Friends of the Wild Swan
PO Box 103
Bigfork, MT 59911

September 20, 2023

Flathead National Forest
PO Box 190340
Hungry Horse, MT 59911
Attn: Gary Blazejewski - Flathead Fuel Break Project
Via electronic submission to: https://cara.fs2c.usda.gov/Public/CommentInput?project=64699

Mr. Blazejewski,

Please accept the following comments on the Flathead Fuel Break Project on behalf of Friends of the Wild Swan. We concur with and incorporate by reference the comments submitted by Swan View Coalition.

• The scoping notice does not provide the treatments that will be done in the fuel breaks but instead leaves it up to on-the-ground personnel to determine what is needed. How can impacts be analyzed if you do not have cutting units, treatments and temporary roads identified?

• Unless the Forest Service has a crystal ball it cannot predict where wildfire will occur and when. During wind events embers can be cast miles ahead of the actual fire and there are numerous examples of fires jumping rivers (such as North Fork Flathead and Columbia) and highways.

The scoping notice objective of the treatments is "to reduce fuel loads by removing the overstory trees and creating space for new trees to be established." So, if you want new trees to establish why cut down the existing trees? Fuel breaks need to be maintained, so your analysis must include future activities and their impacts to other resources.

The analysis must evaluate and consider whether the proposed treatments to reduce fire and fuel hazard will actually have the desired effect during the timeframe before vegetation regrows.

The analysis must consider that in order to maintain this so called “fire proof” condition on the ground it entails repeated entries that have negative consequences for water quality, fish, wildlife and weed spread. This was not analyzed as a cumulative effect. In addition, the combination of repeated entries is a programmatic vegetative management practice that represents a significant departure from the Goals and Objectives and Desired Future Condition envisioned and analyzed by the Forest Plan. We are facing “perpetual management” to the detriment of wildlife, fish and water quality.

This issue is addressed in Rhodes and Baker's Fire Probability, Fuel Treatment Effectiveness and Ecological Tradeoffs in Western U.S. Public Forests (attached) that was not considered or analyzed in the EA. Following are excerpts:

Using Equation 1, our results indicate that if treatments were repeated every 20 years across all USFS lands in the West, it would take about 720 years (36 cycles of treatments), on average, before it is expected that high-severity fire affects slightly more than 50% of treated areas while fuels are reduced. Treatments would have to be repeated at 20-year intervals for 340 years (17 cycles of treatments) before high-moderate severity fire is expected to encounter more than 50% of treated areas. Even after this duration of repeated treatments, it is likely that almost 50% of treated areas will be cumulatively affected by repeated treatments without compensatory benefits from reduced fire severity. These West-wide estimates provide perspective, but include forest types, such as subalpine forests, typified by low frequency, high-severity fire, where fuel treatments are unlikely to encounter fire [4]. Other forests, such as ponderosa pine, burn more often.

Even in ponderosa pine forests that burn relatively frequently, our regional analysis indicates that after 17 cycles of treatments, only slightly more than 50% of treated areas could potentially have fire severity reduced, on average. Our results indicate that high-severity fire is far from inevitable in areas left untreated and is, instead, expected to affect only a relatively small fraction of such areas at the broad scale of our analysis. Factoring in the probability of fire, using our framework, can significantly improve the assessments of the risks posed to aquatic systems by treating or not treating forest fuels. Where site-specific data on fire probabilities exist, the framework can be used to help locate treatments where they are most likely to encounter higher severity fire, increasing the likelihood of treatment benefits. In fact, our results indicate that such efforts are crucial.

There are several important factors that influence the aquatic tradeoffs among fuel treatments, fire, and aquatic systems that our framework does not address. Although the probability of outcomes is critical to assessing the expected value of options, the ecological costs of the outcomes of treatment vs non-treatment are also important in assessing the expected value of these options. With respect to the aquatic context, there is an ongoing need to fully evaluate tradeoffs such as the severity and persistence of the negative and positive impacts on watersheds and aquatic populations from fuel treatments and higher severity fire [8, 45]. An additional related issue is how effective treatments are when they encounter fire under a broad array of conditions affecting fire behavior [3]. While our analysis does not address these factors, it refines evaluation of net impacts of fuel treatment vs non-treatment by providing a framework for estimating the likelihood of fire occurrence in a given time frame.

At the scales of our analysis, results indicate that even if fuel treatments were very effective when encountering fire of any severity, treatments will rarely encounter fire, and thus are unlikely to substantially reduce effects of high-severity fire.

• Fuels treatments do not always result in the desired outcome. For example, the thinning done around Swan Lake in the Six Mile Fuels Reduction project resulted in all the leave trees blowing down leaving large clearcuts. Blowdown is a real concern that must be analyzed.

• A categorical exclusion does not preclude the need to analyze the impacts to wildlife, fish, water quality, wetlands, old-growth forest habitat, threatened, endangered and sensitive species. How will the project impact elk and grizzly bear security habitat? How will new temporary roads impact fish and water quality? How will lynx critical habitat be affected? How will wolverine, goshawk, fisher, marten, whitebark pine and other species be impacted? How will connectivity be maintained? How will soils and myccorhizal fungi be impacted/compacted? How will spreading weeds impact wildlife forage?

• What are the cumulative impacts with other projects?

• What have nearby landowners done to secure their property from fire danger? In addition to Cohen's work on defensible space near structures, new science is validating the same approach in Syphard, et al., The role of defensible space for residential structure protection during wildfires (attached) which concludes:
"Structures were more likely to survive a fire with defensible space immediately adjacent to them. The most effective treatment distance varied between 5 and 20 m (16–58 ft) from the structure, but distances larger than 30 m (100 ft) did not provide additional protection, even for structures located on steep slopes. The most effective actions were reducing woody cover up to 40% immediately adjacent to structures and ensuring that vegetation does not overhang or touch the structure. Multiple-regression models showed landscape-scale factors, including low housing density and distances to major roads, were more important in explaining structure destruction. The best long-term solution will involve a suite of prevention measures that include defensible space as well as building design approach, community education and proactive land use planning that limits exposure to fire."

So, what is important for protecting homes and other structures is what the homeowners do on their property, not the thinning the Forest Service does miles away.

• How will climate change impact your assumptions about this project?

A categorical exclusion does not preclude the need to comply with the Endangered Species Act, Clean Water Act, National Forest Management Act, National Environmental Policy Act and Forest Plan. Given the presence of numerous threatened and endangered species an Environmental Impact Statement is warranted.

/s/Arlene Montgomery
Program Director

Fire Probability, Fuel Treatment Effectiveness and Ecological Tradeoffs in Western U.S. Public Forests

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Abstract: Fuel treatment effectiveness and non-treatment risks can be estimated from the probability of fire occurrence. Using extensive fire records for western US Forest Service lands, we estimate fuel treatments have a mean probability of 2.0-7.9% of encountering moderate- or high-severity fire during an assumed 20-year period of reduced fuels.

INTRODUCTION

Fuel treatments to reduce fire impacts have been promoted as a public forest restoration priority by policy [1] and the Healthy Forests Restoration Act of 2003. It is difficult to generalize about the effectiveness of fuel treatments under all conditions [2, 3], but treatments are not universally effective when fire affects treated areas [4]. Factors influencing effectiveness include forest type, fire weather [4], and treatment method [5].

However, treatments cannot reduce fire severity and consequent impacts, if fire does not affect treated areas while fuels are reduced. Fuels rebound after treatment, eventually negating treatment effects [3, 6]. Therefore, the necessary, but not sufficient, condition for fuel treatment effectiveness is that a fire affects a treated area while the fuels that contribute to high-severity fire have been reduced. Thus, fire occurrence within the window of effective fuel reduction exerts an overarching control on the probability of fuel treatment effectiveness. The probability of this confluence of events can be estimated from fire records. Although this probability has not been rigorously analyzed, it has often been assumed to be high [7].

The probability of future fire occurrence also abets assessing the ecological risks incurred if fuels are not treated. Therefore, analysis of the likelihood of fire is central to estimating likely risks, costs and benefits incurred with the treatment or non-treatment of fuels.

Assessing fire occurrence and its effect on fuel treatment effectiveness also has merit because treatments can incur ecological costs, including negative impacts on aquatic systems [8], soils [7], and invasion by non-native plants [9, 10]. Here, we use watershed and aquatic systems as a specific context for evaluating tradeoffs involved with treatment and non-treatment of fuels on western public lands. However, the analysis applies to upland ecosystems as well.

The effects of fire on watersheds and native fish vary with several biophysical factors, including watershed and habitat conditions, the condition of affected populations, and fire severity and extent [11]. If treatments reduce the watershed impacts of severe fire, they may provide benefits that outweigh treatment impacts because high-severity fire can sometimes trigger short-term, severe erosion and runoff [12] that can negatively affect soils, water quality, and aquatic populations. However, fuel treatments can also have impacts on aquatic systems. The magnitude and persistence of these treatment impacts vary with treatment methods, location, extent and frequency.

Although some fuel-treatment methods could have lower impacts, ground-based mechanical treatments are often employed because other methods generate activity fuels [7] and are more costly. Ground-based methods and associated machine piling, burning of activity fuels, construction and increased use of roads and landings can increase soil erosion, compact soils, and elevate surface runoff [8, 13, 14]. Although the effects of prescribed fire on watersheds are typically limited and fleeting, it can increase soil erosion and sediment delivery, sometimes significantly and persistently [15], especially if fires escape and burn larger and more severely than planned.

When impacts are extensive, proximate to streams, or in terrain with erosion hazards, treatments can increase runoff and sediment delivery to streams. Road activities that increase sediment production, such as elevated road traffic, often affect stream crossings where sediment delivery is typically efficient and difficult to control [16]. Elevated sediment delivery to streams contributes to water quality degradation that impairs aquatic ecosystems [17].

The extent and frequency of treatments may be significant. Stephens and Ruth [18] suggested treating fuels on 9.4 million ha, or ~53% of USFS lands in the Pacific Northwest and California. Agee and Skinner [7] suggested repeating treatments every 10-20 years, due to transient effects on fuels.

Repeated treatments increase the potential for cumulative effects on aquatic ecosystems due to the persistence and additive nature of watershed impacts over time [19] and may increase the establishment of non-native plants [9]. The chronic watershed impacts from repeated treatments may be more deleterious to native fish than pulsed disturbances from wildfires [8].

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Additional degradation of aquatic habitats on public lands may hamper efforts to protect and restore aquatic biodiversity. These habitats are increasingly important as cornerstones for restoring aquatic ecosystems and native fish [14].

Where fuel treatments might incur soil and watershed impacts, the risks from treatment and non-treatment should be assessed [7]. Although the respective impacts of treatments and fire are influenced by numerous factors, the occurrence of fire strongly affects the net balance between costs and benefits. If fire does not affect treated areas while fuels are reduced, treatment impacts on watersheds are not counterbalanced by benefits from reduction in fire impacts.

We provide a framework for quantitatively bounding the potential effectiveness of fuel treatments and the likelihood of fire affecting untreated watersheds, based on the probability of fire and the duration of treatment effects on fuels. This can be used to help statistically estimate the expected value associated with treatments or non-treatment based on the probability of possible outcomes and their associated costs and benefits [20]. Previous assessments of watershed trade-offs from treatment and non-treatment [21, 22] did not include these in quantifying risk to aquatic systems associated with treatment versus non-treatment of fuels.

We use geographically-explicit data on fire on public lands in the western US to estimate, at a broad-scale, the probability that fuel treatments will be affected by fire during the period when fuels have been reduced. We also estimate the risk of higher severity fire occurring in watersheds if fuel treatments are foregone. These estimates provide a broad-scale bounding of treatment effectiveness and potential return from the fiscal and environmental costs of fuel treatments.

METHODS

The Analytical Model

Our analysis is based on the simple conceptual framework that unless fire occurs while fuels are reduced, fuel treatments cannot affect fire severity. We examine the probability of discrete classes of fire severity because fire impacts on watersheds vary with severity [11]. For instance, lower-severity fire has minimal, transient watershed impacts [11].

Future fire occurrence in specific locations cannot be predicted with certainty, but its probability can be estimated from empirical data. The probability of fire of a particular severity affecting treated areas can be estimated using the standard formula for the probability of an event occurring during a specific time frame:

$$q = 1 - (1 - p)^n \tag{1}$$

where q is the probability that a fire that would be of a specific severity in the absence of treatment occurs within n years, p is the annual probability of fire of a specific severity at the treatment location, and n is the duration, in years, that treatments decrease fuels and can reduce fire severity. In Equation 1, q provides an estimate of the mean fraction of an analysis area likely to burn at a specific severity within a given time frame in the absence of fuel treatments, which also represents the upper bound of potential effectiveness of treatments in reducing fire, since treatments cannot lower fire severity unless a fire occurs.

Both n and p can be estimated from available data. The duration of post-treatment fuel reduction, n, likely varies regionally with factors affecting vegetation re-growth rates, but fuels in western U.S. forests generally return to pre-treatment levels in 10-20 years [3, 7]. To estimate the upper limit of treatment effectiveness, we assume $n = 20$ years. We estimated the annual probability of fire of various severities, p, for each analysis area based on standard methods [23]:

$$p = (F*r)/(A*D) \tag{2}$$

where p is the annual probability of fire of a specific severity, F is total area burned at any severity within the analysis area over the duration of the data record, r is the estimated fraction of F that burned at the specified severity over the analysis area, A is the total analysis area, and D is the total duration of the data record, in years.

We based our estimates of the annual probability of fire on post-1960 fire records rather than reported natural fire return intervals for two primary reasons. First, evidence indicates that natural fire regimes no longer operate in many forests, because of direct fire suppression and indirect changes in fuels from livestock grazing, logging and fire exclusion [24]. Annual burned area has also increased in some forest types, likely due to climatic warming [25]. Recent fire data ostensibly integrate these alterations, reflecting how fires are likely to burn in the near future under current conditions and management. Natural fire return intervals do not capture these alterations. Second, there is considerable uncertainty regarding the accuracy of reported natural fire intervals [23, 24]. However, we stress that our approach can easily accommodate alternate estimates of annual fire probability using more geographically-refined data or where management changes might alter future fire probability.

We confined analysis to USFS lands in 11 western states, the focus for most proposed fuel treatments on public lands. The probability of fire varies geographically with several factors, including weather, ignition, fuels, and forest types. To bracket this effect, we estimated the annual probability of high-severity fire, p, for (*i*) all landcover types and (*ii*) more frequently burning ponderosa pine (*Pinus ponderosa*) forests at the scale of U.S. Forest Service (USFS) administrative regions that are the finest scale at which extensive data allow estimation of fire severity. We focus on high-severity fire, but also analyze fires of broader severity, including (1) either high- or moderate severity and (2) any severity.

Our estimates represent an initial, broad-scale first approximation of the potential of fire to affect areas within a given time frame, based on the assumption that fire and treatments are random. Although fire is not random, data are insufficient to accurately quantify more local patterns. Our approach provides a valid mean result at our scale of analysis, based on data from more than 40,000 fires across the western U.S. Site-specific data could be used in future, local studies where the probability of fire is known to depart considerably from the regional mean. Ideally, fuel treatments may not be randomly located, but instead focused in areas where fire is most likely. However, this is not assured by current policy [26]. Widely used methods for assessing the risk of high-severity fire may have limited accuracy [27].

Therefore, our analysis assumes random treatment location, as a first approximation.

West-Wide Analysis

To provide a broad-scale perspective of potential fuel treatment efficacy, we estimated mean annual probability, *p,* of fire for all USFS lands in the 11 western U.S. states, excluding Alaska, for the entire duration that data on total annual fire area are available (1960-2006). Data on fire area from 1993-2003, reported by agency ownership [28], were used to estimate mean annual fraction of total fire area on USFS lands, which was extrapolated to estimate mean annual fire area on USFS lands from 1960-1993 and 2004-2006, for which fire area data were reported [29], but not by agency ownership. Annual fire area on USFS lands in the 11 western states was assumed proportional to the fraction of total USFS area in these states. Total number of fires on western USFS lands from 1960-2006 is not reported, but based on the foregoing areal partitioning, the fire area data are from several hundred thousand fires on western USFS lands. The estimated annual fire area on these western USFS lands from 1960-2006 was summed to yield *F* in Equation 2.

The fraction of total fire area, *r*, that burned at high severity and high-moderate severity was estimated from data in USFS burned area emergency rehabilitation reports (BAER) for 470 fires in the 11 western states from 1973-1998 in six western USFS regions [30].

Regional Analysis of Fire in Ponderosa Pine

Because ponderosa pine forests are a key forest with more frequent fire, we estimated the mean annual probability of fire by severity in these forests on USFS lands: 1) on a regional basis, in six western USFS regions; and 2) West-wide. We used geographical information system (GIS) data for 40,389 fires in these forests for the entire period of data availability, 1980-2003 (Fig. **1**). Data were in a GIS point dataset, containing burned area for each fire, maintained by the Bureau of Land Management [31] and derived from a systematic National database [32]. We quality controlled these data for our study area, removing a few duplicate records.

A GIS map of ponderosa pine forests was obtained by selecting codes 5-7 (ponderosa pine) in the Westgap map from the GAP program, which includes national vegetation mapping from satellite imagery [33]. A GIS map of U.S. Forest Service regions is from the agency [34]. We converted all maps to Albers projection, Clarke 1866 datum, then used these to extract all fire records (n = 40,389) for ponderosa pine forests on USFS land in the 11 western states. We used USFS maps to subset fires by region, and then: (*i*) areas of individual fires were summed to yield *F* in Equation 2; (*ii*) the GIS was used to obtain *A*, and (*iii*) fire severity data by USFS region from 1973-1998 [30] were used to estimate *r* by severity.

RESULTS AND DISCUSSION

West-Wide Analysis

For the period 1960-2006, an estimated mean of ~220,000 ha, or a decimal fraction of 0.0037 of USFS western lands burned annually at any severity. Despite the approximations involved, our estimate of the mean annual fraction of areas burning at any severity compares reasonably with independent estimates by falling between them. Fire of any severity annually burned a mean fraction of ~0.0014 of the Deschutes National Forest in Oregon, from 1910-2001 [35], and ~0.0046 of 11 national forests in the Sierra Nevada, California, based on data from 1970-2003 [36].

Together with fire severity data [30], our West-wide estimate yields an estimated mean annual probability, *p*, of 0.001 and 0.002 for high- and high-moderate severity fire, respectively (Table **1**). Based on these estimates of *p*, Equation 1 yields a probability, *q*, of 0.020 and 0.042, respectively, for high- and high-moderate-severity fire. Substituting space for time, our results indicate that, on average, approximately 2.0 to 4.2% of areas treated to reduce fuels are likely to encounter fires that would otherwise be high or high-moderate severity without treatment. In the remaining 95.8-98.0% of treated areas, potentially adverse treatment effects on watersheds are not counterbalanced by benefits from reduced fire severity. These results also provide an estimate of the likelihood of high-severity fire affecting forests, if fuels are untreated. On average, over a 20-year period, about 2.0-4.2% of untreated areas would be expected to burn at high or high-moderate severity, respectively.

Using Equation 1, our results indicate that if treatments were repeated every 20 years across all USFS lands in the West, it would take about 720 years (36 cycles of treatments), on average, before it is expected that high-severity fire affects slightly more than 50% of treated areas while fuels are reduced. Treatments would have to be repeated at 20-year intervals for 340 years (17 cycles of treatments) before high-moderate severity fire is expected to encounter more than 50% of treated areas. Even after this duration of repeated treatments, it is likely that almost 50% of treated areas will be cumulatively affected by repeated treatments without compensatory benefits from reduced fire severity.

These West-wide estimates provide perspective, but include forest types, such as subalpine forests, typified by low-frequency, high-severity fire, where fuel treatments are unlikely to encounter fire [4]. Other forests, such as ponderosa pine, burn more often.

Regional Analysis of Ponderosa Pine

For ponderosa pine forests, the probability, *q*, of treated areas being affected within their window of effectiveness varies regionally from 0.020 to 0.040 for high-severity fires and from 0.042 to 0.079 for high-moderate severity fires (Table **1**). As expected, *q* in these forests is higher than for the West-wide analysis of all cover types. The highest probabilities, as expected, are in the Southwest and in the Northern Rockies, with its dry summers (Table **1**).

In these forests with more frequent fire, it is likely that fuel treatments can potentially reduce fire severity on a small fraction of treated areas. The results (Table **1**) indicate that in 92.1-98.0% of treated areas, fuel treatment impacts on watershed processes are not likely to be counterbalanced by a reduction in higher-severity fire.

Across the six regions, treatments would have to be repeated every 20 years for 340 to 700 years (17 to 35 times), on average, before it is expected that high-severity fire affects more than 50% of treated areas during periods of treat-

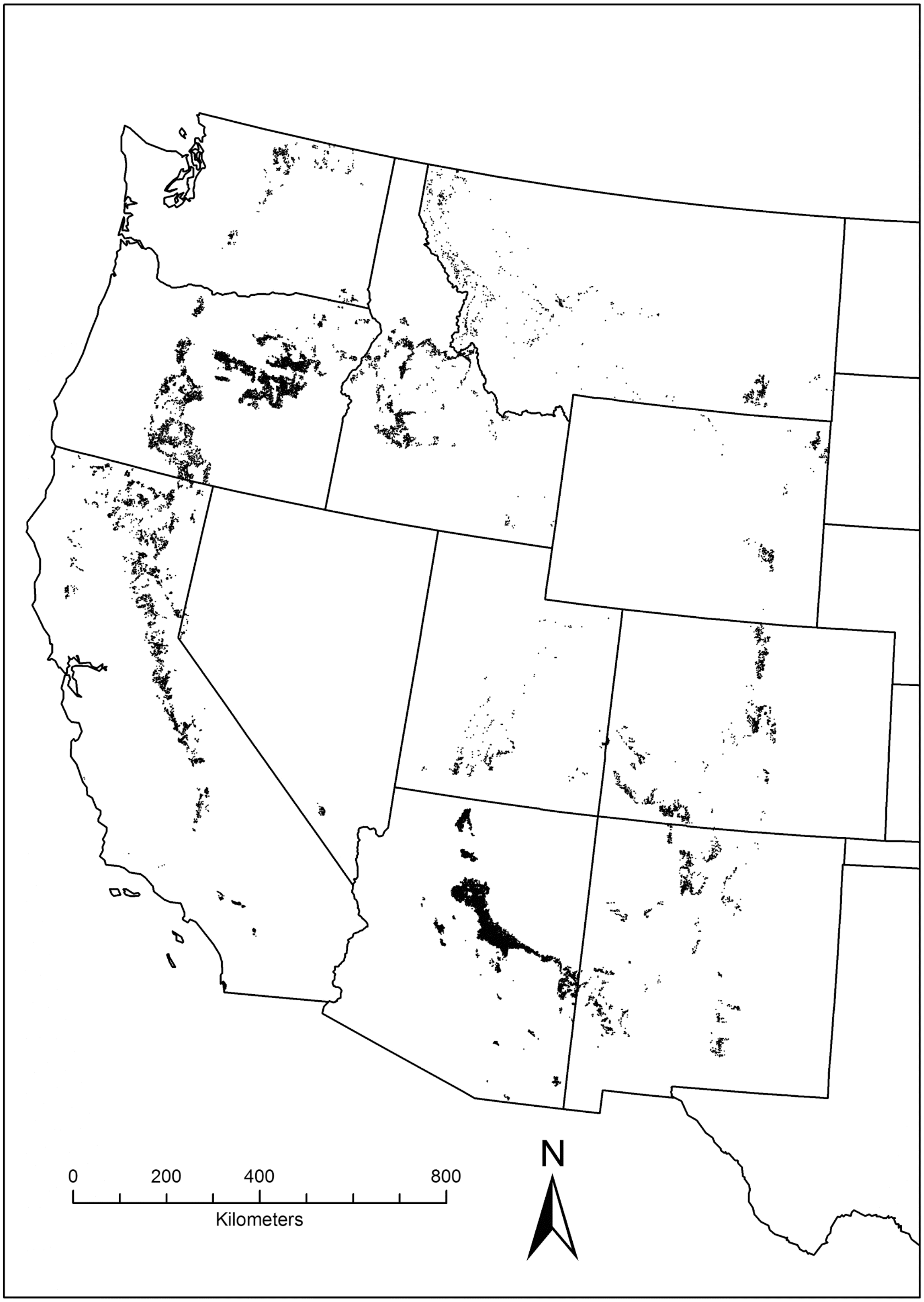

Fig. (1). Ponderosa pine forest fires (n = 40,389) in the western United States from 1980-2003. This is the dataset used in the regional analysis.

ment effectiveness. Treatments would have to be repeated for 180 to 340 years (9 to 17 times) before more than 50% of treated areas are expected to be affected by high-moderate severity fire. On average, these repeated treatments would affect watersheds and, potentially aquatic systems, depending on treatment practices, without providing reduction in fire severity on almost 50% of treated area.

An alternative method for estimating the risk of fire in the absence of fuel treatments is to use the fire rotation rather than mean annual probability of fire. The fire rotation indicates how long it takes, on average, for a particular area to burn one time and how often fire may return to a particular point in the landscape [23]. The fire rotation is calculated by:

$$B = 1/p \tag{3}$$

Table 1. Estimated *p* and *q* for Fires in Ponderosa Pine (PIPO) Forests. Data are Shown for Three Fire Severity Classes by USFS Region, and for All Forests on USFS Lands West-Wide

USFS Region	Any Severity		High-Moderate Severity		High Severity	
	p	q	p	q	p	q
1 N. Rockies	0.0070	0.1311	0.0036	0.0693	0.0020	0.0402
2 C&S Rockies	0.0059	0.1116	0.0041	0.0786	0.0014	0.0269
3 SW	0.0053	0.1008	0.0025	0.0487	0.0016	0.0307
4 Gt. Basin	0.0090	0.1654	0.0037	0.0715	0.0013	0.0257
5 Calif.	0.0046	0.0881	0.0031	0.0603	0.0017	0.0338
6 NW	0.0037	0.0715	0.0022	0.0421	0.0010	0.0198
West-wide: PIPO	0.0054	0.1026	0.0031	0.0602	0.0015	0.0295
West-wide: All types	0.0037	0.0715	0.0021	0.0416	0.0010	0.0203

where B is the fire rotation for fire of a specific severity and p is, again, the mean annual probability of fire of a specific severity.

Based on our analysis, the mean annual probability, p, of high-severity fire in ponderosa pine forests West-wide is 0.0015 (Table **1**), implying a fire rotation, B of about 667 years, varying from 500 to 1,000 years among individual regions. Based on the results in Table **1**, the fire rotation for high-moderate severity fire is about 323 years in ponderosa pine forests West-wide, varying from 244 to 454 years in individual regions, based on data in Table **1**. These results suggest that western ponderosa pine forests are not currently being rapidly burned by high or high-moderate severity fire, counter to other previous work [37].

Relaxing the Assumptions and Some Caveats

In some cases, the occurrence of fire of any severity may be of interest. Such cases include areas where fire of any severity might lead to high-severity fire. In ponderosa pine forests, the probability of fire of any severity encountering treatments within 20 years is approximately 7.15-16.5% across the six regions (Table **1**). Thus, if it is assumed that fuel treatments that encounter fire of any severity might be effective, the results indicate fuel treatments, on average, would not have the potential to reduce fire impacts on aquatic systems in 83.5-92.8% of the area treated. Based on Equation 1 and Table **1**, treatments would have be repeated every 20 years for 80-200 years, on average, before fire of any severity affects more than 50% of the treated areas in ponderosa forests in these USFS administrative regions.

However, the assumption that treatments that encounter low-severity fire convey benefits may not be warranted. Low-severity fires are commonly and easily extinguished under current management whether or not they encounter fuel treatments. Further, low-severity fire has minimal adverse impacts on watershed processes while conveying benefits, including maintenance of forest structure and fuel levels.

Our probabilistic approach does not explicitly address factors that can strongly influence fire area and severity, such as fuel conditions. Although spatially-explicit modeling of fire behavior can directly investigate the effects of such conditions, such models are unlikely to provide accurate estimates of the probability of occurrence of fire of a given severity because a host of other factors that influence fire area and severity cannot be deterministically predicted, including the frequency and location of ignitions and weather conditions during fire. Methods of assessing the risk of high-severity fire that are primarily based on fuel conditions have been shown to be an ineffective predictor of the actual severity at which fires burn [38]. In contrast, extensive recent data from numerous fires, as used in our analysis, does provide a robust estimate of the mean probability of the occurrence of fire of a given severity, because it integrates the many factors that influence fire occurrence and severity.

Our estimates likely represent the upper bound for fuel treatment effectiveness at the scale of analysis. In many cases, less than 4.16-7.86% of treated area is likely to experience high-moderate severity fire during the duration of treatment effectiveness, because q decreases with decreases in n, the duration of treatment effectiveness. This duration is often less than the 20 years assumed in our analysis. In the Sierra Nevada of California, fuels returned to pre-treatment levels within 11 years [39]. At the values of p in Table **1**, reducing n from 20 to 11 years (Eq. 1) reduces the probability that higher-severity fire affects treatments by ~45%.

Moreover, fuel levels rebound after treatment, eventually negating potential treatment effectiveness. If the reduction in effectiveness over time is such that mean effectiveness over the duration, n, is half the initial degree of effectiveness, the probability that fuel treatments reduce high-severity fire is approximately half the value of q for any value of p and n calculated using Equation 1.

Finally, available data indicate that fuel treatments do not always reduce fire severity when fire affects treated areas while fuels are reduced [4]. Our analysis does not address these effectiveness issues. For these combined reasons, Equation 1 likely estimates the upper bound of potential fuel treatment effectiveness in reducing fire impacts on aquatic systems.

Although our analysis focuses on higher-severity fire in bounding the effectiveness of fuel treatments and their net watershed effects, these fires do not have solely negative effects. Higher-severity fire benefits watersheds and aquatic ecosystems in several ways, including providing a bonanza of recruitment of large wood and pulsed sediment supply that can rejuvenate aquatic habitats and increase their productivity [8, 14]. High severity fire is also a key process for the restoration of structural heterogeneity in forests, which is important for biodiversity [27, 40].

Our analysis intrinsically assumes some degree of climatic stationarity, which may not be warranted. Climatic variability influences the area annually burned in forests [25, 41]. However, the relatively recent fire data used in our regional analysis incorporates recent climatic fluctuation and possibly directional change, which would not be reflected in estimates based on natural fire return intervals. For instance, the data in our analysis of ponderosa pine forests come primarily from years in which annual fire area had increased due to climatic warming [25]. However, the analysis framework is flexible enough to accommodate projected values of the mean annual probability of fire, p, based on forecasts of climatic change or changes in fire management.

Current findings suggest treatment effects on fire severity are mostly confined to treated areas [3], but theory suggests a dense network of treatments might slow fire spread and reduce intensity, yielding a landscape-scale effect on fire severity [42]. However, empirical evidence of severity reduction was seen in the lee of only three of several dozen treatments in two Arizona wildfires [43]. Nonetheless, if dense treatment networks are shown to work in the future, our approach can aid in estimating their costs and benefits, because fire must still affect treated areas while fuels are reduced for networks to reduce fire severity.

CONCLUSIONS

Our analysis provides West-wide and regional first approximation of the likely upper bound of fuel treatment effectiveness. While valid at these two scales, they are not applicable to all smaller analysis areas, due to spatial variation in annual fire probability. However, the framework is flexible enough to allow more spatially explicit analyses of q where local estimates of n and p are available. The framework allows analysis of uncertainty, by using a range of plausible values for n and p. The analysis can also estimate the number of treatments to reach a specified q, abetting estimation of cumulative effects on ecosystems from repeated treatments.

Our approach also provides a method for quantitatively assessing the imminence of high-severity fire effects in the absence of fuel treatments and the degree of urgency of response. Based on available data, these are shown to be much lower than previously estimated in some work [37].

Our results and analyses can improve the assessment of risks to watersheds inherent in the treatment or non-treatment of forest fuels, because it accounts for the probability of fire and the transient nature of fuel treatments. For instance, previous work [22], evaluating treatment and non-treatment impacts, assessed the risks associated with fuel treatments based on the assumption that a single treatment significantly reduces fire risk on all treated areas, subsequently reducing consequent watershed impacts from fire. Other evaluations of these tradeoffs [21] compared the erosional effects of fuel treatments with high-severity fire under the explicit assumption that high-severity fire was inevitable without treatment and the implicit assumption that treatments always reduce or eliminate the potential for high-severity fire. Our analysis indicates that these assumptions are unwarranted and likely mischaracterize the outcomes and associated impacts of treatment options.

The approach can be extended to aid in assessing the risk to other ecosystem elements and processes that may be adversely affected by either fuel treatments or high-severity fire. For instance, non-native vegetation can be influenced by high fire severity [44] and some fuel treatments [10], especially if the treatments are repeated [9].

Even in ponderosa pine forests that burn relatively frequently, our regional analysis indicates that after 17 cycles of treatments, only slightly more than 50% of treated areas could potentially have fire severity reduced, on average. Our results indicate that high-severity fire is far from inevitable in areas left untreated and is, instead, expected to affect only a relatively small fraction of such areas at the broad scale of our analysis. Factoring in the probability of fire, using our framework, can significantly improve the assessments of the risks posed to aquatic systems by treating or not treating forest fuels. Where site-specific data on fire probabilities exist, the framework can be used to help locate treatments where they are most likely to encounter higher severity fire, increasing the likelihood of treatment benefits. In fact, our results indicate that such efforts are crucial.

There are several important factors that influence the aquatic tradeoffs among fuel treatments, fire, and aquatic systems that our framework does not address. Although the probability of outcomes is critical to assessing the expected value of options, the ecological costs of the outcomes of treatment *vs* non-treatment are also important in assessing the expected value of these options. With respect to the aquatic context, there is an ongoing need to fully evaluate tradeoffs such as the severity and persistence of the negative and positive impacts on watersheds and aquatic populations from fuel treatments and higher severity fire [8, 45]. An additional related issue is how effective treatments are when they encounter fire under a broad array of conditions affecting fire behavior [3]. While our analysis does not address these factors, it refines evaluation of net impacts of fuel treatment *vs* non-treatment by providing a framework for estimating the likelihood of fire occurrence in a given time frame.

At the scales of our analysis, results indicate that even if fuel treatments were very effective when encountering fire of any severity, treatments will rarely encounter fire, and thus are unlikely to substantially reduce effects of high-severity fire.

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The role of defensible space for residential structure protection during wildfires

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Abstract. With the potential for worsening fire conditions, discussion is escalating over how to best reduce effects on urban communities. A widely supported strategy is the creation of defensible space immediately surrounding homes and other structures. Although state and local governments publish specific guidelines and requirements, there is little empirical evidence to suggest how much vegetation modification is needed to provide significant benefits. We analysed the role of defensible space by mapping and measuring a suite of variables on modern pre-fire aerial photography for 1000 destroyed and 1000 surviving structures for all fires where homes burned from 2001 to 2010 in San Diego County, CA, USA. Structures were more likely to survive a fire with defensible space immediately adjacent to them. The most effective treatment distance varied between 5 and 20 m (16–58 ft) from the structure, but distances larger than 30 m (100 ft) did not provide additional protection, even for structures located on steep slopes. The most effective actions were reducing woody cover up to 40% immediately adjacent to structures and ensuring that vegetation does not overhang or touch the structure. Multiple-regression models showed landscape-scale factors, including low housing density and distances to major roads, were more important in explaining structure destruction. The best long-term solution will involve a suite of prevention measures that include defensible space as well as building design approach, community education and proactive land use planning that limits exposure to fire.

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Introduction

Across the globe and over recent decades, homes have been destroyed in wildfires at an unprecedented rate. In the last decade, large wildfires across Australia, southern Europe, Russia, the US and Canada have resulted in tens of thousands of properties destroyed, in addition to lost lives and enormous social, economic and ecological effects (Filmon 2004; Boschetti *et al.* 2008; Keeley *et al.* 2009; Blanchi *et al.* 2010; Vasquez 2011). The potential for climate change to worsen fire conditions (Hessl 2011), and the projection of continued housing growth in fire-prone wildlands (Gude *et al.* 2008) suggest that many more communities will face the threat of catastrophic wildfire in the future.

Concern over increasing fire threat has escalated discussion over how to best prepare for wildfires and reduce their effects. Although ideas such as greater focus on fire hazard in land use planning, using fire-resistant building materials and reducing human-caused ignitions (e.g. Cary *et al.* 2009; Quarles *et al.* 2010; Syphard *et al.* 2012) are gaining traction, the traditional strategy of fuels management continues to receive the most attention. Fuels management in the form of prescribed fires or mechanical treatments has historically occurred in remote, wildland locations (Schoennagel *et al.* 2009), but recent studies suggest that treatments located closer to homes and communities may provide greater protection (Witter and Taylor 2005; Stockmann *et al.* 2010; Gibbons *et al.* 2012). In fact, one of the most commonly recommended strategies in terms of fuels and fire protection is to create defensible space immediately around structures (Cohen 2000; Winter *et al.* 2009). Defensible space is an area around a structure where vegetation has been modified, or 'cleared,' to increase the chance of the structure surviving a wildfire. The idea is to mitigate home loss by minimising direct contact with fire, reducing radiative heating, lowering the probability of ignitions from embers and providing a safer place for fire fighters to defend a structure against fire (Gill and Stephens 2009; Cheney *et al.* 2001). Many jurisdictions provide specific guidelines and practices for creating defensible space, including minimum distances that are required among trees and shrubs as well as minimum total distances from the structure. These distances may be enforced through local ordinances or state-wide laws. In California, for example, a state law in 2005 increased the required total distance from 9 m (30 ft) to 30 m (100 ft).

Despite these specific guidelines on how to create defensible space, there is little scientific evidence to support the amount and location of vegetation modification that is actually effective at providing significant benefits. Most spacing guidelines and laws are based on 'expert opinion' or recommendations from older publications that lack scientific reference or rationale (e.g. Maire 1979; Smith and Adams 1991; Gilmer 1994). However, one study has provided scientific support for, and forms the basis of, most guidelines, policy and laws requiring a minimum of 30 m (100 ft) of defensible space (Cohen 1999, 2000). The modelling and experimental research in that study showed that flames from forest fires located 10–40 m (33–131 ft) away would not scorch or ignite a wooden home; and case studies showed 90% of homes with non-flammable roofs and vegetation clearance of 10–20 m (33–66 ft) could survive wildfires (Cohen 2000). However, the models and experimental research in that study focussed on crown fires in spruce or jack pine forests, and the primary material of home construction was wood. Therefore, it is unknown how well this guideline applies to regions dominated by other forest types, grasslands, or nonforested woody shrublands and in regions where wooden houses are not the norm.

Some older case studies showed that most homes with non-flammable roofs and 10–18 m (33–ft) of defensible space survived the 1961 Bel Air fire in California (Howard *et al.* 1973); most homes with non-flammable roofs and more than 10 m (33 ft) of defensible space also survived the 1990 Painted Cave fire (Foote and Gilless 1996). Also, several fire-behaviour modelling studies have been conducted in chaparral shrublands. One study showed that reducing vegetative cover to 50% at 9–30 m (30–ft) from structures effectively reduced fireline intensity and flame lengths, and that removal of 80% cover would result in unintended consequences such as exotic grass invasion, loss of habitat and increase in highly flammable flashy fuels (A. Fege and D. Pumphrey, unpubl. data). Another showed that separation distances adequate to protect firefighters varied according to fuel model and that wind speeds greater than 23 km h^{-1} negated the effect of slope, and wind speed above 48 km h^{-1} negated any protective effect of defensible space (F. Bilz, E. McCormick and R. Unkovich, unpubl. data, 2009). Results obtained through modelling equations of thermal radiation also found safety distances to vary as a function of fuel type, type of fire, home construction material and protective garments worn by firefighters (Zárate *et al.* 2008).

Although there is no empirical evidence to support the need for more than 30 m (100 ft) of defensible space, there has been a concerted effort in some areas to increase this distance, particularly on steep slopes. In California, a senate bill was introduced in 2008 (SB 1618) to encourage property owners to clear 91 m (300 ft) through the reduction of environmental regulations and permitting needed at that distance. Although this bill was defeated in committee, many local ordinances do require homeowners to clear 91 m (300 ft) or more, and there are reports that some people are unable to get fire insurance without 91 m (300 ft) of defensible space (F. Sproul, pers. comm.). In contrast, homeowner acceptance of and compliance with defensible space policies can be challenging (Winter *et al.* 2009; Absher and Vaske 2011), and in many cases homeowners do not create any defensible space.

It is critically important to develop empirical research that quantifies the amount, location and distance of defensible space that provides significant fire protection benefits so that guidelines and policies are developed with scientific support. Data that are directly applicable to southern California are especially important, as this region experiences the highest annual rate of wildfire-destroyed homes in the US. Not having sufficient defensible space is obviously undesirable because of the hazard to homeowners. However, there are clear trade-offs involved when vegetation reduction is excessive, as it results in the loss of native habitats, potential for increased erosion and invasive species establishment, and it potentially even increases fire risk because of the high flammability of weedy grasslands (Spittler 1995; Keeley *et al.* 2005; Syphard *et al.* 2006).

It is also important to understand the role of defensible space in residential structure protection relative to other factors that explain why some homes are destroyed in fires and some are not. Recent research shows that landscape-scale factors, such as housing arrangement and location, as well as biophysical variables characterising properties and neighbourhoods such as slope and fuel type, were important in explaining which homes burned in two southern California study areas (Syphard *et al.* 2012; 2013). Understanding the relative importance of different variables at different scales may help to identify which combinations of factors are most critical to consider for fire safety.

Our objective was to provide an empirical analysis of the role of defensible space in protecting structures during wildfires in southern California shrublands. Using recent pre-fire aerial photography, we mapped and measured a suite of variables describing defensible space for burned and unburned structures within the perimeters of major fires from 2001 to 2010 in San Diego County to ask the following questions:

1. How much defensible space is needed to provide significant protection to homes during wildfires, and is it beneficial to have more than the legally required 30 m (100 ft)?
2. Does the amount of defensible space needed for protection depend on slope inclination?
3. What is the role of defensible space relative to other factors that influence structure loss, such as terrain, fuel type and housing density?

Methods

Study area

The properties and structures analysed were located in San Diego County, California, USA (Fig. 1) – a topographically diverse region with a Mediterranean climate characterised by cool, wet winters and long summer droughts. Fire typically is a direct threat to structures adjacent to wildland areas. Native shrublands in southern California are extremely flammable during the late summer and fall (autumn) and when ignited, burn in high-intensity, stand-replacing crown fires. Although 500 homes on average have been lost annually since the mid-1900s (Calfire 2000), that rate has doubled since 2000. Most of these homes have burned during extreme fire weather conditions that accompany the autumn Santa Ana winds. The wildland–urban interface here includes more than 5 million homes, covering more than 28 000 km^2 (Hammer *et al.* 2007).

Property data

The data for properties to analyse came from a complete spatial database of existing residential structures and their

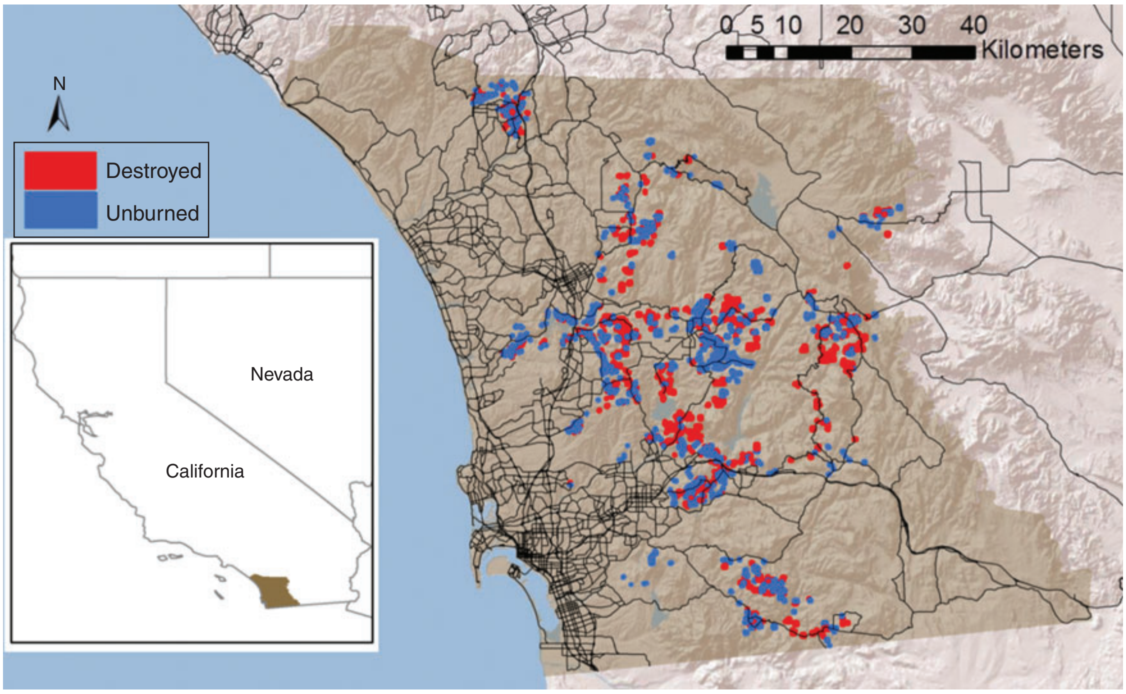

Fig. 1. Location of destroyed and unburned structures within the South Coast ecoregion of San Diego County, California, USA.

corresponding property boundaries developed for San Diego County (Syphard *et al.* 2012). This dataset included 687 869 structures, of which 4315 were completely destroyed by one of 40 major fires that occurred from 2001 to 2010. Our goal was to compare homes that were exposed to wildfire and survived with those that were exposed and destroyed. To determine exposure to fire, we only considered structures located both within a GIS layer of fire perimeters and within areas mapped as having burned at a minimum of low severity through thematic Monitoring Trends in Burn Severity produced by the USA Geological Survey and USDA Forest Service. From these data, we used a random sample algorithm in GIS software to select 1000 destroyed and 1000 unburned homes that were not adjacent to each other, to minimise any potential for spatial autocorrelation. Our final property dataset included structures that burned across eight different fires. More than 97% of these structures burned in Santa Ana wind-driven fire events (Fig. 1).

Calculating defensible space and additional explanatory variables

To estimate defensible space, we developed and explored a suite of variables relative to the distance and amount of defensible space surrounding structures, as well as the proximity of woody vegetation to the structure (Table 1). We measured these variables based on interpretation of Google Earth aerial imagery. We based our measurements on the most recent imagery before the date of the fire. In almost all cases, imagery was available for less than 1 year before the fire.

Our definition of defensible space followed the guidelines published by the California Department of Forestry and Fire Protection (Calfire 2006). 'Clearance' included all areas that were not covered by woody vegetation, including paved areas or grass. Although Google Earth prevents the identification of understorey vegetation, woody trees and shrubs were easily distinguished from grass, and our objective was to measure horizontal distances as required by Calfire rather than assess the relative flammability of different vegetation types. Trees or shrubs were allowed to be within the defensible space zone as long as they were separated by the minimum horizontal required distance, which was 3 m (10 ft) from the edge of one tree canopy to the edge of the next (Fig. 2). Although greater distances between trees or shrubs are recommended on steeper slopes, we followed the same guidelines for all properties. For all structures, we started the distance measurements by drawing lines from the centre of the four orthogonal sides of the structure that ended when they intersected anything that no longer met the requirements in the guidelines. A fair number of structures are not four sided; thus, the start of the centre point was placed at a location that approximated the farthest extent of the structure along each of four orthogonal sides.

We developed two sets of measurements of the distance of defensible space based on what is feasible for homeowners within their properties *v.* the total effective distance of defensible space. We made these two measurements because homeowners are only required to create defensible space within their own property, and this would reflect the effect of individual homeowner compliance. Therefore, even if cleared vegetation extended beyond the property line, the first set of distance measurements ended at the property boundary. The second set of measurements ignored the property boundaries and accounted for the total potential effect of treatment. For all measurements, we recorded the cover types (e.g. structure >3 m (10 ft) long, property boundary, or vegetation type) at which the distance measurements stopped (Table 1). Because property

Table 1. Defensible space variables measured for every structure

Urban veg, landscaping vegetation that was not in compliance with regulations within urban matrix; wildland veg, wildland vegetation that was not in compliance with regulations; orchard, shrub to tree-sized vegetation in rows; urban to wildland, landscaping vegetation that leads into wildland vegetation; structure, any building longer than 3 m (10 ft)

Variable	Definition
Distance defensible space within property	Measure of clearance from side of structure to property boundary calculated for four orthogonal directions from structure and averaged
Total distance defensible space	Measure of clearance from side of structure to end of clearance calculated for four orthogonal directions from structure and averaged
Cover type at end of defensible space	Type of cover encountered at end of measurement (urban veg, wildland veg, orchard, urban to wildland, structure)
Percentage clearance	Percentage of clearance calculated across the entire property
Neighbours' vegetation	Binary indicator of whether neighbours' uncleared vegetation was located within 30 m (100 ft) of the main structure
Vegetation touching structure	Number of sides on which woody vegetation touches main structure (1–4) Structure with more than 4 sides were viewed as a box and given a number between 1 and 4
Vegetation overhanging roof	Was vegetation overhanging the roof? (yes or no)

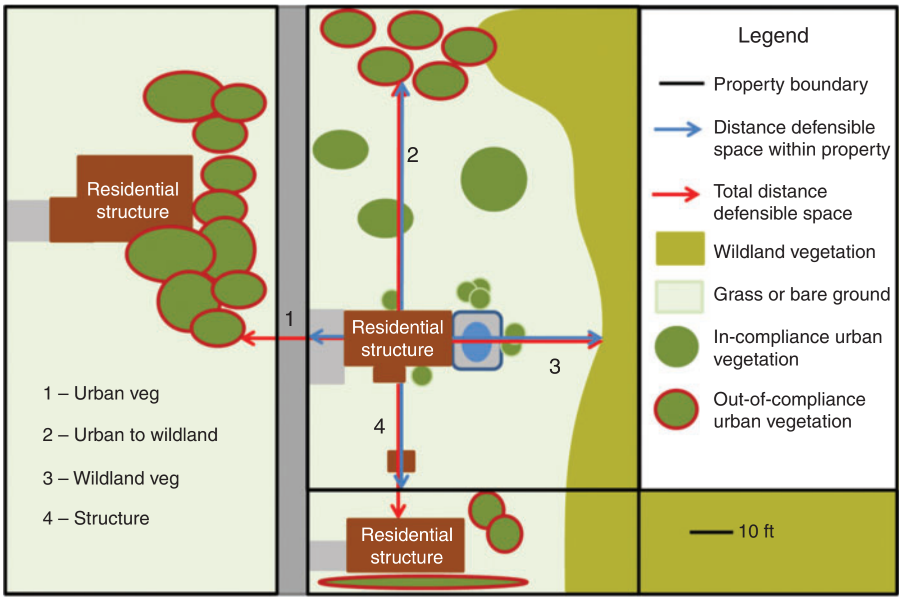

Fig. 2. Illustration of defensible space measurements. See Table 1 for full definition of terms.

owners usually can only clear vegetation on their own land, it is possible that the effectiveness of defensible space partly depends upon the actions of neighbouring homeowners. Therefore, we also recorded whether or not any neighbours' un-cleared vegetation was located within 30 m (100 ft) of the structure.

To assess the total amount of woody vegetation that can safely remain on a property and still receive significant benefits of defensible space, we calculated the total percentage of cleared land, woody vegetation and structure area across every property. This was accomplished by overlaying a grid on each property and determining the proportion of squares falling into each class. Preliminary results showed these three measurements to be highly correlated, so we only retained percentage clearance for further analysis. To evaluate the relative effect of woody vegetation directly adjacent to structures, we also calculated the number of sides of the structure with vegetation touching and recorded whether any trees were overhanging structures' roofs.

In addition to defensible space measurements, we evaluated other factors known to influence the likelihood of housing loss to fire in the region (Syphard *et al.* 2012, 2013). Using the same data as in Syphard *et al.* (2012, 2013), we extracted spatial information from continuous grids of explanatory variables for the locations of all structures in our analysis. Variables included interpolated housing density based on a 1-km search radius; percentage slope derived from a 30-m digital elevation model (DEM); Euclidean distance to nearest major and minor road and fuel type, which was based on a simple classification of US Forest Service data (Syphard *et al.* 2012), including urban, grass, shrubland and forest & woodland.

Analysis

We performed several analyses to determine whether relative differences in home protection are provided by different distances and amounts of defensible space, particularly beyond the legally required 30 m (100 ft), and to identify the effective treatment distance for homes on low and steep slopes.

Categorical analysis

For the first analysis, we divided our data into several groups to identify potential differences among specific categories of defensible space distance around structures located on shallow and steep slopes. We first sorted the full dataset of 2000 structures by slope and then split the data in the middle to create groups of homes with shallow slope and steep slope. We divided the data in half to keep the number of structures even within both groups and to avoid specifying an arbitrary number to define what constitutes shallow or steep slope. The two equal-sized subsets of data ranged from 0 to 9%, with a mean of 8% for shallow slope, and from 9 to 40%, with a mean of 27% for steep slope. Within these data subsets, we next created groups reflecting different mean distances of defensible space around structures. We also performed separate analyses based on whether defensible space measurements were calculated within the property boundary or whether measurements accounted for the total distance of defensible space.

Within all groups, we calculated the proportion of homes that were destroyed by wildfire. We performed Pearson's Chi-square tests of independence to determine whether or not the proportion of destroyed structures within groups was significantly different (Agresti 2007). We based one test on four equal-interval groups within the legally required distance of 30 m (100 ft): 0–7 m (0–25 ft), 8–15 m (26–50 ft), 16–23 m (51–75 ft) and 24–30 m (76–100 ft). A second test was based on three groups (24–30 m (75–100 ft), 31–90 m (101–300 ft) and >90 m (>300 ft) or >60 m (>200 ft)) to evaluate whether groups with mean defensible space distances >30 m (>100 ft) were significantly different from groups with <30 m (<100 ft). When defensible space distances were only measured to the property boundary, few structures had mean defensible space >90 m (>300 ft). Therefore, we used a cut-off of 60 m (200 ft) to increase the sample size in the Chi-square analysis. In addition to the Chi-square analysis, we calculated the relative risk among every successive pair of categories (Sheskin 2004). The relative risk was calculated as the ratio of proportions of burned homes within two groups of homes that had different defensible space distances.

Effective treatment analysis

In addition to comparing the relative effect of defensible space among different groups of mean distances, as described above, we also considered that the protective effect of defensible space for structures exposed to wildfire is conceptually similar to the effect of medication in producing a therapeutic response in people who are sick. In addition to pharmacological applications, treatment–response relationships have been used for radiation, herbicide, drought tolerance and ecotoxicological studies (e.g. Streibig *et al.* 1993; Cedergreen *et al.* 2005; Knezevic *et al.* 2007; Kursar *et al.* 2009). The effect produced by a drug or treatment typically varies according to the concentration or amount, often up to a point at which further increase provides no additional response. The effective treatment (ET50), therefore, is a specific concentration or exposure that produces a therapeutic response or desired effect. Here we considered the treatment to be the distance or amount of defensible space.

Using the software package DRC in R (Knezevic *et al.* 2007; Ritz and Streibig 2013), we evaluated the treatment–response relationship of defensible space in survival of structures during wildfire. To calculate the effective treatment, we fit a log-logistic model with logistic regression because we had a binary dependent variable (burned or unburned). We specified a 2-parameter model where the lower limit was fixed at 0 and the upper limit was fixed at 1. We again performed separate analyses for data subsets reflecting shallow and steep slope, as well as from measurements of defensible space taken within, or regardless of, property boundaries. We also performed analyses to find the effective treatment of percentage clearance of trees and shrubs within the property.

Multiple regression analysis

To evaluate the role of defensible space relative to other variables, we developed multiple generalised linear regression models (GLMs) (Venables and Ripley 1994). We again had a binary dependent variable (burned versus unburned), so we specified a logit link and binomial response. Although the proportion of 0s and 1s in the response may be important to consider for true prediction (King and Zeng 2001; Syphard *et al.* 2008), our objective here was solely to evaluate variable importance. We developed multiple regression models for all possible combinations of the predictor variables and used the corrected Akaike's Information Criterion (AICc) to rank models and select the best ones for each region using package MuMIn in R (R Development Core Team 2012; Burnham and Anderson 2002). We recorded all top-ranked models that had an AICc value within 2 of that of the model with lowest AICc to identify all models with empirical support. To assess variable importance, we calculated the sum of Akaike weights for all models that contained each variable. On a scale of 0–1, this metric represents the weight of evidence that models containing the variable in question are the best model (Burnham and Anderson 2002). The distance of defensible space measured within property boundaries was highly correlated with the distance of defensible space measured beyond property boundaries ($r = 0.82$), so we developed two separate analyses – one using variables measured only within the property boundary and the other using variables that accounted for defensible space outside of the property boundary as well as the potential effect of neighbours having uncleared vegetation within 30 m (100 ft) of the structure. A test to avoid multicollinearity showed all other variables within each multiple regression analysis to be uncorrelated ($r < 0.5$).

Surrounding matrix

To assess whether the proportion of destroyed structures varied according to their surrounding matrix, we summarised the most common cover type at the end of defensible space measurements (descriptions in Table 1) for all structures. These summaries

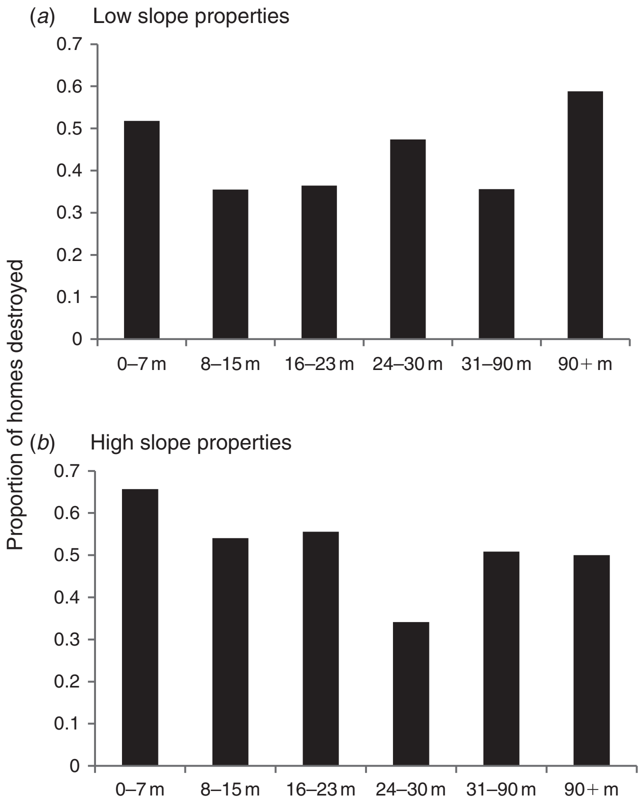

Fig. 3. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance within property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

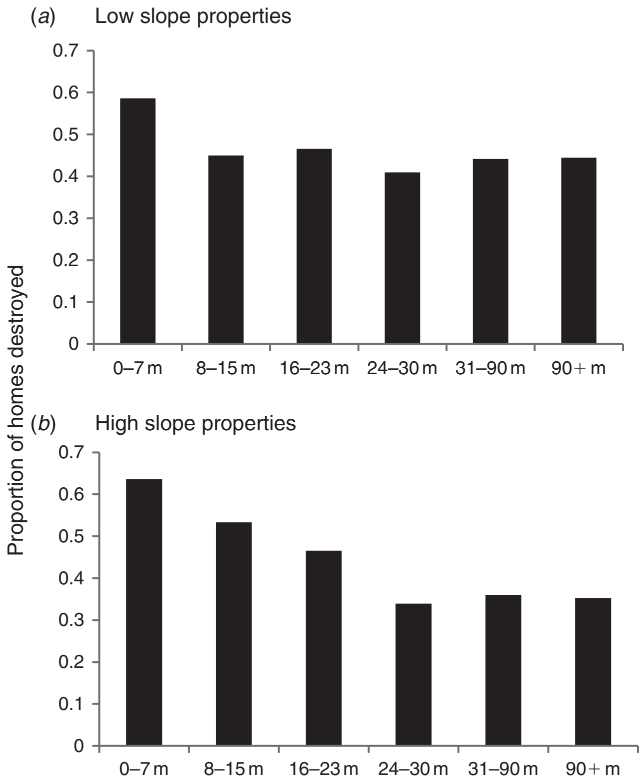

Fig. 4. Proportion of destroyed homes grouped by distances of defensible space based upon total distance of clearance regardless of property boundary, for structures on (*a*) shallow slopes (mean 8%) and (*b*) steep slopes (mean 27%).

were based on the majority surrounding cover type from the four orthogonal sides of the structure. We also noted cases in which there was a tie (e.g. two sides were urban vegetation and two sides were structures).

Results

Categorical analysis

When the distance of defensible space was measured both 'only within property boundaries' (Fig. 3) and 'regardless of property boundaries' (Fig. 4), the Chi-square test showed a significant difference ($P < 0.001$) in the proportion of destroyed structures among the four equal-interval groups of distance ranging from 0 to 30 m (0–100 ft). This relationship was consistent on both shallow-slope and steep-slope properties, although the relative risk analysis showed considerable variation among classes (Table 2) There was a steadily decreasing proportion of destroyed structures at greater distances of defensible space up to 30 m (100 ft) on the steep-slope structures with defensible space measured regardless of property boundaries (Fig. 4*b*). Otherwise, the biggest difference in proportion of destroyed structures occurred between 0 and 7 m (0–25 ft) and 8–15 m (26–50 ft) (Figs 3*a–b*, 4*a*).

When the distance of defensible space was measured in intervals from 24 m (75 ft) and beyond, the Chi-square test showed no significant difference among groups ($P = 0.96$ for shallow-slope properties and $P = 0.74$ for steep-slope properties) (Figs 3, 4), although again, the relative risk analysis showed considerable variation (Table 2).There was a slight increase in the proportion of homes destroyed at longer distance intervals when the defensible space was measured only to the property boundaries (Fig. 3*a–b*). This slight increase is less apparent when distances were measured regardless of boundaries (Fig. 4*a–b*).

The relative risk calculations showed that the ratio of proportions was generally more variable among successive pairs when the distances were measured within property boundaries (Table 2). For these calculations, the risk of a structure being destroyed was significantly lower when the defensible space distance was 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft) on both shallow- and steep-slope properties. On the steep-slope properties, there was an additional reduction of risk when comparing 24–30 m (75–100 ft) to 16–23 m (50–75 ft). However, the risk of a home being destroyed was slightly significantly higher when there was 31–90 m (101–225 ft) compared to 16–23 m (50–75 ft). For distances that were measured regardless of property boundary (total clearance), the only significant differences in risk of burning were a reduction in risk for 8–15 m (25–50 ft) compared to 0–7 m (0–25 ft).

Table 2. Number of burned and unburned structures within defensible space distance categories (m), their relative risk and significance

A relative risk of 1 indicates no difference; <1 means the chance of a structure burning is less than the other group; >1 means the chance is higher than the other group. The relative risk is calculated for pairs that include the existing row and the row above. Confidence intervals are in parentheses

	Distance within property				Total distance			
	Burned	Unburned	Relative risk	*P*	Burned	Unburned	Relative risk	*P*
Shallow slope								
0–7	200	186			162	114		
8–15	109	198	0.69 (0.12)	<0.001	108	132	0.77	0.002
16–23	51	89	1.03 (0.30)	0.850	78	90	1.03	0.770
24–30	36	40	1.30 (0.39)	0.110	50	70	0.90	0.430
31–90	28	47	0.79 (0.24)	0.220	79	99	1.06	0.640
60 or 90+	10	6	1.67 (0.63)	0.040	8	9	1.01	0.830
Steep slope								
0–7	245	128			224	128		
8–15	174	148	0.82 (0.10)	0.001	158	139	0.84	0.008
16–23	85	68	1.03 (0.16)	0.750	73	83	0.87	0.210
24–30	29	56	0.61 (0.17)	0.004	26	50	0.73	0.080
31–	29	28	1.49 (0.48)	0.050	39	68	1.06	0.760
60 or 90+	5	5	0.98 (0.47)	0.950	4	8	0.91	0.830

Table 3. Effective treatment results reflecting the distance (in metres, with feet in parentheses) and percentage clearance within properties that provided significant improvement in structure survival during wildfires

The property mean is the average distance of defensible space or percentage clearance that was calculated on the properties before the wildfires and provides a means to compare the effective treatment result to the actual amount on the properties

	All parcels effective treatment ($n = 2000$)	Parcel mean	Shallow slope (mean 8%) effective treatment ($n = 1000$)	Parcel mean	Steep slope (mean 27%) effective treatment ($n = 1000$)	Parcel mean
Defensible space within parcel	10 (33)	13 (44)	4 (13)	14 (45)	25 (82)	11 (35)
Total distance defensible space	10 (32)	19 (63)	5 (16)	20 (67)	20 (65)	18 (58)
Mean percentage clearance on property	36	48	31	51	37	35

Effective treatment analysis

Analysis of the treatment–response relationships among defensible space and structures that survived wildfire showed that, when all structures are considered together, the mean actual defensible space that existed around structures before the fires was longer than the calculated effective treatment (Table 3). Regardless of whether the defensible space was measured within or beyond property boundaries, the estimated effective treatment of defensible space was nearly the same at 10 m (32–33 ft).

The effective treatment distance was much shorter for structures on shallow slopes (4–5 m (13–16 ft)) than for structures on steep slopes (20–25 m (65–82 ft)), but in all cases was <30 m (<100 ft). Although longer distances of defensible space were calculated as effective on steeper slopes, these structures actually had shorter mean distances of defensible space around their properties than structures on low slopes (Table 3).

The calculated effective treatment of the mean percentage clearance on properties was 36% for all properties, 31% for structures on shallow slopes and 37% for structures on steep slopes (Table 3). In total, the properties all had higher actual percentage clearance on their property than was calculated to be effective. However, this mainly reflects the shallow-slope properties, as those structures on steep slopes had less clearance than the effective treatment.

Multiple regression analysis

When defensible space was measured only to the property boundaries, it was not included in the best model, according to the all-subsets multiple regression analysis (Table 4). However, it was included in the best model when factoring in the distance of defensible space measured beyond property boundaries (Table 5). In both multiple regression analyses, low housing density and shorter distances to major roads were ranked as the most important variables according to their Akaike weights. Slope and surrounding fuel type were also in both of the best models as well as other measures of defensible space, including the percentage clearance on property and whether vegetation was overhanging the structure's roof. The number of sides in which vegetation was touching the structure was included in the best model when defensible space was only measured to the property boundary. The total explained deviance for the multiple regression models was low (12–13%) for both analyses.

Table 4. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured within property boundary only. Top-ranked models include all those ($n = 12$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	12
Distance to major road	1	−0.0005	12
Percentage clearance	1	−0.02	12
Slope	1	0.03	12
Vegetation overhang roof	1	0.5	12
Fuel type	0.67	Factor	9
Vegetation touch structure	0.49	0.07	6
Distance defensible space within property	0.45	−0.0002	5
South-westness	0.36	−0.0007	3
Distance to minor road	0.28	−0.0002	1
D^2 of top-ranked model			0.123

Table 5. Results of multiple regression models of destroyed homes using all possible variable combinations and corrected Akaike's Information Criterion (AICc)

Includes variables measured beyond property boundary. Top-ranked models include all those ($n = 6$) with AICc within 2 of the model with the lowest AICc. Relative variable importance is the sum of 'Akaike weights' over all models including the explanatory variable

Variable in order of importance	Relative variable importance	Model-averaged coefficient	Number inclusions in top-ranked models
Housing density	1	−0.003	6
Distance to major road	1	−0.0005	6
Total distance defensible space	1	−0.004	6
Percentage clearance	1	−0.01	6
Vegetation overhang roof	0.99	0.4	6
Slope	0.99	0.03	6
Fuel type	0.86	Factor	4
South-westness	0.42	−0.0009	2
Distance to minor road	0.36	−0.0009	2
Neighbours' vegetation	0.27	0.08	1
Vegetation touch structure	0.27	0.18	1
D^2 of top-ranked model			0.125

Surrounding matrix

The cover type that most frequently surrounded the structures at the end of the defensible space measurements was urban vegetation, followed by urban vegetation leading into wildland vegetation, and wildland vegetation (Fig. 5). Many structures were equally surrounded by different cover types. There were no significant differences in the proportion of structures destroyed depending on the surrounding cover type. However, a disproportionately large proportion of structures burned (28 *v.* 9% unburned) when they were surrounded by urban vegetation that extended straight into wildland vegetation.

Discussion

For homes that burned in southern Californian urban areas adjacent to non-forested ecosystems, most burned in high-intensity Santa Ana wind-driven wildfires and defensible space increased the likelihood of structure survival during wildfire. The most effective treatment distance varied between 5 and 20 m (16–58 ft), depending on slope and how the defensible space was measured, but distances longer than 30 m (100 ft) provided no significant additional benefit. Structures on steeper slopes benefited from more defensible space than structures on shallow slopes, but the effective treatment was still less than 30 m (100 ft). The steepest overall decline in destroyed structures occurred when mean defensible space increased from 0–7 m (0–25 ft) to 8–15 m (26–50 ft). That, along with the multiple regression results showing the significance of vegetation touching or overhanging the structure, suggests it is most critical to modify vegetation immediately adjacent to the house, and to move outward from there. Similarly, vegetation overhanging the structure was also strongly correlated with structure loss in Australia (Leonard *et al.* 2009).

In terms of fuel modification, the multiple regression models also showed that the percentage of clearance was just as, or more important than, the linear distance of defensible space.

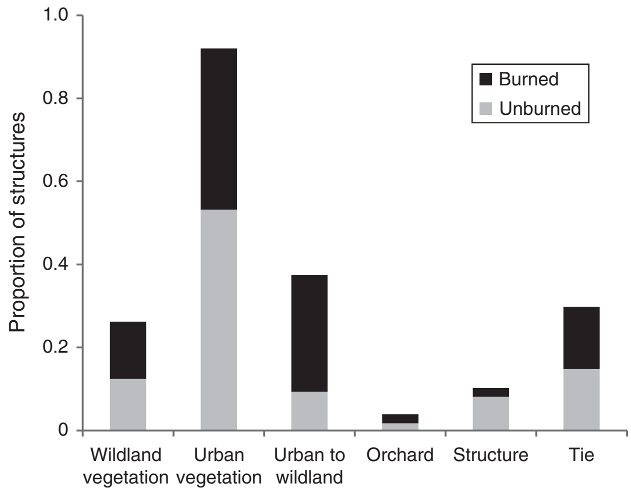

Fig. 5. Proportion of destroyed and unburned structures based on the primary surrounding cover type at the end of defensible space measurements. There were no significant differences in the proportion of burned and unburned structures within cover types ($P = 0.14$). Cover types are defined in Table 1.

However, as with defensible space, percentage clearance did not need to be draconian to be effective. Even on steep slopes, the effective percentage clearance needed on the property was <40%, with no significant advantage beyond that. Although these steep-slope structures benefited more from clearance, they tended to have less clearance than the effective amount, which may be why slope was such an important variable in the multiple regression models. Shallow-slope structures, in contrast, had more clearance on average than was calculated to be effective, suggesting these property owners do not need to modify their behaviours as much relative to people living on steep slopes.

Although the term 'clearance' is often used interchangeably with defensible space, this term is incorrect when misinterpreted to mean clearing all vegetation, and our results underline this difference. The idea behind defensible space is to reduce the continuity of fuels through maintenance of certain distances among trees and shrubs. Although we could not identify the vertical profile of fuels through Google Earth imagery, the fact that at least 60% of the horizontal woody vegetative cover can remain on the property with significant protective effects demonstrates the importance of distinguishing defensible space from complete vegetation removal. Thus, we suggest the term 'clearance' be replaced with 'fuel treatment' as a better way of communicating fire hazard reduction needs to home owners.

The percentage cover of woody shrubs and trees was not evenly distributed across properties, and we did not collect data describing how the cover was distributed. Considering the importance of defensible space and vegetation modification immediately adjacent to the structure, it should follow that actions to reduce cover should also be focussed in close proximity to the structure. The hazard of vegetation near the structure has apparently been recognised for some time (Foote *et al.* 1991; Ramsey and McArthur 1994), but it is not stressed enough, and rarely falls within the scope of defensible space guidelines or ordinances.

In addition to the importance of vegetation overhanging or touching the structure, it is important to understand that ornamental vegetation may be just as, if not more, dangerous than native vegetation in southern California. Although the results showed no significant differences in the cover types in the surrounding matrix, there was a disproportionately large number of structures destroyed (28% burned *v.* 9% unburned) when ornamental vegetation on the property led directly into the wildland. Ornamental vegetation may produce highly flammable litter (Ganteaume *et al.* 2013) or may be particularly dangerous after a drought when it is dry, or has not been maintained, and species of conifer, juniper, cypress, eucalypt, *Acacia* and palm have been present in the properties of many structures that have been destroyed (Franklin 1996). Nevertheless, ornamental vegetation is allowed to be included as defensible space in many codes and ordinances (Haines *et al.* 2008).

One reason that longer defensible space distances did not significantly increase structure protection may be that most homes are not destroyed by the direct ignition of the fire front but rather due to ember-ignited spot fires, sometimes from fire brands carried as far as several km away. Although embers decay with distance, the difference between 30 and 90 m (100 and 300 ft) may be small relative to the distance embers travel under the severe wind conditions that were present at the time of the fires. The ignitability of whatever the embers land on, particularly adjacent to the house, is therefore most critical for propagating the fire within the property or igniting the home (Cohen 1999; Maranghides and Mell 2009).

Aside from roofing or home construction materials and vegetation immediately adjacent to structures (Quarles *et al.* 2010; Keeley *et al.* 2013), the flammability of the vegetation in the property may also play a role. Large, cleared swaths of land are likely occupied at least in part by exotic annual grasses that are highly ignitable for much of the year. Conversion of woody shrubs with higher moisture content into low-fuel-volume grasslands could potentially increase fire risk in some situations by increasing the ignitability of the fuel; and if the vegetation between a structure and a fire is not readily combustible, it could protect the structure by absorbing heat flux and filtering fire brands (Wilson and Ferguson 1986).

The slight increase in proportion of structures destroyed with longer distances of defensible space within parcel boundaries was surprising. However, that increase was not significant in the Chi-square analysis, although there were some significant differences in the pairwise relative risk analysis. Nevertheless, the largest significant effect of defensible space was between the categories of 0–7 m (0–25 ft) to 8–15 m (26–50 ft), and it may be that differences in categories beyond these distances are not highly meaningful or reflect an artefact of the definition of distance categories. These relationships at longer distances are likely also weak compared to the effect of other variables operating at a landscape scale. Although the categorical analysis allowed us to answer questions relative to legal requirements and specific distances, the effective treatment analysis was important for identifying thresholds in the continuous variable.

The multiple regression models showed that landscape factors such as low housing density and longer distances to major roads were more important than distance of defensible space for explaining structure destruction, and the importance of these variables is consistent with previous studies (Syphard *et al.* 2012, 2013), despite the smaller spatial extent studied here. Whereas this study used an unburned control group exposed to the same fires as the destroyed structures, previous studies accounted for structures across entire landscapes. The likelihood of a fire destroying a home is actually a result of two major components: the first is the likelihood that there will be a fire, and the second is the likelihood that a structure will burn in that fire. In this study, we only focussed on structure loss given the presence of a fire, and the total explained variation for the multiple regression models was quite low at ~12%. However, when the entire landscape was accounted for in the total likelihood of structure destruction, the explained variation of housing density alone was >30% (Syphard *et al.* 2012). One reason for the relationship between low housing density and structure destruction is that structures are embedded within a matrix of wildland fuel that leads to greater overall exposure, which is consistent with Australian research that showed a linear decrease of structure loss with increased distance to forest (Chen and McAneney 2004). That research, however, only focussed on distance to wildland boundaries and did not quantify variability in defensible space or ornamental vegetation immediately surrounding structures. Thus, fire safety is important to consider at multiple scales and for multiple variables, which will ultimately require the cooperation of multiple stakeholders.

Conclusions

Structure loss to wildfire is clearly a complicated function of many biophysical, human and spatial factors (Keeley *et al.* 2009; Syphard *et al.* 2012). For such a large sample size, we were unable to account for home construction materials, but this is also well understood to be a major factor, with older homes and wooden roofs being most vulnerable (Franklin 1996; Cohen 1999, 2000). In terms of actionable measures to reduce fire risk, this study shows a clear role for defensible space up to 30 m (100 ft). Although the effective distances were on average much shorter than 30 m (100 ft), we recognise that additional distance may be necessary to provide sufficient protection to firefighters, which we did not address in this study (Cheney *et al.* 2001). In contrast, the data in this study do not support defensible space beyond 30 m (100 ft), even for structures on steep slopes. In addition to the fact that longer distances did not contribute significant additional benefit, excessive vegetation clearance presents a clear detriment to natural habitat and ecological resources. Results here suggest the best actions a homeowner can take are to reduce percentage cover up to 40% immediately adjacent to the structure and to ensure that vegetation does not overhang or touch the structure.

In addition to defensible space, this study also underlines the potential importance of land use planning to develop communities that are fire safe in the long term, in particular through their reduction to exposure to wildfire in the first place. Localised subdivision decisions emphasising infill-type development patterns may significantly reduce fire risk in the future, in addition to minimising habitat loss and fragmentation (Syphard *et al.* 2013). This study was conducted in southern California, which has some of the worst fire weather in the world and many properties surrounded by large, flammable exotic trees. Therefore, recommendations here should apply to other non-forested ecosystems as well as many forested regions.

Acknowledgements

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