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Regional and local controls on historical fire regimes of dry forests and woodlands in the Rogue River Basin, Oregon, USA

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ABSTRACT

Fire regimes structure plant communities worldwide with regional and local factors, including anthropogenic fire management, influencing fire frequency and severity. Forests of the Rogue River Basin in Oregon, USA, are both productive and fire-prone due to ample winter precipitation and summer drought; yet management in this region is strongly influenced by forest practices that depend on fire exclusion. Regionally, climate change is increasing fire frequency, elevating the importance of understanding historically frequent-fire regimes.

We use cross-dated fire-scars to characterize historical fire return intervals, seasonality, and relationships with climate beginning in 1650 CE for 13 sites representative of southwestern Oregon dry forests. Using systematic literature review, we link our local fire histories to a regional dataset and evaluate our data relative to more intensively studied conifer/hardwood forest types in California.

Fire-scars show that fires in the Rogue Basin were frequent and regular until disrupted in the 1850s through 1910s, corresponding with forced displacement of Native Americans and Euro-American settlement. Median historical fire return intervals were 8 years at the stand-scale (< 25 ha), with site medians ranging from five to 14 years and no significant differences between sampled vegetation types. Burn seasonality was broadly distributed with 47% of recorded fires in the latewood (midsummer), 30% at the ring boundary (late summer and fall), and 23% in the earlywood (spring and early summer).

The number of sites recording fire each year was associated with Palmer Drought Severity Index (PDSI) and El Niño Southern Oscillation Index (ENSO). Fires were detected in the study area every other year, and synchrony among sites was associated with stronger annual drought. The ENSO synchronization of fire suggests an herbaceous fuel signal, with warm winters/wet summers two years prior to widespread fire-years, a pattern observed globally in fuel-limited systems.

Stand-scale fire histories in the Klamath, southern Cascades, and northern Sierra Nevada ecoregions resemble Rogue River Basin stand-scale fire histories. Across dry mixed conifer, yellow pine, and mixed evergreen forests, fire return intervals converged on 8 years. Moist mixed conifer and red fir forests exhibited 13-year fire return intervals. Across ecoregions, fire periodicity was weakly correlated with climatic water deficit, but well-modeled by elevation, precipitation, and temperature. These data highlight the need for decadal fire and burning outside of the contemporary fire season for forest restoration and climate adaptation in the dry forests of the Rogue Basin.

1. Introduction

Fire is a critical force shaping plant, animal, and human communities worldwide, often strongly influenced by anthropogenic activities (Bowman et al., 2009; Ryan et al., 2013; Moritz et al., 2014). Globally, historical fire regimes have been disrupted (Marlon et al., 2013) and there is a real need to understand how historical fire regimes functioned, in order to reinitiate a more beneficial role of fire (Prichard et al., 2017).

In the western United States, conservation efforts in forests that developed with frequent fire often emphasize protecting remaining old growth forests (U.S. Fish & Wildlife Service, 2011; Ager et al., 2013) or promoting resilience and the historic range of variability (Covington et al., 1997; Allen et al., 2002; Haugo et al., 2015; Hessburg et al., 2016). Historical references are useful for understanding how forests function and change, but must be interpreted in the context of a changing climate and the variety of conditions that influence forest resilience, such as landscape context and disturbance frequency, severity, and patch size (Stephens et al., 2013; Higgs et al., 2014; Long et al., 2014). Networks of cross-dated fire-scarred trees from across the American West reveal a pervasive pattern of historically frequent-fire regimes disrupted in the late 1800s CE, followed by the contemporary fire-suppression era of relatively infrequent-fire (Falk et al., 2011; Marlon et al., 2012; Taylor et al., 2016). A century of effective fire exclusion and growing fire deficit intersects with climate change predictions of increasing wildfire likelihood in western North America (Westerling et al., 2006; Whitlock et al., 2008; Littell et al., 2009; Marlon et al., 2012) and elevates the need to understand how frequent-fire forests function (Falk et al., 2011; Stephens et al., 2013).

Diverse responses to new and old fires, including recovery from

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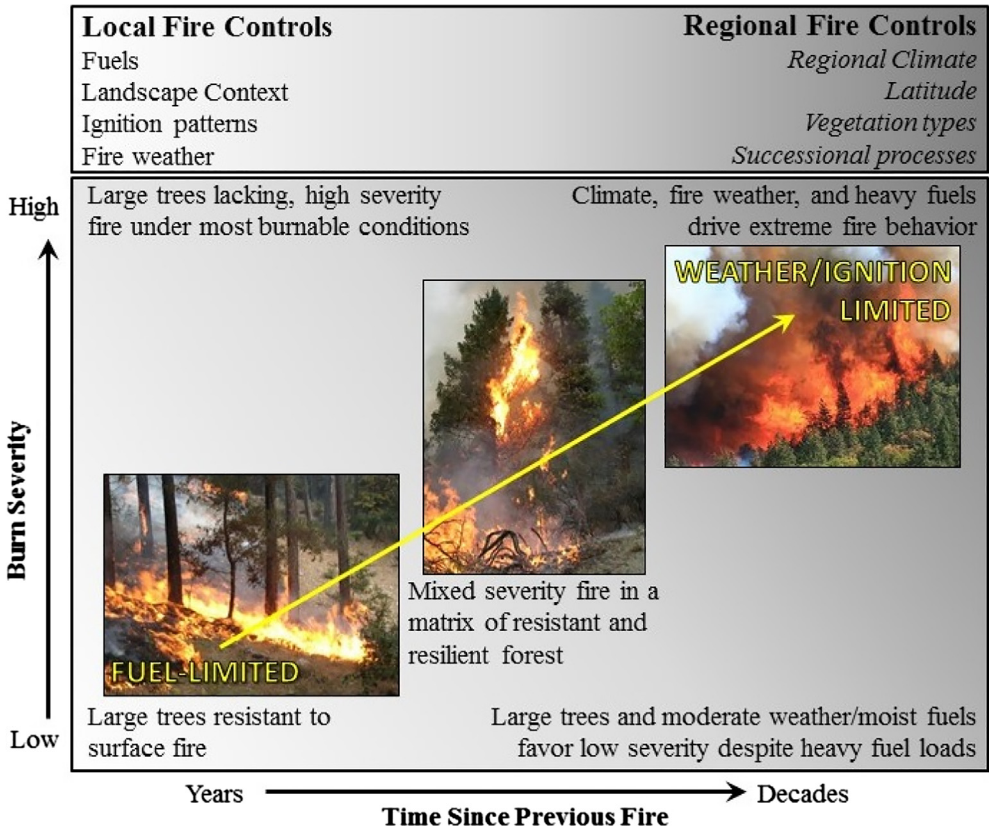

Fig. 1. Fire frequency interacts with a number of factors to determine burn severity. The relative importance of regional top-down (orange) and local bottom-up (white) fire controls is a complex gradient influenced by a number of interacting factors. Time since fire may elevate the importance of regional fire controls and under extreme regional forcing local fire controls may be less important. Photo credits: lower left and center Keith Perchemlides, upper right Scott Harding.

small high-severity patches, drive landscape-scale heterogeneity and forest resilience (Donato et al., 2009; Halofsky et al., 2011; Perry et al., 2011), a globally important principle for understanding fire resilience (Prichard et al., 2017). Fire frequency plays a critical role, with prior fires altering conditions for subsequent fires by reducing fuel availability and limiting subsequent fire intensity (Fig. 1 lower left corner; Taylor and Skinner, 2003, Collins and Stephens, 2010, Lydersen et al., 2014, Harris and Taylor, 2017). However, if large fire-resistant trees have been removed by severe fire or logging then even subsequent low-intensity fires can impede development of late-seral forest (Fig. 1 upper left corner; Taylor and Skinner, 2003, Thompson et al., 2007, Collins and Stephens, 2010, Miller et al., 2012, Lauvaux et al., 2016). With longer fire-return intervals, fuel accumulates with plant growth and large fire-resistant trees can re-grow, homogenizing landscape pattern and reducing the influence of old fires (Teske et al., 2012; Lydersen et al., 2014). Long fire-return intervals and higher-severity fires driven primarily by regional controls are more prevalent in moist forests and can become important in dry forests under active fire-suppression (Fig. 1 right side).

Disruption to the historical fire regimes of the western United States was driven by many factors, including weather, intensive grazing, and suppression of all fires, including Native American burning (Taylor and Skinner, 1998; Stephens et al., 2003; Fry and Stephens, 2006; Taylor et al., 2016). Local controls on the fire regime, (e.g. fuel availability, human ignitions, and heterogeneous landscape patches) have been diminished by fire regime disruption, increasing the relative importance of broad regional fire controls (e.g., climate) (Heyerdahl et al., 2008; Taylor et al., 2008; Falk et al., 2011; Taylor et al., 2016). Systems shifted to the right of Fig. 1 have an increased likelihood of high-severity fire, even if they historically burned frequently at low- to mixed-severity. With larger high-severity patch sizes, the pattern reinforces itself at a landscape-scale with long-term transitions from forest to other habitats (Thompson and Spies, 2010; Hessburg et al., 2016; Harris and Taylor, 2017; Lydersen et al., 2017).

Temperate mixed conifer/hardwood forests of the 4.8 million ha Klamath Mountains Ecoregion are among the most diverse in North America (Ricketts et al., 1999; Vander Schaaf et al., 2004) and because of this outstanding diversity they are a global conservation priority (DellaSala et al., 1999; Vander Schaaf et al., 2004). Similar to other ecosystems with a mediterranean-type climate, these forests face significant conservation threats, including a future climate increasingly conducive to fire (Klausmeyer and Shaw, 2009). Thirty-two percent of the Klamath Ecoregion is in Oregon, with much of it in the 1.3-million ha Rogue River Basin, framed by the Siskiyou Mountains, southern Cascades, and Coast Range (Rogue Basin, Fig. 2). In the California Klamath, fire history studies suggest that prior to Euro-American settlement, lightning ignitions and Native American burning drove a frequent-fire regime, with severity patterns influenced by topography, weather, prior fire influences, and fuels (see Skinner and Taylor 2006, Skinner et al., 2006 and sources therein).

Understanding the role of fire in the Oregon portion of the Klamath Ecoregion has lagged due to limited fire history study and over-generalized assumptions that conflate these relatively drier forests with the wetter forests of the western Cascades and Coast Range further north. Consequently, managers and the public have supported even-aged timber management and aggressive fire suppression, widespread-practices that provide short-term advantages but carry significant long-term wildfire risk—reflecting the global challenge of incorporating fire ecology into management (Bowman et al., 2009; Ryan et al., 2013; Moritz et al., 2014). Local management plans increasingly utilize ecological forestry and introduction of managed fire (e.g. Franklin and Johnson, 2012, Halofsky et al., 2016, Jackson County and Josephine County, 2017), even when managing for species that require complex old-growth forest (U.S. Fish & Wildlife Service, 2011, Davis et al., 2015). Greater understanding of historical fire regimes is critical to long-term success of these efforts.

Diverse Native American cultures of the Rogue River Basin managed an array of resource values across grassland, savanna, woodland, and

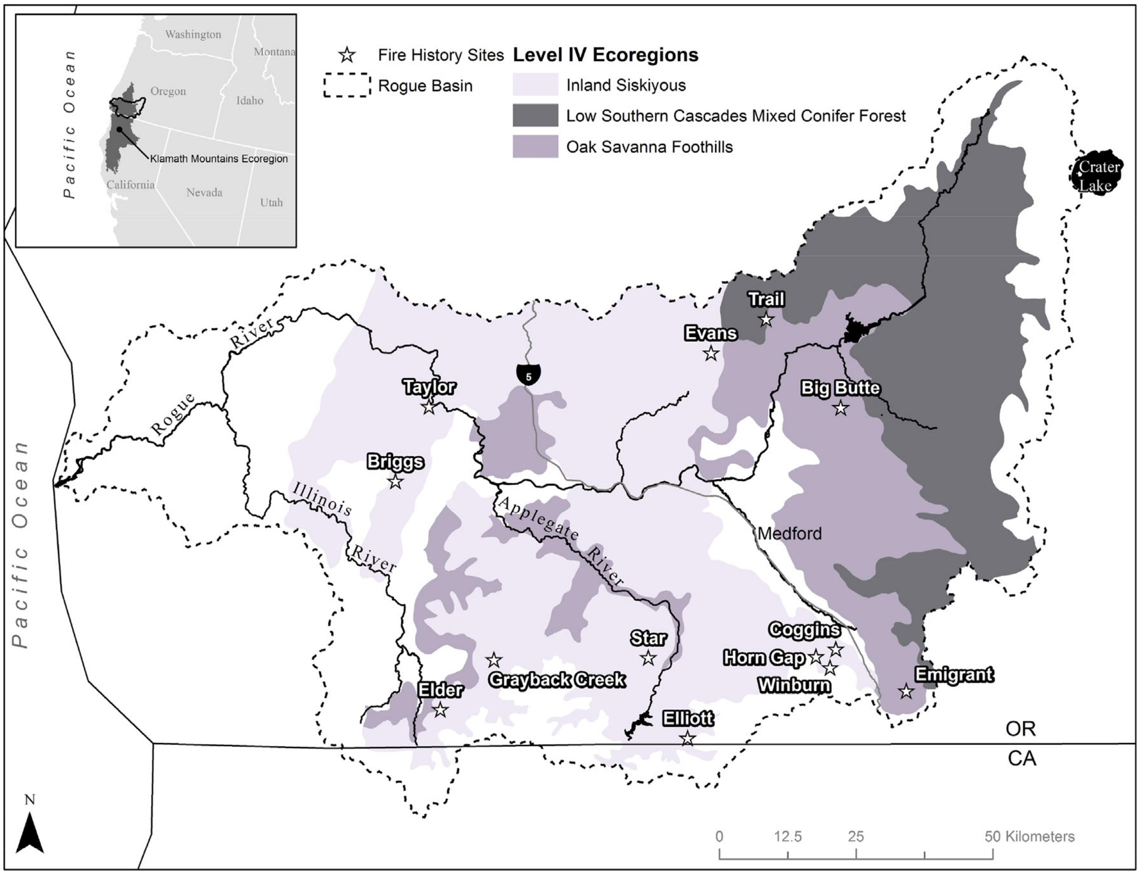

Fig. 2. Locations of fire history sites developed by this study, distributed across the Rogue River Basin in the Klamath Ecoregion. Ecoregions are Level IV Ecoregions from the Environmental Protection Agency.

forests, with fire as a primary management tool (Pullen, 1996; LaLande and Pullen, 1999); with evidence of Native Americans in the Rogue Basin for at least 10,000 years (Connolly, 1988; Minor, 2014). Their populations were forcibly removed to distant reservations in 1856 after decimation by multiple epidemics of European diseases and several years of armed conflict during Euro-American settlement in the 1850s (Gray, 1987; Douthit, 2002). Limited historical observations and ethnographic accounts of fire-use are drawn from Athabaskan groups living along the coast (Coquille, Chetco, Tolowa, and bands of the Tututni) to further inland (Chasta Costa, Umpqua, Galice and Applegate), and from Penutian Takelma and Hokan Shasta language groups inhabiting the interior Rogue Valley (Gray, 1987; Pullen, 1996; LaLande and Pullen, 1999; Long et al., 2016). Native groups of the Rogue Basin typically lived in larger winter villages along rivers and streams and dispersed in summer to upland camps (Gray, 1987; LaLande, 1989), with fire-use widespread and occurring throughout the year for diverse purposes (Pullen, 1996; LaLande and Pullen, 1999; Long et al., 2016), augmenting and modifying fires ignited by lightning.

Existing fire history studies in southwestern Oregon suggest that fire regimes may have been significantly different from moister forests of western Oregon and Washington (Weisberg and Swanson, 2003) and have been significantly disrupted for around 100 years across old-growth conifer (McNeil and Zobel, 1980; Agee, 1991; Taylor and Skinner, 1998; Colombaroli and Gavin, 2010; Sensenig et al., 2013) and lowland and mixed conifer riparian forests (Messier et al., 2012). However, these studies lack temporal precision because they are based on non-cross-dated fire-scars (Agee, 1991; Colombaroli and Gavin, 2010; Sensenig et al., 2013), aging cohorts of trees (Agee, 1991; Messier et al., 2012), or very limited numbers of cross-dated fire-scars (McNeil and Zobel, 1980; Messier et al., 2012).

Studies of sedimentary charcoal show that for millennia the Klamath Mountains, and more specifically the Siskiyou Mountains (Fig. 2) experienced fluctuations in fire activity over centuries, driven by regional changes in climate and vegetation (Briles et al., 2005; Whitlock et al., 2008; Colombaroli and Gavin, 2010; White et al., 2015) as well as anthropogenic burning (Crawford et al., 2015). Modern vegetation patterns established 3000–4000 years before present during a relatively cool period, with highest period of burn activity in forests of modern composition 700–1000 years before present (Briles et al., 2005; Whitlock et al., 2008; Colombaroli and Gavin, 2010; White et al., 2015), a time-period of significant Native American populations as well (Connolly, 1988; Minor, 2014). In the last 500 years there has been a decreasing trend in fire episodes at one site (White et al., 2015) but increasing trends at other sites up to the period of Euro-American immigration (Briles et al., 2005; Whitlock et al., 2008; Colombaroli and Gavin, 2010), consistent with west-wide patterns of climate that increasingly favors more frequent fire but a contemporary *fire deficit* driven by fire suppression (Marlon et al., 2012). This is valuable information, but resolution of sedimentary charcoal records is typically limited to decades at best (Whitlock et al., 2004; Higuera et al., 2010) and detection is biased toward recording high-severity events (Higuera et al., 2005), resulting in a need for complementary fine-scale, high-resolution fire histories.

Characterizing fire histories from cross-dated fire-scars (dendrochronology) can be temporally limited, particularly compared to sedimentation-based studies. However, it has been shown to be both efficient and accurate (Fulé et al., 2003; Swetnam and Baisan, 2003; Van Horne and Fulé, 2006; Farris et al., 2010; Farris et al., 2013) and with adequate sample size provides much greater temporal and spatial resolution than other fire history methods (Whitlock et al., 2004; Van Horne and Fulé, 2006; Allen et al., 2008; Higuera et al., 2010).

Here we use dendrochronology and a systematic literature review to

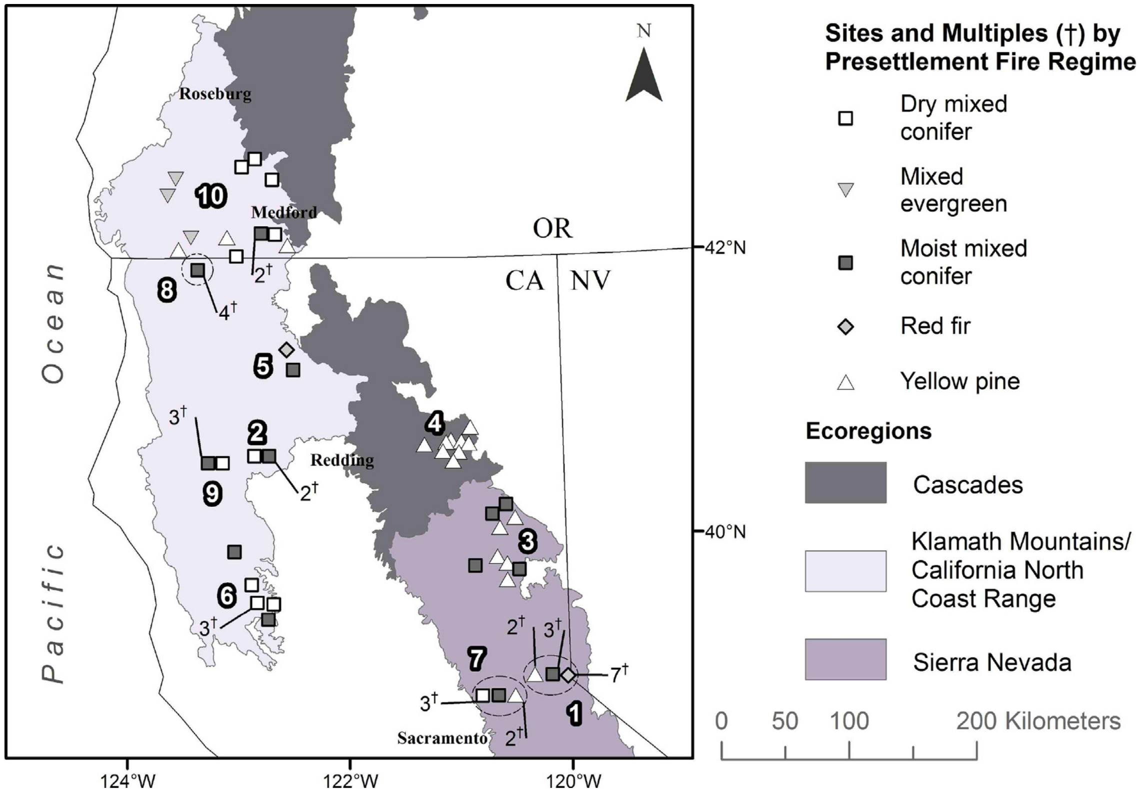

Fig. 3. Regional, cross-dated fire history sites by presettlement fire regime vegetation class from studies (1–10) reporting on fire-scar data meeting our requirements. Ecoregional distinctions are Level III from the Environmental Protection Agency. Studies are: 1 - Beaty and Taylor (2007); 2 – Fry and Stephens (2006); 3 – Moody et al. (2006); 4 – Norman and Taylor (2005); 5 – Skinner (2003); 6 – Skinner et al. (2009); 7 – Stephens and Collins (2004); 8 – Taylor and Skinner (1998); 9 – Taylor and Skinner (2003); 10 – This manuscript.

develop fire histories for summer-dry forests of the Rogue Basin and to evaluate those data in the context of published fire history studies from other mediterranean conifer and hardwood forests in the surrounding region (Fig. 3). We characterize historical fire-periodicity and -seasonality at 13 sites (stand-scale, *sensu* Falk et al., 2011) distributed across the Rogue Basin (Fig. 2). Fire regime disruption is used to define the historical fire period and to understand how much fire regimes have changed. We test for differences in historical fire-return interval among sites and among vegetation classes, hypothesizing that drier forests will have more frequent and less variable fire-intervals than moister forests and we document historical burn seasonality. Using our network of 13 sites (regional-scale, *sensu* Falk et al., 2011), we evaluate how climate synchronized fires among sites under the historical fire regime from 1650 to 1900 CE, hypothesizing that dry/warm conditions would be associated with fires synchronized at more sites. Using our regional dataset, we test the importance of local and regional processes for explaining patterns in median fire return intervals across this broad geography.

2. Methods

2.1. Study area

We quantified fire histories at 13 conifer/hardwood forest sites (< 25 ha each) across the Rogue Basin (Fig. 2) in the Inland Siskiyous, Oak Savanna Foothill, and Low Southern Cascades Mixed Conifer Forest level 4 EPA ecoregions (Omernik and Griffith, 2014). Fire history sites ranged in elevation from 517 to 1412 m, located largely on midslopes and ridges with predominantly south, east, and west aspects (Table 1). All sites were historically dominated by large, fire-resistant conifers, with higher proportions of hardwoods at the lower-elevation sites. The site size of < 25 ha was chosen because it is common to find relatively homogenous patches of this size locally, and because published fire history studies tend to either fall within this size class or be at much larger scales.

Sites were selected to be representative and unbiased, using Geographic Information Systems to identify potential sites before going into the field. Southwestern Oregon was stratified by modeled potential vegetation type (PVT; Halofsky et al., 2014), solar insolation index (Davis and Goetz, 1990, modified by Vander Schaaf et al., 2004), and topographic position index (Jenness, 2006). Solar insolation was split into warm and cool categories using natural breaks and topographic position was split into ridge, midslope, and bottom. Within bottom slope positions, warm and cool insolation settings were merged resulting in five biophysical settings in each plant series. Sites were selected to be dispersed across the Rogue Basin, in relatively undisturbed sites, within large patches of warm insolation, midslope/ridge topographic positions in the most abundant PVTs. Two of the sites (Briggs, and Grayback) were selected because recent timber harvest facilitated collection of fire scarred samples from stumps. For the other sites, U.S. Forest Service and Bureau of Land Management records were used to identify stands that had not been extensively harvested or burned by wildfire in the 20th century.

In the field, we classified the vegetation into plant associations (Atzet et al., 1996), then assigned them to PVT using a crosswalk developed by the Region 6 Forest Service ecologists and collaborators (Appendix A.1). The PVTs sampled at these sites (Table 1), cover 812,000 ha of the 1.3 million ha Rogue Basin (as mapped by Halofsky et al., 2014) and all are classified as fire regime group I, defined broadly as having a historical fire return interval of < 30 years (Barrett et al., 2010). Fire regime group I forest PVTs comprise 86% of Rogue Basin forests and of these 39% are Dry *Pseudotsuga menziesii*, 25% are Intermediate *Abies concolor*, 15% are Dry *Notholithocarpus densiflorus* (syn. *Lithocarpus densiflorus*) – *P. menziesii*, 3% are *Pinus jeffreyi* and 2% are

Table 1
Vegetation types and physical settings of the thirteen Rogue Basin fire history sites. Potential vegetation type is based on plant association. Presettlement fire regime (PFR) is classified as in (Van de Water and Safford, 2011), based on a crosswalk from biophysical setting from (LANDFIRE, 2008). See Appendix A.1 for the full breakdown of vegetation classifications.

Site	Potential vegetation type	Presettlement fire regime	Elevation (m)	Aspect	Slope (%)	Latitude	Longitude
Big Butte	Dry *Pseudotsuga menziesii*	Dry mixed conifer	745	292	15	42.56	−122.68
Briggs	Dry *Notholithocarpus densiflorus – P. menziesii*	Mixed evergreen	927	140	28	42.43	−123.68
Coggins	Dry *P. menziesii*	Dry mixed conifer	1036	102	15	42.16	−122.70
Elder	*Pinus jeffreyi*	Yellow pine	853	242	17	42.06	−123.57
Elliott	*Pinus ponderosa*	Dry mixed conifer	1412	217	22	42.01	−123.03
Emigrant	Dry *P. menziesii*	Yellow pine	1121	42	16	42.09	−122.54
Evans	Dry *P. menziesii*	Dry mixed conifer	905	120	26	42.65	−122.97
Grayback Creek	Dry *N. densiflorus – P. menziesii*	Mixed evergreen	576	184	7	42.14	−123.45
Horn Gap	Cool *Abies concolor*	Moist mixed conifer	1424	152	14	42.15	−122.74
Star	Dry *P. menziesii*	Yellow pine	1056	169	30	42.15	−123.11
Taylor	Dry *N. densiflorus – P. menziesii*	Mixed evergreen	517	268	15	42.56	−123.60
Trail	Dry *P. menziesii*	Dry mixed conifer	699	98	10	42.70	−122.85
Winburn	Intermediate *A. concolor*	Moist mixed conifer	1420	193	18	42.13	−122.71

Dry *Pinus ponderosa*. Our sites provide samples of these fire regime group I potential vegetation types roughly in proportion to their abundance across the landscape, but oversample Dry *N. densiflorus – P. menziesii* and undersample Intermediate *A. concolor* forests.

To relate our sites to regional fire history studies we also use the vegetation classifications of biophysical setting (BpS; LANDFIRE, 2008) which allows a consistent characterization of vegetation across the United States. We used BpS to assign presettlement fire regime (PFR; Van de Water and Safford, 2011) to sites. The PFR class for five of the Rogue Basin sites are dry mixed conifer, three are mixed evergreen, two are moist mixed conifer, and three are yellow pine (Table 1). The BpS to PFR crosswalk is available in Appendix A.1. BpS and BpS descriptions updated by LANDFIRE in 2018 are available in Appendix A.2.

2.2. Fire-scars

We quantified fire histories from the best-preserved fire-scar samples distributed across each site, with a target of collecting fire-scar samples from at least 10 trees. Fire-scars are basal cross-sections (full or partial) of trees that contain evidence of non-lethal fires, as well as tree growth over centuries. This methodology has been shown to be both efficient and accurate for quantifying fire regimes (Fulé et al., 2003, Swetnam and Baisan, 2003, Van Horne and Fulé, 2006; Farris et al., 2010, 2013). Each fire-scar sample was sanded to a high polish and cross-dated using standard dendrochronological techniques (Stokes and Smiley, 1968, Swetnam et al., 1985) with a local tree-ring chronology (chronology originator L.J. Graumlich; https://www.ncdc.noaa.gov/paleo/study/3298). Specimens that could not be visually cross-dated were cross-dated using the program COFECHA to suggest ring dates by statistically comparing the time series from measured annual rings of our samples with established chronologies (Grissino-Mayer, 2001). Statistical suggestions from COFECHA were verified by checking patterns in the wood. Seventeen samples could not be cross-dated, usually due to complacency (rings were not responsive to environmental fluctuations), and these were not used for analysis.

Fire-scar data were analyzed using the Fire History Analysis and Exploration System 2.0.0β (www.frames.gov/fhaes), a software program that manages fire event data and facilitates analysis of fire frequency, seasonality, and relationships between fire and climate. Fire histories at individual sites were developed using only fires that were recorded on two or more trees (samples) at a site to minimize reporting on very small fires with limited ecological impact. To ensure sufficient sample depth to accurately quantify fire-interval, fire histories were calculated only for periods of time with at least three trees recording fire (sensu, Yocom Kent et al., 2014). Using the sample size analysis tool in FHAES 2.0.0 each site was inspected to determine whether sample size was sufficient to characterize the fire return interval. For 11 of 13 sites this analysis clearly illustrated sufficient samples to characterize fire frequency. With five fire-scarred trees, the Horn Gap site weakly approached the asymptote, suggesting that fire-intervals may have been missed and that the fire history for this site is overly conservative. Only three trees were obtained at the Elder site, but those samples recorded 19, 17, and 12 fires each, demonstrating a weakness of using a sample size calculation that evaluates the number of fire-scar samples rather than the number of sampled intervals. We determined that given the number of fires in each of the Elder samples, we had sufficient sample depth to include it in this study. Given variable sample depth, the period of record varied by site (Table 2).

We identify the end of the time period characteristic of the historical fire regime (fire regime disruption) in two ways. First, at six sites, the fire return interval was consistent until the final fire, after which ample sample density extends the fire-scar record into the fire-suppression era. Second, at four sites, the final fires recorded followed fire-free intervals that were more than 3× the standard deviation in fire return interval for the record despite 6–9 recording trees available to record fire and they occurred after documented Euro-American immigration (Walling, 1884; Hickman and Christy, 2011). This classification is more conservative than the 2× standard deviation used by Yocom Kent et al. (2017) and strongly suggests that these outlier intervals are not representative of the historical fire regime. We do not ascribe a fire regime disruption date for three sites for which sample depth diminished in the early to mid-1800s. Quantification of historical fire regimes excludes intervals occurring after fire regime disruption as described above.

To statistically evaluate changes in fire return interval over time we also analyze fire return intervals by era, framed broadly by patterns of Euro-American settlement consistent with other fire history studies from the Klamath Ecoregion (Taylor and Skinner 1998, 2003, Fry and Stephens, 2006): presettlement < 1850, settlement 1850–1905, and fire-suppression era > 1905. Due to a limited number of intervals between fires in the fire-suppression era (seven for the entire study), we only tested for differences between the presettlement and settlement eras within each site using non-parametric methods.

Fire return interval comparisons among sites were conducted using non-parametric tests. Differences in fire return interval among vegetation classifications were determined using nested mixed-model analysis of variance with site nested within the vegetation classification as a random variable and interval square root transformed to meet the assumption of a normal distribution.

We analyzed seasonality for all fire observations for which seasonality could be ascribed. We treated individual fire years as replicates, summing all observations of seasonality in a year for a site. We combined early-early wood, mid-early wood, and late-early wood into one class (earlywood). The proportion of observations among seasons

Table 2
Summary of trees sampled and fire intervals recorded on more than one tree at each site. Date of fire regime disruption was not recorded (nr) at three sites due to inadequate sample depth. Composite historical fire return interval excludes fires occurring after fire regime disruption (see text for definition).

Site	Trees	Intervals	Fire record[†]	Disruption	Last fire	Historical fire return interval		
						Mean	Median	Range
Big Butte	9	23	1737–2000	1852	1930	8	5	1–40
Briggs	8	13	1768–1997	1906	1906	11	6	3–26
Coggins	7	26	1703–1886	1852	1866	7	6	3–20
Elder	3	12	1728–1883	1855	1855	11	9	3–33
Elliott	9	15	1772–1985	1905	1905	9	7	2–26
Emigrant	10	24	1698–1880	nr	1873	7	6	1–24
Evans	9	7	1729–1869	nr	1842	16	14	12–29
Grayback Creek	12	14	1662–1970	1905	1905	17	12	2–52
Horn Gap	5	19	1772–1910	1892	1910	7	8	1–18
Star	11	12	1680–1851	nr	1823	12	9	4–31
Taylor	7	14	1795–2004	1905	1924	9	9	4–19
Trail	8	18	1748–1910	1891	1891	8	8	2–20
Winburn	8	18	1649–1945	1883	1883	13	10	2–37

† Period with more than three recording trees.

within sites was tested using non-parametric methods.

We used fires recorded between 1650 and 1900 to evaluate relationships between climate and synchronicity of fires among sites, using the superposed epoch analysis feature in FHAES. We assessed fire teleconnections with the NINO3 reconstruction of the El Niño – Southern Oscillation (ENSO; data citation Cook, 2000, D'Arrigo et al., 2005), Western North American Summer Temperature (WNAT; Briffa et al., 2002) and Palmer Drought Severity Index (PDSI; Cook and Krusic, 2004). The NINO3 reconstruction (D'Arrigo et al., 2005) was selected because it is based on a region of the Pacific Ocean from −5 to 5°N and from 90 to 150°W which is strongly associated with interannual variability in US winter (December-February) precipitation (Schoennagel et al., 2005). All climate proxies were normalized so that the average departure for 1650–1900 was zero. Significant relationships were determined by comparing observed values to 1000 simulations using a random seed with a significance threshold of $p < 0.05$.

Relationships between ENSO and climate for USA Pacific Coast are spatially dependent, and tend to vary markedly with a north-south dipole around 40° latitude (Dettinger et al., 1998; Wise 2010), roughly the area of southern Oregon and northern California. Given some uncertainty around ENSO effects on the Rogue Basin (see discussion), we correlated the ENSO proxy (D'Arrigo et al., 2005) with climate variables collected at the Rogue Valley International Airport over 75 years (1928–2003). Stepwise regression was parameterized with annual, winter, and summer mean, minimum, and maximum temperature and precipitation for a parsimonious model of ENSO and local climate.

2.3. Systematic review of regional published fire history

To provide regional context for the Rogue River Basin fire history sites, we built a fire history database, using Google Scholar to search the terms "Fire Ecology", "Fire History", "Fire Return", "Fire Rotation", "Fire Regime", and "Fire Scar". We constrained the geographic extent of this search (Fig. 3) based on level 3 EPA ecoregions (Omernik and Griffith, 2014) of northern California and southwest Oregon, focused on the range of BpS (LANDFIRE, 2008) sampled in the Rogue Basin (Table 1). We included BpS subsumed by the range of sampled BpS (e.g. California red fir) but excluded systems with strong coastal influences (e.g. redwoods). This search identified 87 articles, which we augmented with recommendations from local ecologists and the references cited in the original sources, for a total of 115, including our own study. We assessed each source and included it in our analysis based on criteria to match our fire history methods: (1) spatial scale < 25 ha, (2) composite fire return intervals from fires scarring > 1 tree within a plot, (3) pre-settlement era fire history (≤ late 19th century to early 20th century, depending on location) distinct from fire-suppression era fire, and (4) georeferenced sites. We accepted 10 sources, including our own study (Fig. 3). The 105 rejected papers and rationale for rejection are in Appendix A.3.

From the accepted studies, we aggregated mean, median, maximum, and minimum fire return interval, as well as the period of record into a database. Study site locations were drawn from information within the source or gathered from the author, and these were entered into ArcMap 10.4 then used to extract biophysical parameters for each site. The elevation of each plot was determined using digital elevation models (data available from the U.S Geological Survey), corroborating and refining reported elevations. We identify PFR (Van de Water and Safford, 2011) for each site using the published descriptions and BpS mapping (BpS; LANDFIRE, 2008), referencing BpS descriptions updated by LANDFIRE in 2018 (Appendix A.2).

We evaluated climate data for mean and standard deviation in climate variables over the period from 1900 to 1929, the thirty-year period for which all data were available that most overlapped with our fire histories. We obtained annual precipitation and temperature mean, maximum, and minimum from the Parameter-elevation Regression on Independent Slopes Model (PRISM; PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, created 10 November 2017; Daly et al., 2008). Water dynamics, as modeled by actual evapotranspiration (AET; millimeters of annual moisture transpiration), an indicator of productivity, and climactic water deficit (CWD; millimeters of moisture deficit/year), a metric of water stress, have been shown to predict forest structure, burn probability, and burn severity in mixed conifer forests (Kane et al., 2015). We obtained CWD and AET from Dobrowski et al. (2013), a national dataset provided at 64 ha resolution. The coarse spatial resolution of this dataset is not ideal, but it was the only moisture deficit dataset available that applied across the entire study area. Stepwise regressions to model median fire return interval across the regional sites started with site PFR and all two-way interaction terms for elevation, latitude, precipitation, temperature, CWD, and AET. Two sites reported by (Skinner 2003) had high potential leverage due to relatively long fire return intervals; we replicated the model with these sites removed and report the results.

2.4. Analytical methods

Statistics were calculated using JMP 13.1.0 unless otherwise stated above. Assumptions of normality, heteroskedasticity, and equal sample sizes were assessed graphically and with Leven's test of heteroskedasticity. When parametric assumptions were met we used nested mixed model analysis of variance. When parametric assumptions were not met, Wilcoxon Rank Sums test was used to test for global differences, followed with Steel-Dwass non-parametric pairwise comparisons, which protect the overall error rate for multiple comparisons (Hsu, 1996). To evaluate relationships between tree-ring climate proxies and Rogue Basin weather and to generate a regional model to evaluate fire return intervals among sites we used stepwise regression (Miller, 2002) to identify significant terms based on Bayesian information criterion starting with relevant geophysical variables and all two-way interactions. To avoid spurious conclusions driven by serial autocorrelation (Katz, 1988), all time-series data (climate proxies and weather data) with temporal autocorrelation were pre-whitened using the ARIMA procedure in MINITAB Release 13.32 (Minitab, State College, Pennsylvania, USA).

3. Results

3.1. Site fire histories

Cross-dated fire-scar samples from 106 trees, recorded 217 fire-intervals among 13 sites (Table 2). The species composition of these trees was, 66% *P. ponderosa*, 12% *P. lambertiana,* 10% *Chamaecyparis lawsoniana*, 5% *P. menziesii,* 4% *Calocedrus decurrens*, and 3% *P. jeffreyi*.

The time-span at each site with at least three recording samples (period of record) varied by site (Table 2). The study period of record with at least one site with at least three recording samples spans 350 years, from 1649 to 2004, but the average period of record for a site spanned more than 200 years. Despite multiple sites with recording samples running into the twentieth century, on average the last recorded fire was 1886, with the most recent fire recorded in 1930 at Big Butte. The average date of fire regime disruption was 1885, with three sites in the 1850s (Big Butte, Coggins, and Elder) notably earlier than other sites and four sites in the 1910s (Briggs, Elliott, Grayback Creek, and Taylor) (Table 2). Fire regime disruption date did not vary by any vegetation classification: plant association (df = 6, 3), PVT (df = 4, 5), PFR (df = 3, 6), BpS (df = 3, 6), or EPA level IV ecoregion (df = 2, 7), all $\chi^2 < 0.294$ and $P > 0.109$.

For fires recording on at least two out of three or more available samples, historical site median fire return interval ranged from 5 to 14 years (Fig. 4) and fire return intervals significantly varied among sites ($\chi^2 = 30.999$; df = 12,196; $P = 0.002$). The median fire return interval across all thirteen sites in the Rogue Basin was 8 years between fires (calculated both as median of site medians and as median of all intervals). We found no significant relationships between fire return interval and plant association (df = 8, 4), PVT (df = 4, 8), PFR (df = 3, 8), BpS (df = 4, 8), or EPA level IV ecoregion (df = 2, 10), all $F < 2.786$ and $P > 0.165$. Historical fire return interval standard deviation, minimum, and maximum also did not significantly vary with these factors, all $\chi^2 < 10.500$ and $P > 0.230$). Fire return interval did not significantly vary between the presettlement (< 1850) and settlement (1850–1905) eras at any site (all $\chi^2 < 3.093$; df = 1, intervals in Table 2; and all $P > 0.049$), though sample sizes at some sites were quite small for the settlement era (e.g. two intervals at Big Butte).

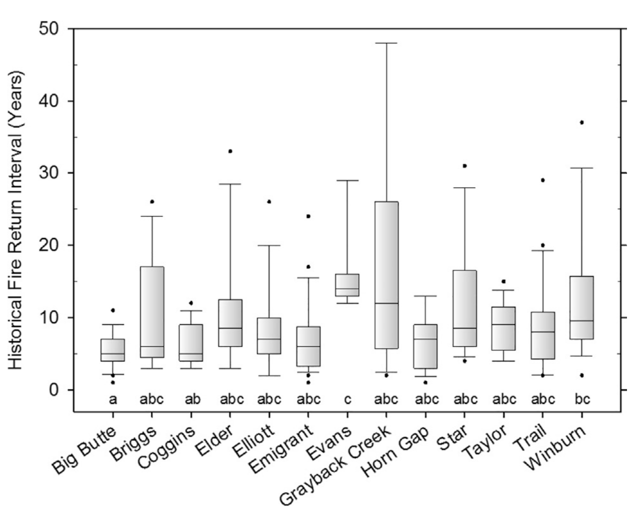

Fig. 4. Historical fire return intervals by site, with boxplots showing median, interquartile ranges, 90th percentiles, and outliers. Different letters indicate significant pairwise differences in median fire return interval ($P < 0.05$).

Seasonality of fire was determined for 548 of 864 cross-dated fire-scars: 56% were in *P. ponderosa*, 22% in *P. lambertiana*, 10% in *C. decurrens,* 8% in *C. lawsoniana,* 3% in *P. jeffreyi* and 1% in *P. menziesii*. Species composition of trees recording burn seasonality was consistent with proportions of trees sampled. Fire-scars were not equally distributed among seasons ($\chi^2 < 50.303$; df = 2, 969; $P < 0.001$) with nearly half recorded in the latewood, more than during dormancy or in earlywood (Table 3). Seven sites exhibited no significant differences in the proportion of fire-scars among seasons. The majority of fire-scars were formed in the earlywood and latewood at Big Butte and Winburn, latewood at Trail, and in latewood and dormancy at Taylor, Grayback, Horn Gap (Table 3).

3.2. Synchronous fire and climate

We focused the analysis of synchronous fires among sites and climate on the 250-year period between 1650 and 1900, for a total of 211 intervals with fires observed on average every other year somewhere in the Basin (Fig. 5). Over this period there were 68 years in which no fire was recorded, and these years were associated with cool/moist conditions indicated by a positive PDSI (Fig. 5). Fires were recorded at a single site in 62 years and these were not associated with any climate variables. Fires recorded at two or more sites were associated with warm/dry conditions as indicated by a negative PDSI in the year of burning.

In 29 years, fires were recorded at four or more sites, with a median return interval of 5 years. These synchronous fire events were associated with negative PDSI in the year of fire, 2.4 times more droughty than years recording fire at more than 2 sites. In addition to drought, fire years recorded at four or more sites were associated with positive ENSO two years prior to the fire year (Fig. 5). The only significant relationships with WNAT were for fire years recorded at five or more sites, even though WNAT tended to be cooler in less synchronous fire years and warmer in more synchronous fire years. Fires were recorded on at least five sites in 13 years: seven sites in 1817 and 1829, six sites in 1762, 1783, 1794, and 1798, and five sites in 1729, 1792, 1807, 1843, 1852, 1869, and 1883.

The final model of ENSO and Medford, Oregon weather included winter temperature, summer precipitation, and spring snow ($r^2 = 0.304$, df = 4, 71). The ENSO was positively associated with average winter temperature ($\beta = 0.2716$, $t = -2.33$, $P = 0.0225$), summer precipitation ($\beta = 0.1830$, $t = 2.16$, $P = 0.0344$), and the interaction term summer precipitation x winter minimum temperature ($\beta = 0.0676$, $t = 2.54$, $P = 0.0132$). Winter minimum temperature was negatively associated with ENSO ($\beta = -0.0981$, $t = -2.33$, $P = 0.0225$). Warm winters and wet summers are associated with positive ENSO in the Rogue Basin.

3.3. Regional fire histories

Nine studies, reporting on 59 sites, met the criteria for inclusion in the systematic review: bringing the total number of sites analyzed to 72 (Table 4). These papers, combined with our results, report on five PFRs: yellow pine, dry mixed conifer, mixed evergreen, moist mixed conifer and red fir forests across three EPA level III ecoregions (Fig. 3). Study intensity varies among PFRs and ecoregions with more studies in dry and moist mixed conifer in the Klamath Ecoregion than in other classes. We found no published studies that met our criteria for inclusion in mixed evergreen PFR.

Median FRI varied significantly among PFRs ($\chi^2 = 18.931$, df = 4,

Table 3
Seasonality of burning at sites, as a percentage of fires observed within a burn-season, across all years recording fire at that site for which seasonality could be determined (Fire Years). Earlywood, latewood, and dormancy infer burning in the spring, summer, and fall respectively from the phenological stage of wood recording fire. Chi square (χ^2) test evaluates differences among seasons within a site with significant pairwise differences ($P < 0.05$) indicated by different letters.

Site	Fire Years	Earlywood (%)	Latewood (%)	Dormancy (%)	χ^2	P
Big Butte	54	36 ab	50 a	14 b	12.491	0.002
Briggs	47	31	46	23	2.867	0.239
Coggins	58	22	38	40	3.549	0.170
Elder	30	17	41	42	4.288	0.117
Elliott	40	23	49	27	5.523	0.063
Emigrant	61	25	36	40	2.765	0.251
Evans	24	18	46	36	2.516	0.284
Grayback Creek	43	15 b	37 ab	49 a	6.000	0.050
Horn Gap	17	7 b	46 a	46 a	6.957	0.031
Star	50	22	43	35	3.789	0.150
Taylor	46	16 b	57 a	27 ab	11.280	0.004
Trail	61	24 b	57 a	19 b	19.833	< 0.001
Winburn	17	32 ab	58 a	10 b	7.831	0.020
Rogue Basin	**548**	**23 b**	**47 a**	**30 b**	**50.303**	**< 0.001**

$P = 0.0008$), with significantly shorter median fire return intervals in dry mixed conifer (8 years) than in red fir (14 years) or moist mixed conifer forest (13 years). Maximum fire return interval also significantly varied among PFRs ($\chi^2 = 12.672$, df = 4, $P = 0.0130$) with significantly longer maximum fire return intervals in moist mixed conifer (50 years) than in yellow pine (22 years).

The Klamath Ecoregion had the most sites represented (35) followed by the northern Sierra Nevada Ecoregion (27) and the southern

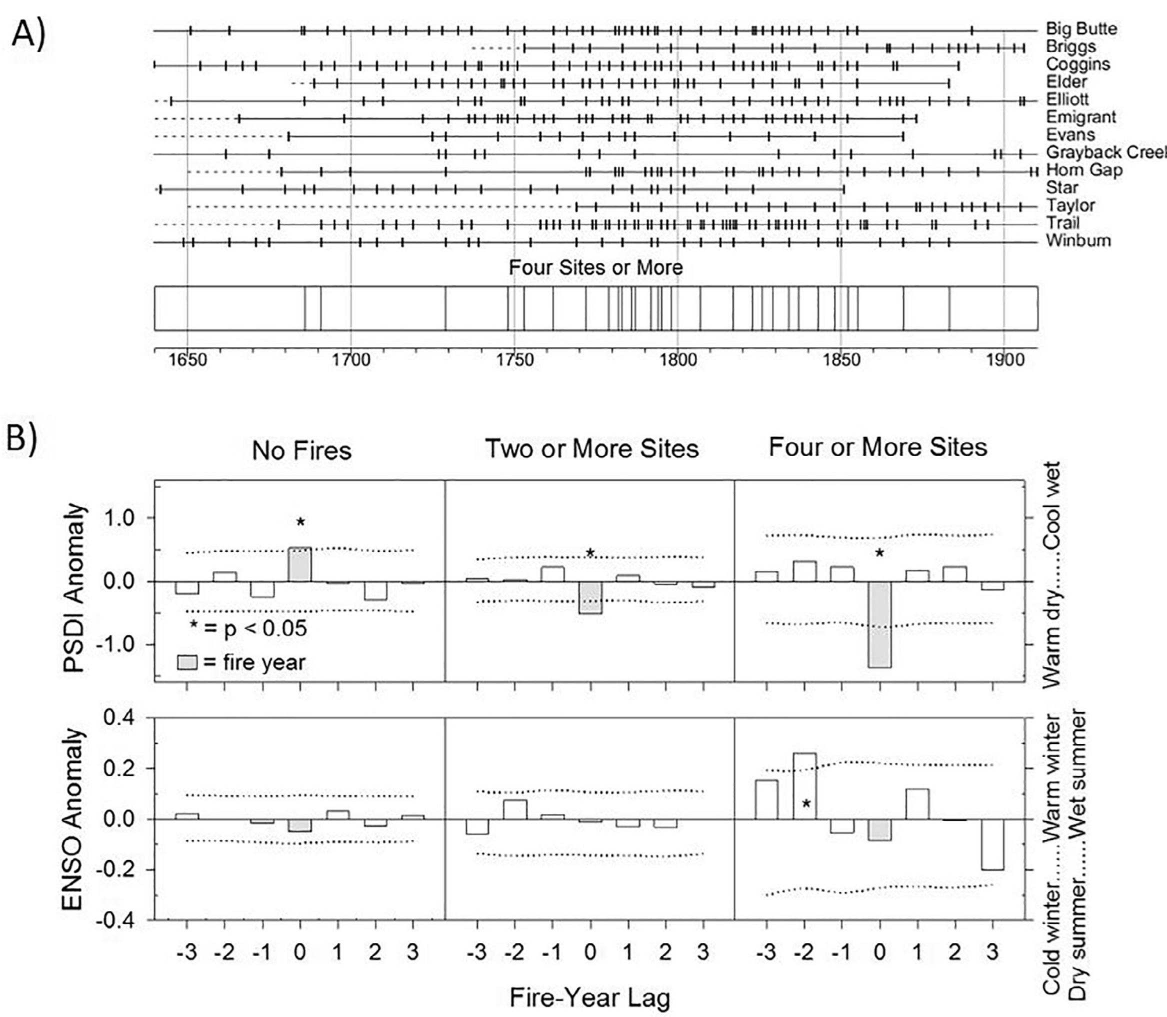

Fig. 5. (A) Incidence of fire across all of the Rogue Basin fire history sites from 1650 to 1900, with fire years recorded at four or more sites in the composite plot. Note that sample densities prior to 1700 were quite low at most sites (Table 2). (B) Teleconnections between climate and fire years (grey bars) recording on progressively more sites. Significant departures in Palmer Drought Severity Index (PDSI), and El Niño Southern Oscillation (ENSO) exceed the dashed 95% confidence intervals and are indicated with an asterisk. There were no significant relationships with fires recorded at one or more sites.

Table 4
Summary of cross-dated, composite fire histories for sites < 25 ha from the systematic review, organized by presettlement fire regime (PFR), and EPA level III ecoregion. Species composition is listed in order of contemporary dominance. Historical composite fire return interval is taken from published literature with each source potentially contributing more than one site. Range of medians is the range in median fire return interval among sites while the absolute range is the maximum and minimum observed interval for that class. Fire-season is the percent of historical fires recorded in fire-scarred wood.

PFR	Ecoregion	Species composition[†]	Historical fire return interval				Fire-season (%)[‡]			Sites	Sources[§]
			Mean	Median	Range of medians	Absolute range	E	L	D		
Dry mixed conifer	Cascades	PSME, PIPO, PILA, ABCO, CADE27, QUKE	8	7	–	2–29	24	57	19	1	10
	Klamath Mountains/CA High N. Coast Range	PSME, PIPO, ARME, QUKE, PILA, CADE27	10	8	5–15	1–76	16	44	40	16	2, 6, 9, 10
	Sierra Nevada	PSME, PIPO, PILA, CADE27, QUKE, QUCH2	9	7	5–10	2–22	0	79	21	4	7
Mixed evergreen	Klamath Mountains/CA High N. Coast Range	PSME, NODE3, ARME, QUKE, QUCH2, PILA	12	9	6–12	2–52	21	47	33	3	10
Moist mixed conifer	Klamath Mountains/CA High N. Coast Range	PSME, ABCO, PIPO, CHCH7, PILA, CADE27	18	14	8–36	1–116	8	25	68	12	2, 5, 6, 8, 9, 10
	Sierra Nevada	ABCO, PSME, PIPO, PILA, CADE27, QUKE	13	11	7–17	1–22	14	23	64	6	1, 3
Red fir	Klamath Mountains/CA High N. Coast Range	PIJE, CADE27, ABCO, ABMA, PICO, PIMO3	50	31	–	7–148	1	4	95	1	5
	Sierra Nevada	ABMA, PIMO3, PIJE, PICO, TSME, CADE27	12	12	8–17	NR¶	11	14	75	7	1
Yellow pine	Cascades	PIJE, PIPO, PICO, CADE27, PILA, JUOC	10	10	5–13	1–27	40	12	48	9	4
	Klamath Mountains/CA High N. Coast Range	PSME, QUCH2, ARME, PIPO, PIJE, QUKE	10	8	6–9	1–33	21	40	39	3	10
	Sierra Nevada	PIJE, ABCO, CADE27, PILA, PIPO, PICO	10	8	5–16	2–33	12	37	50	10	1, 3, 7

¶NR = Not recorded.

[†] ABCO = *Abies concolor*, ABMA = *Abies magnifica*; CADE27 = *Calocedrus decurrens;* CHCH7 = *Chrysolepis chrysophylla*; JUOC = *Juniperus occidentalis* NODE3 = *Notholithocarpus densiflorus* syn. *Lithocarpus densiflorus;* PICO = *Pinus contorta*; PIJE = *Pinus jeffreyi,* PILA = *Pinus lambertiana*; PIMO3 = *Pinus monticola;* PIPO = *Pinus ponderosa;* PSME = *Pseudotsuga menziesii;* QUKE = *Quercus kelloggii;* QUCH2 = *Quercus chrysolepis*

[‡] E = Earlywood, L = Latewood, D = Dormant.

[§] 1 – Beaty and Taylor (2007); 2 – Fry and Stephens (2006); 3 – Moody et al. (2006); 4 – Norman and Taylor (2005); 5 – Skinner (2003); 6 – Skinner et al. (2009); 7 – Stephens and Collins (2004); 8 – Taylor and Skinner (1998); 9 – Taylor and Skinner (2003); 10 – This manuscript.

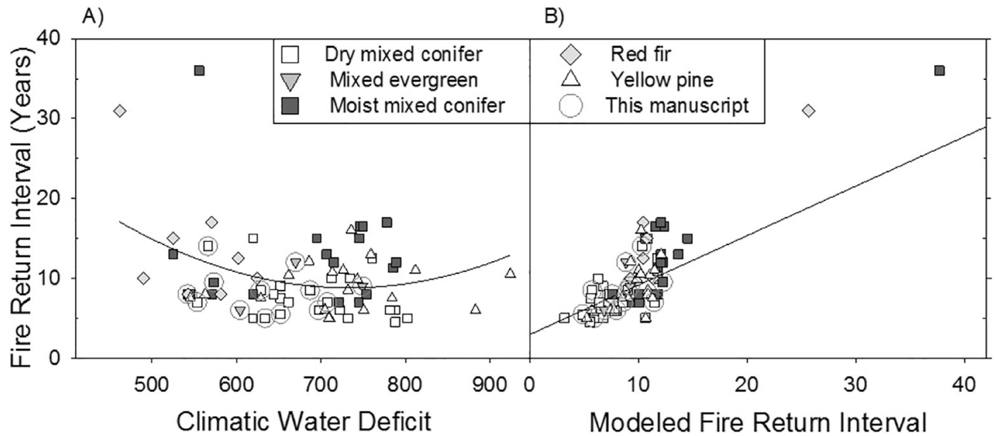

Fig. 6. (A) Correlation between median fire return interval and climatic water deficit; quadratic model $r^2 = 0.115$. (B) Correlation between median fire return interval and regional precipitation and temperature driven model of median fire return interval; linear model $r^2 = 0.731$.

Cascades Ecoregion (10) with uneven representation of PFR among ecoregions (Table 4). Median and maximum fire return interval did not vary by ecoregion for dry mixed conifer, moist mixed conifer, or yellow pine (all $\chi^2 < 3.453$, df = 2, $P > 0.1779$). Maximum fire return intervals were significantly higher ($\chi^2 = 4.361$, df = 1, $P < 0.0368$) in moist mixed conifer forests of the Klamath Ecoregion (57 years) than the northern Sierra Nevada Ecoregion (20 years), but there were only three sites in the northern Sierra Nevada Ecoregion.

Climatic water deficit exhibited a weak quadratic relationship with median fire return interval ($r^2 = 0.115$, F = 4.486, $P = 0.0147$; Fig. 6a) and the fire histories from this study fit consistently with regionally published values. The best predictive model of median fire return interval reflected exclusively local factors: elevation, annual precipitation, annual mean, minimum, and maximum temperature, the standard deviation in mean temperature, maximum temperature, AET, and three interaction terms (Appendix A.4). The most influential relationship was lengthening fire return interval with increasing precipitation, and decreasing fire return interval with decreasing minimum temperature. The model fit the data well, correlating linearly with observed median fire return intervals ($r^2 = 0.73$, F = 16.541, $P < 0.0001$; Fig. 6b) and the model weakened but remained significant with the leveraged red fir and moist mixed conifer sites from Skinner (2003) removed ($r^2 = 0.43$, F = 4.366, $P < 0.0001$).

4. Discussion

4.1. Increasing resolution from dendrochronology

Our results provide evidence that from the 1600s to the 1900s dry mixed conifer, moist mixed conifer, yellow pine, and mixed evergreen forests of the Rogue Basin all experienced frequent-fire. Across our 13 sites we found a median fire return interval of 8 years, and 90% of fire return intervals ranging from 3 to 30 years. The fire histories provided in this study support a substantial body of similar studies in the Klamath, south Cascade, and northern Sierra Nevada ecoregions (Table 4). Site level analysis of fire return intervals across this area (Table 4, Fig. 3) suggests that contemporary vegetation classifications, largely driven by local site productivity, may be too fine to characterize fire regimes that historically converged, possibly driven by the strong seasonal-drought and abundant ignitions that characterize the region.

Fire return intervals in Rogue Basin dry forests have been previously reported ranging from 9 to 21 years (McNeil and Zobel, 1980; Agee, 1991; Sensenig et al., 2013) with the most robust dataset to-date documenting a mean fire return interval of 17 years across the interior Coast range, Siskiyous, and Cascades (Sensenig et al., 2013). Importantly, while these fire histories were developed from fire-scars, they were not cross-dated or samples sizes were very small, resulting in likely underestimates of fire frequency. While frequent-fire forests dominate in the Rogue Basin, PVTs such as *Tsuga mertensiana*, Cool *A. concolor,* and *Pinus contorta* make up 16% of the forests in the Rogue Basin and they have fire regimes with distinctly longer fire return intervals that are more difficult to characterize with fire-scars (e.g. McNeil and Zobel, 1980; Forrestel et al., 2017). These forests are classified as fire regime groups III, IV, and V, they were not sampled in this study and we cast no inference to them.

Even when fire-scars are available, sample density and method of analysis can alter interpretations. In our study, the composite fire return interval of all fires was on average seven years, compared to an 8-year composite fire return interval of fires recorded on at least two trees at a site. Point fire return intervals, fires recorded at individual trees, were on average 60% longer than composite fire return intervals, 13 years between fires. These patterns are consistent with studies evaluating the precision of fire-scar-derived fire histories by comparing them to fire atlases in California, Arizona, and New Mexico (Van Horne and Fulé, 2006. Collins and Stephens, 2007; Farris et al., 2010, 2013). While documenting more frequent fire than previous publications in the Rogue Basin, these data are likely still conservative because absence of a fire-scar is not evidence that there was no fire (Gutsell and Johnson, 1996, Stephens et al., 2010).

This study does not directly evaluate historical fire severity, but fire return intervals reported here are consistent with low- to mixed-severity fuel-limited fire regimes with rare climate-driven high-severity events (Fig. 1, lower left). Evidence for a preponderance of frequent low- to mixed-severity fire does not preclude high-severity fire as an important part of historical fire regimes, but it does support the need for dry-forest restoration (*sensu* Franklin and Johnson, 2012; Stephens et al., 2013; Hessburg et al., 2016). This differs somewhat from the interpretations of historical fire regimes proposed by several authors (i.e. Baker, 2006, 2012; Williams and Baker, 2012; Baker, 2014; Odion et al., 2014). The methods used by these authors have been rigorously evaluated and found to be flawed (Fulé et al., 2006, 2013; Stevens et al., 2016; Levine et al., 2017), but see rebuttals by Odion et al. (2014) and Baker and Williams (2018). The ongoing debate regarding severity of historical fire regimes is identified and addressed thoroughly by others (e.g. Fulé et al., 2006; Stephens et al., 2015b; Hessburg et al., 2016; Safford and Stevens, 2017).

4.2. Fire regime disruption

Our results correspond with fire history studies across the dry forests of western United States that have documented a dramatic shift in fire frequency and character beginning in the late 1800s and becoming pronounced by the early 1900s (Falk et al., 2011; Marlon et al., 2012; Taylor et al., 2016). This is consistent with structural changes identified in riparian (Messier et al., 2012) and old growth forests (Sensenig et al., 2013), and to some extent in oak woodlands and chaparral (Detling, 1961; Riegel et al., 1992; Hosten et al., 2006; Duren et al., 2012) of the Rogue Basin. There is evidence for a matrix of historically open and dense vegetation in the Rogue Basin (Leiberg, 1900; Sensenig et al., 2013), particularly in oak woodlands and chaparral (Hosten et al., 2006; Duren et al., 2012; Dipaolo and Hosten 2015). However, evidence of a landscape mosaic of forest conditions does not conflict with our interpretation of frequent-fire disruption. Most of the fire histories closest to the Rogue Basin have also found that presettlement fire regimes were abruptly ended in the late 1800s in the California Klamath Mountains (Taylor and Skinner, 2003; Fry and Stephens, 2006; Skinner et al., 2009) and Blue Mountains of eastern Oregon (Heyerdahl et al., 2001; Johnston et al., 2016).

Patterns of Euro-American contact and settlement and associated declines in Native American populations – along with the introduction of agriculture and widespread, intensive livestock grazing – have been closely linked to changing fire regimes across the region (Taylor and Skinner, 1998, 2003; LaLande and Pullen, 1999; Norman and Taylor, 2003; Stephens and Collins, 2004; Fry and Stephens, 2006; Skinner et al., 2009; Taylor et al., 2016). In the California Sierra Nevada four distinct fire periods have been identified, pre-Euro-American settlement prior to 1775, the Spanish-Colonial Period from 1775 to 1849, Gold Rush-Settlement (1865–1904), and the fire-suppression period after about 1904 (Taylor et al., 2016). Contrasting with staged disruption in fire regime over many decades, we document an abrupt end to historical fire regimes reflecting local patterns of Euro-American settlement and a short, intense period of Native American forced removal.

Fire regime disruption was been variable across the Klamath Ecoregion, and ranged from the 1850s to the 1910s for our Rogue Basin sites. Disruption in the historical fire regime was evident in the mid to late 1800s in the south Klamath Mountains (Fry and Stephens, 2006), Modoc Plateau (Miller et al., 2003; Riegel et al., 2018), and California North Coast Range (Skinner et al., 2009). These early dates are consistent with our sites near developing Euro-American settlements of Eagle Point (Big Butte site), Ashland (Coggins site), and Waldo (Elder site) while the other sites were more remote from Euro-American influences in the 1850s (Walling, 1884; Hickman and Christy, 2011). Conflict between Euro-Americans and Native Americans precipitated the forced removal of the Native Americans from the Rogue Valley by 1856 (Gray, 1987; Pullen, 1996; Douthit, 2002). Thirty years later, General Joseph Lane observed in the Rogue Valley, "*The hilltops now covered by dense thickets of manzanita, madrone or evergreen brush were then devoid of bushes and trees because of the Indian habit of burning over the surface to remove obstructions to their seed and acorn gathering*" (Walling, 1884), and similar anecdotes were regionally common (Pullen, 1996; LaLande and Pullen, 1999).

More commonly fire regime disruption has not been documented in the California Klamath and south Cascades ecoregions until about 1905 (Taylor and Skinner, 1998; Taylor, 2000; Bekker and Taylor, 2001; Norman and Taylor, 2003; Taylor and Skinner, 2003; Taylor, 2010), or even the mid twentieth century in some higher elevation forests (Skinner, 2003). Fire regime disruption in the 1910s is congruent with our four relatively remote sites (Briggs, Elliott, Grayback Creek, and Taylor) as well as the establishment of the United States Forest Service in 1905 and institutionalized, aggressive fire suppression following the fires of 1910.

4.3. Seasonality of burning

We found fires recorded in all intra-ring positions throughout the period of record; the majority of fires were recorded in the latewood, followed by dormant season fires, and then by fires in the earlywood, with proportions varying by site. A few notably dry sites skewed to more early-season fires (e.g. Big Butte) and the more productive sites exhibiting more dormant-season fires (e.g. Horn Gap). This pattern supports the synthesis forwarded by Skinner (2002) that seasonality of fuel curing largely drives burn-seasonality in the Klamath Ecoregion because ignitions were historically abundant and not limiting. Underlying this synthesis is the finding that, in the Klamath Ecoregion, wetter sites have a higher proportion of dormant-season fires (Taylor and Skinner, 1998; Skinner, 2003; Taylor and Skinner, 2003; Skinner et al., 2009) and in more dry settings a higher proportion of growing-season fires (Skinner et al., 2009). This has been observed as well at much larger-scales, with significantly greater proportions of earlywood fires recorded at drier sites in the Blue Mountains (Heyerdahl et al., 2001) and more growing-season fire at low elevation *P. jeffreyi* forest types than higher elevation *Abies* dominated sites in the southern Cascades (Taylor, 2000).

Landscape position can also influence burn-seasonality. *Pinus ponderosa* forests embedded in shrub steppe of the South Cascades (Miller et al., 2003) and in pine meadows of the South Cascades (Norman and Taylor, 2003) are more herbaceous systems than much of the Klamath Ecoregion and correspondingly exhibit more early-season fire. Similarly, Fry and Stephens (2006) and Skinner et al. (2009) both found higher proportions of early-season burning in relatively productive forests, possibly due to their location at the edge of the Sacramento Valley (Fig. 3) exposed to grasslands and greater likelihood of Native American burning. Similarly, some of our sites in the Inland Siskiyou Mountains are in close proximity to the Oak Savanna Foothills (Fig. 2) or Inland Valleys and Grasslands (not shown) which historically were focal areas for Native American populations (Gray, 1987; Pullen, 1996).

Native Americans utilized all parts of the burn-season to accomplish specific cultural and ecological objectives (Pullen, 1996; LaLande and Pullen, 1999; Long et al., 2016). For example, in the Klamath Mountains July and August burning was used at higher elevations to culture bear grass (Anderson, 2005). In southern Oregon, late summer burning was used in the prairies and savannas to aid collecting seeds of *Madia* spp and in oak woodlands for acorn collection and to culture basketry materials (Pullen, 1996; LaLande and Pullen, 1999; Long et al., 2016). Late summer burning by the Native Americans, and resultant loss of forage for livestock, was acknowledged by the trappers guiding the Wilkes U.S. Exploring Expedition who traveled through the Rogue Basin in September 1841, and they observed an elderly woman igniting the grasses near present day Ashland (Boyd, 1999), between the Coggins and Emigrant fire history sites.

Growing-seasons fluctuate from year to year with annual variation in weather (Wright and Agee, 2004), so direct attribution of burn-seasonality from evidence in growth rings does not directly translate to month of burning. At our latitude in the Klamath and Cascades mountains physiology of the dominant tree species suggests that earlywood scars likely formed in late spring/early summer, latewood scars likely formed in mid-to-late summer, and dormant-ring boundary scars likely formed in late-summer or fall (Taylor and Skinner, 1998; Beaty and Taylor, 2001). Fires recorded between growth rings (dormancy) in a mediterranean climate likely took place in the dry fall rather than in the early spring when surface fuels are characterized by green herbaceous vegetation, even in years of limited precipitation (Taylor and Skinner, 1998, 2003).

Tree physiology studies to corroborate seasonality of earlywood and latewood development are limited locally, particularly for *P. ponderosa*, and cambial physiology varies somewhat by species (Waring, 1970). In the Siskiyous, onset of dormancy in conifers comes with physiological-drought, which peaks in September (Waring, 1969). In the Rogue Basin, *P. menziesii* diameter growth, and thus earlywood formation, is tied to temperature, but currently initiates in February and March (Ford et al., 2016) and diameter growth cessation, and thus dormancy, occurs in October (Ford et al., 2017). Initiation of latewood formation in *P. menziesii* has been identified from June through August, with seasonal variation due to moisture availability (Emmingham, 1977). Finally, a study of *P. menziesii* across western Oregon found that earlywood formed during May-July and latewood formed during the summer-drought, from July-September (Lee et al., 2016). These relatively local data are from *P. menziesii*, but a large-scale study of *P. ponderosa* and *P. menziesii* physiology found that growth responses to water availability were quite similar, though *P. ponderosa* was more responsive to the current year and late summer precipitation (Watson and Luckman, 2002). Together these data suggest that fires recorded in earlywood could be as early as February and as late as July, that latewood fires could occur in June through September, and fires in the dormant-season may be as early as August, but more likely occurred in or after September.

Earlywood and ring boundary fires are largely outside of the contemporary seasonality of lightning-ignited fires. From the period 1983–2017 lighting ignited 85% of wildfire acres burned in Jackson and Josephine counties (Oregon Department of Forestry, 2017), similar to findings of Miller et al. (2012) for northwestern California. Lightning-fires were recorded in January through October, with ignitions peaking in August (37%) and July (31%), followed by September (15%), June (10%), May (5%), and October (1%). Assuming historical lighting ignitions were patterned similarly, with 52% of ignitions in August and September minimal Native American ignitions would have been needed for substantial fall burning. The area burned by contemporary lighting-ignited fires is concentrated in mid-summer, with 63% of the acres burning in July, 35% in August, and 1% in September, largely overlapping with the seasonality of latewood. Mismatch between August-September ignitions (52%) and acres burned (36%) reflects fire suppression effectiveness. Changes in burn-season between historical and contemporary fire regimes reflect changes in human use of fire and fire-suppression responses, and suggest a contemporary deficit of early- and late-season burns.

4.4. Fire and climate

4.4.1. Historical fire synchrony and climate

Climate metrics were not significantly associated with 25% of fires, those recorded only at single sites. This suggests local controls, such as fuel availability and ignition sources, were key drivers of the historical fire regime, enabled by pronounced annual summer drought. Despite these strong local controls on historical fire, at a regional scale climate drove synchrony of fire among sites – a result supported by Yocom Kent et al. (2017). Regional control on fire occurrence was indicated by fire years synchronized among increasing numbers of sites associated with increasingly negative PDSI; a teleconnection reported for both historical and contemporary fire regimes (e.g., Wright and Agee, 2004; Trouet et al., 2010; Marlon et al., 2012; Taylor et al., 2016; Yocom Kent et al., 2017).

Fire years recorded at four or more sites were also associated with positive ENSO (El Niño) two years prior to burning, suggesting yet another mechanism for regional control on fire synchrony. This pattern is important where herbaceous fuels are responsible for fire spread; it is thought the moist years help to build up herbaceous fuels that become available in subsequent dry years and contribute to widespread fire (Swetnam and Anderson, 2008). This pattern has been documented globally, with evidence suggesting that it is most likely where resources tend to be limiting or highly variable (Krawchuk and Moritz, 2011).

This effect of positive ENSO differs from the pattern commonly found in the relatively productive Pacific Northwest where the negative ENSO (La Niña) anomaly is associated with moist years preceding the fire year (Hessl et al., 2004; Trouet et al., 2010), largely due to the north-south dipole of the ENSO effect along the West Coast (Dettinger et al., 1998; Wise 2010). A spatially broad analysis of fire history in the south Cascades found the positive ENSO effect was transient over time (Swetnam and Anderson, 2008; Taylor et al., 2008). Interestingly, a broader study of fire/climate in the Klamath Ecoregion did not find an ENSO association using sites largely dominated by forests rather than woodlands (Trouet et al., 2010). Further support for the importance of herbaceous fuel buildup as a driver of widespread fire years comes from the California North Coast Range, in the southern Klamath Ecoregion adjacent to the Sacramento Valley, where widespread fire years lagged three years behind particularly wet years, though a significant relationship with ENSO was lacking (Skinner et al., 2009).

Correlations between ENSO and climate for northern California and southern Oregon are spatially dependent, but in general ENSO is negatively correlated with precipitation and correlations between ENSO and climate tend to fulcrum at 40° latitude (Dettinger et al., 1998; Wise, 2010) and be stronger north of 42° latitude, (Trouet et al., 2009). We found that from 1928 to 2003, 90th percentile ENSO years exhibited 48% more summer precipitation than the average, winter temperature 7% higher than average, and less spring snowfall at the Rogue International Airport. However, it is important to note that the nature of the ENSO effect is not only transient over time but spatially as well (McAfee and Wise, 2016). Though it is informative, the record of 1928–2003 may not be representative of the ENSO effects that occurred during our fire history reconstruction (1650–1900).

The spatial dispersion of our sites is most similar to that of Norman and Taylor (2003), and similar to their study, herbaceous fuels were likely important carriers of widespread fire as much of the Rogue River Basin is connected by expanses of grasslands and woodlands. Also similar to Norman and Taylor (2003), many of our most synchronous fire years were in relatively quick succession (Fig. 5a), with a surprising number of years (29) recorded at 30% (four) or more of our sites over a 250 year period of record. Of these, 59% occurred within five years of the previous widespread fire year and 83% occurred within 10 years of a synchronous fire year. It is possible that the dispersion of our sites, and the sites of Norman and Taylor (2003) is better suited to detecting a link between herbaceous fuels and widespread fire than studies that focus on a more limited geographic extent or areas dominated by either forests or shrublands. Following low- to moderate-severity fire, shrubby fuels in the Siskiyous require more than 5–10 years to accumulate cover approaching pre-fire levels (Stephens et al., 2015a) suggesting historically frequent fires may not have been carried by shrubs. Further, areas of herbaceous fuels are likely more sensitive than forest or shrublands to short-term variation in precipitation (Swetnam and Anderson, 2008) and thus may have provided our study area a sufficiently rapid fuels response to support the historically frequent and widespread fires.

Our data suggest that historical fire in the sampled vegetation types, under a mediterranean climate with consistent winter precipitation and summer-drought, were sufficiently frequent to maintain fuel-limited conditions whereby local controls strongly influenced burn periodicity, overriding even differences in vegetation types (Fig. 1, lower left). This contrasts with contemporary burn patterns which generally occur following much longer intervals and vary significantly by vegetation class (Miller et al., 2012, Reilly et al., 2017). However, disruption of the historical linkage between fires and climate has been demonstrated across the American West (Marlon et al., 2012; Taylor et al., 2016). In the Klamath Mountains connections between burn patterns and climate have significantly strengthened since 1910; but the relationship between drought years and burn extent has weakened while the amount of precipitation at the time of ignition has increased in importance due to precipitation's influence on efficiency of suppression forces (Miller et al., 2012).

4.4.2. Fire and climate change

Climatic water deficit has been found associated with contemporary burn probability and has been used to forecast potential fire dynamics under a changing climate across the western United States (Westerling et al., 2006; Higuera et al., 2015; Mann et al., 2016; McKenzie and Littell, 2017; Tepley et al., 2017). At broad spatial and temporal-scales these correlations are bi-modal, with burn probability increasing with increasing CWD until fuels become limiting (Mann et al., 2016; McKenzie and Littell, 2017) or increasing with increasing AET until the amount of available moisture reduces burn probability (Mann et al., 2016). In contrast, we found only a weak relationship between fire return interval and CWD. One likely reason for this weak relationship could be a mismatch between the 64-ha spatial resolution of our CWD data and the complex topography of the region. Additionally, the changing relationship between fire and moisture deficit over time (e.g., Higuera et al., 2015) could explain this difference; studies that identify a relationship between CWD and fire analyzed burn patterns are largely contemporary (e.g. from 1976 to 2000 (Mann et al., 2016) or 1916 to 2003 (McKenzie and Littell, 2017).

The limitations of sampling a relatively small range of potential CWD could also confound correlations between CWD and historical fire periodicity. The CWD that we used (Dobrowski et al., 2013) ranges from 542 to 749 mm/yr (mean 636, SD 66) for the Rogue Basin sites which is a greater range than any other of the nine comparable studies of historical fire. Across all sites analyzed in this manuscript, CWD ranges from 462 to 924 mm/yr (mean 679, SD 94). In comparison, climate water deficit for the Klamath Ecoregion ranges from 216 to 1197 mm/yr (mean 661, SD 160) and the Cascade and northern Sierra Nevada ecoregions exhibit similar distributions of CWD. Overall the ecoregions have greater variability (60% larger standard deviation) in CWD than sampled sites. This suggests the potential for a broader range of relationships between CWD and fire return interval than characterized by existing cross-dated fire history studies, though existing studies do sample the predominant CWD in forests and central tendency in CWD for the ecoregions.

Climatic water deficit is predicted to increase substantially in the Klamath Ecoregion under climate change (Dobrowski et al., 2013; McKenzie and Littell; 2017) and will likely be accompanied by increased burn probability (Littell et al., 2010; Mann et al., 2016; McKenzie and Littell, 2017) and burn severity and patch size (Miller et al., 2012; van Mantgem et al., 2013; Kane et al., 2015; Tepley et al., 2017) while reducing the resilience of forest to fire by reducing climactic opportunities for regeneration (Lutz et al., 2010; Davis et al., 2016; Tepley et al., 2017). Extreme weather can override some fuel limitations (e.g., Lydersen et al., 2017), but our results elevate the complexity of climate and fire interactions (Bowman et al., 2009) and add credence to the possibility that local factors, such as fuel limitation, could dampen some of these predicted responses to a changing climate (Higuera et al., 2015; McKenzie and Littell, 2017). Thus, where resilient forests are the long-term goal, a climate adaptation strategy that avoids large areas of rapid state changes with effective risk mitigation strategies, including fuel reduction and forest restoration, may be appropriate (McKinley et al., 2011; Stephens et al., 2013; Millar and Stephenson, 2015; Golladay et al., 2016). Given potential for transformative wildfires in the Klamath Ecoregion (Thompson and Spies, 2010; Davis et al., 2015; Rockweit et al., 2017; Tepley et al., 2017) and worldwide (Prichard et al., 2017), these strategies may be particularly important in the Rogue River Basin (Halofsky et al., 2016) and all forests that historically were very fire resistant (lower left of Fig. 1).

5. Conclusions

Globally, anticipating management needs of dry forests under a changing climate requires understanding fire regimes that historically supported frequent-fire forests (Prichard et al., 2017). This research suggests that historically, dry forests and woodlands of the Rogue River Basin burned frequently, with median fire return intervals of 8 years and 90% of fire-intervals between 3 and 30 years. The historical synchrony of fires with climate validates the paradigm that annual drought supports widespread fire, and suggests that historical fuel beds had a significant herbaceous component. We document disruption of historical fire regimes as early as the 1850s at several sites corresponding with Euro-American settlement and displacement of Native Americans, and in the 1910s for more remote sites corresponding with programmatic fire suppression.

Regionally, we found fire regimes of mixed conifer, yellow pine, and mixed evergreen forests across the Klamath, southern Cascades, and northern Sierra Nevada Ecoregions converged on an eight-year fire return interval, likely driven by wet winters and dry summers of the overarching mediterranean-type climate. Median fire return intervals in moist mixed conifer and red fir forests were thirteen years and more variable. Historical fire return intervals were well predicted by precipitation, temperature, and elevation.

Throughout the 20th century, forest managers of western North America have attempted to ensure ecosystem services and protect human communities in-part through fire exclusion, even in dry forests. Visionary land managers are increasingly incorporating elements of historical low- to mixed-severity fire regimes into land management strategies that emphasize restoring open forest structure with mechanical thinning and controlled burning. Here we provide critical data that suggest decadal fire may be necessary to support low- to mixed-severity fire regimes. Correspondingly, adapting these forest systems to a changing climate while continuing to provide ecosystem benefits for people will require significant investment and innovative approaches to managing fire, such as increased spring and fall burning.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.foreco.2018.07.010. These data include Google maps of the most important areas described in this article.

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