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

Carlin Frances Starrs1,2, Van Butsi1, Connor Stephens1 and William Stewart1

1 Department of Environmental Science, Policy and Management. University of California Berkeley 130 Mulford Hall #3114, Berkeley CA, 94720-3114, United States of America
2 Author to whom any correspondence should be addressed.

E-mail: carlinstarrs@berkeley.edu

Keywords: wildfire, matching statistics, GIS, panel data, forest management, rangeland management

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.
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 firefighting responsibility to maximize efficiency, particularly in areas where a legacy of 19th century railroad firefighting 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 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%             |
| Forest noncomm  | 41 479            | 1 737 142             | 6%              |

*balanced hectares are those where ownership and firefighting falls under different agencies, e.g. federal lands with state firefighting.

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 (CWHRS) vegetation classification, 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-Habits](http://www.wildlife.ca.gov/Data/CWHR/Wildlife-Habits).
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, Westering 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.
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

Figure 2. California spatial distribution of ownership, firefighting responsibility, reserve status, and vegetation type.
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 confound 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 Koeping 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). 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:

\[
N_{Bi} = B_0 + B_1 \cdot Ownership + B_2 \cdot Firefighting + B_3 \cdot Reserve Status + B_4 \cdot Times + B_{57-135} \cdot Controls + B_{16-30} \cdot Interactions + u_i + e_{it}
\]

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 only 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)                            | 95%                                               |
| 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 (0.03)                            | −10%                                              |
| FORESTNONCOMM   | Non-Federal | Federal    | 0.89 (0.06)                            | 32%                                               |
| FORESTNONCOMM   | Federal    | State       | 1.73 (0.07)                            | 147%                                              |
| FORESTNONCOMM   | Federal    | Federal    | 2.08 (0.04)                            | 75%                                               |

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<sup>a</sup>| 0.03       | 0.00        | 14.47   | p < 0.01 |
| Effect of soil moisture        | −0.03      | 0.00        | −14.31  | 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<sup>a</sup>| 0.02       | 0.00        | 5.20    | p < 0.01 |
| Effect of soil moisture        | −0.02      | 0.00        | −4.16   | 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<sup>a</sup>| 0.05       | 0.00        | 13.54   | 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<sup>a</sup>| 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    |

<sup>a</sup> 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...
Effects of federal ownership 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 straightforward. 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

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).
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, Spiegel 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 Wagten-donk 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 unobservable 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.

**Acknowledgments**

We thank Scott Stephens, Lynn Huntsinger, Brandon Collins, Anu Kramer, and the Stephens Lab for their guidance and assistance with this project. This research was supported in part by the California Agricultural Extension Station, the College of Natural Resources at the University of California, Berkeley, a contract from the California Natural Resources Agency, and support from the Berkeley Research Impact Initiative (BRII) sponsored by the UC Berkeley Library.

**ORCID iDs**

Carlin Frances Starrs  @  https://orcid.org/0000-0003-2009-4203

**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 2003 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, Kuenmerle T and Radeloff V C 2015 The relative effectiveness of protected areas, a logging ban, and sacred areas for old-growth forest protection in southwestern 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 I and Ludwig I. 2011 An econometric analysis of land development with endogenous zoning 87 412–32

CAL FIRE 2012 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/fireprevention/)

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
Liu J C, Mickley L J, Sulprizido 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 Wagendonk 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

McKee T C, 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 Moto P 2004 Climatic change, wildfire, and conservation Conserv. Biol. 18 890–98

Miller J D, Collins B M, Lutz J A, Stephens S L, van Wagendonk J W and Yasuda D A 2012b Differences in wildfires among ecoregions and land management agencies in the Sierra Nevada region, California, USA Ecosphere 3 a808

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. Geog. 32 878–88

National Wildfire Coordinating Group (Glossary A-Z 2017 NWCG (www.nwcc.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, Hsioging I 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 J D and DeBenedetti S H 1979 Impact of fire suppression on a mixed-conifer 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-Arrilola 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, Szwab R, Franklin J and Lawson D M 2010 Habitat fragmentation and altered fire regime create trade-offs for an obligate seeding shrub Forest Ecol. Manage. 269 20–27

Rhodes J J and Baker W L 2008 Fire probability, fuel treatment efforts, and fire suppression efficiency of large wildland fire suppression effort: a stochastic frontier analysis J. Environ. Manage. 166 227–3

Regan H M, Crookston J B, Szwab R, Franklin J and Lawson D M 2010 Habitat fragmentation and altered fire regime create trade-offs for an obligate seeding shrub Forest Ecol. Manage. 269 20–27

Rhodes J J and Baker W L 2008 Fire probability, fuel treatment effectiveness and ecological tradeoffs in western US public forests Forest Sci. 54 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

Spiegal S, Huntsinger L, Hopkinson P and Bartolome J W 2016 Range Ecosystems Ecosystems of California ed H A Mooney and E S Zavala (Berkeley, CA: University of California Press) 835–84

Steel Z L, Safford H D and Viers J H 2013 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
Stonefer 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 air tankers 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 J I 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 Wagtenendonk 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 5231–49
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 123009
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