Historic Fire Regimes of Eastern Great Basin (USA) Mountains

Reconstructed from Tree Rings

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Doctor of Philosophy

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ABSTRACT

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Management of natural landscapes requires knowledge of key disturbance processes and their effects. Fire and forest histories provide valuable insight into how fire and vegetation varied and interacted in the past. I constructed multi-century fire chronologies for 10 sites on six mountain ranges representative of the eastern Great Basin (USA), a region in which historic fire information was lacking. I also constructed tree recruitment chronologies for two sites. I use these chronologies to address three research foci. First, using fire-scar data from four heterogeneous sites, I assert that mean fire interval (MFI) values calculated from composite chronologies provide suitable estimates of point MFI (PMFI) when sample area size is ≈½ ha. I also suggest that MFI values for single trees can be used to estimate PMFI after applying a correction factor. Next, I infer climate effects on regional fire patterns using 10 site chronologies and tree-ring-based indices of drought and of El Niño Southern Oscillation (ENSO) and Pacific Decadal Oscillation ([PDO), Pacific Ocean surface temperature variability known to affect North American climate. Regional fire years (≥33% of recording sites) were synchronized by wet-dry cycles where the probability of occurrence was highest in the first year of drought following a wet phase and lowest when climate conditions transitioned from dry to wet. Regional fire probability was highest when ENSO and PDO were negative (Southwest pattern). Local fire years occurred under a broad range of conditions. Fire seasonality was bimodal with early and late-season fires dominant. I imply that Native American burning practices were responsible for differences in historic and modern fire seasonality. Lastly, I assess fire regime and tree recruitment variability within two fire-sheds. PMFI varied more than 10-fold within each site. A mixed-severity regime was dominant. A majority (>60%) of fires were small (<10 ha) but together accounted for a minor proportion of area burned. Recruitment pulses varied spatially from stand to landscape-scales and were often synchronous with multi-decade, fire-quiescent periods. I recommend that management strategies employ fire and fire-surrogate treatments to restore disturbance processes to these and similar landscapes at spatial and temporal scales consistent with the historic record.

Keywords: fire scars, dendrochronology, point mean fire interval, climate-fire interactions, anthropogenic fire, mixed-severity fire, mixed-conifer forests, fire restoration

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INTRODUCTION

Fire functions as a "keystone disturbance" for most natural vegetation types in western North America dictating plant community structure and the direction and pace of ecosystem processes (Frost 1998; Keane et al. 2002). The manner in which fire is manifest over time and space is collectively referred to as the fire regime. Fire regime parameters are quantifiable at multiple temporal and spatial scales as frequency, extent, pattern, seasonality, and predictability (Morgan et al. 2001). An additional parameter, fire severity, is a measure of fire-induced ecosystem change (Ryan and Noste 1985; Romme et al. 2003) and may vary through time and space. These indicators of fire regime are interdependent. For example, severity generally increases as frequency decreases due to greater accumulations of fuels over time (Wright and Bailey 1982). Similarly, fire severity also correlates with seasonality, extent, and pattern (Wright and Bailey 1982; Keane et al. 2002). Fire characteristics are affected locally by past events as well as current events on adjacent landscapes (temporal and spatial autocorrelation; Morgan et al. 2001).

Fire regimes are both heterogeneous and dynamic. Local, spatial scale variation of fire regime is related to topography (i.e. elevation, aspect, slope, connectivity) through the effects of topographic position on species composition (fuel types and arrangement), productivity, fuel desiccation rates, fuel continuity, and wind speed (Swetnam and Baisan 1996; Taylor and Skinner 1998; Brown et al. 2001; Heyerdahl et al. 2001; Iniguez et al. 2008, 2009). Superimposed over local variation, regional synchronization of fire patterns is driven by climate through the effects of precipitation and temperature on fuel production and desiccation. For example, in historic fire regimes of southwest pine forests, major fire years are correlated with severe drought years, particularly when preceded by 1 to 3 years of above average

precipitation (Swetnam and Baisan 1996; Brown and Shepperd 2001; Brown et al. 2001; Kitzberger et al. 2001; Margolis and Balmat 2009). This relationship is strongest for dry forest types and becomes weaker with increasing effective precipitation (i.e. elevation). Donnegan et al. (2001) and Sherriff and Veblen (2008) observed a similar pattern for forests of central Colorado and implicated the importance of inter-annual climate cycling for the sequential production and conditioning of fuels prior to major fire events. In addition to inter-annual synchronization of fire by climate, decadal to millennial scale variation in fire occurrence has been correlated to same-scaled climate variation (Touchan et al. 1995; Swetnam and Baisan 1996; Swetnam and Betancourt 1998; Grissino-Mayer and Swetnam 2000; Brown et al. 2001; Donnegan et al. 2001; Heyerdahl et al. 2002a; Whitlock et al. 2003; Kitzberger et al. 2007; Morgan et al. 2008).

Climate may also produce a seasonal signature in fire regime reconstructions. In the Southwest, pre-monsoonal dry lightning storms often follow warm dry spring conditions resulting in a high proportion of early season fires (Swetnam and Betancourt 1990; Swetnam and Baisan 1996; Brown et al. 2001). Conversely, late season fires become dominant at higher latitudes where springs are relatively cool and wet and fuel desiccation and thunderstorms activity is delayed (Bekker and Taylor 2001; Donnegan et al. 2001; Brown and Shepperd 2001; Heyerdahl et al. 2001; Schmidt et al. 2002).

Locally, unexpected frequency, regularity, or seasonality might be interpreted as evidence of anthropogenic influence on fire regime (Swetnam and Baisan 1996; Allen 2002). Changes in fire regime associated with changes in anthropogenic uses and management have been documented in North America (for examples see Arno et al. 1997; Fulé and Covington 1999, Shumway et al. 2001, Guyette et al. 2003, Parshall et al. 2003), Europe (Niklasson and

Granstrom 2000, Lloret and Mari 2001), and Australia (Ward et al. 2001). Regionally, synchronized changes in fire regime are associated with Euro-American settlement and fire suppression policy implementation (Swetnam 1993; Covington and Moore 1994; Swetnam and Baisan 1996; Fulé et al 1997; Brown and Sieg 1999; Bekker and Taylor 2001; Heyerdahl et al. 2001; Keane et al. 2002).

Fire regime reconstructions, or fire histories, are developed to describe past fire activity through one or more parameters of interest for a specific geography and time period, and are critical for understanding the relationships between fire regimes, regional and local controls, and vegetation dynamics (Arno et al. 1997; Taylor and Skinner 1998; Bekker and Taylor 2001; Morgan et al. 2001). To effectively uncover the relationship between regional and local controls on fire regime, reconstructions must have resolution at the same scales over which variation occurs (Ricklefs 1987; Weins 1989; Levin 1992).

A variety of methods are employed for fire history development, each with limits in temporal and spatial resolution. Those methods capable of fixing fire events with annual accuracy are critical for elucidating fire-climate relationships at inter-annual and decadal scales. Only cross-dated dendrochronological (tree-ring) approaches are effective in achieving that level of accuracy for multi-century time periods. However, tree-ring based fire frequency values aggregated from large or ambiguous landscapes can be misleading because they tend to overestimate local fire frequencies and because they may fail to identify topographically-induced variation in fire regime (Swetnam and Baisan 1996; Baker and Ehle 2001; Falk et al. 2007). Temporal accuracy and spatial resolution of fire frequency reconstructions can be improved through careful selection of the size, number, and spacing of sample points across the landscape (Heyerdahl et al. 2001; Iniguez et al. 2008; Margolis and Balmat 2009).

Although specific boundaries vary by definitions employed, the Great Basin encompasses the northern, elevated section of the "Basin and Range Province" of western North American and includes most of the state of Nevada and significant portions of Utah, Idaho, California, and Oregon (Fig.I.1). The region is characterized by over 100 relatively narrow mountain ranges with a generally north/south orientation and separated by broad, internally-drained desert valleys. Elevation for 33 of these ranges exceeds 3,048 m (10,000 ft.; Grayson 1993). The climate is generally dry due to rain shadow effects of the Sierra Nevada and Cascade Ranges to the west and Rocky Mountains to the east (Peterson 1994). Seasonality of precipitation varies along a geographic gradient with the importance of winter and spring Pacific frontal storms decreasing and summer monsoons increasing as one travels from north to south and from west to east. Precipitation is further modified locally by elevation and orographic position. In short, regional climate patterns are intermediate or transitional between those of the interior Pacific Northwest and the interior Southwest.

Great Basin plant community types form more or less distinct zones across elevational gradients reflecting parallel gradients in temperature and precipitation (Holmgren 1972; Harper et al. 1978). Zonation is further modified by slope, aspect, and substrate. Drought tolerant sub-shrubs and grasses dominate plant communities of arid valleys and dry foothills. Sagebrush (*Artemisia* spp.)-grass steppe communities occupy a broad zone from valley floors to dry mid-elevation montane sites. An upward and downward expanding, pinyon(*Pinus monophylla*)-juniper (*Juniperus* spp.) woodland belt is superimposed through the center of the various sagebrush-grass steppe types on all but northern ranges (Tausch et al. 1981). Various combinations of pine (*Pinus*), fir (*Abies*, *Pseudotsuga*), and spruce (*Picea*) occupy the mixed conifer and sub-alpine forests. A treeless alpine zone is found on the highest peaks.

In spite of significant progress in reconstructing historic fire regimes for forested landscapes across much of western North America, the interior Great Basin has largely been ignored prior to this work (Heyerdahl et al. 1995), subsequently, little is known of corresponding historic fire regimes and the processes that drive them; knowledge critical for science-based fire and fuels management programs. This study begins to address that need. My focus is in the mountains of the east-central portion of the Great Basin corresponding to western Utah and eastern Nevada. I have collected tree-ring derived fire history data from 10 sites located on six mountain ranges. I use subsets of that data to develop three independent papers, presented here as chapters, that address contemporary issues in fire history research. Each paper is written for publication in a scientific journal and is organized into standard sections including: introduction, methods, results, discussion, acknowledgements, literature cited, tables and figures.

Chapter 1 is titled, **Composite and Single-tree Fire Chronologies from Heterogeneous Landscapes: Strategies for Estimating Point Mean Fire Interval**, and deals with ways in which fire history data are collected and used to infer fire frequency. Specifically, I explore benefits and limitations to amalgamated fire chronologies from multiple trees as proxies for point fire frequency using data from four eastern Great Basin sites. I also propose a way in which single tree chronologies can be used to estimate point fire frequency. The fire history community has wrestled with this issue over the past decade with important contributions from Baker and Ehle (2001), Van Horne and Fulé (2006), Falk et al. (2007), however, none have provided a satisfactory method for restricting composite records so that they provide defensible estimates of point fire frequency, especially for topographically and vegetationally heterogeneous landscapes such as those found in the mountains of the eastern

Great Basin. That is the purpose of Chapter 1 and the method is subsequently applied on data from two fire-sheds in Chapter 3.

Chapter 2 is titled, **Climate and Human Influences on Historical Fires (1400-1900) in the Eastern Great Basin (USA)**. In this paper I use data from 10 sites to infer the fire-synchronizing role of climate in this region that is geographically transitional between the Southwest and Interior Northwest. Sites are located near 40˚ N Latitude, a proposed pivot point for a dipole pattern of fluvial and drought conditions that is strongly influenced by Pacific surface sea temperature and associated climate variability patterns, namely El Niño Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO; Mock 1996; Dettinger et al. 1998). Significant relationships between fire occurrence and independent reconstructions of drought, ENSO, and PDO have been observed north and south of the pivot point in numerous studies (for examples see Westerling and Swetnam 2003; Kitzberger et al. 2007; Heyerdahl et al. 2008). The importance of climate as a driver of fire near the pivot point is less well understood. That understanding will improve fire-climate modeling at the sub-continental scale and may be critical in future non-analog climate scenarios. I also use fire seasonality evidence to explore the possibility of human influence on pre-1900 fire regimes in the region. Although many uses of fire by Native American have been documented (Williams 2004), finding evidence for a human influence in historic fire regime reconstructions has been difficult (Allen 2002; Griffin 2002). In this paper I offer credible evidence of a human fire footprint in fire seasonality data. Results have implications for management and restoration strategies of associated mountain ecosystems.

In the final chapter I explore "**Historic Fire Regime and Forest Variability on Two Eastern Great Basin Fire-sheds (USA)**". Although somewhat artificial, the term "fire-shed"

as used here designates a topographic unit somewhat sympatric to one or more small watersheds and corresponding to an area within which barriers (i.e. bare ridges, cliffs, waterways, etc.) are sufficiently permeable to allow fire to spread among all components of the landscape. Sampling methodology was the same at both sites. Fire-sheds differed in bio-physical character and historical use by humans and are generally representative of eastern Great Basin mountains. I identify spatial and temporal variation in fire frequency, extent, and severity using a combination of approaches and relate findings to vegetation histories at both sites. Implications for present and future management options are addressed.

Literature Cited

Allen, C.D. 2002. Lots of lightning and plenty of people: an ecological history of fire in the upland Southwest. In: Fire, native peoples, and the natural landscape. Washington, D.C.: Island Press, p. 143-193.

Arno, S.F., H.Y. Smith, and M.A. Krebs. 1997. Old growth ponderosa pine and western larch stand structures: influences of pre-1900 fires and fire exclusion. Research Paper INT-RP-495. Ogden, UT: USDA Forest Service, Intermountain Research Station. 20 p.

Baker, W.L. and D. Ehle. 2001. Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. Canadian Journal of Forest Research 31:1205-1226.

Bekker, M.F. and A.H. Taylor. 2001. Gradient analysis of fire regimes in montane forests of the southern Cascade Range, Thousand Lakes Wilderness, California, USA. Plant Ecology 155:15-28.

Brown, P.M. and C.H. Sieg. 1999. Historical variability in fire at the ponderosa pine – northern Great Plains prairie ecotone, southeastern Black Hills, South Dakota. Ecoscience 6:539-547.

Brown, P.M. and W.D. Shepperd. 2001. Fire history and fire climatology along a 5° gradient in latitude in Colorado and Wyoming, USA. Paleobotanist 50:133-140.

Brown, P.M., M.W. Kaye, L.S. Huckaby, and C.H. Baisan. 2001. Fire history along environmental gradients in the Sacramento Mountains, New Mexico: Influences of local patterns and regional processes. Ecoscience 8:115-126.

Covington, W. W. and M. M. Moore. 1994. Post-settlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. In: Sampson, R. N. and D. L. Adams (eds.) Assessing the forest ecosystem health in the Inland West. Journal of Sustainable Forestry 2:153-181.

Dettinger, M.D., D.R. Cayan, H.F. Diaz, and D.M. Meko. 1998. North-south precipitation patterns in western North America on interannual-to-decadal timescales. Journal of Climate 11:3095-3111.

Donnegan, J.A., T.T. Veblen, and J.S. Sibold. 2001. Climate and human influences on fire history in Pike National Forest, central Colorado. Canadian Journal of Forest Research 31:1526-1539.

Falk, D., A.C. Miller, D. McKenzie, and A.E. Black. 2007. Cross-scale analysis of fire regimes. Ecosystems 10:809-823.

Frost, C.C. 1998. Presettlement fire frequency regimes of the United States: a first approximation. Tall Timbers Fire Ecology Conference Proceedings 20:70-81.

Fulé, P.Z. and W.W. Covington. 1999. Fire regime changes in La Michilía Biosphere Reserve, Durango, Mexico. Conservation Biology 13:640-652.

Fulé, P.Z., W.W. Covington, and M.M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7:895-908.

Grayson, D.K. 1993. The Desert's Past: A Natural History of the Great Basin. Smithsonian Institution Press, Washington. 356 p.

Griffin, D. 2002. Prehistoric human impacts on fire regimes and vegetation in the northern intermountain West. In: Fire, native peoples, and the natural landscape. Washington, D.C.: Island Press, p. 77-100.

Grissino-Mayer, H.D. and T.W. Swetnam. 2000. Century-scale climate forcing of fire regimes in the American Southwest. The Holocene 10:207-214.

Guyette, R.P., D.C. Dey, and M.C. Stambaugh. 2003. Fire and human history of a barren-forest mosaic in southern Indiana. American Midlands Naturalist 149:21-34.

Harper, K. T., D. L. Freeman, W. K. Ostler, and L. G. Klikoff. 1978. The flora of Great Basin mountain ranges: Diversity, sources, and dispersal ecology. P. 81-104 *In* Intermountain biogeography: A symposium, ed. K. T. Harper and J. L. Reveal. Great Basin Naturalist Memoirs 2.

Heyerdahl, E.K., D. Berry, and J.K. Agee. 1995. Fire history database of the western United States. U.S. Environmental Protection Agency Report EPA/600/R-96/081.

Heyerdahl, E.K., L.B. Brubaker, and J.K. Agee. 2001. Spatial controls of historic fire regimes: a multiscale example from the interior west, USA. Ecology 82:660-678.

Heyerdahl, E.K., L.B. Brubaker, and J.K. Agee. 2002. Annual and decadal climate forcing of historical fire regimes in the interior Pacific Northwest, USA. Holocene 12:597-604.

Heyerdahl, E. K., P. Morgan, and J. P. Riser II. 2008. Multi-season climate synchronized historical fires in dry forests (1650-1900), Northern Rockies, USA. Ecology 89:705-716.

Holmgren, N. 1972. Plant geography in the Intermountain Region. Pages 77-161 *in* A. Cronquist, A.H. Holmgren, N.H. Holmgren, and J.L. Reveal, eds. Intermountain flora. Vol. 1. Hafner Publishing Company, New York.

Iniguez, J.M., T.W. Swetnam, and S.R. Yool. 2008. Topography affected landscape fire history patterns in southern Arizona, USA. Forest Ecology and Management 256:295-303.

Iniguez, J.M., T.W. Swetnam, and C.H. Baisan. 2009. Spatially and temporally variable fire regime on Rincon Peak, Arizona, USA. Fire Ecology 5:3-21.

Keane, R.E., K.C. Ryan, T.T. Veblen, C.D. Allen, J. Logan, and B. Hawkes. 2002. Cascading effects of fire exclusion in Rocky Mountain ecosystems: a literature review. General Technical Report. RMRS-GTR-91. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 24 p.

Kitzberger, T., T.W. Swetnam, and T.T. Veblen. 2001. Inter-hemispheric synchrony of forest fires and the El Niño-Southern Oscillation. Global Ecology & Biogeography 10:315-326.

Kitzberger, T., P.M. Brown, E. K. Heyerdahl, T.W. Swetnam, and T.T. Veblen. 2007. Contingent Pacific-Atlantic Ocean influence on multi-century wildfire synchrony over western North America. Proceedings of the National Academy of Science (USA) 104:543-548.

Levin, S.A. 1992. The problem of pattern and scale in ecology. Ecology 73:1943-1967.

Lloret, F. and G. Mari. 2001. A comparison of the medieval and the current fire regimes in managed pine forests of Catalonia (NE Spain). Forest Ecology and Management 141:155-163.

Margolis, E.Q. and J. Balmat. 2009. Fire history and fire climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA. Forest Ecology and Management 258:2416-2430.

Mock, C.J. 1996. Climate controls and spatial variations of precipitation in the western United States. Journal of Climate 9:1111-1125.

Morgan, P., C.C. Hardy, T.W. Swetnam, M.G. Rollins, and D.G. Long. 2001. Mapping fire regimes across time and space: Understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire 10:329-342.

Morgan, P., E.K. Heyerdahl, and C.E. Gibson. 2008. Multi-season climate synchronized forest fires throughout the 20th Century, Northern Rockies, USA. Ecology 89:717-728.

Niklasson, M. and A. Granstrom. 2000. Numbers and sizes of fires: long-term spatially explicit fire history in a Swedish boreal landscape. Ecology 81:1484-1499.

Parshall, T. D.R. Foster, E. Faison, D. McDonald, and B.C.S. Hansen. 2003. Long-term history of vegetation and fire in pitch pine-oak forests on Cape Cod, Massachusetts. Ecology 84:736-748.

Peterson, K.L. 1994. Modern and Pleistocene climatic patterns in the West. In: K.T. Harper, L.L. St. Clair, K.H. Thorne, and W.M. Hess (eds.) Natural History of the Colorado Plateau and Great Basin. University Press of Colorado, Niwot, CO. pp. 27-53.

Ricklefs, R.E. 1987. Community diversity: relative roles of local and regional processes. Science 235:167-171.

Romme, W.H., T.T. Veblen, M.R. Kaufmann, R. Sherriff, and CM. Regan. 2003. Part 1: Historical (Pre-1860) and current (1860-2002) fire regimes. In: Ecological Effects of the Hayman Fire. General Technical Report RMRS-GTR-114. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. pp. 181-195.

Ryan, K.C. and N.V. Noste. 1985. Evaluating prescribed fires. In: J.E. Lotam, B.M. Kilgore, W.C. Fisher, and R.M. Mutch (Eds.) Proceedings, symposium and workshop on wilderness fire. General Technical Report INT-182. Ogden UT: USDA Forest Service, Intermountain Research Station. pp. 230-238.

Schmidt, K.M., J.P. Menakis, C.C. Hardy, W.J. Hann, and D.L. Bunnell. 2002. Development of course-scale spatial data for wildland fire and fuel management. General Technical Report RMRS-GTR-87. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Sherriff, R.L. and T.T. Veblen. 2008. Variability in fire-climate relationships in ponderosa pine forests in the Colorado Front Range International Journal of Wildland Fire 17:50-59.

Shumway, D.L., M.D. Abrams, and C.M. Ruffner. 2001. A 400-year history of fire and oak recruitment in an old-growth oak forest in western Maryland, U.S.A. Canadian Journal of Forest Research 31:1437-1443.

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

Swetnam, T.W. and C.H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. In: C.D. Allen (technical editor), Fire Effects in southwestern

forests-Proceedings of the second La Mesa fire symposium. General Technical Report RM-GTR-286. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. pp. 11-32.

Swetnam, T.W. and J.L. Betancourt. 1990. Fire-Southern Oscillation relations in the southwestern United States. Science 249:1017-1020.

Swetnam, T.W. and J.L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate. 11:3128-31-47.

Tausch, R.J., N.E. West and A.A. Nabi. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. Journal of Range Management 34:259-264.

Taylor, A.H. and C.N. Skinner. 1998. Fire history and landscape dynamics in a late successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management 111:285-301.

Touchan, R., T.W. Swetnam, and H.D. Grissino-Mayer. 1995. Effects of livestock grazing on pre-settlement fire regimes in New Mexico. In: J.K. Brown, R.W. Mutch, C.W. Spoon, and R.H. Wakimoto (eds.). Proceedings: Symposium on Fire in Wilderness and Park Management. Missoula, MT. General Technical Report INT-GTR-320. Ogden, UT: USDA Forest Service, Intermountain Research Station. pp. 268-272.

Van Horne, M.L. and P.Z. Fulé. 2006 Comparing methods of reconstructing fire history using fire scars in a southwestern United States ponderosa pine forest. Canadian Journal of Forest Research 36:855-867.

Ward, D.J., B.B. Lamont, and C.L. Burrows. 2001. Grasstrees reveal contrasting fire regimes in eucalypt forest before and after European settlement of southwestern Australia. Forest Ecology and Management 150:323-329.

Westerling, A.L. and T.W. Swetnam. 2003. Interannual to decadal drought and wildfire in the western United States. EOS Transactions, American Geophysical Union 84:545-560.

Whitlock, C., S.L. Shafer, and J. Marlon. 2003. The role of climate and vegetation change in shaping past and future fire regimes in the northwestern US and the implications for ecosystem management. Forest Ecology and Management 178:5-21.

Wiens, J.A. 1989. Spatial scaling in ecology. Functional Ecology 3:385-397.

Williams, G.W. 2004. American Indian fire use in the arid west. Fire Management Today. 64:10-14.

Wright, H.A. and A.W. Bailey. 1982. Fire Ecology: United States and Canada. John Wiley and Sons, New York.

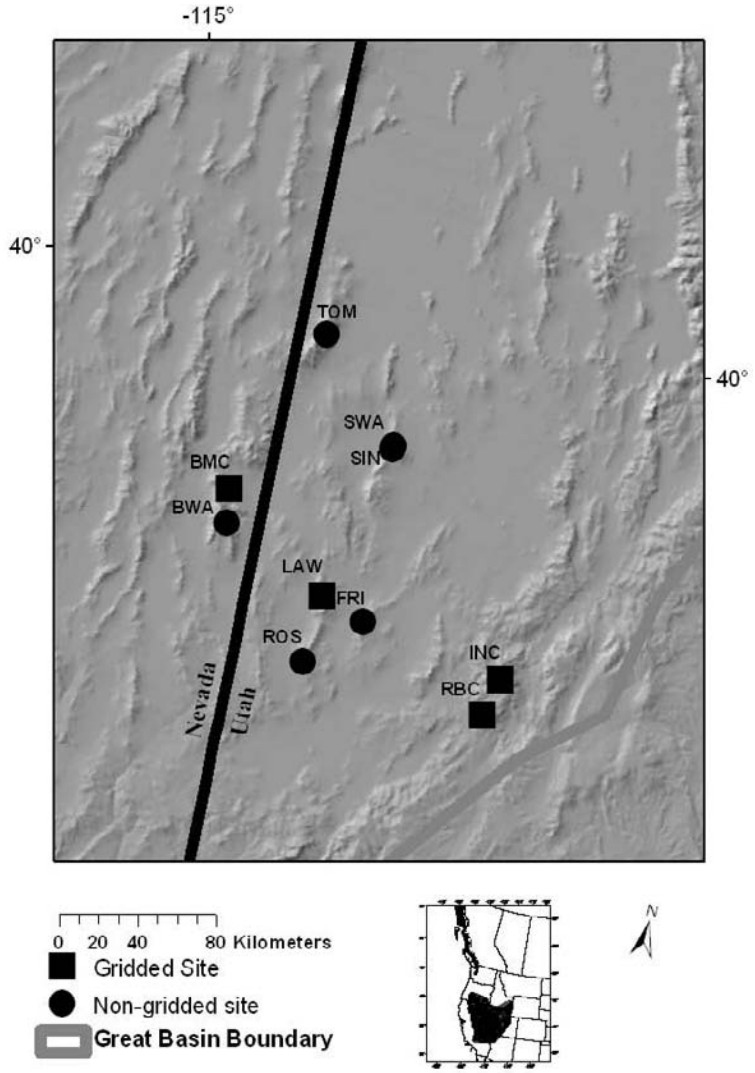

Fig. I.1. Map of eastern Great Basin mountain ranges with locations of 10 fire history sites. Study sites include Right Fork of Beaver Creek (RBC) and Indian Creek Canyons (INC) on the Tushar Mountains, Frisco Peak (FRI) on San Francisco Mountain, Lawson Cove (LAW) and Rose Spring Canyon (ROS) on the Wah Wah Range, Burnt Mill Canyon (BMC) and Big Wash (BWA) on the South Snake Range, Sinbad Springs (SIN) and Swasey Mountain (SWA) on the House Range, and Tom's Creek Canyon (TOM) on the Deep Creek Range. Sites represent a mix of gridded and targeted sampling strategies.

CHAPTER 1 – COMPOSITE AND SINGLE-TREE FIRE CHRONOLOGIES FROM HETEROGENEOUS LANDSCAPES: STRATEGIES FOR ESTIMATING POINT MEAN FIRE INTERVAL

Brief Summary: The effect of sample area size on fire frequency statistics generated from fire-scarred trees was investigated on four mixed-conifer forest sites in the eastern Great Basin. Results suggest that composite records from ½-ha sample areas and adjusted single-tree records could be used to estimate historic point fire interval statistics.

Abstract. Fire-induced scarring of trees provides temporally precise and spatially fixed proxy records of non-lethal fire. Single-tree chronologies are incomplete records of point fire history resulting from fires that do not produce scars and to scar erosion. Composite chronologies are used to correct for these errors of omission but are subject to increasing risks of errors of false inclusion associated with increasing sample area size. I used four eastern Great Basin fire chronologies to investigate the effect of sample area size on composite mean fire interval derived from heterogeneous landscapes. Composite fire frequency statistics were calculated for eight sets of six nested sample areas ⅛ to 128 ha in size, and six additional ⅛ and ½-ha nested pairs. Across nested series, fire frequency differed from 2 to 11 fold between ⅛ and 128-ha sample areas. An optimal spatial scale of ½ ha offers the best compromise for minimizing the competing risks of errors of omission and those of false inclusion, and thus provides a defensible estimate of point mean fire interval. Mean single-tree and 'best-tree' fire totals were 48 and 72% of the ½-ha composite total, respectively. Restricted-area (½ ha) composite and single-tree chronologies provide reasonable means for estimating point fire frequency across heterogeneous landscapes.

Additional key words: fire history, mixed-conifer forests, Great Basin

Introduction

Fire regimes for most western North American dry pine and mixed conifer forests changed dramatically after exploitation by Euro-American settlers (Cooper 1960; Covington and Moore 1994; Arno et al. 1997; Keane et al. 2002a). Quantifying the nature of pre-settlement fire regimes and the magnitude and consequences of post-settlement fire regime change is essential for developing science-based restoration and management strategies for affected forests (Fulé et al. 1997; Brown et al. 2001; Heyerdahl et al. 2001; Keane et al. 2002b; Swetnam 2005; Baker et al. 2006; Sherriff and Veblen 2006). Specifically, detailed information regarding historic patterns of fire frequency, severity, and extent and how these parameters varied at multiple spatial and temporal scales are needed (Taylor and Skinner 1998; Brown and Sieg 1999; Kaufmann and Huckaby 2000; Bekker and Taylor 2001; Heyerdahl et al. 2001; Morgan et al. 2001; Baker 2006; Falk et al. 2007; Beaty and Taylor 2008). Tree-ring based fire chronologies, or histories, are valuable tools for developing reasonable estimates of these measures of past fire regimes.

Fire-induced scars on the boles of trees producing annual growth rings provide temporally precise and spatially fixed proxy records of non-lethal fire. Many tree species (especially conifers) form fire scars under suitable conditions. Species which develop thick insulating bark are adapted to survive the frequent, repeated fires characteristic of low severity fire regimes and are thus capable of recording multiple fires through time. This protective adaptation generally inhibits scarring on older, un-scarred trees (Vines 1968). Consequently, many trees that experience the same fires as nearby fire-injured trees never scar. Although fire scars may occur on only a fraction of the trees in a stand, the probability of scarring increases dramatically after initial scar formation for trees that might otherwise be resistant to scarring.

However, even trees predisposed by existing wound surfaces to new scar formation often fail to form scars in response to some fires that burn at their base (Dieterich 1980; Dieterich and Swetnam 1984). Evidence of fire may also be lost as scar-bearing wood is consumed by subsequent fires or lost through processes of scar erosion. Consequently, it must be assumed that single-tree fire chronologies are incomplete records of surface fire for any reference period (Dieterich 1980; Dieterich and Swetnam 1984; Baker and Ehle 2001).

Investigations of past surface fire patterns at the stand to watershed scale typically involve the collection of scar-bearing cross-sections from several trees or tree remnants. Fire-scar records are most frequently used to assess the frequency of non-lethal fire although they are also useful when investigating fire severity and extent. Generally, no attempt is made to insure that all fire-scarred trees within the area of interest are sampled (for an exception see Van Horne and Fulé 2006), rather sample trees are selected based upon their quality (i.e. number of fire scars, soundness of wood) and actual discovery. Because search protocols and effort vary among and within studies, the degree to which the fire records acquired from sampled trees represent the population of fire records available in the area of interest must also vary. Regardless of search and sampling strategies employed, once sample tree locations are geographically fixed on the landscape (e.g. UTM coordinates) the net result is a unique collection of partial fire records with known spatial distribution yet unknown completeness.

The need for defensible strategies for the amalgamation, analysis, and interpretation of these records into accurate descriptions of fire regime temporal and spatial variability continues to be cause for debate (Baker and Ehle 2001; Stevens and Collins 2004; Swetnam 2005; Baker 2006; Baker et al. 2006; Fulé et al. 2006; Kou and Baker 2006a,b; Van Horne and Fulé 2006). Composite fire chronologies are assembled by combining the individual records of

at least two but generally several fire-scarred trees and are used to provide more complete and often longer histories of non-lethal fire (Dieterich 1980). This practice provides an effective strategy for reducing the probability of errors of omission, or failing to identify and include one or more fire years, and has become a common practice in the investigation of historic fire patterns. However, the fire frequency statistics associated with composite fire chronologies vary as a function of the size of the sample area and the relative abundance of small fires. This led Baker and Ehle (2001) to conclude that the composite mean fire interval (MFI) is unsuitable as a measure of fire frequency because of this dependency upon sample area size. They argued that, as it is typically used composite MFI overestimates the population MFI across the landscape. In a study using data from a ponderosa pine (*Pinus ponderosa* Douglas ex Lawson and Lawson) forest, Falk and Swetnam (2003) observed that composite MFI "was strongly scale dependent" and suggested that the relationship between MFI and sample area size could be predicted in a log-linear fashion. They suggested that the slope of the composite MFI to sample area size relationship might be used as an indicator of mean fire size or fire synchronization at different spatial scales. In another ponderosa pine study utilizing a complete census of all fire-scarred trees from a 1-km^2 study site, Van Horne and Fulé (2006) observed a 50 percent reduction in composite MFI as sample area size increased from 4 to 100 ha. Thus, when the objective is to estimate fire frequency with high spatial resolution (point MFI), amalgamation of individual, spatially-dispersed fire records increases the risk for errors of false inclusion, or including one or more fire years in which fires were not common for all points of the area in question. In summary, composite fire chronologies represent a strategy for reducing errors of omission but increase the risk for errors of false inclusion. Errors of omission result in over-estimation of the length of fire-free intervals while errors of false

inclusion have an opposite effect. In addition, composite fire frequency estimates derived from larger landscapes conceal useful information regarding spatial variability in point fire frequency.

Filtering from the composite record those fire years not represented on at least a minimum percentage of sampled trees (typically 10 or 25%) is a frequently used method for reducing the risk of overestimating fire frequency (e.g. Brown and Sieg 1996; Swetnam and Baisan 1996; Stephens and Collins 2004). A weakness of this approach is that there is no accounting for the spatial distribution of the recording trees used in the composite. Subsequently, different-sized fires which produce scars on a high percentage of closely-grouped trees may not be distinguishable from those that scar a similar number of trees spread across a wider landscape (Falk and Swetnam 2003). Filtering modifies estimates of fire frequency by eliminating from the composite record small, poorly recorded, poorly preserved, or poorly sampled fire events. Conversely, what is often needed is a more accurate measure of the variability in point fire frequency, a point being defined as an area equal to the average area occupied by a single mature tree because that is the minimum area in which a fire can be recorded. Filtering composite records may yield estimates that are similar to what is predicted for point fire frequency. However, the accuracy of results are difficult to determine because the process is indirect and eliminates from the record relatively small fires rather than directly measuring all fires that cross individual points on the landscape. Furthermore, filtering appears to support an assumption of uniform fire frequency across the landscape while knowledge of fire frequency variability in a population of points could yield critical insights into fire regime spatial dynamics.

My objective in this study is to address questions pertinent to the development of point MFI estimates from the partial fire records recovered from individual fire-scarred trees. In contrast to work by Falk and Swetnam (2003) and Van Horn and Fulé (2006) which addressed related questions using fire-scar records collected from southwest ponderosa pine forests located on relatively uniform topography, this study will utilize data collected from sites that are topographically and vegetationally more variable. Subsequently, one might anticipate greater spatial variability in fire regime and in the preserved fire record associated with heterogeneous environments than what would be predicted for more uniform environments. This variability might also be expected to result in greater sensitivity to different sampling and compositing strategies for estimating point fire frequency. My first questions address spatial scaling of sample areas when selecting trees to include in composite fire chronologies. Specifically, what is the optimal spatial scale, or sample area size, that best balances the competing risks of errors of omission against those of false inclusion and what impact does sample-tree number have on the optimal spatial scale? Any resolution of these questions will enable researchers and managers to better assess the utility of existing and future composite fire records for estimating point fire frequency. Additional questions consider the potential of using single-tree fire chronologies and associated frequency statistics to generate estimates of point fire frequency. Specifically, how well does the mean or 'best' single-tree fire record predict the spatially-constrained composite record? Here I define best trees as those identified from a set of spatially-proximate sample trees that record the greatest number of fires during a period of interest. Answers to this question have value for developing strategies for estimating fire frequency on landscapes where salvageable records from fire-scarred trees are uncommon

or widely dispersed, resulting in insufficient clusters of proximal fire-scarred trees for assembling spatially-constrained composite chronologies.

Study Sites

I collected samples from fire-scarred trees at four eastern Great Basin (USA) study sites located on three, north-to-south oriented mountain ranges (Fig. 1.1). Two sites, Burnt Mill Canyon (BMC) and Big Wash (BWA) are located 20 km apart in Great Basin National Park in east-side drainages of the South Snake Range, White Pine County, Nevada. Maximum elevation is 3,982 m (Wheeler Peak) with substantial portions of the range above 3,000 m. The BMC site (39° 2’ N 114° 16’ W) includes approximately 200 ha in the Burnt Mill Canyon and Mill Creek drainages. Elevations range from 2,560 to 2,900 m and slopes vary from 5 to 50%. The BWA site (38° 52’ N 114° 14’ W) starts on the park boundary on the southern edge of the Big Wash South Fork drainage and follows a National Park Service trail west for approximately 700 m. This site is located on a relatively uniform north to northeast aspect somewhat dissected by a series of shallow draws. Elevation is 2,490 to 2,560 m and slopes vary from 10 to 30%.

Two additional sites administered by the USDI Bureau of Land Management are located 65 and 95 km southeast of BWA on the north Wah Wah and the San Francisco Mountain Ranges in Utah’s Millard and Beaver Counties, respectively. Maximum elevation in the north Wah Wah range is 2,737 m. Considerable rock is exposed on ridge tops, as cliff faces of 1 to 30 m in height, and as talus slopes creating significant barriers to surface fire spread. The LAW study site (38° 37’ N 113° 34’ W) ranges in elevation from 2,240 to 2,630 m and

includes approximately 400 ha across two branches of the Lawson Cove drainage. At 2,944 m, Frisco Peak is the highest point of the San Francisco Mountains. Talus slopes near and at the summit are common and likely function as significant barriers to surface fire spread. The FRI study site (38° 32’ N 113° 17’ W) ranges in elevation from 2,580 to 2,770 m and follows the ridgeline of this range north of Frisco Peak.

Vegetation on north and east facing slopes at all sites is primarily mixed-conifer forest. Important tree species include Douglas-fir (*Pseudotsuga menziesii* var. *glauca* (Beissner) Franco), white fir (*Abies concolor* (Gordon & Glendinning) Lindley ex Hildebrand), ponderosa pine (*Pinus ponderosa* var. *scopulorum* Englemann) and lesser amounts of limber pine (*Pinus flexilis* James). At LAW and FRI, Douglas fir is uncommon and limber pine is absent. Stands of Great Basin bristlecone pine (*Pinus longaeva* D.K. Bailey) are present near these two sites with a few individuals scattered within the WAH site. Stands are generally mixed and 0.1 to 2.0 ha-sized patches dominated by single species are common. Based upon the abundance of observed stumps, the impact of Euro-American logging varies by site from minimal, represented by the removal of a handful of easily accessed ponderosa pine trees in drainage bottoms of the LAW site, to extensive, represented by the harvest of more than 80% of mature ponderosa pines at BMC. There is little or no evidence that other species were commercially harvested at any site. Where trees were harvested, secondary growth dominated by white fir and Douglas fir is abundant. Charred snags and log fragments are common throughout all sites with the exception of some old-growth pinyon (*Pinus monophylla* Torrey & Frémont)-juniper (*Juniperus osteosperma* (Torrey) Little and *J. scopulorum* Sargent) stands within the LAW site.

South and west facing slopes are occupied by mountain sagebrush (*Artemisia* spp. L.)-steppe, mixed mountain shrub, curlleaf mountain mahogany (*Cercocarpus ledifolius* Nuttall in Torrey & Gray), and pinyon-juniper communities. Mixed stands of pinyon and juniper occupy all aspects at lower elevations at the LAW site. Somewhat sharp ecotones among community types reflect topographic control (elevation and aspect) of plant community assemblages and corresponding fuels matrices. Fire-scarred trees are primarily ponderosa pine, however scattered, fire-scarred representatives of most tree species are also present.

In short, all four sites have stands of mixed conifer forest on north and east-facing slopes and sufficient fire-scarred trees to support a study evaluating the practice of combining individual, tree-ring-based fire records into composite fire histories. All sites but BWA also have significant non-forested elements (mountain shrublands), primarily on south and west aspects. BMC and BWA are somewhat more mesic than LAW and FRI suggesting greater capacity for post-fire fuel recovery for these sites and less restrictive topographic barriers to surface fire spread.

Field Methods

Systematic grids (500-m intervals) were superimposed over the BMC and WAH study sites to stratify sampling effort (Brown et al. 2008). For these sites, a thorough search for fire-scarred trees (live, snags, stumps and logs) was completed for an area of at least 2 ha (80 m radius) centered on each grid point. One to several cross-sections were removed from scarred surfaces (cat-faces) of most fire-scarred trees located within search radii using chainsaws and standard methods (Arno and Sneck 1977). A few (estimated <20 per study site) fire-scarred

live trees and standing snags located near grid points were not sampled due to safety concerns. Additional fire-scarred trees were sampled opportunistically as they were encountered between grid points. Inter-point search and sampling effort was considered aggressive involving one to four individuals for 1 to 4 hours per grid-point and were focused primarily on those portions of the landscape occupied by mixed conifer forest where fire-scarred trees proved to be most abundant. Although trees with multiple scars were selected preferentially, numerous trees with one to three scars were also sampled. These watershed-scale grids encompassed a broad range in elevation, however, only the mid-elevation portions where fire-scarred trees were at the highest densities are included in the BMC and LAW study areas considered here. A targeted search for fire-scarred trees at the BWA site was limited to a 300-m wide belt centered over the above-described Park Service trail. A similar search effort at the FRI site extended up to 200 m on either side of the Frisco Peak ridgeline road for a distance of 1800 m. Although numerous search hours were spent at each of these locations, the effort was not systematic or comprehensive in nature.

Universal Transverse Mercator (UTM) coordinates were determined at the time of sampling (2002-2003) for most sampled trees at BMC and LAW using hand-held GPS units accurate to within 15 m. Trees at BWA and FRI and a few previously sampled trees at BMC and LAW were relocated (2003-2005) using field notes and their UTM coordinates determined as described above. The locations of seven previously sampled trees could not be resolved resulting in the elimination of these from use in this study. Species, condition (live, snag, stump, or log) and number of apparent scars were noted for each tree at the time of sampling. Samples were labeled and wrapped in shrink-wrap for field preservation.

Laboratory Methods

Sample pieces were glued together and to plywood as necessary for stabilization and surfaced using appropriate combinations of band saw, power planer, belt sander, and hand sanding until cell structure in individual rings became visible using a binocular microscope. Each sample was independently cross-dated and cross-checked, by a minimum of two analysts using a combination of locally and regionally-developed master ring-width chronologies (skeleton plots) and lists of marker (narrow) years (Stokes and Smiley 1968). Specimens that could not be clearly cross-dated were excluded from further analysis. Fire scars that could not be dated with annual accuracy and injuries of questionable causation were also excluded.

Intra-ring position of each fire scar was recorded as early, middle, and late early-wood; late-wood; ring-boundary; or unknown. Ring-boundary scars can result from either late season fires that occur after ring growth is complete or early season fires before ring growth is initiated. By convention, ring-boundary scars are assigned to the following year (early-season fire) in the southwest United States (Baisan and Swetnam 1990) and to the preceding year (late-season fire) at northern latitudes (Heyerdahl et al. 2001; Schmidt et al. 2002) based upon the seasonality of modern fires and the relative abundance of early-wood and late-wood scars in fire-scarred specimens. The intermediate geographic position of these study sites suggests the need for caution before adopting either of these conventions as a rule. Therefore, fire years for ring-boundary scars were assigned to pre-boundary years when evidence of a late season fire (late early-wood or late-wood scar) in that year was found in at least one tree from the study area and similarly to the post-boundary year when evidence of an early season fire (early or middle early-wood scar) was collected from the study area for the same year. Based upon results from a pilot study (Kitchen and McArthur 2003), fire year was assigned to the post-

boundary year (Southwest convention) when fire year could not be conclusively determined using these criteria. Fire seasonality was classified as unknown when it could not be accurately determined due to narrowness of rings or eroded fire scar surfaces related to subsequent fire, rot, or the presence of insect galleries.

Recording years were assigned to each tree from the date of the first injury to the last year where the scar surface remained intact. Years were excluded as recording years if the ring structure indicated that the area of injury had healed over completely (common for very young trees with rapid growth rates) or had been burnt-off or otherwise damaged or lost to the point that evidence of potential fire scars would likely be missing.

A uniform period of analysis (POA) from 1650 to 1850 was selected after all samples had been dated to facilitate across-site comparisons (Fig. 1.2). All fire count and fire interval statistics considered here apply to this 201-year time frame only. Trees that did not have at least one fire or a minimum of 50 years as a recording tree (e.g. 1650 to 1699) during the POA were excluded from further analysis.

Sample tree locations were plotted on site maps using UTM coordinates. One to three sets of six sample areas each were located on each site map for a total of eight sets. Sample areas were circular and nested with radii of 20, 40, 80, 160, 320, and 640 m resulting in corresponding areas of approximately ⅛, ½, 2, 8, 32, and 128 ha. The ⅛-ha sample areas were positioned first on site maps within areas of relatively high sample tree density and include clusters of 2 to 10 sample trees with multiple fire scars corresponding to the POA. Larger sample areas were then positioned to include all trees located in smaller sample areas assigned to the same nested series and the maximum possible number of additional sample trees on the

study site regardless of topographic variation. Six additional nested pairs of ⅛ and ½-ha sample areas were selected for a more complete evaluation of composite chronologies generated from these smaller topographic units.

Fire totals corresponding to the POA were determined for each tree and sample area. I identified 'best' trees as those individual trees within each sample area with the most fires recorded during the POA. I computed composite fire chronologies and interval statistics for each sample area using program FHX2 (Grissino-Mayer 2001). Single-tree mean fire interval (MFI) was computed for all sample trees with two or more fire events during the POA. Composite interval statistics include MFI and Weibull Median Probability Interval (WMPI). WMPI is the estimated interval at which there is a 50% probability of a longer (or shorter) interval, based upon the population of intervals analyzed and is considered more appropriate than MFI as a measure of central tendency in fire frequency data due to non-normal distribution of intervals around the mean (Swetnam and Baisan 1996, Grissino-Mayer 1999). When fire interval data are normally distributed then MFI and WMPI are the same. Although MFI and WMPI are both reported, I report MFI for discussions of fire frequency here because it is the measure of fire frequency most consistently reported in the literature (Van Horne and Fulé 2006; Baker and Ehle 2001). I did not test for statistical significance of differences among means because of spatial autocorrelation and lack of independence in the sampling design (Van Horne and Fulé 2006).

Results

One to several fire-scarred samples was secured from each of 214 trees. Of these, samples for three trees could not be cross-dated and the sample for one was lost. Chronologies for 30 of the remaining trees did not meet the POA criteria and were not included in this analysis leaving a total of 180 study trees from the four study sites (Table 1.1). Tree species were 151 ponderosa pine, 12 limber pine, eight Douglas-fir, seven white fir, and two Rocky Mountain juniper sampled as 62 live trees, 46 stumps, 48 snags, and 24 logs. Sample depth varied among study sites through time (Fig 1.3). Trees from LAW and FRI tended to be older with earlier first fire dates (Figs. 1.2 and 1.3). Seventy-nine percent of FRI and 45% of LAW trees were recording at the beginning of the POA (1650) while only 6% of BMC trees were recording by this date (Fig. 1.3). No BWA trees were in recording status until 1694. At the end of the POA the proportion of sample trees per site still in recording status varied from 60 to 83%.

Each site experienced numerous fires during the POA (Fig. 1.2) with a mean of 44 fire years per site (Table 1.1), representing 22% of all possible years. Across all sites, the mean length of the single-tree recording period was 139 years. Among all trees, the number of fires per tree during the POA varied from 0 (recording for at least 50 years but no fire scars during POA) to 12 with a mean of 3.8 fires. Mean single-tree MFI across all sites was 35.4 years. On average, more fire scars and shorter fire intervals were observed on trees sampled at BMC and BWA than from those sampled at LAW and FRI.

Total tree and fire numbers and fire interval statistics for composites records from the eight nested series and six additional nested pairs are listed in Tables 1.2 and 1.3, respectively.

Because MFI and WMPI values differed by less than 2 years in 50 of 60 paired comparisons, I considered them ecologically unimportant. Differences in numbers of POA-fire years from smallest to largest sample areas varied from 14 to 33 (Table 1.2) with a mean increase of 23 fire years. Similarly, changes in MFI varied from -5.8 to -48.8 years with a mean change of -17.1 years representing a 69% average decrease in the length of fire free intervals associated with increased sample area size. Tree and fire numbers and fire frequency statistics for the six nested pairs in Table 1.3 are within the range of those for the eight, ⅛ and ½-ha paired sample areas listed in Table 2, suggesting that all 14 pairs can be considered in a single group when evaluating variability in these smaller sample-area size classes. Variability in MFI was greatest for ⅛ and ½-ha sample areas and became incrementally smaller with increasing sample area size (Fig. 1.4). Comparisons in fire number and MFI between ⅛ and ½-ha sample areas reveal substantial increases in fire number (MFI decrease) associated with the ½-ha sample areas for some pairs but not others (Tables 1.2 and 1.3). In one case (LAW-2) the addition of a single sample tree with two new fire years resulted in an increase in MFI because the new fire years occurred later than the last fire in the ⅛-ha chronology, creating two new intervals, one of which was long enough to result in an increase in the composite MFI for the sample area.

Commonality of fire record among closely-associated individual trees (inter-tree distance ≤ 40 m) varied considerably. When two such trees recorded multiple fires over the same time period, the proportion of recorded fire-years shared by both trees was typically between 50 and 100%. Lack of complete agreement was considered evidence of incomplete recording of fire events by one or both trees. Record completeness was improved by amalgamation of the fire records of closely associated trees into composite chronologies. Commonality among eight sets of closely associated composite records (maximum inter-tree

distance 75-140m) varied between 0 and ~60% and was not correlated with the number of trees or fires evaluated in each comparison (Table 1.4).

Across all sites, 'best' trees for ½-ha sample areas recorded from three to 12 fires during the POA (Table 1.5). The best-tree mean was 72% of the composite for total fires in this sample-area size class. The mean number of fires recorded by single trees varied from 2.0 to 7.6 across all ½-ha sample areas and represents 48% of the mean composite fire total. In general, the completeness of record from individual 'best' trees or single-tree means decreased as the number of fires in the composite increased (Fig. 1.5).

Discussion

A primary objective of this study was to identify an optimum spatial scale from which tree-ring based records of surface fire could be combined into composite records that could in turn be used as reasonable estimates of point fire chronologies. I assumed that this optimum spatial scale would fall within the size range of sample areas tested and that the methods used would produce evidence of both errors of omission and of false inclusion. This evidence could then be used to identify a spatial scale that would best balance the risks associated with the two sources of error.

Errors of omission are most likely to occur when sample tree number is low. Sample tree number was ≤ 5 for 11 of 14, ⅛-ha sample areas and for 7 (50%) of the ½-ha sample areas (Tables 1.2 and 1.3) suggesting that evidence for errors of omission should be strongest for these small sample areas with the fewest trees. Subsequently, I expected an inverse relationship between tree number and MFI for these small sample areas. However, tree number to MFI

correlation values for ⅛ and ½-ha sample areas (Fig. 1.6*a*,*b*) were not significant ($p \leq 0.05$). The considerable scatter in MFI associated with low tree numbers implies that completeness of record was difficult to predict from tree number alone and that errors of omission were likely in some, but perhaps not in all cases. This point is reinforced by instances where composite MFI was reduced by as much as 45% after inclusion of the few trees located on the additional ⅜ ha of the ½-ha sample areas (see Table 1.2; BWA-2 and Table 1.3; BMC-3 and 4, and FRI-2). At the same time, composite MFI for ⅛ and ½-ha sample areas with as many as five additional trees differed by less than 2 years in six of 14 comparisons. I hypothesized that at 2 ha, sample area size would be large enough that sample tree number (mean = 9) would be sufficient to minimize the risk for errors of omission. If this is true, then the probability of discovering more fire years (shorter MFI) on additional fire-scarred trees from within the sample area would be low. My hypothesis of the adequacy of 2-ha sample areas to reduce the risk of errors of omission is supported by the lack of correlation between tree number and MFI for the 2-ha sample areas (Fig. 1.6*c*). Stronger correlations between tree number and MFI for 8, 32, and 128-ha sample areas (Fig. 1.6*d*-*f*) are unrelated to completeness of record at the point scale (errors of omission) but are instead an indication that years with novel small fires are added to the composite when more sample trees are added at the larger spatial scales (Falk and Swetnam 2003; Van Horne and Fulé 2006). Thus, with sample areas larger than 2 ha there is an unacceptable risk for errors of false inclusion in this study.

Across the eight nested series, fire number differed ~2 to 11-fold and MFI differed ~2 to 10-fold between the ⅛ and 128-ha sample areas (Table 1.2). The composite MFI to sample area size relationship was essentially linear for sample areas of 2 to 128 ha when plotted on a natural log scale (Fig 1.4), suggesting that, within this spatial range, aggregation of new fire

years proceeded at a relatively constant rate as the sample area expanded, as predicted by Falk and Swetnam (2003), and indirectly supports the idea that the optimal spatial scale was ≤ 2 ha in size. New trees or clusters of trees included in composite records as sample-area perimeter expanded typically lacked evidence of fire years common among trees located within short distances. Evidence from eight examples (Table 1.4) suggests that sample areas of ½ to 2 ha (diameter 80-160 m) are sufficiently large to result in errors of false inclusion. In summary, the risk for errors of omission when using the ⅛-ha sample areas was unacceptably high in many cases due to too few sample trees, however, I was not able to predict the acuteness of the risk from sample-tree number alone. Increased risk for errors of false inclusion was manifest at inter-sample distances as short as 80 m and was often excessive at spatial scales of about 2 ha. Thus, in the context of the topographically and vegetationally variable landscapes sampled in this study, a defensible approach for generating composite records as estimates of point MFI would be to include the greatest number of recording trees possible from sample areas limited to ~½ ha (40 m radius). An alternative approach would be to calculate fire interval statistics based upon two sample area scales (e.g. ⅛ and 2 ha) and use the results to bracket a range in the estimate of fire frequency (Baker and Ehle 2001). Additional research is needed to test these conclusions using existing or new data sets from other geographic settings.

Analysis of spatial variability in fire frequency requires that the density and distribution of fire frequency estimates be sufficient to represent the landscape of interest. Fire-scarred tree distribution is typically uneven on many forested landscapes, restricting spatial representation by composite-based fire frequency estimates. Suitable fire-scarred trees are often too widely scattered for amalgamation into spatially-constrained estimates of point MFI. Public policy, resource limitation, and other circumstances can also make it inappropriate or infeasible to

sample at the intensity required to generate a representative array of point fire frequency estimates using the restricted-composite method. Defensible point fire frequency estimates based upon individual tree fire records may provide alternative measures of fire frequency when composite-based estimates are not possible or practical, and can be used to fill gaps on landscapes where composite-derived estimates are spatially limited.

Because single-tree fire records have a high risk for errors of omission, single-tree metrics should be adjusted before application as estimates of point MFI. In this study the average single-tree fire number was 48% (range of 32 to 92%) of the ½-ha composite fire total, respectively (Table 1.5). Assuming similar sampling strategies, single-tree MFI values could converted to estimates of PMFI be adjusted using a ~0.5 multiplier as a correction factor to convert single-tree MFI values to estimates of point MFI. Alternatively, estimates might be bracketed between high and low values in recognition of the broad range in completeness-of-record among single-tree means. High and low multipliers of ~0.3 and 0.9 might be appropriate based on this study. Thus a single-tree MFI of 35 years (overall mean for this study) could be adjusted to a single estimated value of 17.5 years or a bracketed estimated range of 10.5-31.5 years using the above multipliers.

Because 'best' trees contain more complete records of fire than average single trees they represent a reduced risk for errors of omission. The mean and range for best-tree fire numbers were 72 and 47-100% of the ½-ha composite total (Table 1.5). These values serve as the basis for correction factors for this class of single-tree records. Thus a best-tree MFI of 25 years converts to a point MFI estimate of 18 years (25 x 0.72) or an estimated range of 12-25 (25 x 0.47-25 x 1.0) years. Although best-tree estimates have the advantage of reduced uncertainty in comparison to estimates based on single-tree means, their application may be

more limited. Problems stem from the difficulty in knowing what constitutes a ‘best’ tree and how to recognize it when it is not one of a group of sampled trees (best compared to what?). If the ‘best’ status is determined by comparison among a closely-associated group of trees that have already been sampled then a restricted-area (i.e. ½-ha) composite would usually be more appropriate. However, if the number of samples taken must be limited, considerable triage of candidate trees is possible through careful examination of wound surfaces prior to sampling, allowing the identification and sampling of probable ‘best’ trees in a field setting. Conversely, fire-scar records that might otherwise qualify as ‘best’ trees based upon the relative number of fires recorded, length of fire intervals or length of fire record are often isolated from other fire-scarred trees that might be used for comparison. Lacking better defined criteria, these trees would have to be classified as ‘best’ trees based upon the judgment and experience of investigators or be treated without the increased precision afforded best-tree status. In either case, caution must be used when this approach is taken so that fire frequency estimates are restricted to the area in close proximity of sampled trees. Additional research is needed to determine how single-tree correction factors might vary with sampling strategy and biophysical setting.

In conclusion, both spatially-constrained composite and single-tree fire histories have application in generating estimates of point fire frequency. The optimal spatial scale that balanced the competing risks associated with errors of omission with those of false inclusion in this study was relatively small at ~½ ha. Composite chronologies based upon multiple fire-scarred trees and this sample area size are defensible as estimates of point MFI. Simple estimates or bracketed ranges of fire frequency based upon single-tree fire chronologies will allow greater spatial representation of fire frequency variability across heterogeneous

landscapes. This study proposes approaches to generating tree-ring-based estimates of fire frequency that are testable and open to refinement through additional research.

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References

Arno SF, Sneck KM (1977) A method for determining fire history in coniferous forests of the Mountain West. USDA Forest Service, Intermountain Forest and Range Experiment Station General Technical Report INT-42. (Ogden, UT)

Arno SF, Smith HY, Krebs MA (1997) Old growth ponderosa pine and western larch stand structures: influences of pre-1900 fires and fire exclusion. USDA Forest Service, Intermountain Research Station Research Paper INT-RP-495. (Ogden, UT)

Baisan CH, Swetnam TW (1990) Fire history on a desert mountain range: Rincon Mountain Wilderness, USA. *Canadian Journal of Forest Research* **20**, 1559-1569.

Baker WL (2006) Fire history in ponderosa pine landscapes of Grand Canyon National Park: is it reliable enough for management and restoration? *International Journal of Wildland Fire* **15**, 433-437.

Baker WL, Ehle D (2001) Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. *Canadian Journal of Forest Research* **31**, 1205-1226.

Baker WL, Veblen TT, Sheriff RL (2006) Fire, fuels, and restoration of ponderosa pine-Douglas fir forests in the Rocky Mountains, USA. *Journal of Biogeography* **33**, 1-19.

Beaty RM, Taylor AH (2008) Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. *Forest Ecology and Management* **255**, 707-719.

Bekker MF, Taylor AH (2001) Gradient analysis of fire regimes in montane forests of the southern Cascade Range, Thousand Lakes Wilderness, California, USA. *Plant Ecology* **155**, 15-28.

Brown PM, Sieg CH (1996) Fire history interior ponderosa pine communities of the Black Hills, South Dakota, USA. *International Journal of Wildland Fire*, **6**, 97-105.

Brown PM, Sieg CH (1999) Historical variability in fire at the ponderosa pine – northern Great Plains prairie ecotone, southeastern Black Hills, South Dakota. *Ecoscience* **6**, 539-547.

Brown PM, Kaye MW, Huckaby LS, Baisan CH (2001) Fire history along environmental gradients in the Sacramento Mountains, New Mexico: Influences of local patterns and regional processes. *Ecoscience* **8**, 115-126.

Brown P.M., Heyerdahl EK, Kitchen SG, Weber MH (2008) Climate effects on historic fires (1630-1900) in Utah. *International Journal of Wildland Fire* **17**, 28-39.

Cooper CF (1960) Changes in vegetation, structure, and growth of southwestern pine forest since white settlement. *Ecological Monographs* **30**, 129-164.

Covington WW, Moore MM (1994) Southwest ponderosa forest structure: changes since Euro-American settlement. *Journal of Forestry* **92**, 39-47.

Dieterich JH (1980) The composite fire interval: A tool for more accurate interpretations of fire history. In 'Proceedings of the fire history workshop'. (Tech coords MA Stokes, JH Dieterich) pp. 8-14 . (technical coordinators). USDA Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report RM-81. (Fort Collins, CO)

Dieterich JH, Swetnam TW (1984) Dendrochronology of a fire-scarred ponderosa pine. *Forest Science* **30**, 238-247.

Falk DA, Swetnam TW (2003) Scaling rules and probability models for surface fire regimes in ponderosa pine forests. In 'Fire, fuel, treatments, and ecological restoration'. (Eds Omi PN, Joyce LA) pp. 301-317. US Forest Service, Rocky Mountain Research Station General Technical Report RMRS-P-29. (Fort Collins, CO)

Falk DA, Miller C, McKenzie D, Black AE (2007) Cross-scale analysis of fire regimes. *Ecosystems* **10**, 809-823.

Fulé PZ, Covington WW, Moore MM (1997) Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. *Ecological Applications* **7**, 895-908.

Fulé PZ, Heinlein TA, Covington WW (2006). Fire histories in ponderosa pine forests of Grand Canyon are well supported: reply to Baker. *International Journal of Wildland Fire* **15**, 439-445.

Grissino-Mayer HD (1999) Modeling fire interval data from the American Southwest with the Weibull distribution. *International Journal of Wildland Fire* **9**, 37-50.

Grissino-Mayer HD (2001) FHX2 – Software for analyzing temporal and spatial patterns in fire regimes from tree rings. *Tree-Ring Research* **57**, 113-122.

Heyerdahl EK, Brubaker LB, Agee JK (2001) Spatial controls of historic fire regimes: a multiscale example from the interior west, USA. *Ecology* **82**, 660-678.

Kaufmann MR, Huckaby L (2000) Ponderosa pine in the Colorado Front Range: long historical fire and tree recruitment intervals and a case for landscape heterogeneity. In:

Crossing the millennium: integrating spatial technologies and ecological principles for a new age in fire management; proceedings, the Joint Fire Science Conference and Workshop. pp. 153-160. University of Idaho (Boise, ID)

Keane RE, Ryan KC, Veblen TT, Allen CD, Logan J, Hawkes B (2002a) Cascading effects of fire exclusion in Rocky Mountain ecosystems: a literature review. USDA Forest Service, Rocky Mountain Research Station General Technical Report RMRS-GTR-91. (Fort Collins, CO)

Keane RE, Parsons RE, Rollins MG (2002b) Predicting fire regimes at multiple scales. In 'Emulating natural forest landscape disturbances: concepts and applications: popular summaries'. Ontario Forest Research Institute, Ministry of Natural Resources. Forest Research Information Paper **149**, 23-25.

Kitchen SG, McArthur ED (2003) Ponderosa pine fire history in a southeastern Great Basin stand. In 'Proceedings of Fire Conference 2000. The First National Congress on Fire Ecology, Prevention, and Management'. (Eds Galley KEM, Klinger RC, Sugihara NG) pp. 152-156. Tall Timbers Research Station Misc. Publ. No. 13. (Tallahassee, FL)

Kou X, Baker WL (2006a) A landscape model quantifies error in reconstructing fire history from scars. *Landscape Ecology* **21**, 735-745.

Kou X, Baker WL (2006b) Accurate estimation of mean fire interval for managing fire. *International Journal of Wildland Fire* **15**, 489-495.

Morgan P, Hardy CC, Swetnam TW, Rollins MG, Long DG (2001) Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. *International Journal of Wildland Fire* **10**, 329-342.

Schmidt KM, Menakis JP, Hardy CC, Hann WJ, Bunnell DL (2002) Development of coarse-scale spatial data for wildland fire and fuel management. USDA Forest Service, Rocky Mountain Research Station General Technical Report RMRS-GTR-87. (Fort Collins, CO).

Sherriff RL, Veblen TT (2006) Ecological effects od changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. *Journal of Vegetation Science* **17**, 705-718.

Stevens SL, Collins BM (2004) Fire regimes of mixed conifer forests in the north-central Sierra Nevada at multiple spatial scales. *Northwest Science* **78**, 12-22.

Stokes MA, Smiley TL (1968) An Introduction to Tree-Ring Dating. University of Chicago Press. 73 pp. (Chicago, IL).

Swetnam TW (2005) Fire histories from pine-dominant forest in the Madrean Archipelago. pp. 35-43, In: Gottfried GJ, Gebow BS, Eskew LG, Edminister CB (comps) 'Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago II'. USDA Forest Service Proceedings RMRS-P-36. US Forest Service, Rocky Mountain Research Station Proceedings RMRS-P-36. (Fort Collins, CO).

Swetnam TW, Baisan CH (1996) Historical fire regime patterns in the southwestern United States since AD 1700. In: Allen CD (ed.) Fire Effects in southwestern forests-Proceedings of the second La Mesa fire symposium. pp. 11-32. USDA Forest Service, Rocky Mountain

Forest and Range Experiment Station General Technical Report RM-GTR-286. (Fort Collins, CO).

Taylor AH, Skinner CN (1998) Fire history and landscape dynamics in a late successional reserve, Klamath Mountains, California, USA. *Forest Ecology and Management* **111**, 285-301.

Van Horne ML Fulé PZ (2006) Comparing methods of reconstructing fire history using fire scars in a southwestern United States ponderosa pine forest. *Canadian Journal of Forest Research* **36**, 855-867.

Vines RG (1968) Heat transfer through bark, and the resistance of trees to fire. *Australian Journal of Botany* **16**, 499-514.

Table 1.1. Site summary information and single-tree fire statistics.

Mean recording years per tree are the mean number of years from the period of analysis (1650 to 1850) that individual trees were in recording status. Trees in recording status have recorded at least one fire, have an open scar surface, and tree-ring structure is sufficiently intact to detect fire scars where they occur. Single-tree fire records include mean fire interval (MFI) and range of MFI for all study trees from each site.

			Mean Recording Years/Tree	Single Tree Fire Record				
Site	Total Trees	Fire Years		Mean fires/tree	Fire number		MFI	MFI Range
					Min	Max		
BMC	53	46	119.0	4.5	1	12	36.8	12.2-154.0
BWA	25	35	124.1	6.2	3	12	20.6	12.9-40.3
LAW	78	68	155.4	3.0	0	9	39.4	14.0-99.0
FRI	24	26	148.2	2.7	0	7	47.4	17.0-156.0

Table 1.2. Composite tree, fire number and fire interval statistics for eight sample-area nested series.

Interval statistics include: mean fire interval (MFI), Weibull Median Probability Interval (WMPI), and minimum and maximum intervals within the period of analysis (1650-1850).

Site/ Nested series	Sample area size (ha)	Total number		Composite		Interval range (yrs)	
		trees	fire yrs	MFI	WMPI	min	max
BMC-1	1/8	10	17	10.4	10.0	3	21
	1/2	15	18	9.8	9.2	2	21
	2	15	18	9.8	9.2	2	21
	8	22	29	6.4	5.5	1	19
	32	27	36	5.1	4.3	1	19
	128	37	41	4.5	3.8	1	19
BMC-2	1/8	3	3	54.5		48	61
	1/2	6	4	44.7	45.0	25	61
	2	10	7	23.7	17.5	2	61
	8	11	8	20.3	15.6	2	61
	32	16	12	12.9	9.4	2	54
	128	37	33	5.7	4.9	1	19

Site/ Nested series	Sample area size (ha)	Total number		Composite		Interval range (yrs)	
		trees	fire yrs	MFI	WMPI	min	max
BWA-1	1/8	6	15	10.1	9.9	4	19
	1/2	7	15	10.1	9.9	4	19
	2	10	20	7.7	7.0	1	19
	8	15	28	5.4	5.0	1	11
	32	23	32	4.7	4.4	1	11
	128	25	35	4.3	4.0	1	10
BWA-2	1/8	4	10	14.9	14.4	2	25
	1/2	5	15	10.1	9.2	2	19
	2	7	15	10.1	9.2	2	19
	8	9	19	7.8	7.3	2	19
	32	22	30	5.1	4.6	1	13
	128	25	35	4.3	4.0	1	10
LAW-1	1/8	7	11	15.2	14.4	2	23
	1/2	11	11	15.2	14.4	2	23
	2	14	14	13.7	12.3	2	26
	8	18	17	11.1	9.5	2	26
	32	21	22	8.5	7.3	2	26
	128	37	35	5.2	4.1	1	23

Site/ Nested series	Sample area size (ha)	Total number		Composite		Interval range (yrs)	
		trees	fire yrs	MFI	WMPI	min	max
LAW-2	1/8	2	5	23.5	23.8	14	32
	1/2	3	7	30.3	26.8	11	77
	2	4	14	14.0	13.6	6	31
	8	4	14	14.0	13.6	6	31
	32	11	23	8.3	7.6	1	17
	128	22	38	5.1	4.6	1	11
LAW-3	1/8	2	5	25.5	25.4	15	36
	1/2	3	7	24.3	24.3	15	36
	2	5	9	18.3	17.2	3	35
	8	7	10	16.2	14.4	2	33
	32	13	12	14.5	12.9	2	33
	128	23	22	7.9	6.5	1	27
FRI-1	1/8	2	6	29.0	28.0	20	56
	1/2	2	6	29.0	28.0	20	56
	2	6	10	17.0	15.8	6	36
	8	9	16	12.1	10.8	2	29
	32	12	18	10.7	9	1	29
	128	17	20	9.6	8.1	1	29

Table 1.3. Composite tree, fire and fire interval statistics for six sample-area nested pairs.

Interval statistics include: mean fire interval (MFI), Weibull Median Probability Interval (WMPI), and minimum and maximum intervals within the period of analysis (1650-1850).

Site/ Nested series	Sample area size (ha)	Total number		Composite		Interval range (yrs)	
		trees	fire yrs	MFI	WMPI	min	max
BMC-3	1/8	3	12	12.7	12.5	5	24
	1/2	5	16	9.3	8.7	2	24
BMC-4	1/8	4	10	14.7	11.9	1	28
	1/2	6	14	11.5	9.2	1	28
LAW-4	1/8	2	4	31.0	31.7	24	35
	1/2	4	4	31.0	31.7	24	35
LAW-5	1/8	4	4	42.0	39.9	16	69
	1/2	6	5	31.5	331.5	16	44
LAW-6	1/8	2	12	15.8	15.9	8	25
	1/2	3	14	12.4	11.2	1	25
FRI-2	1/8	4	6	25.6	25.2	15	40
	1/2	7	10	14.2	11.8	2	36

Table 1.4. Composite fire-year commonality between proximal clusters of fire-scarred trees.

Total trees and fire numbers for eight pairs of fire-scarred tree clusters indicating the number of trees and fire years in each cluster. Number of common fires is the total fire-years in the POA shared by both members of a set. Cluster area was ≤ 1/2 ha. Maximum inter-tree distance is the greatest distance between any two trees from each paired cluster.

Site-Series ID	Tree number		Maximum inter-tree distance (m)	Fire number		Number of common fires
	Cluster			Cluster		
	A	B		A	B	
BMC-2	6	4	115	4	5	2
BMC-4	4	2	90	10	7	3
BWA-1	7	3	140	15	13	8
BWA-2	4	3	120	10	8	3
LAW-1	11	3	140	11	4	1
LAW-2	3	1	100	7	9	2
FRI-1	2	4	115	6	5	1
FRI-2	4	3	75	7	4	0

Table 1.5. Individual tree and composite (½-ha sample area) fire number comparisons.

The 'best' tree is that tree from among those assigned to each composite with the greatest number of fires recorded for the period of analysis (1650-1850). Single-tree mean is the mean number of fires per tree of those assigned to each composite.

Site-Series ID	Composite	Best tree	Single tree mean
	---------- Fire number ----------		
BMC-1	18	12	6.1
BMC-2	4	3	2.0
BMC-3	16	8	7.6
BMC-4	14	8	4.5
BWA-1	15	11	7.1
BWA-2	15	7	5.0
LAW-1	11	9	3.7
LAW-2	7	5	3.7
LAW-3	7	7	5.7
LAW-4	4	4	2.5
LAW-5	5	4	2.7
LAW-6	14	9	6.7

Site-Series ID	Composite	Best tree	Single tree mean
	---------- Fire number ----------		
FRI-1	6	6	5.5
FRI-2	10	6	3.7
mean	9.9	7.1	4.8

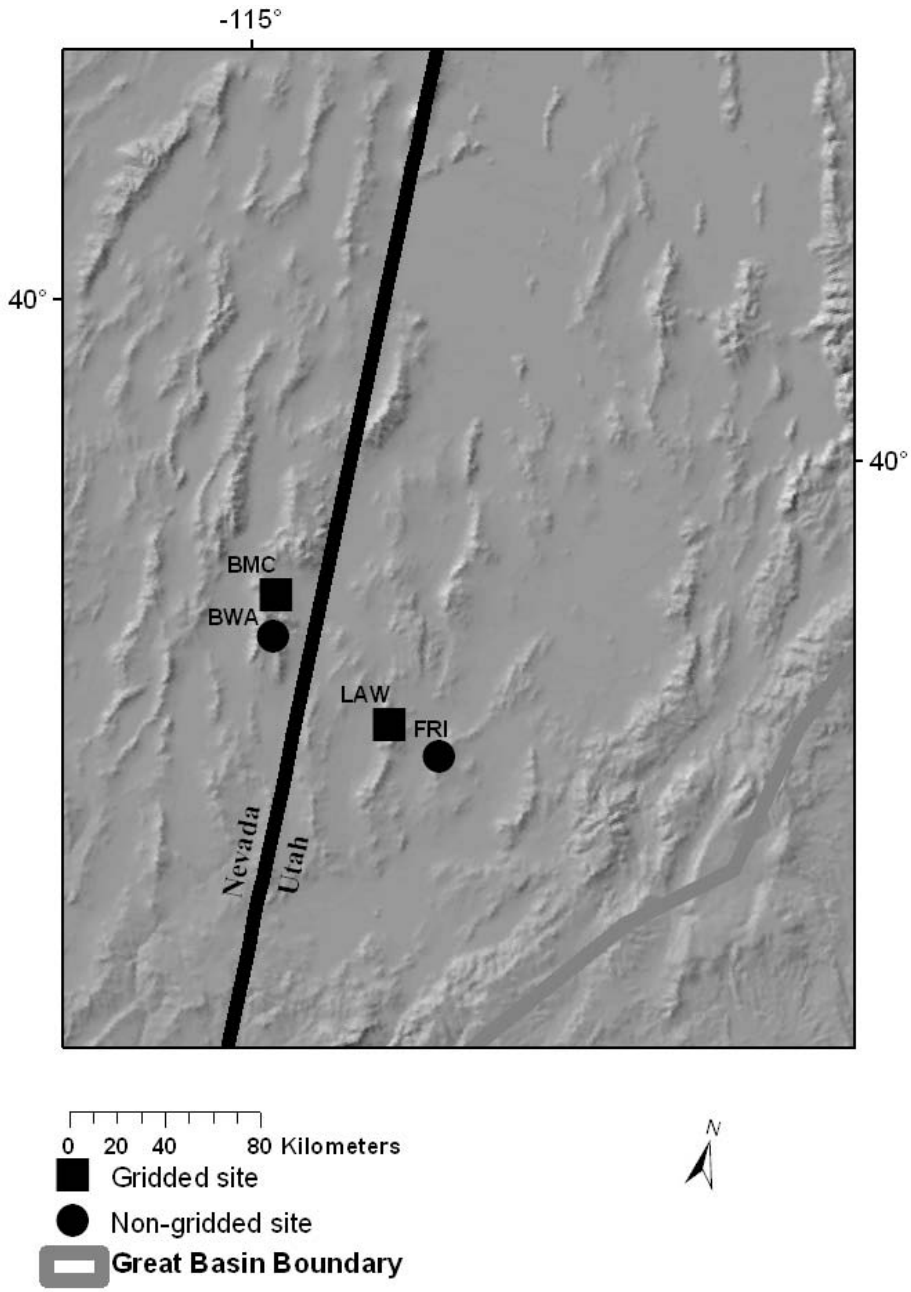
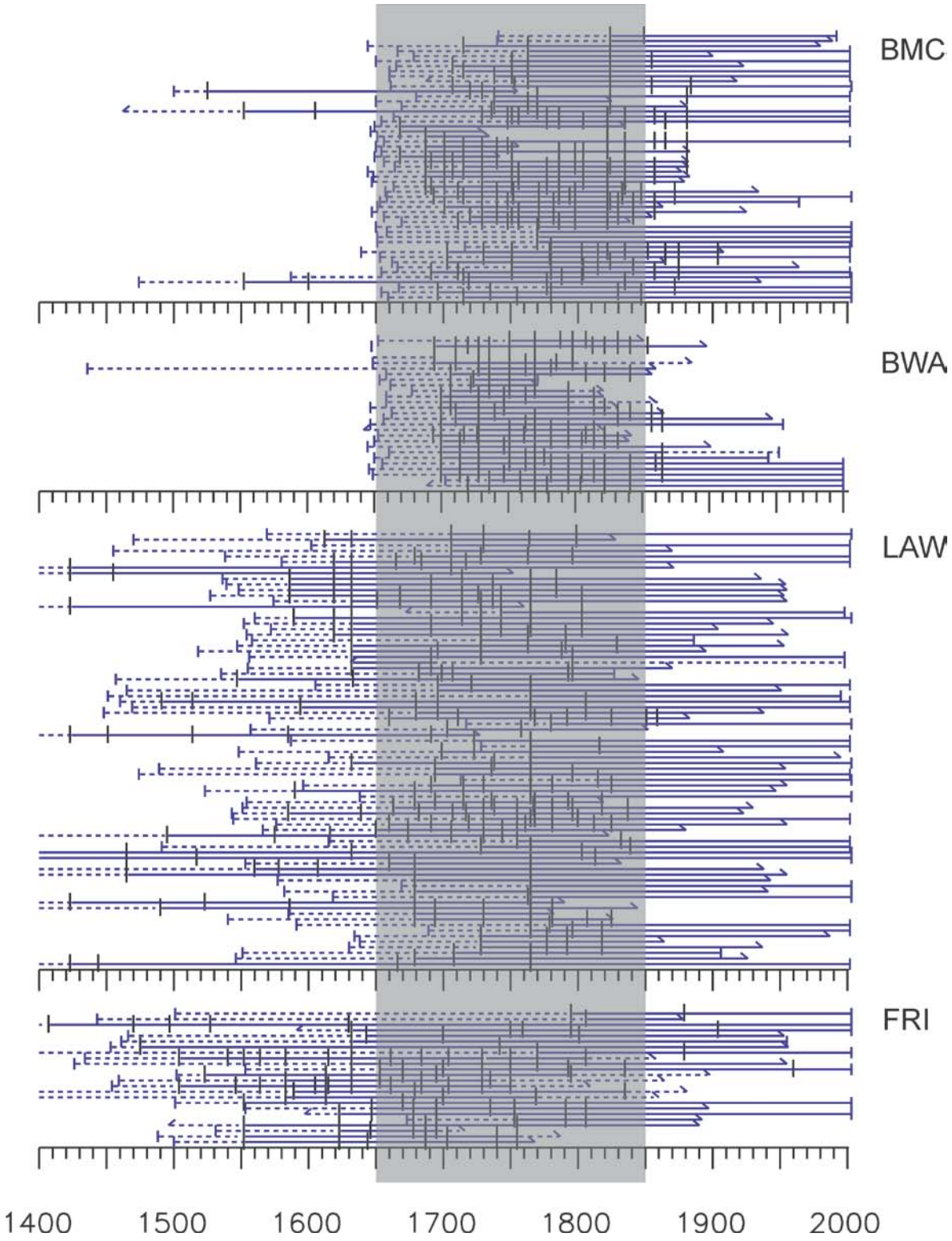


Figure 1.1. Location of four fire history study sites within the eastern Great Basin (USA). Sites are; Burnt Mill Canyon (BMC) and Big Wash (BWA) on the Snake Range, Lawson Cove (LAW) on the Wah Wah Range, and Frisco Peak (FRI) on the San Francisco Range.

Figure 1.2. Fire history chronologies for four eastern Great Basin fire history study sites. Sites are arranged from top to bottom; Burnt Mill Canyon (BMC) and Big Wash (BWA) from the Snake Range, Lawson Cove (LAW) from the Wah Wah Range, and Frisco Peak (FRI) from the San Francisco Range. Horizontal lines represent individual sample trees. Solid lines indicate trees are in recording status. Short vertical lines are placed to indicate years with fire scars for each individual tree. Shading demarks the 1650-1850 period of analysis.


BMC
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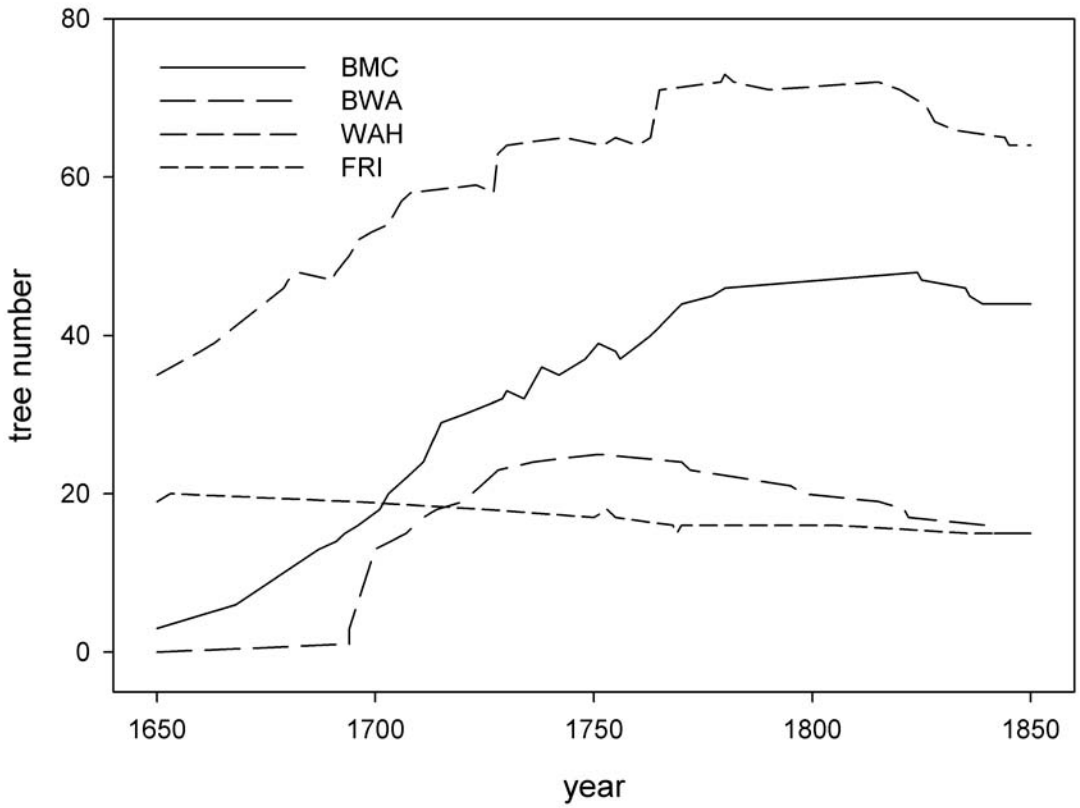

Figure 1.3. Cumulative number of trees in recording status at each of the four study sites (BMC = Burnt Mill Canyon; BWA = Big Wash; LAW = Lawson Cove; FRI = Frisco Peak) during the period of analysis (1650-1850).

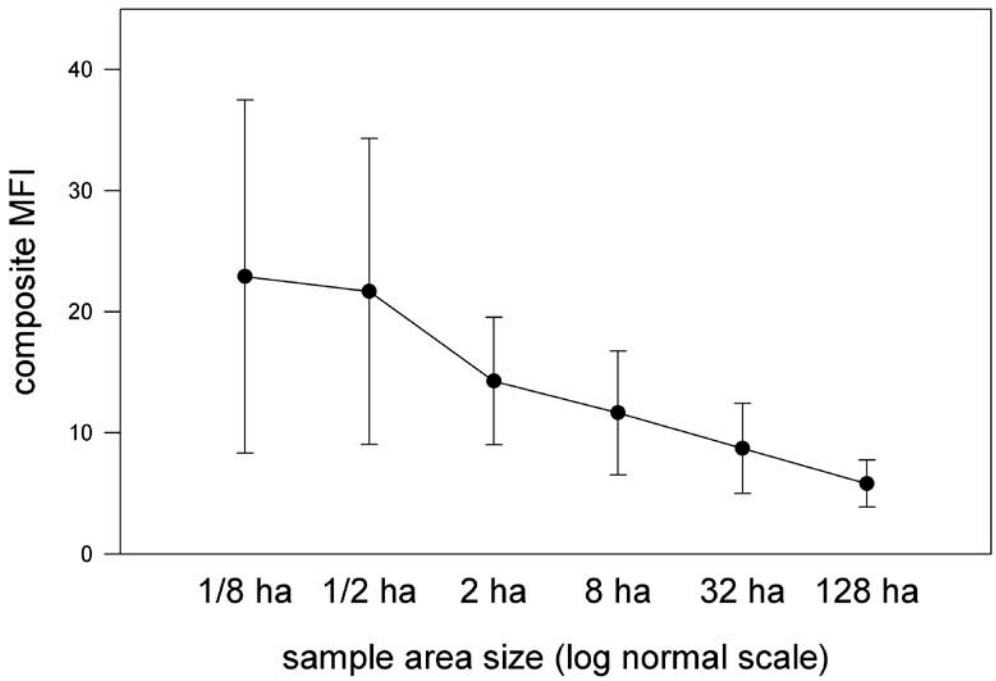

Figure 1.4. Relationship between sample area size and composite MFI across the eight nested series. Error bars = one standard deviation. Sample size area is plotted on the X axis using a log normal scale.

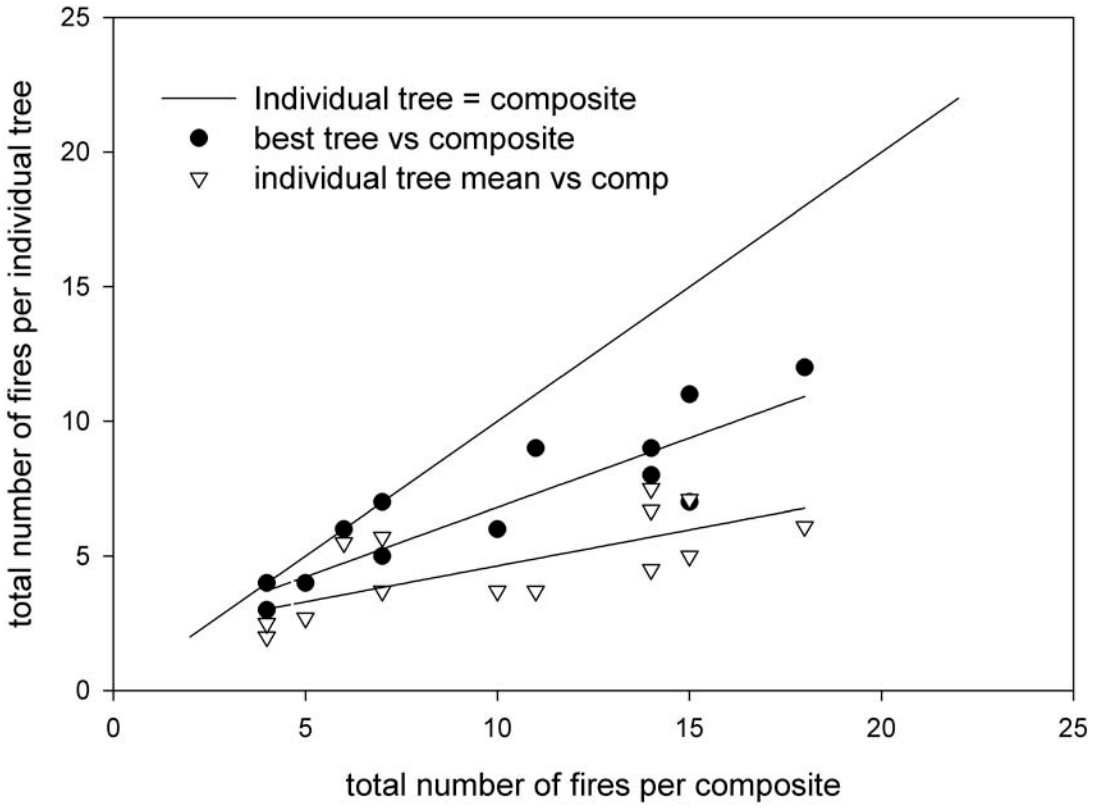

Figure 1.5. Relationship between single-tree mean (inverted triangle) and 'best' tree (circles) total fire numbers and ½-ha composite fire numbers. Proximity to the diagonal line (representative of when single tree and composite totals are equal) is an indication of the relative completeness of the fire record of individual tree chronologies.

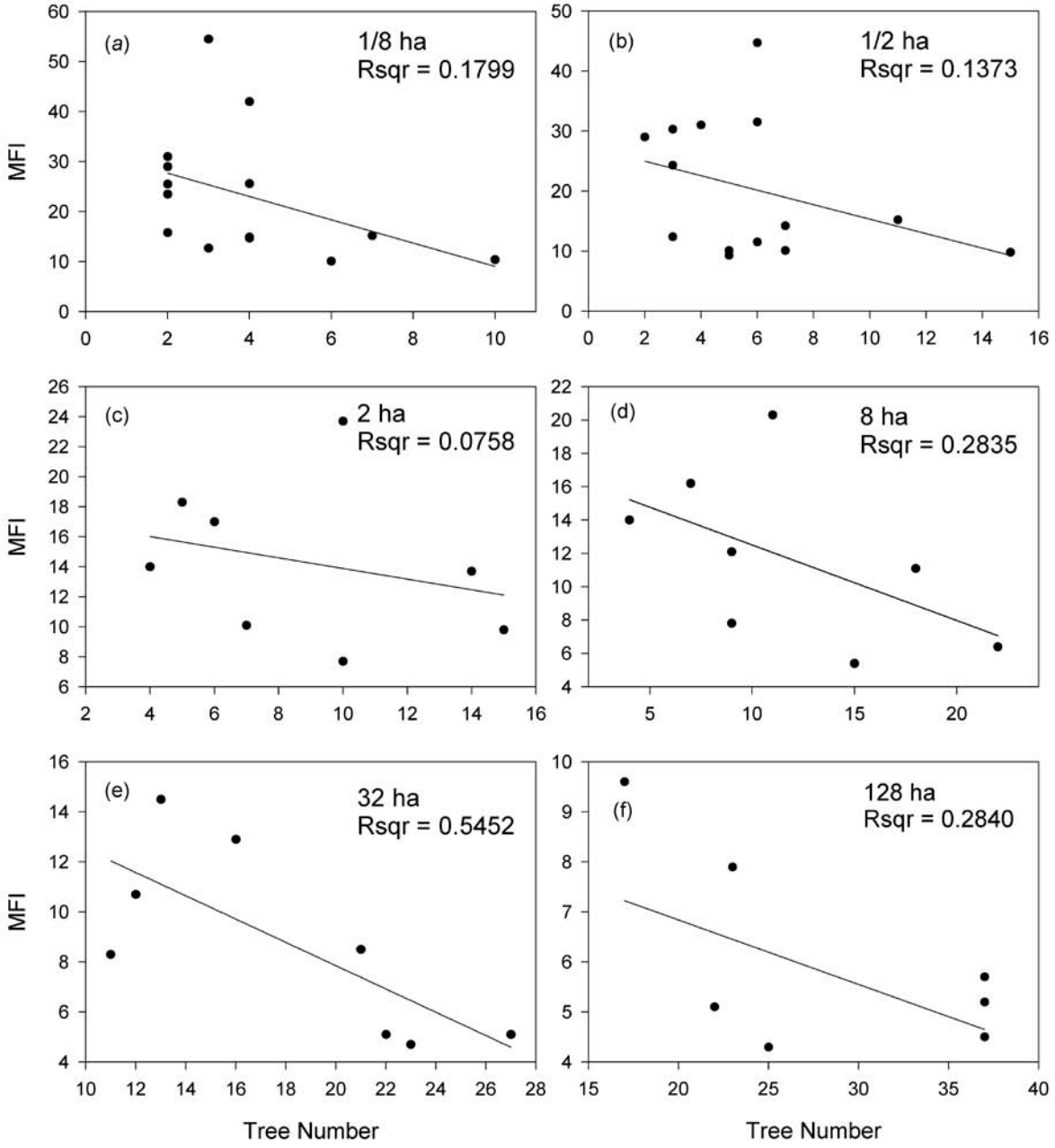

Figure 1.6. The relationship between MFI and sample tree number within each of six sample area size classes using eight nested series and six nested pairs (⅛ and ½-ha sample areas only).

CHAPTER 2 – CLIMATE AND HUMAN INFLUENCES ON HISTORICAL FIRES (1400-1900) IN THE EASTERN GREAT BASIN (USA)

Abstract: High fire activity in western North America is associated in historic and modern records with drought, which is influenced by Pacific Ocean surface sea temperature anomalies and associated atmospheric circulation patterns. Drought and increased fire activity are associated with negative El Niño Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO) phases in the Southwest and with positive phases in the Northwest. Historic and modern fire seasonality patterns also differ geographically. My objectives were to infer climate effects on historic fire patterns for 10 sites in the eastern Great Basin, a dry region geographically transitional between the Southwest and Northwest, and to identify evidence of a human signal in reconstructed fire histories. I constructed surface fire chronologies from 2,173 fire scars associated with 555 trees and 651 fire events. I identified 67 regional and 247 local fire years and 187 no-fire years from 1400 to 1900 C.E. Fire seasonality varied among sites and was bimodal across sites with both early- and late-season fires more numerous than mid-season fires. This pattern is distinct from that observed for modern lightning-caused fires which peak in mid-season, suggesting a human influence on historical ignition patterns. I compared fire chronologies with tree-ring reconstructions of summer temperature, the Palmer Drought Severity Index (PDSI), ENSO, and PDO. Fires were significantly more common during drought (negative PDSI) for four sites and for regional fire years. Conditions were significantly wetter 2 years prior to regional fire years and drier during the 4 years prior to no-fire years, providing evidence of the effect of antecedent precipitation (or the lack thereof) on the probability of fire occurrence. Regional fire years were associated with negative ENSO and positive-to-negative PDO transitions while no-fire years were associated with positive ENSO

and negative to positive PDO. Local fire years occurred under a broad range of climate conditions. Results suggest that climate was an important synchronizer of fire at the regional scale and that locally fire regimes were the product of climate-regulated fuels and some combination of human and lightning ignition patterns that likely varied through time and space.

Key Words: anthropogenic fire; climate-fire interactions; El Niño-Southern Oscillation; fire history; fire seasonality; Great Basin; Pacific Decadal Oscillation; Palmer Drought Severity Index

Introduction

Climate influences fire probability and pattern through its effect on the accumulation and conditioning of fuels. Analysis of multi-century tree-ring based fire chronologies from the Southwest (Swetnam and Betancourt 1990, 1998; Brown and Wu 2005; Margolis and Balmat 2009), Central Rockies (Donnegan et al. 2001; Grissino-Mayer et al. 2004; Schoennagel et al. 2005; Sibold and Veblen 2006; Sherriff and Veblen 2008;), Northern Rockies (Heyerdahl et al. 2008a), Black Hills (Brown 2006), Northwest (Heyerdahl et al. 2002; Norman and Taylor 2003; Hessl et al. 2004; Heyerdahl et al. 2008b; Taylor et al. 2008), Pacific Southwest (Swetnam 1993; Stevens and Collins 2004; Taylor and Beaty 2005; Skinner et al. 2008), and Intermountain highlands (Brown et al. 2008) and of written records of modern, regional fire patterns (Collins et al. 2006; Westerling et al. 2006; Morgan et al. 2008) provide consistent documentation of the link between drought and increased fire occurrence in western North America. In dry forest types, antecedent wet conditions may increase the probability for major fire years by increasing the production and continuity of fine fuels needed for surface fire

spread (Donnegan et al. 2001; Sheriff and Veblen 2008; Brown et al. 2008; Margolis and Balmat 2009). Historically, regional wet-dry cycles synchronized fire activity at parallel spatial scales (Donnigan et al. 2001; Kitzberger et al. 2001, 2007; Grissino-Mayer et al. 2004; Heyerdahl et al. 2008a,b; Morgan et al. 2008; Skinner et al. 2008; Taylor et al. 2008).

Climate variability in western North America and associated patterns of fire occurrence have been coupled to fluctuations in Pacific Ocean sea surface temperatures (SST) including the El Niño-Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO; Westerling and Swetnam 2003; Kitzberger et al. 2007). In the Southwest negative phases of ENSO and PDO result in relatively dry conditions and high fire occurrence and positive phases result in wet conditions and low fire occurrence (Swetnam and Betancourt 1990, 1998; Margolis and Balmat 2009). The effects of one or both of these climate patterns appear to extend into the Central Rockies (Donnegan et al 2001; Sherriff and Veblen 2008), Colorado Plateau (Brown et al. 2008), and as far south as Baja California (Skinner et al. 2008). ENSO and PDO effects on precipitation and fire activity in the Northwest tend to be opposite those of the Southwest; negative phases are wet and positive phases are dry, which in turn favor higher than average fire activity (Heyerdahl et al 2002, 2008b). When ENSO and PDO phases are in sync (+/+ or –/–) their effects on climate and fire activity are magnified (Gershunov et al. 1999; McCabe and Dettinger 1999; Brown et al. 2008; Heyerdahl et al. 2008b; Skinner et al. 2008). It has been suggested that the geographic fulcrum that separates these opposing climate-fire modes lies at ~40° N latitude (Mock 1996; Dettinger et al. 1998; Brown and Comrie 2004; Schoennagel et al. 2005; Brown et al. 2008). An understanding of the nature of past and present climate-fire relationships on landscapes proximal to this alleged transition zone is

lacking and may be critical for assessing geo-spatial variation in fire patterns under changing climate conditions.

Fire is also regulated by the frequency and timing of ignitions. Successful ignitions are generally the result of dry lightning storms or human activities. Teasing out the relative importance of each in historic fire chronologies can be difficult where the probability of ignition is high from both sources (Allen 2002: Griffin 2002). Allen (2002) proposed that indirect evidence of human influence on historic fire regimes might come in the form of unexpected patterns in fire frequency or seasonality, or in decreased climate-fire correlations.

My objectives were to: (1) infer climate drivers of historic surface fires for eastern Great Basin forests located south of, but in close proximity to 40˚ N latitude, (2) assess temporal and spatial variability in fire seasonality and occurrence and (3) identify evidence of a possible human footprint on past fire regimes. I explore relationships among multi-century, tree-ring derived fire chronologies and existing tree-ring based indices of regional climate that capture annual variability in summer temperature and drought as well as large scale climate patterns that have been shown to fluctuate over longer time periods (ENSO and PDO). I infer that differences between historic fire seasonality patterns and those observed in modern fire records or predicted from climate and biogeographic settings constitute indirect evidence for human influence on fire regimes (Allen 2002; Williams 2004).

Study Area

Biogeographic setting

The Great Basin encompasses the northern, elevated section of the Basin and Range Province of western North American and includes most of the state of Nevada and significant portions of Utah, Idaho, California, and Oregon. The region is characterized by over 100 relatively narrow mountain ranges with a generally north/south orientation and separated by broad, internally-drained valleys. The climate is generally dry due to rain shadow effects of the Sierra Nevada and Cascade Ranges to the west and Rocky Mountains to the east (Peterson 1994). Seasonality of precipitation varies along a geographic gradient with the importance of winter and spring Pacific frontal storms decreasing and summer monsoon increasing from north to south and from west to east. Six mountain ranges are represented in this study including the Tushar Mountains on the eastern rim of the Great Basin and San Francisco, Wah Wah, Snake, House and Deep Creek Ranges extending to the north and west (Fig. 2.1). The study area covers approximately 1.7 degrees latitude and 1.9 degrees longitude and is immediately south of 40° N (Table 2.1), the proposed pivot point for the dipole pattern of droughts and pluvials associated with ENSO and PDO variability.

Eastern Great Basin plant communities are stratified into more or less distinct zones across gradients in elevation reflecting parallel gradients in temperature and precipitation (Holmgren 1972; Harper et al. 1978). Zonation is further modified by slope, aspect, and substrate. Sagebrush-grass communities occupy a broad zone from valley floors to dry mid-elevation sites. Woodlands of singleleaf (*Pinus monophylla*) or Colorado pinyon (*P. edulis*) and Utah and Rocky Mountain junipers (*Juniperus osteosperma*; *J. scopulorum*) occupy a mid-elevation thermal belt that is superimposed more or less through the center of the sagebrush (*Artemisia* spp.) zone and have expanded up and down-slope over the last century (Tausch et al. 1981). Stands of mountain mahogany (*Cercocarpus ledifolius*) and other tall shrubs are

most abundant at mid-elevations and are sometimes expansive. Mixed and monotypic stands of ponderosa pine (*P. ponderosa*), white fir (*Abies concolor*), and/or Douglas fir (*Pseudotsuga menziesii*) occupy mid-elevations and transition to limber (*P. flexilis*) or bristlecone pine (*P. longaeva*) and sometimes Engelmann spruce (*Picea engelmannii*) with increasing elevation. Quaking aspen (*Populus tremuloides*) is often present but is less abundant than in mountains immediately east of the Great Basin. A treeless alpine zone is found on the highest mountains such as the Tushar, Snake, and Deep Creek Ranges. As a member of the generally more massive and mesic central Utah highlands, vegetation of the Tushar Mountains differs from that of the other ranges included in this study with the addition of subalpine fir (*Abies lasiocarpa*), the greater abundance of quaking aspen, and the diminished importance of high-elevation pine (scattered limber pine only). Ponderosa pine, a key species for developing multi-century fire histories, is present on all six study ranges but decreases in abundance from south to north.

Human occupation

The Great Basin has been inhabited by humans for at least 13,000 years. The study area encompasses a region where, immediately prior to Euro-American settlement, Western Shoshone, Ute, and Southern Paiute cultures converged in what is now western Utah (Simms 2008). The mobile hunter-gatherer economies practiced by these Numic-speaking inhabitants contrast with the more sedentary, semi-agricultural model of the Fremont that appears to have dominated the area from 900 to 1300 C.E. (Simms 2008). For millennia before the Fremont, Archaic foraging strategies included the desert-mountain settlement pattern with wide-ranging seasonal migrations, and the less-mobile wetland settlement pattern in which economies benefited from resource-rich environments such as those found at lake margins and marshes

(Janetski and Madsen 1990; Simms 2008). Undoubtedly, fire had many useful applications and was an essential tool for manipulating natural environments (Williams 2004) as well as an accidental consequence of occupation (Kay 2007) throughout this long pre-history. However, knowledge regarding specific practices used in this region and how they or their impacts on vegetation might have varied through space and time is lacking (Griffin 2002). Steward's (1938) ethnographic studies shed some light on the uses of fire by Numic inhabitants at about the time of Euro-American settlement. The purposes for setting fire given by his informants were generalized by Griffin (2002) into three categories typical of hunter-forager societies namely: (1) to increase the quantity and quality of useful vegetation, (2) to improve or maintain habitat for important animals, and (3) to drive game in hunting. Numerous human-set fires on the eastern edge of the Great Basin were observed and recorded in the fall of 1776 by the Dominguez-Escalante expedition which passed through the eastern margins of the Great Basin while searching for an alternative route from Santa Fe to California (Chavez and Warner 1976). In spite of these and other records (see Griffin 2002), the relative importance of human-set fires on the historic fire regimes of the eastern Great Basin and the plant communities they structured is not well understood and remains a point of scientific disagreement.

As with other parts of the continent, the first impacts of European colonization on Native Americans in the eastern Great Basin were likely from diseases to which they had little resistance. Elsewhere, repeated epidemics in the 1500s to 1700s reduced Native American populations by 70% or more (Thornton 1987; Reff 1991; Butzer 1992). Evidence for a disruption in human fire ignition patterns associated with such drastic depopulation might be anticipated in the Great Basin.

Euro-American settlement within the study area began with the arrival of Mormon settlers in the 1850s (Young et al. 1979). Settlements were small and widely dispersed, especially west of the better-watered valleys located adjacent to mountains forming the eastern rim of the Great Basin. Farming and ranching were the principal economic activities of early settlers. Horses, cattle, and sheep were introduced at the time of settlement. Regionally, stocking rates were generally low until the 1880s when the number of sheep trailed to the area for winter-spring grazing increased dramatically and remained high for 50+ years (Murdock and Welsh 1971). Large numbers of domestic livestock impacted surface fire regimes throughout the western United States by reducing the quantity and continuity of fine fuels needed for fire spread (Pyne 1982; Mandany and West 1983; Covington and Moore 1994). The timing and intensity of livestock-induced impacts on fire regimes within the study area likely varied among and within mountain ranges in response to such factors as surface water availability, ruggedness of terrain, and trailing distance. Preferred tree species were selectively cut by early settlers for fuel-wood and for general construction needs. In subsequent decades mining operations on most ranges in the area increased demands for water, lumber, and other natural resources. Today, most of the mines are closed and the regional population remains sparse. The need for fire prevention was debated until 1910 when the federal government adopted an aggressive policy of fire suppression on all public lands (Pyne 1982; Keane et al. 2002). Similar to the effects of livestock, the application of this policy within the sparsely inhabited and undervalued study area must have varied considerably and may have been essentially non-existent on remote mountain ranges before post-World War II mechanization extended agency capacity to fight fire on most landscapes.

Methods

Historical surface fire chronologies

Fire-scar-based fire histories were reconstructed for 10 sites from six study ranges with a combined range in elevation of 1,035 m (Fig. 2.1; Table 2.1). A systematic sampling strategy using plots spaced at 500-m intervals was used to sample across the range of elevation within small watersheds at RBC, INC, BMC, and LAW study sites (see Brown et al. 2008). I employed a targeted sampling approach for the remaining sites (Van Horne and Fulé 2006). Sample tree number and sample area size varied among sites (Table 2.1). Species, condition (live, snag, stump, or log), and elevation were noted for each sample tree. Universal Transverse Mercator (UTM) coordinates were determined for sample trees using hand-held GPS units accurate to within 15 m. One or more cross-sections were removed from scarred surfaces (cat-faces) of fire-scarred trees using chainsaws and standard methods (Arno and Sneck 1977). Although trees with multiple scars were selected preferentially, numerous trees with 1 to 3 scars were also sampled.

Sample pieces were stabilized with glue and plywood as needed and surfaced using appropriate combinations of band saw, power planer, belt sanders and hand sanding until cell structure was discernible using a binocular microscope. Each sample was independently cross-dated by a minimum of two analysts using a combination of locally and regionally-developed master ring-width chronologies (skeleton plots) and lists of marker (narrow) years (Stokes and Smiley 1968). Specimens that could not be clearly dated and fire scars that could not be dated with annual accuracy were excluded from the analysis.

Fire seasonality

I determined a calendar year for each fire scar and assigned each an intra-ring [i.e. early (EE), middle (ME), late early-wood (LE) or late-wood (LW)] or inter-ring [ring boundary (RB)] position when fire-scar condition permitted (Dieterich and Swetnam 1984). Seasonality of fire scars associated with very narrow rings or eroded scar-ring structure was classified as unknown (UNK). Ring-boundary scars are caused by either late season fires that occur after ring growth is complete or by early season fires that burn before ring growth is initiated. Typically, these inter-ring scars are assigned to the following year in the southwest United States and to the preceding year at more northern latitudes based upon the seasonality of modern fires and the relative abundance of early-wood and late-wood scars in fire-scarred specimens in historic fire studies (Schmidt et al. 2002, Kitchen and McArthur 2003). The intermediate geographic position of these study sites suggests caution is warranted before adopting either convention as a rule. First, I assigned inter-ring scars to preceding years when evidence of a late season fire (LE or LW scar) in that year was found in at least one tree from the study area, and similarly to the post-boundary year when evidence of an early season fire (EE or ME scar) was collected from the study area for the same year. Based upon results from a pilot study (Kitchen and McArthur 2003), fire year was assigned to the post-boundary year (Southwest convention) when fire-year could not be conclusively determined using these criteria.

I assigned seasonality as unknown, dormant, early, middle, late, or multi-season for each fire event (site by year combination) using predetermined classification criteria that assess, as a group, the ring positions of the complete list of fire scars associated with each fire event (Table 2.2). Fire events classified as multi-season always included EE and LE or LW fire scars and generally also included ME and RB fire scars (Baisan and Swetnam 1990; Grissino-

Mayer et al. 2004). This classification was intended to capture fires that might have burned through the majority of the fire season across the sample area. Fire seasonality through time was assessed for all sites combined by comparing the proportion of fire events in each seasonal class among 50-year bins from 1400 to 1900.

Climate effects on fire occurrence

All years with at least one assigned fire scar were composited into a single fire chronology for each site, using program FHX2 (Grissino-Mayer 2001). It is common practice to require a minimum of two trees per site with evidence of fire for a year to be considered a fire year based upon the assumed greater possibility that single injuries within a year may not have been fire-induced (e.g. tree-fall, lightning; see Brown and Wu 2005; Brown 2006; Brown et al. 2008). With the incomplete sampling strategies employed in this study there was a similar likelihood of capturing either one or two fire-scars per year among all sampled trees per site, making the arbitrary designation of a two-scar threshold for fire-year classification difficult to defend. My decision to include single-tree fire years in this study is further justified because a large proportion (76%) of the scars in question were imbedded (not the first) within scar series on individual fire recording trees, suggesting a high probability that they were indeed fire-caused. A site was considered to be a recording site when a least one tree from the site was in recording status. A tree was in recording status when it had experienced at least one fire, the post-fire scar surface was sufficiently intact to detect subsequent fire scars and the cat-face remained open subjecting the tree to a high probability for injury from new fire events. Low- and high-filter regional fire chronologies were created from the individual site chronologies representing those fire years in which fire was recorded on at least 33 or 50% of recording sites, respectively. Local fire years, or years when fire was recorded on < 33% of

recording sites, and years with no fire scars for any of the 10 sites (no-fire years) were also tabulated.

I compared individual site and regional fire chronologies to four indices of climate variability derived from independent, tree-ring reconstructions of climate variables to assess fire-climate relationships. These indices include: (1) warm-season temperature (April-September) at one grid point near the study area (grid point 17; Briffa et al. 1992); (2) Palmer Drought Severity Index (PDSI), a measure of June through August drought, averaged from four grid points closely associated with the study area (grid points 71, 72, 86, and 87; Cook et al. 2004); (3) winter (December through February) NINO3, an index of ENSO variability that fluctuates on a sub-decadal (3-8 year) scale (D´Arrigo et al. 2005); and (4) Pacific Decadal Oscillation (PDO) an index of annual north Pacific temperature patterns that vary at decadal scales (McDonald and Case 2005). I used superposed epoch analysis (SEA) to compare average annual climate conditions to climate conditions associated with fire and non-fire years in the regional and individual site fire-climate analyses, and to conditions prior to (5 years) and following (2 years, Temperature, PDSI and NINO3 and 3 years PDO) the event year (Swetnam 1993; Swetnam and Baisan 1996). I identified significance of departures at two levels based upon 95 and 99% confidence intervals determined by bootstrapping techniques (1000 trials; Grissino-Mayer 2001).

Results

Between four and 167 sampled trees per site were cross-dated for a total of 555 fire-scarred trees from the 10 sites (Table 2.1). Although datable scars were sampled from 11 tree

species, most were taken from ponderosa pine (58%), Douglas-fir (15%), or limber pine (9%). These same species typically had the most scars per tree, with one ponderosa pine stump from the ROS site recording 35 fire dates. Dead trees (snags, logs and stumps) made up 64% of cross-dated trees.

The earliest and latest fire dates were 1205 and 1960. Because five of the sites (BMC, LAW, FRI, INC, and ROS) were in recording status by the year 1400 (Fig 2.2.), I selected this date as the fire-climate analysis starting point for these sites and for the regional analysis. More recent dates were selected as starting points for the other individual sites (RBC 1500; SWA and SIN 1550; TOM and BWA 1700) corresponding to dates when fire records for each site began. The end year for all fire-climate analyses was 1900 corresponding to a drop-off in fire occurrence at most sites in the mid to late 1800s (Fig. 2.2). Between 1400 and 1900, I identified a total of 342 fire years with 67 and 24 regional fire years, respectively (Table 2.3; Fig. 2.2) using the low- and high-level filters. Regionally, low- and high-filter mean fire intervals were 9 and 26 years with a 1-year minimum interval and 33- and 109-year maximum intervals. There were 247 local fire years and 187 no-fire years.

Fire seasonality

Ring position was assigned to 1,691 of the 2,173 dated fire scars. Three sites dominated the region-wide fire scar totals due to the greater number of fire-scarred trees sampled at these sites (Table 2.1; Fig. 2.3*a*). Ring boundary scars dominated fire-scar ring positions for eight sites (range 46-68%) and accounted for 27% of scar positions for the other two (INC and RBC; Fig 2.3*a*). Across all sites EE scars accounted for 3 (SIN) to 26% (INC) and ME scars accounted for 7 (FRI) to 30% (INC) of assigned fire-scar ring positions. Late season scars (LE

and LW combined) contributed between 8 (BWA) and 32% (RBC) of totals. Overall, no clear among-site patterns were apparent in the seasonal distribution of fire scars. Regionally, nearly half of the scars assigned ring positions were assigned an RB position with the EE, ME, and LE approximately equal to each other and double the number of LW scars (Fig. 2.3*b*).

The among-site inequality in the regional analysis is partially overcome by switching the focus from fire scars to fire events (compare Figs. 2.3*a* and 2.4*a*). Unknown seasonality was assigned to 92 of 651 fire events (Fig. 2.4*b*). At least one RB scar was associated with 59, 44, and 71% of early-, late-, and multi-season fire events, respectively. I observed that the proportion of fire events classified as dormant (RB scars only) ranged from 11 (RBC) to 62% (BWA) among sites (Fig. 2.4*a*) and, excluding unknowns, represents just over one in three fires, regionally (Fig. 2.4*b*). Among sites, early-season fires ranged from 6 (SIN) to 27% (INC) and middle-season fires ranged from 5 (BWA) to 27% (INC). Taken together they account for another one in three fires at the regional scale. Late season fires varied from 15 (BWA) to 48% (RBC) and with multi-season fires account for the final ⅓ of fire events across the region. Multi-season fires were only detected on the five largest sites. As with the fire scar seasonality analysis, no clear geographic patterns (e.g. north to south) emerged from the analysis of fire event seasonality although site differences are apparent. Results demonstrate a somewhat balanced importance between early and late season fires across the region (Fig. 2.4*a* and *b*), and the more or less equal likelihood that RB scars would be formed from both early or late fire events.

Dormant-season fire events as a percentage of all fires with assigned seasonality (unknowns excluded) were most common from 1450-1549 and from 1850-1899 (~50%) and only dropped below 30% (23%) during the 1650-1699 50-year bin. If I further restrict the

analysis to exclude dormant-season events, then early-season fires were most common from 1400-1499 (~47%), least common from 1600-1649 (16%) and ranged from 24 to 36% for the balance of the analysis period (Fig. 2.5). Middle-season fires ranged from 43% (1450-1499) to 19% (1750-1799) with a 500-yr mean of 27%. Late-season fires were least common during the 1400s (~20%), increased during the 1500s (~30%), peaked at in the early 1600s (53%) and remained high (~43%) for the next 250 years.

Climate effects on fire occurrence

Departures in summer temperature for the year of fire (or no fire) were not significant in individual site or regional analyses and will not be considered further here. PDSI was negative for 50 of 67 low-filter regional fire years and 22 of 24 high-filter regional fire years (Fig. 2.6*a*). Departures in PDSI were significantly negative (indicating drought) during fire years for four sites (Fig. 2.7) and for both classes of regional fire years (Figs. 2.8*a* and *d*). The first year after fire was significantly negative for two sites (Fig. 2.7) and for both classes of regional fire years (Fig. 2.8*a* and *d*). Conditions during at least one year before fire were significantly PDSI-positive (indicating wet conditions) for four sites (Fig. 2.7). PDSI departures 2 years before regional fire years were significantly positive, resulting in a pattern of wetter than average conditions before and dry conditions during and immediately after regional fire years (Fig. 2.8*a* and *d*). I observed no significant departures in PDSI associated with local fire-years (Figs. 2.6*d*, 2.8*g*). Although conditions during and after no-fire years were not different than average, departures for the 4 preceding years were significantly negative (Fig. 2.8*j*), suggesting that fire was least common after extended periods of drought.

NINO3 was negative for 36 of 67 and 15 of 24 low- and high-filter regional fire years, respectively (Fig. 2.6*b*). Departures in NINO3 during fire years were significantly negative for three sites (Fig. 2.7) and for both classes of regional fire years (Fig. 2.8*b* and *e*). No significant departures in NINO3 were observed for local fire years and associated lag years (Fig. 2.6*e*, 2.8*h*), but the positive departure for no-fire years was significant (Fig. 2.8*k*).

PDO was strongly positive from the mid 1400s to late 1500s and again for shorter periods in the mid-1700s and mid-1800s (Fig. 2.6*c*). Although fire years are spread throughout the analysis period, the frequency of regional fire years appears to be greatest when PDO was near neutral or negative. Departures in PDO during regional fire years and for 3 years after are significantly negative (Fig. 2.8*c* and *f*) suggesting that regional fire years were most likely to occur at or just after positive-to-negative phase changes in PDO. A tendency for this pattern is apparent in eight individual site analyses, and is supported by significant departures in four (Fig. 2.7). Conversely, departures in PDO for no-fire years and 2 years after are significantly positive (Fig. 8*l*), suggesting that fire is least likely to occur during a negative-to-positive PDO phase change.

Discussion

Fire chronologies for the 10 sites in this study reveal a general pattern of frequent surface fires that was sustained over a long period of time (Fig 2.2). This is particularly so on larger sites with greater numbers of sample trees and broader sample tree dispersal, as indicated by sample area size (Table 2.1), suggesting the importance of the contribution of small fires to the site-level composite fire records. Because sites differed in sample strategy,

effort and area, it would be inappropriate to make intra-site comparisons regarding fire frequency. Fire histories for five sites begin before 1400 C.E. providing at least partial fire records for a period recognized as pre-Little Ice Age. Visual examination of these composite chronologies (Fig. 2.2) would seem to suggest that site-level fire return intervals were longer before ~1500. However caution is needed in interpreting site-level fire frequencies for this portion of the records as they were constructed from fewer trees than what contributed to the composite fire chronologies in later centuries. Thus it would be difficult at best to tease out the effects of this fading record from any real differences in fire frequency between pre-1500 and later time periods. On the other hand variation in fire seasonality should not be impacted by the fading record, assuming that the fire records preserved from the earliest centuries are representative of the period.

Fire seasonality

The annual timing of new tree-ring growth varies among tree species and individuals within species and is related to variations in yearly weather, elevation, and topographic setting. Overriding this variation, I observed a predominance of RB (inter-ring) fire scars that were associated with substantial proportions of both early- (59%) and late-season (44%) fire events and were the only class of fire scars associated with 204 of the 559 fire events in which seasonality was assigned (Fig. 4b). These results demonstrate that the historic fire season in the eastern Great Basin was potentially long, beginning before and ending after the period of annual ring growth. This should come as no surprise given the generally dry climate in the region expressed by both early spring snowmelt and a weak or absent summer monsoon. If the ambiguous, dormant season fire events are split more or less evenly between the early- and late-season fire groupings, a bimodal pattern emerges with a 36% early-, 15% middle-, 45%

late- and 4% multi-season fire distribution. Considerable variation in fire seasonality was observed among sites (Fig. 2.4*a*). Altering the relative percentages in the division of dormant fires to early and late fire groupings to reflect the seasonal patterns encountered on individual sites would change the relative proportions of early and late season fires but the bimodal shape of the distribution would largely be preserved.

Interestingly, the among-site variation and bimodal pattern in fire seasonality observed here are similar to those identified for the San Juan Mountains (Grissino-Mayer et al. 2004), a study area located slightly south (~37˚ 30´ N) and roughly 500 km east of these eastern Great Basin sites on the Colorado Plateau. The authors in that study suggested that wet conditions during the summer monsoon would reduce fire occurrence during mid-summer resulting in a bimodal fire seasonality pattern but offered no explanation for among-site differences. Among-site variability in fire seasonality has been reported for site clusters elsewhere in western North American studies including the Blue Mountains in Oregon (Heyerdahl et al. 2001) and the Sierra San Pedro Mártir in northwestern Mexico (Skinner et al. 2008). Topographic variability expressed at fine spatial scales might explain some differences in fire seasonality due to the effects of features such as aspect, slope, and elevation on the rate of fuel desiccation and timing of tree-ring growth. Conversely, if we allow for a human element in fire ignitions, the variability in patterns of human occupation or burning strategies might easily explain major among-site differences in early and late-season fire dominance such as those observed in this study.

Long-term climate variability has been used to explain temporal variability in fire seasonality through the interaction of temperature and precipitation (timing and abundance) on the production and desiccation of fine fuels (Grissino-Mayer and Swetnam 2000). I evaluated

the potential influence of climate variation on fire seasonality in the eastern Great Basin by observing proportional changes among the 50-year bins from 1400 to 1900 in this record (Fig. 2.5). Early-season fires dominated in the 1400s suggesting lower winter-spring precipitation and wetter summer monsoons relative to the Little Ice Age that would follow. As the climate cooled and summers dried through the 1500s, fire seasonality would have been in transition with fewer early-season fires and an increasing number of late-season fires relative to the 1400s, as was observed. The proportional shift from early- to late-season fires reached its maximum during a cool-wet period in the early 1600s (Fig. 2.6*a*) after which a relative balance between early- and late-season fires was reached by the late 1700s. That balance persisted until the late 1800s where fire seasonality shifted somewhat back towards late-season fires once again. Grissino-Mayer and Swetnam (2000) described a gradual shift for a cluster of Southwest sites (northwest New Mexico) from predominantly middle- and late-season fires prior to 1800 to predominantly early-season fires by the end of the 19th Century. They postulate that as the Southwest warmed at the end of the Little Ice Age, the dominant precipitation mode shifted from winter-spring to summer monsoon and fire seasonality changed accordingly. Although the nature and pace of fire seasonality change for the Great Basin sites was similar to that of the Southwest sites, the direction was not. The summer monsoon is much weaker and sporadic and winter-spring precipitation more reliable in the eastern Great Basin than in the Southwest, precluding any late to early-season shift in fire seasonality.

Modern (post 1980) fire seasonality distribution for the eastern Great Basin, measured either in number of fires or area burned, has a broad middle- to late season (July and August) peak (Griffin 2002; Schmidt et al. 2002; Westerling et al. 2003). A narrower peak from mid-July to mid-August is suggested for lightning-caused fires based upon a 42-year record of fire

from the Snake Range (Nevada) and surrounding area (data on file at the US Forest Service, Shrub Sciences Laboratory, Provo, Utah). The predicted effect of the warming trend that began in the mid-1800s would be to lengthen the fire season resulting in more early- and late-season fires and dampening rather than strengthening a mid-season peak. This prediction is confirmed by a lengthening of the fire season in the western United States associated with the accelerated warming trend during the last three decades (Westerling et al. 2006). Although similar in some ways (late season dominance) to the fire seasonality distribution inferred for the late 1800s, the modern, lightning-caused distribution lacks the bimodal pattern apparent for most individual sites and regionally for 500 years suggesting that regulation of historic fire seasonality likely involved one or more factors besides climate.

Unexpected seasonality patterns may constitute an indirect clue of a human footprint on fire regimes (Allen 2002; Williams 2004). Unexplained among-site differences in fire seasonality such as those described here and in the San Juan Mountains (Grissino-Mayer et al. 2004) might also infer a human-modified fire regime due to local differences in burning practices. In this study, the case for an anthropogenic source of ignitions is perhaps strongest for the Sinbad Springs (SIN) site. The ~1 ha area of this site is defined by a small cluster of mature ponderosa pine trees associated with a perennial spring. Ponderosa pines are located along and near the bottom of a ravine and end where surface water drops off a 30-m limestone cliff. Beyond the influence of the spring the vegetation changes sharply to old growth pinyon-juniper and mountain mahogany woodland that is poorly adapted for frequent burning. Therefore, the probability of fire spreading to the 1-ha study from the surrounding landscape is extremely low. Consequently, the ignitions that sustained the high frequency fire regime recorded in fire scars (4 sample trees) almost certainly originated from within the site. Given

the small and recessed nature of the study site relative to the surrounding landscape, the likelihood of lightning being the primary source of ignitions is remote. The presence of a reliable water source in this generally dry landscape would have been an attraction to both humans and animals that would constitute important food sources. Thus, the best explanation for the fire chronology found at this site is that it represents, perhaps exclusively, a pattern of intentional (and possibly accidental) ignitions by Native Americans dating back to late 16th Century (Swetnam and Baisan 1996). Late- and dormant-season fires are approximately equal in number and together they account for nearly 90% of the fires at this site suggesting that burning occurred primarily in late summer or fall. This may have been the only time that the spring-fed vegetation was sufficiently cured to burn most years and is in agreement with historical accounts that indicate that fall was a preferred time for Native American burning in this part of the West (Chavez and Warner 1976; Griffin 2002). Fall is also a time of reduced lightning-caused fire occurrence in modern records (Griffin 2002) strengthening the argument for human ignitions at this site. Reasons for intentional burning of this site are potentially numerous (Williams, 2004) and include clearing vegetation to improve visibility for ambush hunting, 'greening' of vegetation to attract large herbivores, or improving growth characteristics of useful vegetation. Although the unique biophysical setting of this site is useful for deducing a probable source of ignitions, fire frequency and seasonality are not extreme when compared to other sites in this study where natural fire ignitions are likely to have been more important. This could be interpreted as an indication of the difficulty of teasing out the human fire 'footprint' from a background fire regime dictated by lightning ignitions. Alternatively, the relative 'normality' of the fire regime for this site when weighed with the

unexpected bimodal distribution and among-site variability in fire seasonality could also suggest that the impact of human ignitions, though locally variable, was likely widespread.

Climate effects on fire occurrence

A strong correlation between regional fire years and drought was expected. I also expected that antecedent wet years (positive PDSI) favorable for fine fuel production would be statistically significant. It is of interest that conditions 2 years prior to regional fire years were significantly wetter than average but conditions 1 year prior were not (Fig. 2.8*a*,*d*). It was also notable that significant drought persisted for two years after regional fire years. This pattern suggests that years of widespread regional fires were not merely a product of one-year droughts but that fire was synchronized by multi-year, wet-dry oscillations. Thus the probability of fire was highest during the first year of strong drought following one or more wet years. The potential impact of earlier wet years (lag -3 to -5) would be muted in comparison due to the natural breakdown of fine fuels as time between fuel production and fire ignition increased. The lack of significant departure for the year immediately before regional fire years could indicate that these years were climatically transitional in the multi-year oscillation. This pattern of fire-timing in response to multi-year precipitation variability is also evident in the individual site SEA of PDSI (Fig. 2.7), though significant departures are scattered among only half of the sites. A reverse pattern of multiple years of drought resulting in subsequent low fire probability (no-fire years) is a logical extension of the pattern and represents a separate segment of the oscillation (Fig. 2.8*j*). Conversely, years in which fire occurred on few sites (local fire years) do not follow this pattern and in fact appear not to be linked to climate variation in any way (Fig. 2.8*g*-*i*). An increased probability for synchronized fire associated with a multi-year positive-to-negative shift in PDSI has been documented

elsewhere in western North America including the Tahoe Basin of eastern California (Beaty and Taylor 2008), Rincon Mountains in southeastern Arizona (Iniguez et al. 2009), Sacramento Mountains, southeastern New Mexico (Brown et al. 2001), and Archuleta Mesa in southwestern Colorado (Brown and Wu 2005).

The relationship between regional fire years and ENSO was clearly Southwestern in nature (Swetnam and Betancourt 1990, 1998) with fires-years correlated with significantly negative and no-fire years correlated with significantly positive NINO3 departures (Fig. 2.8*a*,*e*,*k*). The relationship appears a bit more muddled in the individual site analyses, indicating that NINO3 was a poor predictor of fire probability for most sites. PDO values for regional fire-years and 3 years thereafter were significantly negative while values for no-fire years and for two years after were significantly positive relative to average. This multi-year pattern is similar to that observed for the PDSI analysis in that the probability of regional fire was greatest as PDO was shifting downward to a series of years of significantly low values, an indication of drought in the southwest. The multi-year positive departure in PDO during and following no-fire years is also consistent with the idea that regional fire years were synchronized by multi-year drivers of climate and that fires were least likely as the climate shifted from average (dry) to wetter conditions. Skinner et al. (2008) reported similar patterns for regional and no-fire years in relation to drought and PDO for northwestern Mexican forests using McDonald and Case (2005) and Biondi et al. (2001) PDO indices. A third index (D'Arrigo et al. 2001) failed to reveal this pattern there or for the Colorado Front Range (Sherriff and Veblen 2008). These differences in PDO-fire relationships reflect differences in the climate proxies themselves and suggest one or more are substantially influenced by additional factors. Although clarification of the PDO-fire relationship will require refinements

in proxy records, the patterns observed here and in northwestern Mexico using the McDonald and Case (2005) and Biondi et al. (2001) PDO proxies, suggest a broad regional fire synchronization in response to multi-year oscillations in precipitation that are at least to some degree responsive to variation in PDO.

Low-filter, regional fire-year frequency was somewhat stable from the late 1400s to 1822 with periods of high frequency from 1619 to 1738 and from 1777 to 1822 (Table 2.3; Fig. 2.2). An anomalously long gap in regional fire years occurred between 1553 and 1579 (Fig. 2.2), a period devoid of major drought (average to high PDSI), muted inter-annual differences in ENSO, and high PDO (Fig 2.6*a*-*c*), conditions favorable for reduced fire activity (Kitzberger et al. 2001). Regional fire frequency remained relatively low until about 1620. During this period of below average fire synchronization, late 16th century drought gave way to wetter conditions (negative to positive PDSI), ENSO oscillations were average, and PDO was mostly positive (indicating wet) and stable. During the next 120 years of high frequency fire, droughts were frequent and severe, ENSO oscillations were about average, and PDO was mostly negative. Average PDO became slightly positive but was highly variable during the 40-year period of somewhat reduced frequency in low-filter, regional fire frequency that occurred between 1738 and 1777. This period of reduced regional fire is synchronous with a distinct hiatus in regional fires documented for southwestern Colorado (Grissino-Mayer et al. 2004). In this study, local fire frequency remained high during this period suggesting that the phenomenon, if real, was weaker in the eastern Great Basin. A second long hiatus in regional fire years occurs from 1822 to 1855 and is more or less synchronous with a lengthening of fire intervals at five sites (SIN, BWA, LAW, FRI, ROS; Fig. 2.2). During this 33-year period severe drought years are few, ENSO oscillations are muted, and PDO is mostly positive and

trending higher (Fig. 2.6*a*-*c*), conditions similar to those of the mid 1500s hiatus already identified. A decrease in fire activity during the early 19th Century has been documented for sites in the interior Northwest (Heyerdahl et al. 2001, 2002), Colorado Plateau (Brown and Wu 2005; Brown et al. 2008), Central Rockies (Donnegan et al. 2001), Southwest (Swetnam and Baisan 1996; Swetnam and Betancourt 1998, Iniguez et al. 2009), northern Mexico (Skinner et al. 2008), and Argentina (Kitzberger et al. 2001). This study supports the conclusions in these earlier studies that this widespread period of reduced fire occurrence was a result of reduced variability in ENSO oscillations but also identifies the lack of severe drought and positive PDO as possible contributing factors.

The last regional fire year was in 1857, roughly synchronous with the last fire recorded for three sites (SWA, SIN, BWA). All of the remaining sites continued to record fires into the 20th Century (although at mostly lower frequencies) with the last local fire year recorded in 1960. A late 1800s loss or reduction in fire-scar-based fire records is a common feature of fire regime reconstructions in the western United States and is generally attributed to the impacts of domestic livestock on fine fuels and changes in human burning practices. The persistence of fire well into the 20th Century on about half of the sites suggests that the timing and intensity of livestock impacts varied within this study area. Post-1900 ignitions may also have been less frequent though both natural and human sources were probable. Earlier disruptions of Native American ignition patterns may have contributed to longer fire intervals as early as ~1800 (see FRI, ROS, and LAW; Fig. 2.2). The deployment of modern fire suppression techniques and equipment after World War II improved suppression efforts on these remote ranges and was more or less synchronous with the last fire-years recorded in the study.

Conclusions

This study revealed a fire seasonality pattern best interpreted as long and bimodal with early and late season peaks and considerable among-site variation. Although fire seasonality varied with climate throughout the period of analysis (1400-1900) it never approached the broad middle to late-season peak observed in modern fire records. The most reasonable interpretation for the cause of this pattern of fire seasonality requires human ignitions and constitutes credible evidence that Native American peoples consistently affected fire regimes in the region. Multi-year, wet-dry cycles synchronized fire at a regional scale with greatest fire activity corresponding to wet-to-dry transitions and least fire activity associated with dry-to-wet transitions. Responses to Pacific-driven oscillations in climate followed a Southwest mode supporting the notion that the pivot-point in the precipitation dipole lies north of the study area. The lack of a clear north-south trend among these sites precludes further speculation regarding the proximity of that pivot-point. Taken together, results support an interpretation that fire regimes were the product of dynamic interactions between a variable climate and human fire use. The relative importance of each of these factors likely varied through time and across the study landscape. Comparable analysis of historic fire seasonality data from other sites are needed and would serve as tests of the interpretations made here. Finally, I must ask whether a completely natural fire regime (lightning ignitions only) is sufficient to maintain pre-1900 forest structure and resilience. The answer to that question will require a better understanding of the extent to which human-ignited fires affected historical fire frequency, severity and extent at both local and broad spatial scales and will ultimately be critical to inform sound long-term forest restoration and maintenance plans.

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Literature Cited

Allen, C. D. 2002. Lots of lightning and plenty of people: an ecological history of fire in the upland Southwest. Pages 143-193 *in* Fire, native peoples, and the natural landscape. Washington, D.C.: Island Press.

Arno, S. F. and K. M. Sneck. 1977. A method for determining fire history in coniferous forests of the mountain west. General Technical Report GTR-INT-42. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah, USA.

Baisan, C. H. and T. W. Swetnam. 1990. Fire history on a desert mountain range: Rincon Mountain Wilderness, USA. Canadian Journal of Forest Research 20:1559-1569.

Beaty, R. M. and A. H. Taylor. 2008. Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. Forest Ecology and Management 255:707-719.

Biondi, F., A. Gershunov, and D. R. Cayan. 2001. North Pacific decadal climate variability since 1661. Journal of Climate 14:5-10.

Briffa, K. R., P. D. Jones, and F. H. Schweingruber. 1992. Tree ring density reconstructions of summer temperature patters across western North America since 1600. Journal of Climate 5:735-754.

Brown, P. M. 2006. Climate effects on the fire regimes and tree recruitment in Black Hills ponderosa pine forests. Ecology 87:2500-2510.

Brown, D. P. and A. C. Comrie. 2004. A winter precipitation ‘dipole’ in the western United States associated with multidecadal ENSO variability. Geophysical Research Letters 31:L09203.

Brown, P. M. and R. Wu. 2005. Climate and disturbance forcing of episodic tree recruitment in a south-western ponderosa pine landscape. Ecology 86:3030-3038.

Brown, P. M., M. W. Kaye, L. S. Huckaby, and C. H. Baisan. 2001. Fire history along environmental gradients in the Sacramento Mountains, New Mexico: Influences of local patterns and regional processes. Ecoscience 8:115-126.

Brown, P. M., E. K. Heyerdahl, S. G. Kitchen, and M. H. Weber. 2008. Climate effects on historic fires (1630-1900) in Utah. International Journal of Wildland Fire 17:28-39.

Butzer, K. W. 1992. The Americans before and after 1492: an introduction to current geographical research. Annals of the Association of American Geographers 82:345-368.

Chavez, F. A. and T. J. Warner, (eds.). 1976. The Dominguez-Escalante journal: Their expedition through Colorado, Utah, Arizona, and New Mexico in 1776. Provo, Utah: Brigham Young University Press.

Collins, B. M., P. N. Omi, and P. L. Chapman. 2006. Regional relationships between climate and wildfire-burned area in the Interior West, USA. Canadian Journal of Forest Research 36:699-709.

Cook, E. R., C. A. Woodhouse, C. M. Eakin, D. M. Meko, and D. W. Stahle. 2004. Long-term aridity changes in the western United States. Science 306:1015-1018.

Covington, W. W. and M. M. Moore. 1994. Post-settlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. In: Sampson, R. N. and D. L. Adams (eds.) Assessing the forest ecosystem health in the Inland West. Journal of Sustainable Forestry 2:153-181.

D'Arrigo, R. D., R. Villalba, and G. Wiles. 2001. Tree-ring estimates of Pacific decadal climate variability. Climate Dynamics 18:219-224.

D´Arrigo, R. D., E. R. Cook, R. J. Wilson, R. Allan, and M. E. Mann. 2005. On the variability of ENSO over the past six centuries. Geophysical Research Letters 32:**L**03711.

Dettinger, M. D., D. R. Cayan, H. F. Diaz, D. M. Meko. 1998. North-south precipitation patterns in western North America on interannual -to-decadal timescales. Journal of Climate 11:3095-3111.

Donnegan, J. A., T. T. Veblen, and J. S. Sibold. 2001. Climate and human influences on fire history in Pike National Forest, central Colorado. Canadian Journal of Forest Research 31:1526-1539.

Gershunov, A., T. P. Barnett, and D. R. Cayan. 1999. North Pacific interdecadal oscillation seen as a factor in ENSO-related North American climate anomalies. EOS, Transactions, American Geophysical Union 80:25-30.

Griffin, D. 2002. Prehistoric human impacts on fire regimes and vegetation in the northern intermountain West. Pages 77-100 *in* Fire, native peoples, and the natural landscape. Washington, D.C.: Island Press.

Grissino-Mayer, H. D. 2001. FHX2 – Software for analyzing temporal and spatial patterns in fire regimes from tree rings. Tree-Ring Research 57:115-124.

Grissino-Mayer, H. D. and T. W. Swetnam. 2000. Century-scale climate forcing of fire regimes in the American Southwest. The Holocene 10:207-214.

Grissino-Mayer, H. D., W. H. Romme, M. L. Floyd, and D. D. Hanna. 2004. Climate and human influences on fire regimes of the southern San Juan Mountains, Colorado, USA. Ecology 85:1708-1724.

Harper, K. T., D. L. Freeman, W. K. Ostler, and L. G. Klikoff. 1978. The flora of Great Basin mountain ranges: Diversity, sources, and dispersal ecology. Pages 81-104 *in* Intermountain biogeography: A symposium, ed. K. T. Harper and J. L. Reveal. Great Basin Naturalist Memoirs 2.

Hessl, A. E., E. McKenzie, and R. Schellhhaas. 2004. Drought and Pacific Decadal Oscillation linked to fire occurrence in the inland Pacific North-west. Ecological Applications 14:425-442.

Heyerdahl, E. K., L. B. Brubaker, and J. K. Agee. 2001. Spatial controls of historic fire regimes: a multiscale example from the interior west, USA. Ecology 82:660-678.

Heyerdahl, E. K., L. B. Brubaker, and J. K. Agee. 2002. Annual and decadal climate forcing of historical fire regimes in the interior Pacific Northwest, USA. Holocene 12:597-604.

Heyerdahl, E. K., P. Morgan, and J. P. Riser II. 2008a. Multiseason climate synchronized historical fires in dry forests (1650-1900), Northern Rockies, USA. Ecology 89:705-716.

Heyerdahl, E. K., D. McKenzie, L. D. Daniels, A. E. Hessl, J. S. Littell, and N. J. Mantua. 2008b. Climate drivers of regionally synchronous fires in the inland North-west (1651-1900). International Journal of Wildland Fire 17:40-49.

Holmgren, N. 1972. Plant geography in the Intermountain Region. Pages 77-161 *in* A. Cronquist, A.H. Holmgren, N.H. Holmgren, and J.L. Reveal, eds. Intermountain flora. Vol. 1. Hafner Publishing Company, New York.

Iniguez, J. M., T. W. Swetnam, and C. H. Baisan. 2009. Spatially and temporally variable fire regime on Rincon Peak, Arizona, USA. Fire Ecology 5:3-21.

Janetski, J. C. and D. B. Madsen. 1990. Wetlands adaptations in the Great Basin. Museum of Peoples and Cultures Occasional papers No. 1. Brigham Young University, Provo, Utah.

Kay, C. E. 2007. Are lightning fires unnatural? A comparison of aboriginal and lightning ignition rates in the United States. Pages 16-28 *in* R. E. Masters and K. E. M Galley (eds.) Proceedings of the 23rd Tall Timbers Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tall Timbers Research Station, Tallahassee, FL.

Keane, R. E., K. C. Ryan, T. T. Veblen, C. D. Allen, J. Logan, and B. Hawkes. 2002. Cascading effects of fire exclusion in Rocky Mountain ecosystems: a literature review. General Technical Report. RMRS-GTR-91. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station.

Kitchen, S. G. and E. D. McArthur. 2003. Ponderosa pine fire history in a southeastern Great Basin stand. Pages 152-156 *in* K. E. M. Galley, R. C. Klinger, and N.G. Sugihara (eds.) Proceedings of Fire Conference 2000. The First National Congress on Fire Ecology,

Prevention, and Management. Miscellaneous Publication Number 13, Tall Timbers Research Station, Tallahassee, FL.

Kitzberger, T., T. W. Swetnam, and T. T. Veblen. 2001. Inter-hemispheric synchrony of forest fires and the El Nino-Southern Oscillation. Global Ecology & Biogeography 10:315-326.

Kitzberger, T., P. M. Brown, E. K. Heyerdahl, T. W. Swetnam, and T. T. Veblen. 2007. Contingent Pacific-Atlantic Ocean influenceon multi-century wildfire synchrony over western North America. Proceedings of the National Academy of Science (USA) 104:543-548.

Mandany, M. H., and N. E. West. 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park, Utah. Ecology 64:661-667.

Margolis, E. Q. and J. Balmat. 2009. Fire history and fire climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA. Forest Ecology and Management 258:2416-2430.

McCabe, G. J. and M. D. Dettinger. 1999. Decadal variability in the strength of ENSO teleconnections with precipitation in the western United States. International Journal of Climatology 19:1399-1410.

McDonald, G. M. and R. A.Case. 2005. Variations in the Pacific Decadal Oscillation over the past millennium. Geophysical Research Letters 32:**L**08703.

Mock, C. J. 1996. Climate controls and spatial variations of precipitation in the western United States. Journal of Climate 9:1111-1125.

Morgan, P., E. K. Heyerdahl, and C. E. Gibson. 2008. Multi-season climate synchronized forest fires throughout the 20th Century, Northern Rockies, USA. Ecology 89:717-728.

Murdock, J. R. and S. L. Welsh. 1971. Land use in Wah Wah and Pine Valleys, western Utah. Science Bulletin, Biological Series, Brigham Young University, vol. XII(4).

Norman, S. P. and A. H. Taylor. 2003. Tropical and north Pacific teleconnections influence fire regimes in pine-dominated forests of north-eastern California, USA. Journal of Biogeography 30:1081-1092.

Peterson, K. L. 1994. Modern and Pleistocene climatic patterns in the West. Pages 27-53 *in* K.T. Harper, L.L. St. Clair, K.H. Thorne, and W.M. Hess (eds.) Natural History of the Colorado Plateau and Great Basin. University Press of Colorado, Niwot, CO.

Pyne, S. J. 1982. Fire in America – a cultural history of wildland and rural fire. Princeton University Press.

Reff, D. T. 1991. Disease, Depopulation, and Culture Change in Northwestern New Spain A.D. 1518-1764. University of Utah Press, Salt Lake City, Utah, USA.

Schmidt, K. M., J. P. Menakis, C. C. Hardy, W. J. Hann, and D. L. Bunnell. 2002. Development of course-scale spatial data for wildland fire and fuel management. General Technical Report RMRS-GTR-87. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Schoennagel, T, T. T. Veblen, W. H. Romme, J. S. Sibold, and E. R. Cook. 2005. ENSO and PDO variability affect drought-induced fire occurrence in Rocky Mountain subalpine forests. Ecological Applications 15:2000-2014.

Sherriff, R. L. and T. T. Veblen. 2008. Variability in fire-climate relationships in ponderosa pine forests in the Colorado Front Range International Journal of Wildland Fire 17:50-59.

Sibold, J. S., T. T. Veblen, and M. E. González. 2006. Spatial and temporal variation in historic fire regimes in subalpine forests across the Colorado Front Range in Rocky Mountain National Park, Colorado, USA. Journal of Biogeography 32:631-647.

Simms, S. R. 2008. Ancient Peoples of the Great Basin and Colorado Plateau. Left Coast Press, Walnut Creek, California, USA.

Skinner, C. N., J. H. Burk, M. G. Barbour, E. Franco-Vizcaino, and S. L. Stevens. 2008. Influences of climate on fire regimes in montane forests of north-western Mexico. Journal of Biogeography 35:1436-1451.

Stevens, S. L. and B. M. Collins. 2004. Fire regimes of mixed conifer forests in the north-central Sierra Nevada at multiple spatial scales. Northwest Science 78:12-23.

Stokes, M. A. and T. L. Smiley. 1968. An Introduction to Tree-Ring Dating. University of Chicago Press, Chicago, IL.

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

Swetnam, T. W. 2005. Fire histories from pine-dominant forest in the Madrean Archpelago. USDA Forest Service Proceedings RMRS-P-36 p35-43.

Swetnam, T. W. and C. H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 11-32 *In* C.D. Allen (technical editor), Fire Effects in southwestern forests-Proceedings of the second La Mesa fire symposium. General

Technical Report RM-GTR-286. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station.

Swetnam, T. W. and J. L. Betancourt. 1990. Fire-Southern Oscillation relations in the southwestern United States. Science 249:1017-1020.

Swetnam, T. W. and J. L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate. 11:3128-3147.

Tausch, R. J., N. E. West and A. A. Nabi. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. Journal of Range Management 34:259-264.

Taylor, A. H. and R. M. Beaty. 2005. Climate influences on fire regimes in the Carson Range, Lake Tahoe Basin, Nevada, U.S.A. Journal of Biogeography 32:425-438.

Taylor, A. H., V. Trouer, and C. N. Skinner. 2008. Climate influences on fire regimes in montane forests of the southern Cascades, California, USA. International Journal of Wildland Fire 17:60-71.

Thornton, R. B. 1987. American Indian Holocaust and Survival: A Population History since 1492. University of Oklahoma Press, Norman, Oklahoma, USA.

Van Horne, M. L. and P. Z. Fulé. 2006. Comparing methods of reconstructing fire history using fire scars in a south-western United States ponderosa pine forest. Canadian Journal of Forest Research 36:855-867.

Westerling, A. L. and T. W. Swetnam. 2003. Interannual to decadal drought and wildfire in the western United States. EOS Transactions, American Geophysical Union 84:545-560.

Westerling, A. L., A. Gershunov, T. J. Brown, D. R. Cayan, and M. D. Dettinger. 2003. Climate and wildfire in the western United States. Bulletin of the American Meteorological Society 84:595-604.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western US forest wildfire activity. Science 313:940-943.

Williams, G.W. 2004. American Indian fire use in the arid west. Fire Management Today. 64:10-14.

Young, J. A., R. E. Eckert Jr, and R. A. Evans. 1979. Historical perspectives regarding the sagebrush ecosystem. In: The Sagebrush Ecosystem: a Symposium. Utah State University, Logan, UT: 1-13.

Table 2.1. Fire chronology sites from the eastern Great Basin arranged from north to south. Names with asterisk were gridded sites.

Site (code)	Mountain range	Location (LAT/LON)	Elevation range (m)	Area (ha)	No. fire-scarred trees	No. of fire years	Recording years (to 2000)
Tom's Creek Canyon (TOM)	Deep Creek	39˚52´ N/113˚52´ W	2475-2480	5	6	16	294
Swasey Mountain (SWA)	House	39˚24´ N/113˚19´ W	2370-2620	30	14	15	443
Sinbad Springs (SIN)	House	39˚23´ N/113˚19´ W	2375-2395	1	4	20	439
Burnt Mill Canyon (BMC)*	Snake	39˚02´ N/114˚16´ W	2365-3230	455	110	99	734
Big Wash (BWA)	Snake	38˚52´ N/114˚14´ W	2480-2560	20	28	40	307
Lawson Cove (LAW)*	Wah Wah	38˚37´ N/113˚34´ W	2195-2685	430	137	124	707
Frisco Peak (FRI)	San Francisco	38˚32´ N/113˚17´ W	2580-2770	60	27	59	796
Indian Creek Canyon(INC)*	Tushar	38˚23´ N/112˚23´ W	2365-2550	130	32	57	754
Rose Spring Canyon (ROS)	Wah Wah	38˚17´ N/113˚36´ W	2240-2330	120	30	116	661
Right Fork, Beaver Creek Canyon (RBC)*	Tushar	38˚12´ N/112˚27´ W	2360-3080	910	167	105	501

Table 2.2. Criteria for assigning fire event seasonality based upon fire-scar ring positions (UNK= unknown, RB = ring boundary, EE = early, early wood; ME = middle, early wood; LE = late, early wood; LW = late wood). Yes (Y) indicates a minimum of one fire scar of that class is listed for the fire year at a given site.

Presence (Y) or absence (N) of each class of inter- or intra-ring fire scar among those assigned to each fire event.					Assigned season
UNK	RB	EE	EM	LE+LW	
Y	N	N	N	N	Unknown
Y or N	Y	N	Y	N	
Y or N	Y	N	N	N	Dormant
Y or N	Y	Y	Y or N	N	Early
Y or N	N	Y/ EE>ME	Y	N	
Y or N	N	Y or N	Y/ME≥EE ME≥LE+LW	Y or N	Middle
Y or N	Y	N	Y	Y	Late
Y or N	N	N	Y or N	Y/LE+LW>ME	
Y or N	Y or N	Y	Y or N	Y	Multi-season

Table 2.3. Regional, local, and no-fire years summaries grouped by century (1400-1999). All dates are low-filter (≥33% of recording sites) except those in bold which are high-filter (≥50% of recording sites) regional fire years.

	Fire years			
	Regional		Local	No-fire years
Century	Dates	Total	Total	Total
1400-1499	**1407** **1423** 1444 1475 1490 1495 1497	7	16	77
1500-1599	1506 1522 1523 1525 **1532** 1538 1543 1547 **1552** 1580 1589 1598	12	50	38
1600-1699	1605 1619 1623 **1630** **1632** 1637 **1645** 1650 1653 1666 1668 1670 1679 **1684** **1685** 1687 1693 1694 1695 1696	20	55	25
1700-1799	1700 1701 **1703** **1707** 1708 1715 **1722** **1728** 1729 **1735** **1736** 1738 **1751** **1755** **1763** 1777 **1780** **1786** 1788 1795	20	66	14
1800-1899	1800 1804 **1806** 1809 **1813** 1822 **1855** 1857	8	59	33
1900-1999		0	14	86

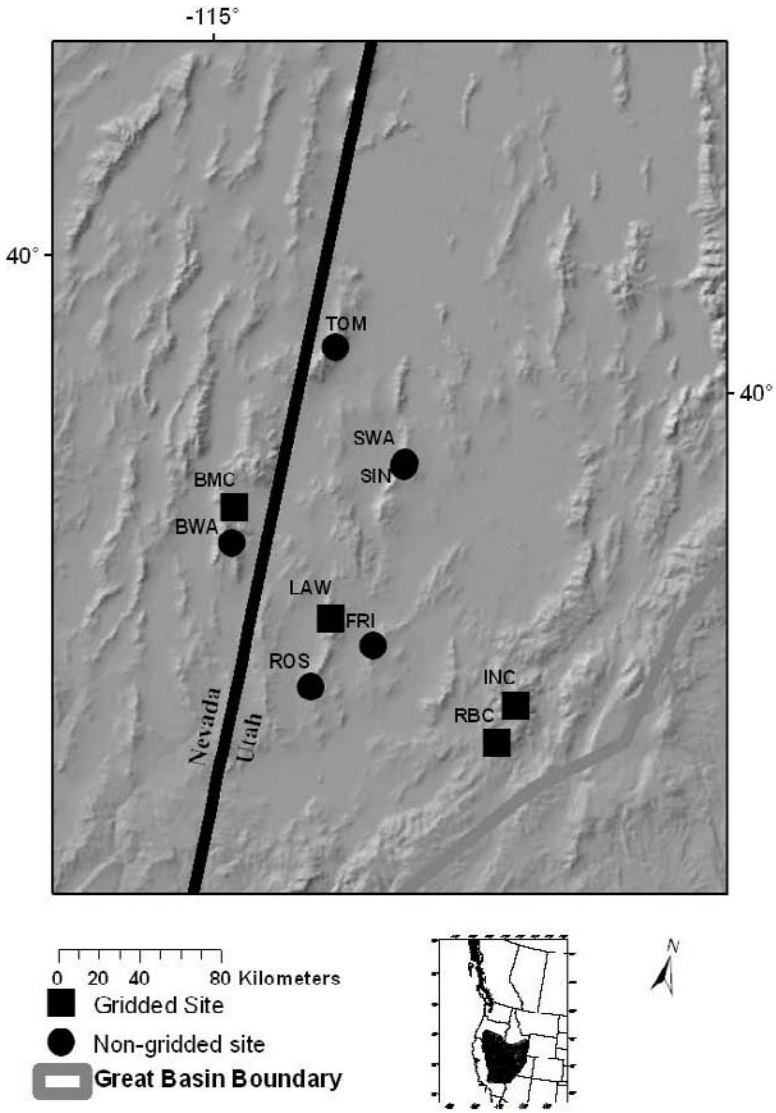

Figure 2.1. Map of eastern Great Basin mountain ranges with the location of 10 fire history sites. Study sites include the Right Fork of Beaver Creek (RBC) and Indian Creek Canyons (INC) on the Tushar Mountains, Frisco Peak (FRI) on San Francisco Mountain, Lawson Cove (LAW) and Rose Spring Canyon (ROS) on the Wah Wah Range, Burnt Mill Canyon (BMC) and Big Wash (BWA) on the South Snake Range, Sinbad Springs (SIN) and Swasey Mountain (SWA) on the House Range, and Tom's Creek Canyon (TOM) on the Deep Creek Mountains. A gridded sampling design was employed for sites marked by squares while targeted sampling strategies were employed at sites marked by circles.

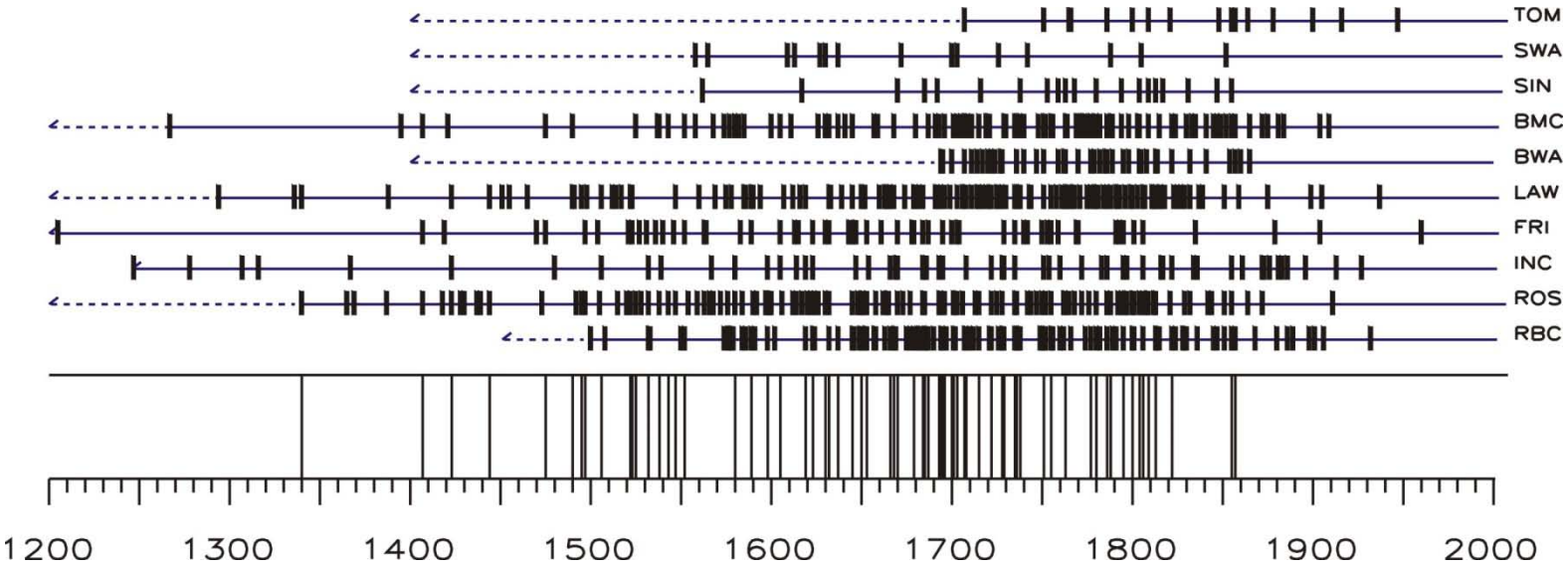

Figure 2.2. Fire chronologies for the 10 eastern Great Basin sites arranged from north (top) to south. Horizontal lines are composites of fire record for each site. Solid lines indicate at least one tree is in recording status. Short vertical lines mark fire dates. Locations of low-filter (

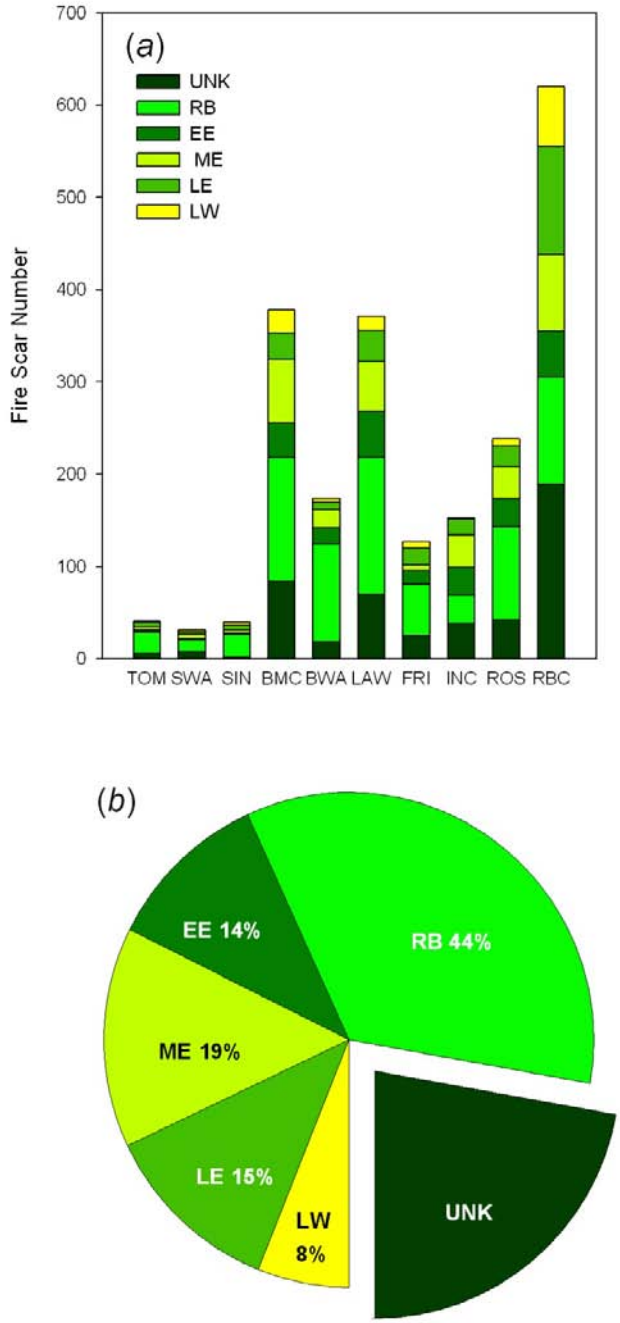

Figure 2.3. Total intra- and inter-ring fire scars by position for individual study sites (*a*) and percentages across all sites (*b*). UNK = Unknown; RB = ring boundary; EE = early early-wood; ME = middle early-wood; LE = late early-wood; LW = late-wood.

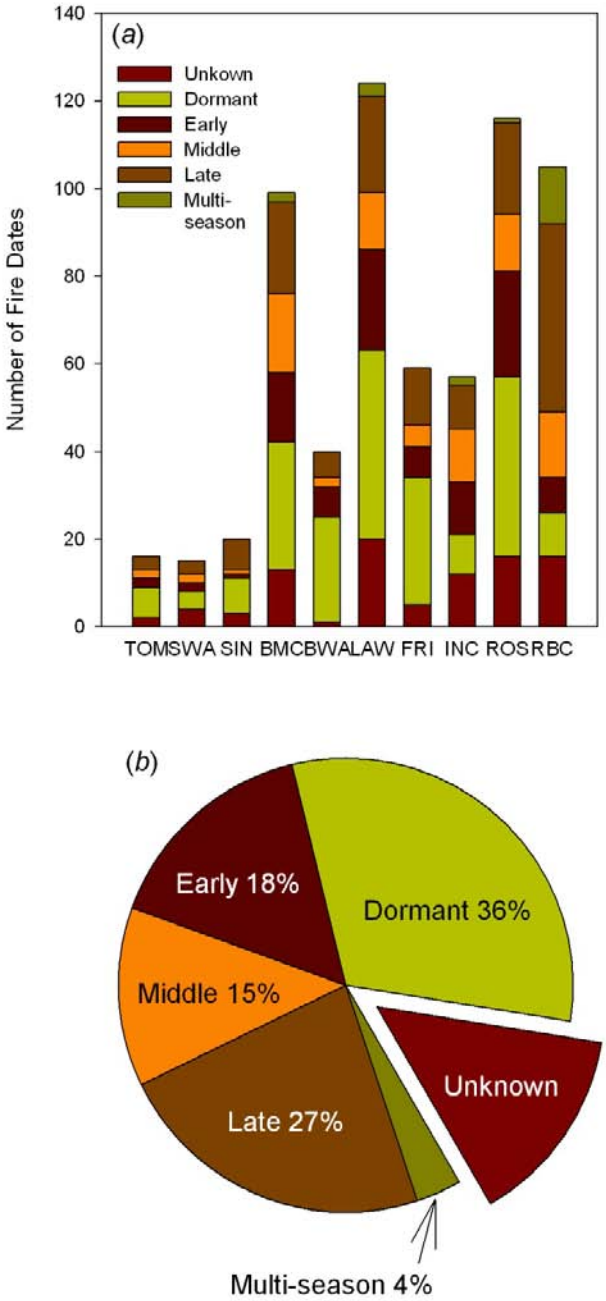

Figure 2.4. Seasonality of fire events for individual sites (*a*) and across all sites (*b*) using established criteria.

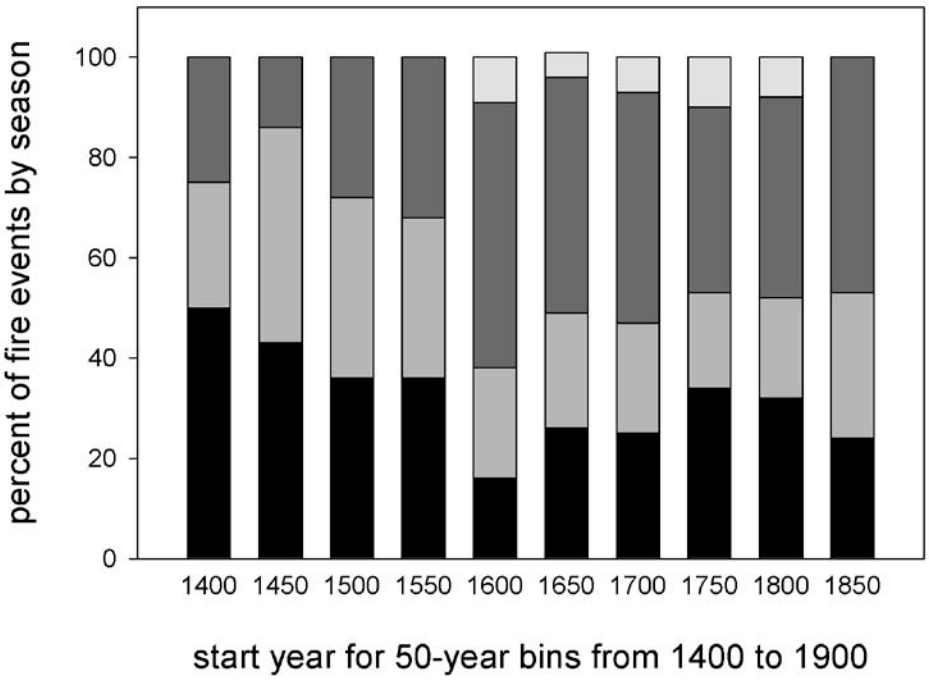

Figure 2.5. Fire event seasonality through time (1400-1900) expressed as a percent of all fire events in each 50-year bin excluding unknown and dormant season seasonal classifications. Seasonality is black = early; medium grey = middle; dark grey = late; and light grey = multi-season. Years listed on horizontal axis indicate the start of 50-year bins.

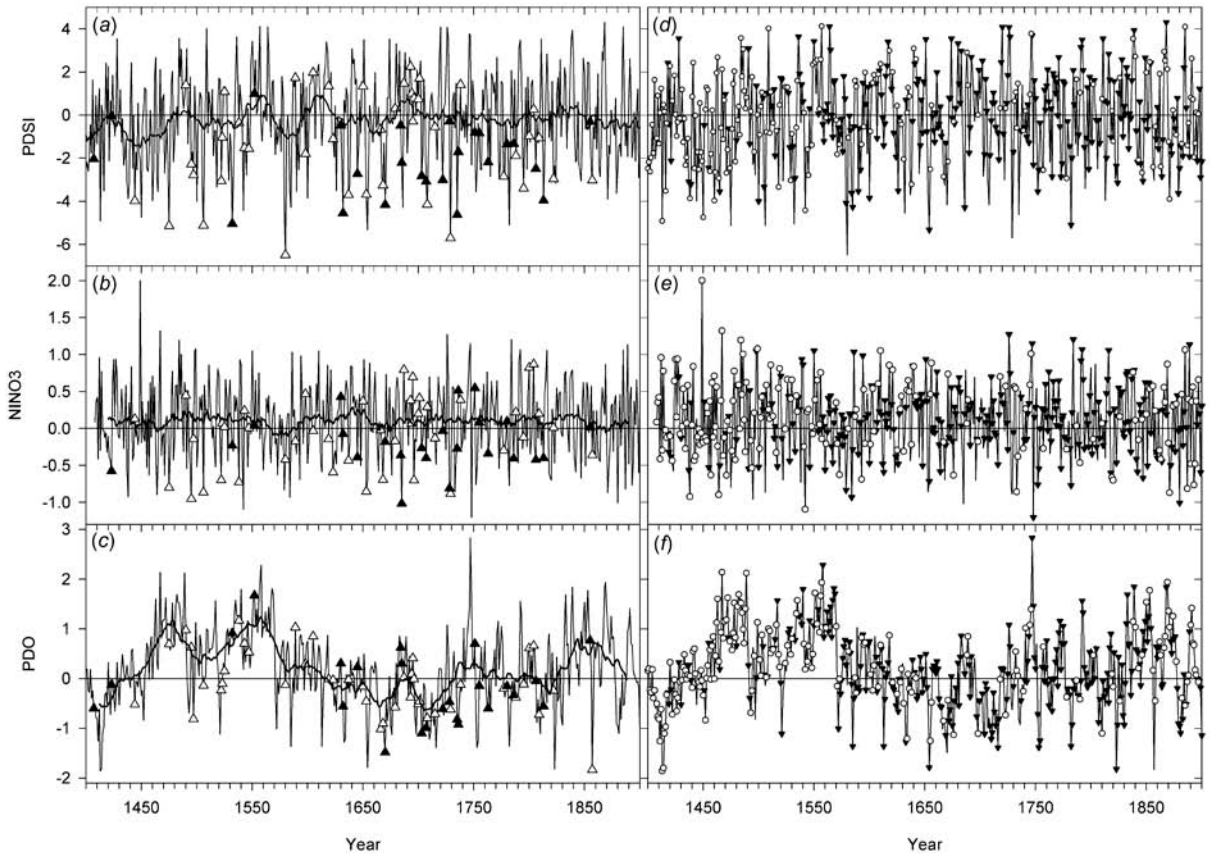

Figure 2.6. Variation (1400-1900) in annual climate indices plotted against low- (large open triangle) and high-level (large filled triangle) regional fire years (*a-c*) and local- (small filled triangle) and non-fire years (small open circle; (*d-f*). Light lines are indices of annual climate variation. Heavy lines (*a-c* only) are climate index smoothed with cubic splines that retain 50% of variation over segments of 25 years.

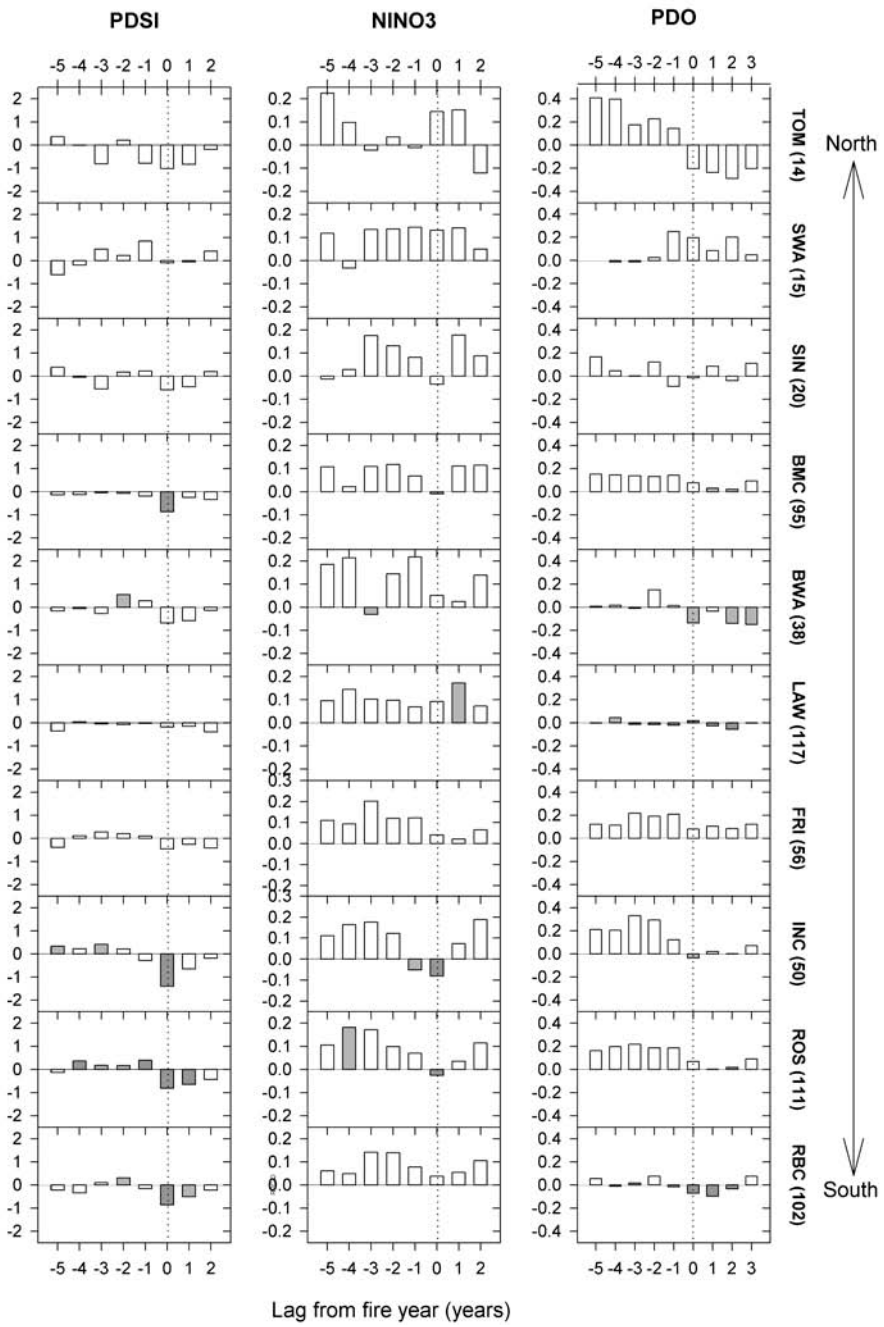

Figure 2.7. Superposed epoch analysis (SEA) of average climate departures during fire years at each of 10 eastern Great Basin sites. Numbers in parentheses are total fire years at each site. Analysis is for 5 years before and 2 or 3 (PDO) years following fire-years. Light and dark shading indicate departures exceeding 95 and 99% confidence intervals.

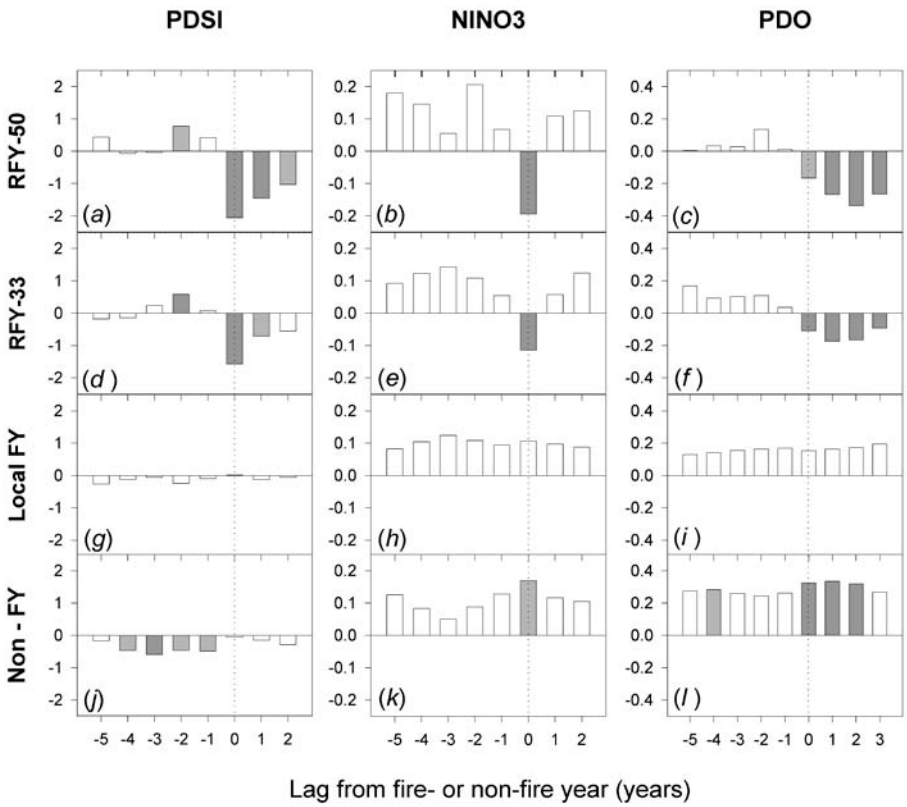

Figure 2.8. Superposed epoch analysis (PDO) of average departures for PDSI, NINO3, and PDO during high- and low-filter regional fire years (RFY-50; *a-c* and RFY-33; *d-f*) and local (*g-i*) and non-fire years (*j-l*). Analysis is for 5 years before and 2 or 3 (PDO) years following fire-years. Light and dark shading indicate departures exceeding 95 and 99% confidence intervals.

CHAPTER 3 – HISTORIC FIRE REGIME AND FOREST VARIABILITY ON TWO EASTERN GREAT BASIN FIRE-SHEDS (USA)

Abstract: Management of naturally forested landscapes requires knowledge of key disturbance processes and their effects on plant community composition and structure. Spatially-intensive fire and forest histories provide valuable information about how fire and vegetation vary and interact on heterogeneous landscapes. I constructed 800-yr fire and recruitment chronologies for two eastern Great Basin fire-sheds using fire-scar and establishment evidence from 48, variable-radius recruitment plots (500 m grid) and from fire-scarred trees between plots. Fire-sheds are located in the Snake Range of eastern Nevada (BMC) and Wah Wah Range of western Utah (LAW) and span a range in elevation and vegetation zones typical for the region. Estimates of point mean fire interval varied more than 10-fold at BMC (7.8-125.6 years) and LAW (13.3-138.4 years). Control of fire frequency variation was largely manifest at the landscape scale at BMC and at finer spatial scales at LAW where topography was more broken. At BMC, a distinct within-fire-shed pattern in fire frequency variation was difficult to explain without invoking the possibility of human-caused ignitions. A majority of fires were small (<10 ha) but large fires (≥100 ha) accounted for 78% at BMC and 89% at LAW of cumulative area burned. Tree recruitment for mid-elevation mixed-conifer stands was somewhat episodic and asynchronous among plots. Recruitment pulses were synchronous with multi-decade fire quiescent periods and often followed landscape-scale fires. I concluded that fire frequency was under strong topographic control and that fire severity was mixed and variable through time and space resulting in a dynamic mosaic of variable-aged vegetation intermixed with long-lived, fire-resilient trees and open shrub-steppe communities. Fire regime and forest composition change began in the early (LAW) and mid-1800s (BMC), causing shifts in composition and structure at the stand scale and homogenization at the landscape scale. I recommend that management

strategies prioritize the use of fire and surrogate treatments on mid-elevation forests that have deviated most from historic conditions and associated shrub-steppe communities where conifer encroachment is greatest. Planned disturbances should be of mixed severity and size to recreate vegetation mosaics at the scale documented in the study.

Key words: Dendrochronology, Point mean fire interval, Mixed-severity fire, Multi-scale analysis, Anthropogenic fire, Fire restoration

Introduction

Successful forest management and restoration strategies require a thorough understanding of natural disturbance regimes and their effects on vegetation. Fire histories provide a means to describe and quantify variation in fire regimes from past eras at various scales of time and space (Morgan et al., 2001; Keane et al., 2002). To effectively assess fire regime variability, fire regime reconstructions must have resolution at the same scales in which variation occurs (Ricklefs, 1987; Weins, 1989; Levin; 1992). Fire histories derived from tree-ring evidence provide a means for assessing the effects of broad-scale (e.g. climatic) to fine-scale (e.g. topographic) drivers of fire regime variability over long periods (Swetnam and Baisan, 1996; Swetnam and Betancourt, 1998; Brown and Shepperd, 2001; Brown et al., 2001; Heyerdahl et al., 2001; Taylor and Skinner, 2003; Grissino-Mayer et al., 2004; Sibold et al., 2006). For example, investigations of the role of climate as a driver of historic fire patterns are possible because tree-ring evidence of past fires can be fixed with annual and often seasonal precision. Consequently, considerable progress has been achieved in assessing the role of climate in synchronizing fire occurrence regionally as the number and spatial representation of cross-dated

fire chronologies increased in recent decades (Grissino-Mayer and Swetnam, 2000; Brown et al., 2008; Heyerdahl et al., 2008a, 2008b; Kitzberger et al., 2001, 2007; Westerling and Swetnam, 2003; Swetnam and Betancourt, 1990, 1998).

Fire histories constructed from spatially-precise sampling strategies are useful for evaluating fine-scale controls of fire regime variation (Heyerdahl et al., 2001; Beaty and Taylor, 2008; Iniguez et al., 2008). Investigations using intensive sampling strategies suggest that topographic variables, interacting with differences in vegetation, affect fire regime characteristics differently, depending on the biophysical context. For example, historic fire frequency varied with aspect in the Blue Mountains (Oregon and Washington) but only when terrain was steep, and when contrasting topographic facets were large and separated by barriers to fire spread (Heyerdahl et al, 2001). In addition, fire frequency did not differ with elevation when fuels matrices were continuous. Similarly, studies in the Klamath Mountains of California (Taylor and Skinner, 1998) failed to detect differences in fire frequency related to elevation (slope position) even though fire severity increased as species composition and age structure changed on intermediate and upper slopes. In contrast, fire frequency differed significantly in relation to differences in elevation and corresponding forest types in the southern Cascade Range, California (Bekker and Taylor, 2001) and was inversely related to fire size. In another California study (Lake Tahoe Basin), fire frequency, fire severity, and cohort patch size varied in mixed-conifer forests with aspect and slope position (Beaty and Taylor, 2008). In the Sacramento Mountains (New Mexico) fire was more frequent at lower than at higher elevations but did not differ between ponderosa pine (*Pinus ponderosa*) and mixed conifer forest types (Brown et al., 2001). Fires were also more common on the west side of the range than on the more topographically heterogeneous east side. A similar pattern was reported for the Santa Catalina

Mountains (Arizona) in which a larger mean fire size was inferred as the reason for the higher fire frequency associated with a more homogeneous landscape (Iniguez et al, 2008). These results inferred that fire frequency was controlled at the landscape rather than stand level, a result similar to what has been observed elsewhere (Heyerdahl et al., 2001; Margolis and Balmat, 2009).

When forest histories are constructed in concert with fire histories they allow examination of linkages between fire regime and vegetation dynamics (Taylor and Skinner, 2003; Brown and Wu, 2005; Margolis and Balmat, 2009; Bekker and Taylor, 2010). Synchronous mortality or recruitment may infer mixed- (Kaufmann et al., 2000; Fulé et al., 2003; Taylor and Skinner, 2003;) or high-severity (Sibold et al., 2006; Beaty and Taylor, 2008; Margolis and Balmat, 2009; Bekker and Taylor, 2010) fire regimes depending on the presence (absence) of fire scars and spatial scale. Demographic data are also used to delineate fire perimeters for fire-size calculation (Everett, 2008; Iniguez et al., 2008, 2009; Margolis and Balmat, 2009; Bekker and Taylor, 2010). Linked analyses facilitate assessment of the relative importance of fire regime in driving vegetation dynamics and of how stable that relationship might be over time (Kaufmann et al., 2000; Fulé et al., 2003; Brown and Wu, 2005; Iniguez et al., 2009).

Existing studies have considerable value for developing ecologically sound management strategies locally, however, it remains unclear how broadly results should be extrapolated. Additional work using intensive sampling strategies is needed (Taylor and Skinner, 2003; Swetnam, 2005; Iniguez et al., 2008) particularly in regions and biophysical settings that have as yet been understudied. For example, most studies completed to date were conducted on landscapes of more or less continuous forest. Thus, additional work is needed to determine how

fire regimes vary on topographically variable landscapes with spatially-complex vegetation patterns. The mountains of the eastern Great Basin represent such an opportunity.

The Great Basin encompasses the northern, elevated section of the Basin and Range Province of western North America. It includes 100+ mountain ranges with a generally north/south orientation, separated by broad, internally-drained desert valleys. Elevation for 33 of these ranges exceeds 3,050 m (10,000 ft.; Grayson, 1993). The climate is dry due to rain shadow effects of the Sierra Nevada and Cascade Ranges to the west and Rocky Mountains to the east (Peterson, 1994). Seasonality of precipitation varies along a geographic gradient with the importance of winter and spring Pacific frontal storms decreasing and summer monsoons increasing as one travels from north to south and from west to east. Locally, steep precipitation gradients are dictated by elevation and orographic position.

Great Basin vegetation types occur in more or less distinct zones across gradients of elevation reflecting parallel gradients in temperature and precipitation (Holmgren, 1972; Harper et al., 1978). Zonation is further modified by slope, aspect, and substrate. Drought tolerant sub-shrubs and grasses dominate plant communities of arid valleys and dry foothills. Sagebrush (*Artemisia* spp.)-grass steppe communities occupy a broad zone from the alluvial bajadas at mountain bases up to dry mid-elevation landscapes. Species diversity increases in this type with increasing elevation and in places stands of tall shrubs such as curlleaf mountain mahogany (*Cercocarpus ledifolius*) displace sagebrush, sometimes forming extensive mono-specific stands. Mosaics of various shrubland and forest types are common at middle elevations. A pinyon-juniper woodland belt is superimposed near the center of the sagebrush-grass steppe type on all but northern ranges and has expanded during the past century into the upper and lower shrublands that bracket its core distribution (Tausch et al., 1981; Miller et al. 2008). Woodlands

occupy the tops of many mountain ranges of more modest elevation. Various combinations of pine (*Pinus*), fir (*Abies* and *Pseudotsuga*), and spruce (*Picea*) with primarily Rocky Mountain affinities occupy mixed conifer and sub-alpine forest zones on taller mountains. Conifer diversity decreases from east to west reflecting bottlenecks in post-Pleistocene dispersal (Wells 1983). A treeless alpine zone is found on the highest peaks. Because of the steepness of many ranges, multiple abrupt changes in vegetation, and therefore fuels matrices, may be juxtaposed over short distances.

Published fire histories for sites from the Great Basin are few (Heyerdahl et al., 1995; Kitchen and McArthur, 2003). This oversight is likely due, at least in part, to a perceived lack of trees, such as ponderosa pine, that function as good recorders of multiple, nonlethal fires. Interestingly, at the initiation of this study little had been published describing fire regimes for mountains of the eastern or southern Great Basin where ponderosa pine is common if not abundant. Published results of studies that ran concurrent with this study begin to fill the void for historical fire regime information relevant to the Great Basin. One addresses landscape-scale variation in fire regime in a pinyon pine dominated landscape in central Nevada (Bauer and Weisberg, 2009). In addition, two masters theses investigate fire history patterns on mountains located in the southern Great Basin where ponderosa pine is present (Jamieson, 2008; Kilpatrick; 2009). These last two studies reveal relatively high frequency for surface fires up to the late 1800s with a reduced frequency extending into the 1900s.

My objective was to produce spatially explicit, multi-century fire and tree recruitment histories for two fire-sheds representative of montane landscapes of the eastern Great Basin and to address five questions with regards to fire regime and tree recruitment. I selected study fire-sheds with geographic proximity to ensure similarity in climatic pattern while allowing for

variability in topography and anthropogenic use history. As used here, the term ‘fire-shed’ designates a topographic unit somewhat sympatric to one or more small watersheds and corresponding to an area within which barriers (i.e. bare ridges, cliffs, waterways, etc.) are sufficiently permeable to allow fire to spread among all components of the landscape. My questions were; (1) How did fire frequency vary within each fire-shed in relation to topography? Specifically I was interested in the range and pattern of variation and spatial scale at which it would be manifest. (2) When did historic fire patterns change in these fire-sheds and what were possible causes for the change? (3) How did fire size vary within fire-sheds and through time? (4) How did species-specific tree recruitment vary through time and space in relation to fire, fire quiescent periods, and climate? (5) What can be inferred about fire severity from tree recruitment patterns?

Study Fire-sheds

I selected two fire-sheds for this study, one on the South Snake Range, White Pine County, Nevada, USA and a second approximately 65 km southeast on the Wah Wah Mountains, Millard County, Utah (Fig. 3.1). The first (BMC) is located within Great Basin National Park and includes forested and non-forested portions of Mill Creek and Burnt Mill Canyons, east slope drainages near the north end of the range. Maximum elevation is 3,344 m at the summit of Buck Mountain. The lower limit of the study area was defined by the park boundary at about 2,300 m. This study area is approximately 4 km long (east to west) and 1.5 km wide, with a total area of ~600 ha (Fig 3.1). Mean annual precipitation at Great Basin National Park headquarters (elev. 2,080 m), located approximately 4 km south of study area, is 336 mm (WRCC, 2009). I estimate that mean annual precipitation for the study area varies with elevation from approximately 340 to 1,000 mm with differences attributable primarily to differences in winter-

spring snowfall. Parent material is predominantly quartzite and granite with some limestone at lower elevations. Mill Creek is a small stream ≤ 2 m across and, aside from a few small springs, is the only perennial surface water in the study area. Conifers dominate on north and east aspects with gradual transitions between subalpine and dry mixed conifer and pinyon-juniper woodland types. Important tree species include Englemann spruce (*Picea englemannii*), limber pine (*Pinus flexilis*) ponderosa pine, single-needle pinyon (or just pinyon) pine (*P. monophylla*), Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), and Utah Juniper (*Juniperus osteosperma*). Pockets of quaking aspen (*Populus tremuloides*) are found along Mill Creek and scattered throughout the mixed conifer and subalpine forest types. Curlleaf mountain mahogany occurs as scattered plants in sagebrush-steppe, mixed conifer, and pinyon-juniper types and as solid stands on some warmer south-facing slopes up to ~3,000 m. Mid-elevation shrub-steppe communities dominated by medium-statured shrubs such as mountain sagebrush and mountain snowberry (*Symphoricarpos oreophilus*) are widespread on drier slopes between mixed conifer forests and pinyon-juniper woodlands and are in various stages of invasion by pinyon, white fir, and mountain mahogany.

The second fire-shed (LAW) is located near the north end of the Wah Wah Range in the Lawson Cove drainage. The high point is at the southern extreme on Ranch Peak at 2,718 m. The study area extends northward from Ranch Peak 3.5 km down two main branches of the drainage to ~2,200 m, near the point where pinyon-juniper woodlands become discontinuous (Fig. 3.1). Study area width varied from 0.5 to 2 km. Total area was approximately 525 ha. Mean annual precipitation on Ranch Peak is 343 mm (1999-2009; data on file USDA Forest Service, Shrub Sciences Laboratory, Provo Utah) from which I estimate mean annual precipitation for this study area to be 240 to 340 mm. Parent material is primarily limestone and dolomite with lesser

amounts of volcanic dacite. Considerable rock is exposed on ridge tops, as cliff faces 1-30 m in height, and on talus slopes. Higher cliffs form natural boundaries for parts of the east and west sides of the study area. There is no perennial surface water within or near the study area. White fir is the most abundant tree species on north and east facing slopes down to about 2,300 m and is also common in canyon bottoms at lower elevations. Pinyon pine and Utah juniper dominate lower elevations and are found throughout the study area, although largely restricted to west and south-facing slopes and dry rocky ridges at upper elevations. Rocky Mountain juniper (*Juniperus scopulorum*) becomes common and partially replaces Utah juniper at upper elevations. Ponderosa pine is found as scattered trees and in small, locally-dominant stands on ridge tops, east and north facing slopes and along drainage bottoms. A relatively large stand (32-ha) of Great Basin bristlecone pine (*Pinus longaeva*) occupies Ranch Peak, where it is variably dominant to co-dominant with white fir. Several additional clusters are scattered within the fire-shed on north and east-facing slopes generally above 2,500 m. Douglas fir is widely scattered on upper north facing slopes and in drainage bottoms. Numerous openings are dominated by low and medium-statured shrubs such as black sagebrush (*Artemisia nova*), green ephedra (*Ephedra viridis*), and littleleaf mountain mahogany (*Cercocarpus intricatus*).

The Great Basin has likely been inhabited by humans for at least 13,000 years. The Snake and Wah Wah Ranges were located near the convergence of influence for Western Shoshone, Ute, and Southern Paiute cultures immediately prior to Euro-American settlement (Simms, 2008). These Numic-speaking inhabitants practiced mobile, hunter-gatherer economies in contrast with the more sedentary, semi-agricultural model of the Fremont that is known to have occupied the area during the thirteenth century and possibly for some time before (Simms, 2008). Undoubtedly, fire was used by all of these groups as an essential tool for manipulating natural

environments (Williams, 2004), however, knowledge regarding specific practices used in this region and how they or their impacts on vegetation might have varied through space and time is lacking (Griffin, 2002).

The first impacts of European colonization on Native Americans in the eastern Great Basin were likely from diseases to which they had little resistance. Elsewhere in North America, repeated epidemics in the 1500s to 1700s reduced Native American populations by 70% or more (Thornton 1987; Reff 1991; Butzer 1992). Evidence for a disruption in human fire ignition patterns associated with such drastic depopulation might be anticipated in the Great Basin.

Although the timing of Euro-American settlement in the valleys adjacent to the fire-sheds was approximately the same (1860's), the nature of settlement-related impacts differs considerably. Pre-1900 logging for ponderosa pine on the South Snake Range was extensive in support of local settlement and mining activities as evidenced by numerous decayed stumps in many areas. Conversely, logging in the Lawson Cove drainage was restricted primarily to a few ponderosa pines in drainage bottoms. The BMC fire-shed was intensively grazed in summer by domestic cattle from the 1860's to 1999 when grazing permits were terminated (NPS, 2009). In contrast, significant numbers of livestock did not have access to the LAW fire-shed until large numbers of domestic sheep were brought to winter in Wah Wah Valley beginning in the 1880's and continued to the mid-1900's (Murdock and Welsh, 1971). Sheep spent summer and early fall months on better-watered ranges to the east. The impact these herds may have had on vegetation of the LAW fire-shed is unknown, but past use would have been limited, as it is today, to periods when snow was present (for drinking water) but not so deep as to hinder movement and access to forage. Although sheep and cattle currently spend winter and spring in the general area, they have not used the rugged terrain of the fire-shed for several decades. Subsequently, because of

the “undisturbed character, typicality, potential for scientific study [related to bristlecone pine presence], and lack of conflict” (Tuhy, 1985) associated with the northern part of the Wah Wah Mountain Range a major portion (including all of the LAW fire-shed) was nominated as a USDI Bureau of Land Management Research Natural Area.

Methods

Field sampling and sample preparation and analysis

Evidence for fire and tree recruitment history reconstructions was collected from 24 plots in each fire-shed and opportunistically from fire-scarred trees found individually and in clusters between plots. I first selected a location for a reference plot in each fire-shed based on an observed abundance of fire-scarred trees. Subsequent plots were placed across the study area at 500-m intervals using Universal Transverse Mercator (UTM) coordinates along cardinal directions from the reference plot and were arranged to span a broad range in elevation and forest type (Fig. 3.1). Plot identity within the resulting grid was specified by alphanumeric couplets designating row (east-west) and column (north-south) location. In the field, plot centers were located using hand-held GPS receivers accurate to 15 m. When plots fell on unsuitable terrain (i.e. roads, cliffs, rock outcrops), plot centers were moved 50 m in a randomly-selected cardinal direction. I determined elevation, aspect, slope, and slope position (lower, middle, upper, ridge) for each plot.

I used an n-tree density-adapted sampling method to select sample trees for each plot (Jonsson et al., 1992, Lessard et al., 2002) with a maximum plot size of 0.5 ha (40 m radius). Sample trees included those nearest to plot center up to a maximum of 36 trees (generally 30)

with at least 20-cm diameter breast height (DBH = 1.4 m). Remnants (stumps, snags, and logs) were included unless judged to not be datable due to rot. Unsampled remnants were tabulated and classified by the presence or absence of surface char. Species, DBH, diameter at sample height, and distance to plot center were determined for each sample tree. Increment cores were removed from live trees without fire scars at 10-20 cm above ground level. Most cores typically did not intersect pith in which case individual trees were cored up to four times to secure samples with inner rings as near as possible to pith. Surface fire evidence was collected from fire-scarred trees with a chainsaw as one or more partial cross-sections cut so as to extend through fire-scarred portions of the bole and pith (Arno and Sneck, 1977). Cross-sections were also cut from sound, non-scarred remnants at a point estimated to have been approximately10-20 cm above ground level. I searched for and sampled additional fire-scarred trees within a minimum search radius of 80 m of each plot center (2 ha). I further employed a targeted approach to find and sample fire-scarred trees between plots (Van Horne and Fulé, 2006) where one to four individuals expended 1 to 4 hrs per plot. Species, condition (live, snag, stump, log), UTM location, elevation, aspect, and slope position were determined for each non-plot sample tree.

All cores and cross-sections were stabilized and surfaced using combinations of band saw, planer, belt sander, and hand sanding until cell structure became visible using a binocular microscope. Each sample was independently cross-dated by at least 2 analysts using a combination of locally-developed master ring-width chronologies (skeleton plots) and lists of marker (narrow) years (Stokes and Smiley, 1968). Samples that could not be dated with annual accuracy were excluded from further analysis. I estimated the recruitment date for each tree as its pith date at sample height. Although a conservative measure for germination year, I employed no correction factors to pith dates because of unknown and variable time lapses between

germination year and the year a tree reached sample height (10-20 cm). I estimated years to pith for samples in which pith was missing by matching sample ring curvature and spacing to that of concentrate rings in a transparent overlay.

Fire frequency analysis

I assigned a calendar year to each fire scar. Fire scars that could not be dated to annual accuracy were not included in the analysis. Abrupt, multi-year changes in ring widths indicative of a sudden growth release or suppression and injuries of uncertain origin were treated as evidence of fire if at least one tree at the study site had a fire scar corresponding to the same year. I assigned an intra- ring (i.e. early, middle, or late early-wood or late-wood) or inter-ring (ring boundary) position to each fire scar when conditions permitted. Fire scars associated with very narrow rings or eroded ring structures were classified as unknown. Inter-ring (dormant season) scars result from fires that occur either late in the season after ring growth is complete or early the following year before ring growth is initiated. Typically, these scars are assigned to a calendar year based upon the predominant pattern of fire seasonality in modern or historic records. I assigned ring-boundary scars to pre-boundary years when one or more trees at the site had evidence (late early-wood or late-wood scars) of late season fire in the same year and to the post-boundary year when evidence (early or middle early-wood) suggested a fire had occurred in the following year. Based upon a pilot study that showed a greater number of early-season fires tha middle- or late-season fires (Kitchen and McArthur, 2003), fire was assigned to the post-boundary year when these criteria proved inconclusive. I assigned fire event (year by site combination) seasonality as unknown, dormant, early, middle, late, and multi-season using criteria that assess, as a group, the ring positions of the complete set of fire scars associated with each event (see chapter 2).

I constructed composite fire chronologies (Dieterich, 1980) for plots using records from all fire-scarred trees located within 40 m of plot centers (0.5 ha; 0-16 trees) and for tree clusters outside of plots (two-five trees) with a maximum inter-tree distance of 80 m. A related study (Chapter 1) suggested that a sample area of ~0.5 ha provided an appropriate balance between the opposing risks for errors of omission and errors of false inclusion at small spatial scales. I calculated mean fire interval (MFI) estimates from composite and single-tree chronologies that included a minimum of three fire years (two intervals) using program FHX2 (Grissino-Mayer, 2001). I treat composite MFI values as estimates of point mean fire interval (PMFI) when the composite was based upon the combined records of three or more trees. PMFI estimates were derived from single-tree and paired-tree records by multiplying MFI values by a correction factor of 0.8 to account for a higher probability of unrecorded fires (errors of omission). Topographic position and elevation values for PMFI estimates derived from two or more trees were based on plot center data or averaged single-tree values (non-plot trees). Single-tree and cluster chronologies were assigned alphanumeric labels based upon the nearest plot (Fig. 3.2). I visually explored fire frequency spatial variation for both sites using 2-dimentional contour plots where contour lines demark hypothesized gradients in PMFI.

Fire size analysis

Many of the fires recorded at these sites probably burned outside of the sampling grids and in irregular patterns inside the grids making quantification of historic burn area difficult. Within fire-sheds, fire-size estimation is further hampered by the low density and uneven distribution (spatial and temporal) of recording trees. In addition, temporally precise (tree-ring) evidence is spatially limited to locations where mature trees were present and fire intensity was conducive for at least some trees surviving with newly-formed fire-scar evidence. Evidence

based upon pulses in recruitment could not be used for fire-size estimation because it generally lacked sufficient temporal precision to link to specific fire events against a background of relatively high fire frequency. Acknowledging these limitations, I estimated two-dimensional relative burn area or fire size for all fires using UTM coordinates from fire-scarred trees. For simplicity, all scars assigned to a given year were treated as though caused by the same fire. I estimated that fire size was equal to the area of the smallest rectangle that could include all coordinates of recording-tree with its sides oriented along cardinal directions (minimum fire size = 1 ha). Although imperfect, this approach minimizes the effects of an unequal record through time and uneven distribution of fire record through space. The latter is particularly important at BMC where significant low and mid-elevation portions of the study area were not historically forested. Given the obvious lack of precision, fire size values are best treated as indices of relative fire size rather than as estimates of actual fire extent. I classified fires as small (<10 ha), medium (≥10 and <100 ha), large (≥100 ha), and landscape (≥200 ha) based on these criteria.

In order to assess whether fire size varied spatially with fire frequency, I used a χ^2 goodness-of-fit test (α =0.05) to determine whether the proportion of fires in three size classes (small, medium and large) varied significantly among three groupings of PMFI estimates. Groupings were based upon PMFI estimates of: ≤25 years; >25 and ≤50 years; and >50 years. Observed values were the proportions of fires in each size class assigned to each of the fire frequency groupings. Expected values were the overall proportions of fires in each size class regardless of fire frequency group. I conducted separate analyses for each fire-shed. I also assessed whether the seasonality of large fires was significantly different than that of all fires using a χ^2 goodness-of-fit test (α =0.05) in which the observed values were the proportion of

large fires classified as early, middle, and late and the expected values were derived from the proportions of fires assigned in each of these three seasonality classes regardless of fire size.

Tree recruitment analysis

I created species-specific recruitment chronologies for all plots by grouping pith dates into 10-yr bins. Pith dates for fire-scarred trees from outside of plots were included with those of the closest plot. Ages for fire-scarred trees were typically older than many plot trees therefore including these non-plot trees in recruitment chronologies provided greater temporal depth in the recruitment record. Values from the latter part of each record to the present were truncated because only trees with a minimum DBH of 20 cm were sampled.

I graphically compared temporal variations in fire-size and tree-recruitment chronologies for each fire-shed to two indices of climate variability derived from independent, tree-ring reconstructions of climate variability. The Palmer Drought Severity Index (PDSI) is a high frequency (annual) measure of June through August drought. I averaged values for four grid points closely associated with the study sites (grid points 71, 72, 86, and 87; Cook et al. 2004). The Pacific Decadal Oscillation (PDO) is a measure of variability in north Pacific temperature patterns that vary at decadal scales and have been shown to influence climate and fire patterns in western North America (Westerling and Swetnam 2003, Kitzberger et al. 2007). I selected the McDonald and Case (2005) reconstruction because its length allowed for a longer period of comparison than did other indices considered.

Results

Plot elevation ranged from 2,365 to 3,231 m at BMC and from 2,195 to 2,689 m at LAW (Fig 3.1). In-plot slope estimates varied from 19 to 57 % at BMC and from 12 to 58 % at LAW. Across both sites, plots were located primarily on upper (40%) and middle (33%) slope positions with lower (17%) and ridge top (10%) positions represented to a lesser extent. Plot aspect at BMC was limited to a north to southeast range in bearing (346-127˚) but was largely representative of the study area. Aspect for 14 of 24 plots at LAW had a northwest to northeast bearing (315-45˚) while aspect for the remaining 10 plots varied somewhat evenly across the remaining spectrum with the exception that no plot was oriented in a predominantly south-facing (135-225˚) direction. Plots at BMC were located in pinyon-juniper and mountain mahogany woodlands at lower elevations, shrubland-steppe (with young conifer and mountain mahogany) and dry mixed-conifer forests at mid elevations, and limber pine and Englemann spruce-dominated subalpine forest with some Douglas-fir and quaking aspen at higher elevations. Plots at LAW were pinyon juniper woodland on west and south slopes across the range in elevation, ponderosa pine and dry mixed conifer forests on north and east aspects, and a bristlecone pine stand on one upper elevation plot. Charred remnants were not tallied on all plots but were found on 7 of 12 plots at BMC and 16 of 22 plots at LAW.

I sampled 674 trees in 24 plots (28.1 plot mean) at the BMC study site. Of these, 79% were live trees and the rest were snags (11%), logs (8%), and stumps (2%). I successfully cross-dated 507 (75%) trees and identified or estimated a pith date for 470 of these. Fire-scars were dated on 38 plot trees. I sampled 85 (40 live) additional fire-scarred trees near plot perimeters and in between plots, most of which were limber pine (45%) or ponderosa pine (30%). Of these, I cross-dated 72 and assigned a pith date to 70. I dated 378 fire scars for a mean of 3.4 per tree. I

treated an additional 46 abrupt ring changes or injuries of uncertain origins as evidence of fire. Most of the trees at BMC that were not cross-dated were either older mountain mahogany (live and dead) with dark heartwood in which ring structure was extremely difficult to decipher or high-elevation (>3,000 m) conifer remnants with complacent ring patterns (little ring-width variability). Of the 526 trees with post-1200 pith dates, most were white fir (35%), limber pine (19%), pinyon pine (18%), and Douglas fir (11%) with lesser numbers of ponderosa pine, Englemann spruce, mountain mahogany, and quaking aspen.

I sampled 730 trees in 24 plots (30.4 plot mean) at the LAW study site. Of these, 74% were live trees and the rest were snags (9%), logs (16%), and stumps (1%). I successfully cross-dated 610 (84%) trees and assigned pith dates to 576 of these. Fire scars were dated on 45 plot trees. I sampled 88 (34 live) additional fire-scarred trees outside of plot perimeters and between plots, cross-dated 84, and assigned pith dates to 80. Ponderosa pine was the most common species represented among sampled fire-scarred trees (74%). I assigned calendar years to 371 fire scars for a mean of 2.9 per tree. I treated an additional 71 abrupt ring-width changes or injuries of uncertain origin as evidence of fire at this site. Most trees not cross-dated were pinyon pine with tight ring structure and many missing rings and Utah juniper which has a tendency to produce abundant missing and false rings. Of the 616 trees with post-1200 pith dates most were white fir (49%), ponderosa pine (20%) , and pinyon pine (19%) with small numbers of juniper (Utah and Rocky Mountain combined), bristlecone pine, and Douglas fir.

Fire frequency

Fire-scarred trees (1-16) were associated with 14 of 24 plots (40-m radius) at BMC representing the full range in plot elevation (Figs. 3.2). PMFI estimates ranged from 11.2 to

106.0 years for nine plots in which fire was recorded on ≥ three trees and ≥ three years. I identified four non-plot clusters of three to five trees for which composite MFI varied from 7.8 to 109.5 years. MFI for six pairs and eight individual trees ranged from 11.9 to 125.5 years before, and from 9.5 to 100.4 years after correction, for a total of 27 PMFI estimates at BMC. Across the fire-shed, PMFI was shortest for mid-elevation (2,500-2,800 m) forest types and became gradually longer with increasing elevation (Fig. 3.3). On average, PMFI estimates were shorter for chronologies sampled from lower and middle slope positions (mean 22.9, range 7.8-43.6 years) than for those taken from upper and ridge positions (mean 84.2, range 11.2-109.5 years). Frequent surface fires most evident at mid-elevation stands ended abruptly in the middle to late 1800s (Fig. 3.2). Although I was not able to date fire-scars from five mountain mahogany trees I sampled from woodland and shrub-steppe communities, their presence provided direct evidence that periodic fire helped to shape these communities historically. I was also unable to assign years to multiple fire scars for eight high elevation limber pine remnants.

Fire-scarred trees (1-13) were sampled from 19 of 24 LAW plots of which 10 had ≥ three fire dates (Fig. 4). Uncorrected, composite MFI ranged from 25.9 to 125.3 years. After correction for three plots (tree number < three), plot-based, PMFI estimates ranged from 24.8 to 100.2 years. I identified 10 non-plot clusters of three to four trees in which composite MFI varied from 14.8 to 68.4 years. Mean fire interval for four pairs and 12 single trees ranged from 16.6 to 173.0 years before correction and from 13.3 to 138.4 years after correction, for a total of 36 estimates of PMFI at LAW. Variation in PMFI estimates and elevation were not related (Fig. 3.3). Although average PMFI estimates were longer for chronologies sampled from lower and middle slope positions (mean 46.8, range 13.3-138.4 years) than for those taken from upper and ridge positions (mean 36.6, range 14.6-97.3 years), the relatively small difference and wide range

observed in both groups suggests that differences were also unrelated to slope position. Old tree ages and the lack of fire evidence in the form of fire scars or char for two low elevation pinyon-juniper woodland plots (2C and 4D; Fig. 3.4) suggest that some stands of this landscape were largely unaffected by fire over very long (≥800 years) time periods. For most point chronologies, evidence of surface fires ends in the late 1700s to early 1800s (Fig. 3.4).

Contour plots (Fig. 3.5) in which lines reveal gradients in PMFI in comparison to topographic maps of fire-sheds (Fig. 3.1), allowed me to visually explore the integrated effects of topographic variables (elevation, aspect, slope) on fire-frequency spatial patterns. Although I expected that the relatively low density and uneven spatial distribution of MFI values would require caution in how this technique was used, I was pleased to see that the effects of topographic position were visually apparent for BMC as evidenced by the similarity between Figs. 3.1 and 3.5a, and to a lesser extent for LAW (Fig. 3.5b). Steep MFI gradients at BMC roughly correspond to steep north-facing slopes with longest MFI values associated with shallower soils on upper slopes. The 10-year contour line in the upper (west) end of the fire-shed (Fig. 3.5a) was an extrapolation effect created by the graphing software as there were no PMFI values from that part of the landscape below 20 years. The contour plot for LAW also shows regions of fire-frequency variation but does not account for stands (plots) with very long periods with no fire (e.g. plot 4D; 800+ years) because fire interval statistics could not be generated where fire scars were not sampled. Despite limitations, the contour plots suggest that point fire frequency changed abruptly over short distances and that those changes were related to variation in topography at moderate to fine scales.

Fire size

There were 99 years with fire at BMC and 124 at LAW. First to last fire dates span 640+ years at both sites (BMC 1267-1909; LAW 1294-1937; Figs. 3.2, 3.4). Most fires at both sites were relatively small. At BMC, 64% were classified as small (<10 ha) and 20% as medium-sized (≥10 and <100 ha). At LAW, 60% were classified as small and 16% as medium-sized (see examples Fig. 3.6). The earliest large (≥100 ha) and landscape (≥ 200 ha) fires at BMC were 1538 (109 ha) and 1632 (366 ha), respectively. The largest fire was in 1835 (379 ha) with other landscape fires in 1691, 1709, 1751, 1782, 1794, and 1824 (Fig. 3.7). Seven additional years recorded large fires. The fire in 1824 was recorded in the highest number of point chronologies (11 of 27) for this site. Mean, minimum, and maximum intervals for large fires were 22, 1 and 67 years. Mean, minimum, and maximum for landscape fires were 29, 11, and 59 years. The largest fire at LAW was in 1423 (Fig. 3.6; 447 ha). I classified 20 additional landscape fire years and eight large fire years (Fig. 7) the largest of which were in 1586 (384 ha), 1660 (307 ha), 1691 (324 ha), 1707 (324 ha), 1765 (408 ha), and 1796 (353 ha). The fire in 1765 was recorded by 16 of 38 point chronologies, the most for any fire at the LAW site. Mean, minimum, and maximum fire intervals for large fire years at LAW were 14, 1, 42 years. Mean, minimum, and maximum fire intervals for landscape fires were 20, 4, and 42 years. Last dates for large and landscape fires were respectively 1865 and 1835 at BMC and 1825 (both classes) at LAW.

Fires of all size classes burned across the range of elevation at both fire-sheds. Mean proportions by fire-size classification for BMC sample points were 41% small, 27% medium, and 32% large. Results of the χ^2 goodness-of-fit tests revealed that size-class proportions did not differ significantly from expected values for high (PMFI ≤25 years) and medium (PMFI >25 and ≤50 years) fire-frequency groupings but did for the low (PMFI >50 years) fire-frequency grouping (α=0.05). In the latter case, fire-size proportions were 54% small, 26% medium, and

20% large, indicating that small fires had greater and large fires had lesser importance for stands that experienced longer fire intervals relative to stands in which fire intervals were shorter. Mean proportions by fire-size classification for LAW sample points were 30% small, 17% medium, and 53% large. Differences were not significant for any fire-frequency grouping indicating that fires of each size-class were proportionately distributed throughout the fire-shed irrespective of variation in fire frequency.

Cumulative burn area for all fires at BMC was 4,507 ha, a total equal to more than seven times the approximate area of the fire-shed. Of this total 3, 19, and 78% were attributed to small, medium, and large fires. Landscape fires accounted for 67% of the large-fire burn area. Cumulative burn area for all fires at LAW was 7,892 ha, a total equal to 15 times the area of the fire-shed. Of this, 1, 10, and 89% were attributed to small, medium, and large fires. Landscape fires accounted for 85% of the large-fire burn area for this fire-shed. Thus, even though small fires accounted for more than 60% of the total number of fires, they were responsible for only a small fraction of the area burned for both sites. Conversely, approximately one in five fires was classified as large fires which were in turn responsible for more than 80% of the area burned. Thus while years with relatively small fires were common in both fire-sheds, the majority of the area that burned did so during the less frequent years with large fires.

I assigned unknown seasonality to 13 and 16% of BMC and LAW fire events, respectively. Ambiguous dormant season fires (inter-ring fire scars only) were the most common for both BMC (29%) and LAW (35%) sites. Of the remaining, I assigned fire seasonality as 28% early, 32% middle, 37% late, and 3% multi-season at BMC and 38% early, 21% middle, 36% late, and 5% multi-season at LAW. Results of the site-specific χ^2 goodness-of-fit tests indicated that fire seasonality (early, middle and late seasons only) of large fires differed significantly

($\alpha<0.05$) from that of all fires at each site. The effect was only marginally significant ($\alpha=0.10$), however, when the analysis was for both sites combined. The largest seasonal differences observed between all fires and large fires were: an increase of early-season fires (29 to 43%) and decrease in late-season fires (38 to 29%) at BMC and a decrease in middle-season fires (22 to 11%) for LAW. Across both sites early-season fires increase from 35 to 44%, middle-season fires decreased from 27 to 19%, and late-season fires remained unchanged at 38%.

Tree recruitment

I assigned a mean of 21.9 (range 4-36) post-1200 pith dates (plot and inter-plot trees combined) to plot recruitment chronologies at BMC. There was a small pulse of recruitment for high elevation limber pine and Englemann spruce (plots 13C, 11C) in the late 1400s and early 1500s (Figs. 3.7 and 3.8). There was no basis for making inferences regarding the role of fire during this period because the fire record lacks sufficient temporal and spatial depth; however, it appears that most of the plots associated with this sub-alpine association experienced sporadic, asynchronous tree recruitment for the 800 years of observation. In contrast, an easily recognized recruitment pulse occurred in the mid to late 1600s (Fig. 3.7) in which ponderosa and limber pines established early within an extended period of low fire activity that followed the landscape fire of 1632 (Figs. 3.6 and 3.7). The pulse is represented by recruitment in plots 13D, 11D, 14F, 11E, 14G, 12F, and 13H (Fig. 3.8) and was the only event of its kind observed for ponderosa pine in the BMC record. The frequency and intensity of yearly drought during this period, as measured by annual PDSI fluctuations, was about average with the long-term record and contrasts with the ~30-year wet period that came immediately before. A multi-species recruitment pulse occurred in the early 1800s during a 30-year period bracketed by landscape fires in 1794 and 1824 and during a time when annual PDSI values were never strongly negative

indicating a lack of severe drought (Fig. 3.7). At the landscape scale, evidence is lacking of a recruitment response to a large-fire hiatus that occurred between 1752 and 1782. However, a strong recruitment surge in plot 14G suggests that the fire that was recorded in that plot in 1751 and the fire-free interval that followed were important at the stand (plot) scale. Recruitment of white fir at BMC appears to have been asynchronous at the landscape scale with temporally distinct pulses occurring in different plots as illustrated for plots 13F, 14G, 14H, 12G, and 13I (Fig. 3.8). The strong post-1900 recruitment pulse of white fir and/or pinyon pine and the lack of evidence for conifer presence before the late 1800s (including remnants) for six plots (13G, 12G, 12H, 13I, 11H, 11I) confirms that the mid-elevation, essentially treeless, shrub-steppe vegetation type was extensive until the late 1800s at BMC; a timing concurrent with the loss of frequent fire in forested stands of similar elevation (see plot 13H, Fig. 3.2). Similar early-1900s pinyon pine expansion into montane shrub-steppe communities was common throughout the Great Basin (Miller et al., 2008). Pinyon recruitment in plots located in persistent woodlands (12I, 12J, 11J) was mostly continuous to sporadic suggesting a lesser role for fire in structuring this type. Evidence of tree recruitment after the early 1900s is missing due to the 20-cm DBH minimum size imposed for inclusion in the study.

I assigned a mean of 25.8 (range 5-39) post-1200 pith dates (plot and inter-plot trees combined) to plot recruitment chronologies at LAW. The tree recruitment record for LAW differs from that of BMC but with some parallels. Long-lived and remnant ponderosa, pinyon, and bristlecone pines and Douglas fir trees established throughout the 1200s (Fig. 3.7). Although numbers are relatively small they represent the survivors of what was likely a larger set of trees, most of which were not preserved as datable wood for the 700+ years that lapsed to the present. These trees provide a contrast to the almost complete void of recruitment dates for the 120 years

that followed starting at about the year 1300. This is contrary to what I expected based upon the climate proxies, both of which point to longer and deeper droughts during the 1200s than during the 1300s (Fig. 3.7). The fire record for this early period is inadequate to make inferences about its effects on recruitment. A pulse of recruitment (primarily ponderosa pine) began after the landscape fire in 1423 (see plots 6A, 4B, and 5C; Fig. 3.9) and persisted for perhaps 30-40 years. Recruitment overlapped a short drought-free period followed by long-term drought as indicated by the smoothed PDSI curve (Fig. 3.7). The record indicates a single relatively large fire in 1465 interrupted an otherwise extended (83 years) fire quiescent period. A second pulse of primarily ponderosa pine recruitment occurs in the 1500s and is captured to some degree in the records of plots 6A, 6D, 5B, 5D, and especially 3B. During this time an 80-year (1506-1586) hiatus in landscape-scale fires was synchronized with what appears to have been a long period of favorable climate as indicated by lack of drought years (PDSI) and above average PDO (Fig. 3.7). Ponderosa pine recruitment after that time has been sporadic with no pulses observed at either landscape or stand scales. Similar to BMC, recruitment of white fir at LAW was somewhat asynchronous at the landscape scale (with one major exception) with temporally distinct pulses occurring in separate plots in response to localized differences in timing of fire and fire-free periods (Fig 3.9). Probable mixed-species recruitment pulses started in the 1590s and persisted for several decades in plots 7B, 3A, 6D, and 4E and appear to be in response to a fire in 1586 that chronologies show burned in the vicinity of each of these plots. Intermediate-sized fires in 1607 (101 ha) and 1632 (181 ha) were exceptions to an otherwise long fire-quiescent period (1586-1660) during which time recruitment would have been somewhat favored by average climate conditions. The period between 1660 and 1730 included nine landscape fires and recruitment was largely flat at broad scales. A post-1700 recruitment pulse of white fir in plot 4B

appears to be a response to a fire in 1696 that burned in that plot (Fig. 3.8). No large fires were recorded and climate conditions were moderate between 1730 and 1763. Recruitment pulses during this period were of mixed species in plot 4A and of pinyon pine in plot 2A (Fig. 3.9). Although charred remnants were present at plot 2A, I found no fire-scar samples in the immediate vicinity of this plot rendering difficult inferences about disturbance and recruitment patterns for this plot. However, six point chronologies associated with nearby plots 3A and 3B recorded a fire in 1706 that if it had also burned in plot 2A, could have induced the observed recruitment surge in this plot. A widespread recruitment pulse in the late 1700s to early 1800s in plots 9X, 7D, 6A, 6C, 7E, 5A, 5C, 5D, and 3C (Fig. 3.9) provides a notable exception to the pattern of among-plot asynchrony and appears to be in response to disturbances associated with a large fire in 1765 (and possibly with another in 1763) and to have not been affected by subsequent landscape fires in 1781, 1792, 1796, 1800, 1818, or 1825 (Fig. 3.7). Smaller pulses later in the 1800s found in plots 9B and 7E were likely due to smaller-scale disturbance. Most old Utah juniper trees that co-dominated on old growth pinyon-juniper plots could not be cross-dated, however as many as 1,445 rings were counted per tree. Although exact ages could not be determined (due to the unknown effects of false and missing rings), these trees were clearly very old. The lack of recruitment evidence in the late 1800s and 1900s is largely due to the 20-cm DBH minimum size limit imposed for inclusion in the. Plots 7C, 6B, 4D, and 2C showed no evidence of episodic recruitment.

Discussion

How did fire frequency vary within each fire-shed in relation to topography?

At the landscape scale, fire was a frequent visitor to both fire-sheds for several centuries, however the spatial distribution of surface fire was highly uneven, resulting in more than 10-fold differences in PMFI estimates in both fire-sheds. Differences in PMFI were somewhat related to variation in elevation and slope position at BMC but not at LAW. I did not attempt an evaluation of the effects of aspect on fire frequency at BMC because fire-scarred trees were sampled almost exclusively from north and east-facing slopes. There were two reasons for this sampling bias. First, the general slope was to the east resulting in no west-facing slopes in the fire-shed. Secondly, south facing slopes were either not historically forested or were occupied by combinations of pinyon, juniper, and mountain mahogany, species not well suited for preserving datable fire scars. Longer fire-free intervals were located on steep slopes where soils would have been relatively shallow and thus less able to produce continuous fine fuels (Figs. 3.1 and 3.5a). Longer fire intervals have been associated with higher fire severity (Fulé et al., 2003; Margolis and Balmat, 2009). However, that was not the case in this fire-shed where evidence of higher fire severity in the form of episodic recruitment was most pronounced for mid-elevation plots where fire was also most frequent (Fig. 3.8) and essentially absent in subalpine stands where evidence of nonlethal surface fire in the form of fire-scarred trees was common. Some evidence of variable recruitment rates in two of the persistent pinyon-juniper plots (plots 11J, 12J) suggests a mixed-severity fire regime with intermediate to long fire intervals for this type.

The effect of aspect on fire frequency was not assessed for LAW for reasons similar to those given for BMC. Although the magnitude of variation in fire frequency at the LAW fire-shed was as great as that of BMC, there was no relationship to elevation. The lack of an elevation effect in fire frequency at LAW was likely due at least in part to the lower maximum elevation for this fire-shed which corresponds to a range in elevation at BMC where fire was most

frequent. PMFI values at LAW were shortest for locations near ridge-tops (3A-4, 6A-2, 6A-3, 7B-2; mean 16.3 years) and near drainage bottoms (3C-2, 4E-2, 5D-2, 5D-4; mean 19.7 years). Stands with longer intervals were spread across the elevational gradient suggesting that fire frequency was under fine-scale topographic control. For example, intermittent long and short PMFI values associated with samples taken from near the escarpment that defines the western boundary of the LAW fire-shed likely segregate based upon fine-scaled differences in soil depth not apparent in the topographic map (Figs. 3.1 and 3.5b). In addition, evidence of higher fire severity in the form of episodic recruitment was abundant but mostly asynchronous (Fig. 3.9) supporting an interpretation of a fire regime that produced mixed effects across stand-level rather than landscape-level spatial scales. I inferred very long fire-free intervals (up to 800+ years) for some old growth pinyon-juniper stands based upon old tree ages and the lack of fire evidence in plots. These stands were unaffected by the many fires that burned on all sides over the course of several centuries confirming the effectiveness of the fine-scaled topographic control of fire on this fire-shed.

With an average of 10.4 years, the shortest PMFI values at BMC were associated with plots 13H and 14G and two closely associated non-plot clusters located in the southern (Burnt Mill Canyon) fork of the fire-shed (Figs. 3.1 and 3.2). Average PMFI for plot 12F and five non-plot clusters near plots 12D, 12E, 12F, and 13F (elev. range 2,682-2932 m) was 26.7 years and represent the highest fire frequency within the northern (Mill Creek Canyon) branch of the fire-shed. Slopes and aspects are similar across much of each of these sub-drainages and there is considerable overlap in elevation. A significant portion of the landscape that separates mid-elevation forest stands in the two branches was occupied by shrub-steppe vegetation before 1900, thus fuels matrices were not continuous, implying some degree of fuel discontinuity between

north and south sections of the fire-shed. However, I was unable to deduce a topographic explanation for the more than two-fold difference in minimum PMFI values associated with these similar landscape elements. Instead I suggest that the higher fire frequency observed for the south drainage may have been due to differential ignition rates related to Native American use. Although clearly speculative, a case could be developed to support the hypothesis based upon an interest in bighorn sheep (*Ovis canadensis*), though alternative motives are plausible (Williams, 2004). Periodic burning would maintain vegetation in an open, park-like or treeless state that this mammal requires (Risenhoover and Bailey, 1985; Singer et al., 2000). It would also result in a freshening of grass growth providing an additional attractant for these large herbivores. Fire might also have been used to drive game during the hunt (Williams 2004). Bighorn sheep also require escape cover in close proximity (<300 m), usually in the form of rocky ledges (Singer et al., 2000; McKinney et al., 2003) such as those found immediately south of Burnt Mill Canyon but only at greater distances from the Mill Creek branch of the fire-shed. Bighorn sheep are known to have inhabited the Snake Range before Euro-American settlement. Whether due to natural or anthropogenic ignitions, fire was likely important in the maintenance of sufficient suitable habitat in the past and the absence of substantial burning over the past century has almost certainly contributed to a shortage of suitable habitat on this and similar mountains today.

When did historic fire patterns change in these fire-sheds and what were the possible causes of change?

The abrupt loss of frequent surface fire at BMC was more or less synchronous with Euro-American settlement suggesting that the probable change was due to livestock-related removal of the fine fuels necessary for fire spread, disruption of Native American burning practices, or some combination of these factors (Pyne, 1982; Mandany and West, 1983; Covington and Moore,

1994; Kay, 1995). A similar change in fire regime occurred approximately 50 years earlier at LAW than at BMC and several decades before the introduction of large numbers of domestic sheep to the area. This pre-settlement change was similar to that observed for other (but not all) sites in a regional fire history study suggesting an alternate cause or causes given the timing of the change was not synchronous across sites (Chapter 2). Numerous studies report multi-decade reductions in either local or regional fire occurrence in the early 1800s (for examples see Swetnam and Baisan, 1996; Heyerdahl et al.; 2001; Brown and Wu, 2005; Skinner et al, 2008; Iniguez et al. 2009). This hiatus in fire activity corresponds to an extended cool period with reduced amplitude in inter-annual wet-dry oscillations, conditions favorable for reduced fire activity (Kitzberger et al., 2001). Thus climate might be invoked to explain the multi-decadal gap between fire regime change and livestock-induced changes in fuels for the LAW fire-shed and similar sites. However, a simple climate-based explanation fails to account for the inter-site variation in the timing of fire regime change observed in the regional study, nor does it clarify why the phenomenon was not observed at BMC or at other eastern Great Basin sites (Chapter 2). Here I offer an alternate hypothesis for the asynchronous truncation of fire regimes observed among BMC and LAW and the other regional sites. I propose that the observed asynchrony could be evidence of differential disruption of human ignition patterns in response to a major perturbation in the regional human population. Although a late-1700s arrival for Euro-American diseases to the region might be considered a bit overdue given much earlier spread in other parts of the continent (Thornton, 1987; Reff, 1991; Butzer, 1992), a delayed, disease-induced depopulation event is plausible given the low population densities of the hunter-gatherer inhabitants that occupied the Great Basin prior to Euro-American settlement (Simms, 2008). A spatially uneven reduction in anthropogenic burning would be expected as survivors repeatedly

adjusted occupation patterns on the heterogeneous landscape. Of course it is possible that changes in both climate and human ignition patterns interacted to produce the observed patterns.

How did fire size vary within fire-sheds and through time?

The apparent differences in the number of large and landscape fires between BMC and LAW are at least in part an artifact of differences in the distribution of fire-scarred trees within the two fire-sheds. The five eastern plots and associated inter-plot landscape at BMC collectively yielded few datable fire scars including just two that corresponded to years that matched fire-years identified for points further west (upslope) in the fire-shed. This and the somewhat narrow shape of this fire-shed effectively reduced the area that could possibly be included in fire-size estimates. In contrast, fire-scarred trees at LAW are spread throughout the fire-shed, including much of the perimeter, and its shape is less narrow than that of BMC. Consequently proportionally fewer estimates of large and landscape fires should have been expected for BMC.

Inferences regarding the spatial distribution of fires by size class have important implications regarding spatial heterogeneity of fire frequency. Iniguez et al. (2008) observed in the Santa Catalina Mountain of southeastern Arizona that study area differences in plot-level fire frequency were due primarily to the relative size and not frequency of widespread fires and implied a somewhat equal contribution from small fires. In that study local fire frequency was subject to the homogeneity and continuity of landscape-level fuels matrices as controlled by the surrounding topography. Though perhaps expressed at a finer spatial scale, I observed a similar pattern at BMC where the importance of small fires increased and large fires decreased in stands where fire intervals were longest. I inferred that this shift in relative importance for small and large fires applies primarily to upper slopes at higher elevations because that was where longer

fire intervals were concentrated. By extension, I also infer that the relatively higher fire frequency experienced on some parts of the landscape was not the product of a disproportionate number of small fires but instead resulted from proportional increases in fire numbers within each fire-size grouping. No such pattern emerged at LAW in spite of a wide range in point MFI estimates and an abundance of both large and small fires. At this site, differences in point MFI reflected parallel and somewhat equal differences in fire occurrence within each fire size class. Thus, it appears that fire frequency at LAW varied independently of fire size, suggesting strong fine-scale control of fire occurrence and spread. This was likely due to the combined effects of the broken topography and the vegetative heterogeneity that characterized this fire-shed.

Fire seasonality analysis revealed that middle-season fires were not only less numerous than early and late season fires (most apparent after dormant season fires are divided between early and late season classes) but that they were proportionately less important when only large fires are considered. This result is counterintuitive for at least two reasons. First, without a strong monsoon which is atypical in the eastern Great Basin, fuel flammability should correlate with ambient temperature maximizing conditions for fire spread at or just after the hottest period of mid-summer. Second, lightning strike densities are less frequent in the early and late parts of summer, thus conditions for natural fire ignitions and fire spread should reach an optimum sometime in mid-season (Griffin, 2002; Schmidt et al., 2002; Westerling et al., 2003). A bimodal pattern in fire seasonality was therefore unexpected (see Chapter 2) and might be interpreted as indirect evidence for anthropogenic ignitions (Allen, 2002; Williams, 2004). An explanation for the direction of seasonal shifts associated with large fires from mid and late-season toward early-season fire remains difficult to explain.

How did species-specific tree recruitment vary through time and space in relation to fire, fire-quiescent periods, and climate?

I observed evidence of episodic patterns of recruitment for conifer species both tolerant (i.e. white fir, Douglas-fir, Englemann spruce) and intolerant (i.e. limber pine, bristlecone pine, ponderosa pine, pinyon pine) to shade at both fire-sheds. Recruitment pulses were often synchronized with fire-quiescent periods manifest at stand to landscape scales. The composition of recruitment pulses, as recovered in the sampled record, varied through time and has likely been modified through differential rates of decay among species. Thus the record for fast-decaying species (i.e. white fir, Englemann spruce) is primarily limited to later centuries. Multiple, temporally-distinct pulses of white fir are evident at the plot scale from the 1700s to early 1900s at BMC and from the 1600s to 1800s at LAW. The older ages at LAW may be due to slower decay and growth rates at this drier site. The asynchronous timing suggests that recruitment occurred during fire-quiescent periods that varied at fine spatial scales and that climate was not a primary factor. The result was a landscape mosaic of different-aged and multi-aged patches of trees and treeless (recently burned) elements. An exception to this pattern was the widespread recruitment synchronization at LAW in the late 1700s and early 1800s, decades before the demise of large fires and about a century before the introduction of large numbers of domestic livestock by Euro-American settlers. This white fir-dominated surge of tree establishment followed a large fire in 1765 (408 ha) suggesting that this fire caused sufficient disturbance to promote new tree establishment across a wide area. A series of six landscape-scale fires over the next 60 years produced no observable negative impact on the young recruits of this fire sensitive species. Thus, knowledge of widespread fire or the lack thereof, is not sufficient to predict tree recruitment events without better understanding of how fire severity varied across

the landscape. Climate likely played a secondary role in regulating the timing of tree establishment as evidenced by the early 1900 recruitment pulse at BMC, a period of above average precipitation and no significant drought.

Identifiable recruitment pulses for ponderosa, limber, and bristlecone pines were few in the 800-year record, all of which occurred prior to 1700 during fire quiescent periods. The most pronounced recruitment episodes for ponderosa pine were in the mid-1600s at BMC and early to mid-1500s at LAW. Both episodes appear to have been influenced by large fires that probably burned at high intensity at the locations where recruitment was most pronounced such as in plot 13H at BMC and plot 3B at LAW. Although numerous fires were recorded at both of these plots after initial stand establishment, all appear to have been of low severity (in the plot), suggesting that crown fires were rare events once plots were dominated by this species. Similarly, there were few recruitment pulses in pinyon pine dominated stands though their timing was more recent reflecting a more rapid decay rate for this species on most sites.

A rapid decline in the cumulative area of recent burns began in early-1800s at both fire-sheds and ended in an almost complete loss of fire by 1825 at LAW and 1865 at BMC. Subsequently, stand densities and associated fuel loads have increased and forest composition and structure has become more homogenized at the landscape scale. The impact is greatest for the mid-elevation forests where fire-free intervals were shortest. In addition, historically non-forested shrub-steppe landscapes at BMC that previously provided natural fire breaks are now largely stocked with young conifer and mountain mahogany trees. The higher-elevation sub-alpine forests where historic fire intervals were longest have been the least affected by the altered disturbance regime. Loss in heterogeneity at both fire-sheds will result in a shift from fine-scale to landscape-level controls of fire increasing risks for catastrophic crown fire across landscapes

including old-growth stands that were previously low risk for severe fire. A recent example of such an event was the Phillips Ranch fire that burned ~670 ha on the west slope of the South Snake Range in 2000. Ignited by lightning in pinyon-juniper woodland, this fire became an active crown fire and raced upslope stopping near tree-line (3,350 m). Intensity was high resulting in near complete tree mortality within the perimeter including a major portion of what was the largest bristlecone pine-Englemann spruce stand on the mountain.

What can be inferred about fire severity from tree recruitment patterns?

The abundance of fire-scarred trees and recruitment pulses at both fire-sheds strongly favors an interpretation of a mixed-severity fire regime except for old growth pinyon-juniper stands at LAW in which fire had little impact for 800+ years. Fire severity varied spatially within individual fires and among fires that burned the same landscape at different times. Locally, severity was low enough through repeated fires to allow white fir, a fire-sensitive species, to survive. Conversely, at infrequent intervals, fires were severe enough to prompt regeneration of entirely new stands of fire-resilient ponderosa and limber pines. The possibility of uniformly severe fires exists but these would have been spatially-limited given the wide distribution of old-aged trees and fire scars. It is also possible, perhaps likely that some fires, including some large fires burned in an entirely low severity, non-lethal manner. What is clear is that fire severity varied sufficiently through time and space to create and maintain mosaics of different-aged and multi-aged forest stands and in the case of BMC non-forested shrub-grass steppes at middle elevations and that the pattern differed at higher and some lower elevations by being more stable resulting in old-age stands.

Conclusions and management implications

In this study I document multi-century variation in fire regime and tree recruitment at two eastern Great Basin fire-sheds. Topographic control of fire frequency was manifest at somewhat broader spatial scales at BMC where elevation and slope position had influence, than at LAW where they did not. This difference is likely due to the drier conditions and more broken topography associated with the LAW fire-shed. I observed no evidence of differences in fire severity associated with high-elevation stands and longer fire intervals. Fire frequency in persistent pinyon-juniper woodland was more difficult to ascertain but clearly ranged from moderate to very long intervals associated with old-growth stands on some fire protected sites at LAW. Differences in mid-elevation fire frequency between north and south branches of the BMC fire-shed are difficult to explain without invoking the possibility of unequal ignitions of human origin. A majority of fires were small but the overall impact of the less-frequent large fires was much greater based upon their larger cumulative burn area. The spatial distribution of fires by size was largely independent of fire frequency except on BMC stands with long fire intervals where small fires became more important relative to large fires. Tree recruitment was mostly continuous in high (BMC) and low (LAW) elevation old-growth stands (plots) but was highly episodic through a broad range of middle elevations where fires were most frequent. Recruitment pulses were usually synchronized at the stand scale by multi-decade, fire-quiescent periods and often followed large fires. Climate likely played a secondary role in moderating disturbance-induced recruitment. The spatial scale and somewhat asynchronous nature of those pulses created vegetation mosaics that were compositionally and structurally dynamic. Few trees were able to establish or persist in the mid-elevation sections of shrub-steppe at BMC prior to 1900. Changes in fire regime that began during the early (LAW) to mid-1800s (BMC) in concert

with selective logging (BMC) and other management practices resulted in a shift in mid-elevation forest composition to one dominated by the shade-tolerant and fire-sensitive white fire. Shrub-steppe communities are in various stages of invasion by white fir, pinyon pine, and mountain mahogany at BMC. Consequently, vegetation is currently more homogeneous and fuels more continuous than any time for at least the past several centuries.

The challenge for managers is to restore fire resilient vegetation to these and similar fire-sheds on comparable landscapes using an appropriate combination of natural processes and active management. A passive management strategy is not recommended given the degree of departure in both fire regime and vegetation composition and structure. Priority should be given to restoring structural heterogeneity at mid-elevations where the fire regime and vegetation (fuels) are furthest removed from historical conditions. Objectives and strategies should be developed for both mixed-conifer forests and shrub-steppe types where they apply. I recommend incorporation of appropriate combinations of spatially-limited fire and fire-surrogate treatments that over time recreate a mosaic of vegetation conditions. Treatment severity should be mixed with care taken to preserve individuals of fire-resilient species (i.e. ponderosa pine) that have declined over the recent past. Early and late-season fires would mimic the historic seasonality pattern and have the added advantage of ease of control. A reduction in the risk for extreme fire events resulting from high fuel loads and landscape homogenization over the past 150 years will require sustained active management by those charged with stewardship of these lands.

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Literature Cited

Allen, C.D., 2002. Lots of lightning and plenty of people: an ecological history of fire in the upland Southwest. In Vale, T.R. (Ed.), Fire, Native Peoples, and the Natural Landscape. Island Press, Washington, DC.

Arno, S.F., Sneck, K.M., 1977. A method for determining fire history in coniferous forests of the mountain west. USDA Forest Service, Intermountain Research Station, General Technical Report INT-GTR-42, Ogden, UT.

Bauer, J.M., Weisberg, P.J., 2009. Fire history of a central Nevada pinyon-juniper woodland. Can. J. Forest Res. 39, 1589-1599.

Beaty, R.M., Taylor, A.H., 2008. Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. Forest Ecol. Manage. 255, 707-719.

Bekker, M.F. Taylor, A.H., 2010. Fire disturbance, forest structure and stand dynamics in montane forests of the southern Cascades, Thousand Lakes Wilderness, California, USA. Ecoscience 17, in press.

Brown, P.M., Shepperd, W.D., 2001. Fire history and fire climatology along a 5˚ gradient in latitude in Colorado and Wyoming, USA. Paleobotanist 50, 133-140.

Brown, P.M., Wu, R., 2005. Climate and disturbance forcing of episodic tree recruitment in a south-western ponderosa pine landscape. Ecology 86, 3030-3038.

Brown, P.M., Kaye, M.W., Huckaby, L.S., Baisan, C.H., 2001. Fire history along environmental gradients in the Sacramento Mountains, New Mexico: Influences of local patterns and regional processes. Ecoscience 8, 115-126.

Brown, P.M., Heyerdahl, E.K., Kitchen, S.G., Weber, M.H., 2008. Climate effects on historic fires (1630-1900) in Utah. Int. J. Wildland Fire 17, 28-39.

Butzer, K. W., 1992. The Americans before and after 1492: current geographical research. Ann. Assoc Am. Geogr. 82, 345-368.

Cook, E., Meko, D., Stahle, D., Cleaveland, M., 2004. Reconstruction of past drought across the conterminous United States from a network of climatically sensitive tree-ring data. Available at URL: http://www.ngdc.noaa.gov/paleo/usclient2.html/ (last accessed 28 Dec 2009).

Covington, W.W., Moore, M.M., 1994. Post-settlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. In: Sampson, R. N. and D. L. Adams (eds.) Assessing the forest ecosystem health in the Inland West. J. Sustain. For. 2, 153-181.

Dieterich, J.H., 1980. The composite fire interval: A tool for more accurate interpretations of fire history. In: Stokes, M.A., Dieterich, J.H. (Eds.). Proceedings of the Fire History Workshop, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-81. pp. 8-14.

Everett, R.G., 2008. Dendrochronology-based fire history of mixed-conifer forests in the San Jacinto Mountains, California. Forest Ecol. Manage. 256, 1805-1814.

Fulé, P.Z., Covington, W.W., Moore, M.M., 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecol. Appl. 7, 895-908.

Fulé, P.Z., Crouse, J.E., Heinlein, T.A., Moore, M.M, Covington, W.W., Verkamp, G., 2003. Mixed-severity fire regime in a high-elevation forest of Grand Canyon, Arizona, USA. Landscape Ecol. 18, 465-486.

Fulé, P.Z., Korb, J.E., Wu, R., 2009. Changes in forest structure of a mixed conifer forest, southwestern Colorado, USA. Forest Ecol. Manage. 258, 1200-1210.

Grayson, D.K., 1993. The Desert's Past: A Natural History of the Great Basin. Smithsonian Institution Press, Washington, DC.

Griffin, D., 2002. Prehistoric human impacts on fire regimes and vegetation in the northern intermountain West. Fire, native peoples, and the natural landscape. Island Press, Washington, DC.

Grissino-Mayer, H.D., 2001. FHX2 – Software for analyzing temporal and spatial patterns in fire regimes from tree rings. Tree-Ring Res. 57, 113-122.

Grissino-Mayer, H.D., Swetnam, T.W., 2000. Century-scale climate forcing of fire regimes in the American Southwest. The Holocene 10, 207-214.

Grissino-Mayer, H. D., Romme, W.H., Floyd, M.L., Hanna, D.D., 2004. Climate and human influences on fire regimes of the southern San Juan Mountains, Colorado, USA. Ecology 85, 1708-1724.

Harper, K.T., Freeman, D.L., Ostler, W.K., Klikoff, L.G., 1978. The flora of Great Basin mountain ranges: Diversity, sources, and dispersal ecology. In: Harper, K.T., Reveal, J. L.

(Eds.) Intermountain biogeography: A symposium. Great Basin Naturalist Memoirs 2. pp. 81-104.

Heyerdahl, E.K., Berry, D., Agee, J.K., 1995. Fire history database of the western United States. U.S. Environmental Protection Agency Report EPA/600/R-96/081.

Heyerdahl, E.K., Brubaker, L.B., Agee, J.K., 2001. Spatial controls of historic fire regimes: a multiscale example from the interior west, USA. Ecology 82, 660-678.

Heyerdahl, E.K., Morgan, P., Riser, J.P. II, 2008a. Multiseason climate synchronized historical fires in dry forests (1650-1900), Northern Rockies, USA. Ecology 89, 705-716.

Heyerdahl, E.K., McKenzie, D., Daniels, L.D., Hessl, A.E., Littell, J.S., Mantua, N.J., 2008b. Climate drivers of regionally synchronous fires in the inland North-west (1651-1900). Int. J. Wildland Fire 17, 40-49.

Holmgren, N., 1972. Plant geography in the Intermountain Region. In: Cronquist, A., Holmgren, A.H., Holmgren, N.H., Reveal, J.L. (Eds.) Intermountain flora. Vol. 1. Hafner Publishing Company, New York. pp. 77-161.

Iniguez, J.M., Swetnam, T.W., Yool, S.R., 2008. Topography affected landscape fire history patterns in southern Arizona, USA. Forest Ecol. Manage. 256, 295-303.

Iniguez, J. M., Swetnam, T.W., Baisan, C.H., 2009. Spatially and temporally variable fire regime on Rincon Peak, Arizona, USA. Fire Ecol. 5, 3-21.

Jamieson, L.P., 2008. Fire history of a pinyon-juniper/ponderosa pine ecosystem in the Intermountain West. M.S. thesis, University of Nevada, Reno, NV.

Kaufmann, M.R., Huckaby, L., Gleason, P., 2000. Ponderosa pine in the Colorado Front Range: long historical fire and tree recruitment intervals and a case for landscape heterogeneity. In: Crossing the millennium: integrating spatial technologies and ecological principles for a new age in fire management: proceedings, The Joint Fire Science Conference and Workshop. Vol. 1. pp. 153-160.

Kay, C.E., 1995. Aboriginal overkill and native burning: implications for modern ecosystem management. West. J. Appl. For. 10, 121-126.

Keane R.E., Parsons R.E., Rollins M.G., 2002. Predicting fire regimes at multiple scales. In Emulating natural forest landscape disturbances: concepts and applications: popular summaries. Ontario Forest Research Institute, Ministry of Natural Resources. Forest Research Information Paper 149, 23-25.

Kilpatrick, M., 2009. A tree ring based fire history of the Clover Mountains, Lincoln County, Nevada. M.S. thesis, University of Nevada, Reno, NV.

Kitchen, S.G., McArthur, E.D., 2003. Ponderosa pine fire history in a southeastern Great Basin stand. In: Galley, K.E.M., Klinger,R.C., Sugihara, N.G. (Eds.) Proceedings of Fire Conference 2000. The First National Congress on Fire Ecology, Prevention, and Management. Misc. Publ. No. 13, Tall Timbers Research Station, Tallahassee, FL. pp. 152-156.

Kitzberger, T., Swetnam, T.W., Veblen, T.T., 2001. Inter-hemispheric synchrony of forest fires and the El Nino-Southern Oscillation. Global Ecol. & Biogeogr. 10, 315-326.

Kitzberger, T., Brown, P.M., Heyerdahl, E.K., Swetnam, T.W., Veblen, T.T., 2007. Contingent Pacific-Atlantic Ocean influenceon multi-century wildfire synchrony over western North America. Pro. Natl. Acad. Sci. 104, 543-548.

Lessard, V.C., Drummer, T.D., Reed, D.D., 2002. Precision of density estimates from fixed-radius plots compared to N-tree distance sampling. Forest Sci. 48, 1-5.

Levin, S.A., 1992. The problem of pattern and scale in ecology. Ecology 73, 1943-1967.

Mandany, M.H., West, N.E., 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park, Utah. Ecology 64, 661-667.

Margolis, E.Q., Balmat, J., 2009. Fire history and fire climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA. Forest Ecol. Manage. 258, 2416-2430.

McDonald, G.M. Case, R.A., 2005. Variations in the Pacific Decadal Oscillation over the past millennium. Geophys. Res. Lett. 32, L08703.

McKinney, T., Boe, S.R., deVos J.C. Jr., 2003. GIS-based evaluation of escape terrain and desert bighorn sheep populations in Arizona. Wildlife Soc. Bul. 31, 1229-1236.

Miller, R.F., Tausch, R.J., McArthur, E.D., Johnson, D.D., Sanderson, S.C., 2008. Age structure and expansion of piñon-juniper woodlands: a regional perspective in the Intermountain West. Res. Pap. RMRS-RP-69. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 15 p.

Morgan, P., Hardy, C.C., Swetnam, T.W., Rollins, M.G., Long, D.G., 2001. Mapping fire regimes across time and space: Understanding coarse and fine-scale fire patterns. Int. J. Wildland Fire 10, 329-342.

Murdock, J.R, Welsh, S.L., 1971. Land use in Wah Wah and Pine Valleys, western Utah. Science Bulletin, Biological Series, Brigham Young University, vol. XII(4).

National Park Service (NPS), 2009. Great Basin National Park. Baker, Nevada 98311-9702. Available at URL: http://www.nps.gov/grba/grbaindex.htm (last accessed 28 Dec 2009).

Peterson, K.L., 1994. Modern and Pleistocene climatic patterns in the West. In: Harper, K.T., St. Clair, L.L., Thorne, K.H., Hess W.M. (Eds.) Natural History of the Colorado Plateau and Great Basin. University Press of Colorado, Niwot, CO. pp. 27-53.

Pyne, S.J., 1982. Fire in America – a cultural history of wildland and rural fire. Princeton University Press.

Reff, D.T. 1991., Disease, Depopulation, and Culture Change in Northwestern New Spain A.D. 1518-1764. University of Utah Press, Salt Lake City, Utah, USA.

Ricklefs, R.E., 1987. Community diversity: relative roles of local and regional processes. Science 235, 167-171.

Risenhoover, K.L., Bailey, J.A., 1985. Visibility: an important factor for an indigenous, low-elevation bighorn herd in Colorado. Proc. Biennial Symposium of the Northern Wild Sheep and Goat Council 2, 18-28.

Sibold, J.S., Veblen, T.T., González, M.E., 2006. Spatial and temporal variation in historic fire regimes in subalpine forests across the Colorado Front Range in Rocky Mountain National Park, Colorado, USA. J. Biogeogr. 32, 631-647.

Simms, S.R., 2008. Ancient Peoples of the Great Basin and Colorado Plateau. Left Coast Press, Walnut Creek, California.

Singer, F.J., Bleich, V.C., Gudorf, M.A., 2000. Restoration of bighorn sheep metapopulations in and near western national parks. Restor. Ecol. 8, 14-24.

Skinner, C.N., Burk, J.H., Barbour, M.G., Franco-Vizcaino, E., Stevens, S.L.,. 2008. Influences of climate on fire regimes in montane forests of north-western Mexico. J. Biogeogr. 35, 1436-1451.

Stokes, M.A., Smiley, T.L., 1968. An Introduction to Tree-Ring Dating. University of Chicago Press, Chicago, IL.

Swetnam, T.W., Baisan, C.H., 1996. Historical fire regime patterns in the southwestern United States since AD 1700. In: Allen, C.D. (Ed.) Fire Effects in southwestern forests-proceedings of the second La Mesa fire symposium. USDA Forest Service, Rocky Mountain forest and Range Experiment Station. General Technical Report RM-GTR-286. pp. 11-32.

Swetnam, T.W., Betancourt, J.L., 1990. Fire-Southern Oscillation relations in the southwestern United States. Science 249, 1017-1020.

Swetnam, T.W., Betancourt, J.L., 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. J. Climate. 11, 3128-3147.

Tausch, R.J., West, N.E., Nabi. A.A., 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. J. Range Manage. 34, 259-264.

Taylor, A.H., Skinner, C.N., 1998. Fire history and landscape dynamics in a late successional reserve, Klamath Mountains, California, USA. Forest Ecol. Manage. 111, 285-301.

Taylor, A.H., Skinner, C.N., 2003. Spatial patterns and controls on historic fire regimes and forest structure in the Klamath Mountains. Ecol. Appl. 13, 704-719.

Thornton, R.B., 1987. American Indian Holocaust and Survival: A Population History since 1492. University of Oklahoma Press, Norman, Oklahoma, USA.

Tuhy, J.S., 1985. A proposal: Research Natural Area designation for Wah Wah Mountain. USDI Bureau of Land Management, Richfield District Office. 29 p.

Van Horne, M.L., Fulé, P.Z., 2006. Comparing methods of reconstructing fire history using fire scars in a southwestern United States ponderosa pine forest. Can. J. Forest Res. 36, 855-867.

Wells, P.V., 1983. Paleobiogeography of montane islands in the Great Basin since the last glaciopluvial. Ecol. Monogr. 53, 341-382.

Western Regional Climate Center (WRCC), 2009. Historical climate data, Desert Research Institute, 2215 Raggio Parkway, Reno, Nevada 89512. Available from URL: http://www.wrcc.dri.edu/ (last accessed 28 Dec 2009).

Westerling, A.L., Swetnam, T.W., 2003. Interannual to decadal drought and wildfire in the western United States. EOS Transactions, Am. Geophys. Un. 84, 545-560.

Wiens, J.A., 1989. Spatial scaling in ecology. Funct. Ecol. 3, 385-397.

Williams, G.W., 2004. American Indian fire use in the arid west. Fire Manage. Today 64, 10-14.

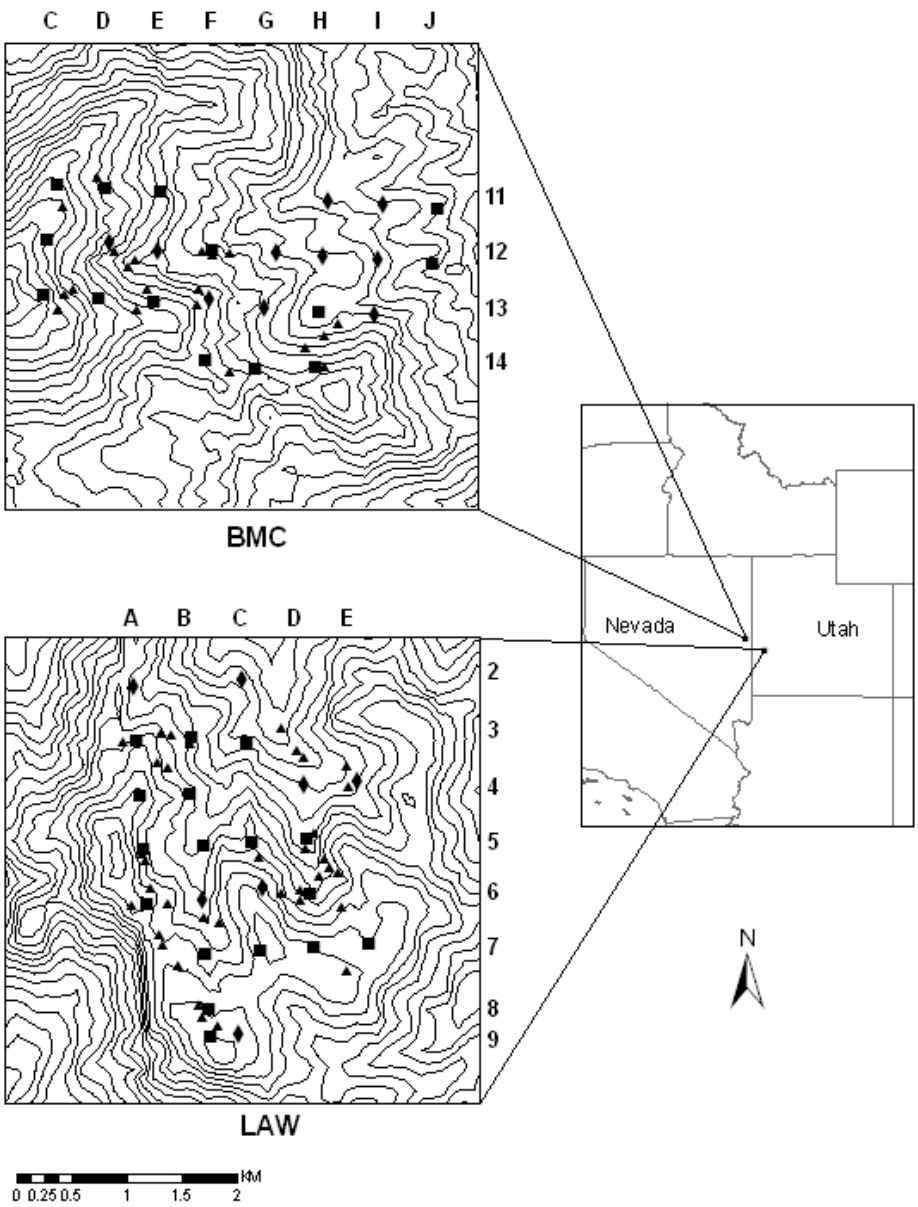
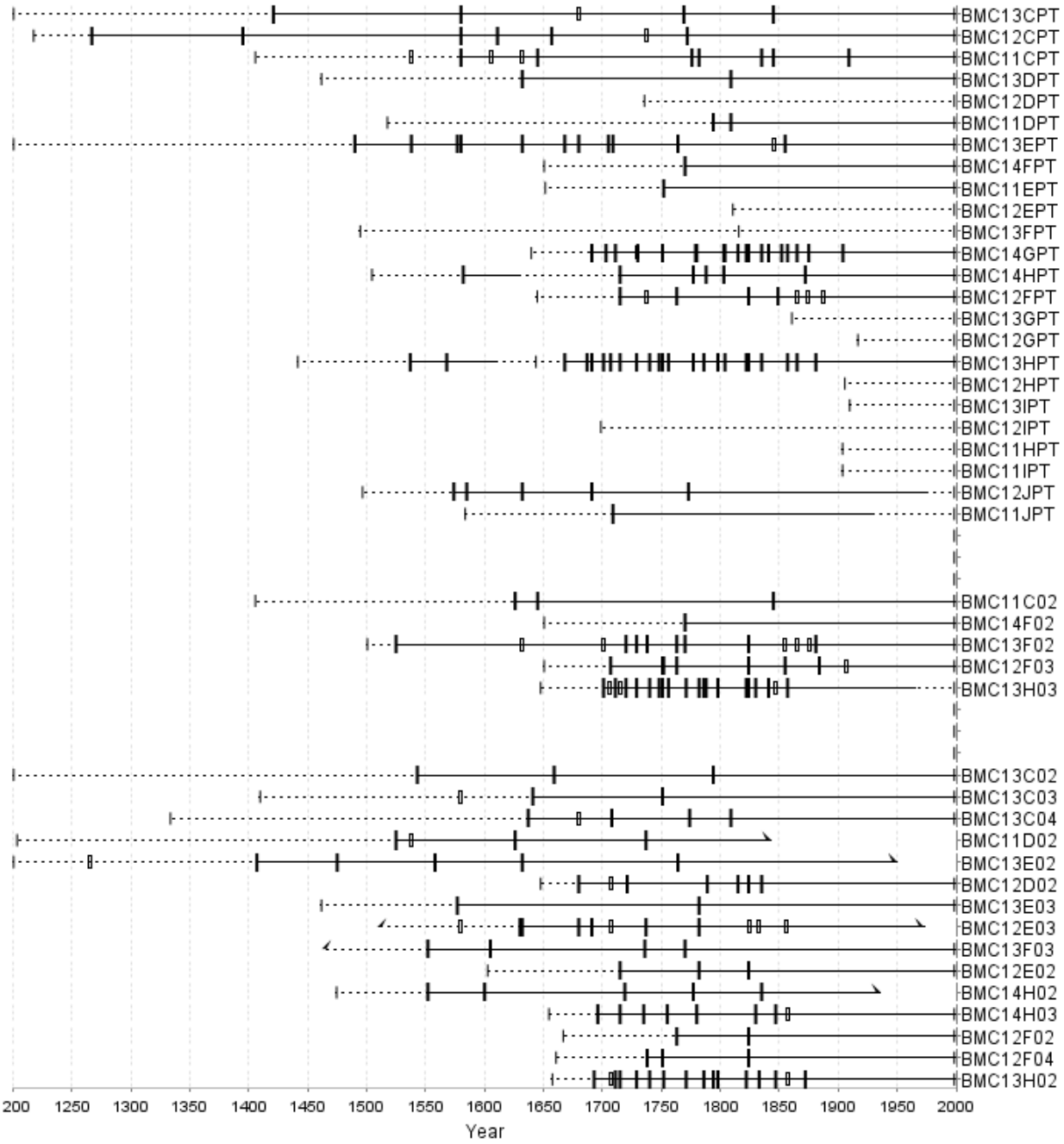


Figure 3.1. Grids of plots (500 m spacing) for fire-sheds located on the South Snake (BMC) and Wah Wah (LAW) Mountain Ranges. Plots are distinguished as squares (fire scars present) or diamonds (fire scars absent). Triangles designate locations for non-plot, fire-scarred sample trees or tree clusters (max area = 0.5 ha). Contour lines indicate elevation changes at 40-m intervals. Approximate area for each fire-shed is BMC 600 ha and LAW 525 ha.

Figure 3.2. Fire history chronologies for the Burnt Mill Canyon (BMC) fire-shed located on the South Snake Range, Nevada. Bottom axis designates calendar year (1200-2000). Horizontal lines represent plot composite chronologies (top group), cluster composite chronologies (≥3 trees; middle group), and paired and single-tree chronologies (bottom group). Within groups, chronologies are arranged by elevation with highest elevation plots or trees at the top. Alphanumeric codes on the right indicate plot or cluster identity. Solid lines indicate chronologies are in recording status. Solid vertical lines are years with fire scars and open vertical lines designate injuries or abrupt changes in ring-widths.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BMC12FPT
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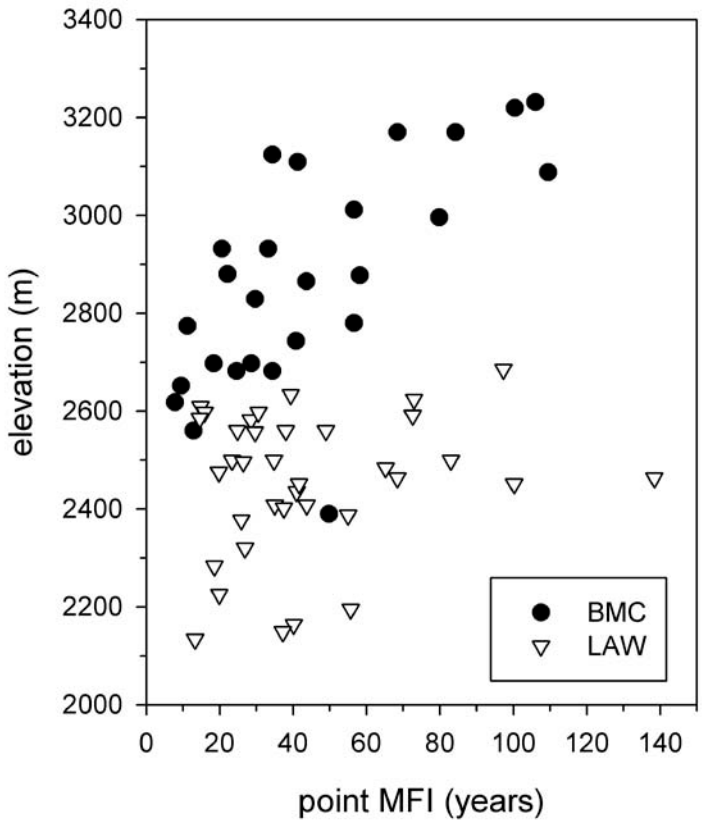
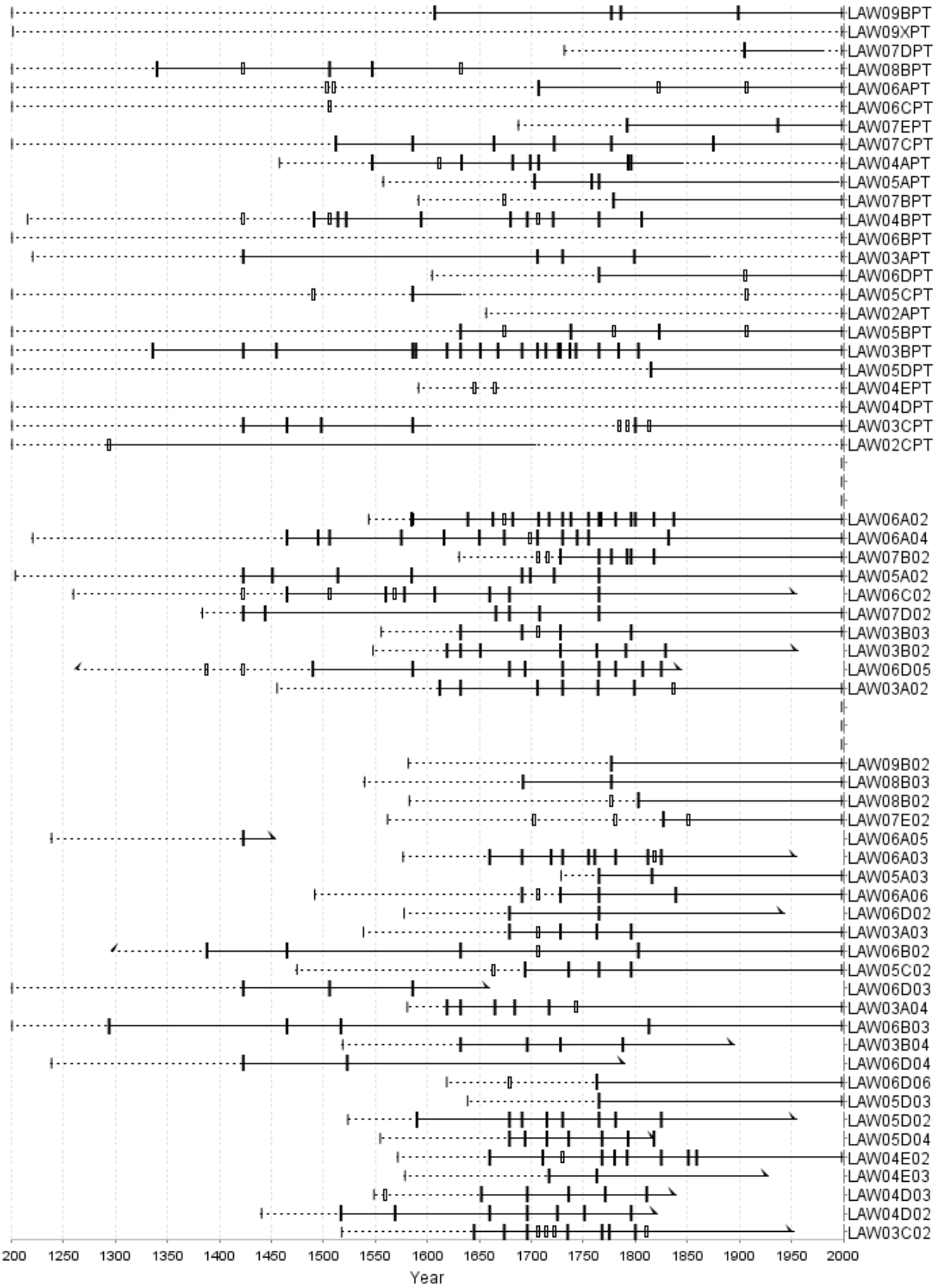


Figure 3.3. Relationship between elevation and PMFI estimates for two fire-sheds in the South Snake Range, Nevada (BMC) and the Wah Wah Mountain Range, Utah (LAW) fire-sheds.

Figure 3.4. Fire history chronologies for the Lawson Cove (LAW) fire-shed located on the Wah Wah Mountain Range, Utah. Bottom axis designates calendar year (1200-2000). Horizontal lines represent plot composite chronologies (top group), cluster composite chronologies (≥3 trees; middle group), and paired and single-tree chronologies (bottom group). Within groups, chronologies are arranged by elevation with highest elevation plots or trees at the top. Alphanumeric codes on the right indicate plot or cluster identity. Solid lines indicate chronologies are in recording status. Solid vertical lines are years with fire scars and open vertical lines designate injuries or abrupt changes in ring-widths.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LAW05D04
LAW04E02
LAW04E03
LAW04D03
LAW04D02
LAW03C02
200 1250 1300 1350 1400 1450 1500 1550 1600 1650 1700 1750 1800 1850 1900 1950 2000
Year

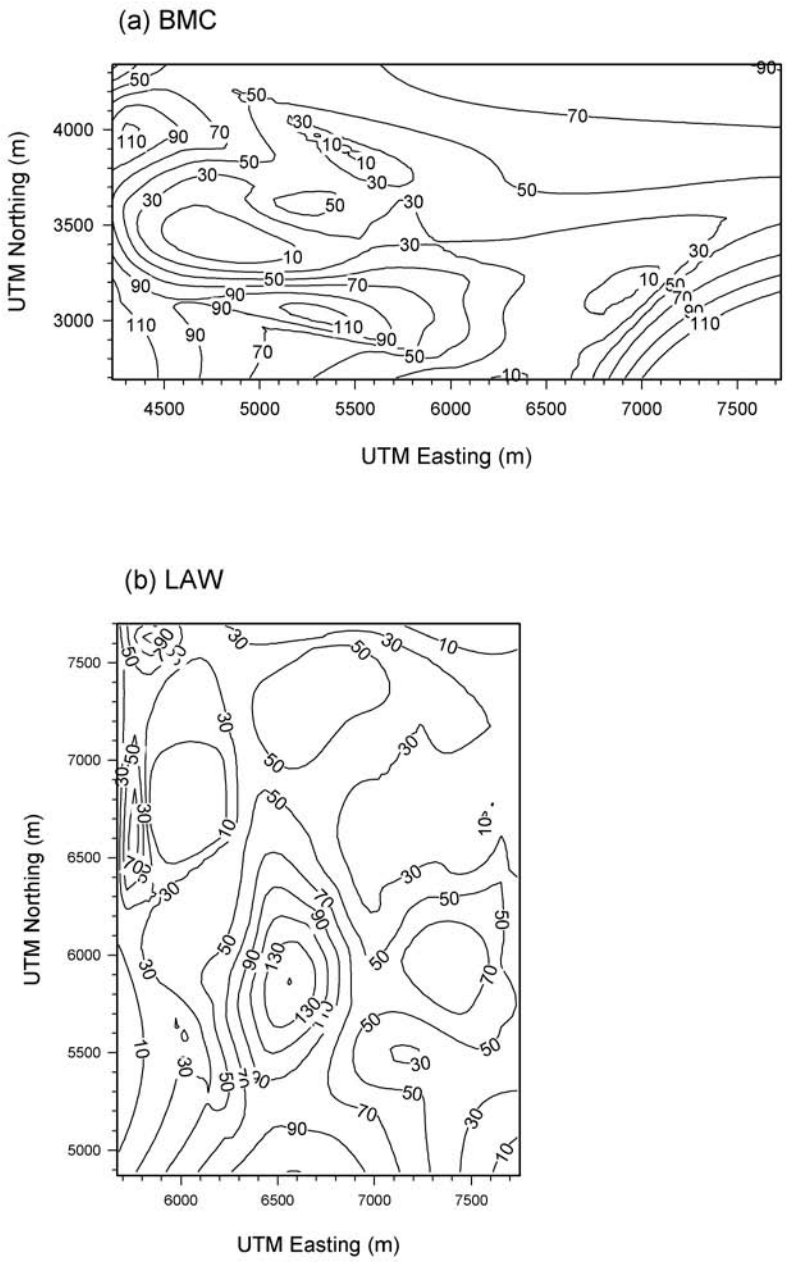


Figure 3.5. Two-dimensional contour plots of fire frequency for BMC (a) and LAW (b) fire-sheds. Contour lines designate gradients in MFI (10-year increments) based upon UTM coordinates for 27 (BMC) and 36 (LAW) PMFI estimates determined from fire-scar-based fire chronologies.

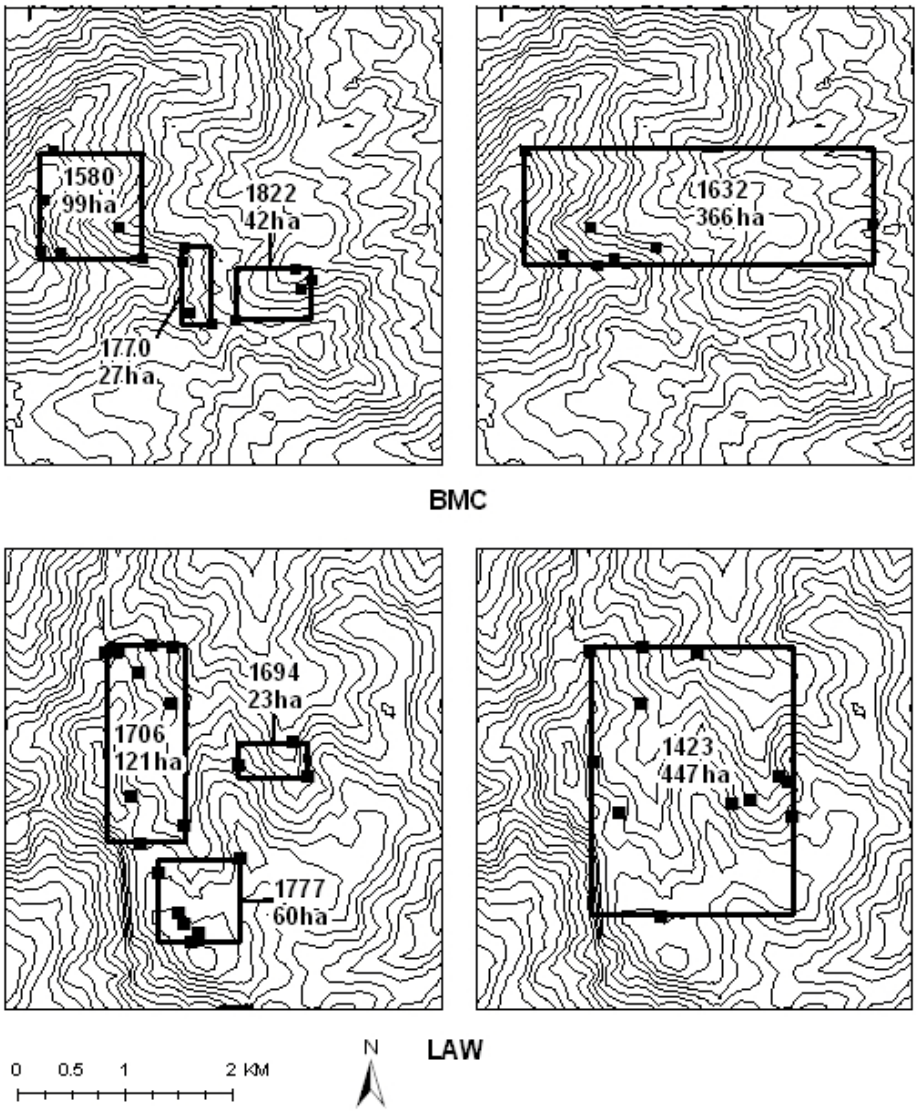
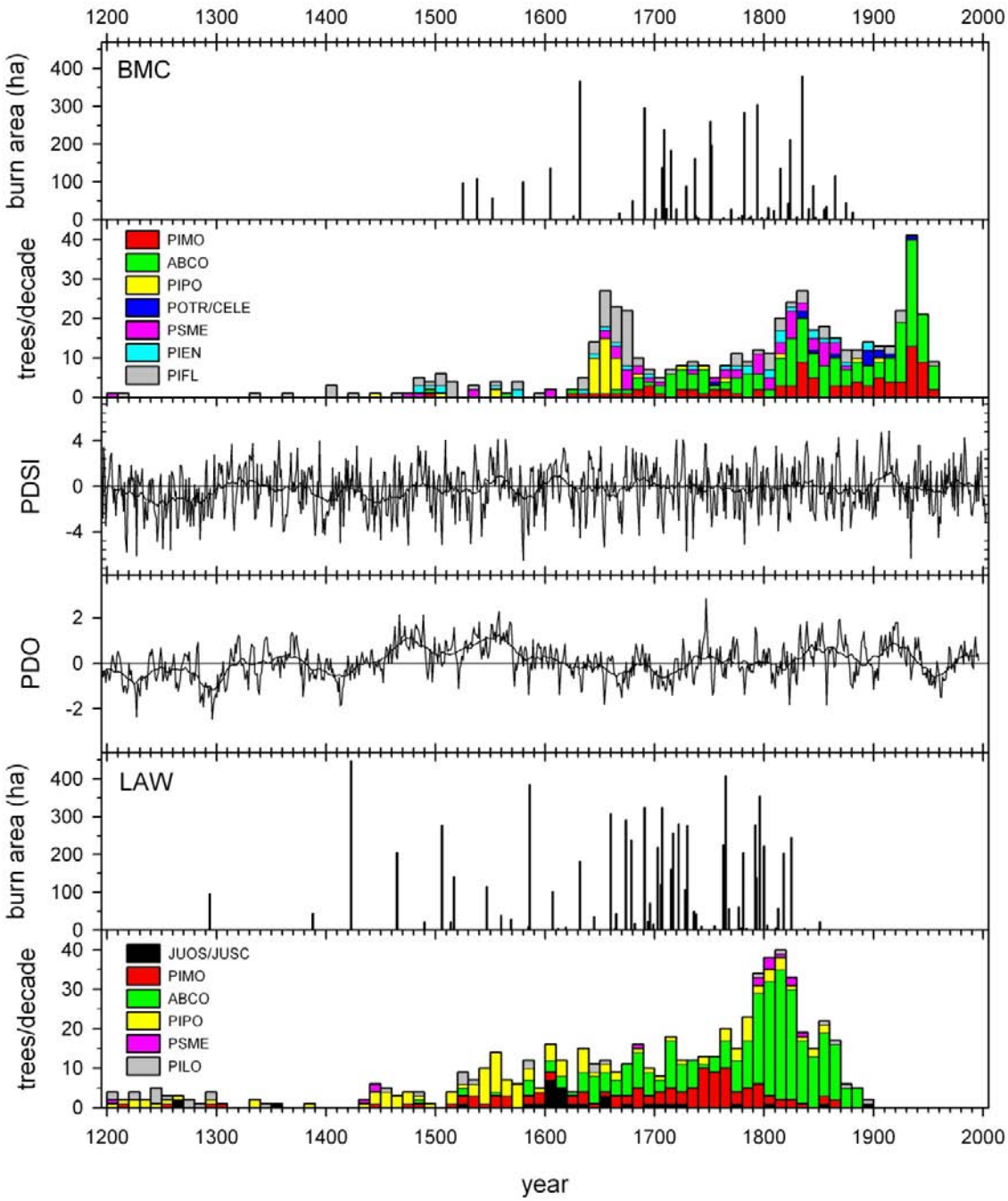


Figure 3.6. Examples of fire size (burn area) estimates from BMC and LAW fire-sheds where size is equal to the area of the smallest rectangle that is able to include all coordinates of trees recording fire for the year with sides oriented in cardinal directions. Fire-shed, year, and estimated burn area (ha) are given for each example. Contour lines indicate elevation changes at 40-m intervals.

Figure 3.7. Fire size (burn area) estimates and tree recruitment for BMC and LAW fire-sheds (1200-2000). Tree recruitment is in 10-year bins. For BMC, red = single-needle pinyon pine, green = white fir, yellow = ponderosa pine, dark blue = quaking aspen and curlleaf mountain mahogany, rose = Douglas fir, light blue = Englemann spruce, and grey = limber pine. For LAW, black = Utah and Rocky Mountain juniper and grey = Great Basin bristlecone pine. Annual variation in Palmer Drought Severity Index (PDSI) and Pacific Decadal Oscillation (PDO) are plotted for visual comparison. Negative PDSI indicates drought where the degree of departure from the mean (center line) infers drought intensity. In the southwest USA, negative (positive) PDO is associated with drought (pluvials). Decadal-scale variation in PDSI and PDO is shown by line smoothed with a cubic spline that retains 50% of variation over segments of 25 years.


BMC
burn area (ha)
trees/decade
PIMO
ABCO
PIPO
POTR/CELE
PSME
PIEN
PIFL
PDSI
PDO
LAW
burn area (ha)
trees/decade
JUOS/JUSC
PIMO
ABCO
PIPO
PSME
PILO
year

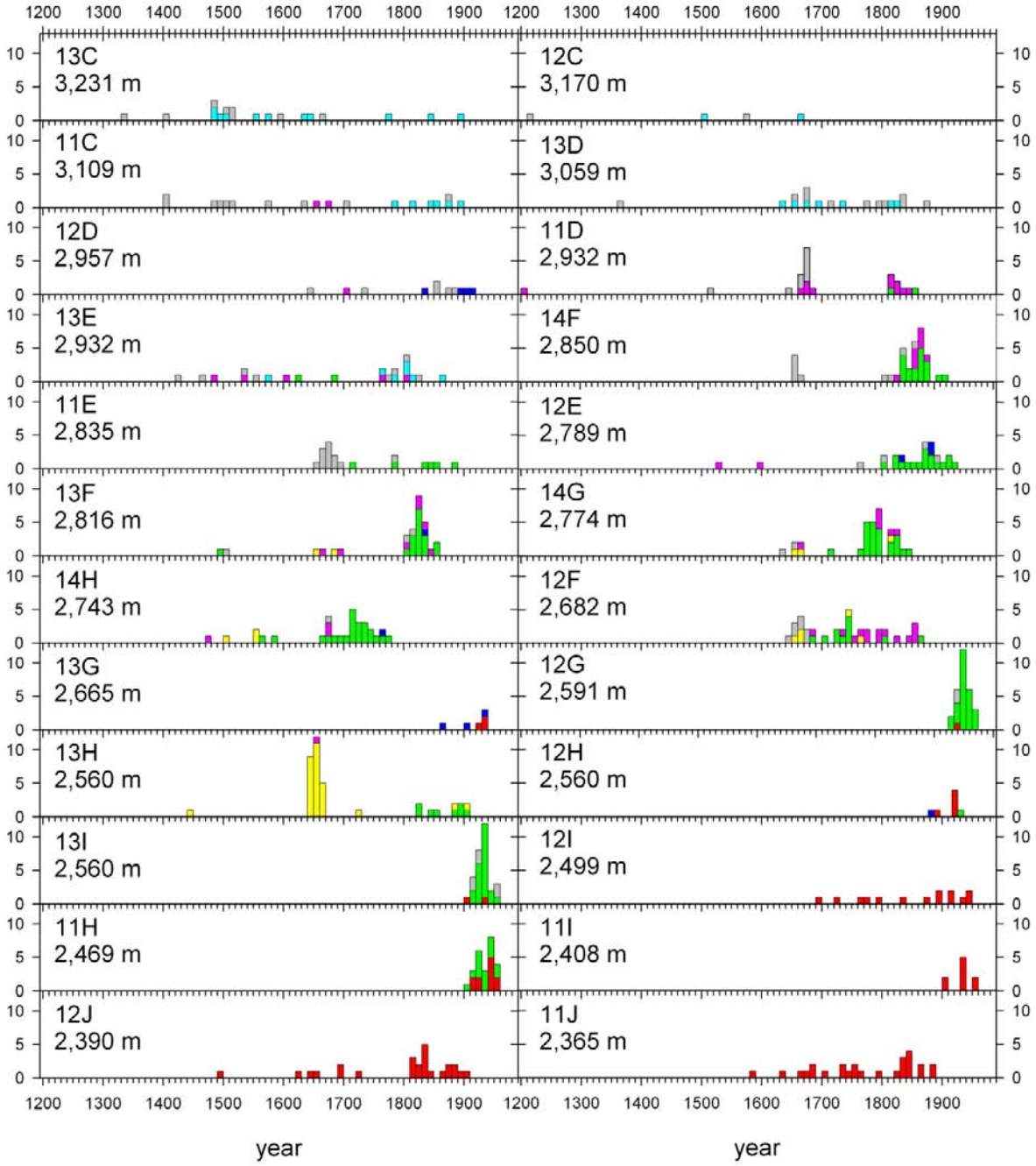


Figure 3.8. Individual panels show tree recruitment by plot (1200-2000) at the BMC fire-shed with plots arranged from top to bottom in order of decreasing elevation. Alphanumeric codes indicate plot locations. Recruitment dates are based on pith dates placed in 10-year bins. Tree species are as follows: red = single- needle pinyon pine, green = white fir, yellow = ponderosa pine, dark blue = quaking aspen and curlleaf mountain mahogany, rose = Douglas fir, light blue = Englemann spruce, and grey = limber pine.

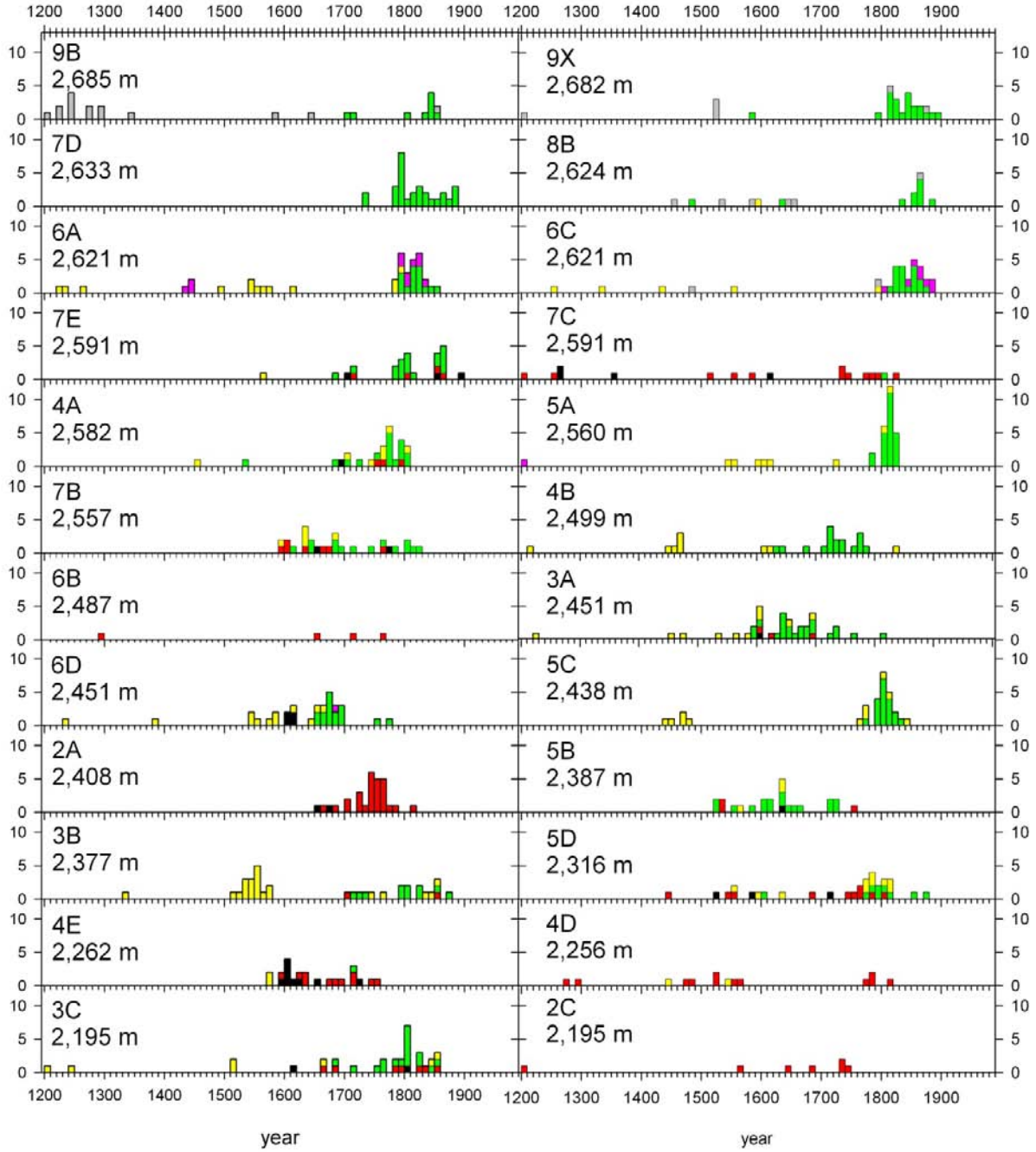


Figure 3.9. Individual panels show tree recruitment by plot (1200-2000) at the LAW fire-shed with plots arranged from top to bottom in order of decreasing elevation. Alphanumeric codes indicate plot locations. Recruitment dates are based on pith dates placed in 10-year bins. Tree species are as follows: black = Utah and Rocky Mountain juniper, red = single-needle pinyon pine, green = white fir, yellow = ponderosa pine, rose = Douglas fir, and grey = Great Basin bristlecone pine.