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Tamm Review: Postfire landscape management in frequent-fire conifer forests of the southwestern United States

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ABSTRACT

The increasing incidence of wildfires across the southwestern United States (US) is altering the contemporary forest management template within historically frequent-fire conifer forests. An increasing fraction of southwestern conifer forests have recently burned, and many of these burned landscapes contain complex mosaics of surviving forest and severely burned patches without surviving conifer trees. These heterogeneous burned landscapes present unique social and ecological challenges. Severely burned patches can present numerous barriers to successful conifer regeneration, and often contain heavy downed fuels which have cascading effects on future fire behavior and conifer regeneration. Conversely, surviving forest patches are increasingly recognized for their value in postfire reforestation but often are overlooked from a management perspective.

Here we present a decision-making framework for landscape-scale management of complex postfire landscapes that allows for adaptation to a warming climate and future fire. We focus specifically on historically frequent-fire forests of the southwestern US but make connections to other forest types and other regions. Our framework depends on a spatially-explicit assessment of the mosaic of conifer forest and severely burned patches in the postfire landscape, evaluates likely vegetation trajectories, and identifies critical decision points to direct vegetation change via manipulations of fuels and live vegetation. This framework includes detailed considerations for postfire fuels management (e.g., edge hardening within live forest patches and repeat burning) and for reforestation (e.g., balancing tradeoffs between intensive and extensive planting strategies, establishing patches of seed trees, spatial planning to optimize reforestation success, and improving nursery capacity). In a future of increasing fire activity in forests where repeated low- to moderate-severity fire is essential to ecosystem

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resilience, the decision-making framework developed here can easily be integrated with existing postfire management strategies to optimize allocation of limited resources and more actively manage burned landscapes.

1. Introduction

The annual area burned by wildfire in forests of the western United States (US) has been rising since the 1990s (Abatzoglou and Williams, 2016). This trend holds in the southwestern US (Southwest), across all forest types (Singleton et al., 2019). As a result, the cumulative area recently burned in this region has increased steadily over the past two decades (Fig. 1).

Accompanying this trend, area burned at high-severity has also increased in the Southwest (Mueller et al., 2020, Parks and Abatzoglou, 2020), including in ponderosa pine and mixed-conifer forests (Singleton et al., 2019) that were historically characterized by frequent, predominantly low- to moderate-severity fire regimes (Swetnam and Baisan, 1996, McKinney, 2019). As in other regions of the western US, this trend has led to a forest management paradigm focused on reducing stand density and surface fuels within these types of fire-suppressed forests, a focus that is warranted (Stephens et al., 2016) given current trajectories of stand-replacing fire and climate warming (Stephens et al., 2013). As burned area has increased, however, capacity to actively manage complex postfire landscapes has not kept pace with the growing management need (North et al., 2019, Fargione et al., 2021).

Postfire management in southwestern forests is currently oriented primarily towards mitigating the short-term risk of erosion and debris flows by attempting to stabilize burned soils using mulching or aerial seeding. These activities are focused in high-severity burned areas due to the well-documented link between increasing burn severity and decreased soil infiltration (Robichaud, 2000), which increases risks to downstream infrastructure. These activities mostly occur through substantial investment in the Burned Area Emergency Response (BAER) program, particularly for federally managed forests (Robichaud et al., 2014). However, BAER operations present their own challenges, including the introduction of non-native or invasive species and the relatively short time horizon over which the money supporting BAER efforts must be spent (Peppin et al., 2011). The appropriate timing for postfire management efforts may extend beyond the short time period in which BAER money is allocated for "emergency response". Further, funding on a per-fire basis precludes working at a landscape-scale.

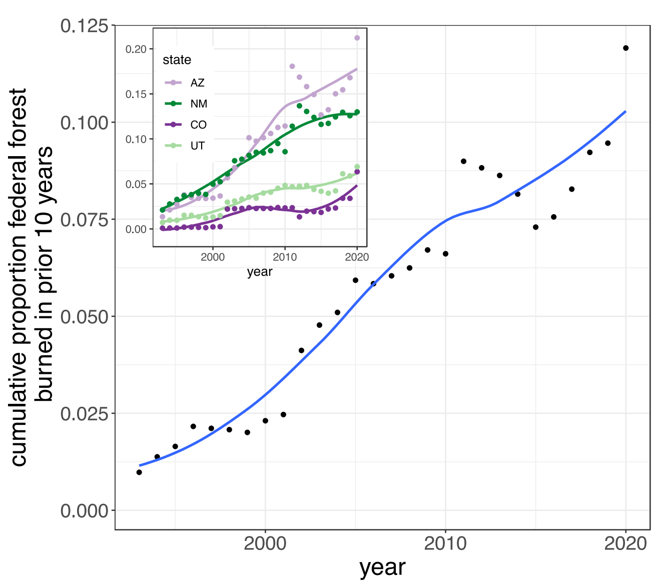


Fig. 1. Cumulative proportion of federally managed forest area burned within the previous ten years in the southwestern US (i.e., Arizona, Colorado, New Mexico, Utah). Federally managed forest includes all National Forest lands in the four-state region, as well as Grand Canyon National Park, Zion National Park, Saguaro National Park, Valles Caldera National Preserve, Bandelier National Monument, and Rocky Mountain National Park. Blue line represents a loess smoothing of the data. Data through 2018 from https://mtbs.gov/ (accessed July 2020); data for 2019 and 2020 from National Interagency Fire Center; https://data-nifc.opendata.arcgis.com (accessed Feb 2021). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

In contrast, longer-term forest management on burned landscapes in the Southwest, as in other regions, has received comparatively little attention (but see North et al., 2019, Meyer et al., 2021, Larson et al., this issue). Particularly in ponderosa pine and drier mixed-conifer forests (especially those containing a component of ponderosa pine) with a history of frequent fire (Reynolds et al., 2013), postfire forest management is challenged by prefire forest structure that is strongly departed from historical conditions (Hagmann et al., 2021). These changes, in conjunction with a warming climate, are leading to larger and more contiguous patches of high-severity fire (Stevens et al., 2017) and the potential conversion of forests to nonforest vegetation for extended periods of time (Coop et al., 2020). Key processes dictating longer term forest resilience, such as tree-seed dispersal, competing vegetation, persistence of nonforest patches, and fuel dynamics, are largely contingent upon the patch size and spatial patterning of heterogeneity in surviving tree cover across the postfire landscape (Hessburg et al., 2016). By resilience, we mean the self-sustaining persistence of conifer forest over some considerable fraction of the landscape in question, through subsequent climate warming, fires, and other disturbances (Johnstone et al., 2016, Hessburg et al., 2019, North et al., 2021), on a scale from decades to centuries.

Fire is well-understood to have played a critical role in sustaining southwestern forests for millennia, and many of the current management challenges are associated with its exclusion (Allen et al., 2002). Whether or not recent increases in burned area produce desirable effects locally, fire will inevitably continue to impact these landscapes, so postfire management must prepare for continued exposure to fire (North et al., 2021). Within ponderosa pine and drier mixed-conifer forest types, long-term resilience may require that forest mosaics are more or less constantly in a postfire condition, e.g., through a rotation of prescribed fire and/or wildfire managed for resource benefits (North et al., 2015b). By using the term postfire, we are referring to relatively recently burned (e.g., within 10 or 20 years) landscapes. Ongoing increases in wildfire activity across the western US means that postfire management actions must anticipate future fires impacting the landscape again, potentially within decades or less (Prichard et al., 2021; North et al., 2021).

Importantly, forest management is not independent of social and cultural values (Hessburg et al., 2019). Decisions to preserve or regenerate conifer forest on postfire landscapes almost always extend beyond ecosystem service considerations to also include important aesthetic, cultural, and spiritual values that emphasize the importance of conifer forests on these landscapes. Postfire landscapes offer an opportunity to develop collaborative approaches to management that include contributions and incorporate values from multiple partners and parties to direct management actions (Stortz et al., 2018).

To address the multitude of challenges to long-term resilience of burned landscapes, we introduce a framework for postfire management decisions that explicitly considers vegetation management strategies for different landscape conditions within a heterogeneous burn matrix, anticipating future fire and climate warming in an adaptive management context. Our focus is on ponderosa pine and mixed-conifer forests of the southwestern US (the states of Arizona, Colorado, New Mexico and Utah), but the strategies we discuss may be applied to similar forest types in other regions, and in some cases other forest types, which we identify where appropriate. We propose strategic and targeted interventions that are spatially explicit, prioritized across space and time, and leverage natural processes to retain or restore a diversity of ecosystem components across burned landscapes. Burned area will continue to increase with ongoing climate change, making a cohesive landscape-scale strategy for long-term forest management of burned landscapes imperative (Meyer et al., 2021), in the southwestern US and beyond.

2. A framework for postfire landscape management

Postfire landscapes are complex across space and time: prefire fuel conditions, fire behavior, topography, species' regenerative traits, and many other factors influence the complex mosaic of vegetation and fuels left behind by wildfire, and the direction and rate of change in that mosaic in the decades that follow. Frameworks to structure decision making in complex environments, however, need not be complex (Puettmann and Messier, 2020). Management actions within complex ecosystems may be guided by relatively simple principles, including the importance of landscape context, retaining diverse ecosystem components to facilitate future adaptation, linkages across scales, a diverse portfolio of operational approaches where uncertainty exists, and adaptive management to learn from actions taken (Larson et al., 2013b, Puettmann and Messier, 2020, Meyer et al., 2021).

With these principles in mind, we propose a three-phase framework (Fig. 2) to guide management of complex postfire landscapes in the southwestern US, which includes 1) partitioning the postfire landscape, 2) evaluating trajectories of change, and 3) prioritizing, implementing, and monitoring actions. These concepts draw from existing frameworks, including the "resist-accept-direct" framework (Aplet and Cole, 2010) that have not always been specifically applied to postfire management decisions (Meyer et al., 2021). In this context, actions may be implemented to "resist" major landscape changes generated by fire. Efforts to maintain forests where fire impacts were less severe represent "resist" strategies. The decision to "accept" potentially novel postfire conditions or trajectories may be appropriate where intervention is not feasible, not a high priority, and/or where changes are acceptable or desirable, for example shifts from conifer forest to aspen forest or to native species of grasses and shrubs. "Accept" should be a deliberate and informed decision, rather than the default due to lack of management capacity. In contrast, the decision to "direct" the landscape toward a new condition recognizes that the postfire environment can present opportunities for

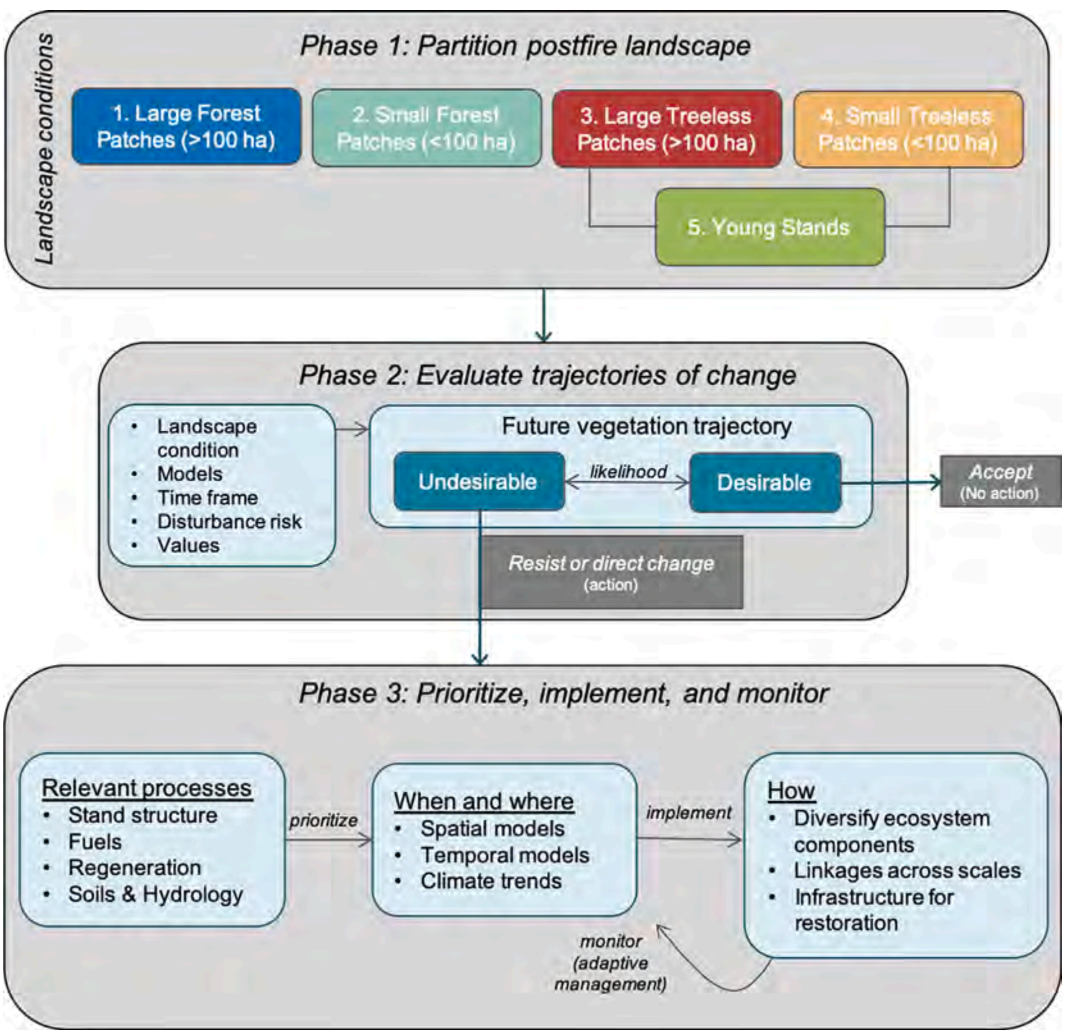


Fig. 2. A framework for postfire landscape management. Under this approach the postfire landscape is first partitioned into five common forest conditions that may warrant management (Phase 1). Then, the likelihood and acceptability of different outcomes over different time horizons for a given forest condition are assessed (Phase 2). Where values dictate undesirable conditions with moderate to high likelihood of occurrence without action, action to resist or direct change (Phase 3) may occur. This phase operationalizes decisions about where and when to act (spatial and temporal prioritization), and how to act (implementation). Monitoring via adaptive management enables future decisions to be made based on outcomes from past decisions.

ecological management toward a system that may be better aligned with future climate. For example, reforestation could "direct" forest composition toward more drought- and heat-resistant genotypes or species.

2.1. Phase 1: Partition postfire landscape

Our framework for postfire management (Fig. 2) adopts a spatially explicit approach that rests on an evaluation of vegetation burn severity via remote sensing (RS, Fig. 3), a mainstay of forest fire science. Landsat-derived burn severity maps (Fig. 3a), based on a single index such as the differenced Normalized Burn Ratio (NBR) (Key and Benson, 2006) or composite indices calibrated to field data (Parks et al., 2019b), are the most common means of assessing vegetation burn severity. These continuous indices are conventionally categorized into low-, moderate-, and high-severity classes based on field-calibrated mortality thresholds (Parks et al., 2019b). High-severity patches are often equated by users with treeless patches, which is often borne out in the field, even when field-calibrated thresholds for categorizing indices as "high severity" are less than 100% mortality (Miller and Quayle, 2015, Lydersen et al., 2016).

Although Landsat is the most widely used RS data source because of its extended temporal record and moderate spatial resolution (30 m), a growing array of sensors and platforms represent complementary or improved options for deriving relevant ecological information (Table S1). Aerial photography (e.g., 1-m National Agriculture Imagery Program (NAIP)) and high-resolution satellite imagery (e.g., 2-m WorldView) provide the ability to map individual live trees within areas of high mortality (Coop et al., 2019, Walker et al., 2019, Chapman et al., 2020), leading to improved predictions of binary forest/nonforest cover on postfire landscapes.

We follow this simplified conceptual approach to categorize postfire landscape conditions into "forest" and "treeless" patches. We define "forest patches" as those specifically containing mature conifer cover that had been present before the fire. Our description of "forest patches" includes low- and moderate-severity patches by the conventional definition, as well as portions of unburned forest within the fire perimeter. Unburned, low- and moderate-severity areas within a fire perimeter, along with adjoining forest outside a fire perimeter, may collectively be considered "fire refugia": places that are disturbed less frequently or severely by wildfire and thereby support postfire ecosystem function, biodiversity, and resilience to changing environmental conditions (Camp et al., 1997, Krawchuk et al., 2016, Meddens et al., 2018b, Krawchuk et al., 2020). We define "treeless patches" as areas that had live forest cover prior to the fire but not after. Where we do use the more conventional term "high-severity areas", we treat it as functionally equivalent to the term "treeless patches". The scope of this framework is restricted to landscapes that were forested pre-fire; climatically or edaphically driven nonforest vegetation types (e.g. meadows) will warrant a different approach.

We further subdivide forest and treeless patches into large (≥100 ha) and small (<100 ha) size classes (Fig. 2), because the function and trajectory of these landscape conditions is size-dependent (Coop et al., 2019). While the use of 100 ha as a breakpoint to distinguish size classes is somewhat arbitrary, it represents a reasonable upper bound for the characteristic scale of treeless patches within historically frequent-fire forests in the western US (Reynolds et al., 2013, Safford and Stevens, 2017), and has precedent as a breakpoint for postfire planning (Meyer et al., 2021). In addition to these four landscape conditions, we also consider a fifth condition of "young stands", representing naturally or artificially regenerating conifers ≤ ca. 50 years old, which may warrant special consideration.

2.2. Phase 2: Evaluate trajectories of change

Phase 2 of our framework is to integrate landscape values with likely trajectories of vegetation change associated with a particular forest condition over different temporal scales (Fig. 2). Values should be explicitly accounted for in this phase because different outcomes may be viewed as more or less desirable by different parties (McWethy et al., 2019). However, the likelihood of an outcome should be assessed based on the best available science (a value-free assessment), independent of the desirability of an outcome (a value-full assessment), to the greatest extent possible (Higuera et al., 2019). A particular advantage of partitioning the landscape (Phase 1) is that different landscape conditions may be deemed more or less valuable based on their abundance on the landscape. For example, if a fire burns at predominantly high severity, then the value of small forest patches may be outsized (Coop et al., 2019) and management to retain these patches, and reforest portions of large treeless patches, may be the highest priority. Conversely, if a fire creates only a few small patches of nonforest vegetation, then nonforest patches

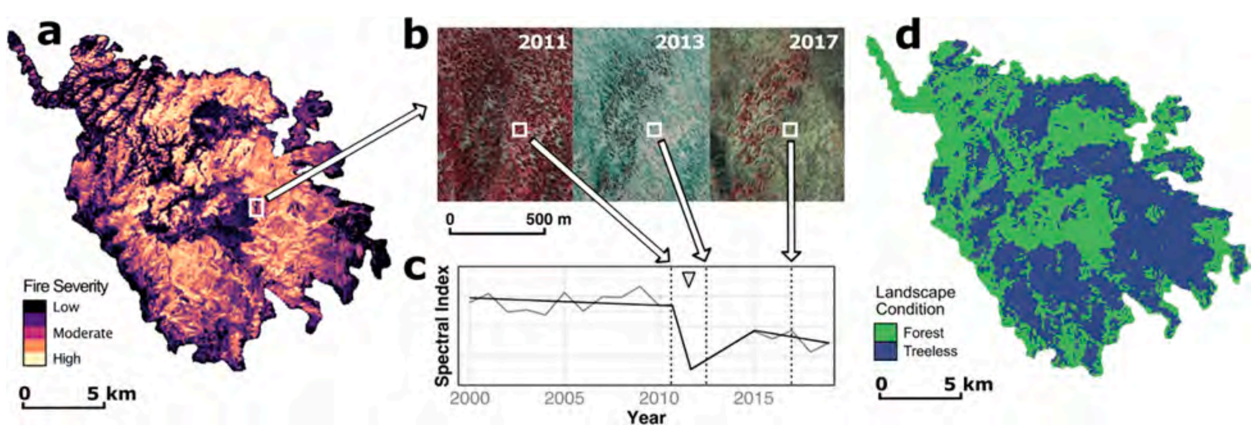


Fig. 3. Remote sensing (RS) provides critical information for the management of postfire landscapes in the southwestern US. (a) Burn severity maps derived from c. 30-m Landsat imagery, and (b) high-resolution (e.g., 1-m National Agriculture Imagery Program [NAIP]) postfire imagery are commonly used to identify the locations of conifer forest and treeless areas. (c) Landsat image time series that track changes in bands/spectral indices indicative of vegetation (in this case, for the small box in (b)), can provide information about postfire successional trajectories. (d) Binary forest/treeless layer of the postfire landscape, based on NAIP imagery, that provides the template for management decision-making and interventions described herein. The example given is for the 2012 Waldo Canyon Fire near Colorado Springs, CO. Panels (b, c) indicate that the fire burned with both low- and high-severity effects at a site near 38.93° N, 104.91° W. Substantial forest cover was lost in the 2012 fire, though some live trees remain in this area (i.e., red areas in b, 2017). Panel (c) shows that after the fire in 2012 (triangle indicates fire occurrence), partial regrowth occurred 2013–2015 and vegetation was relatively stable 2015–2019 without full recovery. White boxes in panels (a) and (b) show the focal areas for panels (b) and (c), respectively. The timing of images in (b) is represented using dotted vertical lines in (c). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

may have comparatively high landscape value (Swanson et al., 2011).

RS tools may again be useful in assessing the likelihood of different vegetation trajectories. For instance, RS-based phenology metrics and spectral indices (Fig. 3c) may help to map successional trajectories, such as evergreen conifer or non-forest herbaceous regrowth (Walker and Soulard, 2019, Vanderhoof et al., 2021). Imagery with higher spatial resolution can be used to characterize locally heterogeneous postfire vegetation (e.g., shrub, forb, bare ground) via object-based image analysis (Vanderhoof et al., 2018, Stevens et al., 2020a). RS data can also help predict postfire vegetation trajectories based on vulnerability to drought (e.g. via long-term signals in productivity that may be linked to hydraulic refugia; Cartwright et al., 2020, Rodman et al., 2020a) or future disturbance (e.g., via RS data describing 3D fuels structure to quantify future fire hazard; Chamberlain et al., 2021, Gale et al., 2021).

2.3. Phase 3: Prioritize, implement, and monitor

Once a decision has been made that intervention is warranted (e.g., to “resist” change and maintain or restore pre-fire conditions, or “direct” the ecosystem to a new state), available actions must be considered (Fig. 2). From a forest management perspective, available actions are generally those that directly manipulate vegetation and fuels. Regardless of the forest condition and suite of management actions under consideration, there are several guiding principles for successful postfire management (Meyer et al., 2021). First, when limited resources prohibit large-scale actions, more localized strategic actions that maximize benefits to identified values will be necessary (Meyer et al., 2021). Second, in the face of uncertainty over future conditions (Millar et al., 2007), a guiding principle for any given action is to “diversify the portfolio” of management outcomes (Schindler et al., 2015). This can be achieved by promoting 1) a diversity of ecosystem components, 2) linkages between components across spatial and temporal scales, and/or 3) specific ecosystem components which might facilitate future transitions to new yet desirable ecosystem states (Puettmann and Messier, 2020). Third, effective management often requires sustained maintenance and monitoring (Hobbs et al., 2011). Ultimately management decisions are most effective in an adaptive management context (Fig. 2), where a flexible plan for postfire management exists before a fire occurs (Stortz et al., 2018), and postfire outcomes are tracked to inform future actions.

3. Management options in different forest conditions

Here we discuss the current state of forest ecosystem science as it pertains to different management actions for the five postfire landscape conditions in our framework (Fig. 2). In each case, we describe common defining characteristics of the condition in question, identify plausible trajectories for the condition, and discuss which values might lead to a decision to accept the new condition on the postfire landscape without further intervention. We emphasize that trajectories are uncertain and dependent on exposure to future fire, conifer regeneration dynamics, and a host of other factors that are context specific. Following our framework, we then discuss 1–2 potential management interventions for each landscape condition to either resist or direct future change over a 30–50 year time horizon, focusing on evidence from ponderosa pine and mixed-conifer forests from the southwestern states but drawing on evidence from other forest types and regions as appropriate.

3.1. Landscape condition #1: Large (>100 ha) forest patches

Even with increasing area burned at high severity, the majority of the area within most fire perimeters in western US forests burns at low to moderate severity (Singleton et al., 2019, Parks and Abatzoglou, 2020), with some proportion of the area within a fire perimeter often remaining entirely unburned (Meddens et al., 2018a). We refer to these areas collectively as “forest patches”, a term roughly analogous to the fire refugia concept discussed above (Krawchuk et al., 2020) but explicitly describing their key characteristic relevant to southwestern forest ecosystems: surviving postfire conifer cover. Despite this key characteristic, fuel and vegetation structure may still vary considerably within and among forest patches, depending on fire behavior and pre-fire forest structure (Fig. 4; Huffman et al., 2017).

“Large” forest patches are further characterized by living mature trees remaining throughout a patch greater than 100 ha, with embedded fine-scale canopy gaps generally ranging in size from single tree crowns to ca. 10 ha (Churchill et al., 2013, Lydersen et al., 2013). In southwestern ponderosa pine and drier mixed-conifer forests, large patches, if not entire landscapes, of recently burned living trees were common historically, because pre-settlement fire regimes were characterized by frequent low- to moderate-severity fire that limited the potential for extensive high-severity fire effects (Swetnam and Baisan, 1996, McKinney, 2019).

Because areas burned at low to moderate severity still often represent the majority of contemporary postfire landscapes in southwestern forests (Fig. 3; Singleton et al., 2019), and forest cover generally (but not always) occurs immediately outside a fire perimeter as well, large forest patches usually comprise the matrix surrounding high-severity patches and therefore have an outsized influence on postfire vegetation and fuel development within fire perimeters. In this context large forest patches have two key characteristics for forest regeneration: seeds from mature trees, and suitable postfire microsite conditions, including bare mineral soil. If high-severity burned areas are outside the climatic regeneration niche for a local tree species, low- and moderate-severity burned areas may provide critical sites for seedling recruitment (Dobrowski et al., 2015), because of the ameliorating effects of the overstory canopy (Owen et al., 2017, Kemp et al., 2019, Korb et al., 2019). Large forest patches also contribute seeds to nearby treeless patches.

3.1.1. Future trajectories and risk

The potential for subsequent high-severity fire is a critical factor in developing a postfire management strategy within large forest patches. Risk of subsequent high-severity fire is typically reduced when the initial fire reduces live tree densities, fuel connectivity, and surface fuel loads (Holden et al., 2010, Hunter et al., 2011, Walker et al., 2018). If fuels are reduced sufficiently, subsequent fires may extinguish naturally or burn at low severity upon entering such patches. Such limitations on fire spread appear to be greatest within five to fifteen years after an initial fire (Parks et al., 2015, Yocom et al., 2019, Buma et al., 2020), while reductions in subsequent fire severity may extend to at least 30 years (Stevens-Rumann et al., 2016).

High fuel loads in large forest patches that increase the risk of subsequent high-severity fire can arise via multiple pathways. On the one hand, the initial fire might not burn hot enough to significantly reduce surface and ladder fuels, tree densities, or canopy fuels (Hunter et al., 2011, Higgins et al., 2015, Huffman et al., 2018). On the other hand, initial fires that burn at moderate severities can lead to significant additional fuels to replace those consumed by the initial fire, particularly if the forest had previously experienced an extended absence of fire (Walker et al., 2018). This could happen via a postfire pulse of resprouting woody vegetation, or via the conversion of live fuels to coarse woody fuels that could increase subsequent burn severity (Stevens-Rumann et al., 2016, Prichard et al., 2017, Collins et al., 2018, Lutz et al., 2020). Other postfire disturbances, including drought and insect outbreaks (Roccaforte et al., 2018), can also modify fuel structures and have the potential to alter subsequent fire behavior (Stephens et al., 2018).

A decision to resist further change (e.g., loss due to future high-severity fire) within large forest patches is a common one (Meyer et al., 2021), which may be driven by values placed on forested landscapes including recreation, cultural resources, wildlife, soils, water, and protection of the wildland-urban interface (WUI). Certain values may take precedence over others, for instance, adjacency to WUI or

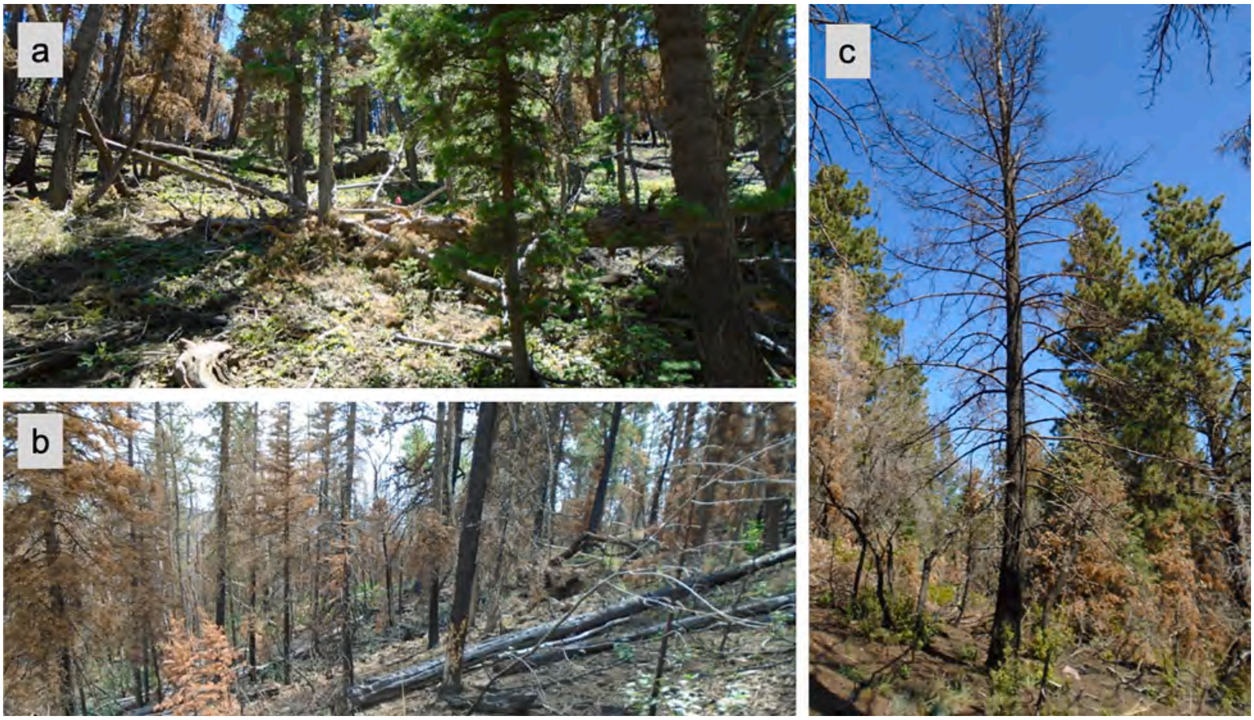


Fig. 4. Variation in postfire stand and fuel conditions within large mixed-conifer forest patches one year after the 2017 Brian Head fire in southern Utah. Portions of the fire had minimal fuel consumption (a), other portions had moderate consumption, mortality of small size class trees, and increased crown base height (b), and still other portions saw the creation of small sub-hectare canopy gaps surrounded by live forest (c). Photo credits: Larissa Yocom, used with permission.

threatened or endangered species habitat may dictate postfire management objectives (e.g., fire suppression) in some forest patches that would otherwise be candidates for future burning based on potential future fire behavior or landscape position. Landscape context, which can be assessed in terms of the postfire burn severity patch mosaic, species ranges, climate gradients, and geography (Haire et al., 2017, Parks et al., 2019a), may further influence decisions to intervene in order to resist future change in large forest patches.

3.1.2. Intervention strategy #1: Fuels management for subsequent fire resistance

Where management goals include sustaining trees that survived the initial fire, and fuel loads present high risk of future high-severity fire, then management actions should maintain or reduce fuels to an acceptable level, using similar techniques as in unburned ponderosa pine and drier mixed-conifer forest (Stephens et al., 2021). Some fuel reduction treatments have been shown to be effective in reducing the risk of subsequent high-severity fire, particularly the combination of thinning and prescribed fire, and this literature has been reviewed elsewhere (Kalies and Yocom Kent, 2016). The whole spectrum of such techniques is available for managing postfire forested landscapes (see https://swfireclime.org/), however we highlight a few options here. Importantly, large postfire forest patches can be managed as landscapes rather than stands, with desired conditions and techniques varying across space, within and among patches.

In postfire forest patches with abundant dead fuel, one management option is piling and burning of fuel. Piles burned in the winter, with little risk of escape, can be effective in reducing heavy fuel loads and subsequent fire severity (Kennedy and Johnson, 2014), although the intense heat associated with burning piles can have negative local impacts on soils, understory plants, and regenerating trees (Korb et al., 2004).

In areas where postfire forest density is still high, mechanical treatments like mastication, cut and pile treatments, lop and scatter treatments, or harvesting are all viable options when not constrained by slope, road access or management designation (North et al., 2015a), although prescribed burning after mechanical treatment is recommended since several studies have shown mechanical treatments alone are less effective in reducing, and may even increase, future fire severity (Raymond and Peterson, 2005, Prichard and Kennedy, 2012).

Where low-severity or moderate-severity wildfire has left fuel loads low or patchy, prescribed fire may be the most cost-effective option to maintain the desired conditions that resulted from the wildfire (Kolden, 2019). Desired conditions (e.g., low fuel loads, reduced tree density, greater abundance of fire-resistant tree species) may require multiple fires over decades (Larson et al., 2013a). This requires patience and a recognition that each individual fire may burn irregularly, leaving unburned patches of fuel and regeneration in patchy patterns over time, which may perpetuate desirable fine grain heterogeneity in future wildfire effects.

Unplanned ignitions have been successfully used to achieve resource objectives within some southwestern regions (Hunter et al., 2014); such "managed wildfires" represent a viable option for repeated fire entry into large postfire forest patches (North et al., 2012, Young et al., 2020), under weather conditions that are milder than those under which most wildfires currently burn. Following policy change in 2009, the use of this management option has increased in other regions of the western US (Young et al., 2020). Managed wildfires in the Southwest are often started by abundant natural lightning ignitions, which are often associated with summer precipitation events (Abatzoglou et al., 2016). High fuel moisture conditions during such managed wildfire events may facilitate slower fire spread, partial log consumption and higher seedling survival within burned areas, which may be desirable. Managed wildfires can reduce fuels and forest density, particularly if they burn with moderate severity (Huffman et al., 2017), and have an additional advantage over prescribed fire in their potential size (e.g. the > 20,000 ha Bear Fire in the Gila National Forest; Hunter et al., 2014). Large postfire forest patches represent an opportunity to manage for conifer forests with large old trees, and low fuel loads that are resilient to future fire, by using multiple methods including repeated burning wherever possible (North et al., 2015b, North et al., 2021).

3.1.3. Key knowledge gaps

There is sparse literature on best practices for directly managing surviving forests within burn perimeters. One key information gap is how different burning windows, or seasons, influence prescribed fire effectiveness for multiple objectives including reducing coarse woody fuel without causing incidental mortality of adult trees. Modeled high-severity fire risk products could be used to assess the probability of high-severity fire across western US forests (e.g., Parks et al., 2018; www.frames.gov/NextGen-FireSeverity), although it is not clear how well these reflect or accurately model fire behavior in recently burned landscapes. Collaboration and co-production of knowledge among managers and researchers can lead to better information on the effectiveness of different management options through monitoring and adaptive management (Fig. 2).

3.2. Landscape condition #2: Small (< 100 ha) forest patches

Small forest patches, which represent the smaller size classes of the more general "fire refugia" concept discussed earlier (Haire et al., 2017, Krawchuk et al., 2020), range in size from individual trees to stands < 100 ha and occur within recent burn perimeters, by definition surrounded by larger treeless patches generally created by high-severity fire (Fig. 5). These islands of remaining forest are further distinguished from large forest patches in that they often represent locally rare forest cover within a broader landscape context. Small forest patches make significant contributions to postfire ecological processes and can provide focal points for conservationists and managers. They are essential landscape components for sustaining fire-sensitive biota through fire and early postfire succession, and for facilitating their eventual recovery in the adjacent high-severity matrix, particularly when they occur near the hotter and drier ends of the prefire forest distribution (Singleton et al., 2021). Their small footprint can still support old-growth attributes and structural complexity (Camp et al., 1997), serve as critical habitat for populations of fire-sensitive trees (Schwilk and Keeley, 2006) and other plant species (Stevens et al., 2015, Stevens et al., 2019, Downing et al., 2020) due to microclimate effects, and contribute disproportionately to biodiversity (Lindenmayer, 2019).

One of the more well-understood ecological functions of small forest patches is their importance to the recovery of conifer forest within the burn matrix. For obligate-seeding trees (i.e., the majority of southwestern conifers), distance to seed source is a key driver of natural postfire regeneration (Haire and McGarigal, 2010, Chambers et al., 2016, Kemp et al., 2016, Rother and Veblen, 2016, Owen et al., 2017, Haffey et al., 2018, Rodman et al., 2020b). Small forest patches within large treeless patches can thus serve as a critical seed source, enabling tree regeneration within adjacent burned areas, proportionate to their seed dispersal distance (Greene and Johnson, 2000). Both proximity and abundance of refugia are strong predictors of postfire natural regeneration by obligate-seeding conifers (Coop et al., 2019, Downing et al., 2019). Even a single surviving tree within a high-severity burn (Fig. 5a) can provide irreplaceable function, for example, by acting as a stepping stone that promotes population connectivity (Manning et al., 2006). Research clearly shows that small forest patches play an outsized role in promoting ecosystem resilience via natural tree regeneration within postfire landscape mosaics (Coop et al., 2019), a process that is likely to increase in importance given observed and expected increases in area burned at high severity (Keyser et al., 2020, Parks and Abatzoglou, 2020).

3.2.1. Future trajectories and risk

Assessment of future trajectories and risks for small forest patches depends on an adequate characterization of their distribution and structure. It can be difficult to adequately identify and fully census small forest patches (which may be only a few square meters in size) within large postfire landscapes using remotely sensed data (Table S1), although the development of high-resolution (e.g. 1–3 m resolution) maps employing UAV, aerial, or satellite imagery could potentially be incorporated into immediate postfire mapping associated with BAER efforts (e.g., Parson et al., 2010). With improved spatial characterization of small forest patches, their landscape position plays an important role in determining their value and vulnerability, which may vary as a function of their size, isolation, and location of the forest patch within larger severely burned landscapes. Very small (e.g. < 1 ha) patches may experience greater postfire drought stress due to a greater proportion of

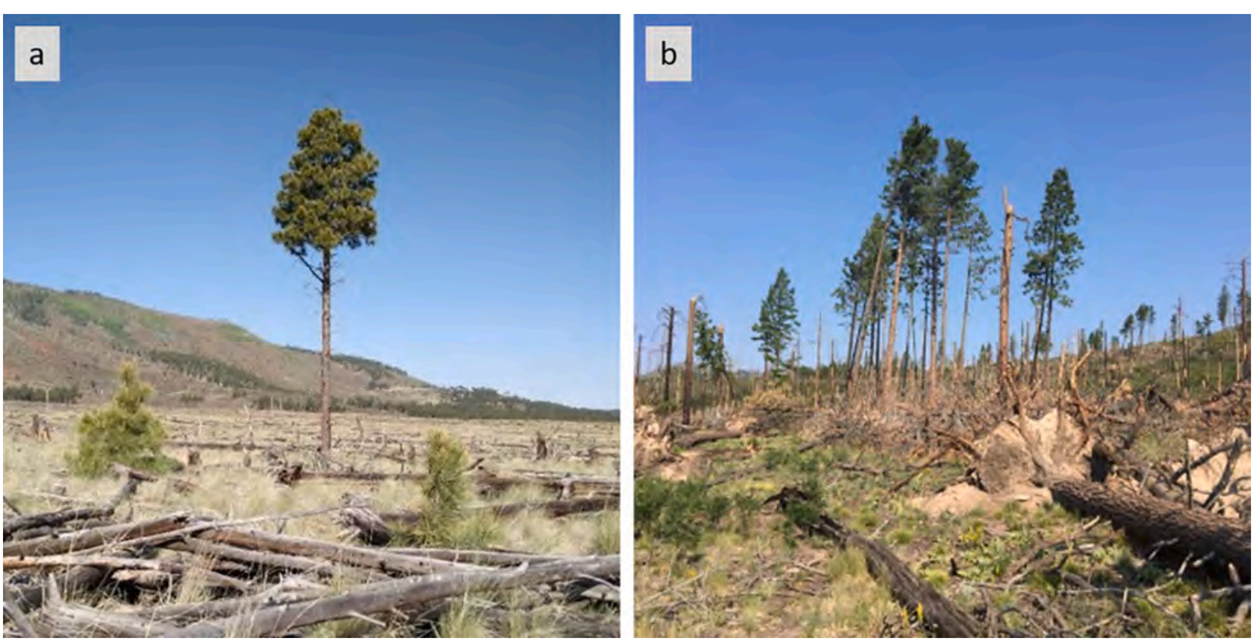


Fig. 5. A single-tree small forest patch within the Pumpkin burn perimeter, northern Arizona (a; photo credit: R.B. Walker, used with permission) and a multi-tree small forest patch within the Las Conchas burn perimeter, northern New Mexico (b; photo credit: J.T. Stevens, used with permission). The small forest patch in the Las Conchas Fire was subsequently impacted by a significant wind event that blew over roughly a quarter of the remaining live trees. This wind event illustrates how the edges of small forest patches can be continuously eroded by postfire disturbances.

edge habitat, which tends to be more exposed to drying or treefall from wind (Fig. 5) and erosion of soils by wind or water in their immediate neighborhood (Haire et al., 2017). Thus, very small patches can be more vulnerable to subsequent interacting disturbances that can weaken resilience (Buma and Wessman, 2011, Buma, 2015), however, isolated patches of forest may also be less susceptible to the spread of some disturbances including crown fire and insect outbreaks (Krawchuk et al., 2020).

Small forest patches can be susceptible to burning at high-severity in subsequent fires (Haire et al., 2017), but with important differences associated with their origins. Patches associated with landscape factors that impede fire spread or reduce fire intensity (e.g., topographic features) are expected to be more resistant to subsequent fire. These types of small forest patches are often more common in valley bottoms, near stream corridors, or in areas of cold air drainage, and also may be more prevalent atop convex landforms (Haire et al., 2017, Barton and Poulos, 2018, Chapman et al., 2020), and are more likely to serve as persistent refugia over multiple disturbances, although extreme burning conditions can override topographic influences on fire severity and refugia formation (Fornwalt et al., 2016, Krawchuk et al., 2016, Collins et al., 2019, Chapman et al., 2020, Krawchuk et al., 2020).

Where small forest patches are not associated with factors that reduce fire spread or intensity, their existence may be a product of stochastic variation in burning conditions or fire behavior (Meigs et al., 2020). Within a burned landscape, the edges between such forest patches and adjacent treeless patches often contain elevated woody fuel loads (Fig. 5b), which may elevate risk of future high-severity fire. Threats posed by these surrounding fuel complexes are compounded by live fuel development following fire (for example, tree regeneration leading to ladder fuels that increase continuity between heavy surface fuels and forest canopies). In the absence of management, this leads to increasing susceptibility to future high severity wildfire (Kolden et al., 2017). Over a series of multiple wildfires, small forest patches can be successively removed from the landscape (Haire et al., 2017), but with remaining fire refugia increasingly fire resistant and more strongly associated with topographic features (Downing et al., 2021)

3.2.2. Intervention strategy #1: Wildfire incident management for small forest patches

The dynamic nature of wildfire spread means that the creation of small forest patches within a broader landscape of high-severity fire can arise through multiple means related to fire behavior, as discussed above. Where fire behavior parameters are within the tactical control of wildland firefighting operations, it may be possible for incident management teams to promote the retention of small forest patches while a wildfire is occurring. Such an approach acknowledges that during critical wildfire incidents producing high-severity effects, the ability to preserve large forest patches within the burn perimeter via suppression tactics may be limited.

Fire suppression in accessible areas is one common spatial prioritization during incident management that can lead to preservation of small forest patches. Applicable techniques include aerial fire suppression, which can be effective at limiting fire spread and severity under moderate weather conditions (Stonesifer et al., 2016), and fireline construction, which can exert a strong influence on fire spread and tree survival. Relatively low-cost actions during incident management that promote refugia formation may lead to major savings in costs and labor otherwise required for artificial regeneration (see landscape condition #3). Suppression activities may also lead to undesirable ecological outcomes (Backer et al., 2004). In particular, aggressive burnout operations—intended to reduce fuels and unexpected fire activity within the burn perimeter—may have the unintended negative consequence of eliminating small forest patches that might otherwise have prevailed through the fire event. One option to reduce the likelihood of this outcome would be conducting nighttime burnout operations when possible.

Proactive planning of suppression activities in advance of fire might lead to enhanced capacity to promote the formation of refugia, where desirable and feasible. Decision support tools and planning approaches, such as the Wildfire Decision Support System (Noonan-Wright and Opperman, 2015) and PODs ("Potential Operational Delineations" for wildfire response; Thompson et al., 2016, Thompson et al., 2020) that are widely used by land management agencies to model fire risk, high-value assets and fire behavior, could be used to identify existing or potential refugia before and during fire operations.

3.2.3. Intervention strategy #2: Fuels and forest management for resilience

If there is concern over the potential loss of high-value small forest patches during subsequent fire, managers could prioritize them for fuels management. Though there may be disagreement among managers regarding the criteria that define "high-value" small forest patches (Martinez et al., 2019), one form of spatial prioritization could target the smallest and most isolated forest patches due to their importance in anchoring natural forest regeneration (Coop et al., 2019). Small forest patches with older, taller trees might be particularly important components of the postfire landscape, both as carbon reservoirs (Powers et al., 2013) and for increased seed dispersal potential (Greene and Johnson, 2000). Patches with high fuel loads at their margins, and/or dense natural regeneration at their periphery as described above, may be identified as particularly vulnerable.

A wide range of postfire fuel management interventions may be deployed in small forest patches prioritized for high value or high vulnerability. High-priority small forest patches may benefit from the use of thinning or prescribed fire to reduce future flammability. As with small treeless patches, described further below, small forest patch resilience to future fire may be enhanced through a strategy of edge hardening, wherein fuels at the interface between forest and treeless patches are targeted via prescribed fire (Box 1) or other techniques that reduce surface fuel loads (Kalies and Yocom Kent, 2016).

Beyond fuels management in small forest patches, the maintenance of their ecological functions may benefit from a host of well-established interventions widely employed elsewhere in forests generally, depending on specific threats and objectives. For example, small forest patches may be priority candidates for actions that shield trees from insect attacks and other pathogens such as pheromone-baited traps to protect trees from bark beetles (Ross and Daterman, 1997). Supplemental reforestation of burned areas between small forest patches using nonlocal seed sources of local tree species has the capacity to increase the genetic diversity of future forest and reduce the risk of poor and seedling quality due to inbreeding (Sorensen and Miles, 1974).

3.2.4. Key knowledge gaps

Generally, the ecological and social values of small forest patches are likely to be best served within a co-production model in which managers and researchers partner to both inform and learn from intervention outcomes (Meadow et al., 2015). For example, one interesting direction for co-produced research could entail testing the effects of water drops on small forest patch retention during firefighting operations. The extent to which small forest patches overlap in space with climate refugia that may be suitable for tree establishment or survival into the future (e.g., Davis et al., 2020, Rodman et al., 2020a) represents another important research priority.

Box 1
: Edge hardening in postfire forest patches.

The occurrence of high-severity fire inevitably creates edges between treeless and forest patches. Moderate-severity effects, which often occur at these edges, can reduce canopy fuel loads but increase surface fuel hazards as snags fall (Huffman et al., 2017). Elevated coarse woody fuel (CWF) loads at patch edges can increase the risk of subsequent fires burning at high-severity, thereby enlarging treeless patches or reducing refugia (Collins et al., 2018), but prescribed fire that reduces these CWF loads can prevent future wildfire from transitioning from surface fire to crown fire (Walker et al., 2018).

At Grand Canyon National Park, an initial fire in 2001 created small treeless patches as a result of high-severity fire in ponderosa pine and mixed conifer forest types (Fig. B1a). During subsequent wildfires in 2009 and 2018, Fire Management Staff saw evidence that treeless patches from the first wildfire were expanding (Fig. B1a). As snags fell, CWF loads at the edge of the surrounding forest patch increased dramatically between fire events, and burned with high intensities and long durations during subsequent fires.

The "High Severity Edge Prescribed Fire Project" was designed to reduce CWF loading underneath and adjacent to live overstory trees on the edge of small treeless patches. The project objectives were to use innovative prescribed fire techniques under mild fire weather conditions to 1) protect overstory "seed" trees at the edge of treeless patches, 2) halt the future expansion of existing treeless patches during future wildfires that may burn under moderate to extreme fire weather conditions, and 3) explore innovative projects to reduce future high-severity fire effects. This project was designed to occur within a late fall weather window, when CWF are available to burn with low to moderate intensities, daytime temperatures are cool, and burn periods are short.

The project was initiated on November 2, 2017, under cool daytime temperatures when CWF was available to burn with low to moderate intensities and short burn periods. Five project staff members utilized ground ignitions targeting CWF (Fig. B1b); fire continued to spread and consume CWF outside of the ignition areas for 5 days. Project outcomes included partial or full consumption of CWF, and minimal fire impact on large living trees in the burned areas (Fig. B1c). The project was implemented with a small staff and a small budget; opportunities for such a project are not always easy to find, nor do they occur on a consistent basis. However, this project shows that the application of prescribed fire by a very small crew on a landscape where the fire is allowed to move freely can lead to long-term benefits for landscape resilience and firefighter safety during future wildfires that impact previously burned landscapes.

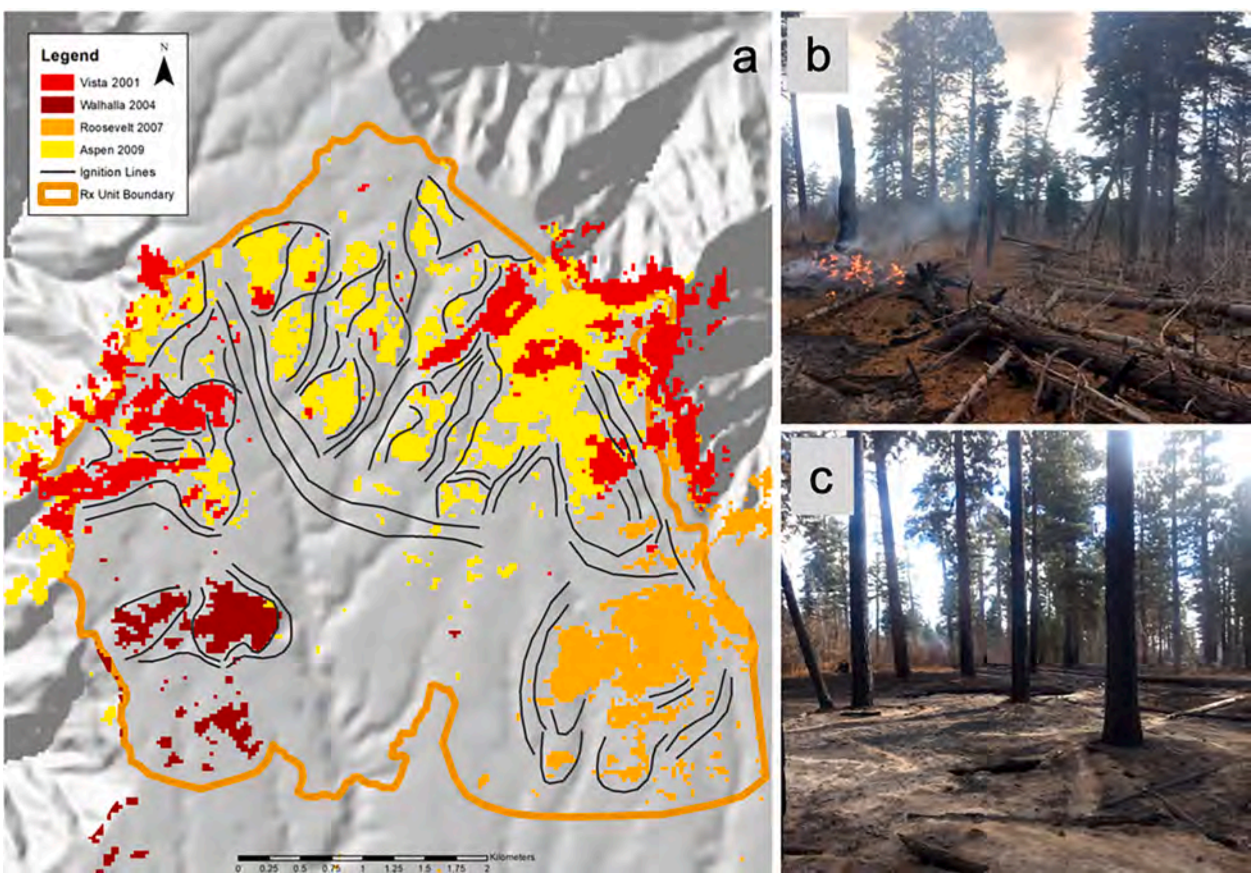


Fig. B1. Project map (a), showing the location of high-severity treeless patches following multiple fires. Patches created in the northeastern portion of the 2001 Vista Fire were enlarged in the subsequent Aspen and Roosevelt Fires. Targeted ignitions were conducted in 2017 within a 2083 ha prescription (Rx) unit boundary, along the ignition lines depicted at the margins of these high-severity patches. Example of coarse woody fuel (CWF) loading is shown at the time of ignition (b) and following completion of the burn (c). Consumption of CWF was high across the study area, yet survival of mature seed trees at the patch edges was also high. The stand in (c) is expected to have greater resilience to future fires. Photos are of different areas within the project; provided by C. Marks and used with permission.

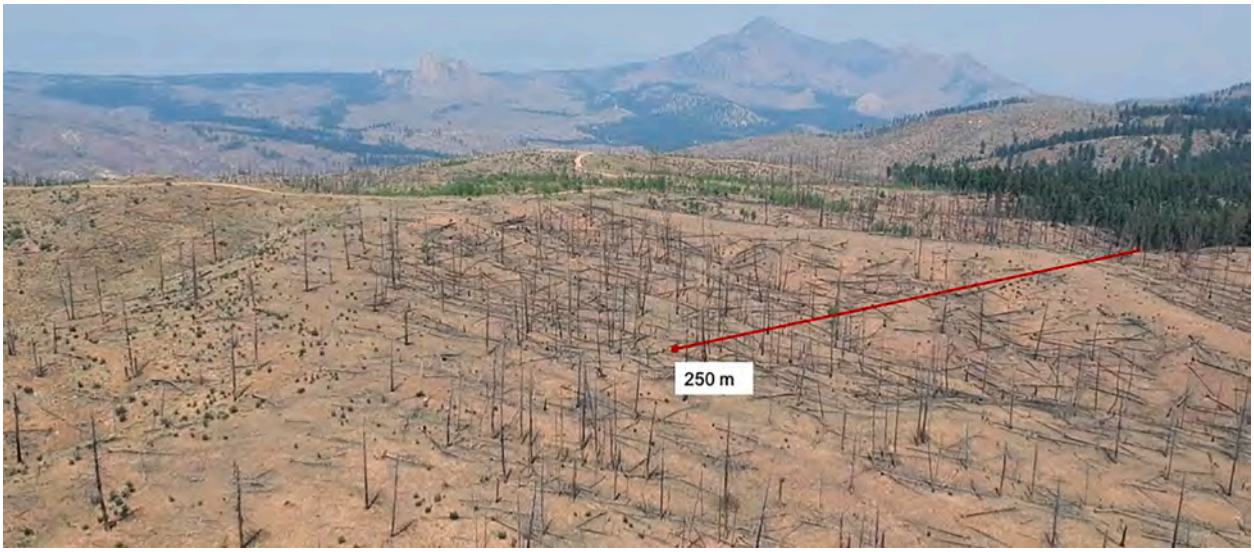


Fig. 6. Photo of a large treeless patch created by the 2002 Hayman Fire, central Colorado. Regenerating shrubs (dark green) are visible in the foreground, while regenerating quaking aspen trees (light green) are visible in the middleground. The remnant 77 ha small conifer forest patch on the right of the photo is dominated by *Pinus ponderosa*. Conifer seedlings at and beyond 250 m (red line; shown for scale) from this patch edge are predicted to be extremely rare or absent, based on data from this and other fires in the region (Chambers et al., 2016). Photo credit: M. Cooney, Colorado College, used with permission. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

3.3. Landscape condition #3: Large (>100 ha) treeless patches

Large treeless patches, for the purposes of this framework, are areas where a wildfire caused complete conifer mortality across a large contiguous area (Fig. 6), although small forest patches can commonly be embedded within them (previous section). Large treeless patches are characterized by low natural conifer regeneration away from patch edges, high woody fuel biomass, high live fuel biomass in the form of resprouting herbs, shrubs and trees, and in some cases high invasive plant abundance (Coop et al., 2020).

Large treeless patches have become an increasingly prevalent component of the southwestern postfire landscape. In the Colorado Front Range, for example, 22 of 23 recent wildfires had one or more large treeless patches > 100 ha, while nine had patches > 1,000 ha and two had patches > 10,000 ha (Chapman et al., 2020). However, prior to the 20th century there is little evidence of widespread treeless patches at this scale in frequent-fire forests of Arizona (Huffman et al., 2015), New Mexico (Guiterman et al., 2018), Colorado (Romme et al., 2003, Sherriff et al., 2014, Fornwalt et al., 2016) or elsewhere in western North America (Hagmann et al., 2021). The rarity of historical large treeless patches in frequent-fire forests is in contrast to higher-elevation, infrequent-fire forests, such as those dominated by lodgepole pine, Engelmann spruce, and/or subalpine fir, where large treeless patches were historically a much more common result of fire owing to the accumulation of fuels during long fire-free periods when climate limited fire spread (Margolis et al., 2011, O'Connor et al., 2014).

The intense disturbance that creates large treeless patches during wildfire is conducive to the establishment of resprouting and ruderal vegetation, which may vary geographically from annual and perennial grasslands and herblands (Abella and Fornwalt, 2015, Coop et al., 2016) to shrublands (Guiterman et al., 2018) to aspen stands (Margolis et al., 2007, Tepley and Veblen, 2015), and which may persist for centuries (Margolis et al., 2007, Roos and Guiterman, 2021). In the short term, disturbed soil and canopy removal can favor non-native invasive plants (Fornwalt et al., 2010), which can influence subsequent fire behavior through complex feedbacks (Kerns et al., 2020), and may be introduced accidentally through postfire rehabilitation treatments.

3.3.1. Future trajectories and risk

Absent any management intervention, the postfire recovery of conifer trees in large treeless patches is predominantly driven by two interacting processes: subsequent high-severity fire (i.e., top-killing shrubs and other vegetation) due to a modified fuel profile (Prichard et al., 2017), and limited seed dispersal and seedling establishment of most dominant conifer species (Korb et al., 2019), both of which are exacerbated by a warming and drying climate (Stevens-Rumann et al., 2018). Collectively, these drivers often portend an increased risk of enduring conversion to alternative non-conifer vegetation types (Coop et al., 2020). This trajectory is not deterministic, however, as these processes are each mediated by many additional factors including snag fall and decomposition rates influencing coarse woody fuel loads (Kennedy et al., 2021), long-distance dispersal events (Owen et al., 2017), climatic variation in the years after fire (Littlefield et al., 2020, Rodman et al., 2020a), and traits of the dominant conifers related to seed dispersal (Greene and Johnson, 2000) and fire resistance (Stevens et al., 2020b). The complexities of these interactions have been extensively described elsewhere (Stevens-Rumann et al., 2018, Hessburg et al., 2019, Coop et al., 2020); here we highlight several emerging trends in postfire trajectories and risks that are likely to influence management response.

Following an initial period of low surface fuel loads immediately after fire (Cansler et al., 2019), surface fuel accumulates quickly as a function of resprouting vegetation and snag density, which together comprise much of the new surface fuel in large treeless patches (Coppoletta et al., 2016, Johnson et al., 2020). Generally, snag fall accelerates (Roccaforte et al., 2012, Fornwalt et al., 2018) and fine woody debris fuel loads peak (Johnson et al., 2020) approximately four to nine years post-fire, while coarse woody debris fuel loads from falling snags peak between six and twenty years post-fire (Roccaforte et al., 2012, Ritchie et al., 2013, Peterson et al., 2015, Johnson et al., 2020), depending on stand basal area, snag fall rates and decomposition rates (Kennedy et al., 2021). The majority of this research has been conducted in the US Pacific Northwest, but limited evidence from the Southwest and California suggests that the peak in coarse woody fuel loads in these regions may be in the earlier 8–10 year range (Passovoy and Fulé, 2006, Ritchie et al., 2013), with complete snag fall possible within 20 years (Guiterman et al., 2015).

This temporal increase in coarse woody debris alone may drive greater severity of subsequent fires (Johnson et al., 2020). However in cases where shrubs or other woody resprouters dominate after fire, cover of this hardwood vegetation can be a better predictor of high-severity fire than coarse woody debris (Coppoletta et al., 2016), and high-severity fire driven by combustion of live woody vegetation can be sufficient to kill regenerating conifers, whether natural or planted, even where coarse woody fuels are low (McIver and Ottmar, 2007, Thompson and Spies, 2010).

High-severity re-burning may reinforce large treeless patches, but it is not required in order for large treeless patches to persist in an alternative stable state (Coop et al., 2020). The dominant effects of conifer dispersal limitation in large treeless patches are well-documented at this point (Haire and McGarigal 2010, Chambers et al., 2016, Welch et al., 2016, Haffey et al., 2018, Shive et al., 2018, Downing et al., 2019, Korb et al., 2019, Rodman et al., 2020b), as are the coincident limiting effects of a warmer and drier climate on natural conifer recruitment (Stevens-Rumann et al., 2018, Davis et al., 2019, Littlefield et al., 2020, Rodman et al., 2020a). These three controls – dispersal, climate, and competing vegetation – may represent a hierarchical set of filters controlling natural reforestation in large treeless patches independent of any re-burn effects. Dispersal limitation is the most consistent barrier to natural reforestation in large treeless patches due to their size (Collins et al., 2017), but all of these filters must be overcome for conifers to establish. Once established, seedlings face additional challenges that include reaching a fire-resistant size before the next fire (assuming the species is fire resistant as an adult; Stevens et al., 2020a,b), and withstanding pest and pathogen attacks. These collective effects enable a potentially prolonged existence for large treeless patches in a non-conifer forest state, absent any intervention.

Per our framework, the decision to intervene in large treeless patches requires a reasonable assessment of the likely future condition (Meyer et al., 2021) and a values-based assessment of the desirability of that future. Here, we set the assumption of a likely future of prolonged non-conifer forest vegetation (caveats notwithstanding; Coop et al., 2020) and discuss values that might drive decision making. A decision to accept large treeless patches might be based on their scarcity on the broader landscape (Hessburg et al., 2015), their ability to provide unmet wildlife habitat requirements normally provided by small treeless patches (Steel et al., 2019, Stephens et al., 2020), or other factors. Conversely, large treeless patches may be undesirable if they fail to provide legally mandated forest cover (e.g., the Forest and Rangeland Renewable Resources Planning Act of 1974), are detrimental to certain wildlife species (Jones et al., 2020), or other reasons. Management of large postfire treeless patches is one of the more contentious issues in western US forest management (Peery et al., 2019), and we do not aim to resolve the issue here. Instead, we examine two management themes that are common to large treeless patches when a decision has been made to restore or direct the landscape condition to a particular target, rather than accept the current trajectory: management to modify fuels and subsequent fire behavior, and management to artificially re-establish conifer forests where it would be unlikely to occur naturally.

3.3.2. Intervention strategy #1: Fuels management

One of the more common management strategies within large treeless patches is the removal of at least a portion of snags within the first several years postfire, commonly referred to as salvage logging (Leverkus et al., 2021). While decisions to salvage log often involve economic interests to obtain merchantable material, they may also be based on safety concerns or ecological concerns around the accumulation of coarse wood created by snag fall increasing future high-severity fire risk (Peterson et al., 2015, Johnson et al., 2020). Salvage logging research from the Southwest is extremely scarce, but evidence from the Pacific Northwest indicates that it generally creates an initial pulse of fine woody debris which remains elevated relative to un-salvaged stands for 5–20 years after treatment (McIver and Ottmar 2007, Keyser et al., 2009, Dunn and Bailey 2015, Peterson et al., 2015, Johnson et al., 2020) while reducing peak coarse woody surface fuel volume relative to unsalvaged stands by 2–4 times (Monsanto and Agee 2008, Dunn and Bailey 2015, Johnson et al., 2020) for up to 50 years (McIver and Ottmar 2007, Peterson et al., 2015).

While considerable evidence supports the effect of salvage logging on reducing peak coarse woody surface fuel loads (Leverkus et al., 2021), there is less evidence of that reduction translating to a reduction in future fire severity (but see Lydersen et al., 2019). Vegetation in salvaged areas appears just as likely to re-burn at high severity as unsalvaged areas (McIver and Ottmar 2007, Thompson et al., 2007, Thompson and Spies 2010, Coppoletta et al., 2016, Johnson et al., 2020), though soil burn severity may be reduced (Monsanto and Agee 2008). This effect appears to be due to the fact that the density of the postfire vegetation can be a more important driver of subsequent burn severity than the coarse woody fuel loading, whether in shrubland (Coppoletta et al., 2016) or conifer plantations (McIver and Ottmar 2007, Thompson et al., 2007). Further, the presence of snags (Tingley et al., 2020) and downed wood (Castro et al., 2011) is often beneficial for postfire wildlife habitat and conifer regeneration, respectively, which has led to increasing interest in exploring variable density salvage logging (Ritchie and Knapp 2014). Other complementary, small-scale targeted approaches to fuels management in large treeless patches, including fuels reduction at their margins (Box 1) or nucleation plantations (see below), may diversify the local structure of the landscape that the next fire encounters and reinforce that variability through repeated burning (Koontz et al., 2020).

3.3.3. Intervention strategy #2: Reforestation

Artificial establishment (i.e., “outplanting”) of conifer seedlings is a useful management tool that may help establish conifers in areas where they are desired but not regenerating naturally. The effectiveness of outplanting is measured by the initial survival and growth of seedlings on the planting site, and the longer-term establishment of forest stands that can tolerate recurring fire and drought events (Dumroese et al., 2016, North et al., 2019). Few studies in the western US have examined survival and growth of planted seedlings in postfire environments (Ritchie and Knapp 2014, Vickers et al., 2018); we are aware of only one study in the southwestern US that examined the success of postfire outplanting, in which inferred survival rates ranged between 0 and 70% (Ouzts et al., 2015). High costs (Dumroese et al., 2019) and continued increases in high-severity burned areas have created a backlog of lands where outplanting is an identified need (Fig. 7), making effective prioritization especially important (Kolb et al., 2019, North et al., 2019).

The prioritization of scarce resources to achieve successful reforestation can be guided by the Target Plant Concept (TPC; Landis, 2011, Dumroese et al., 2016), which incorporates the entire reforestation pipeline (Fig. 8), including *objectives*, *site selection*, *plant material selection*, *outplanting techniques*, and *post-planting activities* (Fargione et al., 2021). Identifying the reforestation or restoration objectives and desired conditions is the first and most critical step of the TPC. The discussion that follows assumes that a manager is interested in returning a large treeless patch to a forested state via artificial regeneration.

Site selection is the process of identifying suitable planting locations, both at a macro (landscape) and micro (local growing environment) scale. At a macro scale, spatial datasets can help to identify the locations of surviving seed trees, so planting can be prioritized farther from live trees where natural regeneration is less likely to occur (Korb et al., 2019, North et al., 2019, Stevens-Rumann and Morgan 2019), in topographic positions with beneficial growing conditions (Morelli et al., 2020). Spatial-statistical models that quantify relationships between natural regeneration and biophysical factors (Davis et al., 2019, Davis et al., 2020, Rodman et al., 2020a, Stewart et al., 2021) may help to identify xeric sites within a species’ range where the survival of planted seedlings is likely to be low based on moisture availability (Fig. 8). At the micro scale, field-based assessments to evaluate local site conditions (North

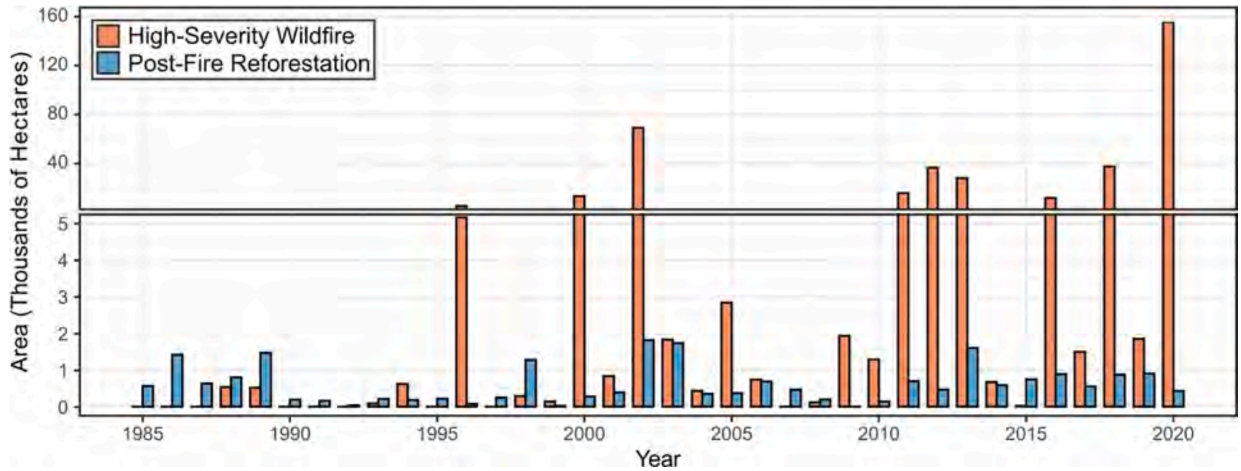


Fig. 7. Trends in high-severity burned area and reforestation area on National Forest land in the southern Rocky Mountains, predominantly in Colorado and New Mexico (area of reference defined in Rodman et al., 2020a). High-severity burned area is based on fire perimeters in the region larger than 1000 ac (405 ha), and defined as relative differenced NBR values > 544 (Chapman et al., 2020) using the composite image methodology of Parks et al. (2018). Postfire reforestation area was calculated from reforestation projects listed in the US Forest Service FACTS database for this region within known fire perimeters, including tree planting and seeding. Reforestation area was limited to National Forest land, so high-severity area was also restricted to these lands for a proper comparison.

et al., 2019, White and Long 2019) should complement macro-scale models to finalize planting locations. Field-based assessments should consider local soil characteristics, vegetation, animal activity, coarse woody debris, and other factors that may influence seedling survival, growth, and the need for site preparation (Lanini and Radosevich 1986, Owen et al., 2020, Puhlick et al., 2021). The combined use of models and field-based assessments, though not widely practiced in the southwestern US currently, provides a useful multi-scaled approach to site selection.

Plant material selection refers to the selection and preparation of growing stock prior to outplanting. The availability of seed is the most critical limitation to outplanting in southwestern US postfire landscapes, and increasing seed collection and storage capacity is essential for future reforestation (Fargione et al., 2021). Beyond seed limitation, ecotype-based seed source selection and population-level genetics are necessary to respond to changing climatic and environmental conditions (Kramer et al., 2019, Fargione et al., 2021). Seed collected from parent trees of priority tree species representing a range of ecotypes and site conditions, accompanied by provenance and progeny tests, would ensure an appropriate match between seed sources and planting sites. Such documentation would also inform the possible use of assisted migration strategies as a climate mitigation tool (Williams and Dumroese 2013).

After the expansion of seed collection to match reforestation needs, the next important investment into the reforestation pipeline for the southwestern US is increasing nursery production capacity. Existing nursery capacity in the Southwest is extremely limited, at approximately 1.5 million containerized seedlings per year as of 2020 (Haase et al., 2020), well short of the one billion seedlings required to meet the current reforestation need of 1.6–4.3 Mha in the Southwest (Cook-Patton et al., 2020).

Outplanting refers to the timing and operations involved in planting nursery-reared seedlings into the field. Although the proper timing of outplanting in the Southwest is largely unknown, a guiding principle should be to minimize stress by avoiding low soil moisture during the months after planting (Pinto et al., 2011), given the importance of soil moisture in limiting natural regeneration in this region (Davis et al., 2019, Littlefield et al., 2020). Therefore, we speculate that particularly in areas with a strong monsoonal cycle (e.g., Arizona and New Mexico), the best time to plant may be during the monsoon season (i.e., July to September) when soil moisture levels are highest (Margolis et al., 2017). Outplanting success in these conditions can be influenced by nursery cultural practices that enhance seedling drought tolerance and efficient transportation of seedlings to a field site (Landis et al., 2010).

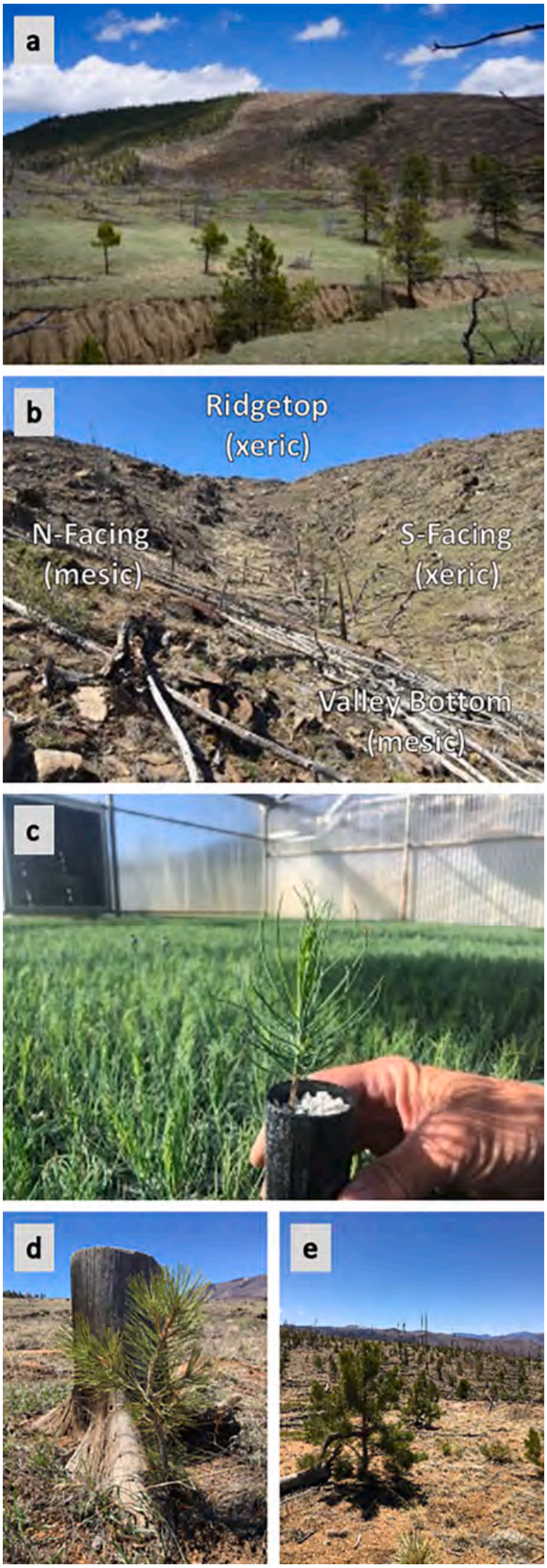
While partial cover by resprouting forbs, shrubs, and trees can benefit seedling establishment and growth rates (Owen et al., 2020), high cover may have a negative impact (Tepley et al., 2017), so planting soon after fire occurrence may be beneficial if resprouting vegetation is a concern. The use of nurse objects at the planting location (i.e., stumps and other plant species; Fig. 8) may mitigate the effects of wind, erosion, high soil temperatures, and moisture stress (Burney et al., 2007, Castro et al., 2011, Lonergan et al., 2014), though these can also be hazardous during future fires (Lydersen et al., 2019); such tradeoffs require further study.

Nucleation planting strategies (i.e., cluster planting surrounded by non-planted areas) are hypothesized to emulate natural regeneration patterns and may be a useful tool in the Southwest (Corbin and Holl 2012, Owen et al., 2017). Once planted trees reach reproductive maturity, which may happen within 30 years for Douglas-fir and ponderosa pine in the Southwest (Rodman et al., 2021), nuclei may faciliate natural regeneration in areas where more extensive outplanting is unfeasible (North et al., 2019). Widely spacing nuclei may reduce the risk of crown fire spread between them, and higher planting densities within nuclei may also improve water balance in certain seedlings within the cluster by reducing solar radiation and/or increasing snow retention, though these hypotheses require testing.

Post-planting activities refer to silvicultural treatments that are designed to enhance seedling growth and survival by alleviating potentially negative influences (Rose and Haase 2006), including adding soil amendments such as mulch or wood chips to reduce competing vegetation and increase soil water retention (Jonas et al., 2019), and using tree shelters to protect against herbivory (Taylor et al., 2006). Monitoring seedling survival and growth for several years following outplanting is a critical component of the adaptive management cycle (Fig. 2). A 'plant and walk away' approach to postfire outplanting activities in the southwestern region may well lead to a loss of investment via regeneration failure, without the ability to learn and improve subsequent interventions. Because investments made earlier in the reforestation pipeline are often substantial (Fargione et al., 2021), resource planning budgets for post-planting monitoring and follow-up treatments are critical to learning from these investments.

3.3.4. Key knowledge gaps

Many of the approaches to plant material selection and outplanting



(caption on next column)

Fig. 8. The success of artificial reforestation treatments in postfire landscapes is measured by the survival and growth of planted seedlings, which can be improved by following the Target Plant Concept. At macro scales, managers can use an understanding of the locations and sizes of treeless areas (a) and climatic and topographic factors (b) that affect plant stress and climatic suitability to identify potential planting sites. Once sites have been identified, selecting the plant materials (c) and the outplanting tools and techniques that most closely align with site conditions (d, e) will increase the chances of success. (Photo credits: a, Kyle Rodman; b, d, e, Paula Fornwalt; c, Owen Burney, used with permission).

discussed here are in their early stages of development and application in the southwestern US, including critical investigations of seedling drought conditioning (Sloan et al., 2020). The influence of vegetation on seedling establishment is particularly complex and requires further research to disentangle survival and growth effects among different conifer species (Tubbesing et al., 2020). Finally, nucleation planting techniques are not widely practiced in the southwestern US and their effective use, along with the fuels management techniques discussed above, will require continued research and monitoring, particularly with respect to how these techniques respond to subsequent fire (Peterson et al., 2007, North et al., 2019, North et al., 2021). Uncertainty of climate change impacts permeates all of these other considerations (Rodman et al., 2020a).

3.4. Landscape condition #4: Small (< 100 ha) treeless patches

Small treeless postfire patches represent fine- to *meso*-scale heterogeneity in canopy disturbance (Perry et al., 2011). These patches are likely consistent with the high-severity component of historical fire regimes, where smaller patches of complete stand mortality typically were < 100 ha (Romme et al., 2003, Swetnam and Baisan 2003, Iniguez et al., 2009) in dry conifer ecosystems, creating heterogeneity within extensive areas of low-moderate severity fire effects. Ecological benefits of the heterogeneity created by small treeless patches include fine-grained edge habitat beneficial for wildlife (Jones et al., 2020) and prolonged snowpack persistence in small gaps and along forest edges (Stevens 2017, Schneider et al., 2019, Moeser et al., 2020).

3.4.1. Future trajectories and risk

A small treeless patch may regenerate naturally to conifer forest, as pulses of conifer seedling establishment are likely to occur several times during an inter-fire period (Iniguez et al., 2016). However, non-conifer resprouting woody vegetation in particular may also be quite stable (Roos and Guiterman 2021) due to the suppression of conifer seedlings in the absence of fire (Marshall and Falk 2020, Tubbesing et al., 2020), repeated but patchy low-severity fire that removes conifer regeneration (Guiterman et al., 2018), or repeated high-severity fire (Coppoletta et al., 2016, Prichard et al., 2017).

To the extent that small treeless patches contribute to fine- to *meso*-scale heterogeneity, this may be desirable to maintain from a management perspective. However, a management objective of maintaining heterogeneity does not necessarily specify whether small treeless patches are to be maintained in situ, or to be created and gradually filled in with conifers as part of a shifting patch mosaic where new small treeless patches are created elsewhere, perhaps on top of some underlying heterogeneity in fuels or topography (Malone et al., 2018). Here, we describe fuel and forest management options for both static maintenance of small treeless patches, and for a shifting patch mosaic.

3.4.2. Intervention strategy #1: Patch maintenance

If the maintenance of existing small treeless patches in a nonforest condition is a management goal, then specific fuel and vegetation management prescriptions to prevent conifer establishment will need to be tailored to the specific climatic, vegetation, and fuel conditions of the patch. Generally, however, the more fire these patches see, the less likely their return to conifer forest will be (Coop et al., 2020). The repeated application of fire may prevent a return to conifer forest within the small treeless patch, but may also enlarge the patch with each successive entry (Box 1). In this case, fuels management around the edges of small treeless patches may be warranted, including via targeted prescribed burning of logs during the cold season following dry periods, where fuel consumption can be high without fire spreading between the logs (Box 1).

3.4.3. Intervention strategy #2: Shifting patch mosaic

If a shifting patch mosaic of nonforest habitat is desired, then conifer regeneration potential and fuels should be assessed within the patch. Generally speaking, natural regeneration potential from nearby seed trees should eliminate the need to plant trees in small treeless patches except under special circumstances. Natural regeneration within small treeless patches will utilize local seed sources and enable local adaptation (Malone et al., 2018). A shifting patch mosaic requires that some natural conifer regeneration within the patch reach maturity and fire-resistant size; partial mortality from repeated fires may still facilitate this (York et al., 2021).

3.5. Landscape condition #5: Young stands (< 100 ha)

Naturally occurring young stands in frequent-fire conifer forests were historically uneven-aged due to the influence of repeated fires, which can produce mortality rates in conifer seedlings ranging from 70% to 95% following fire intervals as short as four years (Sackett 1984, Battaglia et al., 2009). This fire-caused mortality is coupled with naturally variable regeneration patterns that are often clustered at scales up to 500 m^2 (Owen et al., 2017). These clumps tend to form in gaps with greater light penetration (Boyden et al., 2005), and generally form during fire-free intervals, which may be but are not necessarily associated with pluvial periods (Brown and Wu 2005, Iniguez et al., 2016). Although the characteristic spatial grain (i.e., patch size) of young stands varies geographically, the fine-scale spatial variability of both tree establishment and tree mortality is an emergent adaptive property of frequent fire forests that creates and maintains low and variable density forest structure, a hallmark of forest resilience to frequent fire (Hessburg et al., 2016).

3.5.1. Future trajectories and risk

We consider "young stands" to refer primarily to planted or natural conifer regeneration < 50 years in age and < 100 ha in size, emerging over time within large and small treeless patches. One of the primary considerations for young stands is how they respond to recurring fire (North et al., 2019, North et al., 2021). Fire behavior in young stands will be influenced by fire weather, species traits, canopy architecture, stand density, and surrounding landscape conditions among many other factors (Bellows et al., 2016), so the effects of fire on young stands can be difficult to predict. In general, however, high-density plantations are particularly vulnerable to burning at high-severity (Thompson et al., 2007, North et al., 2019), particularly if maintenance treatments such as pre-commercial thinning are not performed.

3.5.2. Intervention strategy #1: Fire management in young stands

The complete loss of young stands in subsequent fires is rarely desirable, particularly if considerable resources have been invested in their establishment (Fargione et al., 2021). However, some degree of mortality may be desirable as a stand maintenance technique (North et al., 2019). Thus, young stands may increasingly need to be considered in a pyrosilvicultural framework where the recurrence of fire is accepted and planned for, rather than discouraged and avoided in favor of mechanical treatments and fire suppression tactics (North et al., 2021).

Research on young stand responses to wildfire is very scarce in the southwestern US, and not much more developed elsewhere. For Douglas-fir and ponderosa pine populations in New Mexico and Colorado, individual-level fire tolerance increases with tree size and age, with individuals of these species estimated to become tolerant of low-severity (<30% basal area mortality in this case) fire above 10 and 6.9 cm DBH, respectively (32 and 25 years of age respectively) under relatively open growing conditions (Rodman et al., 2021). Initial evidence from California, where fall prescribed fire was introduced to small (<1 ha) stands of different ages, suggests that a significant reduction in fire-caused mortality in mixed-conifer species occurs between 22 and 32 years, with mortality dropping from ~ 40% to ~ 20% (York et al., 2021). Spring burns appear to cause greater mortality in young trees than fall burns (Bellows et al., 2016), perhaps because of increased sensitivity during annual growth initiation. Even among 12-year-old stands, fall prescribed fire mortality rates were around 50% rather than complete stand mortality (York et al., 2021). If initial tree densities are high (>800 trees ha^{-1} in York et al., 2021), such mortality may foster the creation of lower density stands and favor more vigorous trees of more fire-resistant species, without creating as much woody surface fuel as would arise from fire in older stands.

Fire suppression, on the other hand, would potentially re-create the hazard of overly dense stands if mechanical follow-up treatments are not implemented. Alternative strategies for artificial stand initiation could include low-density plantations that attempt to mimic natural stand heterogeneity, with groups, gaps, and widely spaced individual trees, from the outset (Churchill et al., 2013, North et al., 2019. The nucleation plantation strategy discussed above may be particularly appealing for young stand resilience, as the dispersion of founder stands across the landscape may represent an alternative form of fire resilience via spatial bet-hedging: some stands may experience complete mortality in future fires but others may partly survive due to variation in fire weather and fuel conditions at the time they burn (Barton and Poulos 2018, McIver and Ottmar 2018, North et al., 2019).

4. Discussion

While the tools and techniques for postfire management presented here are varied and context-dependent, collectively they highlight the need for all postfire management actions to be placed into a landscape context. Given well-documented increases in burned area (Fig. 1), management of recently burned forests would be well served by a strategic approach, rather than treated as an afterthought (Meyer et al., 2021). Importantly, recently burned forests present an opportunity: the myriad effects created by wildfires, particularly those burning through previously fire-excluded forests, introduce heterogeneity that may be beneficial for multiple ecosystem processes (North et al., 2019). Postfire landscapes can thus be leveraged to reinforce conditions where they are desirable and to redirect trajectories where undesirable.

When viewed in a landscape context, postfire forest conditions can be managed holistically (Box 2). For instance, fuels management can be implemented around valued small forest refugia and small treeless patches, while at the same time nucleation plantings can be installed into climatically suitable portions of large treeless patches (Box 2). These types of targeted interventions are a break from conventional management strategies that apply prescriptions more broadly, yet they are facilitated and prioritized using a landscape perspective. Such actions that are more limited in scope still serve to promote fine-grained heterogeneity on the landscape, by using existing landscape features as the template to restore future variability in forest structure.

Box 2
: Postfire landscape partitioning and management: Jemez Mountains (NM) Case Study.

The 2011 Las Conchas fire in the Jemez Mountains of northern New Mexico burned 61,025 ha and left a patchwork of forested and treeless patches of varying sizes (Fig. B2a), as determined by a high-resolution aerial imagery analysis (Walker et al., 2019). Approximately 72% of the postfire landscape is in large (>100 ha) treeless patches, 3% is in small treeless patches, 6% is in small conifer forest patches, and 19% is in large conifer forest patches. The fire spanned multiple jurisdictional boundaries, including National Forest, National Park Service, Tribal, and private lands.

In 2016, land managers, scientists, Native American tribal members, and NGOs convened a working group, called the East Jemez Landscape Futures (EJLF), to identify strategies that would address the restoration needs in the fire-, flood-, and drought-altered landscapes of the Jemez Mountains. Since the launch of the EJLF group, several successful projects have been funded, focusing on planning and implementation of different management actions based on the configuration of postfire patches (Stortz et al., 2018). Specifically, implementation efforts have included a reforestation effort aimed at nucleation planting to create "bottom-up" heterogeneity, and reduction of "legacy" fuels through mechanical thinning and controlled burns.

Large treeless patches (Fig. B2b) were targeted for reforestation within Bandelier National Monument, the Santa Fe National Forest, and Santa Clara Pueblo. A reforestation model of future (years 2031–2060) climate suitability for ponderosa pine was applied to large treeless patches > 120 m from patch edges (Rodman et al., 2020a), which then guided the placement of planting units. Importantly, some of the target locations and species were selected by Native American groups because of traditional or ceremonial purposes and needs. Units were not uniformly planted, but rather were planted beginning in 2020 using a nucleation strategy of 0.1–0.5 ha plots spaced at least 240 m apart, with a mixture of ponderosa pine and Douglas-fir, the ratio of which varied according to the reforestation model and tribal considerations.

Small forest patches, and the edges of certain larger forest patches, are under consideration for targeted fuels management (Fig. B2c) to "harden" the patch edges, reduce subsequent wildfire intensity, and reduce the risk of further tree mortality within the small patch. Options for fuels management include mastication of live vegetation such as shrubs, hand piling and burning of dead logs, and controlled burn operations to remove coarse woody fuels (Box 1). Buffer zones extend 70 m on either side of the forest patch edges, which is a reasonable distance over which burn severity decreases as fire enters a treated forest edge (Safford et al., 2012), leaving some untreated area within even small forest patches. Priority edges were identified based on patches forming the boundary between larger intact forest and larger treeless patches. Small, isolated forest patches surrounded by large treeless areas (red arrows, Figure B3c) are prioritized for conservation (Coop et al., 2019).

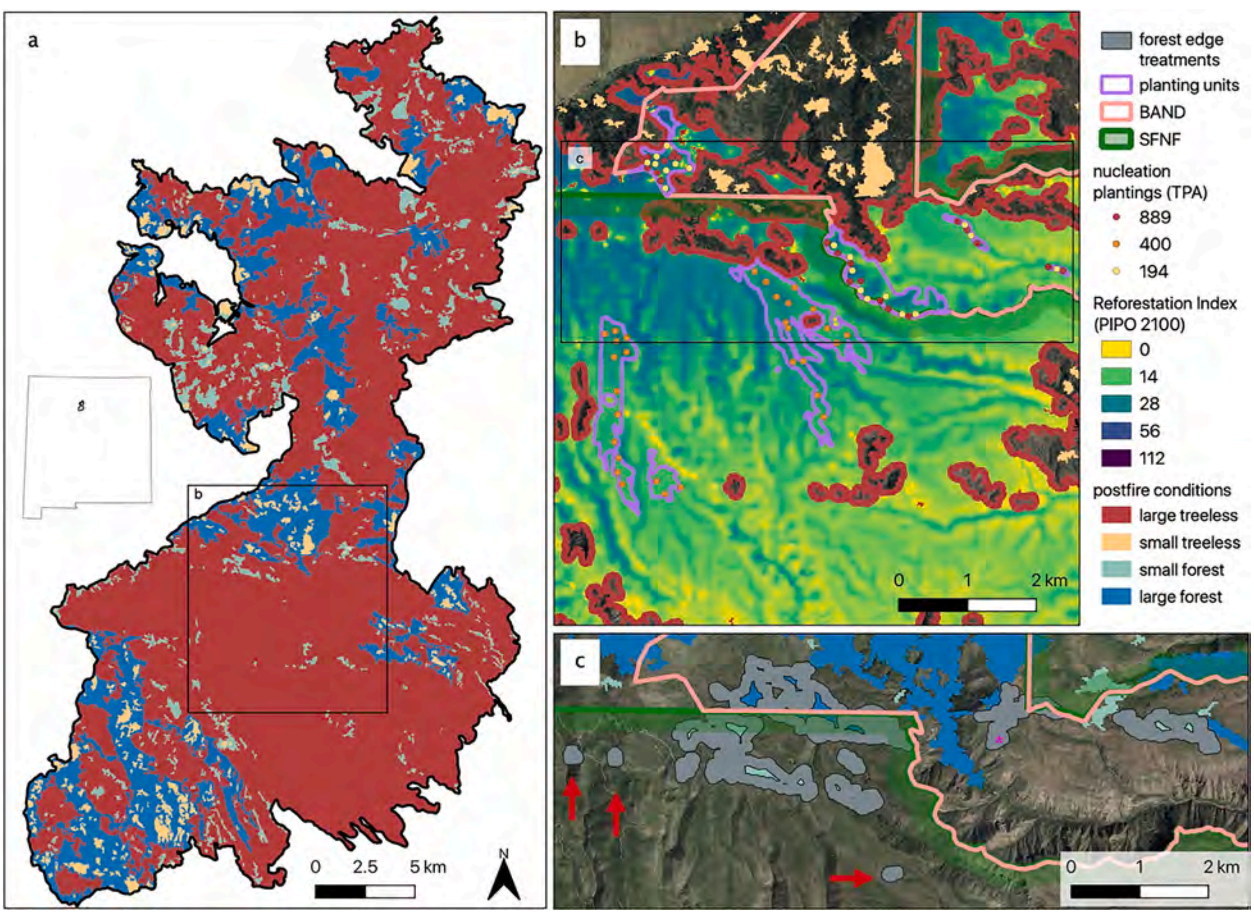


Fig. B2. An example postfire landscape management strategy for the Las Conchas fire in northern New Mexico (a). Inset area (b) shows a postfire management strategy for large treeless patches. A unitless model of future regeneration suitability (Reforestation Index; RI) for ponderosa pine (PIPO) was applied to the interior of large treeless patches, with larger RI values more climatically suitable for regeneration. Planting units were designated for Bandelier National Monument (BAND) and the Santa Fe National Forest (SFNF), based on RI, access and topographic constraints, and distance from live trees. Nucleation plantings are designed for three different densities (Trees Per Acre; TPA) of tree seedlings. Second inset area (c) shows a postfire management strategy for small and large forest patches ("refugia"), where a 70-m buffer inwards and outwards from the edges of refugia may be targeted for fuel treatments, e.g. via burning of down heavy fuels (Box 1). Highest priority refugia are shown with red arrows. The pink dot in c shows the location of the small forest refugia photographed in Fig. 5. Base imagery in (b, c) provided by Google. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Despite the breadth of landscape conditions discussed here, three unifying themes emerge to guide postfire management. First, an accurate assessment of postfire conifer cover is a critical foundation for management. While surviving conifer cover is not by itself predictive of other important postfire ecosystem conditions such as fuels, it does represent a critical form of ecological memory, providing residual carbon storage, regeneration potential, and structural complexity that can take decades to develop once lost (Johnstone et al., 2016).

Second, anticipation of future fire and climate change would benefit all management actions (Box 1, 2; North et al., 2021). In particular, management may need to plan for multiple climate futures that will all likely involve more fire than western forests have experienced in the recent past (Hessburg et al., 2019). This uncertainty may require managers to apply a broad suite of management interventions to a particular landscape, flexibly using available tools at many scales and at different times to guide or adjust landscape trajectories.

Third, management can be deliberate in its goals but still flexible in its approaches (Fig. 2). Some of the approaches discussed here are novel and experimental; thus an adaptive management approach that implements multiple solutions and, critically, invests the time and resources to explicitly monitor and learn from that implementation, will dramatically increase the utility of management techniques in a landscape context – even those that fail to meet objectives (Larson et al., 2013b, Puettmann and Messier 2020). In many cases a substantial proportion of the landscape may be 'accepted' to continue on its current trajectory (Fig. 2), a decision likely to be driven by a complex mix of considerations including local capacity, funding limitations, manager priorities, jurisdictional mandates, and biophysical constraints (Stortz et al., 2018).

A focus on wildlife habitat is one consideration that squarely places postfire management actions into a landscape context. Surviving forest cover anchors our landscape management framework, and assessing its value as wildlife habitat can be a starting point for directing conservation and management, especially for at-risk species (Andrus et al., 2021). A wildlife perspective recognizes that animals move between habitat patches of differing fire histories across a range of spatial scales (Nimmo et al., 2019), and that birds (Hutto et al., 2016, Tingley et al., 2016) and bats (Steel et al., 2019, Blakey et al., 2021) in particular utilize a complex mosaic of burn severity at multiple scales. Managing for wildlife habitat can bridge a diverse suite of values to define common ground for addressing the unique challenges of these complex postfire landscapes, integrate scientific and traditional knowledge and values (Slaton et al., 2019), and engage citizen science over the geographic extent of large fire events (Kirchhoff et al., 2021).

Though the focus of this paper is on postfire landscape management, we would be remiss to overlook the importance of interventions prior to fire in fire-excluded, formerly frequent-fire forests. We do not emphasize postfire management at the expense of pre-fire management, but in recognition that in fire-prone forests, all landscape conditions are subject to short-interval and potentially dramatic shifts in vegetation. Burned landscapes are increasingly recognized as providing value to adjacent unburned landscapes, for instance as anchors for managed wildfire, and as fuel treatments themselves (Parks et al., 2015, Thompson et al., 2016, Huffman et al., 2020). The opportunity to use burned areas as anchors is time sensitive due to fuel recovery rates, and their effectiveness for this purpose may be greatest within 5–15 years of the initial fire (Yocom et al., 2019). Management of unburned forests may benefit from even more explicit consideration of how pre-fire actions are likely to manifest in postfire landscape patterns, e.g., how fuels treatments might promote refugia formation or retention during future wildfires.

Because local forests, fire regimes, and fire incidents vary widely over time and space, and management objectives vary by jurisdiction and valued priorities, postfire landscape management is not "one-size-fits-all", nor will appropriate management actions remain the same over time. In recognition of this need for flexibility, the Southwest FireCLIME project, in conjunction with the Northern Institute of Applied Climate Science (NIACS), has developed a "menu" of adaptation strategies and approaches for managing forests and fire in the context of climate change. The menu was co-produced by scientists and managers in the Southwest based on many of the concepts discussed here, and is available to remind practitioners of the various options available and to spark new ideas about managing landscapes today. The menu is available at https://swfireclime.org/fire-climate-adaptation-tools/.

Toolboxes, menus of options, and science-based postfire recommendations should be viewed as starting points for managers working on burned landscapes, particularly those impacted by stand-replacing fire. In many cases, the uncertainties associated with managing these landscapes may seem daunting. Starting with lessons learned and vetted approaches can help reduce the uncertainties surrounding where to act and what to do, but unique landscape conditions and social values will determine the ultimate actions. Additionally, many communities with ties to the landscape are also recovering, physically and emotionally. Engaging with stakeholder groups to identify restoration action and interventions can help to further reduce the uncertainty people may feel. As community members adapt to different conditions, so too may their values and expectations of landscape conditions. Through engagement with local community groups, land managers can play a part in the postfire recovery and healing of fire-impacted communities by creating situations that are "win–win," with favorable ecosystem management and human community outcomes.

CRediT authorship contribution statement

Jens T. Stevens: Conceptualization, Formal analysis, Investigation, Project administration, Formal analysis, Visualization, Writing – original draft, Writing – review & editing. **Collin M. Haffey:** Conceptualization, Funding acquisition, Investigation, Writing – original draft, Writing – review & editing. **Jonathan D. Coop:** Investigation, Writing – original draft, Writing – review & editing. **Paula J. Fornwalt:** Investigation, Visualization, Writing – original draft, Writing – review & editing. **Larissa Yocom:** Investigation, Visualization, Writing – original draft, Writing – review & editing. **Craig D. Allen:** Funding acquisition, Supervision, Writing – original draft, Writing – review & editing. **Anne Bradley:** Funding acquisition, Writing – original draft, Writing – review & editing. **Owen T. Burney:** Visualization, Writing – original draft, Writing – review & editing. **Dennis Carril:** Writing – original draft, Writing – review & editing. **Marin E. Chambers:** Writing – original draft, Writing – review & editing. **Teresa B. Chapman:** Writing – original draft, Writing – review & editing. **Sandra L. Haire:** Writing – original draft, Writing – review & editing. **Matthew D. Hurteau:** Writing – original draft, Writing – review & editing. **Jose M. Iniguez:** Writing – original draft, Writing – review & editing. **Ellis Q. Margolis:** Funding acquisition, Supervision, Writing – original draft, Writing – review & editing. **Christopher Marks:** Visualization, Writing – original draft, Writing – review & editing. **Laura A.E. Marshall:** Visualization, Writing – original draft, Writing – review & editing. **Kyle C. Rodman:** Formal analysis, Visualization, Writing – original draft, Writing – review & editing. **Camille S. Stevens-Rumann:** Writing – original draft, Writing – review & editing. **Andrea E. Thode:** Writing – original draft, Writing – review & editing. **Jessica J. Walker:** Visualization, Writing – original draft, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2021.119678.

References

Forest and Rangeland Renewable Resources Planning Act of 1974 (88 Stat. 476, as amended; 16 U.S.C. 1601-1610).
Abatzoglou, J.T., Kolden, C.A., Balch, J.K., Bradley, B.A., 2016. Controls on interannual variability in lightning-caused fire activity in the western US. Environ. Res. Lett. 11, 045005.
Abatzoglou, J.T., Williams, A.P., 2016. Impact of anthropogenic climate change on wildfire across western US forests. Proc. Natl. Acad. Sci. 113, 11770–11775.
Abella, S.R., Fornwalt, P.J., 2015. Ten years of vegetation assembly after a North American mega fire. Glob. Change Biol. 21, 789–802.
Allen, C.D., Savage, M., Falk, D.A., Suckling, K.F., Swetnam, T.W., Schulke, T., Stacey, P. B., Morgan, P., Hoffman, M., Klingel, J.T., 2002. Ecological restoration of Southwestern ponderosa pine ecosystems: A broad perspective. Ecol. Appl. 12, 1418–1433.
Andrus, R.A., Martinez, A.J., Jones, G.M., Meddens, A.J.H., 2021. Assessing the quality of fire refugia for wildlife habitat. For. Ecol. Manage. 482, 118868.
Aplet, G. H., and D. N. Cole. 2010. The trouble with naturalness: rethinking park and wilderness goals. Beyond naturalness: Rethinking park and wilderness stewardship in an era of rapid change. 12, 21–22.
Backer, D.M., Jensen, S.E., McPherson, G.R., 2004. Impacts of Fire-Suppression Activities on Natural Communities. Conserv. Biol. 18, 937–946.
Barton, A.M., Poulos, H.M., 2018. Pine vs. oaks revisited: Conversion of Madrean pine-oak forest to oak shrubland after high-severity wildfire in the Sky Islands of Arizona. For. Ecol. Manage. 414, 28–40.
Boyden, S., Binkley, D., Shepperd, W., 2005. Spatial and temporal patterns in structure, regeneration, and mortality of an old-growth ponderosa pine forest in the Colorado Front Range. Forest Ecology and Management 219, 43–55. https://doi.org/10.1016/j.foreco.2005.08.041.
Battaglia, M., Smith, F.W., Shepperd, W.D., 2009. Predicting mortality of ponderosa pine regeneration after prescribed fire in the Black Hills, South Dakota, USA. Int. J. Wildland Fire 18, 176–190.
Bellows, R.S., Thomson, A.C., Helmstedt, K.J., York, R.A., Potts, M.D., 2016. Damage and mortality patterns in young mixed conifer plantations following prescribed fires in the Sierra Nevada, California. For. Ecol. Manage. 376, 193–204.
Blakey, R.V., Webb, E.B., Kesler, D.C., Siegel, R.B., Corcoran, D., Cole, J.S., Johnson, M., 2021. Extent, configuration and diversity of burned and forested areas predict bat richness in a fire-maintained forest. Landscape Ecol. 36, 1101–1115.
Brown, P.M., Wu, R., 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. Ecology 86, 3030–3038.
Buma, B., 2015. Disturbance interactions: characterization, prediction, and the potential for cascading effects. Ecosphere 6 (art70).
Buma, B., Weiss, S., Hayes, K., Lucash, M., 2020. Wildland fire reburning trends across the US West suggest only short-term negative feedback and differing climatic effects. Environ. Res. Lett. 15, 034026.
Buma, B., Wessman, C.A., 2011. Disturbance interactions can impact resilience mechanisms of forests. Ecosphere 2 (art64).
Burney, O., Wing, M.G., Rose, R., 2007. Microsite influences on variability in Douglas-fir seedling development. West. J. Appl. For. 22, 156–162.
Camp, A., Oliver, C., Hessburg, P., Everett, R., 1997. Predicting late-successional fire refugia pre-dating European settlement in the Wenatchee Mountains. For. Ecol. Manage. 95, 63–77.
Cansler, C.A., Swanson, M.E., Furniss, T.J., Larson, A.J., Lutz, J.A., 2019. Fuel dynamics after reintroduced fire in an old-growth Sierra Nevada mixed-conifer forest. Fire Ecology 15, 16.
Cartwright, J.M., Littlefield, C.E., Michalak, J.L., Lawler, J.J., Dobrowski, S.Z., 2020. Topographic, soil, and climate drivers of drought sensitivity in forests and shrublands of the Pacific Northwest, USA. Sci. Rep. 10, 18486.
Castro, J., Allen, C.D., Molina-Morales, M., Marañón-Jiménez, S., Sánchez-Miranda, Á., Zamora, R., 2011. Salvage logging versus the use of burnt wood as a nurse object to promote post-fire tree seedling establishment. Restor. Ecol. 19, 537–544.
Chamberlain, C.P., Sánchez Meador, A.J., Thode, A.E., 2021. Airborne lidar provides reliable estimates of canopy base height and canopy bulk density in southwestern ponderosa pine forests. For. Ecol. Manage. 481, 118695.
Chambers, M.E., Fornwalt, P.J., Malone, S.L., Battaglia, M.A., 2016. Patterns of conifer regeneration following high severity wildfire in ponderosa pine – dominated forests of the Colorado Front Range. For. Ecol. Manage. 378, 57–67.
Chapman, T.B., Schoennagel, T., Veblen, T.T., Rodman, K.C., 2020. Still standing: Recent patterns of post-fire conifer refugia in ponderosa pine-dominated forests of the Colorado Front Range. PLoS ONE 15, e0226926.
Churchill, D.J., Larson, A.J., Dahlgreen, M.C., Franklin, J.F., Hessburg, P.F., Lutz, J.A., 2013. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. For. Ecol. Manage. 291, 442–457.
Collins, B.M., Lydersen, J.M., Everett, R.G., Stephens, S.L., 2018. How does forest recovery following moderate-severity fire influence effects of subsequent wildfire in mixed-conifer forests? Fire Ecology 14, 3.
Collins, B.M., Stevens, J.T., Miller, J.D., Stephens, S.L., Brown, P.M., North, M.P., 2017. Alternative characterization of forest fire regimes: incorporating spatial patterns. Landscape Ecol. 32, 1543–1552.
Collins, L., Bennett, A.F., Leonard, S.W.J., Penman, T.D., 2019. Wildfire refugia in forests: Severe fire weather and drought mute the influence of topography and fuel age. Glob. Change Biol. 25, 3829–3843.
Cook-Patton, S.C., Gopalakrishna, T., Daigneault, A., Leavitt, S.M., Platt, J., Scull, S.M., Amarjargal, O., Ellis, P.W., Griscom, B.W., McGuire, J.L., Yeo, S.M., Fargione, J.E., 2020. Lower cost and more feasible options to restore forest cover in the contiguous United States for climate mitigation. One Earth 3, 739–752.
Coop, J.D., DeLory, T.J., Downing, W.M., Haire, S.L., Krawchuk, M.A., Miller, C., Parisien, M.-A., Walker, R.B., 2019. Contributions of fire refugia to resilient ponderosa pine and dry mixed-conifer forest landscapes. Ecosphere 10, e02809.
Coop, J.D., Parks, S.A., McClernan, S.R., Holsinger, L.M., 2016. Influences of prior wildfires on vegetation response to subsequent fire in a reburned Southwestern landscape. Ecol. Appl. 26, 346–354.
Coop, J.D., Parks, S.A., Stevens-Rumann, C.S., Crausbay, S.D., Higuera, P.E., Hurteau, M. D., Tepley, A., Whitman, E., Assal, T., Collins, B.M., Davis, K.T., Dobrowski, S., Falk, D.A., Fornwalt, P.J., Fulé, P.Z., Harvey, B.J., Kane, V.R., Littlefield, C.E., Margolis, E.Q., North, M., Parisien, M.-A., Prichard, S., Rodman, K.C., 2020. Wildfire-driven forest conversion in western North American landscapes. Bioscience 70, 659–673.
Coppoletta, M., Merriam, K.E., Collins, B.M., 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecol. Appl. 26, 686–699.
Corbin, J.D., Holl, K.D., 2012. Applied nucleation as a forest restoration strategy. For. Ecol. Manage. 265, 37–46.
Davis, K.T., Dobrowski, S.Z., Higuera, P.E., Holden, Z.A., Veblen, T.T., Rother, M.T., Parks, S.A., Sala, A., Maneta, M.P., 2019. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proc. Natl. Acad. Sci. 116, 6193–6198.
Davis, K.T., Higuera, P.E., Dobrowski, S.Z., Parks, S.A., Abatzoglou, J.T., Rother, M.T., Veblen, T.T., 2020. Fire-catalyzed vegetation shifts in ponderosa pine and Douglas-fir forests of the western United States. Environ. Res. Lett. 15:1040b1048.
Dobrowski, S.Z., Swanson, A.K., Abatzoglou, J.T., Holden, Z.A., Safford, H.D., Schwartz, M.K., Gavin, D.G., 2015. Forest structure and species traits mediate projected recruitment declines in western US tree species. Glob. Ecol. Biogeogr. 24, 917–927.
Downing, W.M., Krawchuk, M.A., Coop, J.D., Meigs, G.W., Haire, S.L., Walker, R.B., Whitman, E., Chong, G., Miller, C., Tortorelli, C., 2020. How do plant communities differ between fire refugia and fire-generated early-seral vegetation? J. Veg. Sci. 31, 26–39.
Downing, W.M., Krawchuk, M.A., Meigs, G.W., Haire, S.L., Coop, J.D., Walker, R.B., Whitman, E., Chong, G., Miller, C., 2019. Influence of fire refugia spatial pattern on post-fire forest recovery in Oregon's Blue Mountains. Landscape Ecol. 34, 771–792.
Downing, W.M., Meigs, G.W., Gregory, M.J., Krawchuk, M.A., 2021. Where and why do conifer forests persist in refugia through multiple fire events? Glob. Change Biol.
Dumroese, K.R., Landis, T.D., Pinto, J.R., Haase, D.L., Wilkinson, K.W., Davis, A.S., 2016. Meeting forest restoration challenges: using the target plant concept. Reforesta 1, 37–52.
Dumroese, R.K., Balloffet, N., Crockett, J.W., Stanturf, J.A., Nave, L.E., 2019. A national approach to leverage the benefits of tree planting on public lands. New Forest. 50, 1–9.
Dunn, C.J., Bailey, J.D., 2015. Modeling the direct effects of salvage logging on long-term temporal fuel dynamics in dry-mixed conifer forests. For. Ecol. Manage. 341, 93–109.
Fargione, J., D. L. Haase, O. T. Burney, O. A. Kildisheva, G. Edge, S. C. Cook-Patton, T. Chapman, A. Rempel, M. D. Hurteau, K. T. Davis, S. Dobrowski, S. Enebak, R. De La Torre, A. A. R. Bhuta, F. Cubbage, B. Kittler, D. Zhang, and R. W. Guldin. 2021. Challenges to the reforestation pipeline in the United States. Front. Forests Global Change 4.
Fornwalt, P., Huckaby, L., Alton, S., Kaufmann, M., Brown, P., Cheng, A., 2016. Did the 2002 Hayman Fire, Colorado, USA, burn with uncharacteristic severity. Fire Ecology 12, 117–132.
Fornwalt, P.J., Kaufmann, M.R., Stohlgren, T.J., 2010. Impacts of mixed severity wildfire on exotic plants in a Colorado ponderosa pine–Douglas-fir forest. Biol. Invasions 12, 2683–2695.
Fornwalt, P.J., Stevens-Rumann, C.S., Collins, B.J., 2018. Overstory structure and surface cover dynamics in the decade following the Hayman Fire. Colorado. Forests 9, 152.
Gale, M.G., Cary, G.J., Van Dijk, A.I.J.M., Yebra, M., 2021. Forest fire fuel through the lens of remote sensing: Review of approaches, challenges and future directions in the

remote sensing of biotic determinants of fire behaviour. Remote Sens. Environ. 255, 112282.
Greene, D.F., Johnson, E.A., 2000. Tree recruitment from burn edges. Can. J. For. Res. 30, 1264–1274.
Guiterman, C.H., Margolis, E.Q., Allen, C.D., Falk, D.A., Swetnam, T.W., 2018. Long-term persistence and fire resilience of oak shrubfields in dry conifer forests of northern New Mexico. Ecosystems 21, 943–959.
Guiterman, C.H., Margolis, E.Q., Swetnam, T.W., 2015. Dendroecological methods for reconstructing high-severity fire in pine-oak forests. Tree-Ring Res. 71, 67–77.
Haase, D.L., Pike, C., Enebak, S., Mackey, L., Ma, Z., Rathjen, M., 2020. Forest nursery seedling production in the United States—fiscal year 2019. Tree Planter's Notes 63, 26–31.
Haffey, C., Sisk, T.D., Allen, C.D., Thode, A.E., Margolis, E.Q., 2018. Limits to ponderosa pine regeneration following large high-severity forest fires in the United States Southwest. Fire Ecol. 14, 143–163.
Hagmann, R. K., P. F. Hessburg, S. J. Prichard, N. A. Povak, P. M. Brown, P. Z. Fulé, R. E. Keane, E. E. Knapp, J. M. Lydersen, K. L. Metlen, M. J. Reilly, A. J. Sánchez Meador, S. L. Stephens, J. T. Stevens, A. H. Taylor, L. L. Yocom, M. A. Battaglia, D. J. Churchill, L. D. Daniels, D. A. Falk, P. Henson, J. D. Johnston, M. A. Krawchuk, C. R. Levine, G. W. Meigs, A. G. Merschel, M. P. North, H. D. Safford, T. W. Swetnam, and A. E. M. Waltz. 2021. Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. Ecological Applications Accepted Author Manuscript e02431.
Haire, S.L., Coop, J.D., Miller, C., 2017. Characterizing spatial neighborhoods of refugia following large fires in northern New Mexico USA. Land 6, 19.
Haire, S.L., McGarigal, K., 2010. Effects of landscape patterns of fire severity on regenerating ponderosa pine forests (Pinus ponderosa) in New Mexico and Arizona, USA. Landscape Ecol. 25, 1055–1069.
Hessburg, P.F., Churchill, D.J., Larson, A.J., Haugo, R.D., Miller, C., Spies, T.A., North, M.P., Povak, N.A., Belote, R.T., Singleton, P.H., Gaines, W.L., Keane, R.E., Aplet, G.H., Stephens, S.L., Morgan, P., Bisson, P.A., Rieman, B.E., Salter, R.B., Reeves, G.H., 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecol. 30, 1805–1835.
Hessburg, P.F., Miller, C.L., Parks, S.A., Povak, N.A., Taylor, A.H., Higuera, P.E., Prichard, S.J., North, M.P., Collins, B.M., Hurteau, M.D., Larson, A.J., Allen, C.D., Stephens, S.L., Rivera-Huerta, H., Stevens-Rumann, C.S., Daniels, L.D., Gedalof, Z.E., Gray, R.W., Kane, V.R., Churchill, D.J., Hagmann, R.K., Spies, T.A., Cansler, C.A., Belote, R.T., Veblen, T.T., Battaglia, M.A., Hoffman, C., Skinner, C.N., Safford, H.D., Salter, R.B., 2019. Climate, environment, and disturbance history govern resilience of western North American forests. Front. Ecol. Evolut. 7.
Hessburg, P.F., Spies, T.A., Perry, D.A., Skinner, C.N., Taylor, A.H., Brown, P.M., Stephens, S.L., Larson, A.J., Churchill, D.J., Povak, N.A., Singleton, P.H., McComb, B., Zielinski, W.J., Collins, B.M., Salter, R.B., Keane, J.J., Franklin, J.F., Riegel, G., 2016. Tamm Review: Management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. For. Ecol. Manage. 366, 221–250.
Higgins, A.M., Waring, K.M., Thode, A.E., 2015. The effects of burn entry and burn severity on ponderosa pine and mixed conifer forests in Grand Canyon National Park. Int. J. Wildland Fire 24, 495–506.
Higuera, P.E., Metcalf, A.L., Miller, C., Buma, B., McWethy, D.B., Metcalf, E.C., Ratajczak, Z., Nelson, C.R., Chaffin, B.C., Stedman, R.C., McCaffrey, S., Schoennagel, T., Harvey, B.J., Hood, S.M., Schultz, C.A., Black, A.E., Campbell, D., Haggerty, J.H., Keane, R.E., Krawchuk, M.A., Kulig, J.C., Rafferty, R., Virapongse, A., 2019. Integrating subjective and objective dimensions of resilience in fire-prone landscapes. Bioscience 69, 379–388.
Hobbs, R.J., Hallett, L.M., Ehrlich, P.R., Mooney, H.A., 2011. Intervention Ecology: Applying Ecological Science in the Twenty-first Century. Bioscience 61, 442–450.
Holden, Z.A., Morgan, P., Hudak, A.T., 2010. Burn Severity of Areas Reburned by Wildfires in the Gila National Forest, New Mexico, USA. Fire Ecology 6, 77–85.
Huffman, D.W., Crouse, J.E., Sánchez Meador, A.J., Springer, J.D., Stoddard, M.T., 2018. Restoration benefits of re-entry with resource objective wildfire on a ponderosa pine landscape in northern Arizona, USA. For. Ecol. Manage. 408, 16–24.
Huffman, D.W., Roccaforte, J.P., Springer, J.D., Crouse, J.E., 2020. Restoration applications of resource objective wildfires in western US forests: a status of knowledge review. Fire Ecology 16, 18.
Huffman, D.W., Sánchez Meador, A.J., Stoddard, M.T., Crouse, J.E., Roccaforte, J.P., 2017. Efficacy of resource objective wildfires for restoration of ponderosa pine (Pinus ponderosa) forests in northern Arizona. For. Ecol. Manage. 389, 395–403.
Huffman, D.W., Zegler, T.J., Fulé, P.Z., 2015. Fire history of a mixed conifer forest on the Mogollon Rim, northern Arizona, USA %J. Int. J. Wildland Fire. 24, 680–689.
Hunter, M.E., Iniguez, J.M., Farris, C.A., 2014. Historical and current fire management practices in two wilderness areas in the southwestern United States: The Saguaro Wilderness Area and the Gila-Aldo Leopold Wilderness Complex.
Hunter, M.E., Iniguez, J.M., Lentile, L.B., 2011. Short- and Long-Term Effects on Fuels, Forest Structure, and Wildfire Potential from Prescribed Fire and Resource Benefit Fire in Southwestern Forests, USA. Fire Ecol. 7, 108–121.
Hutto, R.L., Keane, R.E., Sherriff, R.L., Rota, C.T., Eby, L.A., Saab, V.A., 2016. Toward a more ecologically informed view of severe forest fires. 7:e01255.
Iniguez, J.M., Swetnam, T.W., Baisan, C.H., 2016. Fire history and moisture influences on historical forest age structure in the sky islands of southern Arizona, USA. J. Biogeogr. 43, 85–95.
Iniguez, J.M., Swetnam, T.W., Baisan, C.H., Sieg, C.H., Fulé, P.Z., Hunter, M., Allen, C.D., Brooks, M.L., Balice, R.G., 2009. Spatially and temporally variable fire regime on Rincon Peak, Arizona, USA. Fire Ecology 5, 3–21.
Johnson, M.C., Kennedy, M.C., Harrison, S.C., Churchill, D., Pass, J., Fischer, P.W., 2020. Effects of post-fire management on dead woody fuel dynamics and stand structure in a severely burned mixed-conifer forest, in northeastern Washington State, USA. For. Ecol. Manage. 470–471, 118190.
Johnstone, J.F., Allen, C.D., Franklin, J.F., Frelich, L.E., Harvey, B.J., Higuera, P.E., Mack, M.C., Meentemeyer, R.K., Metz, M.R., Perry, G.L.W., Schoennagel, T., Turner, M.G., 2016. Changing disturbance regimes, ecological memory, and forest resilience. Front. Ecol. Environ. 14, 369–378.
Jonas, J.L., Berryman, E., Wolk, B., Morgan, P., Robichaud, P.R., 2019. Post-fire wood mulch for reducing erosion potential increases tree seedlings with few impacts on understory plants and soil nitrogen. For. Ecol. Manage. 453, 117567.
Jones, G.M., Kramer, H.A., Whitmore, S.A., Berigan, W.J., Tempel, D.J., Wood, C.M., Hobart, B.K., Erker, T., Atuo, F.A., Pietrunti, N.F., Kelsey, R., Gutiérrez, R.J., Peery, M.Z., 2020. Habitat selection by spotted owls after a megafire reflects their adaptation to historical frequent-fire regimes. Landscape Ecol.
Kalies, E.L., Yocom Kent, L.L., 2016. Tamm Review: Are fuel treatments effective at achieving ecological and social objectives? A systematic review. For. Ecol. Manage. 375, 84–95.
Kemp, K.B., Higuera, P.E., Morgan, P., 2016. Fire legacies impact conifer regeneration across environmental gradients in the U.S. northern Rockies. Landscape Ecol. 31, 619–636.
Kemp, K.B., Higuera, P.E., Morgan, P., Abatzoglou, J.T., 2019. Climate will increasingly determine post-fire tree regeneration success in low-elevation forests, Northern Rockies, USA. Ecosphere 10, e02568.
Kennedy, M.C., Johnson, M.C., 2014. Fuel treatment prescriptions alter spatial patterns of fire severity around the wildland–urban interface during the Wallow Fire, Arizona, USA. For. Ecol. Manage. 318, 122–132.
Kennedy, M.C., Johnson, M.C., Harrison, S.C., 2021. Model predictions of postwildfire woody fuel succession and fire behavior are sensitive to fuel dynamics parameters. Forest Science 67, 30–42.
Kerns, B.K., Tortorelli, C., Day, M.A., Nietupski, T., Barros, A.M.G., Kim, J.B., Krawchuk, M.A., 2020. Invasive grasses: A new perfect storm for forested ecosystems? For. Ecol. Manage. 463, 117985.
Key, C.H., Benson, N.C., 2006. Landscape assessment: remote sensing of severity, the Normalized Burn Ratio. Pages LA25 - LA41. In: D. C. Lutes, editor. FIREMON: Fire effects monitoring and inventory system Ogden, Utah: USDA Forest Service, Rocky Mountain Res. Station, Fort Collins, Colorado, USA.
Keyser, A.R., Krofcheck, D.J., Remy, C.C., Allen, C.D., Hurteau, M.D., 2020. Simulated increases in fire activity reinforce shrub conversion in a southwestern US forest. Ecosystems 23, 1702–1713.
Keyser, T.L., Smith, F.W., Shepperd, W.D., 2009. Short-term impact of post-fire salvage logging on regeneration, hazardous fuel accumulation, and understorey development in ponderosa pine forests of the Black Hills, SD, USA. Int. J. Wildland Fire 18, 451–458.
Kirchhoff, C., Callaghan, C.T., Keith, D.A., Indiarto, D., Taseski, G., Ooi, M.K.J., Le Breton, T.D., Mesaglio, T., Kingsford, R.T., Cornwell, W.K., 2021. Rapidly mapping fire effects on biodiversity at a large-scale using citizen science. Sci. Total Environ. 755, 142348.
Kolb, T., Dixit, A., Burney, O., 2019. Challenges and opportunities for maintaining ponderosa pine forests in the southwestern United States. Tree Planter's Notes 62, 104–112.
Kolden, C.A., 2019. We're Not Doing Enough Prescribed Fire in the Western United States to Mitigate Wildfire Risk. Fire 2, 30.
Kolden, C.A., Bleeker, T.M., Smith, A.M.S., Poulos, H.M., Camp, A.E., 2017. Fire Effects on Historical Wildfire Refugia in Contemporary Wildfires. Forests 8, 400.
Koontz, M.J., North, M.P., Werner, C.M., Fick, S.E., Latimer, A.M., 2020. Local forest structure variability increases resilience to wildfire in dry western U.S. coniferous forests. Ecol. Lett. 23, 483–494.
Korb, J.E., Fornwalt, P.J., Stevens-Rumann, C.S., 2019. What drives ponderosa pine regeneration following wildfire in the western United States? For. Ecol. Manage. 454, 117663.
Korb, J.E., Johnson, N.C., Covington, W.W., 2004. Slash pile burning effects on soil biotic and chemical properties and plant establishment: recommendations for amelioration. Restor. Ecol. 12, 52–62.
Kramer, A.T., Crane, B., Downing, J., Hamrick, J.L., Havens, K., Highland, A., Jacobi, S. K., Kaye, T.N., Lonsdorf, E.V., Ramp Neale, J., Novy, A., Smouse, P.E., Tallamy, D. W., White, A., Zeldin, J., 2019. Sourcing native plants to support ecosystem function in different planting contexts. 27, 470–476.
Krawchuk, M.A., Haire, S.L., Coop, J., Parisien, M.-A., Whitman, E., Chong, G., Miller, C., 2016. Topographic and fire weather controls of fire refugia in forested ecosystems of northwestern North America. 7, e01632.
Krawchuk, M.A., Meigs, G.W., Cartwright, J.M., Coop, J.D., Davis, R., Holz, A., Kolden, C., Meddens, A.J., 2020. Disturbance refugia within mosaics of forest fire, drought, and insect outbreaks. Front. Ecol. Environ. 18, 235–244.
Landis, T., Dumroese, R., Haase, D.J.U., Forest Service, Washington DC, 2010. Seedling processing, storage, and outplanting, Vol. 7, The container tree nursery manual.3-11.
Landis, T.D., 2011. The target plant concept-a history and brief overview. In Riley, LE; Haase, DL; Pinto, JR, tech. coords. National Proceedings: Forest and Conservation Nursery Associations-2010. Proc. RMRS-P-65. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. p. 61-66.
Lanini, W.T., Radosevich, S.R., 1986. Response of 3 conifer species to site preparation and shrub control. Forest Science 32, 61–77.
Larson, A.J., Belote, R.T., Cansler, C.A., Parks, S.A., Dietz, M., 2013a. Latent resilience in ponderosa pine forest: effects of resumed frequent fire. Ecological. Applications.
Larson, A.J., Belote, R.T., Williamson, M.A., Aplet, G.H., 2013b. Making monitoring count: project design for active adaptive management. J. Forest. 111, 348–356.

Leverkus, A.B., Buma, B., Wagenbrenner, J., Burton, P.J., Lingua, E., Marzano, R., Thorn, S., 2021. Tamm review: Does salvage logging mitigate subsequent forest disturbances? For. Ecol. Manage. 481, 118721.
Littlefield, C.E., Dobrowski, S.Z., Abatzoglou, J.T., Parks, S.A., Davis, K.T., 2020. A climatic dipole drives short- and long-term patterns of postfire forest recovery in the western United States. Proc. Natl. Acad. Sci. 202007434.
Lonergan, E.R., Cripps, C.L., Smith, C.M.J.F.S., 2014. Influence of site conditions, shelter objects, and ectomycorrhizal inoculation on the early survival of whitebark pine seedlings planted in Waterton Lakes National Park. 60, 603–612.
Lutz, J.A., Struckman, S., Furniss, T.J., Cansler, C.A., Germain, S.J., Yocom, L.L., McAvoy, D.J., Kolden, C.A., Smith, A.M.S., Swanson, M.E., Larson, A.J., 2020. Large-diameter trees dominate snag and surface biomass following reintroduced fire. Ecological Processes 9, 41.
Lydersen, J.M., Collins, B.M., Coppoletta, M., Jaffe, M.R., Northrop, H., Stephens, S.L., 2019. Fuel dynamics and reburn severity following high-severity fire in a Sierra Nevada, USA, mixed-conifer forest. Fire Ecology 15, 43.
Lydersen, J.M., Collins, B.M., Miller, J.D., Fry, D.L., Stephens, S.L., 2016. Relating fire-caused change in forest structure to remotely sensed estimates of fire severity. Fire Ecology 12, 99–116.
Lindenmayer, D., 2019. Small patches make critical contributions to biodiversity conservation. Proceedings of the National Academy of Sciences 116, 717–719. https://doi.org/10.1073/pnas.1820169116.
Lydersen, J.M., North, M.P., Knapp, E.E., Collins, B.M., 2013. Quantifying spatial patterns of tree groups and gaps in mixed-conifer forests: Reference conditions and long-term changes following fire suppression and logging. For. Ecol. Manage. 304, 370–382.
Malone, S., Fornwalt, P., Battaglia, M., Chambers, M., Iniguez, J., Sieg, C., 2018. Mixed-severity fire fosters heterogeneous spatial patterns of conifer regeneration in a dry conifer forest. Forests 9, 45.
Manning, A.D., Fischer, J., Lindenmayer, D.B., 2006. Scattered trees are keystone structures – Implications for conservation. Biol. Conserv. 132, 311–321.
Margolis, E.Q., Swetnam, T.W., Allen, C.D., 2007. A stand-replacing fire history in upper montane forests of the southern Rocky Mountains. Can. J. For. Res. 37, 2227–2241.
Margolis, E.Q., Swetnam, T.W., Allen, C.D., 2011. Historical stand-replacing fire in upper montane forests of the Madrean Sky Islands and Mogollon Plateau, Southwestern USA. Fire Ecology 7, 88–107.
Margolis, E.Q., Woodhouse, C.A., Swetnam, T.W., 2017. Drought, multi-seasonal climate, and wildfire in northern New Mexico. Clim. Change 142, 433–446.
Marshall, L.A., Falk, D.A., 2020. Demographic trends in community functional tolerance reflect tree responses to climate and altered fire regimes. Ecol. Appl. 30, e02197.
Martinez, A., Meddens, A., Kolden, C., Hudak, A., 2019. An assessment of fire refugia importance criteria ranked by land managers. Fire 2, 27.
McIver, J.D., Ottmar, R., 2007. Fuel mass and stand structure after post-fire logging of a severely burned ponderosa pine forest in northeastern Oregon. For. Ecol. Manage. 238, 268–279.
McIver, J.D., Ottmar, R., 2018. Fuel mass and stand structure 13 years after logging of a severely burned ponderosa pine forest in northeastern Oregon, U.S.A. For. Ecol. Manage. 424, 505–518.
McKinney, S.T., 2019. Systematic review and meta-analysis of fire regime research in ponderosa pine (Pinus ponderosa) ecosystems, Colorado, USA. Fire Ecology 15, 38.
McWethy, D.B., Schoennagel, T., Higuera, P.E., Krawchuk, M., Harvey, B.J., Metcalf, E. C., Schultz, C., Miller, C., Metcalf, A.L., Buma, B., Virapongse, A., Kulig, J.C., Stedman, R.C., Ratajczak, Z., Nelson, C.R., Kolden, C., 2019. Rethinking resilience to wildfire. Nature Sustainability 2, 797–804.
Meadow, A.M., Ferguson, D.B., Guido, Z., Horangic, A., Owen, G., Wall, T.J.W., Climate, and Society. 2015. Moving toward the deliberate coproduction of climate science knowledge. 7:179-191.
Meddens, A.J.H., Kolden, C.A., Lutz, J.A., Abatzoglou, J.T., Hudak, A.T., 2018a. Spatiotemporal patterns of unburned areas within fire perimeters in the northwestern United States from 1984 to 2014. Ecosphere 9, e02029.
Meddens, A.J.H., Kolden, C.A., Lutz, J.A., Smith, A.M.S., Cansler, C.A., Abatzoglou, J.T., Meigs, G.W., Downing, W.M., Krawchuk, M.A., 2018b. Fire refugia: what are they, and why do they matter for global change? Bioscience 68, 944–954.
Meigs, G.W., Dunn, C.J., Parks, S.A., Krawchuk, M.A., 2020. Influence of topography and fuels on fire refugia probability under varying fire weather conditions in forests of the Pacific Northwest, USA. Can. J. For. Res. 50, 636–647.
Meyer, M.D., Long, J.W., Safford, H.D., 2021. Postfire restoration framework for national forests in California.
Millar, C.I., Stephenson, N.L., Stephens, S.L., 2007. Climate change and forests of the future: Managing in the face of uncertainty. Ecol. Appl. 17, 2145–2151.
Miller, J.D., Quayle, B., 2015. Calibration and validation of immediate post-fire satellite derived data to three severity metrics. Fire Ecology 11, 12–30.
Moeser, C.D., Broxton, P.D., Harpold, A., Robertson, A., 2020. Estimating the effects of forest structure changes from wildfire on snow water resources under varying meteorological conditions. Water Resour. Res. 56:e2020WR027071.
Monsanto, P.G., Agee, J.K., 2008. Long-term post-wildfire dynamics of coarse woody debris after salvage logging and implications for soil heating in dry forests of the eastern Cascades, Washington. For. Ecol. Manage. 255, 3952–3961.
Morelli, T.L., Barrows, C.W., Ramirez, A.R., Cartwright, J.M., Ackerly, D.D., Eaves, T.D., Ebersole, J.L., Krawchuk, M.A., Letcher, B.H., Mahalovich, M.F., Meigs, G.W., Michalak, J.L., Millar, C.I., Quiñones, R.M., Stralberg, D., Thorne, J.H., 2020. Climate-change refugia: biodiversity in the slow lane. 18, 228–234.
Mueller, S.E., Thode, A.E., Margolis, E.Q., Yocom, L.L., Young, J.D., Iniguez, J.M., 2020. Climate relationships with increasing wildfire in the southwestern US from 1984 to 2015. For. Ecol. Manage. 460, 117861.
Nimmo, D.G., Avitabile, S., Banks, S.C., Bliege Bird, R., Callister, K., Clarke, M.F., Dickman, C.R., Doherty, T.S., Driscoll, D.A., Greenville, A.C., Haslem, A., Kelly, L.T., Kenny, S.A., Lahoz-Monfort, J.J., Lee, C., Leonard, S., Moore, H., Newsome, T.M., Parr, C.L., Ritchie, E.G., Schneider, K., Turner, J.M., Watson, S., Westbrooke, M., Wouters, M., White, M., Bennett, A.F., 2019. Animal movements in fire-prone landscapes. 94, 981-998.
Noonan-Wright, E.K., Opperman, T.S., 2015. Applying the Wildland Fire Decision Support System (WFDSS) to support risk-informed decision making: The Gold Pan Fire, Bitterroot National Forest, Montana, USA.
North, M., Brough, A., Long, J., Collins, B., Bowden, P., Yasuda, D., Miller, J., Sugihara, N., 2015a. Constraints on mechanized treatment significantly limit mechancial fuels reduction extend in the Sierra Nevada. J. Forest. 113, 40–48.
North, M., Collins, B.M., Stephens, S., 2012. Using fire to increase the scale, benefits, and future maintenance of fuels treatments. J. Forest. 110, 392–401.
North, M.P., Stephens, S.L., Collins, B.M., Agee, J.K., Aplet, G., Franklin, J.F., Fulé, P.Z., 2015b. Reform forest fire management. Science 349, 1280–1281.
North, M.P., Stevens, J.T., Greene, D.F., Coppoletta, M., Knapp, E.E., Latimer, A.M., Restaino, C.M., Tompkins, R.E., Welch, K.R., York, R.A., Young, D.J.N., Axelson, J. N., Buckley, T.N., Estes, B.L., Hager, R.N., Long, J.W., Meyer, M.D., Ostoja, S.M., Safford, H.D., Shive, K.L., Tubbesing, C.L., Vice, H., Walsh, D., Werner, C.M., Wyrsch, P., 2019. Tamm Review: Reforestation for resilience in dry western U.S. forests. For. Ecol. Manage. 432, 209–224.
North, M.P., York, R.A., Collins, B.M., Hurteau, M.D., Jones, G.M., Knapp, E.E., Kobziar, L., McCann, H., Meyer, M.D., Stephens, S.L., Tompkins, R.E., Tubbesing, C. L., 2021. Pyrosilviculture needed for landscape resilience of dry western United States forests. J. Forest.
O'Connor, C.D., Falk, D.A., Lynch, A.M., Swetnam, T.W., 2014. Fire severity, size, and climate associations diverge from historical precedent along an ecological gradient in the Pinaleño Mountains, Arizona, USA. For. Ecol. Manage. 329, 264–278.
Ouzts, J., Kolb, T., Huffman, D., Sánchez Meador, A., 2015. Post-fire ponderosa pine regeneration with and without planting in Arizona and New Mexico. For. Ecol. Manage. 354, 281–290.
Owen, S.M., Sieg, C.H., Fulé, P.Z., Gehring, C.A., Baggett, L.S., Iniguez, J.M., Fornwalt, P. J., Battaglia, M.A., 2020. Persistent effects of fire severity on ponderosa pine regeneration niches and seedling growth. For. Ecol. Manage. 477, 118502.
Owen, S.M., Sieg, C.H., Sánchez Meador, A.J., Fulé, P.Z., Iniguez, J.M., Baggett, L.S., Fornwalt, P.J., Battaglia, M.A., 2017. Spatial patterns of ponderosa pine regeneration in high-severity burn patches. For. Ecol. Manage. 405, 134–149.
Parks, S.A., Abatzoglou, J.T., 2020. Warmer and drier fire seasons contribute to increases in area burned at high severity in western US forests from 1985–2017. Geophys. Res. Lett. 47:e2020GL089858.
Parks, S.A., Dobrowski, S.Z., Shaw, J.D., Miller, C., 2019a. Living on the edge: trailing edge forests at risk of fire-facilitated conversion to non-forest. Ecosphere 10, e02651.
Parks, S.A., Holsinger, L.M., Koontz, M.J., Collins, L., Whitman, E., Parisien, M.-A., Loehman, R.A., Barnes, J.L., Bourdon, J.-F., Boucher, J., Boucher, Y., Caprio, A.C., Collingwood, A., Hall, R.J., Park, J., Saperstein, L.B., Smetanka, C., Smith, R.J., Soverel, N., 2019b. Giving ecological meaning to satellite-derived fire severity metrics across North American forests. Remote Sensing 11, 1735.
Parks, S.A., Holsinger, L.M., Miller, C., Nelson, C.R., 2015. Wildland fire as a self-regulating mechanism: the role of previous burns and weather in limiting fire progression. Ecol. Appl. 25, 1478–1492.
Parks, S.A., Holsinger, L.M., Voss, M.A., Loehman, R.A., Robinson, N.P., 2018. Mean composite fire severity metrics computed with Google Earth Engine offer improved accuracy and expanded mapping potential. Remote Sensing 10, 879.
Parson, A., P. R. Robichaud, S. A. Lewis, C. Napper, and J. T. J. G. T. R. R.-G.-F. C. Clark, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 49 p. 2010. Field guide for mapping post-fire soil burn severity. 243.
Passovoy, M.D., Fulé, P.Z., 2006. Snag and woody debris dynamics following severe wildfires in northern Arizona ponderosa pine forests. For. Ecol. Manage. 223, 237–246.
Peery, M.Z., Jones, G.M., Gutiérrez, R., Redpath, S.M., Franklin, A.B., Simberloff, D., Turner, M.G., Radeloff, V.C., White, G.C., 2019. The conundrum of agenda-driven science in conservation. Front. Ecol. Environ. 17, 80–82.
Peppin, D.L., Fulé, P.Z., Sieg, C.H., Beyers, J.L., Hunter, M.E., Robichaud, P.R., 2011. Recent trends in post-wildfire seeding in western US forests: costs and seed mixes. Int. J. Wildland Fire 20, 702–708.
Perry, D.A., Hessburg, P.F., Skinner, C.N., Spies, T.A., Stephens, S.L., Taylor, A.H., Franklin, J.F., McComb, B., Riegel, G., 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and northern California. For. Ecol. Manage. 262, 703–717.
Peterson, D.W., Dodson, E.K., Harrod, R.J., 2015. Post-fire logging reduces surface woody fuels up to four decades following wildfire. For. Ecol. Manage. 338, 84–91.
Peterson, D.W., Hessburg, P.F., Salter, B., James, K.M., Dahlgreen, M.C., Barnes, J.A., 2007. Reintroducing fire in regenerated dry forests following stand-replacing wildfire.In: Powers, Robert F., tech. editor. Restoring fire-adapted ecosystems: proceedings of the 2005 national silviculture workshop. Gen. Tech. Rep. PSW-GTR-203, Albany, CA: Pacific Southwest Research Station, Forest Service, US Department of Agriculture: p. 79-86.
Pinto, J.R., Marshall, J.D., Dumroese, R.K., Davis, A.S., Cobos, D.R., 2011. Establishment and growth of container seedlings for reforestation: A function of stocktype and edaphic conditions. For. Ecol. Manage. 261, 1876–1884.
Powers, E.M., Marshall, J.D., Zhang, J., Wei, L., 2013. Post-fire management regimes affect carbon sequestration and storage in a Sierra Nevada mixed conifer forest. For. Ecol. Manage. 291, 268–277.

Prichard, S.J., Kennedy, M.C., 2012. Fuel treatment effects on tree mortality following wildfire in dry mixed conifer forests, Washington State, USA. Int. J. Wildland Fire 21, 1004–1013.
Prichard, S.J., Stevens-Rumann, C.S., Hessburg, P.F., 2017. Tamm Review: Shifting global fire regimes: Lessons from reburns and research needs. For. Ecol. Manage. 396, 217–233.
Prichard, S.J., Hessburg, P.F., Hagmann, R.K., Povak, N.A., Dobrowski, S.Z., Hurteau, M. D., Kane, V.R., Keane, R.E., Kobziar, L.N., Kolden, C.A., North, M., Parks, S.A., Safford, H.D., Stevens, J.T., Yocom, L.L., Churchill, D.J., Gray, R.W., Huffman, D.W., Lake, F.K., Khatri-Chhetri, P., 2021. Adapting western North American forests to climate change and wildfires: ten common questions. Ecological Applications Accepted Author Manuscript e02433. https://doi.org/10.1002/eap.2433.
Puettmann, K.J., Messier, C., 2020. Simple guidelines to prepare forests for global change: the dog and the frisbee. Northwest Sci. 93 (209–225), 217.
Puhlick, J.J., Laughlin, D.C., Moore, M.M., Sieg, C.H., Overby, S.T., Shaw, J.D., 2021. Soil properties and climate drive ponderosa pine seedling presence in the southwestern USA. For. Ecol. Manage. 486, 118972.
Raymond, C.L., Peterson, D.L., 2005. Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. Can. J. Forest Res.-Revue Canadienne De Recherche Forestiere 35, 2981–2995.
Reynolds, R.T., Meador, A.J.S., Youtz, J.A., Nicolet, T., Matonis, M.S., Jackson, P.L., DeLorenzo, D.G., Graves, A.D., 2013. Restoring composition and structure in southwestern frequent-fire forests: a science-based framework for improving ecosystem resiliency.
Ritchie, M.W., Knapp, E.E., 2014. Establishment of a long-term fire salvage study in an interior ponderosa pine forest. J. Forest. 112.
Ritchie, M.W., Knapp, E.E., Skinner, C.N., 2013. Snag longevity and surface fuel accumulation following post-fire logging in a ponderosa pine dominated forest. For. Ecol. Manage. 287, 113–122.
Robichaud, P.R., 2000. Fire effects on infiltration rates after prescribed fire in Northern Rocky Mountain forests, USA. J. Hydrol. 231, 220–229.
Robichaud, P.R., Rhee, H., Lewis, S.A., 2014. A synthesis of post-fire Burned Area Reports from 1972 to 2009 for western US Forest Service lands: trends in wildfire characteristics and post-fire stabilisation treatments and expenditures. Int. J. Wildland Fire 23, 929–944.
Roccaforte, J.P., Fulé, P.Z., Chancellor, W.W., Laughlin, D.C., 2012. Woody debris and tree regeneration dynamics following severe wildfires in Arizona ponderosa pine forests. Can. J. For. Res. 42, 593–604.
Roccaforte, J.P., Sánchez Meador, A., Waltz, A.E.M., Gaylord, M.L., Stoddard, M.T., Huffman, D.W., 2018. Delayed tree mortality, bark beetle activity, and regeneration dynamics five years following the Wallow Fire, Arizona, USA: Assessing trajectories towards resiliency. For. Ecol. Manage. 428, 20–26.
Rodman, K.C., Veblen, T.T., Andrus, R.A., Enright, N.J., Fontaine, J.B., Gonzalez, A.D., Redmond, M.D., Wion, A.P., 2021. A trait-based approach to assessing resistance and resilience to wildfire in two iconic North American conifers. J. Ecol. 109, 313–326.
Rodman, K.C., Veblen, T.T., Battaglia, M.A., Chambers, M.E., Fornwalt, P.J., Holden, Z. A., Kolb, T.E., Ouzts, J.R., Rother, M.T., 2020a. A changing climate is snuffing out post-fire recovery in montane forests. Glob. Ecol. Biogeogr. 29, 2039–2051.
Rodman, K.C., Veblen, T.T., Chapman, T.B., Rother, M.T., Wion, A.P., Redmond, M.D., 2020b. Limitations to recovery following wildfire in dry forests of southern Colorado and northern New Mexico, USA. Ecol. Appl. 30, e02001.
Romme, W.H., Veblen, T.T., Kaufmann, M.R., Sherriff, R., Regan, C.M., 2003. Ecological effects of the Hayman Fire part 1: historical (pre-1860) and current (1860-2002) fire regimes. In: R. T. Graham, editor. Hayman Fire case study. USDA Forest Service General Technical Report RMRS-GTR-114, Rocky Mountain Research Station, Fort Collins, CO, USA, Pages 181-195.
Roos, C.I., Guiterman, C.H., 2021. Dating the origins of persistent oak shrubfields in northern New Mexico using soil charcoal and dendrochronology. The Holocene: 09596836211003255.
Rose, R., Haase, D.L., 2006. Guide to reforestation in Oregon. College of Forestry, Oregon State University, Corvalis, OR.
Ross, D.W., Daterman, G.E., 1997. Using Pheromone-Baited Traps To Control the Amount and Distribution of Tree Mortality During Outbreaks of the Douglas-Fir Beetle. Forest Science 43, 65–70.
Rother, M.T., Veblen, T.T., 2016. Limited conifer regeneration following wildfires in dry ponderosa pine forests of the Colorado Front Range. Ecosphere 7, e01594.
Sackett, S.S. 1984. Observations on natural regeneration in ponderosa pine following a prescribed fire in Arizona. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station.
Safford, H.D., Stevens, J.T., 2017. Natural Range of Variation (NRV) for yellow pine and mixed conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. USDA Forest Service, Pacific Southwest Research Station. General Technical Report PSW-GTR-256, Albany, CA.
Safford, H.D., Stevens, J.T., Merriam, K., Meyer, M.D., Latimer, A.M., 2012. Fuel treatment effectiveness in California yellow pine and mixed conifer forests. For. Ecol. Manage. 274, 17–28.
Schindler, D.E., Armstrong, J.B., Reed, T.E., 2015. The portfolio concept in ecology and evolution. Front. Ecol. Environ. 13, 257–263.
Schneider, E.E., Affleck, D.L.R., Larson, A.J., 2019. Tree spatial patterns modulate peak snow accumulation and snow disappearance. For. Ecol. Manage. 441, 9–19.
Schwilk, D.W., Keeley, J.E., 2006. The role of fire refugia in the distribution of *Pinus sabiniana* (Pinaceae) in the southern Sierra Nevada. Madrono 53 (364–372), 369.
Sherriff, R.L., Platt, R.V., Veblen, T.T., Schoennagel, T.L., Gartner, M.H., 2014. Historical, observed, and modeled wildfire severity in montane forests of the Colorado Front Range. PLoS ONE 9, e106971.
Shive, K.L., Preisler, H.K., Welch, K.R., Safford, H.D., Butz, R.J., O'Hara, Kevin L., Stephens, S.L., 2018. From the stand scale to the landscape scale: predicting the spatial patterns of forest regeneration after disturbance. Ecol. Appl. 28, 1626–1639.
Singleton, M.P., Thode, A.E., Sánchez Meador, A.J., Iniguez, J.M., 2019. Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. For. Ecol. Manage. 433, 709–719.
Slaton, M.R., Holmquist, J.G., Meyer, M., Andrews, R., Beidl, J., 2019. Traditional Ecological Knowledge Used in Forest Restoration Benefits Natural and Cultural Resources: The Intersection between Pandora Moths, Jeffrey Pine, People, and Fire. 39 %J Natural Areas J. 411, 461–471.
Sorensen, F.C., Miles, R.S.J.S.G., 1974. Self-pollination effects on Douglas-fir and ponderosa pine seeds and seedlings. 23.
Steel, Z.L., Campos, B., Frick, W.F., Burnett, R., Safford, H.D., 2019. The effects of wildfire severity and pyrodiversity on bat occupancy and diversity in fire-suppressed forests. Sci. Rep. 9, 16300.
Stephens, S.L., Battaglia, M.A., Churchill, D.J., Collins, B.M., Coppoletta, M., Hoffman, C. M., Lydersen, J.M., North, M.P., Parsons, R.A., Ritter, S.M., Stevens, J.T., 2021. Forest restoration and fuels reduction: convergent or divergent? Bioscience 71, 85–101.
Stephens, S.L., Collins, B.M., Fettig, C.J., Finney, M.A., Hoffman, C.M., Knapp, E.E., North, M.P., Safford, H., Wayman, R.B., 2018. Drought, tree mortality, and wildfire in forests adapted to frequent fire. Bioscience bix146-bix146.
Stephens, S.L., Westerling, A.L., Hurteau, M.D., Peery, M.Z., Schultz, C.A., Thompson, S., 2020. Fire and climate change: conserving seasonally dry forests is still possible. Front. Ecol. Environ. 18, 354–360.
Stevens, J.T., 2017. Scale-dependent effects of post-fire canopy cover on snowpack depth in montane coniferous forests. Ecol. Appl. 27, 1888–1900.
Stevens, J.T., Boisramé, G.F.S., Rakhmatulina, E., Thompson, S.E., Collins, B.M., Stephens, S.L., 2020a. Forest vegetation change and its impacts on soil water following 47 years of managed wildfire. Ecosystems 23, 1547–1565.
Stevens, J.T., Collins, B.M., Miller, J.D., North, M.P., Stephens, S.L., 2017. Changing spatial patterns of stand-replacing fire in California conifer forests. For. Ecol. Manage. 406, 28–36.
Stevens, J.T., Kling, M.M., Schwilk, D.W., Varner, J.M., Kane, J.M., 2020b. Biogeography of fire regimes in western U.S. conifer forests: A trait-based approach. Glob. Ecol. Biogeogr. 29, 944–955.
Stevens, J.T., Miller, J.E.D., Fornwalt, P.J., 2019. Fire severity and changing composition of forest understory plant communities. J. Veg. Sci. 30, 1099–1109.
Stevens, J.T., Safford, H.D., Harrison, S., Latimer, A.M., 2015. Forest disturbance accelerates thermophilization of understory plant communities. J. Ecol. 103, 1253–1263.
Stevens-Rumann, C.S., Morgan, P., 2019. Tree regeneration following wildfires in the western US: a review. Fire Ecol. 15, 15.
Stevens-Rumann, C.S., Prichard, S.J., Strand, E.K., Morgan, P., 2016. Prior wildfires influence burn severity of subsequent large fires. Can. J. For. Res. 46, 1375–1385.
Stevens-Rumann, C.S., Kemp, K., Higuera, B.P., Harvey, E.B., Rother, J.M., Donato, T.D., Morgan, C.P., Veblen, T.T., 2018. Evidence for declining forest resilience to wildfires under climate change. Ecol. Lett. 21, 243–252.
Stewart, J.A.E., van Mantgem, P.J., Young, D.J.N., Shive, K.L., Preisler, H.K., Das, A.J., Stephenson, N.L., Keeley, J.E., Safford, H.D., Wright, M.C., Welch, K.R., Thorne, J. H., 2021. Effects of postfire climate and seed availability on postfire conifer regeneration. Ecol. Appl. 31, e02280.
Stonesifer, C.S., Calkin, D.E., Thompson, M.P., Stockmann, K.D., 2016. Fighting fire in the heat of the day: an analysis of operational and environmental conditions of use for large airtankers in United States fire suppression %J. Int. J. Wildland Fire. 25, 520–533.
Stortz, S., Haffey, C.M., Kimball, C., 2018. East Jemez Landscape Futures needs assessment and recommendations: identifying cross-boundary opportunities for management in altered landscapes. https://www.eastjemez.org.
Swanson, M.E., Franklin, J.F., Beschta, R.L., Crisafulli, C.M., DellaSala, D.A., Hutto, R.L., Lindenmayer, D.B., Swanson, F.J., 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Front. Ecol. Environ. 9, 117–125.
Swetnam, T.W., Baisan, C.H., 1996. Fire histories of montane forests in the Madrean Borderlands. 0277-5786, United States Department of Agriculture Forest Service General Technical Report RM (1996).
Stephens, S.L., Collins, B.M., Biber, E., Fulé, P.Z., 2016. U.S. federal fire and forest policy: emphasizing resilience in dry forests. Ecosphere 7. https://doi.org/10.1002/ecs2.1584 e01584-n/a.
Stephens, S.L., Agee, J.K., Fulé, P.Z., North, M.P., Romme, W.H., Swetnam, T.W., Turner, M.G., 2013. Managing forests and fire in changing climates. Science 342, 41–42. https://doi.org/10.1126/science.1240294.
Sloan, J.L., Burney, O.T., Pinto, J.R., 2020. Drought-Conditioning of Quaking Aspen (Populus tremuloides Michx.) Seedlings During Nursery Production Modifies Seedling Anatomy and Physiology, vol. 11. doi: 10.3389/fpls.2020.557894.
Singleton, M.P., Thode, A.E., Sánchez Meador, A.J., Iniguez, J.M., Stevens, J.T., 2021. Management strategy influences landscape patterns of high-severity burn patches in the southwestern United States. Landscape Ecology. doi: https://dx.doi.org/10.1007/s10980-021-01318-3 (in press).
Swetnam, T.W., Baisan, C.H., 2003. Tree-ring reconstructions of fire and climate history in the Sierra Nevada and Southwestern United States. In: Veblen, T.T., Baker, W., Montenegro, G., Swetnam, T.W. (Eds.), Fire and climatic change in temperate ecosystems of the western Americas. Springer-Verlag, New York, NY, pp. 154–191.
Taylor, T.S., Loewenstein, E.F., Chappelka, A.R., 2006. Efect of animal browse protection and fertilizer application on the establishment of planted Nuttall oak seedlings. New Forest 32, 133–143.

Tepley, A.J., Thompson, J.R., Epstein, H.E., Anderson-Teixeira, K.J., 2017. Vulnerability to forest loss through altered postfire recovery dynamics in a warming climate in the Klamath Mountains. Glob. Change Biol. 23, 4117–4132.

Tepley, A.J., Veblen, T.T., 2015. Spatiotemporal fire dynamics in mixed-conifer and aspen forests in the San Juan Mountains of southwestern Colorado, USA. Ecological Monographs.

Thompson, J.R., Spies, T.A., 2010. Factors associated with crown damage following recurring mixed-severity wildfires and post-fire management in southwestern Oregon. Landscape Ecol. 25, 775–789.

Thompson, J.R., Spies, T.A., Ganio, L.M., 2007. Reburn severity in managed and unmanaged vegetation in a large wildfire. Proc. Natl. Acad. Sci. 104, 10743–10748.

Thompson, M.P., Bowden, P., Brough, A., Scott, J.H., Gilbertson-Day, J., Taylor, A., Anderson, J., Haas, J.R., 2016. Application of wildfire risk assessment results to wildfire response planning in the southern Sierra Nevada, California, USA. Forests 7, 64.

Thompson, M.P., Gannon, B.M., Caggiano, M.D., O'Connor, C.D., Brough, A., Gilbertson-Day, J.W., Scott, J.H., 2020. Prototyping a geospatial atlas for wildfire planning and management. 11, 909.

Tingley, M.W., Ruiz-Gutiérrez, V., Wilkerson, R.L., Howell, C.A., Siegel, R.B., 2016. Pyrodiversity promotes avian diversity over the decade following forest fire. Proc. Roy. Soc. B: Biol. Sci. 283.

Tingley, M.W., Stillman, A.N., Wilkerson, R.L., Sawyer, S.C., Siegel, R.B., 2020. Black-backed woodpecker occupancy in burned and beetle-killed forests: Disturbance agent matters. For. Ecol. Manage. 455, 117694.

Tubbesing, C.L., York, R.A., Stephens, S.L., Battles, J.J., 2020. Rethinking fire-adapted species in an altered fire regime. Ecosphere 11, e03091.

Vanderhoof, M.K., Burt, C., Hawbaker, T.J., 2018. Time series of high-resolution simages enhances efforts to monitor post-fire condition and recovery, Waldo Canyon fire, Colorado, USA. International Journal of Wildland Fire.

Vanderhoof, M.K., Hawbaker, T.J., Ku, A., Merriam, K., Berryman, E., Cattau, M., 2021. Tracking rates of postfire conifer regeneration vs. deciduous vegetation recovery across the western United States. Ecol. Appl. 31:e02237.

Vickers, L.A., Houser, J., Rooni, J., Guldin, J.M., 2018. Restoring ponderosa pine in the Davis Mountains of west Texas: impacts of planting practices on seedling survival. Pages 82-88 In: Kirschman, Julia E., comp. Proceedings of the 19th biennial southern silvicultural research conference; 2017 March 14-16; Blacksburg, VA. e-Gen. Tech. Rep. SRS-234. Asheville, NC: US Department of Agriculture, Forest Service, Southern Research Station.

Walker, J.J., Soulard, C.E., 2019. Phenology patterns indicate recovery trajectories of ponderosa pine forests after high-severity fires. Remote Sens. 11, 2782.

Walker, R.B., Coop, J.D., Downing, W.M., Krawchuk, M.A., Malone, S.L., Meigs, G.W., 2019. How much forest persists through fire? High-resolution mapping of tree cover to characterize the abundance and spatial pattern of fire refugia across mosaics of burn severity. Forests 10, 782.

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

Welch, K.R., Safford, H.D., Young, T.P., 2016. Predicting conifer establishment post wildfire in mixed conifer forests of the North American Mediterranean-climate zone. Ecosphere 7 e01609-n/a.

White, A.M., Long, J.W., 2019. Understanding ecological contexts for active reforestation following wildfires. New Forest. 50, 41–56.

Williams, M.I., Dumroese, R.K., 2013. Preparing for climate change: forestry and assisted migration. J. Forest. 111, 287–297.

Yocom, L.L., Jenness, J., Fulé, P.Z., Thode, A.E., 2019. Previous fires and roads limit wildfire growth in Arizona and New Mexico, U.S.A. Forest Ecol. Manage. 449, 117440.

York, R.A., Noble, H., Quinn-Davidson, L.N., Battles, J.J., 2021. Pyrosilviculture: Combining prescribed fire with gap-based silviculture in mixed-conifer forests of the Sierra Nevada. Can. J. For. Res. 51, 781–791.

Young, J.D., Evans, A.M., Iniguez, J.M., Thode, A., Meyer, M.D., Hedwall, S.J., McCaffrey, S., Shin, P., Huang, C.-H., 2020. Effects of policy change on wildland fire management strategies: evidence for a paradigm shift in the western US? Int. J. Wildland Fire 29, 857–877.