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UNDERSTANDING MIXED CONIFER FORESTS IN YOSEMITE NATIONAL PARK: AN HISTORICAL ANALYSIS OF FIRES REGIMES AND VEGETATION DYNAMICS

A Dissertation in

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By

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ABSTRACT

This dissertation presents the results of an historical ecological analysis that quantifies the temporal and spatial variation in forest structure and fire regimes in the mixed conifer forests of Yosemite National Park. I used stand structural analysis and reconstructed fire regime characteristics to determine the relationship between fire regimes and forest structure (i.e. age, size, composition, and spatial patterning) at the plot scale. The structure and composition of the contemporary forest varied spatially with several environmental factors (i.e. slope aspect, TRMI, elevation), although there were no significant differences in fire regime parameters across environmental gradients and forest compositional groups. The contemporary forest was significantly different from the reconstructed forest in basal area, density and mean diameter. The contemporary forest was denser, contained more basal area, but the mean diameter decreased due to the infilling of younger trees since fire suppression. In addition, the majority of the increased density in the forest was due to increases in the number of fire intolerant species (e.g. white fir, incense cedar). The spatial pattern of trees in the contemporary forest also changed from the reconstructed forest with a smaller number of plots demonstrating a clustering of trees of similar ages. Several fire regime parameters varied at the study area scale (extent, severity). Fires of all sizes were present in the study area, although the majority of the fires were < 250 ha in size. There was also a combination of low to moderate severity fires, with no evidence of high severity fires occurring. The relationship between fires and changing climate conditions was assessed by comparing the occurrence and extent of fires with several climate variables (e.g. Palmer Drought Severity Index, temperature). Strong relationships between climate variability and fire occurrence and extent were identified at both the interannual and interdecadal scale. At the interannual scale fire extent was linked to regional drought. At the interdecadal scale, fire occurrence and extent was significantly correlated with distinct phases of several interhemispheric climate patterns (e.g. El Niño-Southern Oscillation, Atlantic Multidecadal Oscillation), in addition to interactions between the climate patterns. The influence of fire regimes and forest structure on carbon sequestration was assessed by calculating the total biomass and carbon levels for both the reconstructed and contemporary forest. Total carbon storage increased significantly due to fire suppression. However, the location of the carbon in the forest changed due to the changing structure of the contemporary forest. More carbon was stored in smaller trees established since fire suppression, and fire intolerant species (white fir, incense cedar) stored a significantly greater amount of carbon than in the reconstructed forest. The results of this study indicate that there was a high degree of variability in forest structure and composition in the study areas, however, not all of the variability was related to fire regime characteristics. The influence of fire on the forest structure and carbon sequestration also varied over time, due to changing climate conditions. In addition, the significant role fire played in the historic forest structure is apparent from the high degree of change that occurred due to the removal of fire from the

system for over 100 years. Contemporary mixed conifer forests in Yosemite National Park are much denser and homogeneous than historic forests.

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Chapter 1

Understanding Mixed Conifer Forests in Yosemite National Park: An Overview

1.1 RESEARCH CONTEXT

One of the primary interests of studying vegetation dynamics is to understand the relationship between the processes at work and the structure of the resulting landscape and ecosystem (Watt 1947, Bormann and Likens 1979). These interactions have been studied from a variety of viewpoints over the years, and have focused on biotic processes (succession, competition) (Glenn-Lewin et al. 1992), abiotic conditions (moisture availability, soil nutrients) (Stephenson 1998, Burke et al. 1999), the role of disturbance events (fire, windstorm, insect outbreak) (Stewart 1989, Taylor 1990, Swetnam and Lynch 1993, Skinner and Chang 1996), and gradually changing climate conditions (Davis 1981, MacDonald et al. 1998). In addition to influencing the structure of the landscape, these interactions can influence the uptake and cycling of carbon within the system (Cohen et al. 1996). Disturbance events in particular can have a significant and long lasting impact on vegetation dynamics (White 1985, Glenn-Lewin et al. 1992, Shugart 1998) and carbon sequestration (Song and Woodcock 2003, Kashian et al. 2006) and ecosystems that experience regular disturbances may respond to changes in the disturbance regime, resulting in changes in the structure, composition and functioning of the ecosystem (Pickett and White 1985, Shugart 1998, Keane et al. 2002). In addition, changing climatic conditions can further influence the system by altering the disturbance regime thereby influencing the vegetation and, subsequently, the sequestration of carbon (Overpeck et al. 1990, Breshears and Allen 2002, Lenihan et al. 2003).

At the same time, vegetation dynamics have been studied at multiple scales from centimeters to 100s ha (Swetnam and Baisan 1996, Taylor and Skinner 1998, Burke et al. 1999). While much of this research has been successful at explaining vegetation dynamics at the scale of the system studied, they have limited utility in explaining larger, landscape scale controls on landscape patterns and dynamics. Consequently, a major stumbling block has been how to apply conclusions derived from small scale, site-specific research to other locations, and at different scales of interest (Lertzman and Fall 1998). This problem is especially acute when viewing research that has been conducted in the mixed conifer forests in the western United States.

Fire is recognized as the primary disturbance agent in mixed conifer forests in the Sierra Nevada (Kilgore 1973, Skinner and Chang 1996), and the interaction between vegetation, climate and topography influences the frequency and extent of fires in these forests (Agee 1993). However, the role of fire in mixed conifer forests has changed over time due to fire suppression (Kilgore 1973, Skinner and Chang 1996) altering the disturbance regime for an extended period of time. This has resulted in forests that are drastically different, structurally and compositionally than they were before fire suppression (Parsons and DeBenedetti 1979, Taylor 2004). In addition, long term variability in climate patterns has also influenced the disturbance regimes in these forests.

Consequently, in order to better understand how these systems may respond to variations in climate and disturbance regimes in the future, it is necessary to understand how they have responded in the past. In addition, understanding how these systems respond to variation in environmental factors can be very useful for resource managers. However, our understanding of the historic range of variability of disturbance regimes and how they have shaped forest structure in mixed conifer forests is very limited, especially at landscape scales (Swanson et al. 1994, Skinner and Chang 1996, Swetnam et al. 1999).

Identification of the historic structure and dynamics of ecosystems is essential to understanding how forested landscapes functioned prior to Euro-American settlement, and how they have changed in the western United States. Information on the historic forests supplies a "reference" point for a period of time prior to the widespread alteration of the environment that accompanied Euro-American settlement and identifies the natural range of variability of the system (Kaufmann et al. 1994, Landres et al. 1999, Swetnam et al. 1999). The historic range of variability within ecosystems is important because the species, and natural disturbances, present have evolved with each other and are considered to be resilient to perturbations of the environment. Changes that move the system outside of its historic range of variability could potentially result in permanent alterations to the ecosystem, such as species extinctions (Swanson et al. 1994, Folke et al. 2004), and under certain conditions this new condition of the system could persist indefinitely.

The concept of a system being able to permanently shift to a new equilibrium is known as the theory of alternate stable states (Lewontin 1969, Holling 1973). One example of an alternate stable state in a forested system can be found in the chaparral patches in the Sierra Nevada conifer forests (Biswell 1974, Nagel and Taylor 2005). Historically, the chaparral patches persist through the occurrence of periodic severe fires which kill all trees in the area. The chaparral then quickly regrows and shades out tree species thereby preventing their establishment. However, if fire is absent from the system for a long enough period of time for trees to grow above the chaparral and fill the canopy, then the shaded conditions may result in the decline of chaparral species (Nagel and Taylor 2005). The new system may then persist under a more frequent, low severity fire regime maintained by the fine fuels produced by the conifers. A shift to an alternate state may be occurring in mixed conifer forests. Fire suppression has resulted in a structural and compositional shift to higher density, fire intolerant species (Vankat and Major 1978, Parsons and DeBenedetti 1979). Therefore, quantification of the historic structure and dynamics of forests is necessary for the identification of the role that disturbances play in the long term development of forests. Consequently, information on the reference conditions of forests is necessary to understand how contemporary forested landscapes have changed and may continue to change under different disturbance regimes or changing climatic conditions.

The overall objective of my dissertation is to determine the relationship between climate variation, disturbance processes (fire regimes) and structural (density, species composition, age structure) variation in mixed conifer forests at the stand and landscape scales in Yosemite National Park. Yosemite National Park was chosen for the research because it harbors large tracts of old-growth forests that have not experienced any logging or grazing, and no fires since 1900. The premise behind this work is that fire regimes vary both spatially and temporally, due to vegetation, topographic and climatic variation, resulting in a forest that is more variable in structure and composition, and therefore, less predictable, than the shifting steady state mosaic model of mixed conifer forests (Bonnicksen and Stone 1981, 1982). Consequently, a non-equilibrium explanation of mixed conifer forest dynamics may be more appropriate. My research explores the variability in these interactions through an historical ecological approach involving reconstructions of past forest structures, fire regimes and climatic patterns. Comparisons with the contemporary forest will allow a better understanding of the relationships between these components and also the impact of the removal of fire on the structure and dynamics of the forests.

Understanding these relationships is also important to resource managers in National Parks who are trying to manage the resources according to historic (pre-Euro American settlement) conditions and processes. In National Parks, prescribed fire is one of the only tools available for forest restoration work; consequently, resource managers need to know the historic fire regimes if they intend to maintain the natural processes and variability in the forest systems. In addition, determining the relationships between the processes (fire regimes), environmental variables (elevation, slope aspect, etc.) and forest structure (composition, density, etc.) will enable resource managers to develop management plans for larger areas within the park. The amount of land covered by mixed conifer forests both within Yosemite National Park and in neighboring national forests makes it impractical to assess every drainage system in similar detail. Therefore, identifying the relationship between forest structure, climate, and fire regimes at both the stand and landscape scales may enable resource managers to apply the data derived to other locations in the region.

1.2 STUDY AREA

Mixed conifer forest structure and fire regimes were studied in old-growth forests located in two drainages in the mixed conifer zone along the western edge of YNP National Park (Figure 1.1). The first study area encompassed 2125 ha in the South Fork of the Tuolumne River watershed along the old Big Oak Flat entrance road north of Yosemite Valley. The second study area extended across 1600 ha in the drainage of the South Fork of the Merced River and was located upstream from the town of Wawona on the southern side of the park.

The study areas were chosen based upon the absence of known disturbances in the 20th century. There are few large, continuous, undisturbed landscapes in the mixed conifer zone in Yosemite National Park. According to park records these sites have never been logged and they have not experienced fire since the early 20th century. The identification of areas that have not burned since the late 19th century was essential for the project because of the tendency of fires to destroy the evidence (wood) needed to reconstruct historic conditions.

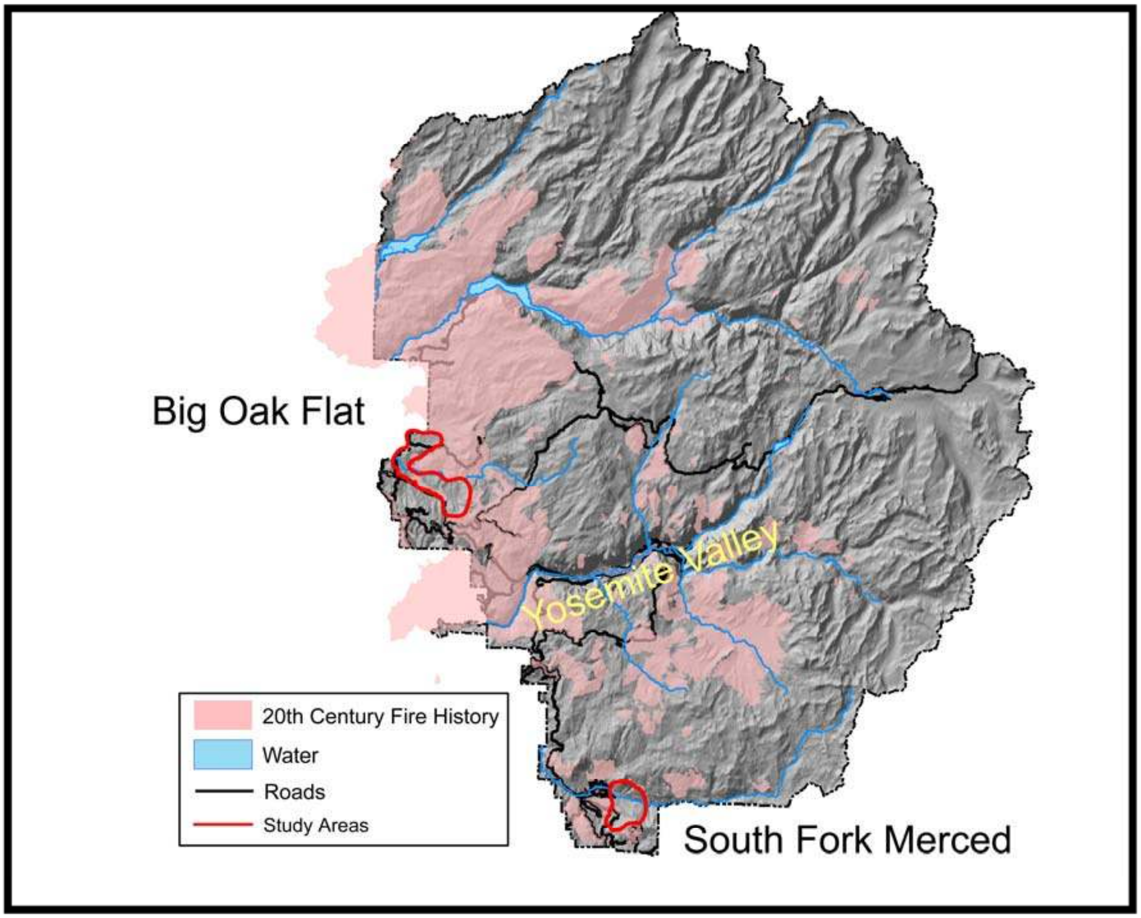


Figure 1.1: Location of the Big Oak Flat (BOF) and South Fork Merced (SFM) study areas, Yosemite National Park, California.

Elevations in both study areas ranged from 1300-2000 m, and included a diverse range of topography between them. The Big Oak Flat (BOF) study area is highly dissected by several small streams, creating a terrain of variable slopes (1-18°) across all aspects. Sizes of individual facets range from ~10 ha to ~100 ha. In contrast, the South Fork of the Merced (SFM) study area is bisected by the Merced River, which cuts a deep east-west running valley through the western slopes of the Sierra Nevada, creating a steep sided valley with strongly contrasting north and south aspects.

The climate of Yosemite National Park is “Mediterranean”, with warm-dry summers and cold-wet winters. Annual precipitation at Yosemite National Park (south entrance station = 1560 m) is 109.1 cm with 86% falling as snow between November and April. Mean monthly temperatures range from 2° C in January to 18° C in July. The study area is located on the western slope of the Sierra Nevada, and it is underlain by Mesozoic aged granite. The terrain is highly dissected by streams, and the terrain is steep and complex. Soils are shallow (<1 m), excessively drained and of medium acidity (Hill 1975, Huber 1989).

Forests in the mixed conifer zone in the Sierra Nevada are dominated by ponderosa pine (Pinus ponderosa Dougl. ex laws.) and incense cedar (Calocedrus decurrens Torr.), with lesser amounts of sugar pine (Pinus lambertiana Dougl.). White fir (Abies concolor Gord. & Glend. Lindl. ex Hildebr.) and Douglas-fir (Pseudotsuga menziesii (Mirb.) Franco) are important associates on wetter, north facing slopes, while California black oak (Quercus kelloggii Newb.) is an important associate on dryer, south facing slopes. Above 1800 m Jeffrey pine (Pinus jeffreyi Grev. and Balf.) replaces ponderosa pine, and red fir (Abies magnifica A. Murr.) is present, while Pacific dogwood (Cornus nuttallii Audobon), bigleaf maple (Acer macrophyllum Pursh), and canyon live oak (Quercus chrysolepis Liebm.) are present below 1300 m (Barbour 1988). Patches of chaparral, composed primarily of greenleaf manzanita (Arctostaphylos patula), are scattered throughout the zone. Dominance in mixed conifer forests can shift among any of the above conifer species and is often related to changes in elevation, topography and latitude (Barbour 1988, Parker 1991, 1994).

The mixed conifer forests of the Sierra Nevada have an extensive history of human occupation and use (Anderson 2005). Native Americans have been present in the Yosemite region since the Early Prehistoric Period 4 (7500-6000 BP) (Moratto 1984). Evidence suggests continuous occupation of the Yosemite region by the Southern Sierra Miwok from approximately 600 BP (Moratto 1999) until the time of Euro-American discovery of Yosemite Valley in 1851. While permanent settlements were located within Yosemite Valley, seasonal camps were more common in the surrounding region in what is present day Yosemite National Park (Clark 1904). The Miwok were hunter-gatherers who actively used fire as a tool. Ethnographic accounts indicate that they used fire to assist in the hunting of animals, by flushing game from cover (Barrett and Gifford 1976), to encourage the growth of desirable seed crops, such as the production of acorns (Lewis 1973) and to promote the growth of plants ideal for weaving baskets (Anderson 1993). The extensive use of fire by Native Americans in the Sierra Nevada, in conjunction with the continuous presence in the region, suggests that forests in the Sierra Nevada may have been altered by human actions over thousands of years. The composition and structure of forests encountered at the time of Euro-American discovery of the Yosemite region was a direct result of hundreds of years of manipulation by the Miwok (Lewis 1973, Anderson and Moratto 1996, Anderson 2005).

The first Euro-Americans in the Yosemite region were an expedition led by Joseph Walker in 1833 that crossed the Sierra Nevada between the Merced and Tuolumne rivers, passing through the region in which the Big Oak Flat study area is located (Sanborn 1981). Following the discovery of Yosemite Valley in 1851 (Bunnell 1892), Euro-American presence in the valley increased steadily, although the surrounding areas did not experience regular visitors until the early 1900s (Greene 1987). At the same time, the Miwok population declined in Yosemite Valley through the late 1800s as a result of the Euro-Americans forcing the majority of them out so that the valley could be settled and developed for tourists (Sanborn 1981). A few Miwok settlements remained, although they slowly dwindled in size due to restrictions imposed on their traditional lifestyles by the Euro-Americans settling in the valley. Only 30 Miwok were present in Yosemite Valley in 1900 (Greene 1987). During this time the Miwok may have still used the surrounding region of the park, but only on a seasonal basis (Clark 1904). Widespread declines in Native American populations also occurred during the 1830s-'50s throughout California and the Sierra Nevada. This has been attributed to large-scale epidemics of diseases brought by Euro-American settlers (Bates and Lee 1990), and the widespread hunting and killing of Native Americans authorized by the State of California during the gold rush (Castillo 1978). In Mariposa County, which encompasses Yosemite Valley, Native Americans were outlawed and killing them was legally permitted (Perlot 1985).

Euro-American populations in the Yosemite region and subsequent land-use practices, including logging, mining, and grazing, increased during the late

1800s. While mining was almost non-existent in the Yosemite region, logging and grazing increased steadily. Major logging operations in the foothills of the Sierra Nevada extended up the Merced River canyon toward Yosemite Valley and resulted in the removal of extensive tracts of forest in the greater Yosemite Region (Johnston 1966). At the same time, sheep grazing increased throughout the Sierra Nevada (Muir 1894). However, logging and grazing activities were limited in current Yosemite National Park due to its early protection under the state of California in 1864, and subsequent establishment as a National Park in 1890. At this time, the military was sent in to patrol the park and to stop any grazing and logging within park boundaries (Runte 1990). With the establishment of the National Forest Service in 1904 and implementation of a policy of fire suppression on federal lands, the presence of wild fires within the mixed conifer zone of Yosemite National Park declined dramatically.

The region of Big Oak Flat (Figure 1.2) experienced even less impact from Euro-Americans than Yosemite Valley. According to records, a limited amount of sheep grazing occurred in the 1870s in the Big Oak Flat region, but was most likely limited to the meadows along the prospective course for the Big Oak Flat road under construction at the time (Paden and Schlictmann 1959, Greene 1987). In addition, the Big Oak Flat region was protected from logging in the 1930s when 19,000 acres were purchased to expand the park boundary (Runte 1990). At the time, logging operations had not yet reached the Big Oak Flat region, and therefore, no logging had occurred there.

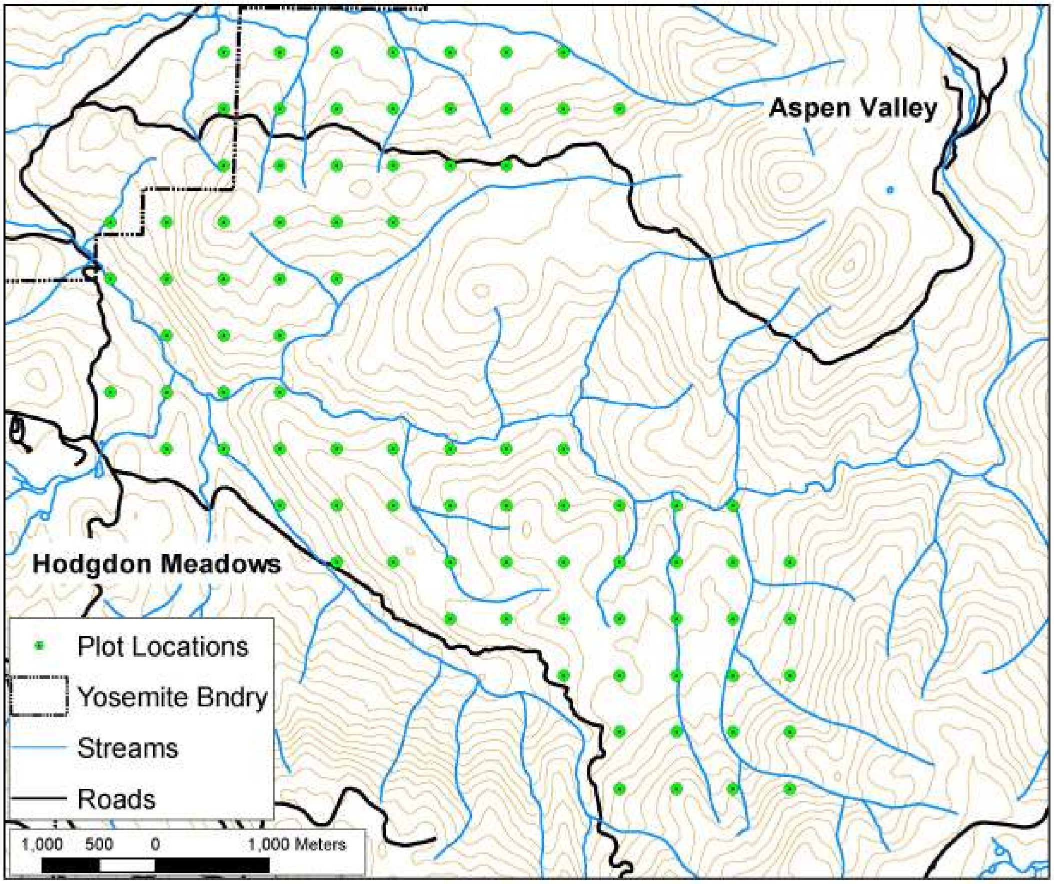


Figure 1.2: Location of study area and sample plots within Big Oak Flat (BOF) region, Yosemite National Park, California.

In the South Fork Merced region (Figure 1.3), Euro-American impact was greater, and occurred earlier than in the Big Oak Flat region. This was due to the first route into Yosemite Valley passing through Wawona. Consequently, the first permanent resident was living in the region in 1856 (Greene 1987), due to the route to the valley, and not for the purposes of logging or grazing. When the cavalry arrived in 1890 for the purpose of stopping illegal grazing, logging and fires, they were headquartered in the Wawona area, which may have led to limited grazing and logging in the area. With the purchase of the Wawona addition in 1932, the region where the SFM study area is located was fully protected from logging and grazing (Greene 1987, Runte 1990).

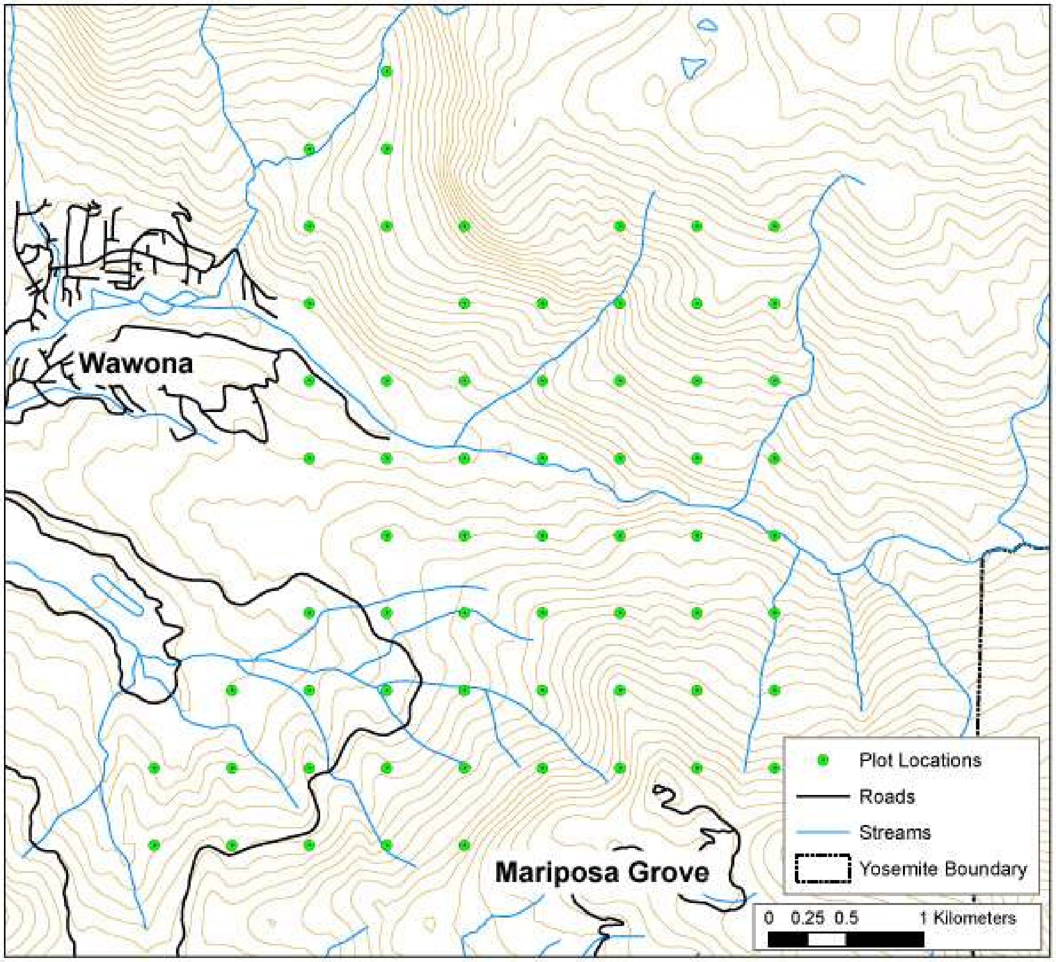


Figure 1.3: Location of study area and sample plots within South Fork Merced (SFM) Region, Yosemite National Park, California.

The study areas were chosen based upon the absence of known disturbances in the 20th century. There are few large, continuous, undisturbed landscapes in the mixed conifer zone in Yosemite National Park. According to park records these sites have never been logged and they have not experienced fire since the early 20th century. The identification of areas that have not burned since the late 19th century was essential for the project because of the tendency of fires to destroy the evidence (wood) needed to reconstruct historic conditions. The limited impact by Euro-American settlers in the two study areas and subsequent protection from fires in the 20th century make the Big Oak Flat and South Fork Merced sites ideal locations in which to study historic fire-climate interactions, and changes in vegetation dynamics and carbon sequestration due to fire suppression.

Chapter 2

Structure and Dynamics of Old Growth Mixed Conifer Forests in Yosemite National Park

2.1 INTRODUCTION

Identification of the historic structure and dynamics of ecosystems is essential to understanding how forested landscapes functioned prior to Euro-American settlement, and how they have changed in the western United States. Information on the historic forests supplies a "reference" point for a period of time prior to the widespread alteration of the environment that accompanied Euro-American settlement and identifies the natural range of variability of the system (Kaufmann et al. 1994, Landres et al. 1999, Swetnam et al. 1999). The historic range of variability within ecosystems is important since the species, and natural disturbances, present have evolved with each other and are considered to be resilient to perturbations of the environment. Changes that move the system outside of its historic range of variability could potentially result in permanent alterations to the ecosystem, such as species extinctions (Swanson et al. 1994, Folke et al. 2004). In addition, quantification of the historic structure and dynamics of forests enables identification of the role disturbances play in the long term development of forests. Consequently, information on the reference conditions of forests is necessary to understand how contemporary forested landscapes have changed

Resource managers are also interested in understanding the natural range of variability of the reference conditions for forested landscapes on public lands. National Park Service policy dictates that managers must manage the resources according to historic conditions and processes. In the case of the mixed conifer forests of the Sierra Nevada, historic refers to pre-EuroAmerican settlement, and processes refer to the disturbances and dynamics of the forests. The National Forest Service, meanwhile, is interested in understanding the historic range of variability, because it is assumed that the historic forests were healthier and less prone to stand replacing fires than contemporary forest. Unfortunately, while reconstructions of fire are common throughout the mixed conifer zone of the Sierra Nevada (Caprio and Swetnam 1995, Skinner and Chang 1996), southern Cascades (Taylor 2000, Beaty and Taylor 2001), and Klamath mountains (Taylor and Skinner 1998, Taylor and Skinner 2003), little information is known about the structure of historic forests in the Sierra Nevada, or the interaction between historic disturbances and forest structure. Consequently, resource managers need information on the reference conditions of the forests in order to develop management plans that mimic historic conditions.

Fire is recognized as the primary disturbance agent in mixed conifer forests in the Sierra Nevada, yet fire's role in mixed conifer forests has changed over time due to fire suppression (Kilgore 1973, Skinner and Chang 1996). This alteration of the disturbance regime for an extended period of time has resulted in forests that are drastically different, structurally and compositionally, than before fire suppression (Parsons and DeBenedetti 1979, Taylor 2004). The density of small trees has increased, and forest composition has shifted from fire tolerant to shade tolerant, fire sensitive species (Parsons and DeBenedetti 1979). At the same time there has been an increase in surface and aerial fuels which have increased the potential for high severity fires in the late 20th century (Skinner and Chang 1996). In Yosemite National Park, fire suppression has been an effective policy since the turn of the century, and the last major fire to burn in either the South Fork of the Tuolumne drainage, or the South Fork of the Merced drainage was in 1899.

While the presence of fire in mixed conifer forests is well documented, there has been little research that links fire regimes with forest structure and regeneration dynamics at either stand or landscape scales (Skinner and Chang 1996, Taylor and Skinner 1998, Beaty and Taylor 2001). A central idea concerning the structure and dynamics of mixed conifer forests is that stands are composed of a mosaic of even-aged patches that developed after low or moderate intensity surface fires (Bonnicksen and Stone 1981, 1982). Frequent, low intensity surface fires are common (median fire return interval = 3-20 years) in these forests and are thought to establish a fine grained mosaic of even-aged patches that result in multi-aged stands (Kilgore and Taylor 1979, Skinner and Chang 1996). The fire-forest mosaic is described as being self-organizing because of the interaction among forest patches, fuel buildup and fire occurrence. For example, a recently burned patch will not burn again until enough fuels accumulate within the forest patch. Consequently, fires in the forest are not likely to burn through a recently burned patch of forest. The result of this process at a landscape scale is a patch mosaic of different regeneration stages which has been described as a shifting mosaic steady state (Bonnicksen and Stone 1981, 1982, Minnich et al. 2000).

However, recent research suggests that the fire-forest mosaic in mixed conifer forests may be more complex (Beaty and Taylor 2001, Heyerdahl et al. 2001, Beaty 2004). Variations in fire regimes along topographic gradients (i.e., elevation, slope aspect) may be critical to differences in forest composition and structure. While some studies have found strong relationships between slope aspect and fire return intervals (Taylor and Skinner 2003), in other studies, this relationship has not been present (Heyerdahl et al. 2001). Likewise, variation in slope position in incised terrain has been connected with variation in fire return intervals in some locations, while there has been no discernable relationship in others (Taylor and Skinner 1998). Furthermore, there is evidence that not all fires in mixed conifer forests are low and moderate severity burns. Historical reconstructions in steep terrain have found fire severity to be related to slope position in some locations (Beaty and Taylor 2001), but not always (Bekker and Taylor 2001). In addition, coarse scale patterns of even-aged trees have been identified in forests that have experienced high severity fires (Beaty and Taylor 2001, Taylor and Skinner 2003).

Species distribution patterns, species regeneration dynamics and forest community structure in mixed conifer forests have all been linked to recurring fire (Morrison and Swanson 1990, Agee 1993, Skinner and Chang 1996, Taylor

2000, Beaty and Taylor 2001). However, the resulting effect of fire on forest structure and dynamics is highly variable due to the spatial and temporal variation of fire regime characteristics (e.g. frequency, size, severity, seasonality) (Chappell and Agee 1996, Swetnam et al. 2000, Taylor and Solem 2001, Taylor and Skinner 2003), and this variability is thought to play an important role in the maintenance of species, and structural, diversity at stand and landscape scales (Martin and Sapsis 1991). Consequently, while several studies in mixed conifer forests have identified forest structural patterns related to recurring fire (Taylor and Skinner 1998, Taylor and Skinner 2003, Nagel and Taylor 2005, Beaty and Taylor 2007, Beaty and Taylor 2008) the variation in patterns identified suggests that fire forest patterns vary geographically. As a result it is difficult to apply the results identified in one location to another location.

The basis of the work presented here is the reconstruction of the historic forest conditions and fire history through the use of dendroecological techniques, including modeling the decomposition of dead trees. The incorporation of dead trees is essential to develop as accurate a reconstruction as possible because the presence of large trees in the historic forest that are dead today could potentially have dramatic impacts on the forest structure. The reconstruction will allow me to compare the characteristics of the contemporary forest with those of the historic forest. The detail of the reconstruction (age and size structure, species composition, spatial pattern, and incorporation of dead trees) is unique to many other mixed conifer forests studied due to the presence of large stands of old growth forest that have experienced no major fires since 1899, leaving all of the dead trees in the forest. In addition, the discovery of data from a forest survey in 1911 (U.S.D.A. Forest Service 1911) which extended into Yosemite National Park in the Big Oak Flat study area allowed me to evaluate the precision of my reconstruction. Consequently the reconstruction complements other research analyzing changes to mixed conifer forests due to fire suppression, and may add more detailed information about the interactions between fire and forest structure at both the stand and landscape scale.

The overall objectives of this study are to examine the structure and dynamics of the mixed conifer forests of Yosemite National Park. The general null hypothesis of this research is that the relationship between fire regimes and forest structure does not vary spatially and temporally. More specifically, this research seeks to answer the following questions: 1.) What were the pre-fire suppression reference conditions of the mixed conifer forest (i.e., species composition, age and size structure, horizontal pattern)?; 2.) How are the contemporary forests different from the reference forests?; 3.) How did the pre-fire-suppression fire regimes vary spatially and temporally?; 4.) What was the role of fire in structuring the pre-fire suppression forests?; 5.) How can resource managers use this information to develop management plans for mixed conifer forests in Yosemite National Park?

2.2 METHODS

2.2.1 Forest Structure and Composition

To characterize variation in forest structure and composition across the two study areas plots were sampled on a grid. Grid points (BOF n = 85, SFM n = 64) were established at 500 m intervals from a starting point chosen randomly on a 1:24000 topographic map, and encompassed the range of variation in slopes, aspects and elevation in the study area. Grid point locations were then identified in the field using a GPS. At each grid point forest characteristics were sampled using three nested circular plots centered on the point. Large trees (live and standing dead) >35.0 cm diameter (conifer) or >15.0 cm diameter (hardwood) were sampled in the large plot (1000 m^2). Trees were measured at breast height (dbh), and their positions (x,y coordinates) were determined by measuring the distance and direction (azimuth) of each live and dead stem from the point at the center of the plot. All live trees were also assigned to one of five height classes based on the relative height and position of the crown (suppressed, intermediate, lower main canopy, upper main canopy, and emergent). The diameter, species, direction of fall, and decay class (Maser et al. 1979) of all logs (>35 cm dbh) rooted in the plot were also recorded. Small trees (10-35 cm dbh conifers, 5-15 cm dbh hardwoods) were sampled in the intermediate plot (250 m^2). Measurements recorded for the small trees were identical to those recorded for large trees. Seedlings (50 cm – 1.4 m tall) and saplings (1.4 m tall – 10 cm dbh) were tallied by species in the smallest plot (100 m^2). Ground cover of shrubs and forbs in the smallest plot was estimated to one of six cover classes (1 = 0-1%, 2

= 1-5%, 3 = 5-25%, 4 = 25-50%, 5 = 50-75%, 6 = 75-100%). The elevation, slope pitch, slope aspect, slope configuration, and topographic position were also recorded for each plot. The last four variables were used to estimate the Topographic Relative Moisture Index (TRMI) for each point. TRMI is an index of relative potential soil moisture based on variation in topography that ranges from 0 (xeric) to 60 (mesic) (Parker 1982).

The age structure of the contemporary forest was determined by coring to the pith all measured trees in the plots at 30 cm height above the soil surface with an increment borer. Cores were sanded to a high polish and annual growth rings were counted and cross-dated using standard dendroecological techniques (Stokes and Smiley 1968). The date of the innermost ring was then used as an estimate of tree age.

Not all trees (BOF = 19%, SFM = 15%) could be aged directly because the increment borer was too short to reach the pith, or the center of the tree was rotten. The ages for trees with incomplete cores were estimated in the following way. First, a regression equation between core length and tree diameter for each species was developed (Table 2.1a, b). All of the regressions were highly significant ($P<0.001$). Second, the missing length for each incomplete core was estimated by subtracting the length of the core from the predicted length. For cores with a predicted length that was shorter than the actual core, the actual core length was used. Finally, the width of the earliest 5 years' growth on complete cores was measured to determine the average number of rings/cm for each species (Table 2.1a, b). The number of years missing from each incomplete core was then calculated (BOF: mean 66 yrs, range 1-216 yrs; SFM: mean 36 yrs, range 5-148 yrs), based on the missing length (BOF: mean 4.3 cm, range 1-37.5 cm; SFM: mean 6.6 cm, range 1-50.3 cm), and added to the age of the core.

Table 2.1: Early growth rates (rings/cm) and coefficient of variation (r^2) values for regressions of dbh on core length for each species in old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas in Yosemite National Park, CA.

Species	**n**	**Dbh-Core Length Regression (r^2)**	**Early Growth (rings/cm)**
ABCO	708	0.89	6.38
CADE	346	0.63	6.01
PILA	132	0.75	5.45
PIPO	184	0.85	3.28
PSME	44	0.65	4.79
QUKE	129	0.61	4.74

a)

Species	**n**	**Dbh-Core Length Regression (r^2)**	**Early Growth (rings/cm)**
ABCO	344	0.90	6.71
CADE	474	0.79	6.73
PILA	95	0.66	5.05
PIPO	310	0.92	3.83
PSME	71	0.81	8.81
QUKE	192	0.60	5.17

b)

2.2.2 Compositional Groups

Groups of plots with similar composition were identified using agglomerative, hierarchical cluster analysis. First, importance values (IV) (maximum value = 200) were calculated for each tree species in each plot as the sum of relative basal area (BA) and relative density. Second, species IV were clustered using relative Euclidean distance and Ward's method using the PC-ORD software package (McCune and Mefford 1997). Ward's method minimizes within group variance, while maximizing between group variance (van Tongeren 1995). Basal area, density and quadratic mean diameter were then calculated for each species in each compositional group.

Variation in species composition across plots was identified by ordinating species IV using detrended correspondence analysis (DCA). DCA is a modified reciprocal averaging technique that simultaneously arranges species and stand scores along orthogonal axes that explain the largest amount of variation among samples (Gauch 1982). This variation is often related to underlying environmental gradients. Environmental variation that was associated with variation in forest composition was identified by correlating the DCA axis scores with environmental variables (elevation, slope pitch, aspect, soil depth and TRMI). Aspect values were transformed using a modified Beers transformation (cosine (45 - aspect) + 1) that scaled values from 0 (southwest slopes) to 2 (northeast slopes) (Beers et al. 1966).

2.2.3 Age Structure Patterns

Patterns in the age structure of the forest were identified at both the plot and landscape scale. At the plot scale, the number of stems for each species in 20-year age-classes was counted in each plot. The number of stems were counted for both all age-classes (20-600 yrs) and for age-classes of stems that established prior to the onset of fire suppression (>100 yrs old). At the landscape scale, groups of plots with similar species age-class distributions were identified using cluster analysis. First, the density of stems for each species in 20 year age-classes were determined for the pre-fire suppression period (stems >100 yrs old). Second, the density of stems per hectare was clustered using Ward's method and relative Euclidean distance. Variation in age structural patterns at the landscape scale and environmental variables were identified by comparing the frequency distribution of species' age-classes for plots grouped by elevation (n= 3), aspect (n=4), and topographic position (ridgetop, upper slope, mid slope, lower slope and valley bottom: n=5) for both all stems and stems >100 yrs old using Kolmogorov-Smirnov two-sample tests.

In addition, compositional shifts that may be related to fire-suppression were identified by ordinating the age-class data. First, the relative density (ha^{-1}) of stems <100 yrs old and stems >100 yrs old of each species was calculated for each of the compositional groups identified above. Then, the age-class data for each compositional group was ordinated using detrended correspondence analysis (DCA). This approach assumes that: 1) the younger stems will replace the older stems in the plot; and, 2) the difference between the density and composition of younger stems versus older stems represents a shift in the regeneration patterns related to fire suppression. Regeneration shifts in compositional groups were represented by joining the younger and older groups in DCA space with a vector. The assumption is that the understory composition represents the composition of the future forest, and the length of the vector represents the magnitude of compositional change occurring in the group (Taylor and Solem 2001).

2.2.4 Reconstruction of Reference Forest Conditions

To determine the amount of change that has occurred since the onset of the fire exclusion period, it is necessary to compare contemporary conditions with a pre-suppression reference. I reconstructed the pre-fire suppression reference conditions using dendroecological methods (Fule et al. 1997). The reference point for the reconstruction was the year of the last widespread fire (1899). Reconstructing forest conditions for an earlier date is problematic because woody material needed for the dendroecological reconstruction may have been consumed by fire in a subsequent year (Fule et al. 1997).

The characteristics of the pre-fire suppression forest were reconstructed using the following procedure. First, stems for all live trees that established after 1899 (i.e., stems ≤103 yrs old) were removed. Next, the diameters of trees (stems >10 cm dbh) that were alive in 1899 were estimated by subtracting radial growth since 1899 from the length of each core for each tree.

To accurately estimate the structural characteristics (basal area, density, size structure) of the trees in the pre-fire suppression forest trees that were dead in 2002, but alive in 1899 needed to be included in the reconstruction. To estimate the date of death for all snags and downed trees in the plots, diameter measurements, decay classes and decomposition rates were used following the method described by Fule (1997). First, dead trees were grouped into one of four classes based on the condition class of the tree: 1) recent snag (unbroken); 2) snag broken above breast height; 3) snag broken below breast height; 4) downed tree). Second, dead trees were grouped into one of seven decay classes: 1) needles present; 2) branchlets present; 3) branches and all bark present; 4) loose bark snag (>75% bark attached); 5) loose bark snag (<75% bark attached); 6) no bark attached; and 7) highly decomposed. Third, tree status and decay class were then converted to decomposition condition class (CONCLASS) 3 through 8 (Table 2.2a). Fourth, decomposition rates (i.e., movement between CONCLASS's) were determined for the average diameter of live trees established prior to the reference point for each species by calculating values for the life of a snag (snag life) and the fall rate of standing snags (snag fall) for each species. Then, decomposition rates between CONCLASS's were calculated (i.e. a tree in CONCLASS 5 must be decomposed from CONCLASS 3 to CONCLASS 4, then to CONCLASS 5). The rate from CONCLASS 3 to 4 is 20% yr^{-1}, and from CONCLASS 4 to 5 is 15% yr^{-1} (Rogers et al. 1984). Decomposition rates (yr^{-1}) between remaining CONCLASS's occur at the calculated snag fall rate for each species (Table 2.2b) (Rogers et al. 1984, Fule et al. 1997). Fifth, decomposition rates were used to calculate the date of death for each tree using the equation:

$$V_n = V_0(1+i)^n \qquad \text{(Fule et al. 1997)}$$

where V_0 is the initial quantity of trees, V_n is the quantity at year n, and i is the rate. Solving for n gives the equation:

$$\ln V_n - \ln V_0 / \ln(1 + i) = n$$

where n equals the number of years the tree has been dead for each CONCLASS. For each successive CONCLASS, the calculated number of years each tree was dead was added to the number of years calculated for the prior

Table 2.2: Decay classes (a) and decomposition rates (b) used to reconstruct historic forest conditions in old-growth, mixed conifer forests in Yosemite National Park, CA. * CONCLASS – Decomposition Condition Class of trees as defined by Fule (1997). ** Decomposition rates and death dates are example values calculated for white fir trees in BOF and SFM study areas.

Tree Status	**Decay Class**		**CONCLASS**
Live tree	Healthy		1
Live tree	Dying		2
2	1-2	=	3
2	3-4	=	4
2	5-6	=	5
3		=	6
4	1-2	=	3
4	3-4	=	4
4	5-6	=	5
4	7	=	8
5	3-4	=	4
5	5-6	=	5
5	7	=	8

a)

CONCLASS	**Decomposition Rate**	**Years Dead****	**Date of Death****
BOF			
3	-	0	2001
4	0.2	3.8	1997
5	0.15	8.8	1992
6	0.07872**	17.9	1983
7	0.07872	27.1	1974
8	0.07872	36.2	1965
SFM			
3	-	0	2001
4	0.2	3.8	1997
5	0.15	8.8	1992
6	0.09119**	16.7	1984
7	0.09119	24.6	1976
8	0.09119	32.6	1968

b)

CONCLASS (Table 2.2b). Sixth, the dbh of trees that were alive in 1899 but dead today were estimated using the following procedure. I first estimated the average annual radial growth rate for each species from 1899 to 2003 by measuring radial growth since 1899 on all tree cores. Then, the amount of radial growth from 1899 to the date of death for each dead tree was determined using the average annual growth measured from the cores. The growth since 1899 was then subtracted from the dbh in 2002 to estimate the dbh for the tree in 1899. Seventh, bark thickness was then added to the 1899 diameter for dead trees without bark. Trees in CONCLASS 3 and 4 had bark attached, therefore no correction was needed. Bark thickness was added to stemwood dbh using inside-outside bark equations for mixed conifer species in the Sierra Nevada (Dolph 1981).

Because estimates of tree death dates are based on decomposition rates which may be highly variable depending on cause of death, climate conditions, etc. (Harmon et al. 1986), I performed a sensitivity analysis to assess the impact of variable decomposition rates on the forest reconstruction. For the sensitivity analysis I determined death dates for trees for the 25^{th} (slowest), 50^{th}, and 75^{th} (fastest) percentile decomposition rates and calculated each of the structural characteristics (basal area, density, quadratic mean diameter) for the 1899 forest. To estimate death dates for different decomposition rates, the dependent variable, V_n, in the decomposition equation is set to the decomposition rate (i.e. setting V_n to 0.5 solves for the 50% decomposition rate). The reconstructions of stand density and basal area using different decomposition rates were then compared to assess the sensitivity of the results to the estimated decomposition rates.

2.2.5 Structure and Composition of 1911 Forest

I also evaluated the precision of my dendroecological forest reconstruction using an unusually detailed data set of key historic measurements of forest conditions in, and adjacent to, the Big Oak Flat study area. In 1911, a forest survey of the Stanislaus National Forest (U.S.D.A. Forest Service 1911) was extended into the South Fork of the Tuolumne river drainage within the current boundaries of Yosemite National Park as part of a National Forest Service inventory to assess timber volume. Several of the Forest Service inventory transects overlapped the study area. The lengths and widths of each transect were recorded allowing me to determine the acreage of each transect which overlapped the study area. In each transect, surveyors recorded the species of each live tree >6 inches (15.2 cm) in diameter, and grouped them by six-inch (15.2 cm) diameter classes. The dbh of dead trees over 18 inches (45.7 cm) was also recorded, but the species of dead trees was not identified, thereby making this part of the dataset less useful. Tree density and basal area per acre were also calculated for each transect, allowing me to compare the 1911 forest survey data with my dendroecological reconstruction.

2.2.6 Comparison of Historic and Contemporary Forest

The change in forest conditions since the onset of fire exclusion was assessed by comparing characteristics of the pre-fire suppression reference forest with contemporary forest characteristics. I calculated mean density, basal area and quadratic mean diameter by species, and Shannon's diversity index (Turner et al. 2001) for each plot, for the 1899 and 2002 forest and then identified any differences using a distribution free Mann-Whitney *U* test. I calculated Shannon's diversity for each date using the density of stems for each species in each size-class (ha^{-1}) in each plot. Changes in forest conditions since fire exclusion were also identified by comparing the density, basal area, quadratic mean diameter and Shannon's diversity index for the 1911 survey with the 1899 reference and contemporary forest characteristics using a distribution free Kruskal-Wallis H test. I also compared the distribution of trees by size-class and age-class for the entire study area and each compositional group for each time period using a kolmogorov-smirnov two-sample test.

2.2.7 Spatial Patterns

Examination of the type, size and intensity of spatial pattern of trees can produce insights into the development of patches, or groups of trees in a forest (White 1979, Pickett and White 1985). For example, if the regeneration of a species is limited to treefall gaps, then the distribution of stems will be clustered at a scale comparable to that produced by treefall. In contrast, a species that readily regenerates in the shade may have a random distribution not limited to the formation of canopy gaps. At the same time, past disturbance events can create opportunities for tree establishment not readily visible on the landscape (Scholl and Taylor 2006). Moderate severity fires kill only a portion of the trees in a stand, allowing regeneration to occur among canopy trees that survived the fire. This can result in clusters of trees of similar age in a multi-aged forest. Therefore, the spatial distribution of tree ages can help identify structural characteristics (i.e. cohorts) related to disturbance events. In addition, the spatial distribution of tree ages can help characterize the relative severity of the disturbance event. Consequently, the spatial structure of tree ages in the plots was tested for spatial autocorrelation. At the same time, the spatial distribution of stems within each plot can help to further identify regeneration patterns within a forest, so the spatial distribution of stems was examined using point pattern analysis to identify clumped, random and dispersed distributions of trees. Finally, a change in a disturbance regime, such as fire in mixed conifer forests, can result in a change in the spatial pattern of trees within the forest. Analysis of changes in spatial patterns between the time of last fire and the contemporary forest can help identify the impact of the removal of fire from the landscape on the spatial distribution of trees.

I analyzed the spatial patterns of trees in the study area in two ways. First, patterns of spatial autocorrelation of tree ages at the plot scale were identified with Moran's *I* statistic (Moran 1950). Spatial autocorrelation is the property of samples (trees) with similar characteristics (e.g. age) being nearer to each other than individuals with dissimilar characteristics (Upton and Fingleton

1985). Moran's *I* coefficient was calculated for 1 m distance classes (*d*) for the contemporary forest (from 1-10 m) and historic forest (from 1-30 m) to identify spatial autocorrelation at the plot scale. Because different species may exhibit different regeneration patterns following a disturbance, I also tested each species independently for spatial autocorrelation. Values of Moran's *I*(*d*) were transformed to standard deviates [z(*d*)], and plotted against distance classes, to aid identification of the spatial structure of tree age-classes. A value of zero for z(*d*) indicates a random distribution; a positive value indicates that similar aged individuals are closer to each other; and negative values indicate that similar aged stems do not occur near each other. Tree ages in both the historic and contemporary forest were examined to identify the characteristics of spatial autocorrelation in the forest at both times.

Second, spatial patterns of the distribution of stems were analyzed using Ripley's K(*t*) statistic (Ripley 1977) to identify patterns of clumping and dispersion within the plots. Whereas Moran's *I* measures the relationship between each point (tree) and its nearest neighbor, the K(*t*) function measures the distribution of all trees at a given radius (Duncan and Stewart 1991, Kenkel et al. 1997). For each stem, the number of points within a radius *t* is counted. The number of trees within the given radius is then compared to the values expected for a random (Poisson) distribution. Because the software, SpPack (Perry 2004), used to calculate Ripley's K(*t*) is calculated for square plots, I inscribed a square inside each circular plot with the grid point as the centroid, and only included stems within the square in the spatial analysis. Ripley's K(*t*) was calculated for 1 m distance classes up to 12 m to identify patterns at the plot scale in 1899. Ripley's K(*t*) was calculated for 1 m distance classes up to 6 m to identify patterns in 2002. Due to the limited number of stems present in the inscribed squares, Ripley's K(*t*) analysis could not be performed on all plots. A minimum of 10 stems was required for the analysis; consequently, in BOF only 40% of the plots were analyzed in 1899 and 2002. All plots in SFM contained fewer than 10 stems in the inscribed squares, therefore I were unable to perform the Ripley's K(*t*) analysis for the study area.

2.2.8 Fire Regimes

Fire regime characteristics (e.g. frequency, fire return interval, severity, extent, seasonality, rotation) were reconstructed using four types of data: (1) written fire records on file at Yosemite National Park; (2) fire dates from partial cross-sections removed from fire scarred trees; (3) radial growth patterns from cored trees; and (4) the age structure of trees in the plots. Fire regimes were quantified from fire scars collected from live (BOF n= 127, SFM n=96) and standing dead (BOF n= 82, SFM n=60) trees in the study areas (Figure 2.1a, b). Fire scarred trees were identified within a 9 ha circular plot centered on each grid point and an average of two fire scar samples (range 1-5) were collected at each gridpoint. Partial cross sections were removed from each fire scarred tree using a chainsaw (Arno and Sneck 1977) and the location of the tree was determined using a GPS and recorded on a topographic map. The date of each fire scar in a fire scarred sample was determined by first sanding each cross section to a high polish and then cross-dating the annual rings with a local tree-ring chronology (Ca064: Lemon Canyon (http://www.ncdc.noaa.gov/paleo/treering.html) (Holmes and Adams 1980) using standard dendroecological techniques (Stokes and Smiley 1968). The year of the tree ring containing a fire scar lesion was then identified, and the calendar year of the fire recorded. The fire history data were analyzed using the program FHX2 (Grissino-Mayer 1997).

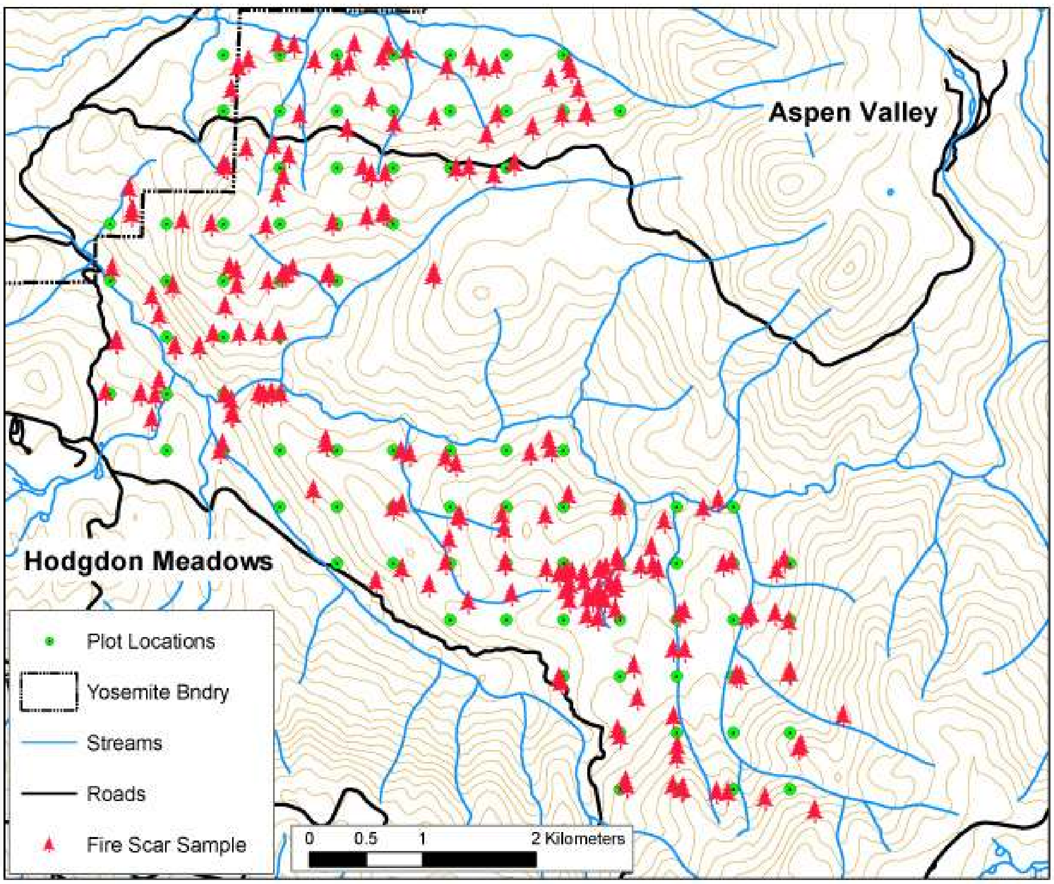


Figure 2.1a: Location of fire scar samples collected in Big Oak Flat (BOF) study area in Yosemite National Park, CA.

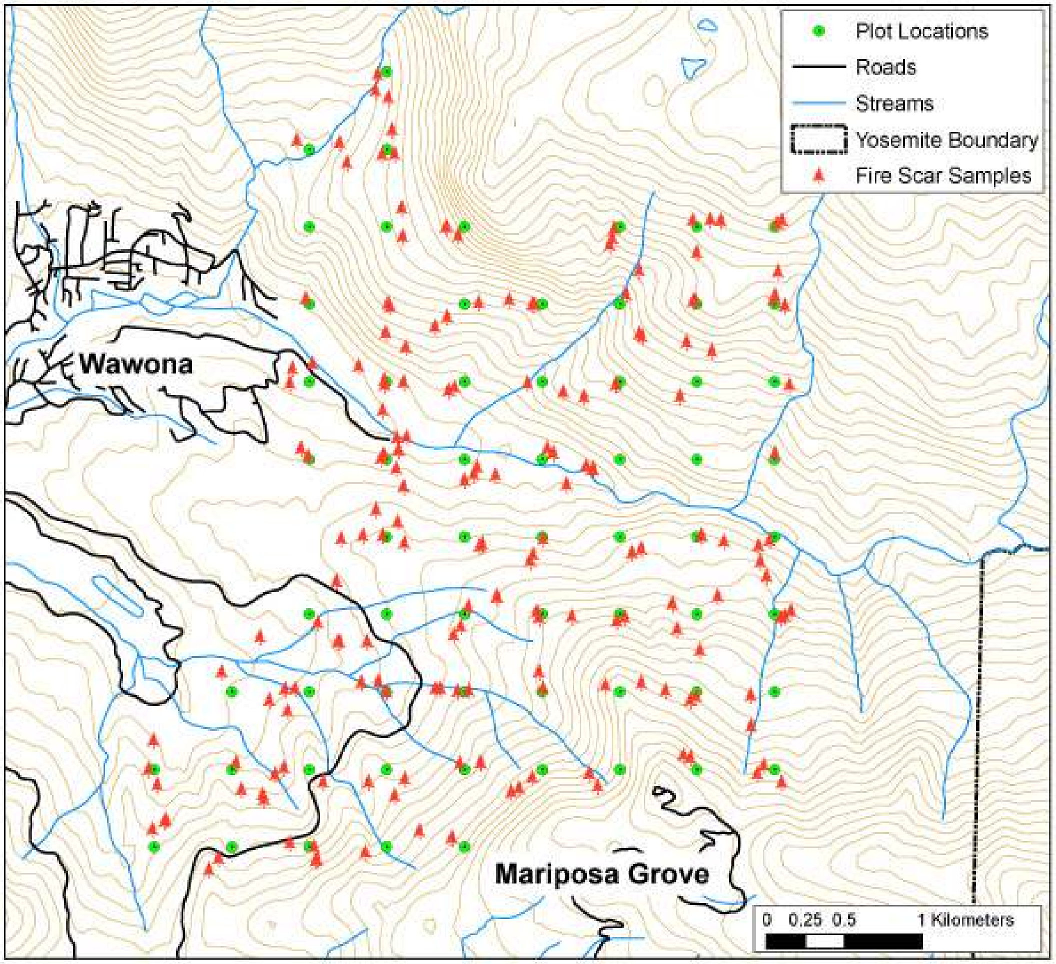


Figure 2.1b: Location of fire scar samples collected in South Fork Merced (SFM) study area in Yosemite National Park, CA.

Fire occurrence at a grid point was also inferred from variation in the radial growth patterns of cored trees. Fires that burn through a stand may affect tree growth, but not scar the tree. A tree damaged, but not scarred, by fire often exhibits a sudden decrease (suppression) in radial growth for several years after a fire, and the date of the fire corresponds with the year before the onset of suppressed growth. Conversely, a fire may kill a tree or group of trees and improve local growing conditions for surviving tree(s) resulting in an increase (release) in radial growth. In this case, the date of the disturbance corresponds with the onset of the sudden increase in radial growth. In contrast to a disturbance, abrupt, single year changes in ring width are often the result of local environmental conditions (e.g., drought). Consequently, to distinguish between radial growth variations caused by disturbances and annual variation in radial growth, suppressions or releases were defined as a 200% change in radial growth for 5 years (compared to the previous 5 years). Dates of growth suppression or release were only used to infer fire occurrence at a grid point if the date was coincident with a fire date in a fire scar sample from an adjacent grid point.

Each fire characteristic I reconstructed is important in describing the variability of the fire regime of the reference forest in Yosemite National Park. In addition, these standard characteristics allow comparisons with fire history studies in other locations. These characteristics are: seasonality; frequency; fire return interval; extent; rotation; and severity (White and Pickett 1985, Skinner and Chang 1996). Seasonality refers to the timing of the fire during the year and is important since it is related to the moisture content of fuels and also the growth of vegetation present. Frequency refers to the mean number of fires that occur during a given period of time and is important since it is related to how often a fire will burn through a given location which will influence the vegetation present. Fire return interval refers to the mean number of years between the occurrence of a fire in a particular location, and is important since it is related to the duration of time in which the site is fire free and influences the vegetation and fuel loadings present. Extent refers to the size of the area burned in a fire and is important in influencing the spatial pattern of the forest. Rotation is the length of time necessary to burn an area equal in size to the study area and is important in evaluating the cumulative area burned by fires which can influence the spatial pattern of the forest and the vegetation present. Severity refers to the impact of the fire on the vegetation present and is important in influencing the type of vegetation, structure, and spatial pattern of the forest. For example, low severity fires tend to only kill seedlings and consume surface fuels, leaving the larger trees intact, while a high severity fire may kill large numbers of canopy trees resulting in an opening in the forest.

2.2.8.1 Fire Seasonality

The relative position of each fire scar within an annual growth ring was used to infer the season each fire burned. Season of burn was identified using the following ring position categories for each fire (Baisan and Swetnam 1990): 1) early (first one-third of earlywood); 2) middle (second one-third); 3) late (last one-third); 4) latewood (in latewood); 5) dormant (at ring boundary). In this strongly winter wet, summer dry climate, dormant season fires occur in late summer or fall after seasonal growth has stopped for the year, rather than in early spring (Caprio and Swetnam 1995).

2.2.8.2 Fire Return Intervals

Spatial Variation - Spatial variation in fire return intervals (FRI) related to slope aspect and forest composition was identified by comparing mean FRIs for different slope aspect and forest compositional groups. First, I composited (i.e. averaged) the fire history for each grid point using all fires recorded for all samples collected at each grid point. Then, I assigned each grid point to a slope aspect (N = 316°-45°, E = 46º-135º, S = 136º-225º, W = 226º-315º) and a forest compositional group. Finally, I calculated the point and composite FRI for each group and the entire study area. Each fire interval measure describes different characteristics about fire in the environment. A mean grid point fire interval (PFI) was calculated for each slope aspect and compositional group from the mean FRI of each individual grid point in the group. Composite fire intervals (CFI) were developed from all of the grid points in a slope aspect or forest compositional group by calculating the average number of years between all fire dates in that group (Dieterich 1980). PFIs provide information on the time required to accumulate enough fuel to support consecutive fires at a specific point and consequently are conservative estimators of the interval between fires (Kitzberger and Veblen 1997). In contrast, CFIs are based on all recorded fires in a given area and some of these fires may not have overlapped other fires. Thus, while CFIs estimate fire occurrence over a wider area and may be less constrained by rate of fuel buildup at a specific location, they are a better measure of the number of ignitions present in an area (Kitzberger and Veblen 1997). However, CFIs vary with the size of the area sampled and they tend to shorten as the size of the study area increases (Arno and Petersen 1983, Baker 1992). Composite and point FRIs for each slope aspect and compositional group were compared using a distribution free Kruskal-Wallis H test.

Temporal Patterns - Temporal variation in fire return intervals that may be related to land use changes was assessed by comparing composite FRIs for three time periods: (1) pre-settlement (up to 1850), (2) settlement (1850-1904), and (3) fire suppression (1904-2002). A composite fire record for the entire study area was used for temporal comparisons because composite records are more sensitive to changes in ignitions or burning conditions that might influence fire occurrence at landscape scales than are point FRIs (Dieterich 1980). The statistical significance of differences in the frequencies of fire between time periods was determined using a t-test.

Fire Size – Fire size was measured indirectly through the number of grid points showing evidence of fire occurrence in a given year. I determined mean fire size for different time periods using a ratio method (Morrison and Swanson 1990, Taylor and Skinner 1998) based on the number of grid points recording a fire in a given year out of the total number of grid points with samples old enough to record the fire. Fire size was then estimated using the equation:

$$A_i = (AT \times NS_i)/(NST - NRE)$$

Where, A_i is the area burned in the ith year, AT is the size of the study area (in hectares), NS_i is the number of gridpoints with a record of the fire in the *i*th year, NST is the total number of gridpoints, and NRE is the number of sites without samples present to record the fire in the *i*th year. Because the accuracy of this method decreases as NRE increases I chose a cutoff date (AD1575) for which I had a large enough sample size (# grid points: BOF = 27, SFM = 11) to reduce errors of interpretation.

In addition, I calculated FRIs for fires of different sizes based on the number of grid points burned. I was unable to use the ratio method to determine mean FRI for fires of any given size (# hectares burned) because the fire size calculated by the ratio method changes with the number of grid points recording any given fire. Grid points were used as an index of fire size, with each grid point representing the 25-ha cell within which it was located. FRIs were then determined for fires that burned different numbers of grid points (BOF = 1-74, SFM = 41). While the number of points showing evidence of fire in a given year is not a direct measure of burn area (Taylor 2000), it is a robust index of fire size, regardless of whether trees were scarred by a single fire or multiple fires (Swetnam and Baisan 2003, Taylor and Skinner 2003). The frequency of fires that burned a given number of grid points also allowed me to compare the relative importance of fires of different size using a Kruskal-Wallis H test.

Last, I assessed the fire return intervals for more widespread fire by calculating composite FRIs for fire events that scarred 10% and 25% of the grid points (Grissino-Mayer 1996). Calculating FRIs for fires burning 10% or more and 25% or more of the grid points filters out spot fires and emphasizes the fire return interval of more widespread fires.

Fire Rotation - Fire rotation was calculated for the entire study area using the fire area estimates derived from the ratio method. Fire rotation is the number of years needed to burn an area equal in size to the study area given the frequency and extent of burning during that period (Heinselman 1973). For a given period, some parts of the study area may have burned more than once, and others not at all. Fire rotations were calculated separately for each century (pre-1700, 1700-1799, 1800-1899, and 1900-2000), and for the pre-settlement (pre-1850), settlement (1850-1904), and fire suppression (1905-present) periods because of the potential changes in fire frequency and extent associated with changes in land-use.

Fire Severity – I assessed fire severity at both the plot and landscape scale indirectly by analyzing the age structure of tree populations in the plots. Because fires burn with variable severity across the landscape, their impact on forest structure can vary from stand to stand, killing many trees in some stands and few trees in others. Stands that have experienced high severity fires which killed most or all trees in a stand are even-aged, while stands with a multi-aged structure may develop under a regime of moderate severity fires that kill only parts of a stand. In contrast, forests that experience mainly low severity fires are multi-aged, but they may have no distinct age-classes that are related to fire events (Agee 1993). Consequently, I used the number of age-classes occupied by trees in a plot as an index of fire severity. Stands with few age-classes will have experienced more severe fire than plots with several age-classes present. I counted the number of 20-year age-classes for each species that were occupied in each plot for all age-classes, and for pre-fire suppression age-classes (>100 yrs). I then used agglomerative hierarchical cluster analysis to identify groups of trees with similar age structures, using relative Euclidean distance and Ward's method. Last, I calculated the mean number of fires for each age structural group to determine the number of fires per age class.

2.2.9 Fuel Limitations and Fire Occurrence

The presence of successive fires at a grid point was used to examine the relationship between the occurrences of a fire on subsequent fires at the same location. Because fires consume fuels on the forest floor, they may influence subsequent burn patterns because of the period needed for fuel to accumulate to the point that a fire could burn the same location again. If a fire occurs at a grid point before sufficient fuels build up since the last fire, then the fire may not burn through the grid point, due to the patchy distribution of fuels. The influence of fire on subsequent fire can be assessed by examining the frequency at which successive fires burn the same grid point. I used the dates of each pair of consecutive fires in the study area to determine the frequency that the second fire in a consecutive pair burned at the same grid point, a different grid point, or the same and a different grid point. I also examined changes in fires burning consecutive grid points over time by measuring the frequencies of consecutive fires at grid points for each century (pre 1700, 1700-1799, 1800-1899, and 1900-

1999) and for the pre-settlement, settlement, and fire suppression periods. In addition, for each successive fire, I counted the number of grid points burned by only the first fire, only the second fire, and both fires in order to assess if fire size has an impact on the occurrence of an area being burned by successive fires.

2.3 RESULTS

2.3.1 Forest Structure and Composition

2.3.1.1 Big Oak Flat study area

Four forest compositional groups were identified from the cluster analysis of species importance values: 1) white fir/ incense cedar; 2) white fir/ incense cedar/ sugar pine; 3) incense cedar/ sugar pine/ Ponderosa pine; and 4) Ponderosa pine/ incense cedar/ California black oak (Table 2.3). Groups were named based on the species with the highest mean importance values.

The white fir/ incense cedar group (WF/IC, n=28) and white fir/ incense cedar/ sugar pine group (WF/IC/SP, n=28) were the most frequent type in the study area, but they were noticeably different in structure and composition. The white fir/ incense cedar group is dominated by white fir and incense cedar, with lesser amounts of sugar pine and ponderosa pine (Table 2.3). On average,

Table 2.3: Next page. Importance value, density and basal area for each compositional group identified by cluster analysis of importance values for plots of old-growth, mixed conifer forests in the Big Oak Flat study area in Yosemite National Park, CA. Compositional groups are classified by species whose importance values, density and basal area are dominant in the group. Mean environmental variables (range) are listed for each compositional group. TRMI = Topographic Relative Moisture Index.

White Fir--Incense Cedar (n=28)

	Importance Value			Density			Basal Area		
Species	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
WF	76.2	32.0	118.6	307.1	70.0	610.0	18.7	8.4	43.7
IC	45.2	0.0	80.7	151.4	0.0	470.0	18.4	0.0	69.0
SP	20.6	0.0	53.8	24.6	0.0	90.0	12.2	0.0	45.9
PP	38.7	0.0	93.3	50.4	0.0	370.0	24.2	0.0	84.9
DF	8.0	0.0	72.3	20.7	0.0	200.0	3.8	0.0	60.2
BO	7.7	0.0	65.7	28.9	0.0	360.0	1.9	0.0	16.6

White Fir--Incense Cedar--Sugar Pine (N=28)

	Importance Value			Density			Basal Area		
Species	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
WF	126.1	90.5	200.0	399.7	140.0	750.0	30.5	11.5	53.9
IC	29.3	0.0	71.8	38.2	0.0	150.0	14.3	0.0	51.2
SP	25.6	0.0	89.8	36.8	0.0	400.0	17.1	0.0	70.5
PP	9.9	0.0	59.8	5.7	0.0	30.0	5.6	0.0	36.4
DF	3.3	0.0	40.8	6.4	0.0	70.0	0.9	0.0	10.7
BO	3.4	0.0	29.7	9.3	0.0	100.0	0.9	0.0	9.3

Incense Cedar--Sugar Pine--Ponderosa Pine (n=18)

	Importance Value			Density			Basal Area		
Species	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
WF	22.9	0.0	64.9	75.6	0.0	180.0	6.3	0.0	30.4
IC	83.2	20.7	140.0	275.0	30.0	800.0	26.6	4.3	51.0
SP	34.9	0.0	98.4	131.7	0.0	380.0	8.1	0.0	27.2
PP	33.3	0.0	91.2	64.4	0.0	470.0	14.3	0.0	46.2
DF	8.7	0.0	75.2	15.0	0.0	80.0	5.1	0.0	58.4
BO	17.1	0.0	79.9	56.7	0.0	320.0	5.7	0.0	43.5

Ponderosa Pine--Incense Cedar--Black Oak (n=11)

	Importance Value			Density			Basal Area		
Species	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
WF	1.4	0.0	15.0	4.5	0.0	50.0	0.2	0.0	2.1
IC	47.8	0.0	85.5	75.5	0.0	250.0	12.1	0.0	31.7
SP	2.6	0.0	12.3	8.2	0.0	40.0	0.5	0.0	4.3
PP	99.4	50.7	174.0	93.6	30.0	230.0	30.8	10.2	45.4
DF	1.6	0.0	17.7	3.6	0.0	40.0	0.4	0.0	4.5
BO	47.3	5.4	106.8	112.7	10.0	410.0	6.2	0.2	15.9

			Aspect						
Group	Elevation (m)		Cardinal	Transformed		Slope (%)		TRMI	
WF-IC	1532.1	(1312-1738)	NW	1.2	(0-2)	10.7	(3-22)	34.3	(12-53)
WF-IC-SP	1575.5	(1334-1920)	N	1.4	(0-2)	12.2	(4-24)	35.2	(17-54)
IC-SP-PP	1525.6	(1372-1653)	SW	1.1	(0-2)	12.3	(4-22)	27.8	(15-51)
PP-IC-BO	1549.9	(1426-1657)	S	0.3	(0-2)	17.1	(4-28)	18.9	(12-28)

white fir has a higher density (307.1 stems ha^{-1}) and IV (76.2) than incense cedar (151.4 stems ha^{-1}, IV = 45.2) but their mean basal areas are similar (WF = 18.7 $m^2\,ha^{-1}$, IC = 18.4 $m^2\,ha^{-1}$). In contrast, ponderosa pine, which has the highest basal area (24.2 $m^2\,ha^{-1}$) has lower densities (50.4 stems ha^{-1}), and IV (38.7). WF/IC groups tend to occur on relatively mesic north facing slopes across all elevations and slope positions.

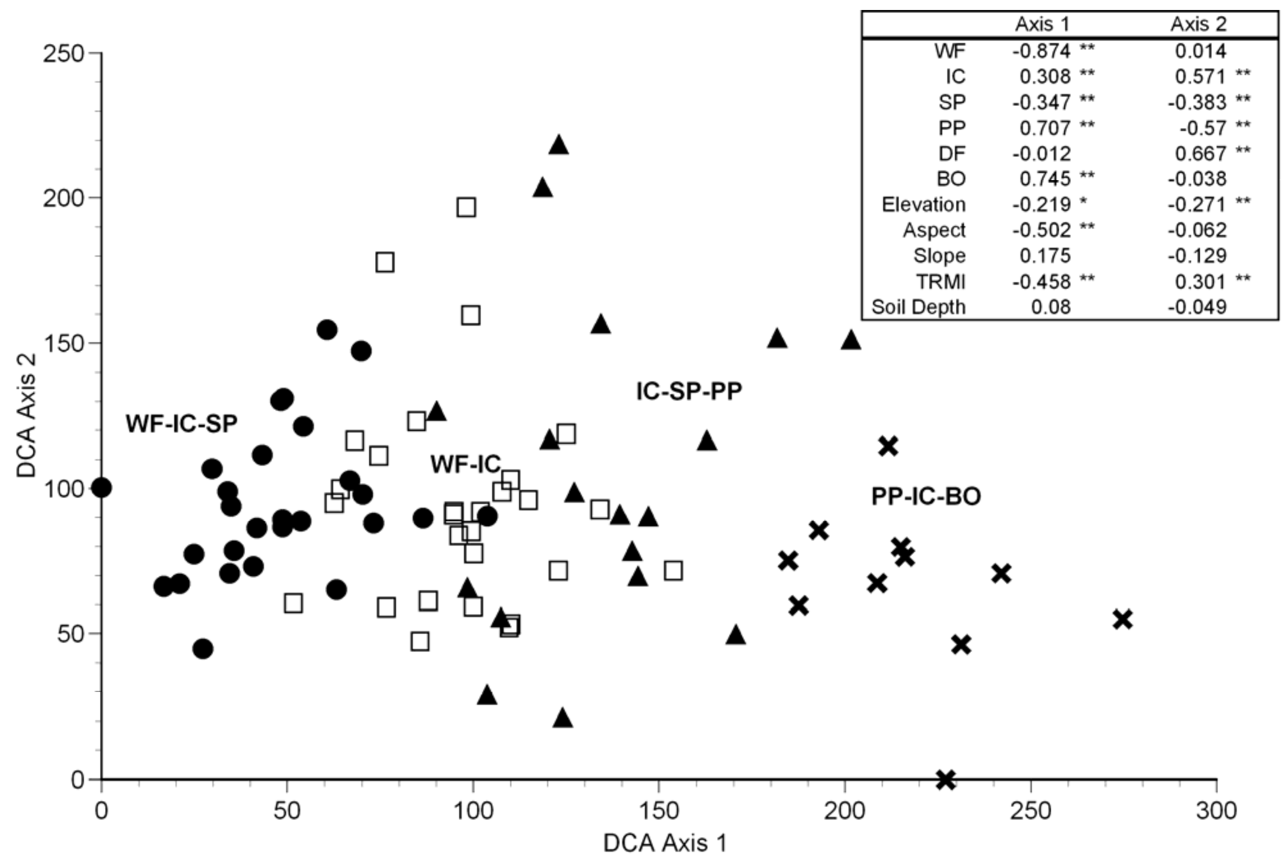
The white fir/ incense cedar/ sugar pine (WF/IC/SP, n=28) group differs from the WF/IC group in relative numbers of incense cedar and sugar pine. The group is strongly dominated by white fir and there are lesser and nearly equal values for sugar pine and incense cedar. Average density of white fir (399.7 stems ha^{-1}) is 10 fold or more greater than incense cedar (38.2 stems ha^{-1}) and sugar pine (36.8 stems ha^{-1}). The pattern for mean basal area was similar with white fir (30.5 $m^2\,ha^{-1}$) two-fold greater than incense cedar (14.3 $m^2\,ha^{-1}$) and sugar pine (17.1 $m^2\,ha^{-1}$), and mean IV, over 4 times higher (WF = 126.1, IC = 29.3, SP = 25.6). WF/IC/SP groups tend to be located on relatively mesic north facing slopes across all elevations and slope positions.

The incense cedar/ sugar pine/ ponderosa pine group (IC/SP/PP, n=18) is dominated by incense cedar, with sugar pine and ponderosa pine important for different reasons. White fir is a common associate in this group. Incense cedar values for all characteristics (IV = 83.2, density = 275 stems ha^{-1}, basal area = 26.6 $m^2\,ha^{-1}$) are higher than for other species. Although the IV for sugar pine (34.9) and ponderosa pine (33.3) are similar, the basal area for ponderosa pine (14.3 $m^2\,ha^{-1}$) is higher than sugar pine (8.1 $m^2\,ha^{-1}$), while the density was lower

(PP = 64.4 stems ha^{-1}, SP = 131.7 stems ha^{-1}). In contrast, while the density of white fir (75.6 stems ha^{-1}) is higher than ponderosa pine, white fir IV (22.9) and basal area (6.3 m^2 ha^{-1}) is lower. IC/SP/PP groups were primarily located at mid-slope positions on south facing slopes.

The ponderosa pine/ incense cedar/ black oak group (PP/IC/BO, n=11) is the smallest group, and is dominated by ponderosa pine with lesser amounts of incense cedar and black oak. Ponderosa pine values for all characteristics are higher (IV = 99.4, density = 93.6 stems ha^{-1}, basal area = 30.8 m^2 ha^{-1}) than for other species. The IV are lower for both incense cedar (47.8) and black oak (47.3), density (IC = 75.5 stems ha^{-1}, BO = 112.7 stems ha^{-1}) and basal area (12.1 m^2 ha^{-1}, BO = 6.2 m^2 ha^{-1}) varies. The relatively low basal areas for incense cedar and black oak are due to high numbers of small stems. The PP/IC/BO plots were located entirely on xeric south and west facing slopes.

The ordination of species IV separated plots based on species composition, which is related to environmental gradients in the study area (Figure 2.2). The first DCA axis is negatively correlated with slope aspect ($r = -0.502$, $p<0.01$), TRMI ($r = -0.458$, $p<0.01$), and elevation ($r = -0.219$, $p<0.05$) and separates more xeric, higher elevation pine dominated sites from more mesic, lower elevation fir dominated sites. Both white fir and sugar pine have significant negative correlations ($p<0.01$) with axis 1, while incense cedar, ponderosa pine, and California black oak have significant positive correlations ($p<0.01$). According to the species correlations with the DCA scores, the first axis appears to represent a complex environmental gradient that ranges from relatively mesic



	Axis 1	Axis 2
WF	-0.874 **	0.014
IC	0.308 **	0.571 **
SP	-0.347 **	-0.383 **
PP	0.707 **	-0.57 **
DF	-0.012	0.667 **
BO	0.745 **	-0.038
Elevation	-0.219 *	-0.271 **
Aspect	-0.502 **	-0.062
Slope	0.175	-0.129
TRMI	-0.458 **	0.301 **
Soil Depth	0.08	-0.049

Figure 2.2: Detrended correspondence analysis ordination of plots (n=85) based on species importance values for old-growth, mixed conifer forests in the Big Oak Flat study area in Yosemite National Park, CA. Compositional groups identified by cluster analysis of species importance values are indicated by different symbols: ● = WF-IC-SP (white fir - incense cedar - sugar pine), □ = WF-IC (white fir - incense cedar), ▲ = IC-SP-PP (incense cedar - sugar pine - ponderosa pine), ✖ = PP-IC-BO (ponderosa pine - incense cedar - black oak). Pearson's product moment correlations of axis scores with species importance values and environmental variables are given in the panel (* $p<0.05$, ** $p<0.01$).

north facing slopes (fir and cedar dominated), to relatively xeric south facing slopes (pine and oak dominated). Although elevation is significantly correlated with Axis 1, the range of elevations in all compositional groups overlaps. The second DCA axis is significantly correlated with elevation ($r = -0.271$, $p<0.01$) and TRMI ($r = 0.301$, $p<0.01$). Douglas-fir and incense cedar are both positively correlated ($P<0.01$) with axis 2 while sugar pine and ponderosa pine have significant negative correlations ($P<0.01$). Axis 2 is related to the subdivision of plots in each compositional group between those dominated by fir or by pine.

2.3.1.2 South Fork Merced study area

Four compositional groups were identified from the cluster analysis of species importance values: 1) white fir/ incense cedar; 2) incense cedar/ ponderosa pine; 3) white fir/ sugar pine; and 4) ponderosa pine/ incense cedar (Table 2.4).

The white fir/ incense cedar group (WF/IC, n=16) is similar in characteristics to the white fir/ incense cedar group identified in Big Oak Flat and is dominated by white fir and incense cedar with lesser amounts of ponderosa pine and sugar pine. White fir has the highest density (331.9 stems ha^{-1}) and IV (72.5) while incense cedar has the highest basal area (20.7 m^2 ha^{-1}). WF/IC groups in both study areas occur in similar locations.

Table 2.4: Next page. Importance value, density and basal area for each compositional group identified by cluster analysis of importance values for plots of old-growth, mixed conifer forests in the South Fork Merced study area in Yosemite National Park, CA. Compositional groups are classified by species whose importance values, density and basal area are dominant in the group. Mean environmental variables (range) for compositional groups. TRMI = Topographic Relative Moisture Index.

White fir -- Incense cedar (n=16)

Species	Importance Value			Density			Basal Area		
	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
WF	72.5	49.1	103.1	331.9	120	710	18.2	4.4	31.3
IC	63.0	23.3	123.7	318.8	30	920	20.7	3.2	60.6
SP	15.7	0.0	36.6	28.8	0	100	10.0	0.0	30.6
PP	31.7	0.0	78.8	33.1	0	100	18.6	0.0	43.5
DF	4.0	0.0	36.1	11.9	0	140	1.4	0.0	16.1
BO	13.2	0.0	51.8	38.8	0	140	4.4	0.0	15.9

Incense cedar -- Ponderosa pine (n=17)

Species	Importance Value			Density			Basal Area		
	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
WF	9.2	0.0	35.4	44.7	0	180	1.7	0.0	11.5
IC	83.1	8.5	132.1	468.8	40	1550	22.0	0.4	51.6
SP	18.3	0.0	62.8	44.7	0	160	7.2	0.0	24.9
PP	35.9	0.0	77.1	98.2	0	510	16.7	0.0	39.9
DF	24.1	0.0	87.6	94.7	0	580	5.8	0.0	22.5
BO	29.5	0.0	98.5	97.6	0	340	8.1	0.0	29.5

White fir -- Sugar pine (n=10)

Species	Importance Value			Density			Basal Area		
	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
WF	103.9	48.5	195.4	425.0	140	1140	24.7	7.2	46.6
IC	14.8	0.0	51.6	53.0	0	140	4.1	0.0	20.2
SP	58.7	0.0	92.2	78.0	0	150	29.8	0.0	52.7
PP	0.0	0.0	0.0	0.0	0	0	0.0	0.0	0.0
DF	17.9	0.0	82.2	39.0	0	210	8.4	0.0	34.2
BO	4.8	0.0	25.3	13.0	0	50	0.9	0.0	4.1

Ponderosa pine -- Incense cedar (n=21)

Species	Importance Value			Density			Basal Area		
	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
WF	2.5	0.0	28.0	6.2	0	110	1.2	0.0	13.5
IC	40.6	0.0	86.8	126.2	0	620	10.3	0.0	29.6
SP	6.7	0.0	60.6	20.0	0	210	1.7	0.0	14.4
PP	127.3	79.9	200.0	218.1	40	750	37.4	16.7	63.8
DF	1.6	0.0	24.1	2.4	0	40	0.2	0.0	2.4
BO	21.3	0.0	69.3	47.6	0	190	3.0	0.0	9.1

Group	Elevation (m)		Aspect			Slope (%)		TRMI	
			Cardinal	Transformed					
WF-IC	1524.2	(1314-1977)	NW	1.0	(0-2)	16.9	(4-26)	32.6	(18-46)
IC-PP	1466.4	(1302-1885)	NNW	1.0	(0-2)	14.1	(3-27)	34.1	(14-55)
WF-SP	1779.9	(1597-1967)	N	1.5	(0.4-2)	23.4	(11-35)	29.8	(19-42)
PP-IC	1566.7	(1262-2058)	SW	0.4	(0-2)	19.1	(4-32)	24.8	(12-51)

The incense cedar / ponderosa pine group (IC/PP, n=17) is dominated by incense cedar with lesser amounts of ponderosa pine. Incense cedar values for all characteristics (IV = 83.1, density = 468.8 stems ha^{-1}, basal area = 22 m^2 ha^{-1}) are higher than for other species. Ponderosa pine is the next most important species, and while values for IV (35.9) and density (98.2 stems ha^{-1}) are less than half that of incense cedar, basal area (16.7 m^2 ha^{-1}) is only slightly less than incense cedar, but over two-fold higher than other species. IC/PP groups tend to occur at mid slope positions across all aspects.

The white fir/ sugar pine group (WF/SP, n=10) is the smallest group and is dominated by white fir with lesser amounts of sugar pine. On average, white fir has higher IV (103.9) and density (425 stems ha^{-1}) than sugar pine (IV = 58.7, density = 78 stems ha^{-1}), while lesser, but similar basal areas (WF = 24.7 m^2 ha^{-1}, SP = 29.8 m^2 ha^{-1}). Other species have much lower IV, density and basal areas, and there are no ponderosa pine stems present in the group. WF/SP groups tend to occur at higher elevations on north facing aspects

The ponderosa pine/ incense cedar group (PP/IC, n=21) is the largest group and is strongly dominated by ponderosa pine across all characteristics (IV = 127.3, density = 218.1 stems ha^{-1}, basal area = 37.4 m^2 ha^{-1}). Ponderosa pine IV and basal area are over three-fold higher than incense cedar (IV = 40.6, basal area = 10.3 m^2 ha^{-1}), and density is nearly double (126.2 stems ha^{-1}). Values for other species are less than half those for incense cedar. PP/IC groups tend to occur on south facing aspects across all elevations and slope positions.

The ordination of species IV in the South Fork Merced study area separated plots along different environmentally related axes (Figure 2.3). The first DCA axis is significantly positively correlated with elevation ($r = 0.262$, $p<0.05$), slope aspect ($r = 0.559$, $p<0.01$), and TRMI ($r = 0.255$, $p<0.05$), and negatively correlated with soil depth ($r = -0.336$, $p<0.01$), and separates lower elevation sites dominated by pines from higher elevation sites dominated by firs. Both white fir and sugar pine have significant positive correlations ($p<0.01$) with axis 1, while ponderosa pine and black oak have significant negative correlations ($p<0.01$ and $p<0.05$ respectively). The first axis appears to represent a complex environmental gradient from relatively xeric, low elevation sites on southerly aspects (ponderosa pine dominated) to relatively mesic, high elevation sites on northerly aspects (white fir dominated), although the range of elevation, aspect and TRMI values overlap between groups. The second DCA axis is only significantly correlated with aspect ($r = -0.379$, $p<0.01$). White fir and sugar pine are both negatively correlated ($p<0.01$) with Axis 2 while ponderosa pine, Douglas fir and black oak have significant positive correlations ($p<0.01$). Axis 2 is related to the subdivision of plots in each compositional group between those occurring on more north-facing slopes with those occurring on more south-facing slopes. Although this was often associated with Douglas-fir, it was not a consistent trend due to the limited number of plots with Douglas-fir ($n = 16$).

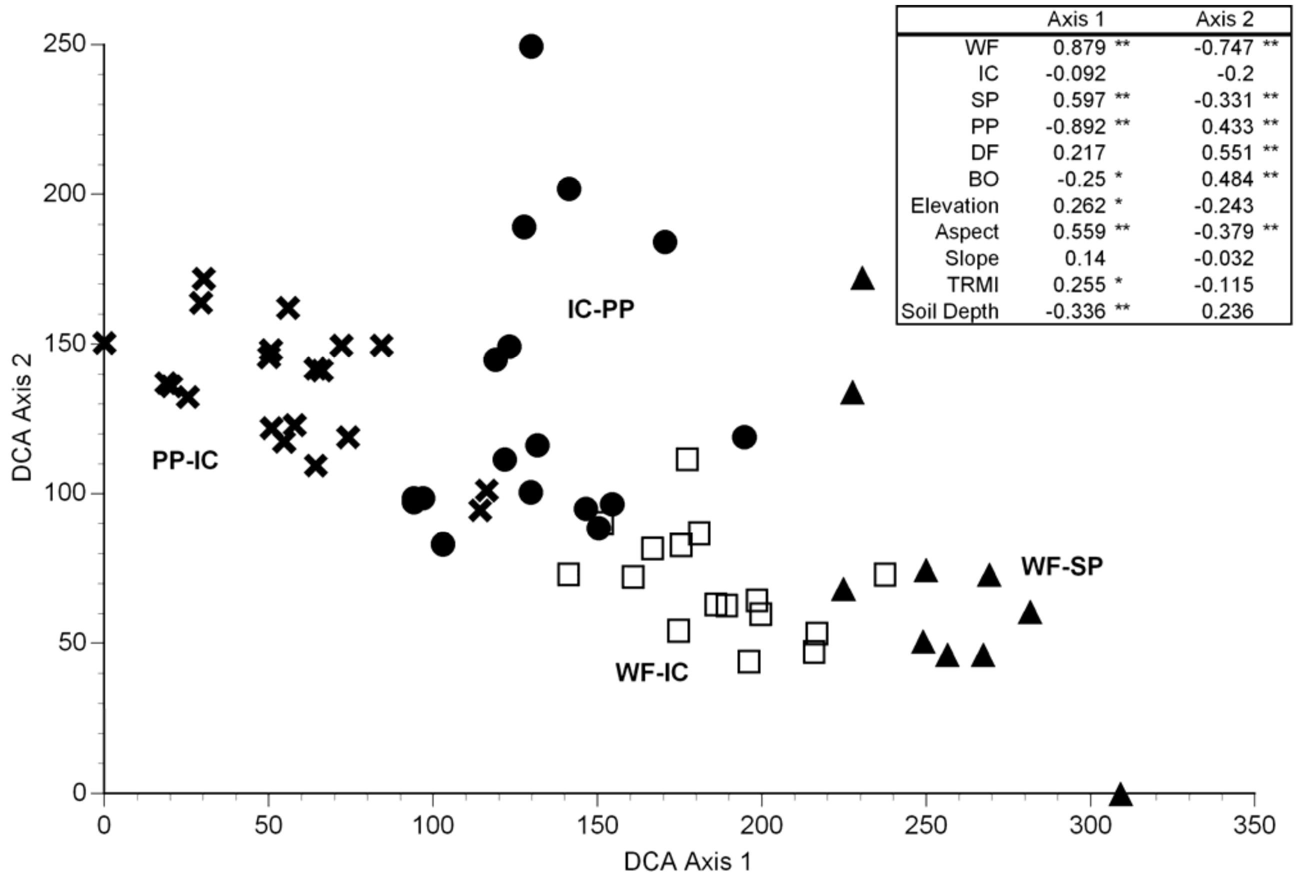


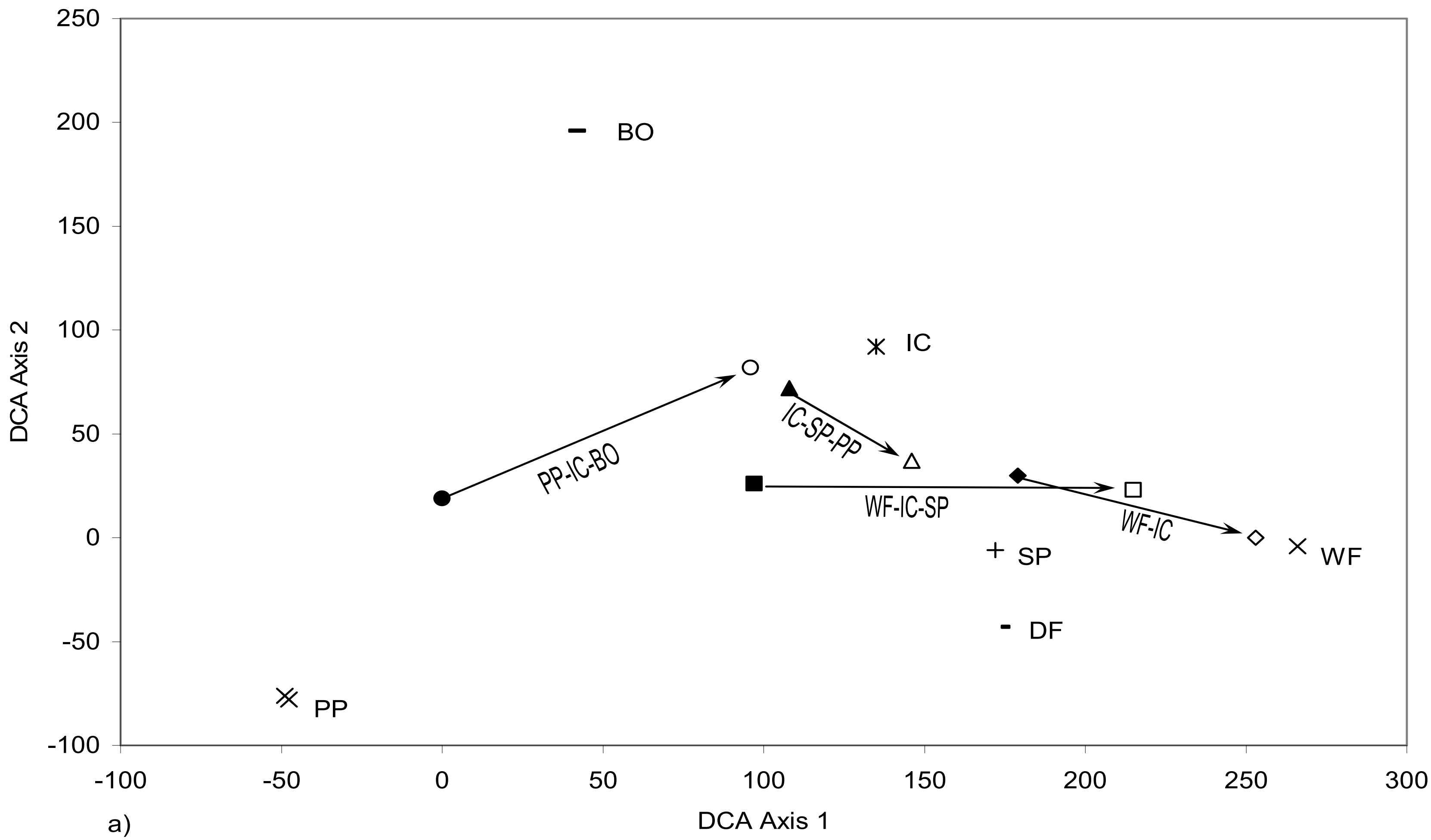
Figure 2.3: Detrended correspondence analysis ordination of plots (n=64) based on species importance values for old-growth, mixed conifer forests in the South Fork Merced study area in Yosemite National Park, CA. Compositional groups identified by cluster analysis of species importance values are indicated by different symbols: ✖ = PP-IC (ponderosa pine - incense cedar), □ = WF-IC (white fir - incense cedar), ● = IC-PP (incense cedar - ponderosa pine), ▲ = WF-SP (white fir - sugar pine). Pearson's product moment correlations of axis scores with species importance values and environmental variables are given in the panel (* $p<0.05$, ** $p<0.01$).

2.3.2 Forest Compositional Change

The ordination of trees <100 yrs old and trees >100 yrs old shows that the majority of the regeneration since fire suppression has been incense cedar and white fir in both study areas (Figure 2.4a, b). The difference between understorey (<100 yrs) and overstorey (>100 yrs) trees is represented by the vectors in the diagram. The vectors are drawn from the overstorey location to the understorey location for a group, and the length of the vector is proportional to the degree of compositional difference between the age groups.

In BOF, the degree of compositional change differed among compositional groups (Figure 2.4a) with the greatest change occurring in the white fir/incense cedar and ponderosa pine/incense cedar/black oak groups. The white fir/incense cedar and white fir/incense cedar/sugar pine groups are moving from a predominantly incense cedar and sugar pine forest to one dominated by white fir. The ponderosa pine/incense cedar/black oak group shows signs of moving from ponderosa pine dominated stands to black oak/incense cedar dominated stands. Plots in the incense cedar/sugar pine/ponderosa pine demonstrated the least change with only increased dominance of incense cedar over time.

Figure 2.4: Next page. Ordination of trees < 100 years old and > 100 years old for each compositional group identified in old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas in Yosemite National Park, CA. Compositional groups are identified: WF-IC-SP = white fir - incense cedar - sugar pine, WF-IC = white fir - incense cedar, IC-SP-PP = incense cedar - sugar pine - ponderosa pine, PP-IC-BO = ponderosa pine - incense cedar - black oak, PP-IC = ponderosa pine - incense cedar, WF-IC = white fir - incense cedar, IC-PP = incense cedar - ponderosa pine, WF-SP = white fir - sugar pine. Compositional centers for each species are indicated. Solid symbols represent canopy trees (> 100 years old) and open symbols represent understory trees (<100 years old).



a)

Figure 2.4: Page 1 of 2.

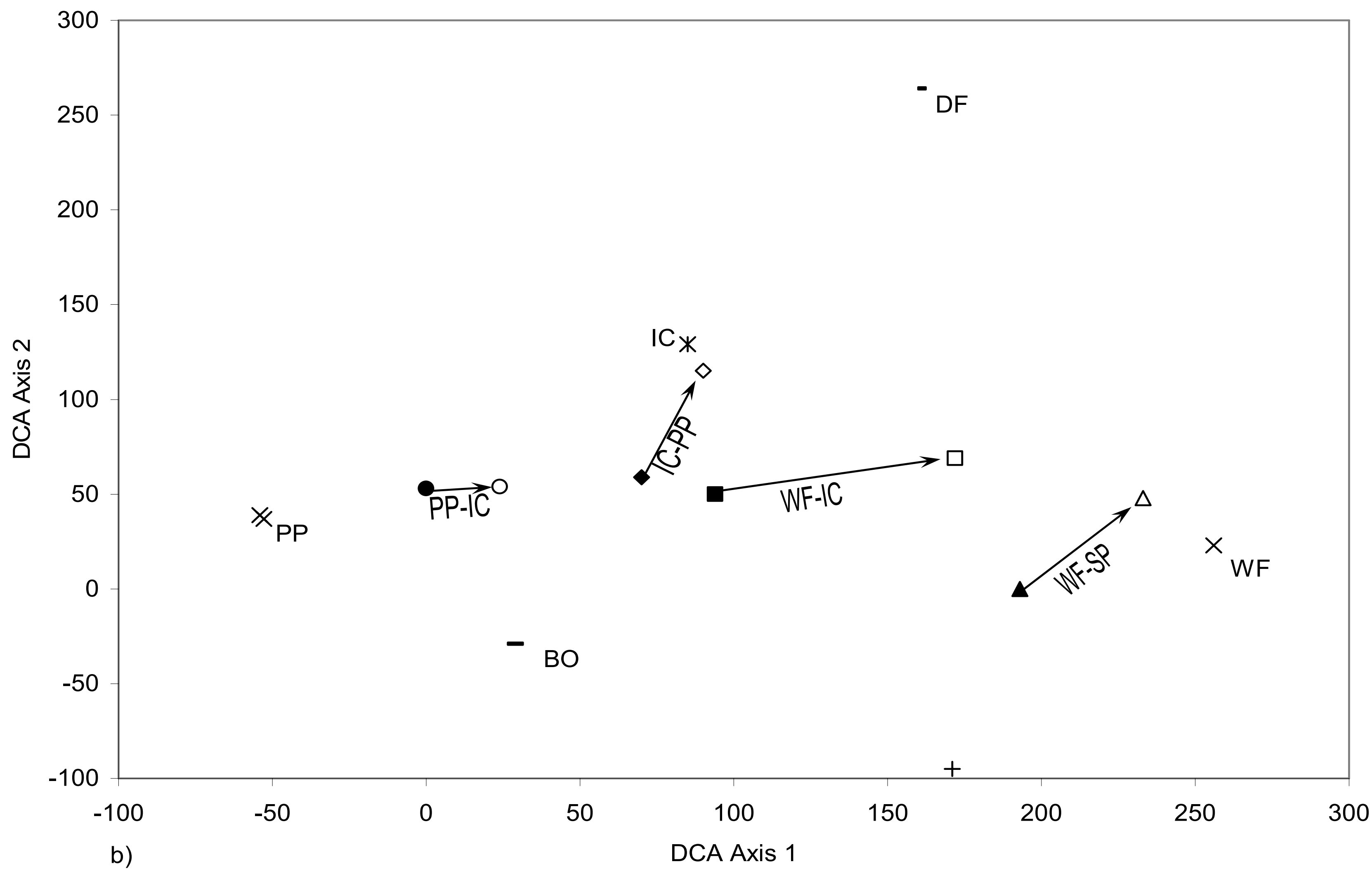


Figure 2.4: Page 2 of 2.

In SFM, the amount of compositional change is different among groups (Figure 2.4b) with the greatest amount of change occurring in the white fir/incense cedar and white fir/sugar pine groups. The white fir/incense cedar group is moving from a predominantly incense cedar structure to one that will be dominated by both incense cedar and white fir, while the white fir/sugar pine group is moving to a structure even more heavily dominated by white fir. Plots in the incense cedar/ponderosa pine group show signs of moving towards even stronger incense cedar dominance. The ponderosa pine/incense cedar group showed the least amount of change, shifting from a pine-dominated structure to one with slightly more numbers of incense cedar.

2.3.3 Reconstruction of Reference Forest Conditions

The sensitivity analysis for the decomposition model used in the forest reconstruction indicates that using different decomposition rates produces only minor differences in the estimated characteristics of the pre-fire suppression forest conditions (Table 2.5a, b). There were no significant differences in the average density, basal area and diameter reconstructed using the different decomposition rates for both study areas ($p>0.05$, Kruskal-Wallis H test). A large number of trees under all three decomposition models (25^{th} = 392, 50^{th} = 467, 75^{th} = 501) were assigned death dates late enough in the 20^{th} century, that when grown backwards, their germination dates were after 1899. These trees were not included in the reconstructions.

Table 2.5: Page 1 of 2. Sensitivity analysis of decomposition rates used in reconstruction of reference forest conditions of old-growth, mixed conifer forests in a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA.

		Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
		Mean	**SD**	**Range**	**Mean**	**SD**	**Range**	**Mean**	**SD**	**Range**
Abco										
	25%	23.1	29.4	0-150	3.9	7.1	0-34.4	37.6	24.1	11-120.9
	50%	20.9	27.9	0-140	3.4	6.5	0-32.5	37.1	23.6	10.1-117.4
	75%	19.2	26.6	0-130	3.1	6.1	0-31.4	38.1	24.2	12.2-115.4
Cade										
	25%	44.1	43	0-290	11.8	11.3	0-56.2	59.4	25.6	12.9-118.5
	50%	43.6	43.3	0-290	11.4	10.8	0-52.9	59	25.2	12.9-118.1
	75%	43.2	43	0-290	11.2	10.6	0-51.2	58.7	25.2	12.9-118.1
Pila										
	25%	14.8	19.9	0-100	7.9	12.9	0-64.8	82.2	49.8	12.8-217.7
	50%	14.5	19.3	0-100	7.6	12.2	0-61.3	81.6	48.9	12.1-208.7
	75%	14.4	19.2	0-100	7.4	11.9	0-59.4	81	48.1	12.1-203.4
Pipo										
	25%	36.5	39.8	0-160	12.4	11.3	0-41.5	71.4	31.2	10.1-157.9
	50%	35.8	38.9	0-160	11.9	10.9	0-39.7	71.1	30.2	17.5-157.9
	75%	34.9	38.7	0-160	11.8	10.8	0-39.1	71.1	30	20.7-157.9
Psme										
	25%	3.1	13.4	0-110	1.3	4.9	0-38.8	79.8	41.7	16.3-134.1
	50%	3.1	13.4	0-110	1.2	4.8	0-37.9	77.7	42.4	16.3-134.1
	75%	3.1	13.4	0-110	1.2	4.8	0-37.5	76.4	43	16.3-134.1
Quke										
	25%	11.8	24.7	0-140	0.8	1.7	0-9.8	28.8	12.1	10.8-65.6
	50%	11.8	24.7	0-140	0.8	1.7	0-9.1	28.4	12.2	10.8-65.6
	75%	11.8	24.7	0-140	0.8	1.6	0-8.7	28.1	12.3	10.8-65.6
All Trees										
	25%	134.5	76.7	30-450	38.1	21.2	4.9-107.1	62.9	20.9	28.4-134.3
	50%	130	75.1	30-450	36.3	20.4	4.9-103.7	62.2	20.6	28.3-131.2
	75%	126.7	75.1	30-450	35.5	20	4.9-101.9	62.6	20.8	29-129.4

a)

Table 2.5: Page 2 of 2.

	Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
	Mean	**SD**	**Range**	**Mean**	**SD**	**Range**	**Mean**	**SD**	**Range**
Abco									
25%	24.1	40.0	0-190	4.4	7.9	0-36.4	51.6	25.5	14.3-114.4
50%	21.6	36.4	0-150	4.1	7.6	0-36.2	51.4	26.7	13-102.2
75%	20.2	34.1	0-140	4	7.4	0-36.1	51.5	27.5	11.7-97.1
Cade									
25%	35.6	42.4	0-200	10.7	11.6	0-57.1	66.8	28.4	18.1-156.2
50%	35.6	42.4	0-200	10.6	11.5	0-56.6	66.5	28.4	17.3-153.7
75%	35.6	42.4	0-200	10.6	11.4	0-56.4	66.2	28.4	16.8-152.3
Pila									
25%	16.3	25.4	0-130	9.8	15.1	0-51	88.2	38.3	11.2-140.9
50%	15.6	24.7	0-130	9.6	14.8	0-51	90.7	35.7	17.6-140.9
75%	15.5	24.6	0-130	9.6	14.7	0-51	90.4	35.6	17.3-140.9
Pipo									
25%	41.1	49.4	0-340	19.3	17.1	0-62.7	82.7	30.9	14-171.7
50%	40.2	49	0-340	19.1	17	0-62.5	82.8	30.7	14-170.7
75%	39.1	49	0-340	19	16.9	0-62.3	84.8	30	18.7-170.2
Psme									
25%	3.9	12.6	0-70	2.4	7	0-38.3	98.3	51.6	11.7-177
50%	3.3	10.5	0-70	2.4	6.9	0-37.5	99.3	49.7	10.6-177
75%	3.1	10.5	0-70	2.3	6.8	0-37.1	108.8	41.4	48-177
Quke									
25%	21.3	30.8	0-120	3.2	4.6	0-20.3	44.6	17.6	13.7-82.3
50%	20.8	30.3	0-120	3.1	4.6	0-20.3	44.4	17.8	11.5-82.3
75%	20.8	30.3	0-120	3.1	4.6	0-20.3	44.1	18	10.2-82.3
All Trees									
25%	142.2	91.2	0-520	49.9	23	5.9-99.2	72.1	22.3	35.5-130.1
50%	137	88.7	0-520	49	22.8	5.9-99	72.6	22.2	36.3-139.9
75%	134.2	88.9	0-520	48.5	22.7	5.3-99	73.6	22.3	36.9-139.9

b)

Estimates of tree density, basal area and quadratic mean diameter for each study area in 1899 was similar ($p>0.05$) between the models. In BOF, the 25th percentile decomposition rate estimated tree densities of 134.5 stems ha^{-1}, a 3.5% increase over the 50th percentile model, while the 75th percentile model estimated stem densities at 126.7 stems ha^{-1}, a 2.5% decrease. A similar trend was present in SFM (25th = 142.2 stems ha^{-1}, 50th = 137.0 stems ha^{-1}, 75th = 134.2 stems ha^{-1}). Differences in the models for basal area were similar. Big Oak Flat basal area for the 25th percentile model was estimated higher (38.1 m^2 ha^{-1}, 5% increase) than the 50th percentile, while the 75th percentile model was estimated lower (35.5 m^2 ha-1, 2.2% decrease) than the 50th percentile. In SFM, basal area was estimated higher for the 25th percentile (49.9 m^2 ha^{-1}, 1.8% increase) and lower for the 75th percentile (48.5 m^2 ha^{-1}, 1% decrease) compared to the 50th percentile. In contrast, the quadratic mean diameter for BOF was different with the 25th and 75th percentile models predicting higher mean diameters (25th = 62.9 cm, 1.1% increase, 75th = 62.6 cm, 1% increase). The pattern in SFM was reversed with the 25th percentile predicting lower mean diameters (72.1 cm, 0.7% decrease) and the 75th percentile predicting higher mean diameters (75th = 73.6 cm, 1.4% increase). When the sensitivity analysis was performed for the individual species, the variation in structural characteristics was similar to that for the whole study area. The greatest amount of variation occurred for white fir in both locations. Due to the limited impact of the different decomposition rates on the structural characteristics of the forest, the middle rate (50th percentile) was chosen for the reconstruction.

2.3.4 Comparison of Forest Structure and Composition

Overall, the pre-fire suppression mixed conifer forests were different than contemporary forests in both study areas (Table 2.6a, b). The contemporary forest is denser, has more basal area, and the average quadratic mean diameter (QMD) of trees is smaller ($p<0.001$). On an individual species basis, white fir and incense cedar showed a two-fold or greater increase in density in both study areas ($p<0.001$). In BOF and SFM, average basal area of white fir and incense cedar increased ($p<0.05$), while the average diameter of both species decreased ($p<0.05$). The size and age structure of the forests in BOF (Figure 2.5a, 2.6a) and SFM (Figure 2.5b, 2.6b) were different between 1899 and 2002. The shape of the diameter distributions for incense cedar was different ($p<0.01$) in both study areas while the size-class distribution of sugar pine was different ($p<0.05$) only in BOF, and Douglas-fir was only different ($p<0.01$) in SFM. The average shape of the age-class distributions was different for all species ($p<0.01$) in both study areas. Within each compositional group there was a greater variety of change between the 1899 and 2002 forest than for the whole study area.

Table 2.6: Page 1 of 2. Comparison of contemporary and reconstructed old-growth, mixed conifer forests in a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA. Calculations are based upon 10 cm size-classes (* $p<0.05$, ** $p<0.01$, *** $p<0.001$).

	Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
	Mean	**SD**	**Range**	**Mean**	**SD**	**Range**	**Mean**	**SD**	**Range**
Abco									
2002	249.4	197.5	0-750	17.5	13.8	0-53.9	32.1	11.9	15.3-70.2
1899	20.9***	27.9	0-140	3.4***	6.5	0-32.5	37.1	23.6	10.1-117.4
Cade									
2002	128.1	152.2	0-800	18	14.8	0-69	56.8	31.8	12.2-118.7
1899	43.6***	43.3	0-290	11.4**	10.8	0-52.9	59	25.2	12.9-118.1
Pila									
2002	48.7	89.4	0-400	11.4	15.5	0-70.5	80.8	59.1	10.2-215
1899	14.5*	19.3	0-100	7.6	12.2	0-61.3	81.6	48.9	12.1-208.7
Pipo									
2002	44.2	75.1	0-470	16.8	18.4	0-84.9	86.2	33.1	22-159.5
1899	35.8	38.9	0-160	11.9	10.9	0-39.7	71.1**	30.2	17.5-157.9
Psme									
2002	12.6	33.4	0-200	2.7	9.7	0-60.2	45.9	25.3	14.5-111.3
1899	3.1	13.4	0-110	1.2	4.8	0-37.8	77.7	42.4	16.3-134.1
Quke									
2002	33.5	67.2	0-360	2.9	6.4	0-43.5	31.9	12.8	13.4-72.9
1899	11.8	24.7	0-140	0.8	1.7	0-9.1	28.4	12.2	10.8-65.6
All Trees									
2002	516.1	242.4	90-1220	69.4	25.9	20.7-150.8	43.5	11.3	19.2-80.6
1899	129.6***	75.5	30-450	36.3***	20.4	4.9-103.7	62.5***	20.9	28.3-131.2

a)

Table 2.6: Page 2 of 2.

	Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
	Mean	**SD**	**Range**	**Mean**	**SD**	**Range**	**Mean**	**SD**	**Range**
Abco									
2002	162	228.4	0-1140	9.3	12.4	0-46.6	28.9	15	14.4-92.8
1899	21.6***	36.4	0-150	4.1*	7.6	0-36.2	51.4***	26.7	13-102.2
Cade									
2002	249.1	321.5	0-1470	15	12.9	0-60.6	35.9	21.9	11.3-121.3
1899	35.6***	42.4	0-200	10.6*	11.5	0-56.6	66.5***	28.4	17.3-153.7
Pila									
2002	37.2	49.1	0-170	9.6	13.4	0-52.7	67.9	44.6	11.1-177
1899	15.6*	24.7	0-130	9.6	14.8	0-51	90.7*	35.7	17.6-140.9
Pipo									
2002	104.7	153.5	0-750	21.4	17.4	0-63.8	72	39.3	18.1-221
1899	40.2	49	0-340	19.1	17	0-62.5	82.8*	30.7	14-170.7
Psme									
2002	34.4	96.3	0-580	3.3	7.2	0-34.2	45.2	30.2	16.5-141
1899	3.3	10.5	0-70	2.4	6.9	0-37.5	99.3**	49.7	10.6-177
Quke									
2002	50.2	65.3	0-300	4.3	6	0-29.5	33.4	11.7	14.8-64.3
1899	20.8*	30.3	0-120	3.1	4.6	0-20.3	44.4**	17.8	11.5-82.3
All Trees									
2002	637.5	412.4	90-2060	63	21.2	29.1-120	40.5	14.5	19.1-84
1899	137***	88.7	10-520	49***	22.8	5.9-99	72.6***	22.2	36.3-129.8

b)

Figure 2.5: Next page. Mean (± SE) density of trees (>10 cm dbh) in 10 cm diameter size-classes in the reference (1899) and contemporary a) Big Oak Flat and b) South Fork Merced mixed conifer forests, Yosemite National Park, CA. Note that the y-axis scale is different on each graph. Species acronyms are Abco (Abies concolor), Cade (Calocedrus decurrens), Pila (Pinus lambertiana), Pipo (Pinus ponderosa), Psme (Pseudotsuga menziesii), Quke (Quercus kelloggii). Values are for trees >10 cm dbh on each date.

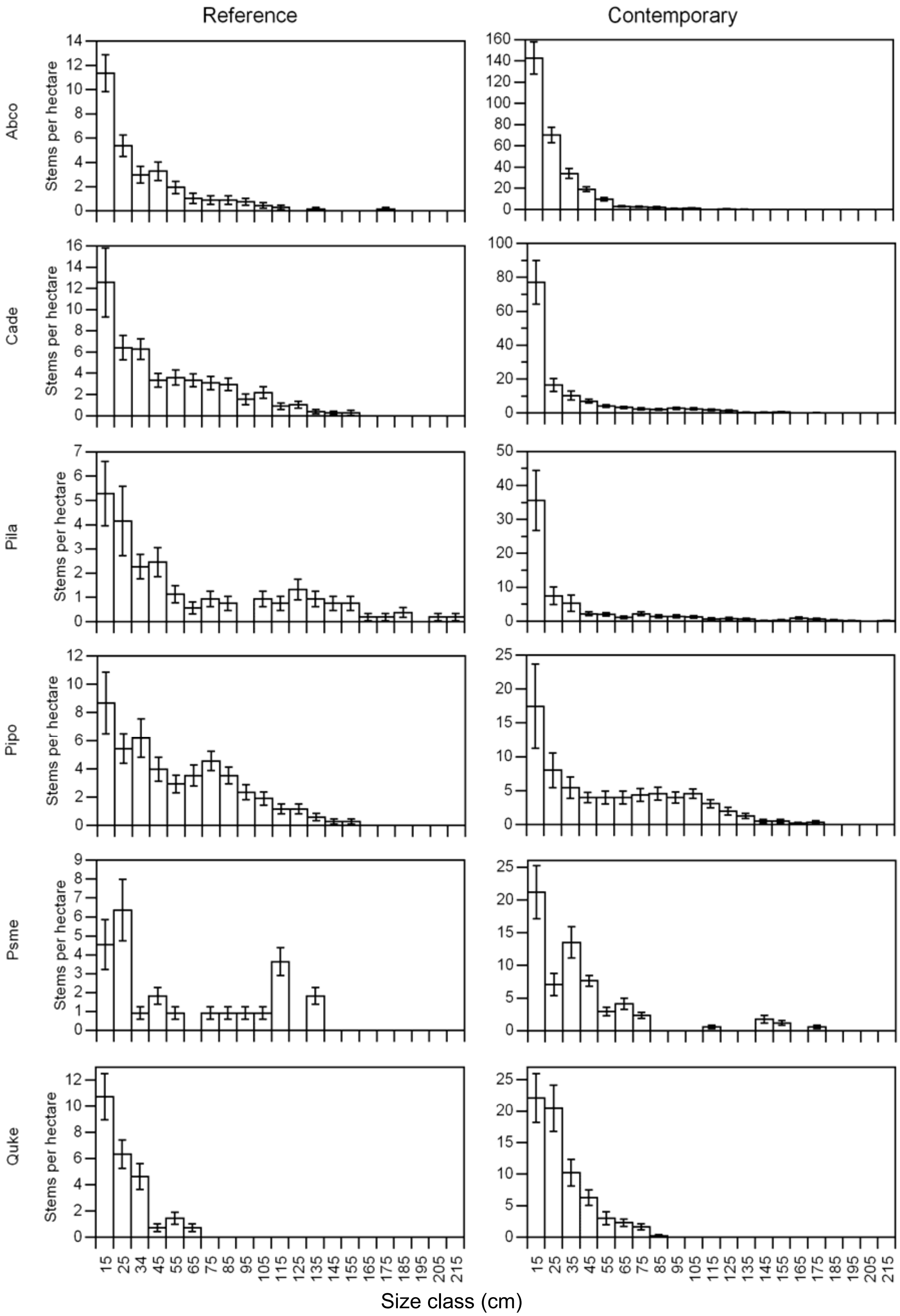


Figure 2.5a: Page 1 of 2.

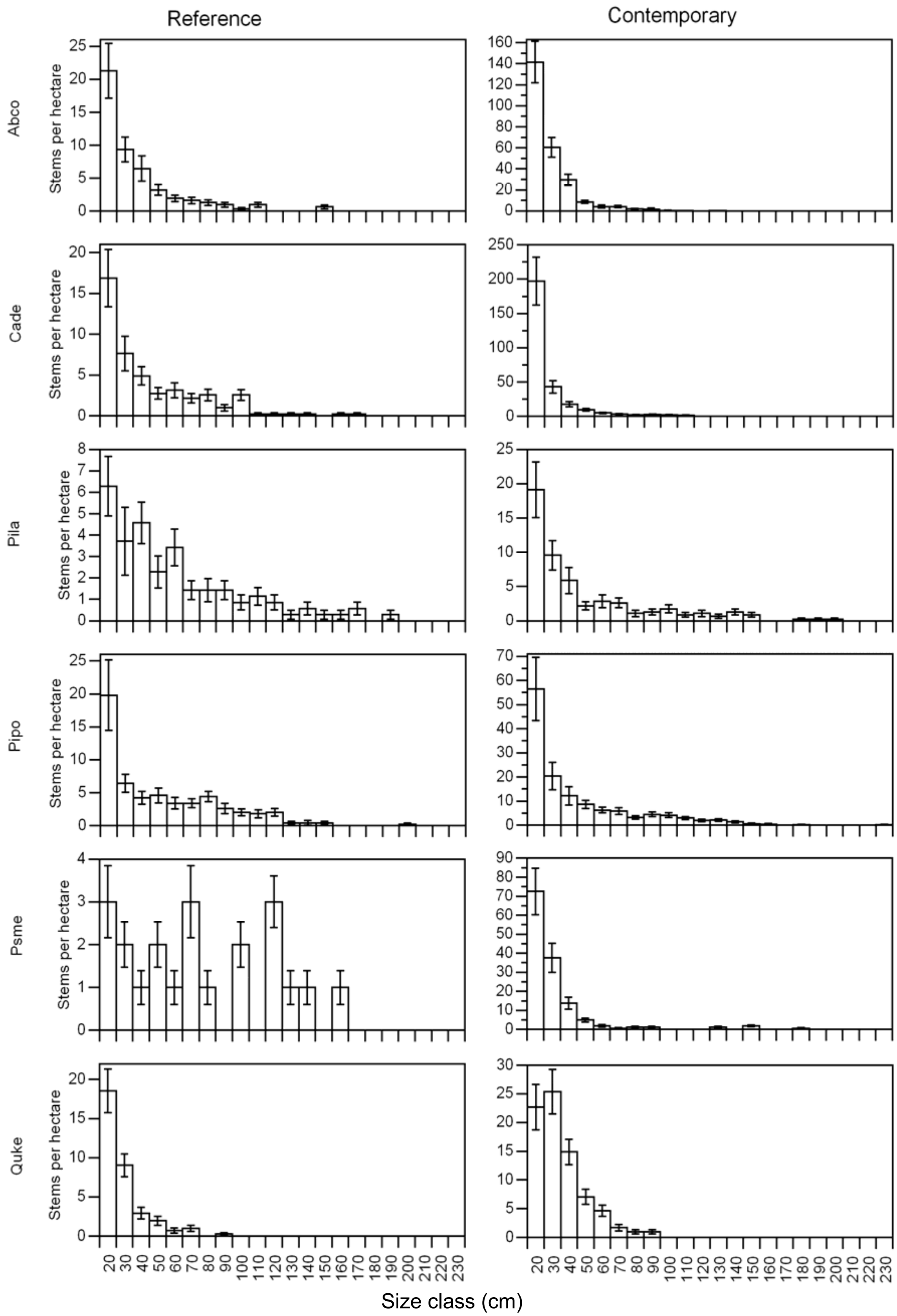


Figure 2.5b: Page 2 of 2.

Figure 2.6: Next page. Mean (± SE) density of trees in 20 year age-classes in the reference (1899) and contemporary a) Big Oak Flat and b) South Fork Merced mixed conifer forests, Yosemite National Park, CA. Note that the y-axis scale is different on each graph. Species acronyms are Abco (Abies concolor), Cade (Calocedrus decurrens), Pila (Pinus lambertiana), Pipo (Pinus ponderosa), Psme (Pseudotsuga menziesii), Quke (Quercus kelloggii). Values are for trees >10 cm dbh on each date.

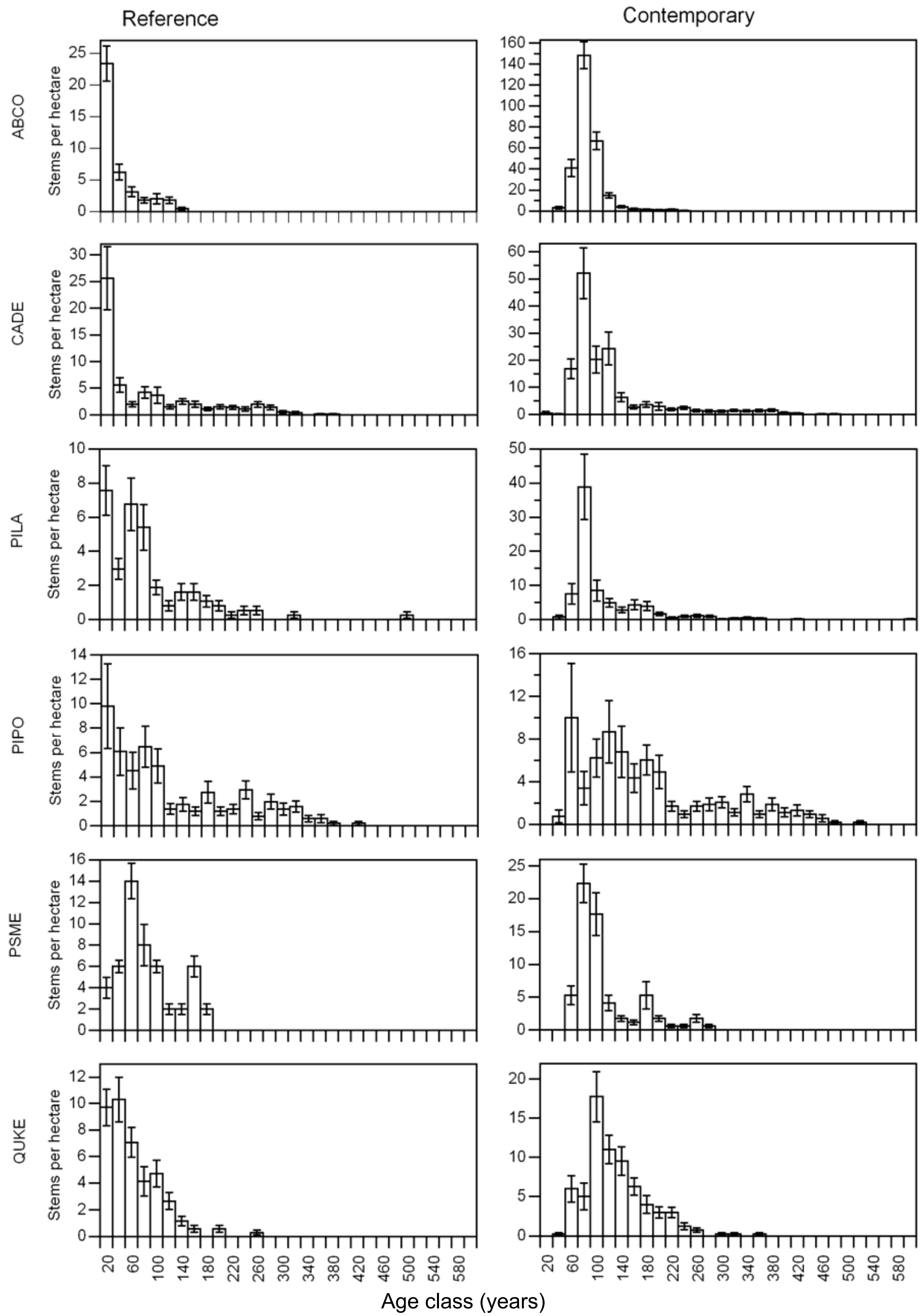


Figure 2.6a: Page 1 of 2.

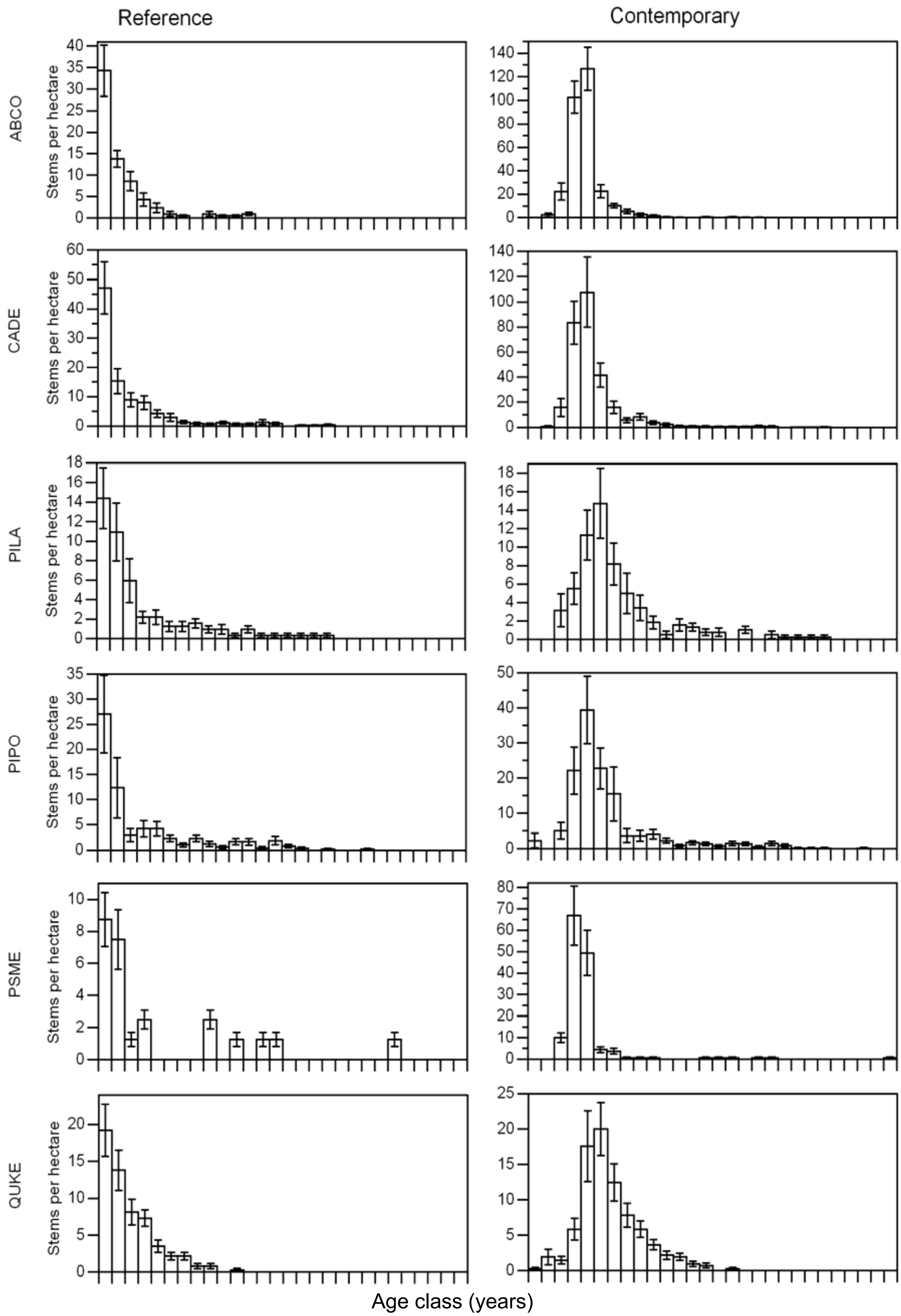


Figure 2.6b: Page 2 of 2.

2.3.4.1 Big Oak Flat study area

White fir-incense cedar – Pre-fire suppression stands in the white fir-incense cedar compositional group were different from contemporary stands (Table 2.7). Overall, contemporary stands are more dense, and have greater basal area, but lower quadratic mean diameter ($p<0.001$). On an individual species basis, contemporary white fir and incense cedar are denser ($p<0.001$) than in pre-fire suppression stands. Contemporary white fir also has a higher average basal area ($p<0.001$). The density, basal area and QMD for the other species were similar in both time periods. The shape of the age-class distributions for all species was different ($p<0.01$) in each time period, while the size-class distribution was different ($p<0.01$) only for incense cedar and white fir.

White fir-incense cedar-sugar pine – Contemporary stands are different from pre-fire suppression stands (Table 2.7). Overall, contemporary stands are denser, have a greater basal area, but a smaller quadratic mean diameter ($p<0.001$). Yet, on an individual species basis, there are only differences for white fir. White fir populations in the group are more dense and have a larger basal area ($p<0.001$). Size-class distributions of ponderosa pine and Douglas-fir were different ($p<0.05$) between the two time periods, while age-class distributions of white fir, incense cedar, Douglas-fir and black oak were different ($p<0.05$).

Table 2.7 a-d: Next page. Comparison of contemporary and reconstructed compositional groups of old-growth, mixed conifer forests in the Big Oak Flat study area, Yosemite National Park, CA (* $p<0.05$, ** $p<0.01$, *** $p<0.001$).

Table 2.7a: Page 1 of 4.

	Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
White fir-incense cedar (n=28)									
Abco									
1899	27.9	33	0-140	5.5	8.6	0-32.5	39.9	28.2	11.3-117.4
2002	307.1***	154.7	70-610	18.7***	7.5	8.4-43.7	32	14.4	15.3-70.2
Cade									
1899	40	30.8	0-130	11.3	11	0-52.9	58.9	23.5	12.9-113.4
2002	147.1***	122.7	0-430	18.4	16.5	0-69	48.3	26.1	12.2-95.5
Pila									
1899	14.3	15.5	0-50	7.1	11.1	0-32.7	71	42.9	12.1-153.7
2002	24.1	23.7	0-90	12.2	13	0-45.9	92	54.5	13.9-185.7
Pipo									
1899	37.9	35.5	0-130	13.3	10.3	0-34.6	71.9	28.5	29.1-128.5
2002	50.4	72.5	0-370	24.2	22.6	0-84.9	92.2*	32.8	25.9-155.8
Psme									
1899	4.6	20.8	0-110	0.8	3	0-14.1	62.2	62.3	23.2-134.1
2002	20.7	49.9	0-200	3.8	11.9	0-60.2	42.8	21	14.5-83.5
Quke									
1899	8.6	16.9	0-70	0.7	1.7	0-7.3	29.4	14.2	10.8-56.7
2002	24.6	72.4	0-360	1.9	4.3	0-16.6	36.7	15.2	16.1-72.9
All Trees									
1899	133.2	70.4	30-400	38.6	20.7	4.9-97.9	61.4	16.1	30-101.7
2002	573.2***	247.4	110-1190	79.3***	26.6	39.6-130.9	44.2***	10.8	23.1-67.7

Table 2.7b: Page 2 of 4.

		Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
		Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
White fir-incense cedar-sugar pine (n=28)										
Abco										
	1899	23.9	23	0-90	3.8	6.3	0-28.6	38.2	23.8	10.1-95.5
	2002	398.2***	153.1	140-750	30.5***	11.1	11.5-53.9	32.5	9	19.2-49.5
Cade										
	1899	40.7	29.6	0-100	13.4	11.1	0-36.6	63.9	24.3	29.3-116
	2002	38.2	41.7	0-150	14.3	12.9	0-51.2	79.1	31.6	20.5-118.7
Pila										
	1899	15	18	0-60	9.1	14.7	0-61.3	87.3	60.1	17.3-208.7
	2002	36.8	83.5	0-400	17.1	20.8	0-70.5	113.7	59.3	17.2-215
Pipo										
	1899	12.5	15.8	0-50	5.8	9.1	0-39.7	72.5	33.5	22.4-142.9
	2002	5.7	9.2	0-30	5.6	9.7	0-36.4	111.4	31.9	50.2-159.5
Psme										
	1899	0.7	2.6	0-10	0.3	1.4	0-7.2	56.2	56.4	16.3-96.1
	2002	6.4	19.5	0-70	0.9	2.8	0-10.7	44.6	12.5	34.1-58.4
Quke										
	1899	5.4	10.7	0-50	0.3	0.7	0-2.8	25.5	10.1	12.4-42.1
	2002	9.3	22.4	0-100	0.9	2.3	0-9.3	32	16.6	15.8-63
All Trees										
	1899	98.2	45.1	30-220	32.6	19.3	5.7-94.6	66.3	23.4	29.5-131.2
	2002	494.6***	181.3	220-960	69.2***	26	20.7-151.1	43.3***	10.5	19.2-65.2

Table 2.7c: Page 3 of 4.

	Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
	Mean	**SD**	**Range**	**Mean**	**SD**	**Range**	**Mean**	**SD**	**Range**
Incense cedar-sugar pine-ponderosa pine (n=18)									
Abco									
1899	16.7	30.5	0-120	1.3	2.5	0-9.1	27.6	8.7	18.5-45.8
2002	75.6***	54.6	0-180	6.3***	6.9	0-30.4	32.2	12.2	17.6-60.4
Cade									
1899	70.6	70.5	20-290	12	12	0.7-45.9	49.2	25.6	17.6-108.1
2002	270.6***	215.7	30-800	26.6**	13.8	4.3-51	41.3	20.3	20.1-92.8
Pila									
1899	21.2	29.3	0-100	8.4	12.4	0-49.5	85	37.4	18.7-125.9
2002	131.7*	132.7	0-380	8.1	9	0-27.2	30.9**	22.7	13.6-88.2
Pipo									
1899	40.6	32.4	10-110	15.3	11.1	0.5-39.1	75	35.3	17.5-157.9
2002	64.4	111.4	0-470	14.3	15.2	0-46.2	69.5	28.9	22-112.7
Psme									
1899	5.6	12.5	0-50	3.6	9.2	0-37.9	88.4	29.7	45.7-118.2
2002	15	25.5	0-80	5.1	14.4	0-58.4	51.5	37.4	15.1-111.3
Quke									
1899	25	43.6	0-140	1.3	2.3	0-9.1	26.1	7.8	13.1-39.3
2002	52.2	77.7	0-320	6.1	11.2	0-43.5	31.4	10.7	16.7-47.7
All Trees									
1899	178.3	99	80-450	41.9	25.6	13.4-103.7	57	22.7	28.3-113.8
2002	609.4***	272.8	130-1220	66.1**	23.8	26.4-117.8	39.3**	11	24.6-68.6

Table 2.7d: Page 4 of 4.

	Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
	Mean	**SD**	**Range**	**Mean**	**SD**	**Range**	**Mean**	**SD**	**Range**
Ponderosa pine-incense cedar-black oak (n=11)									
Abco									
1899	2.7	6.5	0-20	0.3	0.8	0-2.5	35.3	6	31.1-39.6
2002	4.5	15.1	0-50	0.2	0.6	0-2.1	23	-	23-23
Cade									
1899	16.4	13.6	0-40	5.7	5.6	0-14.4	65.5	30.2	25.4-118.1
2002	75.5*	82.7	0-250	12.1	11	0-31.7	56.1	38.9	14.1-116.1
Pila									
1899	4.5	5.2	0-10	3.7	7.3	0-23.2	88.3	56.7	31.6-172
2002	8.2	16	0-40	0.5	1.3	0-4.3	32.5	36.1	10.2-74.1
Pipo									
1899	81.8	54.2	0-160	18.8	9.7	0-32.5	59.9	19.9	21.7-89
2002	93.6	64.7	30-230	30.8**	9.8	10.2-45.4	74.4	25.9	30.5-102.3
Psme									
1899	0.9	3	0-10	0.9	3	0-10.1	113.3	-	113.3-113.3
2002	3.6	12.1	0-40	0.4	1.4	0-4.5	38.1	-	38.1-38.1
Quke									
1899	14.5	19.2	0-50	1.2	1.8	0-5.5	35.1	17.8	18.8-65.6
2002	87.3**	80.6	10-250	6.2	5.8	0.2-15.9	27.7	10	13.4-42.4
All Trees									
1899	120.9	71.8	30-230	30.6	9.3	17.2-47.6	64.6	22.9	32.3-110.1
2002	272.7**	147.9	90-550	50.2**	14.8	28.7-73.4	52	12.5	37.4-80.6

Incense cedar-sugar pine-ponderosa pine – Pre-fire suppression stands were different from contemporary stands (Table 2.7). Overall, contemporary stands are more dense ($p<0.001$), have a greater basal area, and smaller average diameter ($p<0.01$) than pre-fire suppression stands. On an individual species basis, contemporary white fir and incense cedar are denser ($p<0.001$), and have a greater basal area ($p<0.01$) than in pre-fire suppression stands. In addition, sugar pine densities increased in 2002 ($p<0.05$), while QMD decreased ($p<0.01$). Size-class distributions of all species except white fir were different ($p<0.05$) between the two time periods. Age-class distributions were different ($p<0.01$) in 2002 for white fir, incense cedar, sugar pine and black oak.

Ponderosa pine-incense cedar-black oak – Contemporary stands varied from pre-fire suppression stands (Table 2.7). Overall, contemporary stands are more dense and have a greater basal area than pre-fire suppression stands ($p<0.01$). Quadratic mean diameter was similar between the two periods ($p>0.05$). On an individual species basis, contemporary black oak populations are more dense ($p<0.01$) while their basal area and QMD was similar ($p>0.05$). In addition, contemporary incense cedar populations are more dense ($p<0.05$) and ponderosa pine basal area increased ($p<0.01$). Size-class and age-class distributions were different ($p<0.05$) for all species between the two time periods.

2.3.4.2 South Fork Merced study area

White fir-incense cedar – Contemporary stands in the white fir-incense cedar compositional group are different from pre-fire suppression stands (Table

2.8). Overall the contemporary stands are more dense and have a smaller quadratic mean diameter ($p<0.001$), while their basal area is greater ($p<0.05$). On an individual species basis, white fir and incense cedar densities increased in 2002 ($p<0.001$), while only white fir basal area was greater ($p<0.001$), and black oak diameter was smaller ($p<0.01$). Age-class distributions were different for all species ($p<0.05$) between the two periods, while size-class distributions were only different ($p<0.01$) for incense cedar, Douglas-fir and black oak.

Incense cedar-ponderosa pine – Pre-fire suppression stands varied from contemporary stands (Table 2.8). Overall, the contemporary stands are more dense and have a smaller QMD ($p<0.001$), while their basal area is greater ($p<0.05$). On an individual species basis incense cedar, sugar pine and Douglas-fir populations are more dense ($p<0.05$) in 2002 than 1899, while white fir and incense cedar diameters are smaller ($p<0.05$). Age-class distributions were different ($p<0.01$) for all species between the two periods, while size-class distributions were different ($p<0.05$) for all species except sugar pine and ponderosa pine.

White fir-sugar pine – Contemporary stands varied slightly from pre-fire suppression stands (Table 2.8). While overall the contemporary stands are more dense and have a smaller QMD ($p<0.05$), only white fir was significantly different between the periods. White fir populations are over 5-fold denser in 2002

Table 2.8 a-d: Next page. Comparison of contemporary and reconstructed compositional groups of old-growth, mixed conifer forests in the South Fork Merced study area, Yosemite National Park, CA (* $p<0.05$, ** $p<0.01$, *** $p<0.001$).

Table 2-8a: Page 1 of 4.

		Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
		Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
White fir-incense cedar (n=16)										
Abco										
	1899	35.6	44.4	0-140	4.6	5.8	0-21.5	42.1	23.7	13-73.3
	2002	326.9***	129.3	120-670	18.2***	8.3	4.4-31.3	27.2	9.5	16.7-54.6
Cade										
	1899	45.6	52.3	0-200	15.3	15.3	0-56.6	74.5	34.7	40.3-153.7
	2002	314.4***	264.9	30-880	20.7	13.5	3.2-60.6	39**	25.1	14.3-87.1
Pila										
	1899	11.9	15.2	0-50	8.3	11.4	0-30.6	88.7	37.4	37.2-139.6
	2002	28.8	33.2	0-100	10	10.2	0-30.6	79.3	45.5	30-177
Pipo										
	1899	30.6	22.6	0-70	18.3	13	0-43.5	92.3	32.5	14-141.3
	2002	33.1	28	0-100	18.6	12.2	0-43.5	96.2	34.9	38-172.7
Psme										
	1899	1.9	7.5	0-30	1.3	5.2	0-20.9	94.2	-	94.2-94.2
	2002	11.9	36.4	0-140	1.4	4.2	0-16.1	44.2	28	24.4-64
Quke										
	1899	21.9	30.2	0-90	3.8	4.7	0-13.8	51	15.4	32.3-82.3
	2002	36.3	39.8	0-140	4.4	5.3	0-15.9	37.6	12.4	17.7-56.5
All Trees										
	1899	147.5	82.6	50-340	51.6	24.1	5.9-89.4	68.7	19.2	38.9-110.8
	2002	751.3***	298.2	340-1190	73.4*	18.6	45-120	37.2***	8.9	24.6-51.6

Table 2.8b: Page 2 of 4.

		Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
		Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
Incense cedar-ponderosa pine (n=17)										
Abco										
	1899	12.4	29.7	0-110	1.7	3.3	0-9.1	56.9	34.3	22.5-102.2
	2002	44.7	66.5	0-180	1.7	3.1	0-11.5	22.4*	9.2	14.4-38.6
Cade										
	1899	44.7	39.9	0-120	14.6	9.9	0-28.9	69.3	27.9	17.3-135.8
	2002	461.8***	461.3	40-1470	22	12.8	0.4-51.6	30.1***	14.3	11.3-63.2
Pila										
	1899	10.6	15.2	0-40	5.1	8	0-24.9	85.9	46.4	17.6-140.9
	2002	44.7*	50	0-160	7.2	8.7	0-24.9	57	48.2	12.8-140.9
Pipo										
	1899	28.8	21.5	0-80	15.1	15.2	0-45.8	77.3	37.7	20.6-170.7
	2002	93.5	124.4	0-470	16.7	14.2	0-39.9	69.1	50.2	18.1-221
Psme										
	1899	7.6	18.2	0-70	3.2	9.4	0-37.5	53.6	32.2	10.6-82.6
	2002	92.4*	164	0-580	5.8	8.2	0-22.5	32.7	13.9	16.5-59.9
Quke										
	1899	37.1	41.3	0-120	5.4	6.4	0-20.3	42	11.4	21.2-61.8
	2002	90.6	94.2	0-300	8.1	8.9	0-29.5	33.9	10.8	17.4-54.7
All Trees										
	1899	141.2	82.8	10-300	45	24.3	6.3-92.6	68.6	23.6	37.8-129.8
	2002	827.6***	510.1	190-2060	61.6*	23.1	35.1-112.9	33.4***	9	19.1-50.3

Table 2.8c: Page 3 of 4.

		Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
		Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
White fir-sugar pine (n=10)										
Abco										
	1899	55	40.9	20-150	14.4	12.3	3.5-36.2	58.9	25.7	23.5-98.9
	2002	421***	350.7	140-1140	24.7	15.5	7.2-46.6	29.5**	12.2	19.2-58.9
Cade										
	1899	20	43.5	0-140	4.2	7	0-19.7	56.4	24.4	34-89.3
	2002	49	53.4	0-140	4	7	0-20.2	26.5	17.1	13.3-53
Pila										
	1899	54	34.7	10-130	36.2	13.8	12.7-51	102.6	30.6	56.9-139.5
	2002	78	60.7	0-150	29.8	17.4	0-52.7	86.5	42.1	36.9-140.1
Pipo										
	1899	-	-	-	-	-	-	-	-	-
	2002	-	-	-	-	-	-	-	-	-
Psme										
	1899	5	5.3	0-10	7.7	9.1	0-24.6	136.9	30.4	93.2-177
	2002	39	71.4	0-210	8.4	12	0-34.2	74.9	46.4	32.6-141
Quke										
	1899	8	15.5	0-50	0.6	1.2	0-3.6	28.2	14.4	11.5-46.2
	2002	13	18.3	0-50	0.9	1.3	0-4.1	27.9	6	22.9-36.2
All Trees										
	1899	142	89.8	50-330	63.1	23.3	27.9-93.6	82.3	23.1	36.3-102
	2002	600***	311.2	190-1180	68	22	29.1-95.7	41.3**	15.7	22-80

Table 2.8d: Page 4 of 4.

		Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
		Mean	**SD**	**Range**	**Mean**	**SD**	**Range**	**Mean**	**SD**	**Range**
Ponderosa pine-incense cedar (n=21)										
Abco										
	1899	2.4	7.7	0-30	0.8	3	0-13.5	63.6	41.2	34.5-92.8
	2002	8.1	32.8	0-150	1.2	3.9	0-13.5	62.5	42.7	32.3-92.8
Cade										
	1899	28.1	34	0-140	7	8.6	0-29.6	59.1	23.5	23-98.4
	2002	122.4*	151	0-620	10.3	9.1	0-29.6	42.2*	25.4	12.2-121.3
Pila										
	1899	4.3	11.2	0-40	1.6	4.4	16.4	68.4	5.3	62.3-72.3
	2002	18.1	42.4	0-170	1.6	3.8	0-14.4	40.7	23.6	11.1-67.5
Pipo										
	1899	75.7	67.1	20-340	32.1	14.9	9.6-62.5	80.5	23.2	41.6-126.1
	2002	218.1***	195.2	40-750	37.4	12.6	16.7-63.8	58**	23.6	23.7-124.6
Psme										
	1899	-	-	-	-	-	-	-	-	-
	2002	2.4	8.9	0-40	0.2	0.6	0-2.4	36.5	12.3	27.8-45.2
Quke										
	1899	12.9	18.7	0-70	2	2.8	0-6.8	47.5	23.7	15-81
	2002	45.7*	52.2	0-190	3	3.2	0-9.1	31.5	13.4	14.8-64.3
All Trees										
	1899	123.3	101.4	40-520	43.6	18.3	22-99	74.2	22.8	38-126.1
	2002	418.4***	351.2	90-1420	53.7	18	31.7-93.1	48.6***	17.3	26.6-84

(p<0.001) than 1899, while average diameter is less than half that in 1899 (p<0.01). Size- and age-class distributions were different (p<0.05) for all species between the two time periods.

Ponderosa pine-incense cedar – Pre-fire suppression stands in the ponderosa pine-incense cedar compositional group varied slightly from contemporary stands (Table 2.8). Overall, contemporary forests are more dense and have a smaller QMD (p<0.05). Ponderosa pine, incense cedar and black oak all have higher densities in 2002 (p<0.05). In addition, the average diameter of both incense cedar and ponderosa pine decreased in 2002 (p<0.05). Age-class distributions were different (p<0.01) for all species between the two periods, while size-class distributions were only different (p<0.01) for white fir and sugar pine.

2.3.5 Comparison with 1911 Forest

The characteristics of the 1911 forest were different than the contemporary forest in the Big Oak Flat study area (Table 2.9). Overall, the 1911 forest was less dense and had a lower basal area than the contemporary forest (p<0.001). The average diameter of trees in 1911 was similar to the average diameter in 2002 (p>0.05). On an individual species basis, white fir and incense cedar populations were less dense and they had a smaller basal area (p<0.001) in 1911 than in the contemporary populations. The quadratic mean diameter of white fir was smaller in 2002 than in 1911 (p<0.01). In contrast, the average diameter of ponderosa pine was larger in 2002 than 1911 (p<0.001).

Table 2.9: Comparison of contemporary and reconstructed forest with 1911 forest conditions based upon a National Forest Service inventory survey of old-growth, mixed conifer forest in Big Oak Flat study area, Yosemite National Park, CA. Oaks are not included (* $p<0.05$, ** $p<0.01$, *** $p<0.001$).

		Density (# trees/ha)			Basal area (m^2/ha)			Quadratic Mean Diameter (cm)		
		Mean	**SD**	**Range**	**Mean**	**SD**	**Range**	**Mean**	**SD**	**Range**
Abco										
	2002	181.2***	140.8	0-500	17.7***	14.1	0-57.3	36.9***	11.2	22.8-70.5
	1911	13.1	19.6	0-63.8	3.4	6.1	0-27.7	72.7	24.5	22.3-122.1
	1899	15.8	22.8	0-110	3.4	6.6	0-34.1	44.8***	24.3	22.9-120.3
Cade										
	2002	74.5***	90.7	0-480	17.8***	15	0-69.4	67.1	27.8	22.8-118.9
	1911	31.8	39.8	1-159	4.6	3.9	0-13.8	54.9	22.2	22.5-104.3
	1899	35.4	28.3	0-140	11.3**	10.8	0-52.4	62.5	24.3	22.9-120.7
Pila										
	2002	31.8	52.6	0-210	11.4	15.6	0-67.4	87.4	57.1	22.8-220.9
	1911	8.5	9	0-39	2.8	3.2	0-10.2	68.5	32.1	13.9-130.1
	1899	12.9	17.6	0-100	7.6	12.4	0-61.8	85.5	48.1	22.9-205.7
Pipo										
	2002	38.1	56.6	0-330	16.8	18.5	0-86.1	87.9***	31.8	22.8-160
	1911	42.9	42.6	0-145	8.9	7.9	0-25.8	62	23.1	21.3-105.9
	1899	33.4	35.8	0-160	11.9	10.9	0-39.6	72.7	29.7	22.9-160
Psme										
	2002	9.8	25.8	0-150	2.8	9.9	0-60.7	54.7*	35.8	22.8-160.5
	1911	1.6	6.1	0-31	0.9	3.7	0-19.2	77.2	7.9	71.9-89
	1899	2.6	9.8	0-70	1.2	4.6	0-35.2	78.6	40.1	22.9-129.5
All Trees										
	2002	335.3***	146.4	30-750	66.6***	27.8	18.1-142	53.1	14	25.4-100.8
	1911	97.9	68	8-216	20.6	10.1	0.7-34.4	56.7	17.2	32.6-98.4
	1899 (no oaks)	99.1	53.2	10-330	35.3***	20.8	4.5-101	69.7***	22.2	34.6-132.3

The forest reconstruction for 1899 and the 1911 survey were similar for some characteristics but not for others (Table 2.9). While the reconstructed forest was similar in density to the 1911 forest ($p>0.05$), it had a greater basal area (41% increase) and quadratic mean diameter (19% increase) than the 1911 forest ($p<0.001$). On an individual species basis, only incense cedar and white fir populations had different characteristics in the reconstructed forest than the 1911 forest. Incense cedar had a greater basal area in the reconstructed forest ($p<0.01$), while white fir had a smaller QMD in the reconstructed forest ($p<0.01$) than the 1911 forest.

2.3.6 Structural Diversity

For both study areas, the pre-fire suppression forest was structurally similar (Shannon's diversity index, mean diversity BOF = 2.0, SFM = 1.8) to the contemporary forest (mean diversity BOF = 1.9, SFM = 1.9) ($p>0.05$) (Table 2.10a, b). The diversity was similar among the compositional groups between the two periods except for the white fir-incense cedar-sugar pine group in BOF ($p<0.05$) and the ponderosa pine-incense cedar group in SFM ($p<0.05$).

2.3.7 Spatial Patterns

The spatial pattern of tree ages varied widely among the plots (Table 2.11a, b) in both study areas. The spatial distribution of tree ages in 2002 varied highly across all distance classes, with evidence present in many of the plots of positive (BOF = 41%, SFM = 44%) and negative (BOF = 55%, SFM =

Table 2.10: Shannon's diversity index for plots in the entire study area and each compositional group for 1899 and 2002 in the a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA. Shannon's diversity index is included for 1911 forest conditions based upon a National Forest Service inventory survey of old-growth, mixed conifer forest in the Big Oak Flat study area (* p<0.05).

	Mean	Range
All Plots (n=85)		
1899	2	0.9-2.8
1911	2.3	0-3.3
2002	1.9	0.8-2.6
WF-IC (n=28)		
1899	2.1	0.6-2.8
2002	2	0.8-2.6
WF-IC-SP (n=28)		
1899	1.9	1.3-2.4
2002	1.7*	0.9-2.6
IC-SP-PP (n=18)		
1899	2.1	1.6-2.5
2002	2	1.4-2.5
PP-IC-BO (n=11)		
1899	1.9	1.1-2.5
2002	1.9	0.8-2.5

a)

	Mean	Range
All Plots (n=64)		
1899	1.8	0-2.8
2002	1.9	1-2.7
WF-IC (n=16)		
1899	1.9	0.5-2.4
2002	1.9	1.3-2.6
IC-PP (n=17)		
1899	1.8	0-2.5
2002	1.9	1-2.7
WF-SP (n=10)		
1899	2	1.3-2.3
2002	1.9	1.1-2.4
PP-IC (n=21)		
1899	1.7	0.7-2.8
2002	1.9*	1.1-2.5

b)

13%) spatial autocorrelation. However, there was a pattern to the spatial autocorrelation present among the plots in both study areas. The highest frequencies of positive spatial autocorrelation in BOF and SFM occurred at distances of 1-6 m and 2-6 m respectively, while the highest frequencies of negative spatial autocorrelation occurred at distances of 6-10 m in both study areas.

In BOF, a similar pattern was present for tree ages in 1899, with the highest frequencies of positive spatial autocorrelation occurring at distances of 1-9 m and 22–30 m. At the same time the highest frequencies of negative spatial autocorrelation occurred at distances of 12-19 m. In contrast, SFM plots

Table 2.11: Frequency of plots in a) Big Oak Flat and b) South Fork Merced study areas with values of Moran's *I* that indicate spatial autocorrelation (p<0.05) by 1 m distance classes for the reference (AD 1899) and contemporary forest (AD 2002). Positive spatial autocorrelation represents distances between similar aged trees, while negative values represent distances between trees of dissimilar age.

BOF

distance		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
2002	"+"	4	4	1	4	3	4	2	1		2																				
	"-"		4	1		2	4	6	2	2	6																				
1899	"+"	3	6	2	8	3	9	6	3	5	3		1			2	1	2	1	1	3	2	4	5	4	6	5	4	4	5	5
	"-"				3	3	3	3	3	3	2	1	4	6	3	5	5	8	7	7	3	5	3	1		3	1	2	1		2

a)

SFM

distance		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
2002	"+"	2	4	6	4	3	4	3	2	3	4																				
	"-"		3	2	2	1	5	3	3	4	5																				
1899	"+"	2		2		3	4	2	1		3	1	2	2	1	3	2		1	1	4	1	1		1	1	3	2			2
	"-"					1	1	2		2		4		1			2	1	5	1	1	1	3	6	1	2	1			1	2

b)

demonstrated no distinct pattern of spatial autocorrelation in 1899, with similar frequencies at all distances.

Clustering of stems was present in both the reference and contemporary forest in BOF (Table 2.12). Similar frequencies of clustering were present in the contemporary forest in 23% of the plots at all distances, while dispersion was only present at 1 m and 6 m. In contrast, 65% of the plots in 1899 showed evidence of clustering of stems. While evidence of clustering was present at all distances, the highest frequencies occurred at 1-2 m and 6-11 m. The majority of dispersion present in 1899 occurred at a distance of 1 m. I was unable to perform the Ripley's K(*t*) analysis of stems in the South Fork Merced plots due to the limited number of stems present in the inscribed squares.

Table 2.12: Frequency (%) of plots in the Big Oak Flat study area with values of Ripley's K(*t*) that indicate spatial dependence ($p<0.05$) of stems by 1 m distance classes for the reference (1899) and contemporary (2002) forest. Positive (+) values indicate clustering of stems, while negative (-) values indicate dispersion of stems.

Distance		1	2	3	4	5	6	7	8	9	10	11	12
2002	"+"	12	9	12	12	6	12						
	"-"	21					3						
1899	"+"	31	27	19	19	27	31	31	31	46	31	31	27
	"-"	50	8	4	4								

2.3.8 Fire Regimes

2.3.8.1 Fire record

In Big Oak Flat a total of 286 fires were identified in the 209 samples collected in the study area. The fire record spanned the period 1575-2002 and the number of fires varied by slope aspect and compositional group (Table 2.13a). The number of fires experienced by each compositional group ranged from 212 (WF-IC-SP) to 90 (PP-IC-BO), while across aspects the number of fires ranged from 221 (west) to 141 (south). In the South Fork Merced a total of 248 fires were identified in the 156 samples collected in the study area. The fire record spanned the period 1575-2002 and the number of fires varied by slope aspect and compositional group (Table 2.13b). The number of fires occurring on each aspect ranged from 8 (east) to 194 (west), while fire occurrence in the compositional groups ranged from 107 (WF/SP) to 177 (WF/IC).

2.3.8.2 Fire Season

The position of fires within annual growth rings indicate that fires burned mainly late in the growing season (latewood: BOF = 38.7%, SFM = 54.2%) or after the growing season ended (dormant: BOF = 39.8%, SFM = 37.6%), but fires were recorded in all seasons (Table 2.14a, b). The seasonal pattern of burning in the different compositional, and slope aspect groups was similar to the pattern for the study area as a whole, except for late and dormant season fires. In BOF, the ponderosa pine/sugar pine/black oak compositional group experienced more fires in the latewood (60.9%) compared to the dormant season (25.5%). In SFM, all compositional groups and aspects experienced similar seasonal patterns of fire except for east aspects which experienced more fires in the dormant season (60%) compared to latewood (35%). This may be an artifact due to there only being one sample collected on an east aspect in the South Fork study area.

Table 2.13: Number of fire scars, number of recorded fires, and length of fire scar record for each compositional and slope aspect group for old-growth, mixed conifer forests in a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA. * 3 samples not associated with a compositional group are included in the breakdown by slope aspect.

Compositional Group	# Gridpoints	# Fire scars	# Fires	Time Period	Area (ha)
WF - IC	28	1031	192	1379-2002	700
WF - IC - SP	28	912	212	1442-2002	700
IC - SP - PP	18	660	197	1442-2002	450
PP - IC - BO	11	275	90	1612-2002	275
Slope Aspect					
North	27	1205	201	1403-2002	675
South	13	389	141	1525-2002	325
East	11	495	159	1379-2002	275
West	34	789	221	1513-2002	850
All	85	2878	287	1379-2002	2125

a)

Compositional Group	# Gridpoints	# Fire scars	# Fires	Time Period	Area (ha)
WF - IC	16	698	177	1499-2002	400
IC - PP	17	723	132	1570-2002	425
WF - SP	10	376	107	1522-2002	250
PP - IC	21	507	152	1561-2002	525
Slope Aspect					
North	27	1113	194	1499-2002	675
South	14	490	147	1564-2002	350
East	1	64	8	1764-2002	25
West	23	686	187	1522-2002	575
All*	65	2353	249	1499-2002	1625

b)

Table 2.14: Page 1 of 2. Seasonal distribution of fires for each compositional and slope aspect group for old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas in Yosemite National Park, CA.

	Seasonality of Burn		Fire Scar position				
Total	**Determined**	**Undetermined**	**Early**	**Middle**	**Late**	**Latewood**	**Dormant**
Number	1700	1178	19	108	238	658	677
Percentage	59.1	40.9	1.1	6.4	14	38.7	39.8
Seasonality by Aspect							
North							
Number	693	512	1	44	106	274	268
Percentage	57.5	42.5	0.1	6.3	15.3	39.5	38.7
East							
Number	287	208	5	24	41	96	121
Percentage	58	42	1.7	8.4	14.3	33.4	42.2
South							
Number	251	138	6	16	33	111	85
Percentage	64.5	35.5	2.4	6.4	13.1	44.2	33.9
West							
Number	469	320	7	24	58	177	203
Percentage	59.4	40.6	1.5	5.1	12.4	37.7	43.3
Seasonality by Compositional Group							
WF-IC							
Number	632	399	6	53	109	214	250
Percentage	61.3	38.7	0.9	8.4	17.2	33.9	39.6
WF-IC-SP							
Number	551	361	7	37	70	187	250
Percentage	60.4	39.6	1.3	6.7	12.7	33.9	45.4
IC-SP-PP							
Number	356	304	5	12	44	159	136
Percentage	53.9	46.1	1.4	3.4	12.4	44.7	38.2
PP-SP-BO							
Number	161	114	1	6	15	98	41
Percentage	58.5	41.5	0.6	3.7	9.3	60.9	25.5

a)

Table 2.14: Page 2 of 2.

Total	Seasonality of Burn		Fire Scar position				
	Determined	Undetermined	Early	Middle	Late	Latewood	Dormant
Number	1387	966	6	21	87	752	521
Percentage	58.9	41.1	0.4	1.5	6.3	54.2	37.6
Seasonality by Aspect							
North							
Number	659	454	4	8	37	344	266
Percentage	59.2	40.8	0.6	1.2	5.6	52.2	40.4
East							
Number	20	44	0	0	1	7	12
Percentage	31.3	68.8	0	0	5	35	60
South							
Number	265	225	0	1	10	160	94
Percentage	54.1	45.9	0	0.4	3.8	60.4	35.5
West							
Number	443	243	2	12	39	241	149
Percentage	64.6	35.4	0.5	2.7	8.8	54.4	33.6
Seasonality by Compositional Group							
WF-IC							
Number	401	297	4	4	22	228	143
Percentage	57.4	42.6	1	1	5.5	56.9	35.7
IC-PP							
Number	452	271	1	6	38	223	184
Percentage	62.5	37.5	0.2	1.3	8.4	49.3	40.7
WF-SP							
Number	210	166	0	3	10	104	93
Percentage	55.9	44.1	0	1.4	4.8	49.5	44.3
PP-IC							
Number	291	216	1	7	17	170	96
Percentage	57.4	42.6	0.3	2.4	5.8	58.4	33

b)

2.3.8.3 Fire return intervals

The statistical analysis for the 85 gridpoints and group composites includes the mean fire interval (MFI, average number of years between fires), median fire interval, and the Weibull median probability interval (WMPI) as measures of central tendency. The WMPI is a measure of central tendency for asymmetrical distributions like those for fire intervals (Grissino-Mayer 2001). Fire interval distributions for BOF and SFM study areas, each aspect and compositional group, had similar measures of central tendency, were positively skewed, and were defined by more short fire return intervals than long intervals (Table 2.15, 2.16). The mean composite FRI for all fires in BOF was 1.5 years (range 1-16 yrs) and for SFM was 1.7 years (range 1-12 yrs), while the mean point FRI for BOF was longer at 12.4 years (range 2-84 yrs) and for SFM was 9.9 years (range 1-60 yrs). In BOF, the average FRI for more widespread fire events (i.e. 10% or more grid points, 25% or more grid points) was longer at 2.6 years (range 1-11 yrs) and 10 years (range 2-28 yrs) respectively. In SFM, widespread fires (10% and 25% scarred) were also longer at 2.5 years (range 1-12 yrs) and 6.9 years (range 1-48 yrs), respectively.

Spatial Pattern - The composite FRI and point FRI for both study areas did not vary by compositional or slope aspect group and they were similar to the FRI for the whole study area ($p>0.05$) (Table 2.15, 2.16). Mean composite FRI for slope aspect groups in BOF ranged from 1.9 to 2.6 years, and in SFM from 2 to 2.8 years (the 18.6 years found on east slopes is most likely an artifact due to a sample size of one) and the range for mean Point FRI for BOF was 10.6 to 12.4 years and in SFM from 10 to 13.4 years. The average composite and point FRI for compositional groups in BOF ranged from 1.9 to 3.5 years, and 10.2 to 14.1 years respectively and for SFM from 2.3 to 3.1 years and 6.2 to 12 years respectively. The similarity in FRI was also consistent for more widespread fires that scarred 10% or more and 25% or more of the samples ($p>0.05$). Fires that scarred 25% or more of the samples had the greatest range in variation in FRI.

Table 2.15: Page 1 of 2. Point and composite fire return intervals (FRI) for a) slope aspect and b) compositional groups for old-growth, mixed conifer forest in the Big Oak Flat study area in Yosemite National Park, CA.

Type of Sample	No. of intervals	Mean, MFI	Median	WMPI	SD	Min.	Max.	Skewness	Kurtosis
Point (PFI)	500	12.4	10	10.7	10.1	2	84	3.1	14.5
Composite	286	1.5	1	1.2	1.6	1	16	5.5	34.5
>10% scarred	129	2.6	2	2.3	1.8	1	11	2	5.7
> 25% scarred	33	10	9	9.2	6.1	2	28	1	0.7
North									
Point (PFI)	500	11.8	9	9.7	12	1	179	6.9	79.5
Composite	200	1.9	1	1.5	3.4	1	43	9.6	109.3
>10% scarred	119	2.5	2	2.3	1.8	1	12	2.2	7
> 25% scarred	36	8.4	6.5	7.1	6.6	1	34	1.7	4.2
East									
Point (PFI)	437	12.3	10	10.8	9	2	69	2	6.2
Composite	158	2.5	2	1.7	5.8	1	69	10	110
>10% scarred	158	2.5	2	1.7	5.8	1	69	10	110
> 25% scarred	50	6.6	4	5	6.9	1	34	2.5	6.9
South									
Point (PFI)	346	10.6	9	9.3	7.8	1	53	2.5	8.3
Composite	140	2.6	2	2	3.9	1	31	5.9	39.3
>10% scarred	113	3.3	2	2.2	7.9	1	84	9.7	96.8
> 25% scarred	72	4	4	3.6	2.4	1	10	0.7	-0.3
West									
Point (PFI)	500	12.4	10	10.7	10.1	2	84	3.1	14.5
Composite	220	1.9	1	1.4	3.4	1	41	8.4	83.9
>10% scarred	118	2.8	2	2.4	2.5	1	19	3.2	14.5
> 25% scarred	36	9.2	7	6.9	10.2	1	52	2.8	8.1

a)

Table 2.15: Page 2 of 2.

Type of Sample	No. of intervals	Mean (MFI)	Median	WMPI	SD	Min.	Max.	Skewness	Kurtosis
WF-IC									
Point (PFI)	500	12.4	10	10.8	9.6	1	77	2.3	7.4
Composite	191	2	1	1.6	2.9	1	34	7.7	77.7
>10% scarred	115	2.6	2	2.3	1.8	1	10	1.7	3.2
> 25% scarred	43	7	5	6	4.8	1	19	0.7	-0.5
WF-IC-SP									
Point (PFI)	500	14.1	11	12	12.3	2	179	5.8	64.8
Composite	211	1.9	1	1.4	3.7	1	52	11.7	152
>10% scarred	119	2.8	2	2.4	2.2	1	12	2.2	5.4
> 25% scarred	32	10.3	9.5	9.3	6.6	1	33	1.3	2.3
IC-SP-PP									
Point (PFI)	500	11.6	9	10.1	8.9	1	84	3	16
Composite	196	1.9	1	1.3	4.8	1	65	11.9	150.3
>10% scarred	138	2.1	2	1.8	1.6	1	12	2.8	11.1
> 25% scarred	43	6.7	4	4.9	8.3	1	48	3.4	12.8
PP-IC-BO									
Point (PFI)	242	10.2	9	8.8	8.7	1	84	4	24.5
Composite	89	3.5	2	2.3	6.5	1	53	6	40.3
>10% scarred	75	4.1	2	2.7	6.9	1	53	5.5	33.3
> 25% scarred	46	4.9	4	4.4	3.1	1	11	0.7	-0.8

b)

Table 2.16: Page 1 of 2. Point and composite fire return intervals (FRI) for a) slope aspect and b) compositional groups for old-growth, mixed conifer forest in the South Fork Merced study area in Yosemite National Park, CA.

Type of Sample	No. of intervals	Mean (MFI)	Median	WMPI	SD	Min.	Max.	Skewness	Kurtosis
Point (PFI)	500	9.9	8	8.7	7.4	1	60	2.3	7.8
Composite	249	1.7	1	1.4	1.96	1	12	3.4	11.68
>10% scarred	137	2.5	2	2	2.34	1	12	2.23	4.91
> 25% scarred	48	6.9	4	5.2	8	1	48	3.32	12.94
North									
Point (PFI)	500	10.9	8	9.4	8.5	1	60	2.1	5.8
Composite	193	2	1	1.5	2.9	1	29	5.9	43.2
>10% scarred	131	2.6	1	1.9	3.4	1	29	4.6	27.8
> 25% scarred	53	6.4	4	5.1	5.7	1	29	1.9	3.8
East*									
Point (PFI)	51	13.4	11	11.4	10.4	2	60	2	5.8
Composite	7	18.6	14	15.3	14.2	2	38	0.2	-1.8
>10% scarred	-	-	-	-	-	-	-	-	-
> 25% scarred	-	-	-	-	-	-	-	-	-
South									
Point (PFI)	426	11	7	8.4	14.2	2	119	4.7	25.9
Composite	146	2.8	1	1.9	4.6	1	39	5	30.6
>10% scarred	103	3.9	2	2.2	10.3	1	98	7.8	66.1
> 25% scarred	52	5.8	4	4.1	7.5	1	39	3.3	11.2
West									
Point (PFI)	500	10	7	8	11.1	1	103	4.6	27.4
Composite	186	2.2	1	1.6	3.1	1	19	4	16.6
>10% scarred	133	2.7	2	2	4.1	1	37	5.6	38.8
> 25% scarred	57	5.3	4	4.4	4.6	1	25	2.4	6.3

a)

Table 2.16: Page 2 of 2.

Type of Sample	No. of intervals	Mean (MFI)	Median	WMPI	SD	Min.	Max.	Skewness	Kurtosis
WF-IC									
Point (PFI)	500	10.1	7	8.5	9.3	2	103	4.2	28.9
Composite	176	2.4	1	1.5	5.3	1	55	7.4	61.6
>10% scarred	122	2.8	2	2	4.1	1	35	5.5	36.4
> 25% scarred	56	5.9	4	4.1	7.8	1	44	3.3	11.6
IC-PP									
Point (PFI)	500	9.5	7	7.7	10.4	1	119	5.3	38.2
Composite	176	2.3	1	1.5	5.5	1	66	9.5	102.7
>10% scarred	132	2.8	1	1.8	6.5	1	66	8	69.4
> 25% scarred	70	4.6	3	3.3	7.9	1	66	6.8	50.2
WF-SP									
Point (PFI)	332	12	9	10.7	8.3	2	53	1.8	4.1
Composite	106	3.1	2	2.2	4.4	1	33	4.2	21.6
>10% scarred	106	3.1	2	2.2	4.4	1	33	4.2	21.6
> 25% scarred	57	4.9	4	3.8	4.7	1	25	2.3	6.4
PP-IC									
Point (PFI)	440	9.2	7	7.4	9.8	1	111	5.2	38.5
Composite	151	2.7	1	2	3.4	1	19	3.1	10.1
>10% scarred	106	2.9	2	2.4	2.7	1	18	2.6	9.3
> 25% scarred	58	5	4	4.5	3.3	1	18	1.5	2.6

b)

Fire return intervals for widespread fires (25%) were shortest on south (BOF = 4 years) and west (SFM = 5.3 years) aspects, and longest on west (BOF = 9.2 years) and north (SFM = 6.4 years) aspects. In BOF, widespread fire return intervals were also shortest for the ponderosa pine/incense cedar/California black oak compositional group (4.9 years), and longest in the white fir/incense cedar/sugar pine group (10.3 years), while in SFM the shortest intervals occurred in the incense cedar/ponderosa pine compositional group (4.6 years) and the longest in the white fir/incense cedar group (5.9 years).

Temporal patterns – Fire frequency varied by time period (Table 2.17a, b). The mean composite FRI for both study areas was the same during the pre-settlement and settlement periods (BOF = 1.2 yrs, SFM 1.4 yrs), but longer (BOF = 6.2 yrs, SFM = 5.3 yrs) during the fire suppression period ($p < 0.01$). Temporal variation in FRI for more widespread fires (10% or more, 25% or more samples scarred) exhibited the same pattern. In BOF, they were similar during the pre-settlement (10% = 2.4 yrs, 25% = 10.7 yrs) and settlement periods (10% = 2.8 yrs, 25% = 7.3 yrs). The same pattern was exhibited in SFM for the pre-settlement (10% = 2.3 yrs, 25% = 6.4 yrs) and settlement periods (10% = 3.7 yrs, 25% = 11.7 yrs). No widespread fires (10%, 25% scarred) occurred during the fire suppression period. FRI only varied by century when all fires were included in the analysis ($p<0.01$). The 18th century experienced the shortest FRIs (1.0 yrs), while the 20th century intervals were the longest (BOF = 5.4 yrs, SFM = 5.1 yrs).

Fire Extent – Fire extent varied among fire years and by time period (Figure 2.7a, b). The average fire size for the entire period (1575-2002) for BOF was 205 ha (range 25-1946 ha), and for SFM was 199 ha (range 26-1075 ha) (Table 2.18a, b). Average fire size varied for all time periods with the largest fires occurring during the settlement period (BOF = 266 ha, SFM = 214 ha) and the smallest fires during the suppression period (BOF = 39 ha, SFM = 46 ha). Average annual area burned also varied by century with the largest fires occurring during the 19th century (BOF = 300 ha, SFM = 290ha). This suggests that average fire size increased before settlement in 1850 (Table 2.18a, b, Figure 2.7a, b). The fire return interval for different sized fires also varied over the whole time period (Table 2.19a, b). The shortest average FRI in BOF occurred for fires burning 1 grid point (mean 6.4 yrs, range 1-37 yrs), and the longest was for fires burning 5 grid points (mean 52.6 yrs, range 16-134 yrs). In SFM, the shortest FRI was for fires burning 1 grid point (mean 7.6 yrs, range 1-84 yrs) and the longest for fires burning 6 grid points (mean 30.4 yrs, range 3-109 yrs). The majority of fires in both study areas (78%) were smaller than 250 ha, while only 4 fires in BOF (1%) and 2 fires in SFM (1%) were larger than 1000 ha. In addition, the percentage of the total area burned by fires of different size was similar in both study areas ($p>0.05$). Fires smaller than 350 ha burned over 50% of the total area burned by fires, while fires over 1000 ha only burned 11% (BOF) and 5% (SFM) of the total area burned.

Table 2.17: Fire return intervals for different land use periods and different centuries for old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas in Yosemite National Park, CA (* p<0.05, ** p<0.01).

All Fires	Time Period	Mean	Median	Range
All Years	1575-2000	1.4	1	1-16
Pre-Settlement	1575-1849	1.2	1	1-10
Settlement	1850-1904	1.2	1	1-3
Fire	1905-2002	6.2**	5	2-16
	1600-1699	1.3	1	1-3
	1700-1799	1	1	1-2
	1800-1899	1.1	1	1-3
	1900-2000	5.4**	3.5	1-16

10% scarred	Time Period	Mean	Median	Range
All Years	1575-2000	2.5	2	1-10
Pre-Settlement	1575-1849	2.4	2	1-10
Settlement	1850-1904	2.8	2	1-9
Fire	1905-2002	-	-	-
	1600-1699	3	3	1-6
	1700-1799	2.1	2	1-4
	1800-1899	2.4	2	1-9
	1900-2000	-	-	-

25% scarred	Time Period	Mean	Median	Range
All Years	1575-2000	10	9	2-28
Pre-Settlement	1575-1849	10.7	9	3-28
Settlement	1850-1904	7.3	7.5	2-16
Fire	1905-2002	-	-	-
	1600-1699	14.6	15	11-19
	1700-1799	11	7	3-28
	1800-1899	7.1	7.5	2-16
	1900-2000	-	-	-

a)

All Fires	Time Period	Mean	Median	Range
All Years	1575-2000	1.7	1	1-12
Pre-Settlement	1575-1849	1.4	1	1-12
Settlement	1850-1904	1.6	1	1-4
Fire	1905-2002	5.3	5	1-10
	1600-1699	2	1	1-12
	1700-1799	1	1	1-2
	1800-1899	1.2	1	1-4
	1900-2000	5.1	5	1-10

10% scarred	Time Period	Mean	Median	Range
All Years	1575-2000	2.4	2	1-12
Pre-Settlement	1575-1849	2.3	2	1-12
Settlement	1850-1904	3.7	3	1-10
Fire	1905-2002	-	-	-
	1600-1699	3.5	2	1-12
	1700-1799	1.9	2	1-6
	1800-1899	1.9	1	1-10
	1900-2000	-	-	-

25% scarred	Time Period	Mean	Median	Range
All Years	1575-2000	6.7	4	1-48
Pre-Settlement	1575-1849	6.4	4	1-48
Settlement	1850-1904	11.7	6	4-25
Fire	1905-2002	-	-	-
	1600-1699	18.6	12	3-48
	1700-1799	6	5	1-16
	1800-1899	4.5	4	1-25
	1900-2000	-	-	-

b)

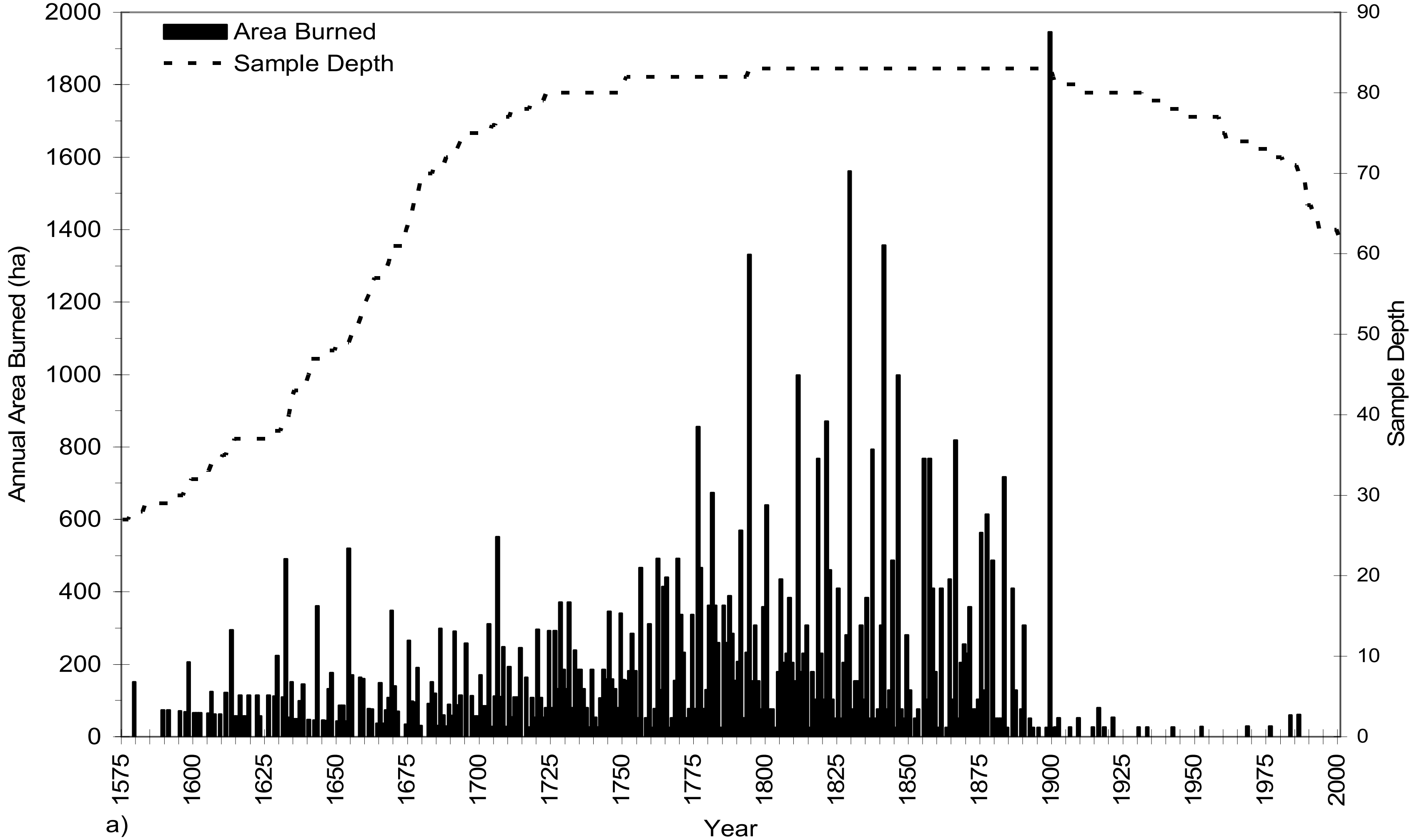


Figure 2.7: Page 1 of 2. Annual area burned (ha) and sample depth from 1575-2000 in old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA.

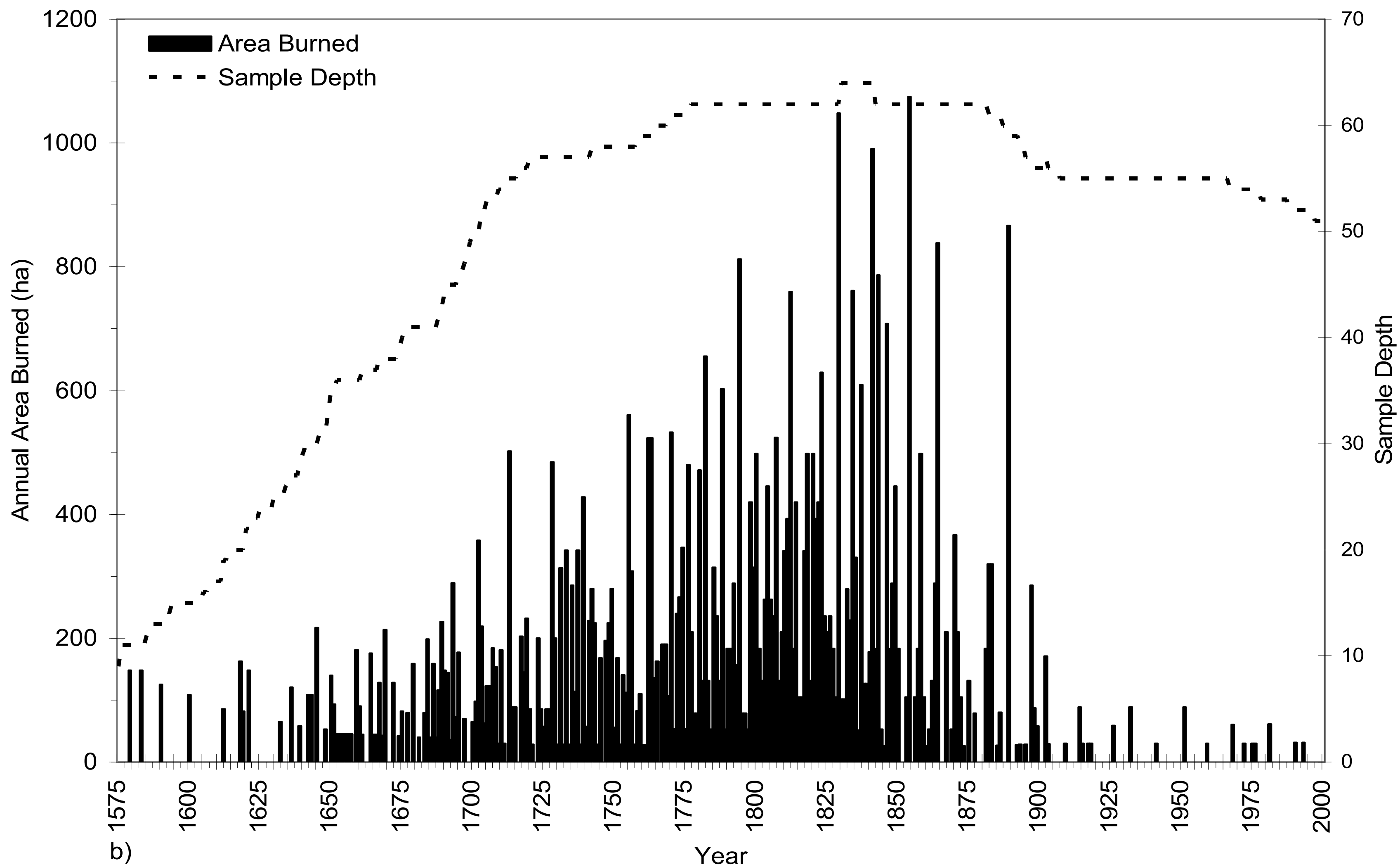


Figure 2.7: Page 2 of 2.

Table 2.18: Average fire size (ha) and rotation by period for old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA.

		Fire Size (ha)			Fire
	Period	Mean	Median	Range	Rotation (Years)
All	1575-2000	205	115	25-1946	16
Pre-Settlement	1575-1849	203	130	26-1562	13
Settlement	1850-1904	266	102	26-1946	10
Fire Suppression	1905-2002	39	28	26-80	378
	1600-1699	120	89	30-520	24
	1700-1799	217	159	26-1331	10
	1800-1899	300	179	26-1946	8
	1900-2000	39	28	26-80	334

a)

		Fire Size (ha)			Fire
	Period	Mean	Median	Range	Rotation (Years)
All	1575-2000	199	131	26-1075	15
Pre-Settlement	1575-1849	209	148	26-1048	11
Settlement	1850-1904	214	118	26-1075	13
Fire Suppression	1905-2002	46	30	30-89	211
	1600-1699	104	86	36-289	33
	1700-1799	200	157	28-813	9
	1800-1899	290	210	26-1075	7
	1900-2000	51	30	29-171	172

b)

Table 2.19: Page 1 of 2. Fire return intervals and area burned by different sized fires for old-growth mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA.

No. Gridpoints	Area (ha)	# Fires	% of Fires	% Area Burned	Cum. Area Burned	% Cum. Area Burned	Mean	Median	Min.	Max.	SD
1	25	63	22.0	3.1	1575	3.1	6.4	4	1	37	6.8
2	50	41	14.3	4.0	3625	7.0	10.1	8	1	67	11.4
3	75	36	12.5	5.2	6325	12.3	9.2	5	1	36	9.4
4	100	25	8.7	4.8	8825	17.1	10.8	7	1	75	15
5	125	6	2.1	1.5	9575	18.6	52.6	33	16	134	49.6
6	150	20	7.0	5.8	12575	24.4	10	6	1	36	9.6
7	175	11	3.8	3.7	14500	28.1	19.5	10	1	54	20.9
8	200	8	2.8	3.1	16100	31.2	26.3	14	3	71	30.3
9	225	10	3.5	4.4	18350	35.6	26.3	22	3	63	23.3
10	250	6	2.1	2.9	19850	38.5	32.6	31	2	60	20.9
11-15	275-375	28	9.8	17.2	28700	55.6	8.7	6	2	29	7
16-20	400-500	14	4.9	11.7	34750	67.3	13.9	8	3	50	13.4
21-25	525-625	4	1.4	4.4	37000	71.7	28.7	9	2	75	40.3
26-50	650-1250	12	4.2	19.3	46975	91.0	9.7	7	2	30	8.3
51-85	1275-2125	3	1.0	9.0	51600	100.0	52.5	53	35	70	24.8

a)

Table 2.19: Page 2 of 2.

No. Gridpoints	Area (ha)	# Fires	% of Fires	% Area Burned	Cum. Area Burned	% Cum. Area Burned	Mean	Median	Min.	Max.	SD
1	25	58	23.1	3.3	1450	3	7.6	5	1	84	11.5
2	50	30	12.0	3.4	2950	7	12.5	9	1	42	10.7
3	75	23	9.2	4.0	4675	11	13.7	13	1	76	16.1
4	100	22	8.8	5.0	6875	16	10.2	5	1	46	11.4
5	125	16	6.4	4.6	8875	20	13.7	13	1	35	9.5
6	150	8	3.2	2.8	10075	23	30.4	18	3	109	36.5
7	175	18	7.2	7.2	13225	30	10.5	10	1	24	7.4
8	200	10	4.0	4.6	15225	35	19.8	22	2	42	14
9	225	6	2.4	3.1	16575	38	12.2	14	2	20	7.6
10	250	7	2.8	4.0	18325	42	27	16	2	92	33.7
11-15	275-375	21	8.4	14.9	24825	57	9.1	7	1	29	7.1
16-20	400-500	18	7.2	18.7	32975	76	8.5	6	1	27	8.3
21-25	525-625	4	1.6	5.5	35375	81	18.3	14	6	35	15
26-64	650-1625	10	4.0	18.9	43625	100	10.6	8	2	25	7.8

b)

Fire Rotation – The fire rotation for the entire period was similar between Big Oak Flat (15 years) and the South Fork Merced (16 years) study areas. Fire rotations also varied ($p<0.01$) by time period (Table 2.18a, b). The fire rotation for the pre-settlement period was 13 years (BOF) and 11 years (SFM), while the settlement period was 10 years (BOF) and 13 years (SFM). This is due to the high number of large fires (> 400 ha) that occurred during these periods. In contrast, the fire rotation for the fire suppression period was much longer at 378 years (BOF) and 211 years (SFM), because only a few, small fires burned during the 20th century. Fire rotation also varied by century in both areas with the shortest rotation during the 19th century (BOF = 8 yrs, SFM = 7 yrs), and the longest during the 20th century (BOF = 334 yrs, SFM = 172 yrs).

2.3.9 Age Structure Patterns and Fire Severity

I identified six age-class structural groups in each study area from the cluster analysis of stems >100 yrs old. All species were present in each structural group, and tree densities varied ($p<0.01$) between groups (Figure 2.8a, b, Table 2.20a, b). White fir was the most prevalent species in each group, although it was often among the youngest trees in each group. Ponderosa pine tended to be the most widely distributed among age classes, and also comprised the oldest trees in the groups.

Figure 2.8: Next page. Mean age-class distribution for each species in the six age-class groups identified by cluster analysis of stems >100 yrs old in 20 yeaqr age-classes in old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA; n= number of plots. Species acronyms are Abco (Abies concolor), Cade (Calocedrus decurrens), Pila (Pinus lambertiana), Pipo (Pinus ponderosa), Psme (Pseudotsuga menziesii), Quke (Quercus kelloggii).

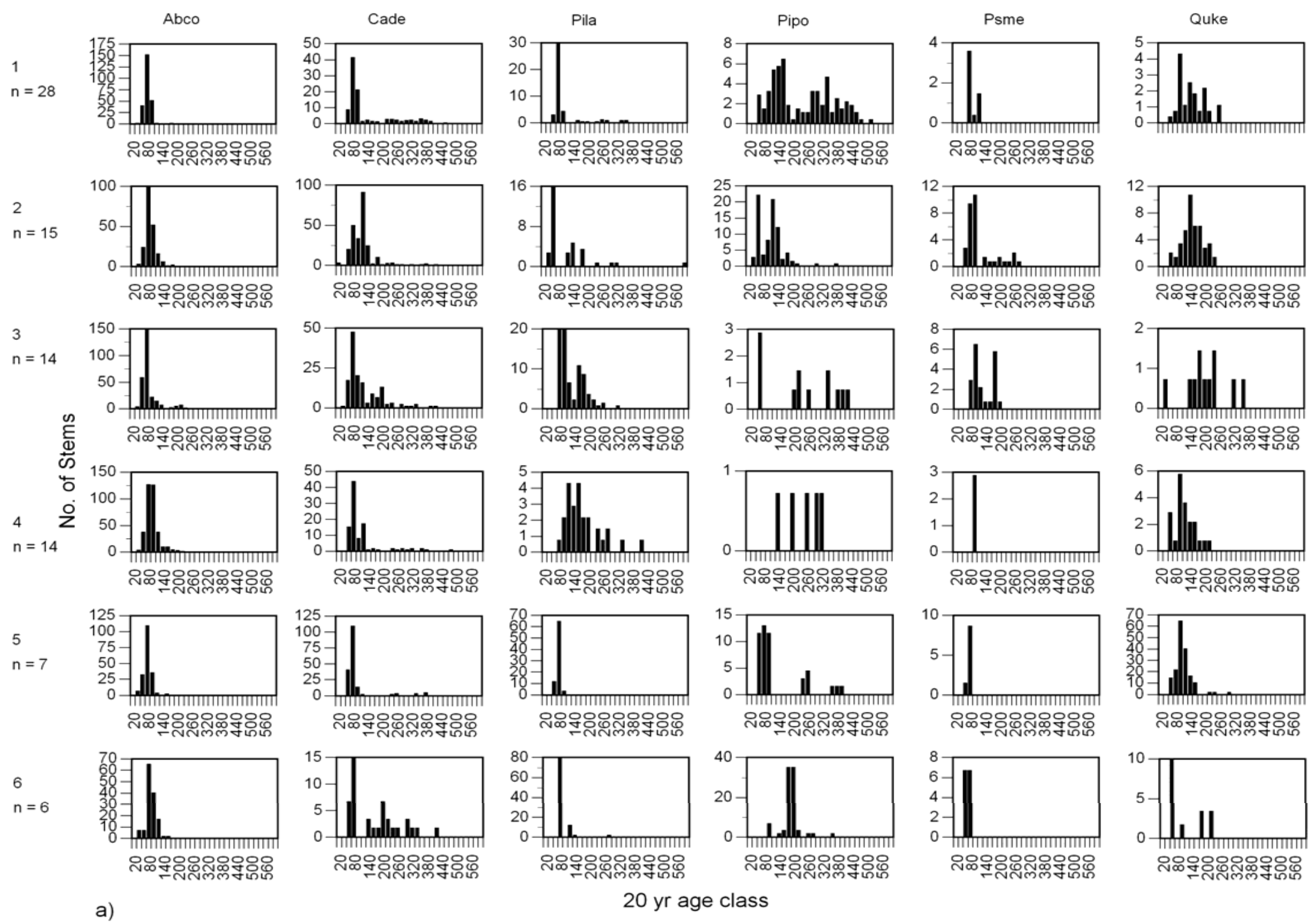


Figure 2.8: Page 1 of 2.

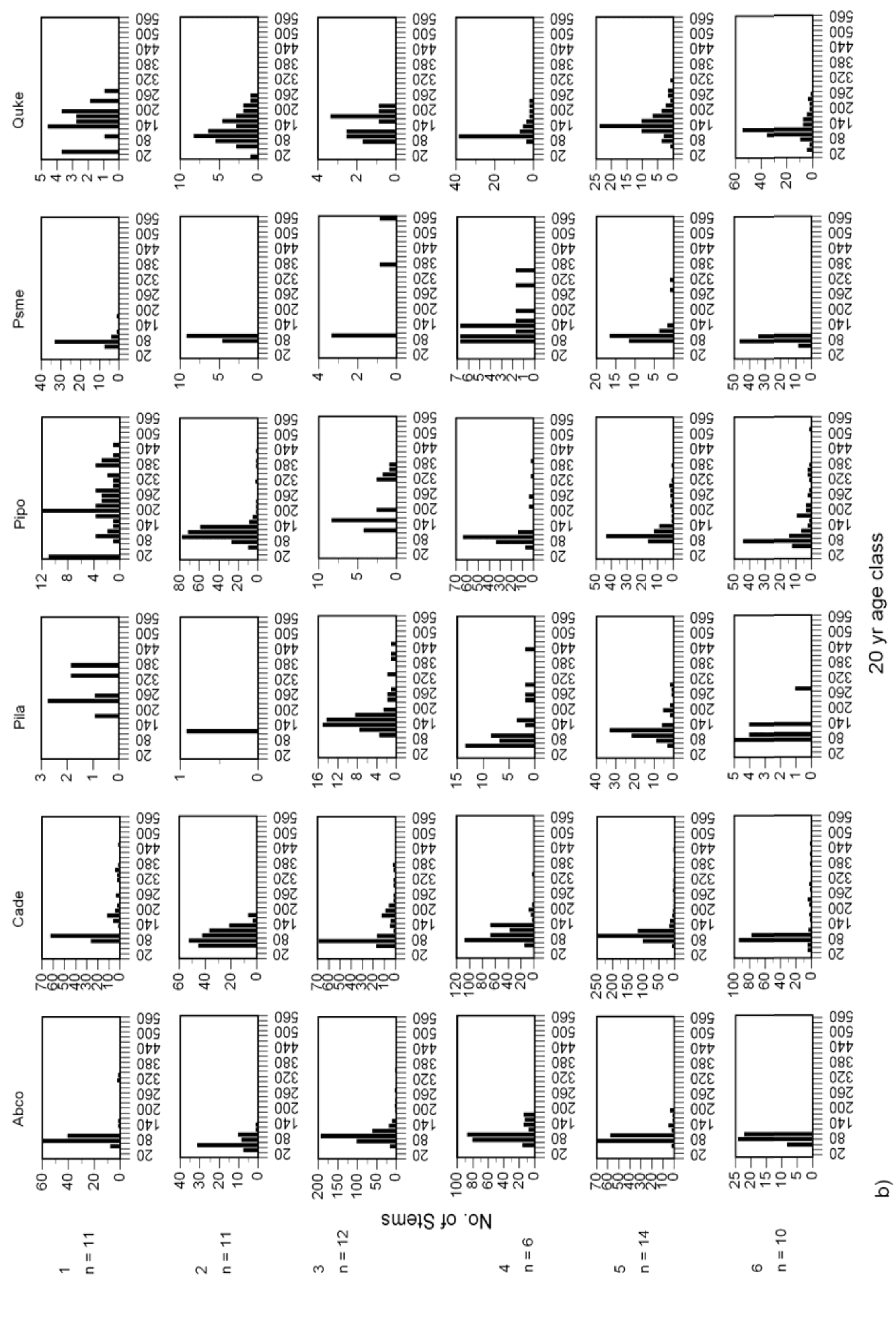


Figure 2.8: Page 2 of 2.

Table 2.20: Mean density and number of occupied age-classes of trees >100 years and all aged trees for groups identified through cluster analysis of trees > 100 years old in 20 year age-classes in plots in old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA. Mean number of fires and number of fires per age class listed for each group.

		Stems >100 yrs (ha^{-1})			All Stems (ha^{-1})			# 20 yr Age classes > 100		# 20 yr age classes all			Fires
Group	n	Mean	Range	SD	Mean	Range	SD	Mean	Range	Mean	Range	Mean # Fires	per age class
1	28	92.5	20-300	57	459	90-960	236	5.5	2-10	7.9	4-12	27	4.9
2	15	257	30-680	187	622	110-1180	315	5.6	3-9	8.2	5-11	25.9	4.6
3	14	151	50-340	100	519	300-820	189	5.9	2-10	8.5	6-11	28.6	4.8
4	14	127	60-340	81	499	270-780	163	4.7	3-7	7.2	5-11	20.9	4.4
5	7	99	30-140	36	664	180-1040	353	4.4	3-6	7.4	6-10	26.7	6.1
6	6	153	90-200	46	405	220-600	161	6	4-7	7.8	5-10	11.3	1.9

a)

		Stems >100 yrs (ha^{-1})			All Stems (ha^{-1})			# 20 yr Age classes > 100		# 20 yr age classes all			Fires
Group	n	Mean	Range	SD	Mean	Range	SD	Mean	Range	Mean	Range	Mean # Fires	per age class
1	11	109	40-230	68	366	90-810	261	4.6	3-7	6.5	3-8	22.8	5
2	11	240	20-1330	369	579	110-1420	417	3.8	2-5	6.4	4-9	14.3	3.7
3	12	217	20-380	108	636	190-1180	325	5.4	2-10	7.4	4-11	19.3	3.6
4	6	235	70-480	158	790	330-1130	369	5.3	3-8	7.7	6-10	19.2	3.6
5	4	315	50-630	187	924	190-2180	548	6	3-9	8.3	5-11	24.6	4.1
6	10	137	40-270	77	586	230-1330	332	5.8	1-8	8.6	5-10	12.8	2.2

b)

2.3.9.1 Big Oak Flat study area

Group 1 plots are the lowest density plots on average with ponderosa pine, incense cedar, and sugar pine present in a wide range of age-classes (Figure 2.8a, Table 2.20a). All of the white fir, Douglas-fir and black oak were less than 300 yrs old. High densities of white fir, incense cedar and sugar pine less than 100 years old are the result of infilling due to fire suppression in the plots. Plots in this group contained an average of 5.5 age-classes >100 yrs (range 2-10) and they burned an average of 27 times. The high number of fires and continuous establishment of ponderosa pine suggest that this group experienced mainly frequent, low severity fires for which most of the established canopy trees survived.

Group 2 plots are the densest plots in the study area. The majority of stems in this group are less than 300 yrs old, with a large number of stems between 80-180 yrs old. Older stems (>300 yrs) of ponderosa pine, incense cedar and sugar pine were present, but infrequent. On average, these plots contained 5.6 age-classes (range 3-9) and burned 25.9 times, suggesting that this plot also experienced frequent, low severity fires. The paucity of trees in older age-classes may be the result of moderate or high severity fires >250 yrs ago.

Group 3 plots are moderately dense, and are distinguished by ponderosa pine, incense cedar, and black oak in a wide range of age-classes. The majority of stems in the plots are white fir, incense cedar and sugar pine < 140 yrs old, with few stems older than 300 yrs for the same species. Plots burned on average 28.6 times, and contained an average of 5.9 age-classes (range 2-10). The pattern of age-classes suggests a punctuated tree establishment following low to moderate severity fires.

Group 4 plots are moderately dense and characterized by very sparse populations of Douglas-fir and ponderosa pine, although the ponderosa pine ranged from 120-320 yrs old. White fir, incense cedar, sugar pine and black oak are found predominantly in the 80-160 yr age-classes, with few older trees. The oldest trees (>400 yrs) in this group are incense cedar. On average, plots in this group contained 4.7 age-classes, and burned 20.9 times. The sporadic establishment of trees suggests that these plots experienced low to moderate severity fires.

Group 5 plots were of very high density with the majority of the trees between 60-120 yrs old for all species. Older trees were present but only incense cedar and ponderosa pine had stems over 300 yrs old. These plots burned an average of 26.7 times and contained 4.4 age-classes (range 3-6). The distribution of age-classes and the high number of fires per age-class (6.1) suggests that these plots experienced frequent low and moderate severity fires.

Group 6 plots were of moderate density with the majority of stems 60-100 yr old white fir and incense cedar. The majority of sugar pine and Douglas-fir in the plots were also in these age-classes. The majority of the ponderosa pine stems present were 160-200 yrs old. Black oak were sparse and <220 yrs old. Plots in this group contained an average of 6 age-classes, and burned 11.3 times. The distribution of age-classes, the high number of age-classes occupied, along with the low number of fires, suggest that this group experienced infrequent fires of low to moderate severity.

2.3.9.2 South Fork Merced study area

Group 1 plots have the lowest average density with ponderosa pine and incense cedar present in a wide range of age-classes (Figure 2.8b, Table 2.20b). All of the Douglas-fir and black oak are less than 300 years old and the majority of white fir and Douglas-fir stems are less than 100 years old. Plots in this group contained an average of 4.6 age-classes > 100 yrs (range 3-7) and burned an average of 23 times. The wide range of occupied age-classes and high number of fires per age-class (5) indicates that these plots experienced mainly frequent low severity fires which most of the existing canopy trees survived.

Group 2 plots are moderately dense and are characterized by very few sugar pine and Douglas-fir in the population. The majority of trees in the population are less than 200 years old, with a few black oak up to 300 years in age. Ponderosa pine is the most prominent species ranging from 60-400 years in age. Plots in this group contained an average of 3.8 age-classes > 100 yrs (range 2-5) and they burned an average of 14 times. The predominance of trees in younger age-classes along with the relatively few fires suggests that this plot experienced infrequent, low to moderate severity fires.

Group 3 plots are moderately dense and characterized by individuals of all species across a wide range of age-classes. Only black oak is limited to stems < 240 years old. The majority of stems in the plot are white fir and incense cedar <

160 years old, with stems extending up to 400 years old for both species. Plots burned an average of 19 times and contained an average of 5.4 age-classes > 100 yrs (range 2-10). The broad range of age-classes present and the high frequency of fires suggest that these plots experienced frequent, low severity fires which most of the canopy trees survived.

Group 4 plots are some of the densest plots in the study area with incense cedar, sugar pine, ponderosa pine and Douglas-fir in a wide range of age-classes. However, the majority of stems for all species are between 60-140 yrs old. All of the white fir and black oak were less than 300 years old. Plots contained an average of 5.3 age-classes > 100 yrs (range 3-8) and burned 19.2 times. The wide distribution of age-classes coupled with the high number of burns suggest that these plots experienced frequent, low severity burns that killed few of the established canopy trees.

Group 5 plots are the densest and are distinguished by the majority of stems < 320 years old for all species. The age distribution of stems was similar across species with the majority of the stems white fir and incense cedar. Plots in this group burned 24.6 times and contained an average of 6 age-classes > 100 yrs (range 3-9). The high number of fires and continuous age structure suggest that these plots experienced very frequent, low intensity fires which most of the canopy trees survived. However, the limited number of stems >320 yrs old (3) suggests that this plot may have experienced a higher severity fire around 320 years ago that killed the majority of trees in the plot at that time.

Group 6 plots are some of the lowest density plots in the study area and are distinguished by incense cedar and ponderosa pine in a wide range of age-classes. In contrast, all of the white fir and Douglas-fir are < 100 yrs old. The majority of stems in the plots are from 60-120 yrs old with few stems > 400 yrs. Plots contained an average of 5.8 age-classes > 100 yrs (range 1-8) and burned 12.8 times. The low frequency of fires along with the sparse number of trees suggests that these plots experienced infrequent, low to moderate severity fires that may have killed several of the canopy trees present.

2.3.10 Fuel Limitations and Fire Occurrence

Consecutive fires tended to burn different sites more than the same sites during all time periods (Table 2.21a, b) ($p<0.01$). Very few consecutive fires burned the same site (BOF 2%, SFM 0%); the majority of fires burned different sites (BOF 57%, SFM 67%). Although a high percentage of fires did burn the same and different sites (BOF 41%, SFM 33%), the number of hectares that re-burned (BOF 4350 ha, SFM 3300 ha) was small compared to the number of different hectares that burned (BOF 47100 ha, SFM 38875 ha). A similar trend was present for different time periods with the same percentage of hectares re-burning (BOF mean 9%, range 7.2–10.7%; SFM mean 7.9%, range 4.5–11.1%), except for the 20th century. During the period of fire suppression, no fires re-burned the same site consecutively. This measure is a conservative estimate of the frequency of sites burning consecutively, because a fire can burn through a grid point without scarring any trees.

Table 2.21: Number and percentage of fires that burned the same, another and both the same and another gridpoint in old-growth, mixed conifer forests in the a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA. Hectares burned are calculated from the number of grid points burned in both fires and those burned only in successive fires.

		Same	Another	Both	n
Whole	n	5	130	92	227
	%	2	57	41	100
	Ha	4350	47100		
Pre-Settlement	n	2	107	78	187
	%	1	57	42	100
	Ha	3550	36650		
Settlement	n	3	18	14	35
	%	9	51	40	100
	Ha	800	10175		
Suppression	n	0	5	0	5
	%	0	100	0	100
	Ha	0	275		
1600-1699	n	1	36	13	50
	%	2	72	26	100
	Ha	425	4500		
1700-1799	n	1	54	35	90
	%	1	60	39	100
	Ha	1350	18700		
1800-1899	n	2	35	44	81
	%	2	43	54	100
	Ha	2525	23625		
1900-2000	n	1	5	0	6
	%	17	83	0	100
	Ha	50	275		

a)

		Same	Another	Both	n
Whole	n	0	129	64	193
	%	0	67	33	100
	Ha	3300	38875		
Pre-Settlement	n	0	105	55	160
	%	0	66	34	100
	Ha	2750	32375		
Settlement	n	0	18	9	27
	%	0	67	33	100
	Ha	550	6125		
Suppression	n	0	6	0	6
	%	0	100	0	100
	Ha	0	375		
1600-1699	n	0	25	3	28
	%	0	89	11	100
	Ha	100	2225		
1700-1799	n	0	58	25	83
	%	0	70	30	100
	Ha	950	15900		
1800-1899	n	0	39	36	75
	%	0	52	48	100
	Ha	2250	20225		
1900-2000	n	0	7	0	7
	%	0	100	0	100
	Ha	0	525		

b)

2.4 DISCUSSION

2.4.1 Forest structure and composition

2.4.1.1 Contemporary Forest

The structure and composition of contemporary mixed conifer forests in Yosemite National Park varied with environmental condition. Tree distribution and abundance patterns were primarily segregated by slope aspect, and secondarily by topographically-influenced soil moisture, and elevation; a pattern consistent with observations in other mixed conifer forests (Parker 1991, 1995, Beaty and Taylor 2001, Bekker and Taylor 2001). Although all species were present in each compositional group, drought tolerant ponderosa pines and oaks were dominant on xeric, south facing slopes, while white fir and incense cedar were more abundant on mesic, north facing slopes. The relatively homogeneous composition in species distribution and abundances in the Big Oak Flat region of Yosemite National Park, compared to other sites in the Sierra Nevada, southern Cascades and Klamath mountains (Vankat and Major 1978, Parker 1994, Taylor and Skinner 1998, Beaty and Taylor 2001, Bekker and Taylor 2001) is probably related to the gentle terrain, without any steep, contrasting slopes. However, other biotic and abiotic factors not evaluated could be influencing the forest composition. Biotic factors such as squirrels could act as both consumers of seed sources, thereby reducing the potential regeneration of certain species, and also dispersers, by caching seeds in different locations in the forest (Vander Wall 1992, 1994). At the same time, abiotic factors such as sunlight could influence regeneration by drying out seedlings that establish in openings in the canopy (Gray and Spies 1997, Gray et al. 2005). In contrast, the greater variation in species distribution and abundance present in the South Fork is probably due to the plots being distributed across a prominent slope aspect (north = 23% slope, south = 19% slope) and elevation gradient (1300-2000 m). Species composition, density and basal area were similar to those found in other mixed conifer forests in Yosemite National Park (Guarin and Taylor 2005).

The size and age structure of the contemporary forest in the Big Oak Flat study area are similar to other mixed conifer forests that have experienced 100 years or more of fire suppression (Vankat and Major 1978, Parsons and DeBenedetti 1979, Taylor 2000, Bekker and Taylor 2001, Guarin and Taylor 2005). Forests were all aged with individuals up to 500 years old. White fir and black oak were restricted predominantly to individuals younger than 300 years. White fir is a fire sensitive species that would not have been dominant in the forest before fire suppression, so there are few older stems, and black oak has a shorter average lifespan (McDonald 1990). Tree densities in Yosemite National Park (525-649 trees ha^{-1}) were less than that encountered in the southern Cascades (2000 trees ha^{-1} (Bekker and Taylor 2001), and 1000 trees ha^{-1} (Taylor 2000)) and Sierra Nevada (800-1700 stems ha^{-1} (Parsons and DeBenedetti 1979) and 900-5000 trees ha^{-1} (Vankat and Major 1978)), but similar to those found elsewhere in Yosemite National Park (680-800 trees ha^{-1} (Guarin and Taylor 2005)). Even though total numbers of stems were fewer in Yosemite National Park than elsewhere, the shape of the age and size distribution of stems were similar, with most of the individuals in the smallest size-classes and less than 100 years old.

The age and size distribution patterns among compositional groups were not consistent with the overall distribution for the forest, mainly due to the grouping process. For example, the white fir-incense cedar group had very few black oak present. A similar pattern was found for white fir in the ponderosa pine-incense cedar-black oak forest type. Although the compositional groups varied from each other, the general trend of high densities of shade tolerant, fire sensitive species (white fir and incense cedar) was consistent across all compositional groups in the study area.

2.4.1.2 Reconstructed Forest

The reconstruction of the 1899 forest is essential in evaluating the relative impacts of fire suppression on the structure and composition of the mixed conifer forests in Yosemite National Park. One of the challenges of the reconstruction technique used is the assumption that remnants of all trees of the historic forest are present in the contemporary forest. However, the absence of any woody material from the historic forest due to either decomposition or consumption by fire can be problematic to a tree-ring based reconstruction (Fule et al. 1997, Stephenson 1999). In addition, in California mixed conifer forests, decay rates are known to vary by species and tree size. Decay rates for fir species are faster than for pine species, and for smaller trees than for larger trees (Kimmey 1955, Harmon et al. 1986, Bull et al. 1997). Consequently, the component of the reconstructed forest derived from trees that were dead in 2002 (31%) will most likely be more reliable for pine species and larger diameter trees. However, the relatively low amount of variation (<7%) between density, basal area and tree size for the reconstructed forest derived under the different decomposition rates suggests that the values are fairly robust.

The relative similarity in values between the reconstructed forest and the 1911 inventory suggests that the techniques used for estimating death dates for the dead trees and subsequent tree sizes are fairly reliable. While there was little variation between the density of the reconstructed forest and 1911 inventory, the variation in basal area and tree size for the whole forest was statistically different. The basal area of the reconstructed forest was 42% greater than the 1911 forest, while average tree size was 19% larger. In addition, only incense cedar and white fir were significantly different from the 1911 forest. Incense cedar basal area was larger (60%) in the reconstructed forest while white fir tree size was smaller (39%). The larger tree size in the reconstructed forest may be the result of the absence of a portion of the smaller diameter individuals than were actually in the forest. The faster decomposition rates for smaller diameter trees could result in limited evidence of them in 2002, and consequently, they would be underrepresented in the reconstructed forest. The presence of younger trees germinating since 1911 would also explain the decrease in average tree size in the contemporary forest. Smaller diameter trees however, have a limited impact on basal area, and their absence probably doesn't explain the difference between the basal area for the reconstructed and 1911 forest. This may be the result of different growing conditions, and decomposition rates between the species, and also between environments in the study area. For example, the presence of incense cedar on both mesic and xeric sites in Big Oak Flat would result in different decomposition rates for the same species, a fact that the reconstruction formulas do not take into account. This is partially due to the limited amount of research involving decomposition rates of mixed conifer species (Harmon et al. 1986). In addition, the decomposition rates were derived from studies in other locations in the mixed conifer zone where climate conditions may vary from Yosemite National Park (Harmon et al. 1987) and, consequently, result in more variation in decomposition than encountered in the study areas. Nevertheless, the similarity between the reconstructed forest and the 1911 survey suggests that the reconstructing method, and its results, provide a reasonable estimate of the pre-fire suppression forest conditions for mixed conifer forests in Yosemite National Park.

My reconstructed 1899 forest is different than other reconstructions of mixed conifer forests in the Lake Tahoe basin (Taylor 2004) and the Sierra San Pedro Martir in Baja CA (Minnich et al. 2000). Reconstructed densities for Jeffrey pine – white fir forests in the Lake Tahoe basin averaged 68 trees ha^{-1}, significantly less than my densities of 130 trees ha^{-1} and 137 trees ha^{-1}. Although the density for Jeffrey pine (55 trees ha^{-1}) in Lake Tahoe was higher than that of ponderosa pine (36 trees ha^{-1} and 40 trees ha^{-1}) in Yosemite National Park, a taxonomically similar species whose ranges overlap and are able to hybridize with each other (Jenkinson 1990, Oliver and Ryker 1990), they were still in the range of densities found in my reconstructions (0-340 stems ha^{-1}). The densities of white fir, however, were similar between Lake Tahoe (13 trees ha^{-1}) and Yosemite National Park (21 trees ha^{-1} and 22 trees ha^{-1}). In contrast, average reconstructed diameters at breast height (dbh) in Yosemite National Park were lower than in Lake Tahoe for both Jeffrey pine (Tahoe = 68 cm, Yosemite = 58.4 cm and 50.7 cm) and white fir (Tahoe = 76.3 cm, Yosemite = 36 cm and 35.6 cm), which is probably due to the presence of smaller trees for both Jeffrey pine (Tahoe range = 54-85.6 cm, Yosemite range = 10.2-157.9 cm and 10.4-194 cm) and white fir (Tahoe range 54.8-113 cm, Yosemite range = 10-176 cm and 10.8-146.6 cm) in forests in Yosemite National Park. Reconstructed basal area, however was similar between the two locations for both Jeffrey pine (Tahoe = 19.4 m^2 ha^{-1}, Yosemite = 11.9 m^2 ha^{-1} and 19.1 m^2 ha^{-1}), and white fir (Tahoe = 5.7 m^2 ha^{-1}, Yosemite = 3.4 m^2 ha^{-1} and 4.1 ha^{-1}). The difference in average density and diameter and similarity in average basal area suggest that while the forests in Yosemite National Park are much denser than those in Lake Tahoe, they are smaller in size, resulting in a similar total basal area present in the forest. Differences between the two forests may be due to the different environments where the reconstructions are based. The Lake Tahoe stands were at a higher elevation and poorer site conditions where ponderosa pine is not present, while the stands in Yosemite National Park were at an elevation below the range of Jeffrey pine. In addition, more species were present in the stands (6) in Yosemite National Park than those in Lake Tahoe (3), and the Lake

Tahoe stands are on the east side of the crest of the Sierra Nevada and therefore receive less precipitation.

The comparison of the reconstruction with the forests of the Sierra San Pedro Martir (SSPM) is slightly different. The data from the SSPM are for the contemporary forest, but since there has been no fire suppression in the region the current forest can be considered representative of the historic forest (Minnich et al. 2000, Stephens and Gill 2005). Forests densities in the SSPM (78-156 trees ha^{-1}) were similar to the average densities found in Yosemite National Park (130-137 trees ha^{-1}). Mean densities in SSPM for white fir (4-105 stems ha^{-1}) and Jeffrey pine (30-110 trees ha^{-1}) were broader than in Yosemite National Park (WF = 21-22 trees ha^{-1}, JP = 36-40 trees ha^{-1}), although not outside the range of densities I reconstructed. Moreover average basal area for the reconstructed forests in Yosemite National Park (36.3-49 m^2 ha^{-1}) was slightly greater than SSPM (19.9-46 m^2 ha^{-1}). Mean basal area for white fir in Yosemite National Park (3.1-4.1 m^2 ha^{-1}) was less than SSPM (1-19 m^2 ha^{-1}), although within the range, while Jeffrey pine was similar (SSPM = 8-22 m^2 ha^{-1}, Yosemite = 11.9-19.1 m^2 ha^{-1}). Average tree sizes were also larger in the reconstructed forest (43.2-50 cm) than in SSPM (32.6 cm), for both Jeffrey pine (Yosemite = 50.7-58.4 cm, SSPM = 30.8 cm) and sugar pine (Yosemite = 56.2-66 cm, SSPM = 40.4 cm), however, white fir diameters were similar (Yosemite = 35.6-38 cm, SSPM = 38.6 cm). In addition, the presence of incense cedar and sugar pine in SSPM allowed the comparisons of those species. My reconstructions contained much higher densities of incense cedar (36-44 trees ha^{-1}) but similar amounts of sugar pine

(16 trees ha^{-1}) compared to SSPM (IC = 13 trees ha^{-1}, SP = 5-21 trees ha^{-1}). Basal area for incense cedar in Yosemite National Park (10.6 – 11.4 m^2 ha^{-1}) was also larger compared to SSPM (6 m^2 ha^{-1}), while sugar pine was more similar (Yosemite = 7.6 – 9.6 m^2 ha^{-1}, SSPM = 1-17 m^2 ha^{-1}). While the density of stems was similar for both the reconstructed forests and the SSPM, the diameter of trees and basal area for Yosemite National Park was larger. In addition, deciduous trees (California black oak) comprised a significant proportion of the trees (15%) in the reconstructed forest. The differences present indicate that historic forests in the central Sierra Nevada were structurally different than the forests of the Sierra San Pedro Martir.

2.4.2 Fire Regimes

There was little spatial variation in fire regime parameters in the study area. The composite median and point fire return intervals were similar across all aspect, and compositional groups, and for the entire study area. This differed from other mixed conifer forests in the Southern Cascades (Taylor 2000, Beaty and Taylor 2001), Klamath mountains (Taylor and Skinner 1998, Taylor and Skinner 2003), and Sierra Nevada (Kilgore and Taylor 1979, Beaty 2004), where the majority of fire history studies identified significant variation in FRIs related to different slope aspects, compositional groups, or both. Median point fire return intervals across aspects in Yosemite National Park (7-11 years) were slightly shorter than those in the southern Cascades (13-32 years (Taylor 2000) and 19-

54 years (Beaty and Taylor 2001)), and Sierra Nevada (mean = 12-24 years (Kilgore and Taylor 1979).

Comparisons of composite FRIs are more difficult since they are dependent upon the size of the sample area, and the larger the study area the smaller the fire return interval. A fire return interval for each grid point, on the other hand, only comprises 2-5 samples from an area of approximately 9 ha surrounding the grid point and can be more easily compared with fire histories based upon several samples for each collection site. Median grid point FRIs for aspect and compositional groups in Yosemite National Park (5-7 years) were similar, albeit at the shorter end, to those identified in the northern Sierra Nevada (8-17 years (Beaty 2004), 9 years (Taylor 2004) and 8-15 years (Stephens and Collins 2004)) and central Sierra Nevada (9-16 years (Kilgore and Taylor 1979) and 2-22 years (Swetnam et al. 2000). A previous study near the South Fork found similar median site FRIs (2-10 years (Swetnam et al. 2000)) as this study.

The similarity in fire regime intervals among slope aspect and compositional groups in Yosemite National Park is most likely related to the topography and fuel bed characteristics of the study area. Terrain in the Big Oak Flat region includes all slope aspects, and is very gentle, so fuels and forest cover are relatively continuous, and there are no major fire breaks present. Consequently, there is little impediment, other than previous burn patches, to the spread of fire from north- to south-facing slopes. In addition, the presence of long needle pines (i.e. ponderosa pine and sugar pine) in every compositional group can contribute to high fire frequency. Pine dominated stands produce fine fuels faster than fir dominated stands in mixed confer forests (Agee et al. 1978, Stohlgren 1988), and fine fuels from long needle pines are less dense and consequently can dry out faster and carry fire more readily than fuels from short needle fir (i.e. white fir and Douglas fir) (Albini 1976, Van Wagtendonk 1998). Although the terrain in the South Fork Merced is more strongly contrasting, the presence of pines on all aspects may contribute to the similar fire return intervals. Moreover, every compositional group was present on each aspect, so there were only small differences in fuel bed characteristics between different aspects, which can contribute to the similarity in group FRIs.

The greatest spatial variation in FRIs was for widespread fires that scarred 25% or more of the samples. However, the variation in fire return intervals was not significantly different among slope aspect or compositional groups. This is most likely due to the relatively low number of gridpoints within the same slope aspect or compositional group being near each other. Whereas composite fire intervals are not dependent on the proximity of neighboring gridpoints, widespread fires within a particular group are related to the adjacency of sites. Most gridpoints within a particular slope aspect or compositional group were only adjacent to a few other gridpoints within the same group. The result of which divided the landscape into clusters, or patches, of gridpoints belonging to the same slope aspect or compositional group. Consequently, it would be difficult for any given fire event to burn through a large number of gridpoints within a particular group. A fire would have to be of sufficient size to burn across gridpoints of different groups in order to burn another cluster of gridpoints in the same slope aspect or compositional group. In the study area, the majority of fires burned 10 gridpoints or fewer, an extent that, while large enough to burn into other groups, was often not large enough to extend into other patches within the same compositional, or slope aspect group.

Fire in Yosemite National Park mixed conifer forests burned near the end (39% and 54% latewood) or after the growing season (40% and 38%). A similar pattern is found elsewhere in the central Sierra Nevada (55% latewood and 38% dormant) (Caprio and Swetnam 1995) and in Yosemite National Park near the South Fork (55% latewood and 24% dormant) (Swetnam et al. 2000). Season of burn in Yosemite National Park is in the middle of the range for mixed conifer forests along the Pacific coast. There is a latitudinal gradient in the season of burn from predominantly early (91 % earlywood) in the Sierra San Pedro Martir in Mexico (Stephens et al. 2003), to dormant (80-90%) in northern California (Taylor and Skinner 1998, Beaty and Taylor 2001, Taylor and Skinner 2003). This seasonal progression of when fires burn is probably related to the later drying out of fuels in the north as a result of the northward movement of the Pacific high pressure cell during the California summer (Western Regional Climate Center 2003). Patterns of season of burn were different for south-facing slopes in both study areas. There were a higher number of late season fires (44% and 60%) than dormant season fires (34% and 36%). This local variation may be due to the southern slopes in the study area drying out earlier than the rest of the slopes, and consequently are more combustible earlier in the year at the time of the highest number of lightning strikes in the region (July-September) (Swetnam et al. 2000).

There was little difference in fire occurrence in the pre-settlement and settlement period for both study areas. This suggests that early settlers did not alter the fire frequency from what existed prior to their settlement of the region. Fire occurrence was significantly less frequent during the fire-suppression period. This pattern was similar to that found in other mixed conifer forests in the Southern Cascades (Beaty and Taylor 2001, Bekker and Taylor 2001), Klamaths (Taylor and Skinner 1998, Taylor and Skinner 2003) and northern Sierra Nevada (Beaty 2004). However, a few studies in the central Sierra Nevada, (Kilgore and Taylor 1979, Caprio and Swetnam 1995) and the Southern Cascades (Norman and Taylor 2005) found fire frequencies dropped slightly during the settlement period, which may be related to grazing of livestock in the mountains (Kilgore and Taylor 1979). The lack of reduction in fire frequency in the study areas suggests that grazing did not have an impact on fires. A dramatic decline in fire frequency occurred after the establishment of both the Stanislaus National Forest in 1897 and the National Forest Service in 1904. One objective of the National Forest Service was to implement a policy of fire suppression. Similar dramatic declines in fire occurrence in the early 1900s, when the forest reserve system was established, have been reported for other mixed conifer forests (Kilgore and Taylor 1979, Caprio and Swetnam 1995, Beaty and Taylor 2001). Nevertheless, fire frequencies were relatively high during the fire-suppression period, but the fires were small. They were spot fires caused by lightning strikes. Both locations have recorded several lightning strikes within the boundary of the study area every year from 1985 – 2000 (J. van Wagtendonk, pers. comm.). Fire return intervals were significantly longer for fires scarring 10% and 25% of samples during the fire suppression period.

Variation in fire size has been found to be related to the presence of natural fire breaks, such as rock outcrops, riparian zones and streams (Taylor 2000, Beaty and Taylor 2001), and climate patterns such as drought (Swetnam 1993, Norman and Taylor 2003, Taylor and Beaty 2005). In my study areas there were few natural fire breaks present, but fire size varied greatly over the period of recorded fires. Mean fire size was 200 ha (range 25-1850 ha) and similar to average fire sizes in mixed conifer forests in the Southern Cascades (106 ha (Beaty and Taylor 2001), 103-145 ha (Bekker and Taylor 2001), and 106 ha (Taylor and Skinner 2003)), and northern Sierra Nevada (203 ha (Beaty 2004)). In contrast, in mixed conifer forests of the Sierra San Pedro Martir, median fire size was 1200 ha (Minnich et al. 2000); significantly larger than in Yosemite National Park (114 ha and 131 ha), although this may be due to differences in how extent was estimated in the two areas. In the SSPM, fire size was calculated from aerial photographs and small fires may not have been included in the fire extent estimations. Fire size based upon fire scars is also a conservative measure, because a fire can burn through a site but not scar any trees and, therefore, leave no evidence behind.

The majority of fires (79%) in both study areas were relatively small in size (≤ 250 ha) and occurred frequently (median FRI = 2 yrs), while the most widespread fires (>800 ha) occurred less frequently (10 yrs). In addition, fires ≤ 250 ha collectively burned 42% of the total area burned, while fires > 800 ha only accounted for 7% and 17% of the total area burned. In fact, fires in each size-class burned an almost equal percentage (mean % of area burned = 3%, range 1-7%) of the total area burned in the study areas. This suggests that in the mixed conifer zone in Yosemite National Park small, frequent fires are just as important at the landscape scale as less frequent, widespread fires.

The average size of fire varied with historical period. Prior to settlement, the mean fire size was 203-209 ha, while from 1850-1904 mean fire size was 214-266 ha. Then, in the 20th century, mean fire size declined significantly to 39-46 ha as a result of fire suppression by both the National Forest Service and National Park Service. The increase in fire size during the settlement period may be related to factors besides Euro-American settlement. Prior to 1775 all fires in both areas were < 500 ha in size. Although the frequency of small fires remained the same after 1775, fires >500 ha burned through the areas. Because the point at which the average fire size shifted is not related to any recorded historical event, it may be the result of changes in larger scale factors, such as climate patterns (see chapter 3).

Fire extent can also be influenced by how recently a previous fire burned the same location. In order for a fire to burn through a spot that was previously burned, enough fuel needs to accumulate on the forest floor to carry the fire. Consequently, the time since last fire at any location can be important to the capacity for a fire to re-burn a previously burned area. This limiting effect can influence where a fire will burn, resulting in the majority of successive fires burning in locations where there has been enough time for sufficient fuel to accumulate to support a fire. This concept of fires tending to burn in locations previously unburned has been described as self-organizing (Holling 1992, Taylor and Skinner 2003). The pattern of successive fires in the study areas suggests that there is a strong tendency towards self-organization in the forest. While nearly half of successive fires burned the same location in addition to a different location, the difference in hectares burned is larger. For all time periods, the percentage of total hectares that burned through a previously burned area ranged from 5-11%, suggesting that when a fire burned into a previously burned area, it quickly burned out, resulting in little overlap in fire areas. At the same time, a similar shift occurred in the frequency of fires that re-burned the same location before and after 1800. Before 1800, 60-70% of fires burned a different location than the fire immediately preceding it, while after 1800 that pattern shifted to only 43-52% of the fires. Half of all fires after 1800 burned a different location in addition to the prior burned location. This shift matches a shift in fire rotation and size at about the same time. Extreme weather (drought) can result in conditions that will cause fires to burn through areas otherwise not conducive to fire spread (Miller and Urban 2000). The change in patterns around 1800 may be the result of changing climate that increased the rate of fuel accumulation, thereby enabling more widespread burns through previously burned patches more frequently than before.

The impact of fire suppression on both fire size and frequency can be seen in the increase in fire rotation at the beginning of the 20th century. Fire rotation in both areas remained similar for both the pre-settlement and settlement periods, but increased 300% in Big Oak Flat and 200% in the South Fork during the fire suppression period. This dramatic increase in fire rotation has been found throughout the southern Cascades and Sierra Nevada (Kilgore and Taylor 1979, Beaty and Taylor 2001, Taylor and Skinner 2003) and is a good indicator of how effective the policy of fire suppression has been. The similar fire rotation and frequency for the pre-settlement and settlement periods suggests that settlers had limited impact. In other locations in the Sierra Nevada, the introduction of grazing into the forests led to a decline in fire frequency and rotation (Vankat 1977, Caprio and Swetnam 1995). Livestock consumed grasses and fine fuels, making it difficult for fires to carry across the forest floor, thereby limiting the size of fires. In Yosemite National Park however, the opposite occurred, with average fire size increasing slightly during this period while fire frequency remained the same, suggesting that some other factor may be influencing the system.

Spatial and temporal variations in fire severity are important sources of structural diversity in forested landscapes, because burns may kill trees and create patches of regeneration. There was limited variation in fire severity present in the study areas. All of the age structural groups identified were multi-aged, and had stems >300 years in age. On average, plots had stems in 5.3 different 20 year age-classes >100 years, and had burned 21.1 times. While the majority of the older trees were fire resistant pines, the presence of older, fire sensitive white fir and incense cedar suggests that the area experienced mostly low or moderate severity fires. The large pulse of trees <100 years is the result of infilling due to fire suppression, rather than a high severity fire. In contrast, the lack of distinct pulses of regeneration present in each group suggests that moderate severity fires occurred in the area historically, creating opportunities for regeneration among surviving trees, resulting in overlapping cohorts. This pattern of primarily low to moderate severity fires is the primary pattern identified in mixed conifer forests (Kilgore and Taylor 1979, Bonnicksen and Stone 1982, Taylor and Skinner 2003), resulting in a mosaic of overlapping patches of trees of similar age that produce a multi-aged forest. In contrast, high severity fires have been identified in some locations in the southern Cascades (Beaty and Taylor 2001, Taylor and Skinner 2003) and the Sierra Nevada (Nagel and Taylor 2005).

However, the mixed conifer forests in Yosemite National Park do not fall completely within either regeneration dynamic. While no high severity fires were identified in Yosemite National Park, there was also limited evidence of very many similar aged patches in the forest. Results of the spatial autocorrelation analysis suggest that the spatial distribution of ages in the forest is a mixture of random regeneration and scattered clumps of similar aged individuals. In addition, the age structural groups suggest that the forest experienced primarily low and moderate severity fires. The result is a forest where most of the canopy trees survive fires, and the openings that are created tend to create establishment opportunities for only a few trees. The result is an all aged forest with scattered clumps of regeneration in the understory mixed in among the canopy trees. Considering the average size of fires, there is always the possibility that the plot size I used was too small to identify the patches of similar aged trees regenerating in openings created by fires, although one study in the Sierra Nevada found relatively small sized patches (6-12 m) of even aged trees (Beaty and Taylor 2007).

2.4.3 Vegetation Change

The absence of fires for 100 years has resulted in very dramatic changes in the mixed conifer forests of the Sierra Nevada (Vankat and Major 1978, Parsons and DeBenedetti 1979, Taylor 2004), the southern Cascades (Taylor 2000, Bekker and Taylor 2001), and the Klamath mountains (Taylor and Skinner 1998, Taylor and Skinner 2003). Similar changes in density, species composition and structure were identified in Yosemite National Park. Although there are few studies reconstructing historic mixed conifer forests in the Sierra Nevada (Bonnicksen and Stone 1982, Taylor 2004), the trends I found were similar to those in other forests that have experienced extended periods of fire suppression (Fule et al. 1997, Fule et al. 2002).

One hundred years of fire suppression allowed the establishment and survival of a large number of seedlings and saplings in the understory; especially among fire intolerant species such as white fir and incense cedar. This has resulted in an infilling of small diameter stems in the forest which has greatly increased the overall density of trees, with relatively little change in the basal area and average diameter for each species, compared to the reference forest. In Yosemite National Park, the number of seedlings and saplings were up to 10-fold greater than the next size-class in some plots. Other studies identified similar increases (Parsons and DeBenedetti 1979) in seedlings and saplings. At the same time, the fire resistant, shade intolerant species (i.e. ponderosa pine, sugar pine) have shown a much smaller relative increase in small diameter trees due to the lack of canopy openings in which to regenerate. One interesting feature is the relative decline in black oak as a percentage of the overall forest structure. With the absence of fire, there are fewer openings in which black oak can regenerate. The decline in regeneration of black oak is more noticeable than in ponderosa pine due to the shorter average lifespan of black oak (McDonald 1990). Most black oak trees in the study area were younger than 200 years old.

The infilling of small diameter trees in all of the plots also resulted in a decline in the structural diversity of the study area between 1899 and 2002. One hundred years of fire suppression has produced a forest that is more homogeneous across the whole study area with trees filling in size-classes that may have been empty in 1899. A similar change has been identified in Lake Tahoe, where Shannon's diversity index declined from the presettlement (mean diversity = 2.4) to the contemporary (mean diversity = 2.1) forest (Taylor 2004). In addition, all of the compositional groups identified have shown a progressive change towards plots that are becoming more dominated by fire-sensitive, shade intolerant species (white fire, incense cedar), from fire tolerant species (ponderosa pine and sugar pine). This convergence to a forest increasingly dominated by fire sensitive species is not limited to mixed conifer forests. Similar successional trends have been identified for higher elevation true fir forests (red fir - white fir) that are becoming increasingly dominated by white fir (Taylor and Solem 2001, Beaty 2004), red fir - western white pine forests shifting to red fir (Bekker 1996, Beaty 1998), lodgepole pine forests being succeeded by white fir and red fir (Bekker 1996, Taylor and Solem 2001, Beaty 2004), and mixed conifer forests being replaced by white fir and incense cedar (Beaty 1998, Taylor and Skinner 2003).

The spatial pattern of mixed conifer forests throughout the Sierra Nevada and southern Cascades has also changed as a result of fire exclusion (Norman 2002, Beaty 2004, Taylor 2004). The relatively low number of plots in the historic forest that demonstrated positive spatial autocorrelation suggests that small (<10 m) and intermediate (22-30 m) sized patches of similarly aged trees (<10m), while a regular part of the forest structure, are not the only pattern present since they only occurred in 10% of the plots. The reconstructed forests in Yosemite National Park were fairly open, and the horizontal pattern of tree ages were not clustered in the majority of plots indicating a forest matrix of trees of different ages, with scattered clusters of even aged patches within it. This can be the result of regeneration occurring among existing canopy trees, or due to successive fires thinning out a cluster of seedlings that established in a cluster where an opening occurred in the canopy. The establishment of trees among existing individuals is further supported by the Ripley’s analysis which identified significant clustering at distances of <10 m in 30-50% of the plots, while positive spatial autocorrelation at distances <10 m was only present in 3-10% of the plots. This suggests that the presence of patches of similarly aged trees in the forest was not very common, and instead, regeneration of individuals among existing trees was more prevalent. This may be due to fires clearing out the understory, while killing few of the canopy trees, and consequently creating opportunities for new individuals to regenerate among the existing canopy trees.

In contrast, the decrease in significant clustering and spatial autocorrelation in the contemporary forest indicates a decline in the amount of clumping of trees, and also a decline in the presence of patches of similar aged trees in both study areas. The similar frequency with which trees of different ages tend to be neighbors (negative spatial autocorrelation), suggests that random regeneration is present in the plots. In addition, several plots were completely random at all distances in 1899 (BOF = 6 plots, SFM = 6 plots) and 2002 (BOF = 22 plots, SFM = 18 plots), and several plots were random in both time periods (BOF = 3 plots, SFM = 6 plots). These patterns suggest that there has been an increase in the degree of random regeneration occurring among the existing trees since the onset of the fire exclusion period. However, due to the minimum coring size of 10 cm, I may have been unable to record any increase in clustering of small diameter and young trees in both study areas. My age structure based upon trees >10 cm dbh identified relatively low densities of trees <80 years in age. However, presence of seedlings (BOF = 1376 stems ha^{-1}, SFM = 850 stems ha^{-1}) and saplings (BOF = 1003 stems ha^{-1}, SFM = 1034 stem ha^{-1}) suggests there has been a continuous establishment of stems over the past

100 years. This increase is related to the increase in establishment of shade tolerant seedlings and saplings that are filling in the space between existing canopy trees. Other mixed conifer forests have shown similar increases in the amount of clumping of stems in the contemporary forests compared to the historic forest. In the Lake Tahoe Basin (Beaty 2004, Taylor 2004) the number of plots showing a clumped distribution doubled for several forest types, while in the southern Cascades (Norman 2002) the distances at which clumping occurred increased in the contemporary forests. The increasing density in the contemporary forests in Yosemite National Park may be resulting in a similar spatial structure that is significantly more clustered than in the reference forests, and this clustering is comprised mostly of fire-sensitive trees <100 years old. However, since the majority of the stems <100 years old are still <10 cm in diameter, the pattern is not evident in the analysis. In addition, the high degree of dispersion present at 1 m is related to the fact that few individuals > 10 cm dbh were within 1 m of each other. The majority of trees that were within 1 m of another individual were seedlings and saplings.

2.4.4 Management Implications

Comparisons of the contemporary forest structure with the reference forest indicate that the mixed conifer forests in Yosemite National Park have changed dramatically over the last 100 years. The historic forests were compositionally different, less dense, and more structurally diverse than the contemporary forest, as a result of fire suppression. The correspondence of all the vegetation changes with the elimination of fires after 1899 indicates the importance of fire as a process in the historic system. Fires in the mixed conifer forests of Yosemite National Park tended to be of low and moderate severity and burned through the forest quite frequently. The effect of frequent fires on the forest structure was to thin out the seedlings and saplings on the forest floor. Fire sensitive species were more susceptible to the low intensity fires and consequently, few of them survived to grow into the canopy. In contrast, the fire tolerant species (ponderosa pine and sugar pine) were more likely to survive the frequent fires and, consequently, those species comprised the majority of canopy trees in the historic forest.

In addition, the self-organizing nature of fires in the forest would aid in the survival of young pines. The amount of time required for enough fuel to build up at a previously burned location allowed younger trees to grow bark thick enough to survive the next fire. Young firs and cedars are very susceptible to fire, so frequent return intervals would result in a high mortality rate among them compared to young pines. Pines are also faster growers than fir and cedar. In the few years between fires at a location, a pine would be able to grow tall enough to have branches above the short flame lengths common in low severity fires in mixed conifer forests.

The spatial pattern of the forests was also directly linked to the frequent, low severity fires. The majority of canopy trees survived the low severity fires, resulting in very few gaps created in the forest structure. Seedlings and saplings that established after a fire, consequently, were not clumped together in openings but scattered through the forest in the small gaps between trees. The mortality of small diameter trees that would have occurred with a successive fire would have further thinned the regeneration layer, creating a forest that is multi-aged with a limited degree of clustering of trees. The removal of a mortality agent for seedlings and saplings has produced the denser contemporary forest containing a higher degree of clustering of stems.

For resource managers interested in restoring a forest to its historic condition, all of the above issues need to be taken into consideration. Simply restoring the structure of the forest (e.g., Species composition, density) will not be a viable option for the long term, nor will it be easy. In addition to the amount of live trees that will need to be removed from the forest, fuel loadings on the forest floor have been estimated in excess of 100 tons ha^{-1} (K. Paintner, pers. comm.; see chapter 3). Without taking into consideration the processes at work that maintained the historic forest, it will be impossible to maintain the historic structure. Eventually the restored forest would return to a condition similar to what is present today. Instead, managers will need to mimic the historic role played by fire in the forest, whether that is through the use of fire, mechanical treatment, or both. In addition, the fire regimes of Sierran mixed conifer forests have been found to vary over time due to long term climate variability (Swetnam and Baisan 2003, Taylor and Beaty 2005). The long term variability in climate conditions and its influence on fire regimes and vegetation dynamics can add an additional level of complexity to the challenge of developing management strategies that mimic the historic range of variability in the structure and processes of the system (Landres et al. 1999, Swetnam et al. 1999).

Chapter 3

Fire-Climate Associations in Mixed Conifer Forests of Yosemite National Park

3.1 INTRODUCTION

Statistical and physical associations between both regional and hemispheric climatic conditions have been identified as key factors controlling the occurrence and extent of fire in coniferous forests of the western United States (Swetnam and Betancourt 1998, Norman and Taylor 2003, Swetnam and Baisan 2003, Hessl et al. 2004, Taylor and Beaty 2005, Sibold and Veblen 2006). At local to regional scales, interannual variability in precipitation is directly related to the occurrence of fire. However, the nature of the relationship varies in different regions of the western U.S. For example, in the Southwest, wetter than normal years precede most years with widespread fire (Veblen et al. 2000, Swetnam and Baisan 2003), while in the Sierra Nevada, antecedent climatic conditions are not always associated with widespread burning (Swetnam and Baisan 2003, Taylor and Beaty 2005). Moreover, while drought is strongly associated with widespread fires, it is not always a requirement (Grissino-Mayer and Swetnam 2000, Taylor and Beaty 2005, Taylor et al. 2008). On a hemispheric scale, several climate teleconnection modes, such as the El Niño-Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO), have been found to correspond with increased fire occurrence across widespread regions (Swetnam and Betancourt 1990, Grissino-Mayer and Swetnam 2000, Norman and Taylor

2003, Swetnam and Baisan 2003, Westerling and Swetnam 2003), and even among continents (Kitzberger et al. 2001). However, the relationship between different hemispheric teleconnection patterns and fire occurrence varies across the western U.S, and over time, making it hard to apply results from one region to another. In addition, the association between climate variability and fire becomes more complex when interactions between different climate patterns are considered. The goal of this analysis is to assess the relationship between fire and climate variability in the Sierra Nevada in an attempt to further understand the influence and variation present in fire-climate interactions across the western U.S.

Interannual variation in precipitation on the U.S. west coast is influenced by the ENSO and PDO, and their mutual interactions (McCabe and Dettinger 1999). During El Niño events (positive or warm phase ENSO), a mid-tropospheric ridge typically develops over the Pacific Northwest, bringing warm, moisture laden air from the tropical and subtropical Pacific over California (Schohner and Nicholson 1989). This results in wetter conditions in the southwestern United States. In contrast, La Niña events (negative or cool phase ENSO) typically bring drought conditions into southern California and the Southwest (Swetnam and Betancourt 1990). The Pacific Decadal Oscillation is an ENSO-like variation in north Pacific sea surface temperature (SST) (Dettinger et al. 2000), that influences precipitation in the western U.S. During the warm phase (positive) of the PDO, the Aleutian Low moves southeastward, bringing conditions similar to El Niño events to the western U.S. (Gershunov et al. 1999). Wetter than normal conditions generally occur in the Southwest, while dryer than normal conditions prevail in the Pacific Northwest. In contrast, cool phases (negative) of the PDO usually bring dryer than normal conditions to the Southwest and wetter than normal conditions to the Pacific Northwest (Gershunov et al. 1999) resulting in a dipole between the Northwest and Southwest U.S. When the Northwest is wet the Southwest is dry, and vice versa (Dettinger et al. 1998). In addition, multidecadal variation in precipitation has been highly correlated with the PDO and the Atlantic Multidecadal Oscillation (AMO) (McCabe et al. 2004). The AMO is a low frequency climate pattern related to sea surface temperature variations in the North Atlantic, that oscillates on a period of 40-50 years (Gray et al. 2004) and is hypothesized to influence north pacific sea surface temperatures and precipitation across the contiguous U.S. (Mestas-Nuñez and Enfield 1999, Enfield et al. 2001). While the precise mechanisms behind the AMO and PDO are still poorly understood, statistically significant patterns have been found between both patterns and decadal scale patterns of precipitation and drought across the entire U.S. (Enfield et al. 2001, McCabe et al. 2004). In addition, changing phases of the AMO influences the strength of the relationship between both PDO and ENSO and precipitation (Enfield et al. 2001, McCabe et al. 2004, McCabe et al. in press).

The identification of strong correlations between fire occurrence and extent in the southwestern United States (Swetnam and Betancourt 1990, Swetnam and Baisan 2003) and central Sierra Nevada (Swetnam 1993, Taylor and Beaty 2005, Moody et al. 2006) with La Niña events suggests a synchronizing effect of the climatic system on fire activity. The result is that the years of widespread fire appear strongly associated with strong La Niña events, and years with small fires are associated with weak La Nina events. Different correlation patterns have been found further north in the Pacific Northwest and northern California, where increased fire extent coincides with El Niño events (Heyerdahl et al. 2002, Norman and Taylor 2003). These opposite patterns are similar to the dipole in precipitation patterns present between the Northwest and Southwest. The region from the central Sierra Nevada to southern Oregon is located along the pivot line between the two poles, and consequently the strength of the fire-climate relationship varies. Several studies in the region have identified a stronger correlation between fire and ENSO (Norman and Taylor 2003, Taylor et al. 2008) than between fire and PDO, while others have found a stronger correlation between fire and PDO (Taylor and Beaty 2005, Moody et al. 2006), than ENSO. Simple associations between climate and fire occurrence cannot be assumed, however, because of the complex interaction of multiple climate teleconnections (i.e. ENSO, PDO, AMO) (Sibold and Veblen 2006, Kitzberger et al. 2007)

Variations in climate teleconnection patterns influence precipitation rates (McCabe and Dettinger 1999), and consequently fire occurrence in the western U.S. (Westerling and Swetnam 2003). El Nino-Southern Oscillation fluctuates on a time scale of 3-7 years while the PDO frequency is 20-30 years. Consequently, when PDO and ENSO are both in a warm phase, they can amplify the influence on precipitation in a region (i.e. constructive), but if the PDO is in a negative phase, and ENSO is in a positive phase, the relative impact on precipitation is diminished (i.e. destructive) (McCabe and Dettinger 1999). In addition, recent studies have identified long-term variation in fire occurrence related to the AMO (Guyette et al. 2006, Sibold and Veblen 2006, Kitzberger et al. 2007). The contrasting results found in the relationship between climate and fire occurrence in the southern Cascades and northern and central Sierra Nevada make it difficult to identify which climate variables have a dominant influence on fire regimes in the region, and where the influence changes from strong correlations with PDO to strong correlations with ENSO. Consequently, the goal of this study is to further clarify the relationship between climate variability and fire occurrence in the central Sierra Nevada. The general null hypothesis of this study is that fire regimes are not influenced by climate conditions and therefore do not vary with climate variability. Specifically, I address the following questions: 1) How are fire occurrence and extent related to regional climate (drought, temperature) variability?; 2) Is interannual variation in fire occurrence and extent related to ENSO, PDO and AMO, and are the relationships stable over time?; 3) Is fire occurrence and extent related to long term fluctuations in the climate teleconnections (ENSO, PDO, AMO) at the interdecadal scale?; and, 4) Do interactions between these climate teleconnections (ENSO, PDO, AMO) influence fire occurrence and extent?

3.2 METHODS

3.2.1 Fire History

Historic (pre-Fire-suppression) fire occurrence and extent were reconstructed in the Big Oak Flat (BOF) and South Fork Merced (SFM) study areas using fire scars in partial cross sections (BOF, n = 209; SFM n = 156 samples) (Figure 3.1a, b) collected from fire scarred trees (Arno and Sneck 1977). In each study area I collected fire-scarred samples on a grid, and points were located at 500 m intervals across the landscape. Sample points included sites on different slope aspects and at different elevations in the study areas.

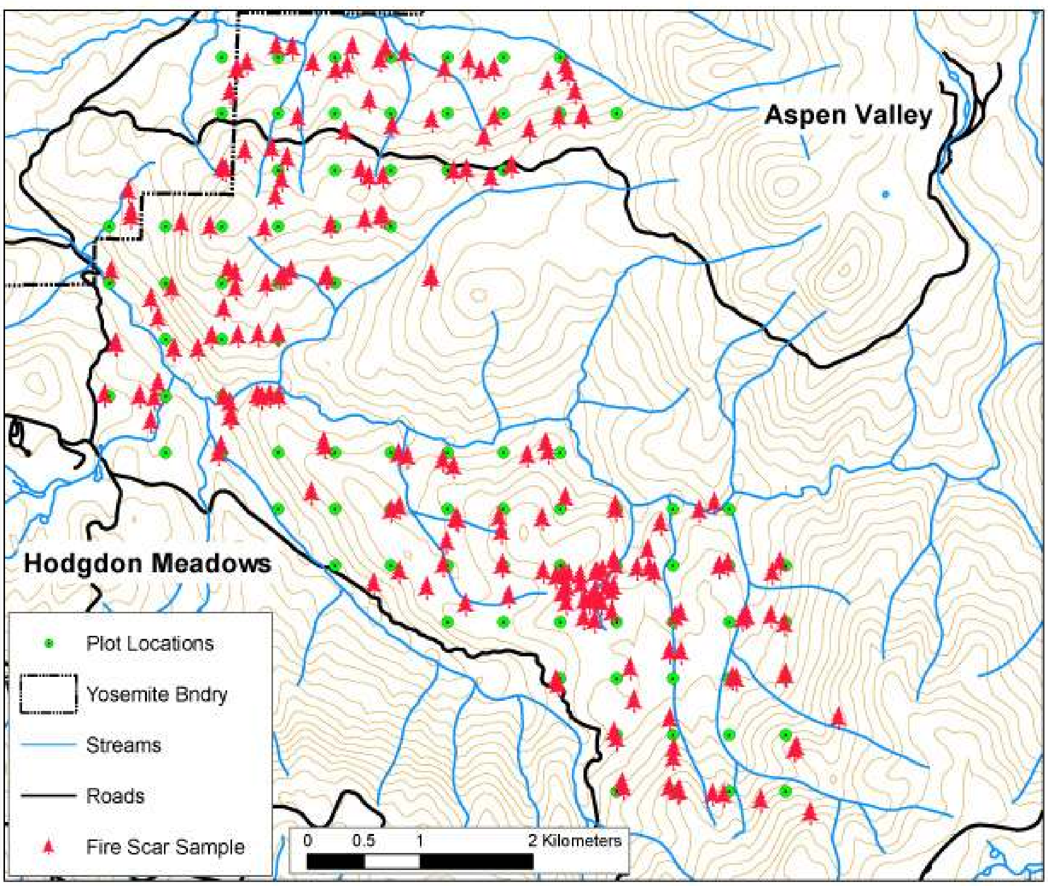


Figure 3.1a: Location of fire scar samples collected in Big Oak Flat (BOF) study area in Yosemite National Park, California.

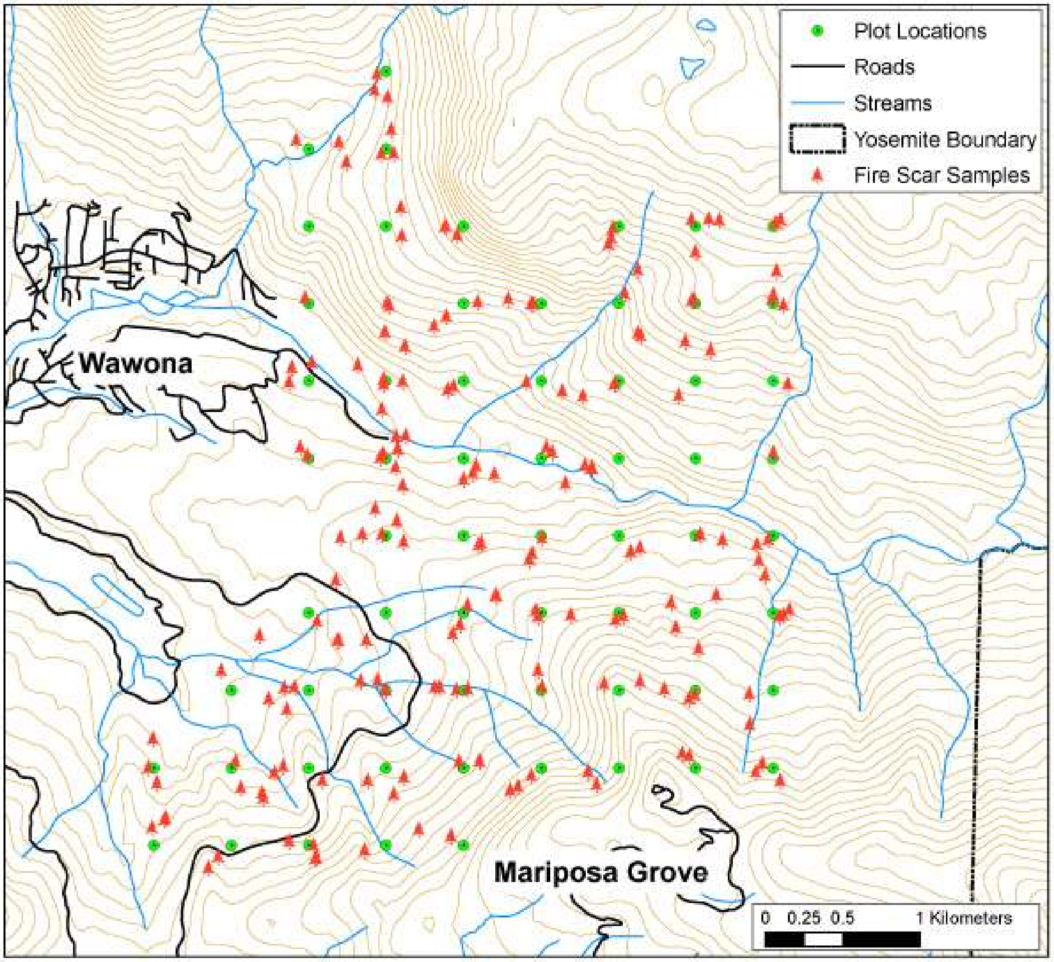


Figure 3.1b: Location of fire scar samples collected in South Fork Merced (SFM) study area in Yosemite National Park, California.

An average of two samples (range 1-5) were collected at each grid point to insure an even distribution of samples across both study areas. Locations of fire scarred samples were determined using a GPS and recorded on a topographic map. The samples were collected primarily from Ponderosa Pine (*Pinus ponderosa*) due to the ability of this species to heal over and survive fires. The date of each fire scar was determined by sanding each fire scarred sample to a high polish and then cross-dating the rings (Stokes and Smiley 1968) with local tree ring chronologies (Ca064: Lemon Canyon http://www.ncdc.noaa.gov/paleo/treering.html) (Holmes et al. 1986). The ring with the scar in it was then determined, and the calendar year of the fire was recorded.

Characteristics of fire occurrence and fire extent were determined for each study area (BOF, SFM) and for both sites together (YOSE), in the following way. First, I determined the fire history for each grid point by aggregating the individual fire events from each sample collected at an individual grid point. Fire size was determined indirectly based on the number of grid points recording a fire in a given year. Each grid point was deemed representative of the 25 ha cell within which it was located, therefore, each grid point burned represented a 25 ha fire. Fire years were then identified for fires burning 25 ha or more (≥1 grid point), 100 ha or more (≥4 grid points), 250 ha or more (≥10 grid points), and 500 ha or more (≥500 grid points) in size for each study area. In addition, I identified fire years where 800 ha or more (≥ 32 grid points) burned in the Big Oak Flat study area. For both study areas combined I identified years when any fire burned, 10% or more, and 20% or more of the grid points in each study area burned. Last, I combined the data for both study areas in order to analyze the fire association between hemispheric teleconnection indices and fire occurrence and extent. I calculated an index of annual fire extent for the region by first calculating the percentage of samples that recorded a fire in each year for each study area. I then summed the percentages for each study area to derive an index for annual fire extent for the region.

Temporal variation in fire regimes were examined in the following way. First, I calculated the 49-year running sums (calculated on the 25th year) for fire frequency (number of fire events) and fire extent (number of grid points burned) in each study area to determine if there was a temporal change in the fire regimes over the entire record. This analysis allows a comparison over time between the number of ignitions and the extent of fires, and can identify changes in the relative frequency of widespread and local fires. Second, differences in frequency of fires of different sizes were identified by comparing fire return intervals for the 1600-1775 and 1776-1900 periods using a *t*-test.

3.2.2 Climate Record

Five reconstructions of climate variables derived from tree rings were used in my analysis of regional fire-climate associations. First, the Palmer Drought Severity Index (PDSI), developed for the south central Sierra Nevada (Gridpoint 47) (Cook et al. 2004), is a proxy for drought severity and is derived from precipitation, temperature and local soil moisture measurements for the current month and prior months, and is a good indicator of local drought conditions (Palmer 1965). Negative PDSI values represent dryer than normal conditions, while positive values represent wetter than normal conditions. The PDSI reconstruction captures 50-70% of the variation in the instrumental data (Cook et al. 1999). Second, I used a reconstruction of western North American summer temperature (TEMP) (Gridpoint 16 (Briffa et al. 1992)) as an index for surface temperature. This reconstruction captures over 50% of the variance in the instrumental record. Third, for an index of ENSO activity I used the Nino3 reconstruction (NINO3) (Cook 2000). The NINO3 index is a measure of SST variations in the equatorial Pacific derived from trees rings from the southwestern U.S.A. and north central Mexico. Positive values of Nino3 occur during El Niño events, while negative values occur during La Niña events. The Nino3 reconstruction captures 43%-52% of the SST variation in the instrumental record (D' Arrigo et al. 2005). Fourth, I used an index of the Pacific Decadal Oscillation (PDO) (Gedalof et al. 2002), derived from coral and tree rings, which measures fluctuations in SST in the North Pacific. A positive value of the index represents the warm phase of the PDO while negative values represent the cold phase. The PDO reconstruction captures 52% of the variation in the instrumental record. Lastly, I used a reconstruction of the Atlantic Multidecadal Oscillation (AMO) (Gray et al. 2004), which measures fluctuations in SST in the northern Atlantic Ocean from 0-70ºN, and was derived from tree rings in eastern North America and western Europe. Positive values for the index represent the warm phase of the AMO while negative values represent the cold phase. This reconstruction captures 66% of the variation in the instrumental record.

The PDSI, TEMP, PDO, NINO3 and AMO reconstructed indices represent variations in climate over large geographic areas, and consequently, may not adequately capture variations in local climate conditions. Therefore, I determined the strength and direction of the relationship between the indices and local climate by calculating the Pearson product moment correlations between each index and local climate data (precipitation, temperature) at both annual and decadal time scales. The local climate data used were mean monthly precipitation and mean monthly temperature from three meteorological stations (Hetch Hetchy, Yosemite Park Headquarters and South Entrance Ranger Station) located in Yosemite National Park and which had long instrumental records. The decadal scale data were derived by calculating a 10 year moving average of the annual data.

3.2.3 Fire-Climate Analysis

The relationship between fire events and local and regional climate patterns was analyzed using correlation analysis, contingency analysis and superposed epoch analysis (SEA) (Haurwitz and Brier 1981, Baisan and Swetnam 1990), at both the interannual and interdecadal scales. In addition, I analyzed fire-climate associations for each half of the record, before and after 1775, and for the entire time period from 1600-1900.

Interannual patterns between fire occurrence and extent and climate were analyzed using SEA. SEA compares the mean climate value (PDSI, NINO3, PDO, TEMP) for years in which a fire occurs, with the five years preceding and two years following each fire year. Monte Carlo simulations (1000 runs) are used to calculate confidence envelopes around the average values surrounding fire years by comparing average climatic conditions before, during and after the event to actual conditions present over the entire record (Veblen et al. 1999). I performed SEA analysis separately for fires of different extent (≥25 ha, ≥100 ha, ≥250 ha, ≥500 ha, and ≥800 ha), and non-fire years for each study area, and for both study areas combined. In addition to the entire record, SEA analysis was performed separately for each half of the record (1600-1775, and 1775-1900). The SEA analysis was limited to the period from 1600 to 1900 to insure adequate sample depth of fire scars and climate reconstructions. Before 1600, fewer than 25% of the grid points were recording fires in either study area, and after 1900 there were very few fires due to fire suppression.

Correlation analysis was used to identify both interannual and interdecadal associations between fire extent and climate variables. Pearson product moment correlation coefficients were calculated between fire extent and each climate variable for each study area and both areas combined. First differences were calculated for each variable (value (year t) – value (year t-1)) to emphasize the interannual variability and remove any temporal autocorrelation present in the data (Swetnam 1993). Interannual scale relationships between variables were identified for the entire record, and for the periods 1600-1775 and 1775-1900. Interdecadal scale relationships were identified by first calculating 10-year means for non-overlapping periods for each variable, then calculating the Pearson Product Moment Correlations between fire extent and each climate variable.

To investigate the relationship between fires and combinations of multiple climate patterns I performed a contingency analysis on all possible phase combinations (positive = "+", negative = "-") of NINO3, PDO, and AMO for fires of different sizes. To test the relationships between all possible phase combinations of the three climate variables at once (e.g. +NINO3/ +PDO/ +AMO) (n=8) and all possible combinations of two of the climate variables at the same time (e.g. +PDO/ +AMO, +NINO3/ -PDO) (n=12), I compared the observed and expected numbers of fire events using a chi-square test. I first identified all of the years in which a fire event occurred (n=280), and all of the years in which the largest fire events (10% or more gridpoints burned) occurred (n=28). I then compared the proportion of all fire years that occurred during each of the phase combinations to determine if fire events occurred randomly during all phase combinations or disproportionately during a particular combination. The distribution of all fires during each phase combination was then used as the expected pattern for the analysis of the largest 10% of fires. I used a Chi-squared analysis to compare the distribution of the largest 10% of fires to an expected distribution derived from all fire events.

3.3 RESULTS

3.3.1 Fire History

The mixed conifer forests of Yosemite National Park experienced a long history of frequent fire. In Big Oak Flat, a total of 286 fires were recorded over the period of 1575 to 2002 (Figure 3.2a), while in the South Fork Merced site, 248 fires were recorded from 1575 to 2002 (Figure 3.2b). The fire history for the combined study areas (YOSE) recorded 316 fires from 1575-2002 (Figure 3.2c). In addition, the fire regimes in the study areas varied temporally over the entire period of analysis. Fire size for each study area and the combined region demonstrated a trend of increasing fire extent from 1620 to 1820 without any noticeable change in fire frequency (Figure 3.3). Over the same period there

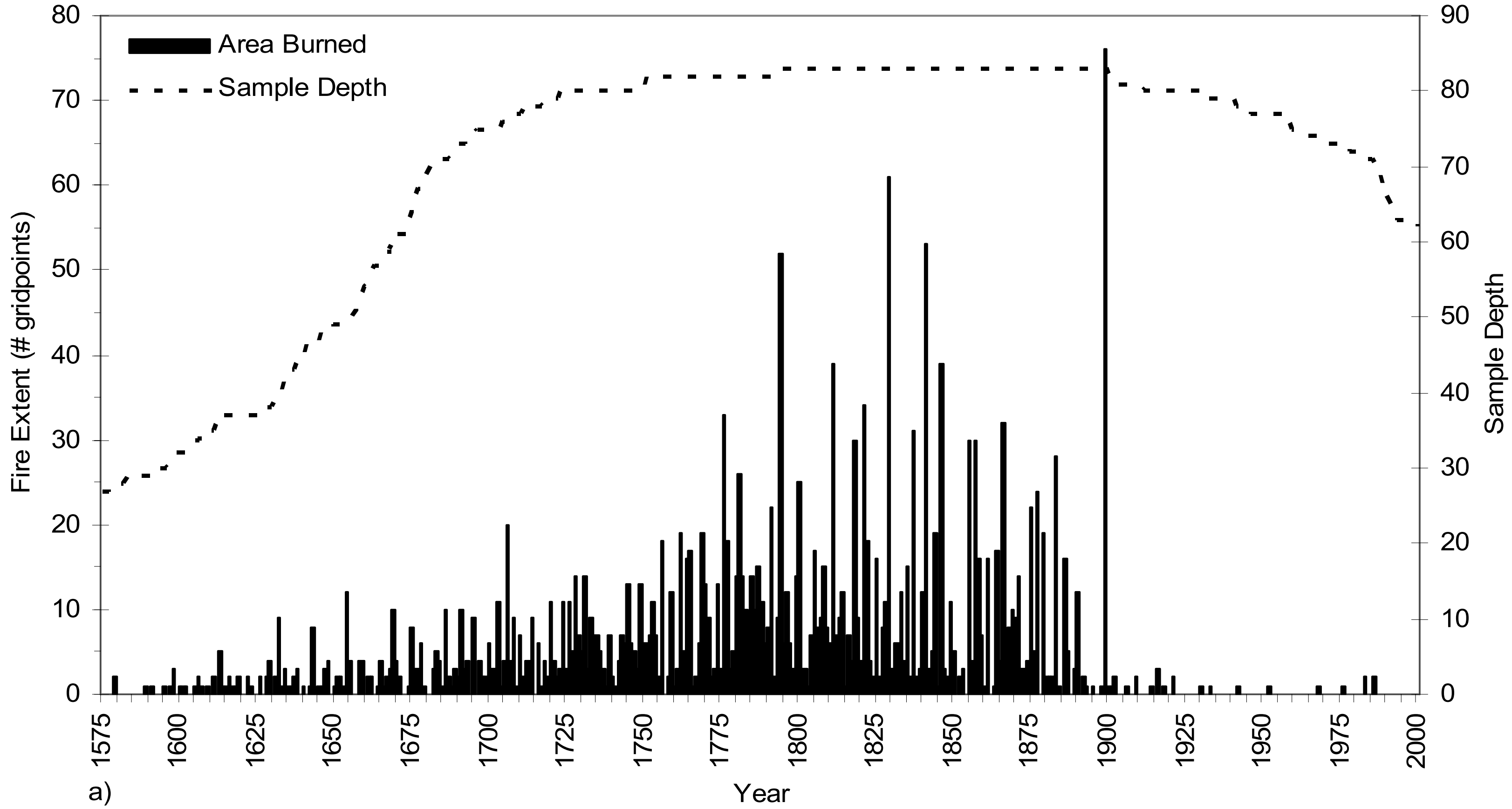


Figure 3.2: Page 1 of 3. Fire extent (# gridpoints) and sample depth from 1575-2000 in old-growth, mixed conifer forests in the a) Big Oak Flat, b) South Fork Merced, and c) both study areas combined, Yosemite National Park, CA.

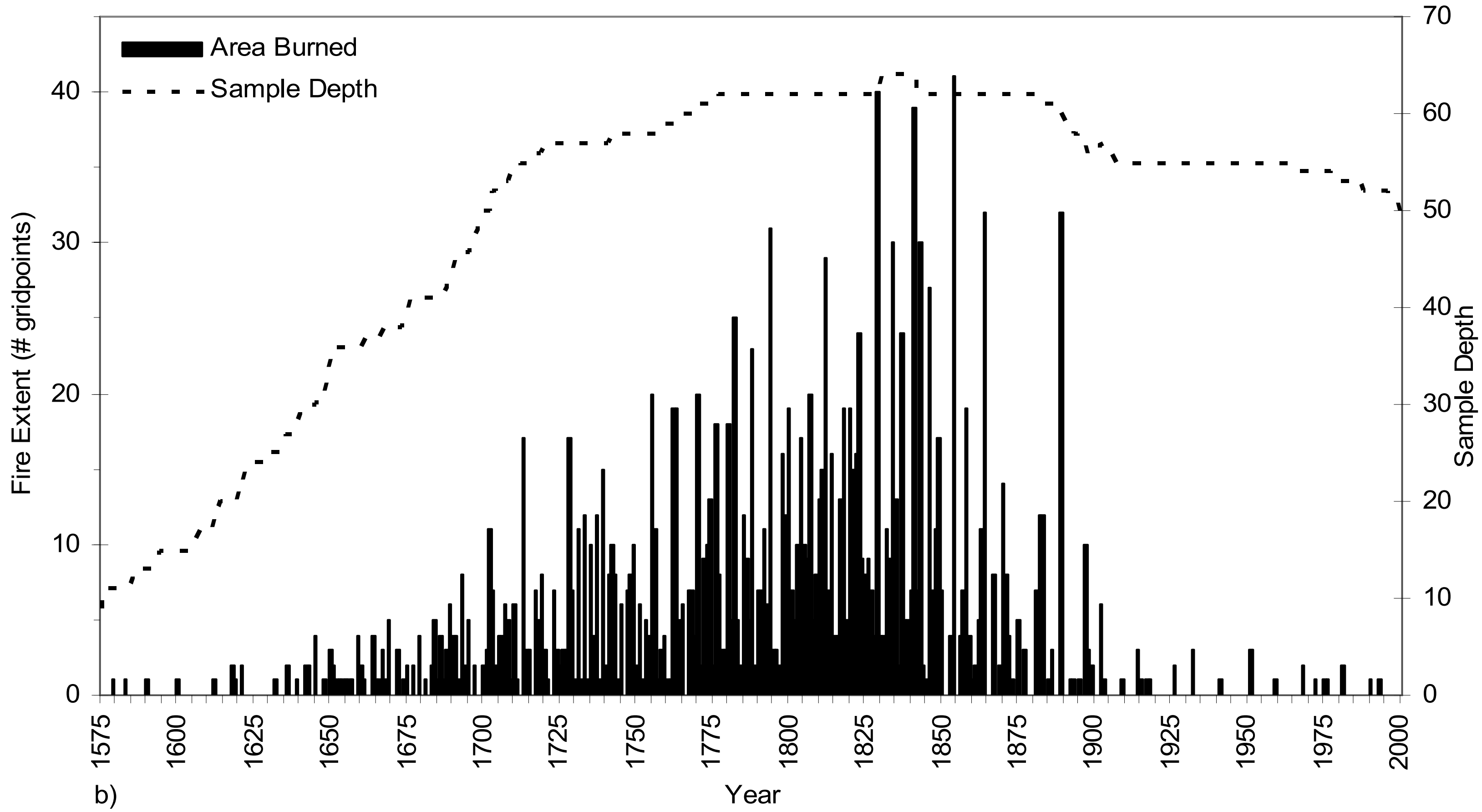
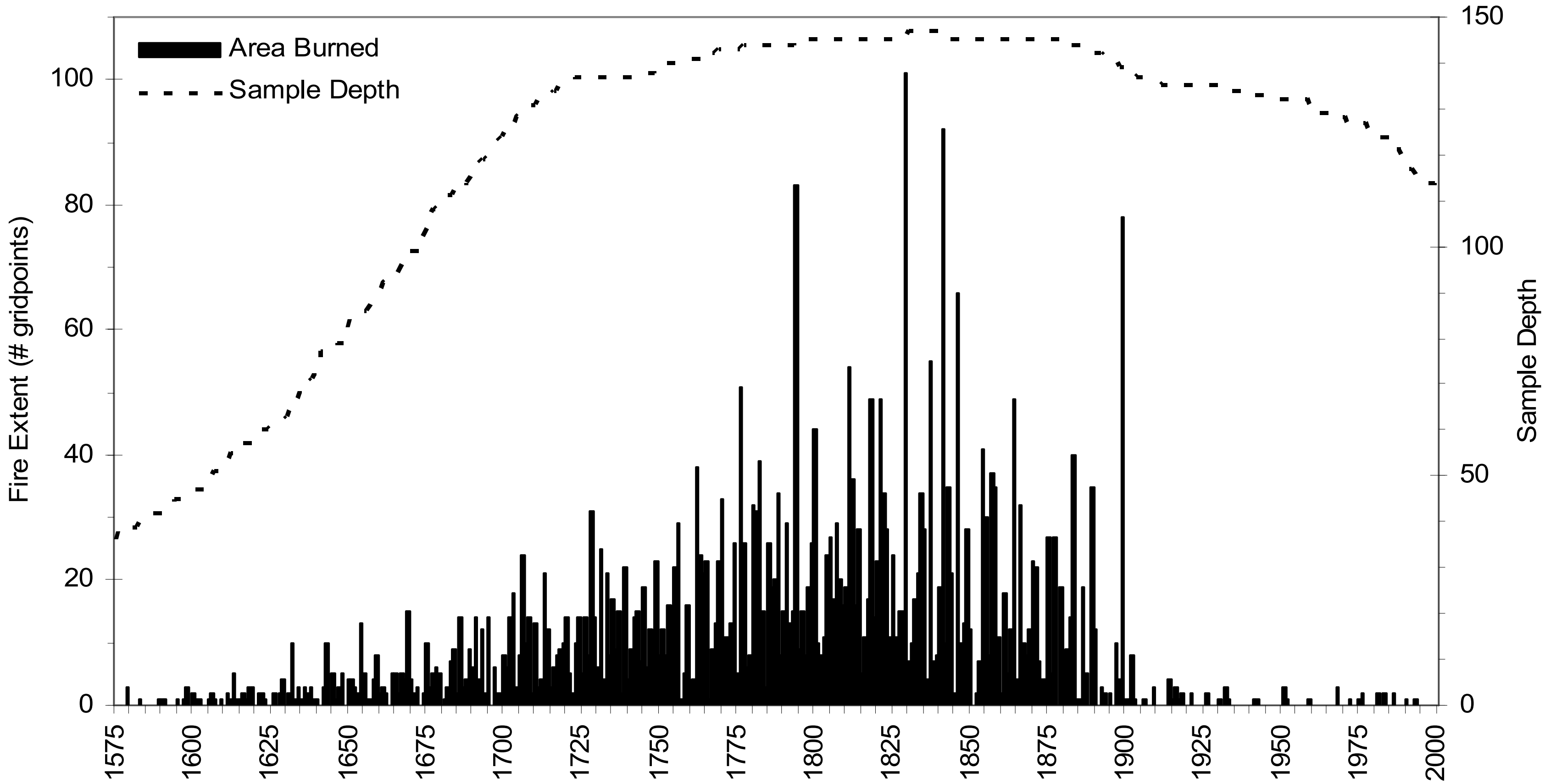


Figure 3.2: Page 2 of 3.



c)

Figure 3.2: Page 3 of 3.

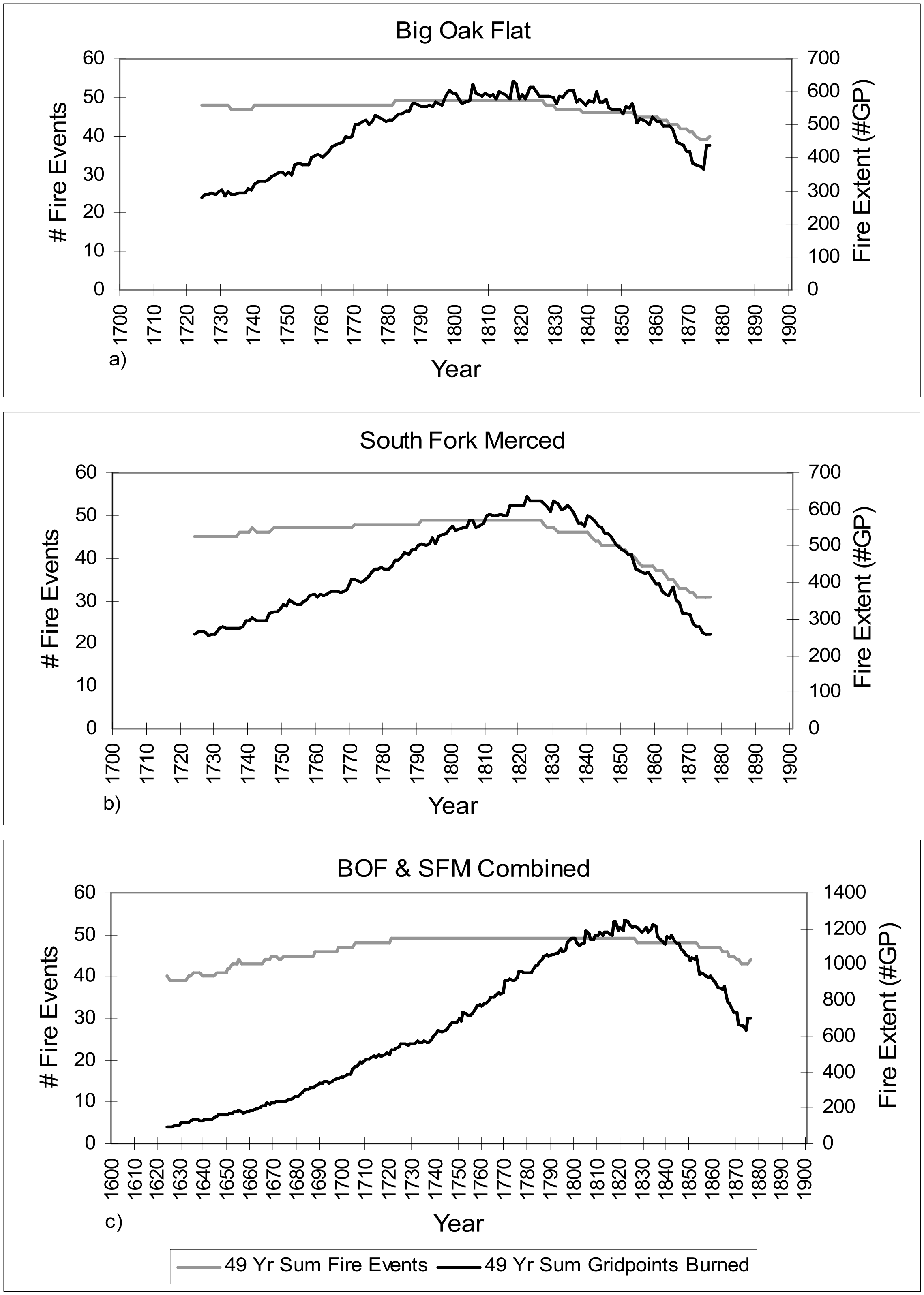
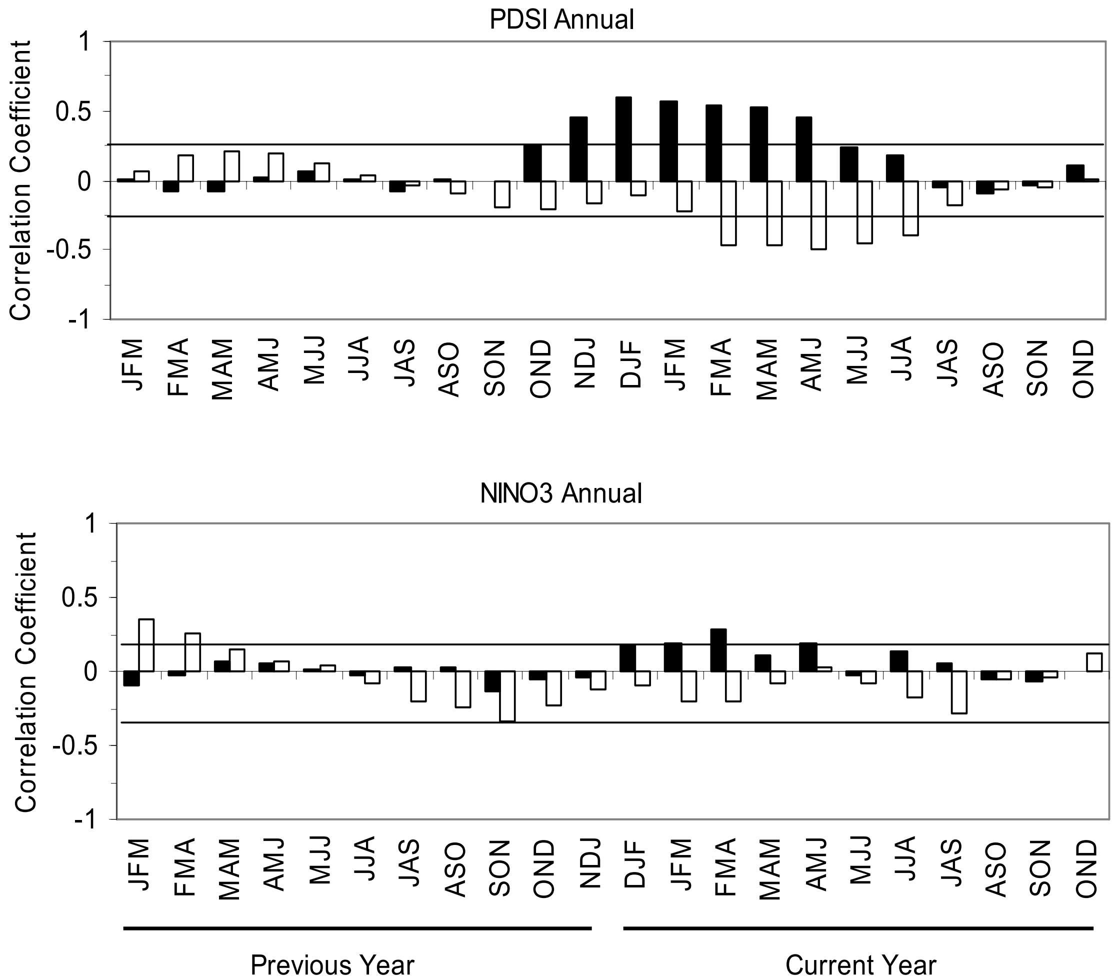


Figure 3.3: Forty-nine year moving sums of fire events and fire frequency for a) Big Oak Flat, b) South Fork Merced, and c) both study areas combined for the period 1600-1900 in old-growth, mixed conifer forests in Yosemite National Park, CA.

was no change in fire frequency for all fires occurring in the region. However, the mean fire return interval for widespread fires that burned 10% or more of the grid points was significantly longer ($p<0.01$, *t*-test) for the period before 1775 (mean 2.7 years) than the period after (mean 1.9 years). During the period between 1600 and 1900 there was a significant amount of synchroneity in fire dates between the two study sites ($X^2 = 194.5$, $p<0.001$); higher than would be expected if the fires occurred randomly.

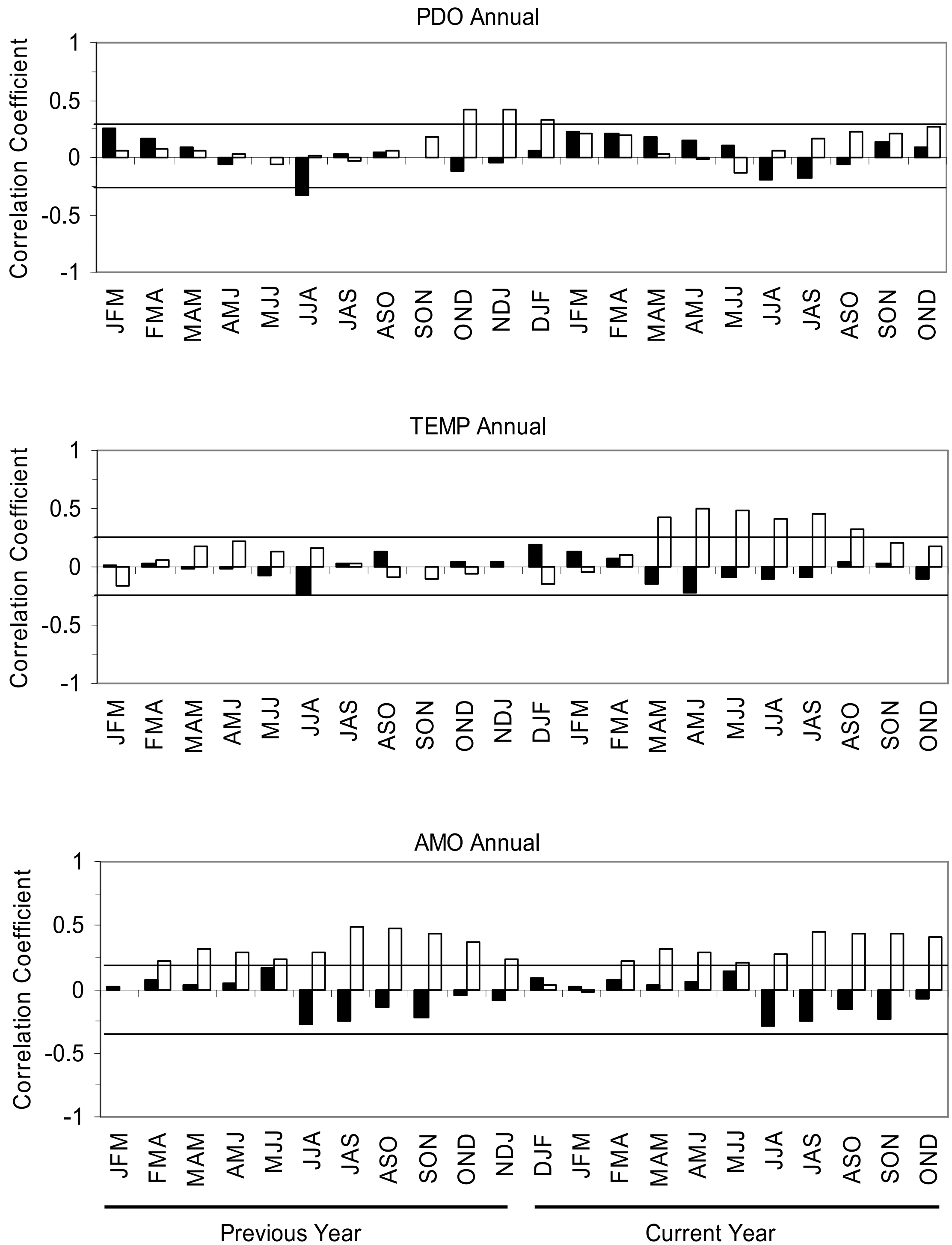
3.3.2 Regional-Local Climate Associations

Annual and decadal scale variation in the climate indices was associated with variation in local climate at annual and decadal time scales (Figure 3.4). Current year PDSI was significantly correlated ($p<0.05$) with precipitation (+) during the wet season (fall, winter, spring) and temperature (-) during the growing season (spring, summer) at annual time scales. At decadal time scales, PDSI was significantly correlated ($p<0.05$) with previous year and current year temperature (+) for all seasons and precipitation during both wet (+) and growing (-) seasons. PDO was correlated ($p<0.05$) with current year winter temperature (+) at annual time scales, and strongly correlated ($p<0.05$) with current and lagged precipitation during winter (+) and summer (-) seasons, and temperature (+) for all seasons, at decadal time scales. Local climate variation was not correlated with NINO3 at annual time scales, but was significantly correlated ($p<0.05$) with temperature (+) for all seasons and precipitation during winter (+) and summer (-) seasons, at decadal time scales. Current year regional



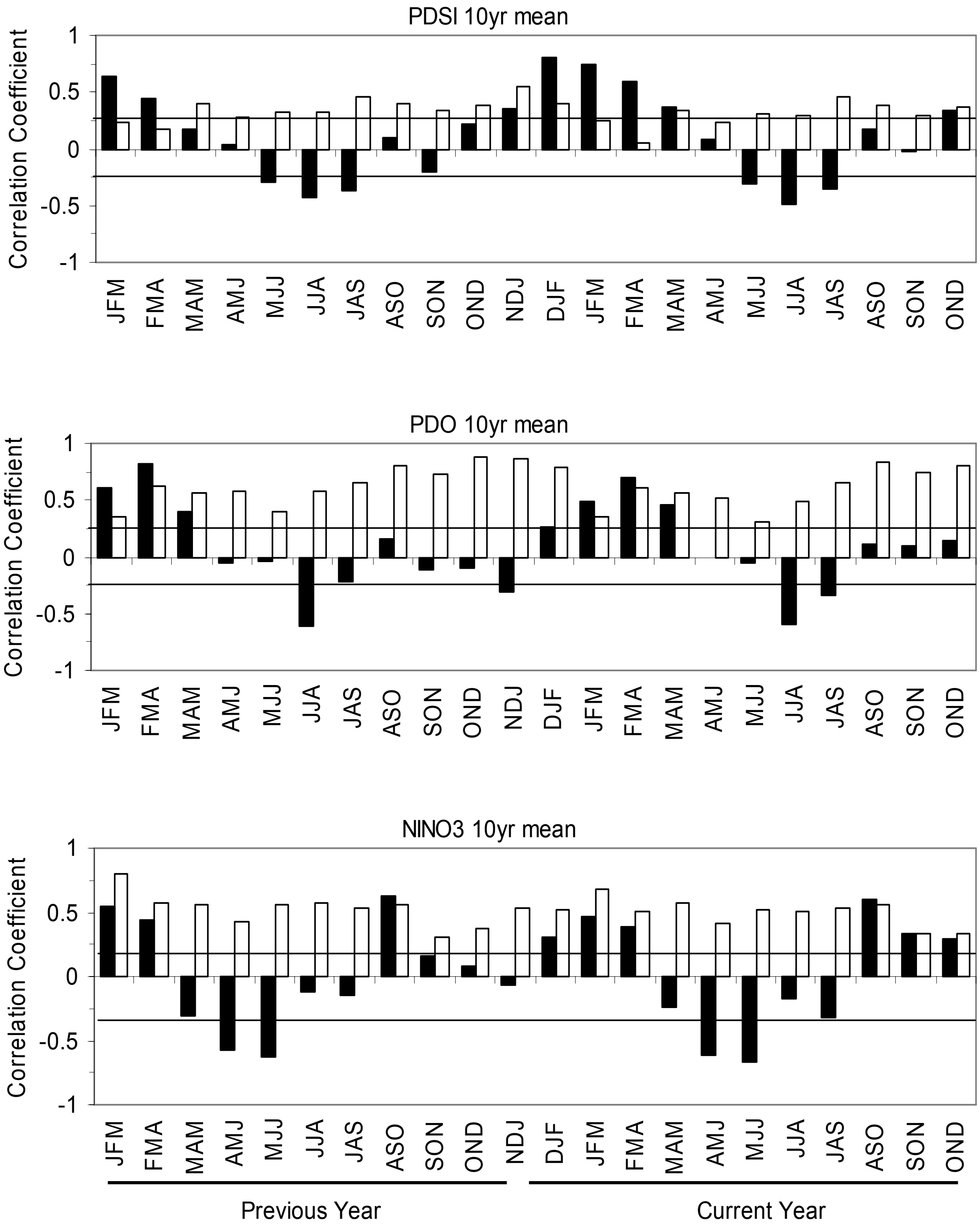
a)

Figure 3.4 a-b: Pearson product moment correlations of local temperature (open bars) and precipitation (solid bars) (1931-1978, n=48 years) with the reconstructed Palmer Drought Severity Index (PDSI, gridpoint 47) (Cook et al. 2004), reconstructed temperature (TEMP, gridpoint 16) (Briffa et al. 1992), reconstructed El Niño index (NINO3) (Cook 2000), reconstructed Pacific Decadal Oscillation index (PDO) (Gedalof et al. 2002), and Atlantic Multidecadal Oscillation index (AMO) (Gray et al. 2004) for the current year and previous year. Correlations were computed for both a) annual and b) decadal (10 year moving average) time-scales. Significance ($p<0.05$) is indicated by the horizontal lines.



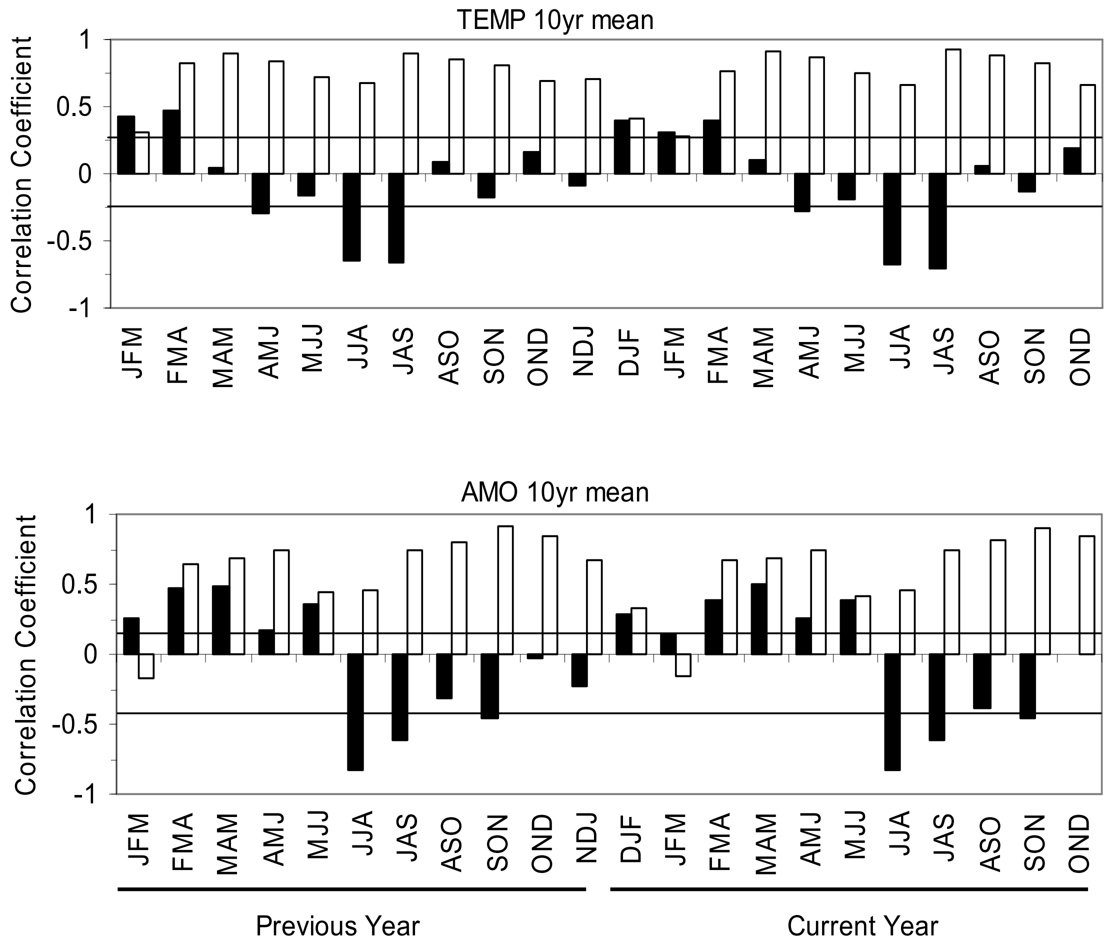
a)

Figure 3.4a: Page 2 of 2.



b)

Figure 3.4b: Page 1 of 2.



b)

Figure 3.4b: Page 2 of 2.

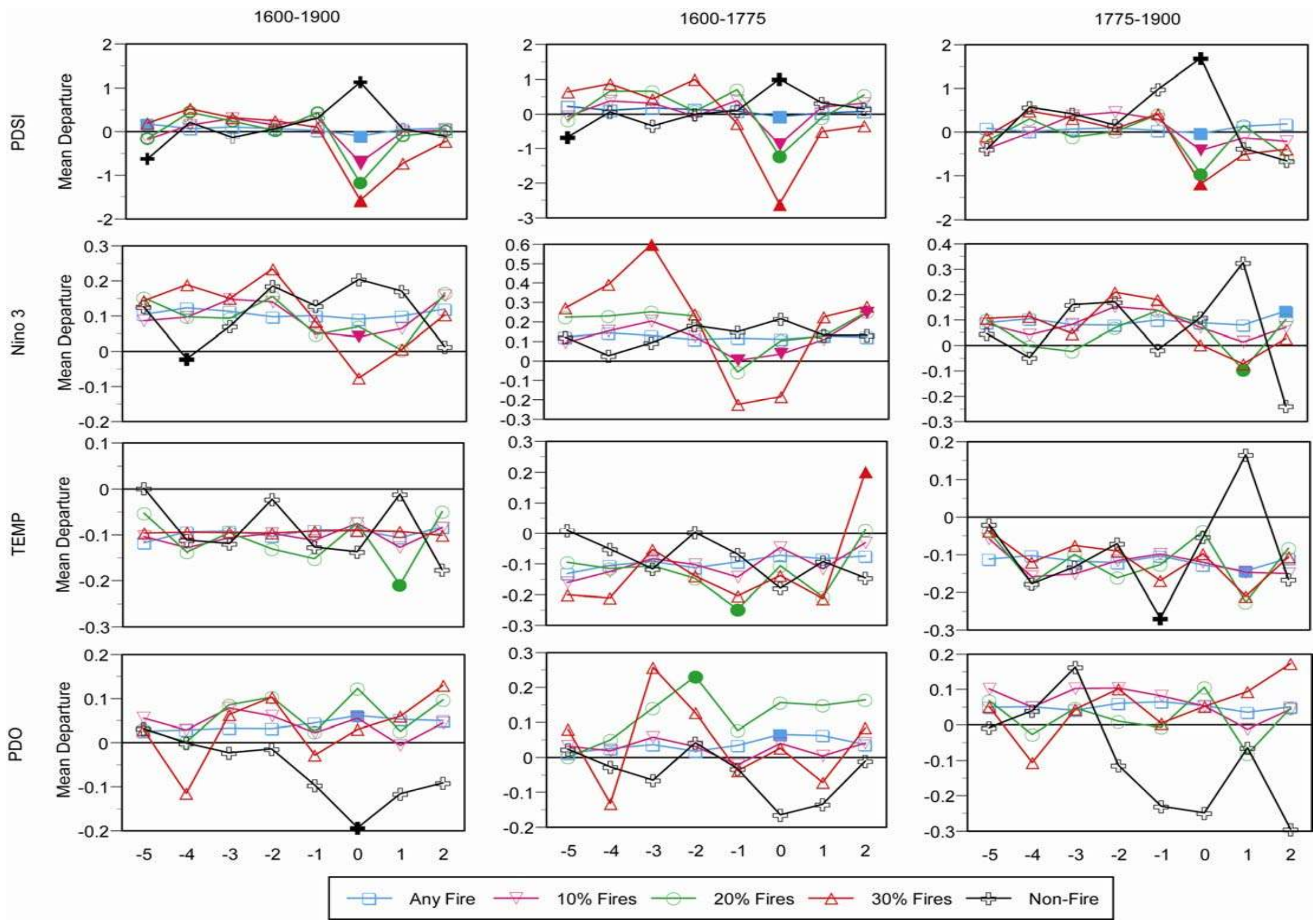
temperature (TEMP) was only significantly correlated ($p<0.05$) with current summer temperature (+) at annual time scales, while at decadal time scales, TEMP was significantly correlated ($p<0.05$) with local temperature (+) for every season, and precipitation during winter (+) and summer (-) seasons. AMO was only significantly correlated ($p<0.05$) with previous and current fall temperature (+) at annual time scales. At decadal scales AMO was significantly correlated ($p<0.05$) with temperature (+) during all seasons, and precipitation during winter (+) and summer (-) seasons, for the current and lagged year.

3.3.3 Fire-Climate Associations

3.3.3.1 Interannual Relationships

The SEA between fire years and PDSI for the entire time period demonstrates a significant negative relationship ($p<0.05$) between moisture conditions and fire events at both study sites (Figure 3.5). In addition, fire extent was related to increasingly dry conditions, with the most widespread fires associated with the driest years. On a regional scale, fire extent was also significantly related to fires for both sites combined ($p<0.01$) (Figure 3.6). In contrast, non-fire years were significantly related with wet conditions ($p<0.001$).

Figure 3.5 a-b: Next page. Superposed epoch analysis (SEA) of reconstructed PDSI, NINO3, TEMP and PDO indices with non-fire years and fire years of different extent for the periods 1600-1900, 1600-1775, and 1775-1900, for old-growth, mixed conifer forests in a) Big Oak Flat and b) South Fork Merced study areas, Yosemite National Park, CA. Values along the X-axis are years preceding and following a fire (fire year = 0). Values along the Y-axis are mean departures of climate indices from the average. Filled in symbols represent statistically significant responses ($p<0.05$).


1600-1900
1600-1775
1775-1900
PDSI
Nino 3
TEMP
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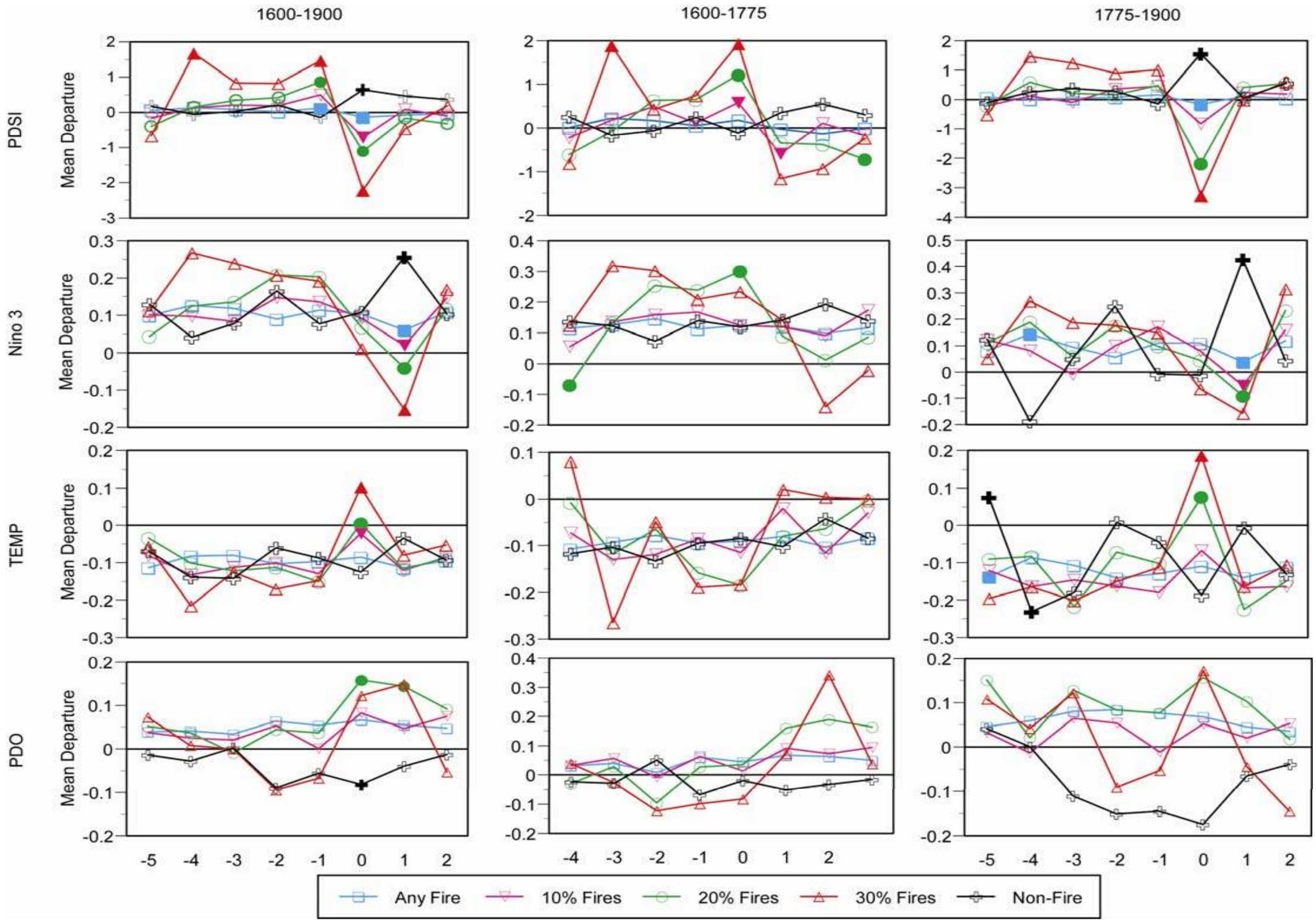
Mean Departure
Any Fire
10% Fires
20% Fires
30% Fires
Non-Fire

Figure 3.5a:



1600-1900
1600-1775
1775-1900
PDSI
Nino 3
TEMP
PDO
Mean Departure
Any Fire
10% Fires
20% Fires
30% Fires
Non-Fire

Figure 3.5b:

Figure 3.6: Next page. Superposed epoch analysis of reconstructed PDSI, NINO3, TEMP and PDO indices with non-fire years and fire years of different extent for the periods 1600-1900, 1600-1775, and 1775-1900, for old-growth, mixed conifer forests for the combined study areas in Yosemite National Park, CA. Values along the X-axis are years preceding and following a fire (fire year = 0). Values along the Y-axis are mean departures of climate indices from the average. Filled in symbols represent statistically significant responses ($p<0.05$).

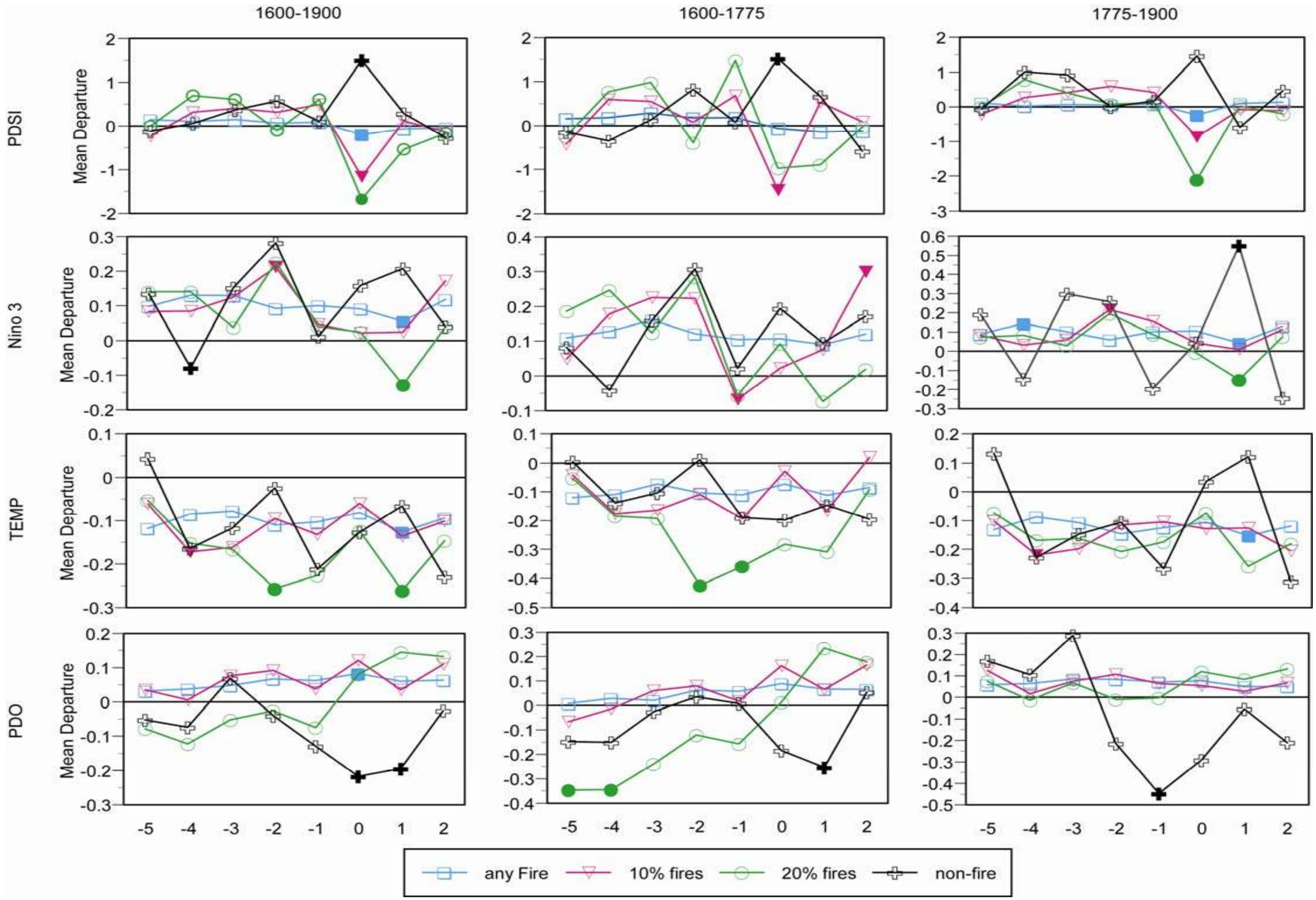

1600-1900
1600-1775
1775-1900
PDSI
Nino 3
TEMP
PDO
Mean Departure
any Fire
10% fires
20% fires
non-fire

Figure 3.6

NINO3 was significantly related to the year after a fire for SFM and the whole region ($p<0.01$), but not for BOF. There was no consistent association between PDO and fire extent (Figure 3.5 & 3.6) except for mid-sized fires in SFM ($p<0.05$), and fires of the smallest size in BOF and for both sites combined ($P<0.05$). However, non-fire years were significantly related to PDO for the entire region and both study areas. TEMP was significantly associated ($p<0.05$) with fires of all extents in SFM. In both BOF and the whole region, TEMP was associated ($p<0.05$) with the year after fire for small and mid-sized fires.

A temporal change in the relationship between PDSI and fire was also present in both study areas (Figure 3.5). Prior to 1775 there was no association between fire and dry conditions in BOF, although in SFM fire was associated with dry conditions at 100, 250, & 500 ha scales. After 1775 there was a significant relationship ($p<0.05$) between fire years and dry conditions, and between non-fire years and wet conditions, at all scales. A similar pattern was present for the whole region (Figure 3.6). A similar temporal change in the association between NINO3 and fire extent was present for SFM and both sites combined (Figure 3.5 & 3.6). Prior to 1775 there was no significant relationship, while after 1775 the year following a fire was strongly associated with NINO3 ($p<0.01$). The relationship between TEMP and fire events demonstrated a similar pattern for SFM, but not for BOF or the entire region. There was no significant relationship prior to 1775, while after 1775, TEMP was significantly associated with fires of increasing extent in SFM ($p<0.01$).

The correlation analysis identified a moderately strong association between fire extent and dry conditions ($p<0.001$) for both sites and the region (Table 3.1, Figure 3.7). The strength of the association changed over time, becoming stronger after 1800 (Figure 3.8). In addition, warm temperatures were correlated with fire extent in both sites before 1775 ($p<0.001$), but only SFM after 1775 ($p<0.05$). The analysis also identified a slow decline in the strength of the correlation between fire and temperature from 1775 to 1825, and no correlation after 1825 (Figure 3.8). The relationship between PDO and fire extent was more variable. Prior to 1775, PDO was correlated with fire extent for the entire region, while after 1800 the only significant relationship was with fire extent in SFM. The correlation analysis between fire extent and PDO was positive from 1700-1900, but was only statistically significant during the periods 1720 to 1770 and 1820 to 1860 (Figure 3.8). Last, NINO3 was significantly correlated with fire extent in the region for the entire record, but was only correlated for the period after 1775 when both halves of the record were compared separately.

Table 3.1: Pearson product moment correlation coefficients for reconstructed time series of fire extent and climate for the period 1600-1900, and subsets of the period, for Big Oak Flat (BOF), South Fork Merced (SFM) and both sites combined (YOSE) in old-growth, mixed conifer forests in Yosemite National Park, CA. The climatic variables are Pacific Decadal Oscillation (PDO), Palmer Drought Severity Index (PDSI), Atlantic Multidecadal Oscillation (AMO), El Niño (NINO3), and summer temperature (TEMP). ENSO values are for the fire year +1. Statistically significant correlations are marked by * (p<0.05), ** (p<0.01), and *** (p<0.001).

Fire Extent (% GP)	PDO	AMO	NINO3	PDSI	TEMP
BOF					
1600-1900	0.096	0.0165	-0.082	-0.267***	0.0653
1600-1775	0.123	0.081	-0.039	-0.346***	0.199**
1775-1900	0.107	0.005	-0.118	-0.276**	0.017
SFM					
1600-1900	0.173**	-0.005	-0.181**	-0.393***	0.255***
1600-1775	0.206**	0.009	-0.102	-0.305***	0.253***
1775-1900	0.177*	-0.002	-0.249**	-0.489***	0.295***
YOSE					
1600-1900	0.164**	0.007	-0.161**	-0.404***	0.195***
1600-1775	0.209**	0.052	-0.091	-0.405***	0.285***
1775-1900	0.171	0.002	-0.221*	-0.461***	0.183*

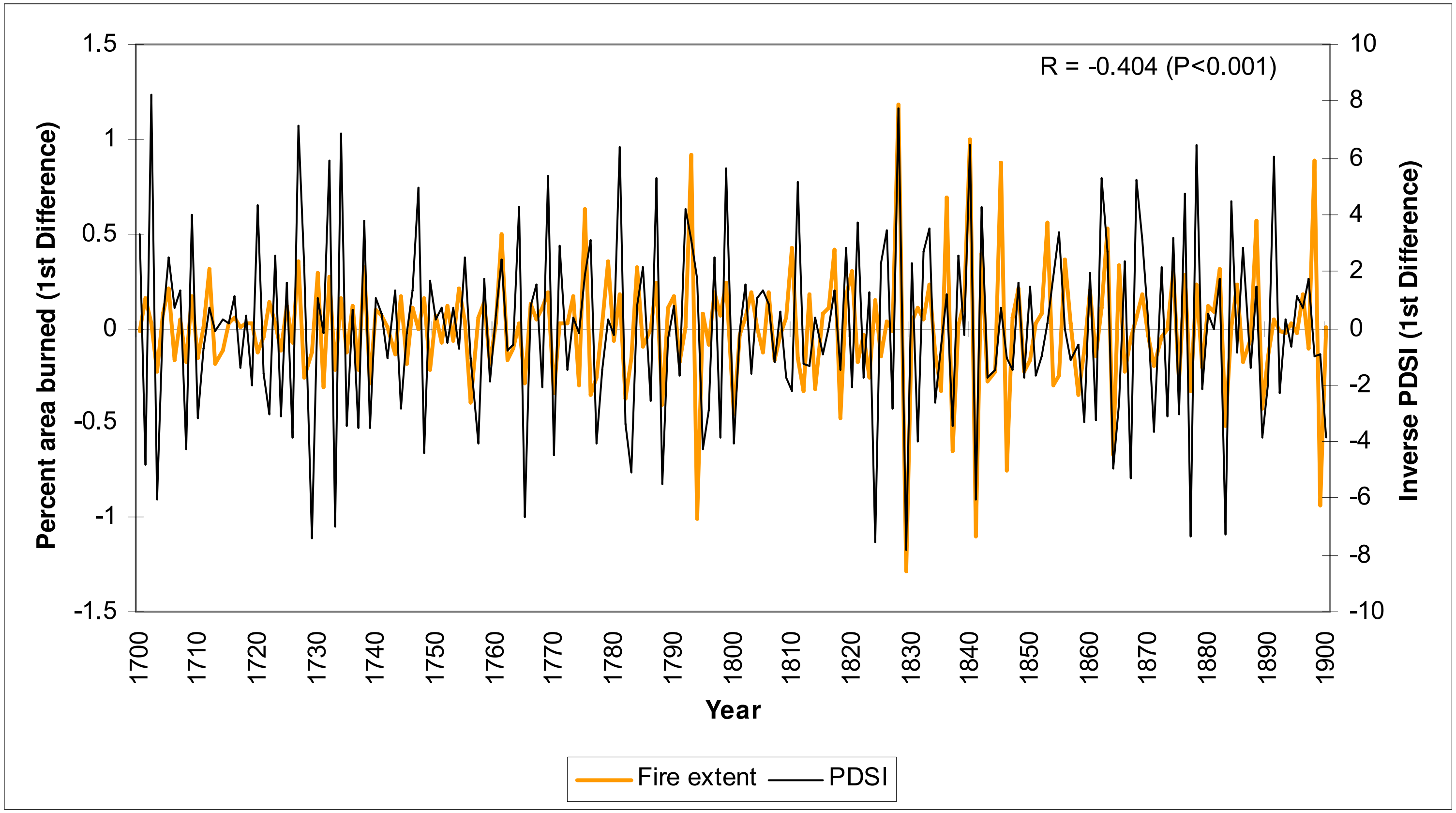


Figure 3.7: Pearson product moment correlation coefficients of first differences for PDSI and regional fire extent for the combined study areas in old-growth, mixed conifer forests in Yosemite National Park, CA. PDSI values were inverted for presentation. Pearson product moment correlation of first differences reported in table.

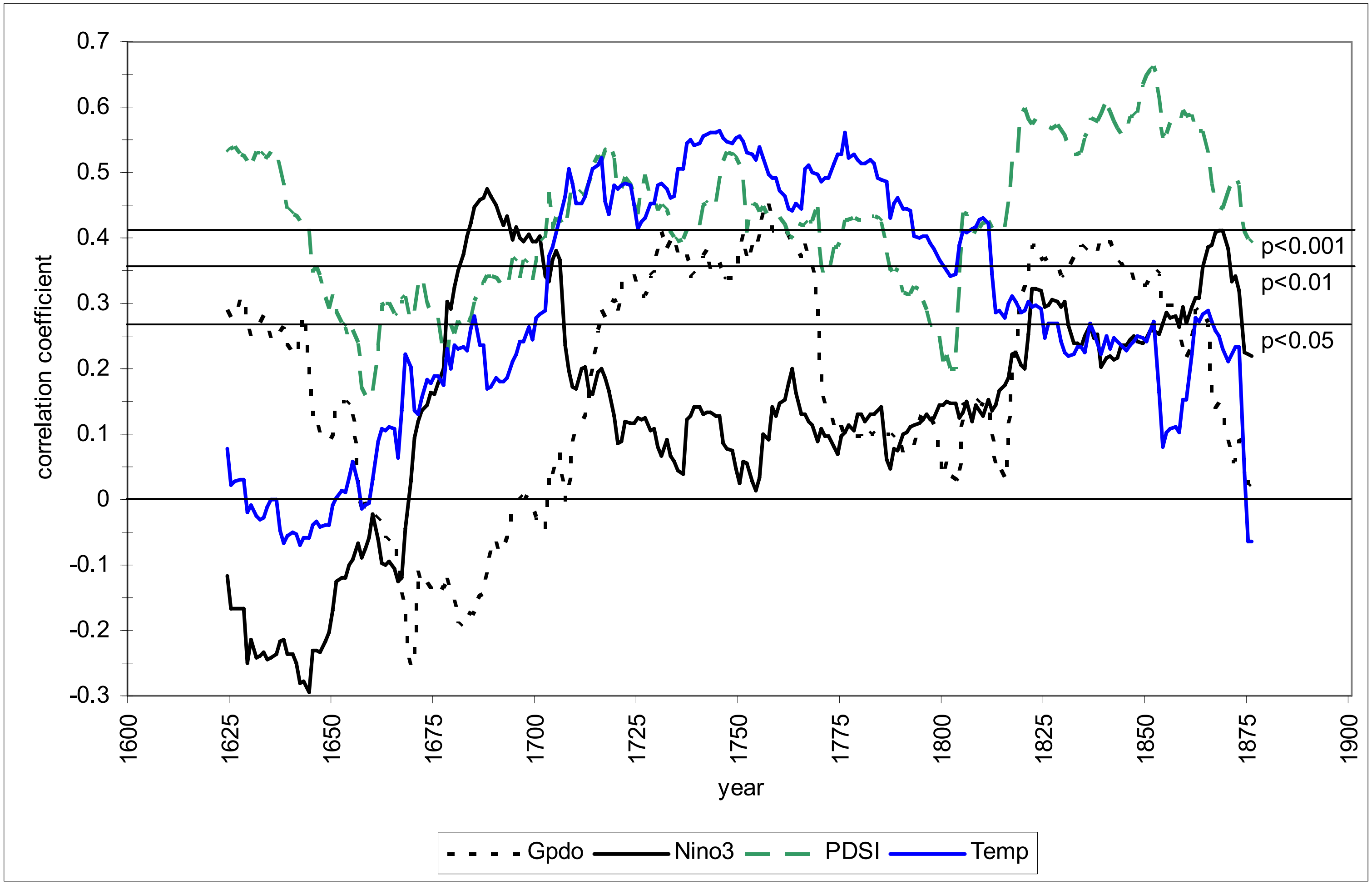


Figure 3.8: Forty-nine year running correlation of first differences, plotted on 25th year of the period, between PDSI, summer temperature (TEMP), NINO3, PDO And fire extent in old-growth, mixed conifer forests in Yosemite National Park, CA. Values for PDSI and NINO are inverted for presentation. NINO values for correlation are for year of fire +1 (year following fire.)

3.3.3.2 Interdecadal Relationships

Fire extent was only significantly correlated with the AMO on a decadal time scale (Figure 3.9). The AMO was negatively correlated with fire extent for the period from 1600-1700 ($r = -0.821$, $p<0.01$) and 1800-1900 ($r = -0.785$, $p<0.01$), but not for the period in between, from 1700-1800. There was no significant correlation between fire extent and the other climate variables at the decadal scale.

Several climate variables were significantly inter-correlated at the annual scale (Table 3.2). TEMP was significantly correlated with PDO ($r = 0.251$, $p<0.001$), NINO3 ($r = 0.256$, $p<0.001$), and PDSI ($r = -0.356$, $p<0.001$) for the entire period. All variables were inter-correlated for the periods before and after 1800, except for PDSI. PDSI was only correlated with TEMP ($r = -0.387$, $p<0.001$) for the period from 1800-1900. Inter-correlation of climate variables was limited at the interdecadal scale (Table 3.2). TEMP was significantly correlated with PDSI ($r=-0.399$, $p<0.05$) and PDO ($r=0.401$, $p<0.05$) for the whole period, and with PDO ($r=0.662$, $p<0.05$) from 1600-1700. NINO3 was correlated PDO ($r=0.476$, $p<0.01$) and PDSI ($r=0.497$, $p<0.01$) for the entire period and with PDSI ($r=0.754$, $p<0.05$) from 1700-1800. AMO was not significantly correlated with any other climate variables.

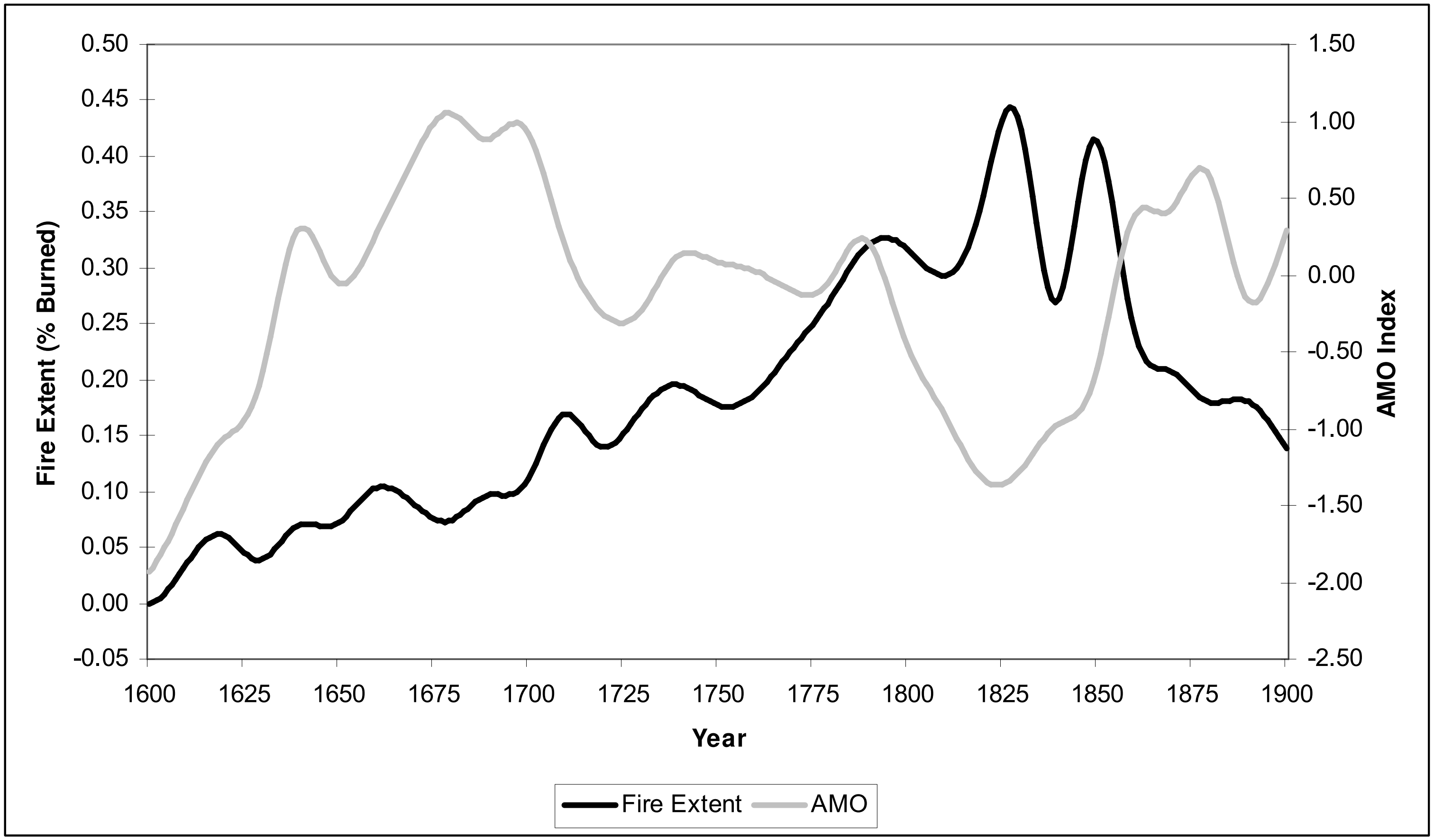


Figure 3.9: Interdecadal variation in AMO and regional fire extent in old-growth, mixed conifer forests in Yosemite National Park, CA. Series were computed from 10 year non-overlapping means (sums in the case of fire extent) and were smoothed using a cubic spline that retained 50% of the variation of the original data for presentation.

Table 3.2: Pearson product moment correlation coefficients for reconstructed time series of correlations between different climatic variables for the period 1600-1900, and subsets of that period. The climatic variables are the Pacific Decadal Oscillation (PDO), Palmer Drought Severity Index (PDSI), Atlantic Multidecadal Oscillation (AMO), El Niño-Southern Oscillation (NINO3), and summer temperature (TEMP). Statistically significant correlations are marked by * (p<0.05), ** (p<0.01), and *** (p<0.001).

	Interannual				Interdecadal			
PDO	AMO	Nino3	PDSI	Temp	AMO	Nino3	PDSI	Temp
1600-1700	0.119	0.124	-0.079	0.101	0.143	0.559	0.062	0.662*
1700-1800	0.035	0.061	-0.181	0.262**	-0.483	0.362	0.385	0.09
1800-1900	-0.008	0.257**	-0.065	0.231*	-0.129	0.499	-0.185	0.479
1600-1900	0.044	0.153**	-0.106	0.199***	-0.010	0.467**	0.100	0.401*
1700-1900	0.009	0.167*	-0.122	0.251***				

AMO	PDO	Nino3	PDSI	Temp	PDO	Nino3	PDSI	Temp
1600-1700	0.119	0.039	-0.068	0.095	0.143	-0.187	-0.184	0.251
1700-1800	0.035	0.014	-0.053	-0.022	-0.483	-0.612	-0.037	-0.479
1800-1900	-0.008	0.006	-0.016	0.065	-0.129	-0.447	0.102	0.542
1600-1900	0.044	0.015	-0.041	0.046	-0.010	-0.261	-0.047	0.279
1700-1900	0.009	0.002	-0.026	0.023				

Nino3	AMO	PDO	PDSI	Temp	AMO	PDO	PDSI	Temp
1600-1700	0.039	0.124	0.299**	0.172	-0.187	0.559	0.264	0.137
1700-1800	0.014	0.061	0.277**	0.163	-0.612	0.362	0.754*	0.038
1800-1900	0.006	0.257**	0.035	0.372***	-0.447	0.499	0.527	-0.057
1600-1900	0.015	0.153**	0.210***	0.227***	-0.261	0.467**	0.497**	0.071
1700-1900	0.002	0.167*	0.171*	0.256***				

PDSI	AMO	PDO	Nino3	Temp	AMO	PDO	Nino3	Temp
1600-1700	-0.068	-0.079	0.299**	-0.103	-0.184	0.062	0.264	-0.554
1700-1800	-0.053	-0.181	0.277**	-0.342***	-0.037	0.385	0.754*	-0.266
1800-1900	-0.016	-0.065	0.035	-0.387***	0.102	-0.185	0.527	-0.346
1600-1900	-0.041	-0.106	0.210***	-0.269***	-0.047	0.100	0.497**	-0.399*
1700-1900	-0.026	-0.122	0.171*	-0.356***				

Temp	AMO	PDO	Nino3	PDSI	AMO	PDO	Nino3	PDSI
1600-1700	0.095	0.101	0.172	-0.103	0.251	0.662*	0.137	-0.554
1700-1800	-0.022	0.262**	0.163	-0.342***	-0.479	0.090	0.038	-0.266
1800-1900	0.065	0.231*	0.372***	-0.387***	0.542	0.479	-0.057	-0.346
1600-1900	0.046	0.199***	0.227***	-0.269***	0.279	0.401*	0.071	-0.399*
1700-1900	0.023	0.251***	0.256***	-0.356***				

3.3.3.3 Contingency Analysis

Several different phase combinations of AMO, PDO and Nino3 were associated with the occurrence of widespread fire events in Yosemite National Park (Figure 3.10). Although fires occurred during all phase combinations of AMO, PDO and NINO3, the distribution of all 280 fires was not randomly distributed across all combinations (χ^2=20.4, p<0.01). The greatest percentage (19.3%) of fires occurred during a -AMO/+PDO/+NINO3 combination. Fires that did not occur during the -AMO/+PDO/+NINO3 combination were randomly distributed among the other possible phase combinations. The only paired combination in which more fires than expected occurred was for +PDO/+NINO3 (χ^2=19.2, p<0.001). Different combinations of climate phase combinations were also related to the occurrence of the largest 10% (n=28) of the fires that occurred from 1600-1900. The distribution of the largest fires during all phase combinations of AMO, PDO, and NINO3 were significantly different than expected (χ^2=35.6, p<0.001). More fires than expected (25%) occurred during a combination of -AMO/+PDO/-NINO3, and fewer fires than expected (0%) occurred during a combination of +AMO/-PDO/+NINO3. The distribution of fires for different paired phase combinations was also significantly different than expected for AMO-PDO (χ^2=16.2, p<0.001), AMO-NINO3 (χ^2=17.0, p<0.001), and PDO-NINO3 (χ^2=18.0, p<0.001). More fires than expected occurred during the following phase combinations: -AMO/+PDO (46.4%); -AMO/-NINO3 (32.1%); and +PDO/-NINO3 (35.7%). Fewer fires than expected occurred during +AMO/-PDO (10.7%), +AMO/+NINO3 (14.3%), and -PDO/+NINO3 (10.7%) phase combinations.

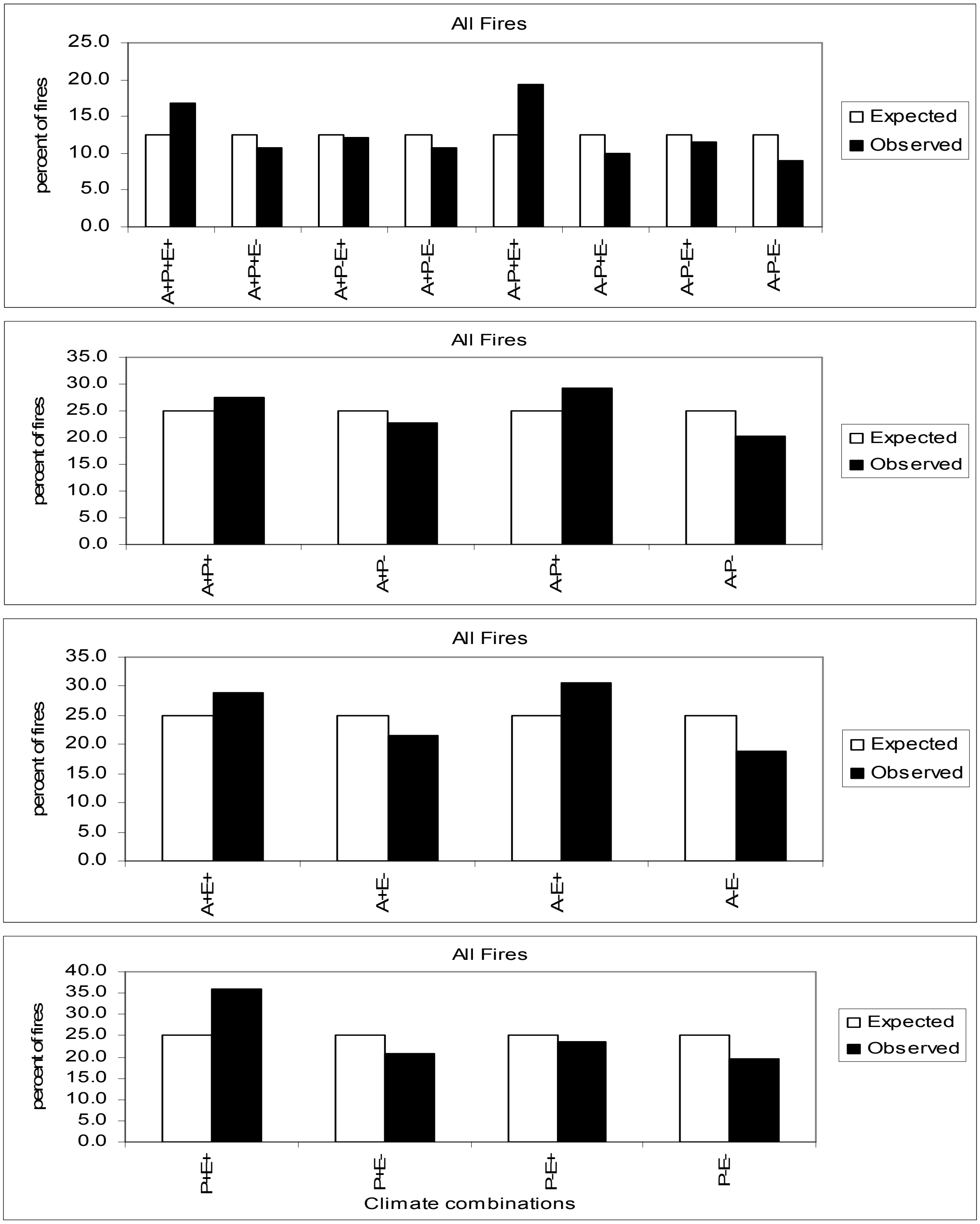


Figure 3.10: Page 1 of 2. The frequency (%) of expected and observed occurrences of all fire events and the largest 10% of fire events for all possible climate pattern phase combinations of the Atlantic Multidecadal Oscillation (AMO), Pacific decadal Oscillation (PDO) and El Niño-Southern Oscillation (NINO3) in old-growth, mixed conifer forests in Yosemite National Park, CA. Abbreviations for climate patterns are AMO (A), PDO (P), NINO3 (E), and positive (+) and negative (-) values. Statistically significant deviations from expected occurrences are marked by * (p<0.05) and ** (p<0.01).

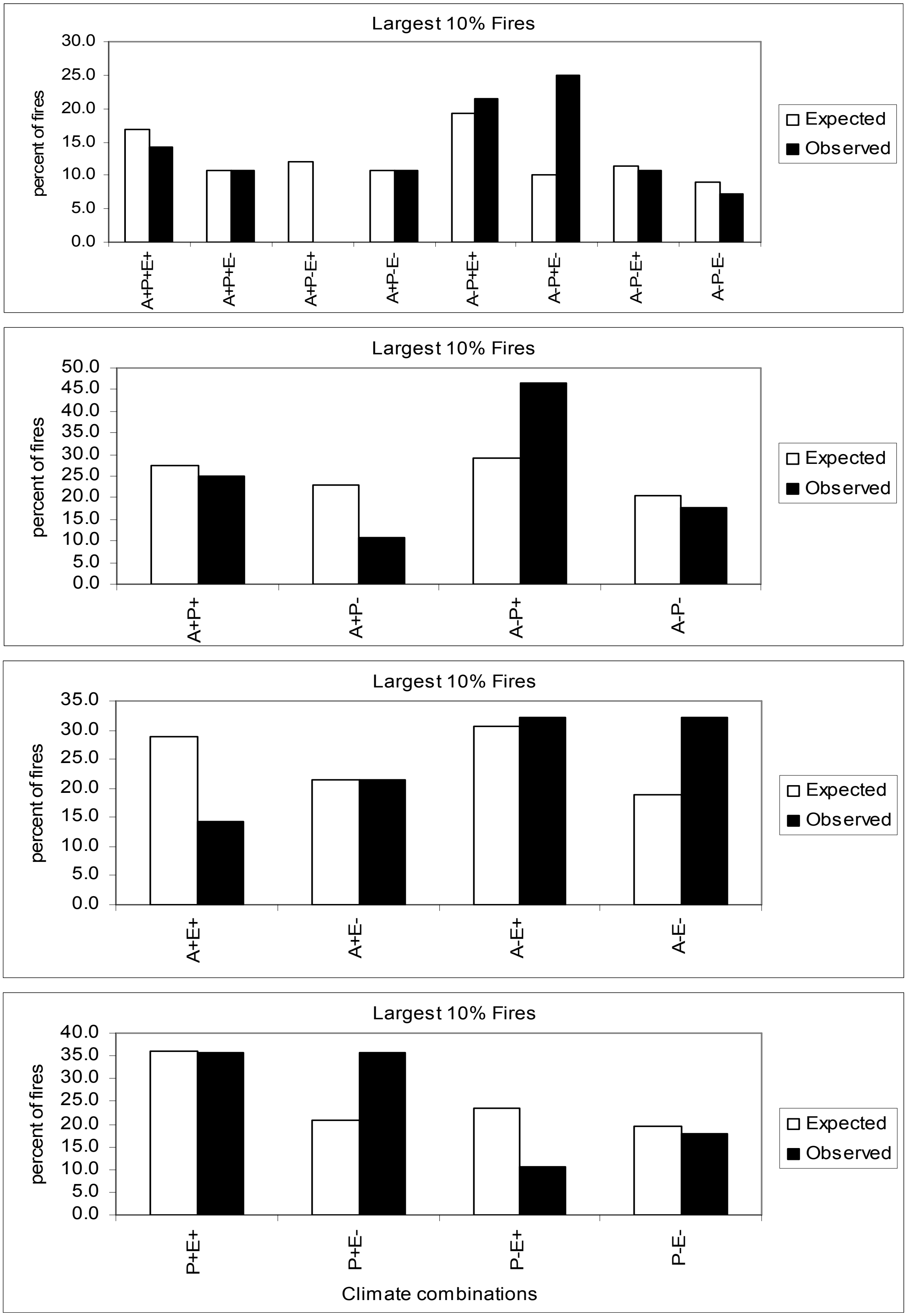


Figure 3.10: Page 2 of 2.

3.4 DISCUSSION

Some strong relationships between climate variability and fire occurrence and extent were identified in the mixed conifer forests of the central Sierra Nevada at both interannual and interdecadal scales. Interannually, fires of all sizes were related to years of low moisture availability, while non-fire years were related to years that were wetter than normal. Interdecadally, the timing and extent of fires were influenced by decadal scale variations in tropical, north Pacific and north Atlantic teleconnections, and their mutual interactions. Long term trends in fire occurrence and extent were also related to long term variability in climate regimes, which are dominated by different teleconnections.

The relationship between drought conditions and widespread burning is well documented in conifer forests of the western United States (Swetnam and Betancourt 1990, Swetnam 1993, Veblen et al. 2000, Norman and Taylor 2003, Swetnam and Baisan 2003, Hessl et al. 2004, Taylor and Beaty 2005). The overall pattern of extreme drought years associated with more widespread fires is generally present in the mixed conifer forests of Yosemite National Park. All fires were associated with years of low soil moisture in both study areas and for the entire region, and more widespread fires were associated with increasingly drier conditions, indicating that drier fuels enable fires to burn across larger areas on the landscape. In addition, non-fire years were associated with conditions that are wetter than normal in both study areas. However, the relationship between non-fire years and wetter soil moisture conditions was not significant for the entire region. This may be due to the extremely high frequencies of fire in both study areas. While there was a high coincidence of fires between both study areas (X^2 = 194.5, p<0.001), there were also large numbers of small fires that only occurred in one of the two study areas, even in years with above normal moisture conditions. Consequently, there were relatively few years (n=6) when no fire occurred in either study area, which may have contributed to the lack of association with wet years.

The influence of antecedent wetter than normal years on widespread fires is highly regional in western conifer forests. In the Southwestern U.S., there is a consistent trend of wetter than normal conditions 2-4 years preceding widespread fires (Brown et al. 2001, Swetnam and Baisan 2003). A similar temporal pattern is present in the Sierra San Pedro Martir (SSPM) in Mexico (Stephens et al. 2003) and the Rocky Mountains (Veblen et al. 2000, Donnegan et al. 2001). The explanation for this pattern is that antecedent wet years result in increased production of fine fuels resulting in high levels of fuels at the time of fire, which increases the connectivity of fuels on the forest floor. When subsequent fires occur in dry years, the result is more widespread fires. In contrast, forests in the Pacific Northwest only demonstrate a contemporaneous relationship between widespread burning and drought; antecedent moisture conditions are not associated with years of fire (Heyerdahl et al. 2002, Hessl et al. 2004). However, this temporal pattern is less clear in the northern Sierra Nevada and southern Cascades where several studies have found antecedent wet years to be associated with fire years (Norman and Taylor 2003, Taylor and Beaty 2005), while other studies found no relationship (Moody et al. 2006, Taylor et al. 2008). In Yosemite National Park the pattern is mixed with a relationship only present in SFM for the year prior to a fire. This may be the result of the strongly contrasting slopes in the study area, thereby making the production of fine fuels necessary to increase the connectivity across the drainage. Consequently, widespread fires in Yosemite National Park are associated primarily with drought conditions in the year of fire and not fine fuel production in proceeding years. A similar temporal pattern was found for giant sequoia (Sequoiadendron giganteum (Lindl.) Buchholz) mixed conifer forests in the central Sierra Nevada (Swetnam 1993). The similarly high fire frequencies between the study sites, small fire sizes and lack of association with preceding wet years (Swetnam 1993, Caprio and Swetnam 1995, Swetnam et al. 2000), suggest that fires in the central Sierra Nevada occur with sufficient frequency that not all fuels may be consumed during a given fire. The evidence of trees with fire scars at intervals of two years supports this view. Consequently, when drought conditions occur, the additional drying results in increased connectivity between the existing fuels, regardless of antecedent wet years, allowing for widespread burning to occur.

Interannual climate variability related to ENSO has been identified as a significant influence on fire extent in forests of the southwestern U.S. (Swetnam and Betancourt 1990, Grissino-Mayer and Swetnam 2000) and the Pacific Northwest (Heyerdahl et al. 2002). However, no relationship between ENSO and fires has been identified in the southern Cascades (Norman and Taylor 2003) or the northern Sierra Nevada (Taylor and Beaty 2005, Taylor et al. 2008). In

Yosemite National Park, the pattern is different, where fire is significantly related to ENSO the year following the fire. A pattern similar to what was found in one study in the northern Sierra Nevada (Moody et al. 2006). This pattern may be the result of the dipole present in precipitation patterns between the northern and southern areas of the western U.S. that is related to ENSO phases (Dettinger et al. 1998), and consequently, to widespread fire patterns (Kitzberger et al. 2007). Positive ENSO (i.e. El Niño) produces wet conditions in the Southwest, while the Pacific Northwest experiences drier than normal conditions at these times. The hinge point between these two poles appears to be in northern California between 40-45º north latitude where the response is much more variable. In Yosemite National Park, the correlation of fires with ENSO values for the year following the fire suggests that the shift in ENSO from a positive (wet) to negative (dry) phase is related to the onset of drier conditions, and consequently more widespread fire events. The large variability in fire-ENSO patterns found in the northern Sierra Nevada may be the result of a weaker influence from ENSO, and consequently other teleconnections (e.g. PDO) having a stronger influence on precipitation and subsequently the occurrence of fires.

In northern California the relationship between fire and ENSO, and fire and PDO is not consistent in mixed conifer forests. Interannual variation in fire extent in the Lake Tahoe Basin (Taylor 2004) and southern Cascades (Norman and Taylor 2003, Taylor et al. 2008) was unrelated to ENSO, but was significantly correlated with a negative PDO (dry phase), whereas in Yosemite National Park, there was no PDO signal present in the fire data. This suggests that in Yosemite National Park, the climatic influence of ENSO is stronger than that of the PDO, and consequently ENSO is the driver of the prevailing interannual precipitation patterns. However, in the Northern Sierra Nevada, the opposite appears to be true. The PDO appears to have a stronger influence on precipitation and, consequently, burning patterns, than ENSO. The influence of ENSO on precipitation in the Yosemite region is further demonstrated by the significant correlation between PDSI and ENSO over the entire record ($r = 0.21$, $p < 0.001$), compared to the relationship between PDSI and PDO ($p > 0.05$).

The influence of interannual climate variations on fire extent fluctuated over time. There is a pronounced increase in correlation between PDSI, PDO, ENSO and fire extent from 1815 to 1870. This pattern coincides with the increase in widespread fires (and average fire extent) that occurred from 1800 to 1900 in both study areas. In addition, the negative relationship between moisture levels and fire extent is significant for the entire period. This pattern is different than that found in the northern Sierra Nevada (Taylor 2004), southern Cascades (Taylor et al. 2008), and the southwestern U.S. (Grissino-Mayer and Swetnam 2000), where the relationship between PDSI and fire changed from the 1700-1800 period to the 1800-1900 period. In the northern Sierra Nevada (Taylor 2004) and southern Cascades (Taylor et al. 2008) PDSI and fire were correlated from 1800-1900 but not 1700-1800. The opposite pattern was present in the southwestern U.S. (Grissino-Mayer and Swetnam 2000) where the relationship was present prior to 1800, but not after. The coincident shift in the correlation between fire extent and PDSI during the 1800-1900 period in the southern

Cascades, Sierra Nevada and southwestern U.S. suggests that the shift is related to climate factors affecting a much broader area.

The increase in correlation between PDSI and fire extent in 1810 coincides with a shift in the correlation pattern between ENSO and fire extent. The relationship between fire extent and ENSO for the year following a fire is negative for the entire time period, but only becomes significant between 1820 and 1875. The SEA analysis shows a similar pattern, with a significant relationship present for the second half of the record but not the first half. In the northern Sierra Nevada and southern Cascades, the correlation pattern between ENSO and fire extent is the opposite, with a significant relationship from 1700-1800, but not from 1800-1900 (Taylor and Beaty 2005, Taylor et al. 2008). The temporal pattern found in Yosemite National Park is similar to that in the Southwest, except that while the relationship between fire extent and dry conditions gets stronger during the 19th century in Yosemite National Park, it weakens or disappears entirely in the southwest (Grissino-Mayer and Swetnam 2000, Swetnam and Baisan 2003). The temporal variation in correlation between ENSO and fire extent in the Southern Cascades, Sierra Nevada, and southwestern U.S. indicates that other factors may be influencing the relative effect of ENSO in the different locations.

A shift in fire climate relationships beginning around 1800 has been identified in locations comprising a broad region in the western United States including the southern Cascades (Taylor et al. 2008), northern Sierra Nevada (Taylor and Beaty 2005), Rocky Mountains (Veblen et al. 2000), Pacific

Northwest (Heyerdahl et al. 2002), and Argentina (Kitzberger et al. 2001), suggesting that the climatic shift occurring was global in scale, but with different effects on regional fire-climate patterns. In the southwestern U.S., and Argentina, the beginning of the 19th century was a period of few fires, but increased synchrony among fires, while in the Rocky Mountains and Pacific Northwest the same period was characterized by a decrease in fire extent and fire frequency. In the southern Cascades and northern Sierra Nevada, the frequency of fires declined during this period while the fire extent remained the same. The increase in the strength of the relationship between dry conditions and fire extent during this period suggests an increase in fire synchrony. The same period (1800-1900) in Yosemite National Park was characterized by an increase in fire extent without a change fire frequency, which coincided with a strengthening of the relationship between fire extent and dry conditions. These shifts in the relationship between PDSI and ENSO with fire extent may be related to a shift in the PDO to a positive phase during the 1810-1870 period, given that the PDO is known to influence ENSO and its effects (Gershunov et al. 1999, McCabe and Dettinger 1999).

In Yosemite National Park, from 1815-1875, ENSO and PDSI were negatively associated with fire extent, while PDO was positively associated with fire extent. This differs from the pattern found in the southern Cascades (Taylor et al. 2008), where the association between ENSO and fire extent weakened during the same period. In addition, the association between temperature and fire extent increases in the southern Cascades during this period, while in

Yosemite National Park, the relationship weakens and eventually disappears. The weakening in the association between temperature and fire extent during the 19th century indicates that summer temperatures became less important for the occurrence of widespread fires, and instead the interannual variability in precipitation associated with the shift from positive to negative phase of ENSO became more important. This pattern is opposite to one found in the southern Cascades, where the association between fire extent and temperature is more significant than the association between fire extent and ENSO.

The temporal pattern in Yosemite National Park suggests that the positive PDO modulates the influence of ENSO by accentuating the shift from wet to dry conditions when ENSO changes from positive to negative phases. The result can be a more pronounced drying effect, and consequently more widespread fires in the region. Likewise, the absence of any association between ENSO and fire extent during the 1700-1800 period seems to reflect the loss of correlation between PDO and ENSO, which resulted in a decline in the impact of ENSO on fires.

Long-term patterns in fire extent were only associated with interdecadal variations in the AMO. Decades in which the AMO was in a negative phase were associated with widespread fires and vice versa. The decadal scale relationship between AMO and fire extent may reflect a long term influence of AMO on precipitation in the region, and consequently long term trends in fuel production. AMO has been strongly correlated with precipitation patterns across the coterminous United States (Enfield et al. 2001, McCabe et al. 2004), however the mechanisms behind the connection are still poorly understood. A dipole similar to that found with ENSO exists with AMO and precipitation between the Southwest and the Pacific Northwest. While positive AMO is significantly correlated with drought in the central and southwestern U.S. (McCabe et al. 2004), positive AMO is negatively correlated with drought in the Pacific Northwest. However, the relationship between AMO and precipitation (and subsequently drought occurrence) is less clear in the Sierra Nevada, where the correlation breaks down between the two poles and is very weak to non-existent.

The relationship between AMO and fire in western forests is just beginning to be identified (Brown 2006, Guyette et al. 2006, Sibold and Veblen 2006, Kitzberger et al. 2007) and is further complicated by the interactions among AMO, ENSO and PDO. Associations between different phases of the three teleconnections and fire occurrence and extent have been identified in the Rocky Mountains (Sibold and Veblen 2006), Black Hills (Brown 2006), Southwest and Pacific Northwest (Kitzberger et al. 2007). In the Rocky Mountains, Southwest, Black Hills and Pacific Northwest regions, high fire synchrony is associated with +AMO/-PDO/-ENSO. This pattern is the result of the influence of the AMO on ENSO, and consequently precipitation in the region (Enfield et al. 2001, McCabe et al. 2004). A positive AMO strengthens the teleconnection to ENSO in the central and southwestern United States. In the Sierra Nevada, the opposite interaction occurs when the AMO is in the positive phase and the teleconnection weakens the influence of ENSO. In Yosemite National Park, fire occurrence and extent are related to combinations of -AMO/+PDO/+ENSO. The AMO was in a negative phase from 1775 – 1850; the same period having a high correlation between the interannual variability of ENSO and fire extent. Prior to 1775, the AMO was in a positive phase, when it weakened the impact of ENSO in the Sierra Nevada, and consequently, there was no relationship between ENSO and fire extent. The long term frequency of the AMO appears to influence the relative strength of the ENSO signal on precipitation patterns and drought in the Sierra Nevada, and consequently, on the extent and occurrence of fires.

3.4.1 Conclusion

This study demonstrates that climate variations at both interannual and decadal scales influence fire regimes in Yosemite National Park. Interannual variability in regional climate conditions was strongly related to the extent and occurrence of fires in the park. However, the relationship was not stable over time. Interdecadal and multidecadal fluctuations in teleconnections influenced the long term trend in fires. Similar relationships between fire and climate in the western U.S. and South America also suggest that there are cross-hemispheric connections in these climate patterns. In addition, the changes in fire regimes associated with changes in climate conditions over time, suggests that future climatic variability may influence fire patterns in the western United States. Understanding this natural range of variability is necessary in trying to understand potential changes in fire regimes, forest structure and composition, and potential carbon sequestration that may occur in the future due to changing climate conditions.

Chapter 4

Changes in Carbon Storage over 100 years of Fire Suppression in Mixed Conifer Forests of Yosemite National Park

4.1 INTRODUCTION

Carbon dynamics in forested ecosystems have been identified as an important component in the global carbon cycle (Dixon et al. 1994, Grace 2005), and whether an individual forested area is a net source or sink of terrestrial carbon is related to several factors, including growth and regeneration dynamics (Hicke et al. 2004, Tremblay et al. 2006), the occurrence of disturbance events (Acker et al. 2000, Wang et al. 2003), human activities (Harmon et al. 1990, Houghton et al. 1999, Casperson et al. 2000), and climate variability (Hollinger et al. 1999, Thornton et al. 2002). Carbon uptake in forested ecosystems is related to the growth of vegetation and production of biomass, while carbon loss is related to the removal of biomass from the system. Consequently, it is necessary to understand how different processes in a forest influence the amount of biomass present in the system, and how those processes and the carbon balance changes over time (Cohen et al. 1996, Law et al. 2004, Morgenstern et al. 2004, Dore et al. 2008).

One common approach to studying how discrete events influence changes in carbon storage in forested systems has been to measure the biomass in a system at two discrete points in time, convert the biomass to carbon content and determine the difference between the two points (Turner et al. 2000).

Although this approach has been used to identify both the amount of carbon loss due to known events such as logging (Harmon et al. 1990) and fire (Harden et al. 2000, Tilman et al. 2000), and the amount of carbon uptake due to post-logging regeneration (Martin et al. 2005), old field succession (Johnston et al. 1996, Tremblay et al. 2006), and post-fire regeneration (Wang et al. 2003, Rothstein et al. 2004), it is only useful when there are data for the system before the event occurred. Consequently, the majority of these studies involve human caused events occurring in the past 50 years, and not natural occurrences such as wildfire, or vegetation change due to historical land-use changes.

Another approach has been to study the regeneration dynamics of forests in their current state and use that to model the biomass and also carbon flux dynamics of the system (Grier and Logan 1977, Gholz 1982, Graumlich et al. 1989, Kurz and Apps 1999, Smithwick et al. 2002). However, the majority of these studies have occurred in old-growth forests in which widespread disturbance events such as fire were rare and infrequent. In addition, whereas the regeneration dynamics of undisturbed old-growth forests can result in no significant net change in carbon storage over time (Graumlich et al. 1989), the occurrence of stochastic events such as wildfire can result in a net loss of biomass and carbon immediately following the fire (Tilman et al. 2000), followed by a net increase in biomass and carbon due to forest re-growth after the event (Rothstein et al. 2004). These fluctuations in vegetation biomass can vary greatly depending on the frequency and severity of the fires that the landscape experiences (Agee 1993, Chang 1996) and the forest system in which they occur

(Cohen et al. 1996, Law et al. 2001, Kashian et al. 2006). Many of the models used to model carbon fluctuations at continental levels are based on data derived from both of these approaches to measure carbon fluctuation in forested ecosystems (Birdsey 1992, Kauppi et al. 1992, Turner et al. 1995, Houghton et al. 1999, Casperson et al. 2000, Houghton and Hackler 2000). However, one of the problems with these studies is that they are based on relatively few studies containing information on the historical landscape and changes that have occurred, especially in regards to woody encroachment and infilling that has resulted from fire suppression (Houghton et al. 1999, Houghton 2003, Schimel 2004). Consequently, estimates are used for both the characteristics (structure and composition) of the historic forests, and the stating date from which the carbon dynamics are modeled.

Woody encroachment and infilling are two processes whereby tree density in a region increases, usually in response to the removal of a disturbance such as fire (Parsons and DeBenedetti 1979, Taylor 2000), or a change in growing conditions related to climatic variation (Vale 1981, Taylor 1995, Swetnam and Betancourt 1998). Woody encroachment refers to the invasion of woody plants into meadows and grasslands (Archer et al. 2001), whereas infilling refers to the increase in density of trees in an existing forest (Covington and Moore 1994). In the western United States both processes have been occurring since the implementation of a policy of fire suppression by the U.S. forest Service at the beginning of the 20th century (Pyne 1982). However, it is difficult to quantify exactly how much change has occurred in these systems, because there are few, if any, detailed inventories of forest data from the start of the fire exclusion period. In addition, many of the existing forests were also logged since that time. Consequently, there are few forests in fire-prone ecosystems with old-growth trees present, making it difficult to determine how much of the increase in carbon storage is due to fire suppression, and how much is due to natural regeneration following logging.

The majority of studies on carbon change in forested ecosystems in the past 100 years involve making estimates concerning the amount of change in biomass that has occurred due to fire suppression (Houghton 2003). One continental scale estimate of the carbon budget for the U.S. (Houghton et al. 2000) attributed a carbon sink of 0.24 Pg C/year (34% - 80% of the total carbon sink of 0.3-0.7 Pg C/year (Pacala et al. 2001)) to fire suppression. Pacala et al. (2001) estimated that about half of that sink was due to woody encroachment and half from infilling of existing forests. However, the limited number of studies of both processes suggests that the actual amount of encroachment and infilling is uncertain (Houghton et al. 1999, Tilman et al. 2000, Pacala et al. 2001).

This project attempts to calculate a more accurate estimate of biomass change due to fire suppression in a mixed conifer forest in the Sierra Nevada through a detailed reconstruction of the pre-fire suppression forest, including the incorporation of trees currently dead that were alive prior to fire suppression. In addition, the detail of my reconstruction will allow me to identify precisely how much biomass has been added to the forest since fire suppression, and whether the increase is due to the addition of young trees infilling the forest, or the increase in diameter of existing canopy trees. However, since the data used for this analysis were collected for a forest structure and fire history study, not all of the data required for a complete carbon accounting were collected (e.g. soil carbon levels, fine fuels). This should not pose a problem, because the purpose of this project is to compare the carbon breakdown of the pre-fire suppression forest with the contemporary forest. The detailed analysis of carbon change in this project will complement other studies attempting to further clarify the carbon cycle in mixed conifer forests in the western U.S. (Black and Harden 1994, Law et al. 2001, Law et al. 2003, Hicke et al. 2004). Understanding the carbon cycling of mixed conifer forests is important because they cover 1.6 million ha (Franklin and Fites-Kaufman 1996) in the Klamath, Sierra Nevada and Cascade ranges (Barbour 1988).

The overall objective of this study is to develop a more accurate accounting of the changes in carbon storage caused by fire suppression in an old-growth mixed conifer forest of Yosemite National Park. The general null hypothesis is that carbon sequestration in pre-settlement forests does not vary from that of contemporary forests. More specifically, this study attempts to answer the following questions: 1) What was the carbon content of the pre-fire suppression forests?; 2) How has the carbon content of the contemporary forests changed from the reference forest?; 3) How has the distribution of carbon within different components of the forest (live versus dead trees, young versus old trees) changed since the onset of fire suppression?

4.2 METHODS

4.2.1 Forest Structure

Historic and contemporary forest structure (size and age) were reconstructed in two study areas, Big Oak Flat (BOF) & South Fork Merced (SFM) from vegetation data collected in plots (BOF, n=85; SFM, n=64). In each study area, plots were located on a grid with gridpoints placed at 500 m intervals across the study areas, and included all slope aspects and elevations present. At each gridpoint, forest characteristics were collected in a series of nested, circular plots centered on the gridpoint. In the largest plot (1000 m^2), all trees (live and standing dead) > 35 cm dbh (conifer) and >15 cm dbh (hardwood) were measured and identified by species. All live trees were cored at 30 cm height, while the diameter, species, and decay class (Maser et al. 1979) of all logs (>35 cm) rooted in the plot were recorded. In the intermediate plot (250 m^2), all trees 10-35 cm dbh (conifers) and 5-15 cm dbh (hardwoods) were sampled in the same way as the large plot. In the smallest plot (100 m^2), all live seedlings (50 cm – 1.4 m height) and saplings (1.4 m height – 10 cm dbh) were counted by species. In addition, fuel loadings were estimated for each plot. Litter and duff depth was measured to the top of the mineral soil layer (cm), and coarse woody debris (CWD) was estimated from a photo series of fuel conditions developed for the Sierra Nevada (Blonski and Schramel 1993). Specific photo series were selected by selecting the series with the closest approximation of conditions within each plot. This included a visual comparison of the photo series with each plot and also a comparison of forest structural conditions listed in the photo series with structural characteristics collected from each plot.

Ages of live trees in the plots were determined by coring each tree measured in the plot to the pith with an increment borer. Each core was then sanded to a high polish and annual rings were cross-dated using standard dendroecological techniques and each ring was then assigned a calendar date (Stokes and Smiley 1968). The year of the innermost annual ring was then used as an estimate of tree age. Tree ages were estimated for trees that could not be aged directly (BOF = 19%, SFM = 15%) because either the increment bore was too short or the pith was rotten. Tree ages were estimated by first developing a regression between tree diameter and core length and then using the regression to determine the missing length of each incomplete core. Then, the average growth rate (rings cm^{-1}) for each species was used to determine the number of years missing from each incomplete core, and those years were added to the age of the core.

4.2.2 Reconstruction of Reference Forest

To estimate the amount of change in carbon stored in the forest that has occurred since the start of fire suppression, it was necessary to determine what the forest looked like prior to the onset of fire suppression. The reference point for the forests was the year of the last widespread fire (1899) to occur in both study areas (see Chapter 1). Reconstructing forest conditions for earlier periods can be problematic, due to the possibility of trees being consumed by subsequent fires (Fule et al. 1997).

Characteristics of the reference forest were reconstructed using the following dendroecological technique developed by Fule et al. (1997) (See chapter 1). First, all live trees that established after 1899 (stems ≤103 years old) were removed. Then, for all trees older than 103 years, I subtracted the amount of growth that occurred since 1899. For trees dead in 2002, I had to determine which trees were alive in 1899. Death dates for snags and logs were estimated based on the diameter, species and decay class (Maser et al. 1979) of the tree using the method described by Fule et al. (1997). First, species specific decomposition rates were calculated between each of the decomposition categories derived from the decay class of the sample. Second, date of death for each tree was determined by subtracting the total number of years derived from the decomposition rates. Third, tree sizes of dead trees in 1899 were determined by subtracting the amount of growth equal to the number of years between 1899 and the year of death of the trees. Last, tree ages were estimated for dead trees based upon species specific age-dbh regressions developed from all live trees cored.

4.2.3 Calculation of Carbon Pools

Carbon amounts were estimated for both study areas for the following tree components: stem wood, stem bark, live and dead branches, foliage, live and dead coarse roots, fine roots, coarse woody debris (CWD) and duff. Estimation of total tree carbon for each study area was determined by: 1) converting tree diameters to carbon amounts for each component; 2) adding all components together to determine the total carbon amount per tree; 3) summing all values on a per hectare basis, and; 4) adding in the carbon pools for CWD and duff in each plot. Species specific equations and procedures were used to calculate the biomass amounts for each pool. Biomass amounts were then converted to carbon by multiplying them by a species specific constant (Table 4.1) (Turner et al. 1995, Schlesinger 1997). Because the carbon amounts were calculated from data collected for a different study, there are no data available concerning understory vegetation and soil, and consequently, no carbon values for those components of the forest. Therefore, the total carbon values calculated only represent the total carbon associated with trees in the system, including CWD and duff.

Table 4.1: Carbon content, turnover time and decay rate constants for different forest types in Sierra Nevada mixed conifer forests, and the species they are used for in this study.

Forest Type	Used for in this study	Carbon content(g/g)*	Decay rate constant (yr^{-1})*	Turnover time (yrs)
Douglas fir	PSME	0.512	0.018	56
Fir-Spruce	ABCO, ABMA	0.512	0.024	42
Ponderosa pine	PIPO, PILA	0.512	0.015	67
Hemlock-Spruce	CADE	0.512	0.029	34
Hardwoods	QUKE	0.496	0.067	15

* From Turner et. al. (1997)

Note: Species abbreviations are: ABCO (Abies concolor); ABMA (Abies magnifica); ABPR (Abies procera); CACH (Castanopsis chrysophylla); CADE (Calocedrus decurrens); CEDAR (multiple cedar species pooled); CONU (Cornus nuttallii); PILA (Pinus lambertiana); PIPO (Pinus ponderosa); PSME (Pseudotsuga menziesii); QUKE (Quercus kellogii); QUSP (Quercus species).

4.2.3.1 Above-ground Tree Carbon

Carbon for live trees was estimated from the diameter of each tree measured in the plots in the following way. First, the biomass of stem wood, stem bark, live and dead branches and foliage were calculated using the BIOPAK program (Means et al. 1994). BIOPAK uses species specific allometric equations derived from a broad set of biomass studies of both regional and specific forests to estimate biomass for individual tree components from tree dbh. All equations used in the analysis were of the form

$$\ln B = A + B \ln X$$

where B is the biomass, X is the tree diameter, and A and B are constants derived from specific studies as used in BIOPAK (Means et al. 1994). All output from BIOPAK was corrected for logarithmic bias. Species specific equations were not available for all components, so substitutions were made where necessary with similar species (Table 4.2). Substitutions for conifers were made with closely related species in the same genus whenever possible because they produced a smaller variation in biomass than between genus substitutions (Smithwick et al. 2002). For Pacific dogwood, no equations were available for the stem wood and stem bark components, so I used equations for total stem biomass. No estimate was made for seedlings and saplings because diameter measurements were not collected.

Table 4.2: Species specific equations and substitutions of closely related species used in BIOPAK software for calculations of biomass for separate components of trees, based upon tree dbh, in old-growth, mixed conifer forests in Big Oak Flat and South Fork Merced study areas in Yosemite National Park, CA. R^2 values for equations listed in parenthesis.

Species	Component					
	Stem Wood*	Stem Bark*	Stem Total*	Live Branches*	Dead Branches*	Total Foliage*
ABCO	ABCO (0.97)	ABCO (0.94)	-	ABPR (0.94)	PSME (0.84)	ABPR (0.99)
ABMA	ABMA (0.98)	ABMA (0.95)	-	ABPR (0.94)	PSME (0.84)	ABPR (0.99)
CADE	CADE (0.96)	CADE (0.94)	-	CEDAR (0.94)	PIPO (0.64)	CEDAR (0.91)
PILA	PILA (0.97)	PILA (0.93)	-	PILA (0.81)	PIPO (0.64)	PILA (0.52)
PIPO	PIPO (0.98)	PIPO (0.97)	-	PIPO (0.99)	PIPO (0.64)	PIPO (0.84)
PSME	PSME (0.93)	PSME (0.85)	-	PSME (0.77)	PSME (0.84)	PSME (0.80)
QUKE	CACH (0.98)	CACH (0.97)	-	CACH (0.89)	CACH (0.81)	CACH (0.81)
CONU	-	-	CONU (0.78)	CONU (0.84)	CONU (0.82)	CONU (0.84)

* From Means et al. (1994)

To accurately determine the amount of carbon stored in the forests, the carbon stored in dead woody material must be included in the forest totals. Carbon for dead trees (standing and downed logs) was calculated in the following way. First, biomass was calculated for each component in the same way as for live trees using BIOPAK. Biomass was only calculated for the components currently present on the tree in 2002. For example, foliage biomass was not calculated for trees without needles present, and stem bark biomass was not calculated for trees with bare trunks. Second, biomass was converted to carbon for each component. Third, carbon for each component was decayed using the equation:

$$C = C_0 \exp(-t/\tau) \qquad \text{(Hicke et al. 2004)}$$

where C is the carbon present in the decayed material, C_0 is the carbon amount present at the time of tree death, t is the number of years since tree death, and τ is the turnover time. Turnover time is the reciprocal of the decay rate constant and represents the amount of time required for dead wood to be reduced to 37% of its initial value. Species specific decay rates were used in the calculation of turnover time when possible, otherwise rates from closely related species were used (Table 4.1) (Turner et al. 1995). Decayed carbon values were then summed up to determine the total carbon stored in each dead tree.

4.2.3.2 Below-ground Tree Carbon

Since no direct measurements were made of below-ground components in the study areas, root biomass was estimated in two different ways. Coarse root biomass (>5 mm) was estimated using allometric equations developed from a study of Douglas-fir forests in the Oregon Cascades (Grier and Logan 1977) which contained three of the species found in the mixed conifer forests in Yosemite National Park; Douglas-fir, sugar pine and Pacific dogwood. The equation was developed from data pooled for all species in the forests and was in the format:

$$Y = \exp(A + B \ln X)$$

where Y is root biomass, X is stem diameter, and A and B are constants based upon stem diameter (Grier and Logan 1977). Values of Y are corrected for logarithmic bias. I assumed that the ratio between dead root biomass and dead tree diameter would be the same as the ratio between live root biomass and live stem diameter, and used the same calculation to determine coarse root biomass for all dead trees in the plots. Fine root biomass (<5 mm) was estimated as ~ 2% of total aboveground biomass (Grier and Logan 1977). Fine root biomass was calculated for both live and dead trees.

4.2.3.3 Coarse Woody Debris (CWD) and Duff Carbon

Because no direct measurements of coarse woody debris (<10 cm) and duff were collected, estimates of biomass were determined using a different approach. Total biomass of CWD was estimated from the photo series developed for quantifying woody fuels in the Sierra Nevada (Blonski and Schramel 1993). Litter and duff carbon pools were estimated from litter and duff depths collected in the plots using an equation developed for experimental plots in Blodgett Forest Research Station (BFRS) in the central Sierra Nevada (Brown et al. 2004). At BFRS, litter and duff depths were measured in multiple plots, and subsamples were collected to determine actual litter and duff biomass. A strong relationship was found between duff depth and carbon content ($r^2 = 0.91$). The resulting equation models the relationship between the combined depth of litter and duff and total carbon as follows:

$$C = 5.4887X + 3.7141 \qquad \text{(Brown et al. 2004)}$$

where C is megagrams carbon per hectare, and X is depth of litter/duff in cm. While it is impossible to determine the amounts of CWD and litter/duff in the historic forests, I included an estimate for comparison purposes. I used values of CWD and litter/duff biomass from an ongoing study in mixed conifer forests in Yosemite National Park that have been burned repeatedly (Keifer et al. 2006). The combined CWD, duff and litter measured 5 years after a prescribed fire was 4.4 kg/m^2 (standard error = 0.5). The amount of fuels measured are probably a reasonable estimate of historic fuel conditions in these forests which experienced a fire return interval of 1-16 years.

4.2.3.4 Total Tree Carbon

To derive the total tree carbon in the plots, all of the above values were combined per tree, converted to a per hectare basis, and then summed within each plot. To assess the relative amounts of carbon stored in trees of different sizes and trees of different ages, I summed the total carbon per plot into 20 year age-classes and 10 cm size-classes. Change in carbon storage since the onset of fire suppression was assessed by comparing the carbon storage between the historic pre-fire suppression and contemporary forests using a distribution free Kruskal-Wallis H test.

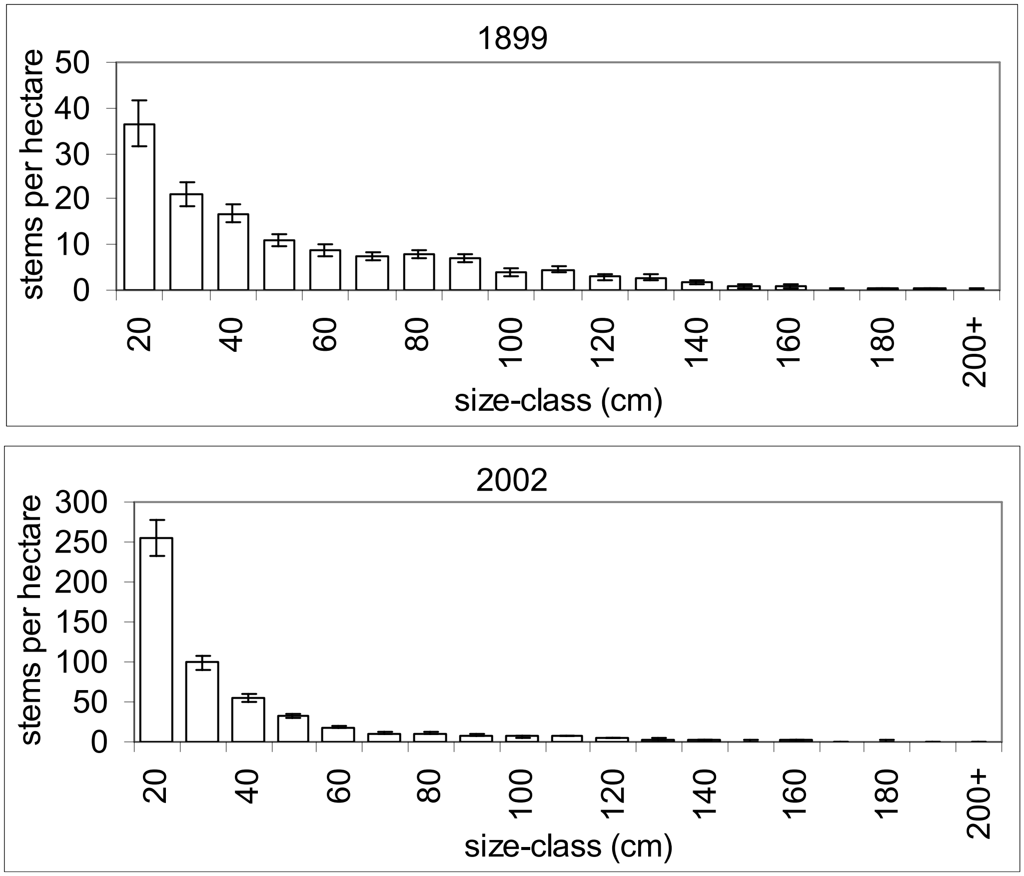
4.3 RESULTS

4.3.1 Forest Structure

Overall, the basal area and density of stems increased significantly ($p<0.01$) between 1899 and 2002 in both study areas (Table 4.3). The maximum size of trees measured in the two study areas were >200 cm in diameter at breast height (BOF = 215 cm dbh, SFM = 221 cm dbh) (Figure 4.1, 4.2). Trees in the contemporary forest ranged in age from 10 years to nearly 600 years old in both study areas (BOF = 10 - 586 years, SFM = 12 - 559 years). The distribution of tree ages and sizes were similar in shape between 1899 and 2002 in both study areas, however, there were more trees present in the contemporary forest. The majority of the increase in density of individuals was in the younger age-classes (< 140 yrs old) and smaller size-classes (<50 cm dbh).

Table 4.3: Density and basal area of plots in Big Oak Flat (BOF) and South Fork Merced (SFM) study areas for 1899 and 2002 in old-growth, mixed conifer forests of Yosemite National Park, CA.

	Density (# trees/ha)			Basal area (m^2/ha)		
	Mean	SD	Range	Mean	SD	Range
BOF						
2002	516.1	242.4	90-1220	69.4	25.9	20.7-150.8
1899	129.6	75.5	30-450	36.3	20.4	4.9-103.7
SFM						
2002	637.5	412.4	90-2060	63	21.2	29.1-120
1899	137	88.7	10-520	49	22.8	5.9-99



a)

Figure 4.1: Page 1 of 2. Mean density (±SE) of trees in a) 10 cm size-classes, and b) 20-year age-classes of old-growth, mixed conifer forests in Big Oak Flat study area for 1899 and 2002 in Yosemite National Park, CA.

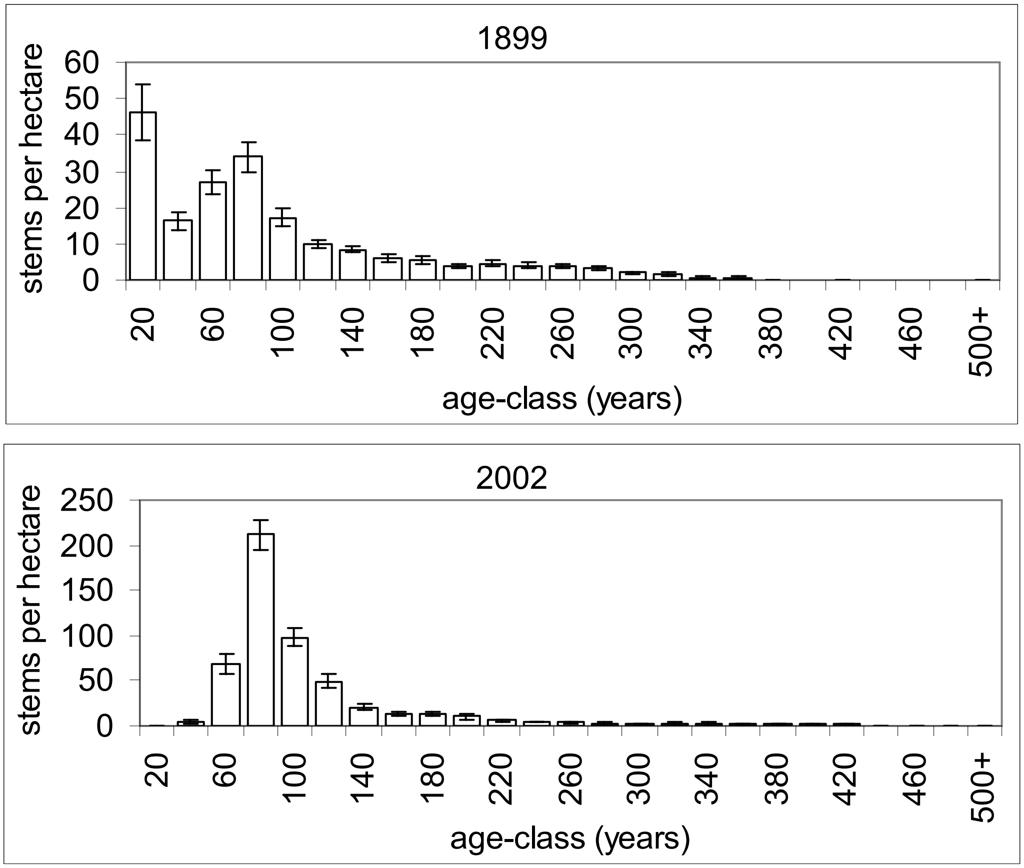


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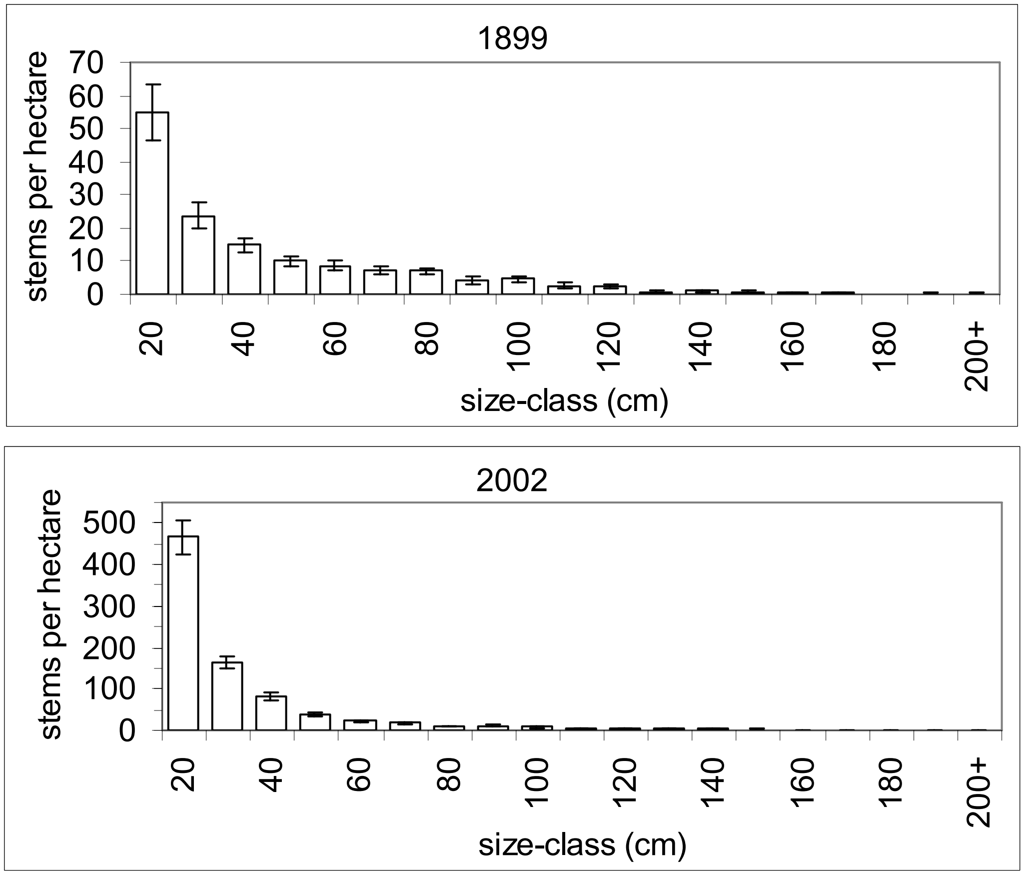
Figure 4.1: Page 2 of 2.



a)

Figure 4.2: Page 1 of 2. Mean density (±SE) of trees in a) 10 cm size-classes, and b) 20year age-classes of old-growth, mixed conifer forests in South Fork Merced study area for 1899 and 2002 in Yosemite National Park, CA.

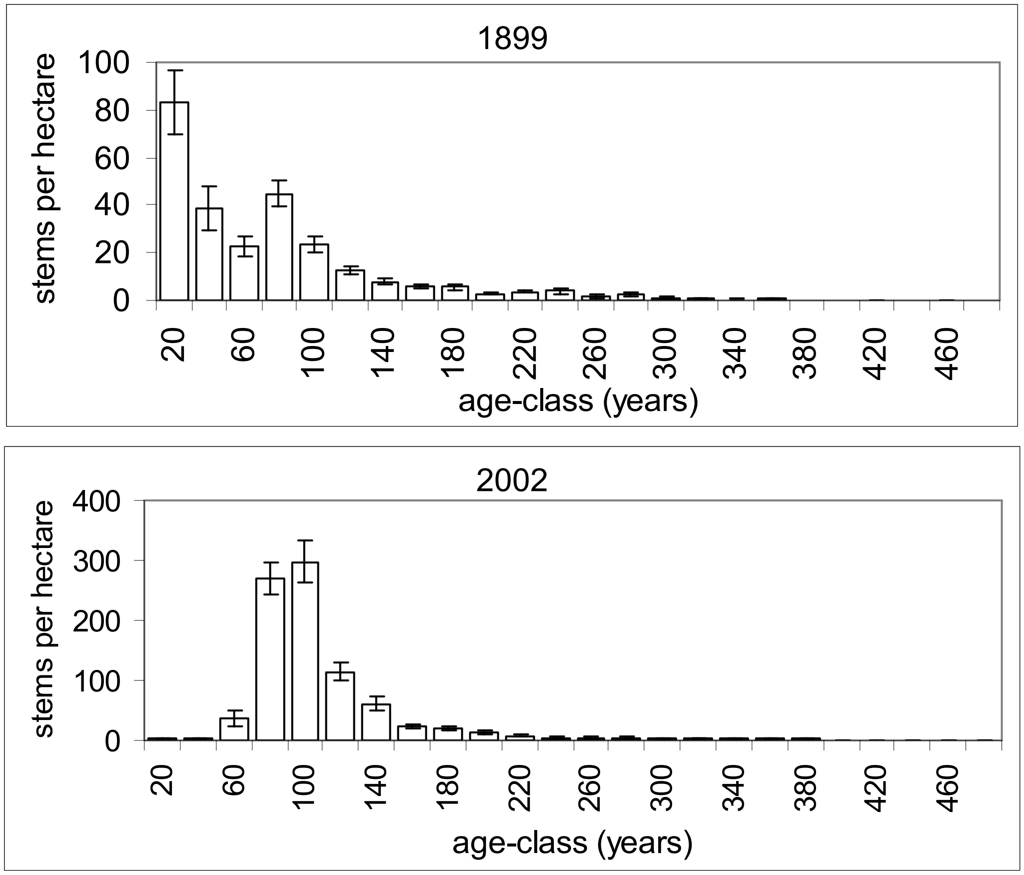


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Figure 4.2: Page 2 of 2.

4.3.2 Carbon Content

The total carbon content for both study areas showed a similar pattern between 1899 and 2002 as the basal area (Figure 4.3). There was a significant increase ($p<0.001$) in carbon in BOF from an average of 222.7 Mg C/ha (range 43.3-860.2 Mg C/ha) in 1899 to 525.9 Mg C/ha (range 183.1-1205.9 Mg C/ha) in 2002. A similar increase ($p<0.001$) was present in SFM from an average of 166.6 Mg C/ha (range 25.5-526.0 Mg C/ha) in 1899 to 459.8 Mg C/ha (range 185.2-939.9 Mg C/ha) in 2002.

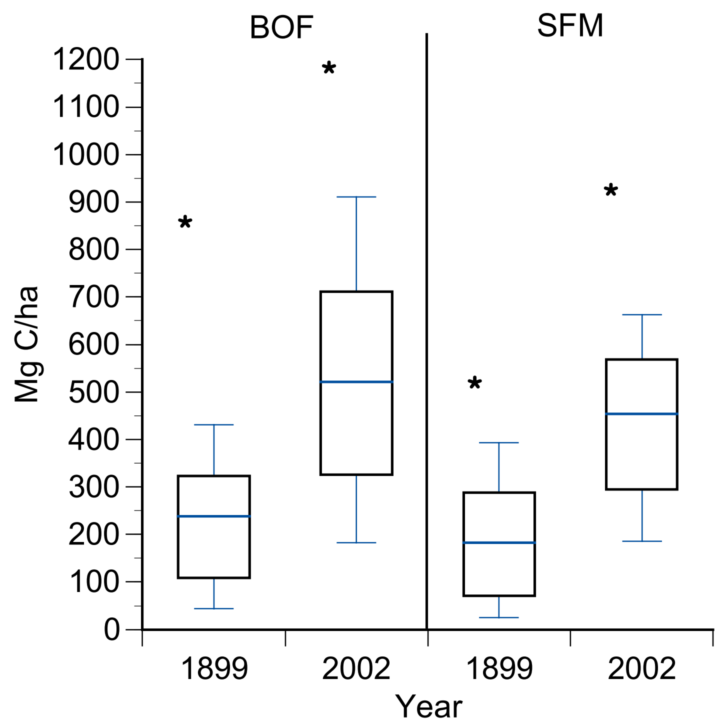


Figure 4.3: Total carbon content (Mg C/ha) for Big Oak Flat (BOF) and South Fork Merced (SFM) study areas for 1899 and 2002 in old-growth, mixed conifer forests in Yosemite National Park, CA. Box length is the interquartile range (middle 50% of the cases). Whiskers are the plots within 1 box length of the upper and lower edges of the box. Asterisks are for most extreme outlier plots in each study area.

The significant increase in carbon from 1899 to 2002 was consistent across all tree component carbon pools (Table 4.4). The largest change in carbon between 1899 and 2002 occurred in the stem wood and litter/duff pools. Although the increase in stem wood carbon (BOF = 64.6 Mg C/ha, SFM = 68.3 Mg C/ha) was greater than the increase in litter/duff carbon (BOF = 54.7 Mg C/ha, SFM = 63.7 Mg C/ha), the relative increase in litter/duff was much more significant at 20% of prior content versus a 2% increase in the stem wood pool. The smallest change occurred in the fine roots and dead branches pools (0-0.4 Mg C/ha). The pool contributing the largest percentage of the total carbon

Table 4.4: Mean, range and % total carbon (Mg C/ha) within each component of the trees in Big Oak Flat (BOF) and South Fork Merced (SFM) study areas in old-growth, mixed conifer forests in Yosemite National Park, CA.

BOF	2002			1899		
Component	Mean	Range	% of Total	Mean	Range	% of Total
Stem Wood	159.5	26.7 - 423.2	30.3	94.9	8.2 - 340	42.6
Stem Bark	56.0	12.7 - 166.2	10.6	32.2	2.8 - 137.2	14.5
Live Branches	51.0	7.8 - 213.8	9.7	34.1	2.7 - 226.6	15.3
Dead Branches	3.8	1.2 - 9.0	0.7	1.8	0.3 - 4.7	0.8
Foliage	10.1	2.6 - 20.9	1.9	5.9	0.7 - 15.1	2.7
Coarse Roots	75.8	15.9 - 201.2	14.4	46.0	4.3 - 158.9	20.7
Fine Roots	5.6	1.0 - 15.6	1.1	3.3	0.3 - 13.3	1.5
Dead Trees	105.4	0.6 - 402.9	20.0	-	-	0.0
CWD	1.1	0.3 - 2.4	0.2	1.5	-	0.7
Duff	57.6	8.0 - 144.0	11.0	2.9	-	1.3
Total	525.9	184.2 - 1217.5	100.0	222.7	25.7 - 856.8	100.0

SFM	2002			1899		
Component	Mean	Range	% of Total	Mean	Range	% of Total
Stem Wood	142.2	34.3 - 424.9	30.9	73.9	1.3 - 269.4	44.3
Stem Bark	45.9	12.9 - 129.8	10.0	24.1	0.7 - 94.2	14.5
Live Branches	42.8	13.0 - 168.2	9.3	22.2	0.5 - 119.9	13.3
Dead Branches	3.9	1.7 - 8.5	0.8	1.5	0.1 - 3.5	0.9
Foliage	10.3	3.1 - 29.2	2.2	4.5	0.2 - 11.7	2.7
Coarse Roots	64.3	20.0 - 165.6	14.0	33.6	0.7 - 106.1	20.1
Fine Roots	4.8	1.4 - 12.4	1.0	2.5	0.1 - 8.1	1.5
Dead Trees	78.2	0.2 - 648.2	17.0	-	-	0.0
CWD	0.8	0.2 - 1.8	0.2	1.5	-	0.9
Duff	66.6	8.0 - 144.0	14.5	2.9	-	1.7
Total	459.8	185.1 - 968.2	100.0	166.6	8.0 - 515.1	100.0

content was stem wood for live trees (BOF = 30.3%, SFM = 30.9%), followed by dead trees (BOF = 20%, SFM = 17%). The smallest contribution to total carbon was CWD (<10 cm) and dead branches, both of which contributed less than 1% to the total tree carbon in the forests. In the historic forests, the largest contributions to the total tree carbon was stem wood (>40%) and coarse roots

(>20%), while the smallest contributions were from dead branches and CWD (<1%).

The contribution of carbon by each species to the total carbon pool varied between both study areas ($p<0.01$) (Figure 4.4). In BOF, significantly more carbon was contributed on average by white fir (ABCO) (102.7 ± 9.4 Mg C/ha), incense cedar (CADE) (79.5 ± 7.5 Mg C/ha) and sugar pine (PILA) (118.6 ±18.6 Mg C/ha), than in SFM (ABCO = 55.7 ± 9.6 Mg C/ha, CADE = 50.7 ± 5.7 Mg C/ha, PILA = 90.5 ± 17 Mg C/ha). In SFM, more carbon was contributed by ponderosa pine (PIPO) (138.6 ± 14.8 Mg C/ha), Douglas-fir (PSME) (25.7 ± 7.6 Mg C/ha) and black oak (QUKE) (20.6 ± 3.6 Mg C/ha) than in BOF (PIPO = 126.9 ± 14.8 Mg C/ha, PSME = 20.2 ± 8.8 Mg C/ha, QUKE = 13.5 ± 3.4 Mg C/ha). The distribution of carbon by species for BOF was different between the contemporary forest and the historic forest ($p<0.05$). In the historic forest, white fir (20.3 ± 5 Mg C/ha) contributed much less carbon than incense cedar (43 ± 5 Mg C/ha), sugar pine (71.2 ± 14.4 Mg C/ha) or ponderosa pine (70.2 ± 7.3 Mg C/ha), while in the contemporary forest, a much greater amount of carbon is contributed by white fir (102.7 Mg C/ha ± 9.4 Mg C/ha) than incense cedar (79.5 ± 7.5 Mg C/ha).

The distribution of total tree carbon among size-classes was similar ($p>0.05$) in both study areas in 1899 and 2002 (Figure 4.5, 4.6), although the total amount of carbon in each size-class was higher in 2002. The majority of the carbon (>50%) in each study area was contained in the intermediate size-classes (70-140 cm) in both 1899 and 2002. The largest increase in proportion of carbon

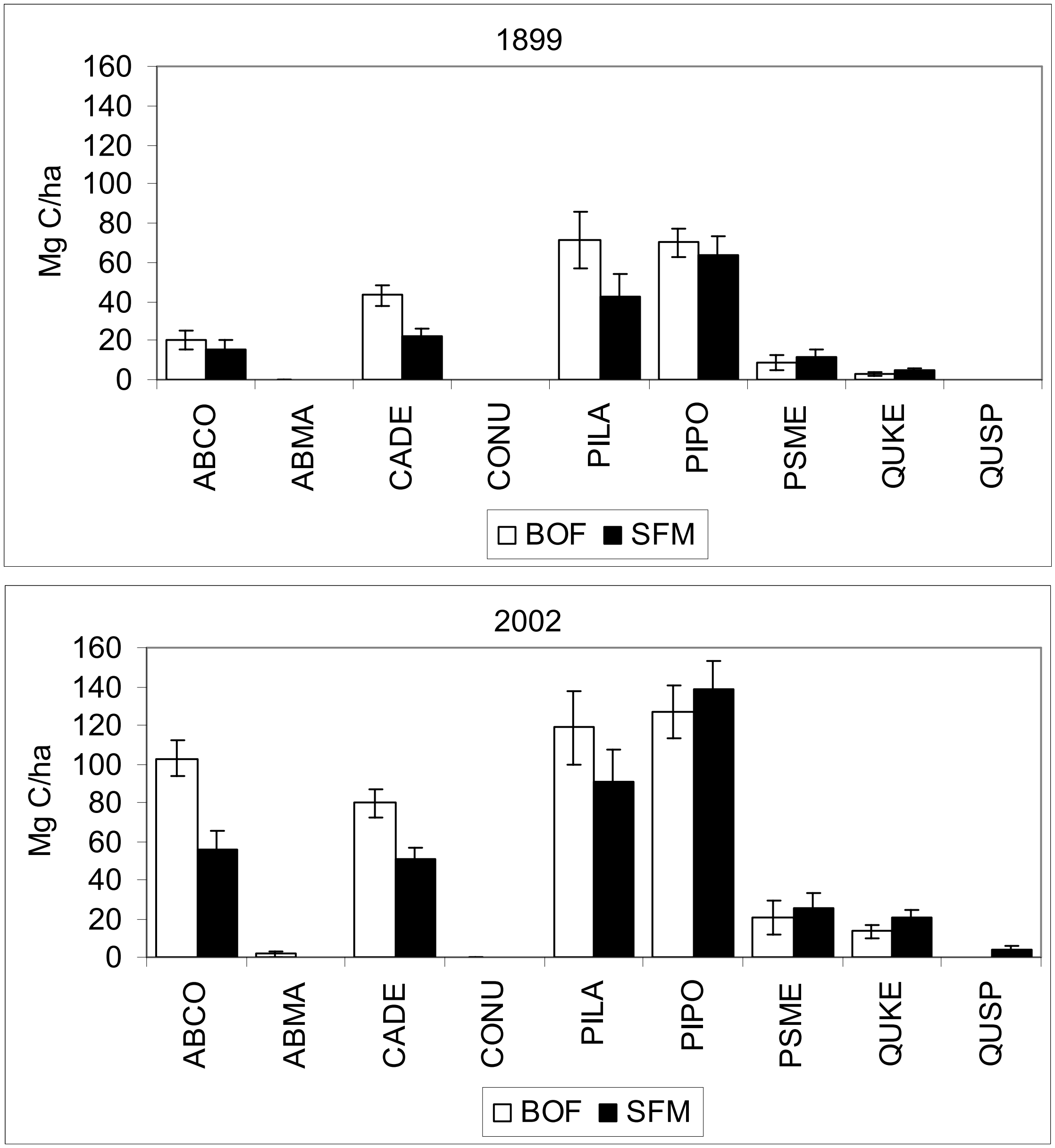
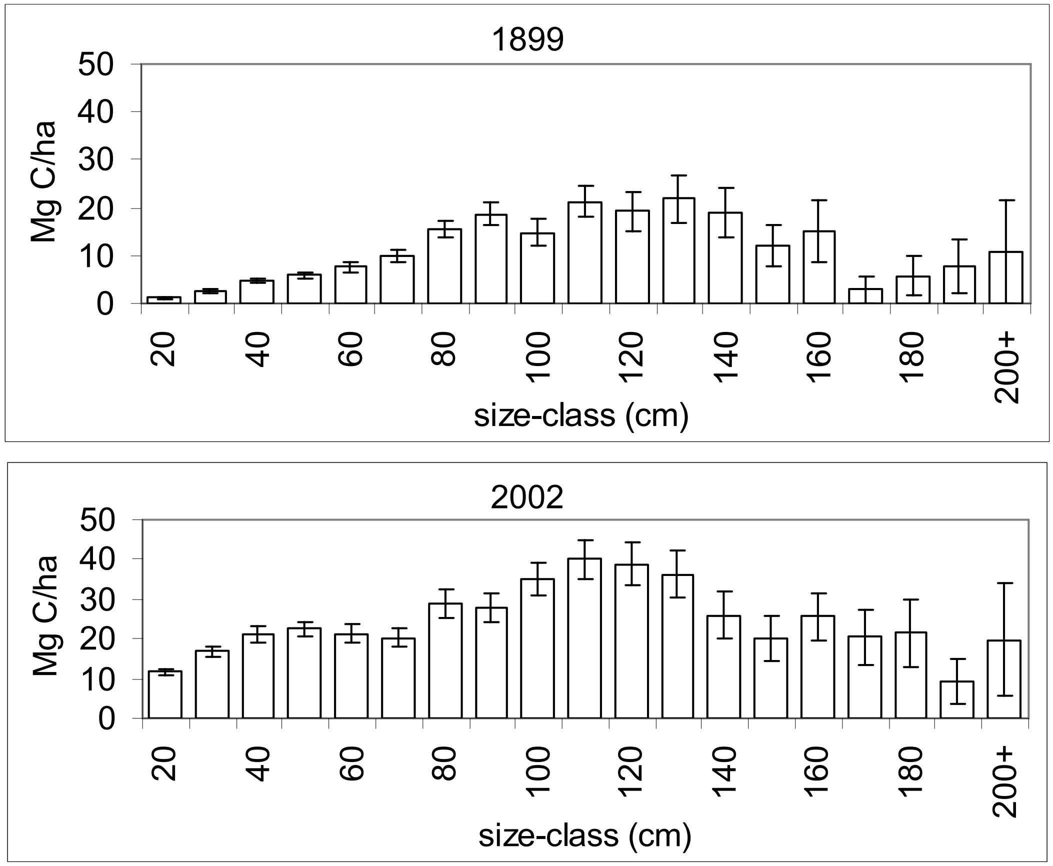


Figure 4.4: Total carbon stores (±SE) by species in Big Oak Flat (BOF) and South Fork Merced (SFM) study areas for 1899 and 2002 in old-growth, mixed conifer forests in Yosemite National Park, CA. Species abbreviations listed in Table 4.1.



a)

Figure 4.5: Page 1 of 2. Mean Carbon content (±SE) for each a) 10 cm size-class, and b) 20 year age-class of old-growth, mixed conifer forests in the Big Oak Flat study area for 1899 and 2002 in Yosemite National Park, CA.

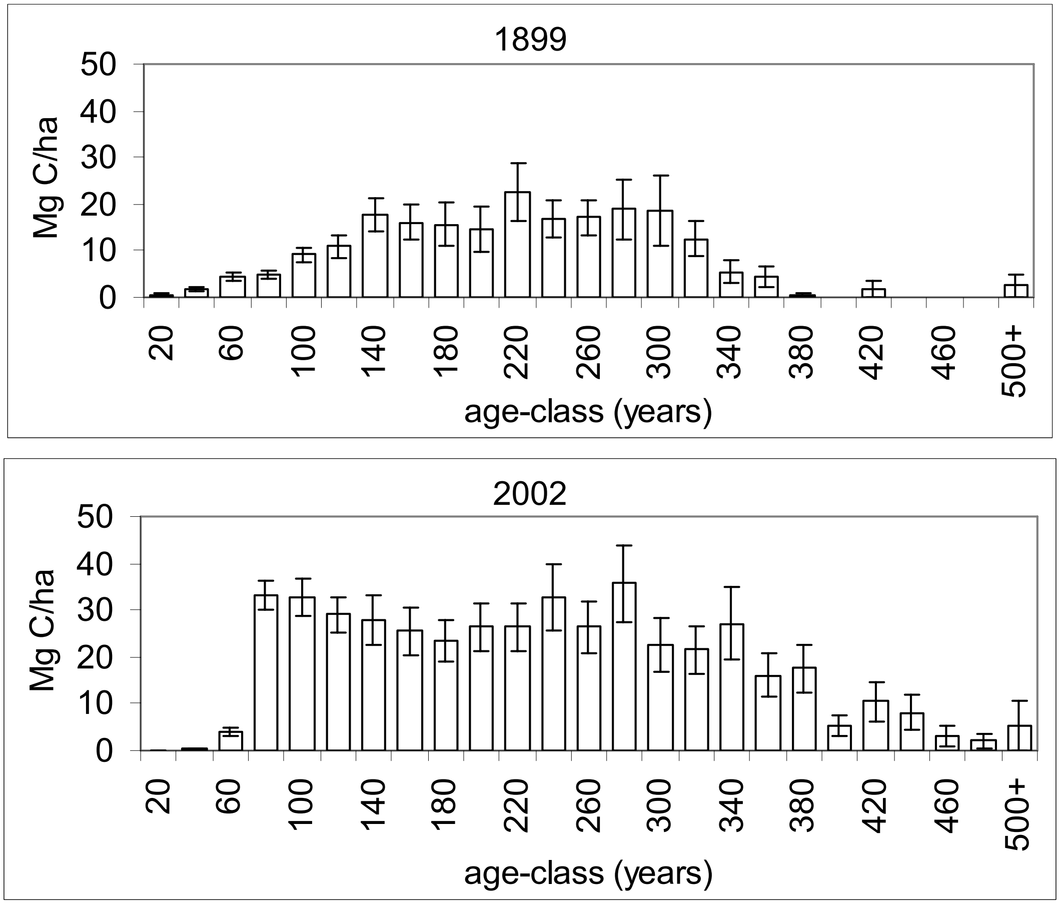


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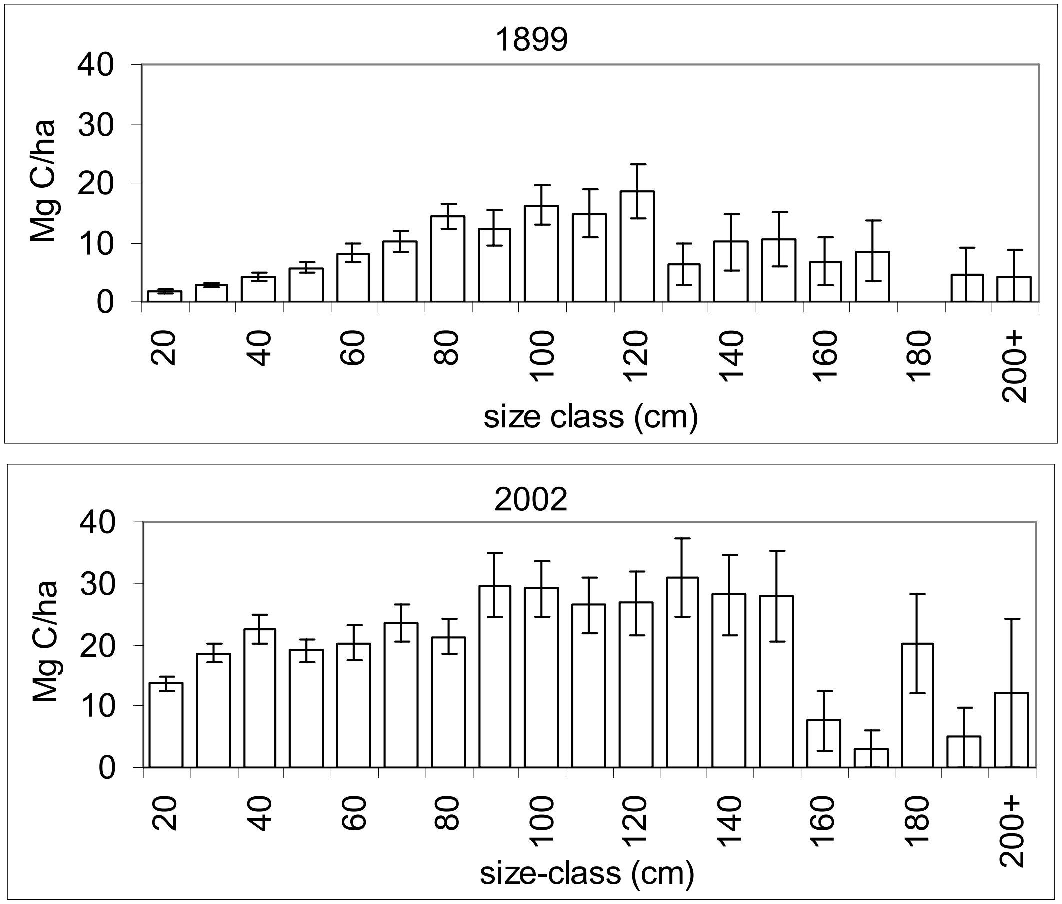
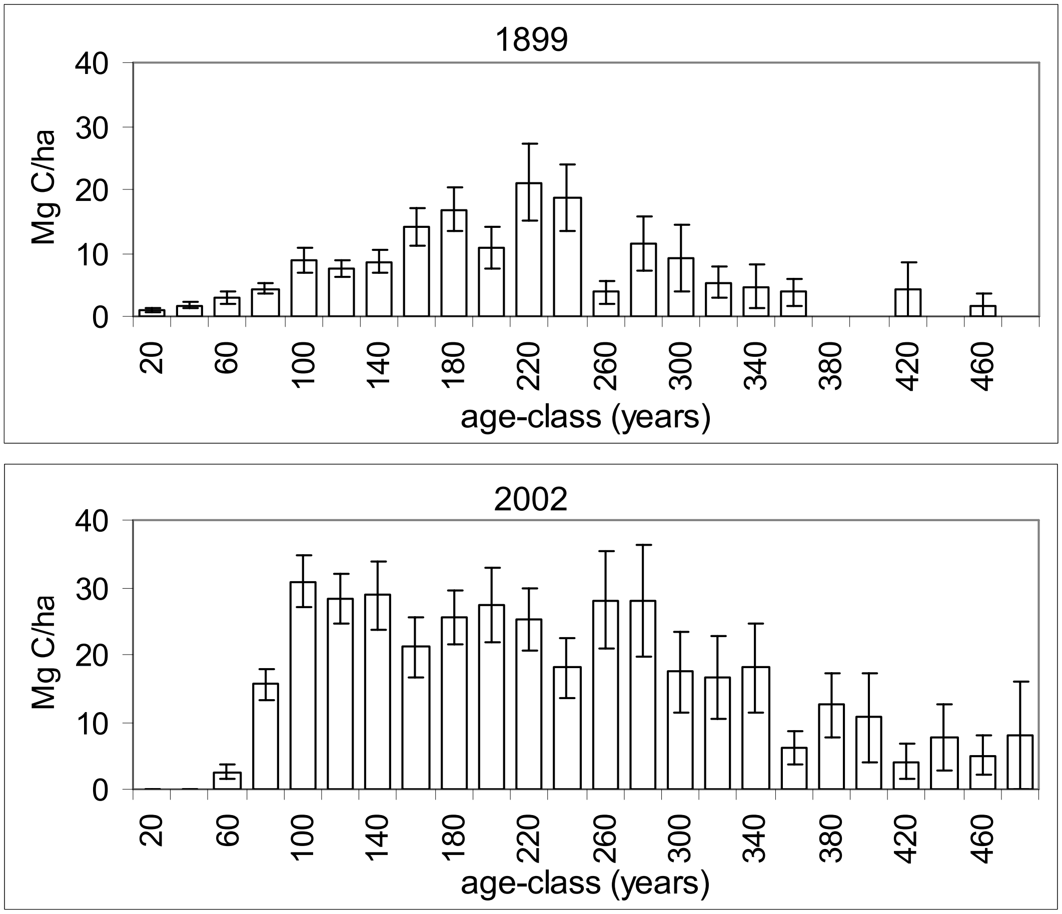
Figure 4.5: Page 2 of 2.



a)

Figure 4.6: Page 1 of 2. Mean Carbon content (±SE) for each a) 10 cm size-class, and b) 20 year age-class of old-growth, mixed conifer forests in the South Fork Merced study area for 1899 and 2002 in Yosemite National Park, CA.



b)

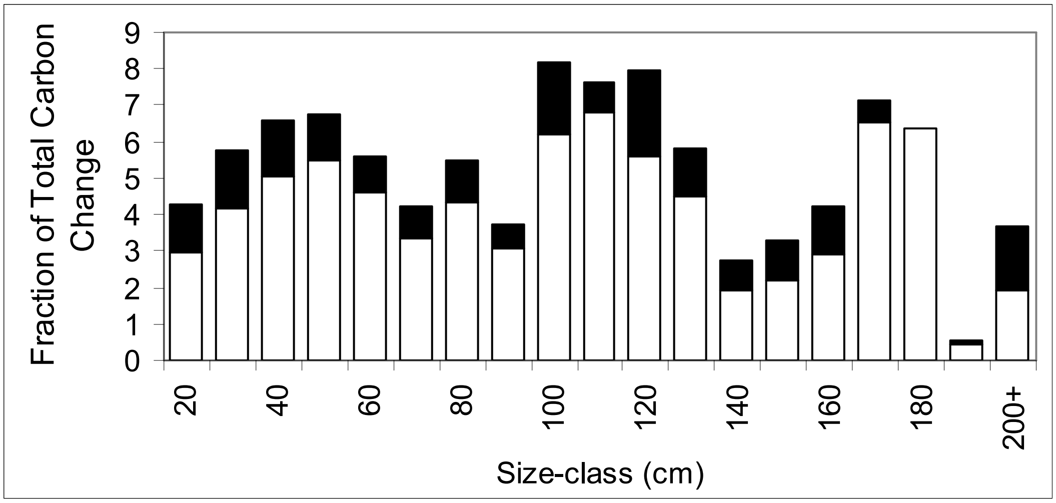
Figure 4.6: Page 2 of 2.

stored occurred in the small diameter trees (<70 cm). In 1899 a relatively small proportion of the total carbon (<15%) was contained in the small diameter trees, while in 2002 the content in the same size-classes was almost two times greater (27%). In contrast, the proportion of carbon contained in the largest size-classes (>140 cm) remained about the same between both time periods for both BOF (1899 = 25.1%, 2002 = 25.2%) and SFM (1899 = 21.6%, 2002 = 19.6%).

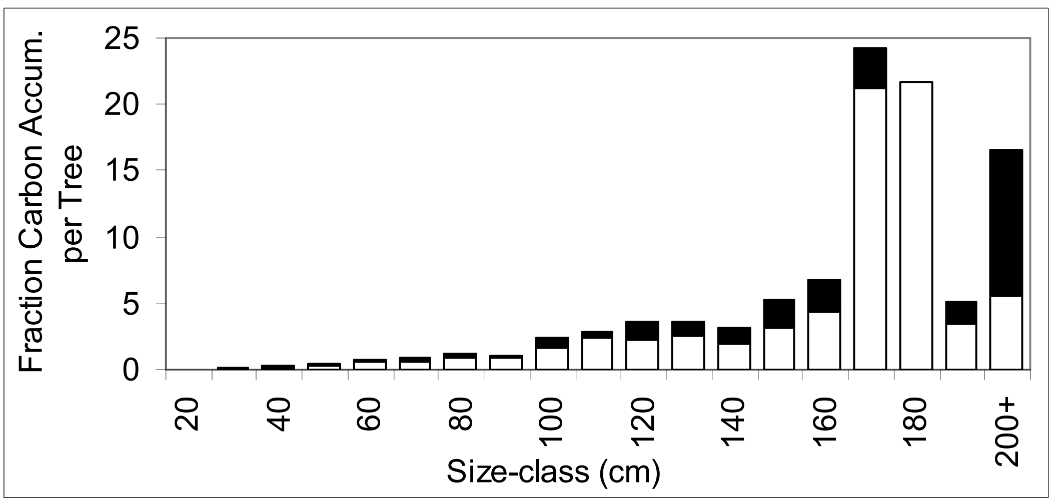
The distribution of total tree carbon among age-classes was significantly different ($p<0.05$) between 1899 and 2002 in both study areas (Figure 4.5, 4.6).

In 1899 the distribution of carbon was a unimodal structure with a peak in the proportion of carbon present (>10% in each age-class) in the middle age-classes (220-240 years), and quickly dropping off in the younger and older age-classes. The shape of the carbon distribution in 2002 was different with a relatively high proportion of the total carbon present in each age-class from 100-300 years (BOF = 5 – 7.7%, SFM = 4.7 – 8%). A relatively small proportion of the total carbon (<10%) was contained in the oldest (>400 years), and youngest (< 100 years) trees.

The change in total carbon between 1899 and 2002 increased across all size-classes in both BOF and SFM (Figure 4.7, 4.9). The greatest amount of increase in carbon storage occurred in the mid- to large-diameter individuals (100 -150 cm), although actual amounts were highly variable both between size-classes and between study areas. A slightly different pattern is present in the change in carbon storage by age-class (Figure 4.8, 4.10). The greatest increase in carbon was in the younger trees (80-200 years) in both study areas, while the oldest individuals (>440 years) contributed a relatively small amount to the total change in carbon storage. In addition, a decline in carbon storage occurred in the youngest age-classes (<80 years) from 1899 to 2002.

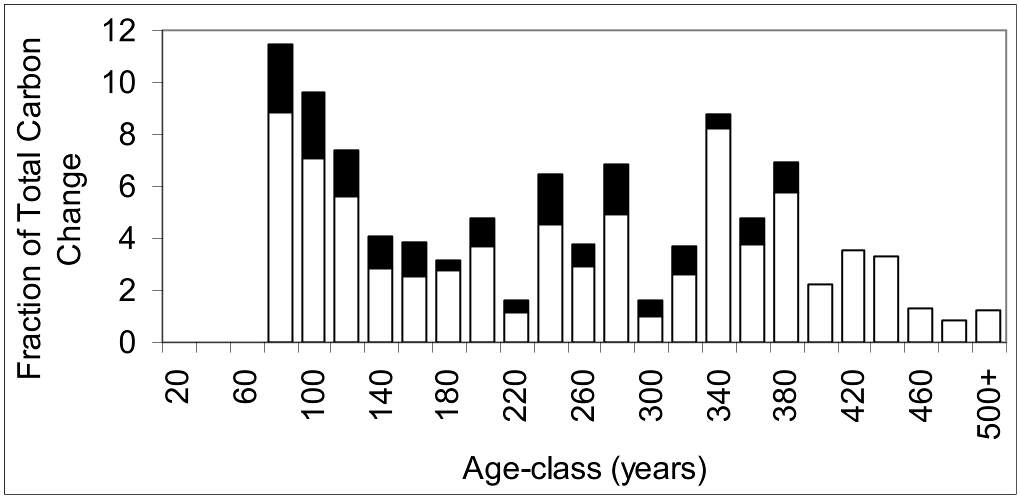


a)

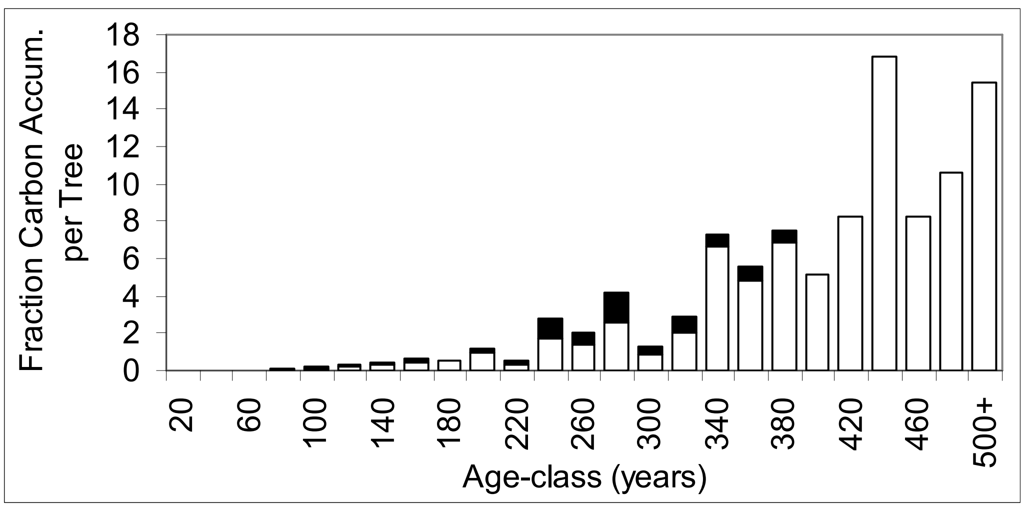


b)

Figure 4.7: Distribution among 10 cm size-classes of live (open bars) and dead (filled bars) trees of the a) Fraction (%) of total carbon change between 1899 and 2002 stored within each size-class, and b) fraction (%) of total carbon accumulation since 1899 on an individual tree basis, for old-growth, mixed conifer forests in the Big Oak Flat study area in Yosemite National Park, CA.

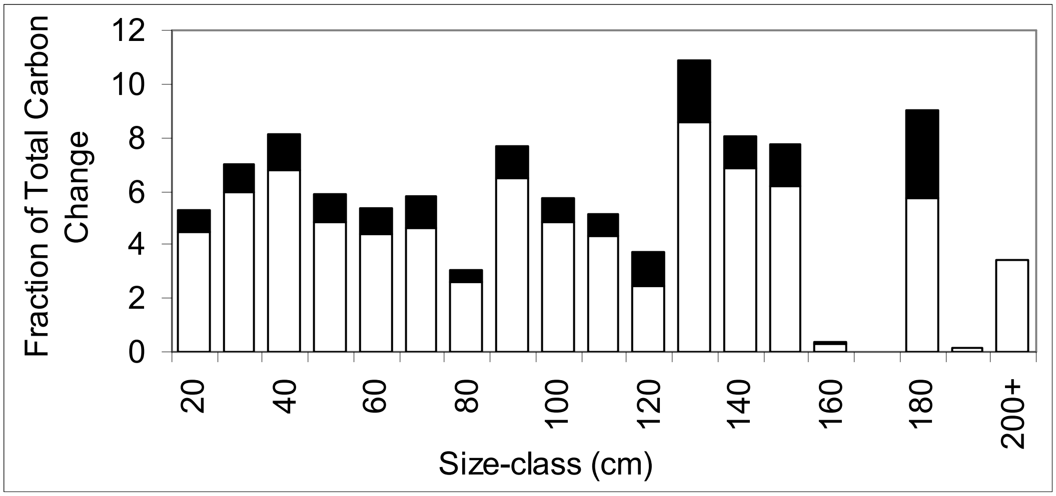


a)

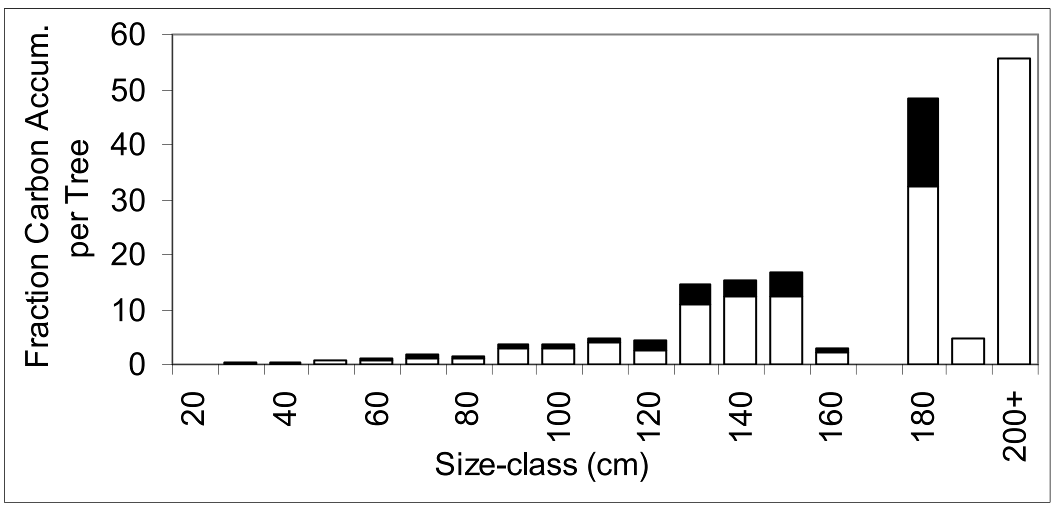


b)

Figure 4.8: Distribution among 20 year age-classes of live (open bars) and dead (filled bars) trees of the a) Fraction (%) of total carbon change between 1899 and 2002 stored within each age-class, and b) fraction (%) of total carbon accumulation since 1899 on an individual tree basis, for old-growth, mixed conifer forests in the Big Oak Flat study area in Yosemite National Park, CA.



a)



b)

Figure 4.9: Distribution among 10 cm size-classes of live (open bars) and dead (filled bars) trees of the a) Fraction (%) of total carbon change between 1899 and 2002 stored within each size-class, and b) fraction (%) of total carbon accumulation since 1899 on an individual tree basis, for old-growth, mixed conifer forests in the South Fork Merced study area in Yosemite National Park, CA.

a)

b)

Figure 4.10: Distribution among 20 year age-classes of live (open bars) and dead (filled bars) trees of the a) Fraction (%) of total carbon change between 1899 and 2002 stored within each age-class, and b) fraction (%) of total carbon accumulation since 1899 on an individual tree basis, for old-growth, mixed conifer forests in the South Fork Merced study area in Yosemite National Park, CA.

On a tree by tree basis, the pattern of the fraction of the change in carbon storage between 1899 and 2002 was dramatically different from the pattern of the total change per age- and size-class. The relative contribution of carbon by individual trees was greatest in the largest individuals (>170 cm) in both study areas (Figure 4.7, 4.9). Small- to mid-sized individuals (<120 cm) contributed relatively small amounts to the total increase in carbon storage since 1899. A similar pattern was present in the relative contribution of each tree in each age-class (Figure 4.8, 4.10). The oldest individuals (>400 years) contributed the largest amount to the overall change in carbon storage that occurred, while the trees <260 years old contributed relatively small amounts individually to the total change in carbon.

4.4 DISCUSSION

4.4.1 Stand level carbon storage

Fire suppression resulted in an increase in carbon storage in both study areas in Yosemite National Park. Over the past 100 years the carbon content of both study areas increased by 42% (BOF) and 36% (SFM) as a result of infilling by young trees and an increase in diameter of existing trees. Since there were no large, discrete patches in the reference forest, woody encroachment was not considered to contribute a measurable amount to the overall increase. The wide range of estimates for plot level carbon storage in the two study areas also suggests that while the forest cover was continuous, it was highly variable, both by species composition and density of stems, a pattern that is common in mixed conifer forests in the Sierra Nevada (Bonnicksen and Stone 1981, Taylor 2004, Beaty and Taylor 2007). However, while the changes in carbon storage contribution by species increased for all species, the reason for the increase varied. The increase in carbon storage by white fir and incense cedar is primarily the result of increases in the density of stems for both species. Both white fir and incense cedar are fire intolerant as young trees, and had relatively low densities in the reference forest. As a result of fire suppression, their densities increased by over 325%. Consequently the large increase in carbon storage contributed by both species is due primarily to infilling by younger individuals. In contrast, the large increase in carbon storage contributed by ponderosa pine and sugar pine is due to the increase in diameter of existing trees. Both species had a much lower increase in density since 1899 (PIPO = 167%, PILA = 133%). The relatively larger increase in carbon accumulation since 1899 in BOF by white fir (82.4 Mg C/ha) compared to the pines (~52 Mg C/ha) indicates that infilling of the forest is contributing more to the total carbon storage than an increase in tree diameter. This is further supported by the breakdown in carbon contribution by size- and age-class where the largest increases are occurring in the youngest age-classes and smallest size-classes. This suggests that at the stand level, the increase in carbon in the forest since 1899 is largely the result of infilling due to fire suppression, instead of the increase in diameter of existing trees.

4.4.2 Plot level carbon storage

Carbon accumulation at the plot level is related to both the increase in diameter of existing trees, and infilling by new stems. On a tree by tree basis, the amount of carbon stored per individual is directly related to the size of the tree and subsequently to the age of the tree. Consequently, the larger and older trees can be expected to contain more carbon per individual than the younger and smaller trees. However, at the plot level, the aggregate effect of total number of individuals must be considered. In both study areas the greatest amount of carbon was actually contained in the mid-sized trees where individuals were of a moderately large diameter and there was a moderately high density of stems. The few large diameter trees (>150 cm dbh) store much less carbon, on a per hectare basis, than do smaller individuals. Even the smallest diameter trees (<80 cm dbh) store the same, if not more, carbon per hectare as the largest trees. A similar pattern is present in the distribution by age class, where the cumulative effect of all the young trees (<150 years) in the plots has surpassed that of the oldest trees (>300 years) in amount of carbon stored. In addition, the total amount of carbon stored in the youngest trees is most likely much higher than calculated here since the smallest diameter trees that were measured and aged were 10 cm. The significant contribution by small diameter trees to total carbon suggests that the high densities of seedlings (50 cm – 1.4 m height) (1376 ha^{-1}) and saplings (1.4 m height – 10 cm dbh) (1003 ha^{-1}) present in the plots could have a noticeable effect on the total carbon stored.

However, on an individual tree basis, the greatest amount of carbon is stored in the largest trees. This is to be expected since carbon content is a function of total biomass which in turn is a function of tree size. At the same time the oldest trees contribute the largest fraction of carbon per tree. However, when the carbon contribution of dead material is considered, an interesting pattern develops. The fraction of carbon stored in individual dead trees is a significant proportion of several of the largest size-classes in the plots. In SFM one-third of the carbon for 180 cm trees in 2002 was contained in dead trees, while in BOF two-thirds of the carbon for trees >200 cm dbh was contained in dead trees. In contrast, the fraction of carbon stored in dead trees was distributed more evenly across age-classes. This suggests that while there is increasing mortality among the largest individuals, they are not necessarily the oldest individuals in the plots. In addition, the carbon contained in the dead trees is slowly released back to the atmosphere as the wood decomposes. As this forest grows into the future and the younger trees continue to increase in size, they will contribute a greater proportion of the total carbon storage present in the plots. Consequently, the distribution of stored carbon may shift to a greater concentration in the larger trees. However, the increased mortality among the largest trees implies that the greatest release of stored carbon will also occur in this group, which may offset the carbon sequestration in the future.

4.4.3 Comparisons with other studies

Total tree carbon in the study areas was greater than that measured in most other forests in the western U.S. (Table 4.5). The only other studies that found very high comparable tree carbon values were from old-growth forests in the Pacific Northwest that have not experienced a frequent disturbance regime. Due to the lack of disturbances, estimates from these forests suggest what are the potential upper limits of carbon storage in forests in the western U.S. (Smithwick et al. 2002). The few ponderosa pine sites from the Pacific Northwest with low levels of total tree carbon are all found on the east side of the Cascades (Table 4.5) (Gholz 1982, Law et al. 2001, Smithwick et al. 2002), a drier

Table 4.5: Comparison of total tree carbon (Mg C/ha) and coarse woody debris (CWD) carbon (Mg C/ha) found in literature from other studies of conifer forests of North America with old-growth, mixed conifer forests in Yosemite National Park, CA.

Forest Type	Location	Study	Forest Age	Tree Carbon (Mg C/ha)	CWD Carbon (Mg C/ha)
Tsuga - Picea	OR Coast	Smithwick et al 2002	150	598	164
Pseodotsuga - Tsuga	OR Cascades	Means et al 1992	450	587	207
Pseodotsuga - Tsuga	OR Cascades	Smithwick et al 2002	450	557	150
Mixed Conifer	**Sierra Nevada, CA**	**BOF, this study 2002**	**old-growth**	**526**	**164**
Abies - Pseudotsuga - Thuja	WA Cascades	Smithwick et al 2002	300-1200	480	158
Tsuga - Picea	WA Coast	Smithwick et al 2002	122-250	479	146
Mixed Conifer	**Sierra Nevada, CA**	**SFM, this study 2002**	**old-growth**	**460**	**146**
Abies - Pseudotsuga	Oregon	Fujimori et al 1976	100-130	440	
Tsuga - Picea	OR Coast	Fujimori et al 1976	100-120	436	
Pseodotsuga - Tsuga	OR Cascades	Grier and Logan 1977	450	435	133
Abies - Pseudotsuga	WA	Fujimori et al 1976	310	422	
Pseodotsuga - Tsuga	OR Cascades	Fujimori et al 1976	500	331	
Pseodotsuga - Tsuga	OR Cascades	Fujimori et al 1976	90-110	331	
Temperate Conifer		Vitousek et al 1988	old-growth	312	
Abies Amabilis	WA Cascades	Grier et al 1981	180	293	195
Tsuga canadensis - Pinus strobus	WI	Crow 1978	225	286	
Psedotsuga menziesii	WA	Keyes and Grier 1981	40	278	
Mixed Conifer	**Sierra Nevada, CA**	**BOF, this study 1899**	**old-growth**	**223**	
Mixed Conifer	**Sierra Nevada, CA**	**SFM, this study 1899**	**old-growth**	**167**	
Tsuga mertensiana	OR	Boone et al 1993	400	158	72
Pinus Ponderosa	E Oregon	Smithwick et al 2002	300-500	113	45
Ponderosa pine	OR Cascades	Law et al 2001	old-growth	108	
Picea englemannii - Abies lasiocarpa	Alberta	Prescott et al 1989	350	102	
Ponderosa pine	OR Cascades	Gholz 1982	old-growth	68	
Ponderosa pine	Front Range, CO	Hicke et al 2004	100-300	24.5	

environment, with less productive forests than those in Yosemite National Park. In contrast, most other studies of conifer forests in the U.S. outside the Pacific Northwest found dramatically lower levels of total tree carbon that were comparable to, or less than, the amounts contained in the 1899 forests in Yosemite National Park. Compared to the old-growth forests in the Pacific Northwest, the comparable high tree carbon totals from this study suggests that the mixed conifer forests in Yosemite National Park may be near their upper limits of carbon sequestration. Consequently, as these forests continue to grow into the future, the increase in carbon stored in the younger trees may become offset by the increasing number of older trees that begin to die, subsequently releasing their carbon through decomposition, resulting in a stable level of carbon storage over time. However, this is assuming that the forests persist in the current condition. Based upon the fire history of the study areas (see chapter 2), fire has historically been a part of the mixed conifer forests in Yosemite National Park, and the current high biomass in the forests is the result of management policies focused on the exclusion of fire from the landscape for the past 100 years. Under a regime of more frequent fire occurrence, the carbon stored in the forests would be less than the levels currently measured. In this case, the total tree carbon levels estimated for the 1899 reconstructed forests may be a more accurate indicator of carbon storage in conifer forests experiencing a frequent disturbance regime. In addition, the 1899 carbon levels are based upon a fire regime and vegetation dynamics that existed under certain climate conditions. Potential changes in climate conditions in the future, however, could lead to changes in the fire regime, which would alter the structure and vegetation dynamics of the forest, and subsequently, could alter the levels of carbon sequestration from the current estimates.

My accumulation rate of carbon is greater than that found in other conifer forests experiencing infilling due to fire suppression (Table 4.6). I estimated annual accumulation rates of carbon at 2.8-2.9 Mg C/ha year^{-1} which was above the upper limits of that estimated for ponderosa pine forests of the western U.S. (2.0-2.5 Mg C/ha year^{-1}). While my rate may be higher than all other listed accumulation rates, my estimates may also be more precise because they are based on the total increase in carbon since the onset of fire suppression. Most other estimates of accumulation rates are derived from measurements taken over a couple of decades, and often begin over 50 years after the onset of fire suppression. Consequently, the rates derived from other studies may be estimates for forests that have already experienced a large degree of infilling and their current growth rates, and carbon accumulation rates, may be slowing down.

4.4.4 Conclusions

Detailed forest measurements of mixed conifer forests in Yosemite National park allowed me to calculate the carbon content of the contemporary forest and also the pre-fire suppression forest in 1899. The estimates enabled me to determine the allocation of carbon within the forest according to tree components, and also by age-class, size-class and species. The result of this

Table 4.6: Comparison of carbon accumulation rates (Mg C/ha $year^{-1}$) found in literature from other studies of conifer forests in North America with old-growth, mixed conifer forests in Yosemite National Park, CA.

Forest Type	Location	Study	Method	Accumulation Rate (Mg C/ha $year^{-1}$)	Notes
Mixed Conifer	Sierra Nevada, CA	BOF, this study	Field Measurements	2.9	1899-2002
Mixed Conifer	Sierra Nevada, CA	SFM, this study	Field Measurements	2.8	1899-2002
Ponderosa Pine	Western US	Houghton et al 2000	Modeling & Field	2.0-2.5	Infilling
white spruce	Quebec	Tremblay et al 2006	Field Measurements	2	50 years after logging
Oak Savanna	Minnesota	Tilman et al 2000	Field Measurements	1.8	soil included
All	Coterminous US	Turner et al 1995	Forest Inventory	1.5	
All	US	Houghton et al 1999	Modeling & Field	1.4	Infilling
All	Coterminous US	Birdsey 1992	Forest Inventory	1.4	
All	US	Pacala et al 2001	Firest Inventory	0.53	
Ponderosa pine	Oregon	Law et al 2001	Field Measurements	0.39	regrowth after logging
All	Colorado	Birdsey 1992	Forest Inventory	0-0.89	
Ponderosa Pine	Front Range, CO	Hicke et al 2004	Field Measurements	0.1-0.7	1980-2001

work was that I was able to determine how structural changes in the forest since the onset of fire suppression influenced carbon sequestration.

My contemporary total tree carbon values were dramatically different from those derived for other conifer forests in the western U.S. In addition, the carbon distribution within the forest changed over time. Initial infilling of the forest resulted in accumulation of carbon comparable to that accumulated in existing trees as they increased in diameter. However, increases in stem diameter contributed a significant amount to the total increase in carbon in the forest. At the same time, increases in mortality of larger trees will result in losses of carbon from the system. The high variability of carbon amounts in the plots and between my study and other estimates, suggests that there is a high degree of variability in carbon sequestration in mixed conifer forests, and consequently, more localized studies are needed to develop a better understanding of the full impact of fire suppression on carbon stores in forests in the western U.S. In addition, studies that estimate the natural range of variability in both the climate and vegetation structure will help to estimate the potential range of carbon sequestration possible under future climate conditions and disturbance regimes.

Chapter 5

Historical Analysis of Fire and Vegetation Dynamics in Mixed Conifer Forests of Yosemite National Park

5.1 SUMMARY OF FINDINGS

Extensive research into mixed conifer forests in the Sierra Nevada has identified fire as an important component in the vegetation dynamics of the forests. Unfortunately, few studies have quantified the relationship between fire and vegetation dynamics beyond identifying the frequent occurrence of fire in historic forests. This understanding has been further complicated by 100 years of fire exclusion which has resulted in forests that are compositionally and structurally different from their historic counterparts. Consequently, our understanding of the vegetation dynamics of historic mixed conifer forests is limited. This dissertation is a first attempt to develop a detailed reconstruction of the mixed conifer forests in the Sierra Nevada in an effort to understand the regeneration dynamics of the forest and their relationship to frequent fire at both stand and landscape scales. In addition, comparing the reconstructed forest to the contemporary forest allows the analysis of the impact of the removal of a regular disturbance (fire) on the forest and its impact on carbon sequestration.

At the stand scale, the structure and dynamics of the mixed conifer forests in Yosemite National Park were quantified through the mapping and aging of vegetation plots and the development of a fire history for the study areas. The cluster analysis of sample plots identified four different compositional groups according to species composition and structure in each study area. Although the vegetation plots in each study area were distributed across all slope aspects and elevations, there was little variation in the distribution of compositional groups according to environmental variation. The only environmental factor related to the variation in vegetation groups was the potential moisture (TRMI) of each plot. In addition, there was no variation in the fire history by forest compositional groups, or by slope aspect. However, the tendency of fires not to burn the same location in consecutive fires, suggests a self-organizing process at work in the forest related to fire occurrence.

At the landscape scale, the fire history pattern of the study areas becomes more apparent. The majority of the fires were relatively small in size (<250 ha), although there were periodic widespread fires that burned across both drainages. In addition, the majority of fires tended to burn locations not burned in the previous fire. This pattern has most likely led to the development of even-aged patches in the forest that were identified by the spatial analysis of tree ages for the reference and contemporary forest. This tendency of fire not to burn the same location consecutively has led to a forest structure similar to the shifting mosaic which has been suggested in other mixed conifer forests. However, in the mixed conifer forests in Yosemite National Park, the patches tend to be more heterogeneous in ages and not as clearly defined as in other forests. This is partially due to the tendency of fires to be of low to moderate severity which allows the majority of the canopy trees to survive any given fire resulting in a mosaic of overlapping eve-aged patches.

The structure and composition of the forests today are dramatically different from the time of last fire (1899). Density and diameter of contemporary trees increased significantly while the composition of the forests is shifting from predominantly fire tolerant species to predominantly fire intolerant. The canopy of the forest is still predominantly composed of fire tolerant species, while trees younger than 100 years old are mostly fire intolerant. In addition, the spatial distribution of trees has shifted in the contemporary forests. There are fewer patches of similarly aged trees present in the forest, while the cluster of stems has increased. Both features are a result of continuous infilling by younger individuals.

To more accurately understand the variability of fire regimes it is also necessary to identify the relationship between fire occurrence and other factors that drive fire regimes. This research indicates that variation in fire extent and occurrence is strongly related to variation in both local and hemispheric climate factors, at both interannual and interdecadal scales. The high frequency and synchrony of fires between both study areas suggest that fire occurrence is not strictly controlled by dry conditions in the region. Instead, the occurrence of fire is more strongly driven by the presence of lightning strikes and the time since last fire. However, widespread fires are strongly related to drought conditions, where the dry conditions result in dry fuels which increases the connectivity of fuels across the study areas. The variation in moisture across the region is related to variations in hemispheric circulation patterns, specifically ENSO and PDO, resulting in associations between their extreme phases and fire extent. However, fire regimes were not only influenced by climatic variations in the tropical and extratropical Pacific. The AMO, a multidecadal circulation pattern in the North Atlantic, appears to influence the relationship between ENSO, PDO and fire extent in the mixed conifer forests in Yosemite National Park. Consequently, patterns of forest structure may be influenced over long time periods by the phase of the AMO: centuries with mostly positive AMO experience fewer widespread fires than centuries with mostly negative AMO and therefore, may develop more regeneration patches in the forest, resulting in a more heterogeneous structure. In contrast, centuries with mostly negative AMO experience more widespread fires and, therefore, may be more homogeneous in regeneration patterns due to widespread synchroneity in regeneration and mortality across the study areas. As a result, the strength of the patch mosaic structure of the mixed conifer forests may be more closely related to long term climate patterns and the influence they have on fire regimes than previously thought.

The historical analysis of the mixed conifer forest also allowed an analysis of carbon sequestration as a result of fire suppression. Global carbon cycling, and in particular carbon sequestration, of atmospheric carbon is a topic of great concern due to increasing levels of CO_2 in the atmosphere and its connection to atmospheric warming, and forested landscapes are considered to be a sink for CO_2. However, estimates of carbon storage in western forests are derived primarily from studies of previously logged landscapes, with educated guesses as to the influence of fire suppression. The results of this study allow us to derive more accurate estimates of the amount of carbon sequestered due to fire suppression through the reconstruction of the historic carbon levels in the forest. In addition, the reconstruction enables us to determine the amount of carbon sequestration due to infilling versus the increase in diameter of existing stems. The resulting values may be important in re-evaluating the amount of forest carbon sequestration due to infilling, and the potential total carbon stored in forests due to fire exclusion. However, the potential carbon stored in the forests may be offset by the increased risk of high severity fires due to the increased fuel present in forests as a result of fire suppression.

5.2 CONCLUSION

Overall, the results of this study have demonstrated the importance of historical analysis in order to more accurately understand the relationship between the disturbance regime and vegetation dynamics under changing climate and environmental conditions (fire exclusion) within the mixed conifer forests of the Sierra Nevada. The detailed reconstruction enabled a more precise quantification of the historic forest structure and also a better understanding of the role played by fire in the patterning of the forest. The wide variation in density and composition of the study plots is a result of the high fire frequency and small fire sizes in the forests. However, the low overall structural diversity of the two areas may be related to the occurrence of widespread fires. Although widespread fires would promote a greater degree of homogeneity in the forest, the relatively low severity of the fires enabled the majority of the pre-existing trees to survive burning. The result is a forest that demonstrated a wide range of variation at the plot level but was relatively similar at the landscape level.

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EDUCATION

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1999 **M.S.**, The Pennsylvania State University, Department of Geography. Thesis: *"The structure and dynamics of an old-growth red fir (Abies magnifica A. Murr.)– western white pine (Pinus monticola Dougl. ex D. Don.) forest in the Carson Range, Nevada, USA."*

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TEACHING EXPERIENCE

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PUBLICATIONS

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