

CLIMATE AND HUMAN DRIVERS OF FOREST VULNERABILITY IN THE US  
SOUTHWEST: PERSPECTIVES FROM DENDROECOLOGY

by

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SIGNED: *Christopher H Guiterman*

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## DEDICATION

To the Navajo Forestry Department,  
in your efforts to maintain Navajo forests for the benefit of the Diné in perpetuity.

## TABLE OF CONTENTS

ABSTRACT.....	9
CHAPTER 1 - INTRODUCTION.....	11
The problem.....	11
Some key questions.....	14
Organization of the dissertation.....	15
Motivation.....	17
CHAPTER 2 - PRESENT STUDY.....	20
Fire history of Navajo forests.....	20
Gambel oak shrubfields and high-severity fires.....	22
Climate-forest growth across Navajo forests.....	24
REFERENCES.....	26
APPENDIX A - SETTLEMENT, PASTORALISM, AND ASSOCIATED CHANGES TO FREQUENT SURFACE FIRE ACTIVITY ON THE NAVAJO NATION.....	35
ABSTRACT.....	37
INTRODUCTION.....	38
STUDY AREA.....	43
Setting.....	43
Human history.....	45
METHODS.....	48
Site selection.....	48
Fire history reconstruction.....	50
Analyses.....	51
RESULTS.....	53
DISCUSSION.....	54
ACKNOWLEDGMENTS.....	60
REFERENCES.....	61
TABLES.....	69
FIGURES.....	71
APPENDIX B - DENDROECOLOGICAL METHODS FOR RECONSTRUCTING HIGH SEVERITY FIRE IN PINE-OAK FORESTS.....	76
ABSTRACT.....	78
INTRODUCTION.....	78
METHODS.....	82
Study site.....	82
1993 Buchanan Fire.....	83
Field sampling.....	84
Laboratory methods.....	85
Analysis.....	86

TABLE OF CONTENTS - *Continued*

RESULTS .....	87
DISCUSSION .....	89
Tree-ring reconstruction methods .....	89
Current vs. historical fire regime .....	92
CONCLUSIONS.....	93
ACKNOWLEDGEMENTS .....	94
REFERENCES CITED.....	94
TABLES .....	99
FIGURES .....	100
APPENDIX C - LONG-TERM PERSISTENCE AND FREQUENT FIRE IN NORTHERN NEW MEXICO SHRUBFIELDS SUGGEST FUTURE INCREASING LANDSCAPE DOMINANCE.....	104
ABSTRACT.....	106
INTRODUCTION .....	106
METHODS .....	111
Study area.....	111
Identification and evaluation of pre-1900 potentially fire-origin shrubfields ....	113
Site selection and field sampling .....	114
Laboratory procedures .....	116
Analyses.....	117
RESULTS .....	118
DISCUSSION.....	121
Potential for fire-origin of pre-1900 shrubfields.....	125
Fire histories of Gambel oak shrubfields and dry conifer forests.....	127
REFERENCES .....	129
TABLES .....	139
FIGURES .....	141
APPENDIX D - RECENT SYNCHRONIZED TREE GROWTH RESPONSE TO CLIMATE ACROSS A LARGE SEMIARID FOREST IN THE SOUTHWESTERN US .....	148
ABSTRACT.....	150
INTRODUCTION .....	151
METHODS .....	156
Study Area .....	156
Site selection .....	157
Field sampling.....	159
Chronology development.....	161
Analyses.....	162

TABLE OF CONTENTS - *Continued*

RESULTS .....	163
DISCUSSION .....	166
ACKNOWLEDGMENTS .....	171
REFERENCES .....	172
TABLES .....	178
FIGURES .....	180

## ABSTRACT

The ongoing drought in the US Southwest (SW) has led to particularly large and severe wildfires, tree die-off events, insect outbreaks, and increased forest stress levels. These disturbances underscore the vulnerabilities of SW dry conifer forests to climate change and past land-uses. Climate projections show a clear upward trend in regional temperatures, which will lead to accelerated heat-related stressors and disturbances in the coming decades. Already, more than 20% of the dry conifer forests of the SW have been severely impacted. This number is likely to grow, but we lack a clear picture of where, when, and to what degree other forest areas will be affected.

Here, I apply dendroecological methods to evaluate patterns and processes that might determine greater or lesser vulnerability in dry conifer forests. Much of this work stems from critical concerns voiced by the Navajo Forestry Department (NFD). Long-term and representative data are necessary for the NFD, as they are responsible for closely managing over 250,000 hectares of forests and woodlands for the traditional products and ecosystem services that their forests provide for the Navajo people.

The first study takes a multi-century perspective on changing fire regimes across Navajo forests, and places current forest structure and recent severe events in a long-term context. We found that surface fires were frequent across the landscape from at least the late-1500s until 1880. Navajo settlement of the area began to affect the fire regimes with added small fires in some areas beginning in 1700. By 1832, the rise of traditional pastoralist practices and transhumant migrations reduced fire activity in areas of greater use. Conditions changed following the establishment of the Navajo reservation in 1868, as livestock herds grew rapidly and initiated a near-synchronous and widespread collapse of fire regimes across the study area by 1880. The legacies of this change in land use are greater forest densities and higher fuel loads in some areas, raising the vulnerability of the forest to more severe fires.

The second study assesses one of the most dramatic long-term consequences of recent high-severity fires in the SW, the rapid post-fire transition of dry conifer forest to oak-dominated shrubfields. To assess probable successional trajectories and interactions with climate change of recently converted forests, we reconstructed the age structures and

fire regimes of some of the largest and oldest shrubfields in the Jemez Mountains of northern New Mexico. We found that shrubfields are a resilient configuration to drought and fire, historically burning at the same rates and under similar climate as dry conifer forests. Dense shrubfields pose a significant challenge to conifer recruitment from competition effects, with our sites persisting for over 100 years in the absence of burning and through periods of favorable climate to conifers. Given future warming favoring oak over pine, and projected trends in fire activity and high-severity fire behavior, we expect much more forest area to convert to shrubfield, especially in the absence of restoration efforts to reduce crown fire potential.

Finally, we evaluate landscape-scale variability in tree growth response to regional climate across the Navajo forest. Projections of climate-induced forest decline often omit upper-elevation and mesic sites because they are not represented in regional tree-ring chronology networks. We found that these stands had much lower response to 20<sup>th</sup> century droughts than mid to lower elevation stands, and that targeted tree-ring sites are consistently more correlated with regional climate. However, as temperature-driven atmospheric moisture demand has remained above average since ~1997 in the study area, the upper-elevation trees are now nearly as responsive to regional climate as lower elevation xeric sites, probably due to increased moisture limitations. Recent warming has thus synchronized tree growth to an unprecedented extent across this large landscape and regionally.

## CHAPTER 1 - INTRODUCTION

### **The problem**

Recent large and severe disturbances in dry conifer forests of the US Southwest underscore forest vulnerabilities to climate change (Williams et al. 2013; Allen et al. 2015; Allen 2016). I define the US Southwest (SW) as primarily Arizona and New Mexico, including proximate locations, but the patterns and processes of change documented in these forests extend to many areas of the western US and other semiarid regions globally. Dry conifer forests of the SW include low-elevation piñon-juniper (*Pinus edulis* – *Juniperus* spp.) woodlands, mid-elevation ponderosa pine (*Pinus ponderosa*) dominated forests, and higher elevation dry mixed conifer forests dominated by Douglas-fir (*Pseudotsuga menziesii*), true fir (*Abies* spp.), aspen (*Populus tremuloides*) and ponderosa pine (*sensu* Romme et al. 2009).

The dry subtropical SW region is characterized by warm temperatures, clear skies, and variable precipitation (Sheppard et al. 2002). The precipitation regime is influenced by both the Pacific North America pattern of the subtropical Jetstream in winter and the North American Monsoon in summer (Adams and Comrie 1997; Sheppard et al. 2002). This creates a bimodal precipitation pattern, with up to 50% of annual precipitation occurring July-September in some areas. Interannual to decadal variability in precipitation is strongly associated with ocean-atmosphere teleconnections, with El Niño Southern Oscillation exerting the predominant control over sub-decadal precipitation variability (Steenburgh et al. 2013).

Climate projections for the region show a clear trend of rising temperatures, especially under “business-as-usual” greenhouse gas emission scenarios (Garfin et al.

2013; IPCC 2014). Much less confidence can be ascribed to projected trends and variability in precipitation, mainly because of considerable uncertainty in monsoon dynamics with climate change. However, based on multiple reconstructed decadal-length “megadroughts” over the last millennium (Woodhouse and Overpeck 1998; Cook et al. 2004), and projected climate variability, the risk of long and severe droughts in the SW is exceedingly high for the near future (Ault et al. 2014; Cook et al. 2015). Already, recent drought episodes across multiple semiarid regions have been among the worst in at least 500 years (Touchan et al. 2008, 2011; Williams et al. 2013; Griffin and Anchukaitis 2014; Belmecheri et al. 2016).

Future warming climate could exponentially increase drought severity (Williams et al. 2013). Temperature amplifies atmospheric moisture demand with concomitant effects on evapotranspiration and plant water stress (Weiss et al. 2009; Breshears et al. 2013). Thus temperature-driven aridity increases drought stress in trees, which reduces growth, promotes forest decline, and raises the probability of mass mortality (Williams et al. 2010, 2011, 2013; Liu et al. 2013; Asner et al. 2015; Charney et al. 2016; Restaino et al. 2016). Rising temperatures reduce soil moisture, which affects plants, but can lead to reduced runoff and streamflow (Woodhouse et al. 2016). Projected losses in water availability due to these mechanisms are likely to have significant societal impacts (Cayan et al. 2010; Seager et al. 2012).

Increased temperatures act to synchronize and accelerate forest disturbance processes. Periods of increased forest drought stress in the SW closely correspond to tree-killing disturbances (Williams et al. 2013). Above average temperatures are associated with heat-related tree die-off events (Allen et al. 2010, 2015), continental-scale bark

beetle outbreaks (Raffa et al. 2008), rising levels of background forest mortality (van Mantgem et al. 2009; McDowell et al. 2016), and a longer fire season leading to increased burn area (Westerling et al. 2006; Littell et al. 2009; Dennison et al. 2014; Westerling 2016). Projections for the SW show major losses of forest area by ~2050 (Williams et al. 2013; McDowell et al. 2016), with the worst years of the worst drought of the last millennium predicted to be more like average conditions (Williams et al. 2013).

Anthropogenic warming accounts for a doubling of the area burned in the western US since 1984 (Abatzoglou and Williams 2016). In some areas, recent fires have included unusually large patches of high-severity burning (Miller et al. 2009; Allen 2016). These patterns of recent forest fire are imprinted on the legacy of a century or longer of fire exclusion. The lack of fires since the late 19<sup>th</sup> century is attributed to human activity as opposed to climate change (Swetnam et al. 2016; Taylor et al. 2016), resulting from large-scale livestock introductions, Native American population declines, industrial logging, and active fire suppression (Pyne 1982; Swetnam and Baisan 1996, 2003; Swetnam et al. 2001; Allen 2007; Fulé et al. 2012). Decades of above-average precipitation in the SW during the early 20<sup>th</sup> century (Woodhouse et al. 2005) spawned regional tree recruitment in dry conifer forests (Savage et al. 1996; Swetnam and Betancourt 1998; Swetnam and Brown 2011). Higher stand densities now contribute to an increased probability of high-severity crown fire (O'Connor et al. 2014; Stephens et al. 2015).

The new climate reality in the SW is thus one of “hot droughts” (Overpeck and Udall 2010) with widespread and severe disturbances (Overpeck et al. 1990). Future hot

droughts are likely to be more frequent, longer lasting, and more severe than during the 20<sup>th</sup> century (Breshears et al. 2005; Adams et al. 2009; Williams et al. 2014). It is these events and their associated severe disturbances that are likely to trigger the greatest changes in future forest structure, composition, and productivity (Jentsch et al. 2007; Jackson et al. 2009; Zhao and Running 2010; Falk 2013). This will impart substantial down-stream impacts to biodiversity, infrastructure, forest products, traditional forest uses, and vital ecosystem services (Wondzell and King 2003; Breshears et al. 2011; Clark et al. 2011; Anderegg et al. 2012; Lynn et al. 2013; Voggesser et al. 2013).

### **Some key questions**

There are various aspects of regional climate, forest configurations, genetics, and disturbance regimes that determine different levels of forest vulnerability to climate change at the global scale (Allen et al. 2015). It is therefore vital that we continue to evaluate how landscape diversity, forest structure, management, and legacies of disturbance and land-use contribute to greater or lesser forest vulnerabilities at sub-regional and finer scales. Because trees tend to operate at the margins of drought resistance (Choat et al. 2012), rising temperatures are likely to affect all forests across the SW and many other regions (Allen et al. 2015). The questions are where, when, and to what degree?

We lack a clear picture of how future drought will be manifested across diverse landscapes. Some areas are very likely to have already reached tipping points due to various climate and human factors that can erode resilience to disturbance (Falk 2013; Allen 2016). Some sites can abruptly transition to new configurations following

disturbances (Scheffer et al. 2001), as is well documented in SW pine-oak forests following high-severity fires (e.g., Savage and Mast 2005). Others may be more resilient due to fine-scale variability in temperature and soil moisture associated with physiographic position and stand structure (Fritts 1976; Allen and Breshears 1998). The legacies of disturbances and disturbance regimes can also impart a degree of resiliency (Roos 2008; Johnstone et al. 2016). Forest management and ecological restoration of forest processes such as fire are avenues toward greater resilience (Stephens et al. 2016). For example, stand structure and tree survival were higher in treated versus untreated areas following a severe fire during an exceptionally dry year (the 2002 Rodeo-Chediski Fire; Strom and Fulé 2007). Thinning stands is also likely to reduce drought effects as more resources are made available to crop trees (Sohn et al. 2013, 2016).

Therefore, there is a degree of uncertainty as to where, when, and to what degree disturbances and climatic stress associated with future hotter droughts will play out within the SW region, despite predictions of widespread, synchronous forest decline (e.g., Williams et al. 2013). The work presented here is intended to contribute to a growing evaluation of forest vulnerabilities.

### **Organization of the dissertation**

In this dissertation, I examine several aspects of forest vulnerabilities to changing disturbance regimes, past human activities, and increased temperature. The first paper (Appendix A) represents a critical first step in understanding present conditions and future trajectories of a large and relatively unsampled landscape in the SW, the Chuska Mountains-Defiance Plateau on the Navajo Nation. The fire history of the landscape

examined in the paper highlights the uniqueness of the place, and the intricacies that its Native American population both past and present has imparted upon it. For restoration efforts to take place there, this detailed long-term context is vital (Stephens et al. 2013), especially with greater knowledge of the interactions between traditional land-use practices and the fire regimes. In the second and third papers (Appendices B and C), I examine one of the most profound consequences of recent high-severity fire: the abrupt transition of conifer-dominated forest to oak-dominated shrubfields. While these transitions are well documented across the western US, there is little long-term information on their successional trajectories or disturbance patterns. Thus, there is some debate regarding whether post-fire shrubfields will persist as alternative stable states to conifer forests or succeed back to conifers over the course of decades (Falk 2013). Our results support the former, especially given future warming in the region. Furthermore, we can expect more shrubfields to form if trends in large-scale high-severity fires continue. This is an example of where forest managers will need to adapt, or get ahead of these transitions with restoration efforts. Finally, the fourth paper (Appendix D) examines the landscape scale variability of climate-forest growth relationships, and tests whether targeted tree-ring site collections might be exaggerating the response of forests to future warming (e.g., Williams et al. 2013; Charney et al. 2016). Our non-selectively sampled sites represent the variety of topographic and elevational site conditions present across a large landscape, and show that drought vulnerability – or at least the response of trees to drought – is much reduced at higher elevations due to generally wetter conditions and lower atmospheric moisture demand. These sites, therefore, have a lower vulnerability to climate change overall, but recent increases in temperature may be beginning to exert

some control, as climatic sensitivity and drought response in tree growth have reached levels unsurpassed over the last century. Increased temperatures may be imparting greater moisture limitation, which implies that future droughts could significantly impact these stands.

## **Motivation**

Much of the research presented in this dissertation was conducted on the Navajo Nation in partnership with the Navajo Forestry Department (NFD). Study sites are located throughout the Chuska Mountains and Defiance Plateau, a 250,000 ha ponderosa pine-dominated forest on the northern Arizona-New Mexico border. This is the primary management landscape of the NFD and is one of the most unique forests in the SW. The Navajo people (or more appropriately, *Diné*) have lived in the Chuska Mountains seasonally for centuries as part of their tradition of transhumance - migrations with livestock - to “sheep camps” set near small lakes on the Chuska Crest. The scattered hogans, cabins, and corrals, with horses, sheep herds, and other animals roaming the forests and meadows of the Chuskas makes this landscape an enchanting place for me to work. The deep histories of this place, and their presence in the tree-ring collections we have made, have left a remarkable impression on me. My awe and respect for it and its people have also raised my concerns for its future, and their futures. I am often overwhelmed by my sense for the impending environmental stressors projected by climate change models and exacerbated by changes in forest compositions and structure related to human land uses here and elsewhere. But this is not something to fear and ignore, it is my driving motivation to make my research count. In other words, I strive to

conduct research with the NFD that is credible, usable, and relevant (Cash et al. 2002; Lemos et al. 2012). There is a great need to act and action requires knowledge. I am privileged to contribute to this effort.

I began working in the Chuska Mountains in 2011, when Jeff Dean, Chris Baisan, and I visited Narbona Pass with the ambition to build a ponderosa pine tree-ring chronology that extended into the period of occupation at Chaco Canyon (850-1130 CE). We were successful, and the Narbona Pass chronology<sup>1</sup> helped to distinguish the Chuska Mountains as the primary timber source area for the iconic great houses of Chaco Canyon (Guiterman et al. 2016). This helps to demonstrate the significance of the Navajo forests in ancient and modern societal development of the SW, as well as a very long history of human interactions. For our initial and later expeditions to the Chuskas, I visited the NFD offices to retrieve sampling permits, and began a years-long discussion with the head forester, Alex Becenti. Our conversations eventually led to his invitation for me to formally meet with and present my findings to the department. Over the course of our day and a half meeting in March, 2013, I described what I do as researcher, what some of my main interests are, and mainly let the tree-ring samples do the talking. I had the opportunity to ask what I can do for them, and what needs they had for information and research? These simple questions launched the projects I describe below. Their primary concern regarded the vulnerabilities of their forests to climate change.

Underlying the more basic scientific investigations and hypothesis tests of my dissertation papers is the inherently applied nature of the work. From the start, this research program has been directed at delivering to the NFD usable information regarding research questions that *they posed*. But I am also addressing what I find to be

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<sup>1</sup> ITRDB NM589: <https://www.ncdc.noaa.gov/paleo/study/19280>

some of the most pressing scientific questions we now face. My research, therefore, falls under the framework of “use-inspired research” (Stokes 1997). To date, I have presented research findings to the NFD four times and will continue this effort. This document will be provided to the NFD and an additional report on these projects will be delivered in 2017. The intention of the NFD is to incorporate these data and interpretations into their next 10-year management plan, and my sincere hope is that our findings answer their questions.

## CHAPTER 2 - PRESENT STUDY

This dissertation includes three research projects described in four appendices. Each appendix is either a published or publishable paper for which I am or will be the corresponding author upon publication. I led in the research design, implementation, analyses, interpretation, and writing. My perspectives and ideas were vastly improved by the input of my co-authors and mentors as the projects developed. In this section I review the context, methods, and key findings of each study.

### **Fire history of Navajo forests**

The Southwest-wide collapse of surface fire activity in the late 19<sup>th</sup> and early 20<sup>th</sup> centuries is associated with large-scale livestock introductions, industrial logging, the removal and decline of Native American populations, and later decades of above-average precipitation and active fire suppression efforts (Pyne 1982; Swetnam and Baisan 1996, 2003; Swetnam et al. 2001; Allen 2007; Fulé et al. 2012). This multitude of coincident factors challenges our ability to disentangle individual causes, but this effort is important for restoration of surface fires as a keystone ecological process in the resilience of forests to drought and high-severity fire (Swetnam et al. 1999, 2016; Allen et al. 2002; Stephens et al. 2016).

The study described in Appendix A takes place across roughly 150,000 ha of ponderosa pine forests in the Chuska Mountains and Defiance Plateau, situated along the northern Arizona-New Mexico border on the Navajo Nation. This area has a unique and relatively well-documented human history that allows for a robust assessment of several key factors in the disruption of surface fire activity. Our study includes eight tree-ring

based fire history sites from our collections and that of a previous study (Savage and Swetnam 1990). We divided the sample sites based on *a priori* information on Navajo settlement and land-use from ca. 1690 to present into “higher-use areas” and “lower-use areas.”

We found changes in surface fire regimes within and between these land-use groups that closely coincided with distinct eras of Navajo history. From 1597 (the earliest year of high confidence in our reconstruction) to 1700 the fire regimes of both groups show frequent surface fire activity during a period when probably few if any people lived within the study area and fires were “free ranging.” From 1700 to 1832 the two groups diverge, with higher-use areas recording nearly twice as many fires as in the lower-use areas, which did not change from the pre-1700 fire regime. This addition of fires in more populated areas is consistent with patterns from many other semiarid and fuel-limited forests (see Swetnam et al. 2016; Taylor et al. 2016). Fire activity declines sharply from 1832-1880 in the higher-use areas while remaining constant in the lower-use areas. The partial and localized disruption of fire activity is coincident with the rise of pastoralism traditions and livestock herd sizes (Weisiger 2004). After 1880, frequent surface activity is synchronously disrupted across the study area. This cross-scale human interaction is strongly associated with a dramatic increase in herd sizes (500,000 to 1,000,000 sheep and goats; Bailey and Bailey 1986) and a socio-economic transition with the 1868 establishment of the Navajo reservation and the arrival of trading posts that opened the Navajo territory to distant markets for sheep and wool products (Kelley 1986).

The Navajo fire history is unique for the region, and demonstrates the potential for fine-scale human interactions with forest disturbance regimes. It is the only fire

history in the SW that represents entirely Native American land uses, thus showing that the decline of Native American populations in other areas had a very small impact on the collapse of fire activity compared to the influx of Euro-American livestock grazing that severely disrupted fine fuel continuity and cover. Finally, our tree-ring based fire histories demonstrate that for hundreds of years widespread fires burned annually somewhere on the Navajo landscape, a very different scenario than at present. These frequent fires shaped forest structure and composition, and provided resilience to extreme climate events and disturbances. Our reconstructions include vital data on the frequency, timing, seasonality, scale, and climate drivers of the fire regimes. These data place recent high-severity fires on the Navajo Nation in a long-term context, and can directly feed into the restoration of natural processes in the fire-prone and heavily used forests of the Navajo Nation.

### **Gambel oak shrubfields and high-severity fires**

High-severity fire has emerged as major driver of forest change across semiarid forests of the Southwest. It is a direct consequence of human land-use change leading to long-term fire exclusion in many frequent-fire forests (Covington and Moore 1994; Allen et al. 2002) and is exacerbated by anthropogenic warming driving down fuel moisture (Abatzoglou and Williams 2016). The lack of frequent fire for a century or longer has increased stand densities and built up fuel loads, including live understory trees that act as ladder fuels and increase the probability of crown fires (Covington and Moore 1994; O'Connor et al. 2014; Stephens et al. 2015). Abrupt post-fire transitions of ponderosa pine-dominated forests to resprouting shrubs are now documented throughout the region

(e.g., Savage and Mast 2005). These new configurations may persist for decades or longer, with down-stream consequences on wildlife habitat, water yield, and carbon sequestration (Stephens et al. 2013).

In Appendix B, published as Guiterman et al. (2015), we tested the accuracy of dendroecological methods for reconstructing high-severity type-changing fires in pine-oak forests. Borrowing from the multiple lines of tree-ring evidence approach of Margolis et al. (2007), we collected (1) Gambel oak regeneration dates, (2) fire scars, (3) death dates, and (4) tree-ring growth changes within and around a 38 ha Gambel oak shrubfield originating from a documented high-severity fire in 1993. The methods proved successful, demonstrating the potential for Gambel oak crossdating and use to reconstruct patch sizes of past high-severity fires. Our findings also underscored the importance of corroborating multiple lines of evidence to confidently reconstruct past high-severity fire events.

In Appendix C, we applied the Guiterman et al. (2015) methods to five of the largest shrubfields in the Jemez Mountains, New Mexico. Initially we sought to reconstruct past high-severity fire events, but did not find evidence consistent with the tree-ring fingerprint of stand replacing fires. Instead, three important patterns emerged. First, through mapping shrubfields in the study area, we found that Gambel oak shrubfields currently occupy the same physiographic space as ponderosa pine and dry mixed conifer forests. This shows that post-fire conifer-to-shrub transitions can occur anywhere Gambel oak is currently established because of its ability to rapidly resprout and occupy a site following fire. Second, all five shrubfields were dominated by Gambel oak cohorts established in 1899, at the time of the last widespread fire event. This

demonstrates the longevity of the stand structure, and its persistence through climate extremes of the 20<sup>th</sup> and early 21<sup>st</sup> centuries. This long-term persistence was further demonstrated by the lack of conifer encroachment in shrubfields since at least 1935, based on comparisons of aerial imagery. Finally, all sites had a long history of frequent surface fire activity. Historically, fires burned through shrubfields at the same general rates and under similar climate conditions as ponderosa pine dominated forests in the study area, which are the most fire-prone forest type in the region.

These results have strong implications for the longevity of recent post-fire shrubfields, especially where patch sizes are relatively large (> 100 ha). Shrubfields represent an alternative metastable state to conifer forests (Beisner et al. 2003; Falk 2013), and can persist in the presence or absence of fires. The stability of shrubs in high-severity patches will hamper conifer recruitment through substantial competition (Shainsky and Radosevich 1986). A warmer and drier climate is likely to favor oak over many conifers, including ponderosa pine, as it is more tolerant of heat stress and drought (Adams and Kolb 2005; Choat et al. 2012). Furthermore, should trends in warming and associated high-severity fire activity continue (Williams et al. 2013; Abatzoglou and Williams 2016), much more pine-oak forest in the region is likely to be converted to oak dominance.

### **Climate-forest growth across Navajo forests**

Projections of forest drought stress in the Southwest point to a future of extreme moisture limitation and probable widespread conifer mortality (Williams et al. 2013). Although worrisome, there may be landscape positions and forest configurations that promote a greater degree of resistance to drought (Allen et al. 2015). High tree mortality

is likely to concentrate on the margins of species distributions (Rehfeldt et al. 2006, 2014), particularly at warmer low-elevation sites (Allen and Breshears 1998; Kelly and Goulden 2008). Species abundances may increase at the middle and upper-elevation sites within the species realized niche, and the niche could expand at the upper-elevation margins of current distributions (Breshears et al. 2008). Regional projections fail to capture these transitions at landscape and finer scales. Furthermore, there is a concern that many studies showing forest growth decline with warming are biased toward warm, dry sites where the vast majority of tree-ring collections have been made (Fritts 1976; Speer 2010). This potential for physiographic sampling bias would exaggerate the degree to which much of the forest area in the SW responds to future droughts.

In Appendix D, we evaluated these concerns with a physiographically representative network of non-selectively sampled tree-ring chronologies of piñon, ponderosa pine, and Douglas-fir spanning 250,000 ha on the Navajo Nation. Many of these plots are in forests above 2600 m, which are rarely represented in studies of forest decline and mortality (e.g., McDowell et al. 2016). We tested how well the forest drought stress index (FDSI; Williams et al. 2013) represented forest growth at our plot locations and across 12 targeted drought-sensitive sites in the study area. FDSI is analogous to the climate-driven portion of tree growth at the regional scale.

We found that correlations of FDSI to targeted tree-ring chronologies were consistently higher than to non-selectively sampled sites, but that both datasets had reduced correlations to FDSI at higher elevations. FDSI did do a good job in explaining growth at mid- and lower-elevation plots, which has implications for the resistance of roughly half of the landscape to future droughts. At upper elevations (~22% of the

landscape), FDSI did not explain a high degree of growth variation because tree growth at these locations declined only slightly during long and severe droughts of the past, including the very dry but relatively cool 1950s drought (Weiss et al. 2009). The pattern may be changing, however. We found that since ~1997 temperature-driven atmospheric moisture demand has remained above the 20<sup>th</sup> century average across the study area, and tree growth at upper elevations responded to an unprecedented degree to interannual climate. It appears that warming has reduced moisture levels, driving up the sensitivity of these forests to precipitation. Thus, increased moisture stress has promoted synchronous tree growth at the landscape and regional scales and moisture limitations have become increasingly important. Other studies have demonstrated the synchronizing effect of increased temperatures on disturbance regimes and insect outbreaks, and this recent synchronicity in tree growth at upper-elevation, mesic sites may portend rapid, widespread changes in the near future.

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APPENDIX A

SETTLEMENT, PASTORALISM, AND ASSOCIATED CHANGES TO FREQUENT  
SURFACE FIRE ACTIVITY ON THE NAVAJO NATION

Settlement, pastoralism, and associated changes to frequent surface fire activity on the Navajo Nation

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## ABSTRACT

In semiarid frequent-fire ecosystems across the globe, humans have been shown to increase fire frequency and reduce fire extent, and conversely to interrupt surface fuel production and/or continuity leading to the disruption of fire activity. We tested these patterns across a 150,000 ha landscape on the Navajo Nation in the northern Southwest USA, where relatively good data exist for Native American settlement and land-use patterns. We made *a priori* designations of relative use intensity and the timing of settlement (“higher use” versus “lower use” areas) at eight tree-ring fire history sites based on historical and ethnographic data. We found that the different areas had nearly identical fire regimes from 1597 to 1700, but higher use areas diverged with nearly two-times as many fire events until 1832. Added fires were almost entirely small, but overall fire extent during the settlement era (1700-1832) was not limited, probably due to relatively low population density, low numbers of Navajo livestock, and seasonal use of the study sites that left fuel continuity relatively intact. The period 1832-1880 marked the growth of Navajo pastoralism and was defined by a marked reduction in fire activity at the higher use areas. This period coincides with the peak of regional raiding for sheep and goat herds. Finally, nearly synchronous and cross-scale collapse of frequent surface fire activity occurred by 1880. This widespread disruption was associated with a nearly two-fold increase in livestock numbers ( $5 \times 10^5$  to  $10 \times 10^5$ ) over the 10 years following the establishment of the Navajo reservation. Livestock herds were encouraged to grow by federal agencies and the opening of distant markets with the arrival of the railroad. Results of this study demonstrate a diversity of human-fire interactions across spatial scales and through time in the Southwest. They underscore the importance of changing land-use as a major contributor to the demise of natural disturbance regimes, and show that the collapse of frequent fire activity across the western USA is much more closely associated with the introduction of livestock than the decline and removal of Native American populations.

## INTRODUCTION

Fire is a particularly sensitive earth system process to anthropogenic impacts (Bowman et al. 2009), but the effects of humans on fire regimes vary depending on local climate and the productivity and flammability of vegetation, as well as on human population density, land-use intensity, and cultural perspectives of fire (Vale 2002; Marlon et al. 2008, 2012; McWethy et al. 2013; Swetnam et al. 2016). These variables produce a range of human influences on fire regimes both globally and regionally that has challenged efforts to develop a generalized model of human-fire interactions (Bowman et al. 2011; Coughlan and Petty 2013; Roos et al. 2014, 2016; Scott et al. 2016). The importance of this effort, however, is underscored as anthropogenic climate change (Abatzoglou and Williams 2016) and past land use (Taylor et al. 2016) drive increases in area burned and fire severity (Miller et al. 2009; Westerling 2016).

In semiarid frequent fire ecosystems, fuel is generally limiting, and thus climate exerts a primary control on fire regimes (Bowman et al. 2009; McWethy et al. 2013). Climate variability drives fire frequency, spread, and regional to intercontinental synchrony (Swetnam 1993; Kitzberger et al. 2001, 2007; Falk et al. 2011). Humans can alter this pattern and over-ride fire-climate relationships in two ways. First, human ignitions tend to increase fire frequency and reduce fire extent. Fire is a self-limiting process, such that with higher fire frequencies there is a finer and more complex matrix of burn patches that reduces the spread of individual fires. This human-fire interaction is a global phenomenon that has been observed in Africa (Archibald et al. 2012), Argentina (Veblen and Kitzberger 1999), Australia (Bliege Bird et al. 2008, 2012), northern Mexico

(Heyerdahl and Alvarado 2003; Stephens et al. 2003), northern Sweden (Granström and Niklasson 2008), the Sierra Nevada of California (Taylor et al. 2016), and the southern and southwestern US (Guyette et al. 2002; Stambaugh et al. 2013; Swetnam et al. 2016). Second, high human population densities (Liebmann et al. 2016) or intensive land use activities such as livestock grazing (Swetnam et al. 2001) can remove or fragment continuity of fine fuels and disrupt fire activity altogether.

In the semiarid forests of the southwestern US, historic fire regimes were primarily composed of high-frequency, low-severity surface fires that were highly synchronized with wet-dry oscillations in seasonal, interannual, and decadal precipitation (Swetnam and Betancourt 1990, 1998; Kitzberger et al. 2007; Margolis et al. *in review*) and decadal to centennial variability in temperature (Trouet et al. 2010). Between ~1870 and 1900, frequent fire regimes collapsed region-wide in response to multiple human influences including widespread introduction of Euro-American livestock herds numbering in the millions of animals, industrial-scale logging, the removal and decline of Native American populations, and active fire suppression (Pyne 1982; Swetnam and Baisan 1996, 2003; Swetnam et al. 2001; Allen 2007; Fulé et al. 2012). Livestock grazing, and its associated effect on fine fuel coverage and continuity (Allen 2007), is particularly important in the abrupt disruption of fire activity, as evidenced by numerous cases of pre-Euro-American fire regime disruptions in the Southwest associated with introductions of Spanish and/or Native American livestock herds (Savage and Swetnam 1990; Touchan et al. 1995; Swetnam and Baisan 1996; Margolis and Balmat 2009).

Studies of human-fire interactions in the Southwest have generally concluded that Native American influences on fire regimes were localized in space and time (Swetnam

and Baisan 1996). Some have found brief periods with anomalously short fire intervals at particular sites (Seklecki et al. 1996; Kaye and Swetnam 1999; Meunier et al. 2014), unusually long fire-free periods (Heyerdahl and Alvarado 2003; Stephens et al. 2003), or changes in fire seasonality patterns (Grissino-Mayer et al. 2004) that might have been human-caused. Although Native Americans used landscape fire for hunting, agricultural land clearing, food production, and in conflict (Williams 2002; Swetnam and Baisan 2003; Fulé et al. 2011; Sullivan and Forste 2014), these influences can be difficult to discern at certain scales because much of the region experiences adequate lightning strikes to support historic fire frequencies (e.g., Allen 2002). Recently, however, Swetnam et al. (2016) utilized a dense network of fire history sites across a relatively large landscape that included high populations of Native Americans living in year-round villages (Liebmann et al. 2016) to show that Native American fire-use followed the global pattern of highly frequent fires of low extent. This pattern persisted for centuries until the removal and decline of village populations.

A common challenge to discerning human-fire interactions is the rarity of historical documentation and proxy records pertaining to fire, climate, and humans for the same time periods and locations that are necessary to discern human impacts (Bowman et al. 2011). Human history is often the least well documented and controlled. For example, we may know that a certain group of people lived or traveled within an area during a certain broad time period, but we rarely know how many there were, what seasons or years they were actually present in specific places, or much about what they were doing to fuels and fire ignitions.

In the case of the Navajo Nation of northeastern Arizona and northwestern New Mexico, documentation of human history is relatively good, at least since the mid-19th century, and some archaeological data cover the 17th and 18th centuries in some areas (Towner 1996). The region has a dense network of drought-sensitive tree-ring chronologies from which precise, high-resolution climate reconstructions relevant to the Navajo Nation are available for study (e.g., Cook and Krusic 2004). One fire history study using tree rings and fire-scarred trees has been completed on the Navajo Nation (Savage and Swetnam 1990), but the spatial extent of these collections was limited to a fairly small part of the Chuska Mountains in northern Arizona.

Our study area covers ~150,000 ha of the semiarid ponderosa pine (*Pinus ponderosa*)-dominated forests on the Navajo Nation. This landscape is unique in the Southwest for three reasons. First, because of its remoteness, it was affected primarily by one group of people (the Navajo) during the last ~300 years, and it was all but uninhabited for ~400 years prior to Navajo settlement. Many other places in the Southwest had multiple overlapping influences, including Pueblo, Apache, Navajo, Hispanic, and/or Euro-American that can confound inferences of historic human presence and specific land-use practices. In addition, unlike most landscapes in the region, the Navajo have remained in their homeland to present, except for a four-year period (1864-1868) in which they were forcibly removed to a distant reservation before being granted return. Second, we have relatively detailed historical and archaeological data on settlement areas, the adoption of sheep pastoralism, and wartime/peacetime periods. Hence, on this landscape, human, fire and climate history can be compared and studied in some detail to assess the nature, variability, and associations among these variables.

Third, the reservoir of forest and fire history legacy information over a large landscape is unusually rich. Discoveries of exceptionally old and well preserved tree-ring material, including fire-scarred trees presented here, make the Navajo forests even more valuable for understanding the complexity of human-induced changes because tree-ring data provide the temporal depth and precision required to assess the transition from natural to human-dominated time periods.

The objectives of this study are to reconstruct the spatiotemporal patterns of historic fire regimes across the study area and to evaluate the relative influences of climate and human activity on the fire regimes. We divide fire history sites into two *a priori* categories relating to human settlement patterns and associated land-use intensity to test for spatial differences in fire frequency and extent, and then compare the fire chronologies to known periods of Navajo history. These historical periods include early Navajo settlement (~1690 - mid-1700s); increased immigration, intensified land-use, and war times (mid-1700s - 1863); exile and the “Long Walk” (1864-1868); and the reservation era (1869-present). We hypothesize that (1) relatively slow immigration over ~100 years and widely dispersed settlement patterns (following Navajo custom) had a relatively small effect on the overall fire regimes, most likely with a transition from widespread fires every 5-10 years (“free-range fire”, *sensu* Swetnam et al. 2016) to highly frequent fires of relatively low extent; and (2) that the rise of pastoralism initiated a reduction in fire activity as found by Savage and Swetnam (1990) but that the aforementioned land-use patterns concentrated this influence to only higher use areas, such as the Chuska Mountains. By the early 1900s, intense grazing pressure was well documented across the Navajo reservation, and so we expect that fire regimes collapsed

across the study area by the end of 19<sup>th</sup> century, well before fire suppression efforts began in the 1930s (Alexious Becenti, Navajo Forest Manager, personal communication).

## STUDY AREA

### **Setting**

The Chuska Mountains and Defiance Plateau are located on the Navajo Nation, along the northern Arizona-New Mexico border (Figure 1). This area encompasses ~250,000 ha and includes nearly all of the ponderosa pine and dry mixed conifer forests within the Navajo Nation. The Chuskas are a narrow (800-1000 m wide) northwest trending Oligocene sandstone crest (Wright 1956; Cather et al. 2008) intruded by younger volcanics (Appledorn and Wright 1957). The crest of the range contains numerous shallow, perennial lakes, although many were dry during field work for this study due to an ongoing drought. The eastern side of the Chuska crest is bounded by a long escarpment, while the western slopes are more gentle but incised by narrow canyons. The Defiance Plateau is lower in elevation, broader, and with less variable terrain, except in the southern portion where there are several canyons. Canyon de Chelly, a 300 m vertical incision into the sandstone that drains much of the western Chuskas, bounds the northern end of the Defiance Plateau.

Forests of the Chuska crest are predominantly ponderosa pine with Gambel oak (*Quercus gambelii*), which is most abundant along meadow edges and in wetter areas. Dry mixed conifer forests (as defined by Romme et al. 2009) composed of Douglas-fir

(*Pseudotsuga menziesii*), aspen (*Populus tremuloides*), white fir (*Abies concolor*), and ponderosa pine are found at the highest elevations, on mesic sites, and along the steep eastern escarpment. Wet mixed conifer forests containing spruce (*Picea engelmannii* and *P. pungens*), fir (*A. concolor* and *A. lasiocarpa*), and aspen are present only on higher elevation, north-facing slopes and in cold-air drainages. The Defiance Plateau forest is nearly pure ponderosa pine with occasional Gambel oak clumps and single-stemmed trees in the understory. Douglas fir is restricted to cold-air drainages where it is mixed with piñon and ponderosa pine, and northern slopes of occasional bluffs. Both areas are bounded by piñon-juniper (*Pinus edulis-Juniperus* spp.) woodlands at the lower elevations, which usually grade into high desert grasslands below ~2,000 m above sea level. Extensive montane grasslands and meadow areas occur in association with perennial lakes on the Chuska crest (Wright 1964).

The climate of the study area is continental and semiarid. The precipitation regime is dominated by summer convective storms (rain) associated with the North American Monsoon, with highly variable winter precipitation (snow) (Sheppard et al. 2002). Mean annual precipitation is 332 mm (1900-2010; based on the Precipitation-Elevation Regression on Independent Slopes Model (PRISM) [Daly et al. 2008] averaged for the study area). Average minimum temperatures range from -8.6<sup>0</sup> C in January to 10.2<sup>0</sup> C in August, and average maximum temperatures range from 4.9<sup>0</sup> C in January to 27.0<sup>0</sup> C in July (1900-2010; PRISM 2016).

## **Human history**

Large prehistoric human settlements in the study area are concentrated along the eastern and southern foothills of the Chuska Mountains and in Canyon de Chelly. Communities and “great house” structures in the Chuska foothills are associated with the 850-1130 Common Era (CE) Ancestral Puebloan community at Chaco Canyon (Powers et al. 1983; Van Dyke 1999). The Chacoan culture was largely reliant on the Chuska landscape for resource and food procurement, which included stone tools (Cameron 2001), pottery (Mills et al. 2011), construction timbers (English et al. 2001; Guiterman et al. 2016), maize (Benson et al. 2003; Grimstead and Benson 2015) and wild and domesticated animals (Grimstead et al. 2014, 2016). Canyon de Chelly was occupied during the 12<sup>th</sup> and 13<sup>th</sup> centuries CE by Ancestral Puebloans living in cliff dwellings and farming the canyon floor (McDonald 1976). Several Puebloan structures and small communities are present on the Defiance Plateau, mainly in association with the occupation of Canyon de Chelly. These settlements were depopulated along with much of the Colorado Plateau during the late 1200s drought (Dean et al. 1985). In the four centuries that followed until Navajo immigration probably began in the late 1600s, there is no evidence for large scale occupation or use of the area (McDonald 1976).

At the time of Spanish arrival in the 1500s, the Navajo were primarily farming in the San Juan River Valley (the *Dinétaah*) (Weisiger 2004). For multiple reasons, including increasing populations, reduced resources, and tensions with the Spaniards and neighboring Native American groups, the Navajo began to migrate west and south in the late 1600s, with depopulation of the *Dinétaah* complete by the end of the 18<sup>th</sup> century (Towner 2008). Settlement in the Chuska-Defiance area began by 1700, but larger groups

did not arrive until the mid-1700s (Hurt 1942; Robinson and Towner 1993). These first settlers brought sheep to the area for the first time, although initial livestock numbers were likely small and localized (Bailey 1980; Kemrer and Lord 1984). By the early 1800s, the Navajo population was roughly 10,000, and several population centers had grown, including in Canyon de Chelly, the Chuska Valley (east of the crest), Ganado, and Fort Defiance, Arizona (Bailey 1980; Andrews 1991; Towner 2008). In 1863, over 12,000 people were forcibly removed to a small reservation at Bosque Redondo (Fort Sumner), and upon their release in 1868, roughly 8,000 Navajo survivors returned to the Chuska-Defiance area (Hoover 1937).

Sheep, goats, horses, and some cattle were acquired initially through raids on the Spanish and neighboring tribes, abandoned or captured Spanish herds during the Pueblo Revolt (1680-1696), and assimilation with other tribes (Kelley 1986; Weisiger 2004). It is believed that prior to ~1700 Navajo did not raise many livestock, and instead consumed them. But within several decades, pastoralism became an important and sacred way of life (Andrews 1991; Weisiger 2004). Through the mid-1800s, Navajo herds grew rapidly as families bred their sheep to be passed to younger generations. Raiding also peaked in the mid-19<sup>th</sup> century, providing thousands of additional animals, and heightening tensions with Spanish-Americans, neighboring tribes, and American settlers (McNitt 1972; Bailey and Bailey 1986). Navajo shepherds were usually children or the elderly, and sheep and goats were rarely taken farther than 8-10 km from homesites in daily circuits that included bringing animals to pasture, water, and then corals at night (Adams 1963). Seasonally, Navajos moved their herds from low-elevation homesites in the winter to pastures at higher elevations in the Chuska Mountains, where surface water and grazing

lands were plentiful (Jett 1978; Andrews 1991). This transhumance pattern was common among families, and took on a deep, spiritual meaning, with some migrations ranging up to 90 km to summer homesites, or “sheep camps” that are often located adjacent to lakes and meadows in the Chuskas (Weisiger 2004). In the late 1800s to the early 1900s, livestock numbers began to rise rapidly (Bailey and Bailey 1986), and large herds under individual ownership (i.e. ranchers) that were previously rare, became more common (Kelley 1986; Weisiger 2009). By the 1930s, the Federal Soil Conservation Service grew concerned over erosion caused by overgrazing in some areas, and instituted the “Stock Reduction” program, along with new grazing laws, which permanently changed Navajo economies and pastoralist practices (Weisiger 2009).

The 19<sup>th</sup> century was also a wartime period for the Navajo, which included many successful raids and the growth of herds and wealth, but probably also incurred heavy losses in turn, through battle and capture, especially of children who were targeted for the slave trade (Brugge 1968). Fighting was fierce on both sides, and Navajos continued to earn their reputation as fearsome raiders, with expeditions ranging far into northern and central New Mexico (McNitt 1972). Following the Mexican-American War (1846-1848), Navajo territories fell under US control. What was initially a peaceful relationship soon escalated as American soldiers committed atrocities against the Navajo and Navajos continued to raid Spanish-Americans living along the Rio Grande River and elsewhere (McNitt 1972). To address the threats and thwart further reprisals, the US military established several forts in Navajo territory, including Fort Defiance within our study area in 1851. These forts were a direct affront to Navajo sovereignty because, among many reasons, they outlawed grazing in their vicinity. Intense fighting between Navajo

and the US Military ensued over the next decade, culminating in the deployment of Kit Carson to Fort Defiance in 1863 to subdue the Navajo and remove them to Bosque Redondo. Agricultural areas were destroyed, along with livestock herds. In 1868, the US government signed a treaty with the Navajo headmen, and established the Navajo reservation over the Chuska-Defiance area. As the survivors returned during 1868 and 1869, they were given two sheep per person, and the US government encouraged them to rebuild herds (Hoover 1937). By the late-1870s, Navajo herds had regained their previous numbers (Kelley 1986).

## METHODS

### **Site selection**

We reconstructed fire history at seven sites spanning roughly 60% (150,000 ha) of the combined Chuska-Defiance landscape (Figure 1). Sites were *a priori* divided between areas that included some of the earliest and most well-documented settlement and use (hereafter “higher use areas”) to contrast areas that were less likely to be used in the earlier periods of settlement (“lower use areas”) (Table 1). Higher use areas include the Chuska crest that was the summer destination for shepherding families from both the western and eastern reaches of the study area (Jett 1978; Andrews 1991). We also include areas near Canyon de Chelly and the Ganado/Wide Ruins area west of the Defiance Plateau, which probably combined to support roughly half of the Navajo population prior to 1863 (Andrews 1991).

Lower use areas were more difficult to designate *a priori*. We based this designation on areas that lack nearby abundant perennial surface water sources prior to

the development of wells and other ground-water resources in the late-1800s and early-1900s. These surface water sources were imperative for longer seasonal stays or permanent settlement. Navajo home sites were traditionally located in proximity to water and grazing resources in order to tend to livestock needs, and were often multiple kilometers from agricultural fields (Bailey and Bailey 1986). Therefore, in areas where we hypothesized traditional Navajo herding practices, that include children and elderly shepherds bringing sheep to corral each night and maintaining proximity to water (Jett 1978), were unfit for the location we had some confidence in designating lower use. Never do we assume that an area remained un-grazed or un-used, but we recognize a likely spectrum in land-use intensity depending on the available of necessary resources.

We assessed two other factors in support of our designations. First, once married Navajo couples traditionally moved to the land-use-area of the wife's mother (Weisiger 2009), and we therefore assumed that areas with large clusters of homesites were settled and used for longer than areas without many home sites. We observed very few home sites located along the Defiance Plateau and in some areas of the lower-elevation, southern Chuska Mountains relative to high densities of modern and historic home sites on the Chuska crest. Second, we analyzed the fire histories of each site to confirm the presence of unusual patterns in fire frequency (increases or decreases) that might relate to human activity, as reviewed above.

In addition to the seven sites sampled for this study, we include the fire history of Savage and Swetnam (1990). Their sampling covers a large area on the Chuska crest that encompasses one of our sites, DKL. We obtained the Savage and Swetnam fire history

data from the International Multiproxy Paleofire Database<sup>2</sup> and note that it includes 13 trees as opposed to the 16 trees reported by Savage and Swetnam (1990).

### **Fire history reconstruction**

At each site we used standard tree-ring fire history methods (Dieterich and Swetnam 1984) to collect specimens primarily from dead trees (stumps, logs, and snags), and occasionally live trees to extend the fire histories to present. With a chainsaw, we removed sections from “cat faces” with  $\geq 3$  fire scars to obtain a complete inventory of all widespread fire events (Farris et al. 2013). On the few live trees we sampled, we cut partial sections, limiting removal to  $\sim 10\%$  of the tree’s basal area in order to minimize injury and maintain tree stability (Heyerdahl and McKay 2008). We targeted old remnant and living trees, as identified by the number of rings, spiral grain and weathering, in order to extend the reconstruction back in time as far as possible, with the target to include time periods prior to Navajo occupation.

In the laboratory, we applied standard dendrochronological methods (Speer 2010) that included sanding fire-scarred specimens to a fine polish in order to accurately see cellular structure for crossdating and identification of fire years and seasonality (Baisan and Swetnam 1990). Seasonality was assessed by the intra-ring position of the fire scars, defined as within the early-earlywood (EE), middle-earlywood (ME), late-earlywood (LE), latewood (L), along the ring boundary (i.e. dormant; D), or unknown (U). Dormant

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<sup>2</sup> Available at <https://www.ncdc.noaa.gov/data-access/paleoclimatology-data/datasets/fire-history>

scars were assigned to the following growth year, based on regional analyses of fire season (e.g., Swetnam and Betancourt 1998).

## **Analyses**

Fire history data were compiled and analyzed in R (R Core Team 2015) using the *burnr* fire history package (version 0.1.1, Malevich et al. 2016). Fire interval statistics were calculated for each site and all sites pooled, based on the time period between the first and last fire scars recorded on  $\geq 10\%$  of trees with a minimum of 2 trees scarred and 2 trees recording. Trees were identified as “recording” following the scar that appeared to initiate the open cat face (Swetnam and Baisan 1996). Fire intervals were assessed at three levels of relative spatial extent: years recorded by any tree, or years found on either 10% or 25% of samples. The percentage filters omit some small fires that scar only one or two trees, allowing us to distinguish more widespread fires that burn a higher proportion of study sites (Swetnam and Baisan 1996). For each filter, a minimum of 2 trees had to be scarred and 2 trees recording. Farris et al. (2010, 2013) demonstrate the efficiency of targeted fire scar sampling for reconstructing temporal and spatial patterns of fire history in Southwestern ponderosa pine forests, and the strong correlation of percentage-filtered fire interval metrics with independent fire atlases (maps) of total areas burned.

We quantified the relationship between fire and interannual climate variability via superposed epoch analysis (SEA; Swetnam 1993) using the *run\_sea* function in *burnr*. We performed SEA on years with widespread (25%) fires, splitting the dataset into years

with only one site recording fire and “synchronous” years in which more than one site recorded fire (Heyerdahl et al. 2008; O’Connor et al. 2014). These fire years were compared to the average of four of the closest  $2.5^0 \times 2.5^0$  gridpoints (numbers 103, 104, 118, and 119) in the North American Drought Atlas (Cook and Krusic 2004). The Drought Atlas provides tree-ring reconstructed summer Palmer Drought Severity Index (PDSI). Although the PDSI reconstructions in this atlas are nominally summer season, it is notable that in the southwestern U.S. the PDSI time series correlates most strongly with cool-season moisture (St. George et al. 2010).

We assessed spatial differences and temporal changes in the fire history reconstructions in a variety of ways. First, we used visual analyses of fire-scar chronologies to look for potential human influences at the scale of individual sites (Table 1) and by assessing variability in fire intervals among sites. Second, we compiled sites in the higher and lower use groups by calculating fire indices, which represent both the frequency and relative extent of past fires (Taylor et al. 2008, 2016). Fire index is calculated for each site as the number of trees scarred divided by the number of trees recording fire. We limited site-level fire indices to the period after which three trees were recording fire. Group-level fire index is calculated as the sum of site fire indices divided by the number of sites recording fire, and was limited to at least two sites recording fire. The period of analysis (1597-2015) was determined by the shorter record between the two groups. Finally, we calculated cumulative fire event time series (Brown and Sieg 1999; Brown et al. 1999; Meunier et al. 2014) for the fire index chronologies and assessed changes in slope relative to Navajo settlement and land-use history of the study area.

## RESULTS

Reconstructed fire histories for ponderosa pine-dominated sites in the Chuska Mountains and Defiance Plateau of the Navajo Nation include eight sites, 90 trees, and 827 crossdated fire scars. Tree-ring chronologies extend over much of the last millennium, with the earliest tree dating to 1072 CE (Table 1). Five of our sites have at least one tree dating prior to 1500. The earliest fire scar is 1323 and the most recent is 2011.

Frequent fire regimes were recorded at every site, regardless of use designation (Figures 2 and 3). Across all sites, mean fire intervals range from 4.6 to 46.8 years considering all fires, and from 10.1 to 78.0 years when filtered for fires that scarred  $\geq$  25% of trees at each site (Table 2). Weibull Median Fire Intervals for widespread fires ranged from 9.24 to 70.09 years. Much of the variability in MFI is related to sampling area (Table 2). Pooling all sites across our 150,000 ha study area shows that a fire burned somewhere on the landscape every 2.2 years for more than four centuries (1469-1871).

The fire-climate relationship shows the importance of interannual climate variability in promoting fuel conditions for synchronous, widespread fires (Figure 4). Fires that burned multiple sites (i.e. synchronous fires) occur during significantly dry years ( $p < 0.01$ ) that are preceded by three significantly wet years ( $p < 0.05$ ). Precipitation in the year prior to fire are also important for widespread fires recorded at a single site ( $p < 0.01$ ), but drought in the year of fire is not significant. Indeed, four single-

site fires (1784, 1839, 1868, 1869) were recorded in extremely wet years (PDSI > 98<sup>th</sup> percentile).

Differences between higher and lower use areas are discernable in the fire index calculations (Figure 5). From 1597 to 1700, each area shows similar fire frequencies and relative extents. No fires were recorded at higher use areas between 1681 and 1699, coincident with the Pueblo Revolt era (1680-1696) (Roberts 2007), while at lower use sites fires were recorded in 1684 (widespread at two sites) and 1691. From 1700 until 1832, the number of fires recorded at higher use areas diverges from lower use areas with the addition of many small fires. These do not appear to limit fire extent, as the relative numbers of higher fire index values is similar between areas. Added small fires, however, accumulate to nearly two times as many fire events as recorded at the lower use areas (62 versus 33). From 1832 until 1876, fire activity at higher use areas declines markedly with the last fire of greater extent recorded in 1871, and very few small fires recorded. The fire regimes at lower use areas remain relatively constant from 1597 to 1880, with a slight increase in the number of small fires ca. 1760. After 1880, frequent fire regimes all but end across the study area (Figure 1). Four fires were recorded between 1880 and 2010, which were all on one tree from the Savage and Swetnam (1990) collection (CHU; Figure 2).

## DISCUSSION

Across semiarid ponderosa pine-dominated forests on the Navajo Nation, reconstructed fire regimes show a pattern of frequent surface fire activity from the late

16<sup>th</sup> century to the late 19<sup>th</sup> century. Climate played a critical role in establishing conditions for widespread, synchronous fires at the landscape scale, but was of less importance in determining the timing of fire at the scale of forest stands. This is in accord with regional analyses (Falk et al. 2011) that indicate the importance of climate variability in synchronizing fire activity at large spatial scales (landscapes and larger), while bottom-up factors (e.g., fuel continuity, people, landscape configuration) can override climate at smaller spatial scales. The cross-scale importance of antecedent moisture on fire activity (Figure 4) reveals a fuel limitation in these forests (Swetnam and Betancourt 1998). This indicates sensitivity of the system to human land uses that affect fuel continuity, including grazing, fuelwood harvesting, and travel (Swetnam et al. 2016). Indeed, we found clear evidence that Navajo settlement and land use played important roles in altering fire regimes at a range of scales – from individual sites (< 28 ha) to the study area (~150,000 ha) – by influencing fire frequencies in opposing directions.

During the period of Navajo immigration to the study area, ~1690 to the early-1800s (Towner 1996), we found a dramatic increase in fire activity (Figure 5). This change was isolated to areas that we *a priori* hypothesized would have higher human use. These added fires constitute a near doubling in fire frequency from 1700-1832, and help to refine the era of Navajo settlement. This pattern is consistent with the global observation that human fire use tends to increase fire frequency and reduce fire extent (Veblen and Kitzberger 1999; Guyette et al. 2002; Heyerdahl and Alvarado 2003; Stephens et al. 2003; Granström and Niklasson 2008; Bliege Bird et al. 2008, 2012; Archibald et al. 2012; Stambaugh et al. 2013; Swetnam et al. 2016; Taylor et al. 2016).

In this study, however, the addition of many small fires did not seem to limit the number of more extensive fires, which continued in both higher and lower use areas throughout the 1700-1832 period (Figure 5). This probably results from the Navajo custom of living in widely dispersed settlement clusters and migrating seasonally sometimes vast distances to take advantage of seasonal climate and abundances in natural resources. In the past, as today, very few if any people resided in the higher elevations (> 2700 m) of the Chuska Mountains outside of the summer season. By contrast, the nearby Jemez Mountains supported a pre-Columbian population of up to 8,000 people living in ~18 large, multi-storied village structures, all in an area of ~13,000 ha above 2100 m (Liebmann et al. 2016). This high population density living in a cold, arid environment for over a century fragmented the fuel layer in proximity to villages and completely disrupted fire activity. In more distant reaches of their landscape fires were common and of limited extent (Liebmann et al. 2016; Swetnam et al. 2016). Without such a fragmented fuel layer in Navajo forests, extensive fires were just as common between higher and lower use areas, and also between this period and the free range fire period (1597-1700).

As pastoralism grew among the Navajo in the early to mid-19<sup>th</sup> century (Weisiger 2004), there is an associated decline in fire activity that includes both small and more widespread fires (Figures 2 and 5). We found that the period 1832-1880 defines the growth of pastoralism, with livestock-related fire decline present only in areas of higher use. We determined the end-date of this era based on the end of fire at lower use sites, but in fact, the fire regime at the higher use sites ends by 1876, with the last widespread fire in 1871. This pattern of reduced fire activity (1832-1880) in association with grazing

differs from the regional pattern of complete fire regime disruption at about the time of livestock introductions to a particular landscape (e.g., Swetnam et al. 2001). The grazing and movements of livestock herds directly reduce the fine fuel cover, inhibit grass recovery because sheep can pull clumps from the ground (promoting erosion), and further fragment fuel continuity with numerous heavily grazed patches, with effects that can last decades or longer (Belsky and Blumenthal 1997; Allen 2007). Here again, we interpret the reduction in fire activity associated with pastoralism, as opposed to complete disruption, to be explained by the relatively low density of Navajo populations and their seasonal and relatively dispersed use of the study area. Livestock numbers during this period were between 300,000 and 400,000 sheep and goats (Savage 1991). The lack of a similar pattern at the lower use areas supports the conclusion that this was a relatively localized impact associated with the initial build-up of livestock herds.

The period from the 1830s until 1863 marks the peak of Navajo raids on neighboring tribes and settlements in New Mexico, which in the build-up of Navajo herds (Brugge 1968; McNitt 1972; Weisiger 2009). Raiding was carried out for this purpose but also in reprisal of previous attacks, stolen livestock, and the capture of Navajo children for the slave trade. These raids were a consistent occurrence during this era, and supported development of an economic system that was rapidly growing more reliant on sheep and goats (Bailey and Bailey 1986). Raiding was considered an affront on American lives and property, and in response the US Military pursued Navajo submission in the 1850s with the construction of several forts in and around Navajo territory. The Navajo wars against the US Military ended with the 1863 campaign led by Kit Carson to remove the tribe to Bosque Redondo internment. Coincident with these military

campaigns, we found a slight increase in the number of fires at certain sites, including those in the higher use areas. We interpret these fires as relating to a reduction in livestock numbers and/or a greater amount of herd mobility in avoidance of the US Military campaigns, which would have aided in the recovery of grasses and re-established the continuity of fine fuels.

The most profound anthropogenic alteration of fire regimes that we found was the nearly synchronous collapse of frequent fire ca. 1880 (Figures 1 and 5). This disruption crossed boundaries between higher and lower use areas, affecting fire regimes throughout the study area, and likely covered all Navajo forests. Synchronous decline of fire activity at this scale signals a dramatic change in the distribution and intensity of land-use from previous centuries, which relates to the establishment of the Navajo reservation and the rise of pastoralism as a common and widespread practice among the Navajo people (Savage and Swetnam 1990; Weisiger 2004, 2009).

Upon their return from Bosque Redondo in 1868, Navajos were given livestock and encouraged to rebuild their lost herds that had numbered approximately 500,000 in 1860 (Bailey and Bailey 1986). About 8,000 Navajos returned from Bosque Redondo, with another 2,000 still living with other tribes or “at large” (Bailey and Bailey 1986). The estimate for the number of livestock at this time is roughly 4,000 animals, but the US government provided 14,000 sheep and 1,000 goats in 1869 as part of the 1868 treaty, and subsequently kept providing animals (including horses and cattle) throughout the next decade (Bailey and Bailey 1986). For many Navajo during this period of recovery, herds were too small and important for simple consumption, and years of drought in the early 1870s slowed the growth of herds as well as crops, further hampering the economic

recovery of the tribe (Bailey and Bailey 1986). By 1880, however, livestock herds had grown substantially through US government provisioning and Navajo breeding efforts to number approximately 1,000,000 sheep and goats. At this point, Navajos had regained their subsistence strategies and all but ended their dependence on federal rations (Adams 1963; Kelley 1986).

At the same time, new markets were opening up that sought to capitalize on Navajo wool products. The number of trading posts on the reservation increased from one in 1870 to nearly a dozen by the mid-1880s (Bailey and Bailey 1986). The Transcontinental Railroad arrived south of the reservation in 1881, and the town of Gallup east of the Chuska Mountains was established as a trading depot. This further intensified trade and fed a high demand for wool in eastern markets, including for prized Navajo rugs (Underhill 1956; Kelley 1986). With over a million sheep and goats on the landscape, their impacts were now spread across the reservation, causing widespread decline in the abundances and continuity of fine fuels, and ultimately leading to the disruption of frequent fires, just as in many other areas of the western US.

It is worth noting that this overall pattern of fire regime disruption is consistent with the timing and probable causes of fire exclusion across frequent-fire forests of the western US, but differs in one, important aspect. The Navajo forests represent the only persistent fire regime disruption associated with Native Americans. Some have attributed the removal and decline of Native Americans as a proximal cause of disruption in fire activity. Our results, in accord with Savage and Swetnam (1990), show that land-use strategies are more important than who conducts them. Fire regime interruptions across

many landscapes of the western US, therefore, relate much more closely to the influx of livestock than the reduction of Native American populations and their burning practices.

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## TABLES

**Table 1.** Fire history sites

Site Name (abbr.)	Landscape/Use category	Mean elevation (m)	Sample area (ha)	Number of trees	Tree-ring dates		Fire dates	
					Earliest	Latest	Earliest	Latest
Chuska Mtns (CHU)	Chuska/higher	2739	1571.3	13	1393	1986	1471	1930
Duck Lake (DKL)	Chuska/higher	2747	0.5	7	1336	1935	1476	1893
Monument Canyon Lower (MCL)	Defiance/higher	2212	0.1	9	1567	2015	1598	1832
Monument Canyon Upper (MCU)	Defiance/higher	2209	1.0	5	1446	1963	1595	1871
Pine Canyon South (PCS)	Defiance/higher	2225	28.0	14	1072	1962	1323	1876
Piney Hill (PNH)	Defiance/ lower	2434	2.7	3	1567	2015	1573	1880
Squirrel Springs North (SQN)	Chuska/ lower	2455	10.0	21	1384	2016	1462	2011
Scattered Willow Wash (SWW)	Defiance/lower	2389	22.0	21	1134	2016	1333	1879

**Table 2.** Fire return intervals (years).

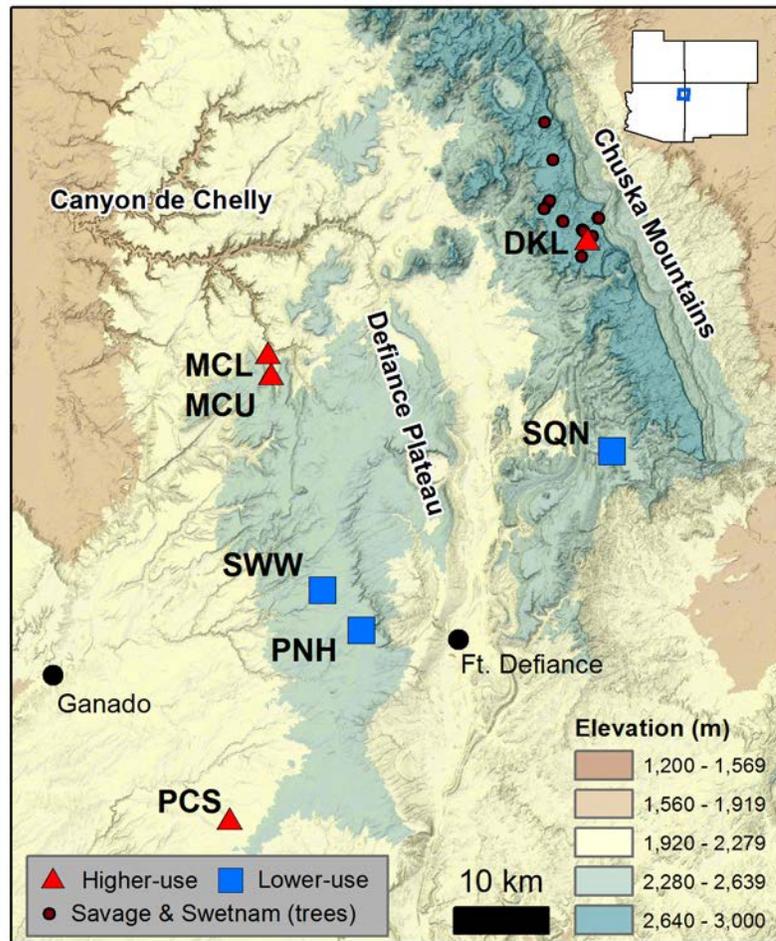
Site / analysis period <sup>1</sup>	Category of analysis <sup>2</sup>	Number of intervals	Mean fire interval (MFI)	Minimum	Maximum	Weibull median fire interval <sup>3</sup>
CHU 1616-1870	All fires	54	4.7	1	23	NA
	10%	27	9.4	1	48	6.30
	25%	14	12.9	2	48	10.35
DKL 1548-1870	All fires	33	9.8	1	30	8.14
	10%	12	26.8	3	187	14.09
	25%	12	26.8	3	187	14.09
MCL 1598-1832	All fires	5	46.8	27	70	46.91
	10%	3	78.0	27	148	70.09
	25%	3	78.0	27	148	70.09
MCU 1595-1871	All fires	27	10.2	3	41	9.10
	10%	18	15.3	4	89	12.08
	25%	18	15.3	4	89	12.08
PCS 1532-1876	All fires	33	10.4	2	43	8.25
	10%	21	15.7	2	58	12.55
	25%	17	19.4	5	58	16.17
PNH 1704-1880	All fires	20	8.8	3	20	8.5
	10%	5	35.2	10	70	32.81
	25%	5	35.2	10	70	32.81
SQN 1562-1886	All fires	71	4.6	1	17	NA
	10%	51	6.8	2	35	5.49
	25%	32	10.1	3	35	9.24
SWW 1469-1868	All fires	70	5.7	1	29	5.01
	10%	35	11.4	1	63	8.76
	25%	26	15.3	3	63	12.75
All sites 1469-1871	All fires	198	2.0	1	14	NA
	10%	63	5.0	1	49	NA
	25%	15	23.3	2	74	18.71

<sup>1</sup> Period of analysis is the time between the first and last fires recorded by a minimum of 2 trees scarred and  $\geq 10\%$  of recorder trees

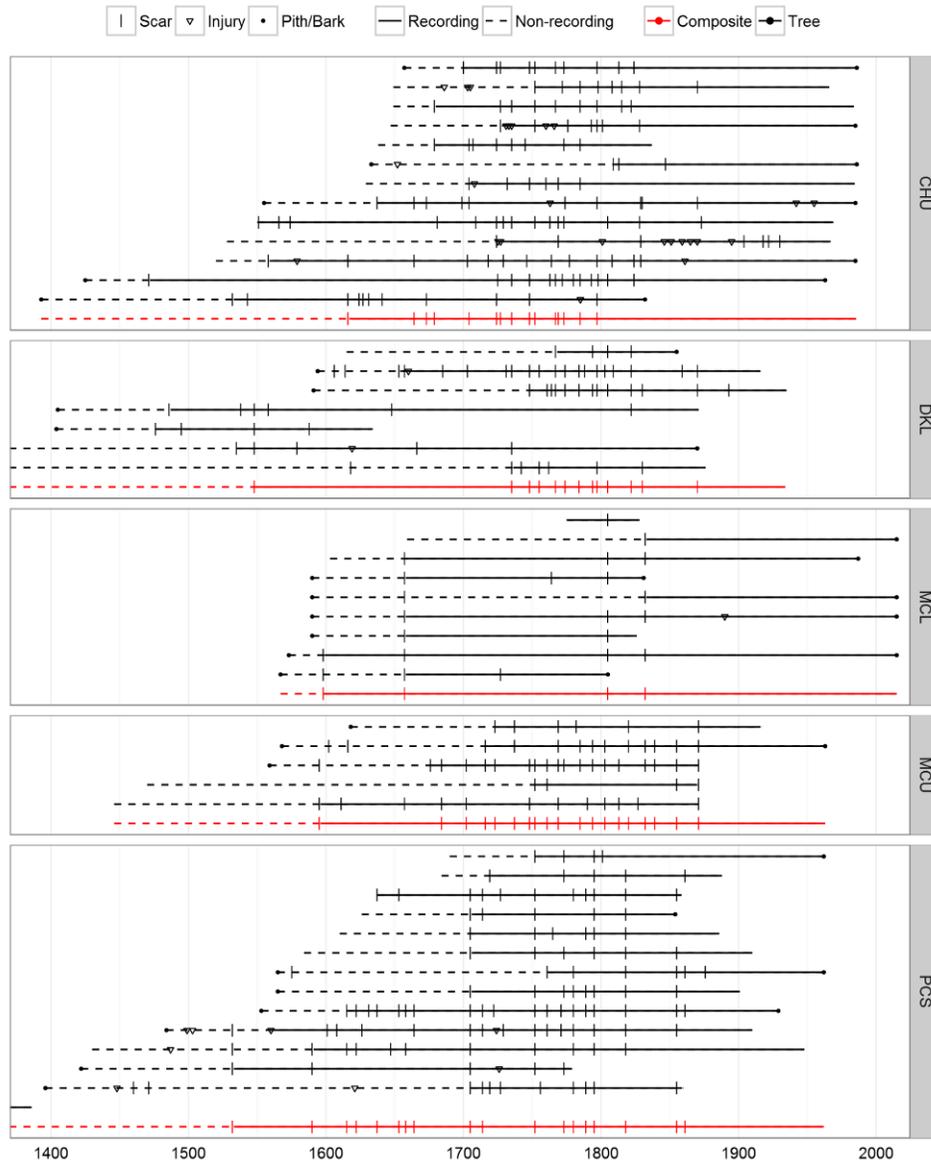
<sup>2</sup> Filters for 10% and 25% of trees scarred also include a minimum of 2 trees scarred and 2 trees recording

<sup>3</sup> Fits of the Weibull density function were assessed with a Kolmogorov-Smirnov test ( $\alpha = 0.10$ ). NA indicates a poor fit (Grissino-Mayer 1999).

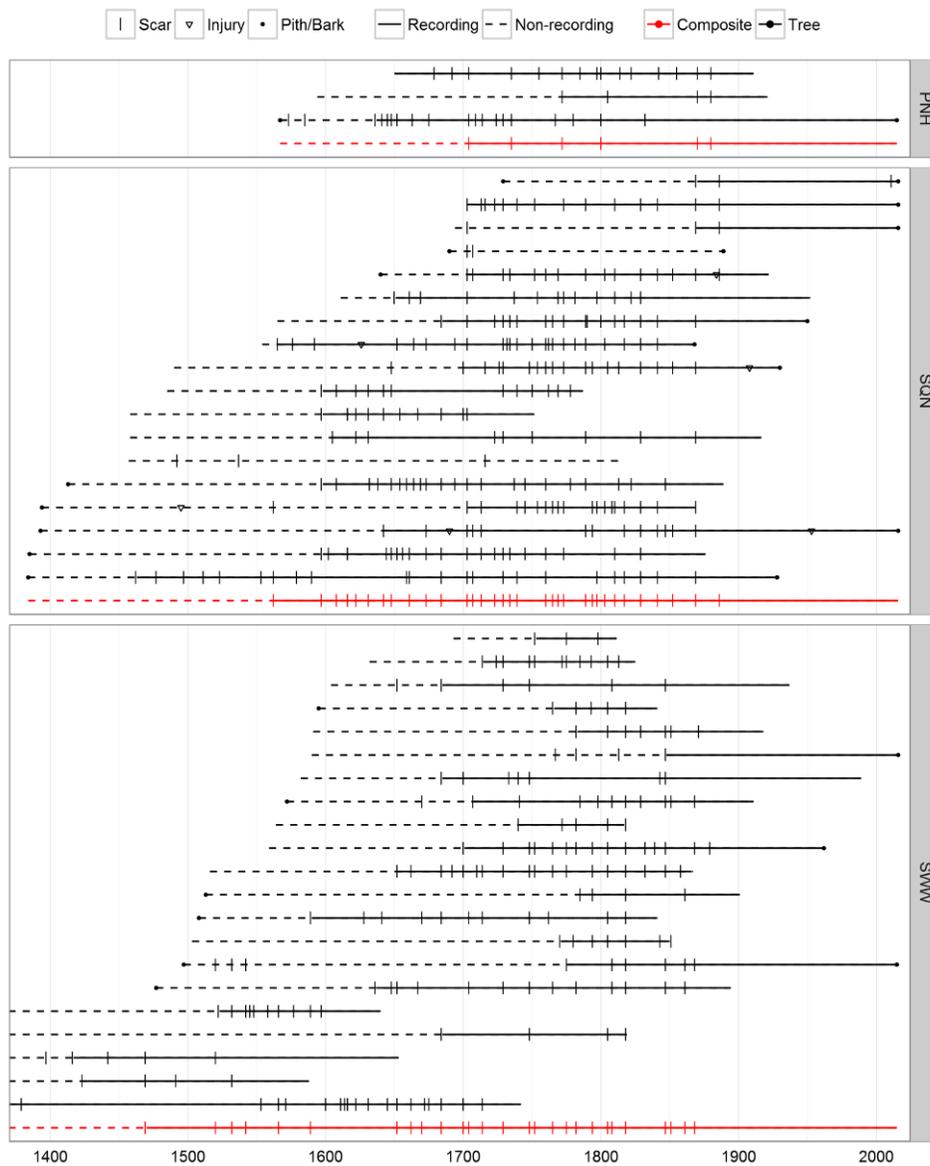
# FIGURES



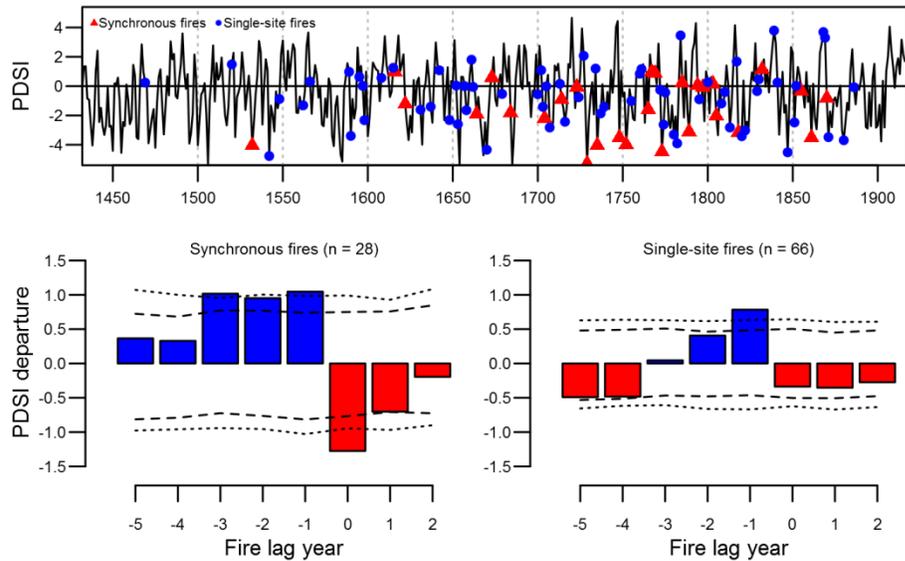
**Figure 1.** Locations of fire history sites on the Navajo Nation. Site names are provided in Table 1. The study area is shown as the blue box on the inset map of the Four Corners states.



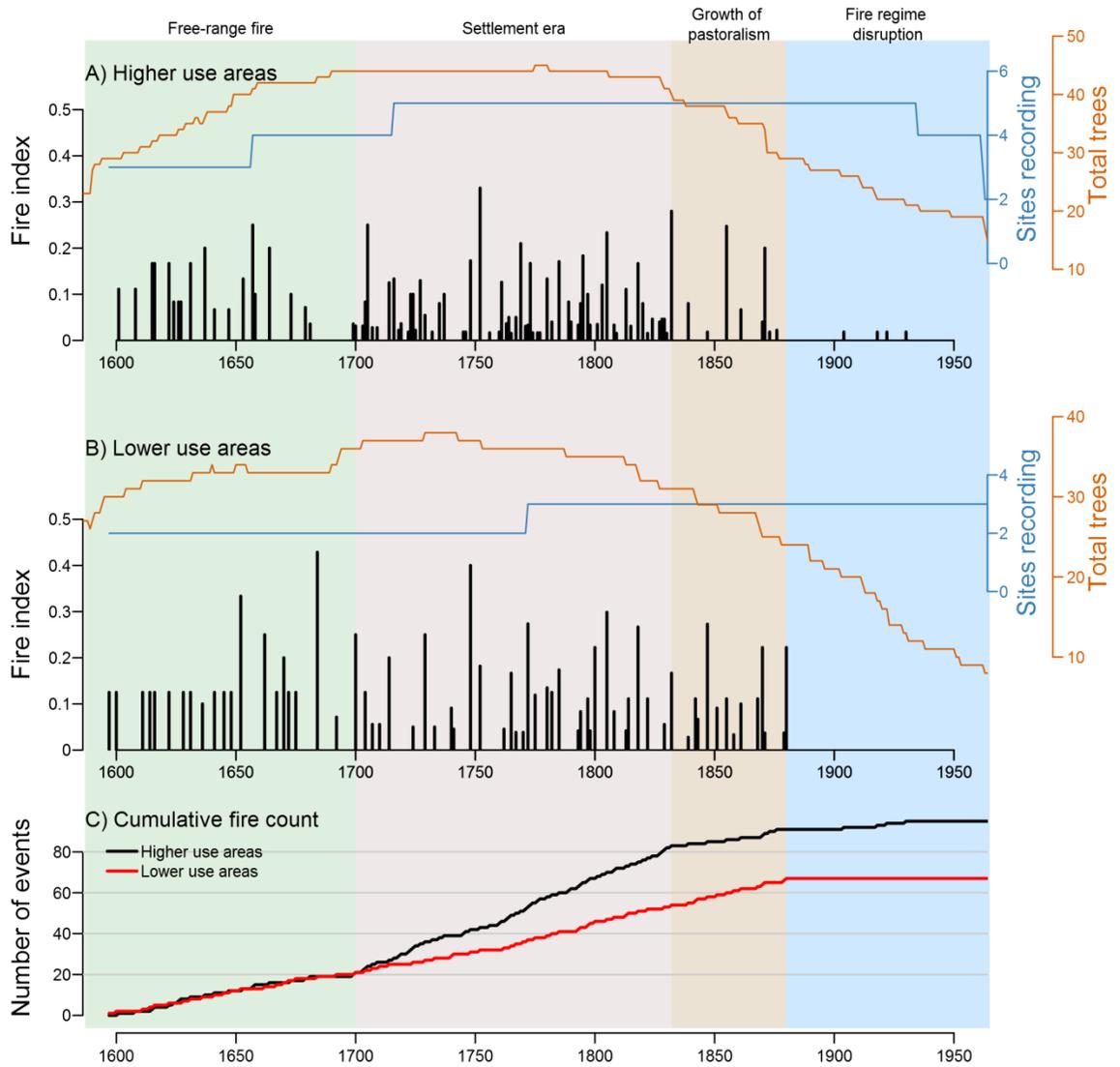
**Figure 2.** Fire history of “higher use areas”. Horizontal lines represent individual trees, except for the red site composite at bottom for each panel. Composites are filtered for  $\geq 25\%$  scarred and a minimum of 2 trees scarred. Site names are provided in Table 1.



**Figure 3.** Fire history of “lower use areas”. Horizontal lines represent individual trees, except for the red site composite at bottom for each panel. Composites are filtered for  $\geq 25\%$  scarred and a minimum of 2 trees scarred. Site names are provided in Table 1.



**Figure 4.** The fire-climate relationship of Navajo forests. At top, years with widespread ( $\geq 25\%$ ) fires recorded across multiple sites (synchronous) and widespread fires recorded at a single site are plotted against tree-ring reconstructed Palmer Drought Severity Index (PDSI) for the study area. At bottom, superposed epoch analyses (SEA) for the same fire years and climate reconstruction. Fire dates for the synchronous group occur during the period 1532-1870 and for single sites during 1469-1886. Positive (negative) values of PDSI indicate wet (dry) conditions. The year of the fire in the SEA is indicated as lag year zero. Bootstrap confidence intervals are provided at the 95% (dashed) and 99% (dotted) levels.



**Figure 5.** Fire indices of higher (A) and lower (B) use areas. Higher fire indices represent greater fire synchrony and relative extent. Cumulative fire event time series (C), show differences between the use areas after ~1700. Shaded periods represent distinct eras of human influences on the fire regimes.

APPENDIX B

DENDROECOLOGICAL METHODS FOR RECONSTRUCTING HIGH SEVERITY  
FIRE IN PINE-OAK FORESTS

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Dendroecological methods for reconstructing high severity fire in pine-oak forests

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## ABSTRACT

Recent high-severity fires in pine-oak forests of the southwestern United States are creating shrubfields that may persist for decades to centuries. Shrubfields embedded in conifer forests that pre-date documentary records are potential evidence of older high-severity fire patches, and may therefore provide insights into the occurrence and extent of past high-severity fires and vegetation type conversion dynamics. In this paper we test whether dendroecological evidence can be used to reconstruct a high-severity, type-changing fire of known date in a ponderosa pine (*Pinus ponderosa* var *scopulorum* Engelm.) dominated forest. Dendroecological evidence included (1) Gambel oak (*Quercus gambelii*, Nutt.) regeneration dates, (2) fire scars, (3) death dates, and (4) tree-ring growth changes. We reconstructed the historical fire regime and fire-climate relationship to evaluate whether the recent high-severity fire was driven by climate or fuel build-up related to a fire regime disruption. The dendroecological evidence correctly dated the year (1993) and season (spring) of the documented fire, and synchronous oak re-sprouts provided a means to estimate the minimum high-severity patch size. The historical fire regime at the site (1625-1871) consisted of frequent, low severity fires occurring in dry years preceded by wet years. Fires stopped in 1871, coincident with increased regional livestock grazing. The 1993 fire occurred under relatively cool and wet conditions, but followed a 122-year fire-free interval (four times the maximum historical interval). Multiple lines of evidence suggested that increased fuel loads from fire exclusion, combined with high winds, were primary drivers of the high-severity fire. The dendroecological approach we outline can be applied to reconstruct high-severity fire across a range of conifer-shrubland ecosystems.

*Keywords:* high-severity fire, shrubland, *Quercus gambelii*, Gambel oak, tree rings, fire scar, pine-oak, ponderosa pine

## INTRODUCTION

Recent large, high-severity wildfires in the western United States underscore the vulnerabilities of forests to climate change and human land-use (Allen *et al.* 2002;

Westerling *et al.* 2006). Over a century of human-caused fire exclusion in the frequent-fire forests of the western U.S. has increased fuel loads and altered forest structure, thereby increasing the probability of high-severity crown fire (Kilgore and Taylor 1979; Covington and Moore 1994; Taylor and Skinner 1998; Allen *et al.* 2002; Hessburg *et al.* 2005; Fulé *et al.* 2009). However, some have challenged this view by proposing that large, high-severity fires are within the natural range of variability in dry conifer forests of the western U.S. (Williams and Baker 2012, but see response by Fulé *et al.* 2014). Nonetheless, high-severity fires are likely to become increasingly common as temperature-driven drought stress is projected to rise in future decades (Williams *et al.* 2013).

One potential consequence of large high-severity fires in the dry conifer forests of the southwestern U.S. is an abrupt post-fire transition from forests to grasslands or shrublands. Historically, species such as Gambel oak (*Quercus gambelii*, Nutt.) commonly existed as a minor sub-canopy component or in the understories of pine-dominated forests and were limited by frequent fires and competition for light (Abella 2008; Abella and Fulé 2008). Large patches of pine mortality in recent fires have led to dominance of Gambel oak and other re-sprouting species (e.g., *Robinia neomexicana* A. Gray, *Arctostaphylos spp.*, and *Quercus spp.*) (Barton 2002; Savage and Mast 2005; Haire and McGarigal 2008; Savage *et al.* 2013; Collins and Roller 2013).

Little is known about the long-term (multi-decadal to century-scale) ecosystem dynamics of post-fire shrubfields in dry conifer forests of this region. Succession of shrubfields back to pine-dominated forests after high-severity fires may be hampered by multiple contingencies for recruitment of ponderosa pine (*Pinus ponderosa* var

*scopulorum* Engelm.) in a xeric post-fire environment (Feddema *et al.* 2013), including the availability of viable seed trees (Puhlick *et al.* 2012). Large high-severity fire patches (100s to 1000s of hectares) represent the most challenging areas for recovery of ponderosa pine because of the generally short-distance dispersal of its heavy, wingless seeds (Larson and Schubert 1970; McDonald 1980). Thus, established ponderosa pine seedlings are rare in high-severity fire patches at distances greater than 250 m from the edge (Haire and McGarigal 2010; Haffey 2014). Furthermore, rapid establishment by re-sprouting shrubs results in a competitive disadvantage for pine seedlings (Shainsky and Radosevich 1986; Oliver 1990) and future disturbances may serve to favor shrubs over pine (Brown 1958; Barton 1999; Savage *et al.* 2013). It is therefore possible that post-fire shrubfields represent alternative metastable states that are more resilient than conifer forests to future disturbance and climate change (Falk 2013).

Given observations of shrubfield persistence for multiple decades following fire (Savage and Mast 2005, Iniguez *et al.* 2009, Savage *et al.* 2013), existing old (pre-1900) patches of Gambel oak or other shrubs could indicate evidence of past high-severity fires in mid-elevation dry conifer forests. There is some evidence from the western U.S. that tree-rings can be used to date shrubfield formation following high-severity fire. In southern California, annual rings of chaparral shrubs (e.g., *Adenostoma spp.*, *Arctostaphylos spp.*, and *Ceanothus spp.*) have been used to indicate stand ages (Keeley 1993; Keeley *et al.* 2008), and synchronous fire scars on isolated stands of bigcone Douglas-fir (*Pseudotsuga macrocarpa* [Vasey] Mayr) were used to reconstruct crown fire history within chaparral shrublands (Lombardo *et al.* 2009). In pine-oak forests on Rincon Peak in southern Arizona, Iniguez *et al.* (2009) used conifer tree-ring evidence of

fire (fire scars and death dates) to reconstruct a 60 ha high-severity fire patch dating to 1867, some of which remains an oak shrubfield. In the oak-dominated shrublands of Mesa Verde, Colorado, Floyd *et al.* (2000) used ring counts on oak and other shrubs presumed to have sprouted following fire to estimate decadal-scale fire rotation, and noted multiple high-severity events in the pre-settlement era. In western Colorado, Brown (1958) documented patches of Gambel oak with stems as old as 212 years based on ring counts, but the origins of the patches were not assessed.

Multiple lines of evidence that include shrub regeneration dates have the potential to reconstruct the year, seasonality, and spatial scale of past high-severity fires in this region, as has been demonstrated in upper-elevation mixed conifer-aspen (*Populus tremuloides* Michx.) forests (Margolis *et al.* 2007). Successful employment of these methods at lower elevations depends on overcoming several hurdles, including the ability to accurately crossdate various shrub species (e.g., *Acer spp.*, *Robinia spp.*, and *Quercus spp.*) to derive post-fire age structures, documenting the temporal distribution of shrub re-sprouting (i.e., immediate versus lagged post-fire regeneration), identifying disturbance-related tree-ring growth responses of surviving trees with substantial fire-related crown loss around the high-severity fire perimeter, as well as finding and dating fire-killed conifers within the high-severity patch to confirm tree mortality. It is also important to consider that other factors may explain the presence of large shrubfields, including site characteristics (exposure, steep and rocky terrain, soils), disturbances other than fire (logging, grazing, drought-induced mortality), or some combination of factors. Where evidence exists to support past fire-related shifts from forest to shrublands, determining the timing, scale, and environmental precursors of these events would provide insights

into type-change behavior. In addition, assessing stand structure and conifer recruitment in old shrubfields related to high-severity fire would provide data on long-term post-fire successional dynamics, or the potential for type conversion.

The goals of our study are to evaluate whether shrubfields in pine-oak ecosystems can be used to reconstruct high-severity fires and to apply our findings to compare the conditions associated with a recent high-severity fire to historical conditions. We focus on a Gambel oak shrubfield that resulted from a documented (1993) crown fire in a ponderosa pine-dominated forest to answer two research questions: (1) can multiple lines of dendroecological evidence reconstruct the timing (year and season), spatial scale, and historical fire regime of a high-severity fire in pine-oak forests? and (2) how does the climate and preceding fire-free interval associated with the documented high-severity, type-changing fire compare to the historical (pre-1880) fire-climatology and fire frequency at the site?

## METHODS

### **Study site**

To test our methods, we chose a recent high-severity fire patch on Peggy Mesa in the southwest corner of the Jemez Mountains in north-central New Mexico (35° 43' 41" N, 106° 48' 5" W, Figure 1). Mean elevation of the sampling area is 2,200 m. The climate is continental and semiarid. Precipitation is dominated by summer convective storms (rain) associated with the North American Monsoon (Sheppard *et al.* 2002), with highly variable winter precipitation (snow). Annual average precipitation is 496 mm, with 41 % of the annual total occurring in July – September (1981 – 2010; PRISM 2012). Annual

average maximum temperature is 16.0° C and annual average minimum temperature is 0.4 ° C (1981 – 2010; PRISM 2012). A warm, dry and windy pre-monsoon period occurs annually in the spring and early summer (April – June), which corresponds with the peak annual area burned (Swetnam and Betancourt 1990; Westerling *et al.* 2006). Soils are composed of sandy loam, derived from local sandstone bedrock. Current vegetation within the sampled high-severity fire patch is dominated by Gambel oak. Surrounding (non-lethally-burned) vegetation is dominated by ponderosa pine with tree-form Gambel oak in the sub-canopy and smaller, shrub-form Gambel oak in the understory. Other tree species present at the site include Douglas-fir (*Pseudotsuga menziesii* Mirb. Franco), white fir (*Abies concolor* (Gord. & Glend.) Lindl. Ex Hildebr.), Rocky Mountain juniper (*Juniperus scopulorum* Sarg.), and piñon pine (*Pinus edulis* Engelm.).

### **1993 Buchanan Fire**

The study site is located within the 4,675 ha Buchanan prescribed fire that burned from April 20 – 23, 1993. In the late afternoon on April 22<sup>nd</sup> a rapid change in wind direction and high wind speeds drove an independent crown fire that burned with high severity in the pine-oak forest and killed one fire fighter (USDA Forest Service 1993). This extreme fire behavior resulted in multiple high-severity fire patches that regenerated as Gambel oak. From 2013 digital aerial photos (e.g., Figure 1), we identified seven patches (totaling 116 ha) dominated by oak shrubs due to clear differences in crown cover, stem height (lack of shadows), foliage color (brown and grey in cool-season imagery), and stand configuration (discontinuous groups) compared to the adjacent,

intact, darker evergreen ponderosa pine forest. We focused on the largest discrete high-severity patch (38 ha) within the fire perimeter for this study (Figure 1).

### **Field sampling**

We adapted the upper elevation high-severity fire reconstruction methods of Margolis *et al.* (2007) to collect four lines of dendroecological evidence for precise dating and assessment of the historical fire regime in the high-severity fire patch. The four lines of evidence included (1) oak regeneration dates, (2) fire scars, (3) death dates, and (4) tree-ring growth changes or injuries (Figure 2). Oak regeneration (re-sprouting) dates were determined from sampling in a random, spatially distributed plot design. In ten circular age structure plots (0.10 ha) we sampled the two largest diameter Gambel oak stems at ground level with a chainsaw. Within and around the edge of the shrub patch, we used a targeted sampling approach to collect the additional lines of evidence. The goal of our targeted sampling was to find the largest, most complete tree-ring record of fire that was spatially dispersed within the study site and near the edges of the high-severity burn patch. Targeted fire-scar sampling is an efficient and accurate approach for obtaining a representative fire history (Farris *et al.* 2013). Fire scars were collected to date the recent fire and reconstruct the historical fire regime. Samples were collected with a chainsaw from living trees located on or near the edge of the patch that survived the 1993 fire and from remnant logs and stumps throughout the patch. For death dates, we collected cross-sections from trees that were potentially killed by the 1993 fire (indicated by burned stubs of fine branches or charred bark) and had a clear waney (under bark) surface, beetle

galleries, and/or intact bark with no signs of erosion of the outermost sapwood. Living trees located on the edge of the patch that lacked a fire scar, but had obvious fire-related crown damage (e.g., Figure 2d), were cored to assess for tree-ring growth changes or anomalies.

As an initial assessment of conifer re-colonization in a shrub patch following high-severity fire, which affects aerial photo interpretation of patch size reconstruction, we recorded the number and species of conifers within each 0.10 ha oak age structure plot. These data were used to calculate overall densities (stems per ha) of post-fire conifer recruitment and maximum distances from the forest edge (measured as the distance between the plot center and the nearest forest edge).

### **Laboratory methods**

All tree-ring samples were prepared and crossdated using standard procedures (Speer 2010) to determine annually accurate and precise fire scar dates, tree death dates, tree injury dates, and Gambel oak regeneration (pith) dates. For all fire scars, we recorded the year of scarring and the intra-ring position (seasonality) of the scar (Baisan and Swetnam 1990). We visually assessed cores collected from potentially fire-injured trees for growth changes, multiple missing rings, and increased density of traumatic resin ducts. The timing of identified anomalies was compared with local, climatically-sensitive ponderosa pine tree-ring chronologies from Fenton Lake and Cat Mesa (Swetnam and Lynch 1993) to determine if they were climate-related. If climate variability did not appear to be the major cause of the growth anomaly, we considered the synchrony of

tree-ring anomalies among trees (and with other lines of fire evidence) to be a response to partial crown mortality from high intensity fire.

### **Analysis**

The four lines of tree-ring evidence were compiled and compared to the known year (1993) and season (April – early growing season) of the Buchanan fire to assess the precision and accuracy of the tree-ring methods. The area of the largest high-severity fire patch was digitized from aerial photos (as described above). Fire frequency statistics and seasonality of the reconstructed fire regime prior to the 1993 fire were calculated using FHX2 software (Grissino-Mayer 2001). The length of the fire-free interval preceding the 1993 fire was compared to the distribution of historical fire intervals before late 19<sup>th</sup> century fire exclusion using the Weibull exceedance interval.

The fire-climatology of the 1993 high-severity fire was quantified using gridded climate data (PRISM 2012) and fire incident reports. Departures of monthly temperature and precipitation for the month and year of the fire (April 1993) and preceding months (October 1992 – March 1993) were compared to average conditions over the full record (1895-2013) and recent decades (1981-2010). Fire-climatology of the historical fire regime was quantified using superposed epoch analysis (SEA; Swetnam 1993). Tree-ring reconstructed inter-annual climate variables used in the SEA include prior cool-season precipitation (prior October through current June) from the Jemez Mountains, New Mexico (Touchan *et al.* 2011) and summer (June through August) Palmer Drought Severity Index (PDSI; Cook and Krusic 2004: gridpoint 133).

## RESULTS

We crossdated tree-ring samples from 45 trees of five species to reconstruct the high severity fire (Table 1). The combination of the four lines of evidence (Gambel oak regeneration, fire scars, death dates, and tree-ring growth anomalies) dated the fire correctly to 1993 at the beginning of the growing season (Figure 3). All sampled Gambel oak ramets re-sprouted the year of the fire (pith date of 1993), with no lagged recruitment detected in our sample. Large inner rings on the oak regeneration indicated favorable initial growth (see photo inset on Figure 3). The “bark-ring” death dates for all sampled trees that appeared to be killed by fire were 1992 (complete ring), indicating that the trees died after the end of the 1992 growing season, but before the onset of growth in the spring of 1993. This is consistent with the timing of the fire in April. The five sampled living ponderosa pine with visible fire-related crown damage (i.e., killed branches on their lower boles, and/or on the side facing the high-severity burn patch) had reduced tree-ring growth beginning in 1993 (see photo inset on Figure 3). For example, one tree had four missing rings between 1993 and 1999, which is a period when only one ring (1996) is occasionally absent on trees from local chronologies. Four of the five fire-scarred trees sampled along the perimeter of the high-severity fire patch had a dormant-season scar occurring between the 1992 and 1993 growth rings. The fifth fire-scarred tree, a 450-year-old live ponderosa pine with 13 fire scars, recorded the fire as a 1993 early-earlywood scar.

The historical fire regime within and around the 1993 crown fire patch was characterized by frequent, low severity fires for over 250 years until fires stopped in 1871 (Figure 3). The mean fire interval across all trees and for all scars during the period 1625-1871 was 10.7 years, with a Weibull median fire interval of 9.5 years. Preceding the 1993 Buchanan fire, there was a 122-year fire-free interval (Figure 3). This interval was over 100 years longer than the upper bounds of the Weibull exceedance interval (19 years) and four times longer than the historical maximum fire interval (30 years). The seasonality of historical fires was predominantly spring and early summer (97% dormant, early-earlywood, and middle-early wood), which was similar to the season of the 1993 fire.

The preceding cool-season climate associated with the 1993 fire was cool ( $> 1$  standard deviation below the mean) and wet ( $> 130\%$  of average) compared to recent decades (1981-2010) and long-term (1895-2013) averages. The month of the fire (April 1993) was dry (21% of average precipitation [1981-2010 and 1895-2013]) and strong winds were reported during the high-severity fire behavior (USDA Forest Service 1993). Historical fire occurrence (1625 – 1871) was associated with below average cool-season precipitation preceding the fire season and above average cool-season precipitation three years prior to the fire (Figure 4). A similar fire-climate relationship (dry during the fire year and wet in prior years) was indicated with PDSI (Figure 4).

Mean density of conifer regeneration in the oak shrub patch 20 years following the 1993 high-severity fire was 11 trees  $\text{ha}^{-1}$ . Conifer seedlings were present in only four of the ten plots. Most (8 of 11) seedlings were ponderosa pine, but Rocky Mountain juniper ( $n = 2$ ) and Douglas-fir ( $n = 1$ ) were also present. Maximum distance of a post-

fire ponderosa pine seedling to the patch edge was 77 m. Rocky Mountain juniper seedlings were found the farthest (135 m) from the forest edge.

## DISCUSSION

### **Tree-ring reconstruction methods**

We successfully used multiple lines of dendroecological evidence to reconstruct the year (1993) and season (spring – early growing season) of a high-severity fire in a pine-oak forest in the southwestern U.S., as verified by documentary records. The multi-proxy, dendroecological approach substantially strengthened our ability to determine the timing, scale, and contributing factors of a high-severity, type-change fire at Peggy Mesa. The possibility of a fall 1992 fire (after the growing season) could not have been ruled out using only regeneration dates, death dates, or tree-ring growth suppressions. A fire scar recorded in the earlywood of 1993 was used to date the fire to the spring of 1993. Although not commonly used for dendrochronology, Gambel oak proved to be a robust species for accurate crossdating back to the pith ring. Additionally, in the case of this spring high-severity fire at Peggy Mesa, Gambel oak pith dates are a reliable proxy for the disturbance year, but require further evidence (e.g., fire scars, tree death dates) to be confidently connected to a past high-severity fire. Based on the success of this multiple lines of evidence approach, we have confidence in this method to reconstruct high-severity fire in older shrubfields that were potentially type converted from pine-dominated forests.

One challenge of reconstructing older high-severity fires from oak patches is the possible rarity of remnant fire-killed trees with intact outside surfaces that can provide

“bark-ring” tree death dates. The most direct evidence of past high-severity fire is the presence of fire-killed trees. Yet, after only twenty years post-fire at Peggy Mesa, very few fire-killed trees were still standing. Studies of snag dynamics in fire-killed forests in Arizona and Oregon indicate similar low rates of snag retention greater than 20 years post-fire among multiple conifer species, with larger snags generally lasting longer (Everett *et al.* 1999; Passovoy and Fulé 2006; Roccaforte *et al.* 2012). Most of the snags that persisted at Peggy Mesa 20-years post-fire were Douglas-fir and Rocky Mountain juniper, although they were a minor part of the pre-fire stand composition. The boles of these snags were in good condition, with charred bark still attached and many charred fine branches (Figure 2c). Nearly all of the fire-killed ponderosa had fallen by 2013 (20 years post-fire), and substantial portions of the tree boles were too decayed for tree-ring sampling. Many of our samples of fire-killed ponderosa pines, therefore, came from the upper portions of the boles that were leaning against oak clumps and stayed off the ground. Since ponderosa pine makes up a majority of conifers killed in high-severity fires within the pine-oak forest type in the southwestern U.S., a lack of fire-killed, tree death dates might prove to be a challenge in interpreting high-severity fire as the cause of older patches of Gambel oak. Some evidence for tree mortality may be attained by mapping logs or by sampling for near-bark outer-ring dates, where possible, if much of the former forest has fallen or decomposed.

Although loss of remnant logs and snags from the pre-existing forest due to decomposition limits the discovery and use of the death-date line of evidence within high-severity burn patches, it is notable that this evidence may still be found with extensive searching. For example, Margolis *et al.* (2007) successfully located fire-killed

conifers as old as 160 years post-fire in mesic mixed-conifer and aspen stands in the southwestern U.S. Also, in a relatively dry pine-oak forest in southern Arizona, Iniguez *et al.* (2009) found old fire-killed trees in shrub-dominated patches that dated to 1867, thereby dating high-severity burn patches that were essentially converted to shrub patches, and that have now persisted for 147 years. It may be that this line of evidence, i.e., remnant, fire killed snags and logs, is relatively more abundant now than in the past because of fire suppression. The lack of fire for more than a century has likely prevented some old, dead wood within shrub patches from burning up during subsequent fires.

Patch size is perhaps the most important metric of high-severity fire regimes (Agee 1993), and is therefore an important aspect of fire history reconstruction. The synchrony of oak recruitment throughout the fire patch (all 1993 pith dates, Figure 3), provided a means to estimate pre-instrumental high-severity patch size. The low density of conifer recruitment (11 trees ha<sup>-1</sup>) into this small shrub patch 20 years post-fire suggests that post high-severity fire shrub patches in the region could be delineated from aerial photos for decades, if not longer. Observations of conifer regeneration from the forest edge inward (e.g., Haire and McGarigal 2008, 2010), suggest that delineating the original edges of the high-severity patch from aerial imagery will become more difficult over time. Additional errors in patch size reconstruction from shrubfields could include dense post-fire conifer recruitment (e.g., Savage and Mast 2005, Iniguez *et al.* 2009) that may not be as clearly visible as shrubfields on aerial imagery. Height differences between the lethally burned and non-lethally burned vegetation may be used to overcome these challenges. Should older shrubfields re-burn, however, the existing shrubfield patch

could be expanded in size (Savage *et al.* 2013), which would inflate estimates of patch size for the older high-severity fire.

### **Current vs. historical fire regime**

The 1993 high-severity fire at Peggy Mesa that abruptly changed a pine-dominated forest to a shrubfield occurred at a site where fire historically burned frequently (10 year return interval), with low severity for hundreds of years (Figure 3). The 122-year fire-free interval that preceded the 1993 high-severity fire was anomalous compared to the previous 400 years (Figure 3). Long fire-free intervals have been associated with pre-1900 high-severity, type conversion fire in the region (Iniguez *et al.* 2009). In this case, a rocky, topographically heterogeneous landscape structure reduced the likelihood of fire spread into an isolated pine-oak forest stand that burned with high severity following a 48-year fire-free interval. In contrast, the end of the frequent fire regime at Peggy Mesa in 1871 is synchronous with the collapse of frequent fire regimes locally (Touchan *et al.* 1996) and at dozens of other dry conifer sites across the region (Swetnam and Baisan 1996). This was clearly initiated by the introduction of large numbers of livestock in the late 1800s that removed the fine fuels that carried surface fires (Allen 2007). The synchronous cessation of surface fire in the late 1800s has been related to increased forest density in piñon-juniper, ponderosa pine, and dry mixed conifer forests throughout the region (Allen *et al.* 2002; Fulé *et al.* 2009; Margolis 2014), and likely resulted in high tree density, high fuel loads, and increased ladder fuels that supported crown fire at Peggy Mesa. All but one of the sampled trees at Peggy Mesa that

were alive in 1993, regenerated after frequent fires stopped in 1871 (Figure 3), suggesting that a high density of young trees fueled the 1993 high-severity fire. The climate associated with the 1993 high-severity fire was mild (wet and cool) compared to the long-term instrumental record and the historical fire-climatology, suggesting that climate was not a primary driver of change in fire severity. Strong winds were reported during the event and likely were a contributing factor (USDA Forest Service 1993). Overall, these findings highlight the important role of human-caused changes in fire regimes and forest structure, combined with fire weather, to produce an ecologically-significant forest-to-shrub transition. This contradicts the conclusions of Williams and Baker (2012), that recent high severity fires in dry conifer forests are not associated with human land use.

## CONCLUSIONS

Abrupt post-fire transitions of ponderosa pine-dominated forests to shrubfields are increasingly common following large high-severity fires in the southwestern U.S. With little known about the long-term (decades to centuries) consequences and ecosystem dynamics of these transitions, reconstructing the origins of existing, old (pre-1900) shrubfields is an important area for future research. In this paper, we have validated dendroecological methods that can be applied to test whether shrubfields originated from high-severity fire, with the potential to provide crucial information on high-severity fire in the historic era and post-fire successional trajectories. The chronological and spatial control associated with these methods can then help to decipher endogenous (e.g., increased fuel loads) versus exogenous (e.g., extreme drought) factors that could have led

to type conversion. We emphasize the need to use multiple lines of evidence in these tests. The combined use of different types of tree-ring evidence provides a cross-check and verification of results from individual proxies, thereby increasing overall confidence in the reconstructions. Characterizing post-fire shrubfields with these methods will provide important data for determining if recent post-fire forest-to-shrub transitions are an early seral successional stage (similar to post-disturbance quaking aspen) or whether they are an alternative metastable state that will persist for centuries.

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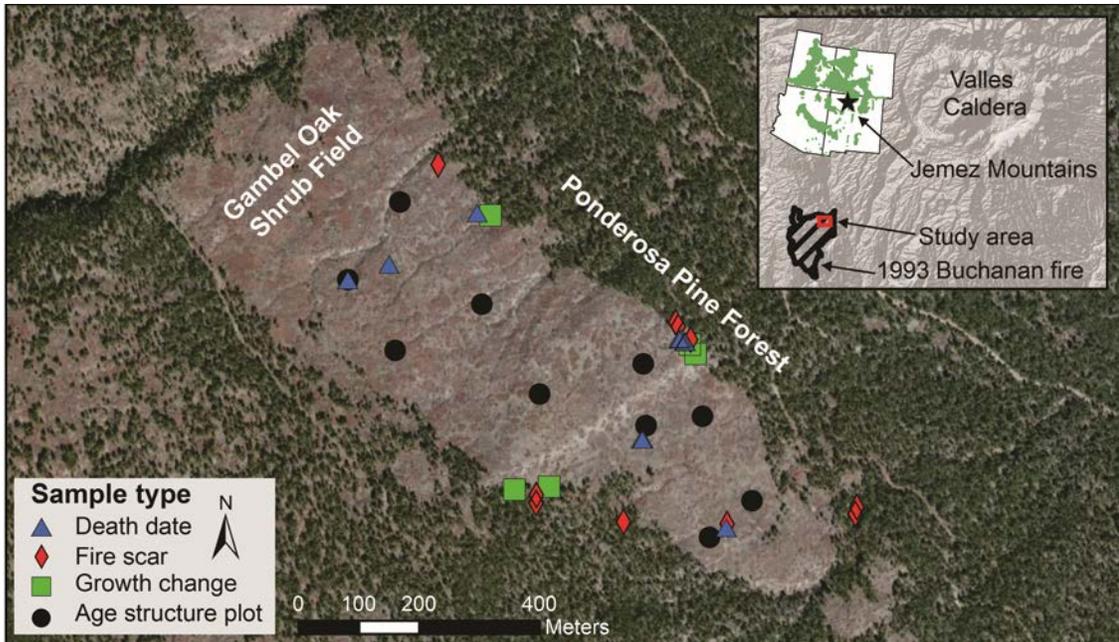
## TABLES

**Table 1.** Number, species, and tree-ring sample type of trees sampled for high-severity fire history reconstruction at Peggy Mesa, New Mexico.

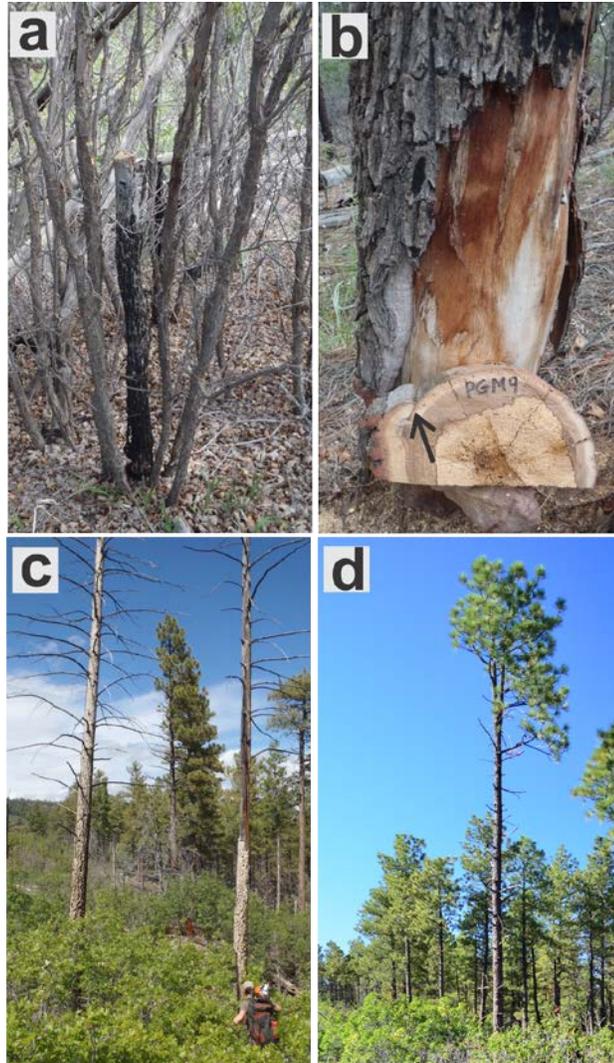
Species <sup>a</sup>	Age structure	Death date	Fire scar	Growth change	Total sampled trees
JUSC		1	2		3
PIED		1			1
PIPO		5	4	5	14
PSME		1	1		2
QUGA	20	3	2		25
All species	20	11	9	5	45

<sup>a</sup> JUSC = *Juniperus scopulorum*. PIED = *Pinus edulis*. PIPO = *P. ponderosa*. PSME = *Pseudotsuga menziesii*. QUGA = *Quercus gambelii*.figures

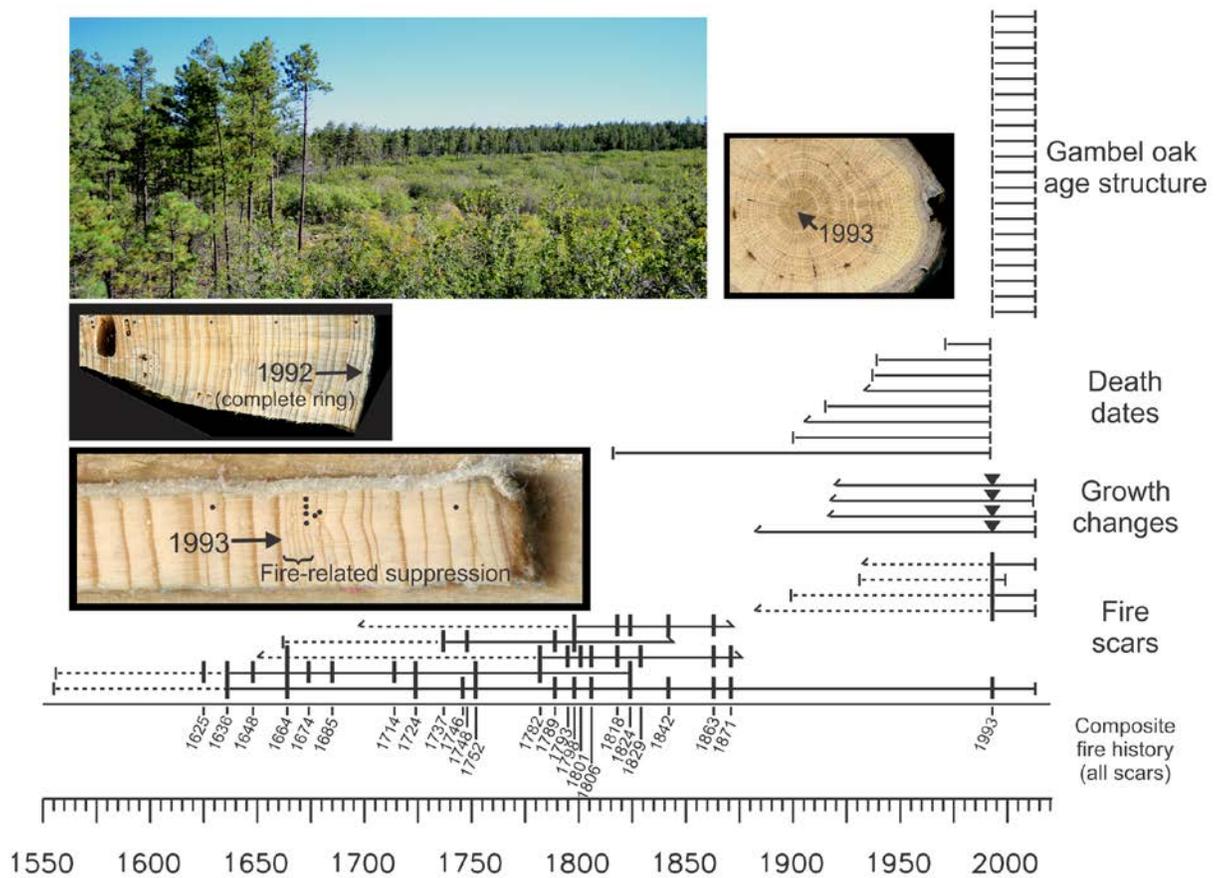
## FIGURES



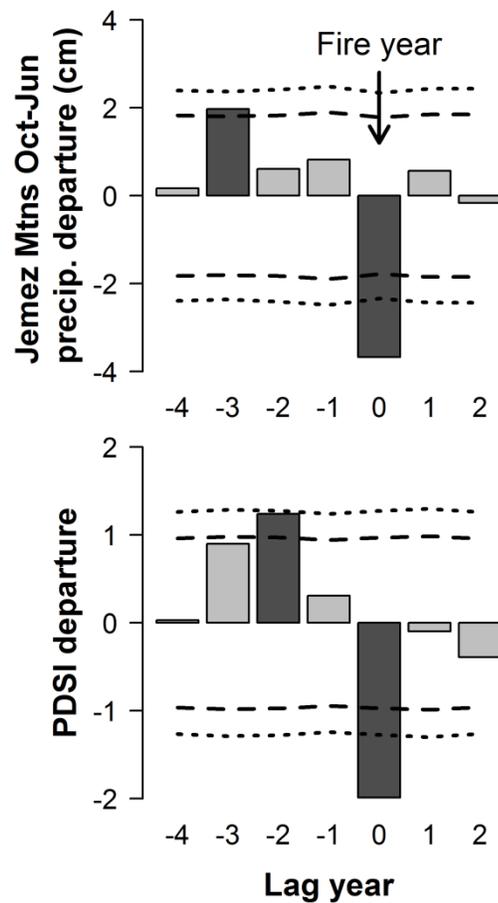
**Figure 1.** Aerial photo of a high-severity patch in the 1993 Buchanan fire (Jemez Mountains, New Mexico) that abruptly changed from a ponderosa pine-dominated forest to a Gambel oak shrubfield. Symbols indicate multiple lines of tree-ring evidence used to test a method for reconstructing high-severity fire in pine-oak. Inset indicates the location of the study area within the Southwest United States and the range of Gambel oak in green (Little 1971).



**Figure 2.** Four lines of dendroecological evidence used to date high-severity fire in pine-oak forests. (a) Gambel oak re-sprouts from near the base of a fire-killed oak stem (the fire-killed stem in this photo was sampled for a death date). The live stems were sampled at ground level to attain a pith date for the year of sprouting. (b) Fire-scarred tree-size oak on the edge of the high severity burn patch. The arrow denotes the fire scar on a cross section of the tree. (c) Fire-killed conifers inside of the high-severity patch sampled for death dates. (d) Live ponderosa pine on the edge of the high-severity fire patch with substantial fire-caused crown mortality that resulted in a tree-ring growth suppression.



**Figure 3.** Four lines of tree-ring evidence used to reconstruct a 1993 early growing season (April 22<sup>nd</sup>) high-severity fire in pine-oak. The composite fire-scar fire history indicates a historically frequent, low severity fire regime (1625 – 1871) followed by an anomalous (122 year) fire-free interval that preceded the 1993 high-severity, type-changing fire. Inset photographs show the Peggy Mesa shrubfield site, a sampled Gambel oak ramet, a fire-killed ponderosa pine, and a tree-ring growth suppression following fire on a surviving ponderosa pine, such as the one left-center in the top image. Each horizontal line represents the timeseries of individual trees, with dotted lines showing non-fire-recording years and solid lines recorder years on fire-scarred trees, vertical lines represent pith or bark rings, arrows represent inner and outer rings, triangles indicate growth changes, and bold vertical lines indicate fire scars.



**Figure 4.** Superposed epoch analysis indicating that historical fire occurrence (1625 - 1871) on Peggy Mesa, New Mexico was associated with dry conditions during the fire year, preceded by generally wet conditions (n = 24 fire years). Dashed and dotted lines represent 95% and 99% confidence intervals determined from 1000 iterations of Monte Carlo simulation.

APPENDIX C

LONG-TERM PERSISTENCE AND FREQUENT FIRE IN NORTHERN NEW  
MEXICO SHRUBFIELDS SUGGEST FUTURE INCREASING LANDSCAPE  
DOMINANCE

Long-term persistence and frequent fire in northern New Mexico shrubfields suggest future increasing landscape dominance

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## ABSTRACT

One of the consequences of recent, large high-severity fires in the interior western United States is the abrupt post-fire transition of mid-elevation dry conifer forests to resprouting shrub species, including Gambel oak (*Quercus gambelii* Nutt.). Little is known about the long-term successional trajectories of these fire-origin shrubfields. We evaluated the persistence of Gambel oak and reconstructed fire regimes in five of the largest pre-1900 shrubfields of the Jemez Mountains, New Mexico. We selected sampling sites by first identifying 349 potentially old shrubfields ranging in size from 5 to 367 ha. We note that fires in the study area since 1977 combined to create high-severity fire patches >27 times larger than the largest pre-1900 shrubfield. Age structures at all five sampling sites related to the last widespread fire in the late 1890s, demonstrating that Gambel oak shrubfields can persist in the absence of fire for more than 115 years. This is supported by aerial imagery from 1935 to present showing sparse conifer recruitment and little change in patch size. Reconstructed fire regimes for the period 1664-1899 showed frequent surface fire activity (median fire intervals of 9 to 13 years) with only occasional and isolated conifer mortality. These shrubfield fire regimes matched the frequency and climate relationships of 39 dry conifer sites in the study area, underscoring the flammability of Gambel oak shrubfields. Our findings indicate that recently converted shrub-dominated areas are not a seral stage of pine-oak forests, but rather possess multi-century stability in the presence or absence of disturbance and drought. Projected increases in forest drought stress and associated high-severity disturbances over the next several decades are likely to continue to drive forest-to-shrub conversions, and further reinforce the persistence of existing shrubfields.

## INTRODUCTION

Across the interior western US, climate change has raised significant concern over the future of fire-prone ecosystems. Fires appear to be worsening and affecting more

natural and cultural resources due to increasing temperature and aridity (IPCC 2014; Abatzoglou and Williams 2016), a longer fire season with increasing area burned (Littell et al. 2009; Dennison et al. 2014; Westerling 2016), larger patches of high-severity fire (Miller et al. 2009), increasing drought-related tree stress (Williams et al. 2013), and extreme fire-related climate conditions (Williams et al. 2014). Forest mortality from moderate to high severity fire is well documented in dry conifer forests, which include mid-elevation dry mixed conifer and ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) dominated forests. Large burn patches are especially problematic for managers, as they can abruptly transition to resprouting shrub species (Barton 2002; Savage and Mast 2005; Haire and McGarigal 2008; Iniguez et al. 2009; Roccaforte et al. 2012; Savage et al. 2013; Collins and Roller 2013; Guiterman et al. 2015; Allen 2016; Coop et al. 2016) (Figure 1). Whether these forest-to-shrub transitions are persistent alternative stable states or a seral stage in pine-oak ecosystems remains uncertain (Falk 2013).

The hypothesis that fire-origin shrubfields can persist for centuries or longer is supported by several advantages of resprouting shrubs over conifers in resisting drought and fire (Clarke et al. 2013; Savage et al. 2013; Zeppel et al. 2014; Pausas et al. 2016), in addition to competitive advantages of shrubs that resprout in dense clumps (Shainsky and Radosevich 1986; Oliver 1990). Recolonization by seed obligate conifers, like ponderosa pine, in large high-severity patches (100s to 1000s of hectares) is hampered by a lack of nearby seed trees (Haire and McGarigal 2010; Chambers et al. 2016) and drought stress to seedlings (Feddemma et al. 2013; Rother et al. 2015). With a warming climate, rapid forest-to-shrub transitions following high-severity fires might be indicative of a

dynamical adjustment to a new bioclimatic regime (Jentsch et al. 2007; Jackson et al. 2009; Turner 2010).

The alternative hypothesis of fire-origin shrub communities as an early seral stage that eventually transitions back to dry conifer forest is analogous to higher elevation mixed conifer-aspen (*Populus tremuloides* Michx.) ecosystems. Following stand-replacing fire, even-aged seral aspen stems that resprout from long-lived clones can dominate for multiple decades until being overtopped by Douglas-fir (*Pseudotsuga menziesii* Mirb. Franco), spruce (*Picea*), and fir (*Abies*) (e.g., Margolis et al. 2007). Documented succession of clonal resprouting shrubfields to dry conifer forest is rare, but as Falk (2013) describes, the co-occurrence of conifer and shrub species over millennia implies some ability to switch between different ecosystem states at some long time-scale. In more mesic environments, succession may be accelerated. For example, in northern California, forests have encroached upon montane chaparral stands, attributed to over a century of fire exclusion (Nagel and Taylor 2005; Airey Lauvaux et al. 2016). Indeed, perhaps the limited documentation of shrubfield persistence indirectly suggests the common-ness of shrub-to-conifer succession, given that the evidence for shrub dominance becomes obscured as conifers overtop the shrub canopy. Results of several studies refutes this, however, and instead document the potential for long-term persistence of Gambel oak (*Quercus gambelii*, Nutt.) in fire-origin shrubfields (Floyd et al. 2000; Ryniker et al. 2006; Iniguez et al. 2009).

Gambel oak is common in post-fire shrubfields in the Southwest. It ranges from northern Utah and southern Wyoming, where spring frosts limit its growth, to the sky islands of Sonora, Mexico where drought and increased competition from Madrean oaks

and chaparral shrubs limit it to cooler, wetter aspects and elevations (Little 1971; Rehfeldt et al. 2006). Its growth character varies along this latitudinal gradient from single stems in southern areas to dense clonal shrubfields toward its northern terminus (Neilson and Wullstein 1983; Harper et al. 1985; Clary and Tiedemann 1992). In much of the central area of its distribution, including northern Arizona and New Mexico, historically frequent surface fires probably kept Gambel oak cover low and maintained the species as an understory shrub with occasional tall, single-stem codominants in ponderosa pine forests (Fulé et al. 1997; Abella 2008). Recent high-severity fires, and past land-use (logging, grazing), raise the possibility that Gambel oak is increasing its distribution and becoming more dominant across the landscape (Abella and Fulé 2008; Kaufmann et al. 2016). It therefore must be ascertained whether the Gambel oak shrubfields in Arizona and New Mexico are a recent phenomenon or a long-established natural component to landscape heterogeneity.

Fire is a vital ecosystem function for landscape heterogeneity in mid-elevation dry conifer forests of the region (Allen et al. 2002). This point is underscored by the fact that recent, large high-severity fires in this ecosystem are largely attributable to over a century without fire, leading to landscape-scale build-up of forest fuels, made highly flammable by higher temperature and drought (Stephens et al. 2013; Abatzoglou and Williams 2016). Under a natural fire regime, variability in fire effects works to regulate landscape heterogeneity through differing vegetation feedbacks. In coastal forests of the western US, montane chaparral stands are supported by fire, with higher fire severities and longer return intervals than the surrounding conifer forests (Keeley et al. 2008; Odion et al. 2010; Airey Lauvaux et al. 2016). Similar patterns have been found in shrublands of

South America (Paritsis et al. 2014). Though we know Gambel oak is a prolific post-fire resprouter, we know little of how fire regulates Gambel oak shrubfields over long periods, because fire regimes in this ecotype have not been documented (Kaufmann et al. 2016). With thousands of hectares of recent fire-origin shrubfields to manage in a warming climate, land managers would benefit from understanding the effects of fire in shrubfields, and its implications for the successional trajectories of shrubfields.

The objectives of this study are to (1) identify Gambel oak dominated shrubfields in the Jemez Mountains, New Mexico, and compare their biogeographic distribution to dry conifer forests; (2) reconstruct the fire histories and age structures of five of the largest shrubfields via dendroecological methods validated for the study area (Guiterman et al. 2015); (3) determine if the shrubfields originated from large, high-severity fires; and (4) compare the fire regimes of shrubfields to dry conifer forests of the Jemez Mountains. Then, we consider the implications of these findings for the long-term trajectories of recent fire-origin shrubfields. Given the broad spatial coverage, scale, and documented persistence (> 50 years, Savage and Mast 2005) of recent forest-to-shrub transitions, we hypothesize that (1) existing, old shrubfields have a similar biogeographic distribution to dry conifer forests, (2) pre-1900 shrubfields may represent historic high-severity fire activity, and (3) that fire regimes of Gambel oak shrubfields are less frequent and tend to be more severe than neighboring dry conifer forests.

## METHODS

### **Study area**

The Jemez Mountains are located in north-central New Mexico, USA. The mountain range arose from volcanic activity, which created its most prominent landform features of the Valles Caldera and Redondo Peak (>3,400 m, the highest point in the Jemez). Bedrock in the volcanic core consists of Quaternary rhyolite and other volcanics, transitioning to Permian sandstones in the northern, southern, and western areas of the range (Anderson et al. 1997). The western Jemez is a long north-south trending ridge, often called the Nacimiento Mountains.

The climate of the region is continental and semiarid, with bimodal precipitation that includes regional-scale winter frontal storms (snow) and summer convective storms (rain) associated with the North American Monsoon (Sheppard et al. 2002). The climate regime is characterized by high interannual variability imprinted on decadal-scale droughts and pluvial periods (Touchan et al. 2011). Annual precipitation totals 496 mm, with 41% occurring in July – September, based on the 1981-2010 normals from the Precipitation-Elevation Regression on Independent Slopes Model (PRISM; Daly et al. 2008). Average annual temperatures range from a low of 0.4<sup>0</sup>C to a high of 16.0<sup>0</sup>C (1981-2010; PRISM 2016). The months of April through June are characterized by a dry and windy pre-monsoon period that corresponds to the peak in annual area burned (Swetnam and Betancourt 1990; Allen 2002; Westerling et al. 2006).

The Jemez Mountains are possibly the most sampled landscape in the world for tree-ring fire history research, now including over 1,377 trees (Swetnam et al. 2016). Changing patterns of fire frequency and spread over the reconstructed time period of

1500-1860 correspond strongly to shifting native American populations and changes in land use associated with Spanish and later Euro-American settlement (Liebmann et al. 2016; Swetnam et al. 2016). Fires were frequent and widespread from the late 1600s until approximately 1860, during which time winter precipitation patterns exerted the greatest control over fire synchrony, with the North American Monsoon playing an active role in limiting and synchronizing warm-season fires (Margolis et al. *in review*). The collapse of the fire regime ca. 1860-1900 (Touchan et al. 1996) is attributed to the arrival of the transcontinental railroad, which encouraged heavy grazing that initiated 20<sup>th</sup> century fire exclusion (Allen 2007). Over the last several decades fires have returned to the landscape, with several large, high-severity fires generating extensive mortality in dry conifer forests that has now resulted in near complete dominance of Gambel oak and New Mexico locust (*Robinia neomexicana*) (Savage and Mast 2005; Haire and McGarigal 2008; Allen 2016; Coop et al. 2016).

Our study sites are located in some of the largest Gambel oak dominated shrubfields in the Jemez Mountains, and ranged in elevation from 2450 m to 2800 m (Figure 2). Understory composition of the shrubfields varied from grasses at lower topographic positions to shrubs, dominated by gooseberry (*Ribes leptanthum*), in the middle and upper slopes. New Mexico locust and aspen were present in more mesic areas of the shrubfields. Conifers were present as scattered individuals or in small clumps within shrubfields, usually closer to the forest edge, and often consisted of ponderosa pine, Rocky Mountain Juniper (*Juniperus scopulorum*), Douglas-fir, white fir (*Abies concolor*), and piñon (*Pinus edulis*). Forests adjacent to the shrubfields were usually dominated by ponderosa pine, but on cooler and wetter sites and aspects (e.g., drainages

and more northerly slopes), forest composition included Douglas-fir, southwestern white pine (*Pinus strobiformis*), and white fir.

### **Identification and evaluation of pre-1900 potentially fire-origin shrubfields**

We identified and mapped all Gambel oak shrubfield sites in the Jemez Mountains over five hectares to provide perspective on likely past high-severity fire activity leading to transitions of conifer stands to oak (Figure 2). To aid us, we used vegetation datasets in a geographic information system (GIS) from the Santa Fe National Forest, Southwest Regional Gap Analysis (SWReGAP; Lowry et al. 2007), Valles Caldera National Preserve (Muldavin and Tonne 2003), and Bandelier National Monument (Muldavin et al. 2011). Visual analyses of aerial photographic imagery, comparison with other GIS layers, and some ground inspection were conducted to ensure that our final mapped shrubfields were dominated by Gambel oak and not misclassified. The vegetation layers we used are current to 2004 in some areas, but do not include large Gambel oak dominated shrubfields created by the 2000 Cerro Grande Fire. They also predated the 2011 Las Conchas Fire, which, in combination with some previous fires, created treeless and shrub-dominated patches of 1,000 to >10,000 ha (Allen 2016).

We compared the biogeography of Jemez shrubfields to dry conifer forests in the study area to evaluate whether shrubfields occupy distinct landscape positions or climatic niches. Similar physiographic and climate settings would suggest the ability of Gambel oak shrubfields to occupy nearly any area within the conifer-dominated landscape. Dry conifer forests are defined as those mapped in SWReGAP as dominated by ponderosa pine (“Southern Rocky Mountain Ponderosa Pine Woodland”) or dry mixed conifer

(“Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland”). Since the latter forests “often have ponderosa pine” (SWReGAP database), it provides accurate coverage of dry mixed conifer forests in the study area as opposed to mesic mixed conifer forests where ponderosa pine is usually absent (Romme et al. 2009). The comparison was done in a Principle Components Analysis (PCA) on 10,000 random points in both the mapped shrubfields (>5 ha) and dry conifer forests. We included the physiographic variables of elevation, slope, aspect, curvature, topographic relative moisture index (TRMI; Parker 1982), and topographic position index (TPI; Jenness 2006). Climate variables came from an 800 m grid of annual precipitation and temperature (max, min, and mean) for the 1981-2010 period (PRISM 2016).

### **Site selection and field sampling**

We selected five of the largest shrubfields sites for field sampling (Figure 2). The selection process included field surveys to evaluate the potential for these sites to be fire-origin during the pre-1900 historical period. Evidence of fire-origin included distinct forest-shrub edges with no clear topographic component, large old Gambel oak clumps (genets), remnant, potentially fire killed trees preserved as logs, snags, and/or occasional stumps, and living trees with elevated crowns and/or fire-scarred branches (see Fig. 2 in Guiterman et al. 2015). We also searched for fire-scarred material in and along the edges of the shrubfields to ensure specimen availability for fire history reconstruction. Each site had soils derived from local bedrock, which at four of five sites was Permian sandstone. One site, Redondo Creek (RDC), was located on the lower slopes of Redondo Peak, and had bedrock composed of rhyolite and basalt, intermingled with tuff.

At each site we collected multiple lines of tree-ring evidence to characterize stand age and fire history. Our methods were validated in the study area, where we reconstructed a 1993 high-severity fire that created a Gambel oak shrubfield in a former ponderosa pine-dominated forest (Guiterman et al. 2015). Similar methods have been used to reconstruct high severity fire patches in upper-elevation mixed conifer-aspen forests (Margolis et al. 2007). The technique employs different sampling strategies and includes the collection of (1) regeneration dates, (2) fire scars, (3) death dates, and (4) tree-ring growth changes or injuries. For oak regeneration (sprouting) dates, we installed at least 10 spatially distributed circular age structure plots (0.10 ha). In each plot we sampled at least two of the largest diameter Gambel oak or other resprouting species (e.g., New Mexico locust, Rocky Mountain maple [*Acer glabrum*], and quaking aspen) at ground level with a chainsaw or increment borer. We used a targeted sampling approach to collect fire-scarred trees, stumps, or logs within and on the edges of the shrubfield to obtain a complete inventory of all widespread fires within the site over the maximum time period encompassed by surviving tree-ring material (Farris et al. 2013). For the death date line of evidence, we targeted all potentially fire-killed trees, especially those with well-preserved outside (bark) surfaces. We cored living trees with any of potential fire-related crown damage in and along the edges of shrubfields to assess for tree-ring growth changes or anomalies related to disturbance. We also cored living trees in the shrubfields that were clearly older than the dominant cohort of oak but otherwise lacked fire scars or crown damage. These were mainly used to build site-level ring-width chronologies useful for crossdating the rest of the collection.

We avoided sampling in shrubfields that were created by or burned severely in recent fires, based on the fire atlas for the Santa Fe National Forest that includes our study area (Snyderman and Allen 1997; data updated through 2014 by K. Beeley, Bandelier National Monument). Four of our sites, however, include some amount of recent burning. The August, 1996 Blue Fire, a small lightning-caused fire, burned through the middle of San Pablo Canyon (SPC) causing oak mortality only in a narrow upslope run. This site also partially burned during the May, 1996 Vacas fire, which was human-caused, but we did not sample within its perimeter. The upper portion (above the road) at Señorito South was burned under prescription during the winter of 1991 for wildlife management purposes (J. Wheaton, Cuba Ranger District, Santa Fe National Forest, *pers. comm.*); this fire was not included in the Santa Fe National Forest Fire Atlas. A 122 ha area in the upper watershed area of Mesa Pinabetosa (MPB) burned in the lightning-caused Diegito Fire of July, 2010, causing relatively little mortality to oaks and no observed conifer mortality. Finally, the July, 2013 Thompson Ridge Fire burned through parts of the Redondo Creek site (RDC) but as with others, caused little mortality in the oak, except where it apparently was driven by wind along the upper-most area of the site. Elsewhere at RDC, the lightning-caused, 9,713 ha Thompson Ridge Fire only burned dead wood and killed oak stems immediately adjacent to burning logs.

### **Laboratory procedures**

We employed standard dendrochronological procedures (Speer 2010) to prepare and crossdate tree-ring samples. We determined annually accurate and precise dates for regeneration (pith) dates, fire scars, death dates, and growth changes. All fire scars were

assessed for the intra-ring position (seasonality) of the scar (Baisan and Swetnam 1990). We looked for and dated tree-ring growth changes across all of our samples, not just those trees cored as suspected growth change trees. Evidence for growth changes included visually obvious growth suppression or release, multiple missing rings, and presence of traumatic resin ducts (Arbellay et al. 2014). We assessed whether these events could be climate-related by comparing the growth patterns to nearby climatically sensitive tree-ring chronologies at Cat Mesa, Fenton Lake, and Mesa Alta (Swetnam and Lynch 1993; Guiterman et al. 2016). If major growth anomalies were not correlated with climate variability, we attributed synchronous growth changes among trees (and with other lines of fire evidence) to be a response to partial crown or root mortality from high intensity fire.

### **Analyses**

The four lines of evidence were compiled for each site in order to compare the reconstructed fire history, timing of tree mortality, and growth changes to major pulses in regeneration of oak and other sprouting species. Fire history analyses were carried out in the R statistical computing program (R Core Team 2015) using the *burnr* library (Malevich et al. 2016). We calculated fire interval statistics on composited fire chronologies by filtering for fire events recorded on a minimum of two or  $\geq 10\%$  of fire-scarred trees. Analyses were limited to the historical period 1664-1899 when four of five sites were recording fire.

Shrubfield fire histories were compared to conifer-dominated sites in the Jemez Mountains in terms of fire frequency and the fire-climate relationship. Jemez-wide fire

frequencies were previously published (Touchan et al. 1996; Swetnam et al. 2016), and we used the fire dates for each site in a comparison of site-level fire-climate relationships. By filtering the 60 available Jemez fire history sites (1,297 trees) in the same manner as the shrubfield sites, and limiting the minimum number of widespread fires to  $\geq 10$ , we narrowed the overall sample to 39 Jemez sites in a superposed epoch analysis (SEA; Swetnam 1993). We ran SEA in the *burnr* R package with the function *run\_sea*, which replicates the FORTRAN program EVENT that is employed by FHX2 software (Grissino-Mayer and Grissino 2001). Climate data for the SEA included tree-ring reconstructed summer (June through August) Palmer Drought Severity Index (PDSI; Cook and Krusic 2004: gridpoint 133). We note that while this tree-ring derived PDSI reconstruction is for summer, it primarily captures variability in cool-season moisture (St. George et al. 2010).

## RESULTS

We identified 349 Gambel oak shrubfields in the study area, totaling 10,809 ha (Figure 2). Roughly a third of the shrubfield area is in 21 individual patches are over 100 ha. The median shrubfield size is 16.5 ha and the largest we identified is 367.5 ha. Gambel oak shrubfields in the Jemez Mountains made up 4.75% of the total forest/shrub component of the study area prior to ~2004.

Principle components analysis revealed no differences between the physiographic and climatic space between shrubfields and dry conifer forests across the Jemez study area (Figure 3), which suggests that Gambel oak shrubfields in the study area are related

to disturbance. The first two principle components in our analysis explained 62% of the variance, with PC1 relating strongly to elevation, precipitation, and temperature. Topographic factors including TRMI, TPI, slope, and curvature were important in PC2. Our observations in the field support this result, as we found no obvious landscape factor that distinguishes shrubfields from conifer forests, other than the dominant vegetation. Although the distribution of Gambel oak shrubfields may be influenced by soil type or other edaphic factors, it is known to be present across a broad range of environments, and it is the growth characteristics (e.g., tall, single-stemmed vs short, shrubby stems) rather than presence or absence that often vary between different soils and parent materials (Kaufmann et al. 2016). Furthermore, our analysis included all shrubfields and most of the range of dry conifer forests by including 10,000 random points per vegetation type, which covered the range of edaphic factors in our study area.

To reconstruct the disturbance dynamics and origins of five of the largest shrubfields in the study area, we collected 336 tree-ring samples from 301 trees of 11 species, and successfully crossdated samples from 281 trees (93%) (Figure 2, Table 1). Shrubfield age structures are derived from 158 trees of 8 species sampled in 72 plots (Figure 4). The majority of species sampled in age structure plots were Gambel oak, reflecting the general composition of shrubfield areas. The age structures show a common pattern of pulsed recruitment in the late 1890s, with 1899 being the primary cohort initiation year at all but one site, SNN, where oak established over a seven-year period (1894-1900) without evidence for a cause of initiation (fire, frost, or logging). The age structure of shrubfields includes the oldest and largest ramets from each site, suggesting that Gambel oak shrubfields can remain relatively stable for >115 years, even

in the absence of disturbance. Indeed, we observed little change in shrubfield area and few conifer trees recruiting within shrubfields in comparing aerial imagery from 1935 and 2010 (Figure 5).

The establishment dates of Gambel oak cohorts are generally synchronous with widespread fire events recorded by fire-scarred trees. Additional lines of dendroecological evidence can indicate higher fire severity in many instances (e.g., Margolis et al. 2007; Guiterman et al. 2015), but few Gambel oak cohorts at these sites are synchronous with other evidences. One exception, RDC, shows all four lines of evidence in the year 1899, indicating a fire event that killed and damaged conifers. This fire, however, did not burn at high severity across the site (76 ha), as multiple older oak individuals survived the 1899 fire, and multiple trees, including oaks, were fire-scarred rather than killed. Another instance of synchrony in multiple lines of evidence is at SNS in 1870, when growth changes and fire scars were recorded on four trees, and two trees had 1869 death dates (i.e. mortality prior to the 1870 growing season). There is no oak cohort associated with this event nor are there any older oaks at the site, although one tree sprouted in 1873, after the next widespread fire event. It is plausible that this fire could have caused some conifer mortality and led to the establishment of the shrubfield, but this could not be confirmed from our data.

The tree-ring fire histories of Gambel oak shrubfields show frequent fire prior to fire exclusion beginning around 1899 (Figure 6). The onset of fire exclusion at our sites is consistent with local and regional collapse of fire regimes around the time of Euro-American livestock introduction (Swetnam and Baisan 1996). Median fire intervals ranged from 8 to 15 years prior to 1900 when widespread fires abruptly ceased (Table 2).

We documented fire events at these sites as early as 1250 CE (SNN), with the majority of fire scars dating from 1664 to 1899. Our sampling was sensitive enough to record fires listed in the fire atlas at MPB (2010), RDC (2013), and SNS (1991). The SPC site included two known fires in 1996, but we only recorded a widespread fire in 1989 across the upper portion of the site. Given its proximity to SNS, where the upper area was prescribed burned in 1991, we think this 1989 fire was an undocumented prescribed burn.

The fire-climate relationship is similar across shrubfields, as well as to the 39 dry conifer sites in the study area (Figure 7). Results from SEA show a pattern of generally (sometimes significantly) above average cool-season moisture during 1-3 years prior to the year of fire. The year of fire was significantly dry ( $p < 0.01$ ) at all shrubfield sites and 36 dry conifer sites.

## DISCUSSION

Critical questions facing managers of post-fire landscapes in the Southwest are whether conifers will return to high-severity burn areas, and if so, how long will that take? Or will shrubfields persist in formerly forested areas?

Our results demonstrate that Gambel oak shrubfields can persist in the same biogeographic space as the dry conifer forests they are currently replacing for a century or longer (Figures 3 and 4). A return to conifer dominance would probably take multiple centuries, given modeling results (Strom and Fulé 2007) and aerial imagery showing only isolated conifer recruitment and nearly constant shrubfield sizes since 1935 (Figure 5).

Furthermore, shrubfields historically supported high-frequency fire regimes similar to those in dry conifer forests dominated by ponderosa pine (Figures 6 and 7). Recurring fires support oak persistence through resprouting, and thus fire suppression has been proposed as a potential strategy to restore conifers in areas transitioned by fire (Savage and Mast 2005). At our shrubfield sites, however, Gambel oak persisted despite a total lack of widespread fires for more than a century. Even favorable climate episodes for conifer regeneration, such as the 1910s (Savage et al. 1996) or the late 1970s to early 1980s (Swetnam and Betancourt 1998), did little to change the composition of the dominant species across our sites. This argues against oak shrubfields as a seral stage in pine-oak forests of the Jemez, with or without fire.

The persistence of shrubfields is attributable to competitive advantages over conifers, especially in combination with disturbance and drought. In the presence of various shrub species, the growth and yield of ponderosa pine is considerably reduced due to competition for soil moisture (Shainsky and Radosevich 1986; Oliver 1990). Shrubs such as Gambel oak appear to favor repeated disturbance, and incrementally gain a competitive edge. Concentrated management interventions in Utah and Colorado, for example, have only achieve limited and short-term effects due to the competitiveness of Gambel oak (Kaufmann et al. 2016). Multiple studies have shown that *Quercus* species are more resilient than conifers to intense drought (Choat et al. 2012; Zeppel et al. 2014). In the case of Gambel oak, it is both more resistant to low moisture levels and less susceptible to periodic drought-induced growth decline than ponderosa pine (Kolb and Stone 2000; Adams and Kolb 2005).

Climate projections for the region suggest that Gambel oak is likely to be increasingly competitive. A trend toward warmer, drier conditions over at least the next several decades (Garfin et al. 2013) will make severe drought episodes more frequent and longer lasting (Adams et al. 2009; Ault et al. 2014; Cook et al. 2015). Already high rates of overstory conifer mortality attributable to intense drought and high temperatures in many areas of the interior western US (Breshears et al. 2005; Williams et al. 2013; McDowell et al. 2016) and globally (Allen et al. 2010) portend future widespread die-offs of conifers and the likelihood of rapidly shifting ecotones (Allen and Breshears 1998). Gambel oak across our study area may have experienced reductions in leaf area and some canopy dieback during some intense drought episodes of the last 20 years (including 2002), but these droughts did little to reduce its overall cover or area occupied (observations by the authors).

Future climate is also likely to increase the numbers and sizes of Gambel oak shrubfields if current trends in area burned and especially the proportion of fires burning at moderate to high severity continue (Abatzoglou and Williams 2016). The largest high-severity patch from the 2011 Las Conchas Fire was upwards of 10,000 ha (Allen 2016), more than 27 times the size of our largest mapped and sampled shrubfield, MPB, at 367 ha. The total area dominated by Gambel oak or resprouting shrubs following the Las Conchas Fire remains uncertain, but is probably relatively high. In areas where the Las Conchas Fire reburned fires since 1977, approximately 86% of forested area transitioned to a different vegetative configuration, with the largest proportion (38%) transitioning to oak shrubfield (Coop et al. 2016). Forest losses such as these, in concert with greater drought stress to conifers, and drought-induced forest mortality, will favor Gambel oak in

its present range and likely result in a much greater percentage of the area dominated by shrubfields in the Jemez Mountains and regionally.

How much of this projected trend toward greater oak dominance at the landscape scale is attributable to human activities is not well known. Intentional burning, livestock grazing, and logging would all encourage repeated resprouting following top-kill of oak stems. With each disturbance, 5-7 new ramets regenerate from adventitious buds on intact, and often unharmed, root systems (Neilson and Wullstein 1983; Clarke et al. 2013). This results in slow dispersal as root systems grow (especially in the absence of fire since the late 19<sup>th</sup> century) and are again disturbed. In addition, Gambel oak produces high numbers of acorns, and can disperse by seed over much of its central and southern range (Neilson and Wullstein 1983), especially in the absence of fire (Rogers 1982). Long-distance dispersal of oak is probably slow (Kaufmann et al. 2016), but it is likely that human activities over the last century or longer increased oak densities and the area it occupies, making it much more likely to rapidly colonize following recent high-severity fires.

Although these various factors tend to promote oak over conifers, we found some evidence for conifer reoccupation at one site, SNN. In the eastern portion of the 292 ha shrubfield, there is an 11 ha patch of white fir-dominated forest that regenerated in the 1940s (Figure 5, bottom). Dead oak stems on the forest floor suggest that it was formerly oak dominated. The area occupies a shallow bench on a steep slope and was clearly more mesic than the surrounding slope, presumably because of seeps or springs. Aspen clones along the stand edges are further evidence of the anomalously mesic nature of the site. We hypothesize that these wet conditions, along with the shade tolerance of white fir,

allowed the fir to establish and grow above the oak to form a dense canopy that shaded the moderately shade-tolerant oaks. Wet conditions such as these are rare in the Jemez shrubfield sites, and this is the only area of its kind that we found. Furthermore, it is likely that if the site had burned since 1899, the fire-intolerant fir would have been killed, and the re-sprouting oaks and aspen would have maintained dominance over the conifers.

### **Potential for fire-origin of pre-1900 shrubfields**

Given multiple observations of forest-to-shrub transition following high-severity fires across interior Western North American forests, we hypothesized that historic high-severity fires were the origins of extant shrubfields in the Jemez Mountains. By contrast, we found that Gambel oak shrubfields predominantly maintained a frequent, low-severity fire regime until frequent, widespread burning ceased in 1899 at nearly every site. The dominant age structure of Gambel oak across our sites more directly represents the last fire, rather than a type-conversion event.

We found only two incidences of potentially tree-killing fire that promoted oak cohorts. These occurred within small areas of the shrubfields and did not match the larger-scale tree-ring fingerprint of high-severity fires in pine-oak that we expected to find prior to conducting the study (e.g., Guiterman et al. 2015). We cannot, therefore, assert that large, pre-1900 shrubfields represent patches of large, high-severity fire during the historic period.

These findings lend support to the long-held idea that fires during the historic period in pine-oak and dry conifer ecosystems were predominantly low-severity surface fires, with isolated patches of high severity up to approximately 100 ha (Swetnam et al.

2001; Iniguez et al. 2009; Yocom-Kent et al. 2015; Bigio et al. 2016). However, it is plausible that shrubfields originated earlier than our tree-ring records can confidently distinguish. The general lack of fire-killed trees dating prior to oak cohort establishment could be due to loss of datable tree-ring samples in subsequent fires, as may be the case for the 1870 fire at SNS. In contrast to mixed conifer-aspen stands (Margolis et al. 2007) or topographically isolated pine-dominated stands (Iniguez et al. 2009) where subsequent fires generally did not occur since the high-severity event, the frequent burning of Jemez shrubfields would have consumed the evidence obtained from fire-killed trees.

For high-severity fire to have burned through one of our shrubfield sites, with fire events every 8 to 10 years on average, a substantial hiatus period in the fire history would be required to allow for the build-up of fuels necessary to carry a stand-replacing crown fire. Indeed, at all sites we found a nearly complete lack of fire from ca. 1818 to 1842, which marks a transition from short fire intervals through the 18<sup>th</sup> and early 19<sup>th</sup> centuries to longer intervals with more widespread burning in the late 19<sup>th</sup> century. A similar change in fire occurrence – from highly frequent small fires to more widespread burning at slightly longer intervals – has been noted in the Jemez Mountains (Touchan et al. 1996; Swetnam et al. 2016), elsewhere in the Southwest (Swetnam and Dieterich 1983; Grissino-Mayer and Swetnam 1997), and northern Mexico (Stephens et al. 2003; Fulé et al. 2012) related to a change in El Niño-Southern Oscillation event frequency or other very broad-scale climate fluctuations (Kitzberger et al. 2001). It remains uncertain, however, whether this 24-year period in the mid-1800s in the Jemez is long enough to promote crown fire. Across the many other areas that experienced a similar gap, crown fire was not evident in subsequent fire events.

Other potential disturbances leading to the dominance of Gambel oak in the historic era could relate to long drought episodes known for widespread tree mortality (e.g., the 1580s; Swetnam and Betancourt 1998) or periods with high human population densities (e.g., Ancestral Puebloans; Liebmann et al. 2016; Swetnam et al. 2016). Further research that utilizes different proxies that have longer temporal reach than tree rings, such as alluvial sediments (e.g., Fitch and Meyer 2014), may provide insights into the timing of initial Gambel oak dominance.

### **Fire histories of Gambel oak shrubfields and dry conifer forests**

Our comparisons of shrubfield fire regimes to those of dry conifer forests in the Jemez Mountains revealed that shrubfields burned historically at the same general rates (Touchan et al. 1996; Swetnam et al. 2016) under similar climate patterns (Figure 7). This is in contrast to studies indicating that shrub-dominated sites have a lower fire frequency and higher severity fire effects than surrounding conifer stands (e.g., Airey Lauvaux et al. 2016). Relatively wet, antecedent climate conditions associated with fire suggest that fine-fuels (grass) were important in supporting fire regimes in both vegetation types. Today, grass is still somewhat common in the understories of shrubfields, especially at lower topographic positions where the oak overstory is the tallest or where densities are relatively low. It is likely, therefore, that historic-era shrubfields included wider-spaced genets with a higher proportion of grass than found today; conditions that would have been conducive to burning frequently at relatively low severity.

As shrubfields have aged, the fuel composition has grown heavier and denser, reducing flammability, except under extreme conditions. Under low to moderate conditions, shrubfields can act to inhibit fire spread (Kaufmann et al. 2016), which is what we observed as the effects of fires in 1996, 2010, and 2013 that only burned isolated areas and caused limited mortality to oaks. Under extreme climate conditions, on the other hand, oak shrubfields can burn at high intensities and be difficult to suppress (Kaufmann et al. 2016). This is illustrated by the high-severity effects of the Las Conchas Fire as it burned through and shrubfields originating from the 1996 Dome and the 2000 Cerro Grande Fires (Coop et al. 2016). The Las Conchas Fire was highly energetic, wind-driven, and occurred during exceptional drought (Williams et al. 2014). The young age and probable low density of the recent fire-origin shrubfields would have also done little to inhibit fire spread. These observations, in accord with our reconstructed fire histories, indicate that 10-15 year-old shrubfields are among the most susceptible to reburning. A fire at this point in stand develop would be especially detrimental to seedling or sapling stage conifers and act to further reinforce shrubfield persistence (Savage and Mast 2005; Savage et al. 2013; Coop et al. 2016).

## **CONCLUSIONS**

Gambel oak is an important component in the landscape heterogeneity in mid-elevation forested ecosystems of the western US. Under projected increases in forest drought stress and associated high-severity disturbances, however, the number and sizes of fire-origin shrubfields are likely to grow. Gambel oak shrubfields in the near future may be much more common. For instance, recent fires in the study area have combined

to create patches of near total conifer mortality and likely future oak dominance that are over 27 times larger than the largest pre-2000 shrubfield we identified. Furthermore, established Gambel oak shrubfields can persist as an alternative stable state for a century or longer. They supported historically frequent surface fire regimes that match those of ponderosa pine-dominated forests, which are the most fire-prone forests of the region. These results demonstrate an affinity for fire in Gambel oak shrubfields but also long-term stability in the absence of disturbance.

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## TABLES

**Table 1.** Shrubfield site information and dendroecological data

<b>Shrubfield site collections</b>					<b>Lines of evidence for high-severity fire</b>			
<b>Name</b>	<b>Code</b>	<b>Shrubfield area (ha)</b>	<b>Mean elevation (m)</b>	<b>No. trees</b>	<b>Age structure dates</b>	<b>Fire scars</b>	<b>Death dates</b>	<b>Growth changes</b>
Mesa Pinabetosa	MPB	340	2570	35	21	51	8	4
Redondo Creek	RDC	76	2660	59	31	61	11	26
Señorito North	SNN	292	2669	69	33	120	17	0
Señorito South	SNS	243	2553	57	32	89	9	11
San Pablo Canyon	SPC	191	2689	61	41	67	12	10
<b>Totals:</b>				<b>281</b>	<b>158</b>	<b>388</b>	<b>57</b>	<b>51</b>

**Table 2.** Fire interval statistics for the historic period.

Site	Analysis period	No. intervals	Mean	Median	WMPI	Min	Max
MPB	1651 - 1896	12	14.8	12.5	13.6	4	29
RDC	1664 - 1899	15	15.7	13.0	14.3	5	41
SNN	1664 - 1899	23	10.2	9.0	9.2	1	24
SNS	1664 - 1899	19	12.4	10.0	10.1	2	47
SPC	1714 - 1899	13	11.8	10.0	11.1	2	25

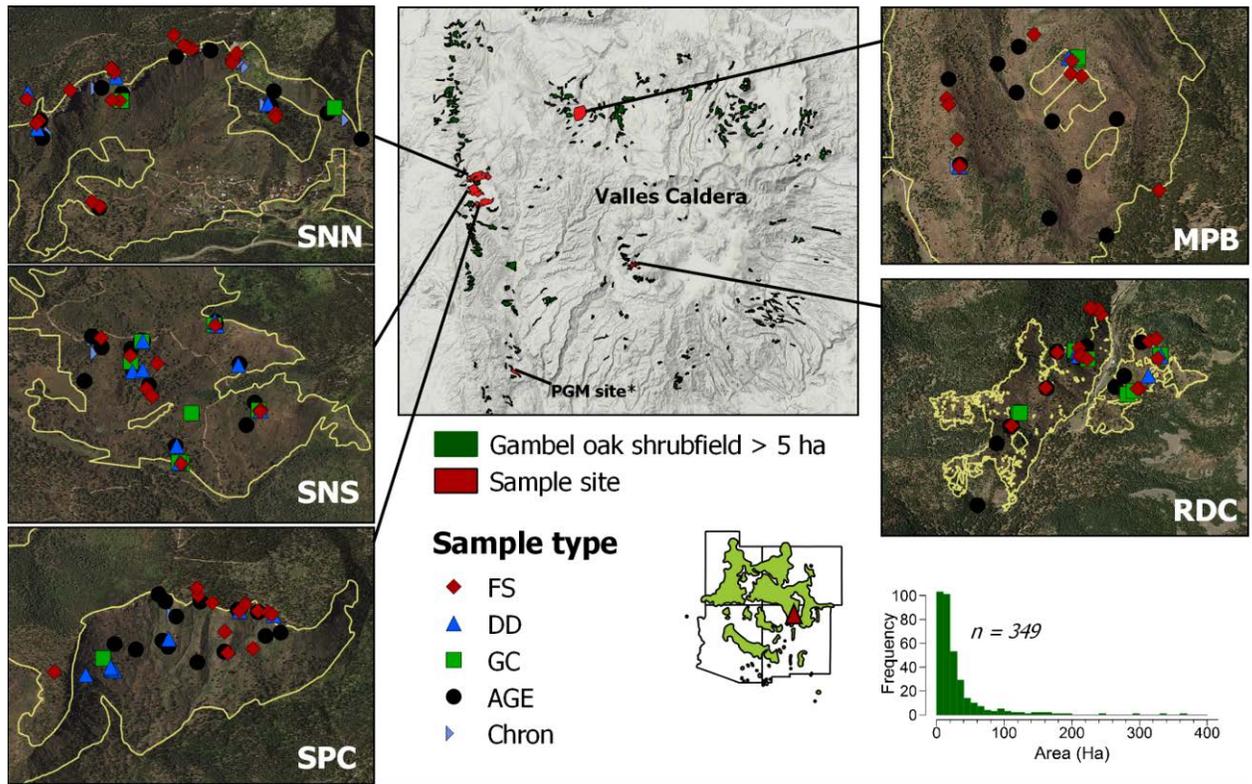
*Note:* WMPI = Weibull median probability interval.

Sites are filtered for 10% scarred and a minimum of 2 trees scarred.

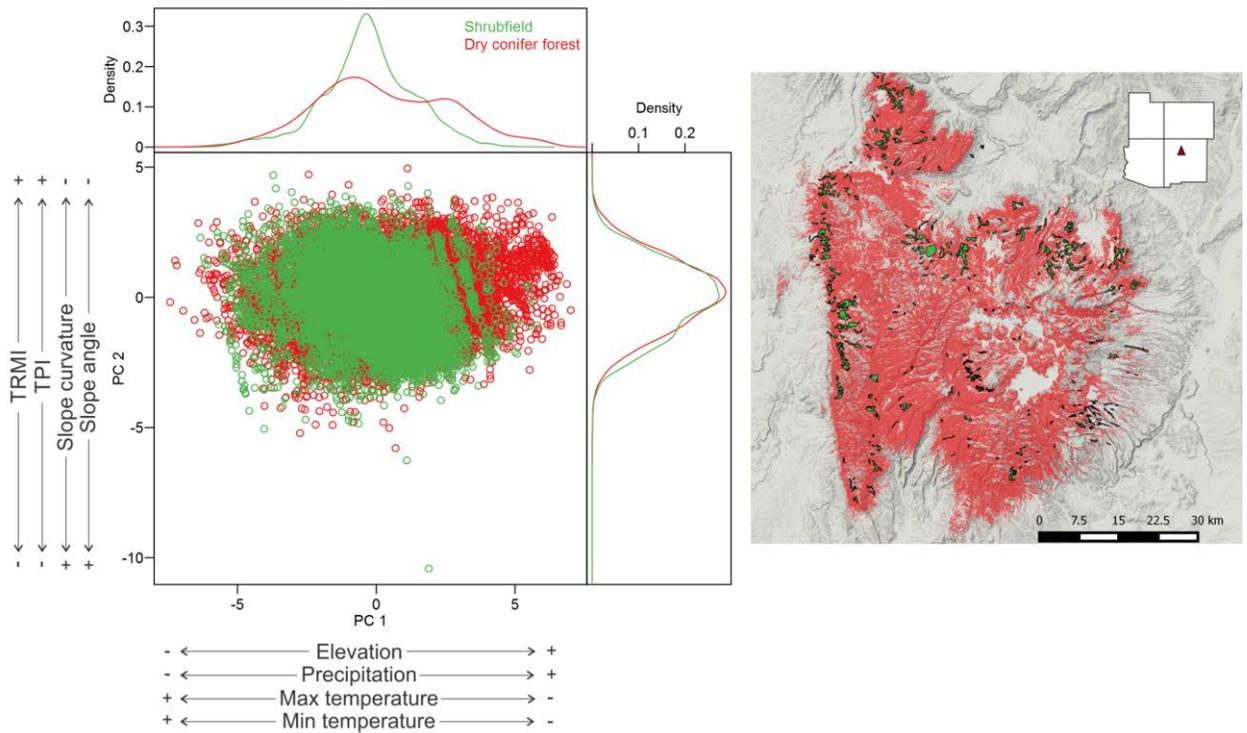
## FIGURES



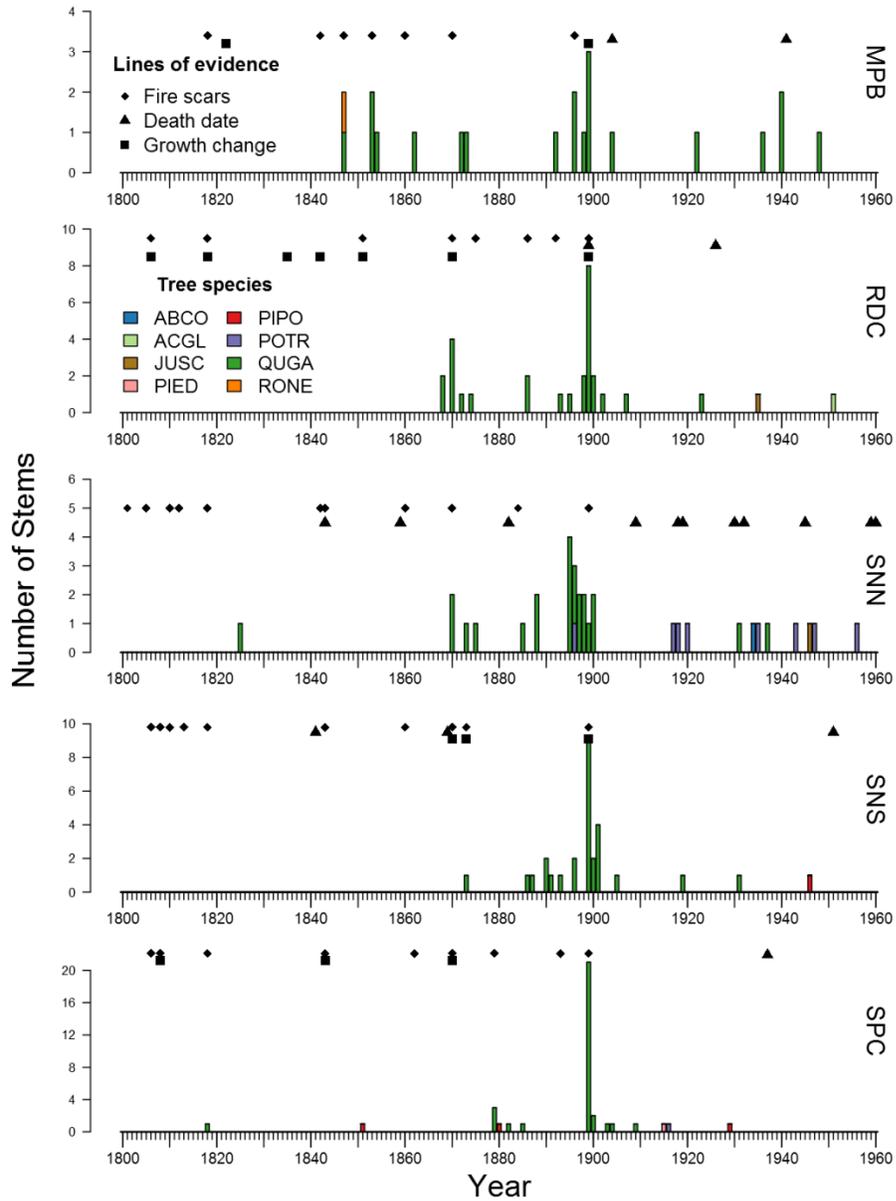
**Figure 1.** Gambel oak shrubfields in New Mexico. Images at top are recent fire-origin shrubfields while those at bottom are extant, potentially fire-origin shrubfields in the Jemez Mountains. *Top left*, Sugarite State Park: 2011 Track Fire; *top right*, Mescalero Apache Reservation, Sacramento Mountains: 1996 Chino Well Fire; *bottom left*, Mesa Pinabetosa site (MPB), northern Jemez Mountains; *bottom right*, San Pablo Canyon site (SPC), western Jemez Mountains. Bottom-right photo by C. Guiterman, all others by E. Margolis.



**Figure 2.** Tree-ring collections in Gambel oak shrubfields of the Jemez Mountains. Site maps have equal scales covering 2.6 km across; site names are provided in Table 1. Notations for sample types are FS, fire scar; DD, death date; GC, growth change; AGE, age structure plot; and Chron, chronology tree. The center-top map shows topographic relief of the study area with all Gambel oak shrubfields > 5 ha; sampled sites are shown in red. Shrubfield polygons represent ca. 2004 data that do not include much larger fire-origin shrubfield areas dating to 2000-2011. The southernmost sampling site, Peggy Mesa (PGM), is described in Guiterman et al. (2015). Insets include a regional map of the range of Gambel oak in green (Little 1971) with the location of the Jemez Mountains (red triangle) and the size distribution of Gambel oak shrubfields.



**Figure 3.** Variable reduction comparison of the physiographic and climate space of Gambel oak shrubfields and forests dominated by ponderosa pine and dry mixed conifers in the Jemez Mountains. At left, principle components reveal broadly similar settings for shrubfields and forests ( $n$  points = 10,000 each), with shrubfields occupying the lower 2/3 of elevation and climate space (PC1). There is no difference between shrubfields and forests by topographic position or physiography (PC2). At right, the map shows the distribution of ponderosa pine and dry mixed conifer forests (red) and the shrubfields >5 ha (green) in the study area.



**Figure 4.** Age structure of Gambel oak shrubfields with additional lines of evidence that indicate fire. Species include ABCO, white fir; ACGL, Rocky Mountain maple; JUSC, Rocky Mountain juniper; PIED, piñon; PIPO, ponderosa pine; POTR, quaking aspen; QUGA, Gambel oak; and RONE, New Mexico locust. Site abbreviations are defined in Table 1.

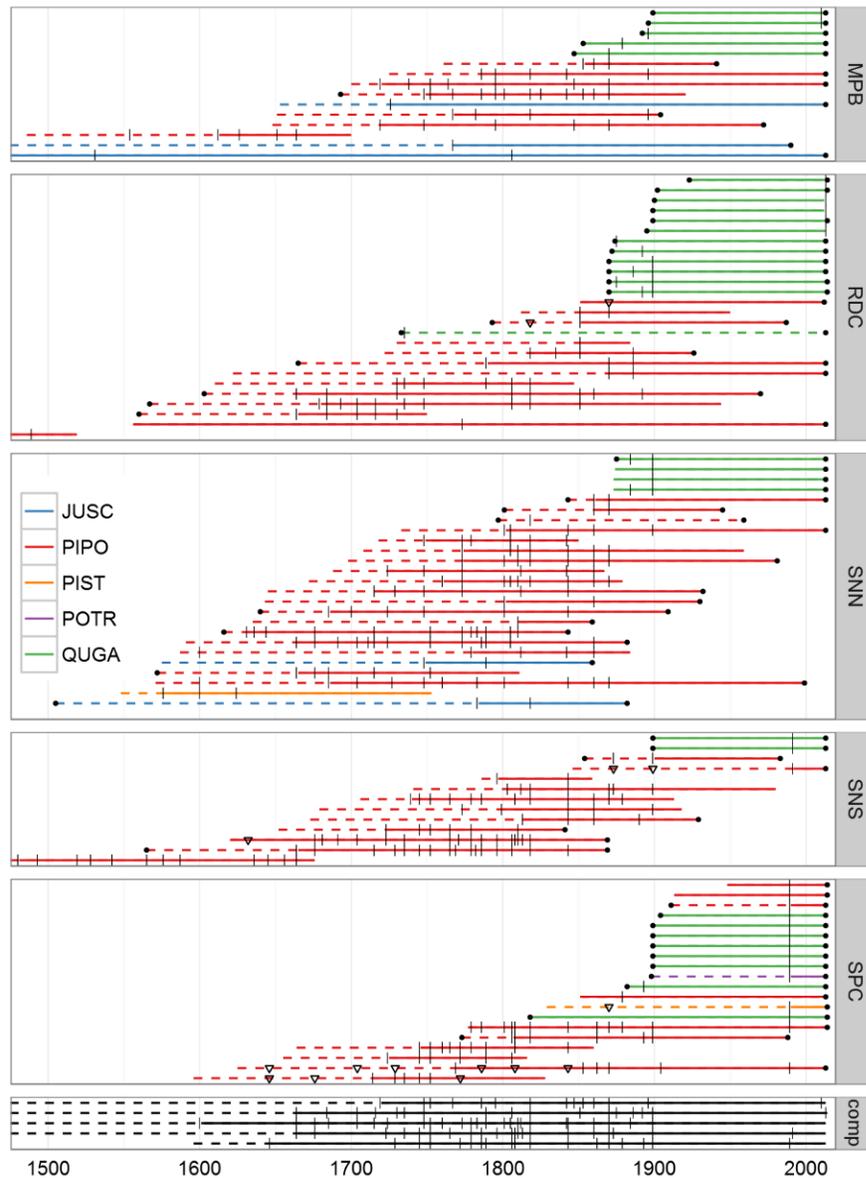
### San Pablo Canyon (SPC)



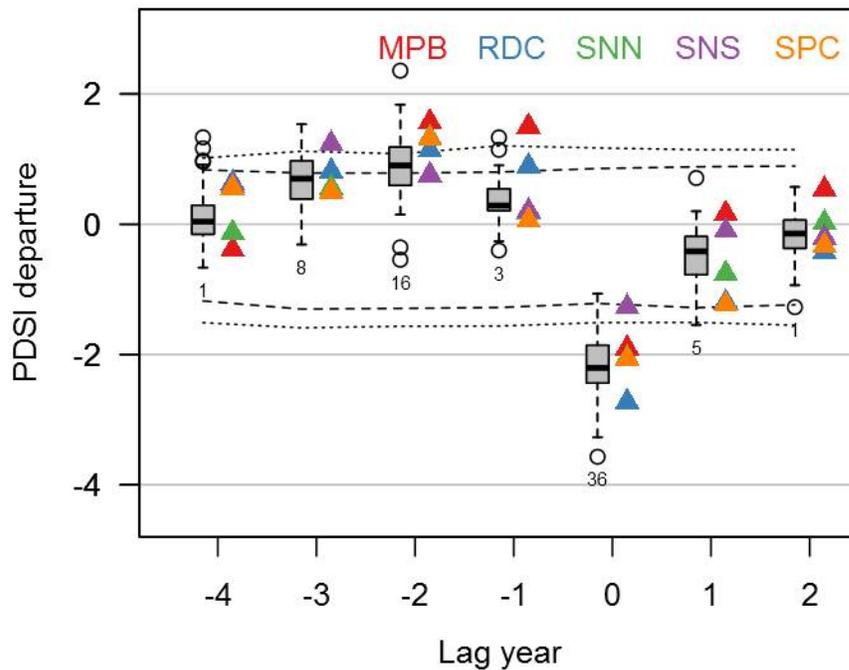
### Senorito North (SNN)



**Figure 5.** Aerial photo comparisons for two shrubfield sites with the most conifer recruitment since 1935.



**Figure 6.** Fire histories of Gambel oak shrubfields. Each colored horizontal line represents the sampled life span of an individual tree ( $n = 100$ ), with pith and bark years indicated by dots. Solid line segments indicate that the tree is recording fires, while dashed segments indicate that a tree is not recording. Vertical tick marks show fire scars ( $n = 388$ ) and triangles show growth changes on fire scar trees. At bottom are composite fire histories for each site, ordered as above, after filtering fire scars and growth changes for a minimum of two trees scarred and 10% scarred/injured. Species include JUSC, Rocky Mountain juniper; PIPO, ponderosa pine; PIST, Southwestern white pine; POTR, quaking aspen; and QUGA, Gambel oak. Site abbreviations are defined in Table 1.



**Figure 7.** Comparative fire-climate relationships for dry conifer sites (box-and-whisker plots) in the Jemez Mountains ( $n = 39$  sites) and Gambel oak shrubfield sites (triangles), based on superposed epoch analysis. The lag year zero is the year of fire. Numbers below the box-and-whisker plots indicate the number of sites with a mean PDSI departure beyond the 95% bootstrapped confidence limit for that lag year. To show the approximate bootstrapped confidence limits, we ran 1000 iterations of 19 random event years; the dashed and dotted lines show the 95% and 99% limits, respectively. We found little difference in bootstraps CIs based on 14-35 random events. There are 14 fire events included at MPB, 17 at RDC, 25 at SNN, 21 at SNS, and 18 at SPC.

## APPENDIX D

### RECENT SYNCHRONIZED TREE GROWTH RESPONSE TO CLIMATE ACROSS A LARGE SEMIARID FOREST IN THE SOUTHWESTERN US

Recent synchronized tree growth response to climate across a large semiarid forest in the southwestern US

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**Guiterman, C.H., Margolis, E.Q., Woodhouse, C.A., Williams, A.P., Falk, D.A., and Swetnam T.W. Recent synchronized tree growth response to climate across a large semiarid forest in the southwestern US**

**ABSTRACT**

Anthropogenic warming has amplified forest drought stress and disturbance processes in semiarid forests across the Southwest USA (SW). Much of what we know about forest response to climate, however, is derived from tree-ring sites targeted for drought sensitivity, which raises the possibility that the strength of climate-growth relationships is exacerbated. We tested this concern across a large (250,000 ha) ponderosa pine (PIPO) and dry mixed conifer forest in the northern SW, focusing on PIPO, Douglas-fir (PSME) and piñon (PIED). We compared correlations of the regional forest drought stress index (FDSI) to tree-ring chronologies derived from both targeted “dendroclimate” sites and non-selectively sampled sites from a network of forest inventory plots. Our plot selection was based on physiographic factors in order to accurately represent the forest area. We found that correlations across all dendroclimate ( $n = 14$ ) and non-selective ( $n = 57$ ) chronologies declined with increased elevation, owing to higher soil moisture and lower vapor pressure deficit (VPD). Dendroclimate sites consistently correlated higher than non-selective plots, which substantiates some of the concern over geographic bias. However, FDSI showed strong coherence ( $r > 0.56$ ) with half of our plots, representing the forest area below  $\sim 2500$  m. At the highest elevations (above  $\sim 2600$  m, 22% of the study area), FDSI did poorly at explaining growth variability ( $r < 0.46$ ). This pattern has changed since  $\sim 1997$ , as VPD has remained above the 20<sup>th</sup> century average, resulting in marked increases in growth variability and correlation to FDSI. Drought periods, in particular, synchronized tree growth across our study area, and the region. More frequent and longer lasting droughts in the near future are likely to exert significant changes to lower elevation forests, but higher elevations are more likely to lag in their response, as they have in our study area until recent decades.

## INTRODUCTION

Climate change appears to be profoundly affecting forest ecosystems, with drought-related tree mortality documented globally (Allen et al. 2010, 2015) and mortality rates on the rise in the western US (van Mantgem et al. 2009; McDowell et al. 2016). In recent decades, roughly 20% of semiarid conifer forests in the southwestern US (SW) have been impacted by drought-related mortality, bark beetle outbreaks, and high-severity wildfires associated with anthropogenic warming (updated from Williams et al. 2010). Regional climate projections suggest that average growing season temperatures will continue to rise in the coming decades (Seager et al. 2007, 2012; Garfin et al. 2013), which would amplify drought stress on plants via an exponential relationship with atmospheric moisture demand (i.e. vapor pressure deficit [VPD]; Weiss et al. 2009; Breshears et al. 2013). By ca. 2050, projected VPD in the SW could drive the average level of forest drought stress to levels that exceed the worst years of the worst drought over the last millennium (Williams et al. 2013). Evidence in the tree-ring record indicates that such extreme and long-lasting droughts led to regional tree die-off events (Swetnam and Betancourt 1998), suggesting that the scale of drought-related impacts may continue to rise (Allen et al. 2015; McDowell et al. 2016).

Forest loss at the regional level could have detrimental effects on society, as ecosystem services may be lost or change in nature and reliance (Garfin et al. 2013). Even small-scale impacts (e.g., severe fire in a watershed) can have large and long-lasting impacts on local communities. Because of generally dispersed rural settlement and reliance on local natural resources, many Native Americans are disproportionately

vulnerable to climate change effects such as these, which are already being observed on the reservations of the SW (Ferguson et al. 2011; Lynn et al. 2013; Voggeser et al. 2013; Cozzetto et al. 2013).

Critical knowledge gaps exist that hamper predictions of how and where forest drought stress will be manifested at subregional to landscape scales. Models of growth niches and projected distributions are promising (e.g., Rehfeldt et al. 2006, 2014a, 2014b), but would benefit from further truthing efforts and a more refined understanding of forest productivity and vulnerabilities. Such an effort requires intensive sampling across ecotones in a representative fashion to capture heterogeneity in climate-tree growth relationships at fine spatial scales. There is a pressing need for such an effort in order to inform and improve ecosystem models and refine our projections of the effects of climate change, as well as to direct forest management toward those areas where action (or conservation) might help to promote forest resilience to climate change.

These uncertainties stem in part from a general lack of tree-ring data representatively sampled across the diversity of SW forests. Many studies utilize the extensive network of tree-ring collections from the International Tree-Ring Databank (ITRDB)<sup>3</sup>. ITRDB “dendroclimate” chronologies most often include trees specifically targeted to capture a climate signal via selection of sites with conditions that promote drought-sensitivity by limiting moisture for tree growth. Such conditions include shallow rocky soil, steep slopes, high insolation levels, low tree density, old age of trees, and few obvious signs of disturbance (fire, logging, roads) (Fritts 1976; Speer 2010). Similarly,

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<sup>3</sup> Hosted by the National Centers for Environmental Information: <http://www.ncdc.noaa.gov/data-access/paleoclimatology-data/datasets/tree-ring>

some dendroclimate sites may be located at the lower ecotone of a particular species, where soil moisture levels are marginal for growth (Fritts et al. 1965). It is at these lower ecotones and at particularly drought-prone sites where rapid, drought-induced mortality is likely to concentrate (Allen and Breshears 1998; Kelly and Goulden 2008).

Furthermore, assessments of climate effects on forest growth at the regional scale (e.g., Williams et al. 2013) or the continental scale (e.g., Williams et al. 2010; Charney et al. 2016) combine networks of chronologies, which further enhances the climate signal because synoptic-scale climate is the only feasible commonality among large sets of drought-sensitive chronologies. Thus, the selection and amalgamation of targeted drought-sensitive tree-ring data for regional assessments of forest health may introduce a systemic bias and may not represent the potential climate response in other parts of species ranges.

Few studies have directly tested this concern with independent tree-ring chronologies from selectively and non-selectively sampled sites and trees. DeRose et al. (2013) found generally good correlations between tree growth on Forest Inventory and Analysis plots across Utah and local dendroclimate chronologies when the data were smoothed to decadal time scales. This suggests that forests and dendroclimate sites respond similarly to long-term climate fluctuations, but forests may be less responsive to short-term events such as drought. Restaino et al. (2016) selected Douglas-fir (*Pseudotsuga menziesii*) sites across its range in the western US specifically to quantify climate-growth relationships. They found similar patterns in drought-sensitivity and projected tree-growth declines due to warming as studies that only used dendroclimate sites (Williams et al. 2010, 2013; Charney et al. 2016).

A growing body of research indicates substantial within stand variability in climate response of tree growth that may further differentiate forest stands from dendroclimate sites, where tree selection standards intentionally mitigate non-climate influences (Speer 2010; Nehrbass-Ahles et al. 2014). Tree age may be an important factor in determining climate sensitivity and vulnerability to climate change, but few studies have tested these effects in the SW region. Studies in Europe have generally concluded that age is not an important factor for most species (Carrer and Urbinati 2004; Esper et al. 2008; Dorado Liñán et al. 2011), but climate sensitivity can increase with age in some species, especially during climatically stressful periods (Linderholm and Linderholm 2004; Carrer and Urbinati 2004). Working with ponderosa pine (*Pinus ponderosa*; a common species in the SW) in the Pacific Northwest, Lloret et al. (2011) found that young trees (< 200 years) could have lower drought-induced growth declines during 1-3 year droughts but may not recover to pre-drought growth rates as well as older trees (> 245 years). Stand structure may also obscure overall patterns, as canopy position influences the light environment and amount of available resources for trees, and thus generates differing responses to climate (Carnwath et al. 2012), except during droughts which synchronized growth responses (Carnwath et al. 2016). Different species often respond to climate differently (Carnwath et al. 2012; Dolanc et al. 2013), owing to life history traits that aid in reducing drought stress or the general site conditions and microclimates under which the trees typically grow. Site conditions can play an important role in that trees growing on mesic or cooler sites show reduced susceptibility to recent warming-induced drought effects compared to the same species in dry and warm sites (Williams et al. 2011; Liu et al. 2013). This difference is reflected in overall lower rates

of tree mortality in wetter eastern Canada versus the generally more arid western Canada (Peng et al. 2011), and even increasing growth rates at high latitude sites (Graven et al. 2013).

In this study we evaluate the representativeness of a regional climate variable to tree growth across a non-selectively sampled landscape and at dendroclimate sites within the study area. Our study area covers ~250,000 ha of semiarid ponderosa pine and dry mixed conifer forests on the Navajo Nation along the northern Arizona-New Mexico border. We include three target species, piñon (PIED, *Pinus edulis*), ponderosa pine (PIPO), and Douglas-fir (PSME). We used the forest drought stress index (FDSI; Williams et al. 2013) derived from gridded climate data to represent regional climate variability. FDSI is analogous to the climate-driven portion of tree growth, as it was fine-tuned ( $r = 0.91$ ) to the coherence of 335 SW dendroclimate chronologies. It is a combination of cool-season precipitation with cool- and warm-season VPD. We are particularly interested in whether projected FDSI portends future forest mortality across our study area as it might across the SW region. Our research questions include:

1. Do targeted dendroclimate sites have higher correlations with FDSI than non-selectively sampled sites? We expect that they do, as site and tree selection at dendroclimate sites are specifically orientated to better capture climate signal.
2. Does soil moisture – as influenced by topography and elevation – reduce correlations between FDSI and forest growth? We expect that it does, based on previous work showing reduced climate sensitivity away from forest ecotones (Fritts et al. 1965).
3. Does the relationship between FDSI and forest growth vary through time? Here, we are specifically testing whether increased temperatures over the last two decades

have strengthened the climate-tree growth relationship. Studies suggest that this may be the case, as drought periods tend to synchronize tree growth (Carnwath et al. 2016).

## METHODS

### Study Area

Our study area is situated along the border between northern Arizona and northern New Mexico, in the Chuska Mountains and Defiance Plateau of the Navajo Nation (Figure 1). This area of ~250,000 ha is the primary forest management landscape for the Navajo Forestry Department and contains some the most vital watersheds for spring-fed and perennial sources of drinking water on the Navajo Nation (NNDWR 2011).

The Chuska Mountains are composed of Oligocene sandstone along a narrow crest at ~2700 m (Wright 1956; Cather et al. 2008) that includes several large volcanic intrusions (Appledorn and Wright 1957). The Defiance Plateau is lower in elevation (~2300 m) with predominantly sandstone-derived soils across relatively gentle terrain. Deep canyons incise the northern and southern areas of the plateau, including the ~300 m deep Canyon de Chelly.

Forests in the Chuska Mountains are predominantly ponderosa pine (PIPO) with dry mixed conifer (as defined by Romme et al. 2009) in more mesic and higher elevation areas. The dry mixed conifer forests are composed of PSME, Gambel oak (*Quercus gambelii*), aspen (*Populus tremuloides*), and PIPO. Forests on the Defiance Plateau are nearly pure PIPO, with understory Gambel oak. Across both landscapes, PIED is present along the lower ecotone (~2000 m) between PIPO and piñon-juniper (*Juniperus* spp.) woodlands.

The climate of the study area is continental and semiarid. The precipitation regime is dominated by summer convective storms (rain) associated with the North American Monsoon, with highly variable winter precipitation (snow) (Sheppard et al. 2002). Mean annual precipitation is 332 mm (1900-2010; based on the Precipitation-Elevation Regression on Independent Slopes Model (PRISM) [Daly et al. 2008] averaged for the study area). Average minimum temperatures range from  $-8.6^{\circ}$  C in January to  $10.2^{\circ}$  C in August, and average maximum temperatures range from  $4.9^{\circ}$  C in January to  $27.0^{\circ}$  C in July.

### **Site selection**

Sampling was conducted on Continuous Forest Inventory (CFI) plots maintained by the Navajo Forestry Department and Bureau of Indian Affairs division of forestry. The CFI plots are located on a systematic grid of 272 plots in the ponderosa pine and dry mixed conifer forests of the study area. The CFI grid consists of squares with 1.6 km sides and a plot every 2.26 km diagonally (northeast) (BIA Division of Forestry 1995). The plots were established in 1974, with remeasurements and some new installations in 1980, 1989, and 2004. Each plot contains three 0.1 ha subplots situated in a triangle with 36 m sides. At each inventory, all trees  $\geq 12.7$  cm dbh (1.37 m above ground) on each subplot are tagged and numbered, and then tallied for condition (live, dead, injuries, and defects) and measured for diameter at breast height (dbh). Ingrowth (new trees reaching 12.7 cm dbh) was tallied on the northeast quarter of the subplots. All trees classified as “woodland species” (piñon, juniper, and oak) with diameter at the root crown (drc)  $\geq 7.62$  cm are tallied in the northeast quad of the first subplot.

To obtain a representative subsample of the CFI grid, we used the 2004 inventory data to determine the number of PIPO, PSME, and PIED trees above diameter limits that roughly equate to tree ages of at least 70-100 years, based on a pilot sample of 111 trees from seven CFI plots we conducted in 2014. The limits were 30 cm dbh for PIPO and PSME and 10 cm drc for PIED. To ensure adequate sample depth for tree-ring chronology development, we then filtered the inventory for plots with at least five PSME or PIED trees and at least 10 PIPO trees; we included more PIPO because of their relative abundance. These estimated lower sampling limits were based on assessment of tree-ring chronology signal strength (Cook and Pederson 2011) derived from random sub-sampling of our 2014 CFI tree-ring chronologies. Filtering by these two criteria reduced the number of available plots to 167, which includes the removal of six plots that burned in the 2014 Asaayi Lake Fire in the central Chuska Mountains.

We then graphed the 167 "selection plots" in the physiographic space of elevation as a function of topographic relative moisture index (TRMI; Parker 1982) (Figure 2). For a balanced sample across these variables, we divided elevation and TRMI into their inner-quartile ranges. Since only 10 PIED plots were available, we selected all of them for sampling. PSME plots were primarily above the upper quartile of elevation, so we randomly selected five PSME plots in each of the lowest and highest quartiles of TRMI for a maximum contrast. PIPO plots were the most broadly distributed across the elevation-TRMI space. We randomly selected PIPO plots along a moisture gradient from low elevation-xeric sites to mid elevation-mesic sites, making our selection consist of five plots in the lowest and middle quartiles zones of elevation and TRMI, and five plots the zone of upper-quartile TRMI and middle quartile elevation. This selection procedure

identified 35 plots, of which we sampled 30 in 2015 due to access issues and time constraints. We combined the 30 plots sampled in 2015 with six of our 2014 sample (one 2014 plot was resampled in 2015) for a broad and representative distribution of 36 sample plots in the study area (Figure 1).

In addition to CFI plots, we include 14 species-specific chronologies from 12 dendroclimate sites (Figure 1; Table 1). Four of the chronologies are original to this study or the authors, one was updated for this study, seven are archived chronologies from the ITRDB, and one was archived at the Laboratory of Tree-Ring Research (Dean and Funkhouser 1995). All but our original chronologies are included in the Williams et al. (2013) FDSI.

### **Field sampling**

During the summer of 2015, we visited 30 plots to tally all previously inventoried trees for status (live, dead, harvested) and dbh across each subplot. We then selected the largest 5-10 live trees to be sampled with an increment borer, in which we cored each tree twice perpendicular to slope at 20-50 cm above ground level. Field data for the tree-ring samples, including location, photographs, social status, sampling heights, diameters at sampling height and breast height, and comments, were collected on a tablet-based system integrated with the Tellervo tree-ring database (Brewer and Guiterman 2016). To constrain the climate-tree growth relationship at each plot, we excluded trees in clearly different topographic positions, such as on a different aspect than the rest of the sample trees. While we visited plots to build the chronology for one specific target species based on our plot selection, we also cored other target species when enough were present to

complete a chronology. Therefore, we were able to obtain a PIPO collection at every PSME plot and at all but three PIED plots. Two plots contained all three species. In some cases, there were too few tally trees in the inventory to complete a collection, either because trees were not tallied across subplots (e.g., PIED) or injuries, rot, or mortality negated sampling of individuals that we intended to sample. In these cases, we added representative trees if they were just outside of subplot boundaries or centered between subplots; for PIED, we often found representative trees on subplots 2 and 3 that were excluded from the inventory.

In our 2014 pilot study, we visited seven plots to tally inventoried trees for status and dbh. Unlike in 2015, we only measured and sampled one subplot, choosing the one with the middle number of trees. We then cored the 10 largest trees and 5-10 trees across the lower diameter classes, regardless of species (only PIPO and PSME were present on these plots).

Sampling at dendroclimatic sites was done according to standards (Fritts 1976; Speer 2010) whereby we targeted sites with conditions that generally promote climatic sensitivity of tree growth as described above. New sites were selected based on the potential for a long chronology, and we sampled remnant materials to improve sample depth in earlier time periods and extend the chronology back in time. We aimed for ~20 live trees and 15-20 remnant trees at each site. At previously sampled sites, our efforts were focused on updating the chronology to present by re-coring older trees and sampling a selection of younger trees, aiming for  $\geq 10$  live trees of each species with good growth form (straight boles) and free of injury or rot.

## **Chronology development**

Followed standards of dendrochronology (Speer 2010), tree cores were mounted, sanded to a fine polish, visually crossdated, and then measured on a Velmex sliding stage (0.001 mm precision) integrated with the Tellervo database (Brewer 2014). Quality of measurements and dating was verified visually in Tellervo and by assessing all tree-ring series for each plot or site in COFECHA software (Grissino-Mayer 2001).

We developed separate tree-ring chronologies for each species at each CFI plot or dendroclimate site. Ring-width series were standardized by division against cubic smoothing splines with a 0.5 frequency response and a wavelength of 50 years (Cook and Peters 1981). The spline length was decided upon after checking curve fits of different wavelengths for every series, and this commonly used approach fit well across our range of site conditions and species. The spline failed (trended negative) on nine series, and we used a simple straight line through the series mean for standardization. Series from dendroclimate sites were standardized in the same way but with 100-year cubic smoothing splines; for data obtained from the ITRDB, we used the residual chronologies provided by the original investigator. For CFI plots and our dendroclimate sites, we produced residual chronologies by modeling and removing the autocorrelation among standardized series prior to applying Tukey's biweight robust mean to average series together (Cook et al. 1990). Standardization, chronology development, and assessment were carried out in the R statistical computing environment using the *dplR* library (Bunn 2008; R Core Team 2015).

## Analyses

We tested for climate sensitivity among our species-level CFI and dendroclimate chronologies by correlating them to the forest drought stress index (FDSI; Williams et al. 2013). We used the 1896-2013 FDSI derived from instrumental climate data, which is a combination of prior November to current March precipitation (PPT) and the average of prior August-October VPD and current May-July VPD derived from PRISM climate data (Daly et al. 2008) spanning the Southwest region. Variability in the correlations between our chronologies and FDSI was evaluated through linear regression against elevation, topographic variables (TRMI, topographic position index, slope, and aspect), and stand structure (basal area and trees per hectare). Some of these variables were highly correlated (e.g., basal area and elevation) and were therefore not combined in a single model. Species differences in climate response (correlation with FDSI) was tested via ANOVA and Tukey's post-hoc means test ( $\alpha = 0.05$ ).

Based on the distribution of correlation to FDSI and ANOVA results, we divided the CFI plot-species chronologies into four groups. We then performed 11-year running correlations between FDSI and the chronologies within each group to test for time-varying trends in climate response. These running correlation time series were compared to PRISM climate data specific to our study area<sup>4</sup>, which is a 3° by 3° area encompassing 81 PRISM grid points. We used the PRISM data to generate the same seasonal PPT and VPD time series as in FDSI and computed anomalies against a 20<sup>th</sup> century base period.

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<sup>4</sup> From <http://www.prism.oregonstate.edu/>

## RESULTS

From 36 CFI plots, we successfully crossdated and measured 899 cores from 469 trees. We use these data to analyze 57 residual tree-ring chronologies that include 11 PIED, 33 PIPO and 13 PSME chronologies (Table 2). The average CFI chronology length is 225 years and the median length is 190 years, while the longest chronology is 487 years. Signal strength among the CFI chronologies demonstrates the strong crossdating and climate response of our trees, despite our non-selective sampling design. The  $\bar{r}_{\text{eff}}$  has range of 0.309 to 0.831, with a mean of 0.615, and EPS has a mean of 0.876 and a range of 0.631 to 0.965. Eighty-percent of our chronologies fall at or above the arbitrary yet widely-used thresholds of acceptance for these statistics, which are  $\bar{r}_{\text{eff}} = 0.5$  and  $\text{EPS} = 0.85$  (Wigley et al. 1984; Briffa and Jones 1990).

We note that because we selected plots with greater numbers of larger living trees, we found little evidence of logging at our plots. There were some stumps of dispersed ages at most plots, which we usually interpreted as light, selective cutting by locals for fuelwood and/or traditional uses over many years. Commercial logging from the 1960s to the late-1980s was more apparent around and within only two plots from our 2014 plot sample in the southern Chuska Mountains.

At one plot we found an anomalous period of tree-ring growth suppression in PSME trees that lasted ~5 years. We interpreted this as evidence for defoliation, probably caused by western spruce budworm (*Choristoneura occidentalis*) (e.g., Swetnam and Lynch 1993). Given its short duration at this plot, and our use of relatively flexible

smoothing splines for standardization, we included this chronology in our tests for climate sensitivity.

Correlations of our CFI chronologies against FDSI were generally quite high and all were highly significant ( $p < 0.001$ ) except for one PSME chronology ( $p = 0.025$ ). The correlations averaged 0.54 (0.56 median) and with a range from 0.23 to 0.70 (Figure 3). Correlations of PIED chronologies had the narrowest range (0.52 – 0.62), with a mean of 0.60. Correlations of PSME chronologies were the lowest, with a mean of 0.48 and range of 0.23 to 0.70. The PIPO chronology correlations averaged 0.54 with a range of 0.29 to 0.69. We found a weak inter-species difference in the correlations to FDSI (ANOVA;  $R^2 = 0.11$ ,  $p = 0.04$ ) that resulted from a slight, but significant difference between the means of PIED and PSME (Tukey's post-hoc means test;  $p = 0.034$ ). There were no differences between the correlations of PIPO and either PSME or PIED ( $p > 0.24$ ). Correlations between FDSI and our 14 dendroclimate chronologies were higher than for the CFI plots, ranging from 0.56 to 0.78 and averaging 0.67. There were no inter-species difference among the dendroclimate sites ( $R^2 = 0.16$ ,  $p = 0.37$ ).

The spatial distribution of correlations with FDSI shows that the Defiance Plateau had higher correlations than the Chuska Mountains (Figure 3). Interestingly, nearly every chronology from the Defiance Plateau, regardless of species, had roughly equal correlations to FDSI. In the Chuska Mountains, conversely, we found far greater variability in correlation to FDSI. The overall lowest correlations came from a group of four plots in the northern area of the Chuskas, where forests were among the most mesic of our sample. Trees on these plots were among the youngest in our network (< 100 years).

We tested a number of topographic, stand structure, and climate variables to explain the distribution of correlations to FDSI. Consistently, elevation explained the most variance ( $R^2 = 0.32$ ,  $p < 0.001$ ). For PIPO and PSME chronologies, the correlation to FDSI declines with elevation (Figure 4). PIED, however, shows a slight increase in correlation with elevation, but there is far less range in both elevation and FDSI correlation than for the other species. Correlations between the dendroclimate chronologies and FDSI also decline with elevation, but less steeply than for CFI chronologies. Also, at a given elevation, correlations for dendroclimate sites are higher on average than for the CFI plots.

Based on these results, we divided the CFI plots into four groups: all PIED, high-correlation PIPO-PSME (at or above the upper quartile PIPO-PSME correlations;  $r \geq 0.62$ ), mid-correlation PIPO-PSME (within the inner-quartile range;  $0.62 < r \leq 0.46$ ; median  $r = 0.54$ ), and low-correlation PIPO-PSME (below the lower quartile;  $r < 0.46$ ). Plots in the high-correlation and mid-correlation groups consist of generally more arid sites at lower elevations, while the opposite is true for the low-correlation group. The ring width index (RWI) chronologies for these groups are shown in Figure 5, which highlight the general differences between the PIPO-PSME groups. Low-correlation chronologies include much less interannual variability during the 20<sup>th</sup> century than the other groups, but the pattern begins to change in the late 1980s when there is considerable more variance.

Running correlations with FDSI among the three groups show a large degree of variability through time (Figure 6). Periods of higher correlation among all groups tend to occur during drought periods such as the 1950s and 2000s. Interestingly, growth

synchrony among plots and to FDSI is most pronounced since ~1997 when a trend in VPD develops with levels that have remained above the 20<sup>th</sup> century average.

## DISCUSSION

Recent studies predict that increased temperatures via anthropogenic global warming will reduce tree growth and increase forest drought stress across semiarid forests of North America (Williams et al. 2010, 2013; Charney et al. 2016; Restaino et al. 2016). These trends are likely to lead to or be associated with higher rates of tree mortality from a variety of drought-related causes (Allen et al. 2015). With one exception (Restaino et al. 2016), these studies evaluated tree growth characteristics across the network of dendroclimate chronologies, which are mostly derived from targeted tree sampling at particularly drought-sensitive sites (Fritts 1976; Speer 2010). Trees growing under these site conditions are also often the most vulnerable to drought-induced declines and mortality (Allen and Breshears 1998; Kelly and Goulden 2008). Thus, there may be some geographic bias to large-scale predictions of drought-induced forest decline and associated projections in forest mortality in semiarid forests.

We tested this concern with a set of 14 targeted dendroclimate chronologies and 57 non-selectively sample plots from a 250,000 ha study area. Our site selection procedures assured representation of this semiarid ponderosa pine and dry mixed conifer forest across a range of elevations and topographically-related soil moisture conditions (Figure 2). Our results substantiate some of the concern over bias, with targeted drought-prone sites showing a greater amount growth synchrony with regional climate (FDSI;

Williams et al. 2013) than non-selectively sampled CFI plot trees (Figure 3). This pattern was consistent across species and elevations, with PSME showing the greatest degree of climate sensitivity across elevations (Figure 4).

It is notable, however, that for half of our CFI plots, FDSI explained between 31% and 49% of the growth variance (median correlation = 0.56). While these values lie far below the regional correlation of FDSI with coherence in tree growth (0.91; Williams et al. 2013) they demonstrate a relatively high degree of landscape-scale response to regional climate. This synchrony implies that if projected trends in VPD under “business-as-usual” greenhouse gas emissions scenarios are realized, these stands – constituting much of the study area below ~2500 m – may begin to experience increased drought stress and associated mortality from disturbances and drought (Williams et al. 2013; Allen et al. 2015).

In this study, PIED showed the most consistent and relatively high correlation with FDSI. PIED is one of the most vulnerable species to drought-induced decline, with near landscape-scale die-offs in the region associated with increased temperatures, reduced tree defenses, and bark beetle infestation (Breshears et al. 2005; Adams et al. 2009; Macalady and Bugmann 2014). Although we sampled across a range of topographic moisture conditions (Figure 2), the elevation range of PIED on our plots was more limited than for the other species, probably because we only sampled PIED at its upper ecotone where it is intermixed with PIPO. The slight increase in correlation with elevation (Figure 4) likely reflects site-to-site variability as opposed to a physiographic relationship; considerable variability exists for the other species at a given elevation. We hypothesize that if we had sampled PIED across its elevation range in the study area (i.e.

downward into piñon-juniper woodlands) that its climate response would have increased similarly to PIPO or PSME, until some level of heightened moisture stress at lower elevations would have made crossdating and chronology development impossible (Fritts et al. 1965). Across Colorado and Utah, however, PIED usually has a strong climate-growth relationship despite a broad range in elevations (observations by C. Woodhouse).

The lowest degree of climate response that we found was in the upper-elevation and mesic sites of the study area, and was reflected equally in PIPO and PSME. The low correlation group included 12 chronologies from 8 CFI plots that correlated with FDSI at  $< 0.46$ , representing roughly 22% of our study area. These upper elevation sites ( $\geq \sim 2600$  m) are associated with reduced temperatures, increased PPT, and lower VPD relative to lower elevations. Regionally, upper-elevation forests are probably the least well represented in predictions of forest drought stress (Williams et al. 2010, 2013; McDowell et al. 2016) and may be an area of greater resilience to climate-induced forest declines and change (Rehfeldt et al. 2014a; Allen et al. 2015). They may also be refuge areas during severe and widespread disturbances, such as crown fire (e.g., O'Connor et al. 2014). In other regions, sites with higher moisture levels can be less responsive to temperature-induced growth declines (Williams et al. 2011; Liu et al. 2013).

The chronologies in the low correlation group had relatively low interannual growth variance through much of the 20<sup>th</sup> century (i.e. a “complacent” pattern) (Figure 5). Even the “extreme drought” of the 1950s (Weiss et al. 2009) caused only a slight reduction in growth compared to trees with higher correlation to FDSI. Beginning in the mid-1980s, however, we found a marked increase in growth variation, with the result that the upper-elevation sites begin to grow similarly to the lower-elevation sites. This

increase in variability is not shared among the other site chronologies. Known drought years, such as 2002-2006, appear to also synchronize growth among plots within this group to an unprecedented degree. We attribute this change in tree growth to drought and rising VPD; however, below-average PPT begins in ~1996 and rising VPD begins in ~1990. Why growth variability and synchrony would initiate 4-5 years earlier than rising VPD is unclear. Logging in the area may be a factor – reducing competition effects and enhancing moisture availability as a primary limiting agent – but we did not find evidence of heavy cutting at most of these plots and there was no logging at several plots. Rising VPD is the best explanation because it would initiate moisture stress as it decreases the persistence of cool-season moisture via early loss of snowmelt or increases the probability of winter PPT falling as rain rather than snow, which would also reduce persistence. The effects of rising temperature on reduced moisture persistence is noted for streamflow rates in the Colorado River (Woodhouse et al. 2016). This reduction in soil moisture would increase its importance as a limiting factor, making the trees more responsive to interannual variability in PPT.

Changing growth at upper elevations is also associated with a greater degree of correlation to FDSI. Running correlations among the low-correlation PIPO-PSME group closely match those from higher correlation groups since ~1997 (Figure 6). This cross-scale growth synchrony is also seen in the late 1950s and early 1960s, following the 1950s drought, but the recent period has lasted longer and includes less within-group variability in running correlations (shading in bottom panel, Figure 6). Thus, it appears that VPD sustained above the 20<sup>th</sup> century average for the last two decades has synchronized growth across our study area and to the region as a whole.

This finding indicates that tree growth across elevations, soil moisture conditions, and the SW region are synchronized during drought periods, especially with rising temperatures. Rising temperatures amplify VPD (Breshears et al. 2013), which causes heat-related drought stress to plants (Allen et al. 2015). Therefore, in areas where moisture was not historically limiting, such as upper-elevation sites, climate change may already be causing greater moisture limitation.

Our results are in accord with studies that have demonstrated the synchronizing and amplifying effects of greater aridity and higher temperatures on forest disturbance processes, including increases in fire season length and area burned (Westerling et al. 2006; Westerling 2016), bark beetle attacks (Raffa et al. 2008; Anderegg et al. 2015), and drought-induced mortality (van Mantgem et al. 2009; McDowell et al. 2016). Cumulatively, the results of this study and of dozens from the region (see Allen et al. 2015) show that climate change has and will continue to exert a strong influence on the health of forests across many landscapes, including nearly all of our study area below ~2500 m. Future research should aim for a better understanding of the tree health, mortality rates, and disturbances at upper-elevation sites, as these are critical areas to restoration efforts (Rehfeldt et al. 2014a) and preservation of ecosystem services.

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## TABLES

**Table 1.** Dendroclimate chronologies

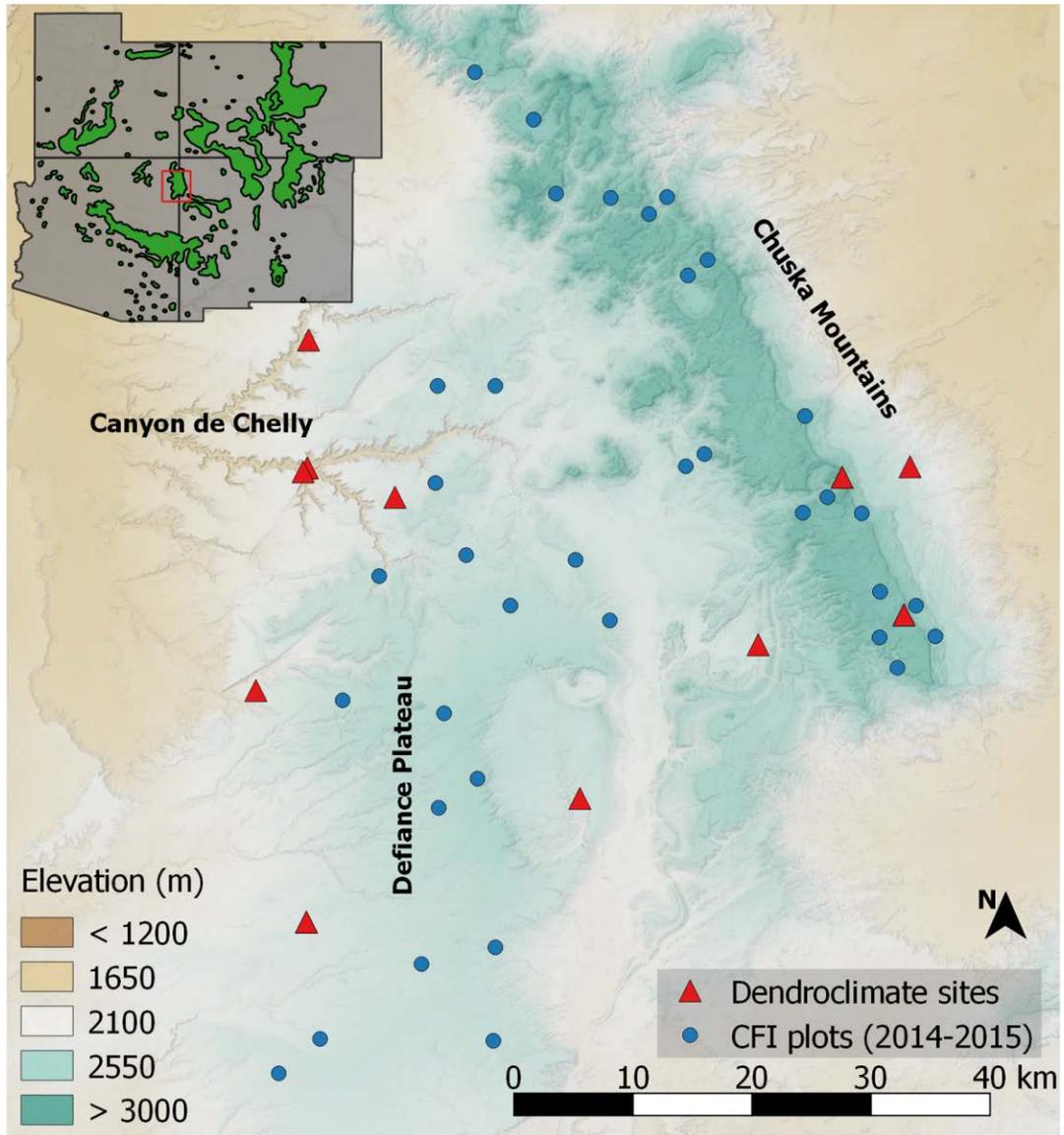
Site name	Species	Year range	Latitude	Longitude	Elevation (m)	Contributor	ITRDB file name
Oak Creek Wash	PSME	1200-2015	35.94	-108.93	2325	This study	-- Update of AZ080
Spider Rock Douglas-fir	PSME	1396-2014	36.11	-109.35	1980	This study	
Spider Rock Overlook	PIED	1601-1971	36.10	-109.36	2134	Dean	AZ081
Small Twin Canyon	PIPO	1656-2014	36.08	-109.27	2152	This study	--
Defiance Cross Canyon	PIPO	1611-1972	35.69	-109.35	2159	Dean	AZ090
Defiance Cross Canyon	PIED	1620-1972	35.69	-109.35	2159	Dean	AZ139
Narbona Pass	PIPO	842-2010	36.10	-108.86	2621	Guiterman	NM589
Narbona Pass	PSME	1314-2010	36.10	-108.86	2621	Guiterman	--
Defiance West	PSME	1474-1965	35.90	-109.40	2121	Stokes	AZ046
Defiance East	PIPO	1554-1965	35.80	-109.10	2210	Stokes	AZ045
Crystal Douglas-fir	PSME	1540-1978	35.97	-108.80	2751	Cleaveland	NM520
Crystal Ponderosa	PIPO	1652-1978	35.97	-108.80	2751	Cleaveland	NM529
Tseh-Ya-Kin Canyon	PSME	1500-1971	36.22	-109.35	1905	Dean	AZ083
Washington Pass piñon	PIED	1599-1976	36.11	-108.79	2225	Dean	--

**Table 2.** Sample depth and chronology statistics for CFI plot chronologies.

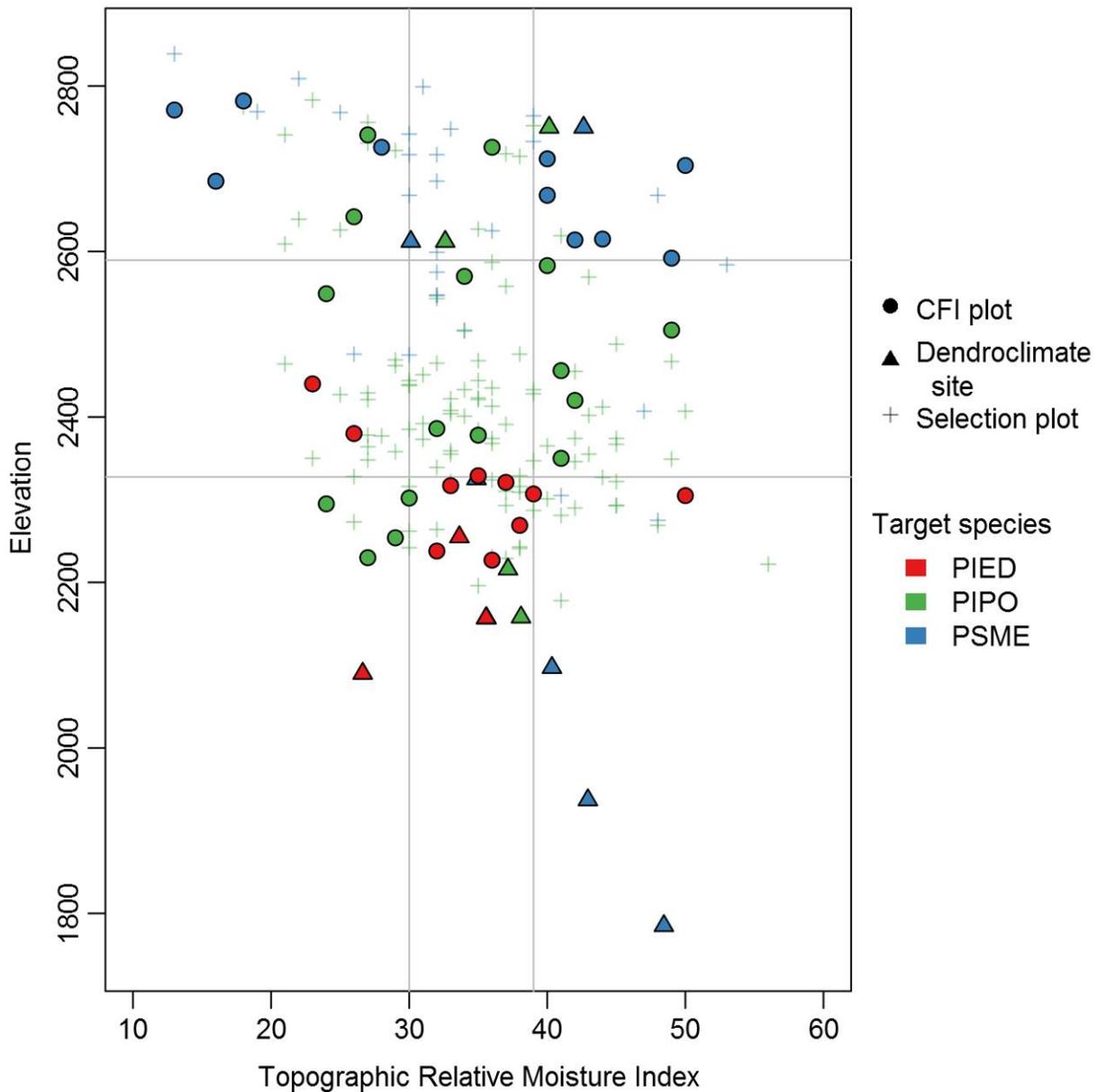
Plot	Species	Inner year	Number of trees (cores)	$\bar{r}_{\text{eff}}$	EPS	SNR	Plot	Species	Inner year	Number of trees (cores)	$\bar{r}_{\text{eff}}$	EPS	SNR
104	PIED	1790	6 (8)	0.519	0.828	4.815	1833	PSME	1551	10 (20)	0.831	0.965	27.882
104	PIPO	1904	5 (10)	0.771	0.878	7.192	1860	PIPO	1867	10 (18)	0.495	0.881	7.41
306	PIPO	1854	10 (20)	0.718	0.955	21.194	1860	PSME	1915	4 (8)	0.661	0.818	4.5
428	PIPO	1764	9 (15)	0.614	0.897	8.676	2059	PIPO	1871	9 (18)	0.349	0.77	3.348
636	PIED	1737	3 (5)	0.646	0.767	3.3	2059	PSME	1927	6 (12)	0.664	0.858	6.03
636	PIPO	1617	9 (15)	0.741	0.953	20.413	2160	PIPO	1813	9 (18)	0.517	0.862	6.267
811	PIPO	1635	10 (20)	0.711	0.941	15.868	2243	PIED	1838	9 (16)	0.584	0.904	9.396
921	PIPO	1855	10 (20)	0.539	0.899	8.918	2243	PIPO	1856	5 (10)	0.826	0.951	19.35
927	PIPO	1900	10 (20)	0.708	0.956	21.737	2255	PIPO	1871	5 (10)	0.309	0.631	1.712
942	PIED	1753	9 (18)	0.646	0.919	11.314	2255	PSME	1853	11 (22)	0.651	0.946	17.684
942	PIPO	1715	5 (10)	0.819	0.943	16.437	2344	PIED	1812	3 (6)	0.693	0.858	6.052
942	PSME	1712	5 (10)	0.799	0.916	10.975	2344	PIPO	1811	3 (6)	0.755	0.891	8.144
948	PIED	1660	6 (10)	0.657	0.897	8.677	2356	PIPO	1870	8 (16)	0.473	0.86	6.135
948	PIPO	1708	8 (16)	0.83	0.954	20.95	2356	PSME	1932	6 (12)	0.469	0.719	2.553
1037	PIED	1676	6 (12)	0.671	0.891	8.174	2840	PIPO	1825	8 (16)	0.555	0.869	6.617
1123	PIPO	1679	10 (20)	0.661	0.905	9.523	2840	PSME	1906	9 (18)	0.644	0.895	8.56
1168	PIPO	1861	9 (13)	0.36	0.801	4.031	2846	PIPO	1861	9 (17)	0.558	0.908	9.873
1206	PIED	1888	3 (6)	0.602	0.8	3.997	2846	PSME	1855	10 (18)	0.487	0.868	6.6
1212	PIPO	1528	9 (18)	0.551	0.821	4.576	2941	PIPO	1797	13 (24)	0.387	0.844	5.414
1248	PIED	1635	6 (11)	0.584	0.84	5.231	2941	PSME	1808	12 (23)	0.495	0.877	7.162
1248	PIPO	1705	8 (16)	0.808	0.951	19.558	3140	PIPO	1845	11 (22)	0.52	0.911	10.201
1334	PIPO	1620	10 (20)	0.71	0.905	9.478	3140	PSME	1900	4 (8)	0.76	0.871	6.742
1465	PIPO	1878	6 (12)	0.443	0.785	3.657	3232	PIPO	1675	16 (28)	0.515	0.853	5.817
1465	PSME	1913	10 (19)	0.576	0.883	7.543	3235	PIPO	1864	14 (26)	0.408	0.803	4.087
1560	PIPO	1789	7 (14)	0.696	0.914	10.68	3330	PIPO	1705	5 (10)	0.692	0.889	8.014
1560	PSME	1851	6 (12)	0.696	0.884	7.603	3330	PSME	1901	12 (24)	0.546	0.785	3.656
1637	PIED	1660	9 (17)	0.591	0.908	9.839	3434	PIPO	1849	14 (28)	0.534	0.93	13.34
1833	PIED	1718	9 (18)	0.669	0.85	5.661	3532	PIPO	1859	17 (32)	0.528	0.928	12.979
1833	PIPO	1623	4 (8)	0.821	0.926	12.495							

Notes:  $\bar{r}_{\text{eff}}$  is the effective chronology signal (Cook and Kairiukstis 1990); EPS is the expressed population signal (Wigley et al. 1984); SNR is the signal-to-noise ratio (Cook and Pederson 2011).

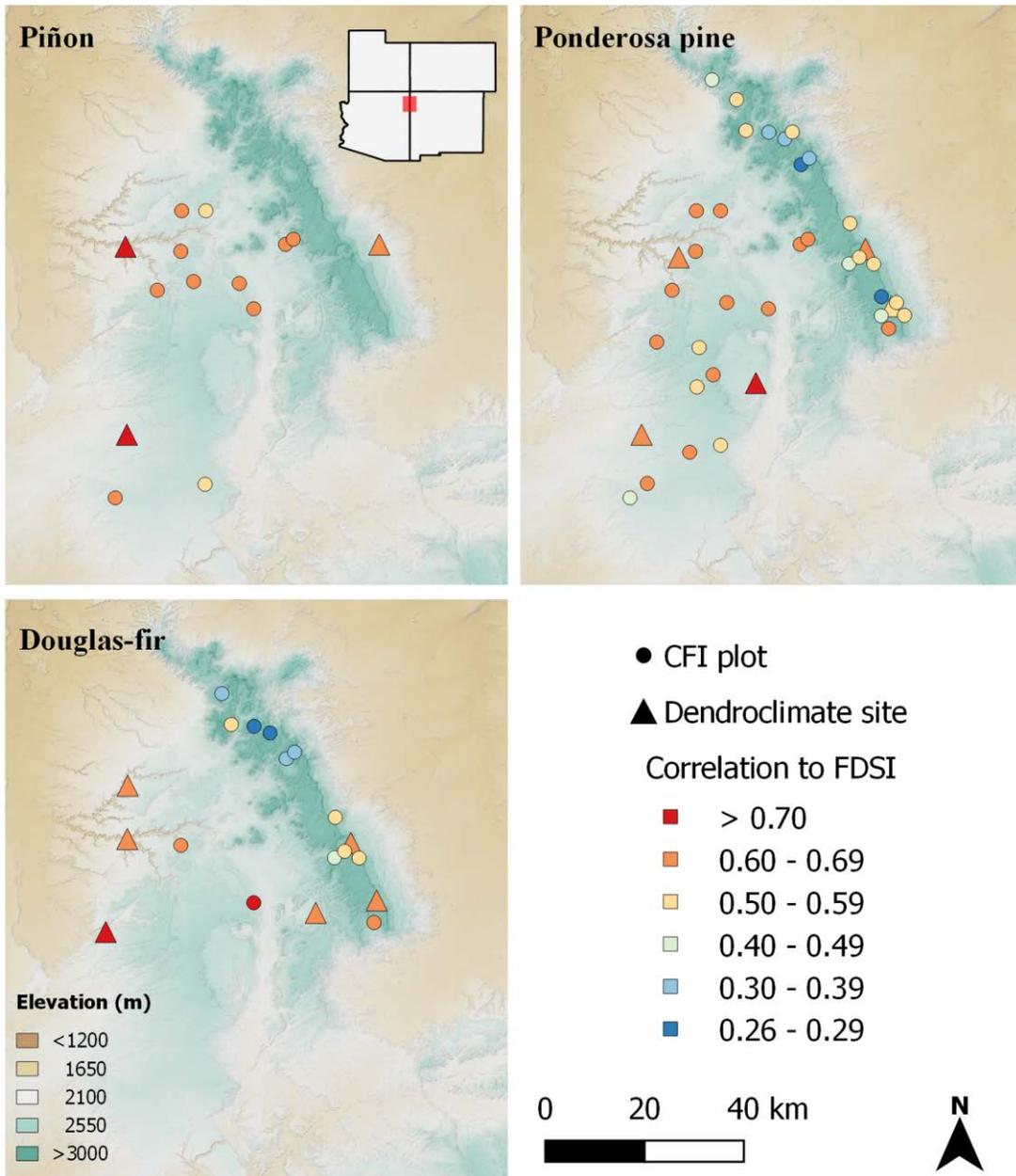
## FIGURES



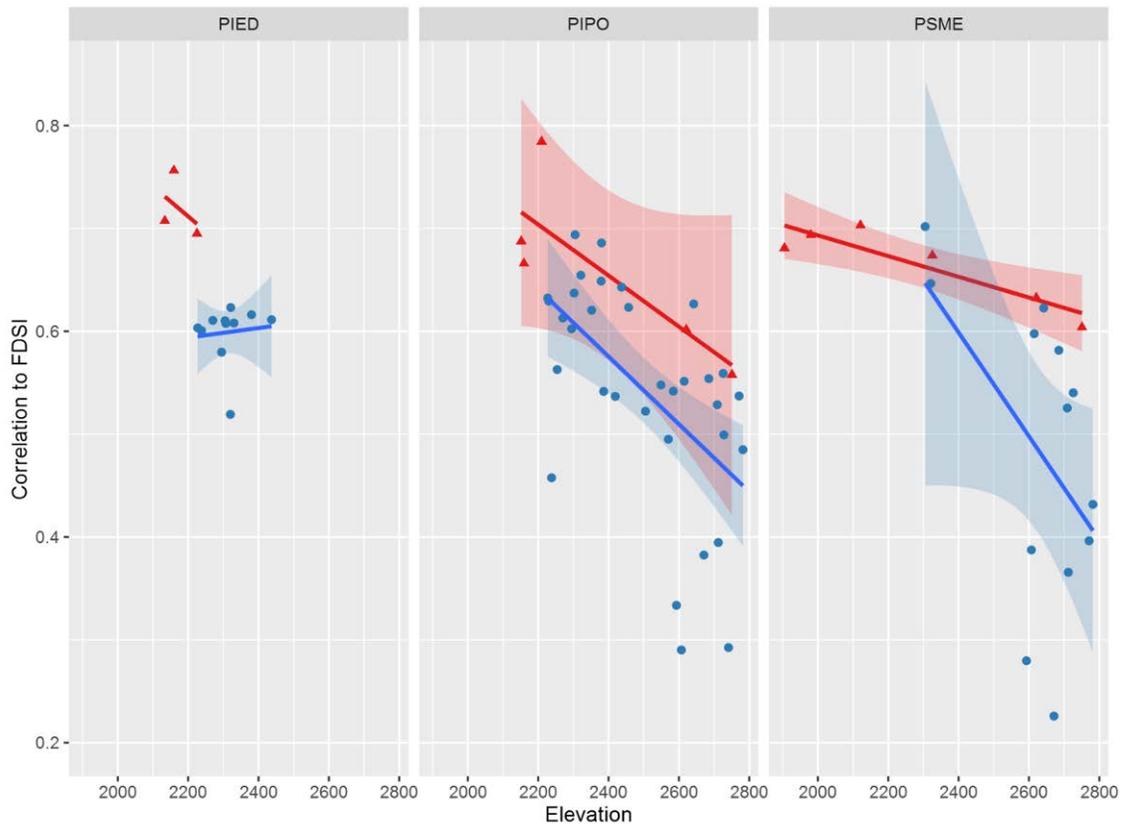
**Figure 1.** Locations of tree-ring collections in the Chuska Mountains and Defiance Plateau. Inset map shows the location of the study area (red box) on the Arizona-New Mexico border and the range of ponderosa pine (green).



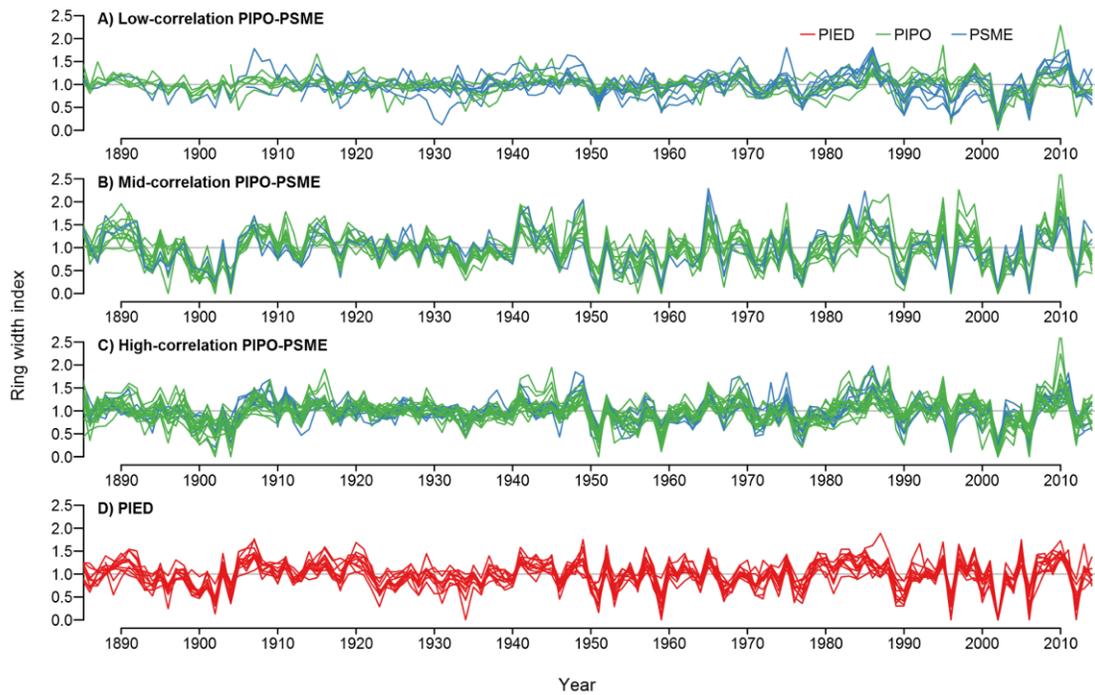
**Figure 2.** Physiographic space of CFI plots and dendroclimate sites. Tree-ring sampled locations from both 2014 and 2015 are shown in dark colors. CFI plots were randomly selected from the subset of plots (“selection plots”,  $n = 167$ ) that met our criteria for chronology development. These criteria and the selection procedure are described in the Methods section. Low (high) values of topographic relative moisture index represent drier (wetter) sites. Grey lines indicate the inner quartile range of elevation and TRMI for the selection plots. Target species include piñon (PIED), ponderosa pine (PIPO), and Douglas-fir (PSME).



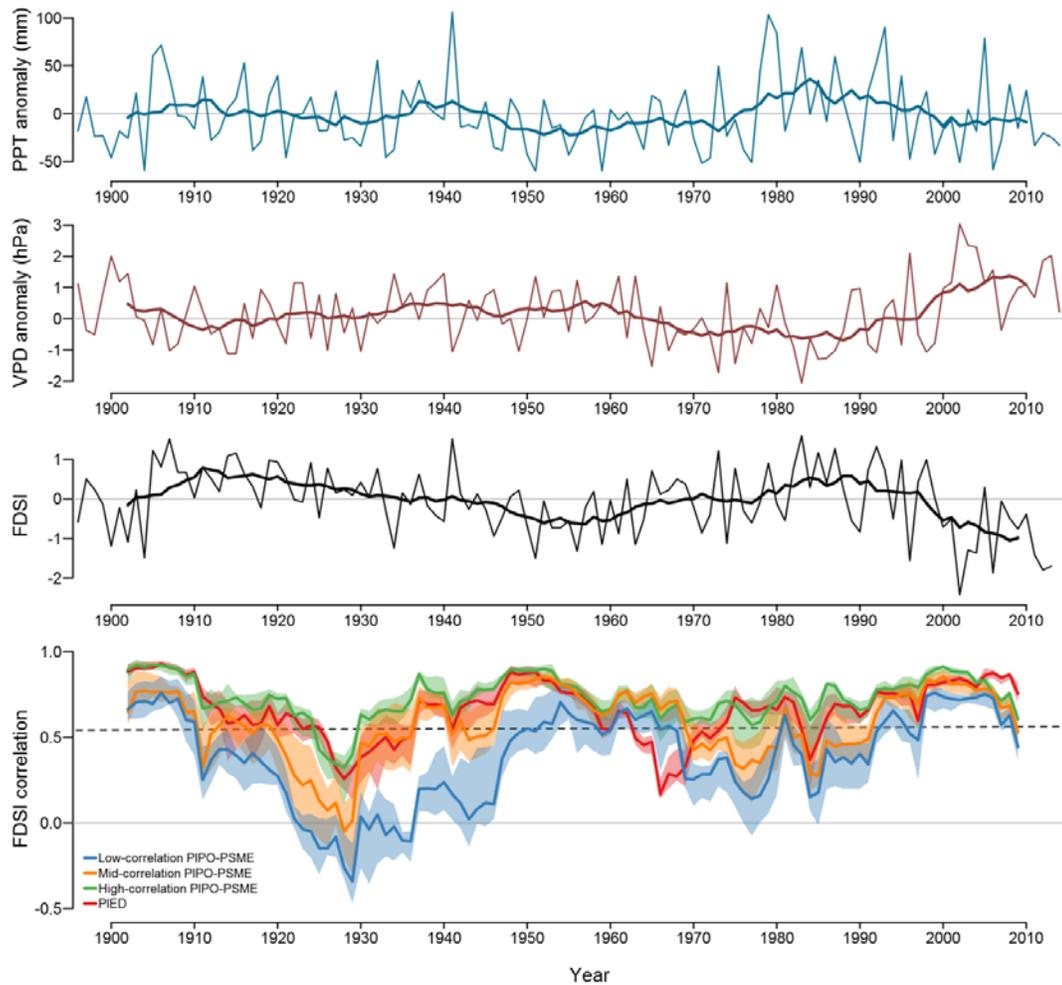
**Figure 3.** Spatial distribution of chronology correlations with FDSI for each of the three target species.



**Figure 4.** Correlations with FDSI as a function of elevation for CFI plots (blue circles) and dendroclimate chronologies (red triangles). Lines show linear regressions with shaded areas for the 95% confidence limits. Species are separated into each panel, with piñon (PIED), ponderosa pine (PIPO), and Douglas-fir (PSME).



**Figure 5.** CFI chronologies grouped according to their species and correlation with FDSI. Interannual variability of the low-correlation plots intensifies beginning in the late 1980s (A), whereas at other sites (B-D) interannual variability is relatively high throughout the 20<sup>th</sup> century.



**Figure 6.** Comparison of precipitation (PPT), vapor pressure deficit (VPD), and forest drought stress index (FDSI) with running correlations of CFI chronologies against FDSI. PPT and VPD are for the same seasons in FDSI and are averaged across the study area. Smoothed lines indicate 11-year running averages. The lower panel shows 11-year running correlations between CFI chronologies in the three different correlation groups, as in Figure 5. Dark lines are the median of the group's correlations and shading shows their inner-quartile range. The dashed line in the lower panel shows the median correlation (0.54) with FDSI across all PIPO and PSME plots. Drought periods in the 1950s and after 1997 tend to synchronize the climate response across all plots.