79.9
Fuelbreaks and Primary Ridges	Yes	10	IC	0.0	1.8	94.1	0.1	39.7
Fuelbreaks and Primary Ridges	Yes	10	PP	0.0	3.8	89.2	0.4	156.4
Fuelbreaks and Primary Ridges	Yes	10	SP	0.0	2.4	84.3	0.4	4.0
Fuelbreaks and Primary Ridges	Yes	10	WF	0.0	1.4	88.8	0.2	230.5
Fuelbreaks and Primary Ridges	Yes	20	BO	3.1	19.8	75.3	4.9	25.3
Fuelbreaks and Primary Ridges	Yes	20	DF	1.7	12.0	72.5	3.3	79.9
Fuelbreaks and Primary Ridges	Yes	20	IC	0.3	5.1	55.7	2.3	39.7
Fuelbreaks and Primary Ridges	Yes	20	PP	1.3	9.5	72.4	2.6	156.4
Fuelbreaks and Primary Ridges	Yes	20	SP	0.9	7.6	75.4	1.9	4.0
Fuelbreaks and Primary Ridges	Yes	20	WF	0.1	4.3	66.5	1.5	92.2
Fuelbreaks and Primary Ridges	Yes	60	BO	5.7	36.8	49.9	18.4	23.5
Fuelbreaks and Primary Ridges	Yes	60	DF	12.2	55.4	63.6	20.2	77.8
Fuelbreaks and Primary Ridges	Yes	60	IC	5.9	26.2	39.1	16.0	36.3
Fuelbreaks and Primary Ridges	Yes	60	PP	8.4	45.2	52.7	21.4	143.7
Fuelbreaks and Primary Ridges	Yes	60	SP	11.8	48.4	66.9	16.0	3.7
Fuelbreaks and Primary Ridges	Yes	60	WF	5.9	38.9	44.4	21.6	84.8

Table A.02-3 Alternative 3: Tree Characteristics by Landscape Position for Individual Tree Species

Landscape Position	DTFC	Year	Species	DBH	HT	PCR	CBH	TPA
Drainages	No	10	BO	1.3	10.9	89.8	1.1	32.6
Drainages	No	10	DF	0.2	5.1	95.5	0.2	10.3
Drainages	No	10	IC	0.1	3.3	96.0	0.1	10.3
Drainages	No	10	OH	2.7	15.9	68.7	5.0	4.9
Drainages	No	10	PP	0.8	7.3	90.3	0.7	51.7
Drainages	No	10	SP	0.1	3.9	89.7	0.4	20.7
Drainages	No	10	WF	0.0	2.9	93.2	0.2	10.3
Drainages	No	20	BO	4.5	26.5	64.8	9.3	13.0
Drainages	No	20	DF	3.3	20.6	64.7	7.3	10.3
Drainages	No	20	IC	1.7	10.8	48.7	5.5	10.3
Drainages	No	20	OH	5.7	30.6	75.0	7.6	1.0
Drainages	No	20	PP	3.9	19.2	75.4	4.7	51.7
Drainages	No	20	SP	2.2	14.8	66.7	4.9	20.7
Drainages	No	20	WF	1.0	8.9	57.4	3.8	10.3
Drainages	No	60	BO	7.3	45.9	42.8	26.2	11.7
Drainages	No	60	DF	17.5	73.2	73.1	19.7	9.9
Drainages	No	60	IC	8.9	40.7	42.9	23.2	9.5
Drainages	No	60	OH	9.1	43.1	49.5	21.8	0.9
Drainages	No	60	PP	14.9	75.7	65.7	26.0	49.0
Drainages	No	60	SP	16.4	67.5	75.8	16.3	19.1
Drainages	No	60	WF	8.7	60.3	47.5	31.6	9.5
Drainages	Yes	10	BO	1.0	10.9	89.8	1.1	32.6
Drainages	Yes	10	DF	0.2	5.1	95.5	0.2	17.2
Drainages	Yes	10	IC	0.1	3.3	96.0	0.1	17.2
Drainages	Yes	10	OH	1.7	15.9	68.7	5.0	4.9
Drainages	Yes	10	PP	0.8	7.3	90.3	0.7	86.2
Drainages	Yes	10	SP	0.3	3.9	89.6	0.4	34.5
Drainages	Yes	10	WF	0.2	2.9	93.2	0.2	17.2
Drainages	Yes	20	BO	4.5	26.5	64.3	9.5	13.0
Drainages	Yes	20	DF	3.3	20.6	64.5	7.3	17.2
Drainages	Yes	20	IC	1.7	10.8	48.4	5.6	17.2
Drainages	Yes	20	OH	5.7	30.6	74.8	7.7	1.0
Drainages	Yes	20	PP	3.9	19.2	74.9	4.8	86.2
Drainages	Yes	20	SP	2.1	14.8	66.0	5.0	34.5
Drainages	Yes	20	WF	1.0	8.9	56.6	3.9	17.2
Drainages	Yes	60	BO	7.2	45.6	42.6	26.2	11.6
Drainages	Yes	60	DF	15.7	77.3	67.0	25.5	16.4
Drainages	Yes	60	IC	8.0	38.3	39.1	23.3	15.7
Drainages	Yes	60	OH	9.0	43.2	49.8	21.7	0.9
Drainages	Yes	60	PP	14.0	74.6	62.0	28.4	81.6
Drainages	Yes	60	SP	15.4	66.7	70.1	19.9	31.8
Drainages	Yes	60	WF	7.4	55.5	42.8	31.8	15.8
Emergency travel routes	No	10	BM	2.5	17.4	81.3	3.3	0.8
Emergency travel routes	No	10	BO	0.8	8.4	87.3	1.1	38.0
Emergency travel routes	No	10	DF	0.2	4.9	97.3	0.1	4.1
Emergency travel routes	No	10	IC	0.0	3.0	96.0	0.1	4.1
Emergency travel routes	No	10	OH	1.4	10.7	68.2	3.4	15.0
Emergency travel routes	No	10	PP	0.8	7.5	95.6	0.3	20.7
Emergency travel routes	No	10	SP	0.2	4.0	89.9	0.4	8.3
Emergency travel routes	No	10	WF	0.0	2.4	92.1	0.2	4.1
Emergency travel routes	No	20	BM	5.2	30.8	75.3	7.6	0.3
Emergency travel routes	No	20	BO	3.7	22.8	51.0	11.2	15.2
Emergency travel routes	No	20	DF	3.0	19.1	62.2	7.2	4.1
Emergency travel routes	No	20	IC	1.6	10.5	47.9	5.5	4.1
Emergency travel routes	No	20	OH	4.3	24.5	59.8	9.9	6.0

Landscape Position	DTFC	Year	Species	DBH	HT	PCR	CBH	TPA
Emergency travel routes	No	20	PP	4.1	21.0	77.5	4.7	20.7
Emergency travel routes	No	20	SP	2.2	15.9	69.1	4.9	8.3
Emergency travel routes	No	20	WF	0.9	8.0	51.3	3.9	4.1
Emergency travel routes	No	60	BM	8.5	40.7	49.5	20.6	0.3
Emergency travel routes	No	60	BO	6.0	40.4	33.8	26.7	13.7
Emergency travel routes	No	60	DF	18.8	60.2	74.2	15.5	4.0
Emergency travel routes	No	60	IC	10.3	43.1	49.2	21.9	3.9
Emergency travel routes	No	60	OH	7.1	33.8	39.5	20.5	5.5
Emergency travel routes	No	60	PP	17.2	81.2	70.9	23.6	20.0
Emergency travel routes	No	60	SP	18.2	69.9	79.7	14.2	7.8
Emergency travel routes	No	60	WF	10.8	62.8	56.1	27.6	3.9
Emergency travel routes	Yes	10	BM	1.5	17.4	81.3	3.3	0.8
Emergency travel routes	Yes	10	BO	0.8	8.4	87.3	1.1	38.0
Emergency travel routes	Yes	10	DF	0.2	4.9	97.3	0.1	6.9
Emergency travel routes	Yes	10	IC	0.0	3.0	96.0	0.1	6.9
Emergency travel routes	Yes	10	OH	1.4	10.7	68.2	3.4	15.0
Emergency travel routes	Yes	10	PP	0.8	7.5	95.6	0.3	34.6
Emergency travel routes	Yes	10	SP	0.2	4.0	89.9	0.4	13.8
Emergency travel routes	Yes	10	WF	0.0	2.4	92.1	0.2	6.9
Emergency travel routes	Yes	20	BM	5.2	30.8	75.3	7.6	0.3
Emergency travel routes	Yes	20	BO	3.7	22.8	51.0	11.2	15.2
Emergency travel routes	Yes	20	DF	3.0	19.1	62.2	7.2	6.9
Emergency travel routes	Yes	20	IC	1.6	10.5	47.5	5.5	6.9
Emergency travel routes	Yes	20	OH	4.3	24.5	59.8	9.9	6.0
Emergency travel routes	Yes	20	PP	4.1	20.9	77.2	4.8	34.6
Emergency travel routes	Yes	20	SP	2.2	15.9	68.9	5.0	13.8
Emergency travel routes	Yes	20	WF	0.9	8.0	51.5	3.9	6.9
Emergency travel routes	Yes	60	BM	8.4	40.6	49.5	20.5	0.3
Emergency travel routes	Yes	60	BO	6.0	40.3	33.8	26.7	13.6
Emergency travel routes	Yes	60	DF	17.7	67.8	73.0	18.3	6.6
Emergency travel routes	Yes	60	IC	9.5	41.3	48.0	21.5	6.4
Emergency travel routes	Yes	60	OH	7.0	33.7	39.5	20.4	5.4
Emergency travel routes	Yes	60	PP	16.3	80.2	70.0	24.1	33.1
Emergency travel routes	Yes	60	SP	17.2	69.4	78.3	15.1	12.9
Emergency travel routes	Yes	60	WF	9.3	58.5	51.4	28.4	6.4
Deer hiding	No	10	BM	2.6	21.0	75.6	5.1	0.1
Deer hiding	No	10	BO	1.3	10.8	85.2	1.6	43.2
Deer hiding	No	10	DF	0.2	4.7	96.3	0.2	5.8
Deer hiding	No	10	IC	0.1	3.4	97.3	0.1	5.8
Deer hiding	No	10	OH	1.7	11.7	62.1	4.4	8.0
Deer hiding	No	10	PP	0.7	6.9	92.5	0.5	28.9
Deer hiding	No	10	SP	0.1	3.6	88.8	0.4	11.6
Deer hiding	No	10	WF	0.0	2.5	92.3	0.2	5.8
Deer hiding	No	20	BM	4.7	30.9	64.8	10.9	0.0
Deer hiding	No	20	BO	4.5	26.5	63.8	9.6	17.3
Deer hiding	No	20	DF	2.7	17.9	61.1	7.0	5.8
Deer hiding	No	20	IC	1.5	10.6	49.5	5.4	5.8
Deer hiding	No	20	OH	5.1	27.6	74.5	7.0	1.6
Deer hiding	No	20	PP	3.8	19.7	72.7	5.4	28.9
Deer hiding	No	20	SP	2.1	14.2	68.4	4.5	11.6
Deer hiding	No	20	WF	0.7	7.3	57.8	3.1	5.8
Deer hiding	No	60	BM	7.5	39.7	43.0	22.6	0.0
Deer hiding	No	60	BO	7.3	45.2	41.9	26.3	15.7
Deer hiding	No	60	DF	17.2	64.2	72.1	17.9	5.5
Deer hiding	No	60	IC	9.4	41.7	47.8	21.8	5.3
Deer hiding	No	60	OH	8.4	39.2	49.0	20.0	1.5
Deer hiding	No	60	PP	15.9	77.9	69.7	23.6	27.5

Landscape Position	DTFC	Year	Species	DBH	HT	PCR	CBH	TPA
Deer hiding	No	60	SP	17.7	68.3	80.6	13.3	10.8
Deer hiding	No	60	WF	9.6	59.3	52.9	27.9	5.4
Deer hiding	Yes	10	BM	2.6	21.0	75.6	5.1	0.1
Deer hiding	Yes	10	BO	1.3	10.8	85.2	1.6	43.2
Deer hiding	Yes	10	DF	0.2	4.7	96.3	0.2	9.6
Deer hiding	Yes	10	IC	0.1	3.4	97.3	0.1	9.6
Deer hiding	Yes	10	OH	1.7	11.7	62.1	4.4	8.0
Deer hiding	Yes	10	PP	0.7	6.9	92.5	0.5	48.2
Deer hiding	Yes	10	SP	0.1	3.6	88.8	0.4	19.3
Deer hiding	Yes	10	WF	0.0	2.5	92.3	0.2	9.6
Deer hiding	Yes	20	BM	4.7	30.9	64.5	11.0	0.0
Deer hiding	Yes	20	BO	4.5	26.5	63.8	9.6	17.3
Deer hiding	Yes	20	DF	2.7	17.9	61.1	7.0	9.6
Deer hiding	Yes	20	IC	1.5	10.6	49.7	5.3	9.6
Deer hiding	Yes	20	OH	5.1	27.6	74.5	7.0	1.6
Deer hiding	Yes	20	PP	3.8	19.7	72.5	5.4	48.2
Deer hiding	Yes	20	SP	2.1	14.2	67.7	4.6	19.3
Deer hiding	Yes	20	WF	0.7	7.3	57.6	3.1	9.6
Deer hiding	Yes	60	BM	7.4	39.5	43.0	22.5	0.0
Deer hiding	Yes	60	BO	7.2	45.0	41.9	26.2	15.6
Deer hiding	Yes	60	DF	15.8	69.3	70.3	20.6	9.1
Deer hiding	Yes	60	IC	8.8	40.3	46.6	21.5	8.9
Deer hiding	Yes	60	OH	8.4	39.2	49.0	20.0	1.5
Deer hiding	Yes	60	PP	15.0	76.7	68.7	24.0	45.8
Deer hiding	Yes	60	SP	16.6	67.2	78.5	14.4	17.9
Deer hiding	Yes	60	WF	8.2	55.3	48.8	28.3	8.9
Deer thermal	No	10	BM	2.6	21.0	75.6	5.1	0.1
Deer thermal	No	10	BO	1.3	10.8	85.2	1.6	43.2
Deer thermal	No	10	DF	0.2	4.7	96.3	0.2	8.3
Deer thermal	No	10	IC	0.1	3.4	97.3	0.1	8.3
Deer thermal	No	10	OH	1.7	11.7	62.1	4.4	8.0
Deer thermal	No	10	PP	0.7	6.9	92.5	0.5	41.4
Deer thermal	No	10	SP	0.1	3.6	88.8	0.4	16.5
Deer thermal	No	10	WF	0.0	2.5	92.3	0.2	8.3
Deer thermal	No	20	BM	4.7	30.9	64.5	11.0	0.0
Deer thermal	No	20	BO	4.5	26.5	63.8	9.6	17.3
Deer thermal	No	20	DF	2.7	17.9	61.1	7.0	8.3
Deer thermal	No	20	IC	1.5	10.6	49.8	5.3	8.3
Deer thermal	No	20	OH	5.1	27.6	74.5	7.0	1.6
Deer thermal	No	20	PP	3.8	19.7	72.6	5.4	41.4
Deer thermal	No	20	SP	2.1	14.2	67.7	4.6	16.5
Deer thermal	No	20	WF	0.7	7.3	57.7	3.1	8.3
Deer thermal	No	60	BM	7.5	39.5	43.0	22.5	0.0
Deer thermal	No	60	BO	7.2	45.1	41.9	26.2	15.6
Deer thermal	No	60	DF	16.2	67.8	70.8	19.8	7.8
Deer thermal	No	60	IC	9.1	40.9	47.3	21.6	7.6
Deer thermal	No	60	OH	8.4	39.3	49.0	20.0	1.5
Deer thermal	No	60	PP	15.2	77.1	69.1	23.8	39.3
Deer thermal	No	60	SP	16.9	67.4	79.3	13.9	15.4
Deer thermal	No	60	WF	8.7	56.9	50.4	28.2	7.6
Deer thermal	Yes	10	BM	2.6	21.0	75.6	5.1	0.1
Deer thermal	Yes	10	BO	1.3	11.1	86.0	1.6	43.2
Deer thermal	Yes	10	DF	0.1	4.5	96.3	0.2	13.8
Deer thermal	Yes	10	IC	0.1	3.3	97.4	0.1	13.8
Deer thermal	Yes	10	OH	1.7	11.5	61.8	4.4	8.0
Deer thermal	Yes	10	PP	0.7	7.0	92.5	0.5	68.9
Deer thermal	Yes	10	SP	0.1	3.5	88.9	0.4	27.6

Landscape Position	DTFC	Year	Species	DBH	HT	PCR	CBH	TPA
Deer thermal	Yes	10	WF	0.0	2.5	92.4	0.2	13.8
Deer thermal	Yes	20	BM	4.7	30.9	63.8	11.2	0.0
Deer thermal	Yes	20	BO	4.6	26.8	64.5	9.5	17.3
Deer thermal	Yes	20	DF	2.6	17.6	60.5	6.9	13.8
Deer thermal	Yes	20	IC	1.5	10.6	49.9	5.3	13.8
Deer thermal	Yes	20	OH	5.1	27.4	74.3	7.1	1.6
Deer thermal	Yes	20	PP	3.8	19.8	72.4	5.5	68.9
Deer thermal	Yes	20	SP	2.1	14.0	67.2	4.6	27.6
Deer thermal	Yes	20	WF	0.7	7.4	57.3	3.1	13.8
Deer thermal	Yes	60	BM	7.5	39.7	42.3	22.9	0.0
Deer thermal	Yes	60	BO	7.4	46.1	42.6	26.5	15.6
Deer thermal	Yes	60	DF	14.5	71.3	66.9	23.6	12.9
Deer thermal	Yes	60	IC	8.3	38.9	44.5	21.6	12.6
Deer thermal	Yes	60	OH	8.2	38.6	48.8	19.8	1.5
Deer thermal	Yes	60	PP	14.4	76.1	66.4	25.6	65.5
Deer thermal	Yes	60	SP	15.9	66.6	75.0	16.6	25.6
Deer thermal	Yes	60	WF	7.4	52.3	44.4	29.1	12.7
Mid-slopes	No	10	BM	2.6	21.0	75.6	5.1	0.1
Mid-slopes	No	10	BO	1.3	10.8	85.2	1.6	43.2
Mid-slopes	No	10	DF	0.2	4.7	96.3	0.2	10.3
Mid-slopes	No	10	IC	0.1	3.4	97.3	0.1	10.3
Mid-slopes	No	10	OH	1.7	11.7	62.1	4.4	8.0
Mid-slopes	No	10	PP	0.7	6.9	92.5	0.5	51.7
Mid-slopes	No	10	SP	0.1	3.6	88.8	0.4	20.7
Mid-slopes	No	10	WF	0.0	2.5	92.3	0.2	10.3
Mid-slopes	No	20	BM	4.7	30.9	64.5	11.0	0.0
Mid-slopes	No	20	BO	4.5	26.5	63.8	9.6	17.3
Mid-slopes	No	20	DF	2.7	17.9	61.1	7.0	10.3
Mid-slopes	No	20	IC	1.5	10.6	49.7	5.3	10.3
Mid-slopes	No	20	OH	5.1	27.6	74.5	7.0	1.6
Mid-slopes	No	20	PP	3.8	19.7	72.5	5.4	51.7
Mid-slopes	No	20	SP	2.1	14.2	67.7	4.6	20.7
Mid-slopes	No	20	WF	0.7	7.3	57.6	3.1	10.3
Mid-slopes	No	60	BM	7.5	39.6	43.0	22.6	0.0
Mid-slopes	No	60	BO	7.3	45.3	41.9	26.3	15.6
Mid-slopes	No	60	DF	15.5	69.8	69.9	21.0	9.8
Mid-slopes	No	60	IC	8.6	39.7	45.7	21.6	9.5
Mid-slopes	No	60	OH	8.3	39.2	49.0	20.0	1.5
Mid-slopes	No	60	PP	14.7	76.2	68.3	24.1	49.2
Mid-slopes	No	60	SP	16.5	67.2	77.9	14.8	19.2
Mid-slopes	No	60	WF	8.0	54.5	47.7	28.5	9.5
Mid-slopes	Yes	10	BM	2.6	21.0	75.6	5.1	0.1
Mid-slopes	Yes	10	BO	1.3	10.8	85.2	1.6	43.2
Mid-slopes	Yes	10	DF	0.2	4.7	96.3	0.2	17.2
Mid-slopes	Yes	10	IC	0.1	3.4	97.0	0.1	17.2
Mid-slopes	Yes	10	OH	1.7	11.7	62.1	4.4	8.0
Mid-slopes	Yes	10	PP	0.7	6.9	92.4	0.5	86.2
Mid-slopes	Yes	10	SP	0.1	3.6	88.8	0.4	34.5
Mid-slopes	Yes	10	WF	0.0	2.5	92.3	0.2	17.2
Mid-slopes	Yes	20	BM	4.7	30.9	64.3	11.0	0.0
Mid-slopes	Yes	20	BO	4.5	26.4	63.8	9.6	17.3
Mid-slopes	Yes	20	DF	2.7	17.8	60.6	7.0	17.2
Mid-slopes	Yes	20	IC	1.5	10.6	49.2	5.4	17.2
Mid-slopes	Yes	20	OH	5.1	27.6	74.3	7.1	1.6
Mid-slopes	Yes	20	PP	3.8	19.7	72.3	5.5	86.2
Mid-slopes	Yes	20	SP	2.1	14.1	69.8	4.3	34.5
Mid-slopes	Yes	20	WF	0.7	7.3	56.5	3.2	17.2

Landscape Position	DTFC	Year	Species	DBH	HT	PCR	CBH	TPA
Mid-slopes	Yes	60	BM	7.5	39.7	42.8	22.7	0.0
Mid-slopes	Yes	60	BO	7.2	45.3	41.9	26.3	15.5
Mid-slopes	Yes	60	DF	13.6	71.7	63.9	25.8	16.1
Mid-slopes	Yes	60	IC	7.9	37.7	41.3	22.1	15.8
Mid-slopes	Yes	60	OH	8.2	38.7	48.8	19.9	1.5
Mid-slopes	Yes	60	PP	14.1	75.9	64.5	26.9	81.9
Mid-slopes	Yes	60	SP	15.4	66.8	71.8	18.8	32.0
Mid-slopes	Yes	60	WF	6.9	50.2	41.9	29.1	15.8
Fuelbreaks and Primary Ridges within SFMA	No	10	BO	0.6	7.8	87.5	1.0	63.4
Fuelbreaks and Primary Ridges within SFMA	No	10	DF	0.2	4.3	92.0	0.3	4.1
Fuelbreaks and Primary Ridges within SFMA	No	10	IC	0.0	2.5	95.4	0.1	4.1
Fuelbreaks and Primary Ridges within SFMA	No	10	PP	0.4	5.4	87.3	0.7	20.5
Fuelbreaks and Primary Ridges within SFMA	No	10	SP	0.1	3.3	90.3	0.3	8.2
Fuelbreaks and Primary Ridges within SFMA	No	10	WF	0.0	1.8	89.9	0.2	4.1
Fuelbreaks and Primary Ridges within SFMA	No	20	BO	3.3	20.8	74.0	5.4	25.3
Fuelbreaks and Primary Ridges within SFMA	No	20	DF	2.6	16.8	70.5	5.0	4.1
Fuelbreaks and Primary Ridges within SFMA	No	20	IC	1.0	7.6	57.8	3.2	4.1
Fuelbreaks and Primary Ridges within SFMA	No	20	PP	2.5	14.3	71.4	4.1	20.5
Fuelbreaks and Primary Ridges within SFMA	No	20	SP	1.8	11.6	73.7	3.0	8.2
Fuelbreaks and Primary Ridges within SFMA	No	20	WF	0.6	6.6	57.8	2.8	3.3
Fuelbreaks and Primary Ridges within SFMA	No	60	BO	6.0	36.2	48.8	18.5	23.4
Fuelbreaks and Primary Ridges within SFMA	No	60	DF	18.0	42.3	77.3	9.6	4.0
Fuelbreaks and Primary Ridges within SFMA	No	60	IC	8.7	34.7	46.0	18.7	3.8
Fuelbreaks and Primary Ridges within SFMA	No	60	PP	13.2	57.5	62.4	21.6	19.4
Fuelbreaks and Primary Ridges within SFMA	No	60	SP	15.6	54.9	77.8	12.2	7.7
Fuelbreaks and Primary Ridges within SFMA	No	60	WF	10.3	54.0	56.6	23.4	3.1
Fuelbreaks and Primary Ridges within SFMA	Yes	10	BO	0.6	7.8	87.5	1.0	63.4
Fuelbreaks and Primary Ridges within SFMA	Yes	10	DF	0.2	4.3	92.0	0.3	6.8
Fuelbreaks and Primary Ridges within SFMA	Yes	10	IC	0.0	2.5	95.4	0.1	6.8
Fuelbreaks and Primary Ridges within SFMA	Yes	10	PP	0.4	5.4	87.3	0.7	34.1
Fuelbreaks and Primary Ridges within SFMA	Yes	10	SP	0.1	3.3	90.3	0.3	13.6
Fuelbreaks and Primary Ridges within SFMA	Yes	10	WF	0.0	1.8	89.9	0.2	6.8
Fuelbreaks and Primary Ridges within SFMA	Yes	20	BO	3.3	20.8	74.0	5.4	25.3
Fuelbreaks and Primary Ridges within SFMA	Yes	20	DF	2.6	16.8	70.5	5.0	6.8
Fuelbreaks and Primary Ridges within SFMA	Yes	20	IC	1.0	7.6	58.7	3.1	6.8
Fuelbreaks and Primary Ridges within SFMA	Yes	20	PP	2.5	14.3	71.3	4.1	34.1
Fuelbreaks and Primary Ridges within SFMA	Yes	20	SP	1.7	11.5	73.6	3.0	13.6
Fuelbreaks and Primary Ridges within SFMA	Yes	20	WF	0.6	6.6	57.5	2.8	5.5
Fuelbreaks and Primary Ridges within SFMA	Yes	60	BO	6.0	36.1	48.8	18.5	23.2
Fuelbreaks and Primary Ridges within SFMA	Yes	60	DF	17.0	48.8	75.7	11.9	6.7
Fuelbreaks and Primary Ridges within SFMA	Yes	60	IC	8.1	33.3	45.8	18.0	6.3
Fuelbreaks and Primary Ridges within SFMA	Yes	60	PP	12.7	57.5	62.3	21.7	32.2
Fuelbreaks and Primary Ridges within SFMA	Yes	60	SP	15.3	55.7	77.5	12.5	12.8
Fuelbreaks and Primary Ridges within SFMA	Yes	60	WF	9.0	50.7	53.1	23.8	5.1
Primary Ridges outside SFMA	No	10	BO	0.6	7.8	87.5	1.0	63.4
Primary Ridges outside SFMA	No	10	DF	0.2	4.3	92.0	0.3	5.3
Primary Ridges outside SFMA	No	10	IC	0.0	2.5	95.4	0.1	5.3
Primary Ridges outside SFMA	No	10	PP	0.4	5.4	87.3	0.7	26.5
Primary Ridges outside SFMA	No	10	SP	0.1	3.3	90.3	0.3	10.6
Primary Ridges outside SFMA	No	10	WF	0.0	1.7	90.3	0.2	5.4
Primary Ridges outside SFMA	No	20	BO	3.3	20.8	74.0	5.4	25.3
Primary Ridges outside SFMA	No	20	DF	2.6	16.8	70.5	5.0	5.3
Primary Ridges outside SFMA	No	20	IC	1.0	7.6	57.8	3.2	5.3
Primary Ridges outside SFMA	No	20	PP	2.5	14.3	71.3	4.1	26.5
Primary Ridges outside SFMA	No	20	SP	1.8	11.5	73.6	3.0	10.6
Primary Ridges outside SFMA	No	20	WF	0.6	6.6	57.8	2.8	4.3
Primary Ridges outside SFMA	No	60	BO	6.1	36.5	48.8	18.7	23.3

Landscape Position	DTFC	Year	Species	DBH	HT	PCR	CBH	TPA
Primary Ridges outside SFMA	No	60	DF	17.4	45.2	76.4	10.7	5.2
Primary Ridges outside SFMA	No	60	IC	8.4	34.0	45.6	18.5	4.9
Primary Ridges outside SFMA	No	60	PP	13.0	57.9	62.6	21.7	25.0
Primary Ridges outside SFMA	No	60	SP	15.2	54.8	77.4	12.4	10.0
Primary Ridges outside SFMA	No	60	WF	9.8	53.1	55.4	23.7	4.0
Primary Ridges outside SFMA	Yes	10	BO	0.6	7.8	87.5	1.0	63.4
Primary Ridges outside SFMA	Yes	10	DF	0.2	4.3	92.0	0.3	8.8
Primary Ridges outside SFMA	Yes	10	IC	0.0	2.5	95.4	0.1	8.8
Primary Ridges outside SFMA	Yes	10	PP	0.4	5.4	87.3	0.7	44.1
Primary Ridges outside SFMA	Yes	10	SP	0.1	3.3	90.3	0.3	17.7
Primary Ridges outside SFMA	Yes	10	WF	0.0	1.8	89.9	0.2	8.8
Primary Ridges outside SFMA	Yes	20	BO	3.3	20.8	74.0	5.4	25.3
Primary Ridges outside SFMA	Yes	20	DF	2.6	16.8	70.5	5.0	8.8
Primary Ridges outside SFMA	Yes	20	IC	1.0	7.6	57.8	3.2	8.8
Primary Ridges outside SFMA	Yes	20	PP	2.5	14.2	71.3	4.1	44.1
Primary Ridges outside SFMA	Yes	20	SP	1.7	11.5	73.5	3.0	17.7
Primary Ridges outside SFMA	Yes	20	WF	0.6	6.6	57.8	2.8	7.1
Primary Ridges outside SFMA	Yes	60	BO	6.0	36.4	48.8	18.6	23.2
Primary Ridges outside SFMA	Yes	60	DF	16.6	52.0	75.4	12.8	8.6
Primary Ridges outside SFMA	Yes	60	IC	7.9	32.7	45.7	17.7	8.2
Primary Ridges outside SFMA	Yes	60	PP	12.2	56.7	61.5	21.8	41.6
Primary Ridges outside SFMA	Yes	60	SP	14.8	55.2	76.7	12.9	16.6
Primary Ridges outside SFMA	Yes	60	WF	8.5	49.3	51.6	23.9	6.6

Table A.02-4 Alternative 4: Tree Characteristics by Landscape Position for Individual Tree Species

Landscape Position	Year	Species	DBH	HT	PCR	CBH	TPA
Mid-slope founder stands	10	DF	0.1	5.2	96.0	0.2	13.1
Mid-slope founder stands	10	IC	0.0	4.1	97.1	0.1	13.1
Mid-slope founder stands	10	PP	1.1	9.1	94.2	0.5	65.6
Mid-slope founder stands	10	SP	0.1	4.6	91.6	0.4	26.3
Mid-slope founder stands	10	WF	0.0	3.2	90.4	0.3	13.1
Mid-slope founder stands	20	DF	3.5	21.3	65.7	7.3	13.1
Mid-slope founder stands	20	IC	2.5	15.0	53.0	7.1	13.1
Mid-slope founder stands	20	PP	5.7	27.2	83.1	4.6	65.6
Mid-slope founder stands	20	SP	2.8	19.2	69.5	5.8	26.3
Mid-slope founder stands	20	WF	1.6	11.4	51.6	5.5	13.1
Mid-slope founder stands	60	DF	15.3	78.3	63.4	28.6	12.2
Mid-slope founder stands	60	IC	9.8	45.5	41.2	26.7	11.9
Mid-slope founder stands	60	PP	16.1	83.4	65.7	28.6	62.2
Mid-slope founder stands	60	SP	16.8	73.5	72.2	20.4	24.1
Mid-slope founder stands	60	WF	9.0	64.3	42.6	36.9	12.0

[1] Remaining slope positions, tree species and tree characteristics are the same as shown in Table A-7

Table A.02-5 Alternative 5: Tree Characteristics by Landscape Position for Individual Tree Species

Landscape Position	Year	Species	DBH	HT	PCR	CBH	TPA
Drainages	10	BO	1.5	12.3	90.7	1.1	32.6
Drainages	10	DF	0.2	5.4	96.6	0.2	11.8
Drainages	10	IC	0.0	4.0	96.9	0.1	3.9
Drainages	10	OH	2.0	14.5	86.8	1.9	0.9
Drainages	10	OH	1.1	9.4	40.7	5.6	4.0
Drainages	10	PP	1.4	10.4	96.7	0.3	128.9
Drainages	10	SP	0.1	4.7	87.9	0.6	35.6
Drainages	10	WF	0.0	3.6	97.3	0.1	30.3
Drainages	20	BO	4.1	24.7	46.3	13.3	13.0
Drainages	20	DF	3.7	22.9	61.8	8.8	12.4

Landscape Position	Year	Species	DBH	HT	PCR	CBH	TPA
Drainages	20	IC	2.3	13.8	44.3	7.7	4.7
Drainages	20	OH	4.4	24.5	50.8	12.1	2.0
Drainages	20	PP	6.3	30.6	83.1	5.2	128.9
Drainages	20	SP	3.2	22.5	67.9	7.2	35.6
Drainages	20	WF	1.7	12.7	41.1	7.4	30.3
Drainages	60	BO	6.2	41.5	30.8	28.7	8.6
Drainages	60	DF	11.7	77.9	58.4	32.4	9.8
Drainages	60	IC	7.6	38.4	36.7	24.4	3.7
Drainages	60	OH	6.7	33.1	33.5	22.0	1.3
Drainages	60	PP	15.3	84.5	63.5	30.9	112.0
Drainages	60	SP	15.3	76.1	67.0	25.1	29.3
Drainages	60	WF	7.3	58.0	38.0	35.9	24.1
Emergency Travel Routes	10	BM	2.4	17.1	82.2	3.1	0.8
Emergency Travel Routes	10	BO	1.2	11.1	89.6	1.2	38.0
Emergency Travel Routes	10	DF	0.2	5.1	94.4	0.3	11.8
Emergency Travel Routes	10	IC	0.0	3.9	96.9	0.1	10.9
Emergency Travel Routes	10	OH	0.1	5.5	44.9	3.0	9.0
Emergency Travel Routes	10	PP	1.4	10.7	95.6	0.5	124.9
Emergency Travel Routes	10	SP	0.1	4.8	92.7	0.3	51.7
Emergency Travel Routes	10	WF	0.0	3.5	94.9	0.2	30.3
Emergency Travel Routes	20	BM	5.3	31.3	58.5	13.0	0.3
Emergency Travel Routes	20	BO	4.5	27.0	51.0	13.2	15.2
Emergency Travel Routes	20	DF	3.2	20.2	57.8	8.5	12.4
Emergency Travel Routes	20	IC	2.7	16.6	54.3	7.6	11.3
Emergency Travel Routes	20	OH	2.4	15.6	27.8	11.3	3.6
Emergency Travel Routes	20	PP	6.3	30.1	82.7	5.2	124.9
Emergency Travel Routes	20	SP	2.6	17.8	64.2	6.4	51.7
Emergency Travel Routes	20	WF	1.7	12.0	40.2	7.2	30.3
Emergency Travel Routes	60	BM	7.9	38.9	38.5	23.9	0.2
Emergency Travel Routes	60	BO	6.8	44.0	33.3	29.4	9.5
Emergency Travel Routes	60	DF	11.1	74.4	56.4	32.4	9.5
Emergency Travel Routes	60	IC	8.5	42.7	41.4	25.0	8.5
Emergency Travel Routes	60	OH	4.1	23.1	18.3	18.9	2.2
Emergency Travel Routes	60	PP	15.5	85.2	63.7	30.9	107.8
Emergency Travel Routes	60	SP	14.1	68.4	64.3	24.4	40.1
Emergency Travel Routes	60	WF	6.9	55.0	36.5	34.9	23.2
Mid-slopes	10	BM	2.5	20.3	75.5	5.0	0.1
Mid-slopes	10	BO	1.3	10.7	84.7	1.6	43.2
Mid-slopes	10	DF	0.3	5.5	97.1	0.2	19.0
Mid-slopes	10	IC	0.0	4.2	96.9	0.1	11.9
Mid-slopes	10	OH	1.5	11.8	78.3	2.6	0.0
Mid-slopes	10	OH	0.6	7.4	44.2	4.1	8.0
Mid-slopes	10	PP	1.1	9.8	93.0	0.7	120.1
Mid-slopes	10	SP	0.2	5.0	93.6	0.3	54.9
Mid-slopes	10	WF	0.0	3.7	96.2	0.1	30.3
Mid-slopes	20	BM	5.2	33.5	61.0	13.1	0.0
Mid-slopes	20	BO	4.1	24.0	45.5	13.1	17.3
Mid-slopes	20	DF	4.1	25.6	65.8	8.8	19.3
Mid-slopes	20	IC	2.4	14.4	43.6	8.1	12.2
Mid-slopes	20	OH	3.3	19.6	36.6	12.4	3.2
Mid-slopes	20	PP	5.6	27.3	81.7	5.0	120.1
Mid-slopes	20	SP	2.7	21.1	62.5	7.9	54.9
Mid-slopes	20	WF	1.7	11.7	49.5	5.9	30.3
Mid-slopes	60	BM	7.8	41.8	40.3	25.0	0.0
Mid-slopes	60	BO	6.2	40.5	30.3	28.2	10.7
Mid-slopes	60	DF	14.0	85.2	62.1	32.3	15.9
Mid-slopes	60	IC	8.0	40.0	38.4	24.6	9.2
Mid-slopes	60	OH	5.3	27.4	24.1	20.8	2.0
Mid-slopes	60	PP	14.7	81.8	61.3	31.7	101.9
Mid-slopes	60	SP	14.7	72.9	64.8	25.7	44.0

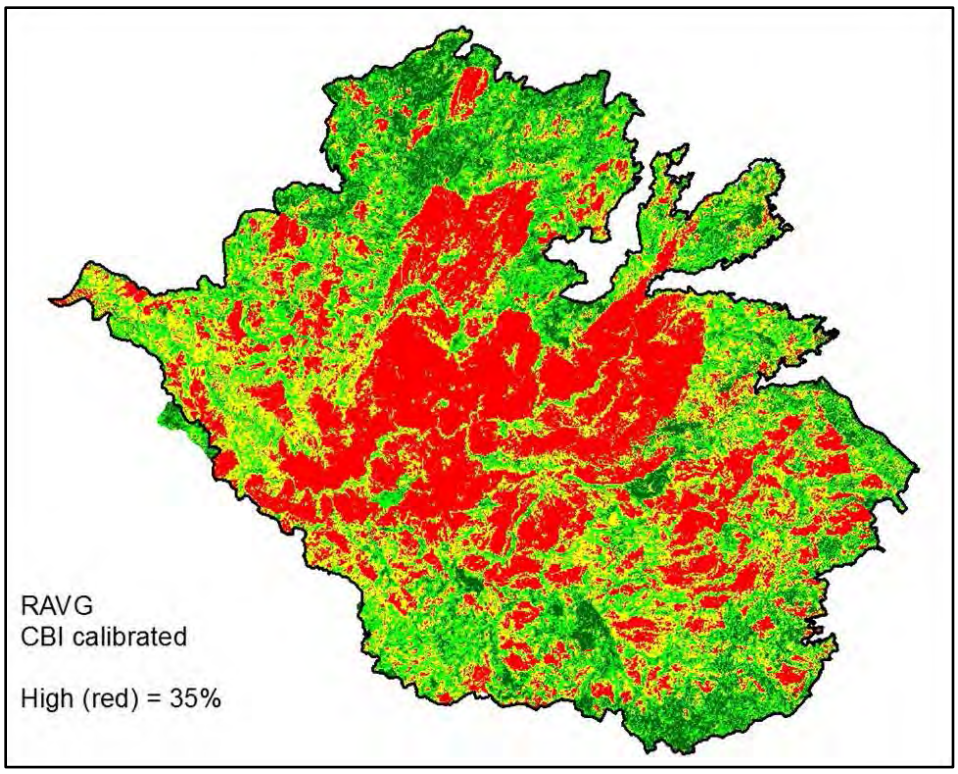
Landscape Position	Year	Species	DBH	HT	PCR	CBH	TPA
Mid-slopes	60	WF	7.3	58.0	40.8	34.3	23.2
Fuelbreaks and Primary Ridges	10	BO	0.6	7.7	84.2	1.2	63.4
Fuelbreaks and Primary Ridges	10	DF	0.3	5.5	94.1	0.3	11.8
Fuelbreaks and Primary Ridges	10	IC	0.0	3.8	98.3	0.1	5.0
Fuelbreaks and Primary Ridges	10	PP	1.3	9.8	95.1	0.5	139.1
Fuelbreaks and Primary Ridges	10	SP	0.1	4.6	94.8	0.2	44.6
Fuelbreaks and Primary Ridges	10	WF	0.0	2.9	94.4	0.2	30.3
Fuelbreaks and Primary Ridges	20	BO	3.1	20.0	39.5	12.1	25.3
Fuelbreaks and Primary Ridges	20	DF	3.7	22.6	65.1	7.9	12.4
Fuelbreaks and Primary Ridges	20	IC	2.5	16.2	51.9	7.8	5.4
Fuelbreaks and Primary Ridges	20	PP	6.1	29.2	81.6	5.4	139.1
Fuelbreaks and Primary Ridges	20	SP	3.0	22.3	68.7	7.0	44.6
Fuelbreaks and Primary Ridges	20	WF	1.2	9.5	48.6	4.9	30.3
Fuelbreaks and Primary Ridges	60	BO	5.0	30.6	26.0	22.6	17.9
Fuelbreaks and Primary Ridges	60	DF	11.8	67.3	58.2	28.1	10.4
Fuelbreaks and Primary Ridges	60	IC	7.5	36.2	39.4	21.9	4.4
Fuelbreaks and Primary Ridges	60	PP	13.8	73.4	61.9	28.0	123.7
Fuelbreaks and Primary Ridges	60	SP	13.6	65.5	65.2	22.8	37.6
Fuelbreaks and Primary Ridges	60	WF	6.0	42.5	39.0	26.0	25.0

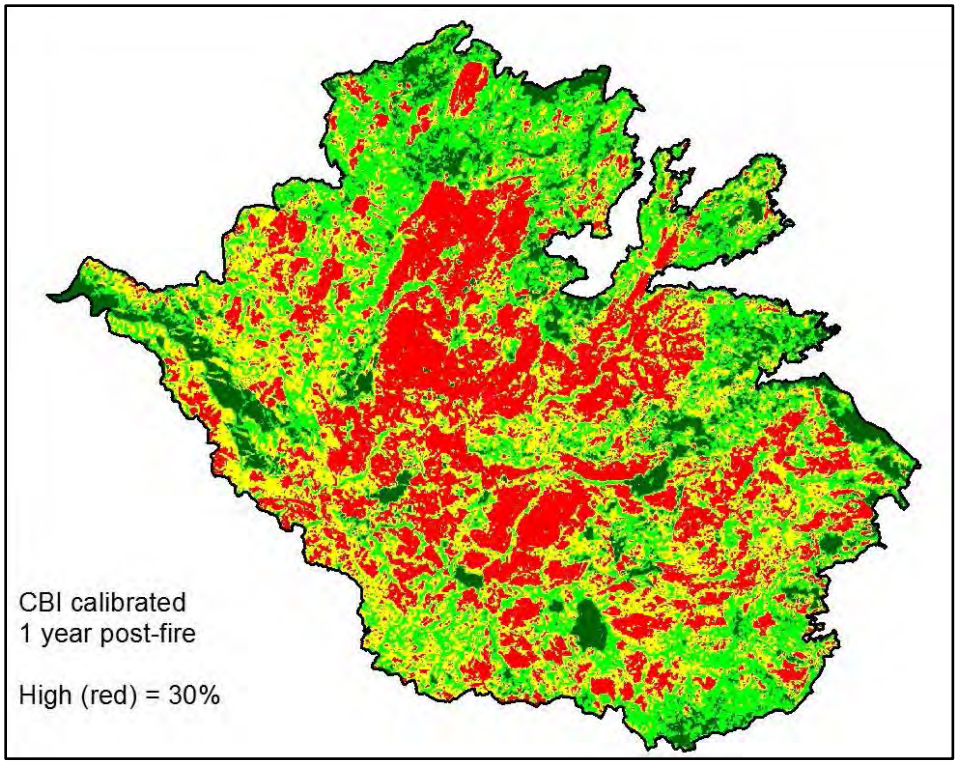
B. Fire Severity Data (RAVG vs. MTBS)

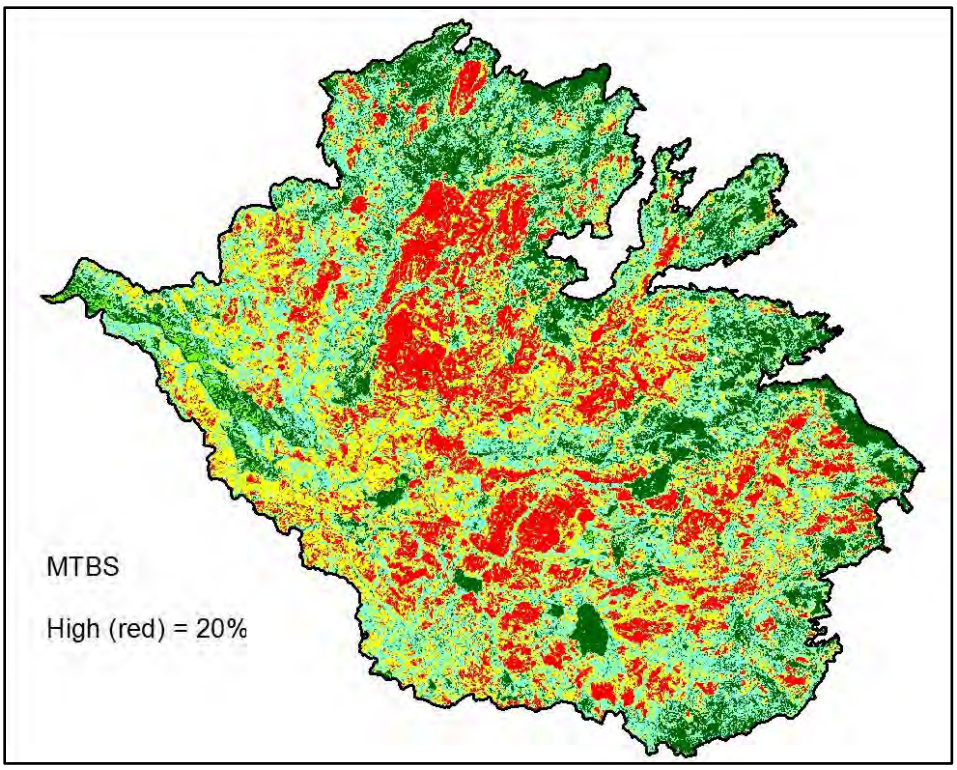
Estimates of high-severity fire in the Rim Fire were derived using data produced by the Rapid Assessment of Vegetation Conditions after Wildfire (RAVG) program. RAVG data were used because the Regional Forester issued a memo on February 5, 2015 (USDA 2015g) directing all Pacific Southwest Region Forests to use RAVG data instead of data produced by the Monitoring Trends in Burn Severity (MTBS) program. RAVG data are produced a few weeks after the fire and compares the change in vegetation before the fire and immediately after the fire. The RAVG process produces several different classifications of vegetative burn severity: Basal Area Loss (four and seven categories), Composite Burn Index (four categories), and Canopy Mortality (five categories). MTBS data compares vegetation before the fire with vegetation some months afterward and uses a five category classification scheme. Since MTBS uses a later post fire image, it may reflect some regrowth of vegetation after the fire. Re-sprouting of grasses and shrubs under a burned canopy could lessen the apparent burn severity in the MTBS image as compared to the RAVG image. Also, since the MTBS definitions of "High, Moderate, and Low Severity" are not the same as the classifications in the RAVG products, direct comparisons of the two are difficult to make.

There are fundamental differences in the mapping methodologies employed by the RAVG and MTBS programs. First, the two programs use two difference satellite indices for producing maps. MTBS uses the differenced normalized burn ratio (dNBR) while RAVG uses a relativized dNBR (RdNBR). The most significant characteristic of dNBR data is that they are correlated to the amount of pre-fire chlorophyll where as RdNBR is not (Miller and Thode 2007). Thus, dNBR values do not always represent the same percent mortality detected by changes in chlorophyll because of differences in species and amounts of vegetative cover within and between fires. Each dNBR image must therefore be individually interpreted. The fact that the dNBR image is correlated to the amount of live pre-fire chlorophyll can therefore be a disadvantage when deriving categorical maps (e.g., low, medium moderate, and high categories) that represent consistent ranges of percent mortality (Safford et al. 2008). Second, although general image interpretation rules are followed, it is common practice for an MTBS image analyst to produce a categorical fire severity map by interpreting an image on a computer screen without any direct knowledge of ground conditions (MTBS methods at http://www.mtbs.gov/methods.html) (Eidenshink et al. 2007). As a result, dNBR derived categorical maps are at risk to subjective evaluation and human error. In contrast, the RdNBR can be calibrated using field data from one fire and applied to another fire (Miller and Thode 2007). Maps produced by RAVG are based upon calibrations from fires in California (Miller and Quayle 2015). Maps derived from RdNBR using one-year post-fire images produce statistically identical classifications in areas with at least 10% pre-fire conifer tree cover (Miller and Quayle 2015). In RdNBR maps derived from immediate and one-year post-fire images calibrated to basal area (BA) change of trees, greater than 85% of field validation plots greater than or equal to 30 meters (one pixel width) inside greater than or equal to 90% BA change polygon boundaries do not have any live trees (Miller and Quayle 2015). Whittier and Gray (2016) also found that MTBS underestimated the proportion of high-severity fire by about 40% and observed an increase in severity 3 to 4 years post-fire due to delayed tree mortality. The inherent limitations of the MTBS data have been recognized and a framework has been suggested on how to overcome them so that the products will support research and planning efforts (Kolden et al. 2015).

The following figures illustrate that the MTBS data for the Rim Fire under predict the amount of high severity by almost 50%.


RAVG
CBI calibrated
High (red) = 35%


CBI calibrated
1 year post-fire
High (red) = 30%


MTBS
High (red) = 20%