



Ecology of the Bald Hills and Little Bald Hills of Redwood National Park: An assessment of vegetation change



ON THE COVER

Images of Oak Woodland in the Bald Hills (top by Eamon Engber) and Jeffrey Pine Woodland in the Little Bald Hills, (bottom by April Sahara) Redwood National and State Parks.

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Executive Summary

Introduction

The open forest, woodlands, and grasslands of the Bald Hills and Little Bald Hills in Redwood National and State Parks are distinctive habitats that support regionally rare plant species and scenic vistas. Although the specific antecedents are not fully understood, the relatively open habitats are believed to be a consequence of unique edaphic conditions and fire regimes that created disclimax conditions. Recently, park managers have noted vegetation changes that suggest these unique systems are changing, primarily through forest encroachment, with important consequences for understory fuel structure and park biodiversity.

Field Studies

This study investigated a suite of questions associated with the dynamics of fire, fuels and vegetation change in the Bald Hills and Little Bald Hills, respectively. The chapters of this study describe distinctive outcomes of specific analyses.

Key Findings

Overstory structure plays a dominant role in fuels and fire behavior in the Bald Hills of Redwood National Park. Across the structural gradients spanning grassland to invaded conifer forest, fuel mass decreased sharply, fuels were more compact, and these sheltered fuels retained more moisture and were less affected by wind. These combined effects reduced fire behavior during prescribed fires and suggest a mechanism for the persistence of forests in these landscapes. Invading conifers are susceptible to fire-caused mortality following prescribed fires in the Bald Hills. Sapling mortality was tightly linked to crown and stem injuries, factors easily monitored following prescribed fires. Understanding these injury thresholds will enable fire managers to adjust prescriptions to meet the reverse tree encroachment in these historically fire-prone ecosystems.

Grassland and the Jeffrey pine savanna vegetation type is rapidly disappearing from the Little Bald Hills. Time-series analysis of the proportional loss of grassland in LBH indicates the savanna-type vegetation will be gone sometime this century, probably within the next 50 years. Low-elevation, north-west facing slopes have been encroached more quickly than flatter, more south-easterly facing areas, which corresponds well to field reconnaissance of the ridgetop areas that remain savanna. Our results also indicate eventual disappearance of grassland areas. We were not able to assess the role, if any, of climate change on tree encroachment, nor could we directly test whether changes in fire regime have led to encroachment. It is possible that LBH is a remnant of an ecosystem managed by the Tolowa peoples who lived in the area before European settlement in 1850. Some evidence for this is illustrated by early tree establishment in the 1850s and 1860s. Variable establishment across the landscape is likely an artifact of spatial correlation; rapid encroachment into the remaining savanna areas seems likely.

Douglas-fir forests of the Little Bald Hills track the same increases in tree establishment as the Jeffrey pine savannas. Large increases in all tree species were found ca. 1850 and peaking around 1900-1920. Among tree species, Douglas-fir, Port Orford-cedar, tanoak, bay laurel, and knobcone pine all follow this pattern. As with the Jeffrey pine results, these increases coincide with the extirpation of the native Tolowa.

Introduction and Background

Within Redwood National Park (RNP), there are two notably open areas that show sharp contrast to the surrounding forest for which the park was established. The Bald Hills is an area with notable Oregon oak (*Quercus garryana*) woodland and a largely herbaceous understory that is notably rich in grassland species. The Little Bald Hills, situated at the edge of the coastal fog zone, contain large sheets of ultramafic soils, with outcrops of serpentinite, which are distinctive in the park and on the northern coast of California. The unique soils and varied microclimates of the Little Bald Hills harbor diverse vegetation assemblages, including disjunct Jeffrey pine (*Pinus jeffreyi*) and knobcone pine (*P. attenuata*) woodlands that contrast sharply with the surrounding redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*) forests. The woodlands harbor a rich array of rare or geographically disjunct plants, such as *Sanicula peckiana*, *Calystegia atriplicifolia* ssp. *buttensis*, *Iris innominata*, *Arnica spathulata*, and others (Smith et al. 2003).

Both areas are insular environments within the larger forest matrix that show evidence of afforestation by more widespread conifer species (chiefly Douglas-fir (*Pseudotsuga menziesii*)). Douglas-fir encroachment has occurred throughout western US grasslands, savannas, and woodlands (Hunter and Barbour, 2001; Stewman, 2001; Heyerdahl et al., 2006). In the Bald Hills, past studies have demonstrated that substantial portions of grassland and Oregon white oak woodland have converted to Douglas-fir-dominated forest since 1850, leading to drastic reductions in understory species diversity within invaded areas (Sugihara and Reed 1987; Fritschle 2008) and changes in fuel structure.

Where fire is excluded from woodlands and savannas, substantial ecological changes occur (Grossmann and Mladenoff 2007; Peterson et al. 2007). Frequently, increases in woody stem densities and advancement of forest boundaries lead to shading of herbaceous layers (Skinner 1995; Hoffman et al. 2003; Devine et al. 2007) and establishment of fire-sensitive trees and shrubs (Sugihara and Reed 1987; Peterson and Hammer 2001). In the Pacific Northwest, fire exclusion has resulted in a suite of structural and compositional changes in woodlands, savannas, and grasslands.

To better understand these changes, we designed a study in the Bald Hills that investigated variability in fuels and microclimate, fire intensity, and their relationships with stand structure and composition. In addition, we evaluated the effects of restoration fire on Douglas-fir saplings. In this report, we outline five questions about the Bald Hills, and we include a description of studies we undertook to answer each:

1. What is the effect of tree invasion on surface fuels?
2. What effect does vegetation structure have on wind speed and fuel moisture?
3. What are the combined effects of community structure on prescribed fire behavior?
4. What are the structure of snags and coarse woody debris in frequently burned Oregon white oak woodlands?
5. What are the effects of prescribed restoration fire on invading Douglas-fir saplings?

Despite its uniqueness, little is known about the flora or forest dynamics of the Little Bald Hills region aside from two vegetation surveys, one by Goforth and Veirs (1989) and a rare plant inventory by Smith *et al.* (2003). In addition, little is known about the management history of the site, although past fires likely affected the presence of several species (Goforth and Veirs 1989). In the Little Bald Hills, scattered large trees among the matrix of small, somewhat even-sized forests suggest substantial changes in forest structure occurred in the recent past.

Here, we report on three main questions we have addressed about the Little Bald Hills:

1. What is the rate and extent of tree encroachment into serpentine grasslands of the Little Bald Hills?
2. What are the composition and vegetation patterns in the Douglas-fir forests in the Little Bald Hills?
3. What is the stand history of the Douglas-fir forests in the Little Bald Hills?

This report is arranged with each smaller study listed separately, with the relevant research questions, the methods used, and the findings.

Research Question 1: How do fuels vary across a diverse overstory structure gradient in the Bald Hills?

Methods

The Schoolhouse Peak, Copper Creek, and Coyote Creek burn units were systematically surveyed and sites representing five structural community types including grassland ($n = 25$), oak savanna ($n = 11$), oak cluster ($n = 15$), oak woodland ($n = 15$), and invaded woodland ($n = 10$) were randomly selected for analysis (Figure 1). In addition to the five structural communities, we established eight plots in oak woodland with an understory dominated by California fescue (*Festuca californica*), a notably flammable native grass (Hastings *et al.* 1997). Herbaceous and woody fuel loading, as well as fuelbed bulk density, were sampled across the structural communities.

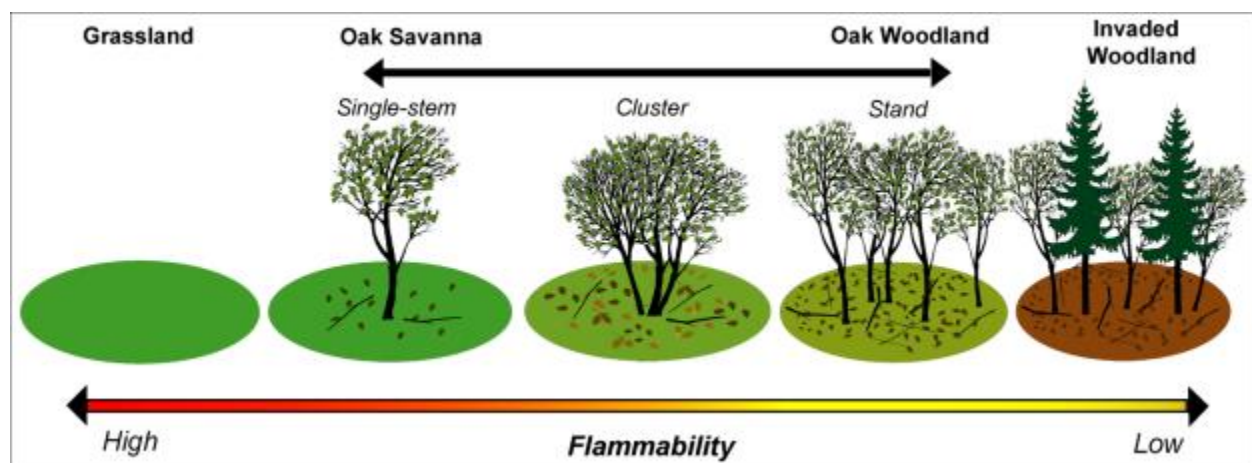


Figure 1. Overstory structural gradient in the Bald Hills oak woodlands of Redwood National Park. Increases in overstory basal area lead to replacement of herbaceous with woody fuels and a reduction in fuelbed flammability.

Fuels were sampled in three strata, including woody, leaf litter, and herbaceous (live and dead combined) fuels, over two field seasons (2008 and 2009). Schoolhouse and Copper Creek burn units were sampled in 2008, while Coyote Creek was sampled in season two (no plots were re-sampled). A plot consisted of three 25×50 cm subplots located at a random azimuth, approximately 1-2 meters from an oak stem (unless in grassland). Grassland plots were located using a random azimuth approximately 5 m from adjacent oak crowns. The three subplots were averaged at the plot level for all analyses. All herbaceous and litter mass within each subplot was removed approximately 1 cm from ground level and transported to the lab for drying and weighing. One and 10-hr (< 2.54 cm diameter) woody fuels were destructively sampled from subplots in the first sampling season, while the planar intercept method (Brown 1974) was employed to estimate 1-, 10-, 100-, and 1000-hr woody fuel loading in the second sampling season. To estimate fuelbed bulk density (kg m^{-3}), fuel depths were measured in each subplot

prior to any destructive sampling to estimate fuelbed volume ($25\text{ cm} \times 50\text{ cm}$ frame area $\times$ measured fuelbed depth). Fuelbed bulk density was calculated in two ways: herbaceous mass divided by fuelbed volume ($\rho_b h$), and a combination of herbaceous, 1 and 10-hr woody fuels, and litter mass divided by fuelbed volume ($\rho_b \text{ tot}$).

Findings

Fuelbed components varied markedly across the structural gradient from grassland to invaded woodland, with mean values differing significantly by structural community type for all fuelbed components ($P < 0.001$; Table 1, Figure 2 and 3). Mean herbaceous mass ranged from 3.38 Mg ha^{-1} in grassland, to 1.8 Mg ha^{-1} in oak woodland, to only 0.03 Mg ha^{-1} in invaded woodland; mean values for savanna and grassland plots were not significantly different. California fescue fuelbeds had the greatest herbaceous mass of any structural community type, averaging 7.76 Mg ha^{-1} , more than double the grassland plots ($P < 0.001$). With few woody fuels and deep fuelbeds, grassland had the lowest values for total fuelbed bulk density ($\rho_b \text{ tot} = 5.44\text{ kg m}^{-3}$), and greatest values for herbaceous fuelbed bulk density ($\rho_b h = 4.76\text{ kg m}^{-3}$) (Figure 2). Similar to other fuelbed components, bulk density did not differ between grassland and savanna communities. Total fuelbed bulk density increased substantially along the gradient from grassland to invaded woodland ($P < 0.001$), with invaded woodland ($\rho_b \text{ tot} = 142.4\text{ kg m}^{-3}$) having considerably greater values than all other structural communities.

Fuelbed characteristics typical of grasslands and lower density oak woodlands (low bulk density and heavy herbaceous mass) are inverted with Douglas-fir invasion in our study area, which results in woody fuel recruitment, reduced herbaceous mass, and ultimately, lower community flammability. By maintaining fuelbeds that are very receptive to fire (i.e., niche construction), grassland fuel structures favor higher-frequency fire, which may serve to limit the recruitment of invading woody species into mature size classes. Fuelbed differences across the oak structural communities (savanna, cluster, woodland) were more subtle than those differentiating grassland from invaded woodland, but still have important implications for fire behavior. Grassland and savanna communities did not differ significantly for several fuelbed metrics, sharing high herbaceous mass and low fuelbed bulk density, suggesting both are fire-facilitating, flammable structural communities. The fact that these communities occur adjacent to each other is intriguing, and underscores the importance of fire in shaping boundaries between the two.

Table 1. Descriptive statistics for measured fuelbed metrics by structural community type in the Bald Hills of Redwood National Park. Means followed by the same letter (e.g., ABC) are not significantly different based on a Tukey-Kramer multiple comparison test.

Fuel Stratum	Structural Community	N	Mean (Mg ha ⁻¹)	SE	CV (%)	Min	Max
Herbaceous	<i>Grassland</i>	25	3.38 A	0.19	27.9	1.49	5.19
	<i>Savanna</i>	11	2.63 AB	0.19	25.08	1.42	3.82
	<i>Cluster</i>	15	2.14 BC	0.18	33.37	1.13	4.02
	<i>Woodland</i>	15	1.80 C	0.12	25.55	1.27	2.77
	<i>Invaded</i>	10	0.03	0.03	298.44	0.00	0.31
	<i>Fescue</i>	8	7.67	0.75	27.52	4.19	9.79
Litter	<i>Grassland</i>	25	0.21 A	0.06	135.97	0.00	1.23
	<i>Savanna</i>	11	0.43 AB	0.07	49.94	0.15	0.94
	<i>Cluster</i>	15	0.61 B	0.15	94.43	0.08	1.64
	<i>Woodland</i>	15	1.82	0.20	42.93	0.47	3.34
	<i>Invaded</i>	10	3.42	0.29	27.30	2.06	4.64
1 Hour	<i>Grassland</i>	25	0.01	0.005	218.27	0.00	0.10
	<i>Savanna</i>	11	0.37 A	0.06	58.1	0.12	0.79
	<i>Cluster</i>	15	0.36 A	0.03	33.1	0.17	0.56
	<i>Woodland</i>	15	0.56 AB	0.09	63.1	0.11	1.34
	<i>Invaded</i>	10	1.68 B	0.33	62.4	0.57	3.40
10 Hour	<i>Grassland</i>	25	0.00	-	-	-	-
	<i>Savanna</i>	11	0.61 A	0.17	90.33	0.01	1.58
	<i>Cluster</i>	15	1.06 AB	0.17	63.81	0.21	2.41
	<i>Woodland</i>	15	1.31 B	0.15	43.38	0.35	2.19
	<i>Invaded</i>	10	2.49	0.34	42.83	0.80	4.56
100 Hour	<i>Grassland</i>	10	0.00 A	-	-	-	-
	<i>Savanna</i>	6	1.67 AB	0.37	53.69	0.84	2.48
	<i>Cluster</i>	10	2.43 B	0.61	79.66	0.00	6.62
	<i>Woodland</i>	10	1.77 B	0.39	69.45	0.00	4.22
	<i>Invaded</i>	5	1.84 AB	0.77	93.05	0.00	3.38
1000 Hour Sound	<i>Grassland</i>	10	0.00 A	-	-	-	-
	<i>Savanna</i>	6	0.00 AB	-	-	-	-
	<i>Cluster</i>	10	0.67 AB	0.67	316.23	0.00	6.75
	<i>Woodland</i>	10	3.10 B	1.29	131.89	0.00	12.80
	<i>Invaded</i>	5	4.39 B	2.82	143.64	0.00	14.60
1000 Hour Rotten	<i>Grassland</i>	10	0.00 A	-	-	-	-
	<i>Savanna</i>	6	0.00 A	-	-	-	-
	<i>Woodland</i>	10	1.24 AB	0.66	171.61	0.00	6.31
	<i>Invaded</i>	5	11.9B	5.15	96.10	0.90	26.86

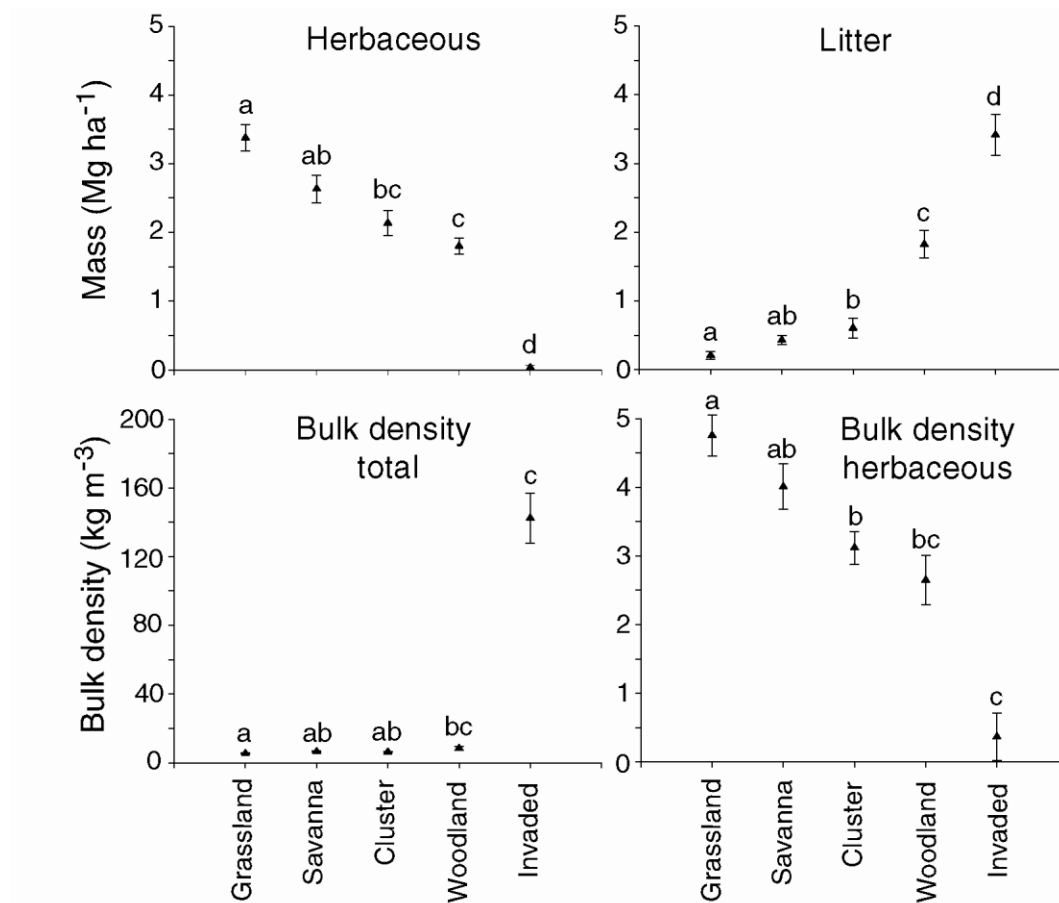


Figure 2. Mean herbaceous fuel loading, litter fuel loading, and fuelbed bulk density (total and herbaceous) across the overstory structural gradient in the Bald Hills of Redwood National Park. Bars represent the +1.0 SE, while means followed by the same letter are not significantly different.

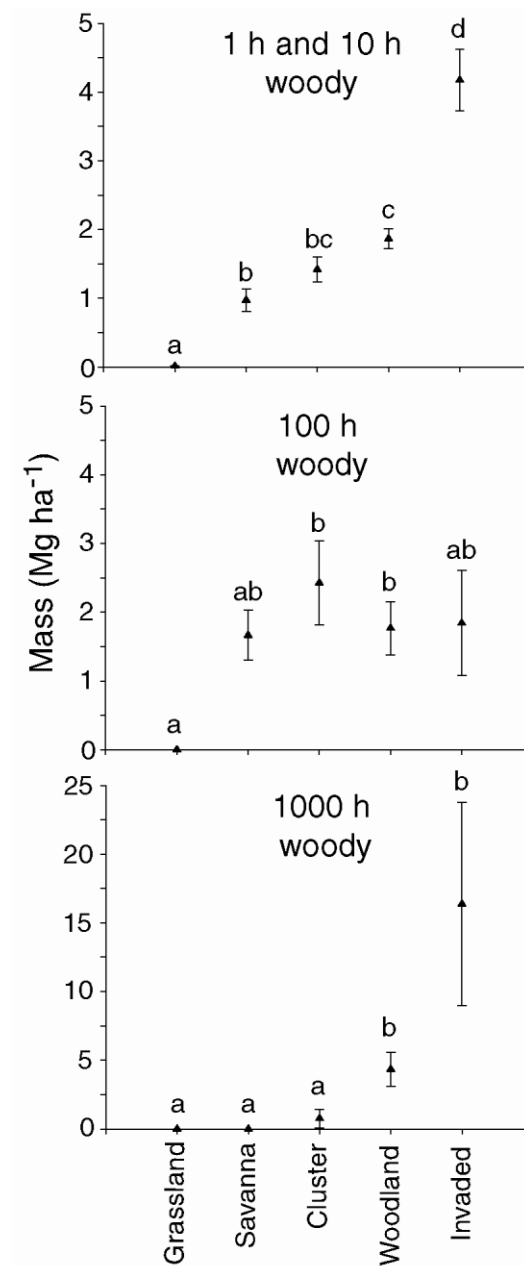


Figure 3. Mean downed woody fuel loading in the 1, 10, 100, and 1000 hour time-lag categories across the overstory structural gradient in the Bald Hills of Redwood National Park. Bars represent the standard error while means followed by the same letter are not significantly different.

Research Question 2: *How do fuel moisture and wind speed vary across a diverse overstory structure gradient in the Bald Hills?*

Methods

Herbaceous and 10-hr woody fuel moisture were sampled across the five structural communities described above ($n = 5$ replicates of each) during Summer/Fall 2008. Herbaceous moisture content was estimated by clipping a 20-35 gram (oven-dry weight) sample at ground level on the south side of tree stems, beneath the canopy (except in grassland where no stems were present). Herbaceous moisture was not sampled in invaded woodland since these sites harbored an insufficient quantity of herbaceous mass. Sampling occurred on two dates (29 August 2008 and 20 September 2008) between 1200 and 1500 hr to minimize variations in environmental conditions. Woody fuel moisture was estimated across all five structural communities with standard 10-hr fuel moisture sticks ($n = 5$ replicates of each, $N = 25$). Sampling occurred on two dates (16 September 2008 and 22 September 2008) between 1200 and 1500 hr. To characterize general differences in microclimate (wind speed, relative humidity, and temperature) across grassland, oak cluster, oak woodland, and invaded woodland, Kestrel data-logging weather stations (pocket weather tracker 4500, Neilsen-Kellerman, Boothwyn, PA, USA) were deployed for six days in late October 2009 (Figure 4). Weather data were recorded on 30-minute time intervals, and values from 1200 to 1600 hr were averaged for reporting (Figure 5).

Findings

Herbaceous fuel moisture was significantly lower in grassland in comparison to oak-dominated plots on both dates ($P = 0.005$ and $P = 0.01$, respectively), although significant differences were not found among savanna, cluster, and woodland (Table 2). Ten-hr woody fuel moisture on 16 September 2008 ranged from 5.0 % in grassland to 6.0 % in invaded woodland, and differed significantly by structural type ($p = 0.02$). Fuel moisture within invaded woodlands did not differ significantly from clusters and woodlands, but was significantly greater than grassland and savanna. Following 3 mm of rain, 10-hr fuel moisture on 22 September 2008 ranged from 11.0 % in grassland to 12.2% in invaded woodland ($p = 0.01$); differences on that date were found only between savanna and invaded woodland (Table 2). Data from weather stations suggest Douglas-fir invaded woodlands have lower and less variable wind speeds, lower temperature, and higher relative humidity in comparison to oak woodlands and grasslands (Figure 5).

Trends in live and 10-hr fuel moisture, temperature, relative humidity, and winds have important implications for fire behavior; results suggest grasslands and lower density oak woodlands (oak savanna and cluster) are the most flammable of the structures studied. Douglas-fir invasion dampens ecosystem flammability through increases in fuel moisture, congruent with findings from our fuelbed sampling.



Figure 4. Kestrel weather station assembly at base of *Quercus garryana* cluster in the Bald Hills of Redwood National Park (E. Engber Photograph).

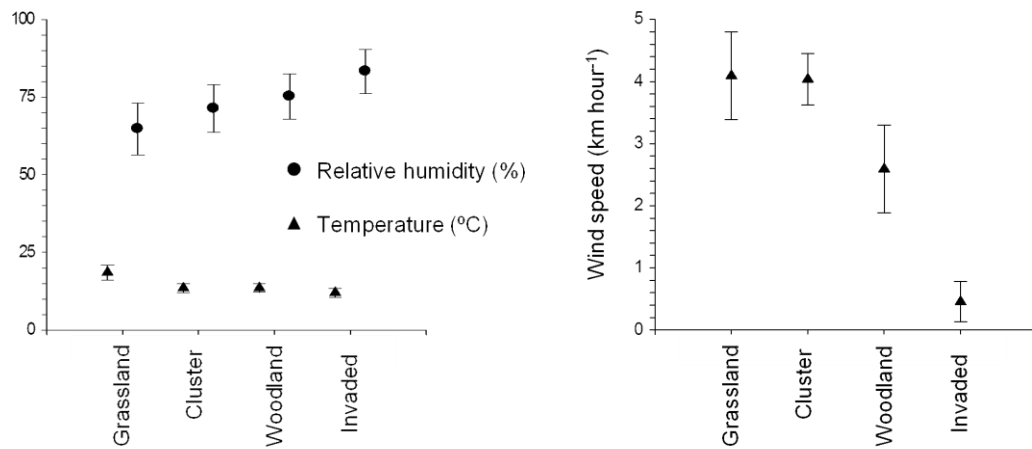


Figure 5. Mean afternoon (1200-1600) air temperature, relative humidity, and wind speed at the study site (6 day period), October 2009. Bars represent +1.0 SE.

Table 2. Live herbaceous and 10-hr woody fuel moisture across structural community types on four dates during the 2008 fire season in the Bald Hills of Redwood National Park. Means followed by the same letter (e.g., ABC) are not significantly different based on a Tukey-Kramer multiple comparison test. Values in parentheses represent 1.0 SE.

Date	Overstory Structure	N	Mean (%)	CV (%)	Range
Live Herbaceous Fuel Moisture					
8-29-08	<i>Grassland</i>	5	33.25 (4.45) A	29.94	16.79 – 43.73
	<i>Savanna</i>	5	51.87 (6.08) AB	26.20	33.63 – 63.87
	<i>Cluster</i>	5	63.36 (7.74) B	27.30	42.77 – 89.12
	<i>Woodland</i>	5	70.23 (6.94) B	22.09	52.96 – 94.65
9-20-08	<i>Grassland</i>	5	36.37 (4.09) A	25.14	27.97 – 51.96
	<i>Savanna</i>	5	43.40 (4.10) AB	21.05	30.80 – 56.42
	<i>Cluster</i>	5	53.28 (3.63) B	15.25	48.04 – 67.70
	<i>Woodland</i>	5	52.40 (1.83) B	7.82	45.70 – 56.36
10-hr Woody Fuel Moisture					
9-16-08	<i>Grassland</i>	5	4.96 (0.16) A	7.62	4.37 – 5.32
	<i>Savanna</i>	5	5.04 (0.29) A	12.68	4.12 – 5.72
	<i>Cluster</i>	5	5.19 (0.23) AB	9.87	4.51 – 5.76
	<i>Woodland</i>	5	5.45 (0.18) AB	7.47	4.87 – 5.91
	<i>Invaded</i>	5	5.98 (0.21) B	7.90	5.392 – 6.44
9-22-08	<i>Grassland</i>	5	10.95 (0.57) A	11.73	9.19 – 12.78
	<i>Savanna</i>	5	10.37 (0.17) AB	3.82	9.85 – 10.94
	<i>Cluster</i>	5	10.81 (0.29) AB	6.12	9.89 – 11.76
	<i>Woodland</i>	5	10.95 (0.22) AB	4.60	10.15 – 11.46
	<i>Invaded</i>	5	12.16 (0.16) B	2.86	11.65 – 12.53

Research Question 3: *How does fire behavior change across a diverse overstory structure gradient in the Bald Hills?*

Methods

Heterogeneity in fire intensity across plant communities is a potential mechanism influencing community structure and species demography in grasslands, woodlands, and forests (Hoffman 1999). Fire temperatures obtained from thermocouples and pyrometers have been used as surrogates for fire intensity. We constructed metal pyrometers with 13 Omegalaq temperature-sensitive paints (Omega Engineering Inc., Stamford, CT, USA) (ranging from 79 to 649° C) applied to thin copper tags (15 cm × 2.5 cm × 0.255 mm thick) in adjacent non-overlapping strips to minimize error. Pyrometers were wired to metal conduits and erected 30 cm above ground, with four per plot (Figure 6). The four pyrometers were averaged at the plot level for all analyses. Manufacturer's protocols were followed to assess melting of paints, using two observers.

Prescribed burns were conducted by Redwood National Park managers in fall of 2008 (Table 3). Prescribed fires in the Bald Hills are generally conducted in the fall due to the sensitivity of the California oatgrass (*Danthonia californica*) to spring burning (Arguello 1994). The Coyote Creek unit was scheduled for burning in fall 2009, but the burn was not accomplished because prescription weather was not met. Additionally, not all plots within burned units burned; these were removed from analysis, leaving a total of 112 pyrometers across 28 plots (see Table 2 for number of replicates per community type).

Findings

Results revealed a decreasing trend in fire temperatures across the gradient from grassland (207.9 ±19 °C) to savanna (202.1±62.9 °C), to cluster (184.9±29.3 °C), to woodland (190.4±31.5 °C), to invaded woodland (74.7±19.6 °C) (Table 4), however, due to limited replication and within-treatment variation in measured temperature (CV = 33-54 %), differences among structural community types were not significant ($F_{4, 23} = 1.53$, $P = 0.23$) (Table 4.). We did find a positive, significant correlation between fire temperature and herbaceous fuel mass ($r = 0.52$, $P = 0.004$), as well as a negative correlation between fire temperature and 1- and 10-hr woody fuel mass ($r = -0.47$, $P = 0.01$) (Figure 7). Additional variation in measured fire temperature could not be explained by including more explanatory variables in a multiple regression model (e.g., live fuel moisture, 10-hr fuel moisture, fuelbed bulk density, or slope). Variability in ignition pattern, distance between ignition points and plots, and local winds likely had a strong influence on fire behavior; these factors present a challenge for fire research in management-scale prescribed burns.

Table 3. Prescribed fire conditions and behavior in the Bald Hills of Redwood National Park, 2008-2009. FL = flame length; Temp. = ambient air temperature; RH = relative humidity.

Burn Unit	Ignition Type	Grassland FL (m)	Oak woodland FL (m)	Wind (gusts) km/hr	Temp. (°C)	RH %	Date Burned
Schoolhouse Peak	Strip and spot head fires	0.9 - 3.0	0.6 - 1.5	3 - 8 (22)	20 - 23	31-71	09-23-2008
Copper Creek	Strip and spot head fires	0.9 - 1.5	0.6 - 0.9	0 - 8 (16)	19 - 23	28-55	10-27-2008

Table 4. Descriptive statistics for pyrometer temperature across structural communities in the Bald Hills of Redwood National Park. Means followed by the same letter (e.g., ABC) are not significantly different based on a Tukey-Kramer multiple comparison test. Values in parenthesis following the mean represent the standard error. "N" refers to the number of plots per community type, while "n" refers to the number of pyrometers per community type.

Structure	N (n)	Mean (SE) °C	CV (%)	Range
<i>Grassland</i>	13 (52)	207.9 (19.0) A	33.02	109.2 - 350.2
<i>Savanna</i>	3 (12)	202.1 (62.9) A	53.92	79.4 - 287.7
<i>Cluster</i>	5 (20)	184.9 (29.3) A	35.41	94.3 - 273.9
<i>Woodland</i>	5 (20)	190.4 (31.5) A	36.97	100.3 - 287.7
<i>Invaded</i>	2 (8)	74.7 (19.6) A	37.06	55.1 - 94.3



Figure 6. Pyrometer assembly at base of single-stem oak in the Bald Hills of Redwood National Park (E. Engber Photograph).

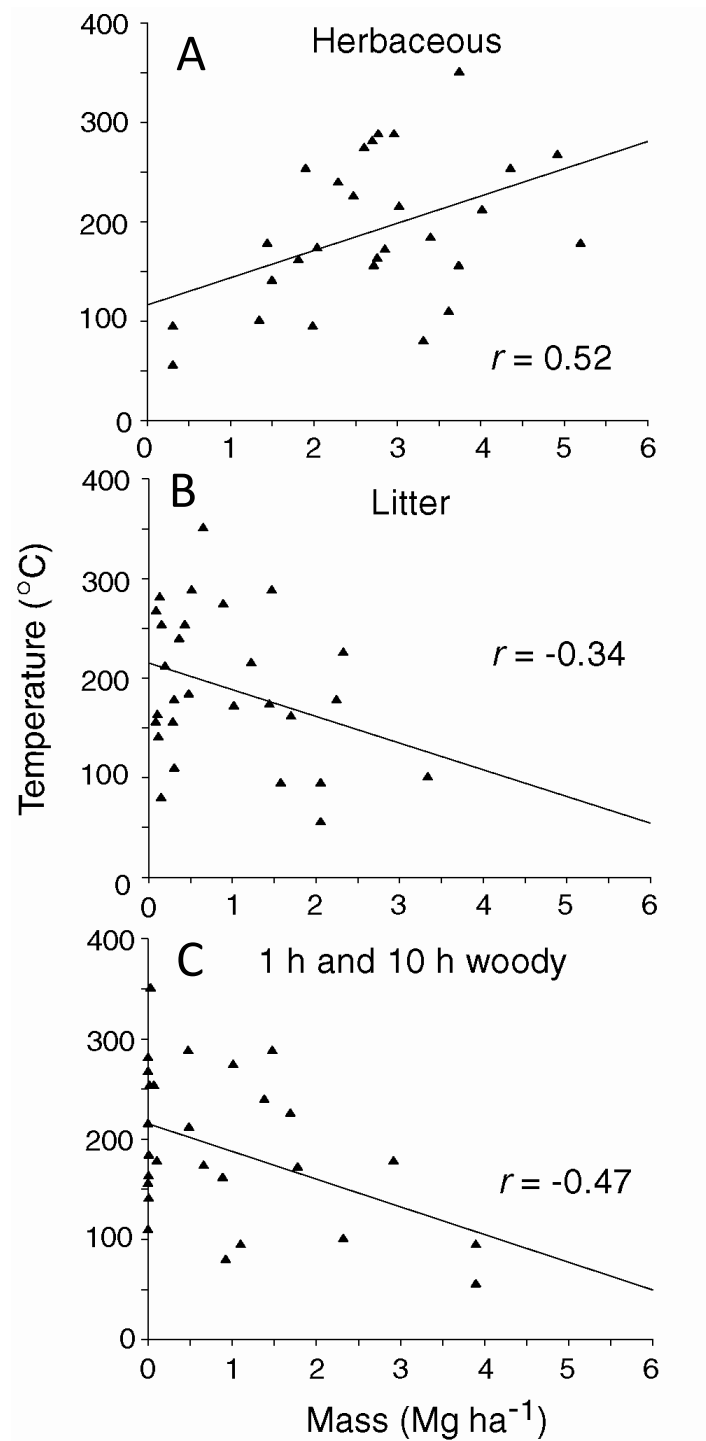


Figure 7. Scatter plots of mean fire temperature (°C) on (A) herbaceous, (B) litter, and (C) fine woody (one and ten-hr) mass (Mg ha⁻¹) from two prescribed burns in the Bald Hills of Redwood National Park, CA, USA 2008.

Research Question 4: *What are the determinants of invading Douglas-fir sapling mortality in prescribed fires in the Bald Hills?*

Methods

The mechanisms of small (e.g., seedling and sapling) Douglas-fir mortality following restoration burns are little understood. Oddly, the voluminous work on fire-induced mortality mostly omits seedlings and saplings even though this size class is among the most prevalent in woodlands and grasslands in the region and a common restoration target given the relatively low fire intensity required to injure trees. While the limited research on young Douglas-fir mortality following fire has focused coarsely on tree size (Gruell et al. 1986, Tveten and Fonda 1999, Regan and Agee 2004), research on larger Douglas-fir has addressed the underlying mechanisms of mortality by including fire-caused injury in mortality models (Ryan and Reinhardt 1988, Hood et al. 2008). The goal of our study was to clarify the relative importance of tree size and fire-induced injury in the post-fire mortality of Douglas-fir seedlings and saplings (< 3 m tall). We also examined patterns of delayed mortality (from ca. 0.5 to 1.5 years post burn), as well as the relationship between crown and bole (i.e., cambium) injury. Results were intended to provide managers with increased knowledge of injury thresholds required for Douglas-fir sapling mortality, and contrast sapling responses to fire with patterns observed in larger Douglas-fir.

Research was carried out in the 40 ha Eastside burn unit within the Bald Hills of Redwood National Park. The Eastside burn unit is positioned on an upper slope at an elevation of ca. 865 m, a mostly NE aspect, and average slope of 20.3 ($\pm$ 7.0) percent. Redwood National Park completed a manual conifer removal in 2002; subsequent Douglas-fir recruitment post-restoration (Figure 8) illustrates the rapidity with which trees invade grasslands and woodlands in the Bald Hills. A first-entry prescribed burn was conducted on 1 October 2009, using a combination of strip and spot head and back fires. Fire behavior ranged from heading (1 to 2 m flame lengths), to flanking (0.3 to 1.2 m flame lengths), to backing (0.3 to 1 m flame lengths), though heading and flanking were most common. Wind speed ranged from 1.5 to 8 km hr⁻¹ (gusts to 16 km hr⁻¹) while relative humidity ranged from 55 to 60 percent. Fuel moisture samples were collected approximately one hour prior to ignition, and ranged from 16.4 ($\pm$ 1.7) percent in cured grass, to 143.1 ($\pm$ 13.9) percent in the shrub layer (primarily *Rubus* spp.), to 134.9 ($\pm$ 10.9) percent in ferns (exclusively *Pteridium aquilinum*). Surface fuel loading was estimated (by clipped plots) to be 5.55 ($\pm$ 2.2) Mg ha⁻¹, with 81.3 percent of the mass comprised of herbaceous fuels, 11.7 percent shrubs (*Rubus* spp.), and 7.0 percent ferns (*Pteridium aquilinum*).

One hundred Douglas-fir seedlings and saplings (ca. 0.5 to 3 m tall) were selected for monitoring in two areas of the unit with heavy conifer invasion. On all trees we measured diameter at ground level (DAG), crown width, and height; all were strongly correlated ($0.72 < R^2 < 0.75$). To estimate bark thickness, 19 Douglas-fir seedlings and saplings ranging from 0.3 to 2.75 m tall were cut at ground level to establish a relationship between bark thickness and DAG. Bark thickness was measured at two points, 90° apart, and least-squares linear regression was conducted to model bark thickness as a function of diameter, where:

$$\log_{10} \text{ bark thickness (mm)} = 0.2486 + 0.0063 (\text{DAG mm}),$$

with DAG explaining 80 percent of the variation in bark thickness.

Approximately three weeks following the prescribed burn, two observers estimated percent crown volume scorched (PCVS; brown and blackened foliage and buds) and percent crown volume consumed (PCVC; only blackened/charred foliage and buds; Figure 9); values from the two observers were averaged for analysis. Twenty of the burned trees spanning a range of crown consumption were randomly selected for removal and were transported to the lab for measurement of bark scorch and cambium injury (these trees were not included in mortality modeling), two important mortality predictors for Douglas-fir and other conifers (Hood *et al.* 2008) (Figure 10). Bark scorch was assessed in two ways: *i.* height of 100% scorch (i.e., scorch fully surrounding bole; BS100); and *ii.* maximum height of discontinuous scorch (BSmax). Bark scorch was clearly distinguishable based on brown discoloration. To assess cambium status on the twenty trees, bark was peeled away from boles to reveal underlying cambium; white or pinkish cambium was assumed living while brown and dry cambium was assumed to be dead (Wagener 1961) (Figure 10). As with bark scorch, height of 100% cambium injury (CI100) as well as discontinuous cambium injury (CImax) were measured separately. Tree mortality was assessed in the field approximately 5.5, 9, and 20 months following the burn. Trees were tallied as living if any green foliage or living buds were found.

Simple linear regression analyses were conducted to assess relationships between crown injury and tree size, and between crown and bole injury. Two-sample *t*-tests were conducted to compare pre-fire tree size and crown injury across dead and living saplings, for the final mortality survey (20 months post burn). Binary logistic regression was conducted to assess the effects of crown injury (PCVS and PCVC) on mortality probability for the final mortality assessment (20 months post-burn) based on 78 trees (20 were removed for cambium assessment and two could not be relocated in the field post burn). No pre-fire tree size variables were included in models because these variables did not differ significantly among dead and living trees at 20 months post-burn (all $P > 0.90$). The predictors PCVS and PCVC were modeled separately because they are not independent ($r = 0.56$). Lastly, logistic modeling equations were developed for each of the three assessment dates based on PCVS and were compared with a published logistic equation for larger Douglas-fir mortality (Ryan and Reinhardt 1988).

Findings

Following the prescribed burn, overall cumulative mortality levels were 47, 74, and 94% at 5.5, 9, and 20 months post-burn, respectively. Comparatively, Ryan and Reinhardt's (1988) logistic mortality model for larger Douglas-fir predicted 99 percent mortality, based on observed PCVS and modeled bark thickness (Figure 11). No significant differences in tree size (height or DAG) were found between dead and living trees 20 months post burn ($p > 0.92$), though small trees died more rapidly (Figure 12). Crown injury (PCVS and PCVC), however, differed significantly between dead and living saplings ($p < 0.002$), with dead saplings suffering ca. 40 to 50 percent greater crown injury (Figure 12). The logistic regression analyses revealed that PCVS and PCVC

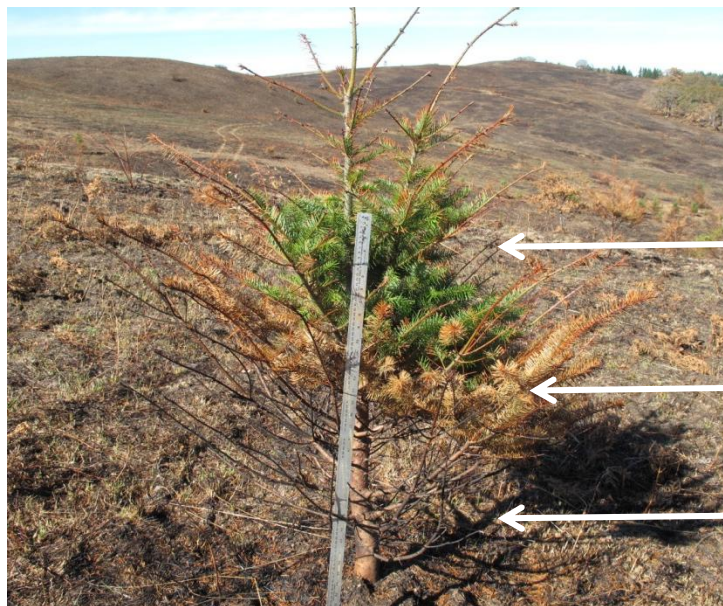
were both strong mortality predictors for small Douglas-fir, with the change in deviance being slightly larger for PCVC (Table 5). If 50 percent mortality probability is taken as a cutoff for tree death (dotted horizontal line in Figure 11), results suggest that levels of PCVS above 20 percent are sufficient to kill trees, while any measurable crown consumption results in mortality probabilities above 50 percent.

Measures of cambium injury (CI_{max} and CI₁₀₀) and bark scorch (BS_{max} and BS₁₀₀) were moderately associated with PCVS ($0.62 \leq r \leq 0.73$) and more strongly associated with PCVC ($0.66 \leq r \leq 0.94$) (Table 6). Maximum height of bark scorch was also strongly correlated with CI_{max} ($r = 0.98$). The strong correlations between crown and cambium injury suggests only one of these variables may need to be measured in the field for Douglas-fir saplings (to account for the variation in tree injury), though Battaglia *et al.* (2009) found tree size, crown injury, and bole char to improve mortality predictions for ponderosa pine (*Pinus ponderosa*) seedlings and saplings.

Sapling injury and mortality following fire is understudied, but it is a relevant topic that warrants further research (Battaglia *et al.* 2009), particularly in areas where managers are using prescribed fire to restore conifer-encroached woodlands and grasslands. This study highlights the importance of crown injury in post-fire mortality, and also suggests the value of assessing cambium injury. Cambium injury appears to be strongly correlated with bark scorch on small Douglas-fir and could therefore be easily monitored in the field following restoration burns. As shown here, mortality assessments conducted too soon (e.g., less than 20 months post-burn) will likely under-predict overall mortality. The discrepancy in injury tolerance between small and large size classes of Douglas-fir is provocative and suggests a direction for future research in post-fire mortality studies.



Figure 8. Douglas-fir invasion into a grassland/oak woodland at the study site, Eastside burn unit, Bald Hills, Redwood National Park (April, 2009). All young Douglas-fir invaded following a Douglas-fir removal project in 2002 (E. Engber Photograph).



Living

Scorched

Consumed

Figure 9. Douglas-fir sapling illustrating crown injury following a prescribed burn in the Bald Hills of Redwood National Park. Notice distinction between crown consumption (lower 3rd of crown), crown scorch (middle 3rd), and living crown above (E. Engber Photograph).

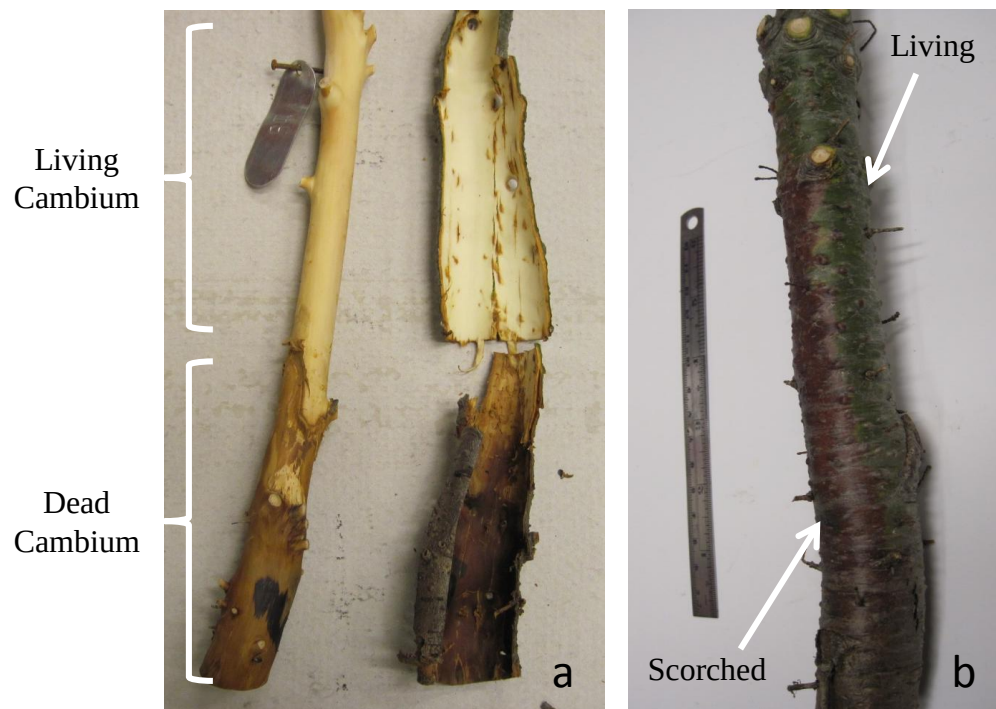


Figure 10.(a) Peeled Douglas-fir sapling depicting areas of cambium injury (brown discoloration, below) and living cambium (white, above). (b) Fire-damaged Douglas-fir bole, illustrating scorched bark (brown) and living bark (green) (E. Engber Photograph).

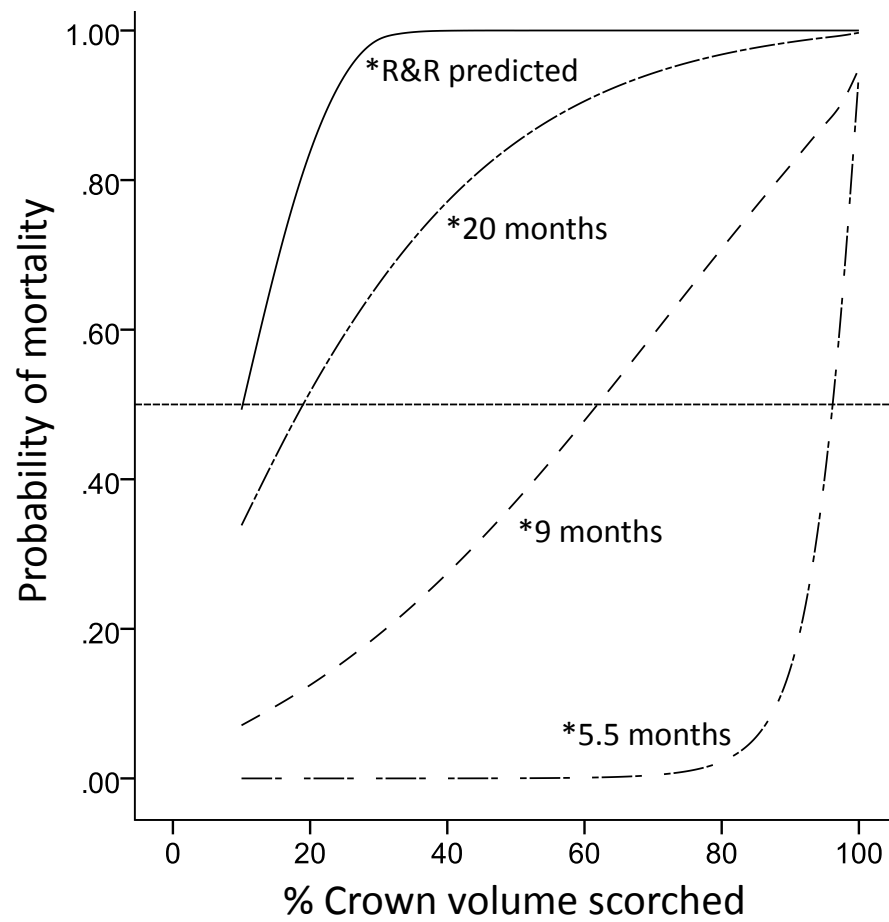


Figure 11. Probability of mortality for Douglas-fir seedlings and saplings in the Eastside prescribed burn, by percent crown volume scorched and months post burn. Also depicted is a mortality model from Ryan and Reinhardt (1988) developed for larger Douglas-fir, using observations from the present study (labeled “R&R predicted”).

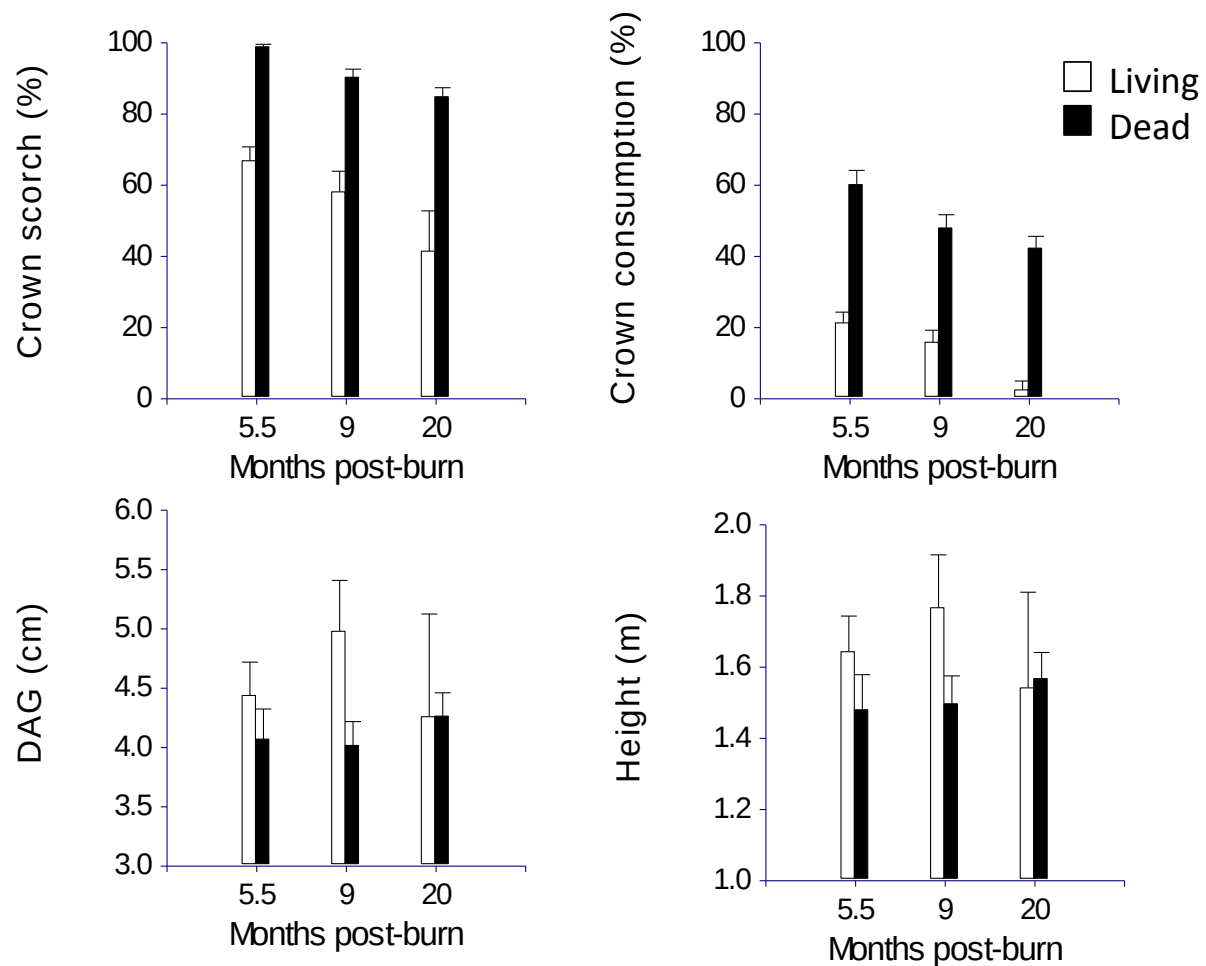


Figure 12. Crown injury (percent crown volume scorched and consumed) and tree size (diameter at ground level and height) for living and dead Douglas-fir seedlings and samplings assessed at 5.5, 9, and 20 months following a prescribed burn. Bars represent the standard error.

Table 5. Logistic regression mortality parameters (β), change in deviance when term is added to model (Dev (χ^2)), p-values for term significance (p), Wald Z value, and ROC value. PCVS = percent crown volume scorched, while PCVC = percent crown volume consumed; both were arcsine-square root transformed prior to analysis.

Model	Parameter			Dev (χ^2)	P	Wald Z	P	ROC
	β_0	β_1	X_1					
1	-2.3419	5.1938	PCVS	12.49	<0.001	2.62	0.008	0.767
2	0.5960	6.9354	PCVC	14.23	<0.001	2.35	0.018	0.761

Table 6. Linear regression equations, Pearson's correlation coefficients (r), and p-values from regression analyses of crown and bole injury variables. Response variables included maximum height of cambium injury expressed as a percent of total tree height (CI_{max}), height of 100 percent cambium injury expressed as a percent of total tree height (CI₁₀₀), maximum height of bark scorch expressed as a percent of total tree height (BS_{max}) and height of 100 percent bark scorch expressed as a percent of total tree height (BS₁₀₀). Predictor variables included percent crown volume scorched (PCVS; arcsine-square root transformed) and percent crown volume consumed (PCVC; arcsine-square root transformed). † refers to Spearman's Rank correlation coefficients (r) where parametric analyses were not appropriate.

Response (y)	Predictor (x)					
	PCVS			PCVC		
	equation	r	P	equation	r	P
CI _{max}	--	0.67†	--	$y = 3.1656 + 5.5081(x)$	0.94	<0.001
CI ₁₀₀	$y = 0.7901 + 3.1652(x)$	0.65	0.002	$y = 2.8009 + 2.8528(x)$	0.66	0.001
BS _{max}	--	0.62†	--	$y = 3.8516 + 4.9620(x)$	0.90	<0.001
BS ₁₀₀	--	0.73†	--	$y = 2.5317 + 5.6989(x)$	0.92	<0.001

Research Question 5: *What is the structure of snags and coarse woody debris in frequently burned Bald Hills oak woodlands?*

Methods

A systematic sample of twenty 0.08 ha plots were installed within the Schoolhouse Peak burn unit in the Bald Hills of Redwood National Park (Figure 13) to assess snag and coarse woody debris abundance and characteristics. The following were measured: live overstory basal area ($\text{m}^2 \text{ha}^{-1}$); snag density, diameter at breast height (dbh), and decay class (1-5); and coarse woody fuel loading (1000-hour: $> 7.62 \text{ cm}$; planar intercept method) and decay status (See table 7 below). Descriptive statistics and simple linear regressions between overstory structural variables and fuel loading are reported.

Table 7. Decay categories used for snag and CWD decay classification in the study.

Decay Classes	CWD Decay	Snag Decay
Class 1	Bark firmly attached. Exposed wood fresh, unstained.	Bark firmly attached, fine branches present.
Class 2	Bark loosened if present. Surface firm, not flaky. May be bleached.	Bark loosened, mostly present. Few fine branches, sound wood.
Class 3	Bark generally absent. Wood firm but surface flaky and soft.	Bark discontinuous if present. No fine branches, surface softening, but generally sound wood.
Class 4	No longer firm, may have some solid chunks. Log oval or flattened, with spongy wood.	No bark, snapped top likely. Holes in stem, rotten areas present, soft surface.
Class 5	Predominately powder wood. Flattened.	Widow maker. Barely standing, rotten.

Findings

Mean live tree basal area was $13.9 \text{ m}^2 \text{ha}^{-1}$ and was positively correlated with snag basal area (mean = $2.8 \text{ m}^2 \text{ha}^{-1}$; $R^2 = 0.21$, $P = 0.04$), although variation was high in both (coefficient of variation (CV) = 63 and 90%, respectively). Snags were abundant (mean snag density = $95.9 \text{ snags ha}^{-1}$) but heterogeneous in distribution, ranging from 0-284 snags ha^{-1} among plots. Snag diameters averaged 19.6 cm and ranged from 7.6-48.5 cm (standard error = 1.27 cm, CV = 0.28) (Figure 14) Mean snag diameter was negatively related to live basal area ($R^2 = 0.37$, $p = 0.007$). 51.0% of snags were categorized in decay class 2 (loosened bark, sound wood), while 40.0% fell within decay class 3 (little bark, surface softening); only 5.2% of snags were categorized in decay classes 4 and 5 (soft, spongy wood) (Figure 15).

Coarse woody fuel loading was variable across plots and consisted of very little highly-decayed wood. Sound and rotten 1000 hour fuel loading (> 7.6 cm) was 8.4 and 15.2 mg ha^{-1} , respectively, with high variation in both ($\text{CV} = 131.9$ and 408.5% , respectively). Most coarse woody fuels fell within decay classes 2 and 3 (35% and 40% , respectively) (Figure 16). Coarse woody fuel loading greater than 12.7 cm (diameter) was almost double that in a large-scale California oak woodland study in stands with similar live tree basal area. No relationships were found between coarse woody fuel loading and overstory structure ($P > 0.05$).

Significant relationships between snag abundance and size, and live basal area suggests stand density is an important factor influencing snag abundance; snag recruitment in higher density stands may be increased by susceptibility of small trees to injury. Decades of fire exclusion allowed surface fuels, snags, and CWD to persist until the prescribed fire program was initiated in the park; the abundance of these features may represent a relict condition that may be altered with future burning. The lack of significant relationships between overstory structure and CWD likely resulted from high variability among plots in both live and dead basal area and fuels; a larger sample size could clarify these relationships. The general lack of decayed wood in this ecosystem suggests the significance of fire in manipulating the character of dead wood in frequently burned ecosystems. Future work should focus on monitoring snag and CWD abundance over time to illuminate the role of frequent fire in recruitment and retention of these features.



Figure 13. Oregon white oak woodland at the study site (E. Engber, Photograph).

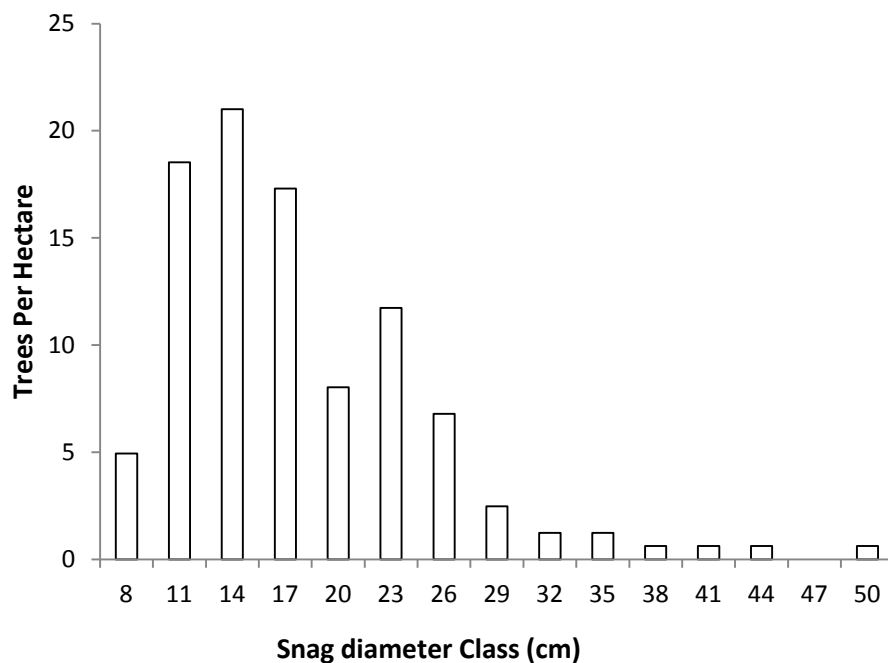


Figure 14. Snag diameter distribution in Bald Hills oak woodlands in Redwood National Park.

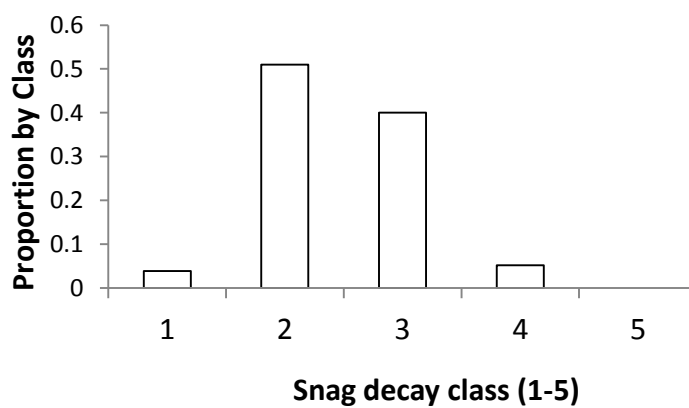


Figure 15. Snag decay status in Bald Hills oak woodlands in Redwood National Park.

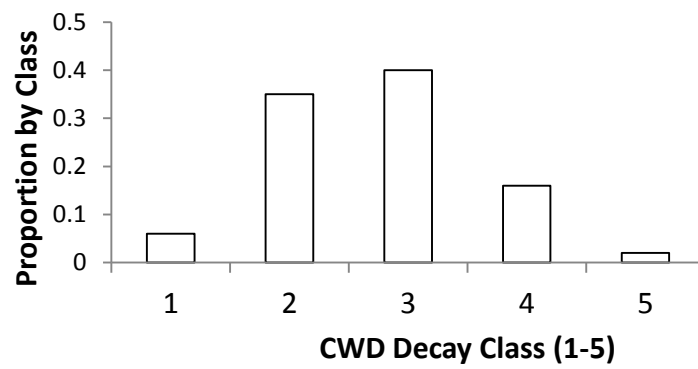


Figure 16. Coarse woody debris decay status by decay class in Bald Hills oak woodlands.

Research Question 6: *What were historical stand conditions in Little Bald Hills?*

Methods

During the summer and fall of 2009, we set up 29 (0.05 ha) circular plots in the Little Bald Hills (LBH) region of Redwood National and State Parks (RNSP). Random plot locations were generated using ArcGIS 9.3.1 (ESRI, Redlands, CA, USA) within an area defined by a combination of a Jeffrey pine savanna GIS layer produced by RNSP and a hand-digitized polygon of Jeffrey pine savanna based on visual assessment of a 1942 aerial photo.

Diameter at breast height (DBH) was recorded for all trees greater than 7 cm DBH, including snags. In all plots, trees less than 7 cm DBH, hereafter juveniles, were tallied by species. A maximum of ten juveniles of each species were cut at ground level in order to determine accurate juvenile establishment year. In 28 of the 29 plots, every tree greater than 7 cm was sampled using an increment borer as close to the base of the tree as possible. One plot had > 200 trees, and trees within this plot were subdivided into 10 cm size classes. Ten trees per size class were haphazardly selected for coring and establishment ages in this plot for uncored trees were assigned proportionally based on the total number of trees establishing across all plots in a given size class.

In order to determine what historical stand conditions were in LBH over as long a period as possible, tree establishment dates and historical sizes were reconstructed using collected tree cores and basal cross-sections. Tree cores and cross-sections were sanded using progressively finer sandpaper, from 120 to 1500 grit, until each ring's structure was clearly visible under a dissecting microscope. All samples were visually cross-dated and the program COFECHA was used as a cross-dating quality control (Holmes 1983). Annual growth rings were measured to 0.001 mm using a Velmex measuring stage (TA 4030H1-S6 Unislide, Bloomfield, New York, USA). For cores that did not intersect the pith, ring-pattern templates of variously sized concentric circles were used to estimate the number of missing rings.

To correct for sampling height, ten saplings of each species were removed from the ground with root material intact and subsequently sanded until the root-shoot interface was visible. Rings were counted at 10 cm intervals from the root-shoot interface, and the mean number of rings present at coring height was added to the earliest visible ring year. Past basal area was calculated using cross-dated raw ring width radial measurements, multiplied by two to get diameter at a given year. A regression-derived bark correction ($\text{bark thickness} = 0.270 \times \text{diameter without bark}$, adjusted $R^2 = 0.62$, intercept forced through 0) was applied to past diameters.

Findings

A total of 825 tree cores and basal cross sections were collected from 755 live trees and five snags. All snags were Jeffrey pine. Of the 760 trees sampled, one Pacific madrone and 17 bay laurel were excluded from the chronology and subsequent establishment analyses. Pacific madrone was excluded because of its low frequency across the landscape; bay laurel was excluded because its multi-stemmed growth habit made accurate age estimates difficult. One Douglas-fir and one Jeffrey pine could not be aged due to irregular ring formation. Cores and cross sections from 681 trees representing three species were used to build species-specific

chronologies spanning 143 years that served as quality controls for visual cross-dating. Additionally, 17 cut stumps greater than 30 cm in diameter were encountered in five of the 28 plots. Of the 17 cut stumps, 12 were located in a single plot, and this plot was removed from subsequent analyses.

Jeffrey pine (*Pinus jeffreyi*) was the most frequently encountered tree ($n = 1123$), followed by Douglas-fir (*Pseudotsuga menziesii*) ($n = 457$), Port-Orford cedar (*Chamaecyparis lawsoniana*) ($n = 80$), bay laurel (*Umbellularia californica*) ($n = 30$), tanoak (*Notholithocarpus densiflorus*) ($n = 6$), and Pacific madrone (*Arbutus menziesii*) ($n = 3$). Jeffrey pine has been the dominant tree species in LBH since 1850, except for a brief period ~1940 when the mean number of Douglas-fir recruits was greater than Jeffrey pine (Figure 17). The proportion of tree species present appears to have remained relatively unchanged since the 1960s (Figure 18), although whether this trend continues as Jeffrey pines senesce and die remains to be determined.

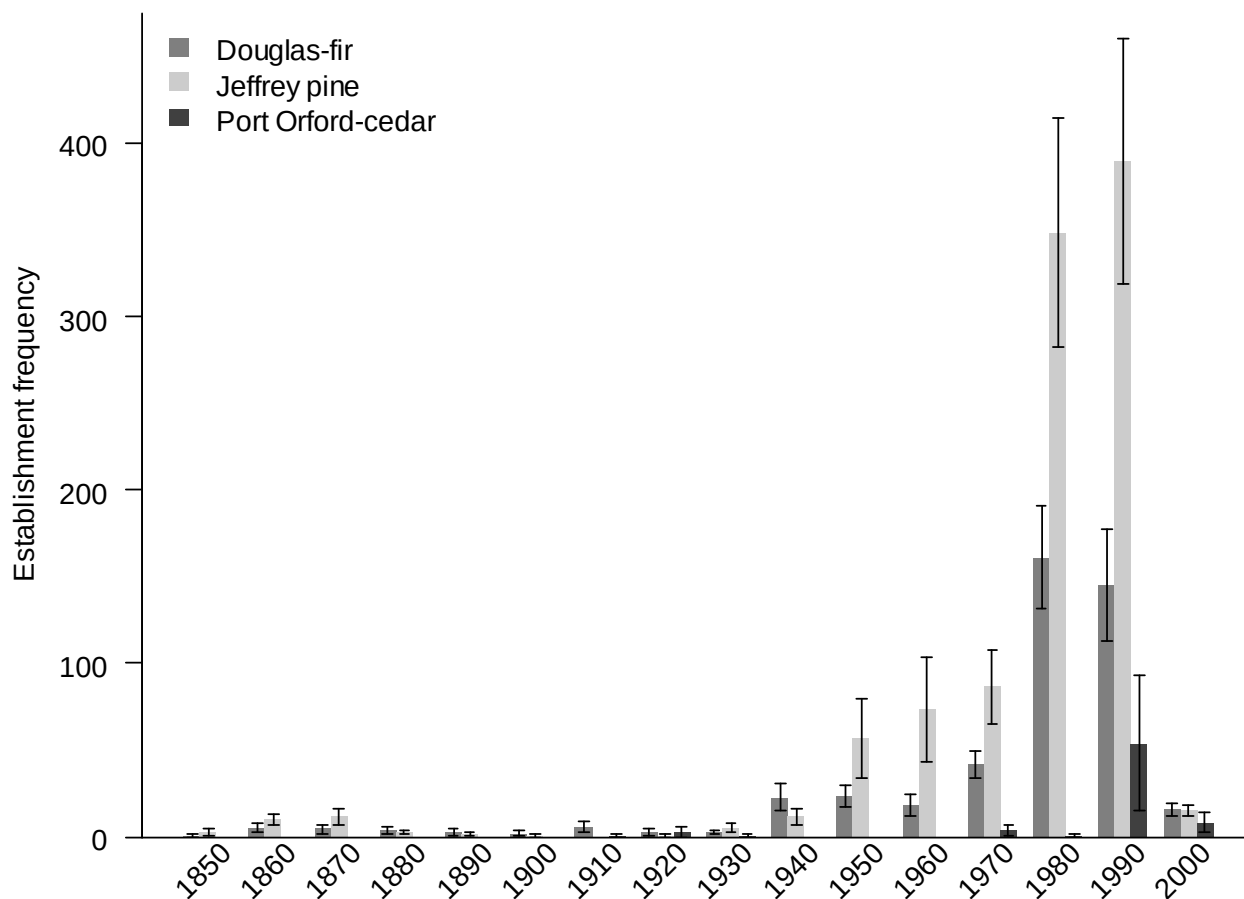


Figure 17. Stem density by decade of the three encountered conifer species in Little Bald Hills since 1850. Error bars represent ± 1 SE.

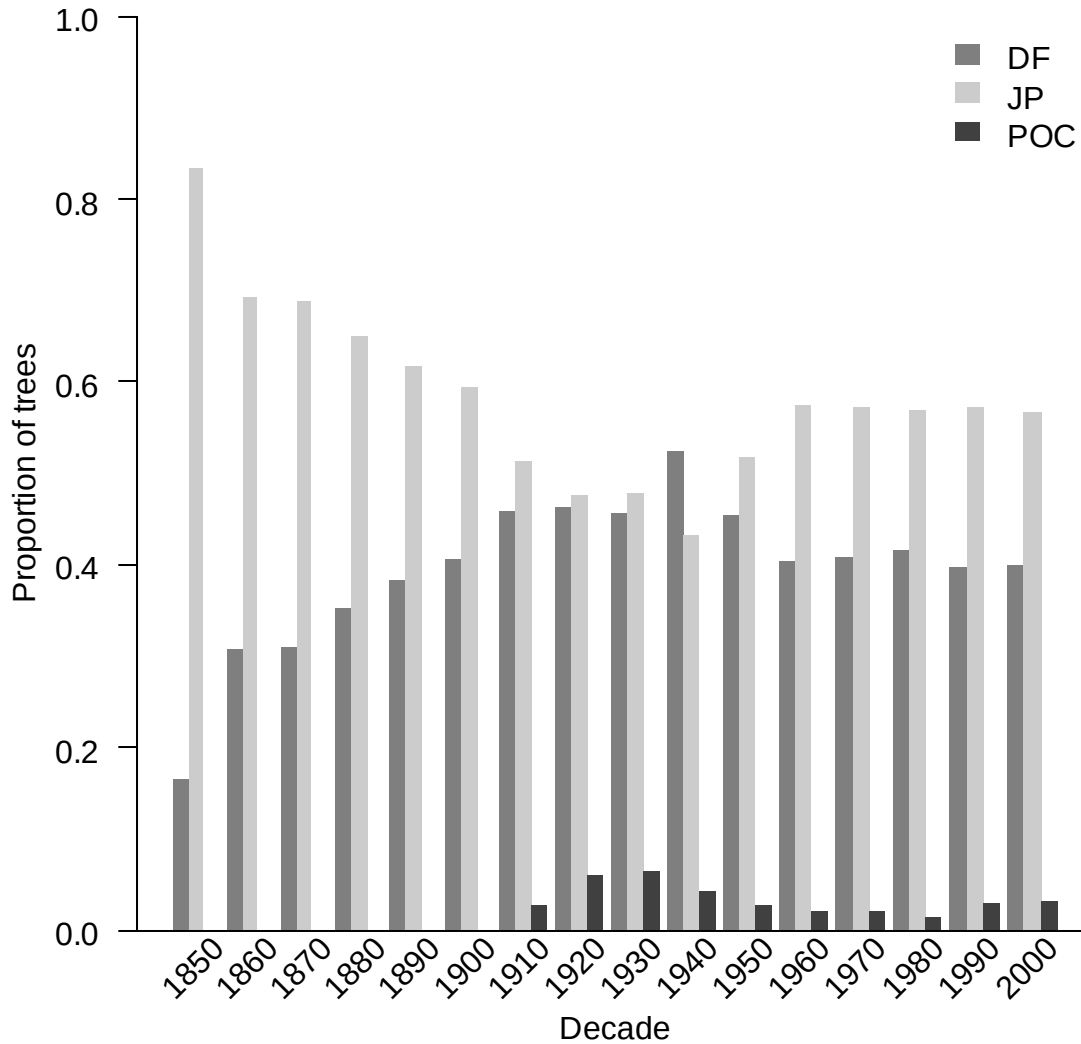


Figure 18. The proportion of trees in Little Bald Hills by decade. Jeffrey pine is the dominant tree species present, and has been consistently since 1850, except for a brief period in the 1940s when more Douglas-fir trees were present.

The mean age of all trees was 43 years. Jeffrey pine, Douglas-fir, and Port-Orford cedar did not have significantly different mean ages (Kruskal Wallis rank sum test, $\chi^2 = 5.19$, d.f. = 2, $p = 0.075$). There were differences in mean tree age by plot (Kruskal Wallis rank sum test, $\chi^2 = 125.46$, d.f. = 26, $p < 0.001$), indicating dates of tree establishment vary significantly across the study site. The oldest trees encountered in this study established in 1855. All tree species exhibited two pulses in establishment, with Jeffrey pine and Douglas-fir showing a pulse beginning ~1940 and another ~1980 (Figure 19 a-d). Port-Orford cedar showed pulses of establishment ~1920 and ~1990.

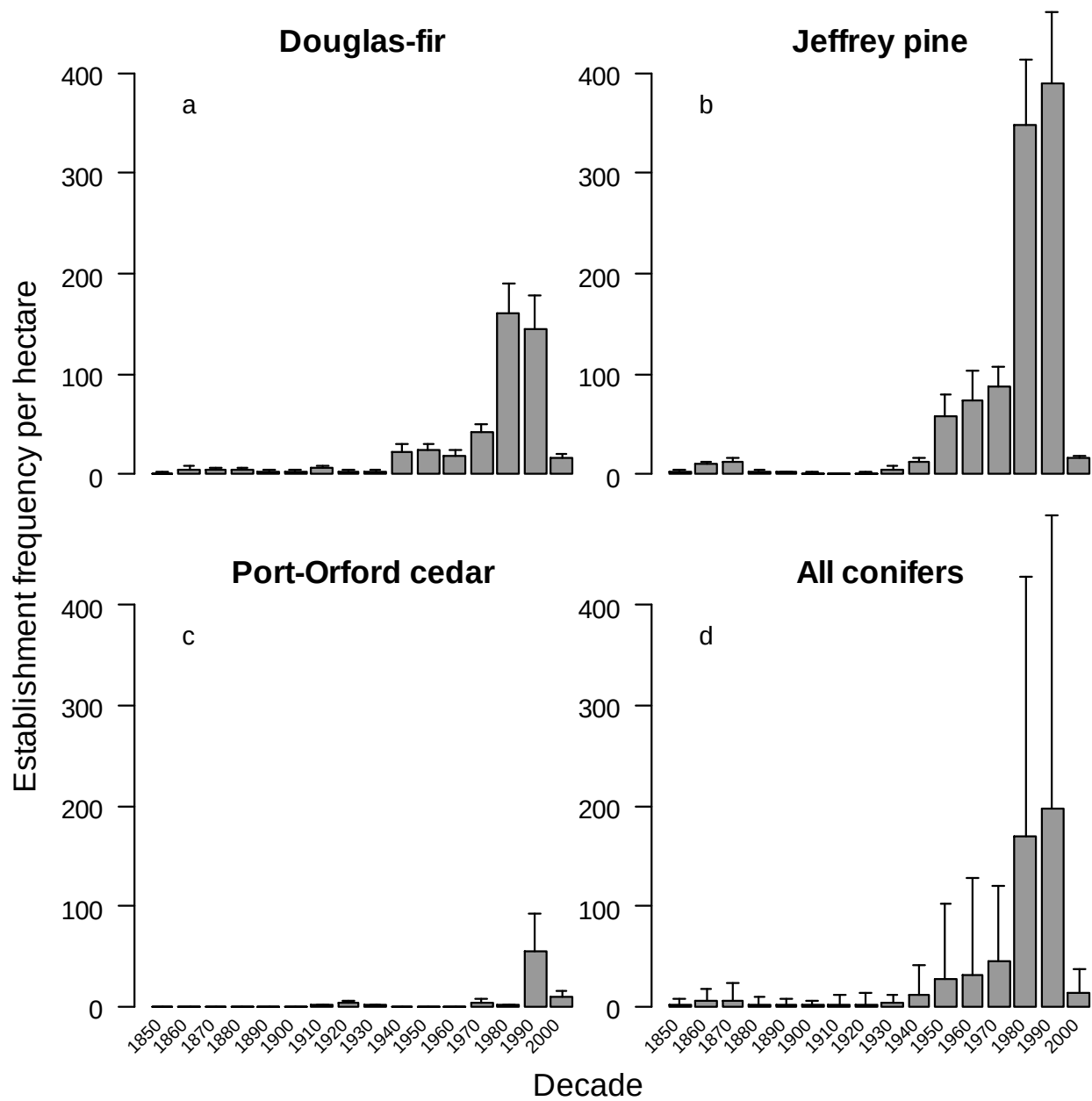


Figure 19. The frequency of tree establishment in Little Bald Hills for the most commonly encountered tree species. Each bar represents the mean number of trees per hectare establishing in a given decade. Error bars are +1 SE. Note the large SE for all conifers combined, an indication of the wide variability in tree establishment across Little Bald Hills.

The number of trees present in LBH has increased dramatically since the decade beginning in 1850, from a mean number of trees per ha of 1.23 ± 1.11 SE to 13.3 ± 4.41 SE in the 2000 decade. The overall stand structure has also shifted throughout time, with medium diameter trees (10-30 cm) making up the majority of trees present on the landscape (Table 8). By 2000, the dominant size class was juvenile trees less than 10 cm in diameter, with a large number of trees greater than 30 cm in diameter (Table 8).

Table 8. Stand structure by diameter size class for four years: 1890, 1920, 1960, and 2000.

	Diameter (cm)	Douglas-fir		Jeffrey pine		Port-Orford cedar	
		Mean no. trees/ha $\pm$ SE		Mean no. trees/ha $\pm$ SE		Mean no. trees/ha $\pm$ SE	
1890	< 10	7.69	$\pm$ 2.50	8.46	$\pm$ 2.76	0.00	$\pm$ 0.00
	10-19.9	2.31	$\pm$ 1.28	9.23	$\pm$ 2.77	0.00	$\pm$ 0.00
	20-29.9	3.08	$\pm$ 1.82	8.46	$\pm$ 3.54	0.00	$\pm$ 0.00
	30-39.9	0.77	$\pm$ 0.77	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00
	40-49.9	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00
	50-59.9	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00
	60-69.9	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00
	>70	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00
1920	< 10	6.92	$\pm$ 2.70	0.77	$\pm$ 0.77	2.31	$\pm$ 2.31
	10-19.9	1.54	$\pm$ 1.07	3.08	$\pm$ 1.44	0.00	$\pm$ 0.00
	20-29.9	6.15	$\pm$ 2.66	10.00	$\pm$ 3.55	0.00	$\pm$ 0.00
	30-39.9	4.62	$\pm$ 2.02	7.69	$\pm$ 2.50	0.00	$\pm$ 0.00
	40-49.9	1.54	$\pm$ 1.07	5.38	$\pm$ 2.09	0.00	$\pm$ 0.00
	50-59.9	2.31	$\pm$ 1.28	0.77	$\pm$ 0.77	0.00	$\pm$ 0.00
	60-69.9	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00
	>70	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00
1960	< 10	43.08	$\pm$ 11.88	50.77	$\pm$ 18.60	3.08	$\pm$ 3.08
	10-19.9	13.08	$\pm$ 4.29	3.85	$\pm$ 1.93	1.54	$\pm$ 1.54
	20-29.9	3.85	$\pm$ 1.58	8.46	$\pm$ 2.76	0.00	$\pm$ 0.00
	30-39.9	3.85	$\pm$ 2.22	3.85	$\pm$ 1.93	0.00	$\pm$ 0.00
	40-49.9	5.38	$\pm$ 2.62	10.00	$\pm$ 3.19	0.00	$\pm$ 0.00
	50-59.9	3.85	$\pm$ 1.93	5.38	$\pm$ 2.09	0.00	$\pm$ 0.00
	60-69.9	3.08	$\pm$ 1.44	2.31	$\pm$ 1.28	0.00	$\pm$ 0.00
	>70	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00	0.00	$\pm$ 0.00
2000	< 10	259.23	$\pm$ 69.17	569.23	$\pm$ 104.61	57.69	$\pm$ 49.85
	10-19.9	30.00	$\pm$ 9.17	45.38	$\pm$ 16.21	2.31	$\pm$ 2.31
	20-29.9	17.69	$\pm$ 4.88	15.38	$\pm$ 6.59	0.77	$\pm$ 0.77
	30-39.9	10.00	$\pm$ 3.37	10.77	$\pm$ 3.55	0.77	$\pm$ 0.77
	40-49.9	6.92	$\pm$ 2.92	7.69	$\pm$ 2.73	0.00	$\pm$ 0.00
	50-59.9	5.38	$\pm$ 2.09	8.46	$\pm$ 2.97	0.00	$\pm$ 0.00
	60-69.9	6.92	$\pm$ 2.70	6.15	$\pm$ 1.85	0.00	$\pm$ 0.00
	>70	5.38	$\pm$ 3.05	1.54	$\pm$ 1.07	0.00	$\pm$ 0.00

Research Question 7: *What is the rate and extent of tree encroachment into serpentine grasslands of the Little Bald Hills?*

Methods

In order to evaluate the spatial extent of encroachment in Little Bald Hills and to project future encroachment trends, we acquired aerial photos for 16 years spanning 1942 to 2009. Digital images for 11 years were downloaded through USGS Earth Explorer (earthexplorer.usgs.gov/), USDA Geospatial Data Gateway (datagateway.nrcs.usda.gov/), or Cal-Atlas (atlas.ca.gov/) and included black-and-white, color-infrared, and true-color images. We created digital images for five additional years of hardcopy black-and-white photos maintained by Six Rivers National Forest by scanning the original 9" x 9" photographs at 600 dpi. We evaluated photos based on scale, area covered, temporal relevance, and image quality and selected nine of the 16 photo years for analysis: 1942, 1960, 1975, 1980, 1988, 1993, 1998, 2005 and 2009.

Images from 1942, 1960, 1975, and 1980 were assigned spatial reference using Erdas Imagine 2010 software. The images were orthorectified to a 10-m digital elevation model (DEM) and referenced to a 1988 digital orthophoto quadrangle. A minimum of ten ground control points (GCPs) were used for reference in each image; the low number of GCPs used was due to the rural setting of LBH and the paucity of consistently recognizable landmarks in the images. Images were then classified using a hybrid classification approach where an eight to 12 class unsupervised classification was first performed. These classes were subsequently merged into three classes: shadow, grassland, and woody vegetation. A supervised classification was then performed on the images. Additional class signatures were gathered if the initial classification was deemed inaccurate. Shadow pixels were eliminated from the images, leaving two types of classified pixels: grass and woody vegetation. Since the earliest photos are black-and-white panchromatic, classifications were run on single band images. In three and four-band images, the green band was selected for classification. Accuracy assessment used the original image as a reference and took into account color, pattern, and texture.

In order to predict when the grassland in LBH might no longer be visible in aerial images, and therefore unlikely to be ecologically functional as grassland, we created an irregular time series of the proportion of grassland classified for each image year. Linear regression requires independent observations, but the proportion of grassland in a given year is not independent of the preceding year's grassland proportion. In order to determine whether there was a temporal trend in the reduction in grassland and to quantify the rate at which grassland is visually disappearing, we fit autoregressive integrated moving average (ARIMA) models, which are the appropriate model choice when dealing autocorrelated time series data. We used an AICc-based model selection approach to evaluate four ARIMA models: 1) a random walk model that fit no temporal trend, 2) a trend model that fit a temporal trend, 3) a 1st-order autoregressive model that fit an autoregressive parameter but no temporal trend, and 4) a 1st-order autoregressive model with one order of differencing that fit both a temporal trend and an autoregressive parameter.

Findings

Total area classified for each image year ranged between 75% and 96% and overall classification accuracies were very high (Figure 20). The 1975 image had the highest overall accuracy rate

(99%), while 2009 had the lowest (90%). Since 1942, the area of LBH classified as grassland has dramatically declined (Figure 21).

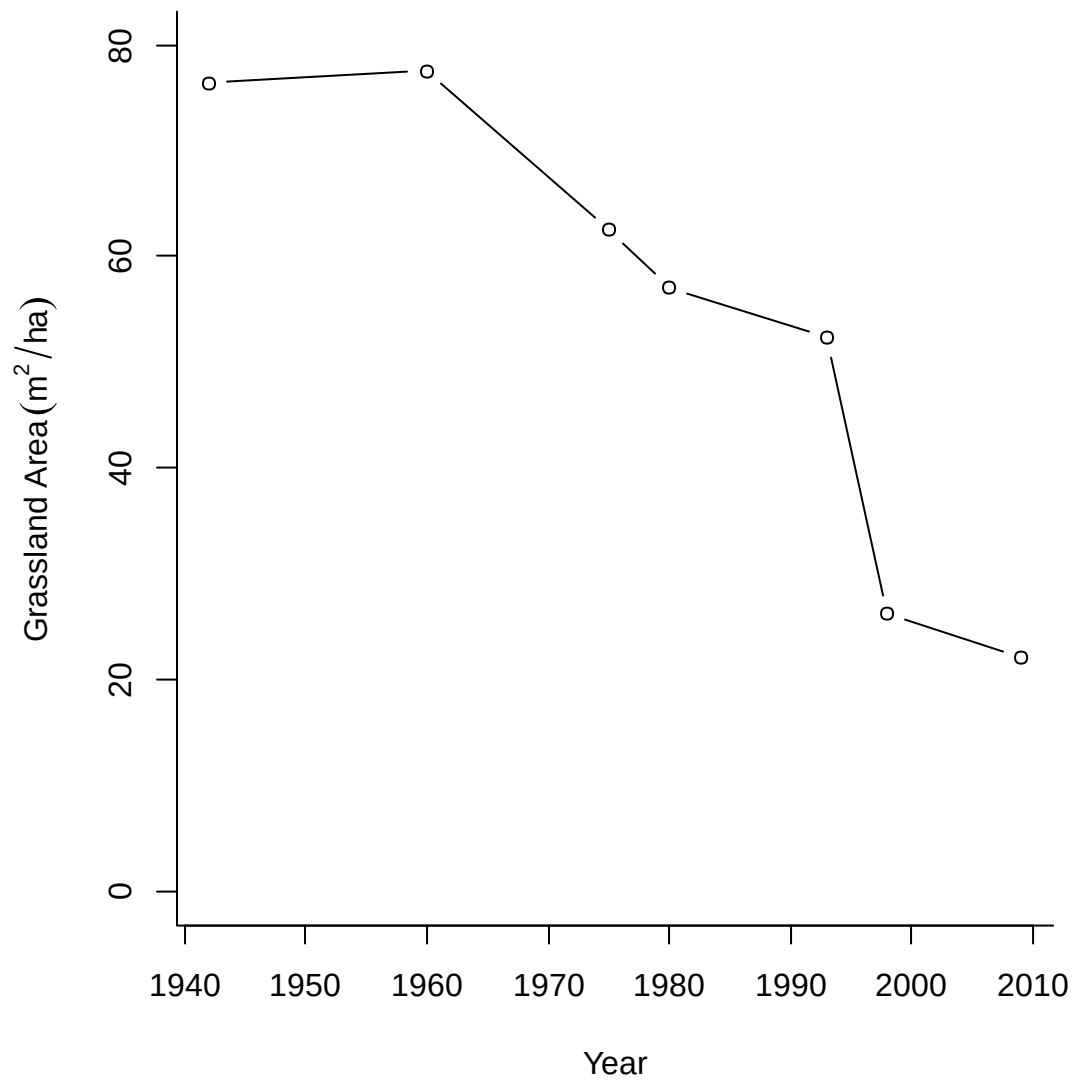


Figure 20. The area in hectares of grassland classified in eight image years. Total grassland area classified in 2009 (22 ha) is less than half the area identified in 1942 (76 ha).

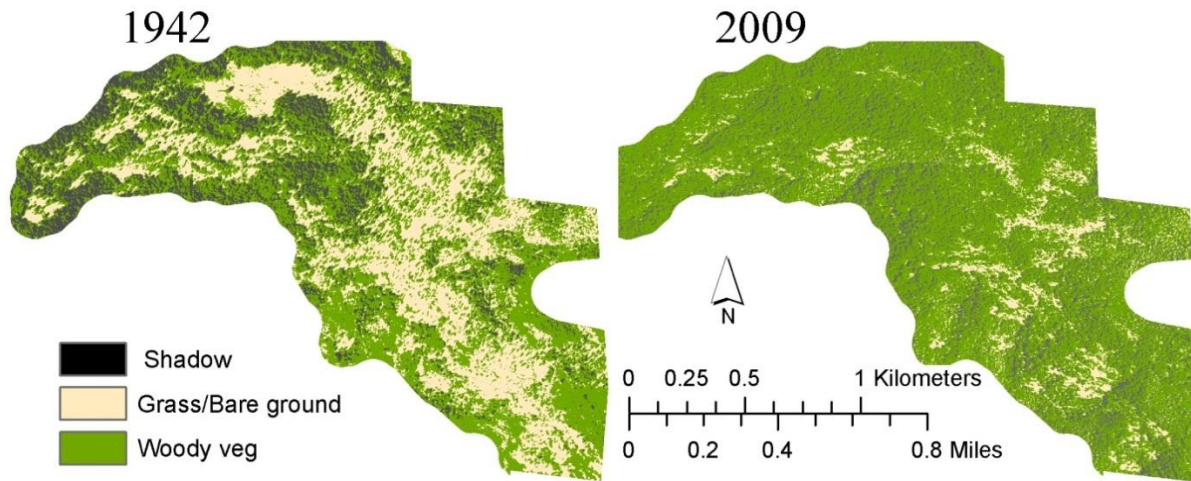


Figure 21. Image classification of LBH into grassland and woody vegetation in 1942 and 2009. Shadow areas were removed from images for analysis.

The time series model with the lowest AICc value was the random walk model (Table 9). Since this time series had only nine observations, which is very few for a time series, it is not surprising that the variability present in the observations is greater than an identifiable trend. However, the random walk model had less than half the model weight, and the second-best model, which had only a temporal trend fit, was weighted nearly equally with the random walk model (Table 9). This is moderate evidence of the ability of this model to predict past and future time trend. The trend model estimated a rate of $37\text{m}^2/\text{year}$ loss of grassland, and indicates a total loss of visible grassland around the year 2035 (Figure 22). The 80% confidence interval around the estimated trend is quite wide, and reflects the limited number of observations available for analysis (Figure 22). Acquisition and classification of additional images is recommended in order to get a better estimate of when visible grassland will disappear in LBH.

Table 9. AICc table for the four fitted time series models.

Model	K	AICc	ΔAICc	AICc Wt	LL	Cum. Wt
Random walk	1	-19.89	0	0.45	11.28	0.45
Trend	2	-19.73	0.16	0.41	13.06	0.86
AR1	2	-17.05	2.84	0.11	11.73	0.97
Trend+AR1	3	-14.4	5.49	0.03	13.2	1

AICc is Akaike's information criterion adjusted for small sample size. The difference between a given model and the best model (ΔAICc) is reported as a measure of model comparison. K is the number of model parameters fit. Weight is the probability that the model is the best of the candidate models. LL is the log-likelihood of the model.

Predicted grassland disappearance

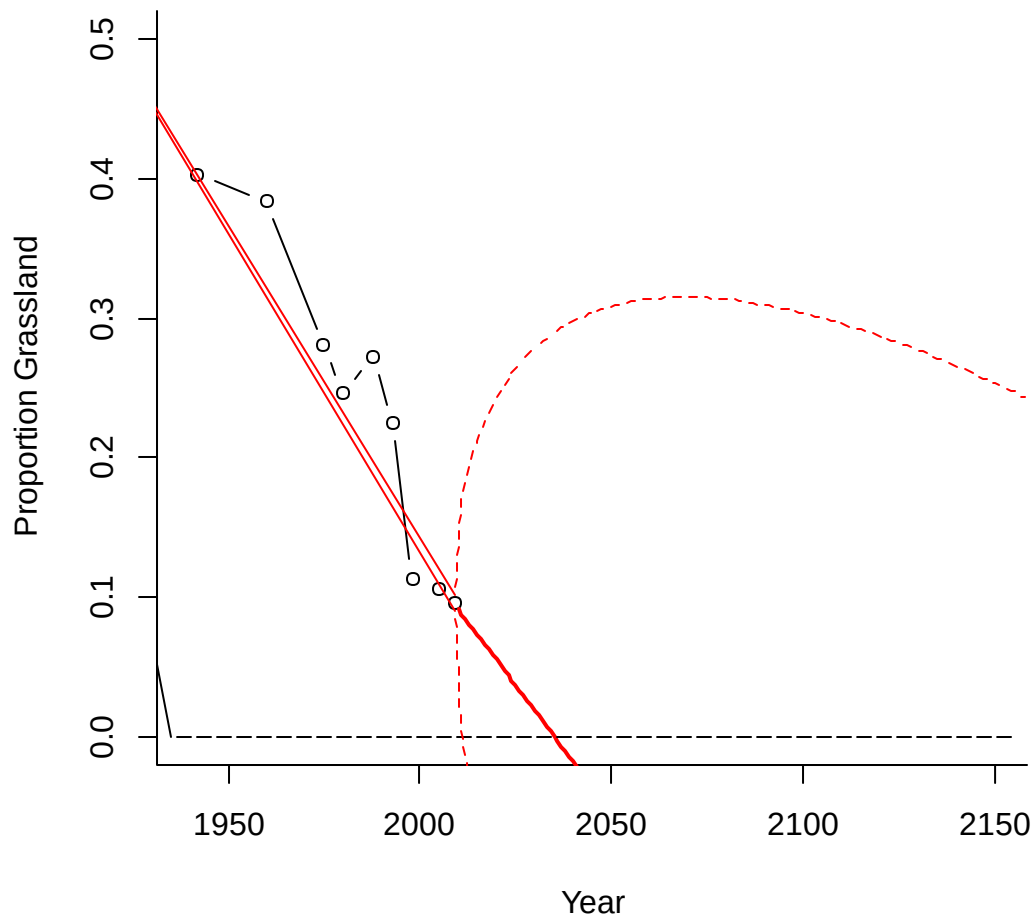


Figure 22. Observations of the proportion of grassland classified from nine vertical aerial photographs and the predicted decline of grassland in red. The dashed red lines are 80% confidence intervals around the estimated trend, and indicate estimated dates of complete grassland loss between the years 2013 and 2300. The confidence intervals are so wide because of the limited number of observations available for analysis.

Research Question 8: *Is encroachment related to site characteristics in Little Bald Hills?*

Methods

For each of the 28 plots established in 2009 in Little Bald Hills (see Research Question 8 methods), we recorded the following environmental variables: canopy cover, slope, aspect, and topographic position (ridge, upper slope, etc.) at the center of each plot. In order for aspect to be used as one continuous variable with a meaningful range for statistical analysis, we treated a transformed aspect value as a proxy for insolation by using the method of Beers et al. (1966) where transformed aspect $A' = \cos(A_{\max} - A) + 1$. $A_{\max}$ in azimuth degrees represents the highest assigned number on the transformed scale, and A is measured aspect. We set $A_{\max} = 45^\circ$ for this insolation index so that northeast-facing aspects had the highest value of 2 and southwest-facing aspects had a value of zero.

We used diameter at breast height (DBH) measurements taken from trees within each plot during our 2009 field season and calculated basal area/ha for each plot. Sixteen linear regression models were evaluated using Akaike's Information Criterion adjusted for small sample sizes (AICc) in order to evaluate whether any site variables were good predictors of modern basal area measurements of Jeffrey pine and Douglas-fir. Topographic position was assigned as a categorical variable; continuous variables were slope, insolation index values (transformed aspect), and elevation. The 16 models included a null model which fit the mean basal area only.

Findings

Plot basal area in 2009 was not well explained by the four measured explanatory variables (Table 10). The best model for both Jeffrey pine and Douglas-fir basal area included no explanatory variables and fit the mean basal area only (Table 10). Although there was limited evidence supporting slope, insolation, and elevation as useful predictors of 2009 basal area, models including these variables did account for some of the variability in basal area between plots (Table 10). Coefficients on slope, insolation (transformed aspect), and elevation for the 2nd, 3rd, and 4th-ranked models were all positive, indicating some support for high basal area values to correspond to northwest-facing, steeper slopes, and higher elevations.

Table 10. AICc table comparing 16 models relating the basal area in 2009 of trees in Little Bald Hills to site variables. The null model has the majority of the model weight, indicating basal area was not well-explained by site variables, although there is limited evidence to support slope, insolation, and elevation as predictors of basal area.

Model	K	AICc	Δ AICc	AICcWt	LL	Cum.Wt
null	2	103.96	0.00	0.40	-49.72	0.40
slope	3	105.48	1.52	0.19	-49.19	0.59
insolation	3	106.29	2.33	0.12	-49.60	0.71
elevation	3	106.40	2.44	0.12	-49.65	0.83
slope+elevation	4	107.67	3.71	0.06	-48.88	0.89
slope+insolation	4	107.99	4.03	0.05	-49.04	0.95
insolation+elevation	4	109.01	5.05	0.03	-49.55	0.98
slope+insolation+elevation	5	110.52	6.57	0.02	-48.76	0.99
topographic position	6	113.41	9.45	0.00	-48.49	1.00
topographic position+elevation	7	116.63	12.67	0.00	-48.20	1.00
slope+topographic position	7	116.73	12.77	0.00	-48.25	1.00
insolation+topographic position	7	117.19	13.23	0.00	-48.48	1.00
slope+topographic position+elevation	8	120.27	16.31	0.00	-47.90	1.00
insolation+topographic position+elevation	8	120.88	16.92	0.00	-48.20	1.00
slope+insolation+topographic position	8	120.93	16.97	0.00	-48.23	1.00
slope+insolation+topographic position+elevation	9	125.04	21.08	0.00	-47.90	1.00

AICc is Akaike's information criterion adjusted for small sample size. The difference between a given model and the best model (Δ AICc) is reported as a measure of model comparison. K is the number of model parameters fit. Weight is the probability that the model is the best of the candidate models. LL is the log-likelihood of the model.

Research Question 9: *What are the vegetation patterns in the Douglas-fir forests in the Little Bald Hills?*

Methods

Field Sampling

Vegetation patterns in the Little Bald Hills were mapped in the 1980s (Goforth and Veirs 1989). The vegetation alliances are illustrated in Figure 23. Within the Douglas-fir forests, we sampled thirty circular 500 m² plots using stratified random sampling. Plots were evenly established in two “zones” representing increasing distance from the Jeffrey pine vegetation alliance described for study question 6: Zone 1 was >200 m from Jeffrey pine savanna and zone 2 was within 200 m of the Jeffrey pine savanna (Figure 23). Vegetation community data were gathered from all plots and tree stand dynamics were gathered from the first 20 of those plots.

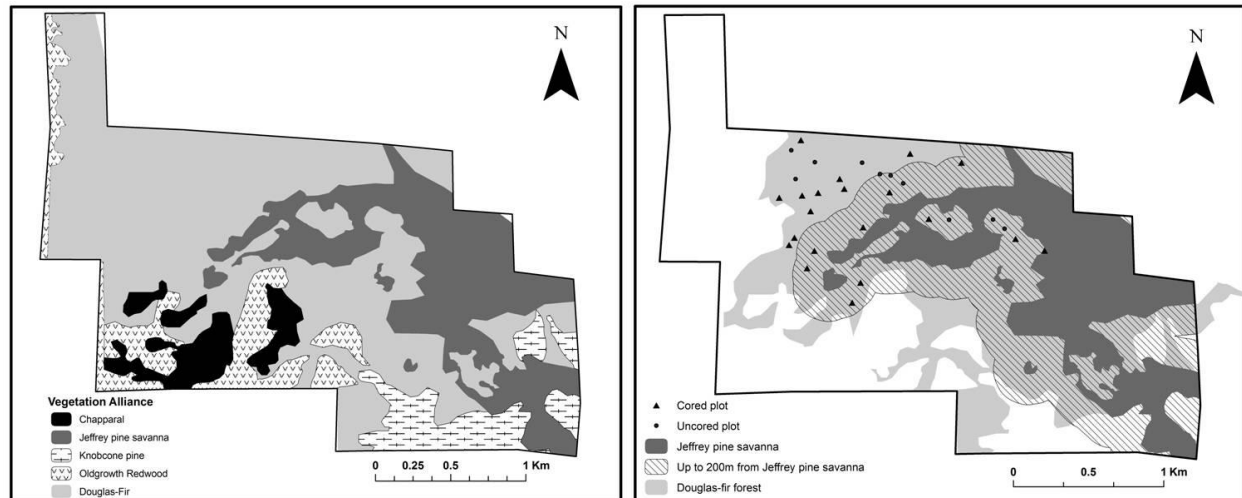


Figure 23. Vegetation alliances in the Little Bald Hills of Redwood National Park. b) Location of field plots for vegetation sampling in the Little Bald Hills.

Forest community data

Within each plot, woody plant species were identified (canopy, understory and shrubs) following Hickman (1993) and abundance was assigned using the Braun-Blanquet scale (Mueller-Dombois and Ellenberg 1974). We recorded environmental variables for each plot such as slope, canopy cover, and tree seedling cover. Any evidence of management (e.g. cut tree stumps) was also noted. The occurrence of herbaceous plants was rare in the plots surveyed and they were not recorded as part of this study.

Vegetation analyses

All plant community analyses were conducted using PC-ORDv.5 (McCune and Mefford 2006). Non-metric Multidimensional Scaling (NMS) was selected as it is the most generally effective ordination method for ecological community data (McCune and Mefford, 2006). NMS also

avoids the assumption of linear relationships among variables and allows the use of distance measures suited to non-normally distributed data (McCune and Grace 2002). An NMS ordination using a Sørensen distance measure, a two-axis solution and 1000 real data runs were selected based on initial autopilot results. Multi-Response Permutation Procedures (MRPP) were used to test predefined groupings within the sampled plots. MRPP is a non-parametric multivariate test for differences between groups (McCune and Grace 2002, Mielke and Berry 2007). The data can be grouped in any permutation and the strength of that grouping can be tested. The Sørensen distance measure was also chosen as the distance measure for the MRPP tests (McCune and Grace 2002, Mielke and Berry 2007).

Findings

Forest community data

The majority (70%) of plots sampled were dominated by Douglas-fir (*Pseudotsuga menziesii*). Port-Orford cedar (*Chamaecyparis lawsoniana*) commonly occurred in these plots. The remaining 30% of plots were dominated by tanoak (*Notholithocarpus densiflorus*), Port-Orford cedar, golden chinquapin (*Chrysolepis chrysophylla*), Jeffrey pine, knobcone pine (*Pinus attenuata*), and Pacific madrone (*Arbutus menziesii*; Appendix A). *Vaccinium ovatum* and *Rhododendron macrophyllum* formed dense shrub-stratum thickets across the site (Figure 24).



Figure 24. Thickets of *R. macrophyllum* (left) and *Vaccinium ovatum* (right) were common across the study site.

The NMS analyses indicated a two axis-solution. The coefficients of determination for the correlations between ordination distances and distances in the original n dimensional space are a good measure for variance explained. Axis 1 accounted for 27.2% and axis 2 accounted for 60.7% of the variance explained (Figure 25).

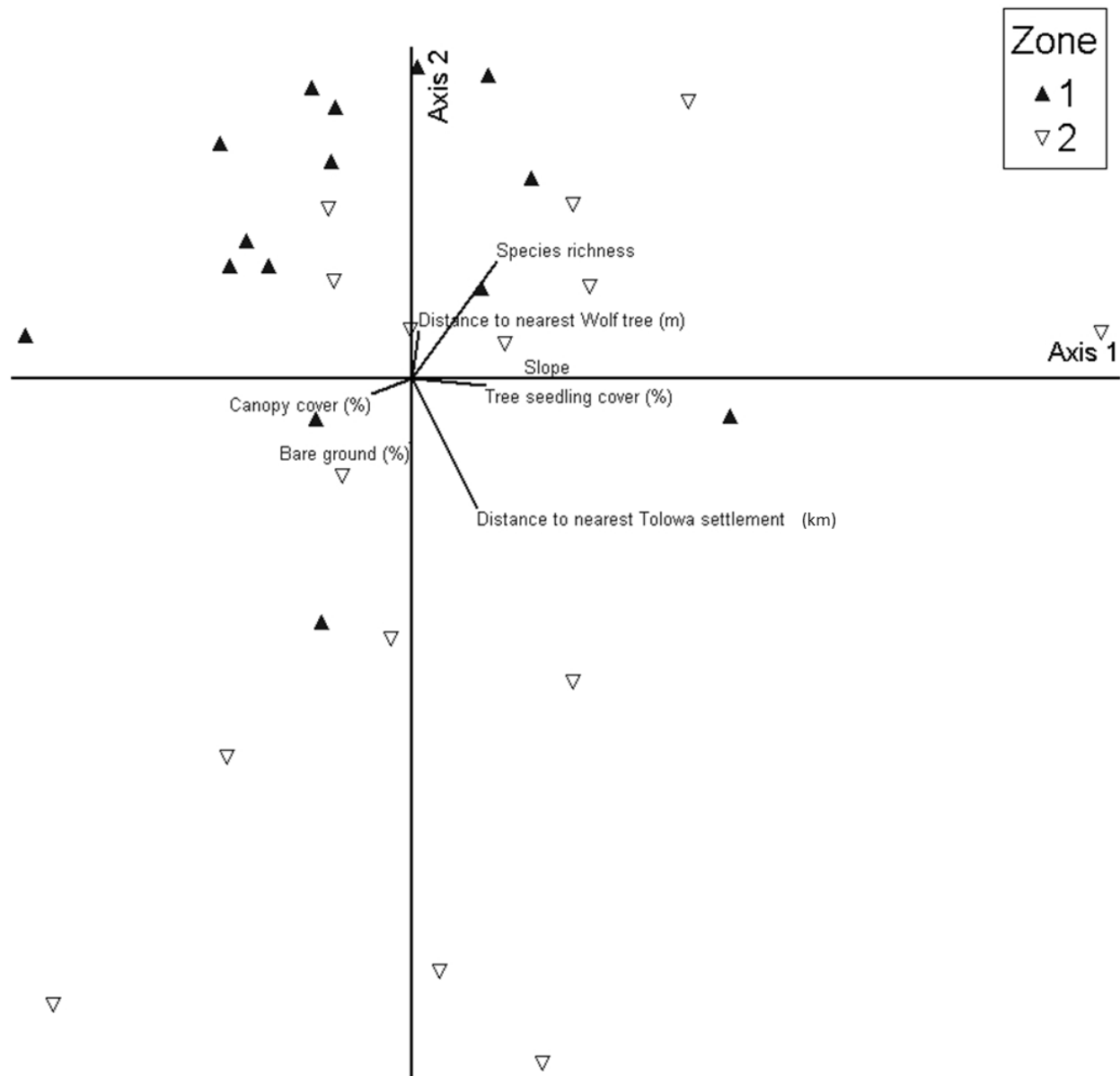


Figure 25. NMS graph of plant community data showing an overlay of the zone each plot was in. Plots in Zone 1 were >200m from Jeffrey Pine savannah. Plots in Zone 2 were <200m from Jeffrey Pine savanna. A biplot of variables measured is also overlain.

Relationships between the graph axes and the plot variables collected were examined visually using biplots and also using Kendall's tau correlation values (Figure 26, table 11). Slope was correlated with axis 1. The plots in the left of the NMS graph were on less steep slopes than those on the right. Tree seedling cover was also correlated with axis 1 (not visible in the biplot as it is parallel with axis 1 in the same direction as slope). Plots to the right of the NMS graph had a lower canopy cover which probably accounts for the increased tree seedling cover. Species richness was correlated with both axes 1 and 2 with plots in the top right quadrant of the graph

being the most species rich. Distance to nearest Tolowa settlement and distance to nearest wolf tree were also significantly correlated with axis 2 (Table 11).

Table 11. Kendall's tau values showing the correlations between the variables gathered per plot and the NMS axes 1 and 2.

Variable	Kendall's tau	
	Axis 1	Axis 2
Canopy cover	-0.109	-0.067
Slope	0.289*	0.093
Bare ground (%)	-0.076	-0.241
Tree seedling cover (%)	0.371**	-0.225
Dead wood (%)	0.058	-0.11
Distance to nearest Wolf Tree	0.034	0.301*
Species richness	0.301*	0.301*
Distance to nearest Tolowa settlement	0.239	0.322*

* $p < 0.05$, ** $p < 0.01$

Axis 2 appears to represent the change in forest habitat communities encountered from Douglas-fir-dominated communities at the top of axis 2 to tanoak-dominated communities at the bottom of the axis (Figure 26). Axis 1 could represent a canopy openness gradient with plots to the left of the graph having high canopy cover and plots to the right of the graph having high tree seedling cover and lower canopy cover. It could also represent a moisture gradient with Port-Orford cedar occurring more to the right of the graph and plots with knobcone pine occurring on the left side. More plots in zone 1 were dominated by Douglas-fir than in zone 2 (Figure 26). MRPP grouped the plots according to the zones ($n=2$, $P < 0.001$, $A=0.06$), but these differences are probably more an indication of the range of forest habitats across the two zones as opposed to two distinctly different vegetation types. Almost all plots that were not dominated by Douglas-fir occur in the lower quadrants of the graph; the majority of these were in zone 2 (Figure 26). This indicates greater variation in the region closer to the Jeffrey pine savanna. Tanoak-dominated, chinquapin-dominated, and Port-Orford cedar-dominated stands were found in zone 2 only (see Figures 27 & 28).

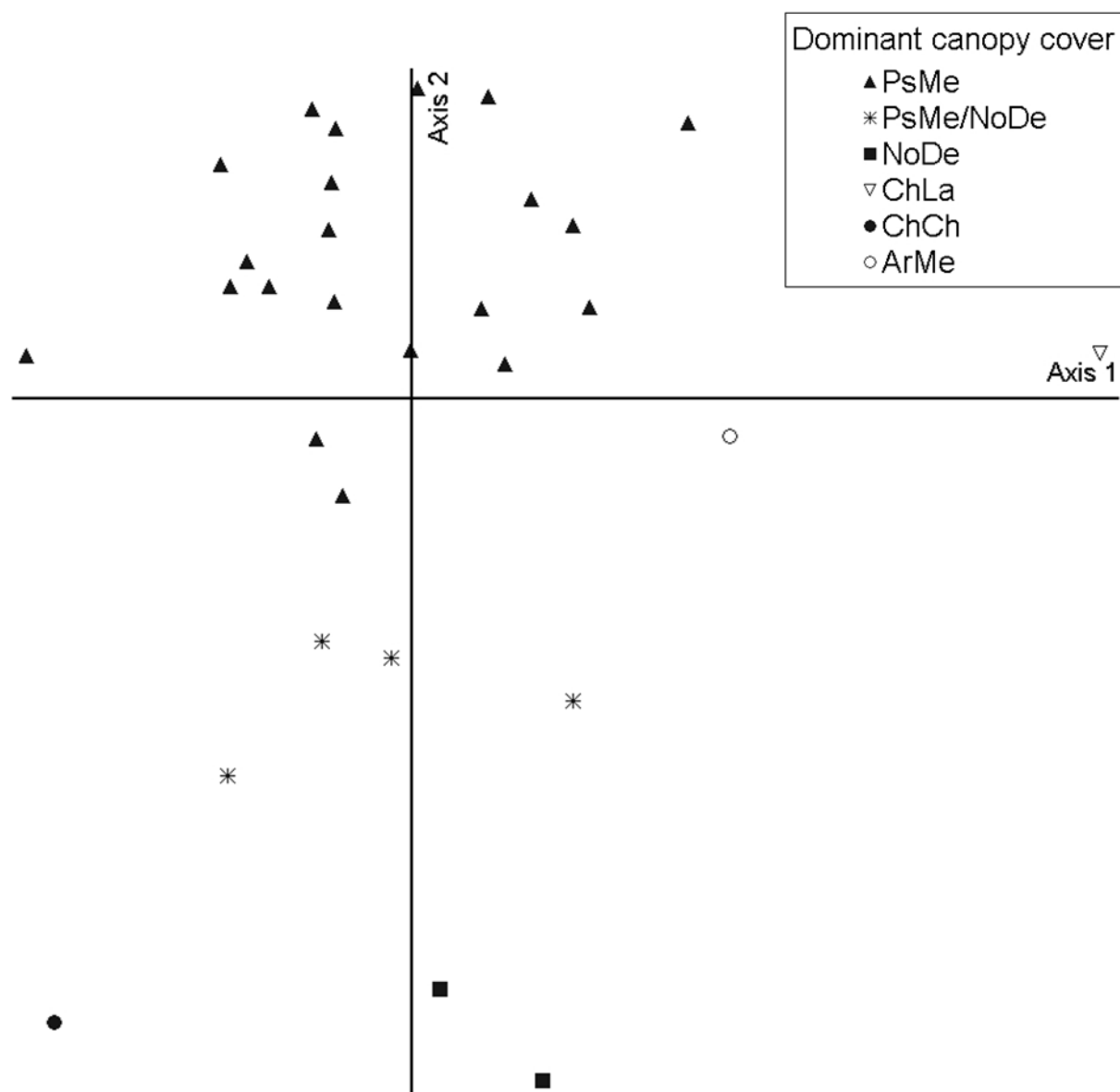


Figure 26. NMS of plant community data showing an overlay of the dominant tree species in that plot. PsMe=*Pseudotsuga menziesii*, PsMe/NoDe- *Pseudotsuga menziesii*/*Notolithocarpus densiflorus* codominance, NoDe=*Notolithocarpus densiflorus*, ChLa=*Chamaecyparis lawsoniana*, ChCh= *Chrysopsis chrysophylla*, ArMe=*Arbutus menziesii*



Figure 27. Douglas-fir dominated stand in the Little Bald Hills with a thick understory of *Vaccinium ovatum*.



Figure 28. Tanoak-dominated stand in the Little Bald Hills. The understory is sparse and is dominated by *Vaccinium ovatum*.

Notholithocarpus densiflorus and *Umbellularia californica* were both present as understory and shrub species as well as canopy tree species (Appendix A). Grouping plots based on the dominant canopy species ($n = 4$, $P < 0.001$, $A = 0.22$) indicated the presence of at least four distinct plant communities: Douglas-fir-dominated; Douglas-fir/tanoak-codominant, tanoak-and

non-Douglas-fir/non-tanoak-dominated assemblages (there was just one plot each of madrone-, chinquapin- and Port-Orford cedar-dominated plots).

Douglas-fir was the dominant canopy species throughout most of the Douglas-fir alliance. The plant community assemblages are similar to those found in Douglas-fir forests throughout the Pacific Northwest (Spies 1991, Sugihara et al. 2006). In those areas where Douglas-fir was not the dominant species, tanoak was most commonly dominant. There were pockets of tanoak-dominated forest throughout the LBHs, particularly in the area closer to the Jeffrey pine savanna.

Research Question 10: *What is the stand history of the Douglas-fir forests in the Little Bald Hills?*

Methods

Sampling

Within the Douglas-fir forests, we used the same plot network surveyed for Question 7. Stand age data were gathered from the first 20 of the 30 plots. Tree diameter at breast height (DBH) was recorded for all trees in the plot. Cores were taken from up to four representative trees across all species and 10 cm DBH size classes present in each plot using an increment borer (Figure 29). Extracted cores were prepared using standard tree ring analysis techniques; they were mounted, air-dried, and sanded with progressively finer sandpaper from 180 to 1500 grit in the lab. Cores were visually cross-dated and aged (Speer 2010).



Figure 29. C. Sullivan coring a large Douglas-fir tree with an increment borer.

Distribution of “wolf” trees and remnant Jeffrey pines

The location and DBH of all encountered “wolf” trees and Jeffrey pines were recorded. Wolf trees were defined as conifers with coarse, heavy-limbs with little effective lateral competition and broad crowns that were open-grown throughout most of their life history (Neitlich and McCune 1997, Thomas and Packham 2007). All wolf trees were visited and their locations and DBH recorded. The same was done for any Jeffrey pine trees encountered outside of the Jeffrey pine savanna area. These data were all digitized in ArcGIS 9.3.

Statistical analyses

Summaries of tree ages were organized by species. Multiple regression analysis was used to investigate whether the establishment date of the plots were influenced by environmental factors. All statistical tests were carried out using SPSS v. 17.

Findings

Tree stand dynamics

In total, 362 trees (10 to 27 trees per plot) were cored. The DBHs ranged from 11-145 cm and the mean dbh was 35cm (SD = 21). Only 6 trees had DBH $\geq$ 100 cm. The majority of the trees cored were Douglas-fir. Tanoak and Port-Orford cedar made up 19 and 11% of cored trees, respectively. *Umbellularia californica*, *Pinus attenuata*, *Arbutus menziesii*, *Chrysolepis chrysophylla*, *Pinus jeffreyi*, and *Tsuga heterophylla* made up the remaining 18% of trees cored (Table 12). The oldest tree cored was a Douglas-fir tree dating to 1660. Douglas-fir trees accounted for the 6 of the 7 trees that established before 1850. The only other tree established before 1850 was a Jeffrey pine that dated to 1728. The earliest establishment date of each tree species cored is shown in Table 12.

Table 12. Tree species cored, number of each, and proportion of total.

Tree species	No. of trees cored	Proportion of all trees cored (%)
<i>Pseudotsuga menziesii</i>	188	52
<i>Notholithocarpus densiflorus</i>	70	19
<i>Chamaecyparis lawsoniana</i>	40	11
<i>Umbellularia californica</i>	25	7
<i>Pinus attenuata</i>	14	4
<i>Arbutus menziesii</i>	12	3
<i>Chrysolepis chrysophylla</i>	9	2
<i>Pinus jeffreyi</i>	3	<1
<i>Tsuga heterophylla</i>	1	<1
Total	362	100

Table 13. Earliest establishment dates for tree species cored in the Little Bald Hills. Tree species are arranged in order of establishment date.

Tree species	Earliest establishment date
<i>Pseudotsuga menziesii</i>	1660
<i>Pinus jeffreyi</i>	1728
<i>Notholithocarpus densiflorus</i>	1853
<i>Chrysolepis chrysophylla</i>	1878
<i>Umbellularia californica</i>	1889
<i>Arbutus menziesii</i>	1902
<i>Chamaecyparis lawsoniana</i>	1904
<i>Pinus attenuata</i>	1907
<i>Tsuga heterophylla</i>	1912

Landscape-level recruitment pattern

All plots, except for plot 1, reveal dates of tree establishment between 1850 and early 1900s. Five trees in plot 1 established between 1660 and 1716. In all other plots there was a sustained recruitment event beginning around the 1850s, peaking in the 1930s. Recruitment appears to reduce dramatically in the 1940s and continues to decline into the 1980s (top left panel in Figure 30).

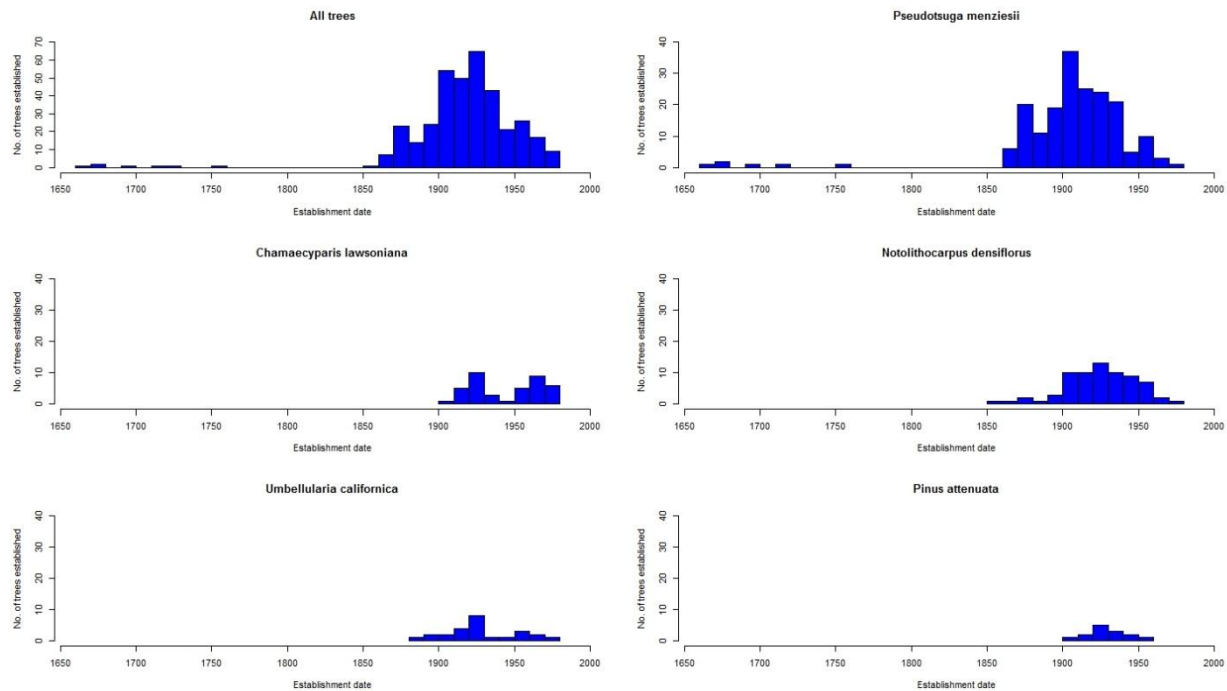


Figure 30. Tree recruitment patterns in the Little Bald Hills. Top left panel shows patterns for all trees cored; other panels show a breakdown by species. Trees that made up <4% of the trees cored are not shown. Note that the change in scale of the y-axis between the top left panel and all other panels.

Tree recruitment data for individual tree species can be seen in Figure 31. The oldest trees cored in the Little Bald Hills were Douglas-fir with 6 trees establishing pre-1760s and the oldest tree dating back to the 1660s. Tanoak and Port-Orford cedar were also common species at the site. The oldest tanoak dated to 1854 with an increase in recruitment around the 1900s. Port-Orford cedar recruitment was confined to after the 1900s but at least one wolf Port-Orford cedar was noted at the site and several trees were not cored due to signs of disease.

Establishment trends

There were no major differences in tree establishment dates between zones (Figure 31). The left panels show the establishment dates for all trees in each zone (blue=>200 from Jeffrey pine savanna, red=<200m from Jeffrey pine savanna). The panels to the right show the establishment trends of the two most common tree species in the plots, Douglas-fir and tanoak. Again, differences are apparent between the zones in Douglas-fir establishment patterns. Several Douglas-fir trees established prior to 1750 in zone 2 compared with just one in zone 1. It is clear that tanoak comprise a more substantial portion of the trees in zone 2.

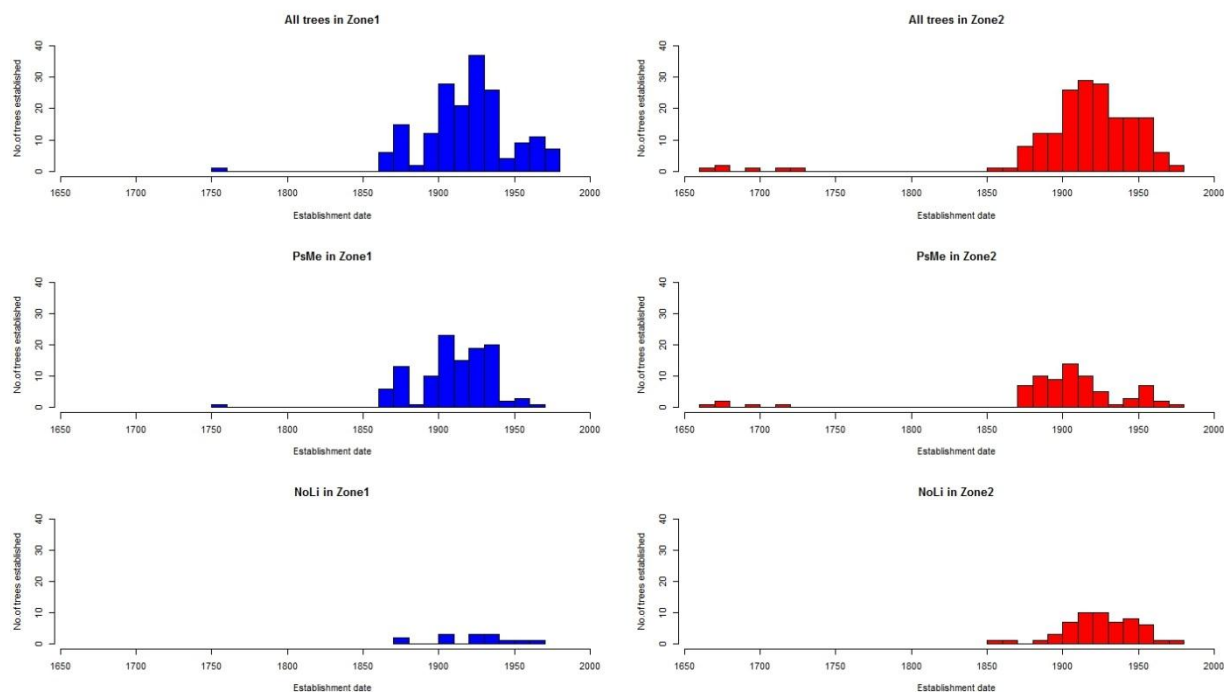


Figure 31. Establishment trends the two zones sampled in the Little Bald Hills. Blue panels represent Zone 1 (plots >200m from Jeffrey pine savanna), red panels represent Zone 2 (plots <200m from Jeffrey pine savanna). Trees that made up <4% of the trees cored are not shown. PsMe= *Pseudotsuga menziesii* NoLi=*Notolithocarpus densiflorus*

The histograms suggest that although some old trees are present in the LBHs, most of the forest established after 1860. The oldest trees at the site were sampled from plot 1 and these were not wolf trees (DBH =84.5) nor were their growth rates indicative of open-canopied environments.

Linear regressions indicated no relationship between the earliest establishment date of the plots (using the average establishment date of the three oldest trees in the plot) and environmental variables such as aspect and slope (R^2 adj=0.058, $F=1.584$, $P=0.24$). Other factors influenced tree establishment dates in different plots. T-tests indicated no significant differences between zones for any variables tested (Table 14).

Table 14. Differences in plot variables between zones were examined using t-tests

Variable	P values
Shannon's diversity index (species)	0.10
Oldest tree	0.60
Average DBH	0.67
Distance to a Tolowa settlement (m)	0.06
Aspect	0.51
Slope	0.22
Distance to nearest wolf tree (m)	0.27
Dead wood (%)	0.89
Shannon's diversity index (age)	0.54

Distribution of wolf trees and Jeffrey pine trees

The location of 70 wolf Douglas-fir trees and one Port-Orford cedar were recorded (Figure 32). The DBH of these trees and whether or not they were snags was also noted. The mean DBH was 144 cm (s.d.= 30.9 cm). Diameters ranged from 97 to 249 cm. The farthest noted wolf tree from the Jeffrey pine savanna was 446 m. The majority (75%) of the wolf trees encountered were <200m from the Jeffrey pine savanna.

The location and DBH of twenty-five Jeffrey pine trees encountered were also recorded. The mean DBH was 70 cm (s.d = 21.2 cm). Diameters ranged from 28 to 99 cm. Just five of these trees occurred more than 200 m from the Jeffrey pine savanna area and no Jeffrey pine tree was found more than 436 m from the Jeffrey pine savanna. More than 70% of the Jeffrey pines encountered were <200m from the Jeffrey pine savanna. It must be noted that this map is not an exhaustive survey of Jeffrey pine trees in the area, but it does highlight the general distribution of Jeffrey pine in the surveyed area.

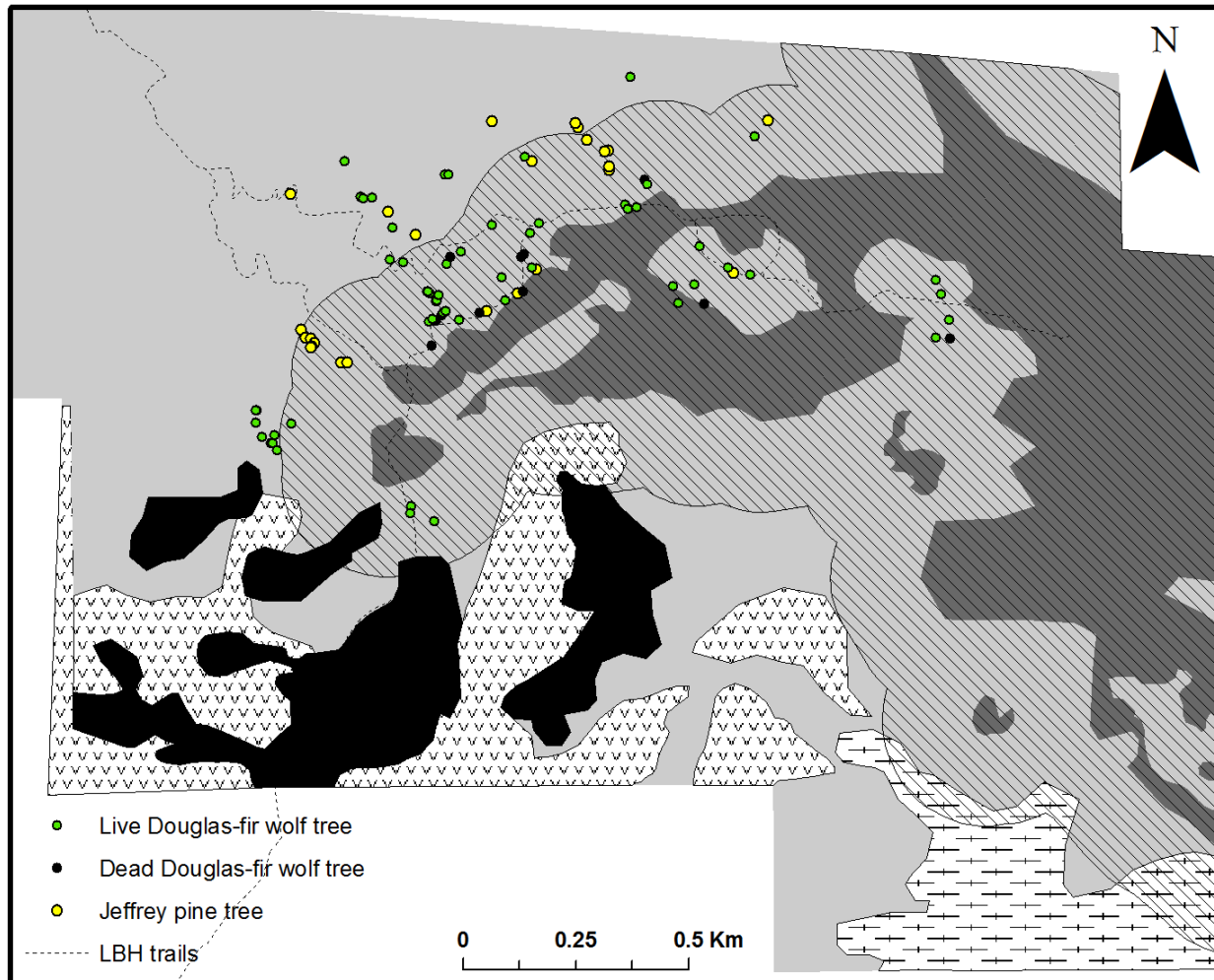


Figure 32. Douglas-fir wolf tree and Jeffrey pine distribution in the study area. Live Douglas-fir wolf trees are represented by green dots and snags are represented by brown dots. Jeffrey pine trees are represented by yellow dots.

Discussion

The stands were overwhelmingly dominated by Douglas-fir and tanoak. There were pockets of tanoak-dominated forest throughout the LBHs, particularly in the area closer to the Jeffrey pine savanna (Appendix C). There was an apparent increase in recruitment of tanoak from 1900. Although we can only speculate, this increase is perhaps due to a removal of tribal orchard management. The Tolowa's population had fallen to 120 by 1910 from around 1000 tribespeople at the time of Anglo-European contact (Bearrs 1969). Western settlers and Native Americans began clashing in the 1850s and clashes with the Tolowa in particular started in 1854 (Bearrs 1969). This perturbation likely resulted in a dramatic change in land management, including abandonment of the Little Bald Hills. The primary food sources, tanoak and chinquapin, may have been much more prevalent before 1850. The present restricted distribution of these two species may have resulted from the removal of Native American management practices in these stands. Both species were economically important for the Native Americans. Removing these management practices likely favored Douglas-fir encroachment from surrounding forests.

Much of the Douglas-fir forest at this site established recently, as 98% of the trees cored were <160 years old. This expansion of Douglas-fir forest may have occurred as a result of significant societal shifts in the mid-1800s. In addition to tribal changes, lumbering commenced in Del Norte County in 1853 with one mill in Crescent city and one mill on the Smith River (Bearrs 1969). The Del Norte County region was a stronghold for the Tolowa tribe who were moved to a reservation in Hoopa Valley in 1868 (Bearrs 1969). The Little Bald Hills region was important economically for the Tolowa. The region is listed as a site of economic importance for acorn gathering and hunting (Drucker 1937). In the late 1800s, the area was economically important for the Western settlers. The Kelsey Trail was built through the site in 1855 and was an important trade route between Crescent City and mining areas in the Klamath Basin. The trail was active until the 1880's (Rohde and Rohde 1994). A shift from Native American management could explain the increase in tree establishment rates from the 1850s onwards (see Figure 32). The presence of trees between 250 and 350 years old at the site with diameters of 85 to 145 cm suggests that Douglas-fir stands were restricted to a much smaller area within the site, perhaps by frequent burning by the Native Americans. A small area of Douglas-fir forest around Plot 1 (Appendix B) may have been present on site for hundreds of years with the expansion of this forest taking place around the 1850s.

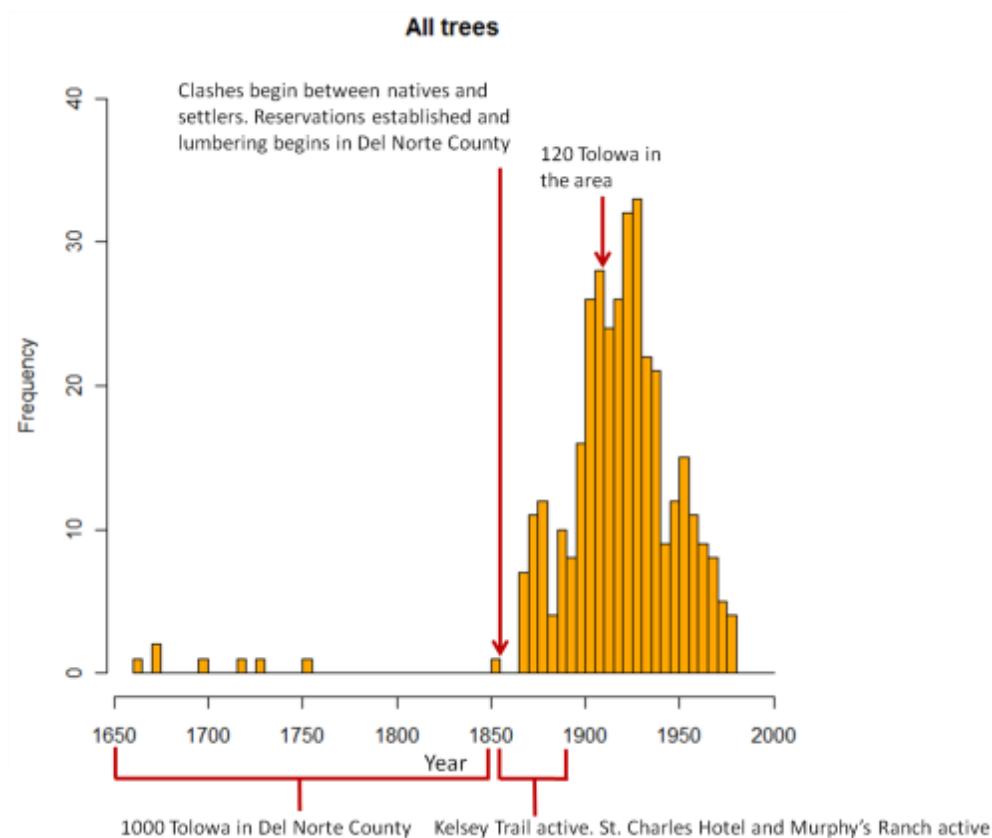


Figure 33. Historical events in the Little Bald Hills in relation to tree establishment dates.

The distribution of wolf trees indicates that there may have been a much greater area of savanna in the past. Evidence from this research indicates that even hundreds of years ago the open areas were patchy (reinforced by Sahara and Jules, *this report*).

Portions of the LBHs have been forested for hundreds of years, but there was an expansion of Douglas-fir and tanoak recruitment between 1860 and 1930. This may have been due to the removal of specific management practices associated with the Tolowa tribes that used the area until this period.

Conclusions and Management Implications

The comparatively open forests, woodlands and grasslands of the Bald Hills and Little Bald Hills are changing rapidly, primarily through woody species encroachment and its effects. There are several alternate and nonexclusive hypotheses for the changes, including cessation of Native American management regimes, increased ground disturbance associated with the legacies of logging and ranching industries, fire exclusion, atmospheric CO₂ enrichment, and possible climatic shifts.

Throughout these woodland communities, there is evidence of historically more open environments and associated values. While the exact mechanisms of forest ingrowth are not fully known at all locations, our research into the fuel dynamics of the Bald Hills suggest that encroachment by conifers changes the fundamental distribution of fuels, including type, size, density, and arrangement, having direct implications for intensity, severity, and spread of surface fires. Encroachment appears to suppress the accumulation of fine grassy fuels, reducing the likelihood of fire spread under conifer-encroached woodlands. This shift in the probability of fire is likely to foster a positive feedback loop that favors further encroachment with conifers leading ultimately to full replacement of oak woodlands and grasslands, and supporting the concept that the Bald Hills landscape can become dominated primarily by coniferous forests at the expense of the species-rich grassland and open woodland habitats that historically dominated these landscapes.

In the Little Bald Hills, we recorded evidence of episodic invasion in both Jeffrey pine woodlands and Douglas-fir forests, which suggests that resources associated with open environments (grasslands and open woodlands) will likely be lost sometime in next century, unless wildland fire, climatic shifts, or active management interrupt current rates and patterns of succession. These patterns were not homogeneous, as low-elevation, north-west facing slopes have been encroached more quickly than flatter, more south-easterly facing areas, which corresponds well to field reconnaissance of the ridgetop areas that remain savanna. Our results also indicate eventual disappearance of grassland areas in the next several decades. If maintenance of the ecological, cultural, and recreational values of these woodlands/grassland habitats is desired, a number of active and passive management approaches might be appropriate. These efforts might include prescribed fire, targeted mechanical treatments (cutting or girdling trees or shrubs), and wildland fire management.

At present, the active prescribed fire program in the Bald Hills is a major activity in the park. The conifer encroachment issues in these woodlands are well documented and prescribed fire is presently the preferred approach to maintaining this habitat in general. This study confirms that when transition to Douglas-fir forest proceeds beyond certain thresholds, ignition and spread of surface fire prove challenging. It may be appropriate to pursue more aggressive mechanical treatments (conifer harvesting) in areas where prescribed fire is unlikely to be effective. Alternatively, a wildland fire policy to allow wildfire to burn in the forested areas of the Bald Hills might be the only passive means to halt and possibly reverse conifer encroachment. Fire records suggest it is unlikely that such a natural ignition would occur on management time

scales. Of course, this option carries its own risks as it is quite possible that wildland fire could spread into the adjacent private timberlands.

In the Little Bald Hills, a similar array of management alternatives exists, but so far, much less has been done. Within the open grasslands and Jeffrey pine woodlands, smaller tree seedlings predominate, and grassy fuels are still sufficiently abundant that the application of prescribed fire, perhaps coupled with lopping or felling of smaller seedlings and saplings, could help to halt or reverse infilling of trees. Our study suggests that such efforts could be targeted towards areas where fire-induced seedling mortality would be high, and where important plant assemblages (i.e., rare plant populations or knobcone pine stands) are likely to gain the most benefit. It seems safe to propose that an integrated fire effects research project that imposes prescribed-fire and / or mechanical treatments and rare plant responses would be valuable at this time. Such an effort would be a great complement to this study.

Within the Douglas-fir forests of the LBH, we noted abundant evidence of forest change, but because of site differences (these are peripheral to the serpentine sites on top of the LBH), and more advanced forest succession, it is unclear whether the forests contain potential habitat for rare plants or other botanical resources. Prescribed fire management may not be an option, given the size of the trees and the magnitude the changes that have already occurred. These forests might be targets for wildland fire management. The ridgetop setting of the LBH is likely affected by offshore winds at periodic intervals, and it seems reasonable to hypothesize that high intensity fire originating in the drier inland forests adjacent to the LBH, capable of killing the invading Douglas-fir trees occurs at least occasionally. Such a fire could potentially create more open conditions and burn knobcone pine stands, allowing reinitiation of new stand cohorts. Of course such events might also kill a number of the older wolf trees. As a component of a wildland fire approach, it might be desirable to remove some of the smaller trees around the older wolf trees in anticipation of such an event. Alternatively, mortality of these older trees may create valuable snag habitat.

To evaluate the wildland fire options at both locations, fuel moisture and climate characterizations might be helpful in determining how likely wildfire is to spread from desired to undesired locations. For example, the LBH is a ridgetop location about 14 km from the coast and at 450-550 m elevation. This location inland and above the marine inversion which characterizes the redwood forest zone, is much more likely to receive warm, dry winds in the summer and to carry fire under offshore wind events in late summer and fall. The redwoods just downslope, typically in concave landscape conditions within the fog zone, are likely to be more mesic and resistant to fire spread. Would fires initiated in the LBH spread into the redwoods of Jedediah Smith State Park or onto adjacent private lands? Obviously this question would be very important for considering a wildland fire option. Modeling of wildland fire scenarios in the LBH could be fostered by detailed fuel moisture measurements across seasons, coupled with modeling scenarios to evaluate the risk of spread.

In conclusion, this study suggests that many of the ecological values of the Little Bald Hills and Bald Hills have been and will continue to be affected by the forest succession in the absence of active park management. Park managers are concerned that the woody species encroachment we have recorded is an unnatural response to past changes in natural and cultural disturbance

regimes. This study provides demographic evidence consistent with this view of rapid change in the last 200 years. Moreover, it suggests that these uniquely open environments will continue to convert to forest in the absence of management actions. Future research may help to target management efforts and evaluate their effectiveness. Specifically, research into the habitat needs and likely prospects of rare plant populations, the impacts and success of prescribed fire, mechanical treatments, or other small scale active management efforts, and modeling and other studies to evaluate the risks of wildland fire would complement the information base developed in this and previous studies.

Project Deliverables

Research papers

Eamon A. Engber, J. Morgan Varner, Leonel A. Arguello, Neil G. Sugihara. 2011. The effects of conifer encroachment and overstory structure on fuels and fire in an oak woodland landscape. *Fire Ecology* 7(2) 32-50.

Eamon A. Engber and J. Morgan Varner. *In review*. Reversing conifer encroachment with prescribed fire: effects of post-fire injury. *Submitted to Restoration Ecology*.

Theses

Eamon A. Engber. Fuelbed heterogeneity, flammability, and restoration of historically fire-frequent oak woodlands with fire. 2010. Master's Thesis. Humboldt State University.

E. April Sahara. 2012. Assessment and prediction of tree encroachment into a serpentine savanna. 2012. Master's thesis. Humboldt State University, Arcata, CA.

Oral Presentations

Engber, E.A. Overstory structure effects on fuelbed components in a frequently burned oak woodland. International Association of Wildland Fire 3rd Fire Behavior and Fuels Conference. October 25 – 29, 2010. Spokane, WA.

Sahara, E. April. Evaluating tree encroachment in the Little Bald Hills (Redwood National and State Parks) using historical photos and dendroecological techniques. California Native Plant Society Conservation Conference. January 12-14, 2012. San Diego, CA.

Sahara, E. April, Daniel Sarr, and Erik S. Jules. *Accepted*. An assessment of tree encroachment into a serpentine pine savanna using remote sensing and dendroecological techniques. Ecological Society of America Annual Meeting, August 5-10, 2012, Portland, OR.

Engber, E.A. Restoring Douglas-fir invaded grasslands with prescribed fire: effects of sapling size and post-fire injury. September 9, 2010. Redwood National Park, Orick, CA.

Sullivan, C. Composition and vegetation patterns in the Little Bald Hills, Redwood National Park. Spring 2011, Southern Oregon University, Ashland, OR.

Sullivan, C. Composition and vegetation patterns in the Little Bald Hills, Redwood National Park. Spring 2011, Redwood National Park, Orick, CA.

Poster presentations

Engber, E. and J.M. Varner . 2009. Fuelbed heterogeneity in a frequently burned oak woodland: how stand structure affects fire behavior. Association for Fire Ecology 4th International Fire Congress, November 30 – December 4, 2009. Savannah, GA.

Engber, E. and J.M. Varner. 2008. Snag and coarse woody debris abundance and characteristics in a frequently burned *Quercus garryana* woodland. (*The International Association of Wildland Fire: The '88 Fires, Yellowstone and beyond*. Jackson Hole, Wyoming. September 22-27th, 2008).

Sahara, E. April. 2010. Assessing vegetation change in a Jeffrey pine savanna, 1973 – 2005. Humboldt State University College of Natural Resources and Sciences Poster Session. Arcata, CA.

Other reports

Sullivan, C. 2011. Forest habitat diversity provides insights into the past at the Little Bald Hills, Redwood National Park. Klamath Kaleidoscope, Spring/Summer, pg. 7.

Sahara, E.A. and E.S. Jules. 2012. Little Bald Hills: a disappearing gem? Kaleidoscope Winter/Spring Issue 6-7.

Engber, E.A., and J.M. Varner. 2008. (Extended Abstract) Snag and coarse woody debris abundance and characteristics in a frequently burned *Quercus garryana* woodland [abstract]. Page 41 in R.E. Masters, K.E.M. Galley, and D.G. Despain (eds.). *The 88 Fires: Yellowstone and Beyond*. Tall Timbers Miscellaneous Publication No. 16, Tall Timbers Research Station, Tallahassee, Florida, USA.

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Appendices

Tree species	12	9	12	11	10	14	15	13	8	2	7	1	7	13	6	14	11	8	1	5	4	9	6	5	2	10	15	4	3	3
<i>Pseudotsuga menziesii</i>	5	5	5	5	5	5	5	5	5	4	4	5	5	4	4	4	4	3	3	5	4	4	3		4		3		1	3
<i>Chamaecyparis lawsoniana</i>							1	1	2	2	2	1	2	2	1	2	1	1	1	2	3		4	1					5	
<i>Umbellaria californica (tree)</i>					4	3	4	2	3	3	2				1	1														
<i>Lithocarpus densiflorus</i>	1	1															1	3	3	1	1	4	1	5	2	5	2			
<i>Chrysolepis chrysophylla</i>																						1			2	3	5			
<i>Pinus attenuate</i>			1												1													1		2
<i>Pinus jeffreyi</i>															1	2	2	1										1		
<i>Arbutus menziesii</i>																								2	1					
Shrubby species																														
<i>Vaccinium ovatum</i>	2	1	2	2	1			1	1	2	1	2		2	2	1	2	3	3	5	2	5	2	3	3	2	4	4	1	2
<i>Lithocarpus densiflorus</i>		2	2	2	2	2	1	2		2		2			2	2	2		1			2	2	2	2		2	1	3	
<i>Rhododendron macrophyllum</i>	1	1	4	4	2	2	2	2		2		4	4		2	1			2	2	3	2		2	2		1	1		5
<i>Polystichum munitum</i>	1	1			1	1	2	1	2	2	2	1	1	2	1	2	1	1	1	2	1	1	2			1	1		2	
<i>Rhamnus californica</i> var. <i>occidentalis</i>		1	2	1		1	2	1	1	2	2	2	1		2	2	2	2			2							2	1	
<i>Vaccinium parviflorum</i>			1			1	1		1		2	1	1	2	1	1	2		1		1			1				1	1	
<i>Holodiscus discolor</i>			1			1	1		1	1	1			1		2												1		
<i>Rhododendron occidentale</i>					1	2	1	2		2					1						2									1
<i>Gaultheria shallon</i>				1		1						1	1						1								2	1		
<i>Umbellaria californica (shrub)</i>										3	3			1			1												1	
<i>Berberis aquifolium</i>									1									1				1								
<i>Quercus vaccinifolia</i>			1					1							2															1
<i>Rubus ursinus</i>										1					1				1											
<i>Lithocarpus densiflorus</i> var. <i>echinoides</i>	2		1								2				1			2												
<i>Arctostaphylos columbiana</i>			1										1					2												
<i>Amelanchier alnifolia</i>											1							1				1								

Appendix A. Plant species data from thirty plots. Plants that occurred in just one plot were excluded. Plots in bold with grey background were from Zone 1 and white plots from Zone 2. Abundance is based on the Braun-Blanquet scale.

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