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# The missing fire: quantifying human exclusion of wildfire in Pacific Northwest forests, USA

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**Abstract.** Western U.S. wildfire area burned has increased dramatically over the last half-century. How contemporary extent and severity of wildfires compare to the pre-settlement patterns to which ecosystems are adapted is debated. We compared large wildfires in Pacific Northwest forests from 1984 to 2015 to modeled historic fire regimes. Despite late twentieth-century increases in area burned, we show that Pacific Northwest forests have experienced an order of magnitude less fire over 32 yr than expected under historic fire regimes. Within fires that have burned, severity distributions are disconnected from historical references. From 1984 to 2015, 1.6 M ha burned; this is 13.3–18.9 M ha less than expected. Deficits were greatest in dry forest ecosystems adapted to frequent, low-severity fire, where 7.2–10.3 M ha of low-severity fire was missing, compared to a 0.2–1.1 M ha deficit of high-severity fire. When these dry forests do burn, we observed that 36% burned with high-severity compared to 6–9% historically. We found smaller fire deficits, 0.3–0.6 M ha, within forest ecosystems adapted to infrequent, high-severity fire. However, we also acknowledge inherent limitations in evaluating contemporary fire regimes in ecosystems which historically burned infrequently and for which fires were highly episodic. The magnitude of contemporary fire deficits and disconnect in burn severity compared to historic fire regimes have important implications for climate change adaptation. Within forests characterized by low- and mixed-severity historic fire regimes, simply increasing wildfire extent while maintaining current trends in burn severity threatens ecosystem resilience and will potentially drive undesirable ecosystem transformations. Restoring natural fire regimes requires management that facilitates much more low- and moderate-severity fire.

**Key words:** forest management; historical range of variability; Pacific Northwest; wildfire.

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## Introduction

Fire is a ubiquitous, driving force in ecosystems across the globe with tremendous ecological, social, and economic impacts (Bowman et al. 2009). Historically, fires played a critical role in sustaining resilient landscapes and were particularly important in maintaining characteristic structures and compositions of many forested ecosystems across western North America (Falk

et al. 2011). More recently, the western United States has experienced a dramatic increase in area burned by wildfire compared to mid-twentieth century (Littell et al. 2009). This increase has been attributed to longer and drier fire seasons, driven in part by anthropogenic climate change (Dennison et al. 2014, Abatzoglou and Williams 2016). By contrast, a significant post-European settlement fire deficit or debt (Lutz et al. 2009) has also been observed for western U.S. forests in millennial-scale reconstructions of climate–fire relationships (Marlon et al. 2012, Parks et al. 2015, Reilly et al. 2017). These deficits are largely attributed to twentieth-century management practices, including wildfire suppression and extensive grazing and logging (Hessburg and Agee 2003).

Patterns of fire activity occurring over centuries to millennia characterize the fire regime for an ecosystem (Sugihara et al. 2006; Table 1). Pre-European settlement historical fire regimes describe baseline reference conditions for sustaining species diversity, resiliency, and ecosystem processes and functions (Keane et al. 2009). The discrepancy between historic wildfire extent and recent trends has led to concern over whether the extent and severity of modern fires are outside of the range of historic conditions to which forest ecosystems are adapted (Mallek et al. 2013). Particularly in dry forests with historical high-frequency, low-severity fire regimes, there is concern that wildfires are now burning more severely, which could increase the rate at which forests permanently transition to non-forest ecosystems (Savage and Mast 2005, Collins and Roller 2013, Tepley et al. 2017, Serra-Diaz et al. 2018). Ecosystem and species reorganization may be more likely in periods of rapid climatic change (Crausbay et al. 2017) and can induce a climate system feedback when conversion from a high-biomass forest to a low-biomass non-forest occurs (Bowman et al. 2013, Hurteau et al. 2016). Uncharacteristic wildfire can also have profound impacts on carbon cycling, species habitat, water quality, and other key ecosystem services (Smith et al. 2011, Adams 2013, Hurteau et al. 2016).

Concern over uncharacteristically severe wildfire is contributing to a focus on fuels' reduction and landscape-scale ecological restoration on western U.S. public forests (Franklin and Johnson 2012, Hessburg et al. 2015, Valliant and Reinhard 2017). Some studies contend that contemporary forest structure and thus fire regimes are not outside the range of historic conditions making fuels' reduction and forest restoration ecologically inappropriate (Baker and Williams 2012, 2018, Odion et al. 2014). Others have challenged the inferences and underlying methodologies of these studies (Fule et al. 2014, Stevens et al. 2016, Levine et al. 2017, Hagmann et al. 2018). Recent advances in consistently quantifying burn severity (Reilly et al. 2017) present an opportunity to compare current fire regimes to the pre-European settlement historical period, providing context to the debate about the role of contemporary wildfire in the western U.S. and the resulting management activities and ecological outcomes.

Building upon recent data and methodological advances, we quantify expected versus observed fire activity for the 20.6 M ha of forestland in the Pacific Northwest (PNW; Fig. 1). We compare the observed extent and severity of all large wildfires in PNW forests over the last three decades

Table 1. Pacific Northwest (PNW) historic Fire Regime Groups following Barrett et al. (2010).

| Hist. Fire Regime Group | Hist. fire freq. (yrs) | PNW extent (ha) | Description |
|---|---|---|---|
| I | 0–35 | 6.4 M | Generally, low-severity fires replacing less than 25% of dominant overstory; can include mixed-severity fires that replace up to 75% of the overstory |
| II | 0–35 | NA | High-severity fires replacing greater than 75% of the dominant overstory vegetation |
| III | 35–200 | 8.0 M | Generally, mixed-severity fires; can also include low-severity fires |
| IV | 35–200 | 0.9 M | High-severity fires |
| V | 200+ | 5.2 M | Generally, replacement-severity fires; can include any severity type in this frequency range |

*Note:* Note that low-, mixed-, and high-severity fires are all present, in varying degrees, in the historic fire regimes for most forest biophysical settings (Tables 3).

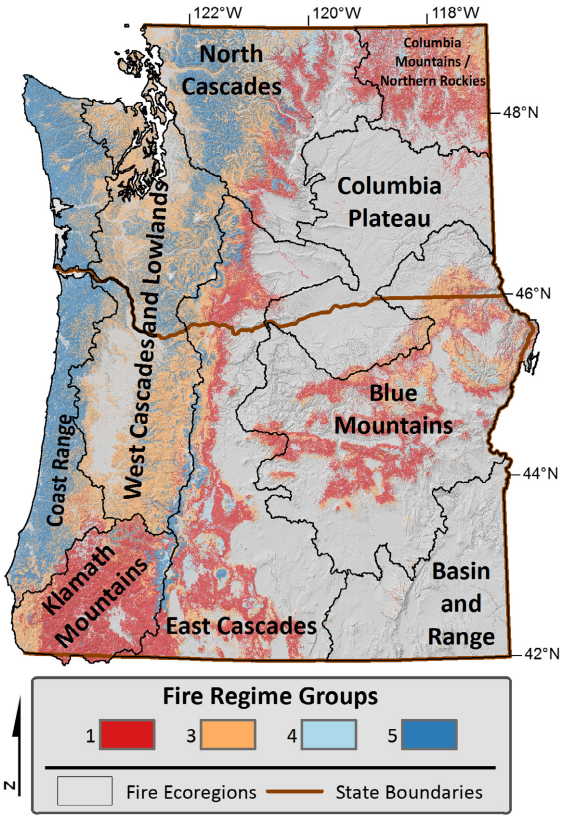


Fig. 1. Pacific Northwest ecoregions and forested historic Fire Regime Groups.

Table 2. Burn severity classes classified by basal area (BA) mortality and associated Relative differenced Normalized Burn Ratio (RdNBR) ranges in Pacific Northwest forests.

| Burn severity class | RdNBR Range | Source |
|---|---|---|
| Excluded from analysis | <1 | Miller and Thode (2007) |
| Unburned | 1–99 | Parks et al. (2014) |
| Low (0–25% BA mortality) | 100–234 | Reilly et al. (2017) |
| Moderate (25–75% BA mortality) | 235–649 | Reilly et al. (2017) |
| High (75–100% BA mortality) | >649 | Reilly et al. (2017) |

(1984–2015) to the expected extent and severity of historical fire regimes for all PNW forest ecosystems. We quantified the extent and severity of contemporary large (>404 ha) fires using the Relative differenced Normalized Burn Ratio metric (RdNBR; Miller and Thode 2007) with consistent, ecologically informed thresholds between fire severity classes (Kolden et al. 2015*b*; Table 2). We developed a regionally consistent and inclusive characterization of historical fire regimes using biophysical setting (BPS) state and transition models (Keane et al. 2007, Rollins 2009), incorporating model updates from the LANDFIRE 2016 Biophysical Settings Review (www.landfirereview.org) and refined simulation methodology from Blankenship et al. (2015). We compare contemporary wildfire and historic fire regimes across PNW forest ecosystems based on (1) the extent of low-, moderate-, and high-severity fire, and (2) the relative proportion of low-, moderate-, and high-severity fire within burned areas. By focusing on both extent and proportion of burn severity classes, we provide a novel quantitative evaluation of whether fires burning under contemporary conditions are within or outside of the historical range. We conceptually build upon and extend previous studies of wildfire extent and severity in Pacific Northwest forests (Reilly et al. 2017, 2018) by using an extensively reviewed and comprehensive set of historical reference conditions covering all forest types and using a broader 32-yr window to represent contemporary wildfire regimes.

## Methods

### *Study area*

We compared contemporary (1984–2015) fire regimes with historic fire regime reference conditions for the 20.6 M ha of forests across Oregon and Washington, USA. The Pacific Northwest is characterized by broad climatic, topographic, and edaphic gradients that result in high ecological complexity and fire regime variability (Agee 1993). Forests within our study area range from Sitka spruce (*Picea sitchensis*) temperate rainforests along the northwest Washington coast with mean annual precipitation >3000 mm per yr, to dry ponderosa pine (*Pinus ponderosa*) forests in southeastern Oregon with mean annual precipitation <400 mm per yr (Franklin and Dyrness 1988). We stratified the study area into nine unique ecoregions (Fig. 1), using our own ecoregion boundaries, which were developed by

setting US Environmental Protection Agency Level 3 Ecoregions (Wiken et al. 2011) to watershed boundaries (10-digit/fifth-level hydrologic unit).

### *Mapping contemporary (1984–2015) burn severity*

We mapped the extent and severity of all large wildfires >404 ha within our study area from 1984 to 2015. We classified the RdNBR data product (Miller and Thode 2007) from the Monitoring Trends in Burn Severity (MTBS, mtbs.gov) program (Eidenshink et al. 2007) into low, moderate, and high burn severity classes based on thresholds derived from a collection of field-based measurements of pre-to-post-fire change in live tree basal area (Meddens et al. 2016, Reilly et al. 2017; Table 2). RdNBR is a remotely sensed measure of post-fire vegetation change using pre- and post-fire scenes from the Landsat satellites. We set RdNBR classification thresholds to correspond with burn severity definitions used by the BPS models, which are based on changes in live tree basal area. This allows for robust comparisons of observation and modeled data and addresses the problem of inconsistent fire severity thresholds within the thematic MTBS classified data product (Kolden et al. 2015*b*). We also removed clouds and cloud shadows utilizing the MTBS cloud mask for each fire. Once classified, the RdNBR raster for each fire was then smoothed with a 3 × 3-pixel neighborhood majority filter to remove sensor spatial errors and finally merged into 32 annual rasters for the entire region (one for each year; Appendix S1: Fig. S1).

### *Simulated historic fire regime reference conditions*

We characterized historic fire regime reference conditions for forests across our study area using biophysical setting (BPS) state-transition models (Table 3). BPS models represent unique potential vegetation units with distinct disturbance regimes based on vegetation, soils, climate, and topography (Pratt et al. 2006, Keane et al. 2007). The models simulate the relative abundance and transitions between vegetative successional states from both deterministic succession and stochastic disturbance processes (Daniel and Frid 2012). The stochastic disturbance processes include low-, moderate-, and high-severity fire as well as other disturbances including insects and disease.

Our BPS models were derived from models developed through the LANDFIRE program (landfire.gov). The LANDFIRE program estimated pre-European settlement rates of succession and disturbance probabilities for each BPS through an intensive literature and expert review process (Keane et al. 2002, 2006, 2007, Pratt et al. 2006, Rollins 2009, DeMeo et al. 2018, LANDFIRE 2018). The BPS models incorporate a range of historic empirical data sources quantifying fire regimes (e.g., pollen and charcoal in sediments, dendrochronological reconstructions, and historical survey records) while providing consistent reference conditions at broad regional spatial scales. The BPS models do not represent a specific year, but instead are designed to capture the variability in ecosystem processes across a range of pre-European settlement climatic conditions. More recent advances in the reconstruction of historical fire history and landscape dynamics for Pacific Northwest forest ecosystems were identified and incorporated into the BPS models through the LANDFIRE 2016 Biophysical Settings Review update process (www.landfirereview.org).

We mapped BPS using the 30 × 30-m pixel Integrated Landscape Assessment Project's Potential Vegetation Type (PVT) dataset (Halofsky et al. 2014), which incorporates updates from subregional vegetation mapping efforts (Simpson 2007, Henderson et al. 2011). The U.S. Forest Service Pacific Northwest Region Ecology Program assigned each PVT mapping unit to a BPS model; this crosswalk was updated from that used in Haugo et al. (2015) and DeMeo et al. (2018; Appendix S1: Table S1).

We simulated the extent and variability of low-, moderate-, and high-severity fire within a 32-yr observation window for each combination of BPS and Ecoregion (*hereafter* BPS + E; Appendix S1: Table S2). We estimated area burned for each severity class for each BPS + E as a range, based on stochastic variation in model runs. Specifically, we captured the mean and the range of variation (5th to 95th percentile) of the simulated occurrence of low-, moderate-, and high-severity fire over a 32-yr window, using ST-Sim version 3.0 (Daniel and Frid 2012). We represented the historical range of variation using the 5th to 95th

Table 3. Biophysical setting (BPS) historic Fire Regime Group and fire return intervals (FRI) by fire severity class, based on the LANDFIRE 2016 Biophysical Settings Review update (www.landfirereview.org).

| BPS name | Hist. Fire Regime Group | Fire severity class | Min return (yr) | Mean return (yr) | Max return (yr) |
|---|---|---|---|---|---|
| Dry ponderosa pine, mesic | I | Low | 2 | 19 | 30 |
| | | Moderate | 50 | 78 | 80 |
| | | High | 100 | 278 | 400 |
| Klamath-Siskiyou lower/upper montane serpentine mixed-conifer woodland | I | Low | 3 | 12 | 35 |
| | | Moderate | 36 | 70 | 100 |
| | | High | 100 | 227 | 400 |
| Mediterranean California dry-mesic mixed-conifer forest and woodland | I | Low | 7 | 12 | 17 |
| | | Moderate | 14 | 32 | 49 |
| | | High | 100 | 333 | 400 |
| Mediterranean California mesic mixed-conifer forest and woodland | I | Low | 10 | 25 | 40 |
| | | Moderate | 15 | 47 | 50 |
| | | High | 170 | 238 | 270 |
| Mediterranean California mixed evergreen forest, interior | I | Low | 5 | 23 | 30 |
| | | Moderate | 15 | 45 | 50 |
| | | High | 100 | 164 | 200 |
| Mediterranean California mixed oak woodland | I | Low | 3 | 12 | 13 |
| | | Moderate | 17 | 34 | 52 |
| | | High | 100 | 294 | 400 |
| Mediterranean California red fir forest | I | Low | 10 | 58 | 90 |
| | | Moderate | 20 | 58 | 200 |
| | | High | 70 | 192 | 500 |
| Northern Rocky Mountain dry-mesic montane mixed-conifer forest | I | Low | 2 | 32 | 35 |
| | | Moderate | 70 | 101 | 175 |
| | | High | 70 | 208 | 400 |
| Oregon white oak/ponderosa pine | I | Low | 5 | 25 | 30 |
| | | Moderate | NA | 900 | NA |
| | | High | 100 | 125 | 300 |
| Pine savannah, ultramafic | I | Low | 10 | 15 | 20 |
| | | Moderate | NA | NA | NA |
| | | High | 100 | 200 | 300 |
| Douglas fir hemlock-dry mesic | III | Low | NA | NA | NA |
| | | Moderate | 50 | 100 | 150 |
| | | High | 250 | 333 | 500 |
| Douglas fir Willamette Valley Foothills | III | Low | 20 | 50 | 80 |
| | | Moderate | 40 | 90 | 150 |
| | | High | 100 | 150 | 400 |
| East Cascades mesic montane mixed-conifer forest and woodland | III | Low | 100 | 270 | 300 |
| | | Moderate | 50 | 128 | 200 |
| | | High | 150 | 244 | 500 |
| Mediterranean California mixed evergreen forest, coastal | III | Low | 150 | 250 | 350 |
| | | Moderate | 45 | 60 | 80 |
| | | High | 150 | 213 | 250 |
| North Pacific dry Douglas fir forest and woodland | III | Low | 40 | 90 | 150 |
| | | Moderate | 40 | 70 | 150 |
| | | High | 100 | 375 | 400 |
| Northern Rocky Mountain ponderosa pine woodland and savanna—xeric | III | Low | 50 | 137 | 150 |
| | | Moderate | 50 | 100 | 200 |
| | | High | 150 | 256 | 450 |
| Northern Rocky Mountain mesic montane mixed-conifer forest | III | Low | NA | NA | NA |
| | | Moderate | 50 | 133 | 150 |
| | | High | 150 | 200 | 500 |

(Table 3. Continued.)

| BPS name | Hist. Fire Regime Group | Fire severity class | Min return (yr) | Mean return (yr) | Max return (yr) |
|---|---|---|---|---|---|
| Pacific silver fir, low elevation | III | Low | NA | NA | NA |
| | | Moderate | 100 | 300 | 400 |
| | | High | 100 | 333 | 800 |
| Subalpine woodland | III | Low | NA | NA | NA |
| | | Moderate | 35 | 80 | 120 |
| | | High | 200 | 300 | 400 |
| Lodgepole Pine, pumice soils | IV | Low | NA | NA | NA |
| | | Moderate | 300 | 450 | 600 |
| | | High | 65 | 125 | 200 |
| Spruce, fir | IV | Low | NA | NA | NA |
| | | Moderate | 285 | 700 | 10000 |
| | | High | 80 | 135 | 270 |
| Subalpine fir | IV | Low | NA | 1250 | NA |
| | | Moderate | NA | 1111 | NA |
| | | High | 150 | 179 | 300 |
| Douglas fir hemlock, wet mesic | V | Low | NA | NA | NA |
| | | Moderate | NA | 1000 | NA |
| | | High | 300 | 400 | 800 |
| North Pacific Mountain Hemlock Forest, wet | V | Low | NA | NA | NA |
| | | Moderate | NA | 5000 | NA |
| | | High | NA | 10000 | NA |
| North Pacific Mountain Hemlock Forest, xeric | V | Low | NA | 1000 | NA |
| | | Moderate | 250 | 588 | 750 |
| | | High | 125 | 303 | 500 |
| Pacific silver fir, high elevation | V | Low | NA | NA | NA |
| | | Moderate | NA | 1100 | NA |
| | | High | 100 | 500 | 800 |
| Sitka spruce, hemlock | V | Low | NA | NA | NA |
| | | Moderate | NA | NA | NA |
| | | High | 300 | 667 | 1000 |

*Notes:* We used the FRI by fire severity class within our simulations of historic fire regimes for each combination of BPS and Ecoregion, following the methodology of Blankenship et al. (2015) to model an expected range in variability in area burned (Fig. 2). Note that cumulative fire return intervals for a BpS (not reported here) are shorter than the fire return intervals for a given fire severity class within the BpS.

percentiles to exclude potential modeling artifacts and because the data to support historical minimum and maximum fire return intervals are less robust than for mean fire return intervals. Post hoc, we evaluated the impact of using the full range of simulated fire values rather than the 5th to 95th percentiles and found no meaningful changes to our results comparing historical references to contemporary fire regimes.

Standard LANDFIRE fire regime parameters represent century-scale historic dynamics (Keane et al. 2009), not a range of variation over 32 yr. To capture a range of variation in fire extent and severity aligned with our contemporary observation conditions, we use modified LANDFIRE model parameters to account for three drivers of variability: (1) the time-period used for model summarization, (2) the number of simulation cells, and (3) variability in fire transition probabilities. Specifically:

1. Shorter time periods result in greater variability in the modeled area burned; therefore, we summarized simulated fire occurrence by severity class for 32 model years to match the 32-yr temporal span (1984–2015) of our contemporary observations.
2. Model cell count, driven by BPS + E extent, also affects model results with smaller cell counts causing greater variability. The state-transition models are non-spatial, so each

simulation cell represents a point sample from a unique ecological unit (Keane et al. 2006). This presented us with two problems: (1) the expected range of variability would differ between BPS + Es of different sizes, and (2) some BPS + Es could be too small to realistically simulate over the 32-yr study period. Therefore, we set the number of independent simulation cells for a BPS + E based on the spatial distribution of each BPS within each ecoregion. To estimate the number of unique ecological units within each BPS + E, we overlaid the BPS + E raster with a 900-ha square grid. We assumed that a BPS + E present in separate 900 ha grid cells would have a degree of ecological independence and thus set the number of simulation cells for each BPS + E as the number of occupied 900 ha grid cells, up to 1000 cells. Based on reconstructions of historic patch size distributions in interior Pacific Northwest forests (Hessburg et al. 2007), the 900 ha grid cell size is larger than the majority of the forest patches created by historic disturbance regimes. Less is known regarding the distribution of forest patch sizes created by historic disturbances in Pacific Northwest coastal and west Cascade forests (Spies et al. 2018). To account for uncertainty in our application of historic forest patch size distributions, we also set a lower threshold for simulation cell count based on a sensitivity analysis of cell count influence on model variation. Across a range of BPS, we found that cell count had relatively low influence on model variation for a 32-yr summarization period when using >100 cells. Consequently, uncommon BPS + Es found in less than 100 grid cells were merged with the next most similar BPS + E within that ecoregion based on ecological similarities and geographic locations to minimize the influence of cell count on model variation (Appendix S1: Table S2).

3. In order to account for uncertainty in fire rotations (Blankenship et al. 2015), we used the methods of Blankenship et al. (2015) to vary fire transition probabilities between Monte Carlo iterations. We automated their heuristic methodology to fit a beta distribution to the range of fire return intervals (FRI) reported by LANDFIRE for each severity class (Fig. 2). ST-Sim uses a transformed version of the beta distribution that takes as inputs: a mean probability multiplier, a standard deviation, and minimum and maximum multipliers. The mean probability multiplier, which adjusts all transition probabilities by the specified factor, was set to 1 in all cases. The minimum and maximum multipliers were used to stretch the distribution between the minimum and maximum FRI reported by LANDFIRE (Blankenship et al. 2015). The standard deviation controls the shape and spread of the distribution. We selected the largest standard deviation possible while meeting two constraints: (1) the maximum of the probability density function be within 10% of the reported mean FRI and (2) the probability density at the minimum and maximum FRI be close to zero. The resulting fire transition probability curves for each fire transition within each BPS model are centered over the LANDFIRE reported mean FRI and with corresponding representative range of variability (Fig. 2).

For each BPS + E, we used 100 Monte Carlo iterations, running each model for 730 yr, with the last 32 yr used for analysis. Model runs were initialized with an equal distribution of simulation cells among the vegetation successional classes and typically stabilized within <400 yr based on area burned by severity class and relative distribution of successional classes per time step.

### *Historical reference versus contemporary fire regime comparisons*

We overlaid our classified burn severity scenes for each year with our mapping of strata to summarize burned area by severity class within each stratum per year. We addressed missing data from scan line errors in RdNBR scenes derived from the Landsat 7 sensor by subtracting the area of scan line errors from total burned area prior to calculating proportions of each burn severity category. We then summarized area burned in each severity class by strata, FRG, and across all forested area in the PNW as defined by the PVT mapping units and BPS reference models.

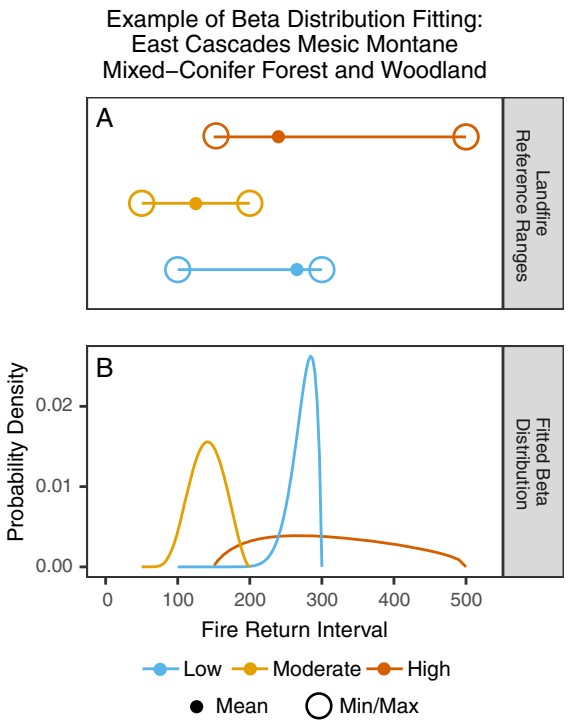


Fig. 2. Example of developing fire probability inputs from LANDFIRE biophysical settings for historic fire regime reference condition state-transition model simulations. In this case, displaying fire return intervals for the East Cascades Mesic Montane Mixed-Conifer Forest and Woodland biophysical setting (A) and the beta-distribution probability densities following the methodology of Blankenship et al. (2015) used as inputs to the state-transition model simulation of historic fire regime reference conditions (B).

We compared observed and reference fire regimes based on both area burned in each severity class and the proportion of each severity class within burned area. We calculated the departure of current fire regimes in terms of both area and proportion as the difference between the observed and the nearest end of the expected range (5th to 95th percentiles), a conservative metric of departure. These comparisons were made for each BPS + E and then summarized across Fire Regime Group, ecoregion, BPS, and Fire Regime Groups within ecoregions. As our datasets provided a census, not a sampling, of all large wildfires for our period observation, we did not assign statistical significance levels to our comparisons of observed versus reference fire regime. In contrast to studies assessing temporal trends over a relatively short time frame (Reilly et al. 2017), our approach used reference conditions that were calculated over the same spatial extents and same time-windows as our observations, comparing both fire rotations (within severity classes) and severity distributions. By modeling expected variation within a 32-yr window, and summarizing observation over the entire 32-yr period, our reference data and observations were temporally aligned, and statistical analysis using highly variable annual data was not needed. The 32-yr period was also consistent with modeling of climatic and fire normals (Arguez and Vose 2011, Lutz et al. 2011), which often use three-decade windows as a baseline for assessment of departure and change. We used the longest possible comprehensive record of contemporary fire extent and severity for Pacific Northwest Forests. We also acknowledge that ideally a longer window of contemporary fire extent and severity would be examined for FRG IV and V forests with naturally longer fire return intervals and greater interannual variability.

FRG IV and V forests are thought to have been characterized by highly episodic and regionally synchronous fire events (Agee 1993, Weisberg and Swanson 2003). Consequently, results for FRG IV and V forests should be interpreted with caution due to the combination of our 32-yr observation window and the inability of our modeling framework to capture such temporal variability and regional-scale synchrony.

## Results

### *An order of magnitude fire deficit over three decades*

Between 1984 and 2015, large wildfires in Pacific Northwest forests burned an area of 1.6 M ha, or 8% of the total forested area. The observed burned area was an order of magnitude less than the 14.9–20.6 M ha expected to burn under historical fire regimes. The Klamath Mountains ecoregion experienced both the greatest overall percentage of total forested area burned in contemporary large wildfires (17%) as well as the greatest fire deficit, with 354,000 ha observed forested area burned compared to an expected burned area range of 4.2–5.0 M ha (Fig. 3; Appendix S1: Table S3).

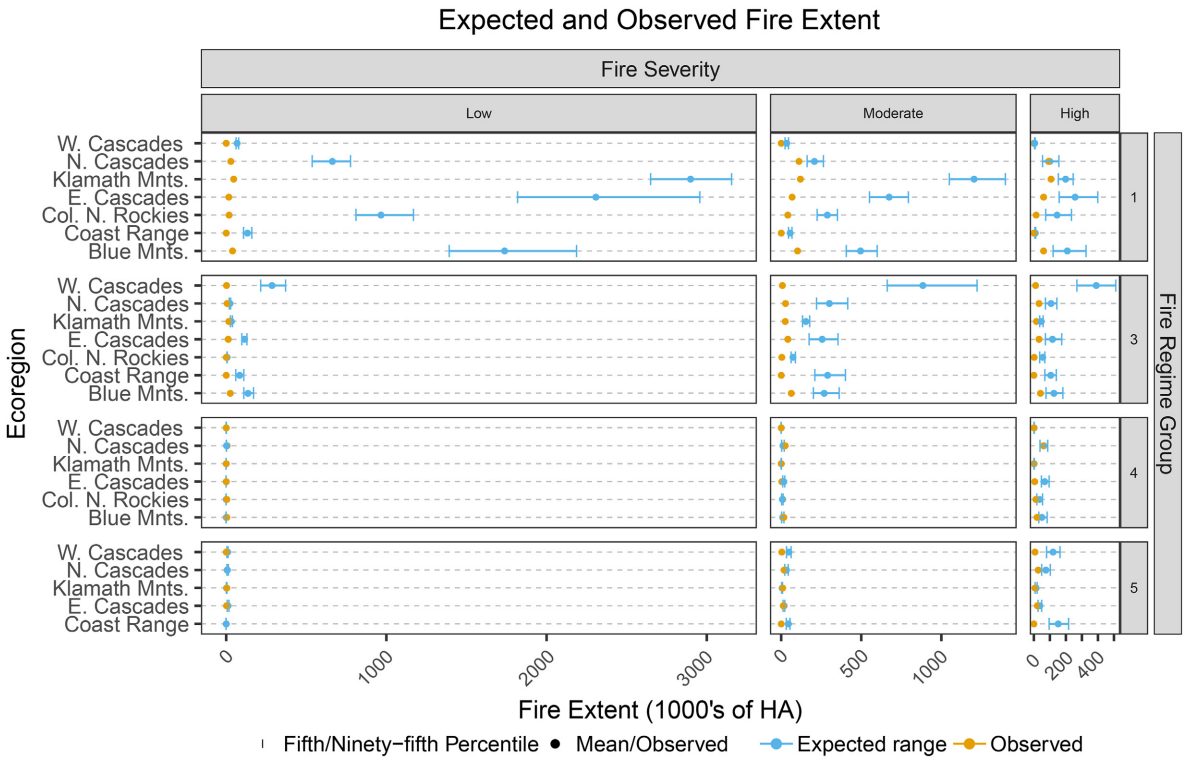


Fig. 3. Fire extent by severity class, plotted as expected (historic fire regime reference; blue bars) versus observed (1984–2015, orange dots) within each historic Fire Regime Group (FRG; Table 1) for Pacific Northwest ecoregions.

### *Observed severity disproportionate to expected severity across fire regimes and ecoregions*

Across the Pacific Northwest, trends were largely driven by the lack of contemporary low-severity fire in Fire Regime Group I (FRG I) forests (Fig. 3, Table 4). The 152,000 ha of low-severity fire observed in FRG I forests represents less than 3% of the area expected to burn at low severity under historic fire regimes (7.4–10.5 M ha; Table 4). Large deficits of low-severity fire were found in all ecoregions with significant areas of FRG I forests, especially the Klamath Mountains, East Cascades, and Blues Mountains ecoregions (Figs. 1, 3). We also found smaller, and in some instances no, deficits of moderate- and high-severity fire in FRG I forests (Table 4, Fig. 3).

When FRG I forests burned, they did so with higher than expected severity across all ecoregions (Fig. 4). High-severity fire represented 36% of the total observed burned area in FRG I forests, compared to a historical range of 6–9% (Table 4). Moderate-severity fire similarly represented a greater proportion of total burned area than expected under historic fire regimes (Fig. 4).

Forests in FRG III, historically characterized by mixed-severity fire regimes, also experienced an overall deficit in fire extent across all severity classes with 0.4 M ha burned compared to 2.8–5.1 M ha expected (Table 4). Deficits in FRG III forests were most pronounced in the West Cascades ecoregion (Fig. 3). FRG III forests experienced an excess of high-severity fire, with high severity representing 36% of total burned area compared to an expected range of 22–25% (Fig. 4, Table 4).

Forests in FRG IV and V (Table 1), historically characterized by episodic high-severity, stand-replacing fires, had substantially less area burned (4.9% of mean total regional expected area burned) and lower overall fire deficits compared to FRG I and III forests (Appendix S1: Table S3). We found a high-severity fire deficit in both FRG

Table 4. Observed and expected (1984–2015) burned area extent by burn severity and forest historic Fire Regime Group (FRG; Table 1) for Pacific Northwest forests.

| Hist. Fire Regime Group | PNW extent, ha | Low severity | | | Moderate severity | | | High severity | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Obs., ha | Expct. 5th %, ha | Expct. 95th %, ha | Obs., ha | Expct. 5th %, ha | Expct. 95th %, ha | Obs., ha | Expct. 5th %, ha | Expct. 95th %, ha |
| I | 6.4 M | 152,000 | 7,379,000 | 10,484,000 | 447,000 | 2,469,000 | 3,525,000 | 337,000 | 573,000 | 1,383,000 |
| III | 8.0 M | 66,000 | 535,000 | 857,000 | 171,000 | 1,669,000 | 3,027,000 | 135,000 | 633,000 | 1,280,000 |
| IV | 0.9 M | 15,000 | 1,000 | 3,000 | 57,000 | 21,000 | 74,000 | 97,000 | 142,000 | 327,000 |
| V | 5.2 M | 13,000 | 20,000 | 47,000 | 45,000 | 106,000 | 195,000 | 63,000 | 260,000 | 554,000 |

*Note:* Expected ranges are the fifth to the ninety-fifth percentile values from biophysical setting simulations.

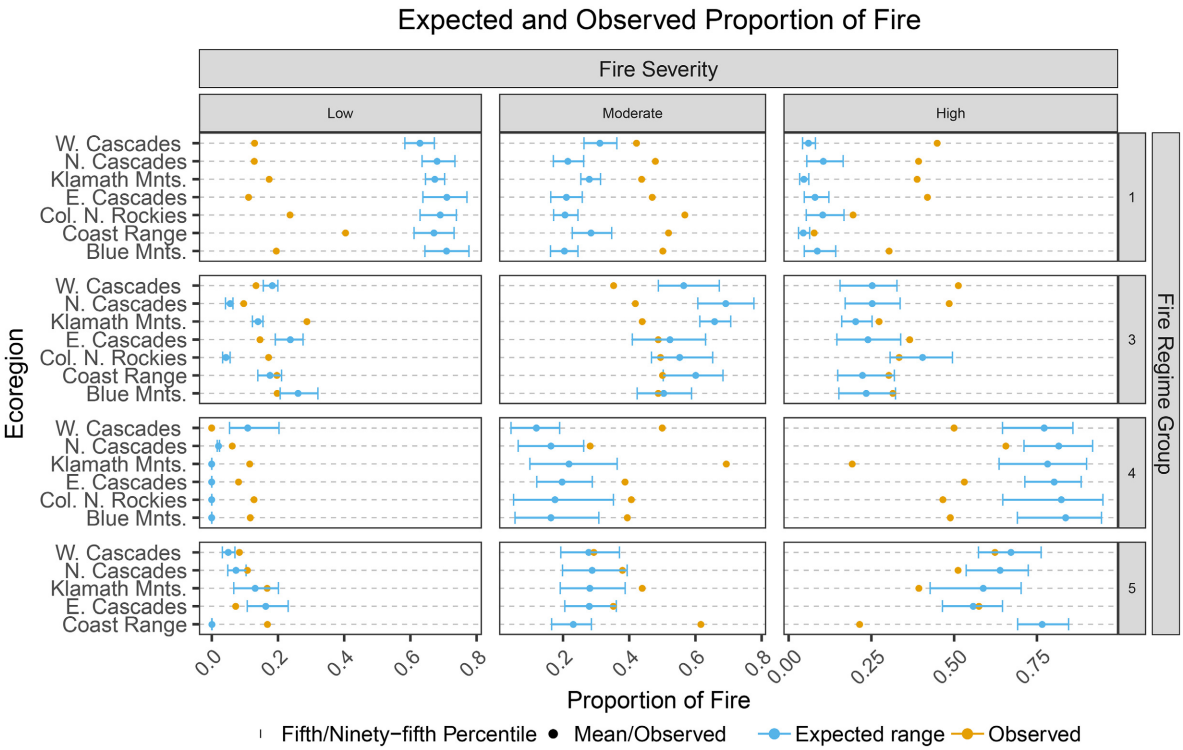


Fig. 4. Relative proportion of total burned area by severity class, plotted for expected (historic fire regime reference; blue bars) versus observed (1984–2015, orange dots) within each historic Fire Regime Group (FRG; Table 1) for Pacific Northwest ecoregions.

IV and V forests; 58% of fire area burned at high severity for FRG IV (compared to an expected 81–88%), and 52% of fire area burned at high severity in FRG V (compared to an expected 67–70%). The high-severity deficit as both total fire extent and as a proportion of burned area was most pronounced among FRG V forests in the Coast Range ecoregion (21% of observed area burned with 69–85% expected; Fig. 4). High-severity fire represented a smaller proportion of the total burned area, and moderate-severity fire a higher proportion, in all ecoregions for FRG IV and most for FRG V (Fig. 4).

## Discussion

Popular perceptions that too much fire has burned in Pacific Northwest forests, particularly

during record wildfire events in 2014 and 2015, are unfounded from an ecological perspective based on burned area alone but supported when stratifying by fire severity. We document that only one-tenth of the area expected to burn in the forests of Washington and Oregon did so over the last three decades. We show that a high percentage of the deficit has occurred where frequent, low-severity fire was expected. In contrast, we also show a small deficit of high-severity fire across all historical fire regimes. Particularly within FRG I forests, we found that contemporary fire severity occurred outside of the ranges of severity to which forest ecosystems are adapted.

The mismatch between modeled historical and recent proportions of fire severity suggests that while recent large fires help address the fire deficit, restoring fire to those ecosystems is more complicated. Different fire severities produce different ecological impacts (Harvey et al. 2014, Stevens-Rumann et al. 2018). Within FRG I forests, we found comparatively small deficits in the overall extent of high-severity fire. However, the higher than expected proportion of high-severity fire in these forests, combined with evidence that high-severity fire favors future high-severity fire in some ecosystems (Lydersen et al. 2017, Prichard et al. 2017), suggests that rather than restoring ecological resilience, these more severe fires may be facilitating transitions to alternative states (i.e., forest to non-forested ecotypes, obligate seeders to resprouters, native to invasive species; Hessburg et al. 2015, Millar and Stephenson 2015). To an extent, fire-mediated transition from forest to persistent non-forest shrublands and grasslands may be reversal of twentieth-century expansion of forest into non-forest areas, driven in part by fire exclusion (Hessburg et al. 2005, Serra-Diaz et al. 2018). However, Pacific Northwest forested landscapes are deficit of late seral forest (DeMeo et al. 2018, Spies et al. 2018) and there is concern that interactions of increasing temperatures, drought, and other stressors with uncharacteristically severe fire will drive large-scale transformations of forested landscapes beyond their natural range of variability (Hessburg et al. 2015, Millar and Stephenson 2015, Serra-Diaz et al. 2018). Such climate- and fire-driven decreases in regeneration of obligate seeders and state transitions of forest to shrubfield have already been documented within other regions in western North America (Kemp et al. 2016, Guiterman et al. 2018). Moving forward, a more complete understanding of the interacting influences of fire and climate change on forested landscapes at regional scales will also require evaluating spatial configuration of high-severity patch interior, distance to seed sources for obligate-seeding species, fire refugia, and forest habitat patches (Cansler and McKenzie 2014, Hessburg et al. 2015, Stevens et al. 2017, Meddens et al. 2018).

Climate change is projected to increase large fire occurrence and area burned (Spracklen et al. 2009, Barbero et al. 2015), elevating the need to understand impacts of fire to ecosystem functions to guide land and resource management decision-making. To that end, there has been an effort to understand both the drivers of fire severity and to determine whether there have been observable trends in severity over the last three decades (Hanson and Odion 2014, Baker 2015, Abatzoglou and Williams 2016, Picotte et al. 2016, Reilly et al. 2017, 2018). The chief shortcomings of these inquiries are that (1) they do not differentiate between fire that is burning within the historical range of variability versus fire that is not, or (2) they use incomplete historic range of variability representations. Analyses that assess temporal trends over three decades for fire regimes where return intervals are often longer than the period of record exclude the influence of natural climate variability (irrespective of global change), which ultimately drives dynamic fire regimes (Marlon et al. 2012). Previous work has also compared contemporary burning to reference conditions and found large area burned deficits in many western North America forest ecosystems (Leenhouts 1998, Marlon et al. 2012, Mallek et al. 2013, Parks et al. 2015, Reilly et al. 2017). The fire deficit we found largely exceeds prior estimates. For example, Parks et al. (2015) estimated a deficit of 4.4 M ha (compared to our 14.9–17.3 M ha) for the ecoregions we assess here but focused on reference conditions from contemporary burning in wilderness and other protected areas.

Our findings demonstrate that humans have not only altered fire frequency and seasonality across the landscape (Balch et al. 2017), but also fundamentally altered fire severity, and in turn, the impacts of fire on ecosystems. The success of U.S. fire suppression in the last century is evident

in the sizeable fire deficit we document for the Pacific Northwest. Further, removal of fire from fire-adapted forests, coupled with other land management practices such as extensive logging and grazing (Hessburg et al. 2015), has both changed the severity of fire that has burned and dramatically altered the composition and structure of many forested landscapes (Haugo et al. 2015). The post-European settlement management footprint is especially evident in the FRG I forests that were historically dominated by low-severity fire but have experienced a disproportionate amount of high-severity fire across the last three decades. In contrast, our results suggest that forest ecosystems historically dominated by high-severity fire experienced a somewhat disproportionate amount of low- and moderate-severity fire. This finding is tempered because our understanding of fire frequency, severity, and variability in FRG IV and V is more limited than in FRG I and III (Spies et al. 2018). For example, Cansler et al. (2018) indicate that there may have been more low and moderate severity historically than previously thought, particularly in high-elevation ecosystems with discontinuous forest cover. It is also unsurprising that the highly episodic and regionally synchronous infrequent large fire events which are thought to characterize FRG IV and V forests (Weisberg and Swanson 2003) were not captured during our 32-yr evaluation window. Instead, our contemporary record largely reflects relatively small fires in FRG IV and V forests burning during moderate conditions. Further, the simultaneous deficit of late seral forests on the landscape (DeMeo et al. 2018, Spies et al. 2018) could mean that if FRG IV and V forests burned with characteristic amounts of high-severity fire, the forest age and structure distribution might further departure from a natural or historic range of variability (Nonaka and Spies 2005).

Additional factors must also be considered to understand the full impacts of contemporary fire in Pacific Northwest forests. We do not address spatial configurations created by fire here, but differential severity also has implications for habitat patchiness and connectivity (Cansler and McKenzie 2014), making it critical to identify where altered patterns of severity eliminate refugia and envelopes of survivability for threatened and endangered species (Kolden et al. 2015*a*). We also do not explicitly evaluate the potential impacts of current and projected future climate change on fire regimes but note that long-term changes in fire regimes may move in different and counterintuitive directions than more immediate changes (McKenzie and Littell 2017, Parks et al. 2018). Nor have we addressed how historic fire regimes may have adapted to climate change in the absence of fire suppression and other twentieth-century land management (e.g., future range of variation; Gartner et al. 2008, Keane et al. 2009). Understanding the range of conditions to which native ecosystems are adapted and present-day departure from those conditions is a necessary component of developing climate adaption strategies. (Stephens et al. 2013, Millar and Stephenson 2015).

Calls for increasing the amount of prescribed fire and wildfire managed for resource objectives used by land managers (Schoennagel et al. 2017) are well-founded based on the magnitude of the fire deficit. Those making these calls must also be cognizant of the proportion of different severities that are appropriate based on evolutionary adaptation and ecosystem function needs for different forest and fire regime types (Falk 2017, Reilly et al. 2018). Restoration of fire is critical to ecosystem function and maintenance of ecosystem services across fire-adapted forests, but fire severity outside of the range of adaptation neither restores nor maintains; rather, it serves to further reduce landscape resilience.

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## Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2702/full