

QUANTIFYING THE INFLUENCE OF PAST WILDFIRES ON THE SEVERITY
AND SIZE OF SUBSEQUENT WILDFIRES

By

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Quantifying the influence of past wildfires on the severity and size of subsequent wildfires

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Wildfire is arguably one of the most important and widespread natural disturbance agents in western U.S. forests. It has a substantial impact on ecosystem structure and function by influencing soils, nutrients, carbon budgets, wildlife habitat, and vegetation. Wildfires also influence fuel amount, type, and structure, potentially influencing the severity and size of subsequent wildfires through site- and landscape-level feedback mechanisms. Until relatively recently, the ability to quantitatively evaluate how these feedback mechanisms operate has not been feasible because of data limitations (i.e. there has not been enough wildfire). However, due to increased fire activity over the last ~25 years, there are a number of examples of wildfires “interacting” with subsequent fires, where a wildfire either burns within the perimeter of a previously burned area (i.e. it reburns) or burns up to (but not in to) a previously burned area. This recent surge in fire activity, along with increased availability of remotely sensed data, now makes it possible to evaluate how wildfires influence subsequent fire severity and size over large landscapes. Some studies have suggested that extreme weather conditions may decrease the strength of the feedback mechanisms associated with interacting fires, and consequently, evaluating the influence of weather on such relationships is increasingly important, especially given that climate change is expected to result in more extreme weather events.

This dissertation is composed of three chapters. The first chapter quantifies how previous wildfire influences the severity of subsequent fires. In my second chapter, I develop and evaluate several approaches to estimate day-of-burning for each point within a fire perimeter using coarse-resolution MODIS fire detection data. Knowing the day-of-burning is essential in order to evaluate the influence of observed weather (e.g., from a nearby weather station) on observed fire-related effects, such as smoke production or the previously mentioned feedback mechanisms of fire. My third chapter evaluates the ability of wildfire to act as a fuel break by limiting the extent (i.e. size) of subsequent fire. Using the methods from Chapter Two to estimate day-of-burning, I was also able to evaluate the influence of weather in weakening the strength of this feedback.

Chapter 3:

Ability of wildfires to limit the extent of subsequent fires

Abstract

Theory suggests that fire size can be limited by previous fires in landscapes with active fire regimes. However, empirical examples of this pattern-process feedback (also termed ‘self-regulation’) are surprisingly rare due to data limitations resulting from an overall lack of fires on the landscape due to fire exclusion policies. Given the increase in fire activity over the last ~25 years in the western US, there are now opportunities to evaluate these spatial feedbacks and explicitly quantify the ability of wildfire to limit the size, or extent, of subsequent fires. Understanding weather’s influence on the ability of wildfires to act as future fuel breaks is also necessary given that extreme fire-conductive weather may moderate this effect and may become more common in the future due to climate change. In this study, I evaluated the ability of wildfire to limit the extent of subsequent fires along a temporal gradient in four large study areas in the western US that have experienced substantial fire activity in recent decades. Using fire progression maps in conjunction with weather station data, I also evaluated the influence of daily weather in modifying the effectiveness of wildfire as a fuel break. Results indicate that wildfires do limit subsequent wildfire spread, but this effect decays over time; wildfires no longer act as fuel breaks ~6-17 years after a fire, depending on the study area. I also found that extreme weather substantially moderates this effect; the ability of wildfire to act as a fuel break is ~halved or more under extreme compared to more moderate weather conditions in three of the study areas. These results will be useful to fire managers who seek to restore natural fire regimes or to exploit recent burns when managing fire.

Introduction

Wildland fire is an important ecological process in many ecosystems (Agee 1993); it alters vegetation composition and structure, consumes biomass, and influences landscape heterogeneity. Such fire-induced changes can influence subsequent fire behavior and effects via site- and landscape-level feedbacks (Agee 1999; Peterson 2002; McKenzie et al. 2011). For example, wildfires reduce fuel loads, and if fires recur before sufficient biomass has accumulated, the size and severity of subsequent fires may be limited (Collins et al. 2009; Parks et al. 2014). These feedback mechanisms are considered fundamental ecosystem properties of fire-adapted ecosystems of the western US (McKenzie et al. 2011) but have been largely disrupted in many ecosystems due to successful fire exclusion dating back to the 1930s (Heyerdahl et al. 2001; Taylor and Skinner 2003). Some areas of the western US, however, have experienced substantial fire over the last three decades, partially because some fires were not actively suppressed (termed “resource benefit fires”); these areas provide crucial natural laboratories to explicitly evaluate how these feedback mechanisms function.

Land managers are increasingly recognizing that fire exclusion is problematic for a number of ecological and social reasons. For example, fire exclusion is often cited as the cause of increased tree density and homogenizations in several forest types (Hessburg et al. 2005; Naficy et al. 2010), which in turn has contributed to increases in area burned and fire severity (Stephens 2005; Mallek et al. 2013). Fire suppression activities are

expensive (Gebert et al. 2007) and have a number of adverse ecological consequences such as high-intensity backburns, fireline construction, and fire retardant pollution (Backer et al. 2004). Finally, there is an increasing awareness that wildland fire is a necessary component of healthy ecosystems (Kilgore 1973; Hutto 2008). These factors, combined with acknowledgement that climate change will likely lead to more frequent fire (Littell et al. 2010; Westerling et al. 2011), have increased interest in explicitly and quantitatively evaluating how feedbacks between wildfire and subsequent wildfire operate.

Several recent studies have shown that fire severity is lower in areas that reburned within a previously recorded fire perimeter compared to those that did not (Arkle et al. 2012; Miller et al. 2012; Parks et al. 2014), clearly indicating a strong feedback between wildfire and subsequent fire severity. However, barring those that are inferential (Price et al. 2012; Parisien et al. In press) or involve fire simulation modeling (e.g., Davis et al. 2010), studies explicitly evaluating feedbacks between wildfires and subsequent wildfire size are extremely limited and inconsistent in their results. For example, Collins et al. (2009) found that wildfire indeed limited the size of subsequent fires in upper mixed-conifer forest in the Sierra Nevada, California, whereas Teske et al. (2012) found this effect in only one of three study areas in central Idaho and northern Montana. The ability of wildfires to act as a fuel breaks depends upon underlying contingencies such as time between fires (Peterson 2002), but only a couple of studies have evaluated this factor (Collins et al. 2009; Price and Bradstock 2010). There is a clear need for more information on how wildfires serve as fuel breaks and how this may change as time between fires increases over a broad range of ecosystems and geographies of the western US.

Another understudied aspect of feedbacks between wildfire and subsequent fire size is the influence of weather. Mortiz (2003) suggested that extreme fire weather may override or moderate the effect of a previously burned area in limiting the extent of subsequent fires; this was substantiated by Collins et al. (2009) and Price and Bradstock (2010), who showed that the ability of a wildfire to act as a fuel break decreased as fire weather became more extreme. Further investigation over a broader range of geography and ecosystem types is needed to develop a more comprehensive understanding of fire-weather relationships, especially given the varying influence of bottom-up and top-down controls on fire regimes (e.g., fuels vs. weather) (Heyerdahl et al. 2001; Mermoz et al. 2005; Parks et al. 2012). Such information would be useful in anticipating how the effectiveness of wildfire as a fuel break may weaken under future climatic conditions, which is important considering that extreme fire weather is expected to become more common in the future (Nitschke and Innes 2008).

The first objective of this study was to determine if wildfires limit the extent of subsequent fires, and if so, how this effect changes as time between fires increases. I hypothesized that the effectiveness of wildfire as a fuel break will be greatest immediately after a fire and decay through time. Assuming a fuel break effect is found, my second objective was to determine if extreme fire-conducive weather conditions modify this effect. I hypothesized that the ability of wildfires to act as a fuel break will be weaker and decay faster with increasing fire weather conditions.

Methods

Study area

I conducted this study within four study areas composed entirely of protected areas (wilderness and national park) (Fig. 1), thereby limiting the confounding effects of mechanical fuel treatments that are common outside such areas. The FCW study area is composed of the Frank Church – River of No Return Wilderness in central Idaho. The adjacent SBW encompasses the Selway-Bitterroot Wilderness in western Montana and north-central Idaho. CCE (Crown of the Continent Ecosystem) is comprised of Glacier National Park and the Great Bear, Bob Marshall, and Scapegoat wilderness areas. Finally, GAL incorporates the Gila and Aldo Leopold Wilderness Areas in western New Mexico. These study areas were chosen because they have experienced substantial fire activity in recent decades and thus have enough data to evaluate the effectiveness of wildfire as a fuel break. Although a proportion of ignitions were managed as resource benefit fires in all study areas, some were also actively suppressed.

FCW (Frank Church – River of No Return Wilderness)

The FCW (9777 km²) is the second largest wilderness area in the lower 48 states. Mean annual precipitation is 871 mm and mean annual temperature is 2.7 °C (Daly et al. 2002). However, there is substantial intra-area variation in both mean annual precipitation and temperature (Fig. 2). In this and all study areas, mean annual precipitation is generally lowest in the low elevation river bottoms and highest on the mountain peaks; temperature exhibits the opposite pattern. FCW is rugged; elevations range from 600 to 3136 m. Topographic features include river breaks, deep canyons, mountains, and glaciated basins (USDA Forest Service 2003). Park-like groves of ponderosa pine (*Pinus ponderosa*) exist below about 1500 m on south and west slopes (Barrett 1988). Denser ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) forests occupy north and east aspects, up to elevations of about 2100 m. Still higher, the vegetation transitions to grand fir (*Abies grandis*), lodgepole pine (*P. contorta*), and Englemann spruce (*Picea engelmannii*). At the highest elevations, subalpine fir (*A. lasiocarpa*), whitebark pine (*P. albicaulis*), and alpine environments predominate (Barrett 1988; Finklin 1988). The fire season runs from early-July to mid-September (USDA Forest Service 2013). Low-elevation, open ponderosa pine forests tend to experience frequent, low-intensity fires, and, generally, fire frequency decreases and severity increases with increasing elevation, moisture, and tree density (Crane and Fischer 1986). Fire suppression became effective in about 1935 (Finklin 1988) although sheep grazing may have excluded fire earlier (Steele et al. 1981). Resource benefit fires began to occur in ~1988 (Beckman 2008).

SBW (Selway-Bitterroot Wilderness)

The SBW (5471 km²) is the third-largest wilderness area in the lower 48 states. It includes the Bitterroot mountain range along the Montana and Idaho border and large portions of the Selway and Lochsa watersheds in Idaho. Mean annual precipitation in SBW is 1221 mm and mean annual temperature is 3.5 °C (Daly et al. 2002). Elevations range from 531 m in the Selway River drainage on the western edge to over 3000 m in the southeast portion of the study area. The vegetation of SBW is diverse. Lower elevations (up to ~1500 m) in the west and northwest portion of the study area are characterized by Pacific maritime forests composed of western hemlock (*Tsuga*

heterophylla), western red cedar (*Thuja plicata*), western white pine (*P. monticola*), and Douglas-fir (Rollins et al. 2002). Ponderosa pine is common at lower elevations in other portions of the study area, particularly on dry south-facing slopes (Brown et al. 1994). As elevation increases, Douglas-fir and grand fir are prominent on mesic sites and ponderosa pine, Douglas-fir, and western larch (*Larix occidentalis*) are common on drier sites. The subalpine forests of the higher elevations (> ~2500 m) are composed of a collection of Engelmann spruce, whitebark pine, lodgepole pine, subalpine fir, and alpine larch (*L. lyallii*) (Rollins et al. 2002). At the highest elevations, alpine environments (i.e., barren or snow/ice) are common, especially along the Bitterroot divide. The fire season in SBW runs from late-June through mid-September (Brown et al. 1994). The fire regime is categorized as mixed: lower-severity surface fires are common in the lower elevations and patchy, stand-replacing fires become more common as elevation increases, although during extremely dry years, stand replacing fires can occur throughout the study area (Brown et al. 1994). Fires were actively suppressed until 1972; resource benefit fires were allowed to burn after this point (van Wagtendonk 2007). Cattle and sheep grazing was evident in the early 1900's (USDA Forest Service 1924), which may have decreased fire frequency within portions of SBW.

CCE (Crown of the Continent Ecosystem)

The CCE is the largest (10,331 km²) of the four study areas. Mean annual precipitation in CCE is 1243 mm and mean annual temperature is 2.2 °C (Daly et al. 2002) (Fig. 2). The CCE straddles both the east and west slopes of the continental divide. The northern portion of is composed of Glacier National Park (GNP), where alpine glacial canyons drain into major river valleys (Barrett et al. 1991). South of GNP lays the Great Bear, Bob Marshall, and Scapegoat Wilderness Areas. Elevations in CCE range from 950 m near Lake McDonald in GNP to over 3100 m on the highest mountain peak (also in GNP). Although dependent upon fire history and soil texture, ponderosa pine, lodgepole pine, Douglas fir, western larch are the dominant tree species in low-elevation areas (< ~1500 m) (Arno 1980; Keane et al. 1994; Keane et al. 2006). Western hemlock and western red cedar are present in low-elevation (< 1500 m) wet areas that have been free of fire for extended periods of time (> ~100 years). As elevation increases the dominant species become lodgepole pine, subalpine fir, and Engelmann spruce. Whitebark pine and alpine larch are present near treeline (1800-2300 m elevation, depending on latitude); alpine environments are common above this elevation. Areas of ponderosa pine and mixed-conifer in CCE were historically maintained by low- and mixed-severity regimes (Arno et al. 2000; Keane et al. 2006); the effects of fire exclusion (dense understory and duff accumulation) are evident in these areas. Most of the study area (excluding alpine environments), however, is characterized by a mixed- to high-severity fire regime (Arno et al. 2000). The fire season runs from mid-July through September (USDA Forest Service 2013). Resource benefit fires began in the Bob Marshall wilderness in 1981 and in GNP in 1994.

GAL (Gila and Aldo Leopold Wilderness)

The GAL (3087 km²) is the driest and warmest of the four study areas; mean annual precipitation is 578 mm and mean annual temperature is 10.4 °C (Daly et al. 2002) (Fig. 2). Elevations range from 1462 to 3314 m. The topography is diverse,

composed of mountains, broad valleys, steep canyons, and extensive mesas. At the lowest elevations, the vegetation is desert scrub and grasslands (*Ceanothus*, *Artemisia*, and *Yucca* spp.). As elevation increases, it transitions to piñon-oak-juniper woodland (*P. edulis engelmannii*, *Juniperus deppeana*, *J. monosperma*, and *Quercus* spp.), and then to ponderosa pine woodland and forest. The highest elevations are composed of Douglas-fir, Englemann spruce, white fir (*A. concolor*), subalpine fir, southwestern white pine (*P. strobiformis*), and aspen (*Populus tremuloides*) forests (Rollins et al. 2002). Although the fire season runs April through September, mid-summer fires are uncommon due to rains associated with monsoonal storms from the Gulf of Mexico (Rollins et al. 2002). Fires in GAL are generally frequent and low-severity surface fires, but fire severity tends to increase with elevation (Swetnam and Dieterich 1985) and varies with aspect, incident radiation and topographic position (Holden et al. 2009). Extensive cattle and sheep grazing began in the 1890's, which substantially reduced fine fuel amount and continuity and caused a decrease in fire frequency (Swetnam and Dieterich 1985; Swetnam and Baisan 1996). Resource benefit fires began to occur in 1975 (Swetnam and Dieterich 1985).

Analyses

Development of geospatial fire atlas

Creating the geospatial fire atlas for each study area was a multi-step process. First, I obtained fire perimeters from the Monitoring Trends in Burn Severity (MTBS) project (Eidenshink et al. 2007), which has mapped the perimeter and severity of fires ≥ 400 ha in the western US from 1984-2011. Next, I supplemented the MTBS fire perimeters by identifying and mapping all fires ≥ 20 ha from 1972-2012 using the entire record of Landsat data, including the multi-spectral sensor (MSS), thematic mapper (TM), enhanced thematic mapper plus (ETM+), and operational land imager (OLI) sensors. This was conducted by obtaining virtually all snow-free images for each study area from the US Geological Survey Center for Earth Resources Observation and Science (USGS-EROS) (available from <http://earthexplorer.usgs.gov/>) and identifying and mapping areas of change between image dates. Identifying and mapping fires with the MSS imagery (circa 1972-1984) relied primarily on evaluating differences between pre- and post-fire NDVI (normalized differenced vegetation index) (dNDVI). For the Landsat TM, ETM+, and OLI data (1984-2012), however, I delineated fire perimeters by evaluating differences between pre- and post-fire NBR (normalized burn ratio) (dNBR) (Key and Benson 2006). I converted the reflective and thermal bands of each Landsat scene into top-of-atmosphere reflectance and brightness temperature respectively, and produced multi-date comparisons of all NDVI/NBR scenes within each year. A linear grayscale was assigned to dNDVI and dNBR imagery typically in the range of -800 to +1100 for best contrast in delineating fire perimeters. To identify and map fires in GAL, I also used two relativized metrics of fire-induced change (RdNBR, Miller and Thode 2007; RBR, Parks et al. 2014) since these severity indices provided higher contrast in the more sparsely vegetated study area. Supplementary spatial data were also used to confirm the presence of fire, including Moderate Resolution Imaging Spectroradiometer (MODIS) fire detections (USDA Forest Service 2013) (2001-2012), National Interagency Fire Management Integrated Database (<https://fam.nwcg.gov/fam-web/kcfast/html/ocmenu.htm>) (1972-2012), Geospatial Multi-Agency Coordination

Group fire perimeters (<http://www.geomac.gov/index.shtml>) (2001-2011), and various regional fire atlases for the Gila Wilderness (Rollins et al. 2001) (1972-1997), Northern Rocky mountains (Gibson 2006) (1972-2003), and the Flathead National Forest (<http://www.fs.usda.gov/detailfull/flathead/landmanagement/gis>) (1980-2012). All geospatial operations were conducted using either ArcMap 10.1 (ESRI Inc. 2012) or the “raster” package (Hijmans and van Etten 2011) within the R statistical program (R Development Core Team 2007).

Numerous MTBS fire perimeters were modified because they incorrectly mapped two fires from different years as one fire or where multiple MTBS fires in a year actually represented one contiguous fire or fire complex. The final product is a geospatial fire atlas for all fires ≥ 20 ha from 1972-2012. All fire perimeters were converted to raster format with a 30 x 30 meter pixel size (matching the resolution of Landsat TM, ETM, and OLI data).

Identifying limiting fire perimeters

Previous wildfires interact with subsequent fire by either stopping the spread or getting reburned by a subsequent fire. As such, I developed an objective and consistently applied rule-set to identify wildfire perimeters, or portions thereof, that either limited or did not limit the spread of subsequent fires. First, each pixel of each fire perimeter was evaluated to determine if it interacted with a subsequent fire, defined by either 1) a fire perimeter pixel is within 375 m of a subsequent fire or 2) a fire perimeter pixel is reburned by a subsequent fire. The 375 m distance threshold allows for error in wildfire perimeter mapping due to the spatial and spectral diversity caused by variability in fire severity, vegetation type, and speed of vegetation recovery (Holden et al. 2005). Next, I determined whether interacting pixels did or did not limit the extent of subsequent fires. If a subsequent fire perimeter was ≤ 375 m as measured outwards from the initial fire perimeter and ≤ 750 m as measured inwards (i.e. the subsequent fire infiltrated the initial fire perimeter by ≤ 750 m), then I assumed that the pixel was limiting the extent of the subsequent fire (Fig. 3); hereafter, these proximal and interacting pixels are referred to as LIMITING. In this case, the 750 m threshold acknowledges that wildfires may limit subsequent fire size even though it may reburn along the perimeter of a previous fire. If a pixel from a subsequent fire perimeter infiltrated > 750 meters and reburned a previous fire, then I assumed that the subsequent fire was not limited in extent by the initial wildfire; hereafter, these interacting pixels are referred to as NOT LIMITING. If a pixel from a subsequent fire was > 375 m from a fire perimeter, I assumed that that there was no interaction and the pixel was excluded from further analyses (Fig. 3). Preliminary analyses indicated that many false-positives resulted from this rule-set (e.g. pixels were mislabeled as LIMITING, see Fig. 3d), prompting an additional step to minimize this occurrence: if greater than 35% of the area of the initial or subsequent wildfire overlapped, then all proximal pixels were identified as NOT LIMITING. All pixels from all fires were thus labeled as LIMITING, NOT LIMITING, or excluded from the analyses. To clarify, the analyses units are pixels along the perimeter boundary, or edge, of the initial wildfire; no pixels from the interior of the initial fire perimeter are analyzed.

Exploratory analyses indicated there are individual cases where the thresholds described above failed and perimeter pixels were seemingly mislabeled as LIMITING or NOT LIMITING. I found that, although changing the thresholds may alleviate this issue

for individual cases, it seemingly mislabeled pixels of other fires. I evaluated alternative thresholds in these exploratory analyses (250 and 500 m vs. 375 and 750 m); the results were surprisingly similar to those reported here, which suggests that minor changes in threshold values do not substantially change the findings of this study.

Statistical model

To quantify the ability of wildfires to serve as fuel breaks, and how this ability may change as time between fires increases, I built logistic regression models (using the logit function) with LIMITING vs. NOT LIMITING as the binary response variable and time between fires (years) as the explanatory variable. I built these models with two sets of data for each study area, one with all fires (≥ 20 ha) and another with large fires (≥ 400 ha). I built two models for two reasons. First, it is probable that some of the smaller fires in my study did not burn in a subsequent fire event although the fire perimeter data would indicate that it did (falsely labeling such pixels as NOT LIMITING). This is due to difficulty in identifying and mapping unburned islands within a fire perimeter. A model including only large fires reduces the chance of this occurring. Second, some have suggested that small fuel treatments are ineffective at limiting fire spread (e.g., Graham 2003); excluding small fires (< 400 ha) acknowledges this notion. Although the fire perimeter data span 41 years, I removed all interactions older than 25 years from the analysis. This was because initial data exploration indicated that there were only small amounts of data beyond 25 years between fires and there appeared to be no effect of wildfire as a fuel break beyond this time, although this could simply be due to the lack of data. Model fits are evaluated with the area under curve calculation for the receiver operating characteristic curve (ROC) as calculated with the 'verification' package in R (NCAR - Research Applications Laboratory 2013).

To test for model significance while minimizing the effects of spatial autocorrelation, which tends to overfit models and inflate statistical significance (Legendre and Fortin 1989; Legendre 1993), I used a subsampling and multi-model approach similar to that described by Parisien et al. (2011). Specifically, for each logistic regression model described above and below, I generated a model ensemble using 2500 random subsets of data; the subsampling frequency was 1% of the full dataset. The model ensemble p-value for each variable (which is the average p-value of each of the 2500 models) was used to test whether or not the independent variables were statistically significant. I chose a 1% subsampling frequency based on Parks et al. (2014) who used ~0.1% subsampling frequency for two-dimensional data; since fire perimeter edges are linear, one-dimensional features, I assumed that this sampling frequency was appropriate. A 1% sampling frequency indicates that, on average, one pixel is selected for every 3 km of interacting fire perimeter in each random subset of data.

Incorporating weather into statistical models

To evaluate how weather conditions may affect the ability of a wildfire to limit subsequent fire extent, I built a second set of logistic regression models for each study area that also included a fire weather index (in addition to time between fires) as an explanatory variable. I used the energy release component (ERC) to represent fire weather, which is commonly used in fire studies (e.g., Abatzoglou and Kolden 2013; Riley et al. 2013). ERC is related to the amount of heat released per unit area at the

flaming front of a fire (Bradshaw et al. 1983) but can also be considered a fuel moisture metric that represents long term drying (Andrews et al. 2003). Daily ERC was generated using Fire Family Plus software (Bradshaw and McCormick 2000) and remote automated weather station (RAWS) data for stations within or in close proximity to each study area (Lodgepole RAWS for FCW, Hells Half Acre for SBW, Spotted Bear Ranger Station for CCE, and Beaverhead for GAL). ERC was calculated using the NFDRS fuel model G for all study areas except GAL, in which I used fuel model K.

I then assigned these daily ERC values to each 30 x 30 m pixel within each large fire that burned between 2001 and 2012 based on the estimated day of burning. Because agency generated fire progression maps were not available for a large number of fires in my study, I estimated day-of-burning using the methods developed by Parks (2014), where day-of-burning for each 30 x 30 m pixel, and hence fire progression, was calculated by spatially interpolating Moderate Resolution Imaging Spectrometer (MODIS) fire detection data (NASA MCD14ML product, Collection 5, Version 1). Due to the coarse nature of the MODIS input data (1 km²), this process was limited to large fires and to fires burning after 2000 to coincide with the operational timeline of the MODIS sensors. MODIS fire detection data depict the date and location (i.e. pixel centroid) of actively burning MODIS pixels, and although the spatial resolution is relatively coarse (pixel size = 1 km²), the fine temporal resolution (there are two MODIS sensors, each passing two times per day) allows day-of-burning to be mapped at finer spatial resolution via interpolation.

The models that incorporate weather employ a subset of data; they include only large fires (≥ 400 ha), and further, those large fires must interact with fires that occurred between 2001 and 2012. For example, a 1000 ha fire from 1990 that interacts with a 1500 ha fire from 1999 is excluded from the analysis because the 1999 fire occurred prior to MODIS; it is also excluded if it interacts with a 300 ha fire from 2003 because the 2003 fire was too small to use day-of-burning interpolation. However, if the same fire interacts with a 1500 ha fire from 2003, then it is included in the analysis since MODIS data can be used to estimate day of burning for the subsequent 2003 fire. For each interacting fire perimeter pixel, I extracted the daily ERC value that was associated with the subsequent fire. In those cases when a wildfire did not technically overlap but was within 375 m from a subsequent fire, I used the day-of-burning estimate, and hence the ERC value, of the nearest pixel of the subsequent fire. I assessed significance of ERC using the subsampling and model ensemble approach described above. Interactions between time and ERC were not evaluated for simplicity.

Results

A total of 1038 fires and 437 large fires were identified between 1972 and 2012 across all study areas. A majority of these ($> 60\%$) interacted with a subsequent fire (Table 1). The FCW had the highest number of large fires and the greatest amount of total area burned. SBW had the most fires (≥ 20 ha) ($n=373$) during this time period, but on average, those fires were smaller compared to the other study areas (average fire size in SBW = 685 ha). GAL (the smallest study area), on the other hand, experienced the least number of fires (≥ 20 ha). Proportionally, CCE burned the least (0.30) over the 1972-2012 time period whereas GAL burned the most (1.12) (Table 1).

In all study areas, the proportion of pixels defined as LIMITING generally decreased as time until subsequent fire increases (Fig. 4) for both sets of wildfires analyzed (all fires and large fires). Consequently, the logistic regression models indicate that the ability of wildfires to limit the extent of subsequent fires is strongest immediately after a fire but decays over time (Figs. 4 and 5). Wildland fires no longer act as an effective fuel break (defined here as a ≤ 0.30 probability of limiting extent of subsequent fire) after ~6 years in GAL and ~16 in the three northern study areas (Figs 4 and 5; Table 2). Overall, the relationship between the effectiveness of fire as a fuel break and time between fires is distinctly different in GAL (i.e. it is weaker and decays faster) compared to the northern study areas of FCW, SBW, and CCE (Fig. 5). Large wildfires in FCW, SBW, and CCE are over 75% effective at limiting the extent of subsequent wildfires for up to four years, diminishing to ~50% 11 years after wildfire (Fig. 5). Model fits, as measured with the ROC statistic, range from 0.72 (FCW) to 0.82 (GAL) for the models including all fires and range from 0.77 (FCW and SBW) to 0.87 (CCE) for those including large fires. The model ensembles with randomly subset data indicate that all models are statistically significant ($p \leq 0.001$).

In all study areas, the ability of wildfire to act as a fuel break weakens with increasing fire-conducive weather conditions (Fig. 6). For example, ten years after wildfire in CCE, the ability of fire to act as a fuel break is very high under moderate conditions (probability = 0.97; 50th percentile ERC) but is very weak and no longer acts as an effective fuel break under extreme conditions (probability < 0.30; 99th percentile ERC). The length of time in which wildfire no longer acts as an effective fuel break (again defined as ≤ 0.30 probability of limiting extent of subsequent fire) is substantially shorter under extreme vs. moderate weather conditions (99th vs. 50th percentile ERC) (Fig. 6; Table 2). In GAL, for example, wildfire no longer acts as a fuel break after two years under extreme conditions compared to eight years under moderate conditions. The influence of ERC was statistically significant ($p \leq 0.03$ in all study areas) according to the model ensembles. Delta ROC values (comparing a model with and without ERC) ranged from 0.00 (FCW) to 0.05 (CCE).

Discussion

Theory suggests that in landscapes with an active fire regime, landscape pattern is shaped by wildfire, but wildfire is also shaped by landscape pattern. This pattern-process feedback loop, also termed self-regulation, is a fundamental concept in disturbance ecology (Turner 1989; Agee 1999) and underscores the importance of wildfire in creating and maintaining resilient landscapes (McKenzie et al. 2011). The results of this study clearly indicate that wildfires act as fuel breaks and limit the extent of subsequent wildfires across my four western US study areas, supporting the notion of self-regulation in landscapes with active fire regimes. The strength of this feedback, however, decays over time and is completely diminished by ~6-16 years after a wildfire, depending on the study area. This suggests that the “ecological memory”, defined as the degree to which ecological processes are shaped by past disturbance events (Peterson 2002), at least in terms of wildfire’s ability to act as a fuel break, is relatively short. However, the pattern-process feedback loop of wildfire not only limits subsequent fire extent, but limits subsequent fire severity (Parks et al. 2014), an effect that can last for decades (Miller et al. 2012), suggesting that the ecological memory of wildfire in terms of subsequent fire

severity is much longer. Since federal agencies spend millions of dollars each year on fuel treatments to reduce fire hazard and risk in fire prone landscapes (Allen et al. 2002), it is critical to understand how wildfires may also serve as fuel treatments, both in terms of how they limit subsequent fire extent and severity. As such, my study has the potential to help managers make more informed decisions about how to best manage a particular wildfire through assessing its potential longevity for constraining future fires and understanding the limitations under extreme weather conditions.

In terms of time between fire events, my findings are broadly similar to those of Collins et al. (2009), who also found that the ability of fire to act as a fuel break decays over time. My findings, however, are less consistent with those of Teske et al. (2012), who found that wildfire limited the extent of subsequent wildfires in only one of the three study areas they examined. I evaluated the same three study areas (FCW, SBW, and CCE) as Teske et al. (2012) and found that wildfires definitively act as fuel breaks in all three areas, especially in the immediate years following a fire, so it is somewhat surprising that our findings are not in agreement. The likely explanation for the lack of agreement involves methodological differences; Teske et al. (2012) did not include a statistical evaluation of time between fires in their analyses, and in not doing so, may have muted the statistical signal of fire as a fuel break. Given my findings that wildfire's ability to act as a fuel break decays relatively quickly and is completely diminished by ~16 years after a fire in these study areas, investigations of this sort should explicitly address time between fires.

In all study areas, the effectiveness of wildfire as a fuel break weakens with increasing fire weather, which was also noted Collins et al. (2009). In fact, my results indicate that, in three out of four study areas, the longevity of the ability of fire to act as a fuel break effect is at least ~halved or more under extreme (99th percentile ERC) compared to more moderate fire-season weather conditions (50th percentile), thereby supporting the assertion that the importance of fuels diminishes during extreme weather events (Bessie and Johnson 1995; Price and Bradstock 2011). Nevertheless, my results indicate that fuels, or lack thereof due to burning, strongly limit fire (probability of limiting subsequent fire ≥ 0.65) in the northern study areas for at least three years following fire even under extreme conditions. Conversely, in GAL, which is generally comprised of dry conifer forest, the ability of fire to act as a fuel break lasts for only two years (probability ≤ 0.3) under extreme fire weather conditions; a study by Price and Bradstock (2010) revealed similar findings in a dry forest in Australia. From a climate change perspective, extreme weather conditions are projected to become more common (Salinger 2005; Nitschke and Innes 2008), and in fact, there is evidence that such changes are already occurring (Collins 2014). As such, the strength and longevity of wildfire in limiting the extent of subsequent fires will be likely be reduced in future years, reinforcing the results from other studies suggesting that climate change will result in higher fire activity in many areas of the western US (Westerling and Bryant 2008; Littell et al. 2010; Moritz et al. 2012).

Some studies have argued that the distribution of fire sizes is dictated by endogenous factors, implicitly implying that fuel availability solely drives fire sizes (Malamud et al. 1998; Turcotte and Malamud 2004). Others, however, have argued that exogenous factors such as weather are responsible for fire size distributions (Boer et al. 2008). Our results suggest that both fuel availability and weather (endogenous and

exogenous factors) are responsible for fire sizes, supporting the assertion of Moritz et al. (2005) who posit that fire size is controlled by multiple factors. Our results further suggest that the influence of weather may vary among regions, being more influential in CCE and GAL (based on improved model fits and relative decreases in the longevity of wildfire to act as a fuel break under extreme conditions [Table 2]). These differences may be due to factors such as variability in vegetation and drought frequency (Wang et al. In press). However, these differences could also be because the fire weather data may imperfectly represent the conditions influencing some fires because the procedure I used to estimate day-of-burning, and therefore ERC, has a moderate degree of uncertainty (Parks 2014), meteorological conditions are highly spatially heterogeneous (Holden and Jolly 2011), and weather station siting may bias observations (Myrick and Horel 2008).

Pyrogeographic differences among the study areas are evident and are likely due to differences in climate and ecosystem response to fire (Keeley et al. 2008; Freeman and Kobziar 2011). The southwest study area in particular, composed of the Gila and Aldo Leopold Wilderness areas (GAL), is strikingly different than the other three study areas in terms of the strength and longevity of wildfire to act as a fuel break. This difference is likely a reflection of differences in climate and fire regime characteristics in GAL. The fire regime in GAL is for the most part characterized by frequent surface fire dependent upon fine fuel availability and continuity (Schoennagel et al. 2004). As such, large fire years tend to occur one to three years after a wet (i.e. high precipitation) year (Swetnam and Baisan 1996); fine fuel growth and accumulation stimulated during wet years therefore erases the effects of the previous fire in terms of its ability to act as a fuel break and, consequently, wildfires are not likely to act as fuel breaks for periods of time exceeding ~6 years. In contrast, the other study areas generally experience less frequent but higher severity fires (Parks et al. 2014) that are more dependent upon ladder and canopy fuels (Schoennagel et al. 2004). Such ladder and canopy fuels take longer to recover after fire, hence the increased longevity of fire as a fuel break in FCW, SBW, and CCE. I suggest similar studies should be conducted in other study areas representing different ecosystems (e.g., chaparral and boreal systems) to gain a broader pyrogeographic perspective. Broader theoretical perspectives may also be necessary, because although fire may act as a fuel break *if* a subsequent fire occurs nearby, the probability of a subsequent fire interacting with a previous fire may be quite low (e.g., Price et al. 2012).

Several aspects of my analyses likely influence the results of this study. First, I assumed that a wildfire limited the extent of a subsequent wildfire if pixels on the perimeters of both wildfires were proximal. Because other features such as mountain ridges or rivers may influence fire boundaries, this assumption may not always hold true. However, given the strong signal of time between fire events, I surmise this assumption has a negligible influence on my results. Second, it is possible that a wildfire limited the extent of a subsequent wildfire even if infiltrated it by more than 750 m (I labeled these pixels as NOT LIMITING). Due to the logistic regression framework utilized in this study, it was necessary to define perimeter pixels in a binary fashion. The implication of this second issue is that I potentially underestimate the strength and longevity of wildfire's ability to limit the extent of subsequent fires. Third, when mapping the fire perimeters with satellite data, it is possible that I may have falsely identified other types of disturbance as fire. I assume, however, that the errors of this sort are negligible since

fuel treatments do not occur in my study areas (because they are inside wilderness or national parks) and vegetation changes due to insect and disease (e.g., bark beetle) are too subtle to be detected using my methods given that their full effects often take multiple years to manifest (Meigs et al. 2011).

Conclusion

My findings show that wildfires clearly limit subsequent fire size. This effect is strongest immediately after fire, decays over time, and lasts for ~6-16 years, depending on the study area. Furthermore, my findings show that increasing fire weather diminishes the ability of fire to act as a fuel break. As such, fire managers can potentially use my results to aid in assessing whether any particular fire scar will act as a fuel break based its age and the projected weather. However, managers should also consider that, even if a past fire scar does not stop the progression of a wildfire and it reburns within a past fire perimeter, the fire severity will likely be limited (Miller et al. 2012; Parks et al. 2014).

More broadly, however, the numerous fires that have occurred over the last couple of decades in the western US potentially provide opportunities for managing fire in a different manner. That is, in forested landscapes that have experienced relatively recent fire (< ~25 years), there are now opportunities to reevaluate fire suppression policies and allow more fires to play their natural ecological role. Although this management strategy may not be advantageous in some landscapes, such as those at risk of invasion by non-native species (Keeley et al. 2011), it has several potential benefits. For example, allowing more fires to burn in certain situations will reduce landscape homogeneity and create more resilient landscapes in which the self-regulating feedback mechanisms of fire can be better realized (Keane et al. 2002), thereby reducing fire suppression costs and increasing firefighter safety. Furthermore, landscapes with active fire regimes may be more resilient to other types of disturbance (i.e. insect and disease outbreaks) (Bebi et al. 2003; Kulakowski et al. 2012). Lastly, ongoing fire disturbance offers the opportunity for establishment of species that are better aligned with the emerging climate, thereby acknowledging that vegetation communities and fire regime characteristics will change with shifts in climate (Westerling et al. 2011; Smith et al. 2014).

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Tables

Table 1. Summary of fires in each study area from 1972-2012.

Study area	Number of fires	<u>All fires</u>		Number of fires	<u>Large fires</u>	
		Number that interact with subsequent fire ^a	Area burned (ha) [proportion of study area]		Number that interact with subsequent fire ^a	Area burned (ha) [percent of study area]
FCW	297	234	862,373 [0.88]	147	123	843,574 [0.86]
SBW	373	225	255,454 [0.47]	125	71	225,698 [0.41]
CCE	189	78	307,228 [0.30]	77	33	297,678 [0.29]
GAL	179	138	345,334 [1.12]	88	56	334,137 [1.08]
Total	1038	675	1,770,389	437	283	1,701,087

^aThese values reflect only those fires that interact with a subsequent fire within 25 years (see Methods).

Table 2. Number of years until wildfires no longer serve as an effective fuel break (defined as having a ≤ 0.30 probability of limiting the extent of subsequent fire). Values reflect model fits (e.g., Figs. 5 and 6) with and without ERC as an explanatory variable.

Study area	All fires ^a	Time only models		Time plus ERC models			
		Large fires ^b	No ERC (n) ^c	ERC 50 th	ERC 75 th	ERC 90 th	ERC 99 th
FCW	16	16	16 (111)	18	17	15	13
SBW	18	18	17 (66)	24	20	17	13
CCE	15	14	14 (32)	24	19	16	10
GAL	6	7	5 (54)	8	5	4	2

^aThese values reflect the model that include fires ≥ 20 ha (Fig. 5a).

^bThese values reflect the model that include fires ≥ 400 ha (Fig. 5b).

^cThese values reflect a model using the subset of fires used in the models that include elapsed time and ERC, but excludes ERC (see Methods); these values are more directly comparable to the values in the columns to the right that include both elapsed time and ERC. The number of fires evaluated in the models evaluating elapsed time and ERC is provided in parentheses.

Figures

Figure 1. Locations of the four study areas in the western US.

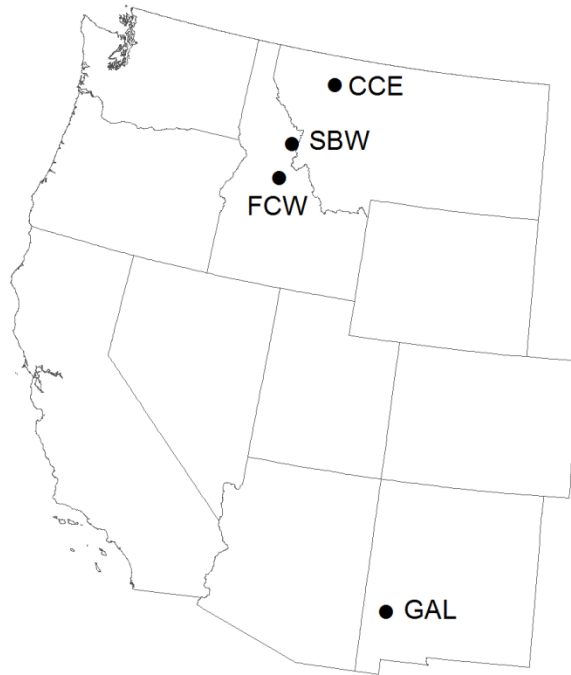


Figure 2. The four study areas for which I evaluated the ability of previous wildfires to limit the extent of subsequent wildfires. The boxplots depict the variability in mean annual precipitation and mean annual temperature within each study area (Daly et al. 2002); boxes represent the inter-quartile range, whiskers extend to the 5th and 95th percentiles, horizontal lines represent the median, and solid dots the mean.

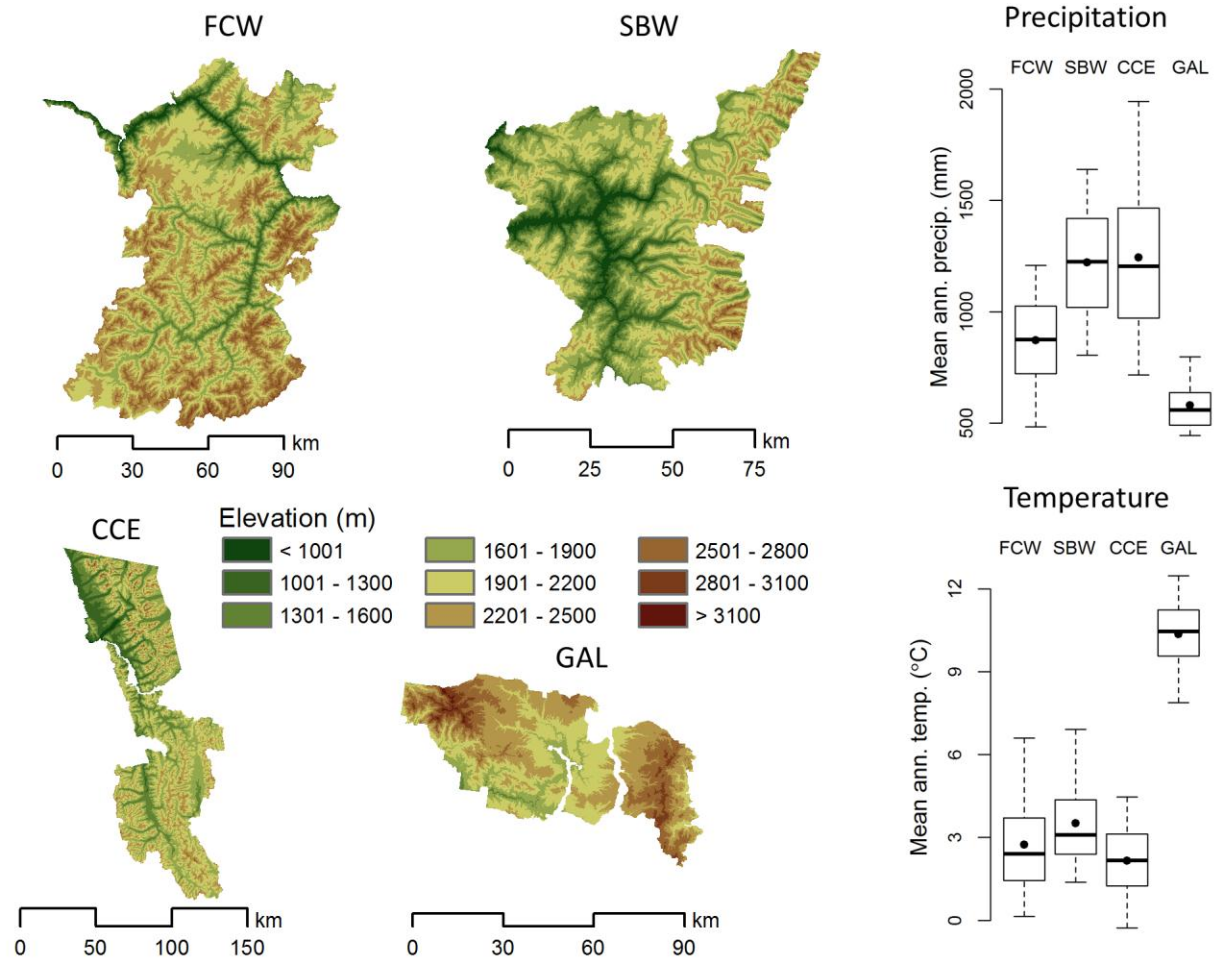


Figure 3. Examples from SBW depicting how pixels were defined as LIMITING or NOT LIMITING. In all examples, the initial wildfire has a blue (LIMITING), red (NOT LIMITING), or brown (not analyzed) perimeter and the subsequent fire is solid gray. In panel (a), a 2007 wildfire that interacts with a subsequent 2008 wildfire. Blue pixels are those defined as LIMITING and are ≤ 375 m (as measured outwards) or ≤ 750 m (as measured inwards) from the subsequent fire perimeter. Those pixels that do not interact with a subsequent fire (brown line) are excluded from the analyses. In panel (b), all pixels from the 2000 wildfire are NOT LIMITING since the 2007 wildfire burned over the entire 2000 wildfire and are > 750 m from the 2007 fire perimeter boundary (as measured inwards). In panel (c), some portions of the 2008 wildfire infiltrate the 2007 wildfire beyond 750 m; such pixels are defined NOT LIMITING. In panel (d), a large proportion of the perimeter of the 2005 wildfire is proximal to the perimeter of the 2012 wildfire. However, since $> 35\%$ of the 2005 wildfire overlaps with the 2012 wildfire, all proximal pixels are labeled NOT LIMITING (see Methods).

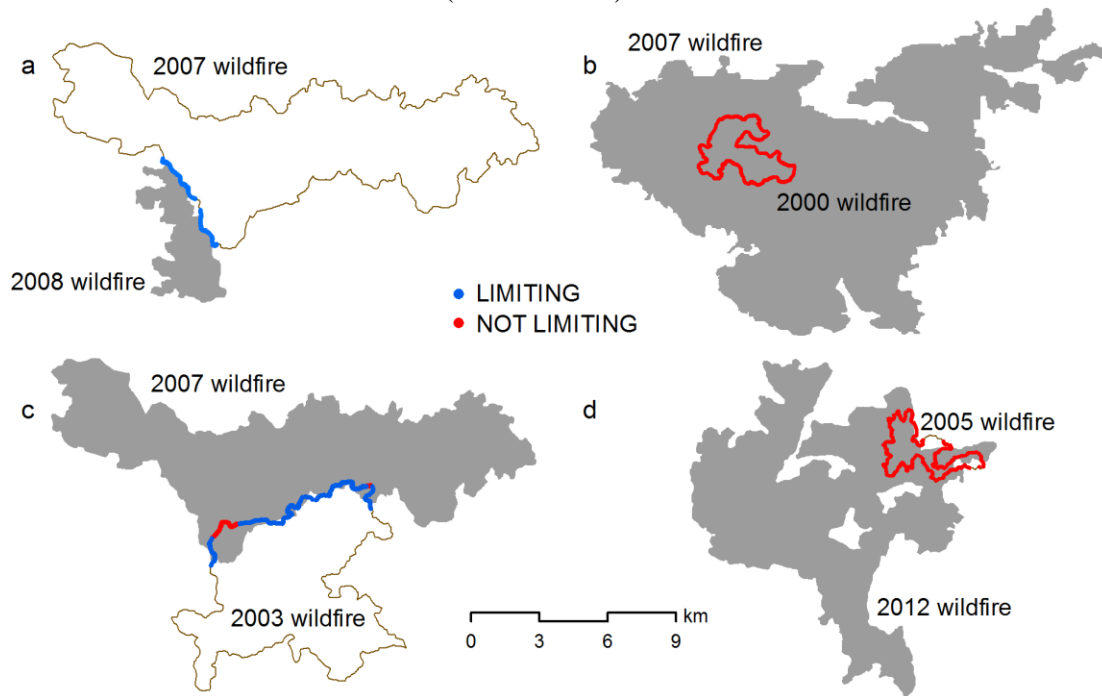


Figure 4. Data depicting proportion of pixels defined as LIMITING (y-axis) along a gradient depicting time until subsequent fire (x-axis). Sizes of circles represent the relative number of pixels for each time until subsequent fire within each study area. Red lines show the predicted logistic regression fit. ROC values are provided in Fig 5.

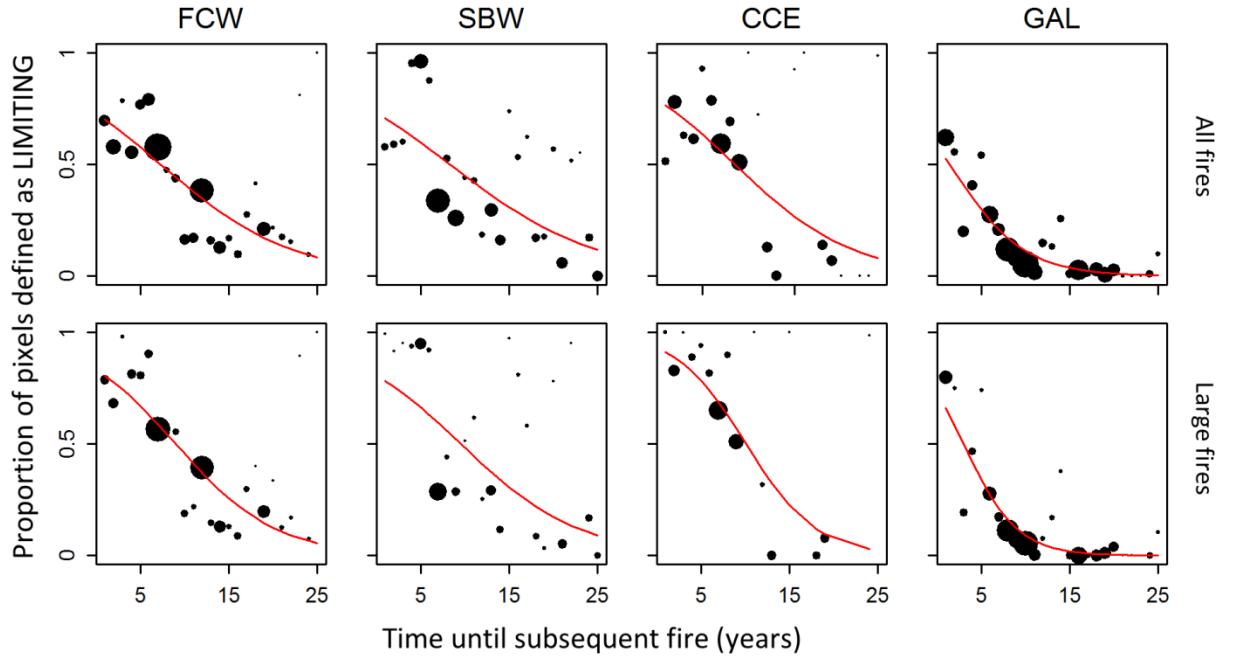


Figure 5. Response curves depicting the probability of a wildfire limiting the extent of subsequent fire over time for each study area for small (a) and large fires (b). The receiver operating characteristic, area under the curve statistic (ROC) is shown for each fit. These models fits were generated using all pixels (the model ensembles were used to test for statistical significance). The horizontal dashed line represents the threshold (0.30 probability) at which wildfires no longer act as an effective fuel break.

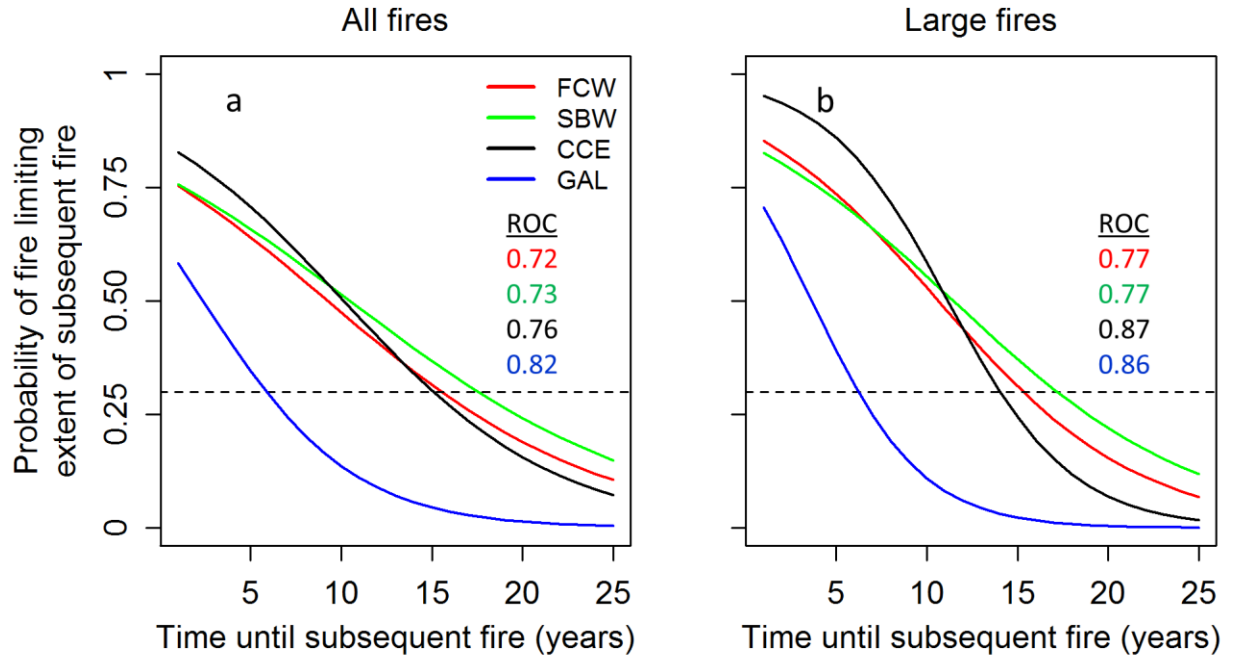
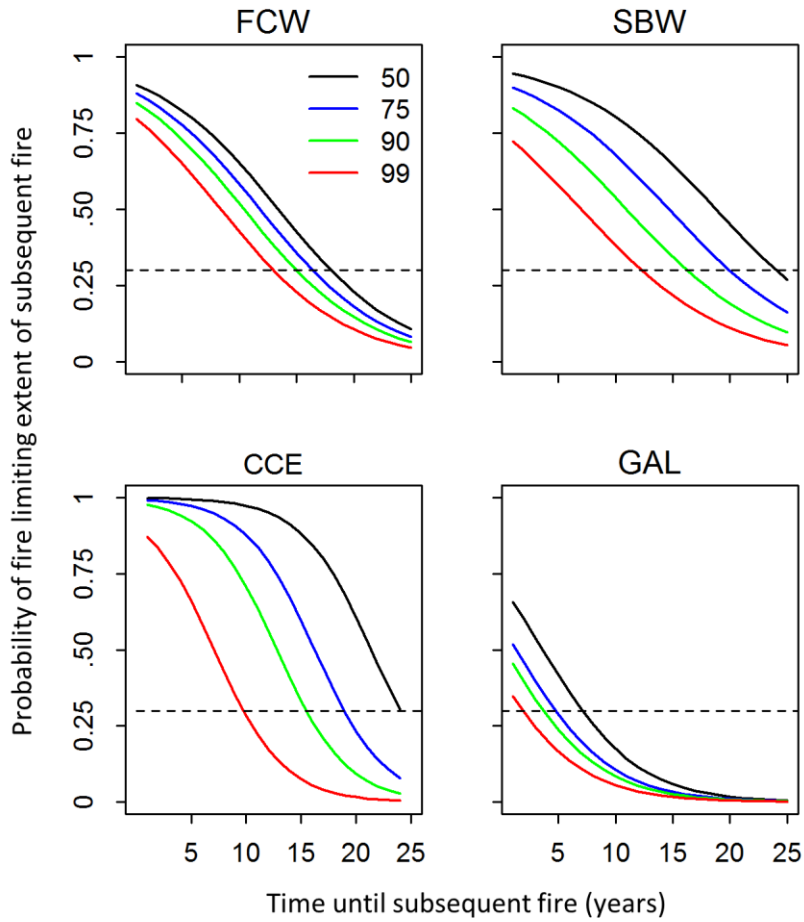


Figure 6. Response curves depicting how the probability of fire limiting the extent of subsequent fire varies by ERC. The contribution of ERC is statistically significant ($p \leq 0.05$) in all study areas according to each 2500 model ensemble. All ERC percentiles are study area specific and determined using ERC values occurring within the fire season; I defined the fire season as the beginning and ending date that encompassed 95% of the MODIS fire detections (USDA Forest Service 2013) for each study area. The horizontal dashed line represents the threshold (0.30 probability) at which wildfires no longer act as an effective fuel break.



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