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LETTER

How will climate change affect wildland fire severity in the western US?

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Abstract

Fire regime characteristics in North America are expected to change over the next several decades as a result of anthropogenic climate change. Although some fire regime characteristics (e.g., area burned and fire season length) are relatively well-studied in the context of a changing climate, fire severity has received less attention. In this study, we used observed data from 1984 to 2012 for the western United States (US) to build a statistical model of fire severity as a function of climate. We then applied this model to several ($n = 20$) climate change projections representing mid-century (2040–2069) conditions under the RCP 8.5 scenario. Model predictions suggest widespread reduction in fire severity for large portions of the western US. However, our model implicitly incorporates climate-induced changes in vegetation type, fuel load, and fire frequency. As such, our predictions are best interpreted as a *potential* reduction in fire severity, a potential that may not be realized due human-induced disequilibrium between plant communities and climate. Consequently, to realize the reductions in fire severity predicted in this study, land managers in the western US could facilitate the transition of plant communities towards a state of equilibrium with the emerging climate through means such as active restoration treatments (e.g., mechanical thinning and prescribed fire) and passive restoration strategies like managed natural fire (under suitable weather conditions). Resisting changes in vegetation composition and fuel load via activities such as aggressive fire suppression will amplify disequilibrium conditions and will likely result in increased fire severity in future decades because fuel loads will increase as the climate warms and fire danger becomes more extreme. The results of our study provide insights to the pros and cons of resisting or facilitating change in vegetation composition and fuel load in the context of a changing climate.

Introduction

Fire regimes in North America are expected to change over the next several decades as a result of anthropogenic climate change (Dale *et al* 2001). Fire activity (i.e., annual area burned and fire frequency) is expected to increase in many regions (Krawchuk *et al* 2009, Littell *et al* 2010) and new research shows that fire seasons are now starting earlier and ending later compared to previous decades (Jolly *et al* 2015). However, the effect of climate change on one very important fire regime characteristic—*fire severity*—is not well-studied or understood (Flannigan *et al* 2009, Hessl 2011). In the context of this paper, we define severity as the degree of fire-induced change to vegetation and soils one year post-fire (Key and Benson 2006, Miller and Thode 2007). For example, a stand-replacing fire in upper-elevation conifer forest is considered high severity because the site has drastically changed one year post-fire compared to pre-fire conditions, whereas a surface fire in a grass-dominated ecosystem is considered low severity because the vegetation is nearly fully recovered one-year post fire.

The severity at which a site burns influences vegetation response and successional trajectory (Barrett *et al* 2011), faunal response (Smucker *et al* 2005), carbon emissions (Ghimire *et al* 2012), and erosion rates and sedimentation (Benavides-Solorio and MacDonald 2005). Furthermore, human safety and infrastructure are influenced by the severity at which a site burns (Miller and Ager 2013), and management responses to fire and allocation of firefighting resources are also influenced by the expected fire severity (e.g., Calkin *et al* 2011). As such, there is a need to better understand how fire severity will respond to a changing climate (e.g., Miller *et al* 2009).

At fine temporal scales, fire severity depends on factors that are highly variable over time, such as fire spread rate and direction (e.g., heading versus backing fire) and weather (Finney 2005, Birch *et al* 2015). At broader temporal scales, however, climate (in terms of climatic normals) is a major influence through its interactive effect on productivity (and hence amount of biomass) and moisture availability (i.e., wet versus dry ecosystems) (Parks *et al* 2014b, Whitman *et al* 2015). Consequently, because fire regimes are intrinsically defined by the characteristics of fires that occur over extended periods of time (years to centuries) (Morgan *et al* 2001), evaluations of fire severity over gradients of observed and predicted climatic normals allows for a formal assessment of how fire severity may respond to climate change.

We seek to quantify how fire severity in the contiguous western United States (US) (hereafter the 'western US') may respond to climate change. We use statistical relationships between observed climatic normals and fire severity (Parks *et al* 2014b, Kane *et al* 2015) to conduct a formal evaluation of future fire severity patterns. Because the relationship between climate and fire regimes is known to be weak in areas of high human impact (Parks *et al* 2014b), we used data from areas with low anthropogenic influence to build a statistical model of fire severity as a function of climatic normals over the 1984–2012 time period. We then predicted contemporary (1984–2012) and future (mid-century; 2040–2069) fire severity using climate data from numerous global climate models (GCMs) for the western US. As far as we know, this study is the first to examine how fire severity may respond to a changing climate over such a broad spatial extent. The results of this study will advance our understanding of fire regimes in the western US in the context of a changing climate and will assist policy makers and land managers to better manage for resilient landscapes.

Methods

Consistent with major fire severity mapping efforts (Key and Benson 2006, Eidenshink *et al* 2007), we define fire severity as the degree of fire-induced change to vegetation and soils. We built a statistical model of fire severity as a function of climate by first partitioning our study area (the western US; figures 1(a) and (b)) into 500 km^2 hexagonal polygons (i.e., 'hexels'). Within each hexel, we summarized fire severity using the delta normalized burn ratio (dNBR) (Key and Benson 2006), a satellite index (resolution: 30 m) that differences pre- and post-fire Landsat TM, ETM+, and OLI images and has a high correspondence to field-based measures of severity such as the composite burn index (CBI; $R^2 \geqslant 0.65$) (van Wagtendonk *et al* 2004, Parks *et al* 2014a). The CBI is a post-fire assessment in which individual rating factors in each of several vertically arranged strata (soil and rock, litter and surface fuels, low herbs and shrubs, tall shrubs, and trees) are assessed on a continuous 0–3 scale indicating the magnitude of fire effects. A rating of 0 reflects no change due to fire, whereas 3 reflects the highest degree of change. Factors assessed include soil char, surface fuel consumption, vegetation mortality, and scorching of trees. Ratings are averaged for each stratum and then across all strata to arrive at an overall CBI rating for an entire plot. The CBI indicates that, as dNBR values increase, there is generally an increase in char and scorched/blackened vegetation and a decrease in moisture content and vegetative cover (Key and Benson 2006). Measurements of fire severity (dNBR and CBI) are generally conducted one year after fire, so any regrowth that occurs within one year will result in reduced severity compared to assessments conducted immediately post-fire; this is particularly relevant for species that recover quickly after fire (e.g., resprouting shrubs, grasses).

Fire severity (i.e., dNBR) data were obtained from the Monitoring Trends in Burn Severity project (Eidenshink *et al* 2007) for all fires ⩾400 ha for the 1984–2012 time period. Raw dNBR values obtained from MTBS were adjusted using the 'dNBR offset' (Key 2006), which accounts for differences due to phenology or precipitation between the pre- and post-fire images by subtracting the average dNBR of pixels outside the burn perimeter. This adjustment can be important when comparing severity among fires (Parks *et al* 2014a). A mean dNBR was calculated using all pixels of all fires that intersected each 500 km^2 hexel; pixels classified as nonfuel were excluded in the calculation of the mean. We square-root transformed mean dNBR values to linearize the relationship to the CBI (figure S1).

We summarized climate normals within each hexel using five variables with known links to fire regimes (e.g., Littell and Gwozdz 2011, Abatzoglou and Kolden 2013, Parks *et al* 2015b): actual evapotranspiration (AET), water deficit (WD), annual

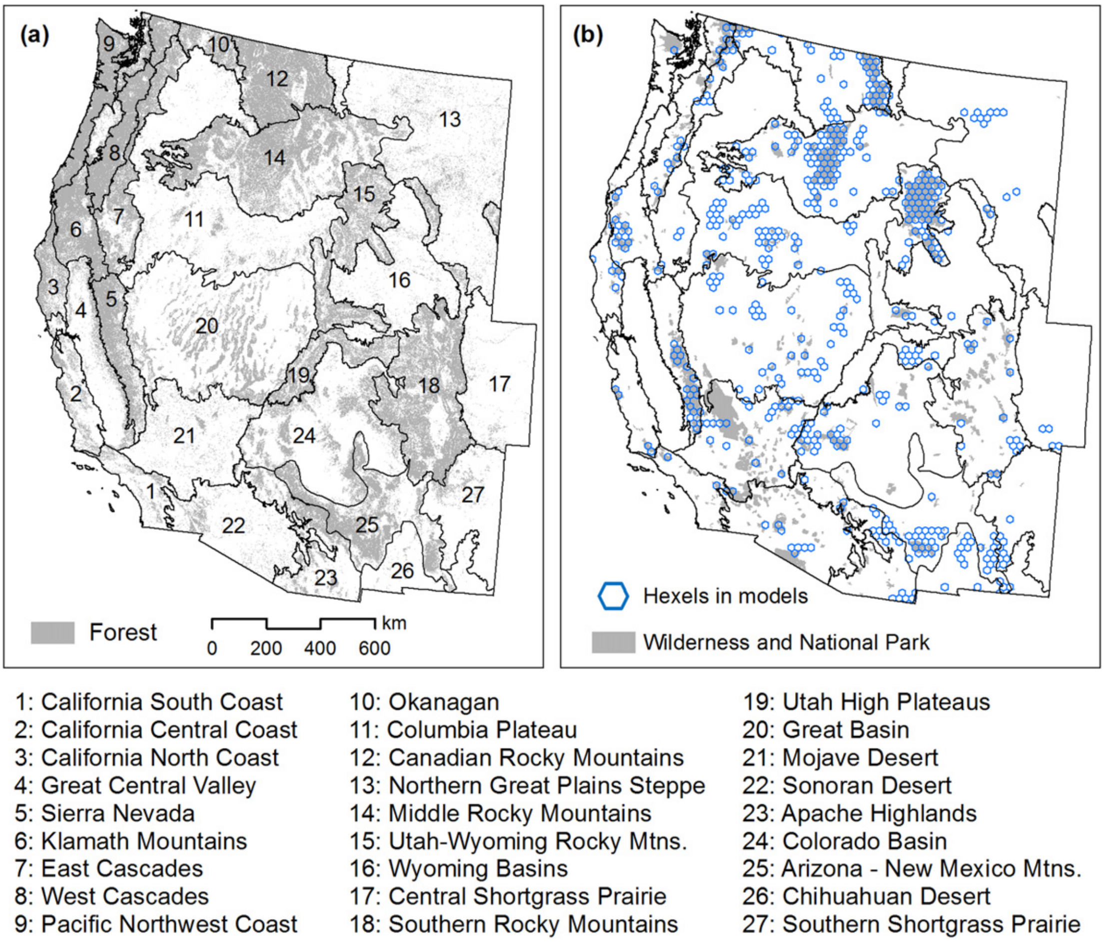

Figure 1. Study area of the western US for which we predicted changes in fire severity under a future climate. Map showing ecoregion boundaries (The Nature Conservancy 2009) and forested areas (in gray) (a) and showing designated wilderness areas and national parks (in gray) as well as the 544 hexels (in blue) used to build the models of fire severity as a function of climate (b). Ecoregion names and boundaries provided for context.

precipitation (PPT), soil moisture (SMO), and snow water equivalent (SWE). Gridded monthly temperature and PPT data were obtained from the parameter-elevation regression on independent slopes model (PRISM; Daly *et al* 2002), which uses weather station data and physiographic factors to map climate at a spatial resolution of ~800 m. In addition, daily and sub-daily surface meteorological variables (~4 km resolution) describing temperature, humidity, winds, solar radiation, and precipitation were produced following Abatzoglou (2013). These data were collectively used to compute climatic water balance following Dobrowski *et al* (2013) to estimate AET, SWE, SMO, and WD. This water balance 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. Each variable was averaged within each hexel for the years 1984–2012, thereby matching the years of the fire severity data. We similarly summarized these five climate variables representing mid-21st century (2040–2069) conditions using 20 global climate models (GCMs) for the RCP8.5 emissions scenario (table S1). These tables were statistically downscaled to the same grid as observed data using the multivariate adapted constructed analogs approach (Abatzoglou and Brown 2012).

Because the relationship between climate and fire is weaker in landscapes that are highly influenced by humans (Parks *et al* 2014b), we built our model using data from a subset of hexels with low human influence (figure 1(b)). We selected only those hexels that were comprised of at least 50% designated wilderness or national park or had an average 'human footprint' (Leu *et al* 2008) $\leqslant$ 2.5 (on a scale of 1–10). We further limited our dataset to include only those hexels with at least 400 ha of total burned area from 1984 to 2012. These selection criteria resulted in 544 hexels that, despite representing a small proportion of our study area (8.7%), are climatically representative of much of the western US, with the notable exception of the wet regions of the Pacific Northwest (figure S2).

Using data from the subset of 544 hexels, we modeled fire severity (dNBR) as a function of contemporary climate (1984–2012) using boosted regression trees (BRT) ('gbm' package) in the R statistical environment (R Development Core Team 2007). BRT is a nonparametric machine-learning approach that does not require *a priori* model specification or test of hypothesis (De'ath 2007). The BRT algorithm fits the best possible model to the data structure, including complex interactions among variables. It does so by building a large number of regression trees, whereby, through a forward stage-wise model-fitting process, each term represents a small tree built on the weighted residuals of the previous tree. The stage-wise procedure reduces bias, whereas variance is decreased through model averaging. The BRT method also employs 'bagging', the use of a random subset of samples, which typically improves model predictions. Comparisons to other modeling techniques indicate that BRT models consistently produce robust predictive estimates (Elith *et al* 2006). We followed the recommendations of Elith *et al* (2008) for selecting BRT options; we set the bagging fraction to 0.5, learning rate to 0.005, and tree complexity to three. We used a custom script from Elith *et al* (2008) to determine the necessary number of trees, thereby reducing the potential for overfitting. We evaluated the model fit using the (a) correlation between predicted and observed fire severity and (b) ten-fold cross-validated correlation between predicted and observed fire severity.

We used the model to predict contemporary (1984–2012) fire severity (dNBR) for all hexels in the western US. However, interpreting dNBR and changes in dNBR under a changing climate is challenging because dNBR units have no direct ecological interpretation. As such, we rescaled these predictions to correspond to the ecologically relevant composite burn index (hereafter 'inferred CBI') that ranges from 0 to 3 (Key and Benson 2006): the lowest predicted severity was given an inferred CBI of 0.1, which is the threshold for 'unchanged' (Miller and Thode 2007), and the highest predicted severity was given an inferred CBI of 3.0. We were then able to infer the CBI of all remaining predictions because the square-root transformation of dNBR linearized the relationship to CBI (figure S1). Consequently, we generated a map representing the inferred CBI for the western US under contemporary climate.

We then predicted fire severity for the mid-21st century (2040–2069) as projected by each GCM using the BRT model. We inferred CBI as previously described using the linear relationship between dNBR and CBI of the observed predictions to make the inferences. Note that the predictions for all hexels in the western US were 'clamped' to avoid predicting outside of the observed range of severity values; all predictions >3 and <0.1 were given values of 3.0 and 0.1, respectively. For each BRT prediction (one for each GCM), we then quantified the predicted change in fire severity by subtracting the inferred CBI of contemporary climate from the inferred CBI of mid-21st century climate. We summarized the results by generating maps of (1) contemporary fire severity, (2) predicted mid-21st century fire severity (averaged over 20 GCMs) and, (3) the average change (for all 20 GCMs) in fire severity (i.e., inferred CBI) between contemporary and mid-century time periods.

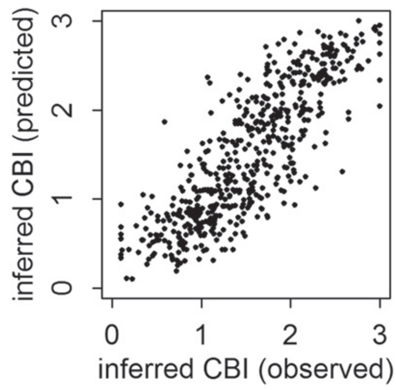

Figure 2. Plot showing predicted versus observed inferred CBI. Inferred CBI of observed data was calculated using similar methods to that of the inferred CBI of predicted data but were bounded by the 0.5 and 99.5 percentile dNBR to ensure extreme values did not overly influence this interpretation. $R^2 = 0.64$.

Results

The correlation between predicted and observed dNBR among the 544 hexels was 0.80 and the cross-validated correlation was 0.72. A plot showing predicted versus observed inferred CBI also indicates a good fit ($R^2 = 0.64$; figure 2). Water deficit and PPT were the most influential variables (relative influence = 41.5% and 29.8%, respectively) (figure 3(a)). Fire severity generally decreased with WD and increased with PPT (figures 3(b) and (c)). The map of predicted contemporary (1984–2012) fire severity indicates that cooler and wetter forested ecoregions (e.g., Pacific Northwest, Northern Rocky Mountains, and Southern Rocky Mountains) experience more high severity fire (inferred CBI $\geqslant 2.25$) compared to warmer and drier forested ecoregions (e.g., Arizona - New Mexico Mountains) (figure 4(a)). Non-forested ecoregions for the most part experience fairly low fire severity (inferred CBI < 1.25). The map of mid-21st century fire severity shows a similar pattern in that the cooler and/or wetter regions generally have higher severity than elsewhere (figure 4(b)), but for the most part, fire severity is predicted to decrease over much of the western US (figure 4(c)). The results of current, future, and predicted changes in fire severity are strikingly similar when we measured fire severity using a relativized metric (the relativized burn ratio; RBR) (Parks *et al* 2014a) instead of dNBR (figure S3).

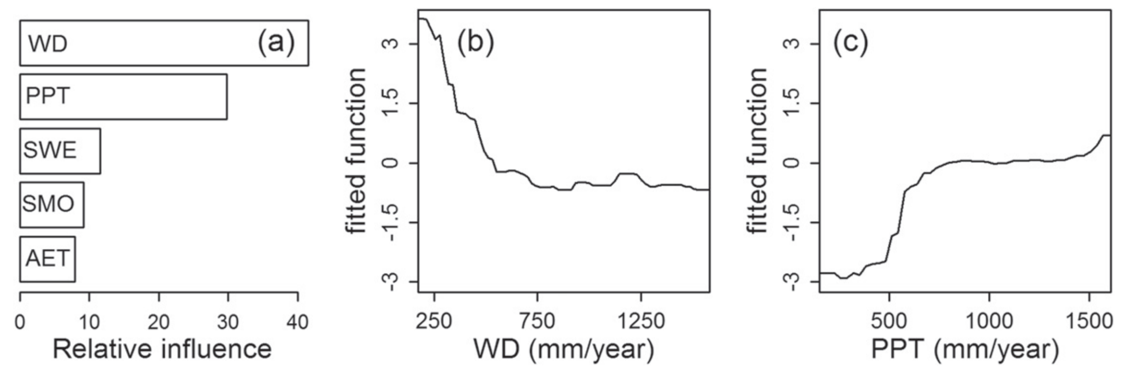

Figure 3. Variable importance in the BRT model (a) and partial dependence plots showing the relationship between dNBR and the two most influential variables (WD and PPT) (b), (c). Note that the partial dependence plots do not reflect interactions between variables and therefore simplify the relationships.

Discussion

Our models based on contemporary fire–climate relationships predict a widespread reduction in fire severity for large portions of the western US by the mid-21st century. Only a very small proportion of the western US is predicted to experience an increase in severity. Our prediction contrasts with those based on the direct influence of climate on fuel moisture and associated fire danger indices that occur at seasonal time scales (Fried *et al* 2004, Nitschke and Innes 2008). Our use of broad-scale climate as a proxy for vegetation composition and fuel load instead emphasizes the indirect influence that climate has on fire regimes (Miller and Urban 1999, Higuera *et al* 2014). Specifically, the predicted decrease in fire severity can be attributed to climatic conditions associated with higher WDs (figures 5(a) and (b)), lower productivity, and less burnable biomass (Zhao and Running 2010, Stegen *et al* 2011).

Our approach and findings are based on an implicit assumption that vegetation composition and fuel load will track changes in climate. Indeed, this is a common assumption that underlies numerous climate change studies, including those that use distribution models to project shifts in habitat ranges (Engler *et al* 2011) and fire activity (Krawchuk *et al* 2009, Moritz *et al* 2012). Specifically, our predictions of overall lower fire severity implicitly assume that vegetation composition and burnable biomass will reflect lower productivity associated with warmer and drier climates (e.g., increased WD; figure 5(b)). As such, our predictions are best interpreted as a *potential* reduction in fire severity, a potential that may not be realized where there is disequilibrium between climate and vegetation. Disequilibrium dynamics are the result of many factors and signals that directional changes in climate may not result in immediate changes in vegetation composition and fuel load (Sprugel 1991, Svenning and Sandel 2013). For example, leading-edge disequilibrium can arise when species are dispersal limited or don't reach reproductive maturity for many years (Svenning and Sandel 2013). Trailing-edge disequilibrium can arise because some species are long-lived and have deep roots, thereby facilitating survival and persistence under substantial inter-annual and decadal fluctuations in climate even though seedlings of the same species are unable to survive (Grubb 1977, Jackson *et al* 2009). To compound this, human-induced disequilibrium has also substantially affected most ecosystems in the western US (and globally) (Parks *et al* 2015b), in that natural disturbances such as fire have been excluded by factors such as livestock grazing, fire suppression, and landscape fragmentation (Marlon *et al* 2008). Both climate- and human-induced disequilibrium underlie present-day concerns about restoration of fire-adapted ecosystems after a century of fire exclusion (Stephens *et al* 2013, Hessburg *et al* 2015).

Consequently, our predictions are more likely to hold up in the presence of an active disturbance regime that catalyzes climatically driven changes in vegetation composition and fuel load (Flannigan *et al* 2000, Turner 2010). Disturbance catalysts are critical components for maintaining a dynamic equilibrium between vegetation and climate and appear to already be occurring with increasing frequency in some regions. For example, many studies have concluded that fire activity has increased in recent years (Westerling *et al* 2006, Kelly *et al* 2013) and widespread tree mortality has been attributed to drought and insect outbreaks (Allen *et al* 2010, Bentz *et al* 2010). In areas recently affected by these disturbances, the post-fire species and vegetation densities may be more tailored to the emerging climate (Overpeck *et al* 1990, Millar *et al* 2007). Although generally considered undesirable, disturbance-facilitated conversions from forest to non-forest vegetation are likely to occur in some situations (Stephens *et al* 2013, Coop *et al* in press), especially when compounded by human-induced disequilibrium.

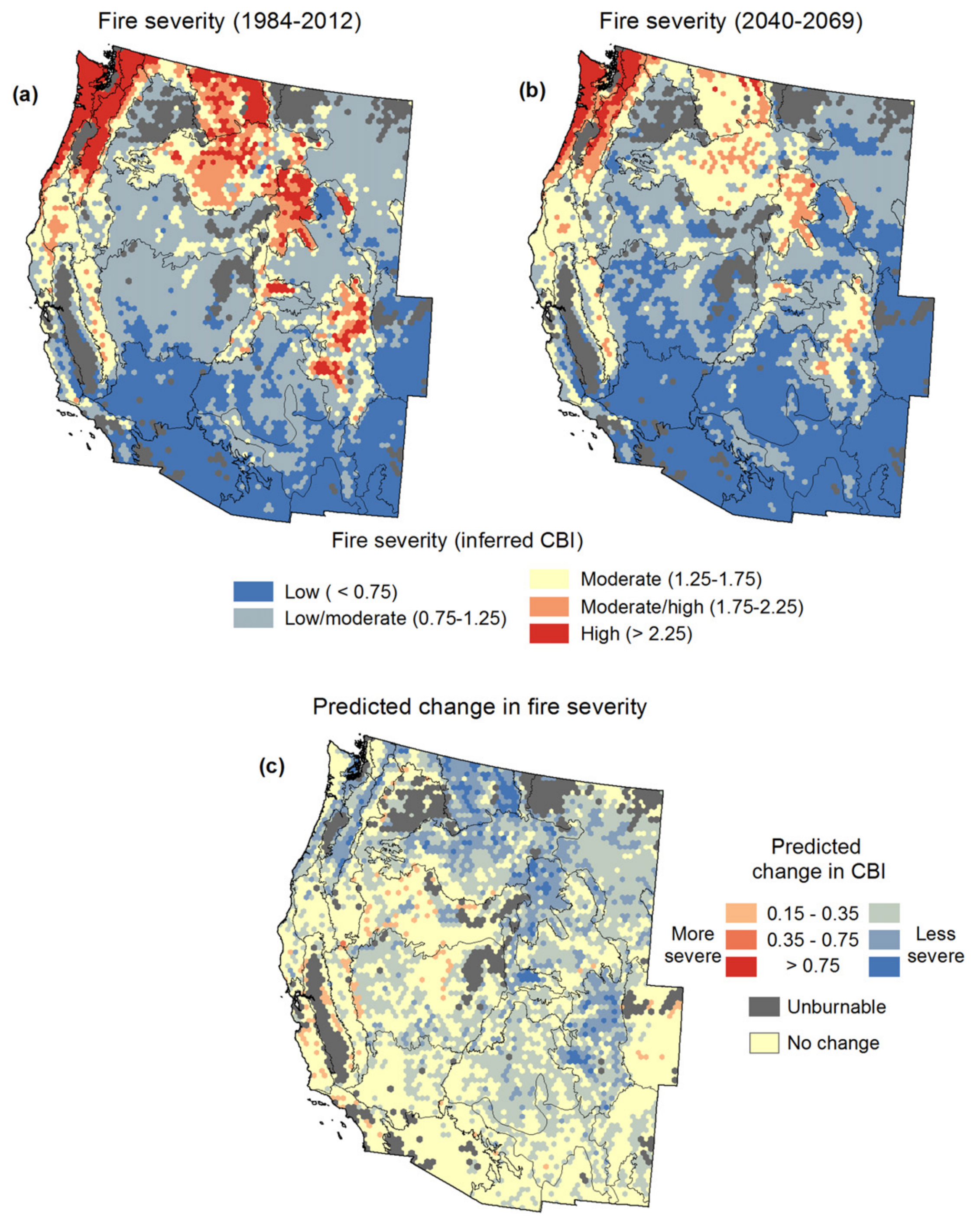

Figure 4. Predicted fire severity under observed (a) and mid-century climate (b). Mean change in fire severity among the 20 predictions (one prediction for each GCM) (c).

Most forested regions in the western US are currently experiencing a 'fire deficit' (Marlon *et al* 2012, Parks *et al* 2015b) because human activities and infrastructure (e.g., fire suppression and roads) exclude fire as an important disturbance agent. Consequently, human-induced disequilibrium between vegetation and climate, coupled with a changing climate, has important implications for future fire severity. We posit that such amplified disequilibrium will likely result in *increased* fire severity in future decades as fuel loads increase, fire seasons lengthen, and fire danger becomes more extreme (Collins 2014, Jolly *et al* 2015). This supposition is consistent with the findings of other studies that found a climate-induced increase in fire severity when assuming static vegetation (Fried *et al* 2004, Nitschke and Innes 2008). Continuing to resist catalysts of vegetation change only increases the probability of undesirable effects given that fire is inevitable (North *et al* 2009, Calkin *et al* 2015). An alternative to this unsustainable cycle is to actively facilitate transition of ecosystems to conditions that are more suited to the future climate by means of managed wildland fire or other restoration treatments (Millar *et al* 2007).

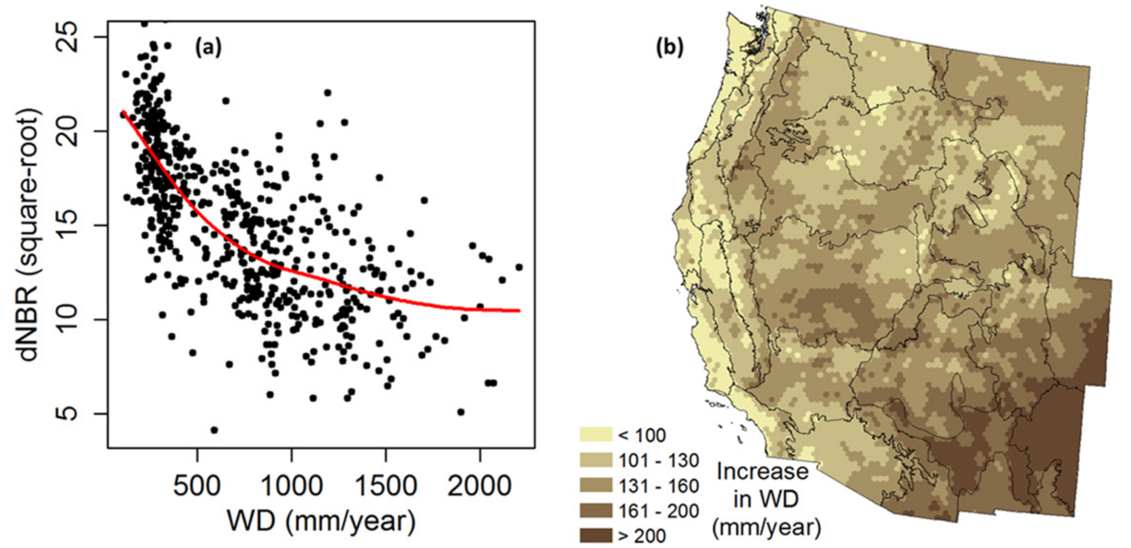

Figure 5. Plot of observed fire severity as a function of observed (1984–2012) water deficit (WD; the most influential variable in the BRT model) for the 544 hexels used to build the model (a). The red line shows the model fit according to a generalized additive model. Map of predicted increase in WD from contemporary to mid-century (2040–2069) climatic conditions (b); values depict the multi-model average change between time periods. According to this simple relationship, increased WD due to climate change will result in decreased fire severity. Note that the relationship flattens out which suggests a weaker response in dry ecosystems.

Our study complements and expands our understanding of controls on fire regimes and how they may respond to a changing climate in the western US. Specifically, predicted increases in fire activity (Littell *et al* 2010, Moritz *et al* 2012) imply that less biomass will be able to accumulate between successive fires, resulting in less biomass available for combustion and a reduction in fire severity. Furthermore, predicted increases in WD (figure 5(b)) are expected to increase water stress and decrease productivity in the generally water-limited western US (Chen *et al* 2010, Williams *et al* 2013), ultimately reducing the amount of biomass available to burn and resultant fire severity. It should be noted, however, that temperature-limited ecosystems (i.e., alpine environments) will likely experience an increase in productivity (and fire severity) under a warmer climate (Grimm *et al* 2013, Goulden and Bales 2014).

Our study relied on observed and predicted climatic normals (i.e., multi-decadal averages) to predict potential changes in fire severity. This is in contrast to other climate change fire studies that used annually or seasonally resolved climate (observed and GCM projections) and fire data to make predictions of potential changes in *fire activity* (i.e., fire frequency or area burned) (Littell *et al* 2010, Stavros *et al* 2014). The latter approach is often used because of the noted importance of climatic extremes on fire regimes (e.g., Westerling *et al* 2006). Although we could have built our model of fire severity using annually resolved data, we posit, for the purpose of predicting future fire severity, using long term averages (e.g., 1984–2012) is more appropriate for at least three reasons. First, although several studies have shown that fire severity responds to annual, seasonal, or daily variability in climate or weather, the relative influence of this variability can be fairly weak (Dillon *et al* 2011, Birch *et al* 2015). This is in contrast to broad temporal scales where the relationship between fire severity and climate has been found to be much stronger (Parks *et al* 2014b, Kane *et al* 2015). Second, because models built at a fine temporal resolution are more focused on the direct influence of climatic variability on fire weather and fuel moisture, they generally fail to incorporate climate- or fire-induced changes in vegetation composition or fuel load (Allen *et al* 2010, Parks *et al* 2015a). We suggest that predictions based on climatic normals implicitly incorporate such changes (Kelly and Goulden 2008, Marlon *et al* 2009). Lastly, GCMs may not adequately simulate annual climatic variability and thus are better suited for predicting long term trends (Stoner *et al* 2009).

Our model used broad scale data and the predictions of widespread reduced fire severity under future climate should be interpreted accordingly. For example, fire severity and climate vary at scales finer than the spatial resolution of the hexel used in this study (Schoennagel *et al* 2004). As such, our analysis does not likely capture finer-scale changes in fire severity that could occur. For example, in alpine environments where localized upward shifts in treeline under a warmer climate are expected to contribute to increases in biomass (Higuera *et al* 2014), fire severity might be expected to increase. Although our model of fire severity (dNBR) as a function of climate performed reasonably well (see section Results), we acknowledge that further error may be introduced due to error in the relationship between CBI and dNBR. However, we posit that the improved ecological interpretation attained by converting dNBR to CBI outweighs any increased error in our predictions.

Our measure of fire severity relied on dNBR (a unitless ratio) and CBI (a composite rating) and, consequently, there is no definable unit of measurement (e.g., grams of carbon consumed m^{-2}). Instead we infer changes in CBI, which integrates several strata (e.g., soil and shrubs) and scales severity from 0 to 3. This is admittedly a somewhat vague framework for assessing potential changes in fire severity, but takes advantage of the widespread availability of satellite-inferred metrics of fire severity and their documented correlation to the CBI. We suggest future research efforts involving fire severity and climate change aim to use more definitive and quantitative units of measurement. On a similar note, fire severity has ecological significance beyond what can be inferred from dNBR and is the result of many complex physical, biological, and ecological factors (Morgan *et al* 2014). For example, in ecosystems that are ill-adapted to fire (e.g., the Mojave Desert), dNBR values may be irrelevant, as any and all fires might be considered 'severe' (Brooks and Matchett 2006). Accordingly, although we used dNBR and CBI as a convenient and standardized way to assess fire severity, predictions for some ecoregions should be carefully interpreted.

Our model does not consider plant physiological responses to a CO_2 enriched atmosphere (e.g., improved water use efficiency and plant productivity) that could lead to increases in fire severity (Drake *et al* 1997, Keenan *et al* 2013). Given that today's atmospheric CO_2 concentration is the highest it's been for at least 650 000 years (Siegenthaler *et al* 2005), this could be a particularly important consideration for extreme water limited ecosystems such as grasslands, where woody plant encroachment could cause changes in biomass amount and structure (Morgan *et al* 2007, Norby and Zak 2011). Consequently, other research approaches using tools such as dynamic global vegetation models may predict different outcomes (Thonicke *et al* 2001).

Although we relied on data from protected areas and other areas of low human influence and thus underrepresented certain climatic environments (see Batllori *et al* 2014), these data represent a surprisingly broad range of ecosystem types in the western US ranging from warm desert (Death Valley National Park (NP) to dry conifer forest (Gila Wilderness) to cold forest (Yellowstone NP) (figure S2). As such, we suggest that under-represented climates have only a marginal effect on our results (see figure S2). Indeed, our analysis (figure S2) indicates that the data we used to build the model adequately represents the climates of most of the western US with the most notable exception being those in the Pacific Northwest where fires were historically and are currently infrequent (Agee 1993).

Conclusions

Our study predicts an overall decrease in fire severity for much of the western US by mid-century (2040–2069) due to changing climatic conditions. These predictions are best interpreted as *potential* decreases in severity that may not be realized unless vegetation composition and fuel load change in parallel with climate. Disequilibrium between plant communities and climate will only escalate, particularly in forested areas, unless natural disturbances and management activities (i.e., prescribed fire and restoration treatments) act as catalysts of vegetation change and push plant communities towards a state of equilibrium with climate. A high degree of disequilibrium between plant communities and climate is generally considered undesirable because the result may be an uncharacteristically severe wildland fire that causes abrupt ecosystem state shifts from, for example, forest to non-forest vegetation (e.g., Coop *et al* 2016).

Our findings support a passive management approach to ecosystem restoration (Arno *et al* 2000), whereby natural disturbance regimes are used to facilitate the transition of plant communities towards a state of equilibrium with the emerging climate. Active restoration treatments may also aid in facilitating these changes in certain situations (Millar *et al* 2007, Stephens *et al* 2010), but the current pace and scale of such treatments is insufficient to make a meaningful impact across the vast forested regions of the western US (North *et al* 2012). In addition, legal (e.g., designated wilderness) and logistical constraints (e.g., steep slopes) make certain activities (mechanical thinning) infeasible across a large proportion of land in the western US (North *et al* 2014). Achieving landscape resilience in a changing climate will likely require increased use of managed wildland fire, especially when weather conditions are not extreme (North *et al* 2015), and in fact, resisting change via activities such as aggressive fire suppression may be counterproductive in the long-run (Calkin *et al* 2015). As such, the results of this study provide insights to policy makers and land managers in the western US as to the pros and cons of resisting or facilitating change in vegetation composition and fuel load in the context of a changing climate.

Acknowledgments

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