

# Living on the edge: trailing edge forests at risk of fire-facilitated conversion to non-forest

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**Abstract.** Forests are an incredibly important resource across the globe, yet they are threatened by climate change through stressors such as drought, insect outbreaks, and wildfire. Trailing edge forests—those areas expected to experience range contractions under a changing climate—are of particular concern because of the potential for abrupt conversion to non-forest. However, due to plant-climate disequilibrium, broad-scale forest die-off and range contraction in trailing edge forests are unlikely to occur over short timeframes (<~25–50 yr) without a disturbance catalyst (e.g., wildfire). This underscores that explicit attention to both climate *and* disturbance is necessary to understand how the distribution of forests will respond to climate change. As such, we first identify the expected location of trailing edge forests in the intermountain western United States by mid-21st century. We then identify those trailing edge forests that have a high probability of stand-replacing fire and consider such sites to have an elevated risk of fire-facilitated transition to non-forest. Results show that 18% of trailing edge forest and 6.6% of all forest are at elevated risk of fire-facilitated conversion to non-forest in the intermountain western United States by mid-21st century. This estimate, however, assumes that fire burns under average weather conditions. For a subset of the study area (the southwestern United States), we were able to incorporate expected fire severity under extreme weather conditions. For this spatial subset, we found that 61% of trailing edge forest and 30% of all forest are at elevated risk of fire-facilitated conversion to non-forest under extreme burning conditions. However, due to compounding error in our process that results in unknowable uncertainty, we urge caution in a strict interpretation of these estimates. Nevertheless, our findings suggest the potential for transformed landscapes in the intermountain western United States that will affect ecosystem services such as watershed integrity, wildlife habitat, wood production, and recreation.

**Key words:** climate analog model; climate change; climatic debt; disequilibrium; disturbance; trailing edge forest; type conversion; wildfire; wildland fire.

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## INTRODUCTION

Forests across the globe provide numerous and important ecosystem services such as carbon sequestration, clean water, wood products, and recreation (Costanza et al. 1997, Goodale et al.

2002, Postel and Thompson 2005, Douglass 2016). However, elevated temperatures and related water deficits associated with a warming climate are increasing rates of tree mortality and raising concerns about forest loss (Allen et al. 2010, Anderegg et al. 2013). Although increased

moisture stress can itself kill trees, it often interacts with other disturbances such as insect outbreaks and fire that catalyze large-scale forest loss (Dale et al. 2001, Bentz et al. 2010). For example, loss of forest and conversion to non-forest have recently been documented in response to drought, stand-replacing fire, and their interaction (Breshears et al. 2005, Savage et al. 2013, Coop et al. 2016, Donato et al. 2016). Climate-induced conversions from forest to non-forest thus impact the ecosystem services forests provide and are a major management concern in the western United States because of recent drought and increased fire activity (Jolly et al. 2015, Westerling 2016).

There are several approaches for evaluating the potential for climate-induced conversions to non-forest. For example, species distribution models (SDMs) and bioclimatic envelope models are often used to evaluate range shifts of individual species (Guisan et al. 2007, McKenney et al. 2007, Iversen et al. 2008). Recently, climate analog models have been used in conjunction with gridded climate data to evaluate potential shifts in vegetation distribution (Batllori et al. 2017, Parks et al. 2018a) and potential effects on ecosystem services such as crop yields (Pugh et al. 2016). Although these and conceptually similar models are useful, they assume that the current and future distribution of any given species or vegetation type is in equilibrium with climate (Heikkinen et al. 2006, Araújo and Peterson 2012; i.e., that changes in climate result in immediate change in vegetation). This is an assumption that is strained for long-lived woody plants (Davis et al. 1998, Boulangeat et al. 2012). Consequently, dynamic vegetation models have been used and developed to evaluate range shifts while accounting for factors such as dispersal, biotic interactions, and disturbances (Cramer et al. 2001, Hickler et al. 2012, Jiang et al. 2013). These models, however, have their own caveats since they incorporate assumptions about complex processes that are difficult to parameterize (Fisher et al. 2010, Williams and Abatzoglou 2016).

Plant-climate disequilibrium, also called climatic debt or resilience debt (Bertrand et al. 2016, Johnstone et al. 2016), is a key consideration when evaluating potential shifts from forest to non-forest. Although rapid climate- and

drought-induced forest to non-forest conversions have been documented (Allen and Breshears 1998), the response of vegetation is often lagged, reflecting a period of disequilibrium between climate and vegetation (Svenning and Sandel 2013). For example, disequilibrium can arise when soil moisture conditions become consistently too dry for shallow-rooted seedlings, yet larger diameter trees with deep roots can survive and persist for decades. Some of the least favorable sites for tree seedling survival are located along the warm and dry edge (e.g., low elevation and southern boundary) of a given species' geographic distribution (Grubb 1977, Jackson et al. 2009). Under a warming climate, these drier peripheral regions are considered the trailing edge of a species' geographic range (Hampe and Petit 2005). In trailing edge locations, disturbances can catalyze changes in vegetation to types more adapted to the emerging climate (Svenning and Sandel 2013, Crausbay et al. 2017). This suggests that to better understand how forests will respond to climate change, explicit attention to both climate and disturbance is necessary (cf. Campbell and Shinneman 2017, Stralberg et al. 2018).

We focus our attention on wildland fire as the catalyzing disturbance agent in trailing edge forests in the intermountain western United States, a region that has experienced substantial fire activity and drought in recent decades. The overall objective of our study was to evaluate the potential for fire-facilitated conversion from forest to non-forest by mid-21st century. We first identify areas that are currently climatically suitable for forest but are projected to become climatically unsuitable. We consider these areas to be trailing edge forests (Corlett and Westcott 2013). We then identify those trailing edge forests that also have a high probability of stand-replacing fire and consider these to be at the highest risk of fire-facilitated conversion to non-forest.

## METHODS

### Study area

We focused our study on ecoregions in the intermountain western United States that were at least 25% forested (Bechtold and Patterson 2005; Fig. 1). We clipped two ecoregions (Canadian Rockies and Middle Rockies) at the Washington and Oregon state boundaries because we did not

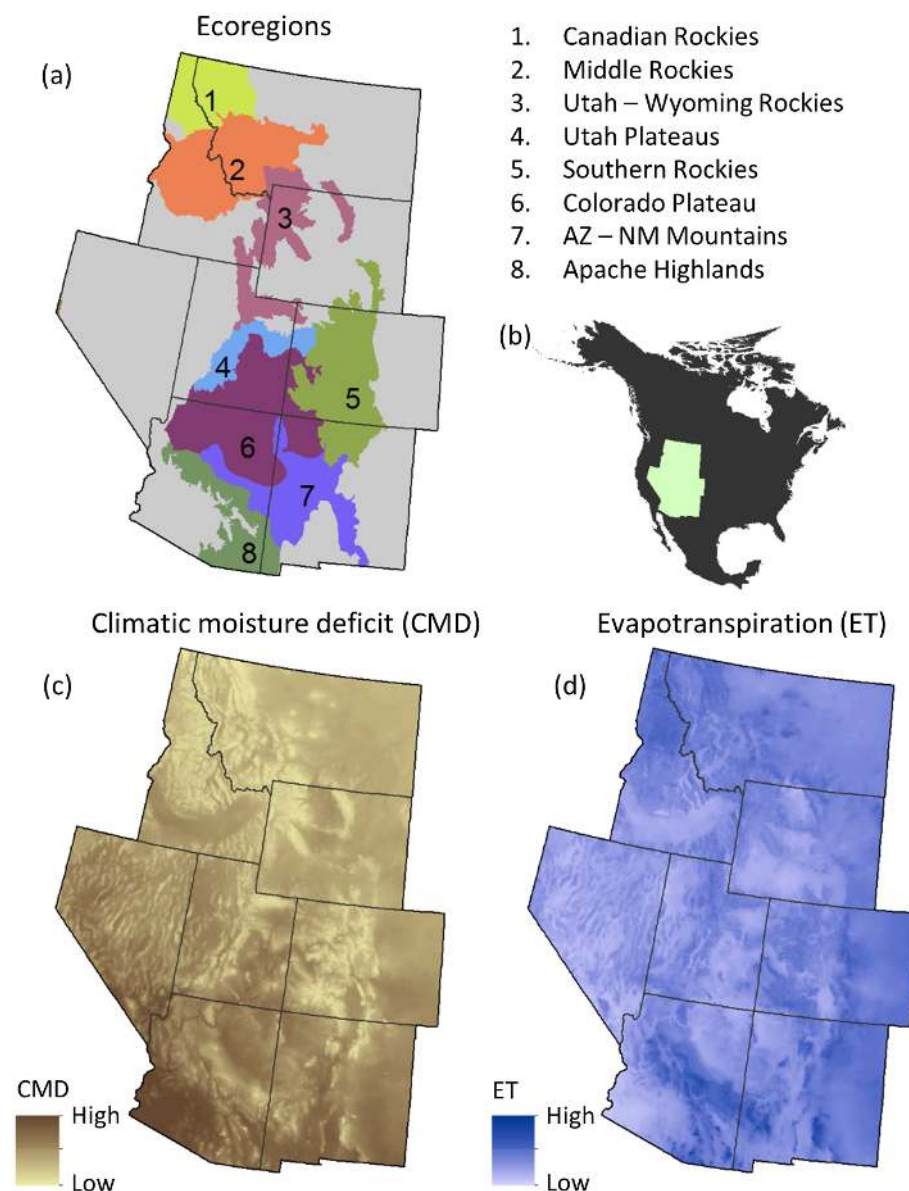


Fig. 1. Maps show the relevant ecoregions (a), location of our study area in relation to North America (b), climatic moisture deficit (c), and evapotranspiration (d).

acquire plot data for these states. The study area spans a range of environments but is primarily semi-arid; mean annual precipitation from 1981 to 2010 across the study area is 54 cm/yr, ranging from <15 cm/yr in the southwest to over 100 cm/yr in the higher elevations and northern reaches of the study area (AdaptWest Project 2015). Mean annual temperature across the study area is 7.4°C, ranging from <2.5°C in the higher

elevations and northern reaches to >15°C in some southern portions of the study area. About half of the study area is forested; the dominant species include pinion pine (*Pinus edulis*), juniper (*Juniperus* spp.), Douglas fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), lodgepole pine (*Pinus contorta*), Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*; Bechtold and Patterson 2005). The other half is

comprised of non-forested land-cover such as alpine, desert, shrubland, grassland, and agriculture.

### Data

We used two gridded climate variables that are strongly related to vegetation and species distributions (Stephenson 1998, Lutz et al. 2010) and are increasingly being used to evaluate the impacts of climate change on such distributions (Ackerly et al. 2015, Parks et al. 2018a). The first variable, climatic moisture deficit (CMD; mm/yr), was obtained from the ClimateNA dataset (resolution = 1 km; AdaptWest Project 2015, Wang et al. 2016). CMD is calculated as ClimateNA reference evaporation (Eref) – ClimateNA precipitation (Wang et al. 2012), assessed at a monthly timestep and summed for the year. Eref from ClimateNA utilizes the Hargreaves approach to estimating reference evapotranspiration (Hargreaves and Samani 1985) which is a temperature-based approach. We developed a second variable to represent an estimate of evapotranspiration (ET; mm/yr; Fig. 1), calculated as Eref – CMD following the logic used to calculate AET as described by Stephenson (1990). CMD and ET are simplifications of the two variables typically used to characterize the water balance (climatic water deficit and actual evapotranspiration, respectively) and are related to temperature and precipitation (their amount and timing). The reference period represents climatic normals (i.e., the average) from 1981 to 2010. We used the 2041–2070 time period to represent future climate (hereafter mid-century). Mid-century climate represents a multi-model ensemble of 15 CMIP5 GCMs in the RCP 8.5 climate forcing (AdaptWest Project 2015).

To identify and characterize forest and non-forest plots in the intermountain western United States, we used U.S. Forest Service Forest Inventory and Analysis (FIA) data (Bechtold and Patterson 2005). The FIA program is a national effort that samples across all forest types and ownerships within the United States at an intensity of approximately one plot per 2430 ha. Plots are visited every ten years, and we acquired data from 2000 to 2015; where a plot was visited more than once in this timeframe (i.e., re-measured), we used data from the most recent visit. An FIA plot is considered forested if the vertically

projected canopy cover of trees  $\geq 10\%$ ; however, recently burned or logged plots are still considered forest even if the canopy cover is  $< 10\%$ . Any plot without an FIA-assigned forest type was classified as non-forest. A total of 33,815 plots overlap with the study ecoregions (Fig. 1a). All FIA plots were attributed with the reference period and mid-century CMD and ET.

Gridded fire severity datasets were obtained from Parks et al. (2018b; available at: <https://www.frames.gov/NextGen-FireSeverity>). These datasets depict the probability of stand-replacing fire (were a fire to occur) under average weather conditions for each 30 m pixel for each ecoregion in the study area (predictions are representative of the year 2016; e.g., Fig. 2). Briefly, these gridded datasets were built using an observed, satellite-derived measure of fire severity (Parks et al. 2014) and statistical models in which the probability of stand-replacing fire was modeled as a function of fuel, topography, climate, and

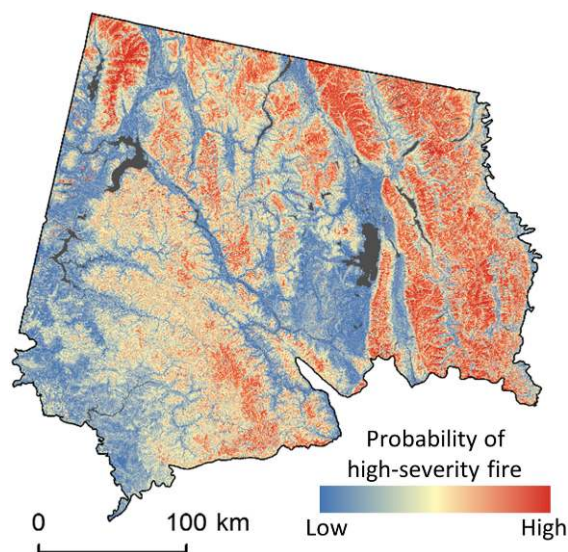


Fig. 2. Probability of stand-replacing fire, were a fire to occur under average weather conditions, for the Canadian Rockies ecoregion (see Fig. 1 to reference ecoregion location). Similar gridded datasets were obtained for all ecoregions from Parks et al. (2018b). These datasets were resampled to a resolution of 1 km and converted to a binary severity classification (stand-replacing and other severity) based on ecoregion-specific thresholds (Fig. 5b; see *Methods*).



weather. For a subset of ecoregions in our study area (Colorado Plateau, AZ–NM Mountains, and Apache Highlands), Parks et al. (2018b) also produced gridded datasets representing the probability of stand-replacing fire under extreme fire weather conditions.

#### *The climate analog model*

Following Parks et al. (2018a), we utilized climate analogs (Hamann et al. 2015, Dobrowski and Parks 2016) to infer the distribution of forest cover for the reference period and mid-21st century (Figs. 3, 4); differences between the two periods provide a means to evaluate potential vegetation shifts. To infer the distribution of forest cover for the reference period, we characterized reference period climate (i.e., CMD and ET) for each 1 km pixel in the study ecoregions. To do this, we identified, for each pixel, the seven nearest (Euclidean distance) FIA plots (including those residing outside of the study ecoregions) that were within  $\pm 1$  mm (after a square-root transformation) for both CMD and ET; we considered these plots to have an analogous (or matching) climate to the pixel of interest. We then used the FIA-assigned forest class (forested vs. non-forested) of the majority of these seven

plots to estimate whether the focal pixel was forested or non-forested. Both the climate bin width (i.e.,  $\pm 1$  mm after square-root transforming) and the number of FIA plots to incorporate ( $n = 7$ ) were chosen based on a validation procedure that minimized classification error (Appendix S1). This approach to characterizing reference period vegetation using climate data is conceptually similar to imputation methods that assign plot-based vegetation and other attributes to pixels that do not have plot data (Ohmann and Gregory 2002, Hudak et al. 2014). The end result was a gridded dataset representing the reference period distribution of forest across the study area (e.g., Fig. 3). To infer the potential distribution and extent of forest under future climate conditions, we used a parallel approach but instead first characterized mid-century climate (i.e., future CMD and ET) for each 1 km pixel. We then identified, for each pixel, the seven nearest FIA plots with an analogous reference period climate (Fig. 4). If the majority of its seven nearest FIA plots were classed as forest, we assumed that the pixel would be climatically suitable for forest under future climate conditions. The end result was a gridded dataset representing the mid-21st-century distribution of forest.

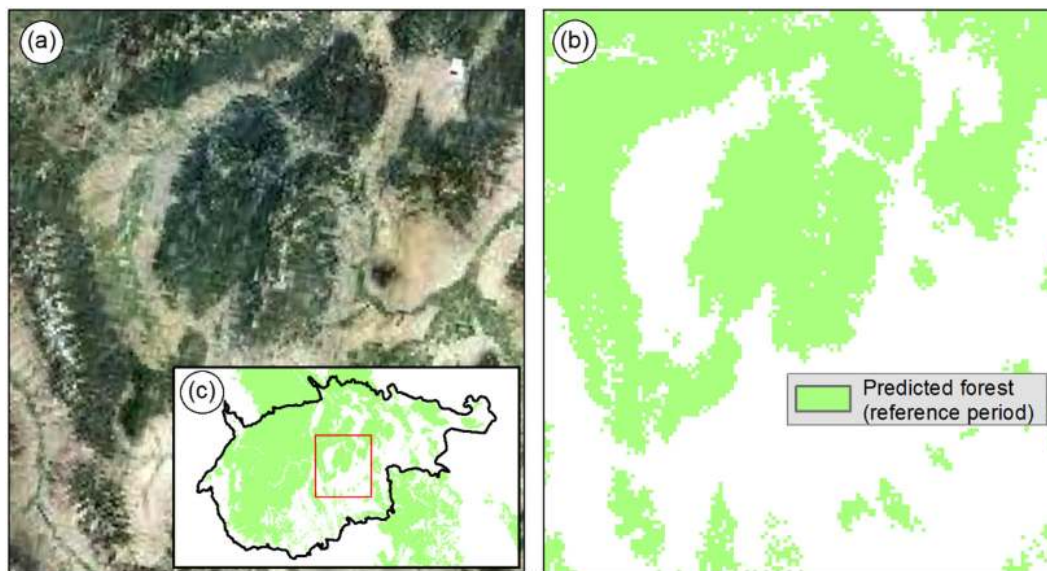


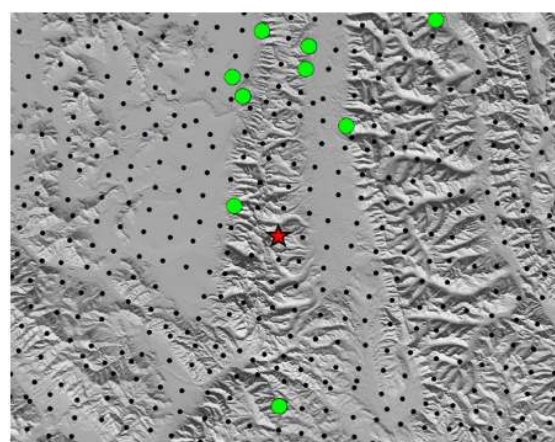
Fig. 3. Aerial imagery shows the distribution of forest for a subset of the Middle Rockies ecoregion (a). Predicted distribution of reference period forest using FIA data and climate analogs (b; see *Methods*). Extent of panels (a) and (b) shown in panel (c). See Fig. 1 to reference location of this ecoregion.

We examined differences between the current and mid-21st-century distribution of forest to evaluate potential climate-induced change in forest extent and distribution. We were particularly interested in those pixels that were mapped as forested in the reference period but were mapped as non-forested (i.e., climatically unsuitable for forest) under mid-century climate; we interpret such pixels to be trailing edge forests.

#### *Incorporating predictions of stand-replacing fire*

We resampled (i.e., averaged) all severity predictions to match the resolution of the climate data and the forest/non-forest predictions (1 km). Each 1 km pixel representing the probability of stand-replacing fire (under average weather conditions) was then classified as either stand-replacing or other severity based on ecoregion-specific severity thresholds. For background, Parks et al. (2018b) used a binary representation of severity (stand-replacing vs. other) as the dependent variable in their models. As such, the prevalence (i.e., the proportion) of observed stand-replacing pixels in Parks et al. (2018b) varied among the input datasets for each ecoregion. Consequently, the severity thresholds we applied to the resampled 1 km resolution predictions are based on the prevalence of stand-replacing pixels in the input datasets used by Parks et al. (2018b). For example, if 35% of the pixels in a given ecoregion were classified as stand-replacing in Parks et al. (2018b), we ensured that the resampled fire severity dataset described here (resolution = 1 km) also had 35% of pixels classified as stand-replacing. For those ecoregions for which we also had datasets representing the probability of stand-replacing fire under conditions representing extreme weather (Colorado Plateau, AZ–NM Mountains, and Apache Highlands), we used the same ecoregion-specific probability threshold to classify as stand-replacing or other severity.

To map areas at risk of fire-facilitated conversions to non-forest, we evaluated spatial coincidence between datasets that satisfied both of the following criteria: (1) pixels identified as trailing edge forest (i.e., climatically suitable for forest during the reference period but climatically unsuitable for forest by mid-century) and (2) pixels with a high probability of stand-replacing fire. We consider areas meeting both criteria to be at elevated



- ★ Focal pixel of interest (CMD=263 and ET=468 mm/yr in mid-century)
- FIA plots that are  $\pm 1$  mm (after square-root transformation) from the focal pixel of interest using reference period climate
- All other FIA plots

Fig. 4. Example of our methods that use climate analogs to evaluate potential forest change in the intermountain western United States. For each pixel within the study area (e.g., red star on map), we characterized the future climate (CMD and ET) as predicted in mid-century and then identified all FIA plots that had analogous climates ( $\pm 1$  mm after square-root transformation) under reference period conditions (1981–2010) (green dots). We compared the forest condition (forest/non-forest) at each focal pixel (see Fig. 3) to that of the seven nearest FIA plots that were identified as climate analogs (e.g., green circles) using a majority rule. The focal pixel (red star) represents reference period forest condition (forest/non-forest; Fig. 3), and the plots identified as climate analogs (green circles) represent potential mid-century forest condition.

risk of fire-facilitated conversion from forest to non-forest. Note that the entire study area was evaluated in terms of the probability of stand-replacing fire under average weather conditions. However, for a subset of ecoregions, we also conducted this evaluation in terms of the probability of stand-replacing fire under extreme fire weather.

## RESULTS

About 490,000 km<sup>2</sup> (51.7%) of the study area is mapped as forested under reference period

conditions (Table 1). However, we identified nearly 176,000 km<sup>2</sup> (35.8%) of this area as trailing edge forest that will become climatically unsuitable for forest by mid-century (Fig 5a). Ecoregional variation is apparent: Whereas <18% of forest in the Canadian Rockies ecoregion is trailing edge forest, ≥44% is trailing edge forest in the Utah-Wyoming Rockies, Colorado Plateau, Arizona–New Mexico Mountains, and Apache Highlands ecoregions (Table 1). Trailing edge forests, on average, are warmer and drier (i.e., higher CMD and lower ET) compared to forests that are expected to remain climatically suitable for forest (i.e., stable forest) by mid-century (Fig. 6).

Within trailing edge forest, ~32,000 km<sup>2</sup> is susceptible to stand-replacing fire under average weather conditions (Table 2, Fig. 5) and therefore meets our criteria for being at elevated risk of fire-facilitated conversion to non-forest. This area amounts to 18.3% of trailing edge forest extent and 6.6% of reference period forest in the intermountain western United States. Again, ecoregional variation is evident, as ≥10% of the reference period forest in the Utah–Wyoming Rockies and Southern Rockies ecoregions is at elevated risk of fire-facilitated conversion to non-forest, whereas <5% is at elevated risk in the Canadian Rockies, Utah High Plateaus, Colorado Plateau, and Arizona–New Mexico Mountains ecoregions (Table 2).

For those ecoregions in which we were able to evaluate the potential for stand-replacing fire

under extreme fire weather conditions (Colorado Plateau, AZ–NM Mountains, and Apache Highlands; Fig. 7), about 45,000 km<sup>2</sup> of trailing edge forest is susceptible to stand-replacing fire under extreme fire weather (Table 3). This represents ~61% of trailing edge forest and ~30% of all reference period forest in these ecoregions.

## DISCUSSION

In the western United States and elsewhere, climate-induced forest range contraction (i.e., conversion to non-forest) in trailing edge locations could result in transformed landscapes in terms of ecosystem services such as watershed integrity, wildlife habitat, aesthetics, and recreation. In the absence of disturbance, however, conversion to non-forest will be slow because of disequilibrium dynamics in which adult trees persist under unfavorable climates due to factors such as deep roots that provide access to water resources (Sprugel 1991, Svenning and Sandel 2013). This can also be considered a storage effect (Warner and Chesson 1985) that serves as a stabilizing process in which mortality does not greatly exceed recruitment over longer timeframes (~10–50 yr; Lloret et al. 2012, Martínez-Vilalta and Lloret 2016). However, disturbance such as fire can alter demographic rates by increasing mortality of adult trees that serve as seed sources, thereby destabilizing the system, particularly under a warming climate in which seedling survival is adversely affected (Stevens-

Table 1. Ecoregional evaluation of reference period forest cover, trailing edge forest, and change in area climatically suitable for forest by mid-century.

Ecoregion name	Ecoregion ID (Fig. 1)	Reference period forest (km <sup>2</sup> )	Trailing edge forest (km <sup>2</sup> )	Reduction in forest area (km <sup>2</sup> )†	Reference period forest (% ecoregion)	Trailing edge forest (% reference period forest)	Reduction in forest area (% reference period forest)†
Canadian Rockies	1	65,142	11,440	8,649	84.2	17.6	13.3
Middle Rockies	2	88,938	22,793	13,446	53.8	25.6	15.1
UT–WY Rockies	3	58,821	25,859	9,050	54.2	44.0	15.4
Utah High Plateaus	4	31,582	7,676	929	68.9	24.3	2.9
Southern Rockies	5	98,218	34,882	14,915	61.0	35.5	15.2
Colorado Plateau	6	59,614	31,755	24,230	30.5	53.3	40.6
AZ–NM Mountains	7	60,520	27,085	23,193	52.6	44.8	38.3
Apache Highlands	8	29,016	14,693	9,463	34.8	50.6	32.6
Total	NA	491,851	176,183	103,875	51.7	35.8	21.1

Note: NA indicates not applicable.

† This represents the total reduction in area climatically suitable for forest by mid-century.



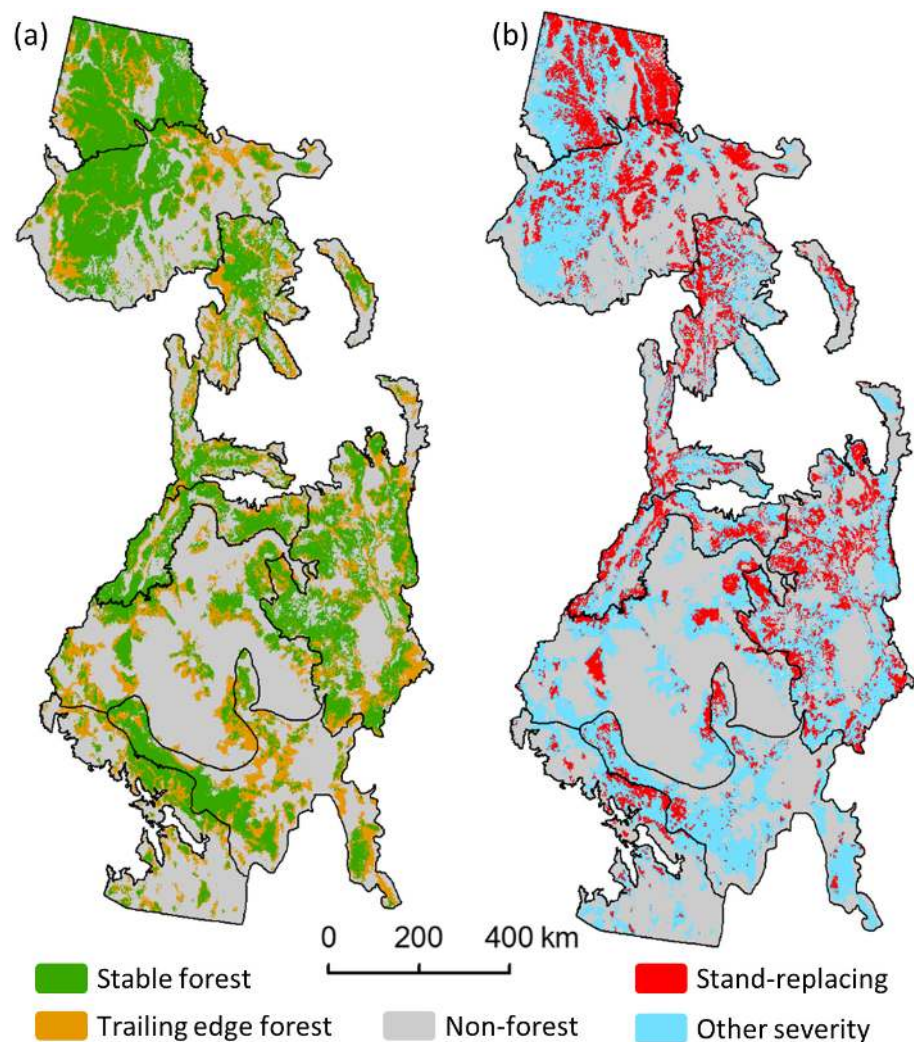


Fig. 5. Stable and trailing edge forest (a). Expected fire severity, were a fire to occur under average weather conditions, across the study domain; (b). Original gridded 30 m datasets (as a continuous variable representing the probability of stand-replacing fire) were resampled to 1 km resolution and converted to a binary representation of severity based on thresholds specific to each ecoregion (see *Methods*).

Rumann et al. 2017, Davis et al. 2019, Kemp et al. 2019). Consequently, disturbance often catalyzes abrupt vegetation change under disequilibrium conditions caused by a changing climate (Turner 2010, Crausbay et al. 2017). Indeed, evidence of fire-facilitated conversions to non-forest is becoming more prevalent (Savage and Mast 2005, Coop et al. 2016, Donato et al. 2016, Walker et al. 2018). Our intersection of estimates of stand-replacing fire with trailing edge forest identified  $\sim 32,000 \text{ km}^2$  (6.6% of currently forested area) in the intermountain western

United States that are susceptible to abrupt conversion to non-forest by mid-century. Although conversion to non-forest may reduce disequilibrium between vegetation and climate, abrupt conversions from forest to non-forest could stress communities dependent on forests and the ecosystem services they provide.

A somewhat unexpected finding was the low amount of forest at elevated risk of fire-facilitated conversion in the southwestern United States (Colorado Plateau, Arizona–New Mexico Mountains, and Apache Highlands) compared to other more



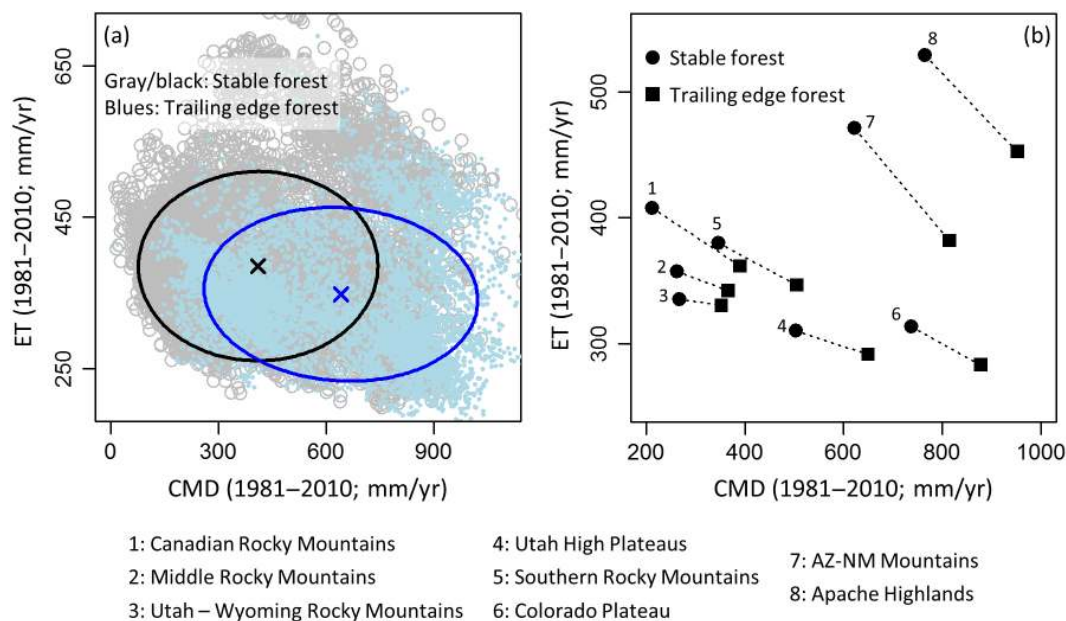


Fig. 6. Reference period climate space of FIA plots considered stable and trailing edge forest (a). The X symbols represent the centroid for each group. Each ellipse encapsulates 2/3 of the data. Ecoregional depiction of the reference period climate space (centroids) of FIA plots considered stable and trailing edge forest (b). In both panels, trailing edge FIA forest plots have higher CMD and lower ET (on average) than their counterparts.

northern ecoregions (specifically the Southern Rockies and Utah–Wyoming Rockies ecoregions; Table 2). This finding is likely because many of the lower treeline forests (i.e., trailing edge) in the southwestern United States are less dense, have limited fuels (Fig. 8), and therefore have a lower probability of stand-replacing fire. In contrast, some of the lower elevation treelines in the Southern Rockies and Utah–Wyoming Rockies ecoregions are well-defined, fairly dense, and have a higher probability of stand-replacing fire (Figs. 5, 8). Consequently, factors controlling these lower treelines (Germaine and McPherson 1999, Sparks and Black 2000) are likely an indirect influence on the expected fire severity and the potential for fire-facilitated transition to non-forest. This said, our evaluation using fire severity predictions under extreme fire weather in the southwestern United States indicated that a large amount (30%) of reference period forest is at risk of fire-facilitated conversion to non-forest. Indeed, fire-driven conversions from forest to non-forest are being increasingly observed in the southwestern United States (O'Connor et al. 2014, Coop et al. 2016, Barton and Poulos 2018, Walker et al. 2018).

Stand-replacing fire is not the only threat to trailing edge forests, as other disturbance agents also have the ability to catalyze shifts to non-forest (Allen and Breshears 1998). For example, severe drought can kill trees (Van Mantgem et al. 2009, Allen et al. 2010) and result in extensive forest die-off (Allen et al. 2015). Insect outbreaks, often in conjunction with moisture stress, can also result in regional forest die-off (Breshears et al. 2005, Adams et al. 2009, Anderegg et al. 2013). Consequently, our estimates of trailing edge forest area that is at elevated risk of abrupt conversion to non-forest could be considered conservative. These additional threats may be pronounced in the southwestern United States, as >44% of forest in the Colorado Plateau, Arizona–New Mexico Mountains, and Apache Highlands ecoregions are considered trailing edge forests. Warming that is likely to occur beyond mid-century only exacerbates these threats.

Our study has limitations which could result in both over- and underestimating risks of fire-facilitated conversions. First, our use of CMD and ET are simplifications of both the water and

Table 2. Area of trailing edge forest affected by each fire severity class (expected fire severity under average weather conditions).

Ecoregion name	Ecoregion ID (Fig. 1)	Other severity (km <sup>2</sup> )	Stand-replacing (km <sup>2</sup> )†	Other severity (% reference period forest)	Stand-replacing (% reference period forest)†
Canadian Rockies	1	10,427	1013	16.0	1.6
Middle Rockies	2	18,175	4618	20.4	5.2
UT–WY Rockies	3	16,997	8862	28.9	15.1
Utah High Plateaus	4	6120	1556	19.4	4.9
Southern Rockies	5	25,047	9835	25.5	10.0
Colorado Plateau	6	29,129	2626	48.9	4.4
AZ–NM Mountains	7	24,858	2227	41.1	3.7
Apache Highlands	8	13,140	1553	45.3	5.4
Total	–	143,893	32,290	29.3	6.6

† These columns should be interpreted as the area and percent of forested area at risk of fire-facilitated conversion to non-forest. They represent pixels that are trailing edge forest and are at risk of stand-replacing fire under average weather conditions.

energy balance. For example, the Hargreaves approach to estimating reference evapotranspiration does not account for wind and relative humidity and has been shown to result in a drying bias under future conditions (Dewes et al. 2017). This would in turn result in overestimating conversion risks of trailing edge forests. In contrast, we may have underestimated the risk of fire-facilitated transitions for several other reasons. First, our study focused on a specific type of transition catalyzed by a single stand-replacing fire. Fires of moderate to high severity that occur at short time intervals (<~15 yr between reburns) have also been shown to shift successional trajectories toward shrub- and grass-dominated systems (Coop et al. 2016, Coppoletta et al. 2016, Harvey et al. 2016, Stevens-Rumann and Morgan 2016, Topley et al. 2017). Second, the estimates for stand-replacing fire we used are representative of 2016 conditions and do not account for expected increases in severity with continued warming and drying (Williams et al. 2012, Abatzoglou et al. 2017). Third, we only considered the severity of fire under “average weather conditions in which fires burn” (Parks et al. 2018b) across the study domain. In the subset of ecoregions for which we evaluated fire severity under extreme fire weather, we show a large increase in the area at risk of conversion to non-forest. Lastly, there is a small directional bias in our classification of forest and non-forest (Appendix S1: Table S2), potentially resulting in

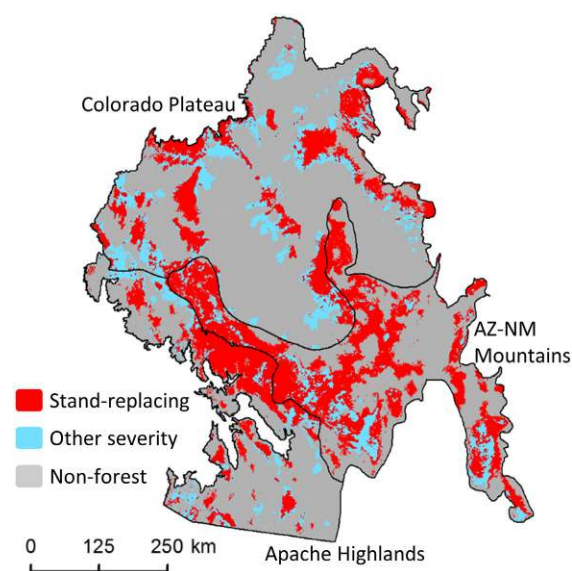


Fig. 7. Expected fire severity, were a fire to occur under extreme weather conditions, for the Colorado Plateau, AZ–NM Mountains, and Apache Highlands ecoregions.

a slight underestimate of reference period and trailing edge forest.

Land management agencies have several available options for reducing the potential for fire-facilitated type conversions and slowing forest loss. For example, forest restoration treatments such as prescribed fire and thinning are effective strategies to reduce the probability of stand-replacing fire (Agee and Skinner 2005, Safford

Table 3. Area of trailing edge forest affected by each fire severity class (expected fire severity under extreme weather conditions).

Ecoregion name	Ecoregion ID (Fig. 1)	Other severity (km <sup>2</sup> )	Stand-replacing (km <sup>2</sup> )†	Other severity (% reference period forest)	Stand-replacing (% reference period forest)†
Colorado Plateau	6	19,845	11,910	33.3	20.0
AZ–NM Mountains	7	2101	24,984	3.5	41.3
Apache Highlands	8	6822	7871	23.5	27.1
Total	–	28,768	44,765	19.3	30.0

† These columns should be interpreted as the area and percent of forested area at risk of fire-facilitated conversion to non-forest. They represent pixels that are trailing edge forest and are at risk of stand-replacing fire under extreme weather conditions.

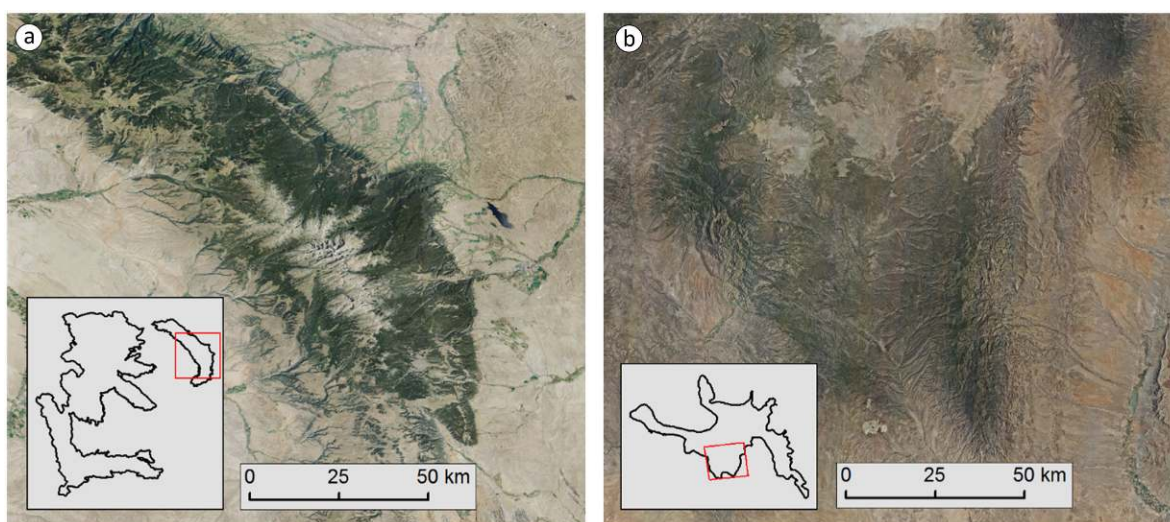


Fig. 8. Example showing abrupt lower treelines of the Utah-Wyoming ecoregion (a) and the relatively diffuse lower treelines in the AZ–NM Mountains ecoregion (b).

et al. 2012, Kennedy and Johnson 2014); such treatments would be particularly relevant for drier, trailing edge sites that have been heavily impacted by fire suppression and historic logging operations. Given that fire severity tends to increase during years of extreme drought (Abatzoglou et al. 2017, Keyser and Westerling 2017, Parks et al. 2018c), judiciously allowing naturally ignited fires to burn during non-drought years could also be a viable option for reducing fuel loads and reducing the probability of stand-replacing fire. Walker et al. (2018), for example, showed that sites with a restored fire regime were less likely to convert to non-forest than sites with altered fire regimes (also see Larson et al. 2013). In cases where forests are substantially

degraded; however, some have suggested that fire should not be reintroduced without first applying treatments such as thinning (Allen et al. 2002). In the driest portions of trailing edge forests or within designated wilderness and other protected areas, allowing nature to take its course (i.e., no management intervention) may be the most appropriate climate change response strategy. Resisting change, for example, by aggressively preventing fire from occurring, could be considered a viable short-term strategy in locations with highly valued resources (e.g., municipal watersheds), but such a strategy may be unsuccessful in the long-run because directional climate change will ultimately cross ecological thresholds (Millar et al. 2007, Walther



2010). Ultimately, a diverse portfolio of climate change response strategies could serve as a bet hedging strategy in trailing edge forests given uncertainty in future disturbances, their interactions, and associated ecological responses (Millar et al. 2007).

Although trailing edge forests comprise about 36% (176,000 km<sup>2</sup>) of the contemporary forest area, some of this potential forest loss may be offset by high elevation alpine areas that will become climatically suitable to forest in the coming decades (i.e., the leading edge). Indeed, many organisms are expected to move upslope and poleward (Parmesan and Yohe 2003) under a warming climate and forest encroachment (i.e., conversions from non-forest to forest) has been documented at higher latitudes and upper elevations (Harsch et al. 2009, Hofgaard et al. 2013). However, we suggest that climate-induced forest encroachment will not completely offset potential forest loss in the intermountain western United States, as a decrease of ~102,000 km<sup>2</sup> of area that is climatically suitable to forest is expected (Table 1). Furthermore, climate-induced forest encroachment will be curtailed by the physical reality of decreasing overall area at upper elevations in mountain ranges (Elsen and Tingley 2015). This said, other factors besides temperature may control upper elevation treelines (Crausbay et al. 2014) and unintuitive responses may arise (e.g., ecotones shifting downward in elevation; Foster and D'Amato 2015), thereby increasing uncertainty in the future distribution of forests.

The spatial resolution of our analysis (1 km) does not capture finer resolution processes that create heterogeneity in the risk of fire-facilitated conversion to non-forest. In particular, fire regime characteristics and vegetation (type and structure) are known to vary according to slope aspect and potential solar radiation (Whittaker 1960, Taylor 2000). Also, the fire severity predictions we used in this study are contingent on a fire actually burning, yet we know the probability of burning varies widely according to factors such as fuel, ignitions, and topography (Ager et al. 2007, Parisien et al. 2011). Future efforts that address fire-facilitated conversion to non-forest could therefore incorporate finer-scale controls on fire severity (Krawchuk et al. 2016) as well as fire probability maps (Short et al. 2016) to highlight high-risk regions.

It is worth noting that our results are likely affected by propagation of model error that results in substantial yet unexplored model uncertainty. Because this error is difficult or impossible to characterize, we urge caution in strictly interpreting our results in terms of the amount and proportion of forest at risk of fire-facilitated conversion. An additional but related point is that our characterization that certain trailing edge forests are at risk of conversion should not be interpreted as a mid-century prediction of changes to the distribution of forest. Instead, our intention is to illustrate vulnerability of forested landscapes in terms of fire-facilitated conversion to non-forest. In summary, our study should be interpreted with the understanding that our findings have an unknown degree of uncertainty.

## CONCLUSIONS

Climate change will influence ecological systems across the planet (Parmesan and Yohe 2003), including expected range shifts in forest biomes (Bonan 2008, Gonzalez et al. 2010). As the climate continues to warm, trailing edge forests have the potential to experience abrupt conversions to non-forest (Allen and Breshears 1998). Such conversions are of particular concern in the semi-arid and fire-prone intermountain region of the western United States where many communities rely on forests for ecosystem services such as clean water, timber production, and carbon sequestration (Bachelet et al. 2001, Lawler et al. 2014). Yet, broad-scale conversions to non-forest are likely to be gradual in the absence of stand-replacing disturbances such as severe fire (Svenning and Sandel 2013). Our study explicitly evaluated the spatial correspondence between trailing edge forest and stand-replacing fire. In doing so, we characterized areas that are primed for change and have the potential for fire-facilitated conversion from forest to non-forest. We found that 6.6% of current forest in the intermountain United States is at risk of such conversions, though this varied among ecoregions. Although this value (6.6% of forest) may not outwardly seem alarming, we note that this is a conservative estimate. When we incorporated fire severity predictions under extreme fire weather in the southwestern United States, we

found that 30% of forest is at risk of fire-facilitated conversion to non-forest. Recent studies in the southwestern United States and elsewhere have documented such conversions. Given our estimate that nearly 36% of forest area in the intermountain United States will be trailing edge forest by mid-century, other non-fire disturbances such as drought, insect outbreaks, and their interactions may put trailing edge forests at further risk (Allen and Breshears 1998).

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