Managing fire-prone forests in a time of decreasing carbon carrying capacity

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Changing climatic conditions are increasing overstory tree mortality in forests globally. This restructuring of the distribution of biomass is making already flammable forests more combustible, posing a major challenge for managing the transition to a lower biomass state. In western US dry conifer forests, tree density resulting from over a century of fire-exclusion practices has increased the risk of high-severity wildfire and susceptibility to climate-driven mortality. Reducing dead fuel loads will require new approaches to mitigate risk to the remaining live trees by preparing forests to withstand future wildfire. Here, we used data from the Teakettle Experimental Forest in California to evaluate different prescribed fire burn frequencies and their impact on accumulated dead fuels after a 4-year drought. Increasing burn frequency could reduce surface fuel build-up but comes with additional challenges that will require creativity and experimentation to overcome.

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Ongoing climate change and climate-driven increases in disturbance are reshaping forests globally. Whether directly through hot droughts (the simultaneous occurrence of drought and extreme heat) or indirectly through increased susceptibility to insects, pathogens, and fire, forest biomass is being reorganized (Williams *et al*. 2013; Kolb *et al*. 2016). As forests respond to changing climate and disturbance, stepwise changes in carbon (C) carrying capacity—the amount of live biomass that can be supported under prevailing climate and natural disturbance conditions (Keith *et al*. 2009)—mean that increasingly large fractions of biomass are dead. While this decrease in live tree biomass and its associated leaf area can leave the remaining trees more drought tolerant because of reduced competition, the surplus of dead biomass heightens forest susceptibility to subsequent stand-replacing wildfire (Goodwin *et al*. 2021; Stephens *et al*. 2022). Fuel- and climate-mediated increases in fire intensity across large landscapes can further erode forest resilience, increasing the potential for transition to a non-forest vegetation type (Falk *et al*. 2022). Managing these climate-driven reductions in forest C carrying capacity is central to sustaining forest area, especially in increasingly flammable forest ecosystems.

In a nutshell:

- Climate change is restructuring forest biomass through tree mortality and disturbance
- Mortality and fire exclusion have put the remaining live trees at risk of high-severity fire
- Managing fuel loads by increasing burn frequency may be possible, but this approach will require considerable experimental research to identify ecologically appropriate solutions

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Climate and disturbance interact with local site factors to determine the amount of C that can be sustained in a given location, and changing climatic conditions combined with human-caused disturbance is reducing the C carrying capacity of many forests globally (Hammond *et al*. 2022). More specifically, increasing atmospheric aridity, which is driving large-scale mortality events (eg hot droughts, bark beetles), is also increasing forest flammability (Dickman *et al*. 2023). Connectivity of live and dead biomass combined with fuel moisture influences both the potential for fire spread and the proportion of a forested area that is susceptible to fire-induced tree mortality (ie high-severity fire; Juang *et al*. 2022; Francis *et al*. 2023). In some ecosystems, this is primarily a climate-driven phenomenon, whereby acute decreases in the moisture content of live and dead fuel due to severe drought is the principal factor driving flammability (Alizadeh *et al*. 2021). In other ecosystems, such as seasonally dry conifer forests in the western US, a legacy of timber management and fire suppression are interacting with atmospheric dryness to increase forest flammability (Stephens *et al*. 2020). Here, we summarize common historical and current drivers of C dynamics in western US frequent-fire forests. We then use data from several decades of changing forest and fuel conditions in mixed-conifer forests in the Teakettle Experimental Forest (hereafter Teakettle)—a 1300-ha section of the Sierra National Forest in

California's Sierra Nevada that has been set aside by the US Forest Service (USFS) for research on forest ecology—to demonstrate the management challenges posed by declining C carrying capacity.

Historical context

Before European settlement, dry western forests were largely fuel-limited systems, where fires ignited by lightning strikes or Indigenous peoples shaped the distribution of biomass for millennia. Dry western forests range from semi-arid ponderosa pine (*Pinus ponderosa*)-dominated forests in the southwestern US, to Sierra Nevada mixed-conifer forests with a Mediterranean climate in California, to dry mixed-conifer forests composed of Douglas-fir (*Pseudotsuga menziesii*) and ponderosa pine in the inland Pacific Northwest and northern US Rocky Mountains (Hessburg *et al.* 2019). High-frequency fire, typically occurring on the order of several years to several decades (Hurteau *et al.* 2019), maintained heterogenous forest structure and moderated surface fuel loads, reinforcing landscape heterogeneity with each fire (Figure 1; Hagmann *et al.* 2021). However, mid- to late-19th century logging and livestock grazing associated with European settlers, which initially reduced fine fuels that supported frequent fires, was accompanied by the extirpation of Indigenous groups from the landscape who served as important ignition sources (Swetnam *et al.* 2016; Knight *et al.* 2022). Subsequent federal fire-suppression policy combined with these land-use changes allowed for the infilling, homogenization, and fuels accumulation of dry conifer forests (Hagmann *et al.* 2021). Collectively, these factors often pushed the forests past their respective C carrying capacities and increased the instability of the stored C (Goodwin *et al.* 2020).

The challenges associated with fire exclusion were recognized during the early to mid-20th century in some western US systems (Show and Kotok 1924) and investigation into management options to help mitigate the effects across the region began in earnest during the late 20th century (Covington *et al.* 1997; Mutch and Parsons 1998). During the climatically benign 1980s and early 1990s, increased wildfire activity was beginning to capture attention, but fire size was relatively small and severity relatively low (Westerling 2016). Beginning in the mid-1990s in the southwestern US, however, increased biomass density from fire exclusion ran headlong into a drying climate, and as the C carrying capacity decreased, ecosystems responded through tree mortality and increasingly large and severe fires (Allen *et al.* 2015; Westerling 2016). Three decades later, the warmer, drier climate is increasing flammability in many forest systems globally and the interaction with fire exclusion is driving extreme fire behavior in the dry conifer forests of the western US (Kirchmeier-Young *et al.* 2019; Abram *et al.* 2021).

Simultaneous to these climatic changes, forest managers began to implement treatments designed to reduce the effects of what was, by the 1990s, nearly a century of fire exclusion. The research community had evaluated different approaches to reducing tree density and surface fuels in dry western conifer forests and recommended combinations of thinning smaller diameter trees and the reintroduction of fire to achieve hazardous fuels reduction and restore ecosystem function (Agee and Skinner 2005; Stephens *et al.* 2020). Yet the speed and scale of forest restoration activity has been insufficient to keep pace with the ongoing aridification and increasing flammability of these forests (Goodwin *et al.* 2021; North *et al.* 2021; Juang *et al.* 2022). Now, the compound effects of fire exclusion are being exacerbated by increased tree mortality due to climate change (Figure 1).

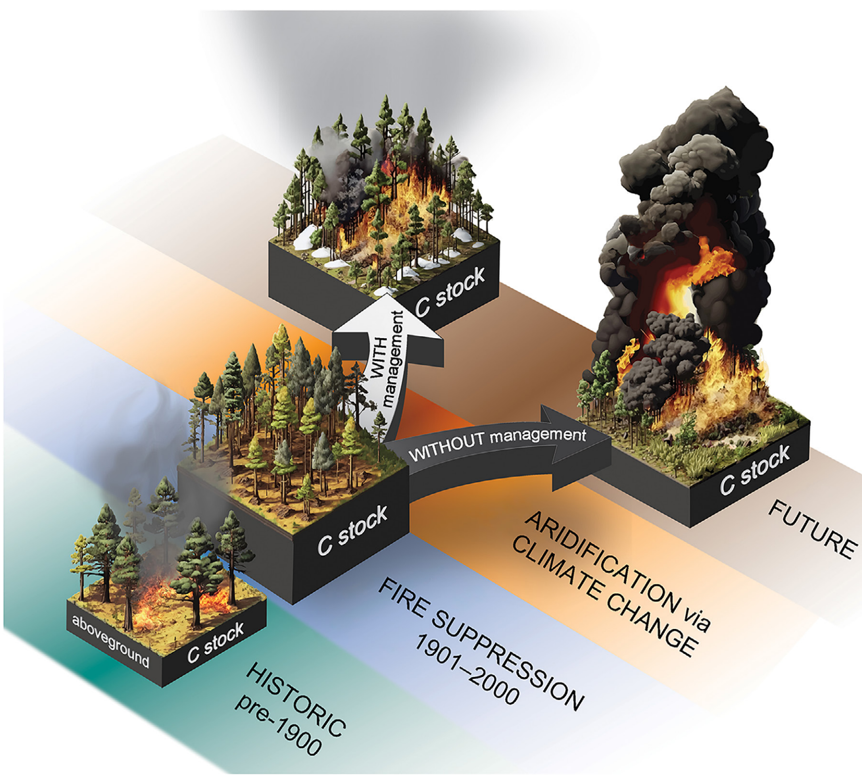

Figure 1. Prior to European settlement, frequent occurrence of fire maintained a more open surface fire-dominated system that had a stable aboveground carbon (C) stock. Fire exclusion practices led to infilling and build-up of surface fuels, both of which increased the aboveground C stock and the potential for high-severity wildfire. Warmer, drier climatic conditions are decreasing the amount of aboveground C that dry forests can support. Fuel-reduction treatments can make forests more resilient to increased flammability and decrease the potential aboveground C loss from type conversion.

An example from the Sierra Nevada

The frequent-fire forests of California's Sierra Nevada are the result of regular fuel accumulation from high productivity, frequent ignitions (lightning strikes, Indigenous burning), and an annual dry season because of the Mediterranean climate (Stephens *et al.* 2020). The large winter snowpack in this system provides sufficient moisture for growth to be maintained through the dry season. Historically, live tree C was aggregated in fewer, larger trees that were resistant to extensive high-severity wildfire (North *et al.* 2009; Harris *et al.* 2019). As with other places throughout the US West, fire exclusion has increased tree density and, in the case of mixed-conifer forest, favored the establishment of shade-tolerant species (Stephens *et al.* 2020). In-growth of shade-tolerant trees and build-up of surface fuels from fire exclusion, combined with hotter droughts and increasing wildfires, suggest Sierran mixed-conifer forests are beyond their C carrying capacity (Stephens *et al.* 2020).

The area burned by wildfire, particularly high-severity fire, has been expanding considerably (Juang *et al.* 2022; Cova *et al.* 2023). In addition, prolonged droughts combined with bark beetle outbreaks have transitioned a substantial proportion of live tree C to the dead tree pool (Goodwin *et al.* 2020). In the southern Sierra Nevada, the 2012–2016 drought, in combination with bark beetle outbreaks, caused as much as 90% overstory tree mortality in some locations (Asner *et al.* 2016). At Teakettle, overstory mortality in fire-excluded forest segments was approximately 30% and concentrated in the largest trees, resulting in a near doubling of dead tree C (Goodwin *et al.* 2020; Steel *et al.* 2021). Beginning in 2021, we sampled 200 plots (where each plot was 0.25 ha) across a 160-ha portion of Teakettle and found dead tree C to be 155 megagrams (Mg) C ha^{-1} (standard deviation [SD] = 102.9 Mg C ha^{-1}), which was nearly as high as live tree C (193 Mg C ha^{-1}, SD = 99.1 Mg C ha^{-1}) (Figure 2). Snags begin to fall about 10–20 years post-mortality and most of this dead tree C transitions to the surface fuel pool (Northrop *et al.* 2024). Forests of the southern Sierra Nevada are currently within that time range. In the space of just a single year (2021–2022), surface fuels in Teakettle increased by 13%. Regardless of whether the dead trees are standing (snags) or on the ground (logs and other coarse fuels), they represent a major source of stored energy that is increasingly available for combustion because of ongoing climate change. As they fall, dead trees present a great fuel risk to the remaining live trees (Stephens *et al.* 2018; Goodwin *et al.* 2021). As the C carrying capacity for this area continues to shift with ongoing climate change, one of the most pressing questions is how to manage surplus dead C in the system to avoid extreme fire behavior and energy release, both of which facilitate undesirable transitions in vegetation state (Figure 1).

To estimate changes in fuel load and the potential to manage the increase in dead fuels with prescribed burning, we used

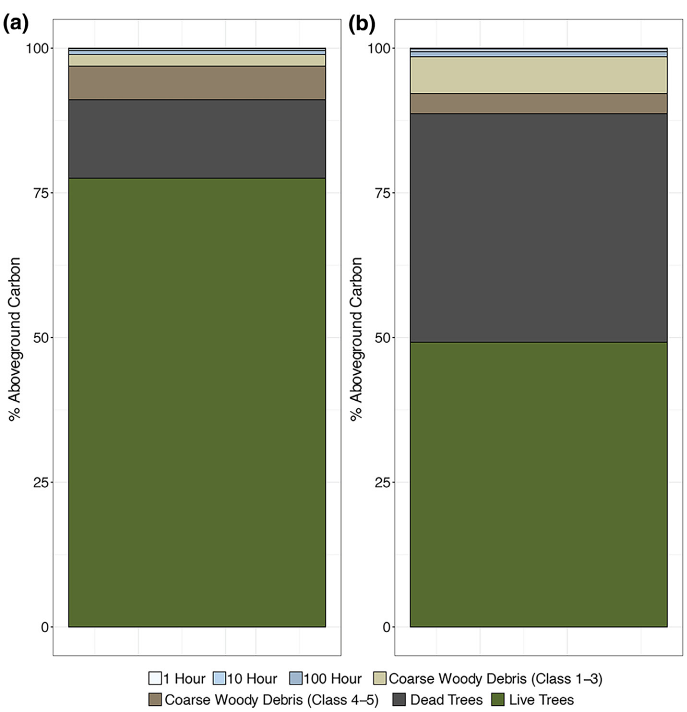

Figure 2. Percent aboveground carbon (C) by pool within an old-growth mixed-conifer forest in the Teakettle Experimental Forest in the southern Sierra Nevada, California. Data were collected in (a) 2011 and (b) 2022 after the 2012–2016 drought, which caused substantial overstory tree mortality among the largest trees. The 2011 and 2022 data were sampled from sites within 500 m of one another. Total aboveground C was 332 and 394 megagrams per hectare (Mg ha^{-1}) in 2011 and 2022, respectively.

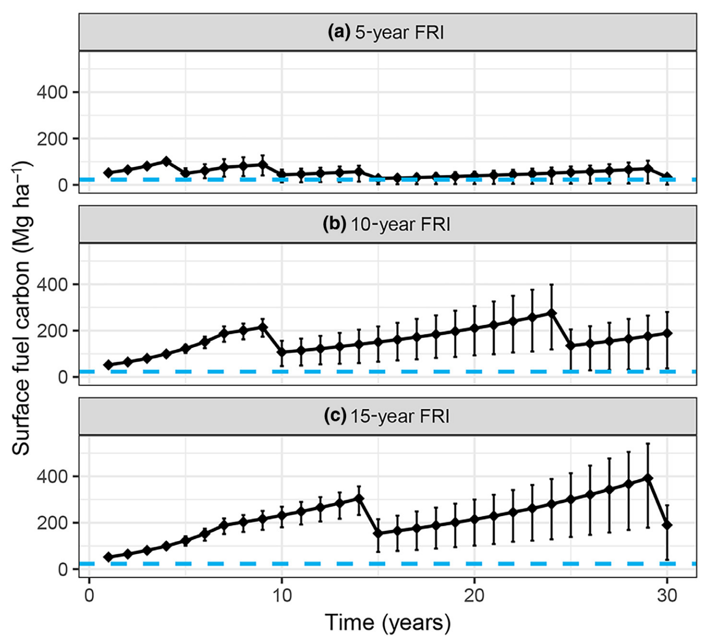

Figure 3. Estimated carbon (C) in surface fuels with three different prescribed fire return intervals (FRIs): (a) 5 years, (b) 10 years, and (c) 15 years. Diamonds are averages and whiskers are upper and lower quartiles from 1000 draws. The dashed horizontal line is the surface fuel C stock (23 Mg C ha^{-1}), above which extreme fire behavior becomes more likely. Monte Carlo simulations used post-drought mortality accumulation for the first 7 years and pre-drought fuel accumulation for the remainder. Fuel consumption from prescribed fire is based on second-entry burn consumption from the Teakettle Experimental Forest, California. Simulations begin with a measured value of 52 Mg C ha^{-1} based on field inventory data from Teakettle.

a Monte Carlo simulation that included distributions for surface fuel accumulation from the period prior to substantial snagfall (2011–2017), surface fuel accumulation for the period of substantial snagfall (2021–2022), and estimates of prescribed fire emissions from the second-entry burn at Teakettle (Goodwin *et al.* 2020). Simulations were performed and plotted in R (v4.3.2) using the *truncnorm* and *ggplot2* packages (Wickham 2016; Mersmann *et al.* 2023; R Core Team 2023). The first 7 years of the simulation sampled from the high snagfall distribution until mean surface fuel increases accounted for the 155 Mg C ha^{-1} mean of dead tree C present in 2022. We simulated prescribed fire at different frequencies, with 23 Mg C ha^{-1} as the post-burn surface fuels target because Stephens *et al.* (2022) determined this was the threshold below which a nearby large wildfire (the 2020 Creek Fire) burned with low- and moderate-severity fire effects. Simulations used typical weather and fuel moisture conditions for prescribing burning in this region.

Given the number of standing dead trees, we assumed substantially higher fuel inputs over the near term (Northrop *et al.* 2024). Maintaining a 15-year fire return interval based on historical frequency meant that surface fuel loads were well above the high-severity threshold determined by Stephens *et al.* (2022) (Figure 3). Increasing the frequency to 10 years was also insufficient to reach the surface fuel target because of the imbalance between fuel inputs and combustive losses. However, when we simulated a 5-year fire return interval for 15 years, combustive losses were sufficient to reduce mean surface fuel C to the target, but mean surface fuels were above the high-severity threshold for much of the next 15 years (Figure 3).

Managing the change

As forests respond to changing climatic conditions and as the C carrying capacity decreases, the role that forests play in climate regulation will diminish, and local challenges and hazards will increase. Teakettle sits within the low- and mid-montane elevation band in California's Sierra Nevada, stretching from the Eldorado National Forest in the central Sierra through the Sequoia National Forest in the southern Sierra, which experienced substantial mortality (Fettig *et al.* 2019). Some of the area has already been subjected to wildfire that burned in such a manner that an operational fire model could not reproduce the extreme rate of spread (Stephens *et al.* 2022). The extreme fire behavior of the 2020 Creek Fire has been attributed to the rare phenomenon of "mass fire" (when a fire generates its own extreme winds), driven by high concentrations of long-burning heavy fuel resulting from the widespread tree mortality 4–6 years prior to the fire (Goodwin *et al.* 2021; Stephens *et al.* 2022). Managing the hazards associated with this extreme fire behavior requires reducing dead tree biomass—a product of uncharacteristic build-up from decades of fire removal interacting with recent climate—in a manner that either breaks down and removes the fuel from the landscape or releases it at a rate that does not further degrade the ecosystem and threaten communities and infrastructure. Physical removal of dead trees at scale is infeasible and a considerable fraction of the landscape is inaccessible to the equipment necessary for removal (North *et al.* 2015). Thus, prescribed fire and managed wildfire are the only tools available for use in many parts of the forest landscape (Figure 4).

Prior estimates of the rates of forest treatments to reduce hazardous fuels across the Sierra Nevada found their pace and scale to be woefully inadequate given the amount of fire that occurred historically in these forests (North *et al.* 2021). Recent federal (HR 5376–Inflation Reduction Act) and state (California–Roadmap to a Million Acres) investments and the USFS Wildfire Crisis Strategy seek to increase the area treated (USFS 2022). If fuel loads from fire exclusion were static, these efforts would begin to reduce the risks associated with atypical wildfire. However, the additional fuels from current and projected climate-driven tree mortality pose a major challenge, and additional investment will be

(a)
(b)
(c)

Figure 4. Mixed-conifer forest in Yosemite National Park, California. (a) Unburned; (b) burned once with low-moderate severity fire; (c) burned three times (this photograph was taken 6 years after the 2012–2016 drought). Image credits: M Meyer.

required to meet surface fuel objectives as the C carrying capacity of dry forests continues to decrease. Even in the absence of substantial overstory mortality, fuel inputs from typical forest dynamics are high enough to require regular maintenance treatment (USFS 2022). Prescribed fire typically occurs on the shoulders of the fire season, but fewer climatically suitable burn days now occur in the fall and spring across much of the western US because of climate change (Swain *et al*. 2023). Higher temperatures co-occurring with more variable precipitation could open burn windows during winter months, but increased potential for degraded air quality due to reduced smoke loft can hinder winter burning. The additional surface fuel inputs from tree mortality will also contribute to future fire emissions. Increasing prescribed fire frequency to manage the risks associated with those inputs will translate into more days with smoke from prescribed burning. Yet as compared to that released from uncontrolled wildfire, the amount of smoke released from individual prescribed burns will be substantially less, greatly reducing the adverse impacts of which on human health (Long *et al*. 2018).

In areas of high tree mortality, hazardous fuel reduction is less about restoring ecologically appropriate fire and more about preparing the forest for the inevitability of wildfire and additional climate change by bringing the C stock into closer alignment with the new C carrying capacity. Late winter and early spring may present additional opportunities to accomplish this objective in years with sufficient snowpack. Given an adequate workforce, fire crews could strategically burn pockets of coarse fuels while overall fuel moisture is high enough to pose less risk from fire escaping planned boundaries. Fuel consumption will be low and the resulting energy release should limit damage to remaining live trees, but this will require multiple entries into the same areas until objectives are met. Using our example of data from the Sierra Nevada, and assuming that live tree C density is within the C carrying capacity of climatic conditions, it could take 15 years of burning at 5-year intervals to reduce the dead tree C from the 2012–2016 drought in the system to a level that does not create additional high-severity fire hazard.

Given the scale and anticipated growth of climate-driven tree mortality, there is substantial area where wildfire will occur before management is possible. If burning under less-than-extreme conditions, these fires could effectively "treat" a large portion of these forests by consuming the dead biomass, albeit with less precision than prescribed fire (Cova *et al*. 2023). However, it is worth noting that the energy release in some areas may lead to near 100% mortality of the remaining live conifers (ie high-severity fire). While this presents a considerable challenge, it also represents an opportunity to facilitate a successional trajectory that is better suited to the likely climate future of a particular location, especially if it is facilitated by post-fire management interventions. In addition, in burned forests with some proportion of remaining live conifers (ie low- to moderate-severity fire, often ≥50% of the burned landscape), there remains an opportunity to further reduce fuels and bolster ecosystem resistance and resilience with prescribed fires or managed wildfires (Table 1).

Table 1. Potential management responses to achieve climate-induced reductions in forest carbon (C) carrying capacity under different disturbance conditions and their potential trade-offs

Unburned and fire-excluded forest	Forests that burn at low or moderate severity	Forests that burn at high severity
Recommendations		
Treat these forests as quickly as possible to bring fuel loads under control	Capitalize on the initial fire entry with another burn that brings fuel conditions closer to the target condition	Transition these areas to new structural or composition conditions that are more aligned with future climate and fire conditions
Use mechanical and prescribed fire treatments to prepare these areas for wildfire	High consumption under wildfire conditions could accelerate reduction in fuel loads faster than prescribed burning	May include transitioning conifer forests to hardwood forests or a non-forest condition
Burn at the hotter end of prescription to increase fuel consumption	Follow-up burns should target removing remaining ladder fuels and post-fire snagfall	Design treatments to increase landscape heterogeneity to prepare the landscape for more fire
Manage natural ignitions in the winter when possible	Follow-up burns should be implemented in a strategic manner to restore landscape heterogeneity	
Increase heterogeneity in fire-excluded patches in a strategic fashion		
Potential trade-offs		
Burning in winter may transfer more heat to the soil, with potential impacts	Increased fire emissions throughout the year as prescribed fires are implemented in the shoulder seasons	Reduction in forest-obligate wildlife habitat, biodiversity, and other forest ecosystem services
Surplus of small-diameter trees will require more costly alternatives to piling and burning to lessen air-quality impacts (this could be offset through increased capacity for wood utilization)	Dedicated prescribed burning crews required to ensure sufficient fire personnel are available	Lower transpiration would cause higher local temperature
Hotter prescribed burns could increase mortality rates of larger trees, resulting in reduced live C pools, and increase risk of escaped fires	Low-severity burned stands may retain high post-fire fuel loads that require both mechanical and prescribed fire treatments	Potential reduction in snowpack persistence due to reduced shading

Notes: lists of recommendations and potential trade-offs are non-exhaustive because site-specific conditions will influence both.

The challenges we highlight using Teakettle data are not unique to this location, nor to the southern Sierra Nevada. Widespread tree mortality associated with a warming and drying atmosphere is occurring globally (Hammond *et al.* 2022). As the atmosphere dries, forests become more flammable (Juang *et al.* 2022). The interaction between changing climate and extensive high-severity wildfire can create a mismatch in the species that are available to seed into an area and the climate space where those species offspring can establish (Liang *et al.* 2017). With increasingly large high-severity patches, artificial regeneration (ie planting seedlings) may be necessary to ensure tree establishment. However, post-fire climatic conditions may be such that simply planting all the high-severity area within an entire burn footprint may not be prudent, nor operationally feasible. A more promising approach might involve using variability in microclimate as a function of topography and vegetation to locate microclimates where tree seedlings are capable of surviving (Crockett and Hurteau 2022; Marsh *et al.* 2022). In those specific locations, being cognizant of the landscape's increasing flammability can help inform planting strategies and improve the chances of creating forest densities and landscape patch configurations that are heterogeneous and better prepared for future wildfire (Table 1; North *et al.* 2021).

While largely focused on fuels reduction, western US forest management has rarely addressed the intensifying drivers or unprecedented fuel loading from hot droughts and bark beetle outbreaks in overcrowded forests. Climate-driven tree mortality is likely to continue as dense, fire-suppressed forests respond to altered climatic conditions.

Where increasing atmospheric dryness is making forests more flammable, the additional dead biomass from this transition poses a substantial challenge for the remaining live trees, especially those that are the largest and oldest. As with many climate-driven changes, forest managers are being forced to make decisions based on incomplete information and with limited resources. Managing this challenge will require creativity, experimentation, and a suite of management actions that take advantage of growing fire effects across the forest landscape (Table 1). Ensuring that we learn from and adapt to change will require regular monitoring, data analysis, and robust science–management partnerships to determine how best to proceed. While uncertainty exists, we must not become paralyzed by it or else we risk losing more old forest habitat and the species that depend upon it.

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Data Availability Statement

Data and code used in this research are available from Dryad at www.doi.org/10.5061/dryad.djh9w0w76.

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