Ecological Applications, 0(0), 2019, e02030

Can wildland fire management alter 21st-century subalpine fire and forests in Grand Teton National Park, Wyoming, USA?

Winslow D. Hansen,[1,4,5] Diane Abendroth,[2] Werner Rammer,[3] Rupert Seidl,[3] and Monica G. Turner[1]

[1]*Department of Integrative Biology, University of Wisconsin–Madison, Madison, Wisconsin 53706 USA*
[2]*Institute of Silviculture, Department of Forest- and Soil Sciences, University of Natural Resources and Life Sciences (BOKU), Vienna Austria*
[3]*Grand Teton National Park, Teton Interagency Fire, Moose, Wyoming 83012 USA*

Citation: Hansen W. D., D. Abendroth, W. Rammer, R. Seidl, and M. G. Turner. 2019. Can wildland fire management alter 21st-century subalpine fire and forests in Grand Teton National Park, Wyoming, USA? Ecological Applications 00(00):e02030. 10.1002/eap.2030

Abstract. In subalpine forests of the western United States that historically experienced infrequent, high-severity fire, whether fire management can shape 21st-century fire regimes and forest dynamics to meet natural resource objectives is not known. Managed wildfire use (i.e., allowing lightning-ignited fires to burn when risk is low instead of suppressing them) is one approach for maintaining natural fire regimes and fostering mosaics of forest structure, stand age, and tree-species composition, while protecting people and property. However, little guidance exists for where and when this strategy may be effective with climate change. We simulated most of the contiguous forest in Grand Teton National Park, Wyoming, USA to ask: (1) how would subalpine fires and forest structure be different if fires had not been suppressed during the last three decades? And (2) what is the relative influence of climate change vs. fire management strategy on future fire and forests? We contrasted fire and forests from 1989 to 2098 under two fire management scenarios (managed wildfire use and fire suppression), two general circulation models (CNRM-CM5 and GFDL-ESM2M), and two representative concentration pathways (8.5 and 4.5). We found little difference between management scenarios in the number, size, or severity of fires during the last three decades. With 21st-century warming, fire activity increased rapidly, particularly after 2050, and followed nearly identical trajectories in both management scenarios. Area burned per year between 2018 and 2099 was 1,700% greater than in the last three decades (1989–2017). Large areas of forest were abruptly lost; only 65% of the original 40,178 ha of forest remained by 2098. However, forests stayed connected and fuels were abundant enough to support profound increases in burning through this century. Our results indicate that strategies emphasizing managed wildfire use, rather than suppression, will not alter climate-induced changes to fire and forests in subalpine landscapes of western North America. This suggests that managers may continue to have flexibility to strategically suppress subalpine fires without concern for long-term consequences, in distinct contrast with dry conifer forests of the Rocky Mountains and mixed conifer forest of California where maintaining low fuel loads is essential for sustaining frequent, low-severity surface fire regimes.

Key words: climate change; forest resilience; fuel limitations; Greater Yellowstone Ecosystem; suppression; wildfire management.

Introduction

In subalpine forests of the western United States that historically experienced infrequent, high-severity fire, whether fire management can shape 21st-century fire regimes and forest dynamics is not known. Wildfire activity is rapidly increasing with warming, particularly in subalpine forests (Westerling et al. 2006, Westerling 2016, Littell et al. 2009, Abatzoglou and Williams 2016, Keyser and Westerling 2019). These trends are expected to accelerate (Westerling et al. 2011, Abatzoglou and Williams 2016, Kitzberger et al. 2017), shifting the baselines associated with fire occurrence and management of fire on public lands. Increasing fire activity is causing suppression costs to rise (National Interagency Fire Center 2018) and damages to mount (Thomas et al. 2017), making it difficult to balance the ecological benefits of fire with undesirable social consequences (Chapin et al. 2008, Calkin et al. 2013, Stephens et al. 2013, Moritz et al. 2014, Steelman 2016, Bentley and Penman 2017, Ingalsbee 2017, Schoennagel et al. 2017, Thompson et al. 2017). Thus, there is an urgent need to evaluate the effectiveness of

Manuscript received 9 April 2019; revised 9 August 2019; accepted 16 September 2019. Corresponding Editor: Tamara J. Zelikova.
[4]Present address: Earth Institute, Columbia University, New York, New York 10025 USA.
[5]E-mail: whansen@ldeo.columbia.edu

current fire management strategies for meeting important social and ecological objectives within the context of projected changes.

Key objectives of federal fire management in the western United States include (1) ensuring people and property are protected from wildfire and (2) maintaining natural fire regimes and fostering mosaics of forest structure, stand age, and tree species composition (National Wildfire Coordinating Group 2009, National Park Service 2019). Given shifting baselines in recent decades, federal agencies responsible for managing fire developed the National Cohesive Wildland Fire Management strategy to provide guidance on how these objectives might be met in coming decades (National Strategic Committee 2014). One management option emphasized in the strategy is managing wildfire for resource objectives (hereafter; managed wildfire use), where lightning-ignited fires are allowed to burn when risk is low, instead of suppressing them. This approach is meant to ensure that fire can continue to play its critical ecological role, and potentially reduce fuel loads, which might lower the probability of subsequent fires that exceed the historical range of variability and threaten people (Jensen and McPherson 2008, Prichard et al. 2017, Schoennagel et al. 2017).

However, implementing managed wildfire use has proven difficult (Thompson et al. 2018). Most 20th-century wildfires in the western United States were suppressed, and those that escaped containment typically burned under extreme drought and wind (Bessie and Johnson 1995, Keane et al. 2008, National Academy of Sciences, Engineering, and Medicine 2017). Many fires are still suppressed because of risk to people and property, even in national parks and wilderness areas where policy has long encouraged allowing lightning-ignited fires to burn under moderate weather (i.e., when fuel moisture is relatively high; Despain and Sellers 1977, Brown 1991). For example, ~60% of lightning-caused fire starts were extinguished since 1988 in Grand Teton and Yellowstone National Parks, the core of the largest intact wildland area in the conterminous United States.

Little guidance also exists for where and when managed wildfire use is likely to lower the risk of subsequent catastrophic fires. Several retrospective studies do suggest recent fires can limit subsequent fire occurrence (Parks et al. 2014, 2018*a*), spread (Parks et al. 2015), and severity (Parks et al. 2013, 2014, 2016, Harvey et al. 2016*a*). However, the feedbacks appear short lived (~5–30 yr) and their strength varies with weather, because fires can spread through forest landscapes with little accumulated fuel when weather conditions are extreme (i.e., low fuel moisture and high wind; Turner and Romme 1994, Turner et al. 1994, Parks et al. 2018*b*).

The effectiveness of managed wildfire use for reducing size or severity of future fires also may vary by forest type. It appears well suited for dry forests where maintaining small fuel loads is essential to sustain frequent, low-severity, surface fire regimes (Allen et al. 2002, Savage and Mast 2005, North et al. 2012, Krofcheck et al. 2018, Parks et al. 2018*c*, Walker et al. 2018). Whether the strategy is as effective in subalpine forests, however, is not clear. In this forest type, most burned area occurs in a few events when weather conditions are extreme (Turner and Romme 1994, Baker 2009, Marlon et al. 2012, Loehman et al. 2018). Historically, subalpine fires were often not limited by fuels because fires were infrequent, tree species were well adapted to the historical fire regime, and trees recovered rapidly (Turner et al. 1997, 2004, 2016), meaning that fuels accumulated quickly relative to the historical fire return interval (Harvey et al. 2016*b*, Nelson et al. 2016). Thus, widespread implementation of managed wildfire use in this forest type would likely have had little influence on subsequent fire during the 20th century (Schoennagel et al. 2004). However, managed wildfire use might influence some characteristics of future fires, perhaps by reducing their size or severity. This is because projected changes in climate and fire will likely compromise postfire tree regeneration or even be incompatible with persistence of dominant tree species in many places (Brown and Johnstone 2012, Martínez-Vilalta and Lloret 2016, Liang et al. 2017, Hansen et al. 2018, Stevens-Rumann et al. 2018, Buotte et al. 2019, Hansen and Turner 2019, Davis et al. 2019), which could cause fuels in subalpine forests to become increasingly limiting.

Determining whether and how widespread implementation of managed wildfire use would shape current and future fire regimes and forest structure in western subalpine landscapes could be exceptionally helpful for managers who are tasked with stewarding these systems during a time of profound environmental change. However, the fingerprint of past suppression in management of subalpine fires makes it difficult to evaluate in the field. Process-based simulation modeling is a promising tool because it allows for a scenarios-based approach, rather than an empirical one (Keane et al. 2019). We conducted simulations of a large landscape in Grand Teton National Park (GRTE), Wyoming, USA to ask: (1) how would subalpine fires and forest structure be different if fires had not been suppressed during the last three decades? And (2) what is the relative influence of climate change vs. fire management strategy on future fire and forests? We contrasted fire and forest dynamics from 1989 to 2098 in GRTE under two scenarios: managed wildfire use, and fire suppression. In both scenarios, we assumed that wildfires could be suppressed or allowed to burn under moderate weather but would remain uncontrollable when weather conditions were extreme. Consistent with historical observations (e.g., Schoennagel et al. 2004), we expected fire patterns and forest structure not to differ between fire management scenarios over the last three decades. However, we expected that effects of managed wildfire use could change during the latter half of the 21st century. Managed wildfire use might begin to limit subsequent burning, relative to the suppression scenario, if changes in future fire and climate begin to cause tree regeneration failure and reduce fuels.

Methods

Study area

We simulated most of the contiguous forest (~40,000 ha) in GRTE (Fig. 1). GRTE is located in the southern Greater Yellowstone Ecosystem on the border of eastern Idaho and northwestern Wyoming. The park was created in 1929 to protect the high peaks of the Teton mountain range, and was expanded in 1950 to include extensive portions of the surrounding Jackson Hole Valley. The park encompasses approximately 1,200 km^2, of which, 42% is forest. The remaining non-forest includes alpine tundra, boulder fields, meadows, grasslands, and shrublands (Knight et al. 2014). Elevation ranges from 1,600 to 3,400 m, with lower tree line at 1,800 m and upper tree line above 3,000 m. Lower treeline is warm and dry in the growing season with a mean July temperature of 27°C and 540 mm of annual precipitation (Western Regional Climate Center 2018). Lower elevation forests are dominated by Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) and occasional stands of aspen (*Populus tremuloides*). Historically, forests at lower treeline burned frequently, at low severity (Houston 1973, Arno and Gruell 1983, Jacobs and Whitlock 2008). Temperatures are cooler and moisture increases at higher elevations, as lower montane forests transition to subalpine forests dominated by lodgepole pine (*Pinus contorta* var. *latifolia*) or mixtures of Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*). Subalpine forests in Greater Yellowstone have burned in large stand-replacing fires at intervals of 100 to 300 yr for millennia (Romme 1982, Millspaugh et al. 2000, Power et al. 2011).

Following decades of universal fire suppression, the U.S. National Park Service implemented a new management policy in 1972 that allowed lightning-ignited fires to burn in remote areas when risk was low (Despain and Sellers 1977). In GRTE, the 1974 Waterfalls Canyon Fire, which burned 1,414 ha along the western shores of Jackson Lake, was one of the first wildfires managed under this policy (Doyle et al. 1998). Such fires were allowed to burn until 1988. That summer, nearly 250 fires consumed >500,000 ha of Greater Yellowstone under conditions of extreme drought and high winds. In GRTE, >6,000 ha burned, which was comparable to the largest known previous fires in the region (late 1870s Jackson Hole fires). The size and severity of the 1988 fires caused a reevaluation of natural fire policies in federal lands. Managed wildfire use was immediately suspended in western U.S. national parks (Knight 1991) and not reinstated until 1992 (Knight et al. 2014). Retrospective evaluation of the 1988 fires revealed that weather was the dominant driver of fire size and severity (Christensen et al. 1989). Fire management policy again shifted away from universal suppression and, in alignment with NPS goals of minimizing consequences for human well-being and maintaining a mosaic of forest ages across the landscape, contemporary fire management policies in GRTE allow lightning-caused fires to burn with close monitoring when risk is low.

Model overview

iLand is a landscape-scale forest model that simulates ecological processes at multiple spatial and temporal resolutions in a hierarchical framework. (Seidl et al. 2012, 2019). iLand simulates tree growth and mortality of individual trees and the interactions among them in spatially explicit landscapes as a function of canopy light interception, radiation, temperature, soil water, and nutrients. The model also explicitly simulates tree regeneration processes, such as seed production (including serotiny), dispersal, and environmental controls (such as drought) on seedling establishment and sapling growth (Hansen et al. 2018). Both sexual reproduction (i.e., by seed) and resprouting are simulated for aspen. Thus, following fire, tree regeneration is influenced by the age of the trees that burned (determining the size of the canopy seed bank for serotinous species), distance to the nearest unburned seed source, soil moisture conditions in subsequent growing seasons, and plant reproductive traits (Hansen et al. 2018). The model has been well tested in Greater Yellowstone (Braziunas et al. 2018, Hansen et al. 2018).

Climate and soil are assumed spatially homogeneous within a 1-ha grid cell, but within-cell variation in light and tree regeneration is simulated at 2×2 m resolution based on forest structure. iLand is forced with daily temperature, precipitation, vapor pressure deficit, and radiation. For this application, we used gridded climate data sets that were statistically downscaled (4-km resolution) with the Multivariate Adaptive Constructed Analogs approach (Abatzoglou and Brown 2012). These included two general circulation models (GCMs), CNRM-CM5 (Voldoire et al. 2013) and GFDL-ESM2M (Dunne et al. 2012, 2013), that represent 20th-century climate well in Greater Yellowstone (Westerling et al. 2011). For each GCM, the first of the five runs from the Inter-Governmental Panel on Climate Change AR5 experiment were downscaled. We also included two representative concentration pathways (RCPs) 8.5 and 4.5, which assume continued increases in radiative forcing to 8.5 W/m^2 by 2100 and stabilization of radiative forcing at 4.5 W/m^2 by 2100, respectively. Both GCMs show similar temperature trends with ~5°C of summer warming by 2099 under RCP 8.5.

iLand dynamically simulates wildfire at $20 \text{ m} \times 20 \text{ m}$ resolution in a modeling framework designed initially for the Northern Rocky Mountains (Keane et al. 2011) and western Oregon (Seidl et al. 2014). For an exhaustive description of the iLand fire module, see Seidl et al. (2014) and Seidl and Rammer (2019). Briefly, fire is

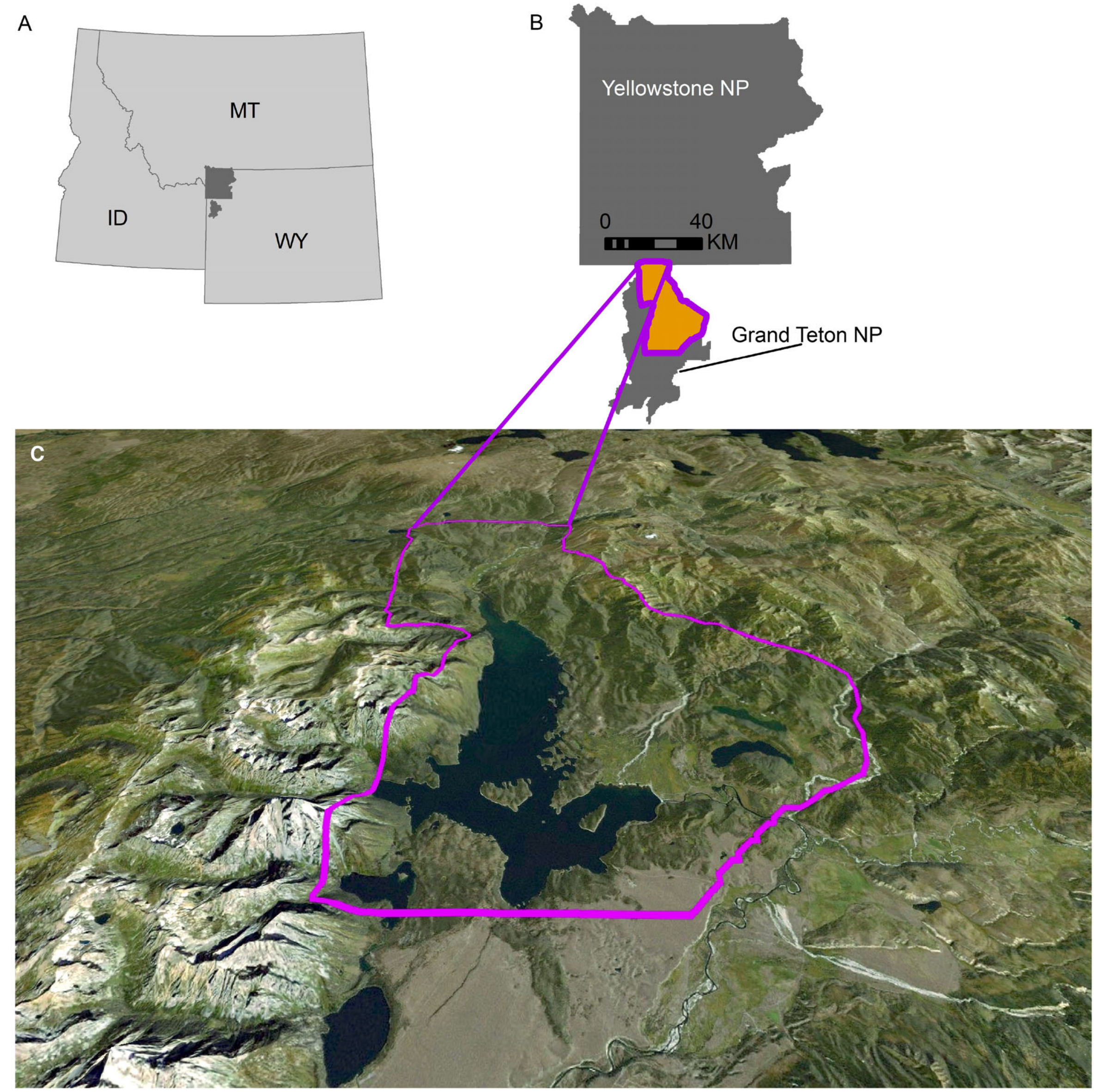

FIG. 1. (A) The study landscape is located in Grand Teton National Park in the Greater Yellowstone Ecosystem, the largest intact wildland area in the contiguous United States. The ecosystem sits at the corners of Montana, Idaho, and Wyoming. (B) We simulated ~40,000 ha of subalpine forest in Grand Teton National Park (80% of forested area in the park and 33% of total area of the park). (C) The landscape sits to the west of the Teton mountain range, centered on Jackson Lake and is representative of subalpine forests across the western United States.

simulated based on statistical distributions of fire occurrence and size, fuel load (including surface litter and downed coarse wood pools, excluding live fuels and dead canopy fuels), and drought (using the Keetch Byram drought index, KBDI; Keane et al. 2011, Seidl et al. 2014, Abatzoglou and Williams 2016). KBDI is a cumulative daily metric of water balance for the fuel layer that accounts for effects of both precipitation and temperature. In the fire module, daily KBDI is summed for each simulation year, and compared to a reference KBDI (1980–2016) to compute a KBDI anomaly.

Fire ignition in any given 20 × 20 m cell that has sufficient available fuels ($\geq$0.05 kg/m^2 or 500 kg/ha; see Appendices S1 and S2 for description of fuels in iLand) is modeled based on the 20th-century fire return interval and adjusted by the KBDI anomaly so that ignition is more likely when conditions are hot and fuels are dry and less likely when conditions are cool and fuels are wet. Fire size is modeled by first drawing a maximum potential fire size from a negative exponential distribution, fit to 20th-century fires, and then dynamically spreading the fire across the landscape using a cellular automaton approach. Because fire size in subalpine forests is strongly driven by aridity, we modified the distributions that determine maximum fire size so that only fires >400 ha are chosen when KBDI anomaly is >1.7 (hot-dry conditions), and only fires <10 ha are chosen when KBDI anomaly is <1 (cool-wet conditions). A KBDI anomaly cutoff of 1.7 delineated the 5% driest years in the contemporary climate record (1980–2016) for the study landscape, and the minimum fire size selected when KBDI anomaly exceeded this threshold was determined based on historical fire records for the region from 1970 to 2016 (Appendix S1). However, many other factors (e.g., winds, flash droughts) can cause fires to become large when our metric of aridity (KBDI anomaly) are at intermediate values. So, when the KBDI anomaly values are between 1 and 1.7, simulated fire size is drawn from a negative exponential distribution with a mean size of 75 ha and a maximum size of 20,000 ha. Thus, large fires can occur at intermediate KBDI anomaly values (between 1 and 1.7), but large fires have a lower probability of occurrence.

Once a maximum fire size is selected, fire spreads dynamically through the landscape, with probability of spread to adjacent cells contingent on fuel load, wind, and slope. Fuel constraints were set so that a fire can only spread if ≥0.05 kg/m^2 (500 kg/ha) of fuel is present in neighboring cells, the same threshold for fire ignition (Keane et al. 2011, Seidl et al. 2014). Wind is simulated with a given speed and direction per fire event (both randomly selected for each event from user-defined ranges). Spread rates differ if fires burn upslope or downslope and vary with slope angle. In the burned cells, fire severity is modeled as percent crown kill based on fuels, KBDI anomaly, tree size, and bark thickness (Seidl et al. 2014). To ensure iLand could re-create 20th-century fire activity with reasonable skill, we parameterized the model and compared the simulated fire regime to historical fire records (1970–2016) from the study area (Appendix S1).

Simulation experiment

To ensure the initial simulation landscape corresponded with the tree-species composition, forest structure, and stand age distributions in the actual landscape, while also creating conditions consistent with the internal model logic, we conducted a 300-yr spin-up under historical climate (CNRM-CM5 period: 1950–2005, climate years randomly chosen with replacement) and fire (Seidl et al. 2019). This procedure generated a simulation landscape similar to the actual landscape in 1989 (Appendix S2). Using the dynamic fire module, we then simulated the resulting landscape from 1989 to 2098 while varying climate conditions (two GCMs and RCPs, as described in *Model overview*) and fire management (two scenarios).

The two fire-management scenarios were designed to be generally consistent with how fire management operates in western subalpine forests, but they were not meant to precisely replicate past management actions in GRTE, nor predict what *will* occur in this landscape, as that is not feasible. The two management scenarios were (1) a managed wildfire use scenario where all fires were allowed to burn naturally and (2) a fire suppression scenario in which fires that ignited when drought was moderate (KBDI anomaly ≤1.7) were suppressed and never grew larger than 0.04 ha, but fires that ignited when drought was extreme (KBDI anomaly >1.7) burned unhindered. Thus, we represented effective suppression of fires when conditions are cool and wet, as was typical of many years in the historical record, and the inability to suppress fires when conditions are hot and dry. We assumed that drought was the dominant factor influencing fire suppression for the purpose of this analysis; other variables (topography, wind speed, proximity to roads, distance to structures) that might influence suppression effectiveness (Coen et al. 2018) were not represented. Because climate projections do not vary within a given GCM and RCP but fire is stochastic in iLand (probability and location of fire ignitions and the fire sizes and severities that result), we simulated 20 replicates for each combination of GCM × RCP × suppression scenario (n = 160).

Model outputs

To determine the relative importance of fire management vs. climate change in the simulations, we analyzed the spatial and temporal patterns of fire (number of fires, area burned, area-weighted mean fire size, and proportion of stand-replacing fire (>90% of mature trees killed in 1-ha grid cells) and forests (forested area, fuel loads, stand age, and dominant tree species in 1-ha grid cells) in the different scenarios. Annual number of fires was tallied within four size classes (<10 ha, 10–100 ha, 100–225 ha, and >225 ha). Fire severity was calculated as the proportion of area in stand-replacing fire (>90% crown kill) within each fire perimeter. Forested area was defined as areas with ≥50 trees/ha (Hansen et al. 2018). We also calculated the median fuel load (coarse and fine downed wood) in forested areas. Stand age was tallied within four classes (<40 yr, 40–150 yr, 151–250 yr, and >250 yr). Species dominance in each 1-ha cell was quantified by using species importance values (IV). Importance values sum the relative abundance (number of individuals of a species divided by number of individuals of all species in a grid cell) and relative basal area (basal area attributed to that species divided by total basal area in a grid cell) for each species on a plot. Thus, species IV ranges from zero (species is not present) to two (pure stand of the species). We then tallied the forested cells in the simulated landscape that were dominated (i.e., IV > 1) by lodgepole pine, Douglas-fir, spruce–fir, or aspen.

Analysis

We quantified differences in response variables among scenarios from 1989 to 2017 (Q1; effect of fire suppression on contemporary landscapes) and 2018 to 2098 (Q2: relative influence of warming and fire suppression on future fire and forest). We evaluated differences among scenarios by comparing means and bootstrapped 95% confidence intervals (CIs). However, in interpreting model results, we emphasize ecologically meaningful differences rather than statistical ones. To aid in the interpretation of our complex dataset, we first address Q1 and Q2 using the GCM (CNRM-CM5) and RCP (8.5) where most area burned in the simulations. Then, we contrast key differences between this scenario and the other GCMs and RCPs.

Results

Initial conditions

Following a spin-up period, iLand represented the contemporary GRTE forest landscape well (Appendix S2). In 1989, 40,178 ha of the simulated subalpine landscape was forested (≥50 stems/ha; Fig. 2A). Approximately 50% of the forested area was of intermediate age (40 to 150 yr old), 19% was young (<40 yr old), 18% was mature (150–250 yr old), and 8% was old growth (>250 yr old; Fig. 2B). Simulated forest area was dominated by lodgepole pine (74%), followed by Douglas-fir (16%), spruce–fir (9%), and aspen (1%; Fig. 2C).

Q1. Effect of fire suppression on contemporary landscapes (GCM CNRM-CM5, RCP 8.5)

From 1989 to 2017, mean annual KBDI across the landscape ranged from 74 to 238 (Fig. 2D) and a total of 93 fires (CI 88–98), or 3.3 fires/yr (CI 2.6–4.1) ignited regardless of whether fires were suppressed under moderate conditions or not. Fire occurred in all years in both scenarios, with the average annual number of fires ranging from one to six. In the fire suppression scenario, nearly all fires were <10 ha in size, as expected given the suppression algorithm, but large fires (>225 ha) occasionally occurred when KBDI values were high (Fig. 3A). Fire sizes were mixed in the managed wildfire scenario, with more fires in the intermediate and large size classes than in the suppression scenario (Fig. 3A). Annual area burned was generally low but varied between the scenarios. On average, 92 ha/yr (CI 53–162) burned in the suppression scenario and 259 ha/yr (CI 171–390) burned in the managed wildfire use scenario (Fig. 3B). However, the proportion of total burned area that was stand replacing did not differ between the two scenarios (Fig. 3B), and averaged 42% (CI 30–54). The area-weighted mean fire size also did not differ between scenarios and averaged 97 ha (CI 71–138; Fig. 3C), indicating that most of the area burned was in larger patches. Modest differences in fire activity between scenarios had little effect on forest extent, stand-age distribution, or tree species dominance by 2017. Forests still occupied the same area within GRTE, and remained dominated by lodgepole pine, with most forests in the intermediate (40–150 yr old) age class.

Q2. Projected future fire and forests (GCM CNRM-CM5, RCP 8.5)

During the 2018–2098 simulation period, KBDI increased, particularly after 2030 (Fig. 2D). In the mid- to late-21st century, mean annual KBDI exceeded 200 in most years and often exceeded 300, particularly after 2050. Similar to the historical period, the suppression scenario was dominated by small fires through 2050, with occasional years when fires burned >225 ha. The managed wildfire use scenario had fires in the intermediate size class during this window, again similar to the historical period. However, consistent with increasing mean annual KBDI values, large fires became increasingly common in both scenarios after 2050. A total of 460 fires (CI 451–471), or 5.6 fires/yr (CI 4.6–6.6) ignited between 2018 and 2098 in both the suppression and managed wildfire use scenarios. The annual number of fires, area burned, and area-weighted mean size sharply increased in the second half of the 21st century (Fig. 3). After 2050, annual area burned exceeded 15,000 ha during many years in both scenarios (Fig. 3B). The area and proportion of stand replacing fire steadily increased during the latter half of the century, again in both fire management scenarios (Fig. 3B). Neither burned area nor area-weighted fire size differed meaningfully between suppression and managed wildfire use scenarios in most years (Fig. 3B,C).

Simulated forest area remained relatively stable at ~43,000 ha until the mid-21st century (Fig. 4A). In concert with increases in large, high-severity fires during the mid-century, forest extent decreased abruptly to ~34,000 ha in less than a decade. The extent of forest again declined abruptly near the end of the century with a second rise in fire number and size. By 2098, only 65%, or 27,365 ha (CI 26,53– 28,084), of the original forested area remained forest in both suppression and managed wildfire use scenarios. During the 21st century, average fuel loads in the landscape remained high and steadily increased through the early- to mid-21st century, to 67,320 kg/ha (CI = 66,966 – 67,735) and 68,467 kg/ha (CI = 68,154 – 68,844) in the suppression and managed wildfire use scenarios. Fuel loads declined slightly in the last three decades of simulation and converged to 62,984 kg/ha (CI 62,577 – 63,368) in both scenarios. Forest age distributions varied through time, but changes were similar in the suppression and managed wildfire use scenarios. Stand-age remained relatively steady through 2050, after which, the extent of intermediate aged stands declined. Young forests expanded late in the 21st century and accounted for 22% of remaining

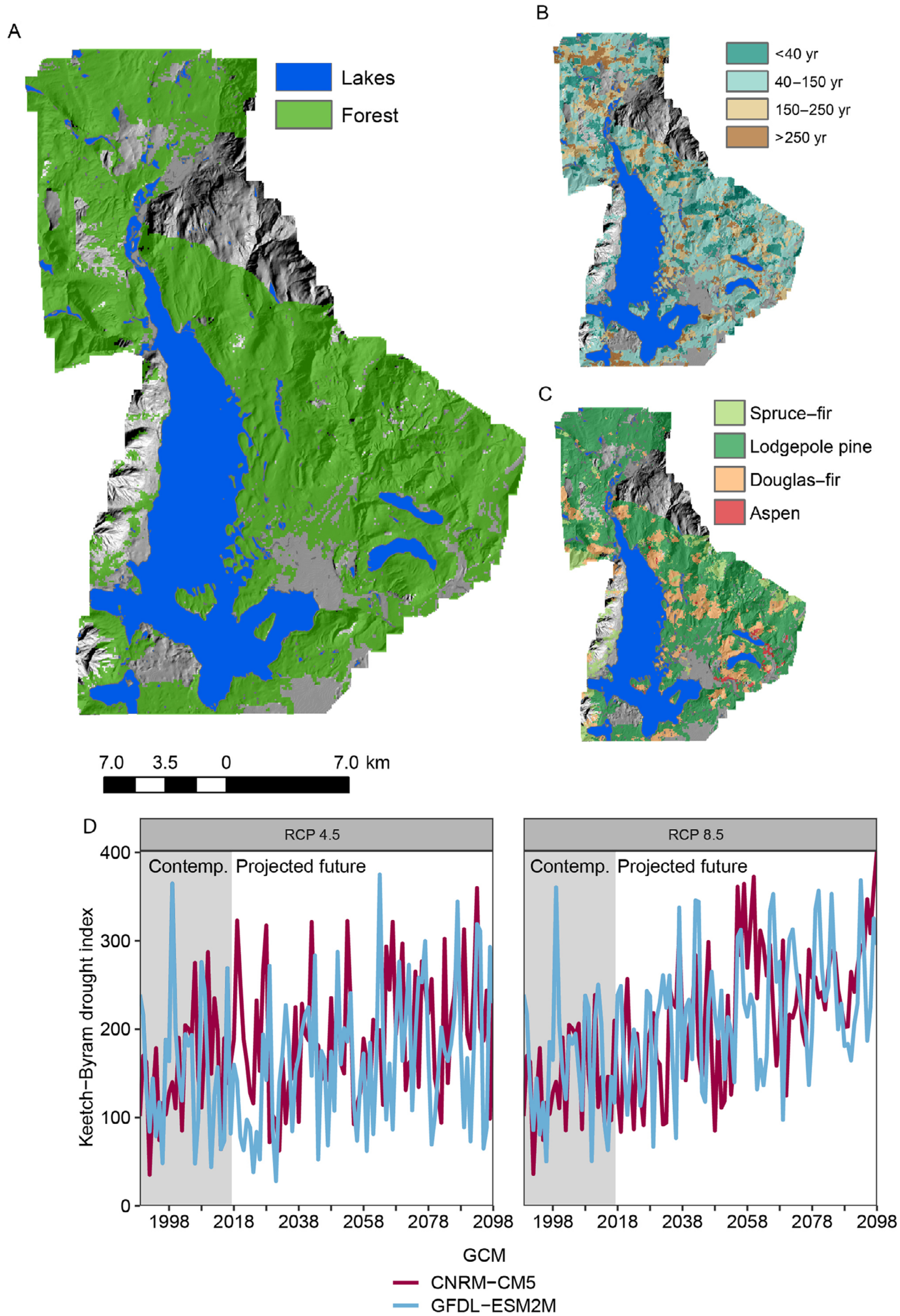

Fig. 2. (A) Forested area, (B) stand age distribution, and (C) tree-species dominance in the initial landscape (1989) used in the simulation experiments. The nonforested area (<50 trees/ha) in the upper right corner is a result of the 1988 wildfire that burned in this landscape. (D) The Keetch-Byram drought index (KBDI) in the simulated landscape between 1989 and 2098 when forced with the general circulation models (GCMs) CNRM-CM5 and GFDL-EM2M, broken out by representative concentration pathway (RCP; 4.5 and 8.5). KBDI is a measure of aridity that integrates daily temperature and precipitation effects on water balance. It scales between 0 (no drought) to 800 (severe drought). Contemp., contemporary.

forest by 2099 in both scenarios (Fig. 4B). Similarly, dominant forest types changed over time but differed little between fire management scenarios. The area dominated by lodgepole pine and spruce-fir forests declined after mid-century, and Douglas-fir increased, eventually replacing lodgepole pine as the dominant

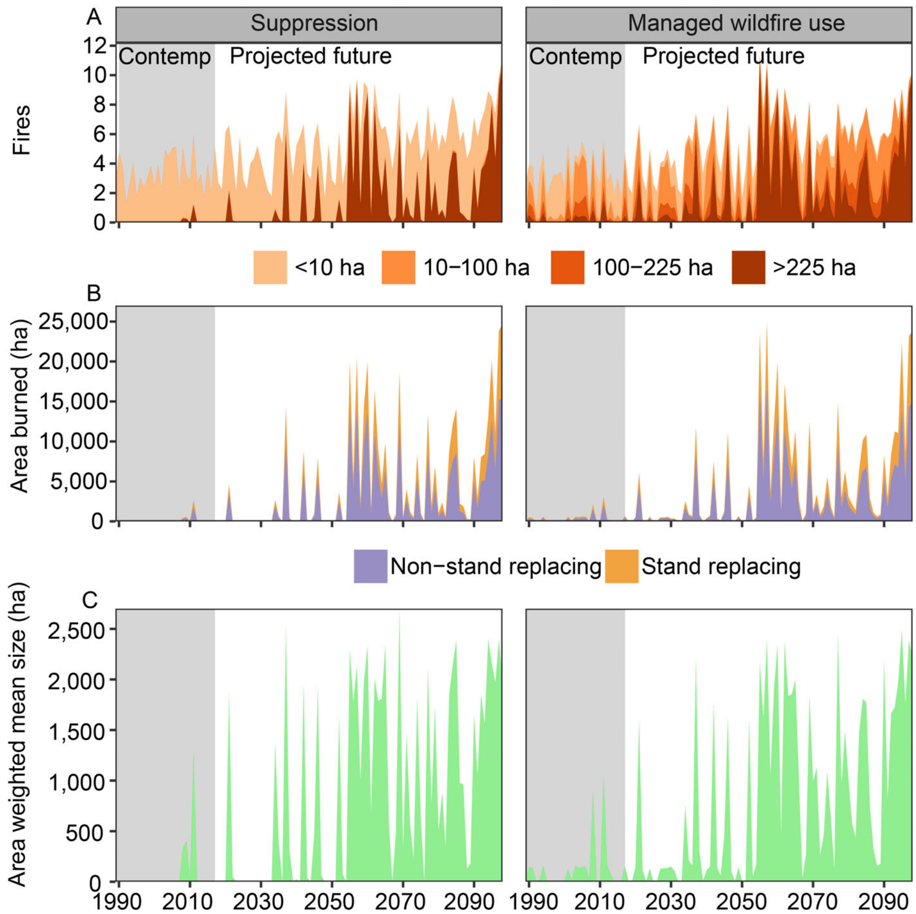

Fig. 3. (A) The number of fires between 1989 and 2098 that burned <10 ha, 10–100 ha, 100–225 ha, and >225 ha in suppression and managed wildfire use scenarios. (B) The annual area (ha) that burned as stand replacing and non-stand replacing between 1989 and 2098 in suppression and managed wildfire use scenarios. (C) Area-weighted mean fire size (ha) between 1989 and 2017. Area weighted mean is a geometric mean, which is more representative of the patch size one would encounter walking around a landscape. All values are means across the 20 replicate simulations of the GCM CNRM-CM5 under the RCP 8.5 scenario.

forest type in both scenarios (Fig. 4C). In 2098, Douglas-fir dominated 63%, or 17,240 ha (CI 16,780 – 17,680), of remaining forest area, while lodgepole pine accounted for only 28%, or 7,633 ha (CI 7,117– 8,168).

Variation among GCMs and RCPs

Consistent with the CNRM-CM5 RCP 8.5 scenario, fire activity increased during the 21st century when simulations were forced with the other GCM climate projections, and differences between fire management scenarios remained minimal. In general, mean number of fires, area burned, and area-weighted mean fire size did not differ between CNRM-CM5 and GFDL-EM2M GCMs when forced with RCP 8.5. However, the timing of big fire years and maximum values for each variable differed (Appendix S3). Between 1989 and 2017, KBDI averaged 162 and ranged between 50 and 360 in the GFDL-EM2M RCP 8.5 scenario (Fig. 2D). The maximum annual number of large fires, area burned, and area-weighted mean fire size were larger in the GFDL-EM2M RCP 8.5 scenario than in the CNRM-CM5 RCP 8.5 scenario (Appendix S3). On average, 4 fires/yr (CI 3.2– 4.8) and 767 ha/yr (CI 584– 1,077) burned between 1989 and 2017. From 2018 to 2098, KBDI increased, averaging 223 and exceeding 300 in many years. Maximum annual area burned was smaller during the projected future under the GFDL-EM2M RCP 8.5 scenario, compared with the CNRM-CM5 RCP 8.5 scenario (Appendix S3). However, large fires (>225 ha) occurred more consistently during the mid- to late-21st century under GFDL-EM2M RCP 8.5. This translated into forest extent beginning to decline earlier and more linearly compared with the CNRM-CM5 RCP 8.5 scenario (Fig. 5). When the CNRM-CM5 and GFDL-EM2M GCMs were forced with RCP 4.5, instead of RCP 8.5, KBDI from 1989 to 2017 did not differ from the RCP 8.5 scenarios (Fig. 2D), but more large fires occurred (Appendix S3). Between 2018 and 2098, KBDI increased more gradually in the RCP 4.5 scenarios (Fig. 2D), resulting in smaller increases in the number of

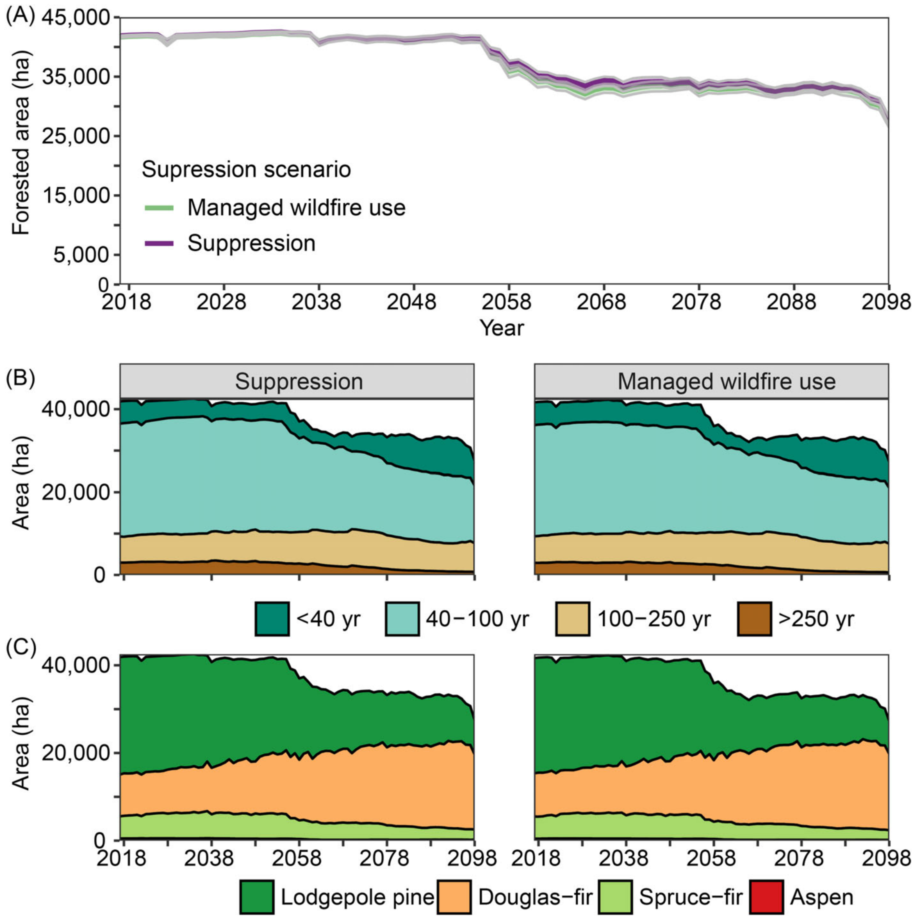

Fig. 4. (A) Forested area (ha) between 2018 and 2098 in suppression and managed wildfire use scenarios. (B) Forested area (ha) that is <40, 40 – 100, 100 – 250, or >250 yr old in suppression and managed wildfire use scenarios. (C) Forested area (ha) dominated by lodgepole pine, Douglas-fir, spruce–fir, or aspen between 2018 and 2098 in suppression and managed wildfire use scenarios. Values in panel A are means and 95% confidence intervals for the 20 replicate simulations of the GCM CNRM-CM5 RCP 8.5 scenario. Values in panels B and C are means for the 20 replicate simulations of the GCM CNRM-CM5 RCP 8.5 scenario.

fires, area burned, and area weighted mean fire size (Appendix S3). Differences in projected future fire activity between RCP scenarios translated into more forest remaining under RCP 4.5 vs. 8.5 for both GCMs (Fig. 5).

Discussion

Results of this study suggest that strategies emphasizing suppression vs. managed wildfire use will do little to alter fire and forest trajectories if climate change (i.e., increasingly severe drought) is the dominant driver of 21st-century fire activity in subalpine forests of the western United States. Under contemporary conditions, fire sizes varied much more in the managed wildfire use scenario than the suppression scenario, fostering landscape heterogeneity. However, hot-dry conditions projected for the middle to end of this century caused simulated fires in GRTE to grow large in all climate and management scenarios. Corresponding with warming and increased fire activity, large forested areas were lost, and tree-species dominance shifted where forest remained. Simulation experiments, such as this one, cannot predict where and when future climate and fire management will affect a particular location. However, our simulations yield useful insights about how complex interactions among climate, fire, forests, and management may emerge in subalpine landscapes. These results quantitatively inform efforts to evaluate fire management strategies in light of expected changes.

21st-century fire and forest trajectories

Fire activity in 21st-century forests of the western United States may exceed the historical range of variability observed over centuries, and possibly millennia (Westerling et al. 2011, Kelly et al. 2013, Kitzberger et al. 2017). Intensifying drought conditions during this century will almost certainly produce many more summers where climate thresholds associated with large high-severity fires are crossed (Westerling et al. 2011, Abatzoglou and

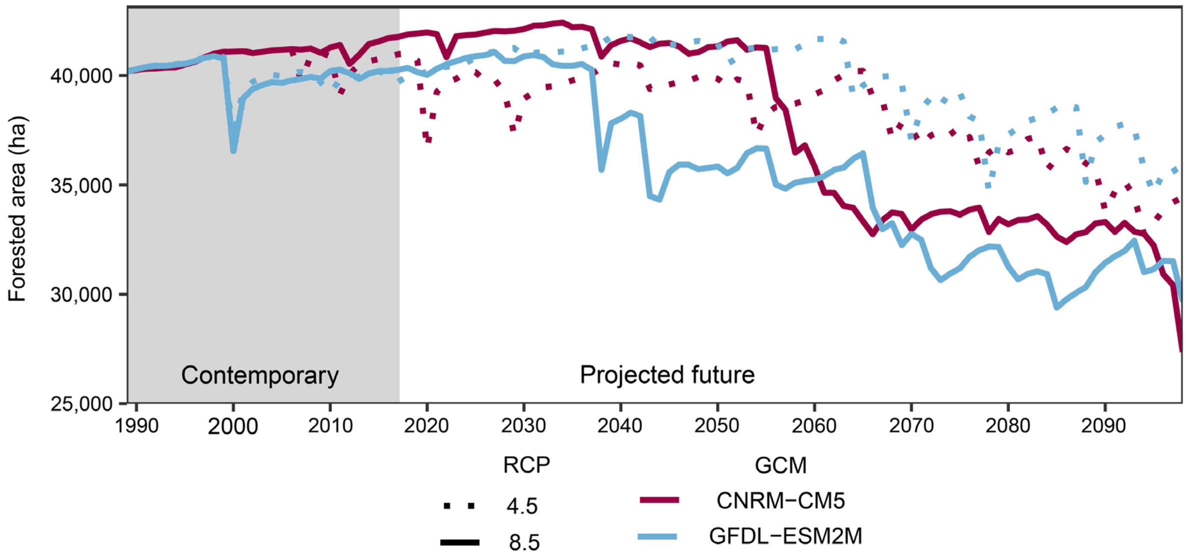

FIG. 5. Forested area (ha) between 1989 and 2098 in suppression and managed wildfire use scenarios when forced with the GCMs CNRM-CM5 and GFDL-EM2M and the RCPs 4.5 and 8.5. Values are means for the 20 replicate simulations of each climate scenario.

Williams 2016). Consistent with expectations (Sommerfeld et al. 2018), we found that future climate was a dominant driver of fire activity, relative to fuels and fire management. Warming-drying trends caused burned area to increase sharply in all climate and fire management scenarios. Average area burned per year was 1,700% greater during the projected future (2018–2098) relative to the contemporary period (1989–2017) in the CNRM-CM5 RCP 8.5 climate scenario. However, key climate forcings (i.e., temperature, precipitation, and vapor pressure deficit) all have their own dynamic patterns of inter-annual and inter-decadal variability layered upon 21st-century trends, and this variability produced distinct patterns of burning (Westerling et al. 2006, Westerling 2016, Keyser and Westerling 2017, Littell et al. 2018) among scenarios. For example, in the CNRM-CM5 RCP 8.5 scenario, a strong multi-decade ocean circulation event in the late 2050s and early 2060s led to drier conditions in the northern Rockies than would be expected from anthropogenic climate change alone (Fig. 2D). This mid-21st century dry excursion caused many fires to escape suppression and grow large (Fig. 3). In the following decades, moisture increased, and fire activity declined, as conditions again became more conducive to suppression. This highlights how managers should plan for effects of future climate variability on fire, in addition to mean climate trends.

It appears that 21st-century conditions may be sufficient to erode subalpine forest resilience, or the capacity of a system to recover from disturbance while retaining structure, functions, and feedbacks (Walker et al. 2006). Forest structure and extent changed profoundly in all simulations with 21st-century climate and fire. By 2098 in the CNRM-CM5 RCP 8.5 climate scenario, for example, 35% of forested area abruptly converted to nonforest (Fig. 4A), which was initiated by increased fire activity during the mid-21st-century dry excursion. Species dominance and stand-age distributions shifted in remaining forests with drought-tolerant Douglas-fir replacing lodgepole pine, and young forests also expanding. Studies have demonstrated that fire and climate, consistent with 21st-century projections, can be powerful forces for change in forests that experience large severe fires (Enright et al. 2014, 2015, Harvey et al. 2016*c*, Johnstone et al. 2016, Hansen et al. 2018, Keane et al. 2018, Serra-Diaz et al. 2018, Stevens-Rumann et al. 2018, Buotte et al. 2019). For example, conversion to nonforest can occur following unusually large or short-interval burns that reduce seed supply and constrain postfire tree regeneration (Brown and Johnstone 2012, Kemp et al. 2016, Hansen et al. 2018, Turner et al. 2019). Even if sufficient seed is available, drought in the first few growing seasons following fire can kill tree seedlings (Walck et al. 2011, Clark et al. 2016, Harvey et al. 2016*b*, Hansen and Turner 2019, Kemp et al. 2019, Davis et al. 2019).

Managed wildfire use in current and future subalpine forests

Whether managed wildland fire use can reduce subsequent fire occurrence, size, and severity hinges on the premise that fire is self-limiting (Hurteau et al. 2019). In other words, recently burned areas have insufficient fuel to carry another fire (Parks et al. 2015, Riley et al. 2018). In some locations, retrospective studies provide compelling evidence for fire's self-limiting nature. Fire occurrence, size, and severity were reduced by previous burns across numerous analyses of recent fires in the western United States (Parks et al. 2013, 2015, 2018*a*, Harvey et al. 2016*a*). For example, in three large northern Rocky Mountain wilderness landscapes, the probability of reburning was low for 15 to 33 yr following the first fire (Parks et al. 2018*b*). However, the length and strength of negative feedbacks varies with weather and forest type. For example, under unusually hot-dry conditions, the length of fire's self-limiting effect was reduced by as much as 46% (from 28 to 15 yr) in the same three wilderness landscapes (Parks et al. 2018*b*). In subalpine forests of the northern Rockies, severity of reburns was actually higher than long-interval fires when more than 12 yr had passed between sequential fires (Harvey et al. 2016*a*).

In our simulations, we found no evidence that managed wildfire use meaningfully affected subsequent fire in a subalpine landscape under contemporary or future conditions. Simulated increases in burned area did alter forest structure and extent, but remaining forests accumulated fuel quickly after fire and stayed sufficiently connected for fires to spread, even without spotting (burning material that travels long distances in the wind and causes new ignitions), a process that is not represented in the model. These results are consistent with recent empirical analyses, which show young postfire forests that burned in 1988 near GRTE rapidly accumulated sufficient fuels to burn again as active crown fire, even under moderate weather (Nelson et al. 2016, 2017). It is also well in line with recently observed fire behavior. In 2016, the Berry and Maple fires in and around GRTE burned almost 18,000 ha of early postfire subalpine forest (16 and 28 yr old; Turner et al. 2019); some stands as young as 7 yr old also burned during the 1988 Yellowstone fires.

While managed wildfire use had little effect on broad-scale patterns of burning during the contemporary or future projected periods, it could still be an important strategy. Fires of intermediate size were more common when simulated aridity was moderate, particularly during the contemporary period when fire activity was generally low. Allowing fires to burn during the contemporary period fostered landscape heterogeneity and created early postfire environments within otherwise contiguous mature forest, which dampens the risk for other disturbances such as bark beetles (Seidl et al. 2016) and provides critical habitat for a variety of species, including cavity nesting bird species (Taylor 1973, Hutto 1995), granivores (e.g., Frock and Turner 2018), and ungulates (e.g., Mao et al. 2005). Further, managed wildfire use may influence fire behavior (e.g., reducing fire intensity), which was not considered in this study, and could have important ecological consequences. For example, reduced fire intensity could foster greater tree regeneration under a managed wildfire use strategy.

Our approach was limited in a few ways. Large fire occurrence and suppression effectiveness were both linked to deterministic climate thresholds. Thus, warming climate and increased drought inherently led to larger fires and lower suppression effectiveness. During the contemporary period, our approach reasonably represented current fire and management patterns (Appendix S1). However, in the future, it assumes that the relationships between climate, fire activity, and suppression effectiveness do not change. This is consistent with other studies that model future fire (e.g., Westerling et al. 2011), but adds uncertainty to the simulated trajectories. First, under future climate, controls on fire spread could diverge from empirical relationships that are based on past observation. The physical multi-scale processes that shape fire spread and size are notoriously complex, and forest landscape models cannot yet adequately capture these emergent dynamics (Finney et al. 2015, Coen et al. 2018). This is an area of active current research. Second, fire management strategies, such as suppression, could become more effective in the future if technology advances or more resources are invested in firefighting. However, dynamically modeling technological innovation and government resource allocation was well beyond our scope. Finally, our simulations do not account for increases in atmospheric CO_2 concentrations, which could cause tree biomass to recover more quickly after fire, if tree seedlings can establish and survive, and if water and nitrogen do not limit primary production (Dusenge et al. 2019). Within the context of these assumptions, our simulations provide valuable insights into how fire management strategies may influence 21st-century climate-fire trajectories in subalpine forests of the western United States.

Management Implications

Key objectives of federal fire management in the western United States include (1) ensuring people and property are protected from wildfire, and (2) maintaining natural fire regimes and fostering mosaics of forest structure, stand age, and tree species composition. The key finding of this paper is that managed wildfire use may foster landscape heterogeneity, particularly under current conditions. However, the strategy will likely not maintain fire within its historical range of variability by ameliorating profound future increases in burned area, driven by climate change. Future changes in the fire regime will almost certainly interact with climate to transform subalpine forests. This is in distinct contrast to low-elevation dry conifer forests across the Rocky Mountains and mixed conifer forests of California, where managed wildfire use will remain a critically important strategy for maintaining frequent, low-severity, surface fire regimes.

While managed wildfire use may still help to meet some resource management objectives in subalpine forests during the next few years to decades, our results also suggest that as climate and fire baselines shift, managers will continue to have substantial flexibility to use suppression (when weather allows) without altering forest dynamics over the long term (Noss et al. 2006, Halofsky et al. 2018). Strategic use of suppression in subalpine forests might be particularly valuable for protecting high-value assets (e.g., buildings and infrastructure) or maintaining essential ecosystem processes and attributes (e.g., old growth forest, wildlife habitat). Exploring the effectiveness of fuels treatments around these social and ecological resources to enhance fire fighter safety and reduce resistance of fires to suppression efforts is also warranted.

Acknowledgments

W. D. Hansen, M. G. Turner, and D. Abendroth designed the study. W. D. Hansen parameterized the model, performed simulations, and conducted analyses. W. D. Hansen and M. G. Turner wrote the paper, and all coauthors contributed to revising the paper. This study benefitted from numerous discussions with forest and fire managers in the Northern Rocky Mountains, including at the 2016 and 2018 Yellowstone Biennial Science Conferences, and workshops and field tours facilitated by the Northern Rockies Fire Science Network. We are grateful to all participants for their suggestions and insights, and to Vita Wright, Anne Black, and Linda Mutch for helping to coordinate these activities. We thank Stephen Carpenter, Kate McCulloh, Anthony Ives, Erika Marín-Spiotta, Kristin Braziunas, Park Williams, and Andy Norman for helpful comments on an earlier version of this manuscript. We acknowledge funding from the National Park Service Fuels Reserve Fund (National Park Service Task Agreement P17AX01470), an NSF Graduate Research Fellowship (DGE-1242789), postdoctoral fellowship from the Earth Institute at Columbia University, and funding from Columbia University's Center for Climate and Life to W. D. Hansen, the Joint Fire Science Program (16-3-01-4), the University of Wisconsin Vilas Trust, and a UW2020 award from the Wisconsin Alumni Research Foundation. R. Seidl and W. Rammer further acknowledge support from the Austrian Science Fund through START grant Y895-B25.

Literature Cited

Abatzoglou, J. T., and T. J. Brown. 2012. A comparison of statistical downscaling methods suited for wildfire applications. International Journal of Climatology 32:772–780.

Abatzoglou, J. T., and A. P. Williams. 2016. Impact of anthropogenic climate change on wildfire across western US forests. Proceedings of the National Academy of Sciences USA 113:11770–11775.

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications 12:1418–1433.

Arno, S. F., and G. E. Gruell. 1983. Fire history at the forest-grassland ecotone in southwestern Montana. Journal of Range Management 36:332–336.

Baker, W. L. 2009. Fire ecology in rocky mountain landscapes. Island Press, Washington, D.C., USA.

Bentley, P. D., and T. D. Penman. 2017. Is there an inherent conflict in managing fire for people and conservation? International Journal of Wildland Fire 26:455–468.

Bessie, W. C., and E. A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology 76:747–762.

Braziunas, K. H., W. D. Hansen, R. Seidl, R. W. Rammer, and M. G. Turner. 2018. Looking beyond the mean: drivers of variability in postfire stand development of conifers in Greater Yellowstone. Forest Ecology and Management 430:460–471.

Brown, J. K. 1991. Should management ignitions be used in Yellowstone National Park? Pages 137–148 *in* R. B. Keiter and M. S. Boyce, editors. The greater Yellowstone ecosystem. Yale University Press, New Haven, Connecticut, USA.

Brown, C. D., and J. F. Johnstone. 2012. Once burned, twice shy: repeat fires reduce seed availability and alter substrate constraints on *Picea mariana* regeneration. Forest Ecology and Management 266:34–41.

Buotte, P. C., S. Levis, B. E. Law, T. W. Hudiburg, D. E. Rupp, and J. J. Kent. 2019. Near-future forest vulnerability to drought and fire varies across the western United States. Global Change Biology 25:290–303.

Calkin, D. E., J. D. Cohen, M. A. Finney, and M. P. Thompson. 2013. How risk management can prevent future wildfire disasters in the wildland-urban interface. Proceedings of the National Academy of Sciences USA 111:746–751.

Chapin, F. S., S. F. Trainor, O. Huntington, A. L. Lovecraft, E. Zavaleta, D. C. Natcher, A. D. McGuire, J. L. Nelson, L. Ray, and M. Calef. 2008. Increasing wildfire in Alaska's boreal forest: pathways to potential solutions of a wicked problem. BioScience 58:531–540.

Christensen, N. L., et al. 1989. Interpreting the Yellowstone fires of 1988. BioScience 39:678–685.

Clark, J. S., et al. 2016. The impacts of increasing drought on forest dynamics, structure, and biodiversity. Global Change Biology 22:2329–2352.

Coen, J. L., E. N. Stavros, and J. A. Fites-Kaufman. 2018. Deconstructing the King megafire. Ecological Applications 28:1565–1580.

Davis, K. T., S. Z. Dobrowski, P. E. Higuera, Z. A. Holden, T. T. Veblen, M. T. Rother, S. A. Parks, A. Sala, and M. P. Maneta. 2019. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proceedings of the National Academy of Sciences USA 116:6193–6198.

Despain, D. G., and R. E. Sellers. 1977. Natural fire in Yellowstone National Park. Western Wildlands 4:20–24.

Doyle, K., D. Knight, D. Taylor, W. Barmore, and J. Benedict. 1998. Seventeen years of forest succession following the Waterfalls Canyon Fire in Grand Teton National Park, Wyoming. International Journal of Wildland Fire 8:45.

Dunne, J. P., et al. 2012. GFDL's ESM2 global coupled climate–carbon earth system models. Part I: physical formulation and baseline simulation characteristics. Journal of Climate 25:6646–6665.

Dunne, J. P., et al. 2013. GFDL's ESM2 global coupled climate–carbon earth system models. Part II: carbon system formulation and baseline simulation characteristics. Journal of Climate 26:2247–2267.

Dusenge, M. E., A. G. Duarte, and D. A. Way. 2019. Plant carbon metabolism and climate change: elevated CO_2 and temperature impacts on photosynthesis, photorespiration, and respiration. New Phytologist 221:32–49.

Enright, N. J., J. B. Fontaine, B. B. Lamont, B. P. Miller, and V. C. Westcott. 2014. Resistance and resilience to changing climate and fire regime depend on plant functional traits. Journal of Ecology 102:1572–1581.

Enright, N. J., J. B. Fontaine, D. M. Bowman, R. A. Bradstock, and R. J. Williams. 2015. Interval squeeze: altered fire regimes and demographic responses interact to threaten woody species persistence as climate changes. Frontiers in Ecology and the Environment 13:265–272.

Finney, M. A., J. D. Cohen, J. M. Forthofer, S. S. McAllister, M. J. Gollner, D. J. Gorham, K. Saito, N. K. Akafuah, B. A.

Adam, and J. D. English. 2015. Role of buoyant flame dynamics in wildfire spread. Proceedings of the National Academy of Sciences USA 112:9833–9838.

Frock, C. F., and M. G. Turner. 2018. Microhabitat conditions and landscape pattern explain nocturnal rodent activity, but not seed removal, in burned and unburned lodgepole pine forests. Landscape Ecology 33:1895–1909.

Halofsky, J. S., D. C. Donato, J. F. Franklin, J. E. Halofsky, D. L. Peterson, and B. J. Harvey. 2018. The nature of the beast: examining climate adaptation options in forests with stand-replacing fire regimes. Ecosphere 9:e02140.

Hansen, W. D., and M. G. Turner. 2019. Origins of abrupt change? Postfire subalpine conifer regeneration declines nonlinearly with warming and drying. Ecological Monographs 89:e01340.

Hansen, W. D., K. H. Braziunas, W. Rammer, R. Seidl, and M. G. Turner. 2018. It takes a few to tango: changing climate and fire regimes can cause regeneration failure of two subalpine conifers. Ecology 9:966–977.

Harvey, B. J., D. C. Donato, and M. G. Turner. 2016*a*. Drivers and trends in landscape patterns of stand-replacing fire in forests of the US Northern Rocky Mountains (1984-2010). Landscape Ecology 31:2367–2383.

Harvey, B. J., D. C. Donato, and M. G. Turner. 2016*b*. Burn me twice, shame on who? Interactions between successive forest fires across a temperate mountain region. Ecology 97:2272–2282.

Harvey, B. J., D. C. Donato, and M. G. Turner. 2016*c*. High and dry: post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches. Global Ecology and Biogeography 25:655–669.

Houston, D. G. 1973. Wildfires in northern Yellowstone National Park. Ecology 54:1111–1117.

Hurteau, M. D., S. Liang, A. L. Westerling, and C. Wiedinmyer. 2019. Vegetation-fire feedback reduces projected area burned under climate change. Scientific Reports 9:2838.

Hutto, R. L. 1995. Composition of bird communities following stand-replacement fires in Northern Rocky Mountain (U.S.A.) conifer forests. Conservation Biology 9:1041–1058.

Ingalsbee, T. 2017. Whither the paradigm shift? Large wildland fires and the wildfire paradox offer opportunities for a new paradigm of ecological fire management. International Journal of Wildland Fire 26:557–561.

Jacobs, K., and C. Whitlock. 2008. A 2000-year environmental history of Jackson Hole, Wyoming, inferred from lake-sediment records. Western North American Naturalist 68:350–364.

Jensen, S. E., and G. R. McPherson. 2008. Living with fire. Fire ecology and policy for the twenty-first century. University of California Press, Berkeley, California, USA.

Johnstone, J. F., et al. 2016. Changing disturbance regimes, ecological memory, and forest resilience. Frontiers in Ecology and the Environment 14:369–378.

Keane, R. E., J. K. Agee, P. Fulé, J. E. Keeley, C. Key, S. G. Kitchen, R. Miller, and L. A. Schulte. 2008. Ecological effects of large fires on US landscapes: benefit or catastrophe? International Journal of Wildland Fire 17:696.

Keane, R. E., R. A. Loehman, and L. M. Holsinger. 2011. The FireBGCv2 Landscape Fire Succession Model. General Technical Report RMRS-GTR-255. USDA Forest Service Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Keane, R. E., R. A. Loehman, L. M. Holsinger, D. A. Falk, P. Higuera, S. M. Hood, and P. F. Hessburg. 2018. Use of landscape simulation modeling to quantify resilience for ecological applications. Ecosphere 9:e02414.

Keane, R. E., K. Gray, B. Davis, L. M. Holsinger, and R. Loehman. 2019. Evaluating ecological resilience across suppression levels under climate and fuel treatment scenarios using landscape simulation modeling. International Journal of Wildland Fire 28:533–549.

Kelly, R., M. L. Chipman, P. E. Higuera, I. Stefanova, L. B. Brubaker, and F. S. Hu. 2013. Recent burning of boreal forests exceeds fire regime limits of the past 10,000 years. Proceedings of the National Academy of Sciences USA 110:13055–13060.

Kemp, K. B., P. E. Higuera, and P. Morgan. 2016. Fire legacies impact conifer regeneration across environmental gradients in the U.S. northern Rockies. Landscape Ecology 31:619–636.

Kemp, K. B., P. E. Higuera, P. Morgan, and J. T. Abatzoglou. 2019. Climate will increasingly determine post-fire tree regeneration success in low-elevation forests, Northern Rockies. USA. Ecosphere 10:e02568.

Keyser, A., and A. L. Westerling. 2017. Climate drives interannual variability in probability of high severity fire occurrence in the western United States. Environmental Research Letters 12:065003.

Keyser, A. R., and A. L. Westerling. 2019. Predicting increasing high severity area burned for three forested regions in the western United States using extreme value theory. Forest Ecology and Management 432:694–706.

Kitzberger, T., D. A. Falk, A. L. Westerling, and T. W. Swetnam. 2017. Direct and indirect climate controls predict heterogeneous early-mid 21st century wildfire burned area across western and boreal North America. PLoS ONE 12: e0188486.

Knight, D. H. 1991. The Yellowstone fire controversy. Pages 87–103 *in* R. B. Keiter and M. S. Boyce, editors. The greater Yellowstone ecosystem. Yale University Press, New Haven, Connecticut, USA.

Knight, D., G. P. Jones, W. A. Reiners, and W. H. Romme. 2014. Mountains and plains: the ecology of Wyoming landscapes. Yale University Press, New Haven, Connecticut, USA.

Krofcheck, D. J., M. D. Hurteau, R. M. Scheller, and E. L. Loudermilk. 2018. Prioritizing forest fuels treatments based on the probability of high-severity fire restores adaptive capacity in Sierran forests. Global Change Biology 24:729–737.

Liang, S., M. D. Hurteau, and A. L. Westerling. 2017. Response of Sierra Nevada forests to projected climate-wildfire interactions. Global Change Biology 23:2016–2030.

Littell, J. S., D. McKenzie, D. L. Peterson, and A. L. Westerling. 2009. Climate and wildfire area burned in western US ecoprovinces, 1916–2003. Ecological Applications 19:1003–1021.

Littell, J. S., D. McKenzie, H. Y. Wan, and S. A. Cushman. 2018. Climate change and future wildfire in the western United States: an ecological approach to nonstationarity. Earth's Future 6:1097–1111.

Loehman, R. A., et al. 2018. Effects of climate change on ecological disturbance in the northern rockies. Pages 115141 in J. E. Halofsky and D. L. Peterson, editors. Climate Change and Rocky Mountain Ecosystems. Springer, New York, New York, USA.

Mao, J. S., M. S. Boyce, D. W. Smith, F. J. Singer, D. J. Vales, J. M. Vore, and H. Evelyn. 2005. Habitat selection by elk before and after wolf reintroduction in Yellowstone National Park. Journal of Wildlife Management 69:1691–1707.

Marlon, J. R., et al. 2012. Long-term perspective on wildfires in the western USA. Proceedings of the National Academy of Sciences 109:E535–E543.

Martínez-Vilalta, J., and F. Lloret. 2016. Drought-induced vegetation shifts in terrestrial ecosystems: the key role of regeneration dynamics. Global and Planetary Change 144:94–108.

Millspaugh, S. H., C. Whitlock, and P. J. Bartlein. 2000. Variations in fire frequency and climate over the past 17 000 yr in central Yellowstone National Park. Geology 28:211–214.

Moritz, M. A., et al. 2014. Learning to coexist with wildfire. Nature 515:58–66.

National Academy of Sciences, Engineering, and Medicine. 2017. A century of wildland fire research. Contributions to long-term approaches for wildland fire management. Proceedings of a workshop. http://dels.nas.edu/Workshop-Summary/Century-Wildland-Fire-Research/24792?bname=besr

National Interagency Fire Center. 2018. Historical wildland fire information: suppression costs. https://www.nifc.gov/fireInfo/fireInfo_documents/SuppCosts.pdf

National Park Service. 2019. Wildland fire management reference manual. https://www.nps.gov/subjects/fire/upload/nps-reference-manual-18.pdf

National Strategic Committee. 2014. The final phase in the development of the National Cohesive Wildland Fire Management Strategy. https://www.forestsandrangelands.gov/documents/strategy/reports/cohesive_strategy_crosswalk_and_strategic_alignment_report.pdf

National Wildfire Coordinating Group. 2009. Guidance for implementation of federal wildland fire management policy. https://www.nifc.gov/policies/policies_documents/GIFWFMP.pdf

Nelson, K. N., M. G. Turner, W. H. Romme, and D. B. Tinker. 2016. Landscape variation in tree regeneration and snag fall drive fuel loads in 24-year old post-fire lodgepole pine forests. Ecological Applications 26:2424–2438.

Nelson, K. N., M. G. Turner, W. H. Romme, and D. B. Tinker. 2017. Simulated fire behaviour in young, postfire lodgepole pine forests. International Journal of Wildland Fire 26:852–865.

North, M., B. M. Collins, and S. Stephens. 2012. Using fire to increase the scale, benefits, and future maintenance of fuel treatments. Journal of Forestry 110:392–401.

Noss, R. F., J. F. Franklin, W. L. Baker, T. Schoennagel, and P. B. Moyle. 2006. Managing fire-prone forests in the western United States. Frontiers in Ecology and the Environment 4:481–487.

Parks, S. A., C. Miller, C. R. Nelson, and Z. A. Holden. 2013. Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. Ecosystems 17:29–42.

Parks, S. A., M. A. Parisien, C. Miller, and S. Z. Dobrowski. 2014. Fire activity and severity in the western US vary along proxy gradients representing fuel amount and fuel moisture. PLoS ONE 9:e99699.

Parks, S. A., L. M. Holsinger, C. Miller, and C. R. Nelson. 2015. Wildland fire as a self-regulating mechanism: the role of previous burns and weather in limiting fire progression. Ecological Applications 25:1478–1492.

Parks, S. A., C. Miller, J. T. Abatzoglou, L. M. Holsinger, M. A. Parisien, and S. Z. Dobrowski. 2016. How will climate change affect wildland fire severity in the western US? Environmental Research Letters 11:035002.

Parks, S. A., L. Holsinger, M. Panunto, W. M. Jolly, S. Dobrowski, and G. Dillon. 2018*a*. High severity fire: evaluating its key drivers and mapping its probability across western US forests. Environmental Research Letters 13:044037.

Parks, S. A., M. A. Parisien, C. Miller, L. M. Holsinger, and L. S. Baggett. 2018*b*. Fine-scale spatial climate variation and drought mediate the likelihood of reburning. Ecological Applications 28:573–586.

Parks, S. A., S. Dobrowski, and M. Panunto. 2018*c*. What drives low-severity fire in the southwestern USA? Forests 9:165.

Power, M. J., C. Whitlock, and P. J. Bartlein. 2011. Postglacial fire, vegetation, and climate history across an elevational gradient in the Northern Rocky Mountains, USA and Canada. Quaternary Science Reviews 30:2520–2533.

Prichard, S. J., C. S. Stevens-Rumann, and P. F. Hessburg. 2017. Tamm Review: shifting global fire regimes: lessons from reburns and research needs. Forest Ecology and Management 396:217–233.

Riley, K., M. Thompson, J. Scott, J. Gilbertson-Day, K. L. Riley, M. P. Thompson, J. H. Scott, and J. W. Gilbertson-Day. 2018. A model-based framework to evaluate alternative wildfire suppression strategies. Resources 7:4.

Romme, W. H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecological Monographs 52:199–221.

Savage, M., and J. N. Mast. 2005. How resilient are southwestern ponderosa pine forests after crown fires? Canadian Journal of Forest Research 35:967–977.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54:393–402.

Schoennagel, T., et al. 2017. Adapt to more wildfire in western North American forests as climate changes. Proceedings of the National Academy of Sciences USA 114:4582–4590.

Seidl, R., and W. Rammer. 2019. iLand wildfire module online documentation. http://iland.boku.ac.at/wildfire?highlight=wildfire

Seidl, R., W. Rammer, R. M. Scheller, and T. A. Spies. 2012. An individual-based process model to simulate landscape-scale forest ecosystem dynamics. Ecological Modelling 231:87–100.

Seidl, R., W. Rammer, and T. A. Spies. 2014. Disturbance legacies increase the resilience of forest ecosystem structure, composition, and functioning. Ecological Applications 24:2063–2077.

Seidl, R., D. C. Donato, K. F. Raffa, and M. G. Turner. 2016. Spatial variability in tree regeneration after wildfire delays and dampens future bark beetle outbreaks. Proceedings of the National Academy of Sciences USA 113:13075–13080.

Seidl, R., K. Albrich, K. Erb, H. Formayer, D. Leidinger, G. Leitinger, U. Tappeiner, E. Tasser, and W. Rammer. 2019. What drives the future supply of regulating ecosystem services in a mountain forest landscape? Forest Ecology and Management 445:37–47.

Serra-Diaz, J. M., C. Maxwell, M. S. Lucash, R. M. Scheller, D. M. Laflower, A. D. Miller, A. J. Tepley, H. E. Epstein, K. J. Anderson-Teixeira, and J. R. Thompson. 2018. Disequilibrium of fire-prone forests sets the stage for a rapid decline in conifer dominance during the 21st century. Scientific Reports 8:6749.

Sommerfeld, A. C., et al. 2018. Patterns and drivers of recent disturbances across the temperate forest biome. Nature Communications 9:4355.

Steelman, T. 2016. U.S. wildfire governance as a social-ecological problem. Ecology and Society 21:3.

Stephens, S. L., J. K. Agee, P. Z. Fulé, M. P. North, W. H. Romme, T. W. Swetnam, and M. G. Turner. 2013. Managing forests and fire in changing climates. Science 342:41–42.

Stevens-Rumann, C. S., K. B. Kemp, P. E. Higuera, B. J. Harvey, M. T. Rother, D. C. Donato, P. Morgan, and T. T. Veblen. 2018. Evidence for declining forest resilience to wildfires under climate change. Ecology Letters 21:243–252.

Taylor, D. L. 1973. Some ecological implications of forest fire control in Yellowstone National Park, Wyoming. Ecology 54:1394–1396.

Thomas, D., D. Butry, S. Gilbert, D. Webb, and J. Fung. 2017. The Costs and Losses of Wildfires. NIST Special Publication 1215. National Institute of Standards and Technology. U.S.

Department of Commerce. https://nvlpubs.nist.gov/nistpubs/SpecialPublications/NIST.SP.1215.pdf

Thompson, M. P., F. R. Silva, D. E. Calkin, and M. S. Hand. 2017. A review of challenges to determining and demonstrating efficiency of large fire management. International Journal of Wildland Fire 26:562–573.

Thompson, M. P., D. G. MacGregor, C. J. Dunn, D. E. Calkin, and J. Phipps. 2018. Rethinking the wildland fire management system. Journal of Forestry 116:382–390.

Turner, M. G., and W. H. Romme. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology 9:59–77.

Turner, M. G., W. W. Hargrove, R. H. Gardner, and W. H. Romme. 1994. Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. Journal of Vegetation Science 5:731–742.

Turner, M. G., W. H. Romme, R. H. Gardner, and W. W. Hargrove. 1997. Effects of fire size and pattern on early succession in Yellowstone National Park. Ecological Monographs 67:411–433.

Turner, M. G., D. Tinker, W. Romme, D. Kashian, and C. Litton. 2004. Landscape patterns of sapling density, leaf area, and aboveground net primary production in postfire lodgepole pine forests, Yellowstone National Park (USA). Ecosystems 7:751–775.

Turner, M. G., T. G. Whitby, D. B. Tinker, and W. H. Romme. 2016. Twenty-four years after the Yellowstone Fires: are postfire lodgepole pine stands converging in structure and function? Ecology 97:1260–1273.

Turner, M. G., K. B. Braziunas, W. D. Hansen, and B. J. Harvey. 2019. Short-interval fires erode the resilience of subalpine lodgepole pine forests. Proceedings of the National Academy of Sciences USA 116:11319–11328.

Voldoire, A., et al. 2013. The CNRM-CM5.1 global climate model: description and basic evaluation. Climate Dynamics 40:2091–2121.

Walck, J. L., S. N. Hidayati, K. W. Dixon, K. Thompson, and P. Poschlod. 2011. Climate change and plant regeneration from seed. Global Change Biology 17:2145–2161.

Walker, B., L. Gunderson, A. Zinzig, C. Folke, S. Carpenter, and L. Schultz. 2006. A handful of heuristics and some propositions for understanding resilience in social-ecological systems. Ecology and Society 11:13.

Walker, R. B., J. D. Coop, S. A. Parks, and L. Trader. 2018. Fire regimes approaching historic norms reduce wildfire-facilitated conversion from forest to non-forest. Ecosphere 9:e02182.

Westerling, A. L. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B: Biological Sciences 371:20150178.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313:940–943.

Westerling, A. L., M. G. Turner, E. A. H. Smithwick, W. H. Romme, and M. G. Ryan. 2011. Continued warming could transform Greater Yellowstone fire regimes by mid-21st century. Proceedings of the National Academy of Sciences USA 108:13165–13170.

Western Regional Climate Center. 2018. NCDC 1958–2016 monthly normals, Moose, WY. https://wrcc.dri.edu/cgi-bin/cliMAIN.pl?wy6428

Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.2030/full

Data Availability

Data are available on the EDI Data Portal: https://doi.org/10.6073/pasta/0107f11a3fe019de1d61fdfe88d72118.