DISSERTATION

DISENTANGLING FIRE, CLIMATE, FOREST STRUCTURE, AND LAND-USE HISTORY INTERACTIONS IN MEXICO'S NORTHERN SIERRA MADRE

Submitted by

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ABSTRACT

DISENTANGLING FIRE, CLIMATE, FOREST STRUCTURE, AND LAND-USE HISTORY INTERACTIONS IN MEXICO'S NORTHERN SIERRA MADRE

The 20^{th} century was a period of profound changes in climate, land-use, forest structure, and fires throughout much of western North America and few montane forests continue to function under historical influences of climate variations and uninterrupted fire regimes. Yet, if we are to manage for resilient forests, understanding these linkages is critical and will depend on both pre-1900 and 20^{th} century observations. My research takes advantage of a unique opportunity in northern Mexico to study forest and fire dynamics before a century of fire exclusion. My research documented a shift in climate – fire relationships in the late 19^{th} century toward an overwhelming importance of antecedent moisture, unlike that seen previously for > 200 years. Tree recruitment peaks were tied to local processes, not broad-scale climate conditions. Antecedent wet conditions that promote fire occurrence suggests that in arid regions of the Southwest, anomalously wet years, still functioning under frequent fire occurrence, may further limit tree recruitment. The importance of fire induced mortality in shaping stand structure underscores the spatial variability of forests and helps explain even-age patches in forests as an artifact of patch survival of seedlings that recruit into the overstory.

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INTRODUCTION

The 20th century was a period of profound changes in climate, land-use, forest structure, and fires throughout most of western North America. Widespread landscape changes were already prevalent by the early 20th century (Leopold 1924) following fire exclusion through extensive overgrazing, fire suppression, and other land-use impacts that accompanied intensive Euro-American settlement in this region. Frequent surface fires were largely eliminated by 1880 in lower-elevation conifer forests, fundamentally altering stand dynamics and fire regimes (Weaver 1951, Covington and Moore 1994, Swetnam et al. 2001). Subsequent increases in stand density and fuel loads, as well as recent increased fire season length, have resulted in increasingly large and severe forest fires in the late 20th century (Westerling et al. 2006). Climate variability, both inter and intra-annual, has also been great, including extensive periods of drought, which have been linked to many of these large fires. Changes have been so pervasive in the last 100 years that few places in western North America continue to function under historical influences of climate variations and uninterrupted fire regimes (Stephens and Fulé 2005). Yet, if we are to manage for resilient forests, understanding linkages among climate, land-use, forest structure, and fire occurrence is critical and will depend on both pre-1900 and 20th century observations (Grissino-Mayer and Swetnam 2000).

Land practitioners are now faced with the daunting task of managing severely disrupted systems, under increasingly variable climate, with little representation of naturally functioning frequent fire forest ecosystems to serve as models for restoring degraded forests. However, in parts of northern Mexico, where intensive livestock grazing did not occur until post-revolutionary land reforms in the mid-twentieth century and fire suppression remains only marginally effective, frequent surface fires continued to occur long after such fires were eliminated from otherwise similar forests on the U.S. side of the border (Dieterich 1983, Baisan and Swetnam 1995). The region of northern Mexico along the international border is an area of high biological diversity, species endemism, and unique history. In fact, the unique human history is, in part, what likely kept land-use impacts from mirroring those on the U.S. side of the border and elsewhere in Mexico. The fear of Apache Indians that occupied this region into the 20th century, well after Geronimo's final surrender in 1886 (Leopold 1937, Seklecki et al. 1996, Goodwin 2002), kept intensive Euro-American settlement out of this region longer than other areas of Mexico and the U.S. Whether due to fear of Apaches or from other factors, the divergent land-use history on either side of the border was apparent well into the 20th century, especially at higher elevations where much of the forested area occurs.

In his foundational cross border comparison work in the mid 20th century, Marshall (1957, 1963) stated that the high elevation forests in the Sierra San Luis (my study area in northern Mexico) were unlogged with practically no grazing, and with abundant evidence of fires. This was in stark contrast to the U.S. side of the border. Aldo Leopold, 20 years prior to Marshall, was also well aware of this divergent history. Upon visiting the northern Sierra Madre he wrote,

> The Sierra Madre [in northern Mexico] offers us the chance to describe, and define, in actual ecological measurements, the lineaments and physiology of an unspoiled mountain landscape. On our side of the line we have few or no natural samples left to measure. I can see here the opportunity for a great international research enterprise which will explain our own history and enlighten the joint task of profiting by its mistakes (Leopold 1937).

The striking differences in land conditions were of course not missed by Leopold, nor were the opportunities to study these differences and learn from them. Leopold defined the northern Sierra as a wilderness laboratory of world-wide interest where conservation could be given “a full and fair test” (Leopold 1937). A “full and fair test” was an opportunity to figure out conservation before intensive settlement, the opposite of the Southwest U.S. where conservation followed deliberate, and successful, attempts to settle the west. Frederick Jackson Turner (1893) argued that settling the frontier was the core defining quality of the U.S. The closing of the frontier following the 1890 census however prompted an equally significant era of public lands conservation. If the frontier indeed defined us as Americans and shaped our conservation history, the lack of a frontier in a longer and often more convoluted history equally shaped Mexico and its own conservation era. In Mexico, a different view of conservation that favored civic projects in general, rather than the preservation of isolated wild spaces like the U.S. system, is rooted in keeping post revolutionary land reform promises that protected areas would not be zones of dispossession (Wakild 2007). Currently, Mexico is unique in that much of the nation’s forests were placed in the hands of communities, in successive degrees of control, beginning in the early decades of the 20th century, a result of the Mexican Revolution (Bray and Merino- Pérez 2002). Mexico contains only half of the global average of terrestrial reserves (Brandon et al. 2005), while 80% of forests are owned at the community level (Klooster and Masera 2000).

In the three decades following Leopold’s experience in the northern Sierra, Mexico’s population more than doubled and included a logging boom to support U.S. timber needs in WWII (Forbes 2004). While the road building Leopold feared has progressed, much of the high country of the northern Sierra remains fairly inaccessible, with a lack of infrastructure limiting timber harvest (Bray and Merino Pérez 2002). Likewise, land-use histories in Mexico are different than in the southwest U.S. and still retain great value in cross border ecological comparisons. There are still forests in northern Mexico, for example, that can serve as reference conditions for largely uninterrupted fire regimes. This includes the least interrupted surface fire regime reported to date in North America (Fule et al. 2011), something practically without analogue north of the border. While some of the defining characteristics and driving factors of Leopold's "full and fair test" have changed, it is not lost. The opportunity to understand linkages among climate, land-use, forest structure, and fire occurrence remains, and is a problem of regional significance as Mexico's northern Sierra and the U.S. southwest share more than a border. Today, northern Mexico presents an opportunity to study forest and fire dynamics *before* a century of fire suppression has fundamentally altered these dynamics.

My dissertation work takes advantage of these opportunities and in three chapters describes: (1) what Leopold found in his transformative experience in Mexico's Sierra and how it relates to conservation in the U.S. – Mexico borderlands; (2) changes in climate – fire interactions over time including how humans activities have modified fire occurrence and climate relationships; (3) the complexity of climate-disturbance-tree recruitment relationships particularly in relation to events such as the 1900s recruitment pulse and the 1950s drought to further try and disentangle climate and anthropogenic effects on recruitment.

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CHAPTER 1

ALDO LEOPOLD'S "FULL AND FAIR TEST": REVISITING CONSERVATION STRATEGIES ON THE U.S. – MEXICO BORDER

Summary

Aldo Leopold's transformation in attitudes toward wilderness, predators, fire suppression, and land health can partly be attributed to his experiences in a remote Mexican wilderness in the mid-1930s. This period coincided with Mexico's conservation era and a push by the United States Government for border parks and conservation areas. Mexico had little use for remote and roadless conservation areas (the U.S. model), instead favoring watershed protection, parks, and community development projects near population centers. This was due in part to post-revolutionary land reforms in Mexico aimed at avoiding dispossession of people, and instead intended to return Mexicans to the land through establishment of community-owned ejidos.

Leopold, however, was most interested in conducting research in Mexico's remote northern wilderness region as a way to better understand the mechanisms of land function, proposing to use the Rio Gavilan area of the Sierra Madre Occidental as a base or "norm" in a continent-wide study of land health. Although Leopold proposed some form of legal protection in the Rio Gavilan, his land health research goals in general fit with Mexico's larger post-revolutionary objective of conservation without dispossession of inhabitants. Conservation in developing nations that incorporates rather than displaces

rural people is relevant today. Likewise, a need for understanding aspects of land-health within working landscapes is increasingly important as informal wilderness areas disappear and an estimated 60% of land globally is in a degraded state.

This article explores Aldo Leopold's vision for international land health research along the U.S. – Mexico border in the mid-1930s as compared to the predominant conservation paradigms between the two countries in the 1930s. It also explores how that era of conservation is relevant to contemporary issues. I propose that Mexico is well suited to be a global leader in community-based conservation efforts that promote healthy land and human communities. Leopold's cross-boundary conservation vision may be even more relevant today with recent changes in global initiatives that align with Mexico's model of conservation, emphasizing community land tenure along with conservation.

Introduction

Aldo Leopold is considered one of America's greatest conservationists and a pioneer in the fields of restoration ecology, wildlife ecology, and environmental ethics. He is also noted for his contributions to the wilderness movement, through cofounding The Wilderness Society and also establishing Wilderness as an official management designation some four decades before the US Federal Wilderness Act (Meine 1988). What is perhaps less known is the significance of Mexico's wilderness in shaping Leopold's thinking on conservation issues, including Wilderness. After trips in 1936 and 1937/38 to Mexico's northern Sierra Madre, he recognized it as a rare example of land, similar to the U.S. Southwest, but in "perfect aboriginal health" (Fig.1.1). Thus he defined the Rio Gavilan as a wilderness laboratory of world-wide interest where conservation could be given “a full and fair test” (Leopold 1937). A “full and fair test” was an opportunity to evaluate conservation *before* intensive settlement, the opposite of the U.S. Southwest where conservation followed deliberate, and successful, attempts to settle the west. Frederick Jackson Turner (1996) argued that settling the frontier was the core defining quality of the U.S. The closing of the frontier following the 1890 census however prompted an equally significant era of public lands conservation. If the frontier helped defined us as Americans and shape our conservation history, the lack of a well defined frontier in a longer and often more convoluted history equally shaped Mexico and its own conservation era.

In Mexico, a view of conservation that favored civic projects emerged, rather than the preservation of wild spaces, and was rooted in keeping post-revolutionary land reform promises that protected areas would not be zones of dispossession (Wakild 2007). Currently, Mexico is unique globally in that much of the nation’s forests are placed in the hands of communities, largely through the ejido system. This process began in the early decades of the 20th century, a result of the Mexican Revolution (Bray and Merino- Pérez 2002).

In the U.S. our collective focus turned to public lands conservation, which created tensions between utilization of resources under U.S. Forest Service doctrine of “wise use” versus nature preservation, as practiced by the National Park Service (NPS). Leopold was at the forefront of both scientific utilization of resources and wilderness preservation, though in 1935, the same year he co-founded The Wilderness Society, he wrote “this impulse to save wild remnants is always, I think, the forerunner of the more important and complex task of mixing a degree of wildness with utility" (Leopold 1999).

For Leopold the opportunity in Mexico lay not in unobstructed land acquisition for wilderness preservation, but in the ability to decipher the mechanisms underlying functioning land as a way to figure out the more complex task of mixing wildness with utility. He defined conservation as "the attempt to understand the interactions of these components of land [soils, water systems, and wild and domesticated plants and animals] to guide their collective behavior under human dominance" (Leopold 1999). Likewise, he had begun to argue that professional conservationists could integrate a degree of wildness into the mosaic of working lands (cultivated fields, pastures, woodlots, and wetlands), what he would eventually call land health (Callicott 2000). It was in Mexico's Rio Gavilan that Leopold first identified "land health" (Fig. 1.2). He wanted to learn from it and did not wish to leave its future to chance alone. A key component of land health is land-use and recognition of the ability of land to restore productivity. Thus, for Leopold a protected wilderness in Mexico was a means to a different end than simply preservation.

Today, more than seven decades after Leopold encountered Mexico's Sierra Madre wilderness, recent reviews of conservation efforts in developing countries have often demonstrated mixed results. In Latin America, for example, human population growth rates near protected areas are nearly double those of surrounding rural areas, with growth rates mirroring international investment and creating new conservation challenges (Wittemyer et al. 2008). Similarly, fighting poverty and promoting sustainable development with community-based conservation has often failed (Leisher et al. 2010). The result in many cases has been increased poverty with heightened demands on natural resources (Leisher et al. 2010). The exception appears to be community-run forests which can protect biodiversity and alleviate poverty rather than marginalize or even criminalize those dependant on resource extraction (Pearce 2011). In Mexico, with only half of the global average of terrestrial reserves (Brandon et al. 2005), 80% of forests are owned at the community level (Klooster and Masera 2000), which creates unique conservation opportunities and challenges.

Our conservation problems have become increasingly global in nature (e.g. climate change mitigation), creating challenges that Leopold could not have imagined 70 years ago. However; with new challenges come new opportunities. While the Mexico wilderness Leopold observed in the 1930s is different today so too are the opportunities to apply “a full and fair test” of conservation. Mexico has been identified as a priority region globally for conservation, ranking among the top five countries of the world for endemism (Brandon et al. 2005). Mexico is also rich in cultural diversity which is inextricably linked to maintaining ecosystem function. It may be prudent as conservation efforts progress to consider a shared conservation history on the border of one of the richest countries in the world (U.S.) neighboring one where half of its citizens live in poverty many of which, particularly rural people, in extreme poverty (Mexico; Walton and Lopez-Acevedo 2005). What was it that Leopold found in Mexico’s Sierra Madre and how can the opportunities Leopold identified be applied to current conservation efforts?

The symphony of wilderness

Aldo Leopold was no stranger to the U.S. Southwest. He was first stationed in the Apache National Forest in 1909, in what was then the Arizona Territory (Meine 1988).

He spent the next fifteen years working in Arizona and New Mexico and would continue to visit and write about the Southwest throughout his life. Leopold was also acquainted with a magnificent Mexican wilderness, having explored the delta of the Colorado River by canoe in 1922. Of this experience he wrote "It is part of the wisdom never to revisit a wilderness, for the more golden the lily, the more certain that someone has gilded it" (Leopold 1949). Leopold was probably well aware that in the same month as his 1922 trip on the Colorado River, the Colorado Compact was signed into law allocating the river's water, laying the groundwork for Hoover Dam, and forever changing the course of the Southwest and Mexico. Yet fifteen years later he would visit another Mexico wilderness that would leave a different impression where he could also imagine a different future.

After making two hunting trips into the Rio Gavilan in the northern Sierra Madre (northwestern Chihuahua) in 1936 and 1937/38, Leopold wrote in an essay titled 'Song of the Gavilan'; "It was [in Chihuahua's Sierra Madre] that I first clearly realized that land is an organism, that all my life I had seen only sick land, whereas here was a biota still in perfect aboriginal health" (Leopold 1949). Perhaps the most poignant point in *Song of the Gavilan* is Leopold's criticism of reductionist thinking, using his own profession in academia, a professor of wildlife ecology, as a case in point. Leopold describes that we examine the construction of plants, animals, water and soils, the components of land, by each selecting one component and spending a lifetime dismembering it (Leopold 1949). He had himself studied the pieces, having written well ahead of his time on issues from fire suppression and soil erosion in the Southwest to game management and the importance of predators. He had also started working on watershed-scale restoration projects in Wisconsin (Meine 1988). In short, by the mid 1930's Leopold had a familiarity with the components of land, and had already begun thinking in a non-reductionist way when he happened upon a system where all the components were there in 'aboriginal health' –the Rio Gavilan, where he at last heard what he termed the full "symphony of wilderness".

Leopold was not one to mince words and his description of "aboriginal health" carries importance. One of the components of the Rio Gavilan that most impressed him was abundant evidence of centuries old, stone check dams or *trincheras*. The Rio Gavilan is reported to have the highest concentration of Paquime trincheras in the Sierra Madre (Howard and Griffiths 1966). The abundance of trincheras led Leopold to speculate that the mountain Paquime people must have "numbered in the thousands", a figure refuted as too large by others (see Forbes 2004). The Rio Gavilan had an abundance of game as well as large predators, it retained its soils, the river ran clear with native trout *and* it had a long history of previous human inhabitation. The Rio Gavilan, which as Leopold stated, came "near to being the cream of creation" was not in spite of people but because of people. This was the onset of one of Leopold's most profound concepts, land health, and a rethinking of wilderness values.

Leopold described land health as the "capacity for internal self-renewal" (Leopold 1941), a concept that was not yet scientifically defined and what ecologists call resilience today (Forbes 2004), but he intended to do just that. In an essay inspired by the Rio Gavilan, 'Wilderness as a Land Laboratory' he wrote: "A science of land health needs, first of all, a base-datum of normality, a picture of how healthy land maintains itself as an organism" (Leopold 1941). It follows that the first lines of this same essay also indicate his changing argument for the principal utility of wilderness, from one based on recreation and aesthetics, to understanding land function: “The recreational value of wilderness has been often and ably presented, but its scientific value is as yet but dimly understood. This is an attempt to set forth the need of wilderness as a base-datum for problems of land-health” (Leopold 1941).

By this point Leopold had been defining the value of wilderness for more than a quarter of a century, his definition evolving over time (Meine 1988, Callicott 2000). He had argued in favor of wilderness on esthetic, biological, recreational, and even in terms similar to Fredrick Jackson Turner’s “cultural” values (Meine 1988). Now he recognized wilderness as a “control against which we could measure our experiment in civilization” (Meine 1988). Much has been written on wilderness values (e.g., The Wilderness Debates; Nash 1967, Cronan 1995, Callicott and Nelson 1998) but ultimately what Leopold recognized was that we knew very little about how to use land without degrading it. Functioning lands like Mexico’s Rio Gavilan were an opportunity to practice conservation with a systems approach from the start rather than a reductive and mechanical approach he himself had once advocated (Callicott 2000). Such an approach acknowledges “wild and tame attributes, all built on a foundation of good health” (Leopold 1939). It follows that a dualistic view of wilderness (uninhabited by people vs. inhabited) or land in general (agrarian vs. wild) unnecessarily divides spaces (Agrawal and Sivaramakrishnan 2000) and retards an ever changing relationship with land and its conservation. Thus, Leopold had begun to define wilderness broadly and with regard to its relationship to land health, with land-use central to maintaining, as well as helping define, this relationship.

Losing quality to gain quantity

At the same time as Leopold's trips into the Sierra Madre, Mexico was undergoing a conservation/protected area boom under the Lázaro Cárdenas administration, beginning with two national parks in 1935 and forty by 1940 (Wakild 2009). During this bonanza of conservation area creation in Mexico, U.S. diplomats became interested in U.S. – Mexico border parks, following the example of Waterton-Glacier International Peace Park on the U.S. – Canada border. U.S. – Mexico border parks were viewed as a way to reaffirm friendly relations with Latin American countries under Roosevelt's Good Neighbor Policy (Wakild 2009). Just four months after Leopold's first trip into the Sierra an international delegation toured potential conservation sites along the border, the crown jewel of which was a site at Big Bend (Wakild 2009). Leopold was aware of these efforts though he had something different in mind. In 1937 he wrote;

> "With the extension of roads, recreation so-called will of course repeat the now familiar process of losing quality as it gains in quantity of human service. Mexican citizens protest that they are going strong on National Parks and Forests. They are particularly proud of the International Park at Big Bend. They do not realize that these devices laudable and necessary as they are, have not exempted us from the inexorable process of losing quality to gain quantity" (Leopold 1937).

After visiting the Rio Gavilan Leopold recognized that the U.S. system of public lands lacked functioning healthy land while Mexico had a wilderness unlike any he had ever seen. This comparison certainly left its mark. He wrote "For it is ironical that Chihuahua, with a history and terrain so strikingly similar to southern New Mexico and Arizona should present so lovely a picture of ecological health, whereas our own states, plastered as they are with National Forests, National Parks and all the other trappings of conservation, are so badly damaged that only tourists and others ecologically color-blind, can look upon them without a feeling of sadness and regret" (Leopold 1937).

Interestingly, the Cárdenas Administration under the guidance of Miguel Ángel de Quevedo, who headed the new (1935) Division of Forestry, also had something else in mind. Because of this different vision, despite more than seven decades of interest in border parks they have largely not come to fruition. As Emily Wakild (2009) points out, the idea of international U.S. – Mexico border parks fell apart due to a generally different view of conservation and post revolutionary circumstances. Quevedo favored central urban parks, reforestation of watersheds adjacent to population centers, and civic projects in general, rather than the preservation of isolated wild spaces like the U.S. system. Quevedo was particularly interested in flood control (Simonian 1995) and most parks were designed to alleviate erosion from deforestation and alteration of hydrological cycles in and around Mexico's central valley and Mexico City (Simonian 1995). Inaccessible wild lands were of little use to Mexicans. As Wakild also points out, a different view of conservation in Mexico is rooted in keeping post revolutionary land reform promises that parks would not be zones of dispossession.

Mexico's model of conservation areas not becoming zones of dispossession has gained traction globally. Consider the United Nations Educational, Scientific and Cultural Organization's (UNESCO) Man and the Biosphere Programme (MAB) created in 1970, which is an intergovernmental scientific program designed to set a scientific basis for the improvement of the relationships between people and their environment (UNESCO 1970). This program aims to involve local communities and incorporate socioeconomic issues in conservation and accounts for the majority of protected areas in Mexico.

Similarly, at the 1992 Rio Earth Summit it was widely recognized that conservation, particularly in developing regions, depended on fighting poverty and creating opportunity for those who directly depend on natural resources. Putting this into practice though has proven more difficult (Pearce 2011).

In recent decades we have seen dynamic changes within the U.S. system of conservation which has moved toward recognizing land-use and private lands as a part of the conservation fabric rather than merely a complicating factor for conservation area development. Examples include the proliferation of community collaborative conservation efforts, ecosystem management, and conservation easements (Knight and White 2009). Across the U.S., easements as a conservation tool are growing exponentially, accounting for 70% of land protected in a given year (Fishburn et al. 2009). While easements vary widely in their application, they represent a new way of conserving land, one that has moved away from a federally dominated "command and control" system while offering a new structure focused on maintaining some degree of local control.

A full and fair test

In a 1937 paper, "Conservationist in Mexico," Leopold wrote that the "Sierra Madre offers us the chance to describe, and define, in actual ecological measurements, the lineaments and physiology of an unspoiled mountain landscape". He goes on to say "I can see here the opportunity for a great international research enterprise which will explain our own history and enlighten the joint task of profiting by its mistakes". While particulars of what Leopold had envisioned for "an international research enterprise" are not entirely clear, several things do stand out.

In September 1936, the same month Leopold returned from his first trip into the Rio Gavilan, he wrote to a geographer at the University of Arizona, Byron Cummings, asking him to look over a problem he intended to pose to his students at the University of Wisconsin. The problem was what the numerous rock masonry dams, or *trincheras,* could have been used for and what crops could have persisted in the absence of intensive management with abundant game populations. Leopold also solicited help from a prominent geographer, Carl Sauer, at the University of California, Berkeley, who was working in northern Mexico at the time. Sauer's research focused on, among other things, long-term, two-way relationships of land and culture combining anthropology, geomorphology, and local interactions (Forbes 2004). While there is no evidence of responses to these inquiries, it is clear that Leopold wasted no time in trying to unravel the secrets of the Gavilan and immediately enlisted the help of geographers, his attention directed at human occupation of the landscape.

Leopold also recognized that the northern Sierra could help us understand mechanisms of a functioning landscape by deciphering soil-water-streamflow relations. In doing so a study of the ecology of flora and fauna would be necessary. Of particular interest were deer-wolf-coyote relations. In a December 1938 letter to Sauer, Leopold stated that the only deer range free from irruptions of deer which he was aware of was the Chihuahua Sierras. He also noted that deer were abundant (but not overabundant) in the presence of an intact predator guild and a conspicuous lack of coyotes in the high country. In the U.S. Southwest, coyotes were abundant at the same elevations in the absence of wolves.

Leopold prefaced what is now a classic debate in ecology concerning the organization of top-down versus bottom-up controls of trophic cascades. In a letter to Carl Sauer (1938) he states “All ecologists know, from both theory and observation, that plants determine animals, but the converse theorem (namely that animals determine plants) is ‘known’ in theory and laboratory only” (Unpublished). The idea of trophic cascades would only begin to gain attention more than two decades later following a now classic paper by Hairston et al. (1960) and it remains the center of debate among many ecologists. Such differing worldviews (top-down vs. bottom-up) clearly have repercussions for the way wilderness is viewed and managed today. NPS “natural regulation policy” (National Park Service 1967), for example, emphasizes eliminating human management actions in National Parks and assumes ungulate populations will “naturally” regulate in the absence of top level predators (bottom-up controls).

In 1941 Leopold responded to an Ecological Society of America committee soliciting ideas for reserves with natural conditions by suggesting “some international joint effort” such as the U.S. financing a research station if Mexico acquired land and provided suitable protection. The same year he wrote his most detailed proposal to the USDA Forest Service soliciting funding for a geographically wide ranging study of the mechanism of deer irruptions. With his proposal he included his 1941 essay “Wilderness as a Land Laboratory”, which reiterated the importance of wilderness as controls in the study of land health. The war years likely detracted from Leopold’s proposals to work on questions of land health in the northern Sierra Madre which went unfunded. He was scheduled to attend the Inter-American Conference on Conservation in Latin America, the first of its kind, in September 1948. His untimely death in April of that year prevented this.

Looking forward

Subsequently, the Mexican landscape has changed. From 1940-1970 Mexico's population grew from 20 million to 48 million and this era included a logging boom to support U.S. timber needs in World War Two (WWII, Forbes 2004). Leopold's son Starker, a student of Carl Sauer, returned to the Rio Gavilan shortly after his father's death to visit the camps he had accompanied his father to in 1937-38 (Fig. 1.3). Starker and a team from Berkeley's Museum of Vertebrate Zoology encountered 14 logging trucks and 12 lumber mills on the way into the Gavilan. The post WWII U.S. housing boom and lumber markets were closing in on the Gavilan. Starker wrote in an essay titled *Adios Gavilan*: "we knew then that instead of…renewed acquaintance with the wilderness we had come to witness its passing" (A.S. Leopold 1949).

However the "good neighborly act well worthy of international consideration" that Aldo Leopold identified is not entirely lost. While the road building he feared has progressed, some of the high country of the northern Sierra Madre remains fairly inaccessible, something that has hampered modern forestry practices and timber utilization; only 30% of the forests in Mexico's six main timber-producing states are physically accessible for harvesting (Bray and Merino Perez 2002). Likewise, land-use histories in Mexico are different from the U.S. Southwest and still retain great value in cross-border ecological comparisons. There are still forests in northern Mexico, for example, that can serve as reference conditions for largely uninterrupted fire regimes. This includes the least interrupted surface fire regime reported to date in North America

(Fulé et al. 2011), something practically without analogue north of the border. It is worth noting that the fire regime reported by Fulé et al. (2011) is one maintained by human land-use under indigenous resource management. Thus the opportunity to study differences in forest structure and processes *before* > 100 years of fire suppression remain, something first identified by Leopold in 1937.

In addition, Mexico's forests did not become zones of dispossession, but are a globally unique case in which much of the nation's forests were placed in the hands of communities (Bray and Merino- Pérez 2002). As of 2004, approximately 25% of forests in developing countries globally were owned by local communities and this is projected to double by 2020 (Scherr et al. 2004). Thus conservation and issues of land health, which include rather than exclude land-use, are inseparable. Today, Mexico's common property community-managed forests appear to be at a scale and level of maturity unmatched anywhere in the world, creating a national laboratory for studying the social and ecological benefits of local community managed forests (Bray and Merino-Pérez 2002).

Leopold recognized that "in our attempt to make conservation easy, we have made it trivial" (Leopold 1949). While the U.S. agenda of borderlands parks and protected areas in the mid 1930s was not in Mexico's best national interest, the opportunities for a cross border conservation vision may be more realistic now than ever as the need for local community conservation grows. Community conservation has many definitions and disparate examples but in developing countries often refers to local involvement in and surrounding protected areas. This raises the question as to what could have resulted from seven decades of scientific collaboration on issues of land health recognizing land-use not as a peripheral problem surrounding protected areas but central to conservation of land in general. Land tenure is a critical concern for conservation, determining the linkage between responsibility and authority over natural resources and incentives for sustainable use (Murphee 1996). It seems that the sciences of both human and ecological dimensions are useful tools to help establish responsibility and authority in communally owned lands and promote land health. *Ejidos*, rural communities with land held in common, are generally managed with some level of government control (Thomas and Betters 1998), which could offer technical support for such an approach. It follows that the idea of land health, which integrates wildness and land-use rather than relegating land to either commodity or wilderness, fits better with Mexico's conservation history and contemporary circumstances.

At small scales Mexico has seen an increase in successful community-based forest management arresting forest degradation and deforestation (Klooster and Masera 2000). Nationally however, forestry is a still a minor priority in Mexico and generally within the forestry sector government promotion of industrial plantations receives greater attention (Klooster and Masera 2000). Leopold recognized the shortcomings of government stating that "when we lay conservation in the lap of government, it will always do the things it can, even though they are not the things that most need doing" (Leopold 1942). Government can serve an important role by helping to link science to community-based conservation, something that has been successfully demonstrated, but needs to be developed to a greater extent in Mexico. It seems the land tenure structure in Mexico is ripe for such leadership and opportunity. Clearly our conservation challenges are not static and will require innovation. Ultimately what Leopold articulated was that conservation requires an ethical relationship with land. He wrote that “ecology is the science of communities, and the ecological conscience is therefore the ethics of community life” (Leopold 1947). While borders have always both united and divided, it appears that the border that divides the U.S and Mexico is well poised to exemplify a cross-boundary conservation vision that can unite us and be a global model for community conservation. An important first step would be to recognize common property systems as a valuable social and ecological context in which to build conservation partnerships centered on issues of land health.

Figure 1.1. Aldo Leopold bow hunting in the Rio Gavilan region in northwestern Chihuahua, Mexico 1937. Photo by A. Starker Leopold provided courtesy of the Aldo Leopold Foundation.

Figure 1.2. Photograph of the Rio Gavilan – Strawberry Park region from Aldo Leopold's journal ca. 1937 showing the intersection of land-uses (agriculture) and open structure pine forests. Photograph provided courtesy of Aldo Leopold Foundation.

Figure 1.3. Starker Leopold packing in the upper reaches of the Rio Gavilan in northwestern Chihuahua, Mexico 1948. Photograph provided courtesy of Aldo Leopold Foundation.

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CHAPTER 2

CHANGES IN INTERANNUAL CLIMATE – FIRE RELATIONSHIPS IN NORTHERN MEXICO

Summary

The 20th century was a period of profound changes in climate, land use, forest structure, and fires throughout western North America. While interest in climate-fire and land-use interactions has greatly expanded in the past decade, disentangling the effects of human and climate influences on fire regimes remains difficult due to pervasive land-use changes and fire exclusion throughout most of western North America by the middle to late 19th century. Yet, understanding linkages among climate, land-use, and fire remains paramount for both long-range forecasting and assessing forest responses to potentially novel future climate conditions. I investigated fire history in the Sierra San Luis of northern Mexico, an area with a spectrum of 20th century land-use changes from areas with little or no history of grazing or logging, to sites that were logged (ca. 1952-1954) and/or grazed (beginning in the early 1930's). Fire regimes closely reflected these differences. Fires were relatively frequent in all three sites until 1932, with the onset of grazing in two sites, but continued to burn in the ungrazed and unlogged site with the last widespread fire occurring in 2000. Composite Weibull median probability intervals ranged from 4.1 years (all fires) to 10.2 years (fires recorded on $\geq$ 25% of recording trees) across sites (45-63 ha in size) for the period from1728–2008. I found a change over time in the role of drought and antecedent conditions for fires. In the period from 1650–1886 (early period), drought was significantly related to fire years with no association with antecedent conditions, whereas from 1887–2003 (modern period) the role of drought diminished, in small fire years, or was not significant in large fire years. Antecedent conditions were significantly wet in both small and large fire years for the modern period. Fire frequency (number per decade) was inversely related to fire size (percent scarred) and percentage of scarring aligned with fluctuations in Palmer Drought Severity Index (PDSI), with higher percent scarring in wet periods and lower scarring in dry periods. Temporal changes in fire frequency for my sites were related to effects of interannual climate variation on fuel abundance. Historical climate – fire interactions are not consistent indicators of modern or future relationships with changing interannual climate variations on fire occurrence.

Introduction

Understanding the influences of interannual climate variability on wildfire occurrence remains a principal objective of fire history research (Kitzberger et al. 2007, Sherriff and Veblen 2008). Interannual moisture variability has important influences on tree regeneration and stand structure (Brown and Wu 2005), quantity and arrangement of fuels (Stephens 2004), and synchrony of fires in many pine dominated plant communities (Swetnam and Betancourt 1998, Fulé et al. 2012). Interannual wet-dry patterns also correspond to widespread fire years in the Southwest, which tend to occur during dry years following one or more wet years that stimulate fine fuel production (Baisan and Swetnam 1990). While this relationship has been widely observed in dry forest types throughout western North America (Sherriff and Veblen 2008) it is particularly pronounced in the arid Southwest (Westerling et al. 2003). A primary source of information for climate – fire dynamics are tree-ring methods, which provide detailed spatial and temporal records for long (>300 years) time periods (Veblen et al. 2003, Biondi et al. 2011). The U.S. Southwest has the longest and most detailed tree-ring records of climate and fire history in the world (Swetnam and Betancourt 1998), though generally fire records end in the mid to late 19th century due to nearly universal and effective fire exclusion, limiting their utility in exploring potential 20th century changes in climate – fire relationships.

Recently, 20th century interannual climate – fire relationships have been described using wildfire area burned data (Westerling 2003, Westerling et al. 2003, Crimmins and Comrie 2004, Littell et al. 2009). These data also have limitations; namely, a lack of site specificity (Littell et al. 2009), a lack of data prior to ca. 1980 (Littell et al. 2009), underreporting (Westerling et al. 2003, Crimmins and Comrie 2004), human intervention via suppression effects, and fires largely of human origin (Westerling et al. 2003). Interannual climate – wildfire area burned analyses have been informative however, demonstrating climatic controls including a strong association of antecedent wet conditions on fire occurrence in the desert southwest in the mid-to-late 20th century (Westerling 2003, Westerling et al. 2003, Crimmins and Comrie 2004, and Littell et al. 2009), though a lack of association between drought conditions and recent fire occurrence (Crimmins and Comrie 2004) seems to depart from paleo climate – fire relationships (Swetnam and Betancourt 1998). Wildfire area burned data for the Southwest have also been used successfully to statistically reconstruct or hind-cast historical area burned, indicating that climate – fire relationships have persisted even with modern land-use changes (Westerling and Swetnam 2003). Interestingly though, Westerling and Swetnam (2003) found greater correlation to drought (Palmer Drought Severity Index, PDSI) with statistical reconstructions of historical tree-ring data than for the modern documentary data used to train the models. Recently, Biondi et al. (2011) generated a hypothetical fire regime using a pyroclimatic model for the central Great Basin, Nevada, with assumed relationships between climate (PDSI) and wildfire occurrence in an effort both to predict past fire years and to project these relationships forward. Biondi et al. (2011) indicate that a decrease in fire occurrence after the mid 1800s has occurred primarily as a result of climate, not modern land-use changes (e.g. grazing and fire suppression). Furthermore, they identify the displacement of Native American tribes (and their use of fire) after the mid-1800s as another contributor to modern changes in fire regimes, implying that Native American ignition sources in this area largely shaped pre-European settlement fire-regimes (Biondi et al. 2011). This suggests that potential anthropogenic effects on fire occurrence should be considered in addition to past and present climate controls.

The role of anthropogenic influences on fire regimes is not limited to post-Euro-American settlement and associated land-use changes. The importance of Native Americans in modifying pre-Euro-American fire regimes has been emphasized particularly in the Southwest U.S. – Mexico borderlands region (Pyne 1982, Swetnam et al. 2001). Ample debate remains on the relative importance of changes in climate vs. human land-use, such as the change from Native American burning to Euro-American extinguishing, on reducing fire frequency in modern times (Betancourt et al. 1993, Biondi et al. 2011). Apache wartime periods, for example, have been proposed as a potential explanation of unusual back-to-back fire years, and overall higher fire frequency periods, unique to several U.S. – Mexico borderland mountains (Kaib et al. 1996, Morino 1996, Seklecki et al. 1996, Kaib 1998, Swetnam et al. 2001), although these influence occurrence in tandem with variation in climate drivers. The termination of fires in the Southwest U.S. – Mexico borderlands following the removal of the Apaches to reservations after 1886 (Seklecki et al. 1996) is consistent with Biondi et al.'s (2011) observations from the central Great Basin. However, to distinguish these effects from modern land-use history changes requires examining climate – fire relationships in conjunction with land-use changes for regions with different land-use history under similar climate controls. Disentangling the effects of human and climate influences on fire regimes is difficult due to pervasive land-use changes and fire exclusion throughout most of western North America by the middle to late 19th century. Yet, understanding linkages among climate and land-use remains paramount for both long-range forecasting systems and assessing forest responses to unique climate conditions that we may face in the future (Morgan et al. 2001).

In parts of northern Mexico, where intensive livestock grazing did not occur until post-revolutionary land reforms in the mid-twentieth century and fire suppression remains generally ineffective even today, frequent surface fires continued to occur long after they were eliminated from otherwise similar forests on the U.S. side of the border (Dieterich 1983, Baisan and Swetnam 1995, Fulé et al. 2011, Fulé et al. 2012). My research is focused on one such site which includes relict forests that have had little or no grazing, logging, or other intensive modern land-use history adjacent to an intensively studied region of the U.S. Southwest (Arizona and New Mexico) with a very different land-use history. My research area has the benefit of being centrally located in the former Chiricahua Apache territory, the international border serving important purposes in Apache raiding and warfare. The mountains of northern Mexico along the international border within and surrounding my study sites were the last refugia for Apaches decades after others were confined to reservations in the U.S. (Leopold 1937, Goodwin 2002). Thus, my research was well poised to address overlapping influences of both historical and more recent climate and human history on fire regimes. My research had two primary objectives: (1) to understand climate – fire interactions over time, particularly for the post-1886 modern period where there is a paucity of such data; and (2) to investigate the relative influence of climate and human activities on fire occurrence. I explicitly considered small or localized fire events relative to more extensive ones to help address these objectives, as a means to investigate potential human or climate controls and to try and understand what turns a small fire into a large event.

Methods

Study area

The Sierra San Luis in northern Mexico encompasses the highland region straddling Chihuahua and Sonora and is comprised of the larger canyons and mountain tops or sub-sierra land features of the northernmost extension of the Sierra Madre Occidental. The Sierra San Luis is part of an array of mountains that are referred to as the Madrean Sky Island Archipelago, which is the confluence of four biogeographical regions: the Rocky Mountains, the Sierra Madre Occidental, and the Sonoran and

Chihuahuan deserts. This convergence results in high biological diversity and species endemism and is part of the two richest floras of mega-Mexico – which ranks as one of the three top mega-diversity centers of the world (Felger and Wilson 1994). It is, however, the divergence in modern land-use history that creates unique opportunities to study ecological differences in Mexico. Aldo Leopold recognized this opportunity in 1937 which he pursued until his death in 1948 (Leopold 1937) as did Joe Marshall two decades later. In his foundational cross-border comparison work in the mid 20th century, Marshall (1957, 1963) states that the high elevation forests in the Sierra San Luis were unlogged with practically no grazing, and with abundant evidence of fires. This was in stark contrast to the U.S. side of the border (Leopold 1924, 1937, Marshall 1963). These differences were in part a result of long Apache occupation, followed by the decade-long Mexican revolution (1910-1920, although outbreaks of warfare continued until 1929) and subsequent unstable land policies (Sanderson 1984, Thompson and Wilson 1994, Heyerdahl and Alvarado 2003). The result was delayed modern land-use impacts (e.g., grazing, logging, mining) and in some cases relict forests that have largely escaped these land-uses (Fulé et al. 2012).

My study sites (Fig. 2.1) take advantage of different land-use histories and are comprised of the larger mountaintops of the Cajon Bonito watershed in the Sierra San Luis. They included three similar physiographic areas that were (1) never logged and had little grazing pressure (Sierra Pan Duro, SPD), (2) were grazed beginning in the early 1930's and logged from 1952 to 1954 (Pan Duro Arroyo, PDA), and (3) were unlogged but grazed in a similar time period (El Pinito Canyon, EPC). EPC had sharper relief and contained mostly lower elevation (~1,800-2,000 m) pine-oak forest limited to narrower confines of a major canyon (Junta de los Cajones), plus Douglas-fir (*Pseudotsuga menziesii*) stands at the highest elevations (~2,440 m). Dominant overstory trees in pine-oak forests included Chihuahua pine (*Pinus leiophylla*), Apache pine (*P. engelmannii*), and mixtures of Madrean oaks (*Quercus arizonica, Q. emoryi, Q. oblongifolia*). PDA and SPD had a greater extent of intermediary elevation (~2,000-2,200 m) ponderosa pine (*P. ponderosa var. arizonica*) forest and occasional southwestern white pine (*P. strobiformus*). In addition, extensive pinyon-juniper woodlands (mostly *Juniperus deppeana* and *P. cembroides*) were present at lower elevations across all three sites.

Northern Mexico lies close to the boundary between mid-latitude (westerly) and tropical (monsoonal) sources of moisture (Metcalfe et al. 1997), producing a strongly bimodal distribution of annual precipitation. Typically, winter precipitation in southeastern Arizona (December – March) accounts for 30%, and summer monsoons (July – September) 50%, of annual precipitation (Weltzin and McPherson 2000). The El Niño-Southern Oscillation (ENSO) accounts for a large source of annual variability in precipitation, with an average interval of 3 to 4 years between relatively cool wet El Niño winters with intervening warm, dry La Niña winters in the southwest (Sheppard et al. 2002). These general climate patterns (monsoon, ENSO) are strongly tied to southwestern seasonal and interannual precipitation and fire patterns (Swetnam 1990, Heyerdahl and Alverado 2003, Brown and Wu 2005).

Fire history

I reconstructed fire history along a 3.3 km stretch of EPC (~45 ha) and 3.0 km stretch of PDA (~63 ha) in 2008, and for eight stands (2 – 15 ha in size, 63 ha in total) within SPD in 2010. I sampled fire scars on trees (both living and dead) exhibiting multiple scarred (>3) “catfaces” within randomly located variable radius plots (Jonsson et al. 1992, Lessard et al. 2002). Plots were either spaced 500 meters apart (EPC and PDA) or were selected from a probability based spatially balanced design (Theobald et al. 2007, Theobald and Norman 2006) for delineable stands (SPD). I also sampled multiple scarred trees opportunistically surrounding my plots. I used non-destructive sampling methods (Heyerdahl and McKay 2008) to remove partial cross sections of both live and remnant fire-scarred trees.

In the laboratory, I sanded samples until the cellular structure of the xylem was clearly visible under magnification (Grissino-Mayer and Swetnam 2000). Samples were subsequently crossdated against master reference chronologies from the nearby Animas and Chiricahua mountains. All fire scars were then dated to their year of formation for cross-dated samples (Dieterich 1983, Dieterich and Swetnam 1984). When possible I determined intra-annual positions of fire scars (Baisan and Swetnam 1990) as an indication of seasonality of fires. I assigned ring-boundary scars to the year containing earlywood immediately after fire scars. Fire events in the Southwest after the cambial growing season ends are relatively rare, instead generally burning in the spring prior to monsoons (Baisan and Swetnam 1990, Fulé and Covington 1997, Heyerdahl and Alverado 2003).

I analyzed fire frequency data with FHX2 software, version 3.2 (Grissino-Mayer 2001). Analysis at each site was for the period of adequate sample depth, defined as the first fire year recorded by $\geq$10% of recording trees, until time of sampling or major disruption of fire events was apparent. Recording trees are those with basal injury leaving them susceptible to repeated scarring by fire (Swetnam and Baisan 1996). I used the same criteria for calculating statistics for the three sites combined (ALL). I calculated fire return intervals for sites (individually and combined) that included all fire years, and for fire years in which ≥10%, and ≥25% of recording samples were scarred. Filtering, based on scarring percentage, provides a meaningful relative index of fire size and is generally used to estimate more widespread or relatively large fire years (Swetnam and Baisan 1996) with 10% and 25% filters commonly reported filtering levels.

Climate- and human – fire interactions

To evaluate climate conditions related to fire occurrence, I used superposed epoch analysis (SEA) in FHX2 version 3.2 (Baisan and Swetnam 1990, Grissino-Mayer 2001). SEA was used to compare independently derived indices of drought (Palmer Drought Severity Index - PDSI) during fire years as well as for five years prior and two years following fire event years. I used the average of four nearest grid points surrounding the study area for North America Drought Atlas PDSI tree-ring reconstructions (Cook et al. 2004). Tree growth of mid-elevation forests typically responds to moisture availability. PDSI is a single variable that represents precipitation and to a lesser extent temperature (Sheppard et al. 2002). PDSI is a commonly used climate variable in paleo-precipitation studies and has been correlated to regional fire occurrence in the Southwest (Swetnam and Baisan 1996). I assessed statistical significance using SEA analyses with confidence levels (95% and 99%) calculated from bootstrapped distributions of PDSI data in 1000 iterations. I split the analysis into two periods 1650-1886, and 1887-2003. While dates of fire cessation vary among individual sites in the Southwest, 1886 was the last widespread fire year (six or more sites recording an event) in regional fire scar chronologies for 31 sites in ten mountain ranges in the Madrean Archipelago (Swetnam 2005). 1886 was also the year Geronimo and his band of Chiricahua Apaches surrendered to General Nelson Miles, effectively ending the Indian wars and prompting intensive settlement and land-use changes throughout the U.S. Southwest (Seklecki et al. 1996). However, fear of small bands of Apaches who continued to occupy the mountains of northern Mexico persisted well into the 20th century after Geronimo's surrender, and likely helped keep land-use impacts in Mexico from mirroring those on the U.S. side of the border (Leopold 1937, Seklecki et al. 1996, Goodwin 2002, Knight 2009).

I was also interested in fires of different size. Generally, filtering to discriminate between large and small fires (Van Horne and Fulé 2006, Farris et al. 2010) is intended to statistically eliminate the influence of small localized fires on estimates of fire intervals and thereby to identify large events for regional climate – fire associations (Swetnam and Brown 2010). I compared small fire events, defined as fires recorded on < 10% of recorder trees, with larger fires, recorded on ≥10% of recorder trees, in an effort to understand how the occurrence of fires of different size has changed over time and to elucidate climate – fire mechanisms that result in crossing thresholds between small and large events. Similarly, I examined climate relationships to years when fires did not burn. I also investigated climate relationships with small or localized fire events to test for competing explanations of human vs. climate controls of fire. This is particularly relevant to periods of frequent, heterogeneous fires that have been attributed to Indian burning (e.g., Fry and Stephens 2006).

Another way in which I measured fire frequency, while also enabling detection of changes over time, was by piecewise regression of cumulative chronological fire dates. Piecewise linear regression procedures (Neter et al. 1989) estimate the number of segments and break points (changes in slope) in fire frequency through time (Brown et al. 1999, Brown and Sieg 1999). I used Analysis of Covariance (ANCOVA) procedures to estimate when there were significant differences in regression slopes between periods. I analyzed climate – fire relationships for high and low slope segment periods and compared resulting segments to human history to examine human vs. climatic explanations for changes in fire frequency.

Results

Fire history

I sampled 194 fire-scarred trees and was able to crossdate 173 (89%) of these, yielding 954 scars in total. Fire-scarred trees were mostly ponderosa pine (55%), but included Douglas-fir (8%), and Chihuahua (16%), Apache (13%), pinyon (7%), and southwestern white pines (1%). About half of the fire-scarred trees I sampled were living (54%). The earliest fire scar identified in the three sites was 1654 and the most recent fire scars were from 2002. Fires burned relatively frequently until 1932 at EPC and PDA and continued to burn in SPD with the last widespread fire occurring in 2000 (Fig. 2.2). Weibull median probability intervals (WMPI) ranged from 5.8 (PDA, 1745-1932) to 10.0 (EPC, 1847-1933) years at individual sites (all fires). WMPI ranged from 7.6 (PDA) to 14.0 (EPC) years when data were filtered to include only fire years in which 25% or more of recording samples were scarred (Table 2.1). With sites combined, WMPI decreased to 4.1 years for all fires and 10.2 years for 25% filtered fire events (1728-2008). While intervals were similar between WMPI and Mean fire intervals (MFI), the Weibull model resulted in shorter return intervals and also generally fit the data well, though I report both to facilitate comparisons to other studies. WMPI is a less biased estimator of central tendency with skewed data and provides a standard way to compare fire regimes across ecological gradients (Grissino-Mayer 1999, Yocom et al. 2010). I was able to determine season of burning for 28% of fire scars (n = 270). Of those, 86% of scars were in the middle earlywood, meaning the majority of fires occurred in the late spring or early summer prior to summer monsoonal precipitation. Another 8% were in earlywood (1st 1/3 of ring), 4% were dormant season fires, followed by 1% each for late earlywood (last 1/3 of ring before latewood) and latewood fire scars.

Temporal changes in fire frequency are evident from slope changes in sequential fire dates for my sites (Fig. 2.3). With Piecewise linear regression I fit multi-line equations for sequential fire dates for each site, which resulted in four break points, five segments, in both SPD (R^2 = 0.999, SE = 0.578) and PDA (R^2 = 0.999, SE = 0.590). EPC was best described with two breakpoints or three segments (R^2 = 0.997, SE = 0.593). While early fire dates in segment one are subject to the greater influence of a small number of fire years (fading record) I see striking similarities among sites. Piecewise regression breaks, for example, detected a period of reduced fire frequency in the early half of the 19th century, a fire quiescent period (“fire gap”) that appears in many (though not all) locations over a broad area from northern Patagonia, Argentina (Kitzberger et al. 2001) to northeastern Oregon (Heyerdahl et al. 2002) and areas in between (Swetnam and Brown 2010). Segments were most similar between SPD and PDA (greatest overlap), which were closer in geographic proximity and had larger sample sizes. The greatest differences in these sites were for the most recent period (segment five), where SPD continued to burn frequently but slope declined for PDA after 1911. Continuous slope for

EPC for segment five, beginning after 1858, was influenced by recent fires in the high elevation stands, but this site had less frequent fires overall.

Climate, human – fire interactions

SEA analyses indicated that in the early period, 1650 – 1886, the role of dry fire years was statistically important (99% confidence level) whereas antecedent wet conditions were not. However, in the modern period, 1887 – 2003, antecedent wet conditions were important (years -1, and -2 at 99% confidence levels) with no apparent relationship of drought in fire years (Fig. 2.4). I identified similar relationships for different levels of filtering and for different climate data (Nino3 SST; Cook 2000, cool season precipitation; Ni et al. 2002, Appendix 2A, 2B).

I further examined the changing relationship between drought and antecedent wet conditions between periods by isolating all years when no fires were recorded in my sites. SEA of PDSI in all non-fire years also demonstrates a changing role of drought and antecedent conditions between periods. In years that fires did not occur in any of my sites, conditions were significantly wet (99% confidence level), as would be expected, for the early period 1650 – 1886 (Fig. 2.4). In the modern period 1887 – 2003, non-fire years were also wet (95% confidence level), but the years preceding non-fire years were also significantly dry (99 % confidence level). I compared small (< 10% recorder trees scarred) and large fire (> 50% recorder trees scarred) years in addition to non-fire years. SEA analyses for the full time span (1650 – 2003) demonstrate that small fire years were significantly dry, whereas the largest fire years in my study sites were not drier than would be expected, but had two significantly wet years preceding large fires (Fig. 2.5). For the ten fire years with the most abundant evidence of burning (highest percent scarred: 1794, 1811, 1819, 1847, 1867, 1877, 1890, 1909, 1921, and 1932), four were in the post 1886 modern period.

Piecewise regression fit to sequential fire years for data pooled among my three sites identified five segments (Fig. 2.6). The resulting segments divide into two periods of high slope/high fire frequency (segments 2 and 4; 1745-1799, and 1848-1947, respectively) and three lower slope/less frequent fire periods (segment 1; 1685-1744, segment 3; 1800-1847; and segment 5; 1948-2002). High slope periods had a large amount of overlap with two identified Apache wartime periods; (1) Apache-Spanish wartime period (WTP1) 1748-1790, and (2) Apache-Mexican/American wartime period (WTP2) 1831-1886 (Seklecki et al. 1996, Kaib 1998, Fig. 2.6). While the overlap of the WTP1 (1748-1790) and segment two (1745-1799) is remarkable, the second high fire frequency period, segment four, continued to burn frequently after 1886, well past the end of the Indian wars and WTP2 (Fig. 2.6). Similarly, a well documented Apache peacetime period (PTP2) between 1787-1830 (Kaib 1998, Seklecki et al 1996) overlaps with a low slope period, but this is also the fire quiescent or “fire gap” period documented broadly over the western hemisphere (Swetnam and Brown 2010). The last segment (5: 1948-2002) corresponds to documented land-use changes that occurred throughout many parts of Mexico, namely the establishment of the *ejido* system of land tenure (Heyerdahl and Alvarado 2003), and also local land-use changes, though less frequent fires are site specific with SPD continuing to burn with relatively high frequency.

SEA of PDSI for high and low slope periods split by early and modern time periods reveal a strong but varying climate signal (Fig. 2.7). Fires burned during drought conditions in low slope, low fire frequency periods (99% confidence level) in both the early and modern periods, and in the early high slope period (95% confidence level) with a change to significantly wet (99% confidence level) antecedent conditions for the modern high slope periods. I found similar differences between high and low slope periods for SEA of each of the five segments considered individually (Appendix 2C), with fires occurring in dry years in low slope segments and after antecedent wet conditions in high slope periods particularly for larger fire years (>10% of recorder trees, Appendix 2C). Mean and distribution of PDSI values were very similar across all years within both high and low slope periods. Comparisons of fire occurrence and PDSI indicate that fire frequency (fires per decade) was inversely related to fire size (percent scarred) with percent scarring more closely aligned with PDSI (Fig. 2.8; both percent scarred and PDSI smoothed with nearest neighbor function) lending further support for a climatic explanation for changing fire frequency through time.

Discussion

Changes in the climate-fire relationship after the late 19th century

Although fire frequency in the Sierra San Luis was similar to fire frequencies documented elsewhere in the Southwest prior to impacts of Euro-American land use, the relationship between fire and climate was different here. Notably, fires in the Sierra San Luis were associated with dry conditions prior to the late 19th century with no association with antecedent wet years. After the late 19th century, the relationship resembled other Southwestern sites, with both antecedent wet years and dry fire years being important – but this was true only when examining all fires regardless of size. For large fire years in the modern period ($\geq$10% scarred), antecedent moisture was the only significant variable; dry conditions during the year of the fire were not important. This was an almost complete reversal in the role of precipitation between time periods before and after the late 19th century for large fire years (Fig. 2.4). These results differ from most other studies in the Southwest that emphasized the role of both antecedent wet conditions and drought in the year of fires for large fire years (Swetnam and Betancourt 1998). However, there are notable exceptions to that general pattern. Morino (1996), for example, found that large fire years in the Organ Mountains in southern New Mexico occurred in drier than average years with little importance of wet antecedent conditions and offered Apache burning as one potential explanation for this unexpected climate – fire pattern. However; that study, like most historical studies, was limited to pre-Euro-American land-use changes and lacked contemporary climate – fire data to decipher potential changes in the modern era. More recently, 20th century fire atlas – climate analyses have also found strong associations of antecedent wet conditions, and little association between moisture conditions during fire seasons (Westerling et al. 2003, Crimmins and Comrie 2004, and Littell et al. 2009) , which corroborate my findings. This suggests that the 20th century change in climate – fire relationships I identified in northern Mexico may be widespread throughout the Southwest via interannual climate effects on fuel abundance.

The role of fuel abundance regulating fire occurrence, particularly in the arid Southwest, is well documented (Swetnam and Baisan 1996, Sherriff and Veblen 2008), but shifting importance of fuel abundance over time is less well understood. Swetnam (1993) identified the role of fuel limitations in frequent fire regimes for giant sequoia (*Sequiadendron* giganteum) groves in the Sierra of California where he reported that relative size of fires enlarged at an increasing rate as average interval between fires lengthened. In my study, I also found an inverse relationship between the number of fires per decade and percent of trees scarred. Percentage of scarring was associated with variation in PDSI – with wet periods corresponding to fewer fires but a higher percent of recorder trees scarring (Fig. 2.8). I would not expect effects of interannual climate – fuel abundance patterns on fire to be static, but late 19th century changes in these relationships are conspicuous in my sites. Long-term patterns indicating a change in importance of fuel abundance are supported by several lines of evidence. Non-fire years, years when no fires were recorded in any of my sites, were significantly wet years, which was expected. However, I again found important differences between periods, with antecedent dry conditions further limiting fire occurrence in the modern period but not historically (Fig. 2.4). This finding points to changes over time in short term interannual climate effects on fine fuel abundance and not simply moisture content of fuels in limiting or promoting fire occurrence in a given year. Similarly, fires in low fire frequency periods burned in significantly dry years for both periods while in high slope periods, where I would expect greater fuel limitations from fuel consumption, fires burned in dry years in the early period, but the only association was with prior wet conditions in the modern period (Fig. 2.7). Limitations of fuel abundance also help explain differences between small and large fire events with small fires occurring in drier than average conditions without antecedent wet conditions to promote fire spread via fuel abundance and continuity. The most widespread fire years in my sites demonstrate opposite relationships with little evidence of dry conditions in fire years, but two significantly wet antecedent years (Fig. 2.5). These findings suggest that fires in the Sierra San Luis have become more fuel limited likely from changes in short term interannual climate variation effects on fine fuels over time.

Climate vs. Human Influences on Fire Regimes

While the importance of Native Americans in modifying pre-Euro-American fire-regimes is difficult to quantify, this study suggests that interannual climate – fire relationships better explain changes in fire frequency than Native American burning practices. Apache burning in the Southwest likely had time and place specific effects (Swetnam 2001). Such effects may be obscured in a system with the potential to burn in almost any given year and regulated more by fuel limitation than ignition (Allen 2002) or climate. The Southwest leads the nation in average number of lightning strikes and subsequent fires (Pyne 2001), and most springs and summers are sufficiently dry for fires to burn (Grissino-Mayer and Swetnam 2000). Meanwhile, for specific periods of increased burning frequency we do see climate signals via effects on fuels (Fig. 2.8). Likewise, I did not find truncated frequency of fire after the end of the Apache wars in 1886. Rather, fires continued to burn across sites with no statistical change in frequency from 1848 to 1947 for sites combined (Fig. 2.6). I expect that my study area was geographically well suited to observe any fire history changes that might have occurred during the Apache wartime periods (Fig. 2.6) as northern Mexico was a region of repeated raids (Wilson 1995) and the last Apache stronghold (Goodwin 2002). The present day border (~16 - 30 km from my sites) was closely aligned with Spanish military presidios established in the late 1600's to deal with Apache depredations (Kaib 1998). The first part of the 19th century was marked by an Apache peacetime period (PTP2) considered one of the more peaceful periods of the northern frontier (~1787 – 1831), but much of this period was also a fire quiescent period observed in many other fire regimes throughout the western hemisphere (Kitzberger et al. 2001, Swetnam and Brown 2010) which suggest coincidence and climate controls of fire occurrence rather than effects of Indian peacetime periods. It is worth noting that Apache groups were independent and while some made peace treaties, others continued raiding, and some did both (Kaib 1998), making generalizations for periods difficult and perhaps all the more time and place dependent. However, attributing either frequent fires pre-Euro-American, or a cessation of frequent fires post-Euro-American settlement, to changes in Native American burning as in California (Fry and Stephens 2006) and Nevada (Biondi et al. 2011) is not applicable to my study in northern Mexico.

The near cessation of fires in EPC and PDA in the early 1930's was anomalous for the past 300 years in this area and coincides with recent documented land-use changes, particularly grazing related reductions in fuel availability as direct fire suppression remains ineffective in this region. That SPD continued to burn frequently without these same land-use changes affirms this finding. I attribute temporal changes in fire frequency for my sites in northern Mexico largely to interannual climate variation. Similar changes in climate – fire interactions (shifts in importance from drought in fire years to antecedent wet) were identified in the late 18^{th} century in central New Mexico, a potential explanation being a change in the summer monsoon (Grissino-Mayer and Swetnam 2000). A fuel limited system with dominant influences of interannual climate on fire occurrence implies greater uncertainty in fire regimes with anticipated increases in southwestern climate variability. Such temporal changes in interannual climate-fire interactions need further attention and may be linked to changes in the relatively poorly understood North American monsoon (Sheppard et al. 2002, Grissino-Mayer and Swetnam 2000), increases in frequency of El Niño events in recent decades (Trenberth and Hoar 1996, 1997), or relatively wet winters since the mid 1970's (Swetnam and Betancourt 1998). I did find evidence of greater amplitude of PDSI beginning in the mid-19th century (Fig. 2.8a) which is particularly conspicuous for longer time series than illustrated in my analysis, and may also be related to changes in climate – fire relationships between periods.

Conclusions

My study highlights fire history in a unique site where frequent fires continue long after similar locations in the Southwest ceased burning despite similar climate conditions. Thus I conclude that cessation of frequent fires in two of my three sites, similar to changes in Southwestern sites adjacent to our sites in northern Mexico, is a product of Euro-American land-use changes rather than climate forcing. My research documented a shift in climate – fire relationships in the late 19th century toward an overwhelming importance of antecedent moisture, unlike that seen previously for > 200 years. My sites have become increasingly fuel limited even in relict sites with absence of modern land-use changes, potentially a result of recent changes in interannual climate variability. Anticipated increases in southwestern climate variability imply greater uncertainty for fire regimes in fuel limited systems where fire occurrence is largely controlled by interannual climate like that of northern Mexico.

Table 2.1. Fire interval metrics (years) for three study sites in the Sierra San Luis, Sonora, Mexico. The analysis period was from the first fire year recorded by > 10% of recording trees at each site until time of sampling or disruption of fire events were apparent. Analyses were further restricted to fires recorded by a minimum of two trees.

Site, analysis period	Category of analysis	No. intervals	Mean fire interval, MFI	Min	Max	Weibull median probability interval, WMPI
Sierra Pan Duro	all scars	31	6.7	2	17	6.3
1794–2008	10% scars	22	9.4	3	26	8.4
	25% scars	17	12.1	3	28	11.3
Pan Duro Arroyo	All scars	31	6.0	1	12	5.8
1745–1932	10% scars	31	6.0	1	12	5.8
	25% scars	24	7.8	3	17	7.6
El Pinito Canyon	all scars	9	9.6	4	14	10.0
1847–1933	10% scars	7	12.3	4	24	11.6
	25% scars	5	17.2	4	43	14.0
Sites Combined	All scars	61	4.5	1	11	4.1
ALL, 1728–2008	10% scars	41	6.6	1	28	5.7
	25% scars	24	11.1	1	28	10.2

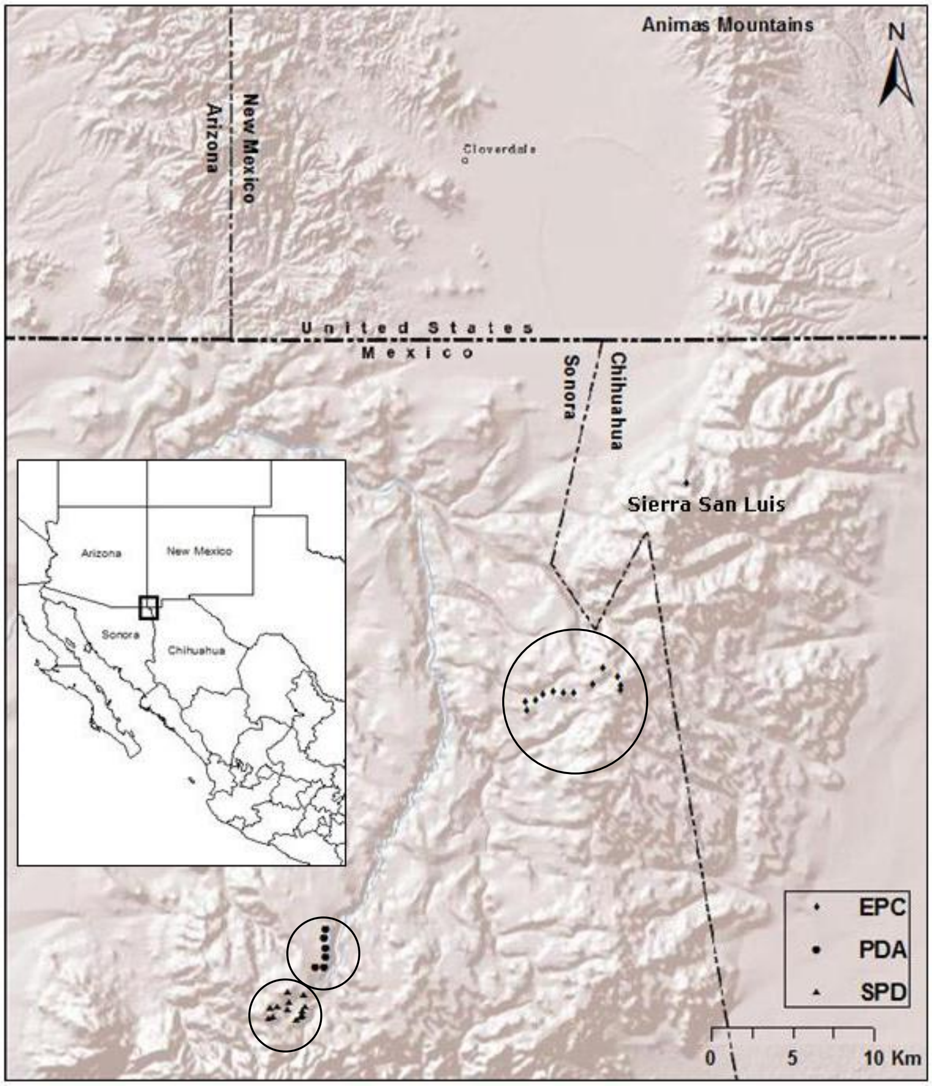

Figure 2.1. Study area in the Sierra San Luis, Sonora, Mexico: Sierra Pan Duro (SPD, little to no grazing or logging), Pan Duro Arroyo (PDA, logged and grazed after 1930), and El Pinito Canyon (EPC, grazed after 1930 but not logged).

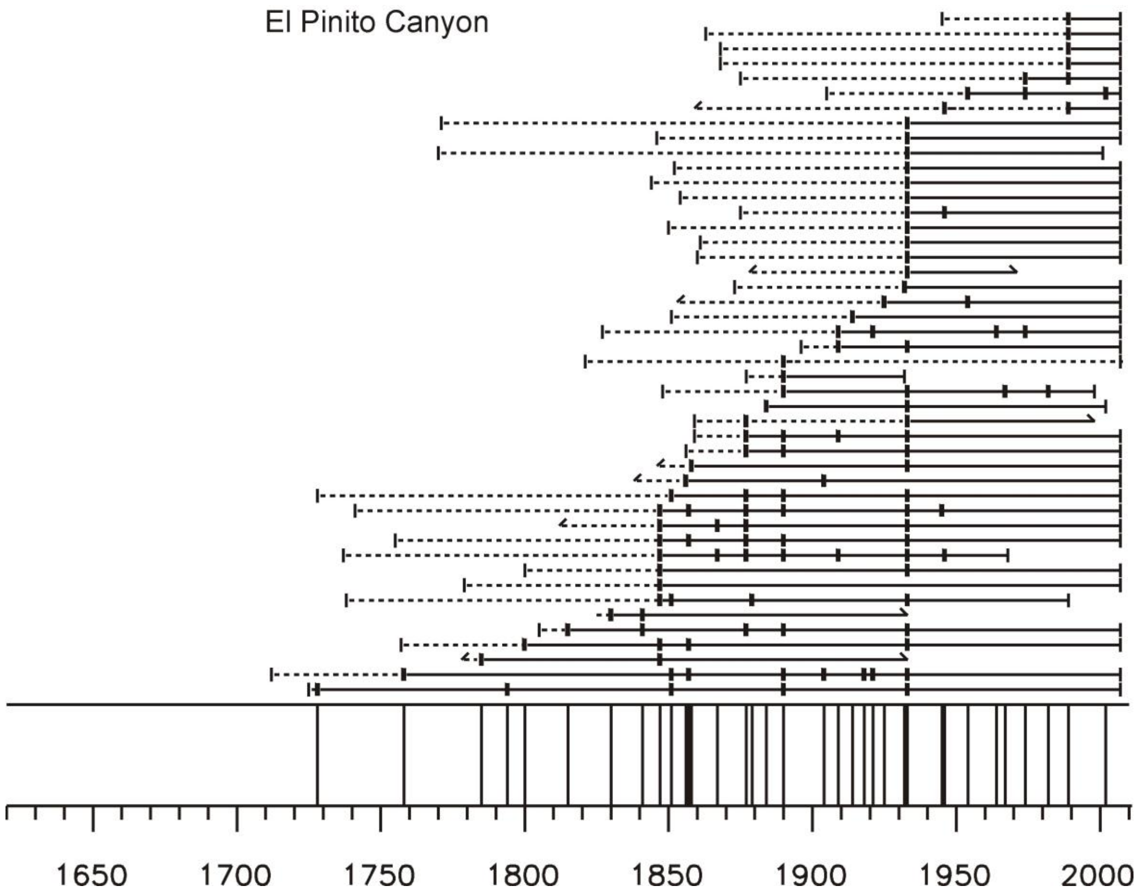

Figure 2.2 a, b, c. Fire year chronologies for (a) El Pinito Canyon (EPC, grazed after 1930 but not logged), (b) Pan Duro Arroyo (PDA, logged and grazed after 1930), and (c) Sierra Pan Duro (SPD, little to no grazing or logging) study sites. Bold vertical lines represent fire years, non-bolded vertical lines represent pith and bark dates, vertical spurs are estimates of pith or bark years, dashed horizontal lines are time spans of individual trees before they became recorder trees (prior to first injury), and solid horizontal lines are time spans of individual recorder trees.

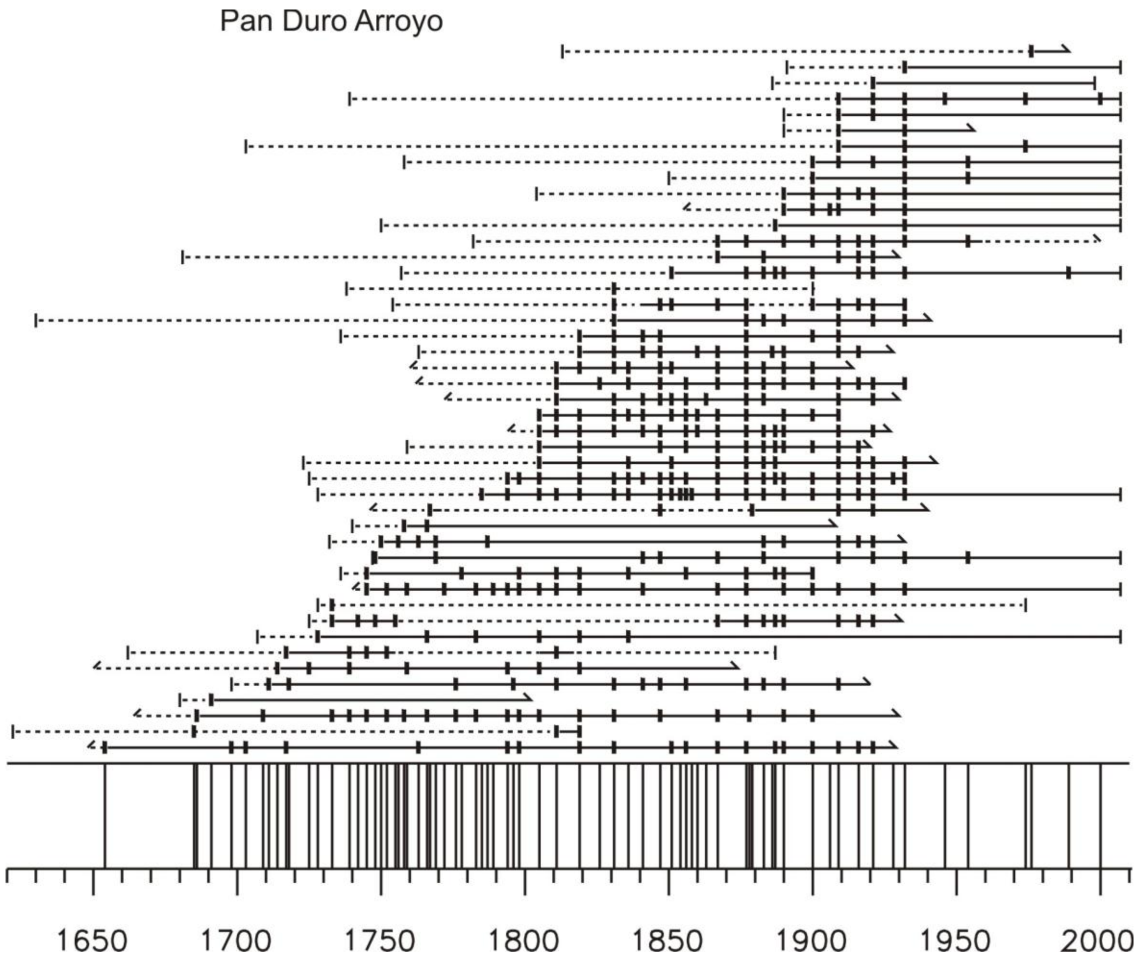

Figure 2.2b

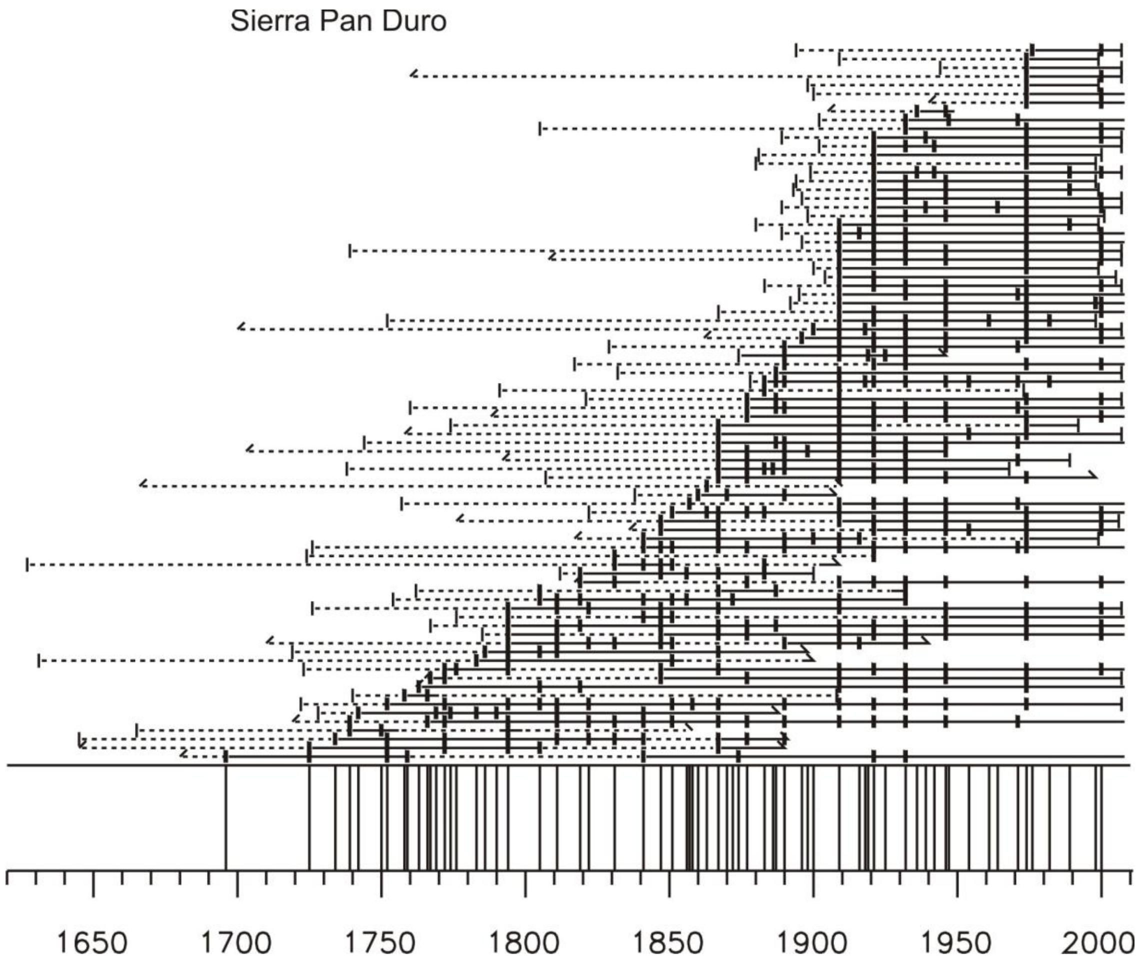

Figure 2.2c

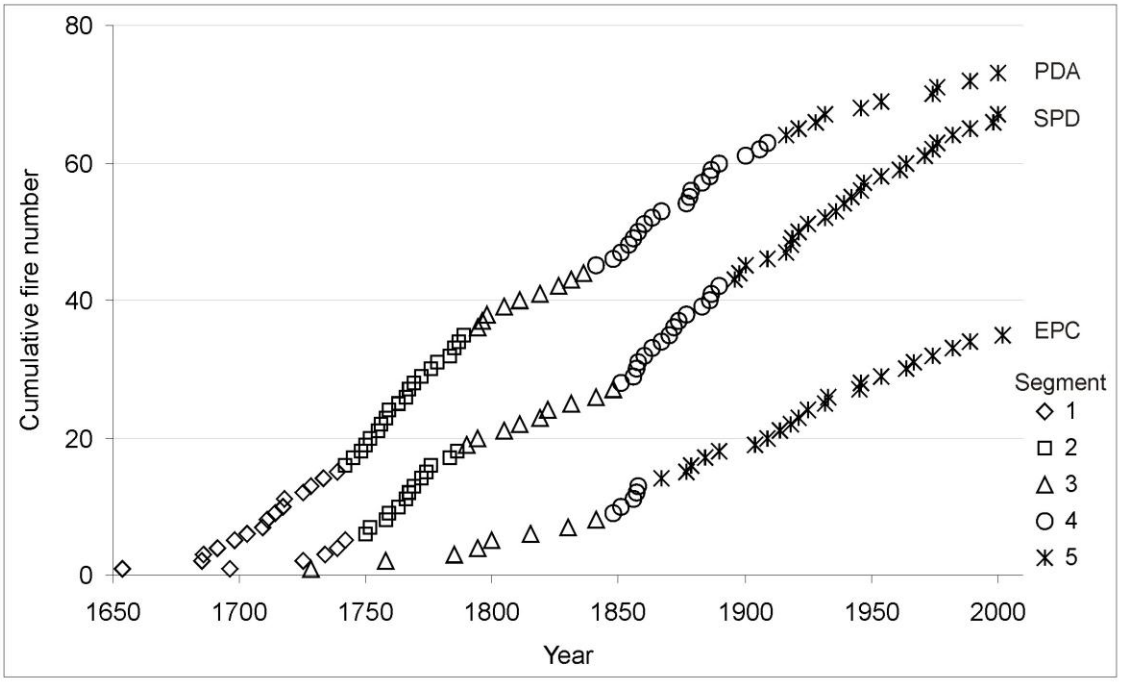

Figure 2.3. Fire frequency for three sites in the Sierra San Luis determined by piecewise linear regression multi-line fit equations and defined by slope changes in cumulative fire dates for three sites: El Pinito Canyon (EPC, grazed after 1930 but not logged, 3 segments), Pan Duro Arroyo (PDA, logged and grazed after 1930, 5 segments), and Sierra Pan Duro (SPD, little to no grazing or logging, 5 segments).

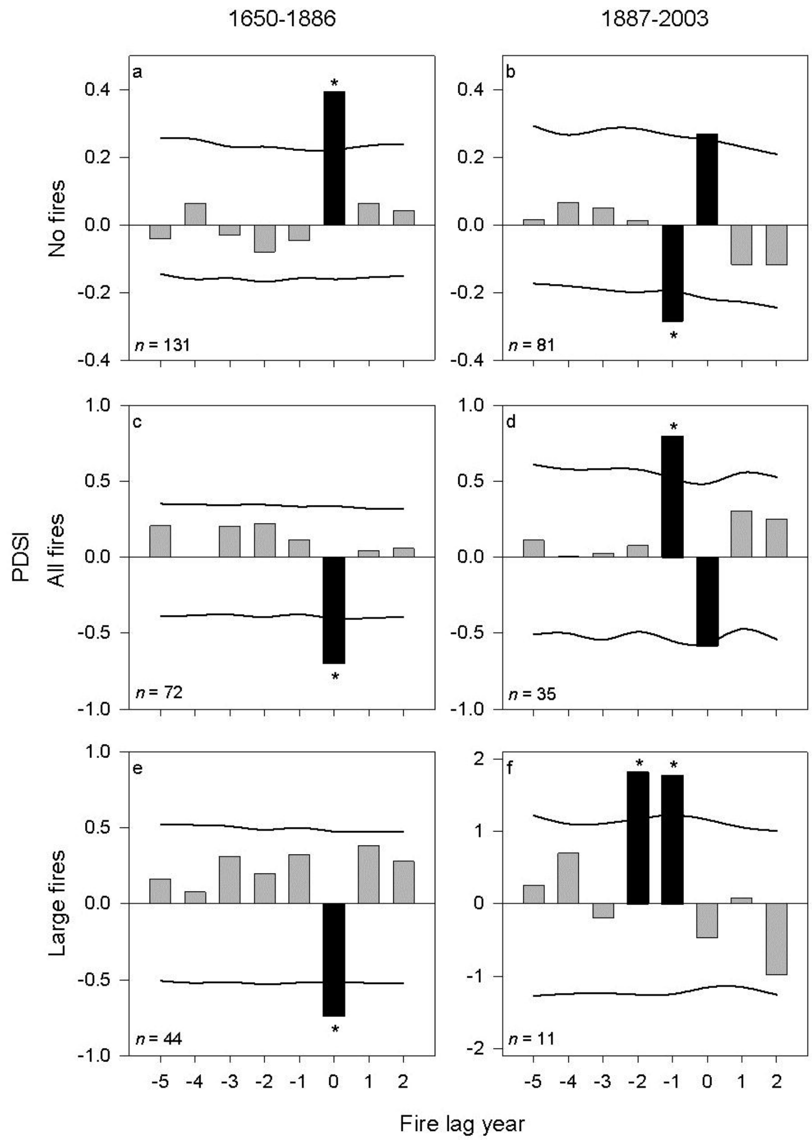

Figure 2.4. Results of superposed epoch analysis (SEA) of tree-ring reconstructions of Palmer Drought Severity Index (PDSI, Cook et al. 2004) for years prior and subsequent to fire event years (year 0). Positive PDSI values indicate wet conditions, negative values represent dry conditions. Data shown are for years when no fires were recorded in any of our sites (no fires, a-b), all fire events without filtering (all fires, c-d) and fires recorded on ≥ 10% of samples (large fires, e-f) for early (1650-1886) and late (1887-2003) time periods. Solid bars indicate PSDI values outside a 95% confidence interval (depicted by lines); asterisk symbols indicate values outside a 99% CI; All CI's are based on 1,000 Monte Carlo simulations of random distributions of annual PDSI. Sample sizes are identified for the number of fire events tested against PDSI data.

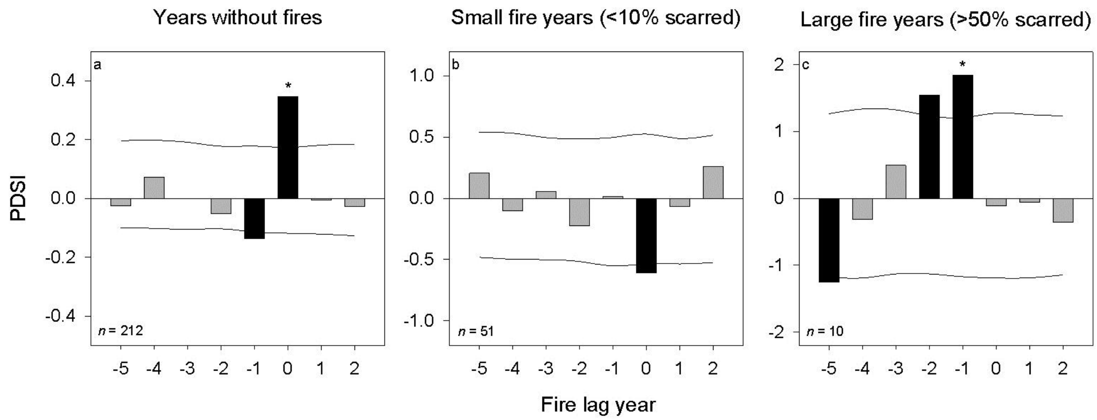

Figure 2.5. Results of superposed epoch analyses (SEA) for tree ring constructions of Palmer Drought Severity Index (PDSI, Cook et al. 2004) for years prior to (-1 to -5) and subsequent to (1 to 2) event years (year 0) from 1650 - 2003. Data shown are for (a) years without fire events in any site, (b) small fire years (only fire events recorded on fewer than 10% of recorder trees) and (c) big fire years (>50% scarred and includes only years with > 10 recorder trees; 1794, 1811, 1819, 1847, 1867, 1877, 1890, 1909, 1921, and 1932). See caption for Figure 4 for details of analysis.

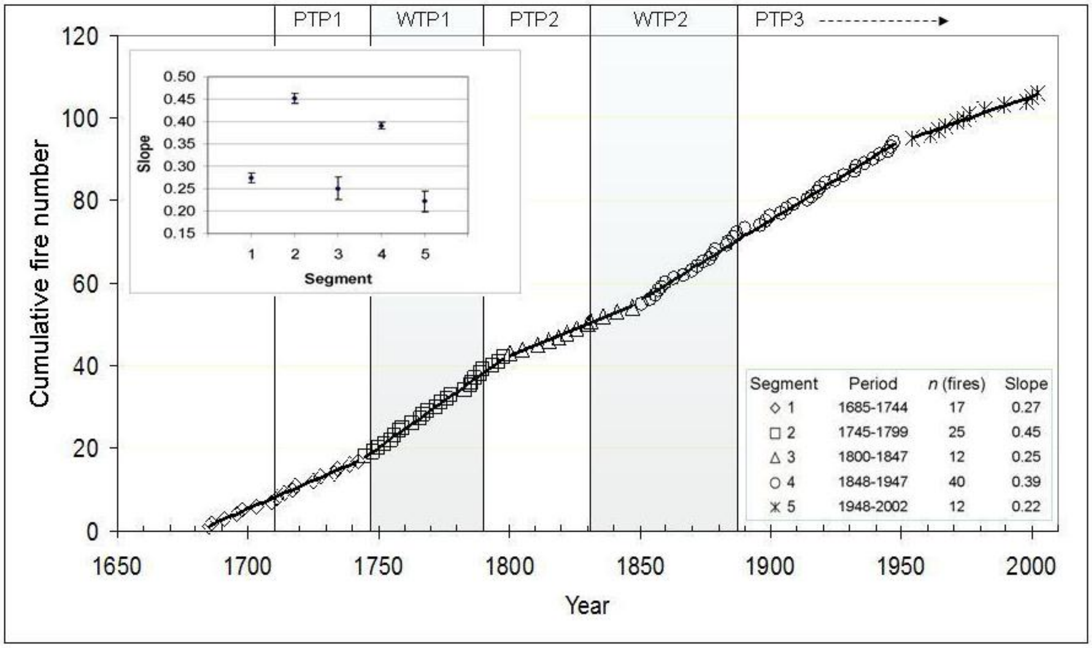

Figure 2.6. Fire frequency determined by piecewise linear regression slopes fit through cumulative fire dates of best fit segments (n = 5) for data pooled among sampling sites (EPC, PDA, and SPD). Slope comparisons for resulting segments and an approximate timeline of Apache peacetime (PTP1-3) and wartime (WTP1-2) periods (Kaib 1998) are illustrated for comparison to high and low fire frequency/slope periods.

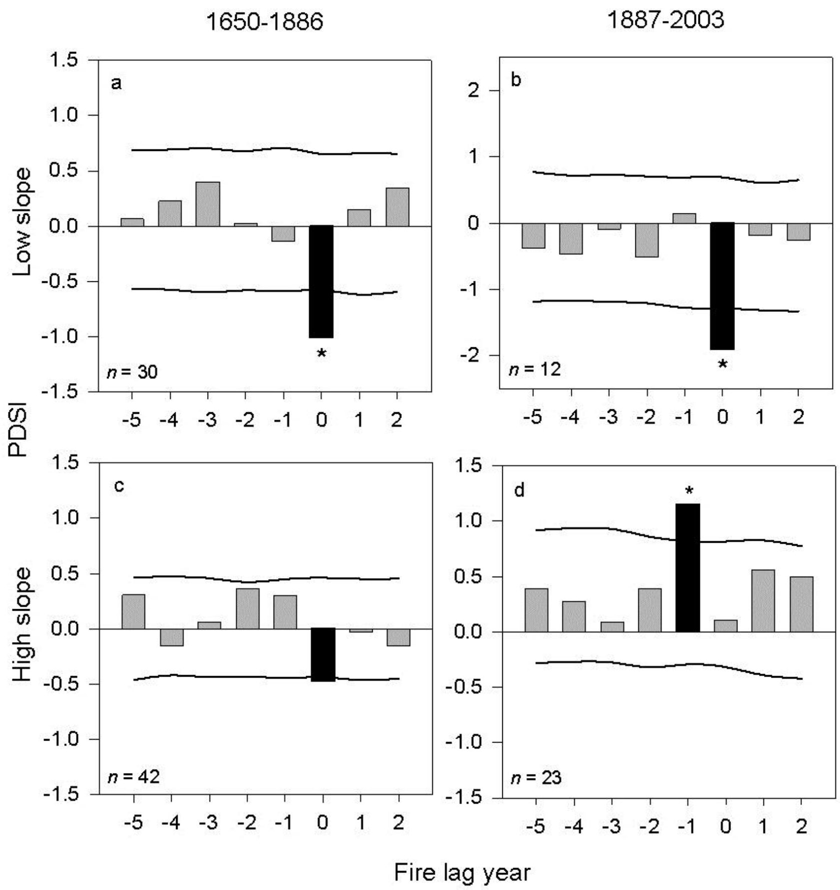

Figure 2.7. Results of superposed epoch analysis (SEA) of tree-ring reconstructions of Palmer Drought Severity Index (PDSI, Cook et al. 2004) for years prior and subsequent to fire event years (year 0). Positive PDSI values indicate wet conditions, negative values represent dry conditions. Data shown are for all fire events for pooled periods of high and low slope fire events determined by piecewise regression (Fig. 6) and PDSI climate data. Solid bars indicate PSDI values outside a 95% confidence interval (depicted by lines); asterisk symbols indicate values outside a 99% CI; All CI's are based on 1,000 Monte Carlo simulations of random distributions of annual PDSI. Sample sizes are identified for the number of fire events tested against PDSI data.

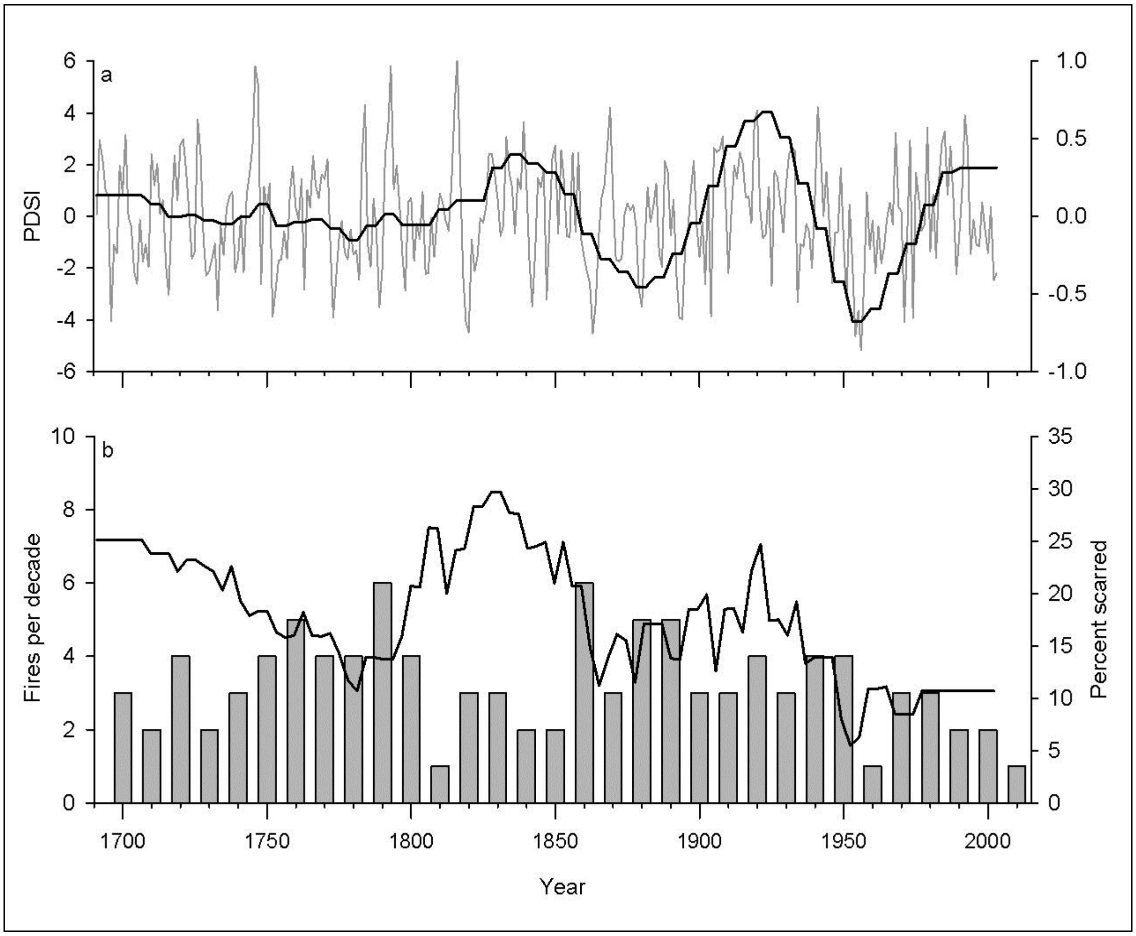

Figure 2.8. Timeline depicting changes in fires as a function of climate (Palmer Drought Severity Index, PDSI), fire size (percent scarred), and frequency (fires per decade). Top graph (a) illustrates PDSI values from tree ring reconstructions (Cook et al. 2004) with a smoothing function (heavy line) and secondary y-axis (right side) to better depict changes in amplitude. Lower graph (b) illustrates the number of fires per decade (bar graph) and a secondary y-axis with percent of recorder trees scarred in a given year with a smoothing function (heavy line). Smoothing functions for both lines are nearest neighbor (0.10 sampling proportion) running averages.

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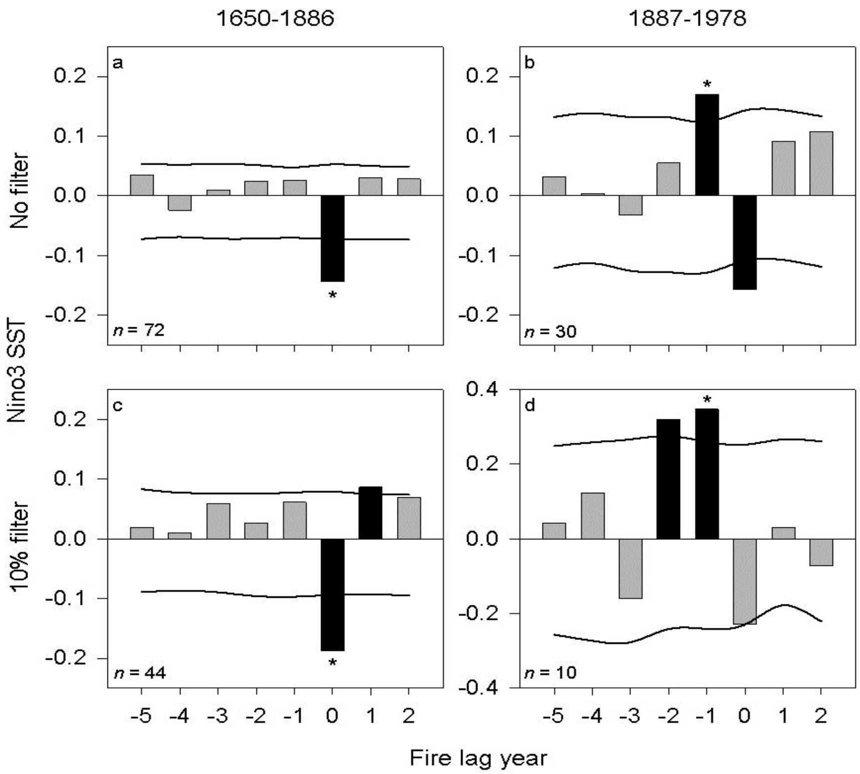
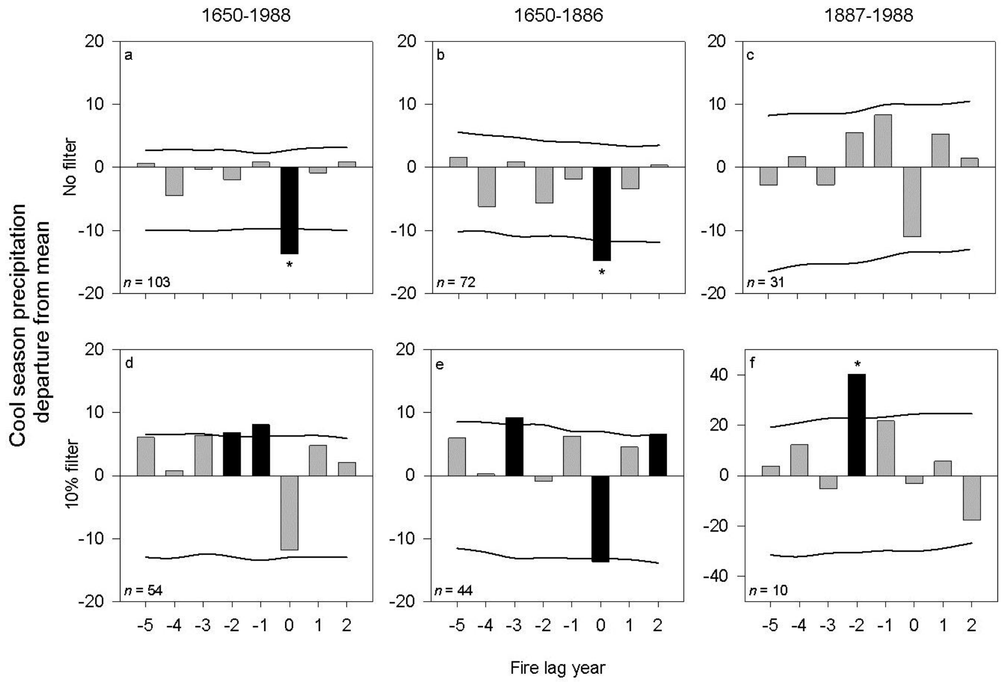
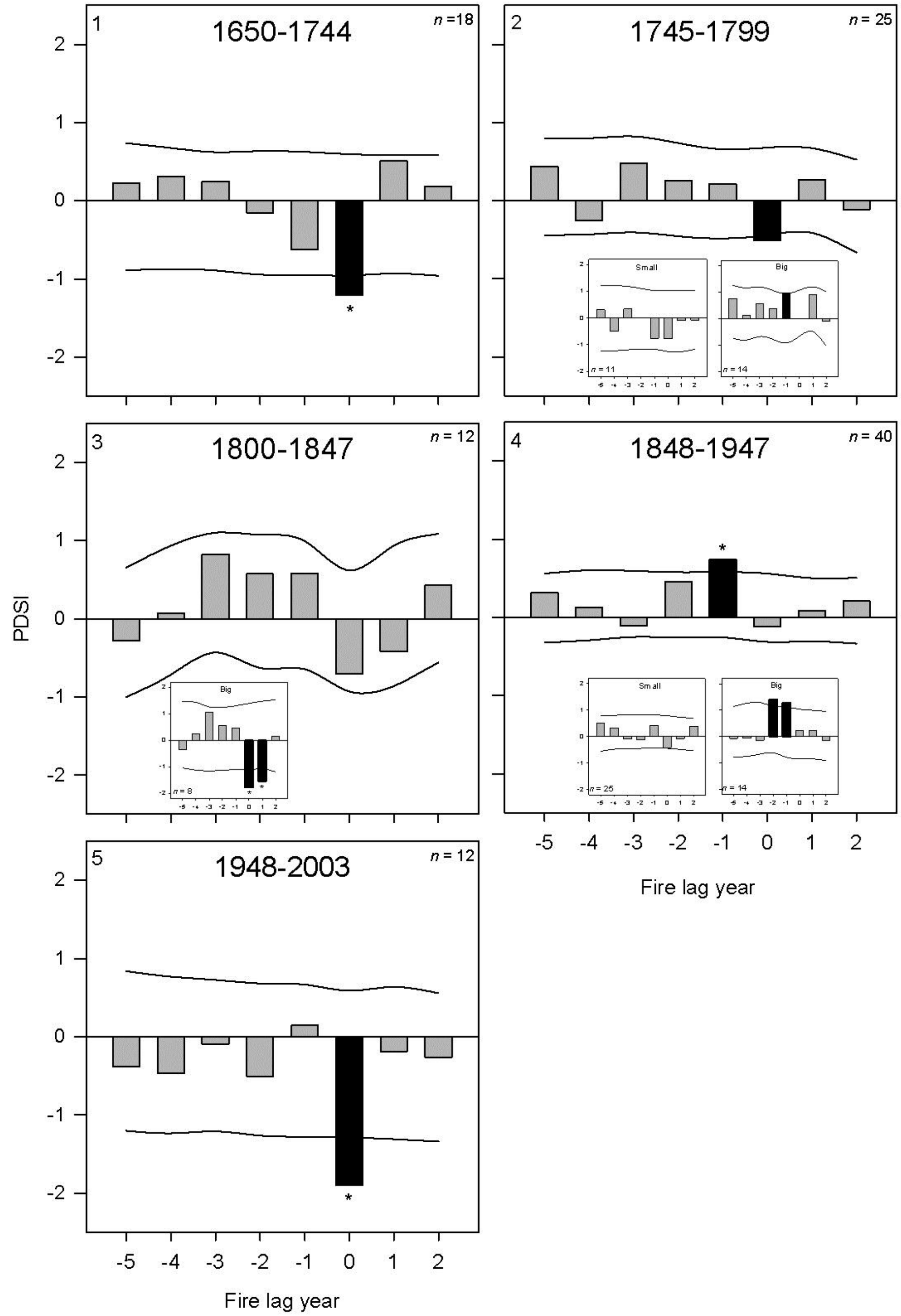


Appendix 2A. Results of superposed epoch analysis (SEA) showing departure from mean value of NINO3 sea surface temperatures (SST, Cook et al. 2000) for years prior and subsequent to fire event years (year 0). Positive Nino3 values indicate wet conditions, negative values represent dry conditions. Data shown are for all fire events (no filter) and fires recorded on $\geq$ 10% of samples for early (1650-1886) and late (1887-1978) time periods. Solid bars indicate PSDI values outside a 95% confidence interval (depicted by lines); asterisk symbols indicate values outside a 99% CI; All CI's are based on 1,000 Monte Carlo simulations of random distributions of annual PDSI. Sample sizes are identified for the number of fire events tested against PDSI data.



Appendix 2B. Results of superposed epoch analysis (SEA) of tree-ring reconstructions of cool season precipitation (November-April, Ni et al. 2002) for years prior and subsequent to fire event years (year 0). Positive values indicate wet conditions, negative values represent dry conditions. Data shown are for all fire events (no filter) and fires recorded on $\geq$ 10% of samples for the entire period with climate data (1650-1998) and for early (1650-1886) and late (1887-1998) time periods. Solid bars indicate PSDI values outside a 95% confidence interval (depicted by lines); asterisk symbols indicate values outside a 99% CI; All CI's are based on 1,000 Monte Carlo simulations of random distributions of annual PDSI. Sample sizes are identified for the number of fire events tested against PDSI data.



Appendix 2C. Results of superposed epoch analysis (SEA) of tree-ring reconstructions of Palmer Drought Severity Index (PDSI, Cook et al. 2004) for years prior and subsequent to fire event years (year 0) for five segments of unique periods based on piecewise regression of slopes. These results were further analyzed for differences by large (≥ 10% recorder trees scarred) and small fire years (< 10% recorder trees scarred) when sample sizes permitted. Positive PDSI values indicate wet conditions, negative values represent dry conditions. Solid bars indicate PSDI values outside a 95% confidence interval (depicted by lines); asterisk symbols indicate values outside a 99% CI; All CI's are based on 1,000 Monte Carlo simulations of random distributions of annual PDSI. Sample sizes are identified for the number of fire events tested against PDSI data.

CHAPTER 3

PATTERNS OF TREE RECRUITMENT IN RELATION TO CLIMATE AND DISTURBANCE IN NORTHERN MEXICO

Summary

The scientific basis for manipulating forest structure in vast areas of montane forests throughout the U.S. Southwest is a documented shift away from historic pre-Euro-American settlement structure. These changes are a product of confounding effects of disturbance, climate, species competition, and modern land-use history. The single greatest forest reproduction event occurred in the early 20th century, a climatically wet period following a period of widespread fire exclusion in the Southwest. This pronounced reproduction event is at the heart of increased forest density and current restoration efforts. Yet we have only a rudimentary understanding of the relative roles of climate and modern land-use changes in driving that event and others like it. I investigated climate, fire, and tree recruitment interactions in the Sierra San Luis of northern Mexico, an area with a spectrum of land-use changes from areas with little or no history of grazing or logging, to sites that were logged (ca. 1952-1954) and/or grazed (beginning in the early 1930's). Fire regimes and tree recruitment closely reflected these differences. While fires were strongly tied to interannual wet-dry cycles of hydroclimate, recruitment peaks were more closely tied to local processes than broad-scale climatically wet conditions.

The greatest pulse of tree recruitment in my sites coincided with a pronounced mid-century drought (1942-57) and a period of reduced fire frequency. The second largest pulse of recruitment in my study sites preceded a well documented period of recruitment elsewhere across the Southwest in the 1910s-20s, which was an anomalously wet period. In my study area, this early 20^{th} century recruitment event also largely coincided with below average precipitation but with remarkable alignment to site specific fire free periods. I found greater spatial clustering and aggregation in older cohorts of trees that indicate a legacy of fire-induced mortality in shaping stand structure. This underscores the importance of frequent fire effects on spatial variability in forests.

Introduction

Current forest structure in western North America is a product of interacting effects of disturbance, climate, species competition, and land-use history. This complexity makes accurately attributing relative importance of past events in shaping current forest structure difficult. A unique set of circumstances, for example, of anthropogenic disturbances, fire exclusion, and anomalously wet climate in the early 20^{th} century resulted in one of the most extensive, and greatest in magnitude, forest recruitment pulses recorded in the western U.S. (Savage et al. 1996). This single pulse of recruitment in long-lived forest ecosystems will continue to shape forest structure for centuries. It is also at the center of one of the most extensive and expensive forest restoration efforts ever undertaken (Larson and Churchill 2012) that aims to restore forests to conditions more representative of pre-Euro-American settlement (Covington and Moore 1994). Yet it remains largely unknown what the effects of anomalous environmental factors (climate, cone crop), versus 20th century land-use history changes (fire exclusion with grazing, fire suppression, and logging) have had in shaping this recruitment event or others like it.

Southwestern forest communities experience regionally synchronous episodic recruitment and mortality (Swetnam and Betencourt 1998, Brown and Wu 2005) with striking connections to synoptically-driven climate variation (Barton et al. 2001, Swetnam and Brown 2010). Such trends have been proposed as evidence of broad scale climate forcing, or anomalous climate years that can override local processes in long-term forest dynamics (Swetnam 1993, Swetnam and Brown 2010). The strong influence of climate rather than disturbance on historical recruitment of ponderosa pine (*Pinus ponderosa*) in the Colorado Front Range led to suggestions that the processes of mortality and regeneration there were temporally uncoupled processes (Boyden et al. 2005). However, in the upper montane zone of the Colorado Front Range pulses of ponderosa pine establishment occurred during periods of extreme drought, following moderate to high severity fire (Sherriff and Veblen 2006, Schoennagel et al. 2011). In southwestern Colorado, Brown and Wu (2005) attribute cohorts of ponderosa pine to favorable climate conditions for recruitment pulses (less suitable to burning) and longer intervals between surface fires, and with little evidence of episodic mortality.

Site and time specific discrepancies in associations of recruitment to disturbance and climate are common and likely reflect spatial and temporal variation as well as limitations of incomplete information. In contrast to regional correlations between recruitment and climate in the Southwest, local comparisons often lack clear connections. Pre-Euro-American settlement tree establishment in Arizona, for example, (Mast et al.

1999, Barton et al. 2001) was not easily related to climate patterns, but instead more closely aligned with reduced fire frequency; though periods of high moisture availability and low fire frequency are often linked. A peak in age structure in the late 19th and early 20th centuries in many studies of ponderosa pine is an extreme example of both favorable climate and reduced fire frequency. Tree recruitment in this period primarily a result of relatively rare climate particularly suitable for recruitment rather than primarily a legacy of fire exclusion has important ramifications for forest restoration efforts. Combining climate and disturbance histories for this period may provide a more complete picture of these dynamics (Mast et al. 1999). Evaluating synchronizing effects of regional climate on forests requires assessing the degree to which local responses of tree populations are driven by large-scale controls versus local factors specific to a site (Barton et al. 2001).

This research was focused on a unique area in northern Mexico with locally different land-use history among sites, and from the adjacent intensively studied U.S. Southwest, making climate versus site specific factor comparisons possible. In northern Mexico frequent fires continued to occur long after fires ceased burning north of the border. This region provided a unique opportunity to study tree recruitment in the presence of frequent fires under similar climate conditions to adjacent forests in the Southwestern U.S.

The goal of my research was to combine climate and disturbance history for an area that had different land-use history both locally (e.g. logged vs. unlogged sites), and regionally (Mexico vs. U.S. Southwest) to provide a more complete picture of tree recruitment dynamics. My primary objectives were to (1) explore the complexity of climate-disturbance-recruitment relationships in three similar sites with different land-use histories, and (2) use extreme events such as the 1900's recruitment pulse and the 1950s drought to disentangle climate and anthropogenic effects on recruitment.

Methods

Study area

The Sierra San Luis in northern Mexico encompasses the highland region straddling Chihuahua and Sonora. It is comprised of the larger canyons and mountain tops or sub-sierra land features of the northernmost extension of the Sierra Madre Occidental. The Sierra San Luis is part of an array of mountains that are referred to as the Madrean Sky Island Archipelago, which is the confluence of four biogeographical regions: the Rocky Mountains, the Sierra Madre Occidental, and the Sonoran and Chihuahuan Deserts (Felger and Wilson 1994). This convergence results in high biological diversity and species endemism and is part of the two richest floras of mega-Mexico – which ranks as one of the three top mega-diversity centers of the world (Felger and Wilson 1994).

It is however, the divergence in modern land-use history that creates unique opportunities to study ecological differences in Mexico. Aldo Leopold (1937) recognized this opportunity as did Joe Marshall two decades later. In his cross border comparison work in the mid 20th century, Marshall (1957, 1963) stated that the high elevation forests in the Sierra San Luis were unlogged with practically no grazing, and with abundant evidence of fires. This was in stark contrast to the U.S. side of the border (Leopold 1924, 1937, Marshall 1963, Swetnam and Baisan 1996). These differences were in part a result of long Apache Indian occupation followed by the decade long Mexican revolution

(1910-1920) and subsequent unstable land policies. After the Mexican Revolution, legislation (Agrarian Law 1915, Mexican Constitution 1917) legalized the redistribution of land to small communities of landless people (*ejidos*; Heyerdahl and Alverado 2003). However, land was generally not reallocated until the late 1930's-early 1940's under the Lazaro Cárdenas administration (Sanderson 1984, Thompson and Wilson 1994, Heyerdahl and Alverado 2003). The result was delayed modern land-use changes (e.g. grazing, logging) and in some places the presence of relict forests that have largely escaped these intensive land-uses (Fulé et al. 2012).

My study sites (Table 3.1, Fig. 3.1) take advantage of different land-use histories and are comprised of the larger mountaintops of the Cajon Bonito Watershed in the Sierra San Luis. They included three similar physiographic areas that were (1) never logged and had little grazing pressure (Sierra Pan Duro, SPD), (2) were grazed beginning in the early 1930's and logged ca. 1952 to 1954 (Pan Duro Arroyo, PDA), and (3) were unlogged but grazed in a similar time period (El Pinito Canyon, EPC). EPC had sharper relief and contained mostly lower elevation (~1,800-2,000 m) pine-oak forest limited to narrower confines of a major canyon, and Douglas-fir (*Pseudotsuga menziesii*) stands at the highest elevations (~2,440 m). Douglas-fir stands within EPC are treated independently for analysis of plot level data as El Pinito Mountain Top (EPMT), but are within the EPC site. Dominant overstory in pine-oak forests included Chihuahua pine (*Pinus leiophylla*), Apache pine (*P. engelmannii*), and mixtures of Madrean oaks (*Quercus arizonica*, *Q. emoryi, Q. oblongifolia*). PDA and SPD had a greater extent of intermediary elevation (~2,000-2,200 m) ponderosa pine (*P. ponderosa var. arizonica*) forest. In addition, extensive pinyon-juniper woodlands (mostly *Juniperus deppeana* and *P. cembroides*) were present across all three sites at lower elevations.

Stand structure

I characterized forest demography using an *n*-tree density-adapted sampling design (Jonsson et al. 1992, Lessard et al. 2002) to collect data from the nearest ~30 trees > 7.5 cm diameter breast height (dbh, cm at 1.35m height) to each plot center. Variable radius *n*-tree sampling is a density-adapted sampling design that permits comparison of patterns within and among plots using a fixed numbers of trees. Plots were randomly located 500 meters apart (EPC and PDA) or from a probability based spatially balanced design (Theobald and Norman 2006, Theobald et al. 2007) for delineable stands (SPD). I sampled both living and remnant trees (stumps, logs, and snags) within plots by coring all living trees for age, and taking partial cross sections from remnant trees and fire-scar samples at 10 cm height to determine recruitment and fire-scar dates. I sampled fire-scarred plot trees with at least three visible scars and also sampled fire-scarred trees opportunistically, collecting samples from multiple scarred trees surrounding our plots (≤ 200 m) to more completely reconstruct fire history for my sites.

In the laboratory, I sanded samples until the cellular structure of the xylem was clearly visible under magnification (Grissino-Mayer and Swetnam 2000). Samples were subsequently crossdated against reference chronologies from the nearby Animas and Chiricahua mountains in the U.S. I dated all fire scars to their exact year of formation for precisely crossdated samples (Dieterich 1983, Dieterich and Swetnam 1984). Tree recruitment or establishment dates refer to the date when trees became successfully rooted as seedlings, rather than exact date of germination (Swetnam and Brown 2010).

Recruitment dates are defined as 10 cm height pith dates (Brown et al. 2008), rather than exact germination dates, as exact dates of “root-shoot” boundaries are difficult to determine without destructive sampling (Savage et al. 1996, Swetnam and Brown 2010). I measured diameters at 10 cm (diameter sample height, dsh) for all trees in addition to dbh for all living trees. Diameter at breast height of living trees was used for conversion of dsh to dbh for remnant trees and to determine stand basal areas. The number of trees < 7.5 cm dbh were counted in each plot for which I did not collect increment cores.

I measured distance and azimuth to all sampled trees from spatially referenced plot centers and converted these measurements into x, y coordinates for all samples. I determined the radius of each plot by calculating the distance from plot center to the center of the farthest tree sampled. Subsequently I determined plot area as circular plots of calculated radii (Table 2, Moore 1954, Lessard et al. 2002, Brown et al. 2008). Stem basal areas of living trees (square meters per hectare) were determined from dbh measurements. I converted dsh for recently dead remnants (with bark) using a linear regression equation derived from dbh/dsh measurements on living trees by species (n = 91 – 299, R^2 = 0.970 – 0.991, P < 0.001). I also corrected recently dead remnant trees with only sapwood to bark diameters using a separate regression equation for estimates of bark diameters (n = 17, Brown et al. 2008). I used empirically derived regression equations (age vs. dbh) by species from 541 trees with pith dates to determine age estimates for trees for which I could not crossdate (n = 259). While calculated ages for trees without pith dates were only estimates, my goal was not to assign exact year of recruitment, but to assign trees to $\geq$ 40 year age-classes into one of four periods of recruitment for spatial analysis (1: pre-1885, 2: 1890-1930, 3: 1935-1975, 4: post 1975).

Spatial analysis

Processes influencing change in structure of forest communities operate through both time and space (Sánchez Meador et al. 2010). It follows that quantitative description of tree recruitment spatial patterns offers insight into the historical and environmental mechanisms influencing forest stand structure (Boyden et al. 2005, Sánchez Meador et al. 2010). I examined spatial aspects of tree recruitment at the tree-neighborhood and within-patch scales for plot level data (n = 28 plots, 766 trees). While my plot areas were generally small (0.04 - 0.37 ha, μ = 0.15 ha), changes in spatial pattern at tree-neighborhood and within-stand patch scales typically are most apparent within 20 m (Frelich et al. 1998) and at scales less than 4 ha (Agee 1998, Kauffman et al. 2007, Larson and Churchill 2012), indicating a utility of local pattern analysis even for small plot areas. Furthermore, I was primarily interested in comparisons among plots with different land-use histories and distribution patterns of tree recruitment. Examples of distribution patterns include even-age cohorts of recruitment, and multi-aged reverse-J distributions, with more numerous young trees and fewer trees in older age-classes.

I used Ripley's K(t) function (Ripley 1976, 1977) to determine how spatial interactions of recruitment change over distance for my plots. The K(t) function estimates spatial dependence between points producing a cumulative distribution function that represents the expected number of trees within a given distance of individual trees (Ripley 1981, Boyden et al. 2005). The model tests point data for departure from a spatially random pattern and I used an $L(t)$ square root transformation to stabilize variance. I computed 95% confidence intervals using a Monte Carlo simulated Poisson process using 100 simulations (Reich and Davis 1998) for an indication of statistical significance. Significant differences ($P \leq 0.05$) between observed and random patterns occur where the $L(t)$ plot falls outside of the simulated confidence envelope. I used a Cramer-von Mises goodness-of-fit test to compare simulated and observed point patterns to test for spatial randomness.

White (1985) suggested that successful tree establishment depended on "safe-sites" where seedlings could establish and grow above lethal flame heights. This implies that seedling survival is low overall, but could lead to patches of high survival in safe-sites missed by fires with heterogeneous burn patterns. Such patterns could lead to a relatively flat age distribution rather than a reverse-J distribution expected in an uneven-aged forest with constant mortality (Mast et al. 1999). Spatially limited recruitment could also lead to greater clumping of tree recruitment in frequent fire periods. I was interested in the spatial distribution of trees based on age-class as a means to determine the relative influence of direct climate effects and competition, where more uniform tree recruitment is expected, versus safe-site limited recruitment, where greater aggregation of trees is expected, particularly in frequent fire periods and sites.

The degree to which trees within an age-class tended to occur close to one another versus other age-classes was calculated with Pielou's index of segregation (Pielou 1961). Pielou's index randomly selects a tree and then its nearest neighbor, recording the type or age-class of each tree to test hypotheses about whether age-classes were segregated or mixed using a Chi-square test under the null hypothesis of no segregation (spatial independence). The test also calculates an index of segregation (S), which is a population parameter with no sampling variance. Pielou's index of segregation is not strongly influenced by plot size or tree spacing making it a robust measure for plots of small and variable sizes. All spatial statistics were computed using Reich and Davis's online spatial library (Reich and Davis, 1998) in R 2.14.0 (R Development Core Team 2008).

Fire, climate, and recruitment interactions

To compare recruitment periods and fire occurrence to climate conditions I used two independently derived measures of hydroclimate; Palmer Drought Severity Index (PDSI, Cook et al. 2004) and cool-season precipitation (November – April) reconstructions (Ni et al. 2002). Both climate variables were averages of the nearest tree-ring based reconstructions. Cool season precipitation was the average of southern Arizona and New Mexico regions (Ni et al. 2002) and PDSI was the average of four nearest grid points surrounding my study area for the North America Drought Atlas PDSI (Cook et al. 2004). PDSI is a single variable that represents precipitation and to a lesser extent temperature (Sheppard et al. 2002). Both climate variables have been correlated to regional tree recruitment and fire occurrence in the U.S. Southwest (Swetnam and Baisan 1996, Swetnam and Brown 2010). Regionally synchronous fires in the Southwest, for example, occurred at the rate of about 10 per century and often coincided with extreme wet-dry cycles (Swetnam et al. 2001). This pattern typically includes 1 to 3 wet years that promote accumulation of fine fuels, followed by a dry year when synchronous fires occur (Baisan and Swetnam 1990, Swetnam and Betancourt 1998). Regional synchrony of episodic tree recruitment in the Southwest also coincides with wet conditions, with large regional pulses in the 1810s-60s and 1890s-1930s, which were exceptionally wet periods (Swetnam and Brown 2010). Thus climate, fire, and recruitment are intricately related.

I graphically and statistically compared tree recruitment and fire year chronologies with climate to assess climate forcing of fire years and recruitment episodes

(Brown and Wu 2005). I used superposed epoch analysis (SEA) in FHX2 version 3.2 (Baisan and Swetnam 1990, Grissino-Mayer 2001) to compare climate during fire years as well as for five years prior and two years following fire event (and non-fire event) years. I assessed statistical significance using SEA analyses with confidence levels (95% and 99%) calculated from bootstrapped distributions of climate data in 1000 iterations. SEA analyses of climate – fire interactions were calculated for fires recorded on $\geq$10% of recording trees, as well as for the most widespread fires recorded on > 50% of recording trees and for years when no fires were recorded in my sites. Recording trees are those with basal injury leaving them susceptible to repeated scarring by fire (Swetnam and Baisan 1996). Filtering, based on scarring percentage, provides a relative index of fire size and is generally used to estimate more widespread or relatively large fire years (Swetnam and Baisan 1996), which is useful in considering "safe-site" limited recruitment.

I also calculated fire return intervals for sites (individually and combined) that included all fire years, and for fire years in which $\geq$10%, and $\geq$25% of recording samples were scarred for the period of adequate sample depth, defined as the first fire year recorded by $\geq$10% of recording trees, until time of sampling or major disruption of fire events was apparent. Fire frequency statistics were calculated for ca. 45 ha in EPC, 63 ha in PDA and 63 ha within SPD. I tested for statistical difference in recruitment distribution among sites using a Kolmogrov-Smirnov test (MATLAB), and analyzed plot level forest demography and fire frequency data with FHX2 software, version 3.2 (Grissino-Mayer 2001). I compared tree recruitment across sites to cool season precipitation and PDSI with 20 year spline smoothing functions and also to fire frequency ($\geq$10% of recording trees).

Results

Stand structure and spatial patterns

I sampled 30 plots among three sampling sites (Tables 3.1 and 3.2). Plot sizes for ponderosa pine dominated sites ranged from 0.051 ha to 0.374 ha; the smallest plot area was 0.036 ha, which was a mix of Chihuahua and pinyon pines. A total of 589 plot trees (73% of all trees collected) were crossdated. Trees that could not be crossdated were primarily trees with too few rings (generally <100 rings) and very tight ring series for confident crossdating with a master chronology. For crossdated samples, 545 (93%) had pith present or pith dates could be determined based on inside ring curvature (within an estimated 10 years of exact pith dates). Regression estimates of age based on dbh performed well (n = 22 – 233, R^2 = 0.829 – 0.943, P < 0.001) and correctly assigned trees with known pre -1885 inner dates, but without exact pith dates, to the correct cohort (cohort one; pre-1885, > 125 yrs old). The majority of trees across all sites were relatively young with 96% under 200 years old (post 1811), and 55% under 100 years (post 1911).

Sites generally demonstrated similar distributions of tree recruitment (Fig. 3.2) with the only significant differences in distribution being between the Douglas-fir stands of EPMT and the pine stands of SPD (α = 0.05). Two periods of high recruitment occurred across all sites in the 1890s-1910s and 1940s-1960s. Interestingly, PDA, which was both grazed (beginning in early 1930's) and logged (early 1950's) had the fewest trees per hectare ($\bar{x}$ = 163.7) and highest basal areas ($\bar{x}$ = 16.3 m^2/ha) of the three pine dominated sites (Table 3.1), though density and basal areas do not differ significantly among sites. While I found similar general distributions of tree recruitment over time, there was considerable variability among plots (Appendix 3A). Age structure tended to follow one of three distributions; (1) a classic reverse-J shape, which is expected with continuous frequent fire disturbance and uneven age structure, (2) a flat even-age cohort establishing around 1900, or (3) a flat even-age cohort establishing around 1950, or some combination of these three. The majority of plots with a strong 1900 recruitment pulse were in SPD. The 1950s recruitment pulse was more common within plots in PDA, while reverse-J distributions were primarily limited to EPC.

Spatial patterns of trees varied among plots and by demographic patterns of recruitment with general agreement between Ripley's $K(t)$ and the Cramer-von Mises test for complete spatial randomness. In general, there was slight clustering of trees in the older tree age-classes limited to near distances (approximately 1-9 m, Fig. 3.3). Clustering disappeared entirely for plots with a reverse-J distribution, which were spatially random. Likewise, most plots with a strong 1950s cohort were also spatially random, with only one plot showing evidence of clustering. Conversely, older age-classes (pre-1895 and 1890-1930) were spatially clumped at near distances ($\leq$ 6 m) with only one spatially random plot. Spatial aggregation (*S* values close to 1 indicate high levels of aggregation within age-classes) of trees was also mostly limited to older age-classes (pre-1895 and 1890-1930, e.g. trees in the pre-1895 age-class were more likely to be aggregated with other trees also in the pre-1895 age-class) and was only found in the high elevation plots of EPC (EPMT) and in SPD, though in general I found only weak evidence of spatial segregation. Aggregation of the pre-1895 age-class was observed in four plots and aggregation of the 1890-1930 age-class in three plots, while the 1935-1975 age-class was found to be aggregated in two plots. In most plots age-classes were spatially independent (*S* values close to zero). Average segregation index values were negative (aggregation between different age-classes) in EPC (*S* = -0.053) and PDA (*S* = -0.062), though not statistically significant, and positive (aggregated) in SPD (*S* = 0.170) and EPMT (*S* = 0.273).

Fire, climate, and recruitment interactions

Fires burned relatively frequently until 1932 at EPC and PDA and continued to burn in SPD until 2000 (Fig. 3.4a-c), although with longer fire intervals after 1890. Mean fire intervals (MFI) ranged from 6.0 (PDA, 1745-1932) to 9.6 (EPC, 1847-1933) years at individual sites (all fires). MFI ranged from 6.0 years (PDA) to 12.3 years (EPC) when data were filtered to include only fire years in which 10% or more of recording samples were scarred (Table 3.3). With sites combined, MFI decreased to 4.5 years for all fires and 6.6 years for 10% filtered fire events (1728-2008). The maximum interval between fires differed among sites. In EPC, prior to 1933, the maximum interval was 14 years (1890-1904) for all fires and 24 years (1909-1933) for fires recorded on $\geq$ 10% of recorder trees. In PDA, which generally had more frequent fires and also more widespread scarring (Fig. 3.5a-c), the maximum interval was 12 years (1819-1831, all fires and also fires recorded on $\geq$10% of recorder trees). SPD had longer intervals from 17 years (1954-1971) for all fires to 26 years (1974-2000) for fires recorded on $\geq$ 10% recorder trees. This changed when considering a common period prior to 1932 for SPD when the maximum interval was 11 years for all fires (1794-1805), and 19 years (1890-1909) for fires recorded on ≥ 10% of recorder trees.

SEA analyses indicate significant interactions of climate and fire occurrence (Fig. 3.5). Both PDSI and cool season precipitation show an association of antecedent wet conditions followed by drought for fires recorded on ≥ 10% of recorder trees, though drought was only statistically significant for PDSI (at 99% confidence level, Fig. 3.4a-b). For the ten fire years with the most abundant evidence of burning (highest percent scarred: 1794, 1811, 1819, 1847, 1867, 1877, 1890, 1909, 1921, and 1932), antecedent conditions were significantly wet for two years prior to large fires with no apparent relationship with drought in year of fires (Fig. 3.5d). Four of these large fires years also covered the early (age-class 1) recruitment period. I further examined the relationship between climate (PDSI) and fire occurrence by isolating all years when no fires were recorded in my sites. These years were significantly wet years, but the antecedent year was significantly dry (95% confidence level) during fire-free years (Fig. 3.5c).

Comparisons of tree recruitment, fire, and climate by site indicate that fire intervals greater than 10 years were important for successful establishment of tree recruits, while climate did not appear to directly influence recruitment aside from effects on fire occurrence (Fig. 3.4a-c). The most obvious manifestation of this was the large spike in recruitment following reduced fire frequency in the mid 20th century, which was also centered on a severe 1950s drought. This period of increased recruitment was closely aligned with changes in fire frequency, which varied by site. In SPD a pulse of recruitment occurred around 1955, with the last widespread fire (≥ 10% recorder trees) occurring in 1946, and the next widespread fire was not until 1971. In PDA this pulse of recruitment began around 1945, the last widespread fires occurring in 1932. Similarly, recruitment in EPC spiked in 1935 following the last widespread fire year in 1933.

Within this period of amplified recruitment there was also evidence that smaller fire years reduced the number of recruits in each site. These relationships were not limited to the cessation or reduced fire frequency period in the mid 20th century. Within the ponderosa dominated sites (PDA and SPD) I found differences in recruitment around the turn of the century that can be explained by fire controls. In SPD the greatest magnitude of recruitment began in 1900 and dropped precipitously in 1910 following a widespread fire year in 1909. There was a 19-year interval between fires in 1890 and 1909. PDA shows a recruitment spike also following a fire in 1890, but this recruitment pulse dropped off with another widespread fire in 1900, a fire year not recorded in SPD. The turn of the century recruitment pulse in PDA was limited to a 10 year fire interval (versus 19-yr in SPD) and was a smaller recruitment event overall. Similarly, in PDA recruitment from 1830-1840 was limited to one plot where there was not evidence of the 1836 fire that was found in surrounding plots. It is noteworthy that this turn of the century recruitment pulse occurred prior to an exceptionally wet period in the 1910s-20s when climate presumably would be favorable for recruitment, but was also a period of frequent and widespread fires in 1909, 1916, and 1921. EPC had fewer fire-scarred trees in general, but a composite fire chronology shows frequent fires in this same period that may have limited successful tree recruitment into the overstory. In EPMT more continuous tree recruitment and few fires were evident, which was expected for higher elevation Douglas-fir sites that have different life history attributes than the lower elevation pine sites.

Discussion

While an increasing number of studies have shown the effect of anomalous climate years in overriding local processes in forest ecosystems (Swetnam 1993, Kitzberger et al. 1997, Brown and Wu 2005, Swetnam and Brown 2010), my research in northern Mexico suggests strong site specific, local controls on forest structure. I found that fire-free intervals, which varied locally within and across sites, had the strongest influence on successful tree recruitment. Recruitment was largely independent of broad scale climate effects. The greatest pulse of tree recruitment coincided with the most extreme drought in over 400 years (1942-57, Swetnam and Betancourt 1998, Sheppard et al. 2002). Similarly, the second largest pulse of recruitment in my sites largely coincided with below average precipitation conditions prior to 1900, and preceded an anomalously wet period in the 1910s-20s when pronounced recruitment occurred elsewhere throughout Western North America.

Recruitment pulses were strongly aligned with periods of reduced fire frequency, with site specific differences in fire and recruitment events. SPD, for example, had a greater recruitment pulse in duration and total numbers from 1890s-1930s than PDA which showed a spike from 1890-1900. The difference can be explained by the fact that PDA had six widespread fires from 1890-1932, while SPD had four, and with less synchrony of burning among plots. The opposite is true for the 1940s-60s when SPD had more fires and less recruitment (Fig. 3.5c) and both PDA and EPC showed a more prolonged recruitment pulse following greatly reduced fire frequency. Greater synchrony of burning among plots in PDA was likely a result of less topographic relief than the other sites and more contiguous forest stands. In PDA, maximum fire-free intervals (recorded on ≥ 10% of recorder trees and minimum of two trees scarred) were between 9-11 years, as opposed to 9-18 years in SPD and 9-23 years in EPC (Table 3.3). EPC had less synchrony in recruitment and fewer fires overall prior to 1935 compared to the other ponderosa pine dominated sites (PDA, SPD). However, EPC also shows a spike in tree recruitment in 1935 following the last fire recorded on ≥ 10% recorder trees in 1933.

Increased recruitment into the overstory following fire cessation has been widely observed in western forests, though generally these changes coincided with a climatically conducive period for recruitment (Savage et al. 1996), not drought. The critical role of fire and a lack of correlation to climate in shaping stand structure were also reported for sites in the nearby (< 90 km) Chiricahua Mountains (Barton et al. 2001).

Climate-fire interactions may help explain an apparent lack of relationship between timing of recruitment peaks and hydroclimate. Climate was a forcing agent for fire occurrence with a notable importance of antecedent conditions for fire years. Fires tended to occur in dry years following two significantly wet years (Fig. 3.5), a common and particularly pronounced pattern in the Southwest (Swetnam and Betancourt 1998). Apparently, the same multiple year wet conditions that might facilitate tree recruitment in fact limited recruitment by promoting fire occurrence. This is especially notable in the largest fire years (fires recorded on > 50% of recorder trees), where there was no apparent relationship to drought in year of fire occurrence, but again two significantly wet years prior to large fires. These climate-fire relationships may help explain why there are not greater magnitudes of recruitment during wet periods such as the 1910s-20s that are prevalent in sites without evidence of fire. Strong associations of wet-dry cycles in promoting fire occurrence may be more limiting to successful recruitment in fuel limited systems of the Southwest (Chapter 2), where significant moisture events are often followed by fire.

Spatial dependence among points was primarily limited to plots dominated by older trees that recruited into plots prior to 1895 (age-class 1) or from 1890-1930 (age-class 2) within approximately 9 m. This pattern was more apparent in SPD, which also had a greater number of plots dominated by age-classes 1 and 2. This finding offers limited support for the hypothesis that increased levels of clustering would be observed in older trees that recruited during periods of frequent fires and limited to “safe-sites” (White 1985), rather than “safe periods” (Brown and Wu 2005). Under this hypothesis, I would have also expected plots dominated by the 1935-1975 age-class, when fires were less frequent, to demonstrate little clustering. This was also supported as only one plot showed evidence of clustering for recruitment in this period.

Alternatively, tree recruitment controlled more by broad scale climate or competition between trees resulting in complete spatial randomness, or small scale regularity, across age-classes was not found. While plots dominated by the younger 1935-1975 age-class had spatially random distributions of recruitment, older age-classes demonstrated greater occurrence of clustering. Harvesting has also been found to increase aggregation of trees (Sánchez Meador et al. 2009), and stand density (Naficy et al. 2010). Interestingly, effects of tree harvest may vary by location. In Southwestern ponderosa pine, Fulé et al. (2002) suggest that historical logging may not produce as strong long-term density feedbacks as observed elsewhere (Naficy et al. 2010). This appears to be the case in my sites as well where PDA (harvested in early 1950s), for example, had lower density (163.7 trees/ha) and slightly higher basal area (16.3 m^2/ha) than similar unharvested sites in SPD (200.8 trees/ha, 15.4 m^2/ha). This difference appears to be an artifact of more frequent fires during 1890-1930 in PDA that limited tree recruitment in this time period and effected current forest structure.

Measuring spatial aggregation of tree cohorts was another way to investigate if broad climate effects resulted in widespread recruitment independent of local site specific factors. Most plots were spatially independent with segregation values (*S*) close to zero, but again I found differences particularly in older age-classes (1-2) where trees in age-class 1 and 2 were more likely to be aggregated with themselves (e.g. trees in age-class 2 aggregate with other trees in age-class 2) while trees in age-class 3 were spatially independent. In general, segregation values were highest (more aggregated) in SPD (mean $S = 0.170$) and EPMT (mean $S = 0.273$), though there was high variability among plots. Comparisons were limited to ~ 30 trees, which limits the ability for interpretation aside from relative comparisons. These results do suggest that not only are trees in plots dominated by older trees (> 80 years, age-class 1-2) clustered in space at near distances, but they are clustered with each other more often than for tree recruitment after 1935 (age-class 3). I found few trees that recruited after 1975 (age-class 4), which is partially a result of sampling only trees > 7.5 cm dbh, but generally few plot trees < 7.5 cm dbh were observed.

Conclusions

This study takes advantage of a unique opportunity to investigate tree recruitment in sites where fires continued to burn long after fires ceased burning in similar locations in the Southwest U.S. despite shared climate conditions. I conclude that fire-induced mortality played a crucial role in shaping current forest structure. Recruitment peaks were closely tied to local processes, notably fire-free periods, not broad-scale climate conditions. Fire-free intervals of > 10 years promote recruitment at local scales. Longer fire-free intervals promoted recruitment events in spite of coinciding drought. Furthermore, the importance of antecedent wet conditions promoting fire occurrence rises the possibility that in arid regions of the Southwest, anomalously wet years, still functioning under frequent fire occurrence, may further limit recruitment by promoting large fires. Forest structure in northern Mexico is a result of patterns of tree recruitment, though I suggest it is less an artifact of suitable climate for seedling establishment as it is seedlings in sites missed by lethal fires. The importance of fire-induced mortality in shaping stand structure underscores the spatial variability in forests and helps explain even-age patches in forests, not as an artifact of rare stand-replacing fire, but patch survival of seedlings that recruit into the overstory.

Table 3.1. Characteristics for three study sites in the Sierra San Luis, northern Sonora, Mexico.

Study site	Site code (*n* plots)	Elevation (m)	Species[ŧ]	Trees/ha	Basal area
El Pinito Canyon	EPMT (4)	2,240–2,430	PSME	263.5	29.0
	EPC (8)	2,104–1,815	PILE/PICE	342.2	10.4
Sierra Pan Duro	SPD (12)	2,217–1,979	PIPO/PIEN	200.8	15.4
Pan Duro Arroyo	PDA (6)	2,050–1,959	PIPO/PIEN/PILE	163.7	16.3

[ŧ] Species codes listed are for Douglas-fir (PSME, *Pseudotsuga menziesii*), Chihuahua pine (PILE, *Pinus leiophylla var. chihuahuana*), Pinyon pine (PICE, *P. cembroides*), Ponderosa pine (PIPO, *P. ponderosa var. arizonica*), and Apache pine (PIEN, *P. engelmannii*).

Table 3.2. Plots sampled to construct stand structure and fire history in the Sierra San Luis, northern Sonora, Mexico. Number of trees is for live trees only.

Site Code	Plot Area (m^2)	No. Trees	Trees/ Ha	BA	Elevation (m)	Aspect	Slope	Forest Type
EPC1	755	29	384	11	2,104	310	10	PILE
EPC2	1590	29	182	3.9	1,916	280	8	PILE/PICE
EPC3	607	15	247	20.4	1,822	320	2	PILE
EPC4	1320	28	212	8.5	1,814	145	2	PILE
EPC5	919	26	283	6.5	1,844	30	48	PICE
EPC6	598	30	501	21.9	1,820	350	13	PILE
EPC7	360	27	751	12.2	1,905	312	38	PICE
EPC8	1146	25	166	2.1	1,881	360	3	PICE
EPMT1	940	28	298	34.9	2,430	310	55	PSME
EPMT2	1134	26	229	23.1	2,440	300	60	PSME
EPMT3	679	11	162	16.1	2,332	60	50	PSME
EPMT4	1075	10	93	3.3	2,311	310	40	PICE
PDA1	1963	27	138	5.6	2,031	190	21	PILE/PIEN
PDA2	824	26	315	15.2	2,049	270	18	PIPO
PDA3	984	23	234	13.2	2,025	110	14	PIEN/PIPO
PDA4	1662	19	114	2	1,992	40	1	PILE/PIEN
PDA5	2827	22	78	3.3	1,992	100	16	PILE/PIEN
PDA6	2827	29	103	58.5	1,959	250	0	PILE
SPD1	3298	17	52	4.3	2,004	50	15	PINE/OAK
SPD2	598	17	284	33.3	1,981	18	42	PIPO/PSME
SPD3	1504	30	199	15.2	2,201	85	18	PIPO/PILE
SPD4	507	19	375	21	2,100	317	40	PIPO
SPD5a	1385	26	188	10.9	2,151	50	19	PIPO
SPD5b	3739	20	53	5.2	2,116	39	45	PIPO
SPD5c	1963	28	143	4.9	2,150	260	45	PIPO/PILE
SPD6	1662	18	108	6.2	1,979	320	43	MIXED
SPD7a	895	28	313	22	2,189	10	40	PIPO
SPD7b	531	25	471	26.2	2,217	312	23	PIPO
SPD8a	1957	30	153	26	2,100	10	40	PIPO
SPD8b	3217	23	71	9.9	2,150	335	25	PIPO

Table 3.3. Fire interval metrics (years) for three study sites in the Sierra San Luis, Sonora, Mexico. The analysis period was from the first fire year recorded by ≥ 10% of recording trees at each site until time of sampling or disruption of fire events were apparent. Analyses were further restricted to fires recorded by a minimum of two trees.

Site, analysis period	Category of analysis	No. intervals	Mean fire interval, MFI	Min	Max	Maximum interval period
Sierra Pan Duro	all scars	31	6.7	2	17	1954-1971
1794–2008	10% scars	22	9.4	3	26	1974-2000
	25% scars	17	12.1	3	28	1946-1944
Pan Duro Arroyo	All scars	31	6.0	1	12	1819-1831
1745–1932	10% scars	31	6.0	1	12	1819-1831
	25% scars	24	7.8	3	17	1819-1831
El Pinito Canyon	all scars	9	9.6	4	14	1890-1904
1847–1933	10% scars	7	12.3	4	24	1909-1933
	25% scars	5	17.2	4	43	1890-1933
Sites Combined	All scars	61	4.5	1	11	1989-2000
ALL, 1728–2008	10% scars	41	6.6	1	28	1946-1974
	25% scars	24	11.1	1	28	1946-1974

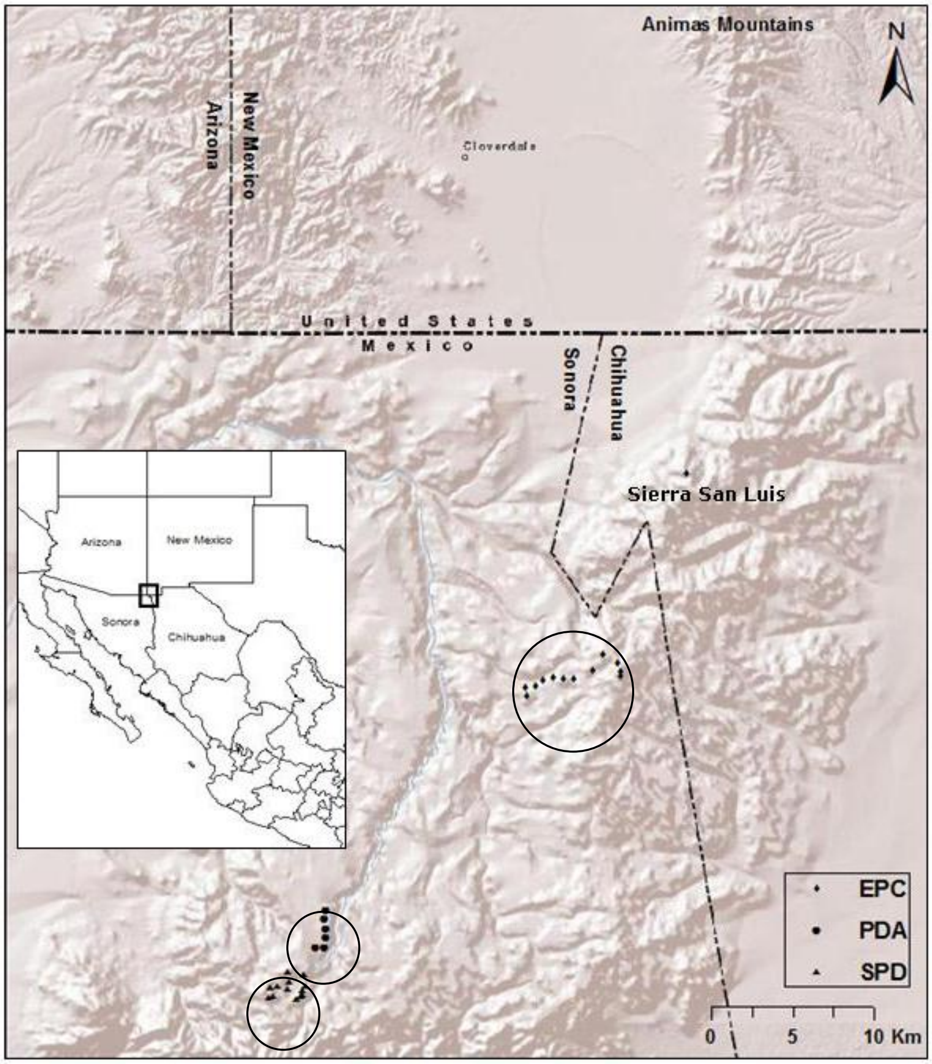


Figure 3.1. Study area and sampling sites in the Sierra San Luis, Sonora, Mexico. Sites were sampled from 2008 – 2010 within and surrounding randomly placed plots at three sites (from south to north): Sierra Pan Duro (SPD, little to no grazing or logging), Pan Duro Arroyo (PDA, logged and grazed after 1930), and El Pinito Canyon (EPC, grazed after 1930 but not logged).

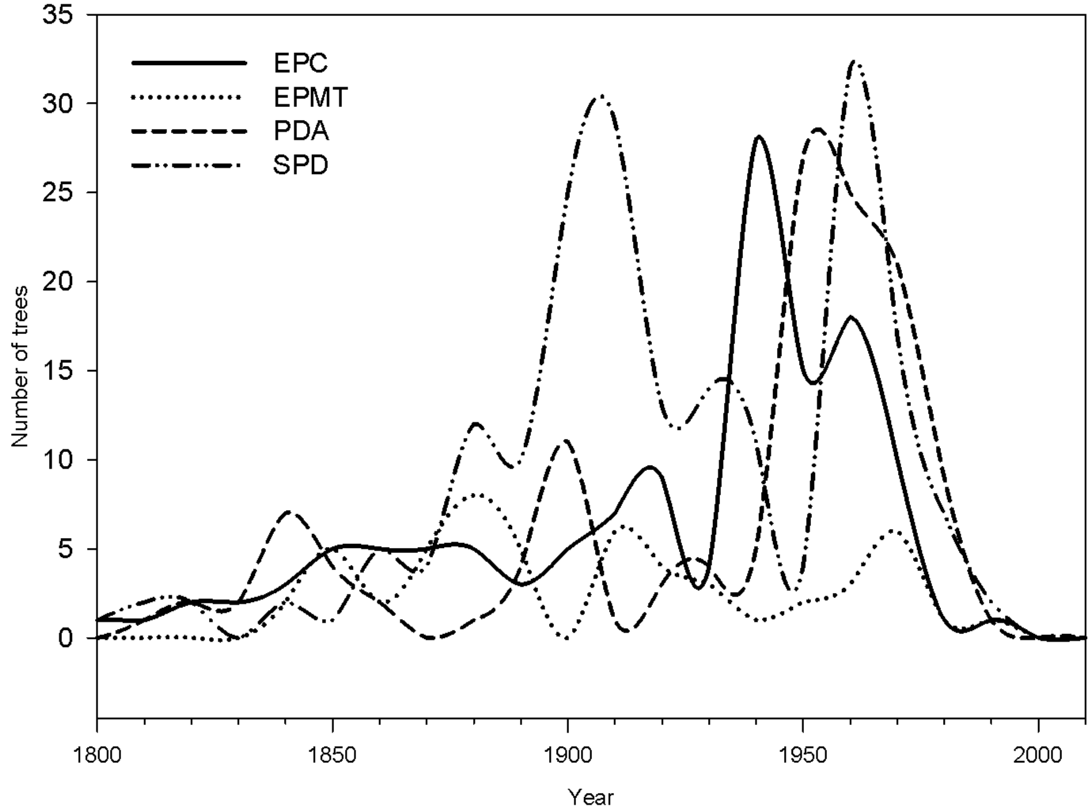


Figure 3.2. Tree recruitment by site for 545 trees with accurate pith dates. Recruitment is defined as 10 cm pith dates. El Pinito Mountatin Top (EPMT) consists of Douglas-fir stands and is separated from EPC and other pine dominated sites (PDA, SPD) for comparison by species.

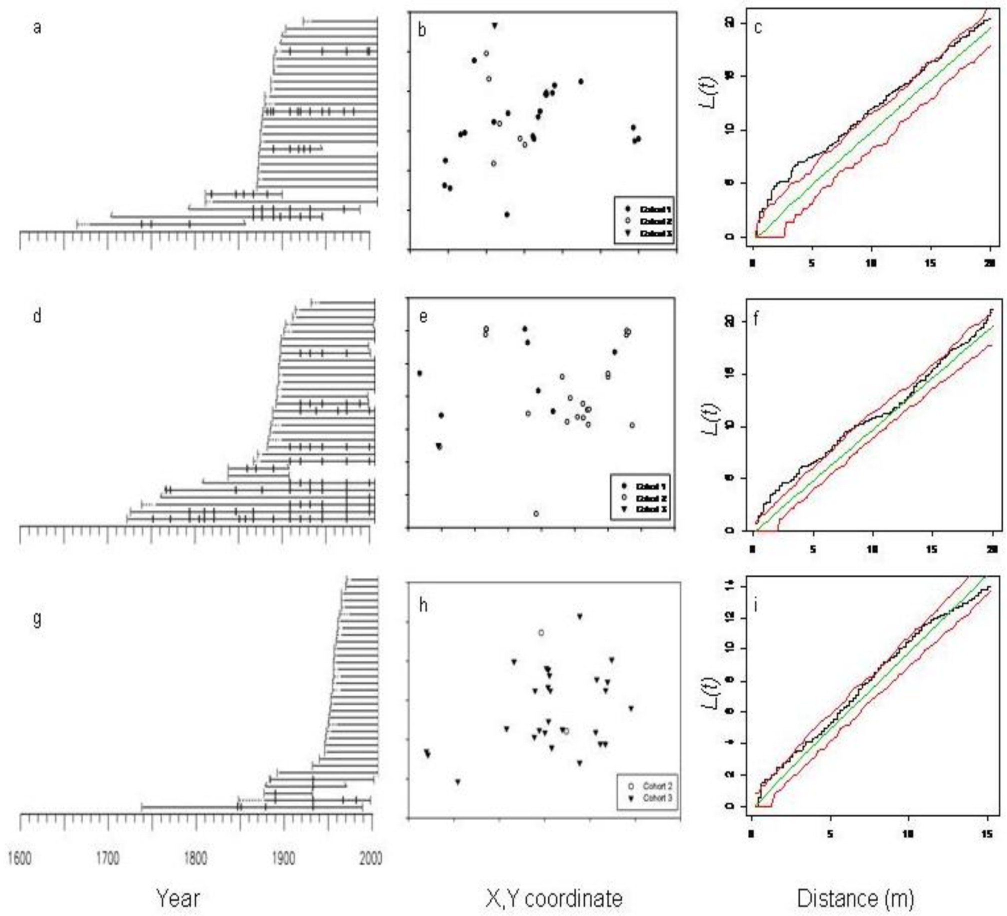


Figure 3.3. Comparison of tree recruitment distribution, spatial distribution, and spatial patterns of trees for three plots that are dominated by recruitment in three age-classes: 1 (a-c) pre-1895, 2 (e-f) 1890-1930, or 3 (g-i) 1935-1975. Tree recruitment distribution includes plot trees and additional fire-scar samples within 200m of plots. Horizontal lines are life spans of individual trees and bold vertical hashes are fire-scars. Spatial distributions are x, y coordinates of individual trees by age-class (b, e, h) and spatial pattern results are from Ripley's $K(t)$ with an $L(t)$ square root transformation for variance stabilization (c, f, i). 95% upper and lower confidence envelopes for a Poisson model are plotted as a function of lag distance from an arbitrary individual. Data above the upper limit (red line) indicate spatially clumped data.

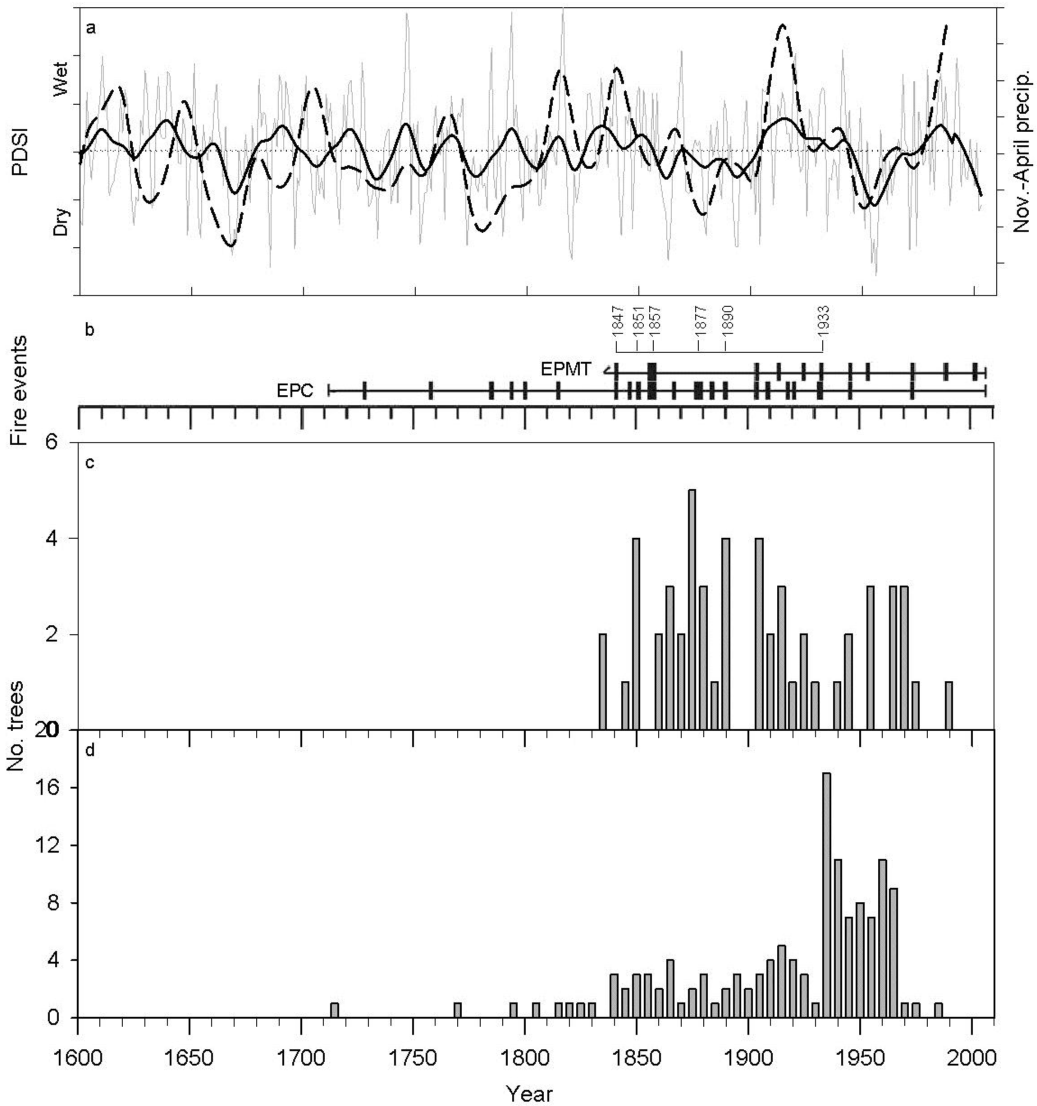


Figure 3.4a. Comparison of hydroclimate, fire-year, and tree recruitment chronologies for El Pinito Canyon. (a) Reconstructed hydroclimate time series, the light grey line shows Palmer drought severity index for the nearest 4 grid points (Cooke et al. 2004) and the solid line is smoothed PDSI. The heavy dashed line is cubic spline smoothed cool season precipitation (Nov.-Apr.) for southeast New Mexico and southwest Arizona (Ni et al. 2002). (b) Composite fire-year chronologies for minimum of two trees scarred for EPMT and EPC, horizontal lines represent composited plots tree time spans and heavy vertical dashes are fire-scars. (c, d) Tree recruitment dates by 5-yr periods for (c) high elevation Douglas-fir stands (EPMT), (d) EPC.

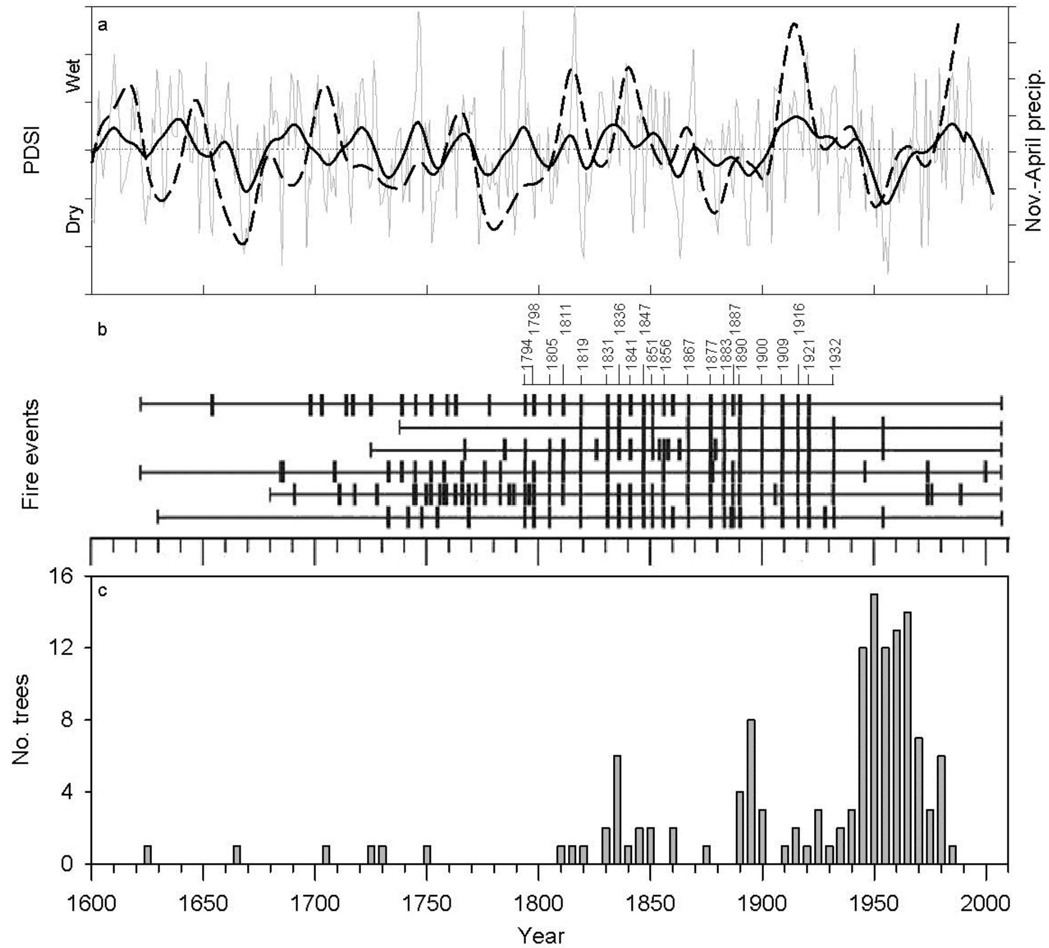


Figure 3.4b. Comparison of hydroclimate, fire-year, and tree recruitment chronologies for Pan Duro Arroyo. (a) Reconstructed hydroclimate time series, the light grey line shows Palmer drought severity index for the nearest 4 grid points (Cooke et al. 2004) and the solid line is smoothed PDSI. The heavy dashed line is cubic spline smoothed cool season precipitation (Nov.-Apr.) for southeast New Mexico and southwest Arizona (Ni et al. 2002). (b) Composite fire-year chronologies for all fire-scar trees scarred by plot. Horizontal lines represent composited tree time spans for each plot and heavy vertical dashes are fire-scars. (c) Tree recruitment dates by 5-yr periods.

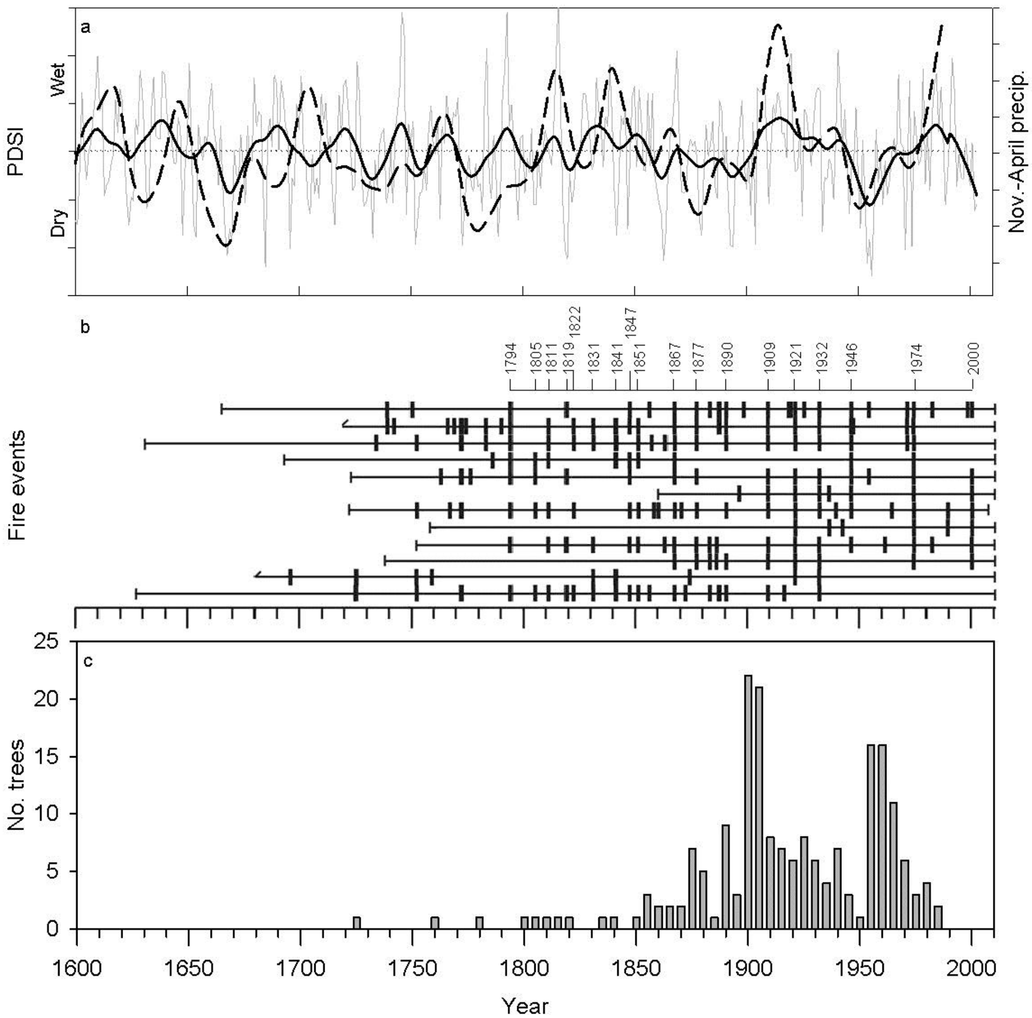


Figure 3.4c. Comparison of hydroclimate, fire-year, and tree recruitment chronologies for Sierra Pan Duro. See caption 4b for details of analysis.

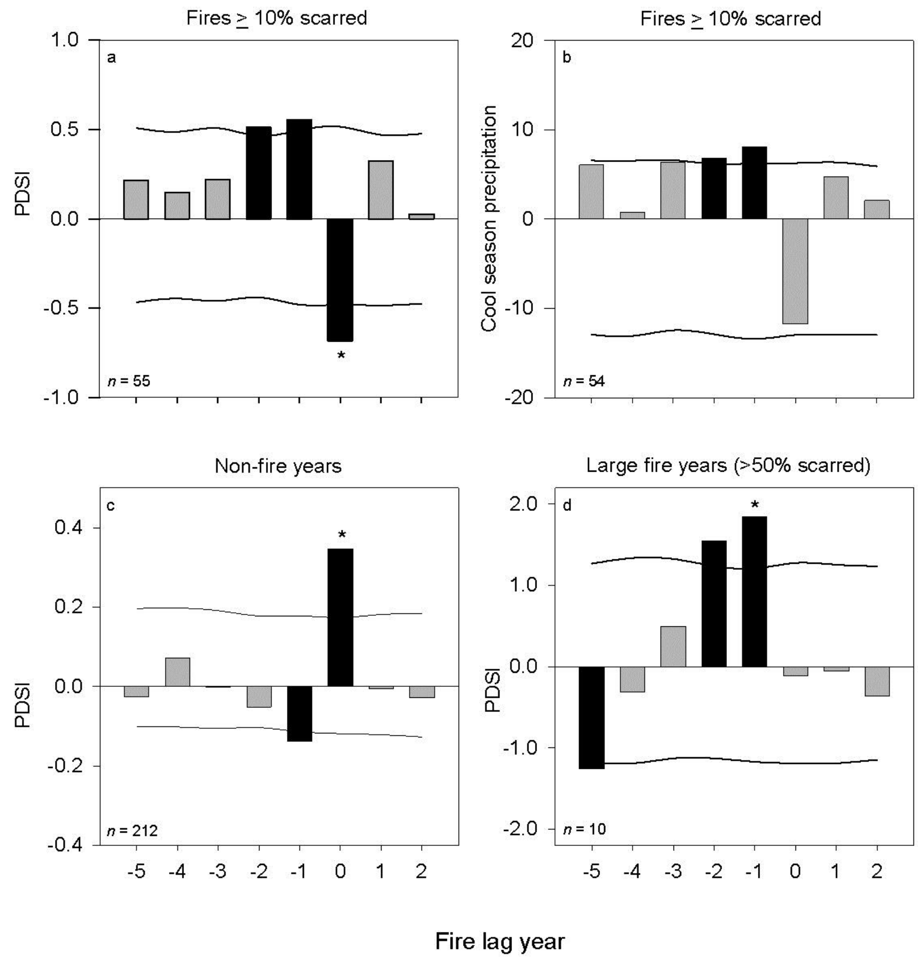


Figure 3.5. Results of superposed epoch analysis (SEA) of tree-ring reconstructions of Palmer Drought Severity Index (a, c, d, PDSI 1650-2003, Cook et al. 2004), and cool season precipitation (b, 1650-1988, Ni et al. 2002) for years prior and subsequent to event years (year 0). Positive PDSI values indicate wet conditions, negative values represent dry conditions. Data shown are for years when fires recorded on ≥ 10% of samples (a, b), no fires were recorded in any of our sites (c), and large fires (d) recorded on > 50% of recorder trees 1650-1886. Solid bars indicate PSDI values outside a 95% confidence interval (depicted by lines); asterisk symbols indicate values outside a 99% CI; All CI's are based on 1,000 Monte Carlo simulations of random distributions of annual PDSI. Sample sizes are identified for the number of fire events tested against PDSI data.

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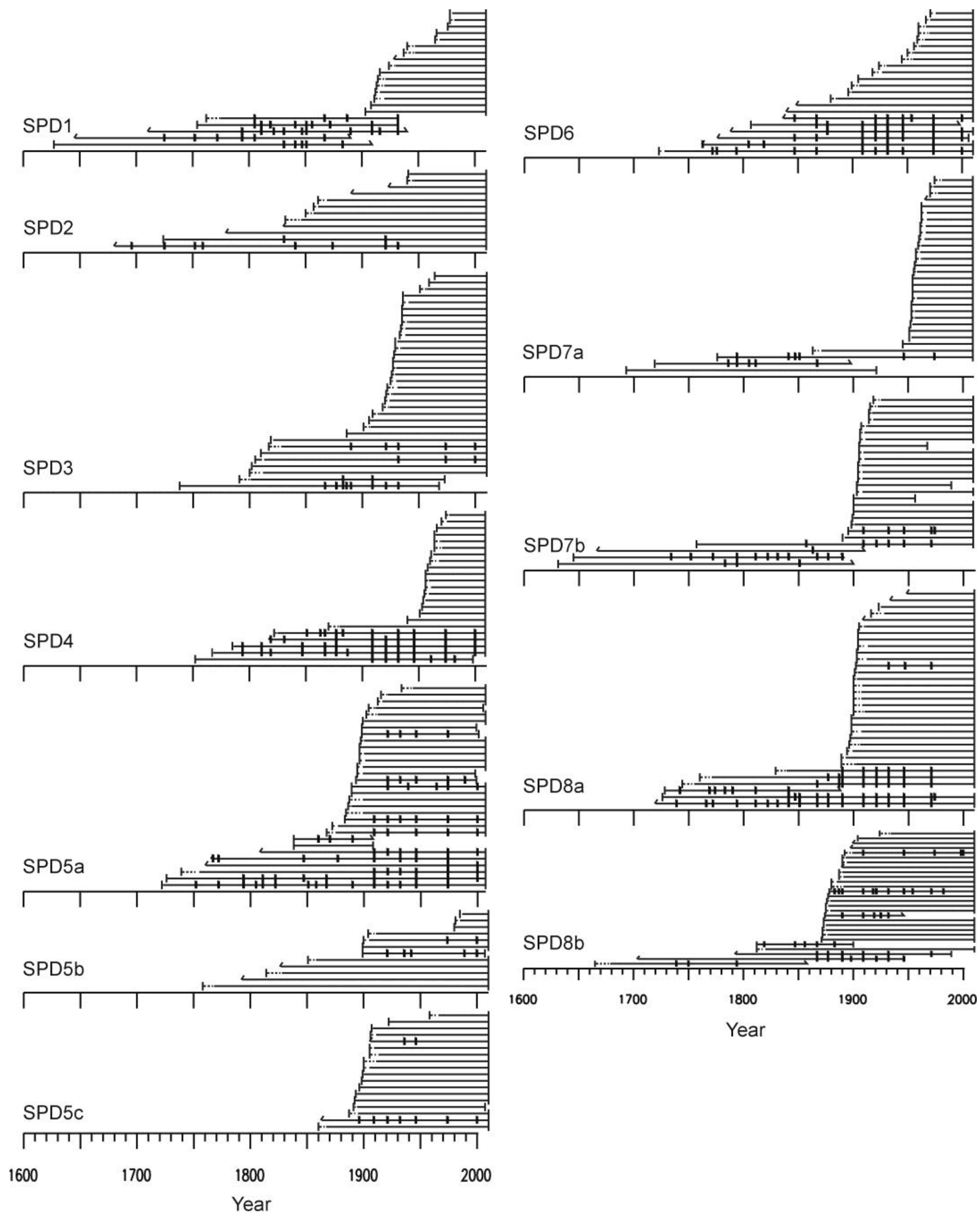
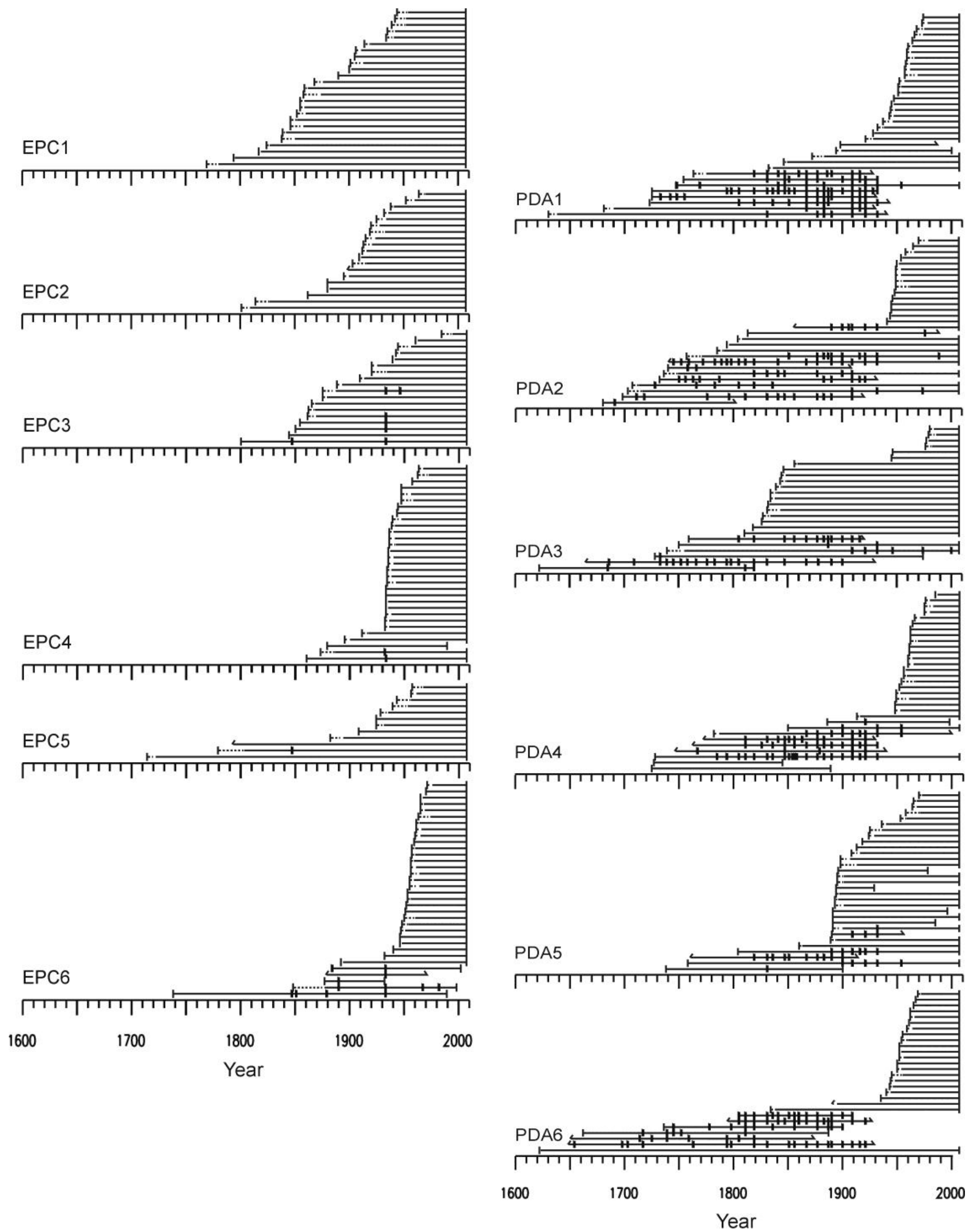
Appendix 3A. Plot level tree demography showing recruitment, mortality and fire events over time. Note: the plot trees are graphed in addition to surrounding fire-scarred trees which are generally older trees that extend the chronology, and which are near the bottom of the graphs.




EPC1
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EPC3
EPC4
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PDA2
PDA3
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