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Wildland fire deficit and surplus in the western United States, 1984–2012

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Abstract. Wildland fire is an important disturbance agent in the western US and globally. However, the natural role of fire has been disrupted in many regions due to the influence of human activities, which have the potential to either exclude or promote fire, resulting in a "fire deficit" or "fire surplus", respectively. In this study, we developed a model of expected area burned for the western US as a function of climate from 1984 to 2012. We then quantified departures from expected area burned to identify geographic regions with fire deficit or surplus. We developed our model of area burned as a function of several climatic variables from reference areas with low human influence; the relationship between climate and fire is strong in these areas. We then quantified the degree of fire deficit or surplus for all areas of the western US as the difference between expected (as predicted with the model) and observed area burned from 1984 to 2012. Results indicate that many forested areas in the western US experienced a fire deficit from 1984 to 2012, likely due to fire exclusion by human activities. We also found that large expanses of non-forested regions experienced a fire surplus, presumably due to introduced annual grasses and the prevalence of anthropogenic ignitions. The heterogeneity in patterns of fire deficit and surplus among ecoregions emphasizes fundamentally different ecosystem sensitivities to human influences and suggests that large-scale adaptation and mitigation strategies will be necessary in order to restore and maintain resilient, healthy, and naturally functioning ecosystems.

Key words: climate; fire deficit; fire departure; fire exclusion; fire occurrence; fire suppression; fire surplus; invasive species; protected areas; wilderness; wildland fire.

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Introduction

The arrival of Euro-American settlers in the late-1800s disrupted the natural ecological role of fire in the western US (Keane et al. 2002), thereby resulting in departures in fire activity compared to earlier periods (Safford and Van de Water 2014). These departures in fire activity have been directly linked to a decoupling of the relationship between climate and fire, which, prior to Euro-American settlement, was fairly strong (Marlon et al. 2012). Consequently, many fire-prone ecosystems in the western US have experienced a reduction in fire activity since Euro-American settlement compared to earlier periods (e.g., Mallek et al. 2013). Such decreases in fire activity have been attributed to human activities such as fire suppression, livestock grazing, logging, land-type conversion (e.g., to agriculture), and other anthropogenic influences (e.g., roads; Savage and Swetnam 1990, Heyerdahl et al. 2001, Allen et al. 2002, Marlon et al. 2008). Conversely, *increases* in fire activity have also been documented in certain ecosystems or localized areas due to factors such as invasive species that facilitate fire ignition and spread as well as high rates of human-caused ignitions (D'Antonio and Vitousek 1992, Syphard et al. 2007, Balch et al. 2013). Although many of these studies definitively demonstrated that fire activity today is much different in the western US compared to historical conditions (e.g., pre Euro-American settlement), some have questioned the relevance of a past time period in defining reference conditions given that contemporary climate change is altering the biophysical environment (Harris et al. 2006).

At broad spatial scales, climate clearly influences fire activity, although the temporal resolution of analysis can result in different interpretations (Parisien et al. 2014). For example, fire activity has been shown to strongly correlate with climatic normals (e.g., 30 year averages; hereafter "climate;" Krawchuk et al. 2009, Parisien et al. 2011, Moritz et al. 2012). This is likely an indirect influence on fuel conditions via climate's effect on productivity, moisture, and dominant vegetation type (Krawchuk and Moritz 2011, Pausas and Ribeiro 2013, Parks et al. 2014). This climatic effect is in contrast to inter-annual *climatic variability* that also influences fire activity, whereby warm and dry years generally correspond to increased fire activity (Westerling et al. 2006, Heyerdahl et al. 2008, Littell et al. 2009). We suggest that, because climate (i.e., climatic normals) is a strong top-down control on fire activity (although it may be an indirect control), a natural level of fire activity can be defined as that which emerges from the climate. As such, the "expected" amount of fire can be statistically modeled as a function of climate. Indeed, this rationale is often invoked in studies evaluating the effect of climate change on future fire activity (Krawchuk et al. 2009, Moritz et al. 2012, Batllori et al. 2013). This rationale also allows for an evaluation of contemporary departures in fire activity by comparing the expected and observed area burned. Where observed fire activity is less than expected, the result is a "fire deficit" because there is less fire than the climate dictates (Marlon et al. 2012). Where observed fire activity is greater than expected, the result is a "fire surplus."

Because ecosystems with disrupted fire regimes are thought to be less resilient to environmental change (Millar et al. 2007), land managers often cite the need to restore wildland fire as a natural disturbance (NWCG 2001). To accomplish this, land managers need to better understand current departures from natural levels of fire activity, especially considering that restoring wildland fire necessitates that our communities learn to co-exist with fire (Moritz et al. 2014). However, quantifying natural levels of fire activity is challenging for two reasons. First, as climate changes, the natural level of fire activity is a moving target. Second, human activities continue to alter fire regimes (as previously described). Fortunately, protected areas and other lands with little anthropogenic impact can provide a benchmark for quantifying natural levels of fire activity as a function of climate. For example, Parks et al. (2014) found that the relationship between climate and fire activity was reasonably strong in the western US when models were built using samples primarily comprised of protected areas; the relationship was weak when models were built using samples irrespective of the protected area composition (Appendix A). Archibald et al. (2010) had a similar finding in their study of fire and climate variability in southern Africa. A comparison of expected (according to the natural benchmark) to observed fire activity then allows for a formal evaluation of fire deficit and surplus for lands of all management designations.

We sought to quantify departures from contemporary natural levels of expected fire activity and identify areas of fire deficit and surplus in the western US for the 1984–2012 time period. This was achieved by developing a boosted regression tree (BRT) model to predict area burned as a function of several climatic variables for 500 km^2 hexagonal polygons (hereafter "hexels"). The model was built using fire and climate data from a subset of hexels in the western US having substantial area in a protected

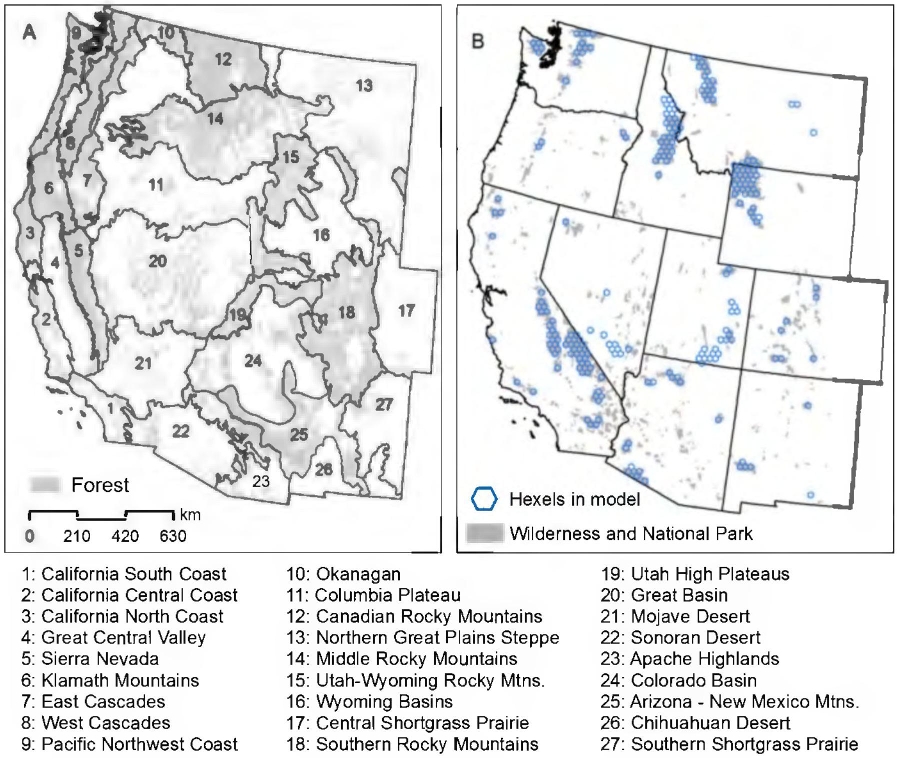

Fig. 1. Study area of the western US for which we quantify departures in expected area burned. Map showing forested areas and ecoregion boundaries (A) and showing designated wilderness areas and national parks as well as the hexels we used to build the model of area burned (B).

status (e.g., wilderness and parks) or otherwise having a low anthropogenic influence. These hexels serve as a natural benchmark (e.g., Appendix A) and are representative of a broad gradient in climate (Appendix B) and include ecosystems such as desert, dry conifer forest, and cold conifer forest. We then quantified the degree of fire deficit or surplus as the difference between expected (as predicted with the BRT model) and observed area burned from 1984 to 2012 for all hexels in the western US and provided ecoregional summaries. Our study is relevant to the contemporary time period (i.e., 1984–2012) and does not reflect prior time periods in which the climate was different or indigenous burning was common.

Methods

We quantified departures in expected area burned from 1984 to 2012 for each 500 km^2 hexagonal polygon (i.e., "hexels") in the western US (Fig. 1). We chose this hexel size based on previous work (Parks et al. 2014), although we acknowledge that the resolution of analysis may influence the relationship between fire and its environment (Parisien et al. 2011, Parks et al. 2011). We built a model of area burned as a function of climate and then used model results to quantify fire departures.

We selected a subset of hexels with low human influence to build the model because the relationship between climate and fire is reasonably strong in areas of low human impact but weakens as anthropogenic influences increase (Archibald et al. 2009, Parks et al. 2014; Appendix A). Consequently, areas of low human impact act as "controls" and therefore characterize the expected relationship between fire and climate were human activities not an overarching influence. We selected hexels comprised of at least 75% designated wilderness and national park or had an average "human footprint" (HFP; Leu et al. 2008) $\leq$1.5 (on a scale of 1–10). We also

Table 1. Climatic variables evaluated as independent variables in the statistical model describing area burned. Only those that had a correlation coefficient <0.9 (bold) to each other were included in the final model ($n = 5$). The coefficient of variation (among years) of the five selected variables were also included in the statistical model for a total of 10 variables. These variables represent climatic conditions from 1984 to 2012.

Variable abbreviation	Description	Units
AET	Actual evapotranspiration (annual average)	mm/year
DMC†	Duff moisture code (annual average)	dimensionless
ERC‡	Energy release component (annual average)	dimensionless
PRECIP	Precipitation (annual average)	mm/year
SWE	Snow water equivalent (monthly average)	mm/month
SMO	Soil moisture (monthly average)	mm/month
TEMP	Temperature (annual average)	°C
VPD	Vapor pressure deficit (monthly average)	kPa
WD	Water deficit (annual average)	mm/year

† ERC is a fire danger index in the National Fire Danger Rating System (NFDRS; Cohen and Deeming 1985).
‡ DMC is a fire danger index in the Canadian Forest Fire Danger Rating System (CFFDRS; van Wagner 1987).

excluded hexels with more than 50% nonfuel (e.g., open water and barren). These selection criteria resulted in 235 hexels for which we built the model of area burned (Fig. 1). Despite representing only a small proportion of the western US (∼4%), the hexels span broad climatic gradients ranging from warm deserts to cold forests (Appendix B). Although human-ignited fires may blur the relationship between climate and fire, only about 8% of the area burned from this subset of hexels was due to such fires (for the years 1992–2012; Short 2014).

Area burned (ha) was calculated within each hexel using a fire atlas covering all fires ≥400 ha from 1984 to 2012 (Eidenshink et al. 2007). Area burned was adjusted to account for unburnable areas (e.g., open water) within each hexel by dividing area burned by the proportion of the hexel that was composed of burnable fuel types (Rollins 2009). Several climatic variables known to influence fire activity were evaluated for inclusion into the model as independent variables (e.g., Littell and Gwozdz 2011, Abatzoglou and Kolden 2013, Parks et al. 2014; Table 1). Gridded monthly temperature and precipitation data were obtained from the parameter-elevation regression on independent slopes model (PRISM; Daly et al. 2002) that uses station data and physiographic factors to map climate at a spatial resolution of ≃800 m. Abatzoglou (2013) combined monthly data from PRISM and hourly estimates from coarser scale analyses from the North American Land Data Assimilation System Phase 2 (NLDAS-2; Mitchell et al. 2004) to produce daily and sub-daily surface meteorological variables at a ≃4-km resolution that captures temperature, humidity, winds, solar radiation, and precipitation, which were subsequently used to estimate the energy release component (ERC; Cohen and Deeming 1985), duff moisture code (DMC; van Wagner 1987), and reference evapotranspiration. These data were collectively used to run the climatic water balance model following Dobrowski et al. (2013) to estimate monthly actual evapotranspiration (AET), snow water equivalent (SWE), soil moisture (SMO), and water deficit (WD). This model operates on a monthly time-step and accounts for atmospheric demand (via the Penman-Monteith equation), soil water storage, and includes the effect of temperature and radiation on snow hydrology via a snow melt model. All variables were characterized by averaging or summing monthly values over each year, then averaging the values over 1984–2012, and finally, averaging within each hexel. Those variables with a correlation coefficient of ≥0.9 (for the 235 hexels used to build the model) were excluded from the model (Table 1). For the five variables that met this correlation criterion, we also included in the model variables representing inter-annual variability (coefficient of variation).

We built our model of area burned as a function of climate using boosted regression trees (BRT) within the R statistical program (R Development Core Team 2007; "gbm" package). BRT is a machine-learning approach that does not require a priori model specification (De'ath 2007). We square-root-transformed area burned to homogenize variance in model residuals and used a Poisson distribution. We followed the recommendations of Elith et al. (2008) for selecting BRT options; we set the bagging fraction to 0.5, learning rate to 0.0025, and tree complexity to 5. We used a custom script from Elith et al. (2008) to determine the necessary number of trees, thereby reducing the potential for overfitting the model. We evaluated model fit using 10-fold cross-validated correlation between predicted and observed values of area burned.

We quantified departures in expected area burned for each hexel in the western US by subtracting predicted from observed area burned for the 1984–2012 time period; negative values indicate less area burned than expected (i.e., a fire deficit), and positive values indicate more area burned (i.e., a fire surplus). We "clamped" predicted area burned to the maximum value in the training hexels to avoid predicting outside of the observed range of area burned. We summarized deficits and surpluses, as well as the expected and observed fire rotation, for each ecoregion (Olsen and Dinerstein 2002; Fig. 1); results for ecoregions with minimal overlap with the study area are not shown (e.g., Black Hills). Hexels on the edge of study area boundary are omitted from the ecoregional summaries if >50% of its area overlaps with the ocean, Mexico, Canada, or areas east of the study area. Hexels are defined as unburnable if ≥50% is characterized as irrigated agriculture, barren, urban, or water (Rollins 2009) and are excluded from the ecoregional summaries.

Results

The BRT performed well; the cross-validated correlation between predicted and observed area burned was 0.80 (see Appendix C for variable importance and response curves for selected variables). Departures from expected area burned from 1984 to 2012 were spatially heterogeneous across the western US (Fig. 2). Some ecoregions exhibited large fire deficits (e.g., California North Coast and Apache Highlands), others a fire surplus (e.g., California South Coast and Columbia Plateau), and others no substantial departure (e.g., West Cascades and Sonoran Desert; Fig. 2, Table 2). Most forested ecoregions experienced a fire deficit, with the notable exception of mesic forested ecoregions in the Pacific Northwest, which showed no substantial departure from expected area burned. Large geographic expanses of fire surplus were apparent in some non-forested regions (e.g., the southern Columbia Plateau and northern Great Basin), although a fire deficit was observed in two non-forested ecoregions (i.e., Central Shortgrass Prairie and Southern Shortgrass Prairie; Fig. 2, Table 2).

Discussion

Our results reveal a striking pattern of departures from expected area burned across the western US from 1984 to 2012. We found that fire deficits were more prevalent in forested ecoregions compared to non-forested ecoregions. In several non-forested ecoregions, large expanses of fire surplus were evident. The contrasting patterns of fire deficit and surplus between forested and non-forested ecoregions emphasize fundamentally different ecosystem sensitivities to human influences. However, there were also large expanses that experienced no substantial departure from expected area burned. Such areas generally corresponded to two types of ecosystems that typically experience little fire: very dry ecosystems that are biomass limited (i.e., warm and cold deserts that have not been invaded by annual grasses) and wet ecosystems (temperate rainforest) that are rarely conducive to burning (Krawchuk and Moritz 2011, Parks et al. 2014).

Most forested areas in the western US experienced a fire deficit from 1984 to 2012. These deficits strongly support the widespread notion that fire activity is reduced by human activities that exclude fire. Although our findings are only relevant to climate and fire over the 1984–2012 time period, they corroborate the findings of Marlon et al. (2012) and multiple fire history studies that indicated a sharp reduction in fire activity since the arrival of Euro-Americans, e.g., in the Sierra Nevada (Taylor 2000, Beaty and Taylor 2008), Klamath Mountains (Taylor and Skinner 2003), California North Coast (Stuart and Salazar 2000), East Cascades (Wright and Agee 2004), Southern Rocky Mountains (Grissino-Mayer et al. 2004), Middle Rocky Mountains (Heyerdahl et al. 2001), Canadian Rocky Mountains (Power et al. 2006), Utah High Plateaus (Stein 1988, Brown et al. 2008), Apache High-

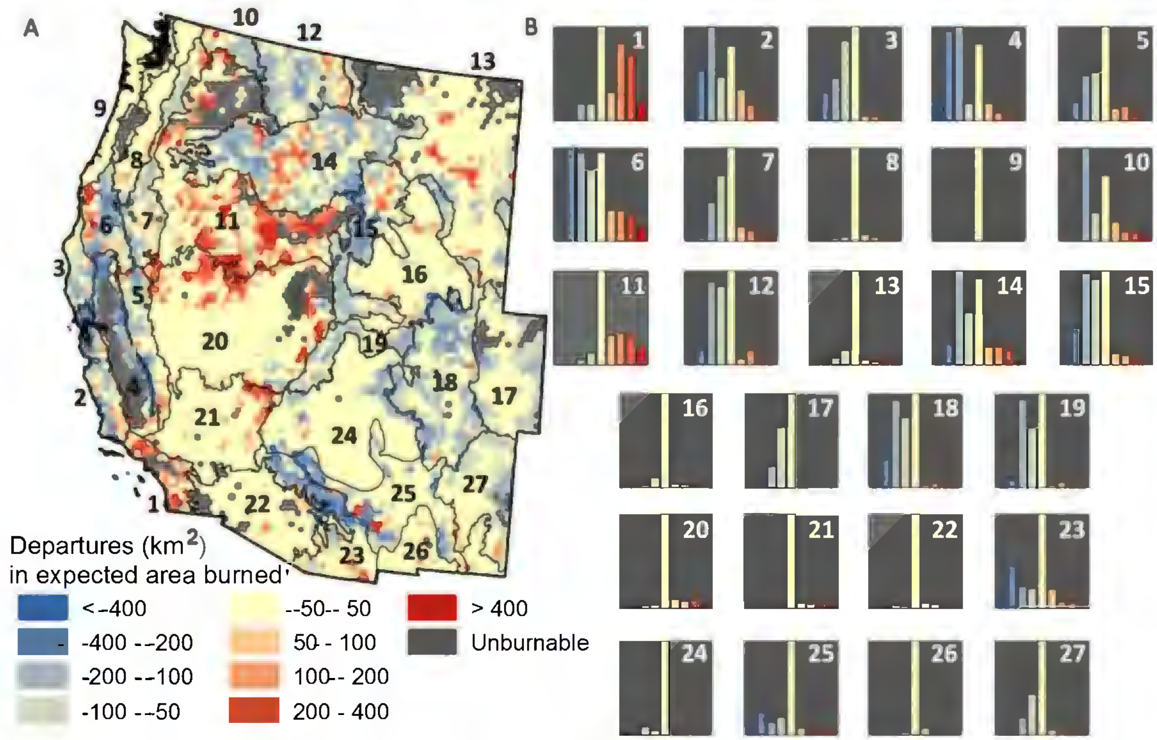

Fig. 2. Map depicting departures in expected area burned from 1984 to 2012 (A). Blue hexels represent less fire than expected (i.e., a fire deficit); red hexels represent more than expected (i.e., a fire surplus); yellow hexels indicate no substantial departure; dark gray hexels are unburnable. Histograms of each ecoregion (B) correspond to fire departure categories shown in the map legend (A).

lands (Baisan and Swetnam 1990), and Arizona-New Mexico Mountains (Swetnam and Dieterich 1985). Decreased fire activity has resulted in substantial changes to some forested ecosystems. For example, reduced rates of burning have resulted in altered species composition (i.e., increases in shade-tolerant species), increased tree density, and increased landscape homogeneity (Taylor 2000, Naficy et al. 2010, Fry et al. 2014, Stephens et al. 2015). A major concern of these site- and landscape-level ecosystem changes is the increased likelihood of uncharacteristically large and severe wildland fire in future years (Hessburg et al. 2005, Stephens 2005, Calkin et al. 2015). Although such fires may contribute to a reduction in the fire deficit, the uncharacteristic manner in which they burn can exceed ecological thresholds and result in conversion to non-forest types (Savage and Mast 2005). A few forested areas did not experience a fire deficit (i.e., Pacific Northwest Coast and West Cascades ecoregions), which is coherent with our knowledge of the fire regime characteristics of these regions where fires are infrequent (fire return intervals of 200–1000 years; Agee 1993).

The pattern of fire surpluses and deficits in non-forested ecoregions is consistent with our knowledge of the impact of human activities on area burned. For example, in the Columbia Plateau and Great Basin ecoregions, the large expanse of fire surplus is likely the result of cheatgrass invasion (*Bromus tectorum*; Fig. 3; Knapp 1996, Bradley and Mustard 2008), a non-native annual grass that was introduced to North America during the 1800s that is known to increase fire activity (Knick and Rotenberry 1997, Balch et al. 2013). Similarly, the fire surplus in the Mojave Desert ecoregion is likely due to invasion by the annual grass red brome (*Bromus rubens*; Hunter 1991, Salo et al. 2005, Brooks and Matchett 2006), another grass introduced from Europe. In many of these affected areas, there is generally a positive feedback where fire facilitates invasive grass establishment, which in turn

Table 2. Ecoregional summary of fire departures in the western US†; values were summed across hexels within each ecoregion.

ID	Ecoregion name	Area (km^2)	Observed area burned (km^2)	Expected area burned (km^2)	Total fire departure (km^2)‡	Expected fire rotation§ (years)	Observed fire rotation§ (years)
1	California South Coast	28376	13256	6268	6987	131	62
2	California Central Coast	39505	7330	14269	−6939	80	156
3	California North Coast	27828	2798	7714	−4916	105	288
4	Great Central Valley	38340	4499	15905	−11406	70	247
5	Sierra Nevada	50000	7121	11793	−4672	123	204
6	Klamath Mountains	50144	10323	15541	−5218	94	141
7	East Cascades	65500	6095	10076	−3981	189	312
8	West Cascades	43000	1774	1992	−217	626	703
9	Pacific Northwest Coast	40177	137	1179	−1042	988	8505
10	Okanagan	24549	3308	5459	−2151	130	215
11	Columbia Plateau	234000	64668	15973	48695	425	105
12	Canadian Rocky Mountains	81230	5957	16871	−10914	140	395
13	Northern Great Plains Steppe	236458	14347	17922	−3575	383	478
14	Middle Rockies	210000	38255	59794	−21539	102	159
15	Utah-Wyoming Rocky Mountains	108500	12393	29836	−17444	105	254
16	Wyoming Basins	132000	3247	5856	−2609	654	1179
17	Central Shortgrass Prairie	108156	650	15094	−14445	208	4825
18	Southern Rocky Mountains	160000	5489	34228	−28739	136	845
19	Utah High Plateaus	45500	2685	9457	−6772	140	491
20	Great Basin	269000	31827	10853	20974	719	245
21	Mojave Desert	128500	8506	475	8031	7845	438
22	Sonoran Desert	101830	3023	2732	291	1081	977
23	Apache Highlands	83987	12359	26610	−14251	92	197
24	Colorado Plateau	197000	3505	12142	−8637	471	1630
25	Arizona-New Mexico Mountains	114572	13491	25820	−12329	129	246
26	Chihuahuan Desert	58407	1347	1221	127	1387	1257
27	Southern Shortgrass Prairie	94031	2761	11402	−8641	239	988

† Hexels are considered unburnable if ≥50% is characterized as irrigated agriculture, barren, urban, or water and are excluded from all calculations.
‡ Negative values indicate a fire deficit and positive values a fire surplus.
§ Fire rotation is defined as the number of years necessary to burn an area the size of the ecoregion and for the 29 year record (1984–2012) is calculated as follows: 29/(expected [or observed] area burned/area of the ecoregion).

increases fire frequency because the invading grass produces a continuous bed of flammable fine fuels (Brooks et al. 2004). Although the Sonoran Desert and Chihuahuan Desert ecoregions exhibited no substantial fire departure from 1984–2012, natural and climate-induced range expansions of invasive annual grasses could put them at risk in the future (Salo 2004, Abatzoglou and Kolden 2011). In addition to the effects from invasive and exotic plant species, the prevalence of human-caused ignitions are implicated in increased fire activity and the associated fire surplus found in the California South Coast ecoregion (Syphard et al. 2007, Keeley et al. 2011, Keeley and Brennan 2012). The fire deficits we found for the Central Shortgrass Prairie and Southern Shortgrass Prairie ecoregions are areas that naturally would support periodic fire, but fire suppression and grazing by domestic cattle has reduced fire activity in these areas (Axelrod 1985, Brockway et al. 2002).

Other studies that also evaluated potential departures in fire activity include Mallek et al. (2013), who used the results of numerous published dendrochronology studies to infer historical annual area burned for several forest types in the Sierra Nevada and made comparisons to contemporary rates of burning. Also, Ager et al. (2014) compared burn probability maps (generated with simulation models) to those of the Landfire "mean fire return interval" product (Rollins 2009) for National Forests in the western US to identify which pixels had higher or lower burn probabilities compared to presumed historical conditions. Similar to our findings, these studies found that contemporary fire activity in forested regions, for the most part, was less than that of the pre-European reference period. However, formal comparisons between our study and these previous studies are not appropriate due to differences in the reference period, spatial resolution, and spatial extent of

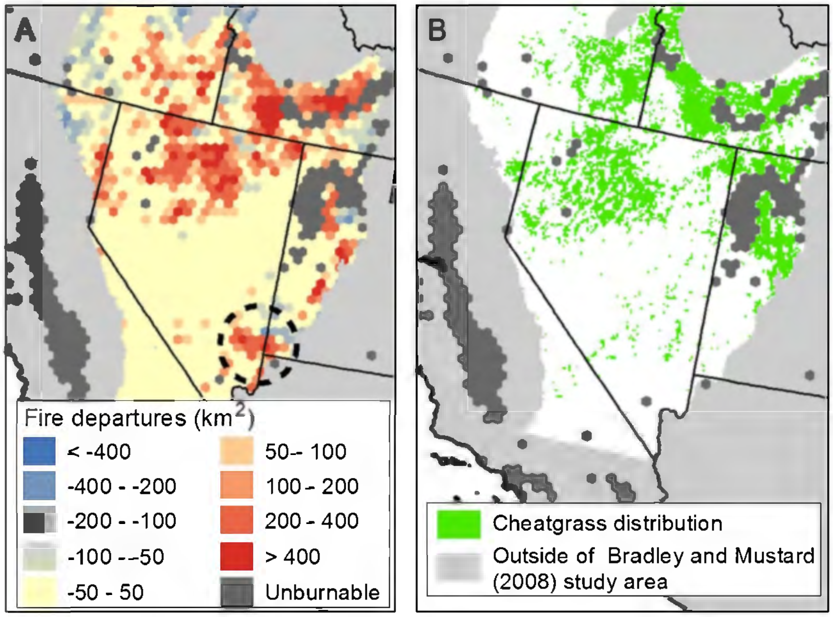

Fig. 3. Departures in expected area burned (km^2) within the Bradley and Mustard (2008) study area (A) and cheatgrass (*Bromus tectorum*) distribution according to Bradley and Mustard (2008) (B). Note the high spatial correspondence between fire surplus and cheatgrass distribution. The large expanse of fire surplus in southeastern Nevada (shown by the dashed circle) is likely due to a different invasive annual grass, red brome (*Bromus rubens*; Salo 2005).

analyses.

Multiple lines of evidence have attributed departures in fire activity between contemporary and historical (i.e., pre-Euro settlement) eras to human activities. Studies using both dendrochronology (i.e., cross-dated fire scar records; e.g., Heyerdahl et al. 2001, Mallek et al. 2013) and charcoal records (e.g., Colombaroli and Gavin 2010) have inferred that humans are at least partly responsible for reductions in rates of burning after about 1900. Our findings suggest that humans continue to influence fire activity today and show how this influence varies among ecosystems. As such, our approach to quantifying departures in fire activity complements earlier research and offers certain advantages. For example, because our model of expected area burned was built from a broad spatial extent, it incorporates all major ecosystems in the western US (e.g., shrub, forest, desert). Specifically, our method is not limited to tree-dominated ecosystems (cf. dendrochronology) or those that have permanent or ephemeral water bodies (cf. charcoal records). Furthermore, because we used contemporary fire and climate data in our study, we defined a contemporary, climatically driven baseline for expected fire activity. As such, in quantifying departures in fire activity, we avoid the use of a past reference period in which climate and indigenous burning practices were quite different (Sheppard et al. 2002, Stephens et al. 2007).

Several factors should be considered when interpreting our results. We assume a space-for-time substitution in calibrating our model; the large sample domain ensures adequate sampling of fires and allows us to infer relationships between fire and climate. Nevertheless, the temporal window of our analysis (1984–2012) is relatively short for many ecosystems (Agee 1993). Consequently, we may not have adequately characterized departures from expected area burned in certain regions, especially in cool/wet forested areas that show an apparent fire surplus only because of large, infrequent fires (sensu Meyn et al. 2007) that have occurred during our study period (e.g., Yellowstone National Park in 1988). For this reason, it is inappropriate to focus on any single hexel or small group of hexels to infer departures in expected fire activity. Instead, it is most appropriate to evaluate fire surplus and deficit over large geographical regions, which explains our rational for providing ecoregional summaries (Fig. 2, Table 2). The temporal window we analyzed (i.e., 1984–2012) also coincides with increased fire activity compared to earlier decades (e.g., 1940–1970; Littell et al. 2009). Consequently, had we analyzed fire and climate data from a different time period, our results would certainly be different. Periodic updates to our model and map are clearly warranted to provide a longer-term perspective on departures in expected fire activity, especially given the heterogeneous nature of fire and that negative feedbacks associated with recurring fires will likely limit future burning in some areas (Héon et al. 2014, Parks et al. 2015). Lastly, our study relies heavily on data from protected areas, which, despite representing the majority of climates in the western US, admittedly do not represent the complete range of climatic conditions (Batllori et al. 2014). Nevertheless, the climatic variability encompassed in the data we used to build the model (Appendix B) translates into a broad range of ecosystem and vegetation types, for example, warm desert (Death Valley National Park [NP]), dry conifer forest (Gila Wilderness), and cold forest (Yellowstone NP). As such, we suggest that poorly represented climates have only a marginal effect on our results.

Conclusions

Previous studies conducted at a variety of temporal and spatial scales identified a sharp drop in fire activity with the arrival of Euro-American settlers in forest-dominated landscapes of the western US (e.g., Marlon et al. 2012, Mallek et al. 2013). Our study makes no comparisons to time periods and climates prior to 1984 but demonstrates that most forested regions continue to experience less fire than expected, or a fire deficit. Our study also demonstrates that some non-forested regions experienced more fire than expected, or a fire surplus, over the 1984–2012 time period. The findings of our study, as well as other efforts such as the Landfire "Vegetation departure" (Rollins 2009) product, suggest that multiple large-scale adaptation and mitigation strategies will be necessary in order to restore and maintain resilient, healthy, and naturally functioning ecosystems (cf. Hessburg et al. 2015). For example, forested areas experiencing a fire deficit may be candidates for restoration treatments (e.g., mechanical fuel treatments, prescribed fire, wildland fire use) intended to promote resilience to future disturbances (Stephens and Ruth 2005, Millar et al. 2007). Conversely, areas experiencing a fire surplus may be indicative of highly altered and degraded ecosystems (Bradley and Mustard 2008, Brooks 2012) that are in need of restoration and ignition management (e.g., fire suppression and prevention). Protected areas and other areas with low anthropogenic influence served as a natural benchmark in which fire activity and climate are relatively tightly linked, and as such, are an invaluable data source for assessing perturbations to fire regimes across multiple ecosystem types.

Acknowledgments

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Supplemental Material

Ecological Archives

Appendices A–C are available online: http://dx.doi.org/10.1890/ES15-00294.1.sm