The University of San Francisco

USF Scholarship: a digital repository @ Gleeson Library | Geschke Center

Master's Projects and Capstones | Theses, Dissertations, Capstones and Projects

Spring 5-14-2020

A Comparison of Fuel Reduction Methods for Wildfire Risk Management and Climate Change Resiliency in Mixed Conifer Forests in the Sierra Nevada

Heather Navle
henavle@dons.usfca.edu

Follow this and additional works at: https://repository.usfca.edu/capstone

Part of the Biodiversity Commons, Environmental Education Commons, Environmental Monitoring Commons, Forest Management Commons, Hydrology Commons, Natural Resources and Conservation Commons, Natural Resources Management and Policy Commons, Other Forestry and Forest Sciences Commons, Soil Science Commons, Sustainability Commons, and the Water Resource Management Commons

Recommended Citation

Navle, Heather, "A Comparison of Fuel Reduction Methods for Wildfire Risk Management and Climate Change Resiliency in Mixed Conifer Forests in the Sierra Nevada" (2020). *Master's Projects and Capstones*. 1033.
https://repository.usfca.edu/capstone/1033

This Project/Capstone is brought to you for free and open access by the Theses, Dissertations, Capstones and Projects at USF Scholarship: a digital repository @ Gleeson Library | Geschke Center. It has been accepted for inclusion in Master's Projects and Capstones by an authorized administrator of USF Scholarship: a digital repository @ Gleeson Library | Geschke Center. For more information, please contact repository@usfca.edu.

This Master's Project

A Comparison of Fuel Reduction Methods for

Wildfire Risk Management and Climate Change Resiliency

in Mixed Conifer Forests in the Sierra Nevada

by

Heather Navle

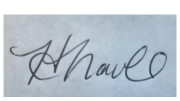
is submitted in partial fulfillment of the requirements for the degree of:

Master of Science in Environmental Management

at the

University of San Francisco

Submitted:



05/10/2020

Heather Navle   Date

Received:

Aviva Rossi, PhDc   Date

Table of Contents

List of Figures

List of Tables

Abstract

Wildfires in the mixed conifer forests of California's Sierra Nevada have been a common and natural disturbance for thousands of years, historically occurring every 3 to 30 years. The flora and fauna of the mixed conifer forest have evolved to depend on low to moderate severity wildfires for reproduction, foraging, and habitat. However, the Sierra Nevada has experienced dramatic environmental changes over the past ~150 years as a result of three main factors: wildfire suppression, climate change, and habitat loss. Because of the threat wildfires pose to human lives, property and timber harvest, they have been suppressed to an extent that has completely altered mixed conifer ecosystems. One of the changes to these ecosystems is increased vegetative fuel density, which can result in stand-replacing mega fires. To mitigate these high-severity mega wildfires, forest managers incorporate various fuel reduction methods into forest management plans. These impacts can have negative effects on forest ecosystems, degrading ecosystem characteristics that are critical for adapting to climate change. Thus, the two main objectives of this paper are to compare and contrast four different fuel reduction methods based on their effectiveness to (I) reduce wildfire risk and (II) promote climate change resiliency. The four fuel reduction methods are: low thinning, canopy thinning, selective thinning, and prescribed fire. These four fuel reduction methods have been compared in syntheses tables for the two main objectives. Qualitative and quantitative metric data, based on a literature review, were used to compare the optimal effects of each fuel reduction method. It was found that prescribed fire or thinning with prescribed fire resulted in the most optimal effects when considering both reduced wildfire risk and climate change resilience. However, tree mortality and the risk of fire escaping controlled boundaries are increased during prescribed fire operations. Additionally, results showed that all four fuel reduction methods displayed both positive and negative effects, depending on the metric used to evaluate the objective, which suggests that appropriate application of fuel reduction methods is highly variable depending on the goals and the environment. For example, canopy thinning alone may have desirable effects when prescribed fire is financially unfeasible or unsafe due to proximity to buildings. Applying prescribed fire is the most optimal fuel reduction method in most forest conditions; however, it is recommended that forest managers evaluate forest structure, density, and tree species prior to selecting the most appropriate fuel reduction method for their situation.

Acknowledgements

I would like to thank all my professors at University of San Francisco in the MSEM program for their guidance and knowledge, especially Aviva Rossi, Gretchen Coffman, Thomas McDonald, Tracy Benning, Amalia Kokkinaki, Thomas MacDonald, Calla Schmidt, April Randle, and Maggie Winslow. I would also like to thank my husband, Dennis, for his support and advice throughout this process. He has been my lifeline during my whole academic venture. I would also like to thank the US military for giving me the opportunity to honorably serve our country for 11 years and providing me with my GI Bill. A higher education would not have been possible without my veteran education benefits.

"The clearest way into the Universe is through a forest wilderness"
(Muir 1979)

1 Introduction

Wildfires are a common natural and anthropogenic ecosystem disturbance across the western United States (Agee and Skinner 2005, Sugihara et al. 2006). However, wildfires pose a major risk to lives and property (Calkin et al. 2014). In recent years, wildfires have become a threat to both the wildland-urban interface (WUI) and to highly urbanized areas (Calkin et al. 2014). Since the year 2000, California has faced some of the most destructive wildfires in its history, especially in the Sierra Nevada (SN) (Keeley and Syphard 2019). The loss of life in the town of Paradise in the 2018 Camp Fire was devastating; and many similarly situated communities live in fear of the next catastrophic wildfire (Keeley and Syphard 2019). Moreover, wildfire smoke can have indirect deleterious effects, such as lung and heart disease (Reid et al. 2016). These health complications due to wildfires are only predicted to increase as urban sprawl expands and population grows (Agee and Skinner 2005, Sugihara et al. 2006). Wildfires are expected to grow in both size and severity due to drought and precipitation changes caused by anthropogenic climate change (Abatzoglou and Williams 2016). Additionally, vegetative fuel build-up occurs due to the suppression of natural disturbance events, such as wildfire (Sugihara et al. 2006). Wildfire management practice includes fuel reduction, which is used to reduce vegetation that has accumulated over time (Sugihara et al. 2006). However, fuel reduction is often constrained by effects on wildlife habitat conservation and unknown climate change complexities (Agee and Skinner 2005). These management practices must consider human expansion into the WUI, which necessitates a complex dynamic of implementing effective practices that protect lives and property, while at the same time protecting ecosystems and wildlife (Jain et al. 2012). This report evaluates and compares the effectiveness of four common fuel reduction methods often prescribed to reduce wildfire risk in mixed conifer ecosystems in the SN and evaluates the effects of climate change on these practices.

1.1 Wildfire: An Ecological Process

Fire is an ecological process that occurs in most of California's highly diverse landscapes (Agee 2006) and is a necessary disturbance for many ecosystems to function (Meyer and Safford 2011, Van Wagtendonk and Fites-Kaufman 2006). Without the function and survival of natural ecosystems, the ecosystem services we rely on for agriculture, drinking water, and fisheries will collapse (Laughlin et al. 2004, Pausas and Keeley 2019). Forest and chaparral ecosystems in California depend on intermediate occurrences of wildfire to promote species richness and successional processes (Goodwin et al. 2018). Species richness is an environmental characteristic that has been shown to create ecosystem resiliency in the face of climate change-induced drought, pests, and habitat fragmentation (White and Long 2019). Wildfire reestablishes successional processes by removing thick duff and litter layers (Laughlin et al. 2004). Without low to moderate severity wildfires, surface vegetation can accumulate and have negative impacts on tree communities, such as shade intolerant pines (Laughlin et al. 2004). Fire reduces surface vegetation so that light can penetrate to the forest surface and allow trees and plants to germinate (Stephens and Moghaddas 2005). Without sunlight penetrating the upper forest canopy, stands can become dense with shade-tolerant trees, which deplete the soil of available water (Dolanc et al. 2014). Shade-tolerant trees such as *Calocedrus decurrens* (incense cedar) will outcompete large, older trees for nutrients and water. *C. decurrens* will grow in dense, monotypic stands, priming the mixed conifer forest for a high-severity wildfire risk. These types of vegetative changes caused by wildfire suppression have rippling effects throughout the mixed conifer ecosystem (Dolanc et al. 2014).

1.1.1 Fire in the Sierra Nevada Mixed Conifer Ecosystem

The Sierra Nevada (SN) is a prominent mountain range that covers roughly 17% of California (Van Wagtendonk and Fites-Kaufman 2006). It stretches approximately 620 kilometers from north to south and roughly 80 kilometers from west to east, covering 69,560 square kilometers (Figure 1) (Van Wagtendonk and Fites-Kaufman 2006, Hamilton 1992). John Muir, an early activist for the preservation of wilderness in America, regarded the SN as the "range of light" because of its radiance and rugged beauty (Hamilton 1992).

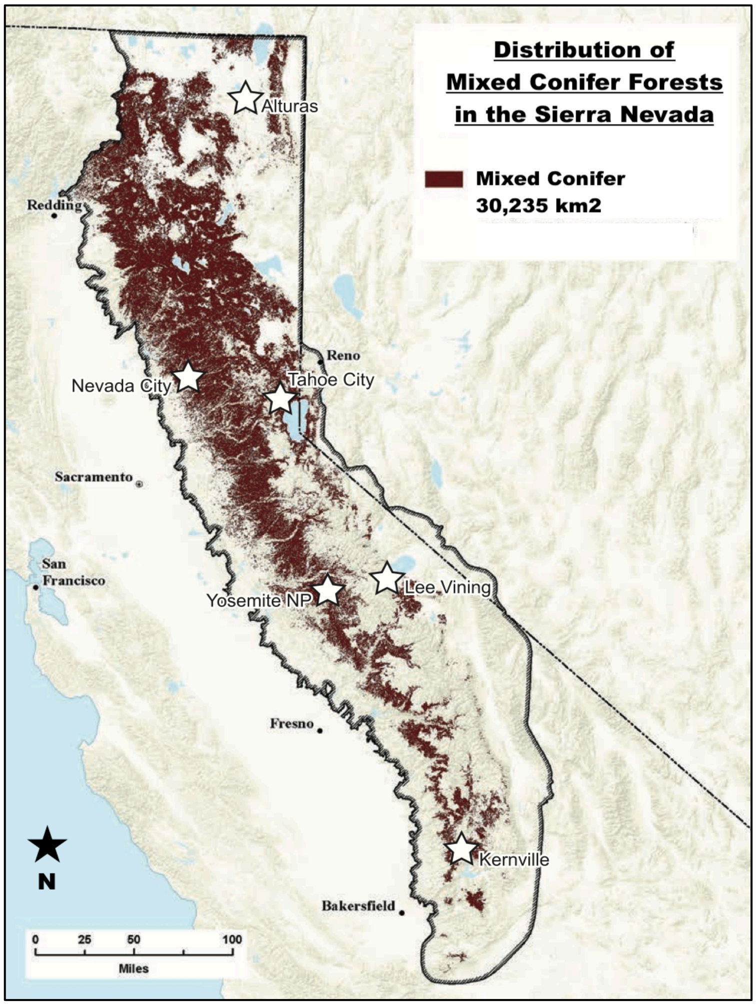


Figure 1. The range of mixed conifer forests in the SN (Safford and Stevens 2017).

1.1.1.1 Sierra Nevada Landform

Approximately 225 million years ago the SN began its iconic granitoid block core formation (Schweickert 1981). Flows of magma nursed the expanding mountain range for 125 million years, creating a granite batholith under the ancient sea floor (Davis et al. 2012). The Cretaceous Period (145–66 mya) brought uplift to the granitoid batholith and exposed the young mountain range to the chemical and physical processes of the atmosphere, which eroded the range into a "proto-Sierra Nevada" (Davis et al. 2012). However, much of the dramatic uplift of the SN occurred in the late Cenozoic Era (~66 mya to present) (Wakabayashi and Sawyer 2001). As the earth began to experience cool temperature cycles, glaciation events periodically carved deep valleys and chiseled upper elevations into what we see today (Huber 1987, Konrad and Clark 1998).

The SN's fault-block mountain range is tilted to the west, so that the eastern slopes rise abruptly from the Great Basin and the western slopes rise gently in terraces and foothills (Wakabayashi and Sawyer 2001). This block tilting has created the highest point in the continental United States, Mount Whitney at 4,417 meters, located in the southeastern section of the SN (Huber 1987, Wolf 1964). Due to the SN's massive formations of metamorphosized granite in quartzites, erosion has occurred slowly, resulting in little soil formation on peaks, ridges, and slopes (Huber 1987). These formations of granite produce heterogenous landscapes and have played an important role in historical fire regimes by forming natural fire breaks in vegetation.

1.1.1.2 Sierra Nevada Climate

The SN's mixed conifer Mediterranean climate has dry, hot summers and cold winters (Johnson et al. 2017). Mean low and high temperatures in the mixed conifer zones typically range from 2°C to 23°C depending on elevation and latitude (Krasnow et al. 2017). The elevation gradient in the SN has an average temperature lapse rate of 3.8°C every 1,000 m (Wolf 1964). Subalpine elevations that are greater than 2,290 m experience microthermal climates which are akin to boreal forest ecosystems. Because of their ability to retain snowpack, these microthermal climates have important cascading hydrologic effects on lower elevations in the SN (Peterson and Arbaugh 1992). The SN's geographical location near the Central Valley, along with the predominant wind direction, produces a rain shadow effect that causes most precipitation to fall over the western slopes (Perterson and Arbaugh 1992). Atmospheric rivers form over the Pacific Ocean and cause 30–40% of the precipitation in the SN to fall in January, February, and March (Van Wagtendonk and Fites-Kaufman 2006). Precipitation typically increases with elevation and will generate heavy snow in the lower and upper montane sections of the mixed conifer zones (Perterson and Arbaugh 1992). However, precipitation generally decreases from the northern to the southern latitudes, so that the southern regions receive significantly less precipitation (Peterson and Arbaugh 1992). For example, southwestern slopes can receive approximately 26 cm of mean annual precipitation while the regions north of Lake Tahoe can receive up to 200 cm (Van Wagtendonk and Fites-Kaufman 2006). These precipitation differences result in varying amounts of fuel accumulation based on latitudinal location in the SN (Peterson and Arbaugh 1992).

Anthropogenic climate change is altering weather patterns in the SN by increasing winter temperatures and altering precipitation events (Abatzoglou and Williams 2016, Safeeq et al. 2016). Regional climate change models predict decreased snowfall and increased rainfall in the western United States within the next century (Wrzesien and Pavelsky 2020). The SN is expected to see continuing shifts in snowmelt and precipitation events to earlier in the water-year (Figure 2) (Wrzesien and Pavelsky 2020). Winters are projected to become shorter, with more variable rainfall amounts; this is anticipated to increase runoff into lower elevations (Wrzesien and Pavelsky 2020). Future climate impacts will affect drought and vegetation aridity, likely increasing wildfire season length and severity throughout the SN (Abatzoglou and Williams 2016).

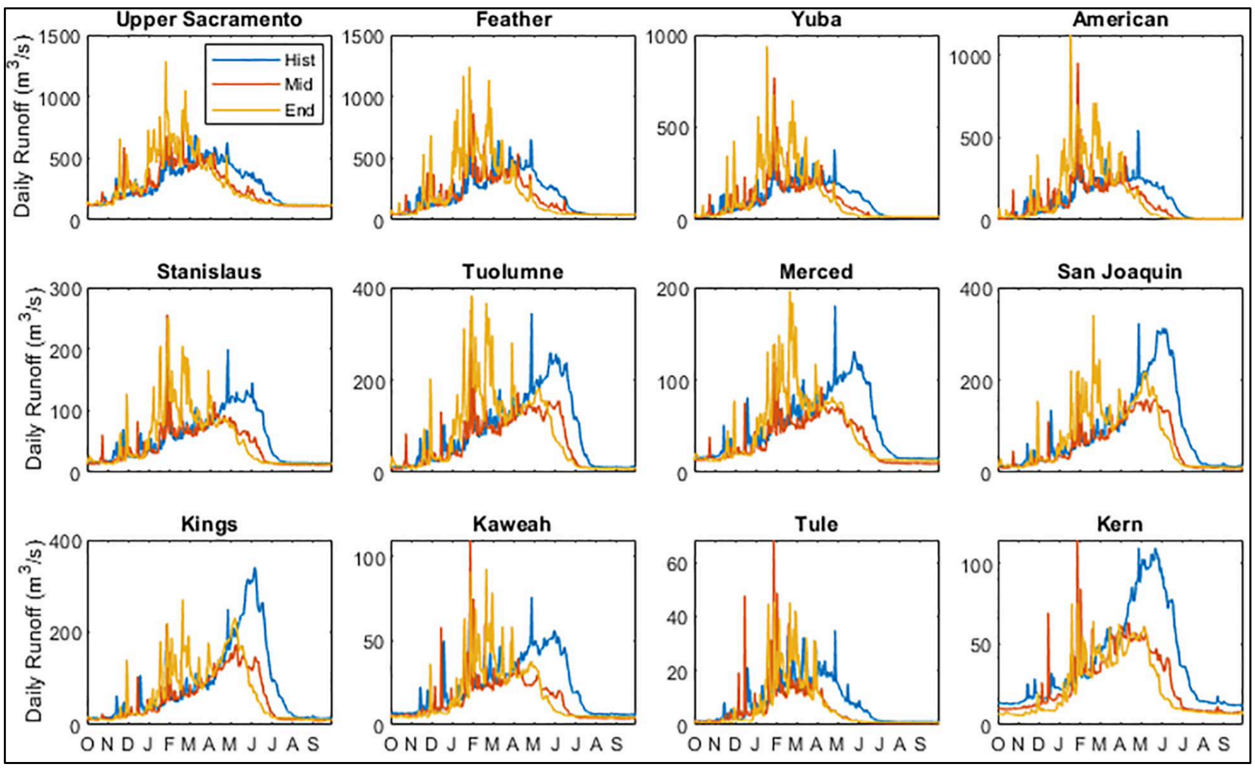


Figure 2. Hydrographs representing snowmelt and precipitation in 12 SN watersheds over three time periods: 1996–2005 blue lines, 2041–2050 orange lines, and 2091–2100 yellow lines. Months are represented the water-year on the x-axis. Note the hydrology retreating into the earlier months of the water-year (Wrzesien and Pavelsky 2020).

1.1.1.3 Sierra Nevada Mixed Conifer Vegetation

Mixed conifer ecosystems are non-coastal, low- to mid-elevation dry forests that exist in the western regions of North America (Odion et al. 2014). 45% of the forests in the SN are composed of mixed conifer habitat (Maloney et al. 2008). Mixed conifer ecosystems are generally dominated by *Pinus ponderosa* (ponderosa pine) and are drier than high elevation or coastal forests (Odion et al. 2014). Healthy mixed conifer forests in the SN consist of heterogenous, patchy landscapes with ~70% of patches containing low densities of shade-intolerant *Pinus* ssp. and ~15% high densities of shade-tolerant species (Dow et al. 2016). These mosaic landscapes also encompass open areas of dead trees (snags), usually caused by disturbance events such as wildfire, wind, or beetle infestation (Stevens et al. 2016). When disturbance events create open areas that encompass 3–6% of the forest, they are generally considered to be healthy and natural (Collins et al. 2016). Due to decreased fire disturbances within the last 150 years, roughly 75% of SN forests have shifted toward a high density of shade-tolerant species (Steel et al. 2018, Safford and Stevens 2017).

Mixed conifer ecosystems are made up of three main types of vegetation strata: ground, surface, and canopy (Agee and Skinner 2005). Ground vegetation generally consists of needles, leaves, rotting biomass, and humus (Steel et al. 2019). Surface vegetation includes woody and non-woody debris in contact with the surface, which can include logs, shrubs, grasses, and saplings (Jain et al. 2012). Canopy strata consist of all layers of vegetation that do not contact the surface (Jain et al. 2012). However, shifting fire regimes have changed forest structure by adding more density to lower canopy layers (Steel et al. 2018). Forest canopies can be broken down into four classes based on height: dominant, codominant, intermediate, and suppressed (Jain et al. 2012). The tallest canopies are referred to as dominant and the lowest canopy levels are suppressed (Jain et al. 2012).

Mixed conifer ecosystems usually occur on the western slopes of the SN at elevations of 1000 m to 2500 m (Peterson and Arbaugh 1992, Safford and Stevens 2017). Generally, mixed conifer forests can be found in lower elevations in the northern sections and higher elevations in the southern sections of the SN (Peterson and Arbaugh 1992). The term "mixed conifer" refers to the mix of evergreen tree species and some deciduous tree species (Stephens and Finney 2002, Walker et al. 2012). Mixed conifer ecosystems occur in the lower and upper montane forest zones, which contain different dominant tree communities (Figure 3) (Stephens et al. 2015). Generally, the upper zones are dominated by *Abies concolor* (white fir) and the lower zones are dominated by *Pinus* ssp. (pine species) and scattered *Quercus* ssp. (oak species) (Stephens et al. 2015). Ten major tree species are found within the mixed conifer ecosystem; some are more prominent depending on topographical, hydrological, and pedological features (Table 1). Elevation, latitude, slope, aspect, soil, and water availability are all driving factors that form forest structure and vegetative communities (Stephens et al. 2015). Thus, north facing slopes tend to retain more soil moisture and will usually be dominated by *Abies concolor* and *Pseudotsuga menziesii* var. *menziesii* (Douglas fir) (Stephens et al. 2015). *S. giganteum* only occurs in approximately 70 protected groves between 1,370 m and 2,190 m on the western slopes. This limited range of *S. giganteum* populations is a result of its narrow ecological niche coupled with centuries of overharvest, making *S. giganteum* particularly vulnerable to climate change and the ill-effects of wildfire suppression (DeSilva and Dodd 2019). Historically, wildfire altered stand composition and structure; however, beginning in the 19th century humans started to manipulate forest structure in the SN (Dolanc et al. 2014). Fire suppression caused stand density to increase, shifting the vegetation composition in lower zones to *A. concolor* and *Calocedrus decurrens* (incense cedar) (Abella and Springer 2015).

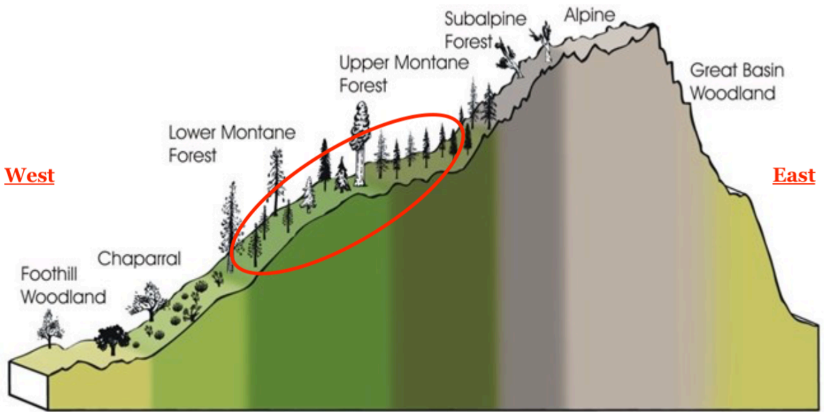


Figure 3. Cross-section (west to east) of the SN. The red oval area roughly indicates the mixed conifer ecosystem covering the upper and lower montane zones (Sourced from Justin Hofman and Meyrl Goldin Rose 2020).

Table 1. Common evergreen and deciduous tree species found in mixed conifer ecosystem.

Species	Common Name	Life Span (yrs)	Height (m)	Wildfire Resistance	Germination	Shade	Elevation (m)
Pinus ponderosa	ponderosa pine	300–600	30–50	resistant	full sun	intolerant	150–1830
Pinus jefferyi	Jeffrey pine	400–500	52–61	resistant	full sun	intolerant	1500–2700
Pinus lambertiana	sugar pine	400–500	61–76	semi-resistant	partial sun	semi-tolerant	600–2300
Abies concolor	white fir	>300	42–55	Not resistant. Only mature trees are semi-resistant.	partial–full shade	tolerant	1500–1440
Pseudotsuga menziesii* var. *menziesii	Douglas-fir	>1,000	61–76	resistant–semi-resistant	partial–full sun	tolerant	1500–1440
Calocedrus decurrens	incense cedar	>500	20–57	Not resistant. Only mature trees are semi-resistant.	partial–full shade	tolerant	600–2100
Sequoiadendron giganteum	giant sequoia	<3,200	76–84	resistant	partial–full sun	semi-tolerant	1400–2000
Quecrus kelloggi	black oak	100-500	9–36	resistant	partial–full sun	semi-tolerant	60–2400
Quecrus chrysolepis	canyon live oak	<300	4.5–30	not resistant	partial–full shade	tolerant	90–2700
Cornus nuttallii	Pacific dogwood	<195	6–22	semi-resistant, sprouts from fire	partial–full shade	tolerant	547–1981
Data Sourced from the USDA, 2019							

The flora of the mixed conifer ecosystem in the SN has evolved over millions of years with low to moderate severity wildfire and has thus developed special adaptations to wildfire (Krasnow et al. 2017). Wildfire acts as a catalyst for the seed germination of many tree and plant species in the mixed conifer ecosystem (Sugihara et al. 2006). Mature *Sequoiadendron giganteum* (giant sequoia) has evolved serotinous cones that open and release seeds with heat (Meyer and Safford 2011). Also, *S. giganteum*, *P. lambertiana*, and *P. ponderosa* grow thick, fire adapted bark that has evolved to withstand low to high-severity fires events (Meyer and Safford 2011, DeSilva and Dodd 2019). Additionally, *Arctostaphylos* sp., the manzanita genus, is a recognized pioneering shrub that proliferates in open spaces created by fire; with its flammable resins, it is thought to be a fire-recruiting species, as well (Keeley 1992). *Chamaebatia foliolosa* (bearclover), another fire adapted species containing flammable resins, has been shown to be the main food source for *Bombus* spp., bumble bees, in mixed conifer ecosystems (Loffland et al. 2017). Pollinators, like *Bombus* spp., play a critical role in overall ecosystem health (United States Department of Agriculture 2019).

1.1.2 History of Fire Regime Alteration in the Sierra Nevada

Historical evidence indicates that Native Americans have been manipulating fire regimes in the SN for approximately 8,000 years (Gassaway 2007). Hunter-gatherers lit wildfires intentionally to move game species, clear vegetation, encourage seed germination, increase foraging resources, and create travel routes (Klimaszewski-Patterson and Mensing 2016). In the mixed conifer ecosystems of Yosemite, the Miwok people ignited wildfires to encourage the growth of *Quercus* spp. (oaks), which provided their main food source of oak acorns (Scholl and Taylor 2010). Furthermore, Native Americans may have influenced forest compositions to favor more shade-intolerant (e.g., *P. ponderosa* and *P. lambertiana*) tree species as a result of their engineering for more open canopies (Klimaszewski-Patterson and Mensing 2016). Increased light through the canopy encouraged the growth of important flora used for food and tools (Gassaway 2007). Some evidence suggests that Native Americans may have completely altered forest structure to *Quercus* spp. and meadow grasses (Anderson and Carpenter 1991). This past alteration may distort historical reference site data when attempting to restore forest composition to pre-settlement conditions (Anderson and Carpenter 1991). The fire regime changes orchestrated by indigenous people were minimal in comparison with the impacts of European settlers, beginning in the 19th century (Taylor and Scholl 2012).

Impacts on fire regimes by Europeans and other non-indigenous people included cattle grazing, introduced plant species, contemporary agriculture practices, and timber harvest (Taylor and Scholl 2012). Protecting timber soon became a priority, and by the early 1900s and complete fire suppression became policy, leading to a persistent alteration in natural fire regimes (Figure 4) (Van Wagtendonk 2007). In 1944 the Forest Service introduced Smokey the Bear, a fictional character who educated the public about wildfire safety, campaigning for wildfire prevention with an insistence that wildfires were purely destructive to natural resources (Forest History Society 2019). It was not until the 1960s that the scientific community began to convincingly demonstrate that fire performed a vital role in forest ecology (Forest History Society 2019). As a result, in 1968 the National Park Service incorporated the concept of fire as a natural ecological process into policy (Van Wagtendonk 2007). Almost a decade later, the Forest Service began to recognize the importance of fire and altered its policy of complete suppression, incorporating fire as a prescription method (Van Wagtendonk 2007). By the 1980s most government agencies had implemented the use of prescribed fire as a preventative measure against uncontrolled wildfires (Van Wagtendonk 2007).

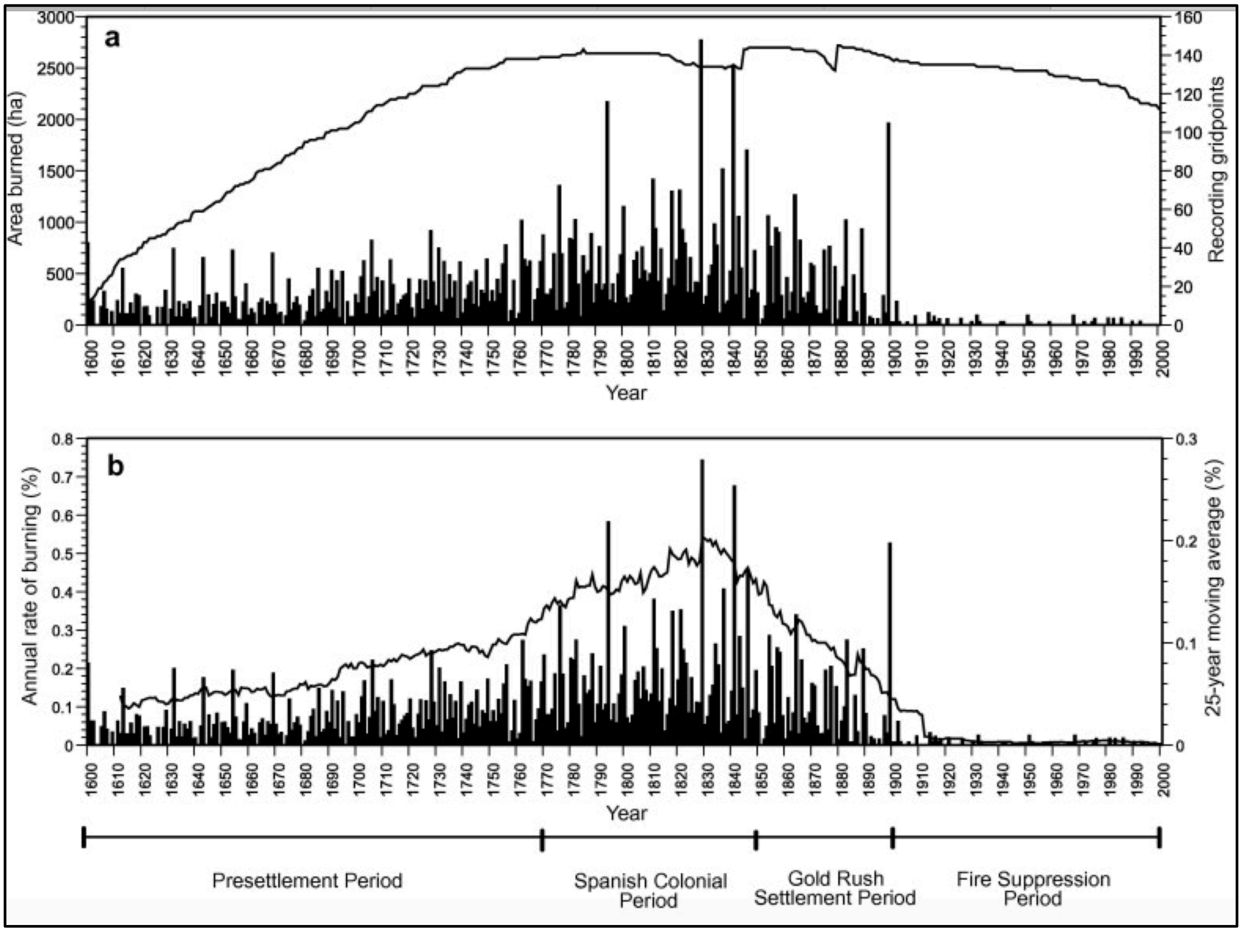


Figure 4. Mixed conifer forests in Yosemite National Park from 1600 to 2000. Annual area burned (a) and annual rate of burning (percent study area burned annually) (b). The left y-axis is associated with the bars and the right y-axis is associated with the line. Timeline highlights: pre-settlement, before 1769, and fire suppression after 1900 (Taylor and Scholl 2012).

As the human population continues to expand into the WUI, prescribed fire (PF) operations are more challenging to conduct, due to the risk of escape and threat to human structures (Van Wagtendonk 2007). Therefore, because of the challenges associated with fuel management operations, the expansion of the WUI landscape can indirectly increase the chances of a large catastrophic wildfire (Van Wagtendonk 2007). Due to the risks associated with PF near the WUI, as communities expand farther into the SN, mechanical vegetation removal has taken precedence over PF in these situations (Kane et al. 2010).

1.1.3 Ecological Impacts of Wildfire Suppression

Forest ecosystems that have endured wildfire suppression beyond their natural fire return intervals can become dense with shade-tolerant trees, which deplete the soil of available resources, such as water (Stephens et al. 2018). In addition to high competition due to tree stand densification, climate change projections show increases in drought events which can exacerbate water stress in trees (Thorne et al. 2018, Restaino et al. 2019). Within these crowded, dry forests, stressed trees cannot fight off bark beetle infestations and pathogens (Stephens et al. 2018). Furthermore, if the dense canopy layers block enough sunlight, herbaceous plants cannot photosynthesize (Collins et al. 2007). It has been suggested that healthy forest ecosystems need intermediate disturbances to retain successional processes (Christensen 2014). The intermediate disturbance hypothesis suggests that there is a correlation between frequency of disturbances and species diversity (DeSiervo et al. 2015). This correlation forms a unimodal curve with the low and high frequency of disturbances producing lower species diversity (DeSiervo et al. 2015). Without disturbance regimes, succession can become skewed by large quantities of impenetrable vegetation competing for resources, which can eventually lead to degraded habitat for wildlife as well as high-severity fires (Christensen 2014). Wildfire suppression has led to infrequent, high-severity wildfires which are uncharacteristic for mixed conifer fire regimes (Richter et al. 2019). It has been shown that high-severity wildfires can decrease native plant biodiversity (Richter et al. 2019). Wildfire suppression can create an ecological chain-reaction that leads to plant and animal biodiversity loss (Van Wagtendonk and Fites-Kaufman 2006, Sugihara et al. 2006, Meyer and Safford 2011).

1.2 Fuel Management Applications

Fire regimes in the Sierra Nevada can be determined through environmental proxy data, like using fire scarring on tree growth rings to estimate fire return intervals (dendrochronology) (Barth et al. 2015). Mixed conifer historical fire regimes as found in fire scars on *S. giganteum* and these can be analyzed to roughly 2,000 years ago (Swetnam 1993). The scars on *S. giganteum* have indicated that the fire return intervals did not exceed 30 years in five *S. giganteum* groves (Swetnam 1993). Sediment cores are also used to collect paleo-fire data, which is identified through microscopic charcoal concentration layers found in lakes (Skinner and Chang 1996). Lakes in Yosemite revealed charcoal concentrations from up to 16,000 years ago (Smith and Anderson 1992). This data indicates that wildfires in Yosemite occurred every 5 to 30 years in mid-elevation forests (Smith and Anderson 1992). The many studies on the fire regimes of the SN have been used in a metanalysis by Skinner and Chang (1996) (Figure 5). Reflecting on these 5 to 30-year fire return intervals, we should consider the ecological changes that have accumulated during fire suppression in the last century (Skinner and Chang 1996). Understanding historical wildfire frequency in the mixed conifer forest is vital to applications in wildfire science and fuels reduction methods (Keeley and Syphard 2016).

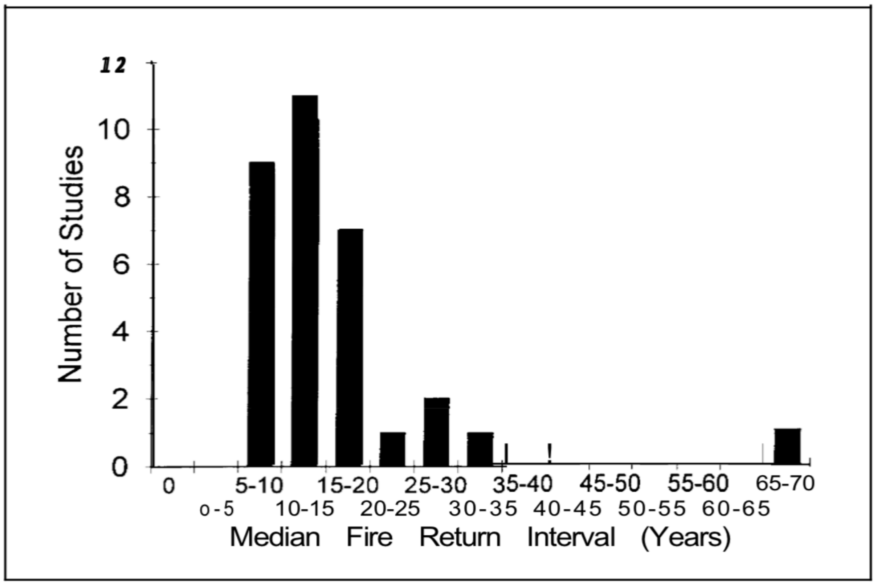


Figure 5. Median fire return interval for the SN. Assessed studies are on the y-axis and median fire return interval of approximately 5–30 years on the x-axis (Skinner and Chang 1996).

Forest manager's objectives can range from merely reducing wildfire risk to producing marketable timber to restoring wildlife habitat (Agee and Skinner 2005). Generally, the main objective of fuel reduction's various thinning methods is to decrease vegetation fuel loads to lower the risk of stand-replacing wildfire (Figure 6) (Kelsey 2019). Controlled burning, the mechanical removal of vegetation, and the stacking of trees, shrubs, and dead vegetation are all basic methods of fuel reduction that are often prescribed by forest managers (Agee and Skinner 2005). Prescribed forest thinning is a silvicultural application that involves the mechanical removal of trees based on size and species (Kane et al. 2010). Different types of fuel reduction treatments are prescribed depending on forest type, density, topography, and available resources (Stephens et al. 2012).

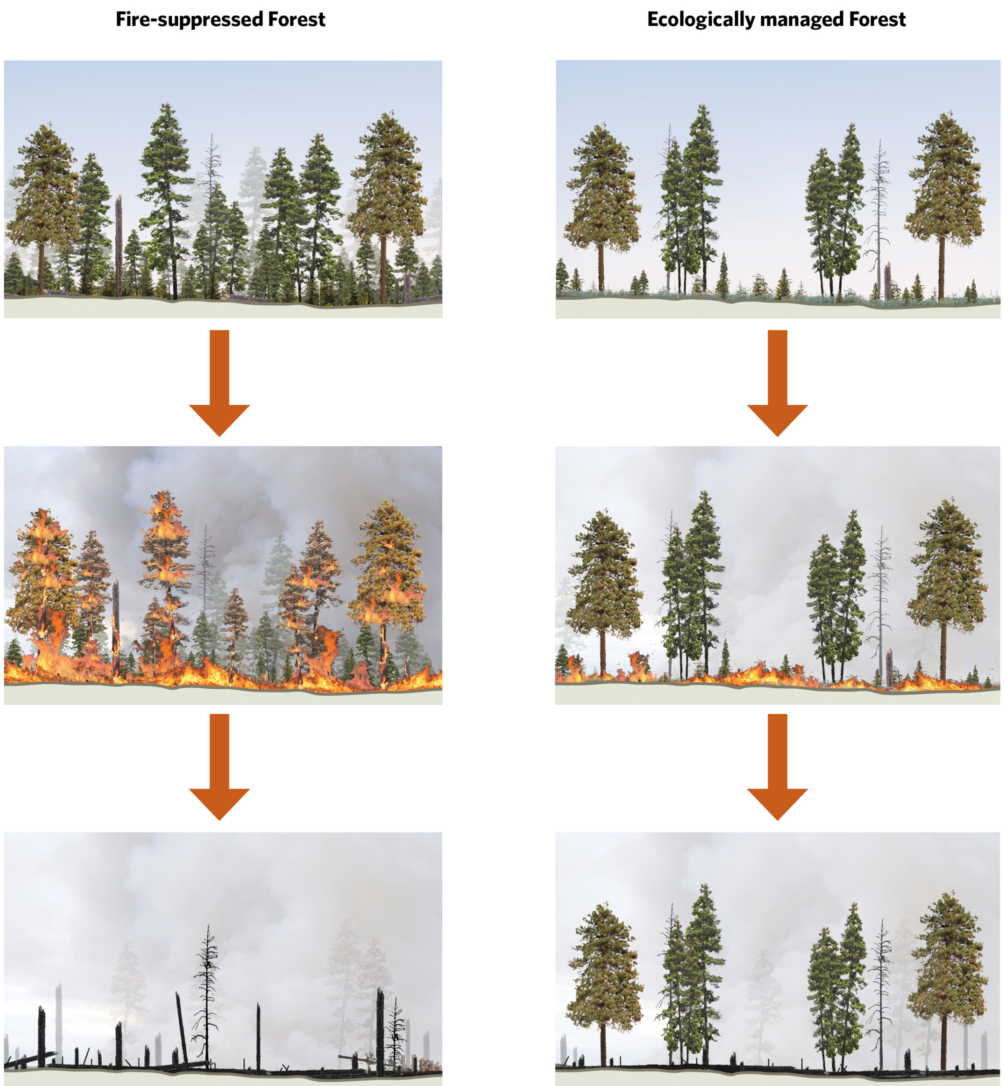


Figure 6. Two examples of forest plots, the left side fire-suppressed and the right side ecologically managed (applied fuel reduction) (Kelsey 2019).

Diameter at breast height (DBH) is a measurement which represents the cross-section of a tree trunk at 1.3 meters above the base of the tree (Bettinger et al. 2017). Tree basal area is a measurement term often used in forestry referring to the total DBH in a unit area (hectares or acres) (Bradford and Bell 2017). Understanding basal area allows forest managers to communicate and assess timber quantitatively within a given area (Bettinger et al. 2017). For example, reducing tree basal area to ~14 m^2 per hectare is a goal that is within the range of a low thinning method.

Sustainable and fire-resilient forest management requires a wide range of professional knowledge from different fields, with specialists needing years of training and experience (Agee and Skinner 2005). Foresters, ecologists, firefighters, and biologists can all be considered experts in certain aspects of forest management (Jain et al. 2012). Therefore, it could take several experts to compose and implement a comprehensive wildfire mitigation plan (Dow et al. 2016). The objectives of a fuel reduction plan should be stated early in the process to allow a shared framework for resolving interdisciplinary conflicts, should they arise. The formation of a fuel reduction plan should incorporate a comprehensive list of environmental factors that must be considered to ensure safe and effective implementation (Table 2). Depending on proximity to homes and other man-made structures, objectives of fuel reduction can differ (Kane et al. 2010, Steel et al. 2018). For example, if the treatment is going to be conducted near a housing development, the objective might be to clear vegetation so that fire suppression operations have ready access to the stand (Jain et al. 2012). Another objective might be to create a more wildfire-resilient ecosystem, which would closely mimic a pre-wildfire suppression forest structure (Steel et al. 2018). Creating heterogenous mixed conifer forests is usually done in wilderness areas, national and state parks, and other natural areas in which ecosystem functionality is vital (Kane et al. 2010, Kelsey 2019, Scholl and Taylor 2010).

Table 2. Environmental factors that must be considered when creating a fuel reduction plan.

Subject of Expertise	Environmental Factor
Vegetation	Federal and state listed plant and tree species in areas of concern
	Vegetation reactions and relationships with wildfire
	Vegetation growth rate, vigor, and resistance to disturbance
	Vegetation structure (canopy layers)
	Forest composition history (prior to fire suppression)
	Pests patterns, such as bark beetle and pathogens
Soil	Soil types, textures, and moisture content
Hydrological	Proximity to waterbodies (e.g. creeks, rivers, ponds, lakes, reservoirs)
Climate	Local weather and climate patters, including climate change projections
Development	Proximity to towns, homes, national parks or monuments, prominent hiking trails, etc.
Fire Behavior	Landscape topography
	Local fire regime history
Data source from: Jain et al. 2012, Stephens et al. 2012, Stephens and Moghaddas 2005	

Thinning operations must be designed in consideration of the different types of vegetation or fuel strata (Jain et al. 2012, Stephens et al. 2012). Removing lower strata can increase the vertical length between the surface strata and the canopy strata. This removal method is referred to as adjusting the canopy base height (Agee and Skinner 2005). Fuel management plans generally aim at removing specific basal areas and crown classes within an area (Jain et al. 2012). The cut vegetation, also known as slash, that results from mechanical thinning can either be left at the site or removed; there are costs and benefits associated with each option. Scattering slash over the forest surface can decrease soil erosion, create wildlife habitat, and cycle ecosystems nutrients; however, slash can also increase wildfire hazard (Stephens et al. 2012).

1.2.1 Low Thinning

Low thinning or understory thinning is a mechanical fuel reduction method used to remove trees in the two lowest canopy classes (suppressed and intermediate), leaving larger trees in the stand (Jain et al. 2012, Agee and Skinner 2005). This common type of thinning has the objective of creating better conditions for trees in the upper canopy classes, which include older, larger tree species (Teraoka et al. 2017). Low thinning eliminates ladder fuels and can increase canopy base height (the vertical length from the surface to the canopy) if the smallest trees (<10 cm DBH) are predominantly harvested (Agee and Skinner 2005). However, small trees usually do not generate timber profit and can increase surface fuels if they are not removed off-site (Agee and Skinner 2005). Low thinning can sometimes impact the average height of certain tree species, such as *Calocedrus decurrens* and *Abies concolor* (Walker et al. 2012). These two shade-tolerant species can thrive in suppressed canopies, which makes them susceptible to removal (Jain et al. 2012, Agee and Skinner 2005). One example of an understory thinning prescription incorporates the removal of all trees from 25 to 76 cm DBH (Innes et al. 2006). All thinning methods, including low thinning, can encompass a range of DBH measurements for removal (Figure 7), which will vary depending on fuel management plan objectives (Agee and Skinner 2005).

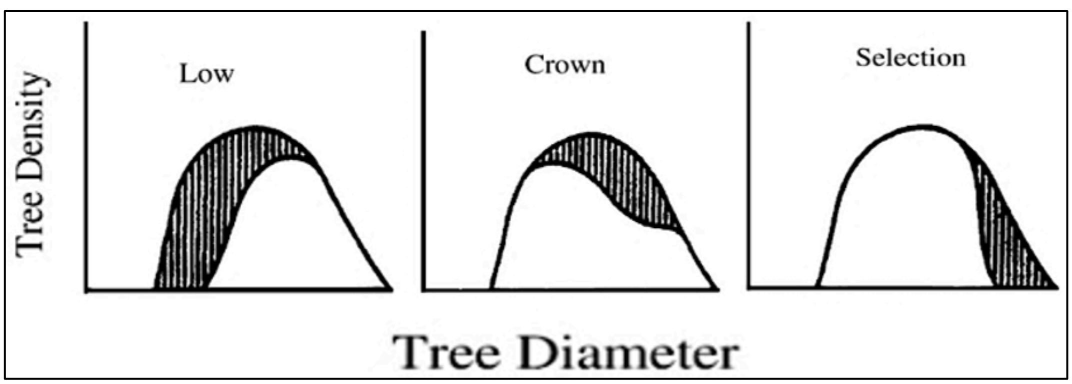


Figure 7. Three methods of thinning in an even-aged plot. The shaded area indicates the approximate tree diameters removed. Low thinning will reduce trees within the lower DBH range. Crown thinning will remove some codominant and dominant trees layers, which tend to have higher DBH ranges. Selective thinning focuses on the largest trees within the plot (Agee and Skinner 2005).

1.2.2 Canopy Thinning

Canopy thinning or overstory thinning is a mechanical fuel reduction method used to decrease the vegetative mass within the canopy layers of a tree stand (Zald et al. 2008, Jain et al. 2012). The target canopy for removal can be the dominant, codominant, intermediate, or suppressed layer (Jain et al. 2012). Larger DBH measurements tend to be associated with higher and denser canopy strata (Ritchie et al. 2013). However, canopy thinning does not always result in the removal of older trees. Canopy thinning allows patches of sunlight to reach the forest surface layers and has been shown to have some positive effects on vegetative biodiversity (Teraoka et al. 2017, Jain et al. 2012) (Figure 8). Canopy thinning has been shown to have positive effects on *Pinus* sp. Regeneration; however, canopy thinning does not increase canopy base height (Zald et al. 2008, Agee and Skinner 2005). If base height is unaddressed, it will lead to the accumulation of ladder fuels (Zald et al. 2008, Agee and Skinner 2005). Canopy removal can negatively impact some wildlife, such as the *Strix occidentalis* (spotted owl), by degrading foraging habitat (Gallagher et al. 2019).

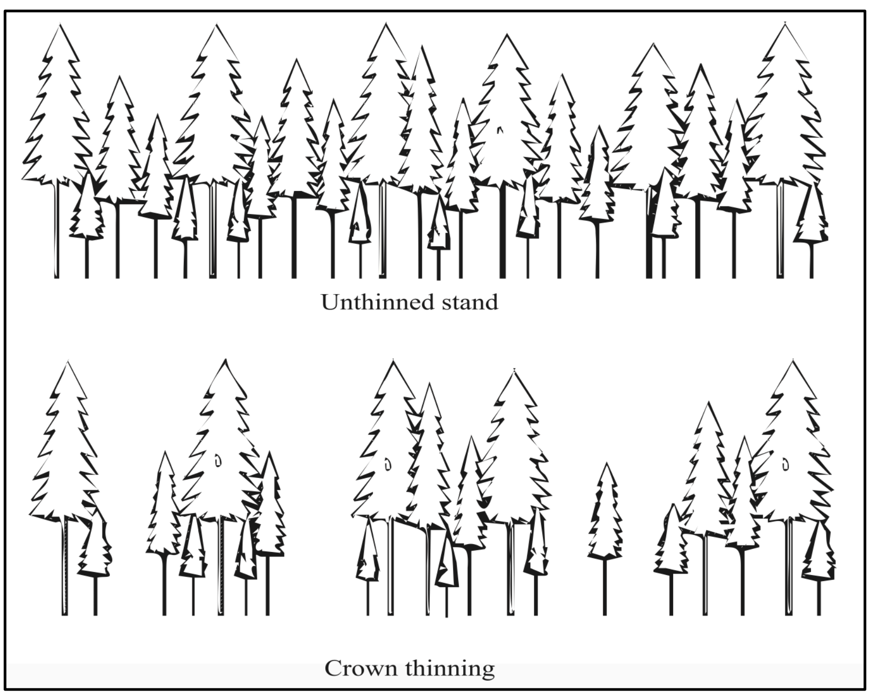


Figure 8. Crown thinning example (Jain et al. 2012).

1.2.3 Selective Thinning

Selective thinning or diameter-limit thinning is a mechanical fuel reduction method that removes trees based on specific DBH measurements and tree species. This type of thinning can be very particular, and it is often used for marketable timber harvest (Jain et al. 2012, Agee and Skinner 2005). Generally, selective thinning will remove the largest tree species due to their higher value. However, selective thinning can vary greatly depending on the objectives of the fuel management plan. Selective thinning does not usually increase canopy base height and is sometimes known to leave high amounts, roughly 60% in dry forest, of small and unmerchantable understory trees (Table 3) (Agee and Skinner 2005). In contrast to canopy thinning, selective thinning can result in the creation of large gaps, which may have negative impacts on wildlife populations (Gallagher et al. 2019).

Table 3. Low, crown, and selection effects on stand canopy structure (Agee and Skinner 2005).

Effect of thinning method on canopy characteristics

Method	Effect on canopy characteristics	
	Canopy base height	Canopy bulk density
Low thinning	I[a]	NE/D
Crown thinning	NE	D
Selection thinning	NE	D

I: increase; D: decrease; NE: no effect.

[a] If unmerchantable small trees also removed.

1.2.4 Prescribed Fire

Prescribed fire (PF) or controlled burning is a fuel reduction and restoration method commonly used by fire-trained personnel (Figure 9) (Jain et al. 2012). A notable difference between controlled burning and mechanical thinning is that PF leaves dead vegetation, mostly trees, standing in the treated area (Stephens et al. 2012). Prior to wildfire suppression, mixed conifer forests historically had a relatively short period of time (5–30 years) between natural wildfires, thus prescribed burning is a method for returning fire to the ecosystem in a way that mimics natural cycles (Taylor and Scholl 2012, Pausas and Keeley 2019, Van Wagtendonk and Fites-Kaufman 2006, Sugihara et al. 2006). Additionally, PF has been shown to be one of the most effective ways to decrease surface fuels (Agee and Skinner 2005). Controlled burning can be used for a variety of applications, including reducing logging remains, decreasing hazardous fuels, preparing ecosystems for vegetative regeneration, improving wildlife habitat, controlling pests, expanding forage opportunities, and reducing competition from other tree species (Jain et al. 2012, Meyer and Safford 2011, Sugihara et al. 2006, Sugihara et al. 2006). PF has also been shown to encourage seed regeneration for *Pinus* sp. and *S. giganteum* (Goodwin et al. 2018, Meyer and Safford 2011, Walker et al. 2012). *S. giganteum* saplings have been shown to grow in moderate to high-severity prescribed burns (Meyer and Safford 2011).

Figure 9. Forest Service firefighters conducting a PF after thinning operations (Photo by Ed Smith, Kelsey 2019).

Despite PF's many benefits, there is always the risk it will escape set boundaries and develop into an uncontrolled wildfire (Jain et al. 2012). In some cases, this has led to the destruction of lives, property, and forest ecosystems (Jain et al. 2012). Prescribed burning also produces short-term smoke, which poses a risk to human health (Reid et al. 2016, Henderson et al. 2011). Additionally, there are challenges in execution due to constraints posed by topographical land access, weather, field crew availability, and antiquated bureaucratic regulations (Stephens et al. 2012). Also, controlled burning generally results in low severity burns, and it has been indicated that specifically low and high-severity burns can decrease plant species richness, while moderate burns are the most ideal burn severity for many ecosystems (Richter et al. 2019). Thus, PF must be conducted with interdisciplinary measures if all fuel reduction objectives are to be met (Jain et al. 2012). For instance, conducting mechanical thinning operations prior to PF is a common practice for meeting interdisciplinary objectives (Jain et al. 2012).

1.3 Objectives

Due to the increased occurrence of destructive wildfires, the changing climate, and loss of biodiversity, reducing forest density has been a priority in California (Liang et al. 2017, Abatzoglou and Williams 2016). Therefore, it is vital to continuously assess best management practices for the future forests of the SN. This report researched two success parameter goals:

- **Goal I: Wildfire Risk Management -** Which fuel reduction treatment combinations are most effective when considering wildfire risk management in mixed conifer ecosystems in the SN?

- **Goal II: Climate Change Resilience -** Which fuel reduction treatment combinations best prepare the mixed conifer forest in the SN for climate change and correlated ecological complications?

I evaluated the following four treatment methods for their effectiveness at meeting Goal I and II:

1) **Low thinning (understory thinning)**
2) **Canopy thinning (crown or overstory thinning)**
3) **Selective thinning (for marketable timber)**
4) **PF or thinning with PF**

Under selective thinning, it should be noted that I have evaluated selective marketable timber harvest thinning specifically. Additionally, I have evaluated two fuel reduction methods as one: PF and mechanical thinning combined with PF. These methods were selected for this research project because of their common use in the forest management industry (Agee and Skinner 2005, Jain et al. 2012, Stephens et al. 2012).

2 Methods

This study is based on a literature review of existing fuel management treatment studies. Four specific treatment methods were selected for this research due to their common use and feasibility in the forest management industry (Agee and Skinner 2005, Jain et al. 2012, Stephens et al. 2012). This process resulted in eight studies for **Goal I: Wildfire Risk** and ten studies for **Goal II: Climate Change Resilience.** Several of these studies have used computer simulation models for thinning and wildfire events; these have been equally assessed to non-modeled studies.

Three databases were used to identify literature: Scopus, Environmental Compete, and Google Scholar. Common terms used to locate literature in these databases were: "low thinning, canopy thinning, crown thinning, timber harvest, selective thinning, prescribed fire, controlled burning or fire, wildfire risk mitigation", and "climate change impacts". Additionally, the terms: "Sierra Nevada, California" and "mixed conifer, California" were added to narrow search results to the location and type of ecosystem. Literature used for each goal has been documented in Tables 4 and 6.

Data was extracted from each study and documented in a correlated synthesis table. In total, three synthesis tables were created: (1) wildfire behavior; (2) climate change resilience; and (3) a combined synthesis table. Four measurable metrics were used for each of the first two tables. Wildfire behavior used the following metrics: type, rate of spread, severity, and intensity. Climate change resilience used these metrics: species richness and percent cover, seed germination, tree mortality, and soil water content. In the synthesis tables for wildfire behavior and climate change resilience, a five-point rating system—very poor (1), poor (2), fair (3), good (4), and excellent (5)—was used to quantify and measure data. The five-point rating system values were assigned to each thinning method to indicate its performance in terms of each metric. Values were chosen based on the data extracted from the literature and careful background investigation on each study. Lastly, the final synthesis table incorporated goals I and II as metrics and the five-point rating system total values were added together for a final comparison analysis.

3 Results and Discussion

The following subsections have introduced Goal I: Wildfire Risk and Goal II: Climate Change Resilience. Each goal includes explanations of the metrics used, syntheses, results, and discussions. The concluding subsections consist of the final synthesis, results, and discussion.

3.1 Goal I: Wildfire Risk Management

Fire regimes are best understood by both quantifying and qualifying their characteristics; this is because wildfire is a complex disturbance and occurs over heterogenous landscapes with varying and unpredictable weather conditions (Stephens et al. 2012). Forest managers and firefighters use these characteristics to predict fire behavior in order to protect lives, property, and ecosystems (Stephens et al. 2012, Agee and Skinner 2005). These wildfire metrics are all tied together and will affect each other depending on fuel, topography, and weather (Stephens 1997). For instance, crown fires usually have higher intensities and severity. Surface fires generally have lower intensities and severities. However, this is a general rule that does not apply under all conditions. Four wildfire characteristics—type, rate of spread, severity, and intensity—were obtained from multiple studies and used as metrics for measuring the efficacy of the four thinning methods (Table 4). These metrics are generally used together to describe the overall fire regime of a specific region, like the SN (Skinner and Chang 1996). Each metric has been assessed based on the type of thinning method that was implemented prior to a wildfire or simulated wildfire. Lastly, the four metrics are defined in detail in the subsequent subsections.

Table 4. Information on eight references used in the wildfire behavior synthesis table (Table 5).

n=8	Reference	Metric	Study Design Description	Location	General Timeframes	Page # of Data Retrieval
1	Finney et al. 2007	ROS	Simulation with Forest Vegetation Simulator.	Stanislaus National Forest	Simulated 50 years of treatment rates and fire.	718
2	Krofcheck et al. 2018	Severity	Simulation with Dynamic Fire Fuels System and LANDIS II.	Dinky Creek Watershed in S. SN	100-year simulation.	734
3	Lydersen et al. 2017	Severity	233 treatment transects within footprint of the Rim Fire (2013).	Stanislaus National Forest and Yosemite NP	18 years of treatments and a ~3-month fire event.	2021
4	Pollet and Omi 2002	Severity	Gathered data on post-fire structure, basal area, and density. Cottonwood Fire (1994).	Tahoe National Forest, Sierra County	11 years of thinning. Data gathered two years after fire.	2
5	Stephens 1997	Intensity, Fire Type	Simulation with FARSITE. 75th percentile weather conditions were used for synthesis table.	North Crane Creek Watershed, Yosemite NP	24-hour simulated fire event.	28
6	Stephens and Moghaddas 2005	Intensity, Fire Type, ROS	Real data from fuel treatments. Simulated fire with Fuels Management Analysis. 90th percentile weather conditions were used for synthesis table.	University of California Blodgett Forest Research Station	25 years of thinning treatments.	375
7	Stephens et al 2009	Severity	Simulation fire behavior with Fuels Management Analysis Plus. Real stand structure data from locations. 80th percentile data used.	University of California Blodgett Forest Research Station	One-year post treatment.	314
8	Van Wagtendonk 1996	Intensity, ROS	Simulation with FARSITE. 75th percentile data used.	Modeled after conditions in Yosemite NP	24-hour simulated fire event.	1162

3.1.1 Metric: Fire Type

The fire type is a term used to describe the type of vegetation strata the wildfire is predominantly burning. For example, a crown fire is a type of fire that burns canopy vegetation (Stephens et al. 2012). There are several different types of fire that are categorized by the way they burn through a forest structure (Stephens 1997). It is important for forest managers to understand wildfire types in order to quickly understand fire severity and intensity (Stephens 1997). Crown fires, surface fires, and ground fires were the three main wildfire types and were incorporated into this evaluation based on the information available in the literature (Lyndon et al. 2019). Two studies were incorporated into the synthesis table to evaluate the types of fires that were either simulated or occurred after the four methods of fuel reduction.

3.1.2 Metric: Rate of Spread

Rate of spread (ROS) is important for forest managers to understand because it represents how fast a fire is moving (Stephens and Moghaddas 2005). It is measured in m/min and is generally reduced by prescribed burning (Stephens and Moghaddas 2005). Reducing rate of spread can afford more time in which to evacuate and gather firefighting equipment (Sullivan et al. 2018). Fuel composition plays a critical role in rate of spread by adding biomass to combustion (Sullivan et al. 2018). Generally, higher amounts of vegetative biomass in combination with dry windy atmospheric conditions will increase rate of spread (Finney et al. 2007). Three studies were incorporated into the synthesis table to evaluate the rate of spread of fires that were either simulated or occurred after the four methods of fuel reduction.

3.1.3 Metric: Severity

Wildfire severity is an assessment of ecosystem impacts in the aftermath of fire (Stone et al. 2004). The higher the severity of the wildfire, the greater the change in the ecosystem (Stone et al. 2004). Levels of severity will have various impacts on the successional abilities of vegetation and soil (Stone et al. 2004). Low severity wildfires will generally only kill fire intolerant trees and surface vegetation, as they are not crown fire types and burn at lower intensities (Lydersen et al. 2017). Four studies were incorporated into the synthesis table to evaluate the severity of fires that were either simulated or occurred after the four methods of fuel reduction.

3.1.4 Metric: Intensity

Intensity is a measurement of the rate of heat (kW/m) produced from a wildfire (Stephens 1997). Intensity is the physical process of fire discharging energy from burning fuels (Skinner and Chang, 1996). Understanding fire intensity allows firefighters to estimate the fire severity level and type of fire (e.g. crown fire or surface fire) (Stephens 1997). Three studies were incorporated into the synthesis table to evaluate the intensity of fires that were simulated or occurred after fuel reduction.

3.1.5 Wildfire Behavior Synthesis

The following wildfire behavior synthesis table was used to gather and display data on the effectiveness of each thinning method in reference to a measurable metric (Table 5).

Table 5. The comparison of four fuel reduction methods with four wildfire behavior metrics. Average values were extracted from studies. Rating system: very poor (1), poor (2), fair (3), good (4), and excellent (5).

Fuel Reduction Method	Fire Type	Rate of Spread m/min	Severity	Intensity kW/m
Low Thinning	Some crowning [1] Surface [2]	3.8 [2] 1.36 [4]	Mean reduction by 16.1–18.4% [7]	795 [2] 84.47 [1]
Rating Total: 14	**Fair**	**Fair**	**Good**	**Fair**
Canopy Thinning	Surface [1] Surface [2]	3.9 [2] 1.49 [4]	Generally reduced [8]	805 [2] 68.75 [1]
Rating Total: 13	**Excellent**	**Poor**	**Fair**	**Fair**
Selective Thinning	Partial crowning [1] Surface [2]	4.0 [2]	>80% small (2–25 cm DBH) tree mortality [5]	817 [2] 114.36 [1]
Rating Total: 6	**Poor**	**Very Poor**	**Poor**	**Very Poor**
PF or Thinning with PF	Surface [1]	Decreased [3] 0.65 [4]	Unchanged–low [6]	40.37 [4] 7.94 [1]
Rating Total: 19	**Excellent**	**Excellent**	**Good**	**Excellent**
References	1- Stephens 1997, 2- Stephens and Moghaddas 2005, 3- Finney et al. 2007, 4- Van Wagtendonk 1996, 5- Stephens et al. 2009, 6- Lydersen et al. 2017, 7- Krofcheck et al. 2018, 8- Pollet and Omi 2002			

3.1.6 Results

Low thinning resulted in some crown type fire formation, which indicated a full crown fire was not developed; tree crowns had only torching of canopy layers (Stephens 1997). Fire intensity had mixed results when low thinning was compared to canopy thinning (Stephens 1997). However, a higher percentile (90^{th}) was simulated, which may explain the result of one high intensity value in canopy thinning (Stephens and Moghaddas 2005). ROS was had the lowest values in low thinning except when compared to PF or thinning with PF (Stephens and Moghaddas 2005, Van Wagtendonk 1996). Severity resulted in an 18.4% tree mortality decrease from the mean reduction percentage (Krofcheck et al. 2018). Low thinning accumulated a total of 14 points in the fuel reduction rating system.

Canopy thinning resulted in surface fire type formation (Stephens 1997, Stephens and Moghaddas 2005). Intensity results were not consistent between studies when canopy thinning was compared to low thinning (Van Wagtendonk 1996, Stephens 1997). However, two different fire weather percentiles were used, the 75^{th} percentile (Stephens 1997) and the 90^{th} percentile (Stephens and Moghaddas 2005). ROS was the second highest in canopy thinning, with selective or clear-cut thinning resulting in the highest ROS. Severity was typically reduced when canopy thinning was applied, however differences were unclear due to minimal details in the study (Pollet and Omi 2002). Canopy thinning accumulated a total of 13 points in the fuel reduction rating system.

Selective thinning resulted in a small amount of crown type fire (Stephens 1997). Selective thinning resulted in the highest fire intensity rates compared to all other fuel reduction methods (Stephens 1997, Stephens and Moghaddas 2005). ROS was also the highest value compared to all other methods. Severity resulted in 80% small tree mortality ranging from 2–25 cm DBH. Severity data was collected from real-world data, while the data for ROS and intensity were simulated (Lydersen et al. 2017, Stephens and Moghaddas 2005, Stephens 1997). Selective thinning accumulated the lowest score in the fuel reduction rating system, a total of six points.

PF or thinning with PF resulted in surface fire only (Stephens 1997). Intensity was the lowest value compared to all other fuel reduction methods (Stephens 1997, Van Wagtendonk 1996). ROS was the lowest value compared to all other methods; additionally, one study suggest ROS was decreased for PF or thinning with PF (Finney et al. 2007, Van Wagtendonk 1996). Severity was reported to be unchanged to low in one real-world study (Lydersen et al. 2017); unchanged to low represents the lowest level of fire severity compared to all other methods. PF or thinning with PF was the highest fuel reduction score overall.

3.1.7 Discussion

PF and thinning with PF produced the most ideal fire behavior characteristics when considering wildfire risk management in mixed conifer ecosystems in the SN. PF and thinning with PF were shown to decrease severity, ROS, and intensity. Results may be due to PF's close resemblance to natural wildfire (Sugihara et al. 2006). PF can remove surface fuels by up to 50%, which decreases surface fire intensity and ROS (Stephens and Moghaddas 2005). The surface fuel reduction caused by PF may explain why the three mechanical thinning methods resulted in higher intensities: surface fuel (slash) was not removed after the three mechanical thinning methods (Figure 10) (Stephens 1997). The positive effects of PF are widely acknowledged in the scientific community; however, it is important to reevaluate findings as our planet's ecosystems alter due to climate change and fire suppression (Pollet and Omi 2002, Stephens and Moghaddas 2005, Stephens 1997, Stephens et al. 2009, Kilgore and Sando 1975). PF or thinning with PF are most effective when considering wildfire risk management in mixed conifer ecosystems in the SN.

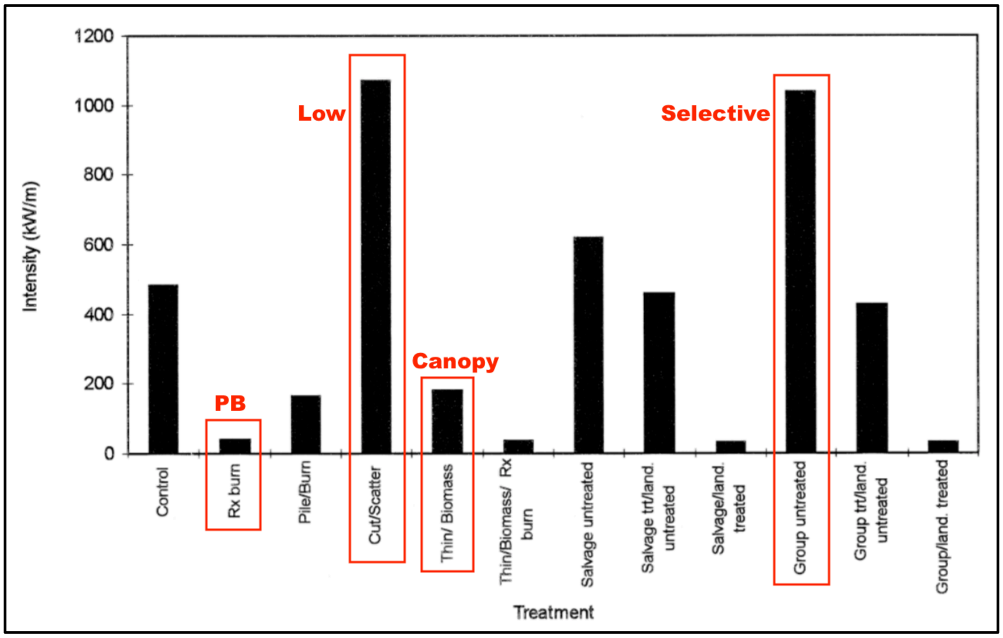


Figure 10. Simulated fire intensity from the 95th percentile weather conditions with the four treatments used in this research paper highlighted in red. Data from the 95th percentile was not used in this synthesis; however, results were comparable to the 75th percentile (Stephens 1997).

Selective thinning resulted in some of the highest risk conditions in fire behavior characteristics. Values were numerically higher for ROS and intensity compared to all other metrics. This may be due to stand structural changes when harvesting trees from one age-group, especially marketable trees. Marketable timber tends to have higher DBH values, and if all marketable trees are harvested, the smaller trees become the dominant canopy layer (Van Wagtendonk 1996). Small tree canopies have a lower canopy base height that can be more prone to crown fire ignition and higher severity levels (Van Wagtendonk 1996). Based on the selective thinning metric data, forest structural changes caused by selective thinning can produce high wildfire risk implications.

3.2 Goal II: Climate Change Resilience

Climate change is one of the most complex and imminent threats to the Sierra Nevada (Thorne 2016). Because climate is the driving force behind the existence and functionality of all ecosystems, it is projected to have rippling impacts on fire regimes, wildlife habitat, and water resources throughout mountain ecosystems, especially the SN (Thorne et al. 2018, Hurteau and

North 2008). Therefore, managing ecosystem resilience and resistance for current and future climate effects must be fused into fuel reduction processes. Resilience is a measurement of an ecosystem's ability to return to its original state after a disturbance (DeClerck et al. 2006). Resistance is a measure of an ecosystem's ability to maintain its original state during a disturbance (DeClerck et al. 2006). Unfortunately, climate change will be a persistent "new normal" and not an occasional disturbance, so many ecosystems may ultimately face elimination or complete alteration from their original state (Berg and Hall 2017, Cayan, D. R. et al. 2007). Models are projecting an increase in fire size, severity, and frequency beyond any natural fire regimes (Abatzoglou and Williams 2016). Fire frequency has been shown to be increasing over time into subalpine ecosystems, which are not adapted to shorter return intervals (Schwartz et al. 2015) (Figure 11). Additionally, plant and animal species may shift to higher elevations and latitudes to avoid warmer temperatures and locate food sources (Galbreath et al. 2009, Wright et al. 2016). Numerous high alpine species may be eradicated completely due to a lack of habitat (Stewart et al. 2015).

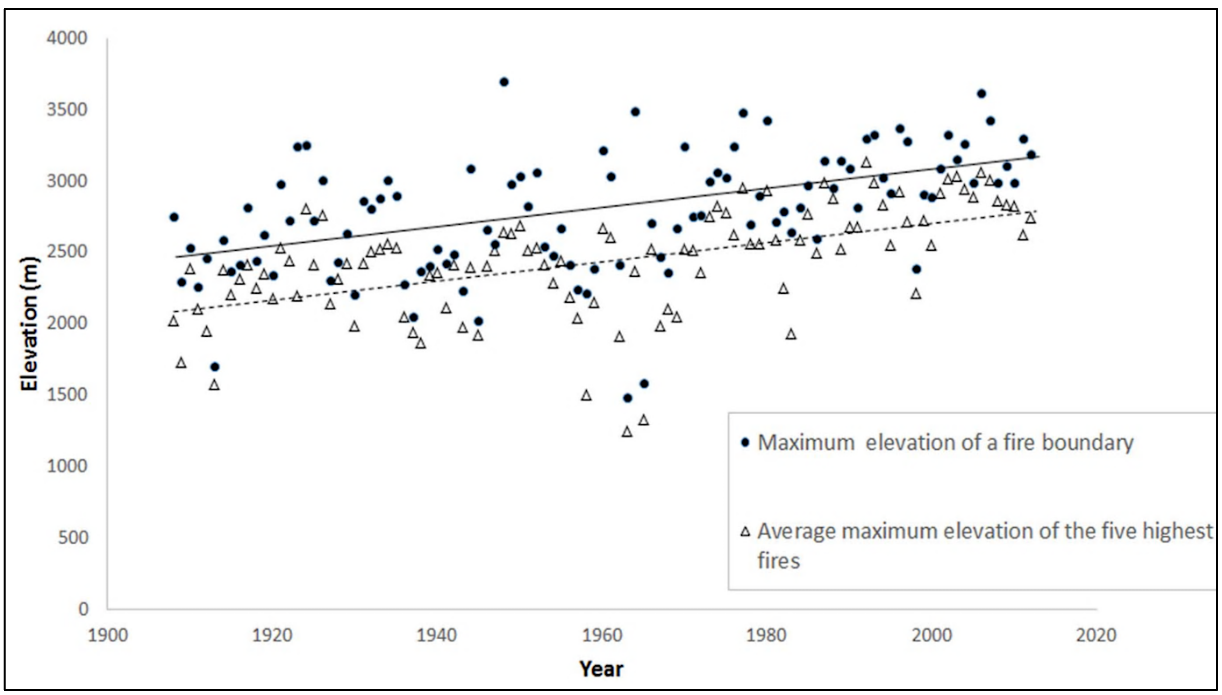


Figure 11. Increase in fire frequency at higher elevations in the SN (Schwartz et al. 2015).

Although there is still uncertainty regarding future climate trajectories and precipitation patterns in California, the scientific community is highly confident temperatures will increase (Figure 12) (Choi et al. 2016, Berg and Hall 2017, Cayan et al. 2007, Morelli et al. 2011). Warming temperatures are leading to decreases in precipitation falling as snow, which is already having measurable impacts on snowpack levels (Figure 13) (Cayan et al 2001, Berg and Hall 2017). Snow water equivalent (SWE) is the amount of water within the snowpack and is commonly used to asses snowpack levels in the SN (Berg and Hall 2017). In the multiyear drought from 2011 to 2015, middle elevation SWE levels in the mixed conifer zone were shown to decreased by 26% due to anthropogenic climate change (Berg and Hall 2017). Loss of snowpack will have a rippling effect on plant and animal species in the mixed conifer forest (Safeeq et al. 2016). For example, *Pinus* ssp. have been found to flower earlier with less snow (Morelli et al. 2011). Earlier flowering will have influences on flowering cues for pollinators to emerge, which could misalign timelines for a whole host of species that depend on pollinators for reproduction (Morelli et al. 2011). Furthermore, global climate models that result in the most extreme climate scenarios (>4°C) show a decrease in percent cover in mixed conifer forest in the SN by 25% (Lenihan et al. 2003). Adopting fuel reduction methods that accommodate climate-driven ecological implications is critical for the health and survival of our forests.

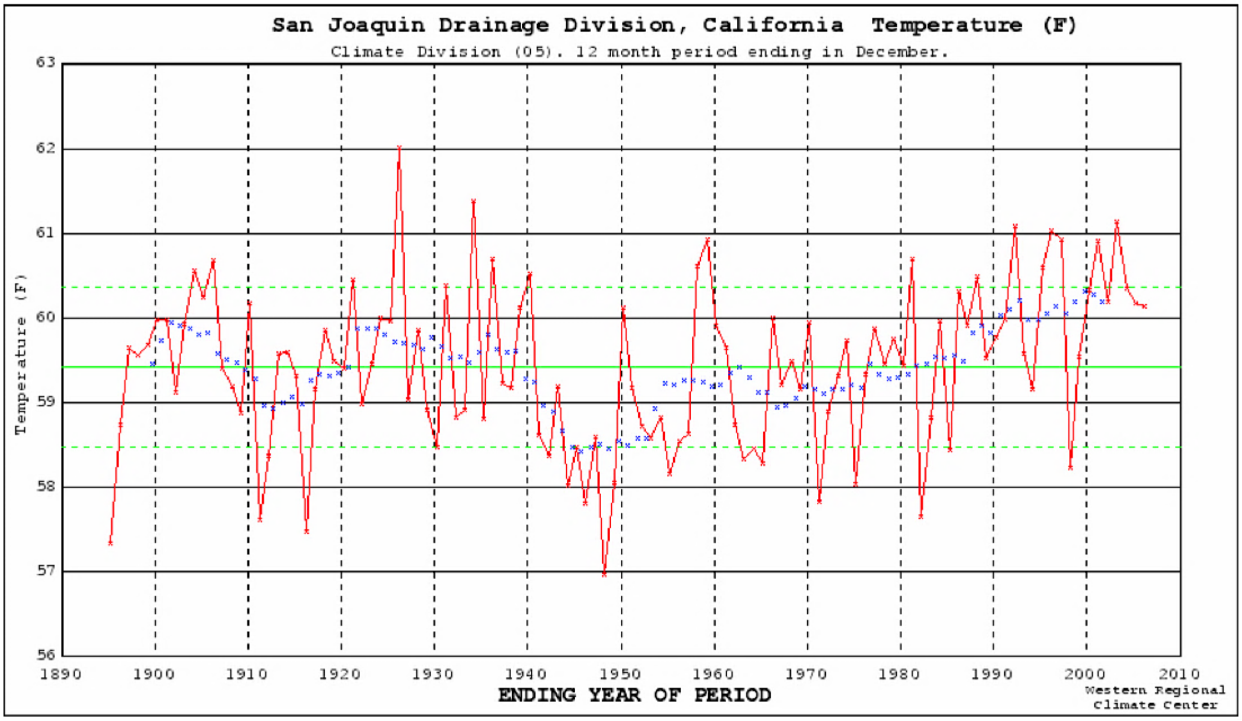


Figure 12. Temperature trends from 1895–2005 within the western central SN. The twelve-month average temperature is represented by red, the ten-year running mean by blue, and the mean by green dashes. Increasing trends are shown in the ~1970–1990s (Davey et al. 2007).

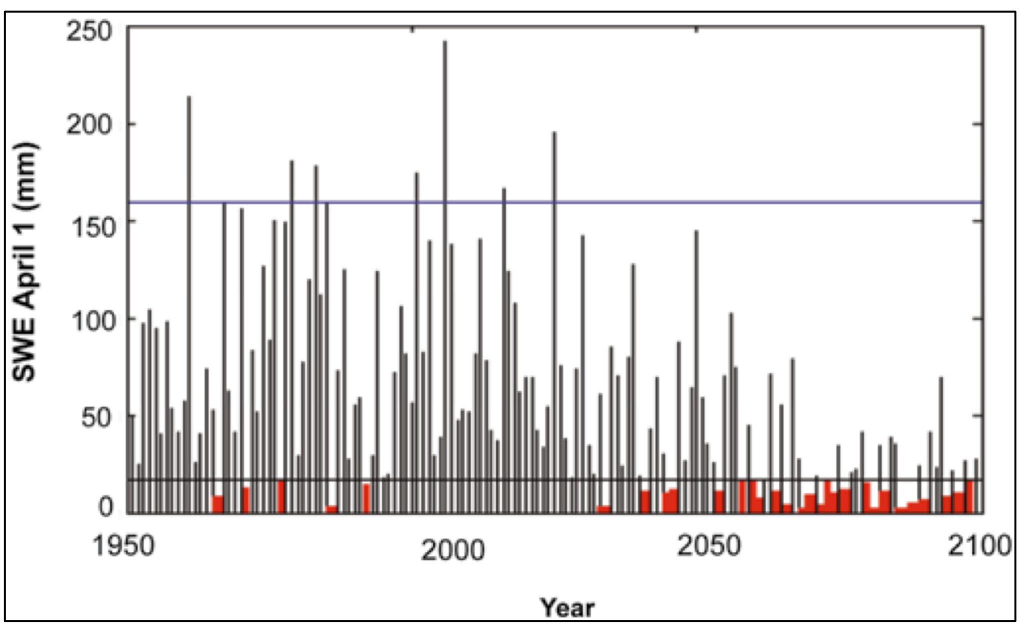


Figure 13. SWE levels in the northern SN from 1950 to 2100. The black and blue horizontal lines represent the 10th and the 90th percentiles, respectively. The red bars represent any occurrences of SWE levels that are less than the 10th percentile (Cayan et al. 2009).

Snowpack slowly releases water into the soil, which is necessary for volumetric water content (VWC); increased VMC provides water for tree roots into the spring (Bales et al. 2011). However, heavy rainfall typically causes surface runoff—water moving over the soil surface—which can produce more fluctuation in VWC (Bales et al. 2011). Low levels of VWC causes trees to become dehydrated and stressed, which lowers their ability to fight off bark beetle infestation (Stephens et al. 2018). In the drought years of 2012–2016, an estimate of 129 million trees died due to bark beetle infestation and drought (Restaino et al. 2019).

The last century has seen a decline in freezing temperatures at night, which are necessary to kill bark beetle larvae (Morelli et al. 2011). Additionally, high populations of bark beetles can easily move through homogenous tree stands (Raffa et al. 2008). These homogenous tree stands, sometimes created by selective thinning or clear-cutting, are less resilient to disturbance and are more prone to high fire severity (Raffa et al. 2008, Stevens et al. 2016). Moreover, when many of the largest trees are killed by bark beetles, there are consequences for climate change resilience (Stevens et al. 2016). The largest trees are the most fire resistant, provide habitat for wildlife, and increase forest structural diversity (Stevens et al. 2016). Although bark beetles are a native species and play an important role in the ecosystem by naturally decreasing tree density, their high populations have dramatically increased dead fuel loads (Stevens et al. 2016). High densities of trees will lead to less VWC in the soil, which can decrease tree vigor and overall resistance to insects and fire (Restaino et al. 2019). Many dead trees killed by bark beetle are not removed due to time and funding constraints, vastly increasing the area of dry, fire susceptible forest stands (Stevens et al. 2016). These standing dead trees, also known as snags, provide some wildlife habit; however, when entire forests are filled with snags there can be damaging effects on ecosystem functionality (Stevens et al. 2016).

Temperatures are warming in mixed conifer forests; therefore, it is imperative to reexamine current fuel reduction methods to ensure the best science is incorporated and applied (Davey et al. 2007). Four climate change resiliency characteristics—species richness/percent cover, seed germination, tree mortality, and VWC—were used as measuring metrics for the efficacy of the four thinning methods (Table 6). These metrics are widely used to assess climate change resilience and resistance (Berg and Hall 2017, Cayan et al. 2007, Bradford and Bell 2017, Abatzoglou and Williams 2016). Each metric has been assessed based on the four types of thinning methods that were implemented.

Table 6. Information on ten references used in the climate change synthesis table (Table 7).

n=10	Reference	Metric	Treatment	Study Description	Location	General Timeframes	Pg.
1	Zald et al. 2008	Gemination, VWC	Low Thin, Canopy Thin, PF w/ UT and OT	Evaluated, sown seed germination, VWC, burn disturbance, soil disturbance. Data averaged from 2002–2005.	Teakettle Experimental Forest, 80 km east of Fresno	Thinning done between 2000 and 2001	173 Table 2 175 Fig 5a
2	Walker et al. 2012	Gemination	PF	Sapling inventory, seedling counts.	USDA Forest Service Lake Tahoe Basin Management Unit	2 years	756, 757 Table 2
3	Kobziar et al. 2006	Mortality	PF	Three prescribed burns with 1300 trees. Mean values taken for seven species.	University of California Blodgett Forest Research Station	Began in fall 2002–2003, eight months total	3229 Table 3
4	Lydersen et al. 2019	Mortality	Selection and PF	Province Creek site data only. Used % dead basal area.	Kings River Experimental Watersheds, Southern SN	2012–2017	506 Table 5
5	Kane et al. 2010	Species Richness	Low Thin (HAND) and PF w/ mastication	Evaluated species richness of native understory plant communities in five plots.	Challenge Experimental Forest, Plumas National Forest	2001–2006	214
6	Ryu et al. 2009	VWC	Low Thin, Canopy Thin, PF	Evaluated soil respiration, temp, C, N, and litter depths in 18 plots.	Teakettle Experimental Forest	2001–2002	1327 Fig 1
7	Collins et al. 2007	Species Richness	PF and Low Thin w/ mastication	Effects of treatments were evaluated the next year after	University of California Blodgett Forest Research Station	2002–2003	107 Fig 6

				treatments in 12 plots.			
8	Goodwin et al. 2018	Percent Cover	Low Thin, Canopy Thin, PF with thinning, PF only	Evaluated plant community in 18 units. Used data from 2003–2017.	Teakettle Experimental Forest	2000–2017	62 Fig 5
9	Wayman and North 2007	VWC	Low Thin, Canopy Thin, PF w/ UT and OT P only	Evaluated relationship between plant community/ composition in 12 plots.	Teakettle Experimental Forest	2000–2003	36
10	Maloney et al. 2008	Mortality	Low Thin, Canopy Thin, PF w/ UT and OT PF only	Compared pathogen mortality after burn treatments to unburned treatments in 18 four ha plots	Teakettle Experimental Forest	2000–2005	3013

3.2.1 Metric: Species Richness and Percent Cover

Species richness is a measurement of the number of different plant or animal species within a given area (Collins et al. 2017). Ecosystems with higher measurements of species richness tend to be more stable and resistant to disturbance events (Collins et al. 2017, Tilman and Downing 1994, Isbell et al. 2015). This may be due to increased redundancy in ecosystem functionality roles by different species (Tilman and Downing 1994). However, most dry forests in the western US have low species richness of trees in comparison to forests in wetter climates (Richter et al. 2019). Additionally, understory percent cover has been used to understand overall plant community structure (Kane et al. 2010). It must be noted that percent cover does not represent species richness; percent cover only represents the percent of plant biomass covering bare ground in a unit area, not the number of different species (Goodwin et al. 2018). For simplicity, the metrics have been represented together in the synthesis.

3.2.2 Metric: Seed Germination

The germination of tree and plant species after wildfire disturbance represents the beginning stages of succession in the mixed conifer forest (Walker et al. 2012). Seedlings germinate best in the mixed conifer ecosystem after low to moderate severity fires (Richter et al. 2019). While fire regimes in the mixed conifer forest have historically included low to mid severity fires, fire severity has increased due to high densities of vegetation resulting from fire suppression (Lydersen et al. 2019). Mechanical fuel reduction methods are also a type of anthropogenic disturbance and should aim to restore the regenerative processes of a natural wildfire. The germination and survival of *Pinus* ssp. generally takes priority in restoration processes, which aims to return forest composition to shade-intolerant, fire-resistant species (Zald et al. 2008). Thus, it has been noted as a benefit in the synthesis table when *Pinus* ssp. are germinated rather than *A. concolor* and *C. decurrens*. However, the germination of all tree species has been considered ideal for forest diversity and heterogeneity considerations (Richter et al. 2019). Germination survival rates have been used as a metric of measurement for climate change resilience.

3.2.3 Metric: Tree Mortality

Understanding the causes of tree mortality after fuel reduction can help forest managers choose better methods based on forest structure and composition (Kobziar et al. 2006). PF can have some unintended tree mortality if duff and surface layers ignite and damage roots and tree boles (Maloney et al. 2008, Kobziar et al. 2006). However, despite the existence of some tree mortality caused by fuel reduction, thinning decreases mortality rates in the long term (Lydersen et al. 2019). Dense stands of fire-intolerant trees that are all similar age can suffer from high mortality rates in high-severity wildfires (Lydersen et al. 2019). Fuel reduction can add structure and age diversity to tree stands, which decreases a wildfire's ability to decimate landscapes (Lydersen et al. 2019). Additionally, bark beetle damage is lower in forests treated with fuel reduction than in untreated forests (Restaino et al. 2019).

The percent of tree mortality after fuel reduction should be quantified and documented for each type of thinning method to ensure applications are performing with efficiency (Kobziar et al. 2006). Mortality has been quantified and documented in the synthesis table by percentage. Mortality has represented the third metric for this analysis.

3.2.4 Metric: Soil Water Content

Soil moisture has been found to be associated with high species diversity (Wayman and North 2007). VWC in soil is a limiting factor in mixed conifer forests and can hinder understory richness during drought conditions (Wayman and North 2007). Higher levels of moisture are also associated with increased soil carbon, due to the presence of high levels of microorganisms and root biomass (Ryu et al. 2009). It should be noted that higher soil moisture levels have been shown to support more shade-tolerant tree species, which may not be a priority in some restoration goals (Zald et al. 2008). Nonetheless, VWC represents the fourth metric for this analysis and provides necessary information for assessing fuel reduction efficacy when considering climate change impacts.

3.2.5 Climate Resilience Synthesis

The following climate resilience synthesis was used to gather and display data on the effectiveness of each thinning method in relation to a measurable metric (Table 7).

Table 7. The comparison of four fuel reduction methods with four climate resilience metrics. Average values were extracted from ten peer-reviewed studies. Rating system: very poor (1), poor (2), fair (3), good (4), and excellent (5).

Fuel Reduction Method	Species Richness and Percent Cover	Seed Germination	Tree Mortality %	VWC %
Low Thinning	7.2 sp./m^2 [1] 10–25% [3] Increase (+0.01 difference) richness after one year [2]	Lowest germinant survival values [5]	9 [6]	10.9–15.9 [9] 16 [10] 8.2 [5]
Rating Total: 11	**Fair**	**Very Poor**	**Excellent**	**Poor**
Canopy Thinning	5–25% [3]	High germinant survival values [5]	12.7 [6]	11.8–21.4 [9] 16.6 [10] 9.7 [5]
Rating Total: 13	**Poor**	**Fair**	**Good**	**Excellent**
Selective Thinning	Inconclusive	Inconclusive	21.4 [7]	Inconclusive
Rating Total: 2	**Inconclusive**	**Inconclusive**	**Poor**	**Inconclusive**
PF or Thinning w/ PF	11.3 sp./m^2 [1] (w/ mastication prior to burn) 15–43% [3] Reduction (-1 difference) in richness after one year [2]	Increased *Pinus* ssp. by 49% and significantly (p=0.04) decreased *A. concolor* and *C. decurrens* [4] Highest germinant survival values for PF w/ canopy thin [5]	18–56 [6] 24.7–25.9 [7] 26.6 [8]	10.6–17 [9] 17–17.5 [10] 6.5 [5]
Rating Total: 14	**Good**	**Excellent**	**Very Poor**	**Good**
References	1- Kane et al. 2010, 2- Collins et al. 2007, 3- Goodwin et al. 2018, 4- Walker et al. 2012, 5- Zald et al. 2008, 6- Maloney et al. 2008, 7- Lydersen et al. 2019, 8- Kobziar et al, 9- Wayman and North 2007, 10- Ryu et al. 2009			

3.2.6 Results

Low thinning resulted in the second highest species richness and percent cover in two out of three studies (Kane et al. 2010, Goodwin et al. 2018). However, results that saw increased species richness due to low thinning did not demonstrate significant differences when comparing PF or thinning with PF (Collins et al. 2007). Low thinning resulted in the lowest rates of seed germination (Zald et al. 2008). However, low thinning also resulted in the lowest mortality, at 9% (Maloney et al. 2008). Low thinning resulted in two of the lowest values of VWC out of three studies, suggesting that soil water retention was suboptimal in low thinning (Ryu et al. 2009, Wayman and North 2007). Mortality rates were the most optimal out of all the metrics for low thinning and seed germination was the least optimal. Both VWC and species richness and percent cover exhibited mixed results. Overall, low thinning resulted in the third highest levels of points for climate resilience.

Canopy thinning resulted in the lowest percent cover (Goodwin et al. 2018). Seed germination and survival post canopy thinning operations was second only to seed survival after PF or thinning with PF (Zald et al. 2008). Tree mortality was the second lowest post canopy thinning (Maloney et al. 2008). VWC was the highest post canopy thinning in two studies (Wayman and North 2007, Zald et al. 2008), and second highest in one study, after PF with thinning (Ryu et al. 2009). Canopy thinning resulted in the second highest levels of points for climate resilience.

Selective thinning resulted in the third lowest tree mortality value, after low and canopy thinning. Selective thinning was not evaluated for the other three metrics—species richness and percent cover, seed germination, and VWC—as it is difficult to find studies that incorporate selective thinning into their methods, perhaps because there is a wide range of selective thinning methods. Selective thinning garnered two points for tree mortality, all other metrics were inconclusive. Selective thinning has been not counted in the combined synthesis due to inconclusive results.

PF or thinning with PF resulted in the highest value of species richness per square meter (Kane et al. 2010). Percent cover has mixed results (Goodwin et al., 2018, Collins et al. 2007). The highest germinant survival rates were found in PF or thinning with PF (Zald et al. 2008); data also showed increased survival rates for *Pinus* ssp., but decreased germinant survival rates for shade-tolerant trees (Walker et al. 2012). Studies that compared PF or thinning with PF with other fuel reduction methods, tree mortality was the highest for PF; however, some data showed a similar value that was not compared with other fuel reduction methods (Lydersen et al. 2019, Maloney et al. 2008, Kobziar et al. 2006). VWC results after PF or thinning with PF were inconclusive; they were found to be both higher and lower in various studies in comparison to low and canopy thinning methods (Wayman and North 2007, Ryu et al. 2009, Zald et al. 2008). PF or thinning with PF garnered the highest level of points.

3.2.7 Discussion

Based on the ratings of the evaluated metrics, PF and thinning with PF produced the most ideal climate change resiliency conditions. Although PF and thinning with PF had the highest tree mortality, all other metrics were rated good to excellent. PF has historically caused higher mortality rates, especially during drought conditions and serious bark beetle infestations (Maloney et al. 2008, Lydersen et al. 2019). However, post PF seed germinant survival rates were excellent, particularly for *Pinus* ssp. (Kobziar et al. 2006). Moreover, it has been shown that canopy cover percentage may have a relationship with PF mortality. *P. ponderosa* mortality has a positive correlation with canopy cover percentage, suggesting that canopy thinning, and PF may complement each other when conducting *P. ponderosa* restoration (Figure 14) (Kobziar et al. 2006). Results indicate that PF may not be optimal for all mixed conifer forest conditions, so a mixture of fuel reduction methods should be applied when drought or bark beetle infestations are prominent. PF should be paired with specific mechanical thinning methods based on stand structure, species, density, and average DBH values. Overall, PF or thinning with PF are the fuel reduction treatments that best prepare the mixed conifer forest in the SN for climate change resiliency

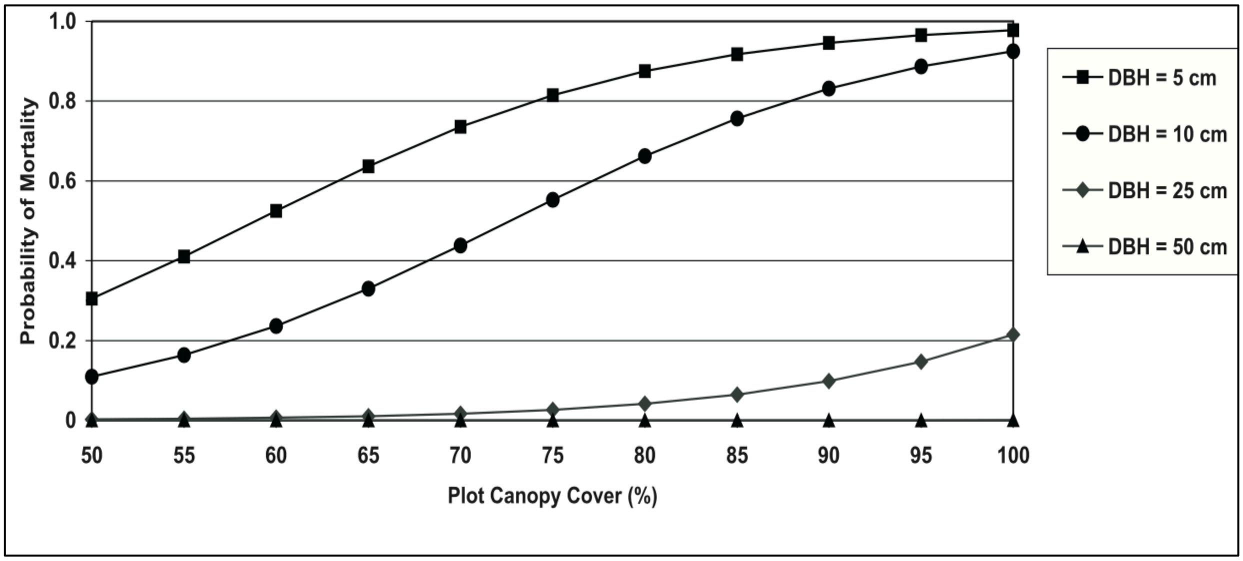


Figure 14. Mortality rates for *P. ponderosa* by DBH based on canopy cover percentage (Kobziar et al. 2006).

Low thinning indicated very poor conditions for seed germinant survival. Low thinning may not provide enough sunlight penetration through the canopy for seedlings (Zald et al. 2008). Additionally, low thinning can allow larger DBH shade-tolerant species to remain on-site, which have been shown to shed substantial amounts of seeds. These seeds can produce reoccurrences of dense, shade-tolerant stands of saplings, limiting space and sunlight for shade-intolerant *Pinus* sp. (Zald et al. 2008). Maintaining a dense surface covering of shade-tolerant trees is in direct opposition to many restoration goals in the fire-suppressed mixed conifer forests of the SN (Zald et al. 2008, Walker et al. 2012).

3.3 Combined Comparison Analysis

The two goals of this paper were to identify the best possible fuel reduction methods in light of wildfire risk and climate change. To this end, I have combined the values of the five-point rating system for each fuel reduction method to come up with total values. The highest values represent the best possible results based on the metrics evaluated. Low values represent poor outcomes based on the metrics. In the discussion I elaborate on the possible reasons for the differences in these findings.

3.3.1 Combined Synthesis

The results from the wildfire risk analysis and the climate resiliency analysis were combined to assess the overall preferred method for forest management (Table 8).

Table 8. Rating sums of the four metrics for Goals I and II. Rating: very poor (1), poor (2), fair (3), good (4), and excellent (5).

Fuel Reduction Method	Goal I: Wildfire Risk	Goal II: Climate Change	Rating Total
Low Thinning	14	11	25
Canopy Thinning	13	13	26
Selective Thinning	6	inconclusive	8
PF or Thinning w/ PF	19	14	33

3.3.2 Combined Results

Low thinning resulted in the second highest value for wildfire risk and the third highest value for climate change. Low thinning had the third highest total value. Canopy thinning resulted in the third highest value for wildfire risk and the second highest value for climate change. Canopy thinning had the second highest total value. Selective thinning resulted in the lowest values for wildfire risk and inconclusive results for climate change (Figure 15). PF or thinning with PF resulted in the highest values for both metrics and the highest total value (Figure 16).

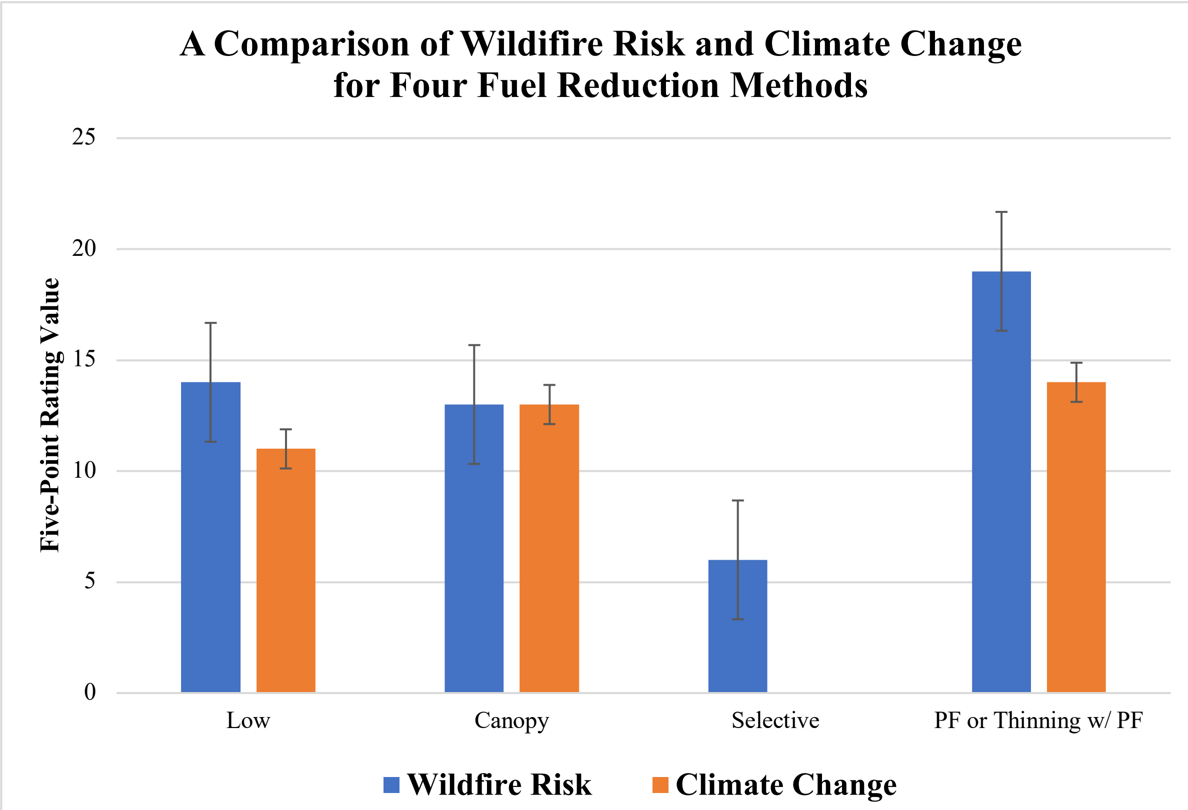


Figure 15. Separate values for Goals I and II (wildfire risk and climate change) based on sums of the five-point rating system for the four fuel reduction methods. Note that selective thinning does not included a climate change value due to inconclusive results. Error bars show data variability. Ratings: very poor (1), poor (2), fair (3), good (4), and excellent (5).

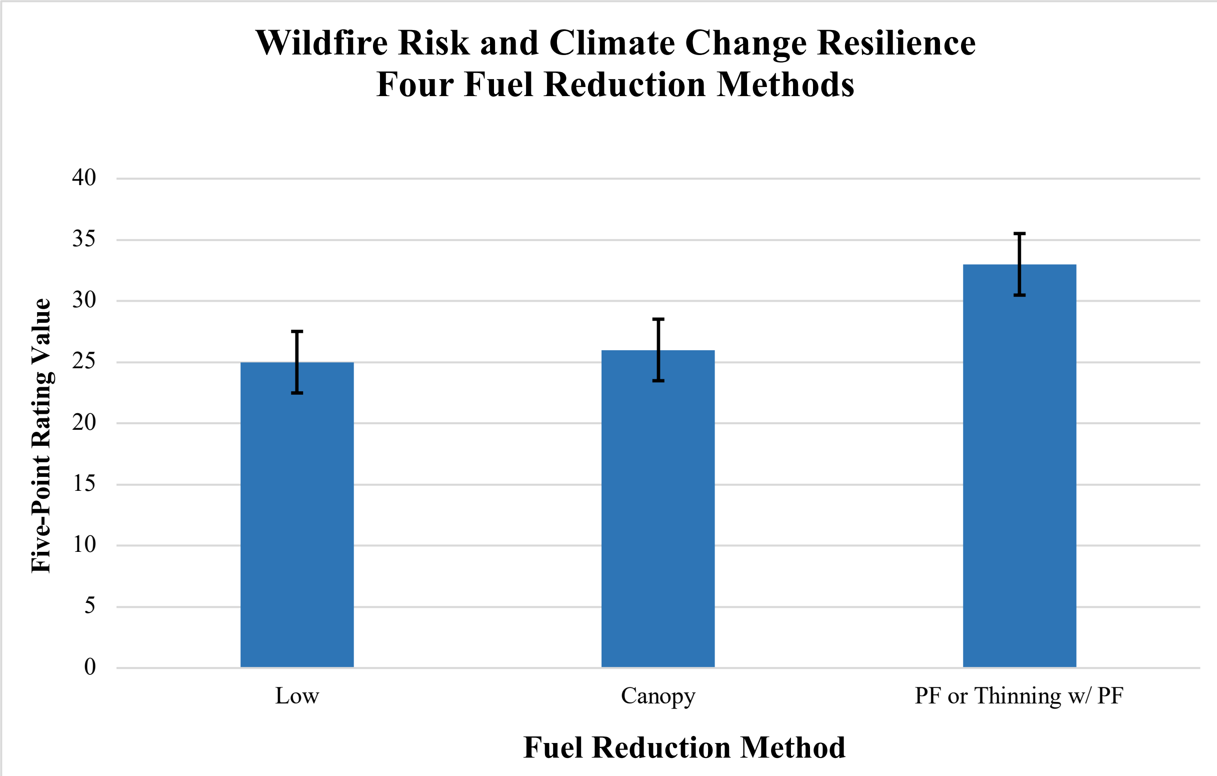


Figure 16. Total combined values based on the sums of the five-point rating system for Goals I and II. Selective thinning is not included due to inconclusive results for climate change resilience. Ratings: very poor (1), poor (2), fair (3), good (4), and excellent (5).

4 Combined Discussion and Management Recommendations

Data found on the metrics for the four fuel reduction methods indicates that each method has benefits and disadvantages. For example, selective thinning did not always result in crown fires, the type of fire that is the most destructive and severe (Stephens 1997 and Stephens and Moghaddas 2005); and PF or thinning with PF did not always result in the highest species richness (Collins et al. 2007). Overall, each method should be tailored to and selected for specific environmental conditions and proximity to manmade structures. There is no "one size fits all" fuel reduction method. However, considering the two goals, the eight metrics used, and the 18 peer-reviewed studies selected for this analysis, PF or thinning with PF has delivered the most optimal results for reducing wildfire risk and building climate change resiliency in mixed conifer ecosystems. This suggests that PF or thinning with PF is an effective fuel reduction choice in most instances.

Selecting a thinning method to be paired with PF should be based on environmental conditions. If the density of understory trees will cause potential issues, such as high ROS (Van Wagtendonk et al. 1997) or decreased species richness (Goodwin et al. 2018), then low thinning may be the best option to pair with PF. Another important aspect of forest management is that PF should not be conducted where there are high amounts of ladder fuels, as this can cause a controlled fire to climb to the canopy and become an uncontrolled crown fire. Based on the findings of this research, canopy thinning can also provide benefits in certain situations. Canopy thinning paired with PF can increase VWC (Wayman and North 2007) and lower the chances of crown fire ignition. Increased VWC due to the removal of canopy layers may be attributed to the ability of rain to contact and percolate into the soil, instead of being intercepted by dense canopy vegetation (Rodríguez-Calcerrada et al. 2008).

Selective thinning resulted in the lowest values for wildfire risk and inclusive results for climate resiliency. It is important to consider that the type of selective thinning I evaluated for this project was mainly based on extracting markable timber. This type of selective thinning creates evenly spaced, homogenous stands with trees of similar ages and heights (Jain et al. 2012, Agee and Skinner 2005). There exist alternative, more sustainable, methods of selective thinning that encourage increased biodiversity and structural heterogeneity (Ares et al. 2010). One example is a relatively new thinning and restoration method known as variable density thinning (VDT) (Knapp et al. 2012). VDT has been shown to recreate structural and spatial patchiness that mimics pre-European forest structure (Knapp et al. 2012). VDT removes tree stands in patches ranging in size from 0.04 to 0.2 ha, creating one open patch every 0.81 ha (Knapp et al. 2012). These patches are randomly scattered throughout an area, as they would be in the aftermath of a wildfire, and generally aim to leave larger trees (Knapp et al. 2012). By only removing patches of trees and leaving non-thinned areas, forest heterogeneity is maintained and wildfire refugia is created for plant and animal species (Blomdahl et al. 2019, Knapp et al. 2017, Knapp et al. 2012). VDT could potentially replace marketable timber harvest methods of selective thinning in order to meet forest restoration goals.

Known limitations in this research include wide ranges in timeframes, design, and thinning methods for each study evaluated. For instance, all the studies that compared low thinning metrics did not apply the exact same low thinning prescription; a limitation of this research is that it contains broad interpretations of each thinning method. Also, the sample size of the studies evaluated was limited due to the specific characteristics necessary to qualify for each thinning method. Lastly, it is important to remember that the four main thinning methods are broad descriptions. There are many specific thinning method prescriptions that exist in forest management that fall under the definitions of the four methods.

My findings address two major contemporary problems with forest management, increases in destructive wildfire, and Earth's changing climate. These two issues are deeply intertwined and managing their effects will be a major endeavor for current and future generations. Before applying the findings of this research to forest management, it is critical that managers have in-depth knowledge of the basic principles of ecology (Odum and Barrett 1971). Forest managers must also understand fire ecology and the role it plays in biogeochemical cycles. In addition, good forest management requires respecting human communities, valuing traditional ecological knowledge, and having the ability to procure often limited financial resources. My research suggests that PF paired with low or canopy thinning is one of the best options in most cases. However, to apply any fuel reduction method identified in my research, forest management must assess the overall forest landscape and select the best fit method based on local conditions, biodiversity, proximity to human structures, and available resources.

4.1 Managing for Biodiversity

Fuel reduction poses a threat to many species, and forest managers must plan to mitigate for endangered and threatened wildlife. The SN supports a large portion of California's wildlife biodiversity (Figure 17). Roughly 400 different animal species, including many migrating birds, occur in the SN (Ruth 1996). The largest percentage of this animal biodiversity encompasses 232 bird species; there are 112 mammal species and 57 reptile and amphibian species (Ruth 1996). Wildlife plays many important roles in ecosystem function in the mixed conifer forest; for example, black bears, rodents, and birds all contribute to seed and fungal spore dispersal (Enders and Vander Wall 2012, Sollmann et al. 2016). Seeds can completely pass through animals' digestive tracts and be deposited in feces, and the surrounding fecal matter can assist in seed gemination (Enders and Vander Wall 2012).

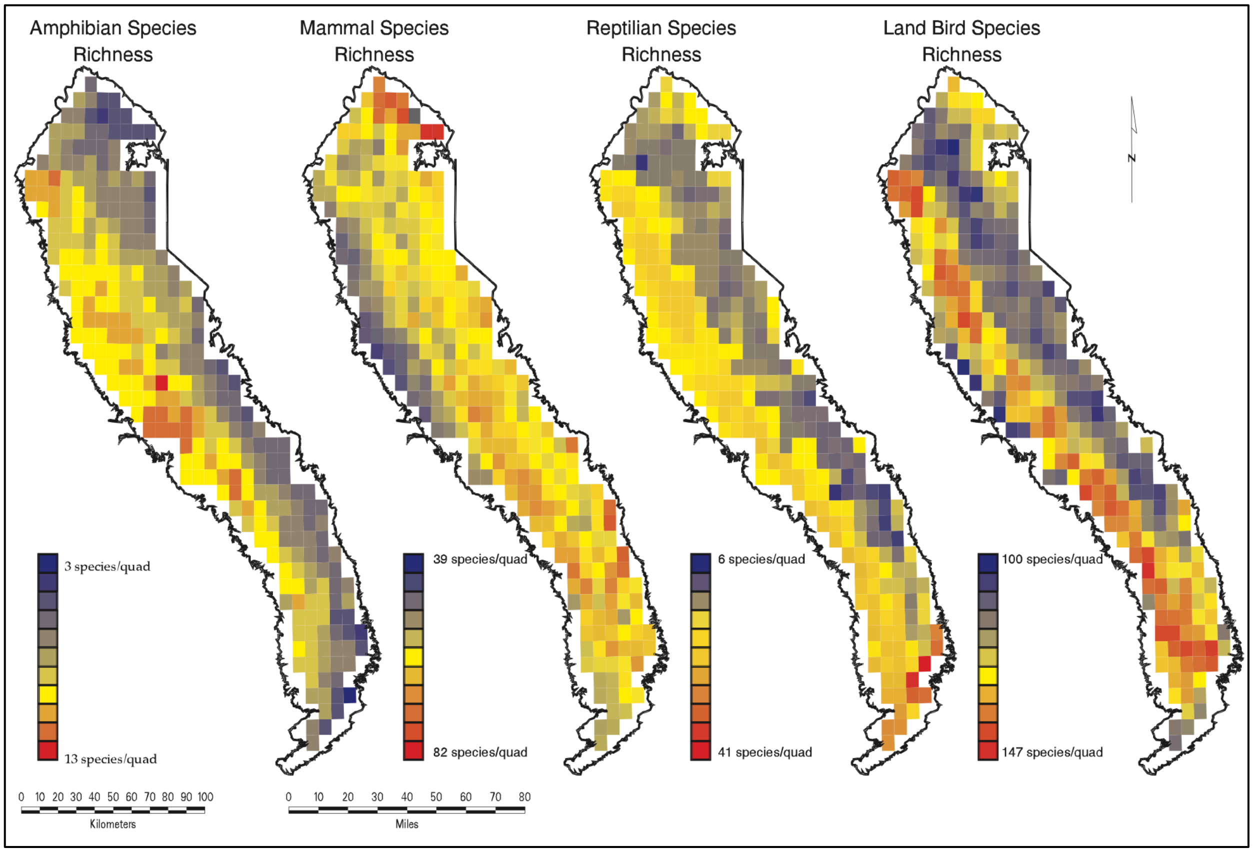


Figure 17. Terrestrial species richness in the SN (Ruth 1996).

Looking at the effects of fuel reduction on mammals should be a priority for forest managers, both for the species themselves and because loss of mammal biodiversity can have rippling effects across ecosystems. Federal and state listed species such as the federally threatened *Pekania pennanti* (fisher) require habitat conservation in mixed conifer forests; however, fuel reduction can have both negative and positive impacts on *P. pennanti* (Blomdahl et al. 2019). *P. pennanti* gives birth in the spring, so PF conducted in spring can have harmful effects on offspring and female *P. pennanti* (Blomdahl et al. 2019). *P. pennanti* prefer canopy cover generally associated with late-seral forest as their main habitat (Sweitzer et al. 2016). PF can create woody structural diversity for denning availability, but canopy thinning can have negative impacts on *P. pennanti* nesting and denning occurrences (Blomdahl et al. 2019).

Since avian diversity is the largest portion of biodiversity in the SN, it is vital to implement avian habitat conservation into fuel reduction plans (White et al. 2013). Many avian species play important roles in mixed conifer ecosystems, especially upper trophic-level species such as *Strix occidentalis occidentalis* (California spotted owl) (White et al. 2013). Predator-prey relationships tend to impact vegetation density (Denno et al. 2005). For example, decreases in *S. occidentalis* populations can permit *Neotoma fuscipes* (dusky-footed woodrat) populations to increase due to reduced predation. *N. fuscipes* can have drastic effects on fuel loads in forest environments by constructing middens out of dry surface debris (Stephens et al. 2014). *S. occidentalis* prefers late-seral forest habitat, and fuel reduction practices can sometimes alter these habitats (Gallagher et al. 2019, Stephens et al. 2014, White et al. 2013). Alterations can decrease canopy cover, which has been shown to have negative impacts on *S. occidentalis* foraging habitat (Gallagher et al. 2019). However, some species benefit from fuel reduction, including *Picoides arcticus* (black-backed woodpecker), which thrives in recently burned open canopy habitats (Blakey et al. 2020). Wildlife in the SN is an essential component in the mixed conifer ecosystem and has irreplaceable intrinsic value. Thus, evaluating the literature on fuel reduction impacts on sensitive species should be included in all fuel reduction goals.

4.2 Further Research Needs

More studies are needed, with a larger sample sizes, to make inferences about any significant benefits from PF on *S. occidentalis* foraging opportunities (Gallagher et al. 2019). *S. occidentalis* may require patches of slash and woody debris for habitat, therefore removing all slash during fuel reduction operations may have negative impacts on *S. occidentalis* habitat (Weatherspoon e al. 1992). However, the threat of large, high severity mega-wildfires could pose a greater risk to *S. occidentalis* by destroying mixed conifer ecosystems entirely, so it is still imperative to conduct fuel reduction operations (Weatherspoon et al. 1992).

Biomass or slash produced from mechanical thinning methods left on-site has caused controversy in forest management (Agee and Skinner 2005). While biomass or slash plays an ecological role in nutrient cycling, erosion protection, and wildlife habitat, slash poses a risk by creating higher fuel loads for wildfire (Evans 2016). Further investigation is required to fully understand the ecological role of downed woody debris in mixed conifer ecosystems. It would be beneficial for wildlife conservation if the scientific community identified an ideal amount (height, bulk density, cover area, fuel moisture) of slash or biomass for mixed conifer landscapes. For example, finding a balance of slash or biomass amounts that could provide ecological roles and minimize wildfire risk.

The role of selective thinning for marketable timber on climate change resilience in mixed conifer ecosystems should be further studied. Data for this research was limited, which created an inconclusive result for climate change resilience. Research centered on the effects of selective thinning on climate change and correlated ecological fundamentals like, species richness, percent cover, seed gemination, and VWC should be considered for the inevitable future of transitioning climates in the SN.

4.3 Conclusion

As wildfires increase in severity, size, and elevation (Schwartz et al. 2015, Abatzoglou and Williams 2016), they are also increasingly responsible for destroying communities and causing human fatalities (Calkin et al. 2014). Although action on climate change is necessary in order to arrest this pattern, it does not provide an immediate answer for addressing this complex issue. In order to lessen the severity of these wildfires, we must reduce fuel loads. If done correctly, fuel reduction can be used to produce marketable timber, restore forest ecosystems, and build climate change reliance (Stephens et al. 2005). Many studies have evaluated and reported on the various aspects of fuel reduction and its broad array of effects on the environment (Wayman and North 2007, Kane et al. 2007, Stevens et al. 2016, Innes et al. 2006, Krofcheck et al. 2009). However, there are not many studies that have reviewed the scientific literature in a comprehensive analysis that consider both wildfire risk and climate change resiliency (Stephens et al. 2012, Knapp et al. 2009). The research in this study was undertaken in hopes of shedding more light on the efficacy of different fuel reduction methods and their role in reducing wildfire risk and building ecosystems in preparation for climate change. Because the SN has evolved with wildfire over the course of thousands of years, it was anticipated that PF is one of the best methods for fuel reduction as my results indicated; however, there are certain circumstances in which PF is not ideal. Low and canopy thinning methods also appear to play important roles in certain environmental conditions, so it is vital to evaluate forest structure and landscape before selecting a fuel reduction method. Fuel reduction must be undertaken with the environment in mind because our forests are limited resources and provide essential ecosystem services. Changes to climate, ecosystems, and habitats are occurring at ever-increasing rates, as is biodiversity loss; this all threatens the ongoing existence of the natural resources that we rely on (Abatzoglou and Williams 2016, Thorne et al. 2018). In 50 to 100 years, all forest ecosystems may be very different from those of today (Odion et al. 2014, Dolanc et al. 2014). Land managers must keep conservation and preservation at the forefront of their professional goals, if they are to help maintain healthy, biodiverse forest ecosystems that can adapt to a rapidly changing planet.

LITERATURE CITED

Abatzoglou, J. T., and A. P. Williams. 2016. Impact of Anthropogenic Climate Change on Wildfire Across Western US Forests. *Proceedings National Academy of Science* **113**:11770.

Abella, S. R., and J. D. Springer. 2015. Effects of Tree Cutting and Fire on Understory Vegetation in Mixed Conifer Forests. *Forest Ecology and Management* **335**:281-299.

Agee, J. K., and C. N. Skinner. 2005. Basic Principles of Forest Fuel Reduction Treatments. *Forest Ecology and Management* **211**:83-96.

Anderson, R. S., and S. L. Carpenter. 1991. Vegetation Change in Yosemite Valley, Yosemite National Park, California During Protohistoric Period. *Madroño* **38**:1-13.

Ares, A., A. R. Neill, and K. J. Puettmann. 2010. Understory abundance, species diversity and functional attribute response to thinning in coniferous stands. *Forest Ecology and Management* **260**:1104-1113.

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

Barth, M. A. F., A. J. Larson, and J. A. Lutz. 2015. A forest reconstruction model to assess changes to Sierra Nevada mixed conifer forest during the fire suppression era. *Forest Ecology and Management* **354**:104-118.

Berg, N., and A. Hall. 2017. Anthropogenic warming impacts on California snowpack during drought. *Geophysical Research Letters* **44**:2511-2518.

Bettinger, P., K. Boston, J. P. Siry, and D. L. Grebner. 2017. Chapter 2 - valuing and characterizing forest conditions. *Forest Management and Planning (Second Edition)* :21-63.

Blakey, R. V., R. B. Siegel, E. B. Webb, C. P. Dillingham, M. Johnson, and D. C. Kesler. 2020. Multi-scale Habitat Selection by Northern Goshawks (*Accipiter gentilis*) in a Fire-prone Forest. *Biological Conservation* **241**:108348.

Blomdahl, E. M., C. A. Kolden, A. J. H. Meddens, and J. A. Lutz. 2019. The importance of small fire refugia in the central Sierra Nevada, California. *Forest Ecology and Management* **432**:1041-1052.

Blomdahl, E. M., C. M. Thompson, J. R. Kane, V. R. Kane, D. Churchill, L. M. Moskal, and J. A. Lutz. 2019. Forest structure predictive of fisher (*Pekania pennanti*) dens exists in recently burned forest in Yosemite, California. *Forest Ecology and Management* **444**:174-186.

Bradford, J. B., and D. M. Bell. 2017. A window of opportunity for climate-change adaptation: Easing tree mortality by reducing forest basal area. *Frontiers in Ecology and the Environment* **15**:11-17.

Calkin, D. E., J. D. Cohen, M. A. Finney, and M. P. Thompson. 2014. How risk management can prevent future wildfire disasters in the wildland-urban interface. *Proceedings National Academy of Science USA* **111**:746.

Cayan, D., M. Dettinger, H. Hidalgo, T. Das, E. Maurer, P. D. Bromirski, R. E. Flick, and M. Tyree. 2009. Climate Change Scenarios and Sea-level Rise Estimates for the California 2009 Climate Change Scenarios Assessment.

Cayan, D., A. L. Luers, G. Franco, M. Hanemann, B. Croes, and E. Vine. 2007. Overview of the California climate change scenarios project. *Climatic Change* **87**:S1-S6.

Choi, J., J. Lu, S. Son, D. M. W. Frierson, and J. Yoon. 2016. Uncertainty in future projections of the North Pacific subtropical high and its implication for California winter precipitation change. *Journal of Geophysical Research: Atmospheres* **121**:795-806.

Christensen, N. L. 2014. An historical perspective on forest succession and its relevance to ecosystem restoration and conservation practice in North America. *Forest Ecology and Management* **330**:312-322.

Collins, B. M., D. L. Fry, J. M. Lydersen, R. Everett, and S. L. Stephens. 2017. Impacts of different land management histories on forest change. *Ecological Applications* **27**:2475-2486.

Collins, B. M., J. J. Moghaddas, and S. L. Stephens. 2007. Initial changes in forest structure and understory plant communities following fuel reduction activities in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management* **239**:102-111.

Davey, C., D. B. Simeral, and K. T. Redmond. 2007. Weather and Climate Inventory, National ParkService, Sierra Nevada network. Natural Resource Technical Report NPS/SIEN/NRTR—2007/042. National Park Service, Fort Collins, Colorado.

Davis, J., D. Coleman, J. Gracely, R. Gaschnig, and M. Stearns. 2012. Magma accumulation rates and thermal histories of plutons of the Sierra Nevada batholith, CA. *Contributions to Mineralogy & Petrology* **163**:449-465.

DeClerck, F. A. J., M. G. Barbour, and J. O. Sawyer. 2006. Species richness and stand stability in conifer forests of the Sierra Nevada. *Ecology* **87**:2787-2799.

Denno, R. F., D. L. Finke, and G. A. Langellotto. 2005. Direct and indirect effects of vegetation structure and habitat complexity on predator-prey and predator-predator interactions. *Ecology of Predator-prey Interactions* :211-239.

DeSiervo, M. H., E. S. Jules, and H. D. Safford. 2015. Disturbance response across a productivity gradient: Postfire vegetation in serpentine and non-serpentine forests. *Ecosphere* **6**.

DeSilva, R., and R. S. Dodd. 2019. Fragmented and isolated: Limited gene flow coupled with weak isolation by environment in the paleoendemic giant sequoia (*Sequoiadendron giganteum*). *American Journal of Botany.*

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

Dow, C. B., B. M. Collins, and S. L. Stephens. 2016. Incorporating resource protection constraints in an analysis of landscape fuel-treatment effectiveness in the Northern Sierra Nevada, CA. *Environmental Management* **57**:516-530.

Enders, M. S., and S. B. Vander Wall. 2012. Black bears *Ursus americanus* are effective seed dispersers, with a little help from their friends. *Oikos* **121**:589-596.

Evans, A. M. 2016. Potential ecological consequences of forest biomass harvesting in California. *Journal of Sustainable Forestry* **35**:1-15.

Finney, M. A., R. C. Seli, C. W. McHugh, A. A. Ager, B. Bahro, and J. K. Agee. 2007. Simulation of long-term landscape-level fuel treatment effects on large wildfires. *International Journal of Wildland Fire* **16**:712-727.

Forest History Society. 2019. United states forest service fire suppression. URL: https://Foresthistory.org/research-explore/us-forest-service-history/policy-and-law/fire-u-s-forest-service/u-s-forest-service-fire-suppression/

Galbreath, K. E., D. J. Hafner, and K. R. Zamudio. 2009. When cold is better: Climate-driven Elevations Shifts Yield Complex Patterns of Diversification and Demography in Alpine Specialist (American Pika, *Ochotona princeps*). *Evolution* **63**:2848-2863.

Gallagher, C. V., J. J. Keane, P. A. Shaklee, H. A. Kramer, and R. Gerrard. 2019. Spotted owl foraging patterns following fuels treatments, Sierra Nevada, California. *Journal of Wildlife Management* **83**:487-501.

Gassaway, L. 2007. Native American Fire Patterns in Yosemite Valley: A cross-disciplinary study. pages 29–39 in R.E. masters and K.E.M. galley (eds.). proceedings of the 23rd tall timbers fire ecology conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, Florida.

Goodwin, M. J., M. P. North, H. S. J. Zald, and M. D. Hurteau. 2018. The 15-year post-treatment response of a mixed-conifer understory plant community to thinning and burning treatments. *Forest Ecology and Management* **429**:617-624.

Hamilton, J. 1992. Visions of the range of light. Sierra **77**:76.

Henderson, S. B., M. Brauer, Y. C. MacNab, and S. M. Kennedy. 2011. Three measures of forest fire smoke exposure and their associations with respiratory and cardiovascular health outcomes in a population-based cohort. *Environmental Health Perspectives* **119**:1266-1271.

Huber, N. K. 1987. The Geologic Story of Yosemite National Park URL: pubs.er.usgs.gov.

Innes, J. C., M. P. North, and N. Williamson. 2006. Effect of thinning and prescribed fire restoration treatments on woody debris and snag dynamics in a sierran old-growth, mixed-conifer forest. *Canadian Journal of Forest Research* **36**:3183-3193.

Isbell, F., D. Craven, J. Connolly, M. Loreau, B. Schmid, C. Beierkuhnlein, T. M. Bezemer, C. Bonin, H. Bruelheide, and E. De Luca. 2015. Biodiversity increases the resistance of ecosystem productivity to climate extremes. *Nature* **526**:574-577.

Jain, T. B., M. A. Battaglia, H. -. Han, R. T. Graham, C. R. Keyes, J. S. Fried, and J. E. Sandquist. 2012. A comprehensive guide to fuel management practices for dry mixed conifer forests in the Northwestern United States. USDA Forest Service - General Technical Report RMRS-GTR :1-331.

Johnson, C., S. Chhin, and J. Zhang. 2017. Effects of climate on competitive dynamics in mixed conifer forests of the Sierra Nevada. *Forest Ecology and Management* **394**:1-12.

Kane, J. M., J. Morgan Varner, E. E. Knapp, and R. F. Powers. 2010. Understory vegetation response to mechanical mastication and other fuels treatments in a ponderosa pine forest. *Applied Vegetation Science* **13**:207-220.

Keeley, J. E. 1992. Demographic structure of California chaparral in the long-term absence of fire. *Journal of Vegetation Science* **3**:79-90.

Keeley, J. E., and A. Syphard. 2016. Climate Change and Future Fire Regimes: Examples from California. *Geosciences* **6**:37.

Keeley, J. E., and A. Syphard. 2019. Twenty-first century California, USA, wildfires: fuel-dominated vs. wind-dominated fires. *Fire Ecology* **15**:24.

Kelsey, R. 2019. Wildfires and forest resilience: The case for ecological forestry in the Sierra Nevada. unpublished report of the Nature Conservancy. Sacramento, California. 12 p.

Kilgore, B. M., and R. W. Sando. 1975. Crown-fire potential in a sequoia forest after prescribed burning. *Forest Science* **21**:83-87.

Klimaszewski-Patterson, A., and S. A. Mensing. 2016. Multi-disciplinary approach to identifying native American impacts on late Holocene forest dynamics in the Southern Sierra Nevada Range, California. *Anthropocene* **15**:37-48.

Knapp, E. E., Estes, B. L., and Skinner, C. N. 2009. Ecological effects of prescribed fire season: a literature review and synthesis for managers.

Knapp, E. E., M. North, M. Benech, and B. Estes. 2012. The variable-density thinning study at Stanislaus-Tuolumne Experimental Forest.

Knapp, E. E., J. M. Lydersen, M. P. North, and B. M. Collins. 2017. Efficacy of variable density thinning and prescribed fire for restoring forest heterogeneity to mixed-conifer forest in the central Sierra Nevada, CA. *Forest Ecology and Management* **406**:228-241.

Kobziar, L., J. Moghaddas, and S. L. Stephens. 2006. Tree mortality patterns following prescribed fires in a mixed conifer forest. *Canadian Journal of Forest Research* **36**:3222-3238.

Konrad, S. K., and D. H. Clark. 1998. Evidence for an Early Neoglacial Glacier Advance from Rock Glaciers and Lake Sediments in the Sierra Nevada, California. *Arctic and Alpine Research* **30**:272-284.

Krasnow, K. D., D. L. Fry, and S. L. Stephens. 2017. Spatial, temporal and latitudinal components of historical fire regimes in mixed conifer forests, California. *Journal of Biogeography* **44**:1239-1253.

Krofcheck, D. J., M. D. Hurteau, R. M. Scheller, and E. L. Loudermilk. 2018. Prioritizing forest fuels treatments based on the probability of high-severity fire restores adaptive capacity in Sierran Forests. *Global Change Biology* **24**:729-737.

Laughlin, D. C., J. D. Bakker, M. T. Stoddard, M. L. Daniels, J. D. Springer, C. N. Gildar, A. M. Green, and W. W. Covington. 2004. Toward reference conditions: Wildfire effects on flora in an old-growth ponderosa pine forest. *Forest Ecology and Management* **199**:137-152.

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

Liang, S., M. D. Hurteau, and A. L. Westerling. 2017. Response of Sierra Nevada forests to projected climate–wildfire interactions. *Global Change Biology* **23**:2016-2030.

Loffland, H. L., J. S. Polasik, M. W. Tingley, E. A. Elsey, C. Loffland, G. Lebuhn, and R. B. Siegel. 2017. Bumble bee use of post-fire chaparral in the Central Sierra Nevada. *The Journal of Wildlife Management* **81**:1084-1097.

Lydersen, J. M., B. M. Collins, and C. T. Hunsaker. 2019. Implementation constraints limit benefits of restoration treatments in mixed-conifer forests. *International Journal of Wildland Fire* **28**:495-511.

Lydersen, J. M., B. M. Collins, M. L. Brooks, J. R. Matchett, K. L. Shive, N. A. Povak, V. R. Kane, and D. F. Smith. 2017. Evidence of fuels management and fire weather influencing fire severity in an extreme fire event: *Ecological Applications* **27**:2013-2030.

Maloney, P. E., T. F. Smith, C. E. Jensen, J. Innes, D. M. Rizzo, and M. P. North. 2008. Initial tree mortality and insect and pathogen response to fire and thinning restoration treatments in an old-growth mixed-conifer forest of the Sierra Nevada, California. *Canadian Journal of Forest Research* **38**:3011-3020.

Meyer, M. D., and H. D. Safford. 2011. Giant sequoia regeneration in groves exposed to wildfire and retention harvest. *Fire Ecology* **7**:2-16.

Morelli, T., M. McGlinchy, and R. Neilson. 2011. A climate change primer for land managers: an example from the Sierra Nevada. *Res.Pap.PSW-RP-262.Albany, CA: US Department of Agriculture, Forest Service, Pacific Southwest Research Station.44 p* **262**.

Muir, J. 1979. John of the Mountains: The Unpublished Journals of John Muir. University of Wisconsin Press.

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.

Odum, E. P., and G. W. Barrett. 1971. Fundamentals of ecology. Saunders Philadelphia.

Pausas, J. G., and J. E. Keeley. 2019. Wildfires as an ecosystem service. *Frontiers in Ecology and the Environment* **17**:289-295.

Peterson, D. L., and M. J. Arbaugh. 1992. Mixed conifer forests of the Sierra Nevada. Pages 433-459 *In* Olson, R. K., D. Binkley, and M. Böhm, editors. The response of western forests to air pollution, Springer New York, New York.

Pollet, J., and P. Omi. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. *International Journal of Wildland Fire* 11(1):1-10
DOI: 10.1071.

Raffa, K. F., B. H. Aukema, B. J. Bentz, A. L. Carroll, J. A. Hicke, M. G. Turner, and W. H. Romme. 2008. Cross-scale Drivers of Natural Disturbances Prone to Anthropogenic Amplification: The Dynamics of Bark Beetle Eruptions. *Bioscience* **58**:501-517.

Reid, C. E., M. Brauer, F. H. Johnston, M. Jerrett, J. R. Balmes, and C. T. Elliott. 2016. Critical review of health impacts of wildfire smoke exposure. *Environmental Health Perspectives* **124**:1334-1343.

Restaino, C., D. J. N. Young, B. Estes, S. Gross, A. Wuenschel, M. Meyer, and H. Safford. 2019. Forest structure and climate mediate drought-induced tree mortality in forests of the Sierra Nevada, USA. *Ecological Applications* **29**:e01902.

Richter, C., M. Rejmánek, J. E. D. Miller, K. R. Welch, J. Weeks, and H. Safford. 2019. The species diversity × fire severity relationship is hump-shaped in semiarid yellow pine and mixed conifer forests. *Ecosphere* **10**:e02882.

Ritchie, M. W., J. Zhang, and T. A. Hamilton. 2013. Aboveground tree biomass for *Pinus ponderosa* in northeastern California. *Forests* **4**:179-196.

Rodríguez-Calcerrada, J., S. Mutke, J. Alonso, L. Gil, J. A. Pardos, and I. Aranda. 2008. Influence of overstory density on understory light, soil moisture, and survival of two underplanted oak species in a Mediterranean montane Scots pine forest. *Forest Systems* **17**:31-38.

Ruth, L. 1996. Conservation and controversy: National forest management, 1960 –95. P. 145-162 in Sierra Nevada ecosystem project: Final report to congress, vol. II. University of California Centers for Water and Wildland Resources, Davis, CA.

Ryu, S., A. Concilio, J. Chen, M. North, and S. Ma. 2009. Prescribed burning and mechanical thinning effects on belowground conditions and soil respiration in a mixed-conifer forest, California. *Forest Ecology and Management* **257**:1324-1332.

Safeeq, M., S. Shukla, I. Arismendi, G. E. Grant, S. L. Lewis, and A. Nolin. 2016. Influence of winter season climate variability on snow–precipitation ratio in the western United States. *International Journal of Climatology* **36**:3175-3190.

Safford, H., and J. Stevens. 2017. Natural range of variation for yellow pine and mixed-conifer forests in the Sierra Nevada, southern cascades, and Modoc and Inyo national forests, California, USA. gen. tech. rep. PSW- GTR-256. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 229.

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

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

Schweickert, R. A. 1981. Tectonic evolution of the Sierra Nevada range. the geotectonic development of California. vol 1. PP. 87-131; ISBN 0-13-3539-385; ILL.

Smith, S. J., and R. S. Anderson. 1992. Late Wisconsin paleoecologic record from swamp lake, Yosemite national park, California. Quaternary Research **38**:91-102.

Sollmann, R., A. M. White, G. L. Tarbill, P. N. Manley, and E. E. Knapp. 2016. Landscape heterogeneity compensates for fuel reduction treatment effects on northern flying squirrel populations. *Forest Ecology and Management* **373**:100-107.

Steel, Z. L., M. J. Koontz, and H. D. Safford. 2018. The changing landscape of wildfire: burn pattern trends and implications for California's yellow pine and mixed conifer forests. *Landscape Ecology* **33**:1159-1176.

Steel, Z. L., B. Campos, W. F. Frick, R. Burnett, and H. D. Safford. 2019. The effects of wildfire severity and pyro-diversity on bat occupancy and diversity in fire-suppressed forests. Scientific Reports **9**.

Stephens, S. L. 1997. Evaluation of the effects of silvicultural and fuels treatments on potential fire behavior in Sierra Nevada mixed-conifer forests. *Forest Ecology & Management* **97**:21.

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

Stephens, S. L., B. M. Collins, C. J. Fettig, M. A. Finney, C. M. Hoffman, E. E. Knapp, M. P. North, H. Safford, and R. B. Wayman. 2018. Drought, Tree Mortality, and Wildfire in Forests Adapted to Frequent Fire. *BioScience* **68**:77-88.

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

Stephens, S. L., J. D. McIver, R. E. J. Boerner, C. J. Fettig, J. B. Fontaine, B. R. Hartsough, P. L. Kennedy, and D. W. Schwilk. 2012. The effects of forest fuel-reduction treatments in the united states. *BioScience* **62**:549-560.

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

Stephens, S. L., and J. J. Moghaddas. 2005. Silvicultural and reserve impacts on potential fire behavior and forest conservation: Twenty-five years of experience from Sierra Nevada mixed conifer forests. *Biological Conservation* **125**:369-379.

Stephens, S. L., and M. A. Finney. 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. *Forest Ecology & Management* **162**:261.

Stevens, J. T., B. M. Collins, J. W. Long, M. P. North, S. J. Prichard, L. W. Tarnay, and A. M. White. 2016. Evaluating potential trade-offs among fuel treatment strategies in mixed-conifer forests of the Sierra Nevada. *Ecosphere* **7**.

Stewart, J. A. E., J. D. Perrine, L. B. Nichols, J. H. Thorne, C. I. Millar, K. E. Goehring, C. P. Massing, and D. H. Wright. 2015. Revisiting the past to foretell the future: Summer temperature and habitat area predict pika extirpations in California. *Journal of Biogeography* **42**:880-890.

Sugihara, N. G., J. W. Van Wagtendonk, and J. Fites-Kaufman. 2006. Fire as an ecological process. *Fire in California's Ecosystems* :58-74.

Sweitzer, R. A., B. J. Furnas, R. H. Barrett, K. L. Purcell, and C. M. Thompson. 2016. Landscape fuel reduction, forest fire, and biophysical linkages to local habitat use and local persistence of fishers (*Pekania pennanti*) in Sierra Nevada mixed-conifer forests. *Forest Ecology and Management* **361**:208-225.

Swetnam, T. W. 1993. Fire history and climate change in giant sequoia groves. *Science* **262**:885-889.

Teraoka, J., P. Mantgem, and C. Keyes. 2017. Low thinning and crown thinning of two severities as restoration tools at Redwood National Park.

Thorne, J., J. Thorne, H. Choe, H. Choe, P. Stine, P. Stine, J. Chambers, J. Chambers, A. Holguin, A. Holguin, A. Kerr, A. Kerr, M. Schwartz, and M. Schwartz. 2018. Climate change vulnerability assessment of forests in the Southwest USA. *Climatic Change* **148**:387-402.

Tilman, D., and J. A. Downing. 1994. Biodiversity and stability in grasslands. *Nature* **367**:363-365.

United States Department of Agriculture. 2019. United states forest service. fire effects information system. URL: https://Www.feis-crs.org/feis/

Van Wagtendonk, J. W. 1996. Use of a deterministic fire growth model to test fuel treatments. in Sierra Nevada ecosystem project, final report to congress (vol. 2, pp. 1155-1166).

Van Wagtendonk, J. W. 2007. The history and evolution of wildland fire use. *Fire Ecology* 3(2): 3-17.

Van Wagtendonk, J. W., and J. Fites-Kaufman. 2006. Sierra Nevada bioregion. *Fire in California's Ecosystems* :264-294.

Wakabayashi, J., and T. L. Sawyer. 2001. Stream incision, tectonics, uplift, and evolution of topography of the Sierra Nevada, *California. Journal of Geology* **109**:539.

Walker, R. F., R. M. Fecko, W. B. Frederick, D. W. Johnson, and W. W. Miller. 2012. Seedling Recruitment and Sapling Retention Following Thinning, Chipping, and Prescribed Fire in Mixed Sierra Nevada Conifer. *Journal of Sustainable Forestry* **31**:747-776.

Wayman, R. B., and M. North. 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. *Forest Ecology and Management* **239**:32-44.

Weatherspoon, C. P., S. J. Husari, and J. W. Van Wagtendonk. 1992. Fire and fuels management in relation to owl habitat in forests of the Sierra Nevada and southern California. *Chapter 12 in: Verner, Jared; McKelvey, Kevin S.; Noon, Barry R.; Gutierrez, RJ; Gould, Gordon I.Jr.; Beck, Thomas W., Technical Coordinators.1992. The California spotted owl: a technical assessment of its current status.Gen.Tech.Rep.PSW-GTR-133.Albany, CA: Pacific Southwest Research Station, Forest Service, US Department of Agriculture; pp.247-260* **133**:247-260.

White, A. M., and J. W. Long. 2019. Understanding ecological contexts for active reforestation following wildfires. *New Forests* **50**:41-56.

White, A. M., E. F. Zipkin, P. N. Manley, and M. D. Schlesinger. 2013. Conservation of Avian Diversity in the Sierra Nevada: Moving beyond a Single-Species Management Focus. *PLOS ONE* **8**:e63088.

Wolf, J. 1964. An analysis of present-day terrestrial lapse rates in the western conterminous United States and their significance to paleo-altitudinal estimates. U.S. Geological Survey.

Wright, D. H., C. V. Nguyen, and S. Anderson. 2016. Upward shifts in recruitment of high-elevation tree species in the northern Sierra Nevada, California. California Fish and Game **102**:17-31.

Wrzesien, M. L., and T. M. Pavelsky. 2020. Projected changes to extreme runoff and precipitation events from a downscaled simulation over the Western United States. *Frontiers in Earth Science* **7**.

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