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LETTER

# The impact of land ownership, firefighting, and reserve status on fire probability in California

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## Abstract

The extent of wildfires in the western United States is increasing, but how land ownership, firefighting, and reserve status influence fire probability is unclear. California serves as a unique natural experiment to estimate the impact of these factors, as ownership is split equally between federal and non-federal landowners; there is a relatively large proportion of reserved lands where extractive uses are prohibited and fire suppression is limited; and land ownership and firefighting responsibility are purposefully not always aligned. Panel Poisson regression techniques and pre-regression matching were used to model changes in annual fire probability from 1950–2015 on reserve and non-reserve lands on federal and non-federal ownerships across four vegetation types: forests, rangelands, shrublands, and forests without commercial species. Fire probability was found to have increased over time across all 32 categories. A marginal effects analysis showed that federal ownership and firefighting was associated with increased fire probability, and that the difference in fire probability on federal versus non-federal lands is increasing over time. Ownership, firefighting, and reserve status, played roughly equal roles in determining fire probability, and were found to have much greater influence than average maximum temperature (°C) during summer months (June, July, August), average annual precipitation (cm), and average annual topsoil moisture content by volume, demonstrating the critical role these factors play in western fire regimes and the importance of including them in future analysis focused on understanding and predicting wildfire in the Western United States.

## Introduction

Globally, wildfires have increased in both number and severity (Pechony and Shindell 2010, Liu *et al* 2010, Flanigan *et al* 2013). This increase is even more pronounced in the western United States (Agee and Skinner 2005, Westerling *et al* 2006, Dennison *et al* 2014, Abatzoglou and Williams 2016). While fires are a natural part of most western ecosystems, current regimes vary greatly from historic baselines in terms of both frequency and severity (Hessburg *et al* 2015, Miller *et al* 2012b, Westerling 2016). Thousands of homes have been destroyed (CAL FIRE 2011), billions of fire suppression dollars have been spent (USFS 2015, CAL FIRE 2014, NIFC 2016), habitat has been destroyed (Stephens *et al* 2016, McKenzie *et al* 2004), vegetation types have been converted (Collins and Roller 2013, Regan *et al* 2010, Zedler *et al* 1983), smoke has impacted public health (Liu *et al* 2016, Williamson *et al* 2015), and lives have been lost.

Although climate change plays an undeniable role in this increase (Westerling *et al* 2006, Rhodes and Baker 2008, Preisler *et al* 2011, Miller and Safford 2012, Berner *et al* 2017), its exact contribution is unclear. Factors such as firefighting, vegetation management, and land ownership are known to impact fire in complex ways (Westerling and Bryant 2008, Miller *et al* 2012a, Barbero *et al* 2014, Parks *et al* 2016), though there is limited research on the role each plays in influencing fire probability. Teasing these drivers apart has proven to be challenging, yet understanding their impacts is crucial to management and policy.

**Table 1.** Hectares of land where fire-fighting responsibility has been traded in the 'balance' of acres. Arrangement for federal and state fire protection in California in selected vegetation types.

| Vegetation type | Balanced[a] hectares | Non-balanced hectares | Balanced % total |
|---|---|---|---|
| Forest | 1 022 874 | 14 455 176 | 15% |
| Rangeland | 466 626 | 15 997 501 | 7% |
| Shrubland | 936 899 | 12 122 251 | 16% |
| Forestnoncomm[b] | 41 479 | 1 737 142 | 6% |

[a] 'balanced' hectares are those where ownership and firefighting falls under different agencies, e.g. federal lands with state firefighting.
[b] Forest without commercial species.

Methodologically, disaggregating the impacts of land ownership and firefighting is difficult. Federal ownerships—especially reserved lands—often have vastly different ecological characteristics than private lands: they are generally less productive, steeper, and more isolated than non-federal ownerships. Since these factors also influence wildfire occurrence, comparisons of federal and non-federal lands are compromised by different baseline fire regimes. Likewise, federal agencies typically only fight fires on federal lands, while state agencies focus on non-federal lands, preventing a clear comparison of the possible influence of federal and state firefighting.

California provides a unique opportunity to analyze how land ownership, firefighting, and reserve status affect wildfire trends. First, half of wildlands in California are in non-federal ownership, providing ample areas to compare federal and non-federal lands. Second, there are considerable areas of reserved lands—designated areas where vegetation management activities and fire suppression actions are limited (USDOI and USGS 2016, Christensen *et al* 2016)—on both federal and non-federal ownerships.

California also serves as an ideal 'natural experiment' due to a policy referred to informally as the 'balance of acres' arrangement (aka the Cooperative Fire Protection Agreement; see Giambattista 2005). Under this agreement, state and federal agencies trade firefighting responsibility to maximize efficiency, particularly in areas where a legacy of 19th century railroad grants has created 'checkerboard' ownership patterns of federal and non-federal ownerships. As a result, some federal lands are protected by state fire protection and vice versa (table 1). This produces four combinations of ownership and firefighting: federal lands with federal firefighting, federal lands with state firefighting, non-federal lands with state firefighting, and non-federal lands with federal firefighting. In sum, with comparable areas of federal and non-federal ownerships, large areas of reserve as well as non-reserve land across federal and non-federal ownerships, and areas with traded firefighting responsibilities on reserve and non-reserve lands, California is an ideal area to test for their effects on fire probability.

Using a combination of pre-regression matching and panel Poisson models, coupled with spatially explicit data on fire parameters, biophysical drivers of fire, vegetation, and human settlements, we estimated the relative effects of land ownership, firefighting, and reserve status on average fire probability in California from 1950–2015 in reserve and non-reserve areas across four different vegetation types (forest, shrublands, rangelands, and forest without commercial species). Specifically, for each vegetation type, ownership, firefighting responsibility, and reserve status we asked:

1. How are ownership, firefighting, and reserve status associated with annual fire probability from 1950–2015?
2. How is ownership related to fire probability over time?
3. How is firefighting responsibility related to fire probability over time?

## Methods

### Study area

Our study area was the entire state of California, USA, with natural vegetation in forest, rangelands, or shrublands. For the purpose of our study, we considered four vegetation types: forests (6 872 681 ha), rangelands (6 940 591 ha), shrublands (5 842 604 ha), and forests without commercial species (i.e. forest where management would not be profitable) (744 476 ha) (figure 2, table S3).

### Data

Vegetation types were identified using the California Department of Forestry and Fire Protection's (CALFIRE) Fire and Resource Assessment Program's (FRAP) 'FVEG' map (table S4). This dataset uses the California Department of Fish and Wildlife's California Wildlife Habitat Relationships System (CWHR) vegetation classification[3], and contains over 59 wildlife habitats (table S4). Habitats were combined into broader vegetation types based on CALFIRE's methodology (figure 2, table S4). Urban, agricultural, and desert lands were excluded from our analysis.

Federal and non-federal ownership was based on CALFIRE's State Responsibility Area (SRA) designation (CAL FIRE 2012). Keeping in mind that the actual firefighting responsibility does not always coincide with the agency owning the officially designated 'Responsibility Area' as explained above, the SRA separates California wildlands into two relevant categories: Federal Responsibility Areas and State Responsibility Areas. Lands classified as State Responsibility Area exclude federal lands and those within city boundaries (CAL FIRE 2012). State Responsibility Area lands were classified as 'non-federal ownership' for the purposes of this study, and contain private, state, and

[3] www.wildlife.ca.gov/Data/CWHR/Wildlife-Habitats.

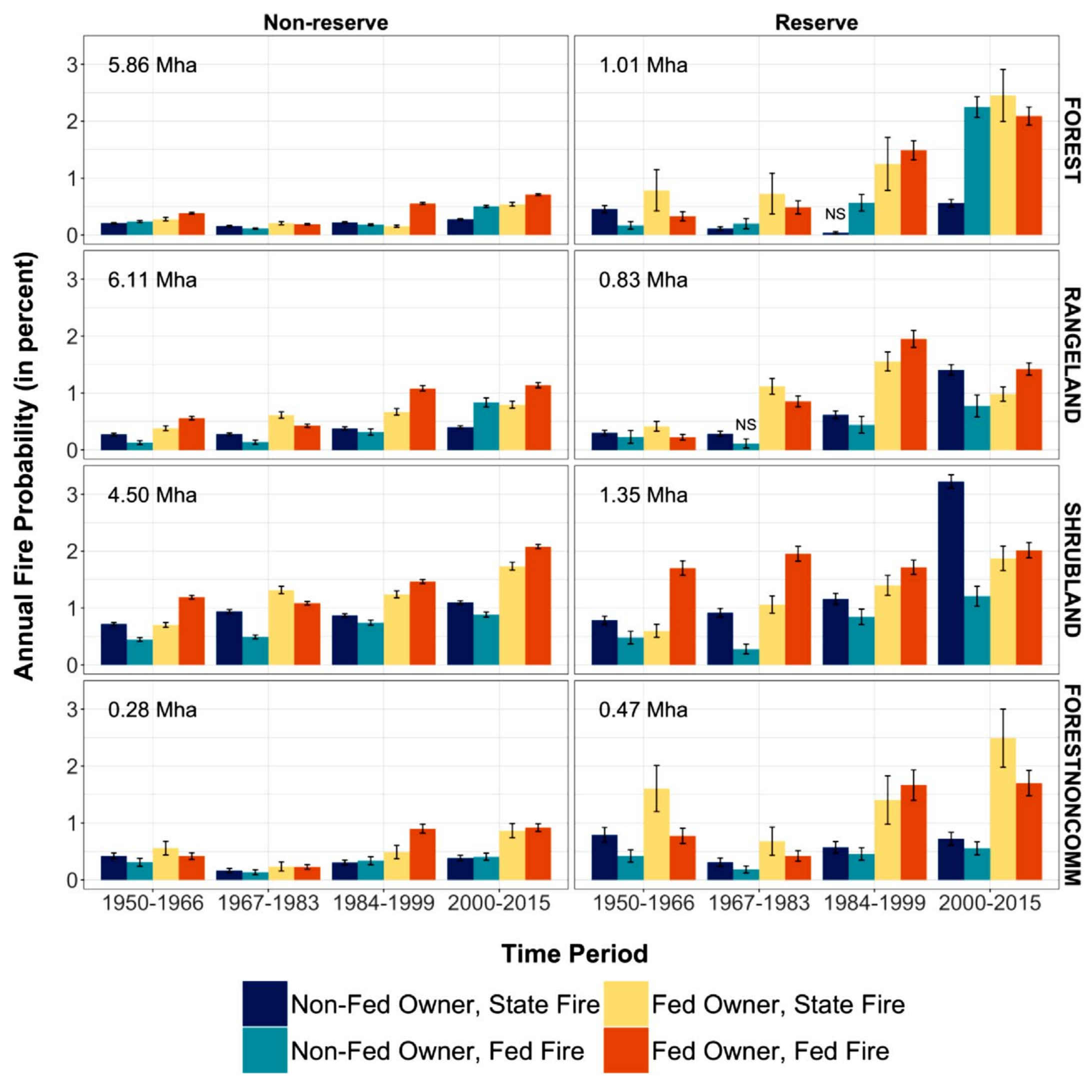


**Figure 1.** Annual fire probability (in percent) over four time-periods by ownership, firefighting agency, and vegetation type. Note: NS = Not statisically significant.

non-profit owned lands. In total, there were 10 136 190 ha of federal land in our dataset and 10 264 144 ha of non-federal land (table S3).

To classify a piece of land as 'reserved' or not, we used the United States Geological Survey's Protected Areas Database under the United States Gap Analysis Program (GAP) dataset to divide land into four categories: (1) lands primarily managed for biodiversity, (2) lands managed mostly for biodiversity, (3) lands that allow multiple uses, and (4) lands with no known mandate for protection (USDOI and USGS 2016). GAP 1 and 2 lands were combined into a 'reserved' category to represent areas where limited management is allowed. GAP 3 and 4 lands were classified as 'non-reserved' to represent areas where more extensive management is permitted, but not necessarily practiced. For forest lands, we applied the USFS Forest Inventory and Analysis program definition of 'reserved forest land' as 'land permanently reserved from wood products utilization through statute or administrative designation' (Christensen *et al* 2016).

Actual firefighting responsibility was determined using the Direct Protection Area designation (figure 2, table S4), which categorizes firefighting as state or federal. Our area of analysis included 10 251 477 ha of lands with federal firefighting and 10 148 875 ha with state firefighting (table S3). Direct Protection Area, State Responsibility Area, vegetation type, and reserve status were assumed to be unchanged throughout the study period. While there have likely been small changes in these variables over time, we were unable to find data documenting these changes. Therefore, we assume that any differences were not of a sufficient magnitude to impact our results.

Topographic, demographic, and climatic variables were used in our model to control for the influence that local geographic variation can exert on ecosystem susceptibility to wildfire (McKenzie *et al* 2004, Narayanaraj and Wimberly 2012, Westerling *et al* 2006). Variables included in the model were elevation (m), slope (degree), aspect, distance (m) to the closest major roadway, distance (m) to and population of the nearest city, average maximum temperature (°C) during summer

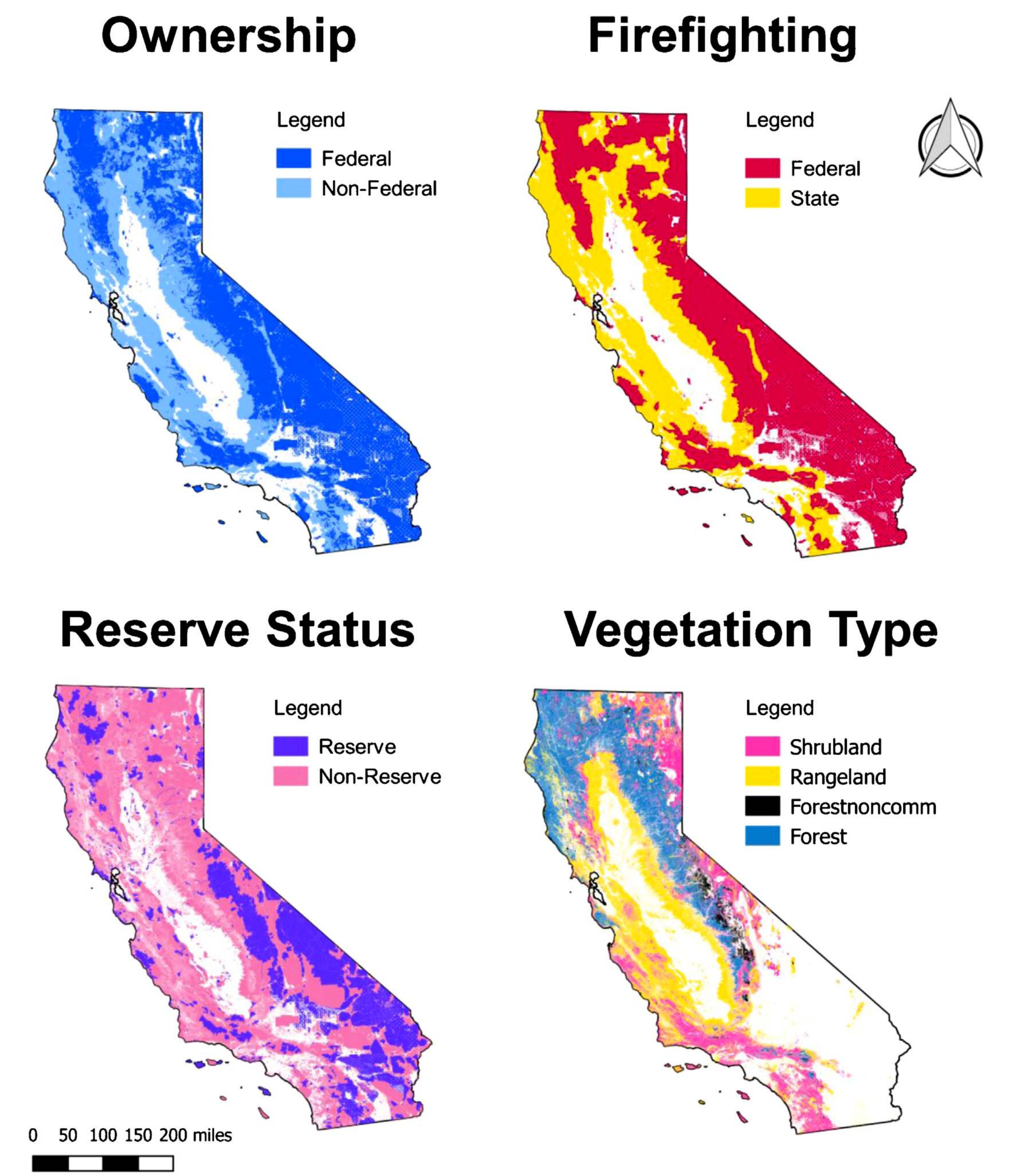


**Figure 2.** California spatial distribution of ownership, firefighting responsibility, reserve status, and vegetation type.

months (June, July, August), average annual precipitation (cm), and average annual topsoil moisture content by volume (mm) (table S4). Climatic variables were averaged over the following time periods: 1950–1966, 1967–1983, 1984–2000, and 2001–2015. These time periods also served as panels for the panel regression analysis. The use of four periods captured broad changes over time while avoiding the highly variable year-to-year rates of fire.

Fire history data were acquired from FRAP maps of historic fire perimeters in California (table S4). This dataset dates back to the 1880s, but due to concern about data accuracy in earlier periods, we started our analysis from 1950. Each fire recorded in the dataset included the year the fire burned and its perimeter. If an area burned more than once since 1950, it had multiple fire perimeters associated with it. In total the dataset contained over 13000 unique fires. The average fire size in the FRAP dataset was approximately 692 ha (1711 acres) (table S10 available at stacks.iop.org/ERL/13/034025/mmedia). Approximately 99% of the total area burned came from fires greater than 121.41 ha (300 acres) in size. Fires greater than 121.41 ha (300 acres) had an average size of 1265 ha (3310 acres) (table S10).

### Estimating the influence of ownership, firefighting, and reserve status on fire probability

For our analysis, we randomly drew 100 000 points each (using GIS software ArcMap) from forested and non-forested land types. The number of points was selected such that after the matching procedure there were sufficient observations in each strata of ownership, reserve status, and firefighting agency to provide robust estimation. We found that 100 000 pre-matched points proved sufficient for this purpose (table S10). For each point, we then assigned the spatial attributes described previously as well as how many times it had burned in each time period. This value—the number of times a point burned in each time period—became our dependent variable for the regression statistics. We chose random selection of the initial points in order to assure that our sample was a valid representation as a whole.

The panel structure of our data allowed us to address two issues which can conflate the relationship between cause and effect (Angrist and Pischke 2008). The first issue occurs if our data is subject to observable bias between data points in the different ownership, firefighting, and reserve status categories (Caliendo and Kopeinig 2008). We suspected that there may be large observable ecological differences between federal and non-federal land, as federal lands are often of lower site quality, at higher elevations, and further from developed areas than non-federal lands (Joppa and Pfaff 2009), potentially obscuring comparisons of fire probabilities on federal and non-federal lands (Andam *et al* 2008).

To control for differences in geographic characteristics by ownership, we used a matching algorithm (nearest neighbor matching without replacement) to find points under federal ownership that were geographically similar to points under non-federal ownership (Ho *et al* 2007). We decided to match on ownership, since firefighting and reserve status vary across ownership. We matched points individually for each vegetation type and matched observations on a number of characteristics including: elevation (m), slope (degree), aspect, distance (m) to the closest major roadway, distance (m) to and population of the nearest city, average maximum temperature (°C) during summer months (June, July, August), average annual precipitation (cm), and average annual topsoil moisture content by volume (mm), and GAP status. The matching algorithm essentially created four new datasets—each comprised of an equal number of federal and non-federal data points—that were similar in the matched covariates. This matching procedure greatly reduced bias in our dataset by allowing us to control for differences in the characteristics of federal and non-federal land. Specifically, we show reduction in the mean differences between matched and unmatched observations (table S5, table S6, table S7, table S8). The matching process reduced the overall number of points used for analysis (table S10), but ensured that federal and non-federal lands selected for analysis were similar in their characteristics, allowing us to better isolate the effects of our variables of interest on fire probabilities.

Second, it is likely that unobservable factors in our dataset played a role in determining the number of times a point burned. For instance, although we observed many factors about a particular point in time, there may have been unobserved fine scale or historical factors that we were not able to control for in our sample selection. Therefore, we used a random effects panel model, which can help control for time invariant factors that are not directly observable (Wooldridge 2012).

All told, using the datasets comprised of the observations that are matched between federal and non-federal lands, we estimated the following panel regression:

$$NB_{it} = B_0 + B_1 * Ownership + B_2 * Firefighting + B_3 * Reserve\ Status + B_4 * Times + B_{57-135}\ Controls + B_{16-30} * Interactions + u_i + e_{it}$$

where *NB* was the number of times a pixel burned per time period, *Ownership* represented federal or non-federal ownership, *Firefighting* designated either federal or state fire protection, *Reserve Status* indicated reserve or non-reserve lands, and the *Control* variable, which included the list of covariates we expected to influence the probability: elevation (m), slope (degrees), aspect, distance (m) to the closest major roadway, distance (m) to and population of the nearest city, average maximum temperature (°C) during summer months (June, July, August), average annual precipitation (cm), and average annual topsoil moisture content by volume (mm). We included all possible interactions between *Firefighting*, *Time Ownership*, and *Reserve Status*, in order to allow the effect of firefighting and ownership to vary across time and reserve status. The one exception to this is the model for non-commercial species where a lack of reserved lands makes this full saturation impossible. In this case we interact *Firefighting*, *Time* and *Ownership*. $u_i$ is the observation-specific (i.e. point) random effect, and $e_{it}$ is the error component for each point in each time period.

Our dependent variable was the number of times a given point burned in each panel, described as a count (i.e. 0, 1, 2,…). Therefore, we used a Poisson panel model specification to estimate the model. We choose the Poisson model over alternatives such as the negative binomial model because robust standard errors can be more easily calculated in Poisson models, and can typically correct for bias in standard errors even when the model is overdispersed (Wooldridge 2012).

Because this model included a number of interaction terms that are used to identify the effect of ownership and firefighting, and changes in these effects over time, directly interpreting the regression results can be difficult. Therefore, to better understand our results, we calculated annual fire probability for each combination of ownership, reserve status, and firefighting using the model coefficients as well as the marginal effects of ownership, firefighting, and reserve status in each time period (table 3, table 4). Standard errors of these calculations are estimated using the delta method (Oehlert 1992, Williams 2012). Full regression results are provided in tables S11–S14.

**Table 2.** Fire probability in time period 4 (2000–2015) and percent increase in fire probability in percent from time period 1 (1950–1965) to time period 4 (2000–2015) by vegetation type, ownership, and firefighting agency for non-reserve lands. Standard Error in parenthesis.

| Vegetation type | Ownership | Firefighting | Fire probability in percent 2000–2015 | % Increase in fire probability from 1950–1966 to 2000–2015 |
|---|---|---|---|---|
| FOREST | Non-Federal | State | 0.28 (0.01) | 33% |
| FOREST | Non-Federal | Federal | 0.5 (0.02) | 108% |
| FOREST | Federal | State | 0.54 (0.03) | 93% |
| FOREST | Federal | Federal | 0.71 (0.02) | 87% |
| RANGELAND | Non-Federal | State | 0.4 (0.02) | 48% |
| RANGELAND | Non-Federal | Federal | 0.84 (0.08) | 546% |
| RANGELAND | Federal | State | 0.8 (0.06) | 111% |
| RANGELAND | Federal | Federal | 1.14 (0.04) | 104% |
| SHRUBLAND | Non-Federal | State | 1.1 (0.03) | 53% |
| SHRUBLAND | Non-Federal | Federal | 0.89 (0.04) | 98% |
| SHRUBLAND | Federal | State | 1.73 (0.07) | 147% |
| SHRUBLAND | Federal | Federal | 2.08 (0.04) | 75% |
| FORESTNONCOMM | Non-Federal | State | 0.38 (0.05) | −10% |
| FORESTNONCOMM | Non-Federal | Federal | 0.41 (0.06) | 32% |
| FORESTNONCOMM | Federal | State | 0.87 (0.12) | 55% |
| FORESTNONCOMM | Federal | Federal | 0.92 (0.07) | 119% |

**Table 3.** Marginal effect of federal ownership, federal firefighting, reserve status, and climate on annual fire probability (AFP). Marginal effects are calculated from period 1 (1950–1966) to period 4 (2000–2015); all other variables in model are held at their means. Standard errors are calculated using the delta method.

| | Percentage point change in AFP | Std. error | *Z*-statistic | *p*-value |
|---|---|---|---|---|
| **Forest** | | | | |
| Effect of federal ownership | 0.15 | 0.01 | 13.79 | $p<0.01$ |
| Effect of federal firefighting | 0.14 | 0.01 | 12.92 | $p<0.01$ |
| Effect of reserved status | 0.54 | 0.04 | 13.17 | $p<0.01$ |
| Effect of temperature[a] | 0.03 | 0.00 | 14.47 | $p<0.01$ |
| Effect of soil moisture | −0.03 | 0.00 | −14.51 | $p<0.01$ |
| Effect of precipitation | 0.00 | 0.00 | −8.59 | $p<0.01$ |
| **Rangelands** | | | | |
| Effect of federal ownership | 0.30 | 0.03 | 11.87 | $p<0.01$ |
| Effect of federal firefighting | 0.17 | 0.03 | 6.31 | $p<0.01$ |
| Effect of reserved status | 0.18 | 0.03 | 6.27 | $p<0.01$ |
| Effect of temperature[a] | 0.02 | 0.00 | 5.20 | $p<0.01$ |
| Effect of soil moisture | −0.02 | 0.00 | −8.16 | $p<0.01$ |
| Effect of precipitation | 0.00 | 0.00 | 12.54 | $p<0.01$ |
| **Shrublands** | | | | |
| Effect of federal ownership | 0.22 | 0.03 | 8.59 | $p<0.01$ |
| Effect of federal firefighting | 0.26 | 0.03 | 10.05 | $p<0.01$ |
| Effect of reserved status | 0.07 | 0.03 | 2.23 | 0.03 |
| Effect of temperature[a] | 0.05 | 0.00 | 15.54 | $p<0.01$ |
| Effect of soil moisture | −0.06 | 0.00 | −20.17 | $p<0.01$ |
| Effect of precipitation | 0.00 | 0.00 | 0.91 | 0.37 |
| **Forest without commercial species** | | | | |
| Effect of federal ownership | 0.17 | 0.04 | 4.39 | $p<0.01$ |
| Effect of federal firefighting | 0.11 | 0.04 | 2.56 | 0.01 |
| Effect of reserved status | 0.28 | 0.11 | 2.62 | 0.01 |
| Effect of temperature[a] | 0.03 | 0.01 | 4.14 | $p<0.01$ |
| Effect of soil moisture | −0.03 | 0.01 | −4.14 | $p<0.01$ |
| Effect of precipitation | 0.00 | 0.00 | 0.02 | 0.98 |

[a] Average maximum temperature during summer months (June/July/August).

### Validation, accuracy assessment, robustness check

We performed a number of robustness checks on our results. First, we were concerned that the results may be sensitive to our choice of four time periods. We ran two alternative models: one with a breakpoint in 1983 and another with a breakpoint in 2000. Results did not change quantitatively. In addition, we suspected that areas that had burned during the previous time period would be less likely to burn in the next period. We therefore included a lag term indicating if a pixel had burned in the past time period. The inclusion of this lagged variable did not change our results and the coefficient was not statistically different from zero, so we report regression results without it.

## Results

### Effects of land ownership, firefighting, and reserve status on annual fire probability

Using pre-regression matching to control for differences in geographic characteristics by ownership and

**Table 4.** Marginal effect of federal ownership and federal firefighting annual fire probability per time period for non-reserved lands. All other values are held at their means; standard errors are calculated using the delta methods. *P*-values are in parenthesis.

| | Time Period | | | | | |
|---|---|---|---|---|---|---|
| | 1967–1983 | | 1984–1999 | | 2000–2015 | |
| **Forests** | | | | | | |
| Effect of federal ownership | 0.07 | ($p < 0.01$) | 0.22 | ($p < 0.01$) | 0.21 | ($p < 0.01$) |
| Effect of federal firefighting | −0.01 | (0.43) | 0.21 | ($p < 0.01$) | 0.32 | ($p < 0.01$) |
| **Rangelands** | | | | | | |
| Effect of federal ownership | 0.35 | ($p < 0.01$) | 0.53 | ($p < 0.01$) | 0.15 | (0.01) |
| Effect of federal firefighting | 0.04 | ($p < 0.01$) | −0.11 | ($p < 0.01$) | 0.27 | ($p < 0.01$) |
| **Shrublands** | | | | | | |
| Effect of federal ownership | 0.25 | ($p < 0.01$) | 0.18 | ($p < 0.01$) | 0.32 | ($p < 0.01$) |
| Effect of federal firefighting | −0.05 | (0.23) | 0.37 | ($p < 0.01$) | 0.39 | ($p < 0.01$) |
| **Forests without Commercial Species** | | | | | | |
| Effect of federal ownership | 0.03 | (0.54) | 0.28 | ($p < 0.01$) | 0.37 | ($p < 0.01$) |
| Effect of federal firefighting | 0.01 | (0.83) | 0.31 | ($p < 0.01$) | 0.16 | (0.08) |

the regression analysis to control for local variables that influence fire probability, we found that average annual fire probability was nearly always higher for points with federal ownership, federal fire protection, and reserve status (table 3). This trend held across all vegetation types (figure 1, table S1). The increase in fire probability associated with the marginal effect of federal ownership and federal firefighting were very similar in magnitude for each vegetation type except rangelands, where the effect of ownership was nearly double the effect of firefighting (table 3). Reserve status had the largest effect on forest lands and the least effect on shrublands (table 3). Reserve status had a larger influence on fire probability than ownership or fire protection for forest lands and a lesser impact on shrublands. Non-federal, non-reserved forests with state fire protection had the lowest fire probability on average, with the exception of shrublands, where non-reserve lands with non-federal ownership and federal firefighting consistently had the lowest fire probability on average (figure 1, table 2, table S1).

### Effects of federal landownership on fire probability over time

Outside of reserved lands where vegetation management is usually not permitted, the effect of federal ownership on annual fire probabilities became more pronounced for all vegetation types from the second time period to the fourth (with the exception of rangelands) (table 4). For forest lands, federal ownership had little effect in the second time period (1967–1983), but by the fourth period (2000–2015), the effect was to increase annual fire probability by 0.20 percentage points. The results for forests without commercial species were even more pronounced. The increase in effect over time in shrublands was less extreme, with a final difference of 0.32 percentage points in the final period. Rangelands exhibited a different pattern, with federal landownership leading to higher annual fire probabilities in each time period, but the difference peaking in the third period (1984–1999) instead of the fourth (2000–2015) (table 4).

### Effects of federal firefighting on fire probability over time

The increase in fire probability associated with federal firefighting became stronger over time for forests, rangelands, and shrublands (table 4). For forests, rangelands, and shrublands, the magnitude of the effect increased from nearly zero in the second time period to 0.32, 0.27, and 0.39 percentage points, respectively, by the fourth. In forests without commercial species, the effect of federal firefighting was not as straight forward. While it increased from 0.01 in the second time period to 0.16 percentage points in the fourth, it peaked at 0.31 in the third period (table 4).

### Effect of climate variables on fire probability

In order to determine potential climate effects on annual fire probability over the study period (1950–2015), we included average maximum temperature (°C) during summer months (June, July, August), average annual precipitation (cm), and average annual topsoil moisture content by volume (mm) for each time period in our marginal effects analysis (table 3). These variables were found to have very little effect for each vegetation type compared to federal ownership, firefighting, and reserve status, with the exception of shrublands, where the effect of reserved status was very low (0.07) and therefore similar to the effect of temperature (0.05) and soil moisture (−0.06).

## Discussion

There is considerable discussion about what is driving increases in wildfires in the Western US, with scientists pointing to a host of potential drivers from increased human ignitions to fuel buildup to climate change (Dennison *et al* 2014, Schoennagel *et al* 2017, Westerling 2016, Littell *et al* 2016). Our results clearly demonstrate that non-federal lands have seen far less of an increase in fire probability than have federal lands. Our results also showed a consistent pattern of lands with federal firefighting—regardless of ownership or reserve status—nearly always having a higher fire probability than lands with state firefighting (figure 1, table 2, table S1).

The effect of federal ownership may be linked to changes vegetation management associated with ownership. From the 1950s to the 1970s, vegetation management practices on federal and non-federal lands were broadly similar (Stewart *et al* 2016). In non-reserve roaded areas, vegetation management activities such timber harvesting, non-commercial thinning of trees and shrubs, grazing by domesticated animals, and prescribed burning were commonly practiced if they produced net revenue for private owners or were supported by federal agencies (Dana and Fairfax 1980, Williams 1989). Beginning in the 1970s, parallel state and federal laws addressing environmental quality, clean water, clean air, and endangered species led to a greater analysis of the potential environmental and ecological impacts of typical practices and to increased emphasis on avoiding them (Burnett and Roberts 2015, Koontz 2007). This reduced the removal of timber and forage (both of which are also flammable fuels) on both federal and non-federal lands, with much larger declines in the removal of timber and forage on federal lands, leading to greater fuel accumulation on federal lands. (Stewart *et al* 2016, Spiegal *et al* 2016)

Substantial management changes on federal lands in California occurred again in the 1990s under the Northwest Forest Plan (USDA and USDOI 1994) and the California Spotted Owl guidelines (Verner *et al* 1992). These policies, designed to protect desired habitats for certain species and to reduce potential risks to other resources, led to substantial decreases in timber harvesting on federal lands (Stewart *et al* 2016, McIver *et al* 2015). Grazing on federal lands also declined approximately 30% over this time period (Wiles and Warren 2016, USFS 2016). This led to a divergence over time in the probability of active vegetation management between federal and non-federal ownerships.

Firefighting practices in California also diverged over time. California's state firefighting agency, CAL FIRE, has consistently stated that their 'fire mission is to protect natural resources on state responsibility areas [SRA] from damage by wildfire' (CAL FIRE 1990). Beginning in the 1960s, the National Park Service began allowing lightning-ignited fires to burn (van Wagtendonk 2007, Stephens and Ruth 2005). The federal shift away from a more aggressive suppression strategy was formalized in a major review of federal wildland fire management in 1995 that began with the 'protection of human life is reaffirmed as the first priority,' and highlighted that 'wildland fire, as a critical natural process, must be reintroduced into the ecosystem' (USDOI and USDA 1995). While state and federal fire agencies share common training, terminology, and incident command structures (National Wildfire Coordinating Group n.d.), the operational differences can be characterized by a relatively greater focus on aggressive direct attack by CAL FIRE (Haight and Fried 2007) compared with more indirect attack by federal agencies (Katuwal *et al* 2016, Stonesifer *et al* 2016).

A low fire probability may be more desirable where primary management objectives pertain to economic or carbon benefits (Berner *et al* 2017, Hicke *et al* 2013), but for lands where management goals are primarily ecologically driven (Stephens *et al* 2013, Hessburg *et al* 2015, Keeley and Safford 2016), increased fire frequency is often a goal for establishing more resilient ecosystems. That is, returning to pre-European settlement fire regimes by allowing fires to burn would necessitate an increase in area burned and an increase in fire probability. Stephens *et al* (2007) estimated that pre-European-settlement, 1 800 000 ha burned per year in California; our data showed approximately 240 000 ha burned per year on average in the most recent time period (2000–2015). The considerably higher fire probability on reserve lands across all vegetation types (figure 1, table S1) may suggest that federal and state land management agencies are successfully pursuing strategies to reintroduce fire into those ecosystems, or that a lack of vegetation management has increased the likelihood of fire. More information is needed to fully assess this.

As with any study, there are certain levels of complexity we were unable to control for. In our case, we have no information on the landscape structure and composition of historical vegetation types. While forest structure is generally similar between landownership types in California (Stewart *et al* 2016), we are unable to control for landscape scale differences which may impact fire frequency. Likewise this study did not examine the effects of fire severity. Without such an analysis, it is impossible to determine the full ecological implications of the increase in fire probability over time. At the time of this study, large-scale fire severity data was only available through the Monitoring Trends in Burn Severity (MTBS) program and did not include data for fires that burned solely on non-federal lands. Regional studies using MTBS fire severity data on federal lands have often found fire severity to be increasing (Miller and Safford 2012, Lutz *et al* 2009), but tree-mortality based classes and MTBS based classes had poor agreement with severity metrics in a study of National Forests in Oregon and Washington (Whittier and Gray 2016). Finally, we did not examine if ignition sources play a role in the differences described here. This is an area of potential future research, as examination of ignition sources in other Mediterranean regions has shown that ignition source can influence the timing, spatial distribution, and size of fires (Curt *et al* 2016).

A surprise in our result was the relatively minor effect of climate variables (average maximum temperature (°C) during summer months (June, July, August), average annual precipitation (cm), and average annual topsoil moisture content by volume (mm)) compared to ownership, firefighting, and reserve status (table 4). Previous studies have correlated increases in the length of fire season, summer temperature, fuel aridity, and droughts have played a significant role in increased fire probability in the western United States (Westerling *et al* 2006, Spracklen *et al* 2009, Barbero *et al* 2015, Abatzoglou and Williams 2016, USFS 2015, Westerling 2016). However, if we compare the effect of one degree Celsius increase in temperature (0.03) to the effect of federal ownership for forest lands (0.14), the corresponding increase in fire probability associated with switching all lands to federal ownership is equivalent to a 4.5 °C increase in temperature (calculated by dividing the effect of federal ownership by the effect of a 1 °C increase in temperature). Considering the wealth of scientific study showing substantial pre-climate change anthropogenic influence on fire probability (Skinner and Chang 1996, Graham *et al* 1999, Agee and Skinner 2005, Westerling and Bryant 2008, Miller *et al* 2012a, Collins 2014, Stephens *et al* 2007, Parsons and DeBenedetti 1979, Pyne 1982, Steel *et al* 2015), it is logical that anthropogenic influence in the form of choices with respect to vegetation management and firefighting continue to play a major role in fire probability even as our climate changes, though the relative role of climate will likely increase.

Our modeling techniques introduce novel ways of controlling for generic differences between federal and non-federal lands by using matching techniques that have been applied for impact evaluation in many other fields (Brandt *et al* 2015, Butsic *et al* 2011). Likewise, the use of a panel data structure allows for control of time invariant unobserved variables that may likewise obscure results. As with all models, there are some caveats. The structure of our data, where many points do not burn, suggest that a model that accounts for a preponderance of zeros such as a hurdle model or a zero inflated Poisson model may be a good fit. However, it is difficult to incorporate such models into a panel setting, and abandoning the panel structure of our data would force us to rely on the assumption that the error term associated with a given point is not correlated over time, a very unlikely case. Therefore, when faced with an imperfect model choice, we believe a panel Poisson model, coupled with pre-regression matching offered us the best choice.

## Conclusion

Our data showed continually increasing wildfire probability across all ownerships, firefighting agencies, reserve statuses, and vegetation types. While our analysis did not show a large effect from three climate variables (average maximum temperature (°C) during summer months (June, July, August), average annual precipitation (cm), and average annual topsoil moisture content by volume (mm)), its relative role may increase over time as our climate changes. However, our analysis suggests that ownership, firefighting decisions, and reserve status play a large role in wildfire probability in the study region.

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## References

Abatzoglou J T and Williams A P 2016 Impact of anthropogenic climate change on wildfire across western US forests *Proc. Natl Acad. Sci. USA* **113** 11770–5

Agee J K and Skinner C N 2005 Basic principles of forest fuel reduction treatments *Forest Ecol. Manage.* **211** 83–96

Andam K S, Ferraro P J, Pfaff A, Sanchez-Azofeifa G A and Robalino J A 2008 Measuring the effectiveness of protected area networks in reducing deforestation *Proc. Natl Acad. Sci.* **105** 16089–94

Angrist J D and Pischke J-S 2008 *Mostly Harmless Econometrics: An Empiricist's Companion* (Princeton, NJ: Princeton University Press)

Barbero R, Abatzoglou J T, Steel E A and Larkin N 2014 Modeling very large-fire occurrences over the continental United States from weather and climate forcing *Environ. Res. Lett.* **9** 124009

Barbero R, Abatzoglou J T, Larkin N K, Kolden C A, Stocks B, 2015 Climate change presents increased potential for very large fires in the contiguous United States *Int. J. Wildl. Fire* **24** 892–9

Berner L T, Law B E, Meddens A J H and Hicke J A 2017 Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States 2003–2012 *Environ. Res. Lett.* **12** 14

Brandt J S, Butsic V, Schwab B, Kuemmerle T and Radeloff V C 2015 The relative effectiveness of protected areas, a logging ban, and sacred areas for old-growth forest protection in southwest China *Biol. Conserv.* **181** 1–8

Burnett R D and Roberts L J 2015 A quantitative evaluation of the conservation umbrella of spotted owl management areas in the Sierra Nevada *PLoS ONE* **10** 1–18

Butsic V, Lewis D J and Ludwig L 2011 An econometric analysis of land development with endogenous zoning **87** 412–32

CAL FIRE 2011 *CAL FIRE Jurisdiction Fires, Acres, Dollar Damage, and Structures Destroyed* (California: Department of Forestry and Fire Protection) (http://cdfdata.fire.ca.gov/pub/cdf/images/incidentstatsevents_270.pdf)

CAL FIRE 2014 *Emergency Fund Fire Suppression Expenditures* (California: Department of Forestry and Fire Protection) (http://www.fire.ca.gov/fire_protection/downloads/SuppressionCostsOnepage.pdf)

CAL FIRE 2012 *What is the State Responsibility Area (SRA)?* (California: Department of Forestry and Fire Protection) (http://www.fire.ca.gov/firepreventionfee/)

CAL FIRE 1990 *Wildfire Activity Statistics* (California: Department of Forestry and Fire Protection) (http://www.fire.ca.gov/downloads/redbooks/2012Redbook/2012_Redbook_Complete.pdf)

Caliendo M and Kopeinig S 2008 Some practical guidance for the implementation of propensity score matching *J. Econ. Surv.* **22** 31–72

Christensen G A, Waddell K L, Stanton S M, Kuegler O and Editors T 2016 *California's Forest Resources: Forest Inventory and Analysis, 2001–2010* (Portland, OR) *Gen. Tech. Rep.* PNW-GTR-913 (US Dep. Agric. Forest Serv. Pacific Northwest Res. Stn. 293) (https://www.fs.usda.gov/treesearch/pubs/50397)

Collins B M 2014 Fire weather and large fire potential in the northern Sierra Nevada *Agric. Forest Meteorol.* **189–190** 30–5

Collins B M and Roller G B 2013 Early forest dynamics in stand-replacing fire patches in the northern Sierra Nevada, California, USA *Landsc. Ecol.* **28** 1801–13

Curt T, Fréjaville T and Lahaye S 2016 Modelling the spatial patterns of ignition causes and fire regime features in southern France: implications for fire prevention policy *Int. J. Wildland Fire* **25** 785–96

Dana S T and Fairfax S K 1980 *Forest Range Policy Its Development in the United States* (New York: McGraw Hill)

Dennison P E, Brewer S C, Arnold J D and Moritz M A 2014 Large wildfire trends in the western United States, 1984–2011 *Geophys. Res. Lett.* **41** 2928–33

Flannigan M, Cantin A S, de Groot W J, Wotton M, Newbery A and Gowman L M 2013 Global Wildland Fire Season Severity in the 21st Century *Forest Ecology and Management* **294** 54–61

FRAP 2003 *The Changing California: Forest and Range 2003 Assessment* (California: Department of Forestry and Fire Protection Fire and Resource Assessment Program) (http://frap.fire.ca.gov/data/assessment2003/toc)

Giambattista J 2005 A Primer: California's Wildland Fire Protection System (www.lao.ca.gov/2005/fire_protection/051205_fire_protection.htm)

Graham R T, Harvey A E, Jain T B and Tonn J R 1999 The effects of thinning and similar stand treatments on fire behavior in western forests *Gen. Tech. Rep.* PNW-GTR-463 (US Dep. Agric. Forest Serv. Pacific Northwest Res. Stn. 27) (www.scopus.com/inward/record.url?eid=2-s2–33367489&partnerID=40&md5=cf7c0b3e3853001917a5d4f0c28d3706) p 1–27

Haight R G and Fried J S 2007 Deploying wildland fire suppression resources with a scenario-based standard response model *INFOR* **45** 31–9

Hessburg P F *et al* 2015 Restoring fire-prone Inland pacific landscapes: seven core principles *Landsc. Ecol.* **30** 1805–35

Hicke J A, Meddens A J H, Allen C D and Kolden C A 2013 Carbon stocks of trees killed by bark beetles and wildfire in the western United States *Environ. Res. Lett.* **8** 35032

Ho D E, Imai K, King G and Stuart E A 2007 Matching as nonparametric preprocessing for reducing model dependence in parametric causal inference *Polit. Anal.* **15** 199–236

Jolly W M, Cochrane M A, Freeborn P H, Holden Z A, Brown T J, Williamson G J and Bowman D M J S 2015 Climate-induced variations in global wildfire danger from 1979–2013 *Nat. Commun.* **6** 7537

Joppa L N and Pfaff A 2009 High and far: Biases in the location of protected areas *PLoS ONE* **4** 1–6

Katuwal H, Calkin D E and Hand M S 2016 Production and efficiency of large wildland fire suppression effort: a stochastic frontier analysis *J. Environ. Manage.* **166** 227–3

Keeley J E and Safford H D 2016 *Fire as an Ecosystem Process Ecosystems of California* ed H A Mooney and E S Zavaleta (Berkeley, CA: University of California Press) 27–46

Koontz T M 2007 Federal and state public forest administration in the new millennium: revisiting Herbert Kaufman's the forest ranger *Public Adm. Rev.* **67** 152–64

Littell J S, Peterson D L, Riley K L, Liu Y and Luce C H 2016 A review of the relationships between drought and forest fire in the United States *Glob. Change Biol.* **22** 2353–69

Liu Y, Stanturf J and Goodrick S 2010 Trends in global wildfire potential in a changing climate *For. Ecol. Manage.* **259** 685–97

Liu Y, Goodrick S and Heilman W 2014 Wildland fire emissions, carbon, and climate: wildfire–climate interactions *Forest Ecol. Manage.* **317** 80–96

Liu J C, Mickley L J, Sulprizio M P, Yue X, Peng R D, Dominici F and Bell M L 2016 Future respiratory hospital admissions from wild fire smoke under climate change in the western US *Environ. Res. Lett.* **11** 124018

Lutz J A, Van Wagtendonk J W, Thode A E and Miller J D 2009 Climate, lightning ignitions, and fire severity in Yosemite national park, California, USA Int *J. Wildl. Fire* **18** 765–74

McIver C P, Meek J P, Scudder M G, Sorenson C B, Morgan T A and Christensen G A 2015 California's forest products industry and timber harvest, 2012 **66** *Gen. Tech. Rep.* PNW-GTR-908 (US Dep. Agric. Forest Serv. Pacific Northwest Res. Stn. 49) (https://www.fs.usda.gov/treesearch/pubs/49762)

McKenzie D, Gedalof Z, Peterson D L and Mote P 2004 Climatic change, wildfire, and conservation *Conserv. Biol.* **18** 890–902

Miller J D, Collins B M, Lutz J A, Stephens S L, van Wagtendonk J W and Yasuda D A 2012a Differences in wildfires among ecoregions and land management agencies in the Sierra Nevada region, California, USA *Ecosphere* **3** art80

Miller J D, Skinner C N, Safford H D, Knapp E E and Ramirez C M 2012b Trends and causes of severity, size, and number of fires in northwestern California, USA *Ecol. Appl.* **22** 184–203

Miller J D and Safford H 2012 Trends in wildfire severity: 1984–2010 in the Sierra Nevada, Modoc Plateau, and southern Cascades, California, USA *Fire Ecol.* **8** 41–57

Narayanaraj G and Wimberly M C 2012 Influences of forest roads on the spatial patterns of human- and lightning-caused wildfire ignitions *Appl. Geogr.* **32** 878–88

National Wildfire Coordinating Group Glossary A-Z 2017 NWCG (www.nwcg.gov/glossary/a-z)

NIFC 2016 *Federal Firefighting Costs (Suppression only)* (Boise, ID: National Interagency Fire Center) (https://www.nifc.gov/fireInfo/fireInfo_documents/SuppCosts.pdf)

Oehlert G W 1992 A note on the delta method *Am. Stat.* **46** 27

Parks S A, Miller C, Abatzoglou J T, Holsinger L M, Parisien M-A and Dobrowski S Z 2016 How will climate change affect wildland fire severity in the western US? *Environ. Res. Lett.* **11** 1–10

Parsons D J and DeBenedetti S H 1979 Impact of fire suppression on a mixed-conifer forest *Forest Ecol. Manage.* **2** 21–33

Pechony O and Shindell D T 2010 Driving forces of global wildfires over the past millennium and the forthcoming century *Proc. Natl Acad. Sci. USA* **107** 19167–70

Preisler H K, Westerling A L, Gebert K M, Munoz-Arriola F and Holmes T P 2011 Spatially explicit forecasts of large wildland fire probability and suppression costs for California *Int. J. Wildl. Fire* **20** 508–17

Pyne S J 1982 Fire in America *A Cultural History of Wildland and Rural Fire* (Princeton, NJ: Princeton University Press)

Regan H M, Crookston J B, Swab R, Franklin J and Lawson D M 2010 Habitat fragmentation and altered fire regime create trade-offs for an obligate seeding shrub *Ecology* **91** 1114–23

Rhodes J J and Baker W L 2008 Fire probability, fuel treatment effectiveness and ecological tradeoffs in western US public forests *Forest Sci. J.* **1** 1–7

Schoennagel T *et al* 2017 Adapt to more wildfire in western North American forests as climate changes *Proc. Natl Acad. Sci.* **114** 4582–90

Skinner C R and Chang C-R 1996 Fire regimes, past and present *Sierra Nevada Ecosystem Project: Final Report to Congress* vol II

Spracklen D V, Mickley L J, Logan J A, Hudman R C, Yevich R, Flannigan M D and Westerling A L 2009 Impacts of climate change from 2000 to 2050 on wildfire activity and carbonaceous aerosol concentrations in the western United States *J. Geophys. Res. Atmos.* **114**

Spiegal S, Huntsinger L, Hopkinson P and Bartolome J W 2016 *Range Ecosystems Ecosystems of California* ed H A Mooney and E S Zavaleta (Berkeley, CA: University of California Press) 835–64

Steel Z L, Safford H D and Viers J H 2015 The fire frequency-severity relationship and the legacy of fire suppression in California forests *Ecosphere* **6** 8

Stephens S L, Agee J K, Fulé P Z, North M P, Romme W H, Swetnam T W and Turner M G 2013 Managing forests and fire in changing climates *Science* **342** 41–2

Stephens S L, Martin R E and Clinton N E 2007 Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands *Forest Ecol. Manage.* **251** 205–16

Stephens S L, Miller J D, Collins B M, North M P, Keane J J and Roberts S L 2016 Wildfire impacts on California spotted owl nesting habitat in the Sierra Nevada *Ecosphere* **7** 1–21

Stephens S L and Ruth L W 2005 Federal forest-fire policy in the United States *Ecol. Appl.* **15** 532–42

Stewart W, Sharma B, York R, Diller L, Hamey N, Powell R and Swiers R 2016 *Forestry Ecosystems of California* ed H A Mooney and E S Zavaleta (Berkeley, CA: University of California Press) 817–34

Stonesifer C S, Calkin D E, Thompson M P and Stockmann K D 2016 Fighting fire in the heat of the day: an analysis of operational and environmental conditions of use for large airtankers in United States fire suppression *Int. J. Wildl. Fire* **25** 520–33

USDA and USDOI 1994 *Record of Decision for Amendments to Forest Service and Bureau of Land Management Planning Documents Within the Range of the Northern Spotted Owl* (Portland, OR: United States Department of Agriculture and United States Department of the Interior) (https://reo.gov/library/reports/newroda.pdf)

USDOI and USDA 1995 *Federal Wildland Fire Management: Policy & Program Review Final Report* (Boise, ID: United States Department of Agriculture and United States Department of the Interior) (https://www.forestsandrangelands.gov/strategy/documents/foundational/1995_fed_wildland_fire_policy_program_report.pdf)

USDOI and USGS 2016 *PAD-US—National Inventory of Protected Areas* (Reston, VA: United States Department of the Interior and United States Geological Survey) (http://gapanalysis.usgs.gov/padus)

USFS 2015 *The Rising Cost of Fire Operations: Effects on the Forest Service's Non- Fire Work* (Washington, DC: United States Forest Service) (https://www.fs.fed.us/sites/default/files/2015-Fire-Budget-Report.pdf)

USFS 2016 Grazing statistical summary FY2015 110 (Washington, DC) (https://www.fs.fed.us/rangeland-management/documents/grazing-stats/2010s/GrazingStatisticalSummaryFY2015.pdf)

Verner J, McKelvey K S, Noon B R, Gutierrez R J, Gould G I J and Beck T W 1992 The California spotted owl: a technical assessment of its current status *Gen. Tech. Rep.* PSW-GTR-133 (US Dep. Agric. Forest Serv. Pacific Southwest Res. Stn. 285) (https://www.fs.usda.gov/treesearch/pubs/43171)

van Wagtendonk J W 2007 The history and evolution of wildland fire use fire *Ecology* **3** 3–17

Westerling A L 2016 Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring *Phil. Trans. R. Soc. Lond. B. Biol. Sci.* **371** 717–28

Westerling A L and Bryant B P 2008 Climate change and wildfire in California *Clim. Change* **87** S231–49

Westerling A L, Hidalgo H G, Cayan D R and Swetnam T W 2006 Warming and earlier spring increase western US forest wildfire activity *Science* **313** 940–3

Whittier T R and Gray A N 2016 Tree mortality based fire severity classification for forest inventories: a pacific northwest national forests example *Forest Ecol. Manage.* **359** 199–209

Wiles T and Warren B 2016 Federal-lands ranching: a half- century of decline *High Ctry. News* 1–8 (13 June 2016) (http://www.hcn.org/issues/48.10/federal-lands-grazing)

Williams M 1989 *Americans and their Forests: A Historical Geography* (Cambridge: Cambridge University Press)

Williams R 2012 The stata journal *Stata J.* **12** 308–31

Williamson G J, Bowman D M J S, Price O F, Henderson S B and Johnston F H 2015 A transdisciplinary approach to understanding the health effects of wild fire and prescribed fire smoke regimes *Environ. Res. Lett.* **11** 125009

Wooldridge J M 2012 *Introductory Econometrics: A Modern Approach* (Mason, OH: Cengage)

Zedler P H, Gautier C R and McMaster G S 1983 Vegetation change in response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub *Ecology* **64** 809–18