Wildland fire as a self-regulating mechanism: the role of previous burns and weather in limiting fire progression

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Abstract. Theory suggests that natural fire regimes can result in landscapes that are both self-regulating and resilient to fire. For example, because fires consume fuel, they may create barriers to the spread of future fires, thereby regulating fire size. Top-down controls such as weather, however, can weaken this effect. While empirical examples demonstrating this pattern–process feedback between vegetation and fire exist, they have been geographically limited or did not consider the influence of time between fires and weather. The availability of remotely sensed data identifying fire activity over the last four decades provides an opportunity to explicitly quantify the ability of wildland fire to limit the progression of subsequent fire. Furthermore, advances in fire progression mapping now allow an evaluation of how daily weather as a top-down control modifies this effect. In this study, we evaluated the ability of wildland fire to create barriers that limit the spread of subsequent fire along a gradient representing time between fires in four large study areas in the western United States. Using fire progression maps in conjunction with weather station data, we also evaluated the influence of daily weather. Results indicate that wildland fire does limit subsequent fire spread in all four study areas, but this effect decays over time; wildland fire no longer limits subsequent fire spread 6–18 years after fire, depending on the study area. We also found that the ability of fire to regulate subsequent fire progression was substantially reduced under extreme conditions compared to moderate weather conditions in all four study areas. This study increases understanding of the spatial feedbacks that can lead to self-regulating landscapes as well as the effects of top-down controls, such as weather, on these feedbacks. Our results will be useful to managers who seek to restore natural fire regimes or to exploit recent burns when managing fire.

Key words: fire history; fire progression; fire spread; fuel break; fuel treatment; interacting fires; reburn; self-limiting; self-regulation; time-dependence; top-down vs. bottom-up controls; wilderness.

Introduction

Wildland fire is an important ecological process in many ecosystems; it alters vegetation composition and structure, consumes biomass, and influences landscape heterogeneity. These fire-induced changes can influence subsequent fire activity, behavior, and effects via site- and landscape-level feedbacks (Agee 1999, Peterson 2002, McKenzie et al. 2011). For example, wildland fires reduce fuel loads and therefore lessen the potential for subsequent fire until sufficient biomass re-accumulates (Héon et al. 2014, Parisien et al. 2014). If a fire does recur, however, both the size and severity may be limited, though this effect diminishes over time due to fuel accumulation (Collins et al. 2009, Miller et al. 2012, Parks et al. 2014*b*). These negative feedbacks are considered fundamental ecosystem properties of fire-adapted ecosystems of the western United States

(McKenzie et al. 2011), and play an important role in promoting ecosystem resilience (Peterson et al. 1998, Gunderson et al. 2009).

In many fire-adapted forests, key pattern–process feedbacks that maintain self-regulation and promote resilience to wildland fire have been disrupted. For example, fire exclusion dating back to the 1930s in much of the western United States has removed a key regulatory feedback of some forest types, resulting in increased tree density and more homogeneous landscapes, particularly in forests that historically experienced periodic fire (Hessburg et al. 2005, Naficy et al. 2010). In contrast to landscapes in which self-regulating feedback mechanisms are better realized (Keane et al. 2002, McKenzie et al. 2011, Larson et al. 2013), these increases in tree density and landscape homogeneity may be eroding resilience to future fire events, as witnessed by recent increases in area burned and fire severity (Stephens 2005, Mallek et al. 2013). A better understanding of how pattern–process feedbacks operate in forested areas is necessary to restore resilience to

wildland fire in landscapes affected by decades of fire exclusion.

Restoring fire-resilient landscapes will require acknowledging that not all fires can or should be suppressed (Donovan and Brown 2007, Hutto 2008). Reestablishing wildland fire as a natural process, however, is difficult because fires pose a risk to lives and property (Gill et al. 2012), especially given the current fuel buildup in some forest types. Political roadblocks and numerous social, economic, and air quality concerns (Zimmerman et al. 2006) also hamper restoring wildland fire to landscapes, as does uncertainty relating to expected fire behavior and final fire size (Thompson and Calkin 2011). These roadblocks are becoming increasingly challenging to overcome due to the rapid expansion of the wildland–urban interface (Theobald and Romme 2007) and the potential for climate-induced changes to the length of the fire season (Flannigan et al. 2013) and frequency of fires (Littell et al. 2010, Westerling et al. 2011). Thus, managers need quantitative information describing the effectiveness of wildland fire to act as a "fuel break," defined in this study as an absolute barrier to the spread of subsequent fire (cf. Syphard et al. 2011). This information is necessary to improve decision making during active fire management and in designing fuel treatments that incorporate past fire into landscape planning efforts.

The ability of wildland fire to act as a fuel break and limit the spread of subsequent fire, and the strength and longevity of the associated negative feedbacks, are likely influenced by several factors. In many forest types, fuels have large influence over the spread of wildland fires; fuel characteristics are dependent on ecosystem productivity and time since previous fire (i.e., biomass accumulation rate) (Minnich and Chou 1997, Peterson 2002). Because ecosystem productivity varies with temperature and precipitation (e.g., Cleveland et al. 1999), it is probable that the ability of fire to act as a fuel break varies geographically and among ecosystems. A better understanding of how wildland fire's ability to act as a fuel break varies with time since fire and ecosystem type would provide managers higher confidence in fire management decision making and planning. Furthermore, some studies have suggested that top-down controls such as weather may override or moderate the effects of previous fire (Bessie and Johnson 1995, Moritz 2003). Given that extreme fire weather is expected to become more common (Nitschke and Innes 2008), a better understanding of this top-down control would help land managers anticipate how the effectiveness of wildland fire as a fuel break may weaken in the future, and will provide scientists a means to better predict potential changes in fire regimes and resulting changes to vegetation (Westerling et al. 2011).

Although numerous anecdotal observations of previously burned areas acting as fuel breaks exist, few empirical studies have explicitly evaluated feedbacks between wildland fire in terms of its ability to limit the spread of subsequent fire. In general, these investigations were limited to one study area, limited in scope (e.g., did not evaluate time between fires), or inconclusive in their results. For example, in a study within a mixed-conifer forest of the Sierra Nevada, California, Collins et al. (2009) found that wildland fire indeed limited the spread of subsequent fire; this effect diminished with increasing time between fires and fire-conducive weather. As another example, in an investigation of three study areas in central Idaho and northern Montana, Teske et al. (2012) found that wildland fire limited the spread of subsequent fire in only one study area; time between fires and weather were not evaluated. Quantitative information on the ability of wildland fire to limit subsequent fire progression, and how this varies with time between fires and weather, is lacking over a broad range of geographies and ecosystems but is necessary to advance ecological theory on interacting disturbances.

We evaluated the ability of wildland fire to limit the spread of subsequent fire in four large study areas in the western United States. We had two specific objectives: (1) to quantify the ability of wildland fire to limit the progression of subsequent fire as time since previous fire increases, and (2) to quantify how weather conditions modify this effect. We expected that the effectiveness of wildland fire to act as a fuel break is greatest immediately after a fire, and decays through time as fuels re-accumulate; we also expected that this effectiveness varies by study area and weakens under extreme weather conditions. The results of our study add to the fundamental understanding of ecological feedbacks between pattern and process on landscape heterogeneity and composition (Turner 1989, Peterson 2002). Land managers will likely find our results useful and informative for active fire planning efforts and for restoring resilience to wildland fire (e.g., by quantifying the number of years that fires remain barriers to subsequent fire spread).

METHODS

Study area

We conducted our investigation within four study areas composed entirely of federally protected areas (wilderness and national park) in the western United States (Fig. 1), thereby limiting potential confounding effects of land management activities that are more common outside such areas (Parks et al. 2014c). The FCW (Frank Church-River of No Return Wilderness) study area is in central Idaho. The adjacent SBW (Selway-Bitterroot Wilderness) study area is in western Montana and north-central Idaho. The CCE (Crown of the Continent Ecosystem) study area comprises Glacier National Park and the Great Bear, Bob Marshall, and Scapegoat wilderness areas. Finally, the GAL study area incorporates the Gila and Aldo Leopold Wilderness Areas in western New Mexico. All four study areas have experienced substantial fire activity in recent decades, thus providing sufficient data with which to evaluate the

FIG. 1. Locations of the four study areas in the western United States. FCW, Frank Church-River of No Return Wilderness; SBW, Selway-Bitterroot Wilderness; CCE, Crown of the Continent Ecosystem; GAL, Gila and Aldo Leopold Wilderness.

effectiveness of wildland fire as a fuel break. Some, but not all, fires were managed as natural events for their resource benefits.

FCW (Frank Church-River of No Return Wilderness).—The FCW (9777 km²) is the second largest wilderness area in the contiguous United 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). Mean annual precipitation is generally lowest in the low-elevation river bottoms and highest on the mountain peaks; temperature exhibits the opposite pattern. Elevations range from 600 to 3136 m and topographic features include river breaks, deep canyons, mountains, and glaciated basins (USDA Forest Service 2003a). Park-like groves of ponderosa pine (*Pinus ponderosa*) occupy south and west slopes below 1500 m (Barrett 1988). Denser ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) forests occupy north and east aspects, up to elevations of ~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 typically 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 around 1935 (Finklin 1988), although sheep grazing may have excluded fire

earlier (Steele et al. 1981). Management of lightning-caused fires for their resource benefits began around 1988 (Beckman 2008).

SBW (Selway-Bitterroot Wilderness).—The SBW (5471 km²) is the third-largest wilderness area in the contiguous United States and includes the Bitterroot Mountains along the Montana and Idaho border and large portions of the Selway and Lochsa watersheds in Idaho. Mean annual precipitation is 1221 mm and mean annual temperature is 3.5°C (Daly et al. 2002). Elevations range from 531 m on the western edge of the Selway River drainage to >3000 m in the southeast portion of the study area. The vegetation of SBW is diverse. Pacific maritime forests occur below ~1500 m in the west and northwest portions of the study area and are 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 (more than ~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. The fire season 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; some resource benefit fires were allowed to burn after this point (van Wagtendonk 2007). Cattle and sheep grazing was evident in the early 1900s (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 and straddles the east and west slopes of the continental divide. Mean annual precipitation is 1243 mm, and mean annual temperature is 2.2°C (Fig. 2 and Daly et al. 2002). Elevations range from 950 m to >3100 m. In this rugged study area, alpine glacial canyons and cirques drain into major river valleys (Barrett et al. 1991, Keane et al. 1994). Ponderosa pine, lodgepole pine, Douglas-fir, western larch are the dominant tree species at low elevations (less than ~1500 m) (Arno 1980, Keane et al. 1994, 2006). Western hemlock and western red cedar also occur at low elevations (<1500 m) in wet areas that have been free of fire for extended periods of time (more than ~100 years). As elevation increases, the dominant species become lodgepole pine, subalpine fir, and
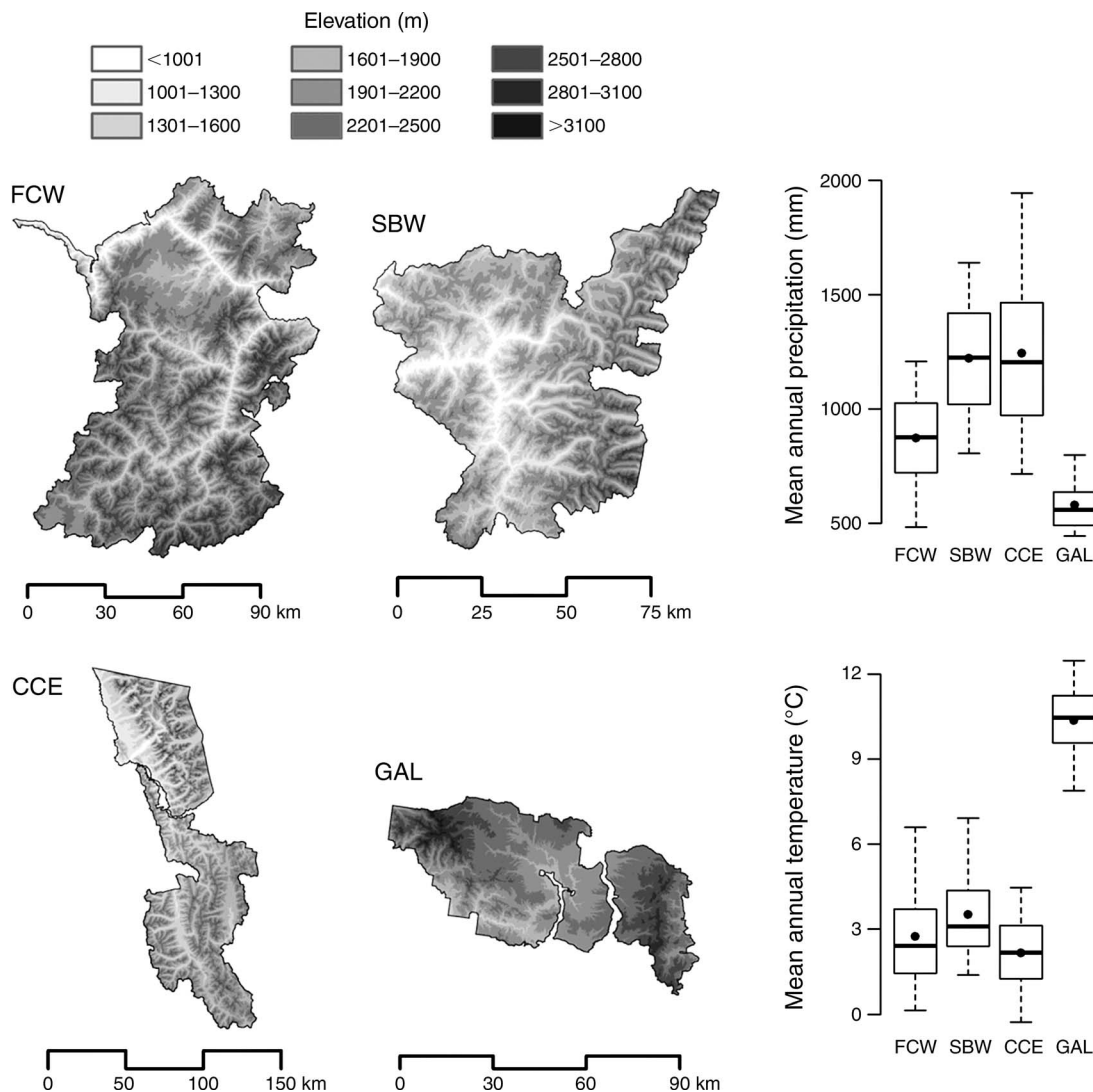
FIG. 2. The physical environments of the four study areas for which we evaluated the ability of wildland fire to limit the spread of subsequent fire. The boxplots depict the variability in mean annual precipitation and mean annual temperature within each study area (Daly et al. 2002); boxes represent the inter-quartile range, whiskers extend to the 5th and 95th percentiles, horizontal lines represent the median, and solid dots the mean. For study area acronyms, see Fig. 1.

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 forest 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. With the exception of alpine environments, most of the study area 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). Although the year in which fire suppression became effective is unknown, resource benefit fires began in the Bob Marshall Wilderness in 1981 and in

GNP in 1994; livestock grazing is assumed to have had a negligible effect on the fire regime in this study area.

GAL (Gila and Aldo Leopold Wilderness).—The GAL (3087 km^2) is the driest and warmest of the four study areas; mean annual precipitation is 578 mm and mean annual temperature is 10.4°C (Fig. 2 and Daly et al. 2002). 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). The fire season runs from early May through mid-July (USDA Forest Service 2013), although fires are less likely after mid-June 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 1890s, 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.—To create the geospatial fire atlas for each study area, we first 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 United States from 1984 to 2011. We modified numerous MTBS fire perimeters to correct for mapping inaccuracies that we encountered using the methods described in the following paragraph. For example, we found a few instances where MTBS had incorrectly mapped fires from different years as one fire, and where MTBS had incorrectly mapped a single contiguous fire as multiple fires.

Next, we supplemented the MTBS fire perimeters by identifying and mapping all wildland fires ≥20 ha from 1972 to 2012 using the entire record of Landsat data, including the multi-spectral scanner (MSS), thematic mapper (TM), enhanced thematic mapper plus (ETM+), and operational land imager (OLI) sensors. We obtained virtually all snow-free images for each study area from the U.S. Geological Survey Center for Earth Resources Observation and Science (USGS-EROS; *available on-line*)[4] and identified and mapped 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, we delineated fire perimeters by evaluating differences between pre- and post-fire NBR (normalized burn ratio) (dNBR) (Key and Benson 2006). We 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.

Although dNBR proved effective at mapping most wildland fires in GAL (~80%), we also used a relativized metric of fire-induced change (RdNBR [Miller and Thode 2007]) to map fires in this study area in an effort to provide higher contrast when pre-fire vegetation is sparse (Zhu et al. 2006, Parks et al. 2014*a*). Although sparse vegetation conditions do exist in the northern study areas, they occur so less frequently (GAL contains ~2.5 times more grassland compared to FCW [Rollins 2009]). This slight difference in methodology between study areas likely has negligible effect on our results, especially given that we also used ancillary data to locate fires.

Supplementary spatial data also aided in locating fires in all study areas, including Moderate Resolution Imaging Spectroradiometer (MODIS) fire detections (USDA Forest Service 2013) (2001–2012), National Interagency Fire Management Integrated Database (1972–2012; *available online*),[5] Geospatial Multi-Agency Coordination Group fire perimeters (2001–2011; *available online*),[6] 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 (1980–2012; *available online*).[7] The final product is a geospatial fire atlas for all fires ≥20 ha from 1972 to 2012. All fire perimeters were converted to raster format with a 30 × 30 m pixel size (matching the resolution of Landsat TM, ETM, and OLI data).

Identifying limiting fire perimeters.—Previous wildland fire interacts with subsequent fire by either limiting the spread of or getting reburned by a subsequent fire. We applied an objective and consistent rule-set to identify where fire perimeters did 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 using two criteria: (1) a pixel on the fire perimeter must be located within 375 m of the perimeter of the subsequent fire, or (2) a pixel on the fire perimeter must be reburned by the subsequent fire. The 375 m distance threshold we used to identify interacting pixels acknowledged uncertainty in fire perimeter mapping due to spatial and spectral diversity caused by variability in fire severity and vegetation type (Holden et al. 2005). Next, we determined whether those interacting pixels did or did not limit the spread of subsequent fires. If a subsequent fire perimeter was within 375 m, as measured outwards from the previous fire perimeter (i.e., the subsequent fire was in close

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[4] http://earthexplorer.usgs.gov/

[5] https://fam.nwcg.gov/fam-web/kcfast/html/ocmenu.htm

[6] http://www.geomac.gov/index.shtml

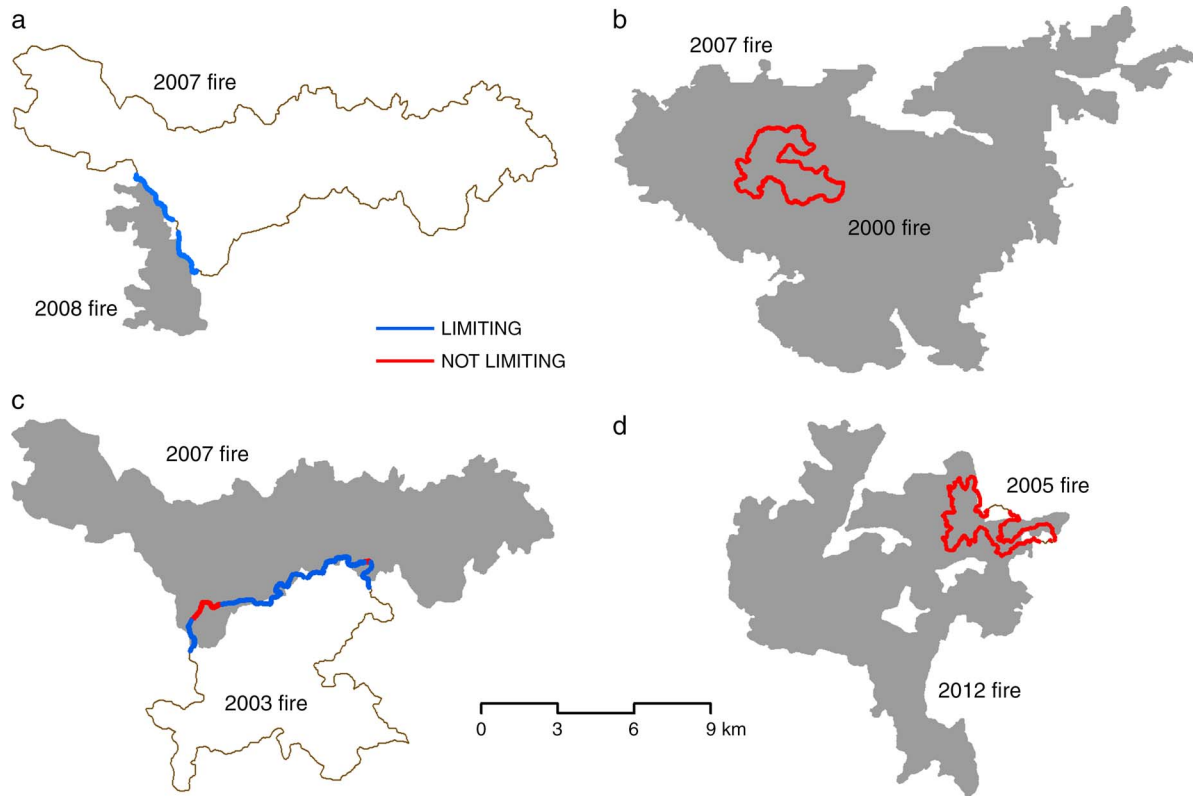[7] http://www.fs.usda.gov/detailfull/flathead/landmanagement/gis

Fig. 3. Examples from SBW depicting how pixels were defined as LIMITING or NOT LIMITING. In all examples, the initial wildland fire has a blue (LIMITING), red (NOT LIMITING), or brown (not analyzed) perimeter and the subsequent fire is solid gray. In panel (a), a 2007 fire interacts with a subsequent 2008 fire. 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 fire are NOT LIMITING since the 2007 fire burned over the entire 2000 fire and pixels are >750 m from the 2007 fire perimeter boundary (as measured inwards). In panel (c), some portions of the 2008 fire infiltrate the 2007 fire beyond 750 m; such pixels are defined as NOT LIMITING. In panel (d), a large proportion of the perimeter of the 2005 fire is proximal to the perimeter of the 2012 fire. However, since >35% of the 2005 fire overlaps with the 2012 fire, all proximal pixels are labeled NOT LIMITING (see *Methods*).

proximity to, but not touching, the previous fire) and no more than 750 m as measured inwards (i.e., the subsequent fire burned into and overlapped the previous fire by ≤750 m), then we assumed that the spread of the subsequent fire was limited by the previous fire and we labeled the pixel of the previous fire as LIMITING (Fig. 3a, c). In this case, the 750-m threshold reasonably accounted for the regulating capacity of wildland fire to limit subsequent fire spread, even though some overlap among successive fires could occur. If a pixel from a subsequent fire perimeter was located >750 m inside of a previous fire perimeter, then we assumed that the subsequent fire's progression was not limited by the previous wildland fire; hereafter, these interacting pixels of the previous fire are referred to as NOT LIMITING (Fig. 3b, c, d). If a pixel from a subsequent fire was >375 m from a previous fire, we assumed that that there was no interaction and the pixel was excluded from our analyses (Fig. 3a, c, d). Preliminary analyses indicated that many false positives resulted from this rule-set (e.g., in our judgment, pixels were mislabeled as LIMITING),

prompting an additional rule: if >35% of the area of either the initial or subsequent fire overlapped, then all proximal pixels were identified as NOT LIMITING (see Fig. 3d). All pixels from all fires were thus labeled as LIMITING or NOT LIMITING subsequent fire spread, or they were excluded from the analyses. To clarify, the units of analysis are pixels along the perimeter boundary, or edge, of the initial fire.

Although the thresholds described above are admittedly arbitrary and can result in pixels seemingly mislabeled as LIMITING or NOT LIMITING, exploratory analyses using alternative thresholds (e.g., 250 m and 500 m vs. 375 m and 750 m) yielded results that were similar to those reported here (see Appendix). We concluded that minor changes in threshold values do not substantially change the findings of this study. All geospatial operations were conducted using either ArcMap 10.1 (ESRI 2012) or the raster package (Hijmans and van Etten 2011) within the R statistical program (R Development Core Team 2007).

Statistical model.—To quantify the ability of wildland fire to serve as a fuel break along a temporal gradient representing time between fires, we 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. Although the fire perimeter data span 41 years, there were relatively few instances of >25 years between fires, and additionally, initial data exploration indicated that there appeared to be no effect of fire as a fuel break beyond this time. As such, we removed all interactions older than 25 years from the analysis. For each study area, we built a model for all fires ($\geq$20 ha) as well as a model for just large fires ($\geq$400 ha). We built a separate model for just large fires for two reasons. First, unburned islands within fires can spatially coincide with previous small fires, but because they can be difficult to identify, they often remain unmapped (Kolden et al. 2012). As a result, we were concerned that some of the smaller fires in our study may have been falsely labeled as NOT LIMITING. A model including only large fires reduces the chance of including these falsely labeled pixels. 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.

To facilitate comparison among study areas, we consider that wildland fire serves as an effective fuel break when the probability of limiting subsequent fire spread is >0.30 as defined by the fit of the logistic regression models; below this probability, fire no longer serves as a fuel break. This threshold is arbitrary. A higher threshold would decrease the perceived longevity of wildland fire's influence on subsequent fire spread, while a lower threshold would potentially overstate the longevity; the latter of these is particularly relevant given the asymptotic nature of logistic regression. We settled on 0.3 as a reasonable compromise. 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 1993), we 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, we generated a multi-model ensemble using 2500 random subsets of data; the subsampling frequency was 1% of the full data set. 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. We chose a 1% subsampling frequency based on Parks et al. (2014*b*), who used ~0.1% subsampling frequency for two-dimensional fire severity data; since fire perimeter edges are linear, one-dimen-

sional features, we 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 wildland fire to limit subsequent fire spread, we built a 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. We 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). Increasing ERC represents increasing fire-conducive weather conditions. 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 we used fuel model K.

To assign these daily ERC values to each 30-m pixel, we used information on day-of-burning. Because agency generated fire progression maps were not available for a large number of the fires, we mapped day-of-burning using the methods developed by Parks (2014), where day-of-burning for each 30 × 30 m pixel, and hence fire progression, was estimated by spatially interpolating Moderate Resolution Imaging Spectrometer (MODIS) fire detection data (NASA MCD14ML product, Collection 5, Version 1). As such, this process was limited to fires burning after 2000 to coincide with the operational timeline of the MODIS sensors. The process was also limited to large fires ($\geq$400 ha) due to the coarse nature of the MODIS input data (pixel size = 1 km^2). 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, 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. Within each large fire that burned between 2001 and 2012, we then assigned these daily ERC values to each 30 × 30 m pixel based on its estimated day-of-burning.

Due to the aforementioned limitations imposed by the operational timeline and spatial resolution of MODIS, the models that incorporate weather employ a subset of data; they include only large fires (occurring any year from 1972 to 2011) that subsequently interacted with large fires that occurred between 2001 and 2012. For each interacting perimeter pixel of the initial wildland

TABLE 1. Summary of fires in each study area from 1972 to 2012.

Study area†	All fires (≥20 ha)			Large fires (≥400 ha)		
	Number of fires	Number that interact with subsequent fire‡	Area burned (ha) [proportion of study area]	Number of fires	Number that interact with subsequent fire‡	Area burned (ha) [proportion 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

† For study area acronyms, see *Methods*.
‡ These values reflect only those fires that interact with a subsequent fire within 25 years (see *Methods*).

fire, we extracted the daily ERC value that was associated with the subsequent fire. In those cases when a previous fire did not technically overlap, but was within 375 m from a subsequent fire, we used the day-of-burning estimate, and hence the ERC value, of the nearest pixel of the subsequent fire. We assessed significance of ERC using the subsampling and model ensemble approach described previously.

RESULTS

A total of 1038 fires, including 437 large fires, were identified between 1972 and 2012 across all study areas. Subsequent fires interacted with a majority of these (>60%), though the exact percentage varied by study area (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) experienced the least number of fires (≥20 ha) but proportionally burned the most (1.12). The CCE (the largest study area) proportionally burned the least (0.30) and had the lowest percentage of interacting fires (~42%) (Table 1).

In all study areas, and for both sets of wildland fires analyzed (all fires and large fires), the proportion of pixels identified as LIMITING generally decreases as time until subsequent fire increases (Fig. 4). Consequently, the logistic regression models indicate that the ability of wildland fire to limit the progression of subsequent fire is strongest immediately after a fire, but decays over time (Figs. 4 and 5). Based on our threshold in which wildland fire no longer acts as an effective fuel break (≤0.30 probability), the effect lasts ~6 years in GAL and ~16 years in the three northern study areas (Figs. 4 and 5; Table 2). Overall, the relationship between the effectiveness of fire as a fuel
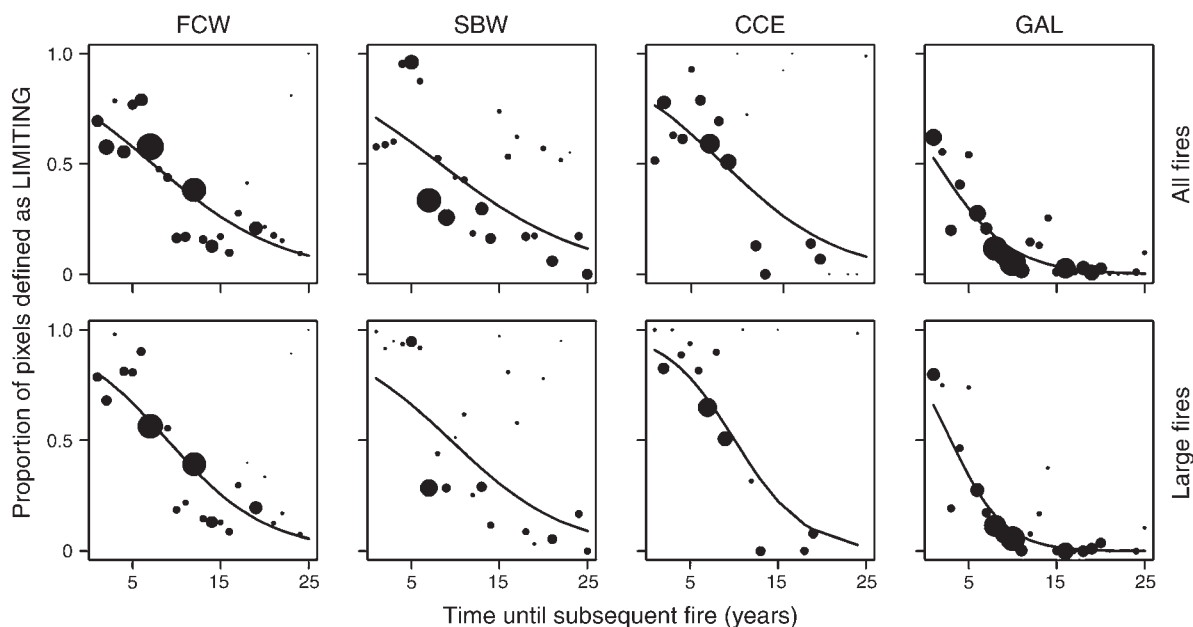

FIG. 4. Data depicting the proportion of pixels defined as LIMITING (*y*-axis) along a gradient depicting time until subsequent fire (*x*-axis). Sizes of circles represent the relative number of pixels for each time until subsequent fire within each study area. Black lines show the predicted logistic regression fit. ROC values are provided in Fig. 5. See Fig. 1 for study area acronyms.

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 wildland fires in FCW, SBW, and CCE are >75% effective at limiting the spread of subsequent fires for up to four years, diminishing to ~50% 11 years after fire (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 ($P \leq 0.001$).

The ability of wildland fire to act as a fuel break weakens with increasing fire-conducive weather conditions, as represented by ERC in all study areas (Fig. 6). For example, 10 years after fire in CCE, the probability that a wildland fire limits the spread of a subsequent fire is very high under moderate conditions (probability = 0.97; 50th percentile ERC), but is quite low under extreme conditions (probability = 0.30; 99th percentile ERC). The length of time for which a fire can effectively serve as a fuel break (i.e., probability >0.30) is shorter under extreme vs. moderate weather conditions (99th vs. 50th percentile ERC) in all four study areas (Fig. 6, Table 2). In CCE, for example, the ability of fire to act as a fuel break lasts 10 years under extreme conditions compared to 24 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 0.04 (CCE and GAL).

DISCUSSION

Theory suggests that in landscapes with an active fire regime, landscape pattern is shaped by fire, but fire is also shaped by landscape pattern (Peterson 2002). This pattern–process feedback is a fundamental concept in disturbance ecology and can lead to self-regulating systems (Turner 1989, Agee 1999), underscoring the importance of wildland fire in creating and maintaining resilient landscapes (McKenzie et al. 2011). Our results clearly indicate that this pattern–process feedback is present in our four western United States study areas; we found that wildland fire limits the spread of subsequent fire in these landscapes. The strength of this feedback, however, decays with time between fires, varies among geographic regions, and is modulated by extreme weather.

Differences among the study areas in the strength and duration of the pattern–process feedback are evident and reflect geographic differences in climate, fire regime characteristics, 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 wildland fire to act as a fuel break. The GAL is dryer and warmer than the other study areas (Fig. 2) and the fire regime is for the most part characterized by surface fires whose occurrence is limited not by the frequency of fire-conducive weather conditions, but rather by the availability and continuity of fine fuel (Schoennagel et al. 2004). As such, large fire years tend to occur when herbaceous fine fuels (i.e., grasses) are abundant: one to three years after a wet (i.e., high-precipitation) year (Swetnam and Baisan 1996). The rapid growth and accumulation of fine fuels during wet years therefore erases the effects of the previous fire in terms of its ability to act as a fuel break and, consequently, wildland fire is not likely to act as a fuel break for periods of time exceeding six years. In contrast, fires in the other cooler, wetter study areas tend to be of higher severity (Parks et al. 2014b) due to the abundance of ladder and canopy fuels that develop during long fire-free intervals (Schoennagel et al. 2004). These ecosystems generally experience fire-conducive weather less frequently; hence the increased longevity of wildland fire as a fuel break in FCW, SBW, and CCE.

In relation to other investigations of feedbacks between wildland fire and subsequent fire spread, our findings are 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. However, our results differ from those of Teske et al. (2012), who examined three of the same study areas we did (FCW, SBW, and CCE), and found that wildland fire limited the spread of subsequent fire in only one of them (FCW). 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 that wildland fire's ability to act as a fuel break decays over time and is no longer effective after ~16 years in these study areas, investigations of this sort should explicitly address time between fires.

In all study areas, the effectiveness of wildland fire as a fuel break weakens as weather becomes more fire conducive, which was also noted by Collins et al. (2009), thereby supporting the assertion that the importance of fuels diminishes during extreme weather events (Bessie and Johnson 1995, Price and Bradstock 2011). Nevertheless, our results indicate that fuels, or lack thereof post-fire, strongly limit fire spread in the northern study areas in the years immediately following fire, even under extreme conditions. Conversely, in GAL, which is generally composed of dry conifer forest, fire acts as a fuel break for a only very short period of time (two years) under extreme 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
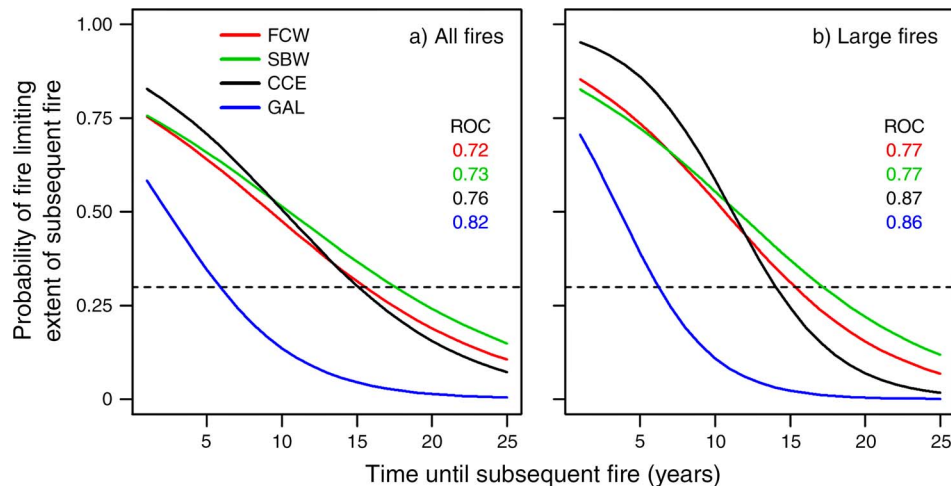
Fig. 5. Response curves depicting the probability of wildland fire limiting the spread of subsequent fire over time for each study area for (a) all and (b) large fires. The receiver operating characteristic (ROC), area under the curve statistic is shown for each fit. These model fits were generated using all pixels (the model ensembles were used to test for statistical significance). The horizontal dashed line represents the threshold (0.30 probability) at which wildland fire no longer acts as an effective fuel break. See Fig. 1 for study area acronyms.

(Collins 2014, U.S. Environmental Protection Agency 2014). As such, the strength and longevity of wildland fire's ability to act as a fuel break will likely be reduced in future years as weather becomes more extreme and less limiting to fire spread, reinforcing the results from other studies suggesting that climate change will result in more large fires and more area burned in many areas of the western United States (Westerling and Bryant 2008, Littell et al. 2010, Moritz et al. 2012). The resulting interaction between increased fire frequency and a warming climate has the potential to cause regionwide shifts in vegetation communities (Westerling et al. 2011, Rocca et al. 2014).

Clearly, land managers are not only interested in fire size and area burned, but also are keenly interested in the manner in which it burns (e.g., fire severity). Although we focused this study on the pattern–process feedback of wildland fire affecting subsequent fire spread, this feedback also affects other aspects of fire.

For example, fire severity is significantly lower when fires burn within a previously recorded fire perimeter (i.e., it reburned) compared to those areas that had an extended fire-free period (Miller et al. 2012, Parks et al. 2014b). This reduction in burn severity lasts at least 20 years and potentially much longer, yet the degree to which topography and climate affects the severity in repeat burns needs further study (but see Parks et al. 2014b). Federal agencies spend millions of dollars annually on fuel treatments (prescribed fire and thinning) to reduce fire hazard and risk (Allen et al. 2002); such treatments, when the area is burned, have resulted in reduced burn severity (Wimberly et al. 2009, Arkle et al. 2010, Prichard et al. 2010, Kennedy and Johnson 2014, Prichard and Kennedy 2014). However, it is critical to understand how *wildland fire* serves as a fuel treatment (in terms of both fire severity and size), given that wildland fire affects a larger proportion of public lands than traditional treatments (USDA Forest

Table 2. The number of years that wildland fire serves as an effective fuel break (defined as having a >0.30 probability of limiting the spread of subsequent fire). Values reflect model fits (e.g., Figs. 5 and 6) with and without energy release component (ERC) as an explanatory variable.

Study area†	Time only models			Time plus ERC models			
	All fires‡	Large fires§	No ERC (n)¶	ERC 50th	ERC 75th	ERC 90th	ERC 99th
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

† For study area acronyms, see *Methods*.
‡ These values reflect the model that includes fires ≥20 ha (Fig. 5a).
§ These values reflect the model that includes fires ≥400 ha (Fig. 5b).
¶ These values reflect a model using the subset of fires used in the models that include elapsed time and energy release component (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.
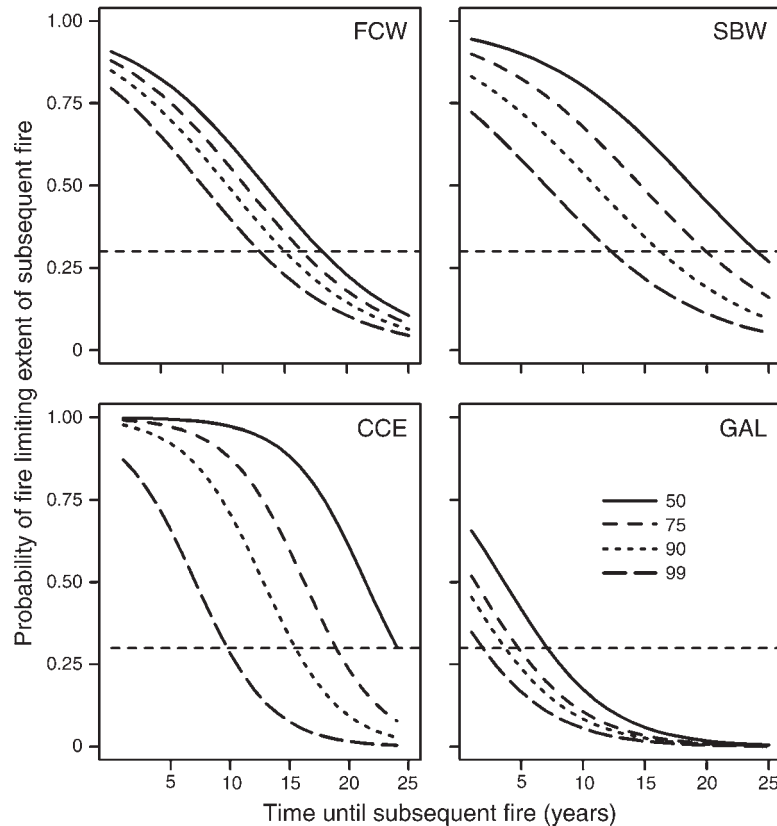
FIG. 6. Response curves depicting how the probability of fire limiting the spread of subsequent fire varies by energy release component (ERC) percentile. The contribution of ERC is statistically significant ($P \leq 0.03$) in all study areas according to each 2500-subset model ensemble. All ERC percentiles are study area specific and determined using ERC values occurring within the fire season; we 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 wildland fire no longer acts as an effective fuel break. See Fig. 1 for study area acronyms.

Service 2003b, NIFC 2010). The findings from this study combined with knowledge about trends in severity will help managers better assess the potential for a particular fire to serve as a fuel treatment under various weather conditions.

Our findings clearly indicate that fire *spread*, or *progression*, is controlled by previous fires (i.e., fuels) and weather. However, a logical extension is that fire *size* is also controlled by these factors. The statistical properties of fire size distributions have been widely examined and interpreted to describe fire regimes and to infer mechanisms of fire spread. Although some studies have suggested that fire size is dictated by fuel availability (Malamud et al. 1998, Turcotte and Malamud 2004, Haire et al. 2013), others have argued that fire size is dictated by weather (Boer et al. 2008), and topography, a bottom-up control, undoubtedly plays a role in fire patterns (Heyerdahl et al. 2001, Parks et al. 2012, Cansler and McKenzie 2013). More likely is that multiple factors are necessary to explain fire spread and size at landscape and regional scales (Reed and McKelvey 2002, Moritz et al. 2005). Our analysis

focused on how the lack of fuel within a previous fire can limit the spread of fire, but we did not account for fine-scale weather events (e.g., rain, shifting winds, and diurnal fluctuations in humidity), topographic features (e.g., valley bottoms and ridges), or natural fuel breaks (e.g., lakes and alpine areas) that may also limit the spread and size of fires. Future investigations should work toward quantifying the relative importance of these and other factors (e.g., roads).

Although our findings provide strong evidence that fire plays an important self-regulating role in fire-adapted ecosystems, the relationships reported in this study may not be representative of all environmental conditions within each of our study areas or generalizable across all forest types. For example, the Frank Church-River of No Return Wilderness ranges from relatively open ponderosa pine savannas to higher-elevation lodgepole pine and spruce–fir forests. Our analysis approach aggregated data from these contrasting vegetation types, even though the strength and longevity of feedbacks between past wildland fire and subsequent fire spread would be expected to differ

between these systems. Thus, care should be taken in making inference to individual forest types within each study area and to landscapes with different environmental constraints. It is also possible that we understated the importance of extreme weather in diminishing the ability of wildland fire to limit subsequent fire spread, as we used only one weather station for each study area, and therefore did not account for spatial heterogeneity in extreme weather. Similarly, MODIS fire detections and the corresponding fire progression maps generated by interpolation are not without error (Parks 2014), and consequently, the day-of-burning and the associated daily weather may not be representative of the conditions under which the fire actually burned. Furthermore, our quantitative results for longevity are conditional to the 0.3 threshold we used to define whether or not wildland fires remained effective fuel breaks; a different threshold would yield different values for the longevity of the treatment effect. Similarly, a change in the distance thresholds we used to define fire perimeters as LIMITING or NOT LIMITING would yield different results (e.g., see Appendix). Finally, we did not account for direction of fire spread (e.g., heading vs. flanking), a factor that surely affects the ability of wildland fire to limit subsequent fire progression given that a heading fire has higher intensity and spread rate than a flanking or backing fire (Finney 2005).

Conclusion

Our study has three important findings: (1) Wildland fire clearly acts as a fuel break and is a barrier to subsequent fire spread; this effect is strongest immediately after fire and decays over time. (2) The ability of wildland fire to act as a fuel break varies geographically, lasting 6 years in the warmest/driest study area and 14–18 years in the cooler/wetter study areas. And (3), extreme fire-conducive weather diminishes the ability of fire to act as a fuel break. These findings will help fire managers assess whether a previous wildland fire will act as a fuel break based on its age, ecosystem type, and expected weather.

More broadly, however, land managers will likely find our results useful and informative for restoring resilience to landscapes, especially in areas in the western United States that have experienced substantial fire activity over the last three decades. Our findings add to and complement a growing body of knowledge that can provide tremendous decision support to fire managers. Knowing that fire recurrence (Héon et al. 2014, Parisien et al. 2014), fire spread (this study and Collins et al. 2009), and fire severity (Miller et al. 2012, Parks et al. 2014b) are limited by previous wildland fire should provide greater flexibility and confidence in managing fire for resource benefit and in a manner in which self-regulation can be better realized. In fact, our results suggest that in the presence of recent previous wildland fire, suppressing a fire presents a lost opportunity to restore self-regulation and resilience, especially during non-extreme weather conditions.

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Supplemental Material

Ecological Archives

The Appendix is available online: http://dx.doi.org/10.1890/14-1430.1.sm

Data Availability

Data associated with this paper has been deposited in these archives for the following study areas:
Frank Church-River of No Return (FCW): http://www.fs.usda.gov/rds/archive/Product/RDS-2015-0021
Crown of the Continent Ecosystem (CCE): http://www.fs.usda.gov/rds/archive/Product/RDS-2015-0022
Gila and Aldo Leopold Wilderness (GAL): http://www.fs.usda.gov/rds/archive/Product/RDS-2015-0023
Selway-Bitterroot Wilderness (SBW): http://www.fs.usda.gov/rds/archive/Product/RDS-2015-0024