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Wildland Fire Reburning Trends Across the US West Suggest Only Short-Term Negative Feedback and Differing Climatic Effects

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B Buma^{1,3} , S Weiss², K Hayes¹ and M Lucash²¹ Dept. of Integrative Biology, University of Colorado, Denver, Colorado, United States of America² Portland State University, Portland, OR, United States of America³ Author to whom any correspondence should be addressed.E-mail: brian.buma@ucdenver.edu**Keywords:** reburning, fire, short interval fire, ecology, resilience, climate change, fire feedbacksSupplementary material for this article is available [online](#)**Abstract**

Wildfires are a significant agent of disturbance in forests and highly sensitive to climate change. Short-interval fires and high severity (mortality-causing) fires in particular, may catalyze rapid and substantial ecosystem shifts by eliminating woody species and triggering conversions from forest to shrub or grassland ecosystems. Modeling and fine-scale observations suggest negative feedbacks between fire and fuels should limit reburn prevalence as overall fire frequency rises. However, while we have good information on reburning patterns for individual fires or small regions, the validity of scaling these conclusions to broad regions like the US West remains unknown. Both the prevalence of reburning and the strength of feedbacks on likelihood of reburning over differing timescales have not been documented at the regional scale. Here we show that while there is a strong negative feedback for very short reburning intervals throughout wildland forests of the Western US, that feedback weakens after 10–20 years. The relationship between reburning intervals and drought diverges depending on location, with coastal systems reburning quicker (e.g. shorter interval between fires) in wetter conditions and interior forests in drier. This supports the idea that vegetation productivity—primarily fine fuels that accumulate rapidly (<10 years)—is of primary importance in determining reburn intervals. Our results demonstrate that while over short time intervals increasing fires inhibits reburning at broad scales, that breaks down after a decade. This provides important insights about patterns at very broad scales and agrees with finer scale work, suggesting that lessons from those scales apply across the entire western US.

1. Introduction

Fires are widespread agents of change in the world's forests (Williams *et al* 2016). Most ecosystems contain many species adapted to their local fire regime as a result (Buma *et al* 2013). However, if fire rates increase, burns should begin to intersect with recent fires, termed 'reburning,' 'short-interval fires,' or more broadly 'interacting disturbances.' Short interval fires can have an outsized impact on ecosystem functioning due to a lack of adaptation to the increased fire frequency (Walker *et al* 2018), sometimes resulting in forest loss (Jackson *et al* 2009). As a result, the impacts of reburning over relatively brief intervals are a major research focus (Buma *et al* 2013, Donato *et al* 2016,

Hart *et al* 2019, Stevens-Rumann and Morgan 2019, Turner *et al* 2019). Rapid, directional ecosystem changes, such as converting those forests to alternate ecological regimes (e.g. grasslands), occur via several mechanisms: reducing the efficacy of adaptive resilience mechanisms (e.g. serotiny, Buma and Wessman 2011; resprouting, Fairman *et al* 2019), interval squeeze (Enright *et al* 2015), reseeding failure (Bowman *et al* 2014), decreased organic layer depth (Brown and Johnstone 2012), and changes in species flammability producing positive or negative feedbacks with future fire probability (e.g. Brooks *et al* 2004, Paritsis *et al* 2015).

How the rate of reburning is changing, and where, is therefore an important question. While not all

portions of the world are expected to see increasing fire rates, most temperate latitudes are likely to see major increases in fire frequency (Mortiz *et al* 2012). Previous work suggests increasing fire rates could limit future fire occurrence—a negative feedback (Parks *et al* 2015). This feedback can be useful, as prescribed burning can limit ‘unplanned’ fires (Price *et al* 2015). But for how long? It is vital that we understand the prevalence, trends, and correlating factors with reburning in recently burned forest ecosystems, but while there are many excellent local and landscape scale investigations of reburning trends and feedback mechanisms (e.g. Parks *et al* 2016, Harvey *et al* 2016, Erni *et al* 2017, Tepley *et al* 2018, Hart *et al* 2019), a broad-scale assessment of trends has not been attempted. It is important to determine if trends and feedbacks suggested at finer scales do indeed scale up to broad regional patterns.

Mechanistically, much has been learned about reburning and feedback mechanisms. Climate and topography strongly influence recovering vegetation that ultimately fuels subsequent fires (Coppoletta *et al* 2016, Erni *et al* 2017, Grabinski *et al* 2017, Parks *et al* 2018) and climate (especially inter-annual moisture deficits, Westerling 2016) affects flammability as well. Extreme fire conditions, such as intense drought, can overwhelm negative feedbacks on burning associated with low fuel loads (Parks *et al* 2018, Tepley *et al* 2018), and increasing aridity will lead to increasing burn probabilities in many locations (Coppoletta *et al* 2016, Littell *et al* 2016, Keyser and Westerling 2017). Looking past the first fire, however, suggests that fuel limitations immediately after a burn will limit reburn activity—at least until the landscape regenerates biomass (Heon *et al* 2014, Coppoletta *et al* 2016). Conceptually, fine-scale studies suggest that climate-vegetation dynamics should result in differential feedbacks—moisture driving increased fire frequency in productive systems that are primarily fuel limited (with droughts limiting reburning; Parks *et al* 2018), and drought driving increased fire frequency in systems that have substantial biomass but are primarily ignition/condition limited. The role of topography is realized at finer scales, with southerly aspects and steeper slopes generally drier and less productive and playing into the pattern of burning in any given event (Harris and Taylor 2017). These factors driving feedbacks between fires have been well documented in local studies (for a review, see Prichard *et al* 2017).

Despite this, the prevalence and temporal nature of reburning likelihood—how long the negative feedback lasts, and when/where it disappears—has not been explored at broad scales. In other words, the ability of previous studies to scale up their conclusions, that there is generally a significant feedback on fire likelihood for ~10–20 years (e.g. Harvey *et al* 2016, Parks *et al* 2018), is unknown. Here we explore that potential for scaling previous, foundational, and site-specific work to the wildland forests of the Western

US. We explicitly do not look at managed areas, deforested areas, or landscapes with intense human presence, which dramatically alters fire likelihood (e.g. Fusco *et al* 2016). The data presented here are the first to document reburning trends at this scale, providing context for this important driver of long-lasting ecosystem change.

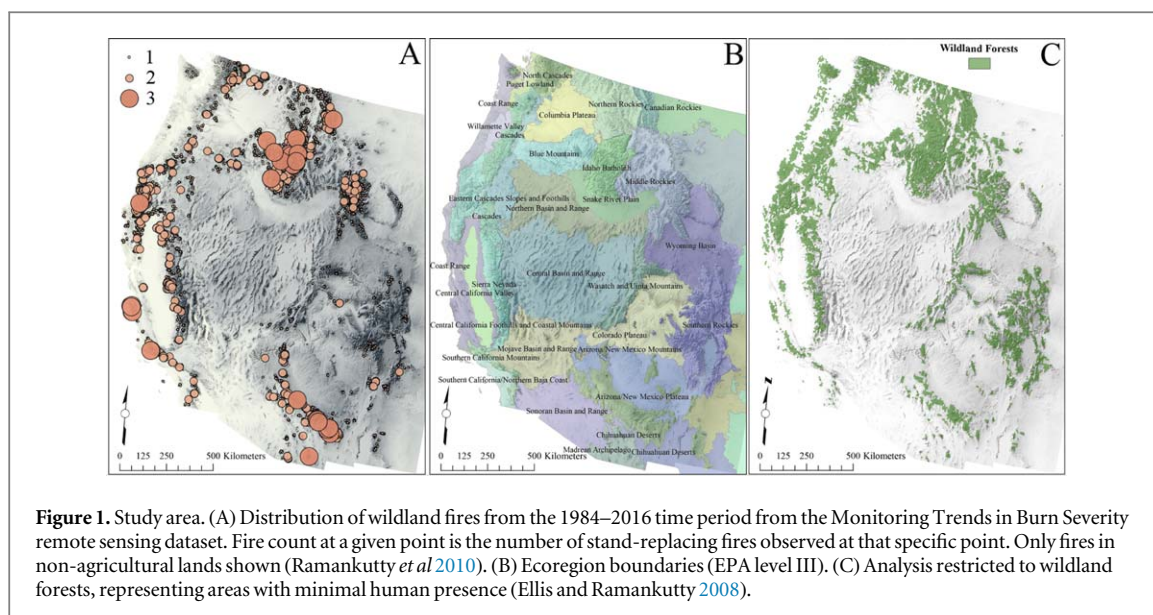
We use the longest spatially-explicit, high resolution fire history mapping (1984–2016, 30 m resolution) available to test that intuition across the Western US. Our primary questions are: Are increasing fire rates inhibiting reburning frequency and at what time-scales? Are those reburns associated with differing topographic contexts, and is that association changing with time? Are reburns associated with droughtier conditions, and is that relationship changing with time?

2. Methods

The analysis region spans the western US (figure 1). The analysis was limited to reburning in wildland forests. This was done using remote sensing derived landcover classification and land use maps (anthromes: Ellis and Ramankutty 2008, agriculture fraction: Ramankutty *et al* 2010). The anthrome dataset was created using population density, land use, and land cover. To focus on forests and limit human influence, we restricted the analysis to pixels classified as ‘remote forests’ or ‘wild forests’ indicating little to no human presence. This was done to avoid unintentional topographic/climatic/ecoregion bias that could be introduced by reburning associated with human causes (Fusco *et al* 2016). We eliminated areas with >1% agriculture in the pixel, a conservative step intended to remove any spurious reburning associated with farming.

2.1. Data sets

For burn data within those wildland forest locations, we used the Monitoring Trends in Burn Severity (MTBS) database (www.mtbs.gov; 1984–2016). The MTBS database is a remote-sensing based mapping of all major (>404 ha) fires in the United States (30 m resolution). As such, small fires <404 ha are not included here. Severity is classified from 1 to 4, with 1 being low severity/unburned and 4 being severely burned. Severity is based on pre- versus post-reflection data. Because this is an automated process, there can be concerns about the cutoffs used between categories and unburned areas may be included within burn perimeters (Kolden *et al* 2015). Therefore, we limited our analysis to locations with severity ≥ 2 ; this cutoff was applied to avoid unburned/low severity locations within fire, avoids errors in burn perimeters, and limits inconsistencies in severity classification between fires.



For drought magnitude, the standardized precipitation-evapotranspiration index (SPEI) was used (Beguería and Vicente-Serrano 2014). SPEI is an extension of the standardized precipitation index that incorporates evapotranspiration, making it more relevant for ecosystem-climate interactions, such as how drought impacts fuel flammability. SPEI was calculated for the 12 month period prior to each year and extracted for each burn year (0.5° resolution, 1984–2016); high SPEI indicates moisture surplus, low a deficit. It is standardized within the time series, and so was used for comparing differences in conditions at a given point rather than direct numerical comparisons between points. SPEI was extracted for each burn point for the year of the fire, incorporating yearly variability in weather. For topography, slope and aspect were extracted for each point (NASA ASTER; 30 m resolution). Aspect was transformed to 0 (north/northeast) –1 (south/southwest) scale using Moisen and Frescino (2002).

Data were grouped according to their Level III EPA ecoregion (US Environmental Protection Agency 2013, figure 1); ecoregions with <10 fires overall were eliminated for the analysis to limit spurious noise. Note that because we delineated forested area via the more detailed anthrome classification and then grouped by ecoregion, some fires are included from ecoregions not typically considered forested (e.g. coastal California) and others were excluded from ecoregions typically forested (e.g. if they occurred within non-forested parts of that ecoregion). For computation efficiency, the region was sampled at a density of approximately 1 point per 2 km^2 (final $n = 185\,423$; figure S1 is available online at stacks.iop.org/ERL/15/034026/mmedia). Percentage of total burn area was calculated as the sum of burned points versus total points. Reburns were defined as single

point locations with multiple burn dates, as defined by the MTBS dataset.

To assess trends in wildland fires, a moving window was used to avoid biasing reburn counts to later years, given the longer period of observation inherent at later time points. Reburns can be arbitrarily defined; here we use four intervals: 5, 10, 20, and 25 year. As the interval increases, the ability to compare across years declines due to the MTBS observation availability (33 years). These intervals were chosen because they match and then approach critical points where reburning threatens forest resilience; 25 years is approximately the timespan when many seed banks of forest tree species begin to re-establish, whereas 5 years is clearly too short for tree species across the US West (Buma *et al* 2013) and forest recovery is unlikely. The moving window approach provides the same temporal interval for each data point, enabling consistency when calculating trends. As a result, the first focal year begins in 1988 and 1993 for 5 and 10 year intervals, respectively (and later for 20 and 25 year intervals). This total was divided by the total number of fires in the focal year to avoid conflating an increase in reburning rates with increased burning rates overall. This was done both overall and for each ecoregion.

2.2. Null model

A reburn is an overlap between a fire in any given focal year and the fire footprints of previous fires. However, one has to also define an interval in the past that constitutes a short-interval fire (for all fires are essentially reburns over some time period). To determine if observed reburning rates are different from a purely random distribution of fires in a given area, a null model for each year was created. First, the percentage of landscape burned in the past interval under consideration was calculated as:

$$B_{interval} = \frac{\sum BurnPoints_{interval}}{TotalPoints} * 100,$$

where $BurnPoints_{interval}$ are all points with an observed fire during the temporal interval prior to the focal year, and $TotalPoints$ are all points in the dataset. This results in a percentage area burned in the preceding time period under consideration. For example, for a 10-year interval, all burned points in the previous 9 years would be summed and the percentage of landscape burned in that time period calculated. Second, the percentage of landscape burned in the focal year t , denoted B_t , was calculated as:

$$B_t = \frac{BurnPoints_t}{Total Points} * 100,$$

where $BurnPoints_t$ is the number of fires in the focal year. This results in the percentage area burned in the focal year.

$B_{interval}$ and B_t both represent percentages of the landscape. These two percentages were randomly distributed in 10 000 cell environments, creating two independent sets of simulated burn locations at the same percentage as observed fires. Expectations for reburning frequency was calculated as the number of cells which overlapped, i.e. the sets intersected:

$$ReburnEvents = Count\ of\ B_{interval} \cap B_t.$$

And the null expected reburn percentage for that focal year was calculated as:

$$ReburnPercent_{null} = \frac{ReburnEvents}{B_t}.$$

This process was repeated 10 000 times for each focal year, and the mean and 95% percentiles calculated.

This null model naturally produces increasing reburn rates with time, a simple consequence of the increase in overall fire rate ($p < 0.001$, $r^2 = 0.85$). What is of interest is the rate of change, and in particular if the rate of increase driven by increasing fire rates (the null model) is matched by observed changes in reburn rates. Rates of change in the null and observed models were compared with a standard ANCOVA (two-tailed). Differences between the null and observed values were also assessed via subtracting the observed rate from the mean null rate, with 95% confidence intervals constructed from the 0.05% and 0.95% quantiles of the null rate distribution. For the ecoregion scale, the process was identical but constrained to only points and percentages within the focal ecoregion.

2.3. Interval

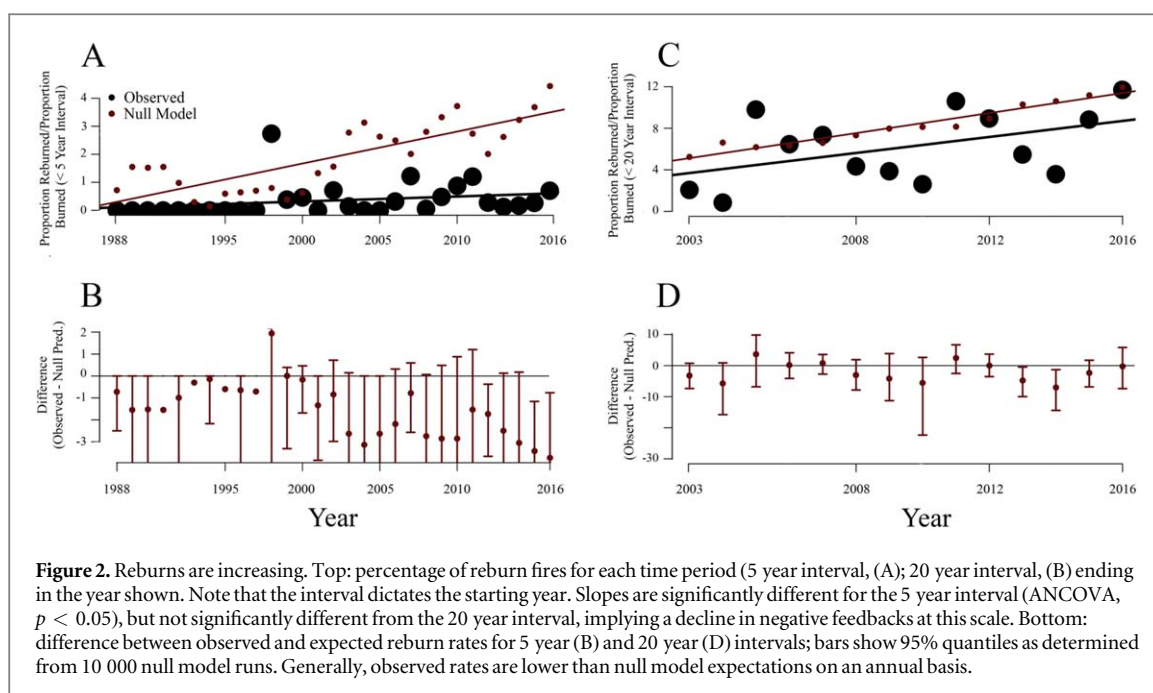
To assess the importance of topography (slope/aspect) and drought on reburning intervals, we used linear mixed modeling. Mixed models are useful for questions regarding the importance of factors at broad scales, especially where factors may vary as a function of location. Multiple model structures were developed using ecoregion as a random (grouping) variable and

slope, aspect, and SPEI as either fixed variables or as random slopes, depending on the model (fixed variables act equivalently across all groups, random slopes allows the variable influence to vary as a function of the grouping variable). The interval between consecutive fires was the response. The most parsimonious model was determined via AIC. Significance was determined via the 95% confidence intervals of effect sizes. A Holm-Bonferroni correction was applied to limit Type II error inflation. Processing was primarily done in ArcGIS, with analyses in R (R Core Team 2014). A universal alpha value was set at 0.05.

3. Results

Overall, fires impacted 15.2% of wildland forests between 1984 and 2016 (figures 1, S2), with 92.5% burned once versus 7.4% burning twice and 0.2% burning thrice. Reburns were spread across the region, but primarily concentrated in southern California, the southern desert mountains, and the Idaho portion of the Rockies (table S1). Although reburns occurred on significantly steeper slopes compared to unburned locations ($p < 0.05$, figure S4 inset) and trended towards southwest aspects (though not significantly, figure S5), the influence of topography on the interval between repeat fires was muted. Overall then, reburns in the observational period were associated with steep, south facing slopes and drought in most ecoregions, though some ecoregions tended to reburn in wetter conditions.

Trends in reburning proportion: overall reburn proportions increased significantly ($p < 0.05$) regardless of how a reburn is classified (5, 10, 20, or 25 years between fires (figures 2; S3). At the ecoregion level, sample size is significantly smaller, and short-interval burns (<10 years) only increased significantly in the Idaho Batholith (~1% per decade increase, $p = 0.02$, $r^2 = 0.36$). Reburns should increase because of increasing fire rates, of course, and so were tested against the null model. At a 5 year reburn interval, the difference between observed reburning rates and the null model results increased with time, suggesting fuel limitation. The rate of observed reburn frequency is nearly zero ($p > 0.05$) despite rapidly increasing fire frequency overall (0.2% per year). Put another way, the slope of the null model is significantly higher than the slope of the observed change (ANCOVA, $p < 0.05$). Meaningfully, this negative feedback disappeared when looking at the 20 year interval trends (0.38% per year, $p = 0.09$; $F = 3.39_{1,12}$), where reburn rates increased just as fast as the increasing fire frequency would predict assuming no interactions (slopes not significantly different: ANCOVA, $p > 0.05$; figure 2). 10 and 25 year intervals agreed with this trend of declining inhibition as intervals increase (figure S3).



Reburning Interval Analysis: for the mixed model, which focused on the interval between fires, the most parsimonious model form included (topographic) slope as a fixed effect and SPEI as a random slope varying by ecoregion (i.e. effect magnitude and direction varying by ecoregion). Aspect was not included in the most parsimonious model. Steeper slopes had longer intervals across all ecoregions (figure S4). While the effect sizes of SPEI on reburn interval overlapped zero for many ecoregions, several estimates were significantly different from zero (figure 3). Coastal regions generally experienced shorter reburn intervals when conditions were wetter (higher SPEI was associated with shorter return intervals), whereas the estimate for several interior regions was the opposite, with wetter conditions (higher SPEI) favoring longer reburn intervals.

4. Discussion

This study demonstrates that reburns are increasing significantly at very broad scales, agreeing with expectations that as fire frequency increases so will short-interval events. However, it also demonstrates a short-term negative feedback that is limiting extremely short-interval fires in most locations. This suggests that patterns observed at finer scales (local, landscape, and regional scale studies, such as Parks *et al* 2015) scale to wildland forests of the US West, and that interpretations of the mechanisms posited by those studies likely apply across broader regions.

The substantial increase in fires across the West has been well documented (Dennison *et al* 2014, Westerling 2016), and with it an expectation for increasing reburns. Reburning now occurs within 7.4% of total observed fires in US West forests. Given

the increased rate of total fires, the significant increase in observed reburns is in line with expectations. The observed rate is less than anticipated due to chance in the short-term, however. This is likely associated with limitations (e.g. Parks *et al* 2015, Harvey *et al* 2016, Prichard *et al* 2017), as the effect is strongest for the shorter time period and gets progressively weaker when longer intervals are considered. However, even the longest intervals considered here (25 years) are still short relative to the life history and reproductive timelines of dominant tree species across North America (Buma *et al* 2013). A 20 year reburn interval still poses a threat to many species whose resilience mechanisms cannot cope with such frequent fires, such as lodgepole pine (*Pinus contorta*), a dominant species across North American mountain forests (Buma *et al* 2013). Thus, the increase in the rate of fire does appear to pose a risk via reburning at broad scales, despite short-term negative feedbacks.

These results suggest 20 year interval reburn rates will continue to increase as fast as the overall fire rate, whereas 5 and 10 year intervals are currently partially inhibited by previous fires. On a yearly basis, there are few significant differences between the null and expected early in the record due to low fire activity overall and subsequent high variability in the null model; in the last decade there are several years with significantly lower reburning than expected (figure 2, bottom) corresponding to recent years of very high fire activity.

Drought strongly influences the interval between fires in many ecoregions (figure 3). Droughty conditions had an opposite effect depending on ecoregion; short intervals were associated with less droughty conditions in more coastal areas. We interpret this as a function of productivity of fuels (review: Prichard *et al* 2017, also explored in

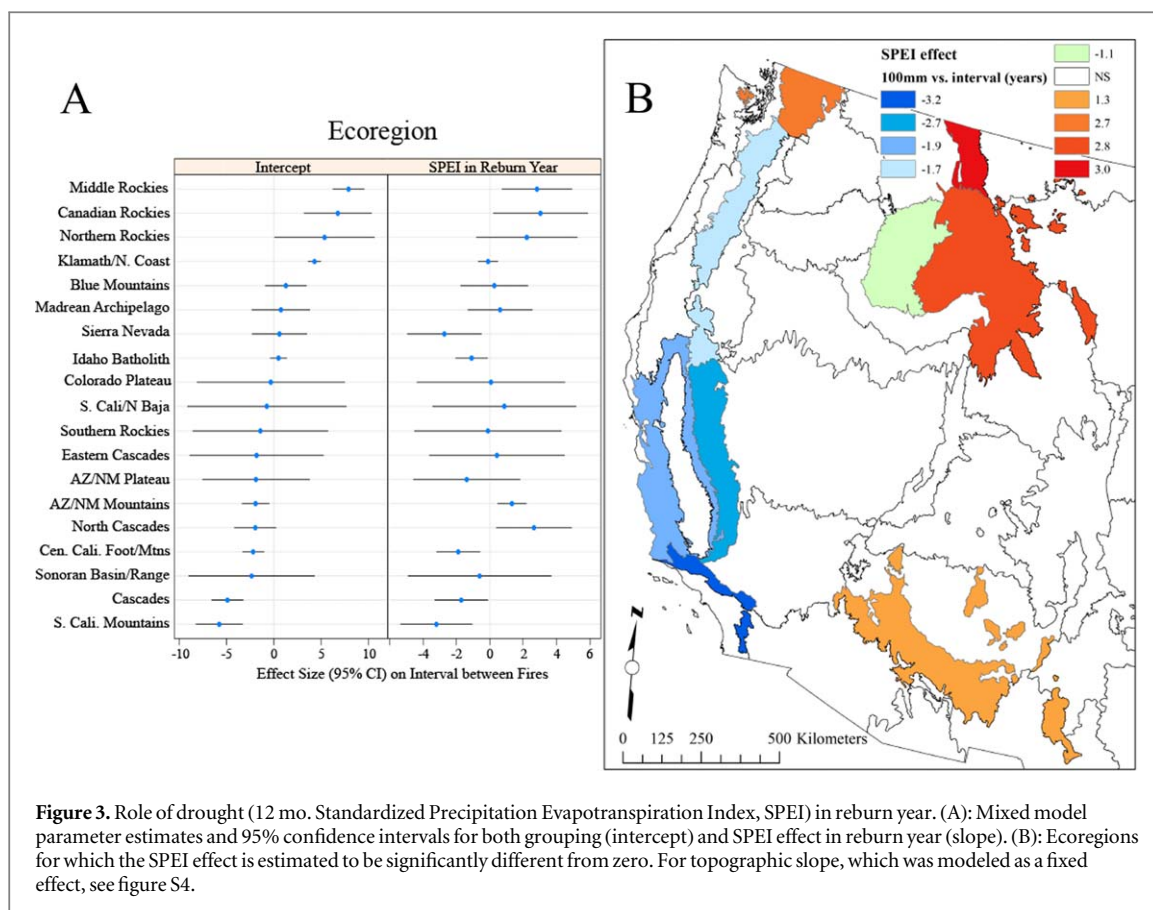


Figure 3. Role of drought (12 mo. Standardized Precipitation Evapotranspiration Index, SPEI) in reburn year. (A): Mixed model parameter estimates and 95% confidence intervals for both grouping (intercept) and SPEI effect in reburn year (slope). (B): Ecoregions for which the SPEI effect is estimated to be significantly different from zero. For topographic slope, which was modeled as a fixed effect, see figure S4.

Parks *et al* 2018), potentially due to increasing fine/grassy fuels associated with wetter conditions and corresponding to a reduction in tree cover with fire (Grabinski *et al* 2017). The drier conditions in interior forests are consistent with this concept as well—fuels (in the form of woody debris) do exist but require extreme drought to carry fires, and the low productivity in general limits fine fuel accumulation, at least over this observational period. It should be noted that drought/moisture conditions vary over multiple timescales, some of which may be longer than the timescale of observation (1984–2016) and the role of drought over longer time periods should be explored with other site-specific proxies. Overall, divergence in terms of reburn response to drought between coastal California forests and the remainder of the West may be significant in the future as climate shifts. Demonstrating that these feedbacks hold over the entire US West is valuable and useful in constraining future broad-scale modeling efforts, as well as validating inferences from finer scale studies.

Positive severity feedbacks (Barker and Price 2018) have been observed at finer scales and the role of management can be critical in managing these complex interactions between climate and topographical tendencies towards reburning (reported here, Prichard *et al* 2017), the local specifics of recovering vegetation (e.g. invasive species; Brooks *et al* 2004), and human actions (management/land use change; Thompson *et al* 2007, Fusco *et al* 2016). Prescribed fire may be a useful tool towards managing reburning impacts after

a major wildfire (Barker and Price 2018) by strongly reducing fuel loads and avoiding intense reburns that may trigger forest loss (Buma *et al* 2013, Enright *et al* 2015). Biome types can influence the efficacy of prescribed fire in reducing future fires via fuel reductions, the same mechanism indicated here (Price *et al* 2015). An interesting question is the role of historical fire suppression (early to late 20th century) and its role in reburn probabilities; fires in suppressed areas can be more intense due to increased fuels (with high variability; Steel *et al* 2015), but the impacts on reburn probabilities after that fire are unknown. Fire prediction and effects modeling must take these spatial and temporal interactions into account, rather than naïve non-spatial modeling, because of the strong role of fire history in future burning at any given locations.

This study demonstrates an increase in reburning across the US West, but also ample evidence for negative feedbacks. It suggests that feedback mechanisms proposed at finer scales are indeed relevant across extremely broad regions, but that those feedbacks are relatively short-lived. Overall, as fires increase so do reburns, and with them new challenges for ecosystems and society.

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