SIERRAN MIXED CONIFER FOREST WILDFIRES:
A BIODIVERSITY COMPARISON BETWEEN ACTIVE AND PASSIVE TIMBER LAND MANAGEMENT

by

Caitlin M. Dalby

A professional paper submitted in partial fulfillment
of the requirements for the degree

of

Master of Science

in

Land Resources and Environmental Sciences

MONTANA STATE UNIVERSITY
Bozeman, Montana

April 2019

Acknowledgement

This project would not have been possible without the support and guidance of many. I would like to extend my gratitude to my instructor and project advisor Dr. William Kleindl, who was a mentor to me for this project and at multiple points throughout my time in the graduate program. I would like to thank my advisory committee members Dr. Scott Powell, Robert Peterson, and Tracy Sterling for their patience and encouragement for me through this process. I would also like to express my appreciation for Dr. Lisa Rew for the inspiration and guidance on this project. In addition to the faculty members who supported me through this project, I would like to thank Eric O'Kelley, Lee Reno, Maureen Dalby, Kirsten Bovee, and Bennie Johnson for their field support. If it were not for their assistance, site selection and field data collection would have been an impossible task. Lastly, I would like to thank my parents, Maureen and Deron Dalby, and my siblings Zachary and Zoie Dalby, for their unwavering support and encouragement throughout this entire process.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

ABSTRACT

The mixed conifer forests of the Northern Sierras have great value in their biodiversity. Timber land management in this region varies by land owner and their objectives, including timber harvest and conservation. An increase in annual wildfires each year indicates a need to understand how different silvicultural strategies before and after fires affect the overall ecosystem biodiversity. Ecosystem functions and services can be affected by many factors involving anthropogenic activities in combination with fires. The aim of this study was to determine to what degree active (private timber company) and passive (United States Forest Service) management after a wildfire may affect plant biodiversity, and to compare those silvicultural approaches to active and passive pre-fire management. The study takes place within the vicinity of the 2012 Chips fire burn scar in Plumas County, California. Using tree canopy and plant species percent cover, in addition to presence and absence data, within frames and nested frames along 50 meter transects, statistical analyses revealed little significant difference between active and passive management. Analysis from data collected in this study concluded that tree canopy cover is significantly different under active post-fire management than under passive post-fire management and pre-fire conditions. There was not a significant difference in understory biodiversity (richness and evenness) among the four treatments. Dissimilarity in plant species composition was significant among the burned and unburned treatments, as well as between the two differently managed burned treatments. While the treatments were significantly dissimilar, there was not enough data collected to account for the high degree of variability seen in the data and so further data collection and analyses across multiple spatial and temporal scales would give better insight into the differences in biodiversity between treatments.

INTRODUCTION

The yellow pine-mixed conifer forest ecosystems of the Sierra Nevada and Southern Cascade region provide many services: provisioning (food, fuel, fiber), supporting (nutrient cycling), regulating (clean air and water) and cultural (urbanization, recreation)(Norris, 2012). These are important forests where biodiversity needs to be maintained. When managed for timber, silvicultural approaches in mixed conifer forests of the Sierra-Cascades depends on and varies by landowner and management goals (Battles et al., 2001). Mixed conifer forest of the Pacific Southwest region covers over 3 million hectares (figure 1) (Safford and Stevens, 2017), of the 6.24 million hectares of the Sierra Nevada bioregion (Davis and Stoms, 1996)**.** These species-rich forest ecosystems have understory vegetation that historically accounted for a majority of plant species in these communities (Easterday et al., 2018).

Fire has and continues to be a central theme in land management approaches, especially in regions that are becoming increasingly susceptible to the effects of climate change (Luce et al., 2012). In the Western United States, and the Southern Cascade-Northern Sierra region, fires are noticeably increasing in size and severity, bringing the discussion of timberland management to the forefront of policy decision-making on a local, state, national, and global scale (Mallek et al., 2013). The historic fire regime for this and surrounding regions is described as low to moderate intensity (Fites-Kaufman et al., 2005). The forests of the western United States changed dramatically after Euro-American settlement. Land use and anthropogenic disturbances shifted from minimal to landscape-scale ecosystem interference with fire-suppression and exploitation of resources for commercial purposes (Stephens and Collins, 2004). In dealing with over a century forest mismanagement, contemporary resource managers need to develop strategies that address landscape level anthropogenic alterations, especially coupled with climate change, to ensure social, economic, and environmental solvency into the future.

As we progress into the 21st century, forest managers are challenged to develop proactive plans that anticipate ecological needs in response to change to ensure ecosystem services persist despite natural and anthropogenic disturbances (Golladay et al., 2016). The task that resource managers are now faced with is to create strategies for adaptive management under global change of our forests at macroscales, which takes into consideration ecosystem assessment of structure and function, public opinion and the social-ecological interface, policy-making, and resource management (Kleindl et al., 2018). With a greater understanding of how human interaction and climate change alters the landscape under multiple conditions, land managers will be able to make more informed decisions when adaptively managing at different scales. This is particularly true for understanding the interaction between biodiversity, land management, and forest fires. The mixed conifer forests of the Sierra Nevada are susceptible to drought stress, changes to vegetation, and high-severity fire. Passive forest management may compromise resilience of these forests due to drought stress and fuel loads. Active management decreases fuel loads and stand density, thus reducing wildfire severity, but may also negatively affect forest conditions necessary to support sensitive species (North et al., 2017). Some studies suggest that by increasing structural heterogeneity associated with ecosystem resilience in these fire-dependent forests, multiple management objectives can be met that include fuels reduction, forest resilience, and ideal forest conditions to support sensitive species (Collins et al., 2011; Miller et al., 2012; Stephens and Collins, 2004; Stephens and Moghaddas, 2005).

Across the Sierra Nevada bioregion, management of timberland varies from passive to varying degrees of active management. Passive management, sometimes referred to as benign neglect, is an approach in which managers let nature take its course (Carey, 2006). Active techniques typical in managing mixed conifer forests of the region include harvesting, thinning, planting and seeding, controlling for pests, among other intentional approaches. From 1990 to 2014, 16.7% of US Forest Service land and 51.3% of private industrial forests were actively managed (North et al., 2017). This study attempts to evaluate the differences between active and passive silvicultural approaches in a comparison of post-fire understory vegetation biodiversity among four different treatment areas. Within a six-year-old burn scar, from the 2012 Chips Fire, one treatment is an active management approach by the Collins Pine Company (CPC). The other post-burn treatment is a passive management approach by the US Forest Service (USFS). As controls, two managed treatment areas (active CPC and passive USFS) are nearby outside the burn area. The data that were collected for this study were intended to empirically quantify differences in biodiversity to give, at the very least, an indication if more research should be done in this field within the realm of silviculture practice and anthropogenic interaction following wildfires. I hypothesized that pre-fire forest biodiversity and canopy cover will not be significantly different between active or passive management, but that there will be significant differences in plant biodiversity and tree canopy cover between the post-fire active and passive treatments and their pre-fire counterparts.

BACKGROUND

Biodiversity in Sierran Mixed Conifer Forests

It has long been recognized that biodiversity plays an important role in the resilience of ecosystems experiencing natural and anthropogenic disturbances (Fischer et al., 2006). Forest ecosystems provide resources for a large share of the world's biological diversity, as well as provide goods and services for society. Managing these forest ecosystems is becoming increasingly important, and the understanding of how regional and global perturbations affect the social-ecological productivity of these forests is essential for adaptive management with increasing human activity and climate change.

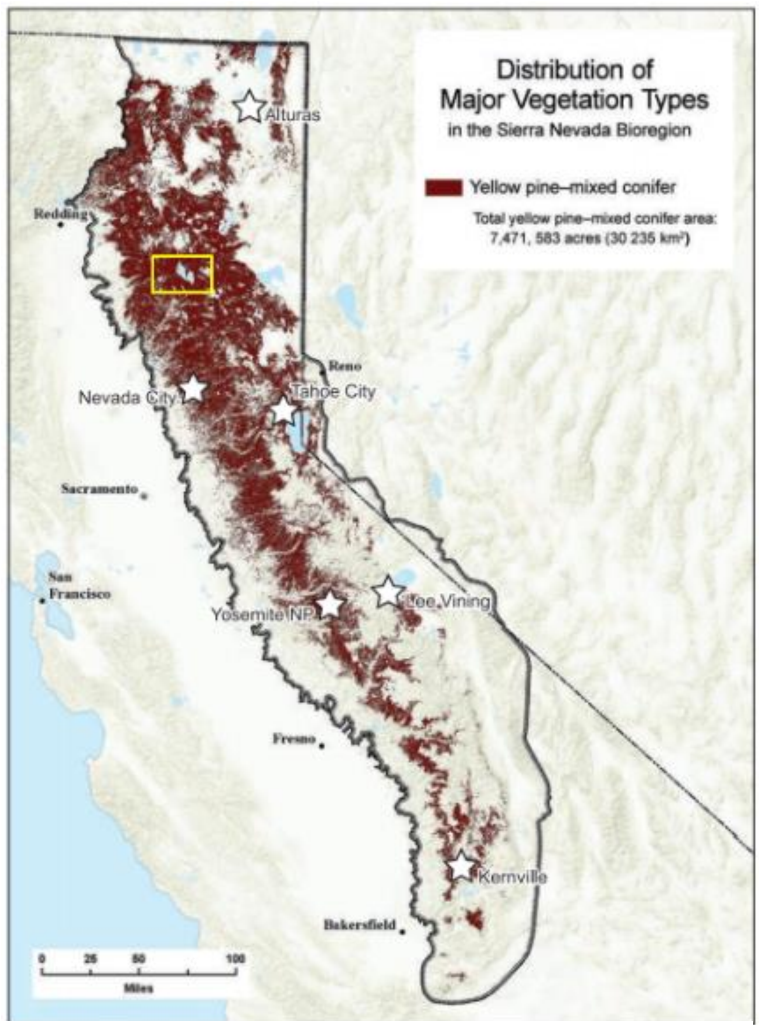


Figure 1. Sierran mixed conifer forest ecoregion (red). Study area for this project falls within the yellow box. (Easterday et al., 2018)

The mixed conifer forests of the Northern Sierra Nevada and Southern Cascade Region in California, also called Sierran mixed conifer forests, provide a variety of ecological, social, and economic goods and services (Franklin et al., 2002). Management of these forests falls under the discretion of many different stakeholders due to the variety of private and public ownership. The US Forest Service manages most public land in this region and the primary goals of public land management are centered around maintaining biodiversity and other ecosystem functions. Multiple private timberland companies also manage portions of these mixed conifer forests. It is the goal of timberland managers to maintain biodiversity and ecosystem functioning while turning a profit and producing goods such as meeting the demand for wood products (Golladay et al., 2016).

Fires in Mixed Conifer Forests

Ignited naturally by lightning and purposefully by the Native Americans, fire was a common occurrence in the Sierra Nevada before suppression efforts on the 1900s. Many of the species of the Sierra Nevada exhibit fire adaptations due to the relatively frequent recurrence of fire in this ecoregion before Euro-American settlement (McKelvey et al., 1996). With the ever-growing prevalence of large, intense wildfires, fire ecology is taking on a central role in land management approaches especially in regions that are becoming increasingly susceptible to the effects of climate change (Stephens and Collins, 2004). Historically, mixed conifer canopy and ground cover patchiness was a result of the variety of fire severity experienced in the region (Hessburg et al., 2007). Before Euro-American settlement, this region was “managed” by local first-nation tribes who regularly set fires in the forest. Historical records from this era indicate a median fire-return interval of about seven years (Stephens, 1998).

After Euro-American settlement in this region, land-use changed dramatically and included mining, livestock grazing, and logging. This shift in land use prompted a belief in the need for fire suppression, which was accepted and implemented early in the 1900s, yet eventually led to undesirable effects including increasing tree densities and higher fuel loads as well as changes in wildlife habitats (Stephens and Collins, 2004). These forest ecosystem and landscape level alterations, which have increased fire risk throughout the Western United

States (Luce et al., 2012), coupled with global climate change, are challenging land management strategies.

The role that fire plays in natural forest ecosystem disturbance is now well understood. It is now widely accepted that the successful fire suppression and land management practices of the twentieth century led to accumulated fuel loads that have greatly increased fire risk (Perry et al., 2011). Before settlement, wildfires in the Sierra Nevada were predominantly surface-level, with very little crown burning or mature tree mortality. This is in stark contrast to what we see today: an increase in fire size and intensity and tree mortality from crown fires. Many factors are contributing to this effect, the primary of which being the almost century of mismanagement and fire suppression. Fire plays an important role in these conifer ecosystems, including seedbed preparation, nutrient cycling, alterations to succession, generating of a mosaic of vegetative patterns that favor wildlife, and reducing hazardous conditions that would otherwise lead to an increase in fire intensity (Kilgore, 1973).

Regional Timberland Management

The mixed conifer forests of the Sierra Nevada are managed in multiple ways, under three broad categories: preservation, passive, and active. Preservation management occurs within the areas that are protected for various reasons such as National Parks and preserves. Passive management occurs in the regions that are generally unmanaged or have minimal management, such as National Forests. Active management occurs when an individual or company applies silvicultural practices for timber production. Several different practices fall into the category of active management, including clear-cutting (even-aged) and selective cutting (uneven-aged). Collins Pine Company, a certified sustainable forestry company, most often utilizes selective, uneven-aged management and occasionally will clear a stand if the decision falls within multiple sustainable ecosystem goals (such as releasing a meadow from conifer encroachment (O'Kelley, 2018). Following a wildfire, land managers decide to apply either active or passive management: to salvage log or to leave the site alone to progress through natural succession practices.

In the project treatment areas, private land is owned and managed by multiple corporate timber companies interspersed throughout federal public land managed by the US Forest Service. On private timberland, following a fire, it is common practice to salvage log burned trees to capture some profit and then plant seedlings in the disturbed footprint left behind, often with an herbicide spray to improve the success of the conifers planted to replace the previous stand. In these stand-replacing events, it is also typical practice to plant a monocrop of a profitable conifer species. Multiple species may be replanted, instead of a monocrop, depending on the land owner's goals and objectives. On passively managed timberland, such as the US Forest Service, it is common that natural succession progresses after a fire, rather than directly interacting and altering the natural succession. There are multiple reasons this may happen. For example, this approach may be part of the current forest management plan, to keep costs low, ease (or difficulty) of access, and litigation are all obstacles for the Forest Service to actively manage post-fire timberlands.

Since this study aims to compare four treatments, I decided to choose a region with both active and passive management, pre and post-wildfire. In 2012, a large, high-intensity fire named "Chips" burned through a portion of Plumas County. Within the area of interest, near Butt Valley Reservoir, the Chips fire burned 30,526 hectares of Plumas and Lassen National

Forests, Collins Pine Company, and other large and small-scale private forest land (Fites et al., 2012). For the control areas of this study, I selected the area just outside the Chips burn area, consisting of a mosaic of private and public land ownership that continues across Humbug Valley and the surrounding mountains.

Questions and Hypotheses

There have been empirical studies and meta-analyses on the effects of wildfire on forests, as well as investigations into the effects of different timberland management practices on biodiversity. As fires grow in intensity and frequency in response to climate change and the rapidly developing urban-wild interface, it is important to continue to develop our working understanding of how different perturbations affect our forest ecosystems. This study aims to improve understanding of how differences in land management strategies and fire affect forest structure and biodiversity.

As a forest ecosystem recovers from a wildfire disturbance, the composition of vegetation, the structure of the forest, and other ecosystem services follow a line of succession depending on multiple environmental conditions. The trajectory of this succession may alter the shift away from pre-fire conditions and services the forest provided, such as erosion control, nutrient cycling, and hydrologic processes (Beaty and Taylor, 2007). These features are affected by many different interacting processes including burn severity and duration, hydrologic function, human land use, and climate. This project addresses the potential impact that different management practices, pre- and post-fire, may have on forest structure and function, by examining vegetation diversity under four different treatments: active timber management pre- and post-wildfire and passive management pre- and post-wildfire. The goal of this project is to determine if there is a difference in understory plant biodiversity or forest canopy between passive and active timber management?

To meet the goal of this project, the following questions will be addressed: (1) Is there a difference in ecological indicators such as tree canopy cover? (2) Does plant species richness and evenness differ among the treatments? (3) Does understory forest composition differ among the treatments? In comparing the four treatment areas, I hypothesize that:

1. There will be differences in forest canopy cover and understory plant biodiversity between the passive (US Forest Service) and active (Collins Pine Company) burned treatment areas due to management differences following a wildfire.
2. Understory plant biodiversity of the two burned treatments will be different from their unburned counterparts.
3. The actively managed post-burn treatment, having undergone the highest degree of disturbance, will exhibit minimal canopy cover, presence of invasive plant species and increased potential for soil erosion.

METHODS

Study Area

The 2012 Chips fire burned in parts of the Plumas and Lassen National Forests of Plumas County, California. Much of the literature on characteristics of conifer forests under different management or response to wildfire has been focused on plots within research forests. This study was not done in a controlled “research” forest, and so it was imperative to control for as many variables, such as climate, aspect, elevation, slope and native forest structure as possible. As such, the study areas were selected to control for such variables, both within and outside the Chips burn scar (figure 2). The Chips burned study area also needed to have multiple timberland management approaches adjacent under the controlled parameters. The site selected within the Chips burn area was located upslope and North-East of the Butt Valley Reservoir.

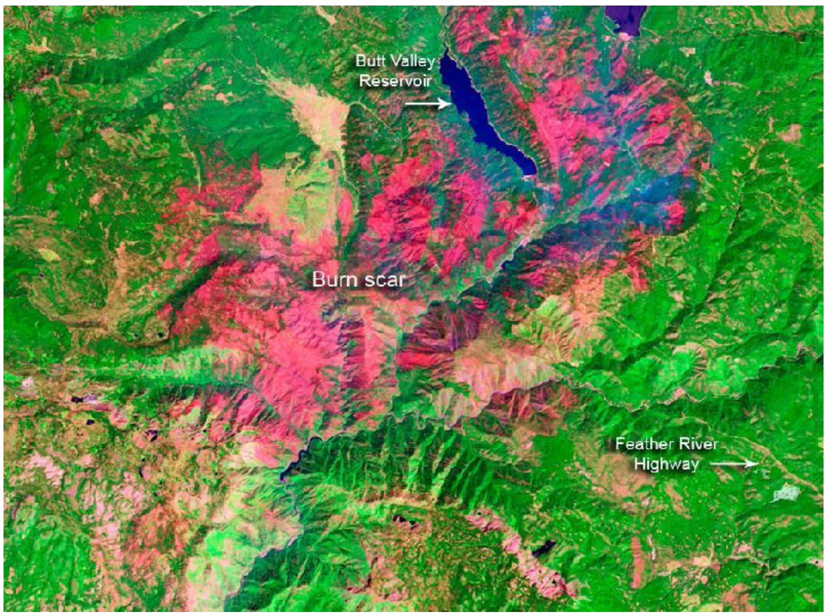


Figure 2: The 2012 Chips Fire burn scar NDVI (Allen and Simmon, 2012)

Within the Chips fire burn scar the treatment plots that were chosen were adjacent USFS- and CPC-managed sites (figure 3a). These selected sites were at an elevation between 1465 and 1550 meters, a south-east to south-west aspect, and a slope between 15 and 22 degrees. For further consistency, transects were always done from east to west. These sites within the Chips burn scar also experienced the same intensity of fire on the same date (figure 3b), providing an additional control variable.

To select the unburned treatment area, I used topographic maps to locate a nearby site with parameters matching the Chips burn area. West of Butt Valley is Humbug Valley, with similar elevation, slope, aspect, forest type, and timberland ownership (USGS, 2017). At this location, both USFS and CPC land was located along the same road, allowing for easy access.

The land under USFS management was under passive management, as it had been in the Chips study site. Similarly, the unburned site managed by Collins Pine Company had been logged using selective logging (uneven-aged management) 12 years prior, falling within the 10-20 year return-interval for selective cut management (O'Kelley, 2018).

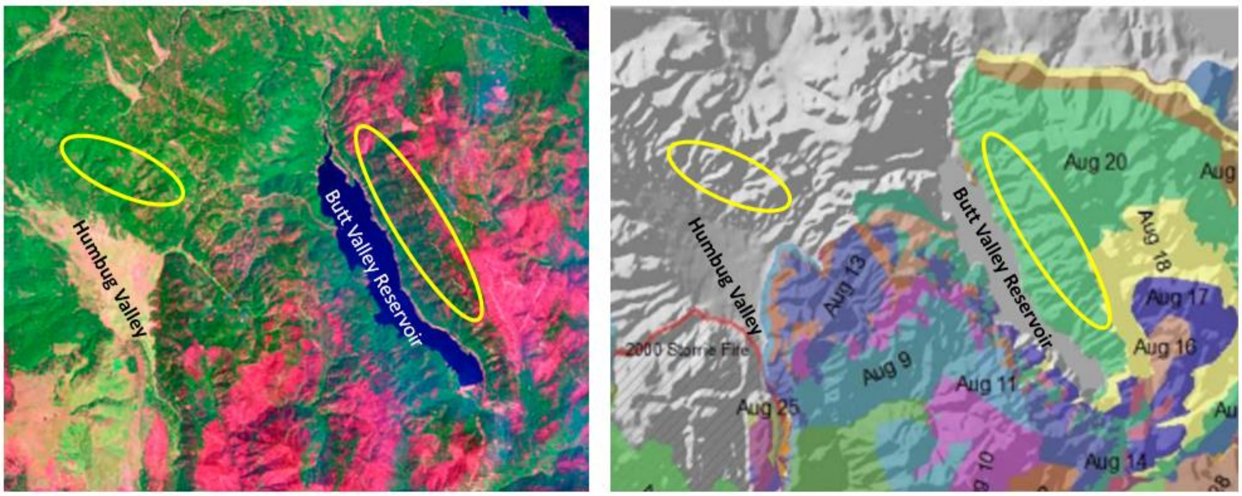


Figure 3 a (left) and b (right): Site selection for burned and unburned treatments. Figure a shows NDVI of treatment areas, b shows ignition dates within the Chips area. Site selection fell within the yellow ovals: Burned treatments to the upper-right of Butt Valley Reservoir and unburned treatments to the upper-right of Humbug Valley.

Data Collection

I collected data along three transects for each of the four treatment areas, for a total of 12 transects (figure 4). All transects ran east-to-west. Transect starting locations were determined by pinning numbered flags on the border of the treatment area every 20 meters in a stretch of 800 meters in length to locate random positions for transect starting points. Each transect was started 30 meters from the road, to control for disturbance factors other than fire, and was 50 meters long. Each transect was divided into 5-meter frames. Using a random number generator, four to five frames of each transect were selected for the 5-meter frame data collection plots to give a total of four or five frames per transect and three transects in each of the four treatment areas.

Figure 4: Project polygons for four treatments- Collins Pine Company (orange) and US Forest Service (green). Pins identify transect starting locations, labels identify burn or no burn and transect number.

Inside each of the 5-meter frames was a nested 1-meter frame, placed using a random number generator. In the five-meter frames, number and size class for each tree species was recorded, and presence/absence of all vegetation was obtained. In each 1-meter frame, percent cover for vegetation, litter, bare ground, downed woody debris, and rock was determined as well as percent canopy cover from the center point of each 1-meter frame.

Data Analysis

Statistical analysis was conducted in R using the Vegan package (Oksanen et al., 2007; RStudio Team, 2015). To evaluate for differences in canopy among treatments, multiple regression with a general linear model and analysis of variance (ANOVA) was performed. Species richness among the four treatments was calculated using a generalized linear model (GLM) with a Poisson distribution. The GLM is applied to transform the distribution of the data to stabilize the variance of non-normal distributions that occur due to small sample sizes and/or data with many zeros (Seavy et al., 2005). To assess alpha species diversity, Simpson's Inverse (1/D) and Shannon's indices were used. ANOVA analysis with a generalized linear model was used to measure significance.

To compare the compositional similarity of the vegetation in each treatment area, Jaccard's and Bray-Curtis dissimilarity indices were used to generate a dissimilarity matrix. An Adonis test was used to test for differences among treatments and within treatments. The individual species cover for each treatment was calculated and graphed by relative abundance to determine differences in evenness and compare the most abundant species within each treatment.

RESULTS

Forest Structure

Canopy

Forest canopy percent measurement was obtained from each one-meter nested frame for a total of 50 observations recorded for canopy cover: 15 in the burned passive (USFS) management treatment area and 12 each in active (CPC) management burned, CPC unburned and USFS unburned treatment areas. The Collins Pine burn treatment has lower forest canopy cover than the other three treatments ($p<0.001$; figure 5).

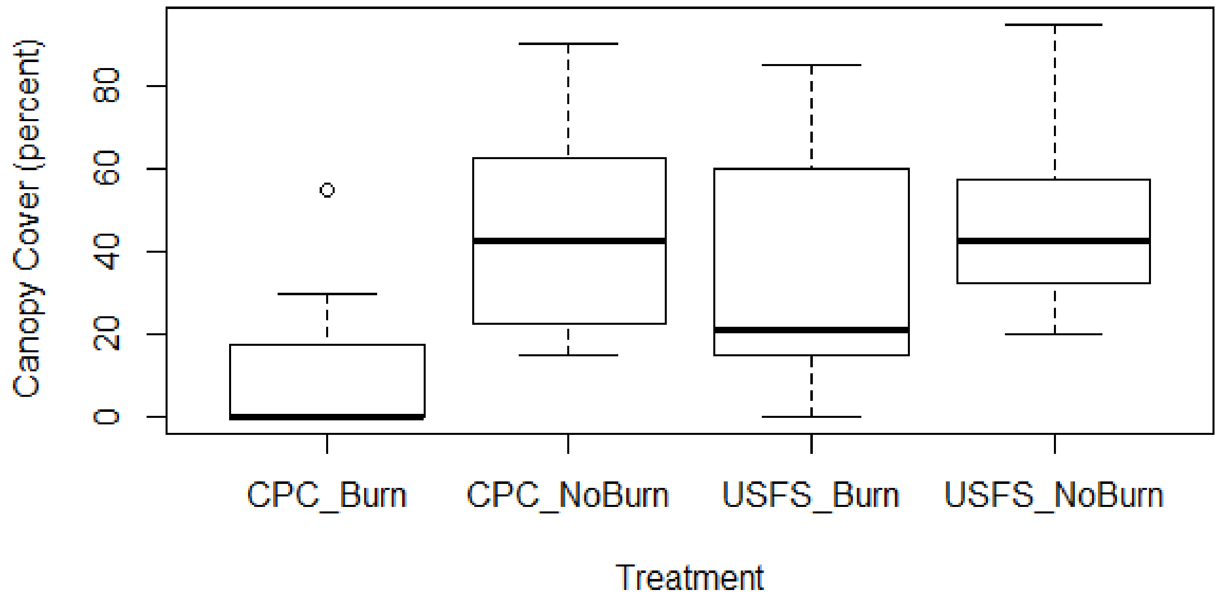


Figure 5: Difference in tree canopy cover percent among treatments

<u>Plant Species Composition</u>

Richness

A total of 34 species were recorded: five conifers, nine shrubs, fifteen forbs, four grasses/sedges, and one moss. In the USFS burn treatment, 22 species were observed. In the CPC burn treatment, 14 species were observed. In the USFS no-burn treatment, 24 species were observed. In CPC no burn treatment, 21 species were observed. All five species of conifer trees were observed in three of the four treatments, except in the CPC burn treatment where no *Abies concolor* (white fir) were observed. Additionally, two invasive species, *Verbascum thapsus* and *Cirsium vulgare* (mullein and bull thistle, respectively), were observed within the CPC burn treatment only. The CPC burn treatment area had much less variation in richness among frames within the treatment area but had the highest mean richness (figure 6). There was a wide range of richness observed within the other three treatments. Although some differences are apparent, diversity in terms of richness was not significantly different (p=0.368).

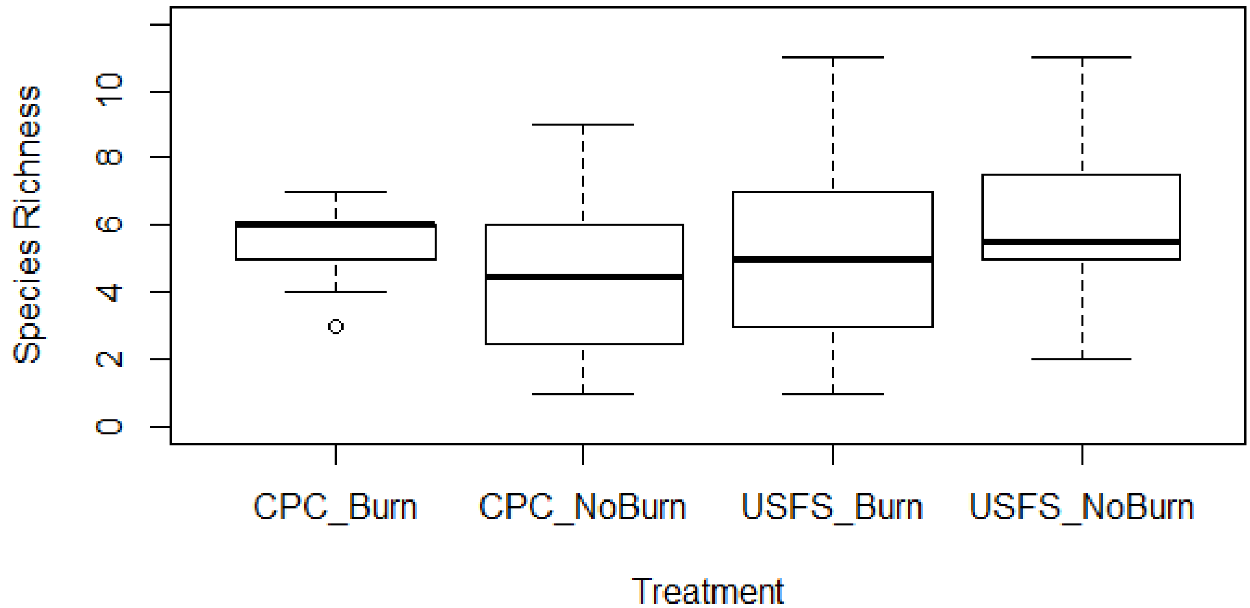


Figure 6: Plant species richness among the four treatments

Alpha and Beta Diversity

Analysis of alpha biodiversity evenness and richness, Simpson's Inverse (figure 7) and Shannon's (figure 8) indices indicate no significant differences among treatments (p=0.302 and 0.353, respectively).

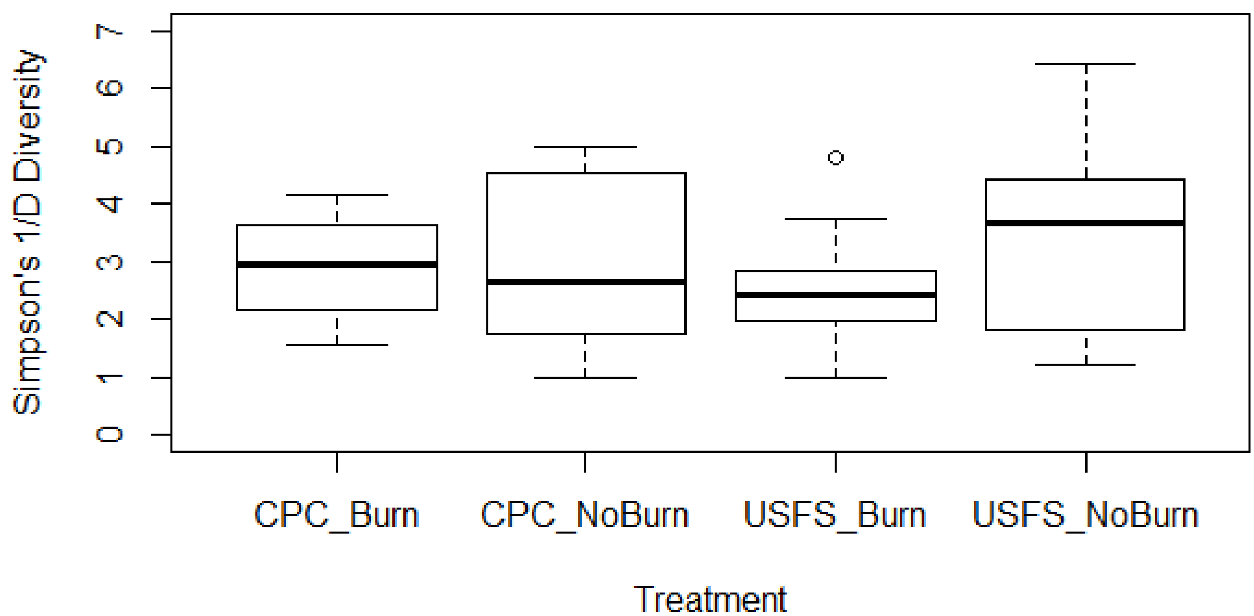


Figure 7: Simpson's inverse diversity differences among treatments

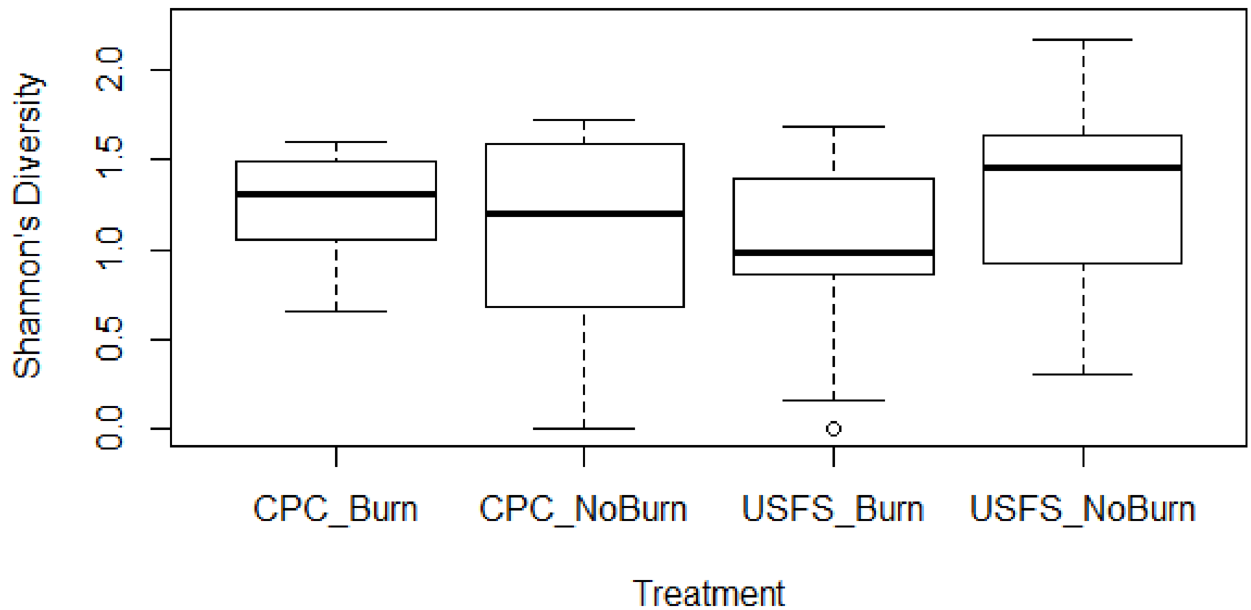


Figure 8: Shannon's diversity differences among timber management treatments

Evaluating beta diversity using Jaccard's index, results demonstrated the most dissimilar treatments are between the CPC no burn and USFS no burn (93% dissimilar) whereas the least dissimilar treatment comparisons were between both the no burn treatments (81% dissimilar) (p=0.001, table 1). Low coefficient of determination (r^2=0.339) indicates the unpredictability of the data. Within a given treatment, one frame does not predict what will be found in another frame.

Table 1: Jaccard dissimilarity values among treatments and within treatments.

	CPC Burn	*CPC No Burn*	*USFS Burn*	*USFS No Burn*
CPC Burn	0.652	0.900	0.874	0.910
CPC No Burn	0.900	0.791	0.930	0.813
USFS Burn	0.874	0.930	0.648	0.906
USFS No Burn	0.910	0.813	0.906	0.708

Bray-Curtis dissimilarity values indicate similar results as Jaccard with similar significance (r^2= 0.452, p=0.001). On the principal coordinate analysis plot for Bray (figure 9), there is much overlap in the no burn treatments indicating the non-burned treatments under US Forest

Service and Collins Pine Company are quite similar. The burn treatment under Collins Pine Company management is more like the no burn treatments than the USFS burn treatment, but the points in the plot for the burned CPC treatment are much more spread out because the data collected from frame to frame within the treatment had a lot of variation and greater dissimilarity from plot to plot.

Table 2: Bray-Curtis dissimilarity among and within the four treatments

	CPC Burn	*CPC No Burn*	*USFS Burn*	*USFS No Burn*
CPC Burn	0.524	0.831	0.790	0.844
CPC No Burn	0.831	0.674	0.879	0.703
USFS Burn	0.790	0.879	0.496	0.837
USFS No Burn	0.844	0.703	0.837	0.565

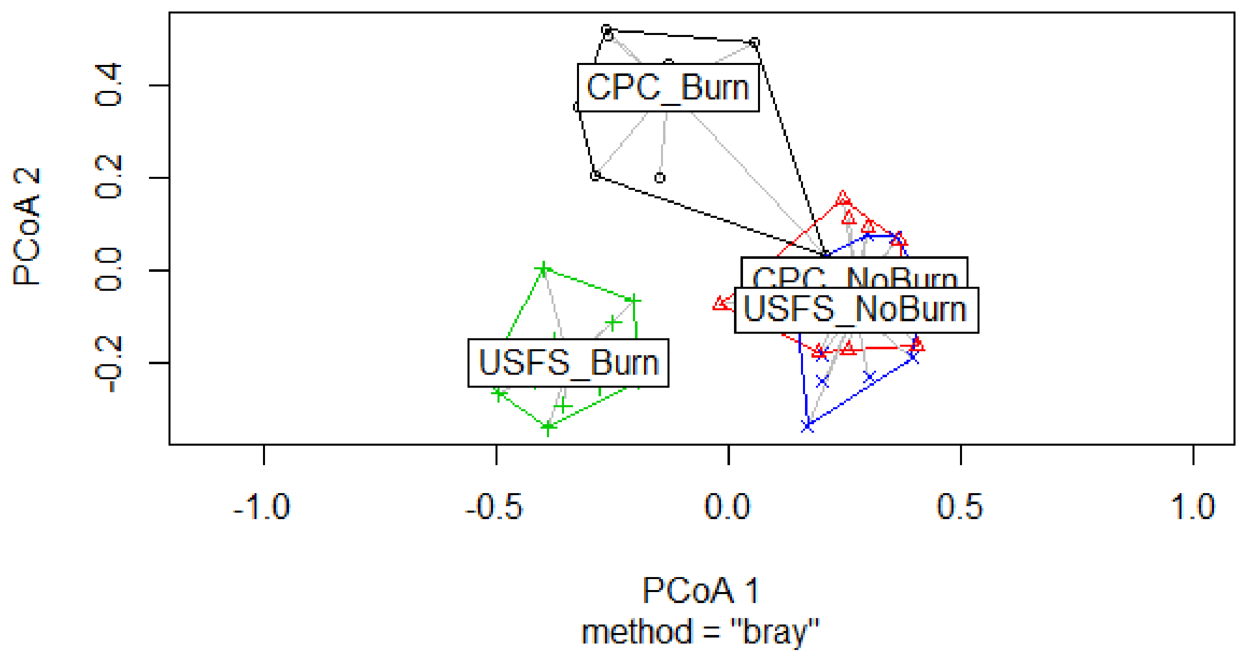


Figure 9: Principal Coordinate Analysis with Bray-Curtis method of the four treatment areas

Across all transects the most abundant species were *Ceanothus prostrates* (mahala mat), *Gayophytum diffusum* (ground smoke), and *Ceanothus integerrimus* (deer brush) (Table 3). The shape of the relative abundance distribution was similar between the burned treatments, where one dominant species stands out in either treatment area (figure 10). In the

USFS no burn treatment, species abundance is more even, indicated by the flatter curve in figure 10.

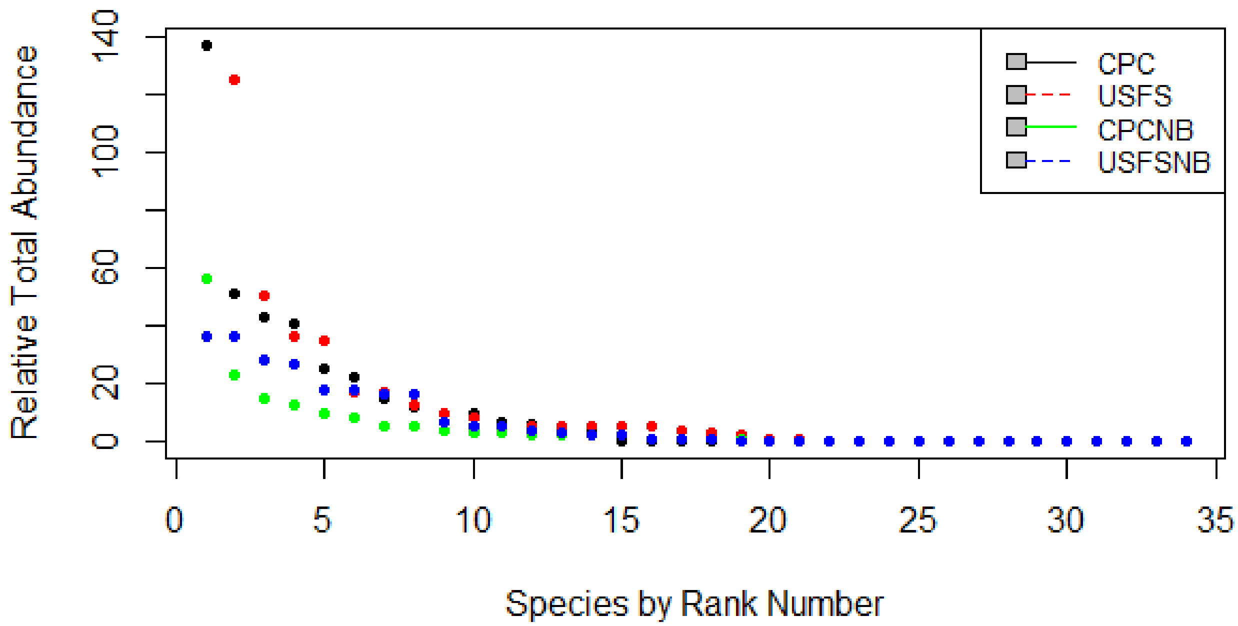


Figure 10: Relative total abundance of species among treatments

Table 3. The top ten most abundant species found in this study across all treatments (overall) and within each treatment. *Some species could not be identified beyond genus.

	Overall Frequency	*Frequency USFS Burn*	*Frequency CPC Burn*	*Frequency USNF No Burn*	*Frequency CPC No Burn*
# frames	50	14	12	12	12
Ceanothus prostratus	64%	43%	36%	92%	83%
Gayophytum diffusum	38%	21%	75%	42%	17%
Ceanothus integerrimus	32%	86%	0%	33%	0%
Gnaphalium spp.*	32%	29%	67%	8%	25%
Pinus ponderosa, jeffreyi	30%	50%	58%	0%	8%
Arctostaphylos patula	30%	7%	8%	58%	50%
Abies concolor	24%	86%	0%	0%	0%
Bromus spp.*	22%	14%	42%	17%	17%
Carex spp.*	22%	21%	0%	33%	33%
Pseudotsuga menziesii	18%	43%	17%	0%	8%

DISCUSSION

In comparing the four treatment areas, the following hypotheses were tested: (1) That there would be differences in forest canopy cover and understory plant biodiversity between the passive (US Forest Service) and active (Collins Pine Company) burned treatment areas due to management differences following a wildfire; (2) That understory plant biodiversity of the two burned treatments would be different from their unburned counterparts; and (3) that the actively managed post-burn treatment, having undergone the highest degree of disturbance, would exhibit factors of disturbance such as minimal canopy cover, presence of invasive plant species and increased potential for soil erosion. The results of this study show that forest canopy cover for timber land that is actively managed after a wildfire is lower than passively managed land. Although alpha diversity was not significantly different, there were differences in beta diversity. Species composition and most abundant species also differed between the two post-fire treatments. The passive and active pre-burn treatments had similar canopy cover, understory plant species composition, and relative species abundance, which were different than the corresponding post-fire treatments. The active management post-fire treatment demonstrated factors of disturbance such as a higher ratio of bare ground, less overall understory vegetation cover, and presence of two invasive species that were not observed in the other treatments. These results provide support for my original hypotheses.

Forest Structure

In this study, forest structure was defined by percent tree canopy cover. Tree canopy cover for the post-burn active management treatment by Collins Pine Company was lower ($p<0.001$) than the other three treatments, which supports the hypothesis that canopy cover would differ between active and passive post-fire management. This difference is most likely due to the post-fire practice of salvage logging and spraying herbicide. Collins Pine did not salvage log all available burned trees within the treatment area and transects for this study were selected randomly within the entire treatment area, which is why there is some variation in the data for tree canopy cover. However, the mean percent canopy cover is near zero, indicating that a large portion of the burned CPC treatment was salvage logged.

Plant Species Composition

All diversity indices indicate that the control areas, the treatments which undergo typical “management” that have not been burned are very similar. There are differences observed in the burned treatment areas. Both burned treatments have differences from each other as well as differences from the respective control treatments. Dominant species in terms of mean percent cover were different. In the Collins Pine burn treatment area, the dominant cover species was ponderosa pine, which aligns with expectations since this is the tree species that was planted the season following the fire with herbicides applied to inhibit other vegetative growth. Also, much of the Collins Pine treatment area was clear-cut for salvage regardless of tree mortality. Any other tree species that may have been present were taken out, which contrasts with the US Forest Service post-burn treatment area. The dominant cover species in the passive USFS burn treatment was an understory shrub, *Ceanothus integerrimus*, commonly known as deer brush. While there were many saplings in this treatment, deer brush begins growing more readily than conifer seeds from the seed bank post-fire, and the seedlings tend to be overtopped and grow slower than if mechanical or chemical removal of shrub species were to occur. This generates a buildup of brush which may be good for mitigating soil erosion but also contributes to future fire hazard potential.

Other Considerations

There were several limitations in this project especially in terms of spatial and temporal analysis. Ideally, this study would have captured many more observations over many more transects to improve confidence in the results. Not only would there be more observations, but these observations collected over a longer period would be preferable in this experimental design. The data were collected in late fall when many plant species are no longer flowering and so are more difficult to identify, so there is also a degree of observer error to consider when discussing the results and conclusions.

Observational comparison of ground cover among treatments revealed an apparent difference in bare ground versus litter. The burned Collins Pine Company treatment appeared to have more bare ground, likely an effect of using herbicide after planting ponderosa pine seedlings post-fire. This contrasts with the Forest Service burn area, which had a greater amount of litter possibly due to several seasons post-fire of surviving trees and shrubs dropping needles and leaves. It is important to note that biodiversity is only metric to consider in terms of analyzing differences among forest management strategies post-fire.

For further investigations on this topic and question, the study would ideally be expanded to include abiotic factors as indicators of landscape-scale ecosystem function, increase the number of transects, repeat sampling throughout an entire season, and repeat sampling over multiple seasons to detect spatial and temporal changes. This project is a snapshot comparison that can only allude to a much more complex interaction among multiple variables over time.

CONCLUSION

In this study, the lack of differences in biodiversity between the two sites were primarily due to the limitations of the study. I hypothesized that greater disturbance would affect the biodiversity of the forest ecosystem. The private timber management company, Collins Pine Company, effectively altered the forest structure by planting a single species of tree after the wildfire. Also, their practices opened the potential for invasive species to invade, and the canopy and ground cover loss in the salvage clear-cut may play a factor in negative impacts to the overall ecosystem function. The burned treatment under Collins Pine Company management has experienced the greatest disturbance yet maintains a relatively comparable level of biodiversity compared to its unburned counterpart and greater diversity than the US Forest Service managed fire area, which was a somewhat surprising outcome in this study. However, in these higher disturbance areas (logging), there was an indication that invasive species may be a management problem as well as higher rates of soil erosion and issues with underlying hydrologic processes due to exposed bare ground.

Though biodiversity was similar among all treatments, the burned treatments each had a single most highly abundant species. This indicates that any disturbance, fire only or fire plus logging and herbicide spraying, affects how species respond following a wildfire. The results of this study warrant further research into the differences in how forest ecosystems respond to different types of management following a wildfire.

REFERENCES

Allen, J., Simmon, R., 2012. Chips Fire Burn Scar [WWW Document]. NASA Earth Observatory. URL https://earthobservatory.nasa.gov/images/79105/chips-fire-burn-scar (accessed 10.28.18).

Battles, J.J., Shlisky, A.J., Barrett, R.H., Heald, R.C., Allen-Diaz, B.H., 2001. The effects of forest management on plant species diversity in a Sierran conifer forest. Forest Ecology and Management 146, 211–222. https://doi.org/10.1016/S0378-1127(00)00463-1

Beaty, R.M., Taylor, A.H., 2007. Fire Disturbance and Forest Structure in Old-Growth Mixed Conifer Forests in the Northern Sierra Nevada, California. Journal of Vegetation Science 18, 879–890.

Carey, A.B., 2006. ACTIVE AND PASSIVE FOREST MANAGEMENT FOR MULTIPLE VALUES. Northwestern Naturalist 87, 18. https://doi.org/10.1898/1051-1733(2006)87[18:AAPFMF]2.0.CO;2

Collins, B.M., Everett, R.G., Stephens, S.L., 2011. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. Ecosphere 2, art51. https://doi.org/10.1890/ES11-00026.1

Davis, F., Stoms, D., 1996. Sierran vegetation: a gap analysis. Nevada Ecosystem Project: Final Report to Congress, Vol. II. Centers for Water and Wildland Resources, University of California, Davis, pp. 671–690.

Easterday, K., McIntyre, P., Kelly, M., 2018. Land ownership and 20th century changes to forest structure in California. Forest Ecology and Management 422, 137–146. https://doi.org/10.1016/j.foreco.2018.04.012

Fischer, J., Lindenmayer, D.B., Manning, A.D., 2006. Biodiversity, ecosystem function, and resilience: ten guiding principles for commodity production landscapes. Frontiers in Ecology and the Environment 4, 80–86. https://doi.org/10.1890/1540-9295(2006)004[0080:BEFART]2.0.CO;2

Fites, J.A., Ewell, C., Bauer, R., 2012. The 2012 Chips Fire, California: A Case Study of Fire Behavior.

Fites-Kaufman, J.A., Noonan, E., Ramirez, David, 2005. Evaluation of Wildland Fire Use Fires on the Sequoia and Stanislaus National Forests in 2003: Effects in Relation to Historic Regimes and Resource Benefits.

Franklin, J.F., Spies, T.A., Pelt, R.V., Carey, A.B., Thornburgh, D.A., Berg, D.R., Lindenmayer, D.B., Harmon, M.E., Keeton, W.S., Shaw, D.C., Bible, K., Chen, J., 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using

Douglas-fir forests as an example. Forest Ecology and Management 155, 399–423. https://doi.org/10.1016/S0378-1127(01)00575-8

Golladay, S.W., Martin, K.L., Vose, J.M., Wear, D.N., Covich, A.P., Hobbs, R.J., Klepzig, K.D., Likens, G.E., Naiman, R.J., Shearer, A.W., 2016. Achievable future conditions as a framework for guiding forest conservation and management. Forest Ecology and Management 360, 80–96. https://doi.org/10.1016/j.foreco.2015.10.009

Hessburg, P.F., Salter, R.B., James, K.M., 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology 22, 5–24. https://doi.org/10.1007/s10980-007-9098-2

Kilgore, B., 1973. The Ecological Role of Fire in Sierran Conifer Forests: Its Application to National Park Management. Journal Quarternary Research 3.

Kleindl, W., Stoy, P., Binford, M., Desai, A., Dietze, M., Schultz, C., Starr, G., Staudhammer, C., Wood, D., 2018. Toward a Social-Ecological Theory of Forest Macrosystems for Improved Ecosystem Management. Forests 9, 200. https://doi.org/10.3390/f9040200

Luce, C., Morgan, P., Dwire, K., Isaak, D., Holden, Z., Rieman, B., 2012. Climate change, forests, fire, water, and fish: Building resilient landscapes, streams, and managers.

Mallek, C., Safford, H., Viers, J., Miller, J., 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. Ecosphere 4, art153. https://doi.org/10.1890/ES13-00217.1

McKelvey, K., Skinner, C., Chang, C., 1996. An Overview of Fire in the Sierra Nevada. Assessments and scientific basis for management options: Davis: University of California, Centers for Water and Wildland Resources, 2.

Miller, J.D., Collins, B.M., Lutz, J.A., Stephens, S.L., van Wagtendonk, J.W., Yasuda, D.A., 2012. Differences in wildfires among ecoregions and land management agencies in the Sierra Nevada region, California, USA. Ecosphere 3, art80. https://doi.org/10.1890/ES12-00158.1

Norris, K., 2012. Biodiversity in the context of ecosystem services: the applied need for systems approaches. Philosophical Transactions of the Royal Society B: Biological Sciences 367, 191–199. https://doi.org/10.1098/rstb.2011.0176

North, M.P., Schwartz, M.W., Collins, B.M., Keane, J.J., 2017. Chapter 5: Current and Projected Condition of Mid-Elevation Sierra Nevada Forests (General Technical Report), PSW-GTR-254. Forest Service, U.S. Department of Agriculture., Albany, CA: Pacific Southwest Research Station.

O'Kelley, E., 2018. Interview with Collins Pine Forester.

Oksanen, J., Kindt, R., Legendre, P., O'Hara, B., Stevens, M.H.H., 2007. The Vegan Package. Community Ecology Package 10, 631–637.

Perry, D.A., Hessburg, P.F., Skinner, C.N., Spies, T.A., Stephens, S.L., Taylor, A.H., Franklin, J.F., McComb, B., Riegel, G., 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management 262, 703–717. https://doi.org/10.1016/j.foreco.2011.05.004

RStudio Team, 2015. RStudio version 1.1.453, RStudio: Integrated Development for R. RStudio, Inc. RStudio: Integrated Development for R. RStudio, Inc., Boston, MA.

Safford, H.D., Stevens, J.T., 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.

Seavy, N.E., Quader, S., Alexander, J.D., Ralph, C.J., 2005. Generalized Linear Models and Point Count Data: Statistical Considerations for the Design and Analysis of Monitoring Studies (Gen. Tech. Rep.). USDA Forest Service.

Stephens, S.L., 1998. Evaluation of the effects of silvicultural and fuels treatments on potential fire behaviour in Sierra Nevada mixed-conifer forests. Forest Ecology and Management 105, 21–35. https://doi.org/10.1016/S0378-1127(97)00293-4

Stephens, S.L., Collins, B.M., 2004. Fire regimes of mixed conifer forests in the north-central Sierra Nevada at multiple spatial scales. Northwest Science 78.

Stephens, S.L., Moghaddas, J.J., 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. https://doi.org/10.1016/j.biocon.2005.04.007

USGS, 2017. The National Map [WWW Document]. The National Map. URL https://nationalmap.gov/3DEP/3dep_prodserv.html. (accessed 9.12.18).

APPENDICES

Detailed description of treatment areas and transects

Burned and unburned treatment areas were selected based off the following criteria: proximity of different types of timber management, ease of access, elevation, slope, aspect, and, in the burn area, burn date (figure 11) and fire intensity.

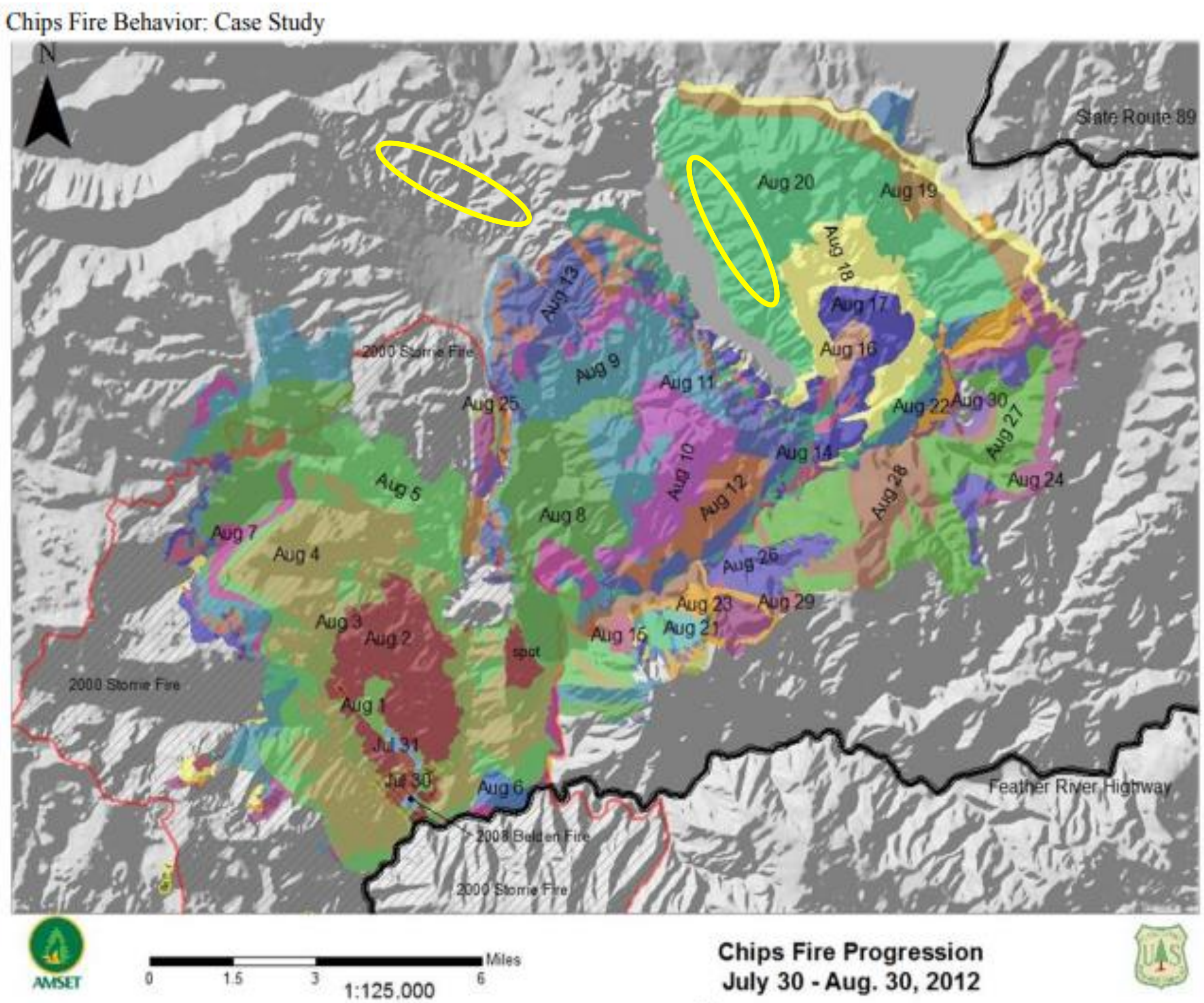


Figure 11: Chips fire progression (Fites et al., 2012)

To assess differences in forest structure and vegetation diversity under different management approaches before and after a fire, two study sites were selected which had forest managed by Collins Pine Company adjacent to US Forest Service. The first study site was above Butt Valley Reservoir within the 2012 Chips Fire burn scar (figure 9). The second study area, the non-burned site, was chosen outside of the burn area, above Humbug Valley, with a similar elevation, slope, and aspect to the first site. Each study site was assessed under two different treatments: 1) managed by Collins Pine Company, a private timber company and 2) managed by the US Forest Service. Three 50 m long transects were placed within each of the four treatments, selected at random distances and started from East to West at 30 m from the road.

Table 4: Eastern transect starting waypoints

	50 m transect Eastern Start Waypoint	*Frame 1 distance (m) from start*	*Frame 2 distance (m) from start*	*Frame 3 distance (m) from start*	*Frame 4 distance (m) from start*	*Frame 5 distance (m) from start*
			Study Site 1- Collins Pine Company Burn			
Transect 1	40.1477300, -121.1537330	10	25	30	45	NA
Transect 2	40.173773, -121.169679	0	20	30	35	NA
Transect 3	40.176179, -121.171288	10	20	25	40	NA
			Study Site 2- US Forest Service Burn			
Transect 1	40.94383, -121.9968	5	15	20	35	45
Transect 2	40.1468447, -121.1532039	0	10	25	30	45
Transect 3	40.1488230, -121.1552630	0	15	25	40	NA
			Study Site 3- Collins Pine Company No Burn			
Transect 1	40.1619906, -121.2603116	0	10	25	40	NA
Transect 2	40.1621362, -121.2592011	5	20	30	40	NA
Transect 3	40.1622858, -121.2571415	0	15	25	35	NA
			Study Site 4- US Forest Service No Burn			
Transect 1	40.1604627, -121.2387677	5	20	25	35	NA
Transect 2	40.1600656, -121.2395389	0	10	35	45	NA
Transect 3	40.1599036, -121.2406372	0	10	30	40	NA

<u>Burned Treatments</u>

US Forest Service – Plumas National Forest Management

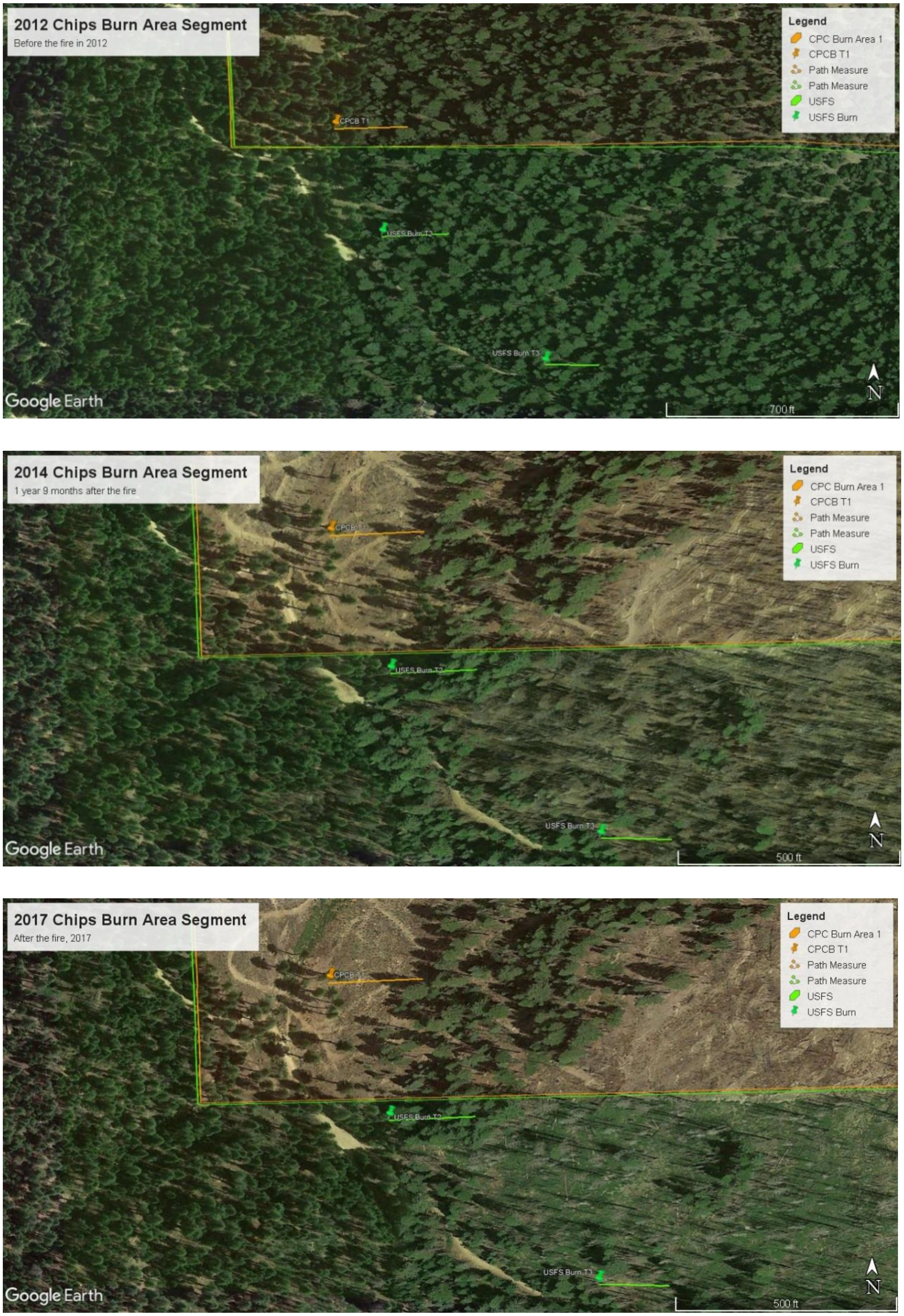
The areas within the Chips fire burn scar managed by the US Forest Service are primarily passively managed. Before the Chips fire, this area had not been logged for timber or managed for vegetation control. After the fire, the area was not treated and burned trees were left standing. Snags were left in place to stand or fall, adding to the amount of downed woody debris in this treatment area.

Figure 12: USFS Burn area near transect 1

Collins Pine Company Management

The area within the Chips fire burn scar managed by Collins Pine Company are actively managed, with a timber harvest return interval of 10-20 years. After the Chips fire, most of this area was 100% salvage logged regardless of tree mortality, replanted with only yellow pine (*Pinus jeffreyi* and *Pinus ponderosa*) then sprayed with herbicide. Some small portions of the Collins Pine property were left untreated. These sites were also captured within the transects of this study.

Figure 13: Collins Pine Company burn area near transect 1 (foreground) with USFS area in background



a)

b)

c)

Figure 14: Satellite images of Collins Pine (orange) and US Forest Service (green) treatment areas before the 2012 Chips fire (a), 1.75 years after the fire (b), and five years after the fire (c)

R Code

```
setwd("C:/Users/Caitlin/Desktop/Biodiversity/Project") #Set working directory
chips<-read.csv("CDalbyProfessionalProjectData.csv")
str(chips)
chips$treatment<-as.factor(chips$treatment)
str(chips)
dim(chips)
env=chips[,1:9]
names(env)
head(env)
cov=chips[,1:9]
names(cov)
head(cov)
veg=chips[,15:48]
names(veg)
head(veg)
tre=chips[,10:14]
names(tre)
head(tre)
pa=chips[,49:82]
names(pa)
head(pa)

#Test for significant difference in canopy among treatments
boxplot(canopy~treatment,data=env)
boxplot(canopy~treatment,data=env,ylab="Canopy Cover
(percent)",xlab="Treatment",main="Differences in canopy cover among treatments")
can.mod=glm(canopy~treatment,data=env)
anova(can.mod,test='Chi')

#Test for significant difference in richness among treatments
rich=rowSums(sign(veg))
boxplot(rich~env$treatment,ylim=c(0,12),ylab="Species
Richness",xlab="Treatment",main="Plant Species Richness Among Treatments")
rmod=glm(rich~env$treatment,'poisson')
anova(rmod,test='Chi')

#To test species diversity
install.packages("vegan")
library(vegan)

#alpha diversity differences
#Simpson’s Inverse index, inverse as in 1/D
Dsim=diversity(veg,'invsimpson')
boxplot(Dsim~env$treatment,ylim=c(0,7),ylab="Simpson's 1/D
Diversity",xlab="Treatment",main="Simpson's 1/D Diversity Differences")
#Simpson’s 1/D significance
anova(glm(Dsim~env$treatment),test='Chi')

Dshan=diversity(veg,'shannon') # Shannon
boxplot(Dshan~env$treatment,ylim=c(0,2.25),ylab="Shannon's
Diversity",xlab="Treatment",main="Shannon's Diversity Differences")
#Shannon's significance
anova(glm(Dshan~env$treatment),test='Chi')


#How dissimilar are plots between treatment?
#Jaccards dissimilarity
vJaccard=vegdist(pa,'jaccard') # get Jaccard dissimilarity (1-Jaccard index)
vJaccard
meandist(vJaccard,env$treatment)
#Significance?
adonis(vJaccard~env$treatment)

#Bray
vdiss=vegdist(pa,'bray')
vdiss
meandist(vdiss,env$treatment)
#Significance?
```

```
adonis(vdiss~env$treatment)

#Dissimilarity Principal Coordinates Analysis
ch.pco=betadisper(vdiss,group=env$treatment)
plot(ch.pco,main="Bray-Curtis Dissimilarity Principal Coordinates Analysis")
attributes(ch.pco)

#Two ways to look at community differences
#1. Calculate the relative abundance across each and compare them
tcov.cpcb=apply(cpcb,2,sum) # take the sum of total cover of each species for burned
collins pine plots
tcov.usfsb=apply(usfsb,2,sum)# take the sum of total cover of each species for burned
USFS plots
tcov.cpcnb=apply(cpcnb,2,sum)
tcov.usfsnb=apply(usfsnb,2,sum)

plot(seq(1:length(tcov.cpcb)),sort(tcov.cpcb,decreasing=T),col='black',pch=20,
ylab="Relative Abundance",xlab="Species by Rank Number",main="Relative Abundance of
Species")
points(seq(1:length(tcov.usfsb)),sort(tcov.usfsb,decreasing=T),col='red',pch=20)
points(seq(1:length(tcov.cpcnb)),sort(tcov.cpcnb,decreasing=T),col='green',pch=20)
points(seq(1:length(tcov.usfsnb)),sort(tcov.usfsnb,decreasing=T),col='blue',pch=20)
legend("topright",30,600,legend=c("CPC_Burn","USFS_Burn","CPC_NoBurn","USFS_NoBurn"),c
ol=c("black","red","green","blue"),lty=1:2,cex=0.8)

#Look at community differences
#Calculate the mean cover by CPC_Burn vs USFS_Burn
mcov.cpcb=apply(cpcb,2,mean) # take the mean cover of each species for Collins burned
plots
mcov.usfsb=apply(usfsb,2,mean)
rbind(mcov.cpcb,mcov.usfsb) # rbind pastes 2 rows together (mcov.l and mcov.ul)
diffb=mcov.cpcb-mcov.usfsb # Subtract USFS from Collins to look at mean difs. in cover
Pdiff=cbind(diffb,seq(1:length(diffb)))
Pdiff[order(Pdiff[,1]),] # nice ordered list of differences in cover between Collins
burned and USFS burned for each species
#second column is the species number in the original list

#Calculate the mean cover by CPC_NoBurn vs USFS_NoBurn
mcov.cpcnb=apply(cpcnb,2,mean)
mcov.usfsnb=apply(usfsnb,2,mean)
rbind(mcov.cpcnb,mcov.usfsnb)
diffb=mcov.cpcnb-mcov.usfsnb
Pdiff=cbind(diffb,seq(1:length(diffb)))
Pdiff[order(Pdiff[,1]),]

#Calculate the mean cover by CPC_NoBurn vs CPC_Burn
mcov.cpcnb=apply(cpcnb,2,mean)
mcov.cpcb=apply(cpcb,2,mean)
rbind(mcov.cpcnb,mcov.cpcb)
diffb=mcov.cpcnb-mcov.cpcb
Pdiff=cbind(diffb,seq(1:length(diffb)))
Pdiff[order(Pdiff[,1]),]

#Calculate the mean cover by USFS_NoBurn vs USFS_Burn
mcov.cpcnb=apply(usfsnb,2,mean)
mcov.cpcb=apply(usfsb,2,mean)
rbind(mcov.usfsnb,mcov.usfsb)
diffb=mcov.usfsnb-mcov.usfsb
seq(1:length(diffb)))
Pdiff[order(Pdiff[,1]),]
```