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Understanding ecological contexts for active reforestation following wildfires

Angela M. White[1] · Jonathan W. Long[2]

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Abstract
To forestall loss of ecological values associated with forests, land managers need to consider where and when to prioritize active reforestation following major disturbance events. To aid this decision-making process, we summarize recent research findings pertaining to the Sierra Nevada region of California, USA to identify contexts in which active reforestation or passive recovery may best promote desirable post-fire ecological trajectories. Based on our synthesis, we suggest conceptual frameworks for assessing landscape conditions and determining areas that may be the highest priorities for tree planting to avoid persistent loss of conifer forests. Field studies have shown that some large patches of high severity burn can have relatively low levels of natural regeneration, especially among desired pine species. The accumulation of fuels and competition with shrubs and resprouting hardwoods may hinder the reestablishment of mature conifer trees. However, severe fires could also play a restorative role, by promoting non-conifer forested communities, such as meadows, shrubfields, and open forests with significant hardwood components. Such communities were historically rejuvenated and maintained by fire but have been replaced by conifer forest due in part to fire suppression. Reforestation in such areas may run counter to restoring ecological function and the ecosystem services that are provided by non-conifer communities. Through this framework, managers and stakeholders may better understand the contexts in which planting and passive recovery may better support ecological restoration.

Keywords Natural regeneration · Succession · Meadow restoration · Shrub competition · Biodiversity · Landscape restoration · Planting

Introduction

Loss of forests is a global concern because they provide many valuable services including wood products, carbon sequestration, wildlife habitat, and watershed protection (Jacobs et al. 2015). Societies around the world are choosing between strategies of assisted

✉ Angela M. White
angelawhite@fs.fed.us

1 Pacific Southwest Research Station, USDA Forest Service, Davis, CA 95618, USA

2 Pacific Northwest Research Station, USDA Forest Service, Davis, CA 95618, USA

restoration and unassisted natural regeneration to restore such services following forest degradation (Chazdon 2008). The justification for active intervention to ameliorate such declines depends on how quickly forests can naturally recover. In the Sierra Nevada of California, this question has been raised following extreme wildfires in recent years (Jones et al. 2016; Young et al. 2017). Active reforestation efforts in this region generally focus on planting conifer seedlings, but often include control of non-tree vegetation as well as other practices intended to accelerate reestablishment of forest cover. Understanding the ecological contexts in which passive recovery may be sufficient or desirable may help focus interventions where they are most needed for ecological restoration (Peterson et al. 2009; Johnstone et al. 2016; Shive et al. 2018), as well as how to craft such interventions to better achieve objectives (Jones 2018).

Most forested ecosystems of the Western USA have evolved with fire, but decades of fire exclusion have resulted in widespread densification, increases in fuels, and shifts toward less fire-adapted species, rendering many forests less resilient to disturbances (Mallek et al. 2013; Miller and Safford 2012; Scholl and Taylor 2010). As wildfires in the interior mountains of California have overwhelmed suppression efforts in recent decades (North et al. 2015), they have produced increasingly large patches of dead trees (Steel et al. 2018; Stevens et al. 2017). Projected warming and drying are expected to exacerbate such trends through increases in large wildfires (Barbero et al. 2015) and large drought-induced mortality events (Young et al. 2017). Reforestation following large-scale tree mortality events such as wildfires has long been a high priority for the U.S. Forest Service, which manages the national forests, and planting nursery-grown trees has been one of the established means for achieving that goal (Dumroese et al. 2005; Nave et al. 2018). Planting may have objectives other than restoration, including promoting various wood production and other services that are important to societies regardless of former ecological conditions (Silva et al. 2018). Understanding complex landscape dynamics can help managers and stakeholders meet any of those objectives by weighing benefits and risks of alternative strategies (Johnstone et al. 2016; Jones 2018).

Active reforestation efforts are motivated by concerns that conifer forests would not recover, or would recover too slowly to sustain key services, without interventions. Studies in the Sierra Nevada in particular have found that natural regeneration was often too low to achieve desired levels of reforestation. For example, Collins and Roller (2013) reported that more than 50% of the patches and approximately 80% all plots lacked tree regeneration 2–11 years after five fires in the Northern Sierra Nevada. Similarly, Welch et al. (2016) investigated conifer regeneration in 14 fires that burned with high severity effects across 10 national forests in the Sierra Nevada between 1999 and 2007 and found that 43% of their plots had no conifer regeneration. Both of those studies found that where regeneration did occur, it was dominated by more shade-tolerant species than pines, especially white fir (*Abies concolor*), Douglas- fir (*Pseudotsuga menziesii*) and incense cedar (*Calocedrus decurrens*). The reduction in pine regeneration is of particular concern, as yellow pine forests (*Pinus ponderosa* and *P. jeffreyi*) have declined substantially from historical periods in many parts of the Sierra Nevada (Knapp et al. 2013; McIntyre et al. 2015; Thorne et al. 2008). Harris and Taylor (2015) attributed the decline of formerly dominant yellow-pine systems to the loss of seed trees and altered vegetation-fire dynamics. Concerns over declines in dry yellow pine forests following wildfires are consistent with studies in Arizona and New Mexico (Ouzts et al. 2015) and in Colorado (Rother and Veblen 2016).

Managers also plant seedlings where natural regeneration may be present but they are concerned that it will be insufficient to meet objectives. Young seedlings often experience high levels of mortality, from 50 to 80% or higher (Shive et al. 2018). To increase growth of planted trees, managers have often controlled understory vegetation to reduce competition (Helms and Tappeiner 1996; McGinnis et al. 2010; Peterson et al. 2009). Snag management is also an important consideration in reforestation, because salvage logging often goes hand-in-hand with conifer replanting for safety reasons (to avoid risks to planting crews from falling snags) and to reduce the future fire risk in planted stands. While areas that are not salvaged can be reseeded or replanted with conifer trees, land managers may be reticent to invest planting resources where they perceive conditions to be highly vulnerable to fire.

Dynamics of non-conifer vegetation communities

Tree mortality and shifts to non-forested states are natural parts of the dynamic landscape of mountain areas such as the Sierra Nevada. While generally accounting for only a small portion of the montane region (Thorne et al. 2008), non-conifer communities collectively support important ecosystem services such as providing food and structural resources that promote biodiversity (Betts et al. 2010; McCullough et al. 2013), regulating fire (Kobziar and McBride 2006), promoting resilience to drought, and fixing nitrogen (Oakley et al. 2006). Although many of these communities may have been relatively short-lived and spatially variable within the landscape due to the unpredictable nature of fire, other patches, including wet meadows and some hardwood groves, were likely long-lived and relatively stable in particular landscape positions associated with relatively deep and mesic soils (Ratliff 1985; Shepperd et al. 2006). Extensive forest homogenization and degradation of formerly diverse vegetation patches could lead to disproportionate losses in forest diversity (Kelt et al. 2017; Shepperd et al. 2006; Sollmann et al. 2015).

Many non-conifer communities can be maintained by low-severity fire, but once overtopped by conifers, shade-intolerant species are competitively excluded in the absence of further canopy opening disturbances (Berrill et al. 2017; Knapp et al. 2013). Conifers generally become more resistant to fire as they grow taller and develop thicker bark, although a variety of factors influence fire resistance across different species (Hood and Lutes 2017). Consequently, stand-replacing disturbances, such as high severity fire, may be necessary to restore non-conifer vegetation (Cocking et al. 2014). Therefore, rather than focusing solely on how high-severity fire can shift conifer stands into a non-conifer state [a pathway described by Johnstone et al. (2016)], conceptual models need to recognize how fire exclusion has also shifted meadows, shrubfields, and open forests with significant hardwood components to conifer forests (Fig. 1). Planting conifer trees in these areas after fire may be contrary to restoring historical or desired non-conifer communities and their associated ecosystem services.

Wetland-associated communities

Riparian communities, including aspen

Riparian zones connect terrestrial and aquatic ecosystems and are complex and dynamic microhabitats that support heterogeneous biological communities that are more diverse than upland habitats (Dwire and Kauffman 2003; Naiman et al. 1993). Fire plays an important role in maintaining and rejuvenating riparian communities by consuming fuels and killing non-resprouting conifers (Arkle and Pilliod 2010; Bêche et al. 2005; Berrill et al.

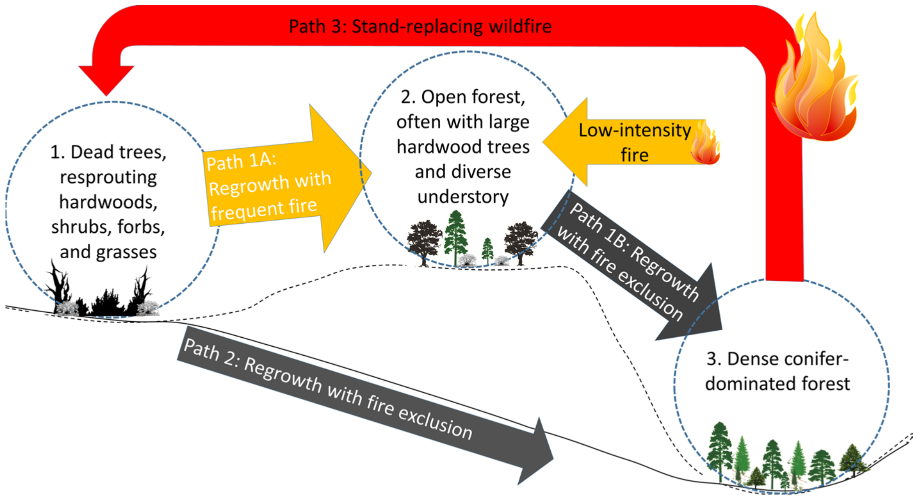

Fig. 1 A "ball and cup" diagram representing how the interactions of tree growth (rightward force) and fires (leftward force) influence alternative forest states. Path 1A represents how frequent, low-intensity fires kill smaller trees and encourage open conditions with large hardwood trees such as oaks and aspens. Path 1B and 2 represents succession to dense conifer forest in the absence of fire. Path 3 represents stand-replacing fires that return to the early successional state without mature trees

2017). For example, quaking aspen (*Populus tremuloides*) is a particularly important hardwood tree in montane riparian areas, supporting understory plant biodiversity, forage, riparian soil moisture, and many wildlife species (Jones et al. 2005; Kuhn et al. 2011; Richardson and Heath 2004). Changes in the historic fire regime and grazing pressure have threatened aspen stands in many places (Jones et al. 2005) and long-term restoration of aspen stands appears to depend on aggressively removing conifers (Berrill et al. 2017; Jones et al. 2005). Although some studies indicate that riparian areas burned with similar or less frequency than adjacent uplands (Dwire and Kauffman 2003), others suggest a more frequent fire regime, perhaps due to Native American burning to stimulate growth of desired understory species (Van de Water and North 2010). Exclusion of fire from riparian areas may allow fuels to accumulate and encourage high severity fire (Pettit and Naiman 2007; Safford et al. 2009). Such fires can reduce conifer encroachment by fire-intolerant species such as lodgepole pine (*Pinus contorta*), while hardwood species such as black cottonwood (*Populus trichocarpa*), willow (*Salix* spp.), and aspen are likely to be top-killed but can vigorously resprout (Kobziar and McBride 2006; Krasnow and Stephens 2015).

Montane meadows

Meadows are productive and diverse ecosystems that cover less than 2% of the Sierra Nevada and southern Cascade ranges of California (Ratliff 1985). Fire suppression and grazing have facilitated conifer encroachment in lower elevation meadows (Norman and Taylor 2005), while warming temperatures and reduced snowpack have made conditions more favorable for conifer establishment in higher elevation meadows (Lubetkin et al. 2017; Maher et al. 2017). The removal of conifers can lead to increases in the water table that are necessary to sustain meadow communities (Sanford 2016). However, Frenzel (2012) found that late fall prescribed burns in seven meadows in the central Sierra Nevada had little efficacy in removing conifers and suggested that more intense burns or cutting may be needed to restore such areas. Boisramé et al. (2017a) found an increase in meadow communities and a decrease in conifer communities in an area that experienced repeated wildland fires in Yosemite National Park. While they described this phenomenon as large-scale conversion, it may represent a return to historical conditions.

Upland non-conifer communities

Complex early-successional communities

High-severity fire induces the development of early successional communities characterized by young shrubs and resprouting hardwoods, and forbs, as well as standing dead trees and other biological “legacies” that are not destroyed by the disturbance. Although such legacies may impart significant structural and compositional diversity, these communities may be mapped from remote sensing simply as annual or perennial grasslands. Vertical and horizontal heterogeneity in complex, early-successional habitats is important, as it is in late seral forests (Tarbill et al. 2015; Tingley et al. 2016; White et al. 2016). Woodpeckers and other cavity-associated species depend on standing dead trees for food (Nappi et al. 2010) and nesting structures (Tarbill et al. 2015). As snags decay and fall, they provide habitat for small mammals, which in turn, are an important food source for forest predators. In the understory, post-fire response to severe burns often includes a flush of understory plant and fungal growth (Keeley et al. 2003; Larson et al. 2016). Such habitats in the Sierra Nevada are relatively short-lived, as most snags will fall and shrubs will dominate within the 1st decade post-fire (Bohlman et al. 2016; Ritchie et al. 2013).

Montane chaparral

Shrubfields, also called montane chaparral, historically ranged in size from tens to hundreds of hectares (Nagel and Taylor 2005; Skinner and Chang 1996). Shrubs provide nesting habitat for breeding landbirds, as well as foraging habitat for the California spotted owl (Gutiérrez et al. 1992) and other species of concern. Although very few studies from the Sierra Nevada have focused on the specific area requirements of shrub associated wildlife species, studies in the eastern USA have demonstrated that most shrub specialists avoided mature forest (Rodewald and Vitz 2005). In the Sierra Nevada, some shrub-associated species, such as the fox sparrow, tend to avoid areas near forest edges and attain their highest abundance in the interior of patches (Campos and Burnett 2014).

Shrub species in the Sierra Nevada are well-adapted to high-severity fire because they can disperse long distances, and many vigorously resprout following fire and/or reestablish via a persistent seed bank (Pausas and Keeley 2014). Like many other non-conifer dominated communities, montane chaparral had a distinct fire regime characterized by slower production of dead fuels and greater live fuel moisture than adjacent forests (Nagel and Taylor 2005; Skinner and Chang 1996). Chaparral shrub seeds are widely distributed throughout mixed-conifer forest (Knapp et al. 2012), indicating that shrub patches are dynamic and may exist in locations that are edaphically suited for trees (Biswell 1974). Shrub patches also occur in relatively fixed positions due to topographic and edaphic factors and associated fire behavior. For example, more xeric, south- and southeast-facing slopes are predisposed to high severity fire (Nagel and Taylor 2005). Fire exclusion during the last century has contributed to the conversion of chaparral to forests, reducing heterogeneity both within and between forest stands (Knapp et al. 2012; Lauvaux et al. 2016; Thorne et al. 2008). For example, Knapp et al. (2013) reported an order of magnitude decrease in shrub cover from 28.6% in 1929 to 2.5% in 2008 in the Stanislaus Experiment Forest, while densities of small and intermediate trees increased three-fold. Similarly, Nagel and Taylor (2005) found that chaparral patches decreased by 62% due to conifer encroachment in the Lake Tahoe Basin. Some evidence shows that fires can lead to recovery of that historical diversity, as Boisramé et al. (2017b) found that shrub area increased by 35% in the Yosemite National Park following 40 years of repeated managed wildfire. The study authors noted that a relatively homogeneous forested patch differentiated into patches of more diverse vegetation composition after fire was restored (Boisramé et al. 2017a). They concluded that a return of a more reference fire regime was consistent with greater resilience to a variety of disturbances.

Montane hardwoods

As with shrubs, species such as California black oak (*Quercus kelloggii*), tanoak (*Notholithocarpus densiflorus*), Pacific madrone (*Arbutus menziesii*), and other species have a long history of being derided as weedy because they occupied space where more valuable conifers could grow (Bolsinger 1988). Such hardwood species were systematically reduced using herbicides in the mid-twentieth century to promote conifer growth on national forest lands (Long et al. 2016). Within mixed-conifer forests, oaks have tended to decline in recent decades (Thorne et al. 2008). Cocking et al. (2012) suggested that wildfires could be restorative, as California black oak may need high-severity fire to overcome encroachment by relatively large conifers. Baker (2014) contended that infrequent, severe fires maintained high heterogeneity and abundant and large patches (up to 9400 ha) of montane chaparral and early-successional forest, including oak-dominated areas. He contended that displacement of oaks by conifers in such areas represented natural succession in the absence of fire rather than "encroachment". Baker's model of infrequent disturbance facilitating a gradual succession to conifer might fit with some areas of the landscape; however, his findings have been challenged as overestimating areas of high-severity fire due to methodological artifacts (Levine et al. 2017). In areas where fires were frequent, they likely maintained an alternative state that included open groves dominated by large, shade-intolerant hardwoods as well as pines (Fig. 1). Such landscape conditions are consistent with conditions desired by Native Americans (Long et al. 2016) and understandings of aspen ecology (Shepperd et al. 2006).

Considerations to guide restorative reforestation

Many of the conifer-dominated forests of interior California evolved with patches of high severity fire limited to small areas, averaging less than 10 ha and generally less than 100 ha (Safford and Stevens 2017; Stevens et al. 2017). Such small patches are believed to have been recolonized relatively quickly by seed fall from nearby surviving, mature trees. Under this fire regime, some communities would shift between forested and non-forested states due to more frequent fire and successional dynamics (Fig. 1) resulting in both fine and coarse scale heterogeneity within and between vegetative communities. In contrast, large areas of deciduous- or shrub-dominated states created by high-severity patches may be self-reinforcing (Bowman et al. 2015; Scheffer et al.

2001). For instance, large quantities of dead woody biomass and resprouting shrubs form fuels that can favor future high-intensity reburns (Coppoletta et al. 2016; Lauvaux et al. 2016) and delay expected succession to forested states.

Landscape restoration efforts increasingly highlight the need for a more natural fire regime outside of developed areas (Lindenmayer 2018; North et al. 2015). However, large fires such as the Rim Fire (Harris and Taylor 2017) have shown a tendency for severe burns to be self-reinforcing by maintaining heavy fuels and low-statured vegetation. Extensive areas of young trees, whether natural or planted, pose a challenge to restoring fire as a disturbance, because young plantations are highly susceptible to fire (Kobziar et al. 2009). Many areas that burned at high severity in the recent large wildfires coincide with areas that were replanted following previous wildfires; for example, in a recent study of a highly mixed private and public forests in Oregon, Zald and Dunn (2018) suggested that the young plantations were a more important contributor to high burn severity than fire suppression in old forests.

As mixed-conifer forests continue to experience large, stand-replacing fires, land managers may consider where, when, and how to promote the return of conifer forest that will increase future landscape resilience to disturbances. In this section, we consider factors that may help to identify places where targeted interventions may be particularly important in advancing ecological restoration. However, we acknowledge that planning specific interventions demands consideration of a variety of social and economic factors, including organizational capability, access, costs, and other constraints (Long et al. 2014). Furthermore, although our framework suggests how to consider where active reforestation might be appropriate or not, it is important to consider how to customize interventions to particular contexts in more subtle ways (North 2012). For example, different tactical approaches to planting, such as even spacing versus cluster spacing, and variable densities, could better align treatments with restoration goals. Targeting favorable microclimates might guide restoration at fine scales, although it may be challenging for planting crews to implement such guidance. Strategies need to consider temporal factors, because timing after disturbance and weather factors exert strong influence on reforestation outcomes (North et al. 2005). In particular, managers have stressed that planting soon after a disturbance can reduce the need for more costly treatments, including herbicides, to encourage development of planted seedlings (McGinnis et al. 2010). Consequently, advance planning of interventions could yield both economic and ecological benefits.

While studies have highlighted low stocking rates as a particular need driving replanting following major disturbances (Welch et al. 2016; Safford and Stevens 2017), several other factors besides the numbers of young trees are also important considerations. These factors include redressing shifts in species composition, especially the reductions in pines mentioned earlier, and promoting resistance to climate change and introduced disease by encouraging particular species and/or genotypes. Planting of trees with genetic resistance to threatening diseases may be a particularly important tool of restoration, as in the case of introduced white pine blister rust (Sessions et al. 2004), which has impacted important species in the Sierra Nevada region such as sugar pine (*Pinus lambertiana*) and western white pine (*Pinus monticola*). Promoting sugar pine regeneration is a priority for restoration in the Sierra Nevada, as it has experienced high mortality in field studies examining planting (Lanini and Radosevich 1986) and prescribed burning (Bellows et al. 2016). Anticipation of climate change has encouraged managers to consider "assisted migration" strategies that would accelerate the expected movement of trees that might be better adapted for future climates, although such policies have not been adopted for national forests (Long et al. 2014).

Prioritizing areas for passive reforestation

Managers needing to procure and plant nursery-grown seedlings after wildfires may lack time to conduct detailed analyses of reference conditions. Instead, they may rely on pre-fire vegetation maps, which are readily available at fine scales. Yet, such maps may not necessarily be a good indication of departure from ecological references, as they are likely to underrepresent the historical extent of non-conifer forest areas due to the shifts resulting from fire suppression discussed previously. While such areas often have potential to support conifers, reforestation may better achieve restoration objectives by targeting areas that have been dominated by conifer forest for longer periods. In addition, more ephemeral non-forested patches are also important to consider for restoring biodiversity and other services (Fig. 2). By embracing the concept of a "shifting mosaic" at large landscape scales (Borman and Likens 1979), management can be designed to sustain the extent of diverse vegetation communities, even as they shift their locations over time, while maintaining connectivity between the persistent patches and the more ephemeral ones.

The historical extent and locations of these communities prior to Euro-American settlement are likely to reflect topography, moisture, and soils, so soil surveys and models based upon elevation, moisture, and disturbance regime (including the Biophysical Setting product from LANDFIRE www.landfire.gov/bps.php) provide useful guidance for restoration in addition to historical maps. Figure 3 provides a simple model for considering edaphic factors based upon an idealized cross-sectional view of a valley going from a steep slope

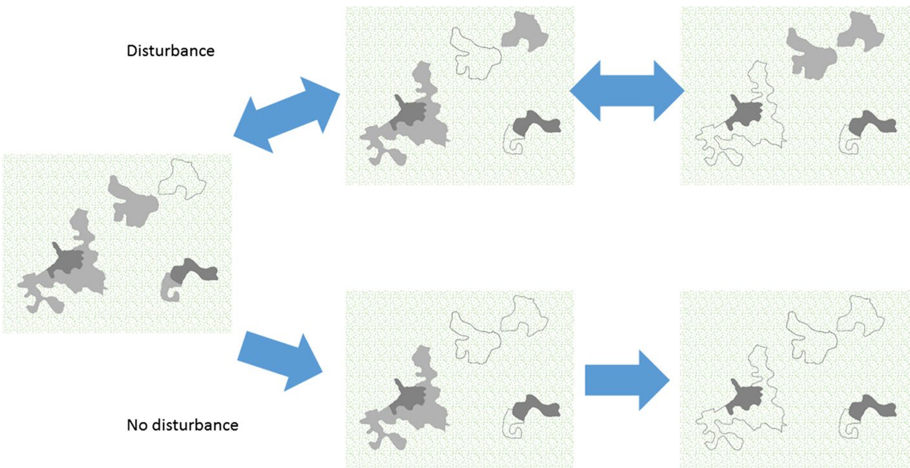

Fig. 2 Hypothetical landscape with grey patches representing non-conifer communities such as meadows within a conifer-dominated landscape. Dark grey patches are relatively fixed in the landscape due to topography and edaphic conditions. Light grey patches represent ephemeral habitat, existing in both non-conifer states and conifer-dominated areas (grey outline) depending on the level of disturbance and the conditions following a disturbance. Ephemeral patches not only contribute to the amount of available habitat, but also play a role in habitat connectivity. Without disturbance, ephemeral patches succeed to conifer-dominated habitat thereby reducing the amount and connectivity of non-conifer patches

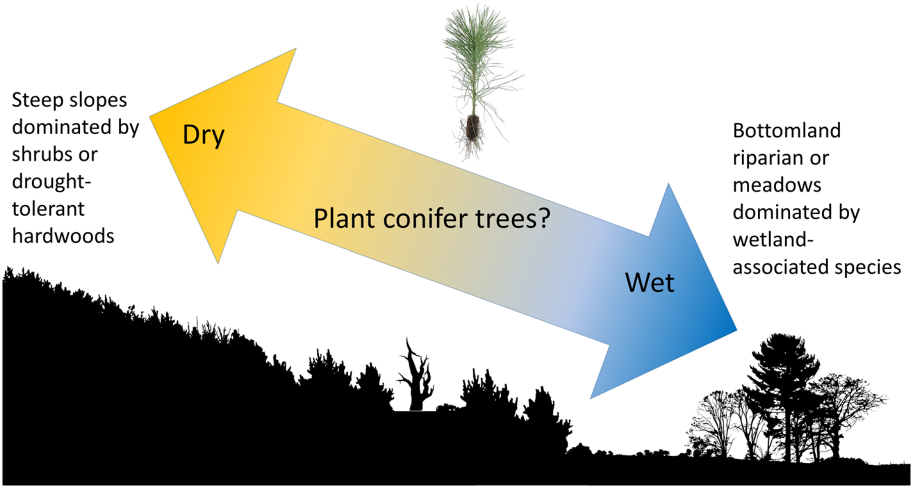

Fig. 3 Conceptual representation of how landscape moisture relationships may influence post-fire reforestation decisions. Conifer planting is more likely to be avoided on steeper, dry slopes where tree survival is likely to be poor, but planting may also be avoided in moist bottomlands where natural regeneration, particularly by desirable non-conifer species, is expected

to a more productive bottomland. Historical studies indicate how species composition varied with moisture, with upper and more xeric slopes often dominated by montane chaparral and hardwood trees (Nagel and Taylor 2005; Taylor and Skinner 2003) and where tree survival is often poor (Rother and Veblen 2016). Steep slopes may also be avoided for interventions due to operational constraints, in which case practical considerations may align with ecological objectives to restore non-conifer communities. Planting might also be discouraged in moist bottomlands where natural regeneration, particularly by desirable hardwood species, is expected. National forest policy under the Sierra Nevada Forest Plan Amendment has restricted planting of any conifers closer than 6.1 m from the crown edge of oaks; although it may be important to consider broader strategies for facilitating growth of hardwoods especially where mature trees are scarce (Long et al. 2016). Although Welch et al. (2016) emphasized that Forest Service silvicultural goals are tied to conifer regeneration, managers can also certify that regeneration objectives have been met based upon hardwoods (R. Tompkins, U.S. Forest Service, *personal communication*).

Prioritizing areas for active reforestation

Targeted interventions may be important for ensuring the growth of large conifer trees on the landscape to support wood products, wildlife habitat, and other ecological services. Where desired tree species are able to resprout, replanting may not be warranted, but for the Sierra Nevada, hardwoods are the predominant resprouting trees. Consequently, conifer planting (alongside other management practices to address fuel loads and shrub removal) may be important for restoring reference composition and large trees. While many wildlife and plant species may depend upon or thrive upon early-successional conditions, reestablishment of large conifers is important to sustaining wildlife habitat for the long term. Large trees are a key habitat feature of California spotted owl habitat (North et al. 2017).

Stephens et al. (2016) suggest that spotted owl habitat is likely to experience a long-term decline due to future high-severity wildfires. Although planting in bottomlands may be discouraged for other reasons (see above), North et al. (2017) suggested focusing management to conserve spotted owl by cultivating tall trees in more productive riparian areas. Large conifers close to streams are important habitat for wildlife such as northern flying squirrel (Meyer et al. 2005). Abundance of such prey animals, combined with large trees, snags, and closed canopies contribute to such areas being preferred habitat for spotted owls and fisher (Underwood et al. 2010). Similarly, for species that depend upon patches of large snags, it is important to consider how long it will take for large trees to develop that can form such habitat.

Planting, as well as shrub control, may also be important for facilitating the development of conifer trees that can survive fires in the frequent fire regimes of California. Fires kill trees particularly when the tree crowns are damaged (Hood and Lutes 2017), so raising the height of the crown is particularly important for increasing tree survival (Bellows et al. 2016). Accelerated growth could also speed the production of seeds, to enable "founder" stands to help naturally reforest larger patches. Lanini and Radosevich (1986) demonstrated that control of shrubs accelerated conifer growth in the Sierra Nevada, finding that shrub canopy volume affected tree canopy volume, although it did not influence tree survival. McDonald and Abbott (1997) found that on a poor site at Mount Shasta, early shrub control afforded ponderosa pines planted an additional 5 m in height after 25 years, yielding trees that were 7.6 m tall on average versus 2.1 m tall. Such findings suggest that trees planted without associated shrub removal may make it harder to restore a more frequent fire regime. McIver and Ottmar (2018) found that planted trees (without follow-up treatments such as prescribed burning or shrub control) did not survive a 12-year reburn, likely due to their small size and abundant fuels. Yet, recent research suggests that fire may be reintroduced to planted stands that are as young as 12 years old without causing widespread mortality, especially by raking litter away from the boles to reduce scorch, and possibly even pruning low branches (Bellows et al. 2016). If conifer plantings can withstand reintroduction of fire within such a frequent return interval, there is potential for meeting goals of restoring low-severity fire as an ecological process while also regenerating conifers, despite the problem that historical plantations have been associated with increased fire severity.

Thinking beyond the high-severity burn patches

Resilience theory asserts that post-fire environments represent a "back loop" stage marked by high unpredictability and often receiving less attention than the growth and conservation phases of resource management (Berkes and Folke 2002). That theory suggests several strategies to avoid shifts in systems by maintaining natural disturbances at smaller scales through intentionally using fire, creating clearings and other kinds of fire breaks, and encouraging patchy landscape. To be more consistent with such thinking, post-fire restoration can move beyond immediate emergency treatments and time-sensitive matters of "salvage" of wood products in severely burned areas, toward reducing the impacts of future burns through promoting heterogeneous vegetation and fuel conditions across the landscape. Many areas that have not yet burned severely are still afflicted by high densities of live trees and heavy accumulations of fuels, including more pieces of small downed wood than occurred historically (Knapp 2015; Knapp et al. 2013). Even areas that burn moderately may still experience an increase in fuels as small fire-killed trees fall. Therefore, the guidance of limiting post-fire harvest only to severely burned areas or only to fire-killed trees, as suggested by Lindenmayer et al. (2006) and Beschta et al. (2004), may not effectively promote landscape restoration.

Unlike many other natural disturbances, land managers can have significant influence on the timing and severity of fire disturbances, and they can recreate and regulate natural wildfires through prescribed fires. In particular, areas adjacent to severe burns may offer key opportunities for leading systems toward more reference conditions, by using recently burned areas as fire breaks or "anchors" for managed wildland fires (North et al. 2015). For a short period of time, such recent burns can act as an important barrier to fire spread, allowing for controlled low severity fires that are particularly effective at moderating future fire severity (Harris and Taylor 2017; Parks et al. 2015). This opportunity closes as older, high-severity fires often are at risk for high-severity reburns as vegetation regrows and fuels accumulate (Harris and Taylor 2017).

Conclusions

The increase in the size and occurrence of stand-replacing disturbances and limited resources for landscape restoration require forest managers to critically evaluate where fire has moved forest conditions both toward and away from more resilient conditions. Prioritizing areas that have long been occupied by conifer-dominated forests, historically experienced frequent fires, and burned recently in large high severity patches, can promote restoration objectives by avoiding shifts into large, persistent non-conifer communities while recognizing that fire-induced recovery of non-conifer communities also supports restoration. Some of those non-conifer communities may be relatively fixed based upon edaphic qualities, while others may be more ephemeral and could occur in a variable, mosaic arrangement over time. While many of the approaches suggested by our framework have been incorporated into management plans, the application of such ideas will remain a topic of active debate for managers and stakeholders.

Continued evolution of silvicultural approaches through consideration of broad landscape dynamics will be important to address restoration challenges. Breaking the cycle of increasing loss of mature forest to wildfires will depend upon on strategic consideration of fire dynamics as well as promoting heterogeneity that naturally increases the resilience of these landscapes to fire. In particular, managers striving to promote overall landscape resilience may focus more on the "green forests" that surround the "black" areas of high tree mortality, both by using burned areas to contain managed fires in the short-term and by ensuring that fuel conditions within green forests are managed to mitigate threats to nearby, highly vulnerable young forests. Such an emphasis on broad landscape contexts should not discount the value of smaller-scale interventions to address local areas of recreational and cultural significance and concerns as well as the survival of large trees. The general patterns outlined in our framework still leave a broad middle ground where the tradeoffs between intervention and passive regeneration may be highly uncertain. Designing post-fire intervention strategies to advance ecological restoration goals, and then evaluating their performance through long-term monitoring and experimentation will be important to reduce uncertainty about outcomes and build support for post-fire strategies.

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