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Interventions to Restore Wildfire-Altered Forests in California

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Cover photo: Upper montane, red fir forest stand burned in a large, high-severity burn patch 27 years after the 1992 Rainbow Fire on the Inyo National Forest. Photo by Marc Meyer.

Abstract

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In California forests that have evolved with frequent fire, very large and severe modern wildfires can put ecological trajectories on a path of departure from reference or desired conditions. On the other hand, such fires may also advance recovery trajectories on parts of a burned landscape. It is important for land managers to understand and explain how postfire interventions may advance restoration and adaptation goals in different areas. This report advances a science-based framework to guide interventions for these situations. Targeted interventions to restore desired conditions and ecosystem integrity can consider a combination of ecological and social factors. Important ecological factors include the size and arrangement of burn severity patches, departures from reference vegetation and fire regimes, and potential for natural regeneration, all of which vary with topography across burned landscapes. Social factors that may influence interventions include costs, whether areas are accessible, and the presence of sites with particular social and cultural values, such as recreation or gathering sites. Achieving increased social and ecological resilience to disturbances will depend on facilitating restoration of more natural roles for fire in the future and limiting persistent losses of valuable ecosystem services afforded by mature forests. This report offers examples from recent large and severe wildfires to illustrate how restoration could be applied to an archetypal yellow pine and mixed-conifer forest landscape. Strategies include targeting interiors of very large patches of high severity for harvest and replanting, appropriately reducing fuels in moderate and low-severity burn patches and unburned adjacent areas, treating ridgelines and other potential control lines to facilitate management of future fires, and encouraging return of desirable fires within and adjacent to burned areas. Monitoring and adaptive management will be important for addressing uncertainty because successful restoration and adaptation outcomes may not be fully evident for many decades and because stressors are increasing and interacting in ways that are likely to shift trajectories toward novel conditions.

Keywords: Adaptation, California, climate change, ecological restoration, ecosystem resilience, fire management, reforestation, wildfire.

Contents

Introduction

Scope

Ecological restoration is a guiding framework for the U.S. Department of Agriculture (USDA) Forest Service in California to remediate historical ecological degradation and promote resilience to future disturbances (USDA FS 2015b). Ecological restoration focuses on reestablishing the composition, structure, pattern, and ecological processes necessary to facilitate terrestrial and aquatic ecosystem sustainability, resilience, and health under current and future conditions (36 CFR 219.19). Although the text of the USDA 2012 Planning Rule for the National Forest System did not define resilience, we consider the ecological dimensions of resilience, as defined by Walker et al. (2004), as “The capacity of a system to absorb disturbance and reorganize so as to still retain essentially the same function, structure, identity, and feedbacks.” Our concept of restoration emphasizes reinstituting the flows of ecosystem services that sustain human and nonhuman well-being (Alexander et al. 2016), which considers social dimensions of resilience. We focus on postfire management interventions to restore conifer-dominated forests that evolved with frequent fire in the mountains of California, which is a subset of mixed, coniferous-broadleaf forest (M261 province in the Bailey ecoregion classification system) (Bailey 1998, Hessburg et al. 2019). We mapped these areas (fig. 1) using data on pre-Euro-American-colonization, frequent, mostly low-severity (non-stand replacing), fire regime group I (Safford and Clark 2022), although many forest areas in that group, especially along the northern California coast and in areas throughout the Klamath and North Coast ranges, are characterized by a frequent, mixed-severity regime (Spies et al. 2018). This report builds on the framework in a companion report, PSW-GTR-270 (Meyer et al. 2021), which suggested approaches to assess conditions and develop restoration options to promote long-term ecological restoration. That previous report featured a case study in mixed-conifer forests; however, the case study focused on conserving giant sequoia (*Sequoiadendron giganteum*) and Pacific fisher (*Pekania pennanti*). In this report, we document in greater detail how to spatially prioritize interventions in forests where large and severe wildfires have occurred to advance broad ecological restoration objectives. Explaining the rationale for interventions is important given the high stakes and contentiousness of forest management during a period of unprecedented wildfires (fig. 1) (Safford et al. 2022).

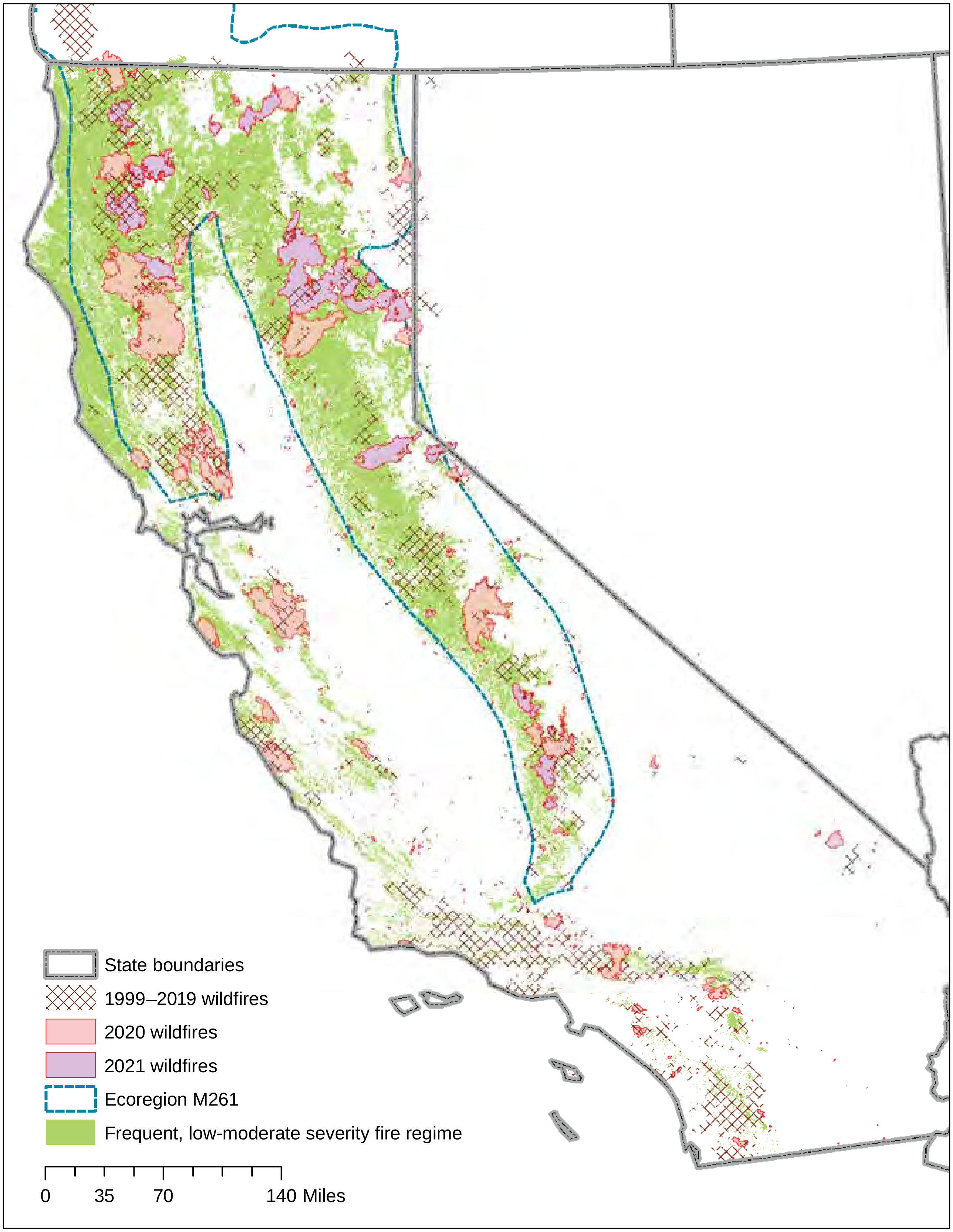

Figure 1—Areas in California that were historically characterized by frequent, mostly low-severity fire (National Fire Plan fire regime group I) and the mixed-conifer ecoregion province overlaid with contemporary wildfires that occurred between 1999 and 2019 (crosshatch) and wildfires in the record-breaking years of 2020 and 2021. Map by Jonathan Long.

Rationales for Forest Interventions

Definition of interventions—

The term "intervention" is commonly used in the medical field to describe an action taken to alter the course of a condition or process to prevent harm or improve functioning. Hobbs et al. (2011) suggested that ecosystem restoration be considered a subset of "intervention ecology." They add that interventions may include not only actions to move a system away from a current undesirable state but also to maintain a system in a current desirable state (Atkinson and Bonser 2020, Holl and Aide 2011, Jones et al. 2018). This report embraces a broad interpretation of interventions, building upon the postfire flow chart in Meyer et al. (2021), which highlighted three general categories of options for influencing recovery trajectories (fig. 2). Where conditions have improved (category I), interventions may be designed to promote and maintain those conditions. Where conditions have deteriorated (category II), interventions may be warranted to restore desired conditions. When the cumulative influence of climate change or other stressors suggests that a restoration approach is no longer feasible or desirable, restoration objectives may need to be realigned (category III) (Millar et al. 2007). However, some interventions, such as intentional use of fire (i.e., prescribed burning, cultural burning, and wildfire managed for resource objectives), may be commonly used under each category, reflecting the critical role of fire in maintaining these systems. Additional documents are valuable for informing intervention approaches, including menus of adaptation approaches developed for California (Swanston et al. 2020) and for tribal contexts (Tribal Adaptation Menu Team 2019) to reduce

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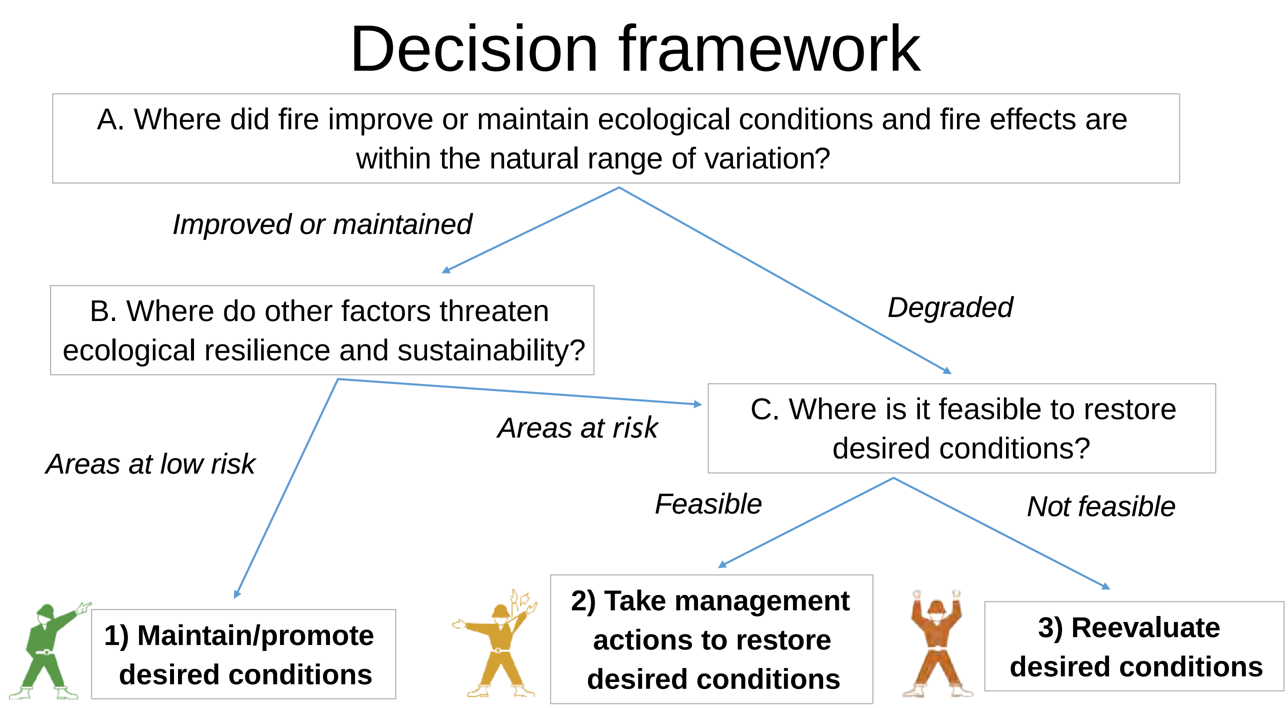

Figure 2—The postfire flow chart is based on three questions (A, B, and C) for the identification of management interventions or "restoration opportunities" (1, 2, and 3) in different portions of the postfire landscape. Adapted from Meyer et al. (2021).

vulnerabilities to climate change and related disturbances. These documents provide examples for translating principles of adaptation into strategies and tactics, several of which are specific to postfire contexts.

This report provides examples of how multiple objectives for ecological restoration can be incorporated into planning postfire landscape interventions. We emphasize broadly restoring ecological functions rather than narrowly trying to re-create past conditions. We also recognize that land managers have multiple objectives that reflect diverse social values, including public health and safety, carbon sequestration, watershed protection, provision of timber supply, and other public benefits that have been characterized as ecosystem services. We recognize that financial considerations, such as net costs of interventions, often drive prioritization; as Franklin et al. (2007) cautioned, ignoring economic objectives will prevent implementation of restoration at scales needed to achieve significant change. Indeed, some proponents of "forest landscape restoration" have articulated more "functional" restoration strategies that strive to meet livelihood needs of local communities and provide ecosystem services (Stanturf et al. 2014). That vision appears consistent with the general guidance established in the 2012 forest planning rule (Spies et al. 2019).

As with the companion framework (Meyer et al. 2021), this report focuses on efforts to promote long-term ecological objectives rather than shorter term emergency rehabilitation measures that are the focus of the Burned Area Emergency Response program. Postfire treatments often include harvest of dead and live trees; conifer tree planting, often in conjunction with control of nonconifer vegetation; and use of prescribed fire. Other potential interventions to achieve long-term restoration goals are only briefly addressed in this report; those include soil erosion control, grazing management, invasive species eradication and control (especially cheatgrass, *Bromus tectorum*), and stream channel treatments.

Land managers in California have long applied postfire interventions, and they have experimented with new approaches that build on past practices. However, many trials of alternative planting designs have been conducted without long-term research or monitoring of outcomes. For example, one can observe examples in the field or find old newspaper articles describing experiments about experimental plantings involving small tree clusters, hardwoods, and giant sequoia, and other noncustomary approaches. However, there have been few published studies on the outcomes from those early experiments.

Historical context for debates over interventions—
The decision of whether to actively intervene after major fires has been a long-running debate in the field of forest management (Chen et al. 2013). In particular, the writings of a prominent conservationist, Aldo Leopold, illustrate the longevity and tenor of these debates. He criticized a model of forestry rooted in a German tradition of planting profitable timber trees "like cabbages," establishing "exotic

plantations," and "purging" nontimber-producing species (Leopold 1939). He called attention to practices that he viewed as contrary to promoting good wildlife habitat, including the planting of pines in open meadow areas and felling snags (standing dead trees) that have high value for wildlife habitat. Instead, he championed a more "nonviolent forestry" that would propagate "owls, woodpeckers, titmice, goshawks, and other useless wildlife." These concerns reverberate in present-day debates over postfire interventions. Some critics of postfire interventions contend that high-severity fires are naturally corrective and that interventions that remove snags, reduce shrubs, and disturb soils with equipment will impede natural recovery mechanisms and reduce habitat for some birds and insects, including many woodpecker and butterfly species (Donato et al. 2006; Swanson et al. 2011, 2014). Although skeptics of interventions may see commonalities in Leopold's writings, many proponents can point to his great interest in replanting trees and investing in "burnt-over" lands as part of efforts to reestablish largely self-sustaining forests (Meine 1991).

Debates regarding postfire restoration are intertwined with a recognition of the importance of using fire to promote resilience. Leopold once counseled against intentional use of fire (Leopold 1920), noting the risk of escapes and culling too many young trees needed to sustain the future forest. However, his later works underscored the value of supporting natural disturbances, such as fire, to promote a state of health on the land, which he described as being "marked by vigorous self-renewal" (Meine 1991). His son Aldo Starker Leopold (Leopold 1963) advanced that legacy by emphasizing the importance of fire to maintaining healthy ecosystems in a report commissioned by the U.S. Department of the Interior National Park Service. In California, that message was advanced through the foundational research of Harold Biswell on the benefits of prescribed burning (Biswell 1999, Miller 2020). Their work has coincided with a growing appreciation of indigenous use of fire in California and other regions—traditions that have long emphasized the rejuvenating benefits of fire (Long et al. 2021).

Theory Regarding Postfire Interventions

Ecological restoration theory suggests that interventions may be necessary to help degraded systems recover from and become more resilient to disturbances, including natural wildfires, drought, and insect outbreaks, as well as more novel stressors (Meyer et al. 2021). In a recent meta-analysis of restoration efforts, Jones et al. (2018) indicated that "passive" restoration was often as effective as "active" restoration/reforestation. However, Atkinson and Bonser (2020) cautioned that such comparisons need to guard against a common bias that active measures may be more commonly applied to more degraded sites, while less degraded sites are more likely to experience passive restoration. Holl and Aide (2011) contended that interventions should target sites where ecosystems are "sufficiently resilient, but where degradation or the landscape context is inhibiting natural recovery" (p. 1561).

Decisions regarding interventions following fire need to consider that fires are a natural disturbance process that can be restorative. Fires can maintain resilient conditions by consuming fuels, killing smaller trees, and reducing overall tree density, and they can restore desirable conditions by promoting canopy gaps and early-seral communities where those are lacking. Concerns regarding interventions include the potential for impeding natural recovery, as some opponents of interventions have noted potential for postfire harvest and planting operations to thwart their own objectives by creating fuel-rich environments or damaging natural regeneration (Donato et al. 2006). Fires can be degradative when they reduce late-successional forest and habitat for associated wildlife in ways that create significant departures in ecological conditions. Evaluating the net effects in terms of ecological departure may depend on the reference values for early- and late-seral conditions and the timeframe for recovery. Many forests in California may be dominated by mid-seral conditions, such that both early- and late-seral communities are underrepresented (Safford and Stevens 2017). Some vegetation communities, such as areas dominated by shrubs (shrubfields or shrublands) areas dominated by shrubs and knobcone pine (*Pinus attenuata*) stands, may be increasing with recent large fires, although their distributions appear to have declined when viewed over a longer period (Airey-Lauvaux et al. 2016, 2022; Reilly et al. 2019). Interpreting the spatial and temporal context of these communities is important but challenging; "shrubfields" are generally regarded as dense shrub patches within a forest matrix that often represent an early successional stage in coniferous forest ecosystems. Shrublands are persistent shrub-dominated ecosystems that were historically dominated by native shrubs and not trees (although tree encroachment may occur in those areas). Researchers have often relied on historical data such as the Wieslander vegetation type maps to infer how vegetation communities have changed across large areas (Dolanc et al. 2014), but such data provide a relatively narrow snapshot of complex landscape dynamics (Donato et al. 2006).

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Debates regarding reference conditions have reflected challenges in deciphering historical datasets on fire and forest vegetation (Fulé et al. 2014). An important concept underlying these principles is the natural range of variation, which is defined as the spatial and temporal variation in ecosystem characteristics under historical disturbance regimes during a reference period or from a reference location (Safford and Stevens 2017). Many restoration efforts have defined reference conditions in California from the period of early Euro-American colonization from 1650 to 1850, which happens to coincide with the Little Ice Age—the coolest period since the early Holocene (Stine 1996). Consequently, a more appropriate reference for contemporary climatic drivers is even farther back in time when conditions were comparably warm and dry (the Medieval Climatic Anomaly, from about 900 to 1350) (Millar and Woolfenden 1999), but information on forest vegetation and fire patterns are much harder to derive for that earlier era. Despite the challenges in deriving reference conditions, current conditions represent an unprecedented combination of warm temperatures and a fire deficit because of exclusion (Marlon et al. 2012, Taylor et al. 2016). As the climate continues to change, it becomes increasingly important to consider how to revise desired conditions to be resilient to natural disturbance regimes.

An important measure of degradation is how much the extent of large, high-severity burn patches (see box 1) has increased over historical references (Miller and Safford 2012, Miller et al. 2009). Recent wildfire events in California have increased the size of high-severity burn patches well beyond the reference for frequent-fire conifer forest areas (Safford and Stevens 2017, Stevens et al. 2017). Although trends have been stronger in the Sierra Nevada, empirical evidence suggest that this trend has also occurred over the past few years in the Klamath Mountains (Bohlman et al. 2021). A postfire development study in the 2007

Box 1: Fire Severity Terms and Metrics

Fire or burn severity is defined as the magnitude of the effect that fire has on the environment, usually in reference to vegetation or soil effects (Sugihara et al. 2018). In forest environments, fire severity is often represented by tree mortality (Keeley 2009). Burn severity is detected, measured, and classified using pre- and postfire satellite imagery and most commonly reported in three different metrics: Composite Burn Index (Key and Benson 2006), percent basal area loss (sometimes referred to as basal area mortality), or percent change in canopy cover. Each metric has its own scale and classification. These metrics are used by land managers and scientists to evaluate fire effects and prioritize postfire restoration including reforestation.

Mixed-severity fire—A complex mix of patches of different fire severities, including patches with no mortality, low severity, moderate severity, or high severity.

High-severity burn patch—A contiguous area of high tree mortality from fire, also described as stand-replacing fire. High severity is sometimes classified based upon a Composite Burn Index of 2.0–3.0, which approximately corresponds with >90–95 percent loss in basal area or canopy cover. Managers also use lower thresholds (e.g., >75 percent basal area loss) in postfire project planning to meet management objectives and to account for areas where very few live trees remain uninjured or delayed tree mortality is expected.

Moderate-severity burn patch—A contiguous area with an intermediate level of tree mortality from fire, in which understory plants, fine surface fuels, and some coarse woody debris may be consumed. Moderate severity covers a broad range of effects, corresponds to a Composite Burn Index of 1.0–2.0, and is often classified as 25–75 percent basal area or canopy cover loss.

Low-severity burn patch—A contiguous area with a low level of tree mortality from fire and consumption of some surface litter and understory plants. Low-severity fire effects correspond with a Composite Burn Index of 0–1.0 and is often classified as < 25 percent basal area or canopy cover loss. This class may also include small areas of unchanged or unburned fire effects.

Moonlight Fire footprint in northeastern California showed that passive management (i.e., allowing "natural" vegetation succession) within such large, high-severity burn patches would likely result in a shift to a persistent shrub-dominated state (Stephens et al. 2020). Furthermore, the fuel complex associated with this altered state (a high continuity of live fuels interspersed with high loads of coarse wood) may allow for repeated high-severity fires in relatively short intervals (Lydersen et al. 2019). Rising incidence of nonfire mortality, including insects and disease, are also increasing the potential for stands to burn at high severity (Stephens et al. 2022). Although actual fire outcomes are regularly influenced by factors other than prefire condition, such as fire weather, recent research has demonstrated that dead biomass and live-tree density are important predictors of large, high-severity burn patches (Halofsky et al. 2011, Safford et al. 2022, Stephens et al. 2022).

A central concern in ecology is to inhibit shifts in key ecological conditions and disruptions of ecosystem services that are likely to persist without intervention or may even be irreversible (Scheffer et al. 2001). Recent research in forests of the Western United States has examined how severe fires have driven such persistent shifts, or vegetation type conversion (VTC), from conifer forest to other vegetative communities, especially shrublands (Coop et al. 2020, Guiterman et al. 2022). Such conversions have been especially associated with short-interval, high-severity reburns (fig. 3) (Coppoletta et al. 2016). Changes in climatic conditions and the scale, severity, or frequency of disturbance regimes have increased the likelihood of such shifts (Safford and Vallejo 2019, Tepley et al. 2017).

Severe wildfires are projected to reduce old and tall trees in California, which are disproportionately important as nesting habitat for California spotted owls (*Strix occidentalis occidentalis*) and other rare wildlife (North et al. 2017, Stephens et al. 2016). A recent study found that California spotted owls avoided severely burned forest when more than 5 percent of their home range was affected by large (>100 ha) high-severity burn patches (Jones et al. 2020). Such concerns have also served as a rationale for interventions to accelerate recovery of such habitat through planting and fuel reduction.

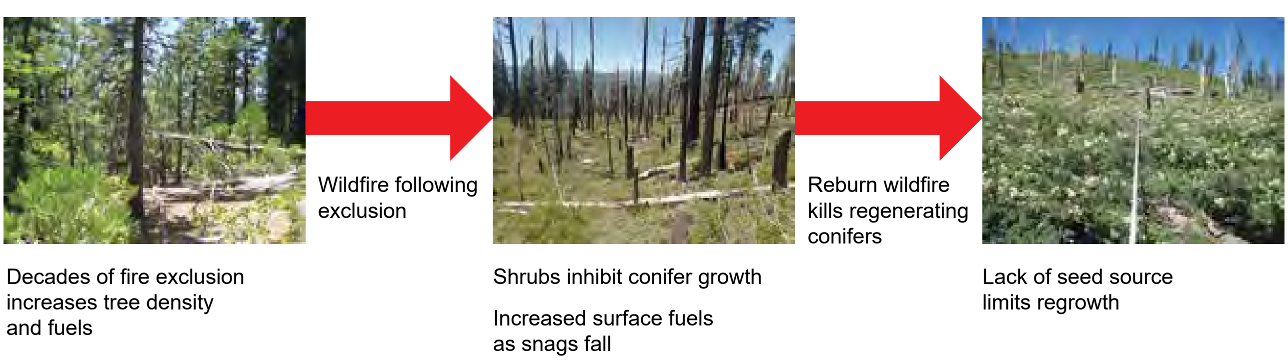

Figure 3—Pathway depicting the potential shift from fire-excluded conifer forest to persistent shrubland following two overlapping high-severity fires. Adapted from Coppoletta et al. (2016).

Additional rationales may be linked to other management goals, especially public safety and timber production, some of which may be associated with legal obligations on the part of land managers (Meyer et al. 2021). We focus on interventions to achieve ecological restoration objectives, while recognizing that other objectives are important and may be intricately linked to restoration objectives. It may be difficult to neatly separate motivations when actions are taken that address multiple objectives. For example, postfire harvest may reduce fuels in the form of dead trees, while also making it safer to plant trees and generate revenues to offset the cost of postfire treatments and support rural economies (Long et al. 2014b).

The rationale for interventions hinges on an expectation that forest ecosystems will not recover without human assistance within an acceptable time frame and also that such interventions will be effective. Larson et al. (2022) stressed the need to distinguish between dispersal limitations, for which intervention may be effective, and climatic limitations, which might render interventions more futile. A variety of tools have been developed to predict the likelihood of natural reforestation (Meyer et al. 2021). Research has helped to project the likelihood of exacerbated fire severity and ecological services within and without fuel reductions (Johnson et al. 2020, Ritchie et al. 2013). As a complement or alternative to predicting trajectories, one could target areas where prefire conditions were most departed before the fire, as indicated by high tree density and fuel accumulation (North et al. 2022), while considering how the increased amount of dead biomass due to wildfire will further exacerbate such departures (Stephens et al. 2022).

Principles for Postfire Restoration

Throughout this report, we emphasize how six principles identified in the previous postfire restoration framework (Meyer et al. 2021) inform goals for interventions: (1) basing restoration on prioritization, (2) considering landscape context, (3) incorporating adaptations to agents of change in support of (4) promoting regional native biodiversity, (5) restoring key ecological processes, and (6) sustaining diverse ecosystem services using a pragmatic approach. Larson et al. (2022) similarly laid out principles to guide postfire restoration through a landscape ecology perspective in the interior Pacific region. Below we translate those principles into goals that can be quantified into measurable objectives.

(1) Promote natural disturbance regimes—
Consistent with the principle of restoring key ecological processes, a central goal in promoting resilience in ecosystems that evolved with fire is to facilitate the reintroduction of fire at an ecologically appropriate frequency, scale, and intensity (North and Keeton 2008). For frequent-fire forests in California (fig. 1), this means facilitating and encouraging fires with predominantly low-severity fire effects rather than placing a heavy emphasis on fire exclusion (Franklin et al. 2007,

North et al. 2014). Fire exclusion is the curtailment of wildland fire via deliberate suppression after ignition as well as unintended effects of human activities, such as intensive grazing that removes grasses and other fuels that carry fire (Keane et al. 2002). Frequent fires provide an important feedback mechanism by maintaining open canopy conditions and favoring large, fire-tolerant trees that are resilient to future fires (Larson et al. 2022). Consequently, a fire-centric approach can reduce the risk of losses of mature conifer forests and associated stored carbon from future wildfires (North et al. 2012, 2021). Promoting the establishment of larger trees, which are often more resilient to fire, could facilitate earlier reintroduction of fire and reduce the rate of conversion of conifer forests. Because using fire and promoting large trees are both important to promoting system resilience, determining how to facilitate increasing frequency of fire while growing young trees to a size that confers resilience is a key challenge in forest ecosystem restoration (Bellows et al. 2016).

(2) Promote heterogeneity at landscape and stand scales—
Forest heterogeneity often refers to variation in forest structure and fuels within stands in horizontal (e.g., single trees, clumps of trees, and gaps of no trees) and vertical (e.g., vegetation at different heights from the forest floor to the top of the forest canopy) dimensions, or across large landscapes (North et al. 2009). Restoring heterogenous stand structure and fuels over large areas may help disrupt ecological synchrony and limit large and severe disturbances (Betancourt 2012) as well as promote biodiversity. Recent research has suggested that restoring stand-scale variability, such as openings and clumps of trees, may promote resilience to drought- and insect-related mortality (Kane et al. 2019). Reducing the amount and continuity of postfire fuels can reduce the potential for very large and severe fires (Collins et al. 2019); this principle also reflects the need to address future fuel accumulation from postfire tree mortality (Larson et al. 2022). Encouraging an appropriate mix of old conifer forest while regenerating conifer forest and other vegetation, including hardwood trees, shrubs, and herbaceous plants, may increase both resilience and habitat diversity (North et al. 2019). A particular challenge following stand-replacing fires is to promote a diversity of age classes. Promoting distributed patches with relatively high tree density, using topography and associated microsite conditions to inform their locations, may support wildlife species that depend on such dense stands as well as future habitats for postfire specialists (Hessburg et al. 2016, North et al. 2019). Treatments that are spatially heterogeneous, both at the stand and landscape scale, are more likely to maximize benefit for wildlife species (Hessburg et al. 2016), although habitat connectivity among stands, particularly for old-forest-dependent species, is also a primary objective for wildlife conservation at the landscape scale (Hessburg et al. 2019). The relationships among vegetation heterogeneity, wildlife habitat, and fire dynamics are complex. An integrative restoration strategy is to promote desirable

wildlife habitat connectivity in some areas, while reducing undesirable wildfire fuel connectivity in others. Specifically, that approach could maintain connectivity in lower slope positions across connecting drainages, which tend to be more productive fire refugia that are important for old-forest-associated wildlife, such as spotted owls (Lesmeister et al. 2018, USDA FS 2019). Meanwhile, this same approach could be used to reduce fuel connectivity across steep upper slopes and ridgetops, where fuel accumulations appear to more strongly drive extreme fire behavior (Airey-Lauvaux et al. 2022, North et al. 2009).

(3) Support early-seral conditions—
Wildfire events are a key process for promoting early-seral conditions (nonforest or preforest vegetation followed by young trees) (Swanson et al. 2014), which support distinctive biological communities, including uncommon wildlife species (Fontaine and Kennedy 2012, Halofsky et al. 2011, Hutto 2006, Saab et al. 2011). Shrubfields and hardwood-dominated woodlands represent alternative stable states in mixed-conifer forest systems (Coppoletta et al. 2016, Odion et al. 2010). These communities contribute to ecosystem services, including conservation of avian biodiversity (Fontaine et al. 2009). Therefore, ensuring provision of early-seral habitats is important to promote regional biodiversity as well as to ensure continual recruitment of areas that will develop into late-seral habitat. Postfire early-seral habitat supports resprouting shrubs and hardwoods and postfire occurrences of plants, fungi, insects, birds, and other animals; it also supports unique legacy features, such as dead trees, downed logs, and burned wood (Franklin et al. 2007). Although large standing trees and fallen logs have disproportionate values as wildlife habitat, dense patches of dead trees are also important as they provide a pulse of prey items for woodpeckers (White et al. 2016). Prefire snags are particularly important to retain for nesting by species, such as the black-backed woodpecker (*Picoides arcticus)* (Nappi and Drapeau 2011). In many areas, fire exclusion has led to the current extent of closed-canopy, old-forest habitat rather than a landscape mosaic of successional stages (Lesmeister et al. 2018). Small- to moderate-sized patches of early-seral habitat, interspersed within larger patches of low- to moderate-severity burned (or unburned) forest habitat, may be particularly valuable in promoting habitat heterogeneity that can benefit species, such as spotted owls (Jones et al. 2020, Lesmeister et al. 2018, Roberts et al. 2011). High-severity fire has the potential to aid restoration of certain tree species that have been displaced through fire exclusion, including various hardwood species that are adapted to recover through resprouting following stand-replacing fire. Only a few conifer species in California, the most common of which is redwood (*Sequoia sempervirens*), demonstrate such resiliency to severe fire (Lazzeri-Aerts and Russell 2014). Furthermore, knobcone pine and several species of western cypresses (*Hesperocyparis* spp.) benefit from high-severity fire and early-seral habitat that promotes their seed dispersal, germination, and establishment (see box 4 on p. 42).

Therefore, some amount of high-severity fire may be restorative where those patches contribute to landscape heterogeneity and do not interrupt habitat connectivity for forest-dependent species (Hammett et al. 2017, Reilly et al. 2019, Thompson et al. 2021a).

However, the extent of high-severity fire is greatly exceeding conditions that existed before Euro-American colonization (by three to six times in 2020 fires, according to Safford et al. 2022). The size of individual high-severity burn patches in mixed-conifer, yellow pine forest is growing and substantially exceeding historically based references, and increased habitat fragmentation (Steel et al. 2018, 2022; Stevens et al. 2017) and losses of mature forest are expected to persist longer than would be expected under a less disrupted fire regime because of the compounding influence of climate change (Guiterman et al. 2022). Therefore, although early-seral conditions are an important consideration in postfire restoration, they are likely to be overrepresented in many current and future forest landscapes and therefore may require far less intervention than the recovery of late-seral conditions (Coppoletta et al. 2016, Fontaine et al. 2009, Lesmeister et al. 2018, Odion et al. 2010).

The extent of high-severity fire and size of individual high-severity burn patches are exceeding conditions that existed before Euro-American colonization, and losses of mature forest are expected to persist longer because of climate change.

(4) Maintain and promote late-seral conditions—
Protecting large-diameter trees and fire refugia is an important postfire management principle (Larson et al. 2022). Furthermore, interventions may be warranted to accelerate establishment of mature trees, particularly within large, high-severity burn patches. Such efforts can restore the flow of important ecosystem services, including seed production for natural regeneration and wildlife, carbon sequestration, timber production, resilience to wildfire, key wildlife habitat associated with large trees and cavities, and connections among late-seral forest areas. It can also provide more options for creating desirable and resilient forest structural conditions in the future. For example, encouraging denser clusters of trees in some productive microsite areas may promote future habitat for spotted owls (North et al. 2019). To achieve these outcomes, such planted stands need to be shepherded to a point at which they can largely withstand the next fire.

Although the size of high-severity burn patches and the proportions of fires that are burning at high severity appear to be increasing, there are often many areas that have burned at low to moderate severity and have moved toward a more desired condition through a spatially variable reduction in tree densities and fuels. In addition to unburned areas, these are areas where mature trees remain on the landscape and provide a valuable seed source, especially when adjacent to high-severity areas. Interventions taken to protect and promote mature trees might include removal of snags (to reduce long-term fuels), prescribed burning (to reduce fuels and reduce fire return interval departures), and thinning of live trees (to further reduce stand densities, remove ladder fuels, and modify species composition to favor fire- and drought-resilient species). All these interventions can build off the beneficial outcomes of the fire and continue moving areas to align more with the natural range of variation or desired conditions.

(5) Promote better adapted composition and densities—
Interventions in frequent-fire conifer forests may be warranted to better align species composition and structure with future fire regimes and climate (Larson et al. 2022). In the Sierra Nevada, fire exclusion and other alterations of disturbance regimes have in many places shifted composition toward increased abundance of younger incense cedar (*Calocedrus decurrens*) and white fir (*Abies concolor*), which are relatively fire and drought intolerant (North et al. 2019, White and Long 2019). In that region and other drier, more southerly conifer forests, ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*P. jeffreyi*), and sugar pine (*P. lambertiana*) have often been prioritized when conifers are planted as they are expected to be more resilient to future fire and warmer temperatures than white fir and incense cedar. In northwest California, including the Klamath region, Douglas-fir (*Pseudotsuga menziesii*) is a dominant conifer that tolerates fire, drought, and shade; it has also been an important component of tree planting (Lopez Ortiz et al. 2019), although it has increased relative to pines and oaks in many areas (Skinner et al. 2006, Spies et al. 2018). Planting adapted stock (e.g., that have resistance to diseases or adaptations for more droughty conditions) may also promote resilience to novel threats from white pine blister rust or climate change, although such strategies may require many years to evaluate their effectiveness. Hardwood trees, including California black oak (*Quercus kelloggii*), tanoak (*Notholithocarpus densiflorus*), and aspen (*Populus tremuloides*) can outpace growth of conifer trees for several decades following stand-replacing wildfires (Long et al. 2018a, Steel et al. 2018, Young et al. 2020). Realigning desired conditions to align more with current and future climatic conditions may be necessary to maintain essential ecosystem functions and services (Millar and Stephenson 2015). Some of the shifts toward hardwoods may reflect both a natural corrective to fire exclusion and a longer term adaptation to a warming and drying climate (McIntyre et al. 2015), including following exceptional drought events (Young et al. 2020). Managers may pursue a heterogeneous mix of conifers, hardwoods, and shrubs to promote wildlife habitat and overall ecological resilience (North et al. 2019).

In addition to considering composition, it is important to consider whether to further lower planting densities to address increased water demand associated with climate change (North et al. 2019). In recent decades, silvicultural guidance in the region has recommended stocking densities ranging from 125 to 200 trees per acre (51 to 81 trees per hectare) in mixed-conifer and Jeffery pine vegetation types, depending on site productivity (USDA FS 1991). In the King Fire, lower target densities (e.g., 40 to 120 trees per acre [16 to 49 trees per hectare]) were suggested for strategic fire management zones, wildland-urban interface (WUI) zones, and middle to upper slope areas, although target planting densities were about 20 percent higher to allow for expected mortality (North et al. 2019: fig. 5). Those target values were lower than the historical values (>300 trees per acre) reported from plantings a half century ago (North et al. 2019). Because managers have observed high seedling survival and reduced support for precommercial thinning, they have reduced their target planting densities, a trend that is discussed further below under planting strategies.

Recognition of the value of nonconifer types and the need to promote less dense, more fire-resistant and fire-resilient forests have been influencing reforestation strategies for several decades. However, these considerations are becoming even more salient as shifting climate and fire regimes increase hardwood components of mixed forests in many regions (Lenihan et al. 2008, McCord et al. 2020, McIntyre et al. 2015). Plans to allocate resources for interventions, such as planting conifer trees, must be continually examined as climate change increases both the need for interventions and their risk of failure.

Responding to Multiple, Overlapping, Large, High-Severity Disturbances

Land managers are increasingly confronted with interacting and repeated large disturbances, such as drought and wildfire (fig. 4). A major concern with the increasing size of disturbance-related tree mortality patches is reduced landscape heterogeneity when forest lands are converted to large areas of nonforested vegetation (McCord et al. 2020) or to forests of a single-age class. For conifer forests in California, a common objective in many restoration projects is to restore uneven or multi-aged forests. Replanting is an important tool for reestablishing forested areas, but planted stands have often not proven resilient to reburns (Levine et al. 2022, Zald and Dunn 2018). For example, the 2013 Rim Fire burned through plantations established after the 1987 Stanislaus Complex fires; the 26-year return period seems unremarkable as it is greater than the reference mean fire return interval in these systems. However, the severity of both events exceeded historically based references, and the best predictor of fire severity in the Rim Fire was how severe the area last burned (Harris and Taylor 2017). Furthermore, burns are increasingly reoccurring after even shorter intervals. For example, the 2018 Camp Fire reburned portions of the 2008 BTU (or Butte) Complex fires (fig. 4). In both instances, areas that burned initially at high severity also reburned at high severity, with high-severity burn patches growing larger in the reburn. This trend could gradually erode later seral forest conditions and associated ecological services across the landscape, and the cumulative effect of homogenizing the landscape may reduce resilience of those forest ecosystems to future disturbances.

The adverse effects of high-severity reburns on forest cover and resilience are evident at landscape scale where overlapping footprints of several fires burned between 2001 and 2021 in the northeastern corner of the Plumas National Forest (fig. 5). Within two decades, six fires burned into, or in the immediate proximity of, one another: 2001 Stream Fire, 2006 Boulder Complex, 2007 Antelope Complex,

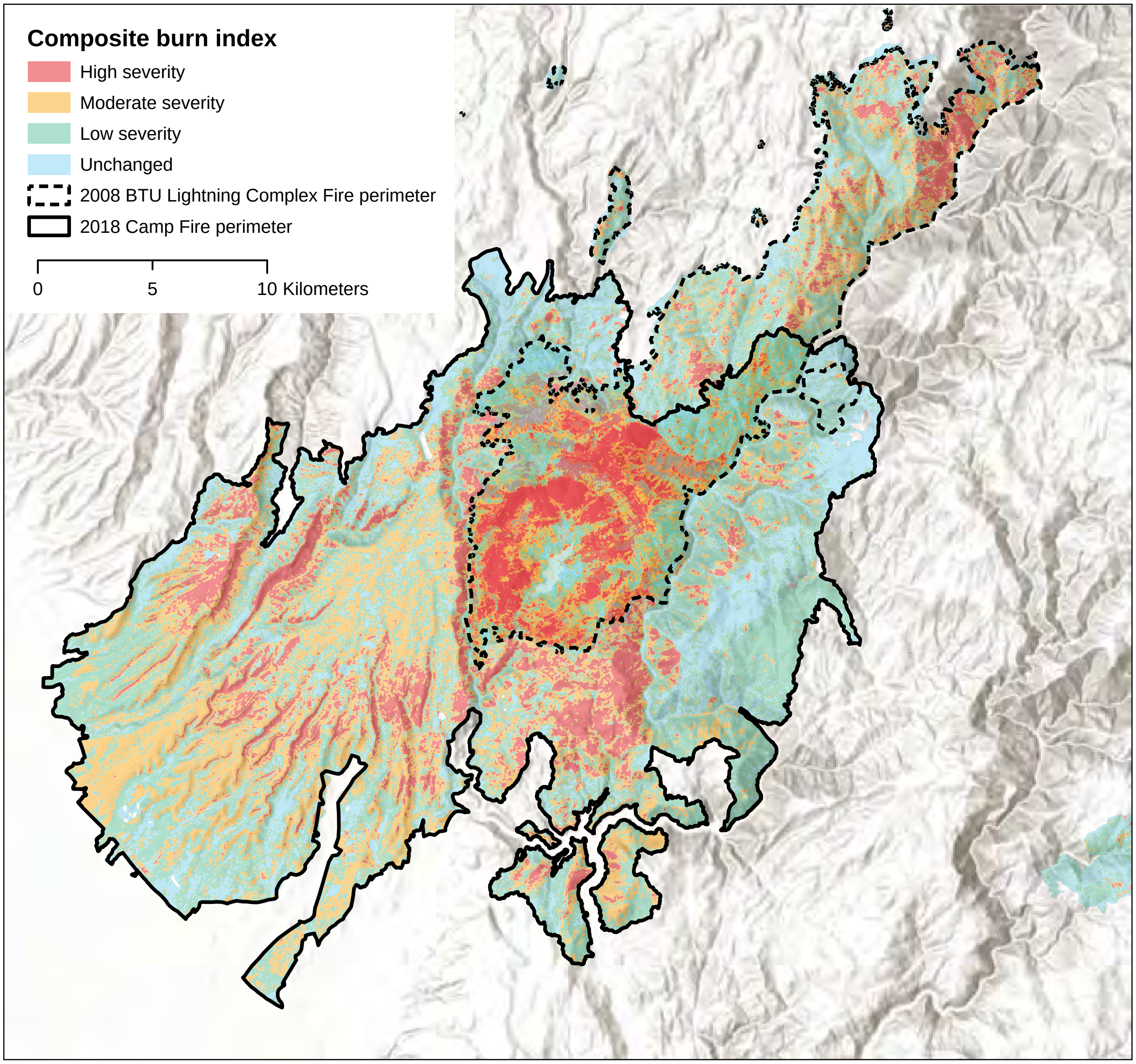

Figure 4—Areas of high severity from the 2008 BTU Complex fires burned with high severity in the subsequent 2018 Camp Fire. Map by Ryan Tompkins.

2007 Moonlight Fire, 2019 Walker Fire, and 2021 Dixie Fire. All these fires had notable proportions of high-severity fire, some of which overlapped, creating an ever-expanding homogeneous area of postfire early-seral vegetation. The 2021 Dixie Fire included four patches of high-severity fire greater than 10,000 ac (4047 ha) in size, which perpetuates homogeneity of early-seral vegetation in the northern Sierra Nevada in this area (fig. 6).

The potential for overlapping wildfires highlights that (1) fuel reduction benefits of wildfire may be short-lived in some cases, particularly in dynamic postfire environments; (2) interventions may need to focus on modifying future fuel profiles; and (3) treating low- and moderate-severity fire patches to reduce fuels and fire is an important tactic in an overall landscape strategy. Postfire restoration provides opportunities to consider the development of secondary cohorts during the initial phase of regeneration.

To promote a diversity of age classes and accelerate recovery of late-seral forests, reforestation efforts need to consider how to expedite development of cone-bearing trees to promote seed production. One idea has been to plant small "founder stands," or small groups of trees strategically planted in mesic and less fire-prone locations to serve as the future seed source for trees in the surrounding area (North et al. 2019). A recent study tested the "nucleation" concept of planting

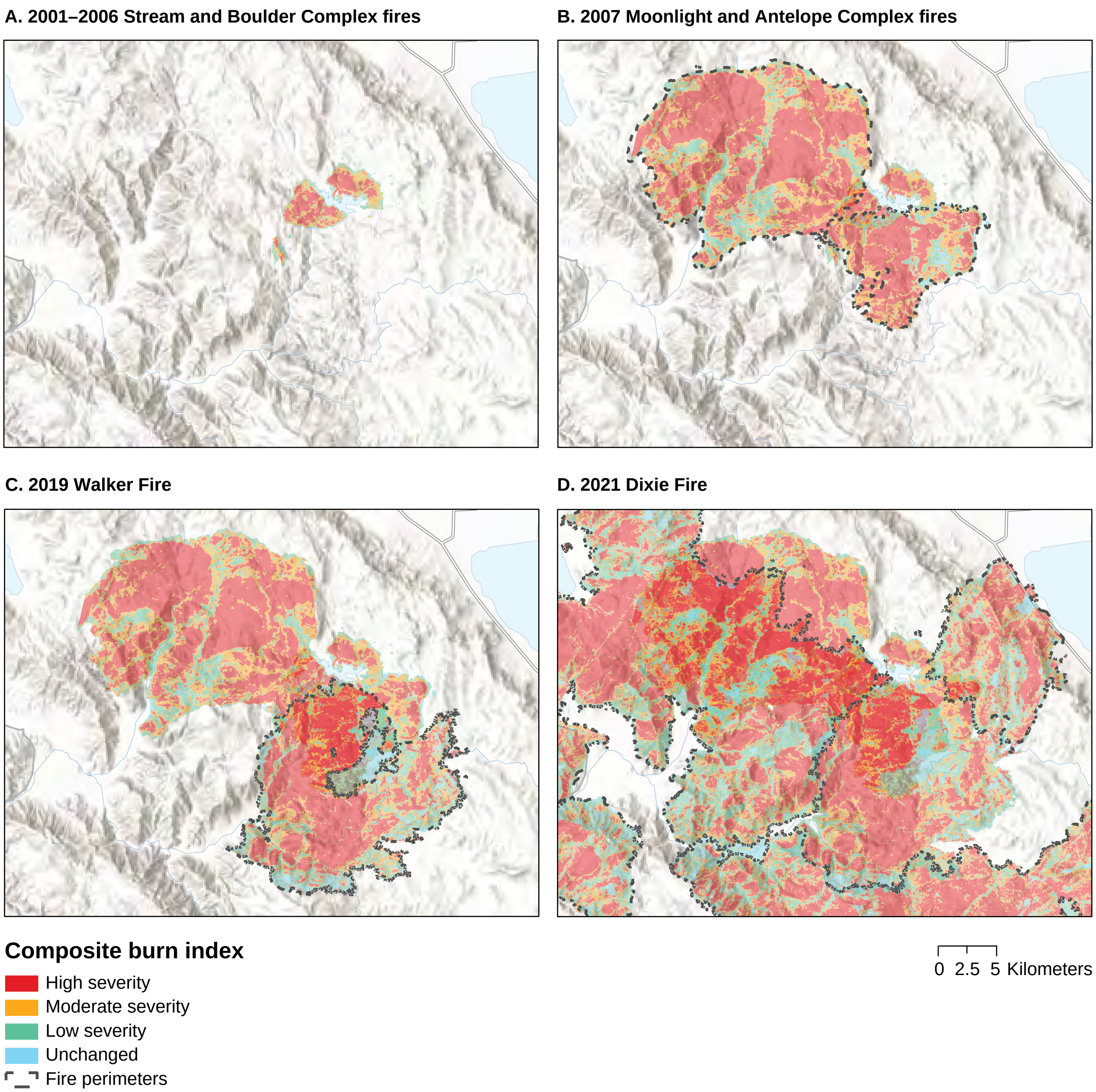

Figure 5—Fire severity patterns within individual large wildfires that occurred between 2001 and 2021 on the Plumas National Forest in the northern Sierra Nevada. All fires had uncharacteristically high proportions of high-severity fire effects. Although some of the reburned areas were classified as low or moderate severity, the Walker Fire and the Dixie Fire reburned substantial portions within the footprints of the Antelope Complex and the Moonlight Fire, respectively, at high severity. Maps by Ryan Tompkins.

trees to increase recruitment within a larger area and found that such plantings may provide a statistically significant but small role in driving recruitment outside of the planted site (Ursell and Safford 2022). Such stands could be planted throughout a larger landscape, on the order of tens to hundreds of acres, to facilitate more gradual reforestation if those founder stands can gradually expand. However, it is important to consider how such founder stands would fare and how they might be conserved in this era of increasing wildfire activity.

Planting at low densities and wider spacing, paired with more intensive investment in competing vegetation control, could facilitate open grown trees that reach cone-bearing age sooner. This approach might reduce the need for precommercial thinning activities to reduce competition among trees, but it would likely increase the need to manage competing vegetation to maximize tree vigor and reduce fuels. Where fuels have been managed to moderate fire behavior, subsequent fires can prepare the sites for seed-fall events and enhance the development of multicohort stands. These approaches could be modified to account for site productivity and expected natural regeneration potential.

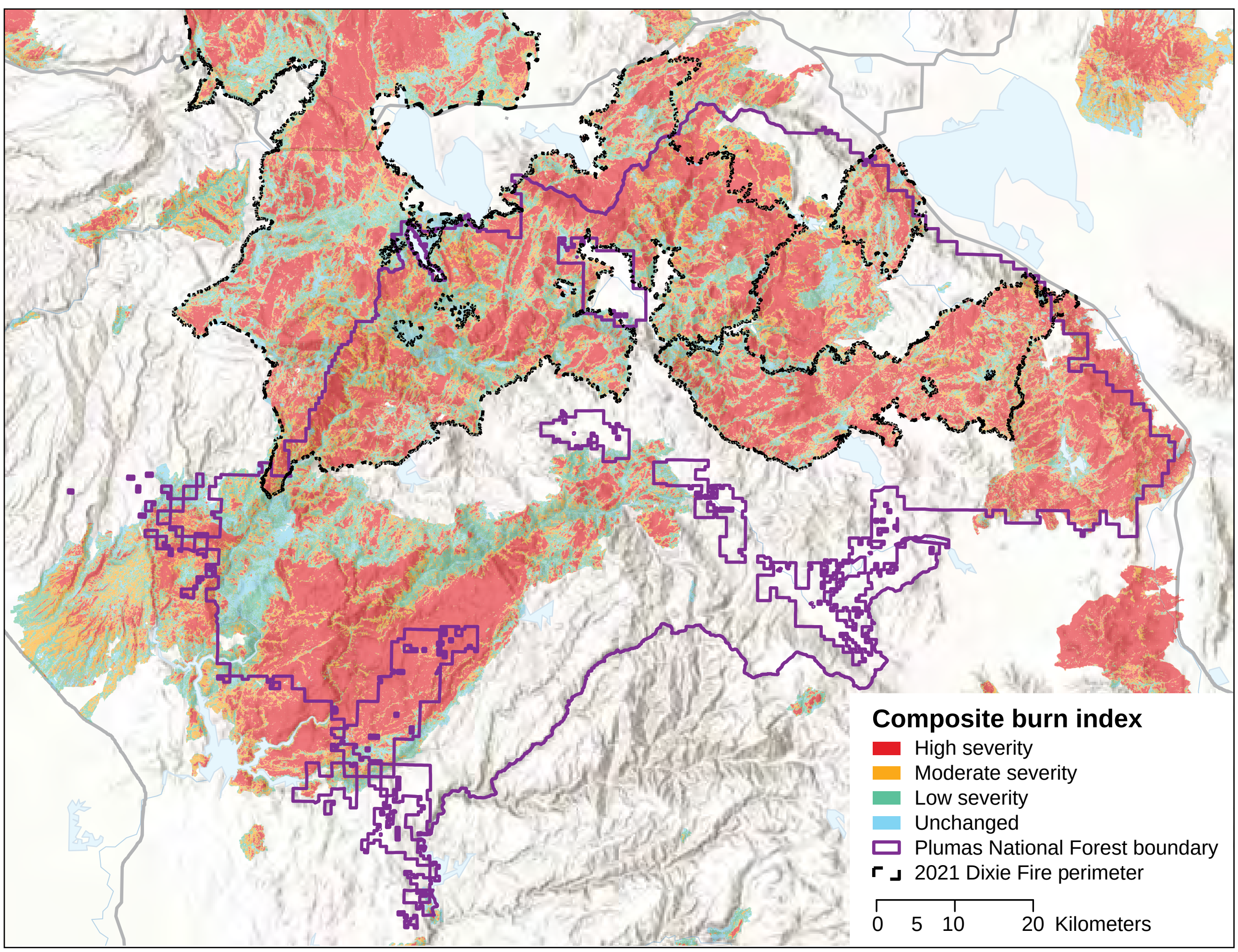

Figure 6—Cumulative areas burned, by severity, on the Plumas National Forest between 2000 and 2021. In areas of overlapping fire footprints, the map depicts the highest fire severity. Map by Ryan Tompkins.

Overview of Postfire Interventions in a Landscape Context

The most common interventions used on postfire landscapes have historically been the removal of dead trees, reduction of surface fuels, and tree planting, often in conjunction with varying forms and degrees of controlling competing vegetation. Prescribed burning and thinning of live trees are also typical practices that have increasingly been suggested as important in postfire landscape restoration (North et al. 2019, White and Long 2019). In this section, we briefly describe how these types of interventions might be combined and applied to achieve the priority restoration goals discussed in the previous section.

The different pathways presented in the decision support framework (fig. 2) may result in a wide range of postfire interventions to address the effects caused by the fire. For example, some of the primary strategies under category I (maintain/promote desired conditions) may include planning for managed fire or proactively implementing prescribed fire as well as monitoring of natural regeneration to ensure that conditions move along a desired trajectory. Interventions under category II (take management action to restore desired conditions) may include harvest of live or dead trees along with treatments to reduce fuels (possibly including prescribed burning) as well as replanting to encourage desired forest structure, age, and species composition. Under category III (reevaluate or realign desired conditions), interventions may include planting to promote a potentially novel vegetative condition that may better sustain desirable ecosystem services. These restoration opportunities can be translated into more specific actions by considering a suite of site-specific factors, such as the size and arrangement of burn severity patches, topography, and proximity to roads and other features of the built environment or WUI. For example, North et al. (2019) suggested applying such factors to identify zones where (1) replanting might be generally unnecessary (owing to availability of seed sources), (2) it was necessary and feasible, and (3) it was infeasible; those distinctions roughly map to the categories in the decision support framework. Although category II areas are expected to be the focus of interventions, treatments may be applied to any of the three categories. However, under category III, one might conclude that because an area would no longer support a previously occurring vegetation type, one might undertake various interventions to foster an alternative type.

Thinning Live Trees and Removing Dead Trees

Most discussions of postfire interventions tend to focus on harvest of dead trees. However, thinning of live trees is an important intervention that can build on the desirable ecological effects of a wildfire, especially in low- and moderate-severity areas, by bringing stand densities and species composition more in alignment with the future range of variation, which can be based on the natural range of variation with consideration of how climate is changing (Meyer et al. 2021, North et al. 2022).

Tree harvest is a conventional tool used to meet a variety of objectives, including timber production, abating hazard trees that pose a risk to the public or forest workers, managing the development of fuel loads over time, or to prepare the site for planting. Harvest of dead or dying trees (snags) has long been referred to as "salvage" when used to recoup economic values, but the term is also used to describe harvests to achieve other objectives, including fuel reduction and public safety after a major disturbance. Postfire salvage logging has been one of the most controversial interventions, particularly because it reduces the habitat/legacy structures of dead trees and can affect conditions for natural regeneration (Beschta et al. 2004, Noss and Lindenmayer 2006). Lindenmayer and Noss (2006) highlighted the importance of retaining high levels of biological legacies (e.g., dead trees), protecting riparian areas and soils from damage, and ensuring maintenance or creation of habitat for species of conservation concern. Franklin and Agee (2003) indicated that strategically planned salvage harvest may be an ecologically appropriate tool in the dry, mixed-conifer forests of Western North America that historically had low-severity fire regimes. Where fires are burning large patches at higher severity, salvage harvest could reduce the quantity and continuity of dead trees, which can reduce the potential for high-severity reburns and ultimately lower the risk of forest type conversion (Coppoletta et al. 2016, Thompson et al. 2007). Removal of dead trees postfire has been shown to effectively reduce the development of postfire fuel loads (Ritchie et al. 2013); however, treatment prescriptions need to effectively target different classes of fuels to achieve overall goals. For example, targeting large trees that have the most economic value may not sufficiently address the smaller fuels (Knight et al. 2022, North et al. 2021). Timing of dead tree removal is also important; while mechanical treatments can reduce woody fuel accumulations, particularly in the 1000-hr fuels, the efficacy of treatment may be impacted by deterioration of wood quality and subsequent breakage into smaller fuel sizes (Moore et al. 2021).

Postfire management plans on national forests generally require retention of some standing dead trees, typically the largest snags, to maximize value for wildlife habitat and to provide long-term inputs of coarse wood into streams and terrestrial environments (fig. 7). Large, fire-killed snags may provide valuable habitat for a suite of postfire-adapted species, and they tend to persist longer than smaller trees (Chambers and Mast 2005, Dahms 1949, Passovoy and Fulé 2006, Ritchie et al. 2013, Russell et al. 2006). Larger snags are important to maintain because of their structural importance for in-stream aquatic habitat when they fall into streams (Long et al. 2014b). Ecologists have suggested that harvests target smaller trees to reduce fuels and retain larger snags for wildlife habitat (Long et al. 2014b). Because of unfavorable economics, such interventions can be challenging to implement at a meaningful scale, underscoring why intentional use of fire is an important alternative to consider. Combinations of tree removal and fire will be necessary to reduce live and dead fuels to promote resiliency (Knight et al. 2022, North et al. 2021).

Figure 7—A large dead tree surrounded by smaller dead trees in an area designated for no treatment as part of the Salt Creek sale, Klamath National Forest. Photo by Mike Hupp.

Brown et al. (2003) suggest that tree removal prescriptions may consider the benefits and risks of leaving snags and coarse wood by recognizing context of place and process. For example, white fir snags may be preferentially retained because they can remain standing longer than ponderosa pine and provide greater habitat value for cavity-excavating birds than incense cedar (Ritchie et al. 2013). Snag retention can vary greatly between forests, with some postfire project managers preferring to leave snags throughout the treatment area to retain the largest snags. Other approaches focus on retaining groups of snags (box 2), which can be both beneficial to wildlife use and provide for better worker safety by limiting exposure of workers in reforestation efforts to dangerous trees. At a fine scale, retaining snags and large, down woody debris as dead shade may also maintain more favorable microsites for tree establishment and help to limit shrub regrowth (Conard and Radosevich 1982). At the landscape scale, leaving clumps with high densities of dead trees may promote several bird species (fig. 8) better than prescriptions that more evenly remove snags from burned stands (White et al. 2016). Different bird species have different preferences, with black-backed woodpeckers preferring dense stands (Tarbill et al. 2015), while birds that consume mammals (like raptors) or flying insects may use snags in more open stands for perching and foraging. Because wildlife relationships are complex and context dependent, consultation with wildlife specialists is important in designing these interventions.

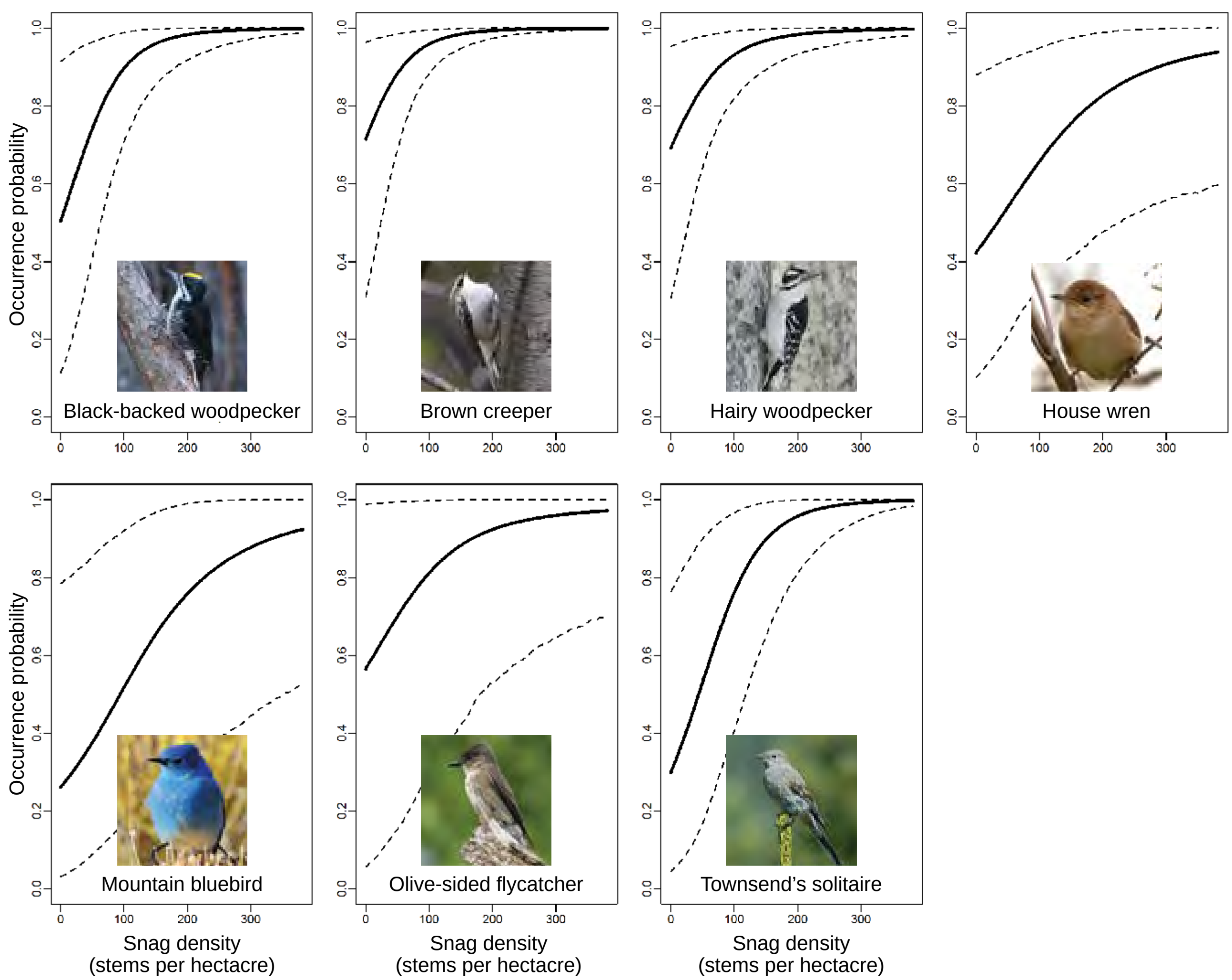

Figure 8—Relationship between density of dead trees (snags) and probability of occurrence among six bird species that are positively associated with high snag densities. Adapted from White et al. (2016). Photos courtesy of Wikimedia Commons.

Box 2: Snag-Retention Patches in the Eiler Fire Footprint, Lassen National Forest

In 2014, the Eiler Fire burned 33,160 ac (13 420 ha) of private and National Forest System lands in northern California. High-severity fire effects accounted for more than two-thirds of the area burned, 75 percent of which was concentrated in one large, contiguous patch that exceeded 17,000 ac (6880 ha). Immediately after the fire, Lassen National Forest managers developed a large-scale salvage and restoration project with the intent of reducing safety hazards, recovering economic value, reducing postfire fuel loads, and reestablishing conifer forest (USDA FS 2015a). One of the goals of the project was to retain key habitat features for wildlife species that benefit from postfire conditions, such as black-backed woodpecker, pallid bat (*Antrozous pallidus*), and fringed myotis bat (*Myotis thysanodes*), while also meeting fuel reduction and economic objectives. To meet this goal, managers identified areas within salvage logging units that would be retained as snag “leave islands.” These areas ranged in size from 2 to 5 ac and comprised about 25 percent of the area in each unit (fig. 9). Leave islands were distributed throughout the units to create heterogeneity in postfire structure, with most located in productive sites. Monitoring conducted 2 years postfire documented higher cover of understory plants and coarse woody debris in unlogged leave islands compared to salvaged areas (USDA FS 2017).

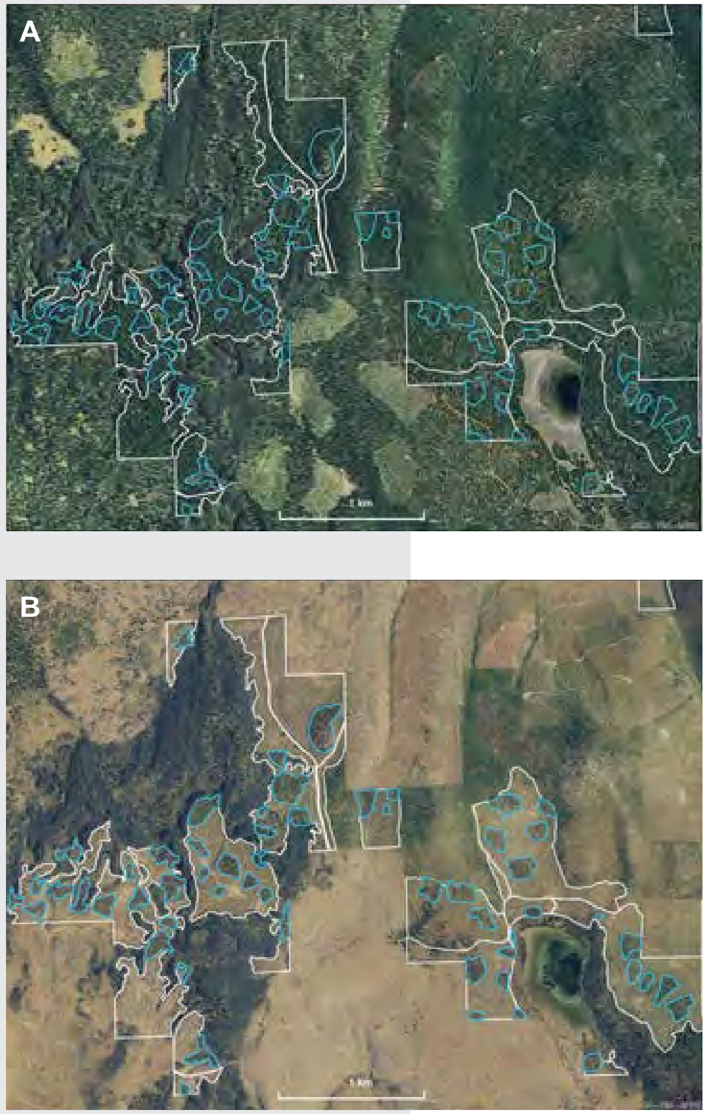

Figure 9—Aerial imagery showing unlogged snag retention “leave islands” (blue outlines) within larger salvaged units (white outlines) in the Eiler Fire area on the Lassen National Forest, (A) prefire and (B) postfire salvage. Maps by Michelle Coppoletta.

Intentional Use of Fire

A postfire restoration framework can also include opportunities to use prescribed burns, cultural burns, and wildfires managed to achieve resource objectives. Outside of postfire contexts, prescribed understory burning is often considered to reduce surface and ladder fuels and restore more natural fire return intervals (fig. 10). However, interest has been growing in applying prescribed burning to postfire landscapes to meet restoration objectives. For example, North et al. (2019) suggested prescribed burning in low- and moderately burned areas; Hessburg et al. (2016) and White and Long (2019) suggested early prescribed burning in areas adjacent to severe burns. Many areas have become so overly dense with trees that they require harvest to reduce fuels before they can be safely and effectively burned. This issue may be especially important when areas burn at moderate severity, resulting in large amounts of dead but mostly smaller trees. Such burning may promote the resilience of both the remaining green forest and the likelihood that naturally regenerating or planted stands within high-severity burn patches may be afforded sufficient respite from wildfire to become resilient.

Postfire restoration using fire can promote the resilience of remaining green forest and regrowing stands in burned areas.

Some managers have also used prescribed burning to fell dead trees and reduce fuel loads within high-severity burn patches. For example, a prescribed burn was conducted in 2016 as part of a Klamath Training Exchange program within the footprint of the Orleans Burn (2013) to reduce fuels (fig. 11). The fire was carried by accumulated limb wood and invasive Himalayan blackberries, and it resulted

Figure 10—A prescribed burn in a previously unburned stand to reduce surface and ladder fuels in the Goosenest Adaptive Management Area on the Klamath National Forest. USDA Forest Service photo.

Figure 11—A 2016 Klamath Training Exchange prescribed burn within the footprint of the 2013 Orleans Complex fire on the Six Rivers National Forest. It was conducted to reduce fuels, including standing snags. Photo by Will Harling.

in patchy consumption of surface fuels and felling and partial consumption of 5 percent of the snags (Will Harling, Mid-Klamath Watershed Council, pers. communication). Prescribed burning within planted areas is another important tactic for promoting long-term resilience. However, this may require increasing tolerance for higher levels of tree mortality in planted stands (Bellows et al. 2016), appropriate timing to allow trees to develop some phenological resistance to fire, and thoughtfully designed prescriptions and firing patterns to minimize impacts to the regenerating stands (North et al. 2019). Another strategy is to design plantings to be surrounded by fuel breaks to facilitate future use of prescribed burns within the planted stands as well as to protect them from fires outside the planted areas. Nemens et al. (2018) suggested that several site preparation methods may be needed to facilitate use of prescribed fire in young stands, including interrupting shrub fuels using herbicide treatment, mastication, or manual removal, as well as seeding of native herbaceous species. Prescribed burning and cultural burning may also serve to create more heterogenous patches of shrublands of different ages.

Managing boundary areas to facilitate future use of fire—
Previous work has suggested the use of roads and ridgetops for fuel breaks that would help to contain fires (wild or prescribed) within smaller, more manageable units (North et al. 2014). This basic approach can extend to more areas, including meadows, riparian areas, aspen groves, oak groves, sparsely vegetated rock outcrops, and high-elevation areas to facilitate use of fire. Furthermore, fuel breaks may have been created or expanded during suppression efforts on a fire.

Figure 12—A boundary between two potential operational delineation units on the Mendocino National Forest follows a road near the ridgeline that separates a wildland-urban interface-dominated watershed from a wildland watershed. The boundary extends through a mixed-conifer, black oak grove interspersed with shrubfields. Photo by Jonathan Long.

It is prudent to maintain control lines/fuel breaks in suitable areas for the long term if they have strategic value and can be used as anchor points to facilitate future beneficial fires (North et al. 2021). These approaches should keep in mind the principle of maintaining habitat connectivity as discussed above under "Promote heterogeneity at landscape and stand scales," and therefore will demand careful consideration of how to integrate multiple ecological processes at landscape scales. Designating such control features can be supported through analyses or conceptual approaches, such as the potential operational delineations framework, to identify effective fire control areas (O'Connor et al. 2017). These boundary areas are critical features for addressing ecological and social considerations at landscape scales (fig. 12).

Planting Conifers

Postfire reforestation is particularly important in areas that have long been forested and have climate conditions that can sustain forests, but in which the size of stand-replacing patches of high-severity fire are so great that the probability of conversion to nonforest vegetation has substantially increased. Traditional reforestation approaches in such areas typically include salvage, site preparation (which includes removing slash and exposing mineral soil for ease of planting and to control competing vegetation for seedling growth and survival), planting, and followup treatments to maximize survival and growth of planted seedlings. In earlier decades, a typical objective of such interventions was to rapidly establish a well-stocked stand of trees (up to 300 to 600 trees per acre [121 to 243 trees per hectare]) that would dominate the site and outcompete vigorous shrubs, followed by treatments to gradually reduce tree stocking and remove undesirable phenotypes (Schubert and Adams 1975). Such intensive approaches were undermined by declining budgets for timber stand improvement programs that support followup treatments (North et al. 2019). Moreover, high-density, regularly spaced plantings are vulnerable to high-severity fire effects in subsequent reburns (Thompson et al. 2007, Zald and Dunn 2018). Consequently, forest managers and researchers have considered opportunities to reduce stocking rates, as well as to diversify seed stock and seedling species mixes with an eye toward future climate resilience

(North et al. 2019). A recent report provides guidance on replanting in California (Stewart 2020); that report notes that the California Board of Forestry recently adopted lower reforestation stocking standards on private and state lands.

Planting where conifer forest is the reference vegetation but seed sources are lacking—
Priority areas for interventions may occur where a shift away from conifer dominance (under reference conditions) seems likely due to uncharacteristic fire severity and where natural regeneration is expected to be intolerably delayed or permanently absent. Areas within historical conifer stands that have experienced stand-replacing fire and lie beyond a threshold distance from green trees may experience long delays in conifer recovery or persistent shifts to nonconifer vegetation. Because of this relationship, interiors of large patches beyond that threshold are often priorities for replanting. These spatial relationships may also be consistent with efforts that account for the fact that conifer trees may have encroached into many areas that were open under historical references; for example, in fire-excluded forests of the Klamath region, the area occupied by openings decreased from 25.8 to 15.6 percent over 41 years (Skinner 1995). From an ecological restoration perspective, prefire conditions may not be an appropriate reference because conditions may have departed as a result of historical fire exclusion and other human influences; reference targets may also factor in effects of climate change (Meyer et al. 2021).

Priority areas for interventions may occur where a shift away from conifer dominance (under reference conditions) seems likely due to uncharacteristic fire severity and where natural regeneration is expected to be intolerably delayed or permanently absent.

To inform decisions about replanting, several threshold values for distance to seed trees have been suggested for mixed-conifer forests in recent publications, including 60, 120, and 200 m. The 120-m range seems to have become a rule of thumb for managers, being cited in an example from the King Fire (North et al. 2019) and used in an analysis of the Rough Fire (Meyer et al. 2021). Other work supported the 120-m threshold as a basis for likely distance of seed dispersal based on the greatest height of mixed-conifer trees in the area (Clark et al. 1999, McDonald 1980). North et al. (2019) referenced a 200-m threshold as a high-end rule of thumb to capture any natural regeneration. That value was informed by research conducted by Greene and Johnson (1996), who suggested that 200 m was an upper limit for spruce species. Clark et al. (1999) presented figures suggesting that 100 m was a threshold for wind dispersal of seed in mixed-conifer forests. McDonald (1980) found that more than 90 percent of conifer seed fell within an area one and one-half times the height of the average dominant tree (200 ft [60 m]), which suggests a threshold above 90 m. McDonald (1980) added that "at least some seeds of ponderosa pine and white fir reach the center of 10-ac [113 m in diameter; circular] clearcuttings, however. Wind gusts, eddies, and possibly convectional lifting, aid in transporting them there." Furthermore, animal dispersal could also contribute to regeneration at greater distances (Vander Wall 1992), and a field study by Welch et al. (2016) found some regeneration in places where potential seed source distance exceeded 200 m.

Varying planting densities and arrangements—
Common practices following wildfires have been to plant individual trees with regular spacing to encourage vigorous tree growth (fig. 13), with plans to thin those stands in future decades. However, some recent reforestation efforts have used lower stocking rates (>100–300 trees per acre) and alternative planting arrangements that modify spacing and planting patterns. These efforts have been proposed to reduce the need for precommercial thinning or other followup treatments, while achieving desirable, heterogeneous stand structures that have been associated with resilience to disturbances (Kane et al. 2019). Although lower stocking densities reduce the need for precommercial thinning, there is also the argument that lower densities result in a longer period to the point of crown closure needed to suppress competing vegetation. Such delays may result in a need for more treatment to reduce competing vegetation in some areas or reduced tree growth or survival (Ritchie et al. 2019). On the other hand, in many dry forest stands, it may be challenging to promote relatively closed canopies while encouraging more open and less dense conditions associated with resilience (North et al. 2022).

Between 1993 and 2017, rates of release and precommercial thinning treatments to control tree density on U.S. national forest lands have steadily decreased largely because of staffing and budget decreases (North et al. 2019). As a result, some managers have shifted toward lower density planting to mitigate the need for reducing tree density within the first few decades of planting. North et al. (2019) also contended that replanting designs that achieved more clumped patterns would promote spatial heterogeneity and resilience to frequent fire and drought. Keys to moderating fire severity include reducing surface and ladder fuels as well

Figure 13—June 2011: Planted areas with both evenly and variable spaced trees, likely the result of natural regeneration (ponderosa pines and California black oaks in the background) following the 1987 Stanislaus Complex (now in the Rim Fire footprint) on the Stanislaus National Forest. Photo by Marc Meyer.

as increasing spatial heterogeneity in vegetation structure (Atchley et al. 2021, Koontz et al. 2020, Ritter et al. 2020). After the 2006 Boulder Fire on the Plumas National Forest, managers implemented a low-density, widely spaced cluster design to promote structural heterogeneity and meet aesthetic visual quality objectives within the recreation area. Postfire salvage operations emphasized reducing fuels and competing vegetation to promote conifer regrowth. This plantation was designed to mimic a naturally occurring, more varied pattern of clumped tree establishment that would improve visual depth into stands and increase horizontal structural heterogeneity. This approach also allowed growing space for any natural regeneration that could occur and augment the development of vertical structural heterogeneity within the reforested stands.

Given that stand densities in Western dry forests are often too high (Hagmann et al. 2021), there are concerns that stands planted at high densities will far exceed reference conditions without high natural mortality or thinning efforts. However, debate continues about whether to try to realign stand diversity during the stand initiation phase, or through subsequent treatments. North et al. (2019) suggest that lower tree densities would reduce moisture stress and may also be more appropriate for areas that are expected to experience greater use of fire, including wildfire protection zones and other strategic fuel breaks, including ridges. A recent study found that clumped spatial patterns of natural tree recruitment may favor the establishment and early growth of regenerating conifers in active-fire forests (Fertel et al. 2022). The study noted that negative effects of competition on tree growth within clumps may not be evident until trees are older. Because these findings challenge conventional reliance on widely spaced plantings, and because of complex interactions with fire, longer-term studies are needed to inform management strategies.

After the 2014 King Fire, managers set targets for spacing between tree clusters, desired density per cluster, and species composition based on stand reconstruction studies for yellow pine and mixed-conifer forests and evaluations to identify where reforestation efforts were likely to be most successful under current and future climate predictions. These targets were further stratified based on slope and proximity to the WUI as discussed below under the landscape strategy. The goal was for the resulting tree clusters to align with more favorable microsites. However, initial planting densities were developed based on assumptions that initial mortality rates would be lower than 10 percent when combined with proposed release treatments; that assumption was based on monitoring in the 2001 Star Fire on the Eldorado National Forest. However, initial survival was lower than expected, with the highest mortality rates observed in species other than ponderosa pine. Several unexpected implementation factors may have contributed to lower initial survival, including the loss of a highly trained workforce and lack of effective control of competing vegetation, in addition to the novel conditions created by the extremely large, high-severity burn patch resulting from this fire. Consequently, the initial planting densities and designs needed to achieve future desired densities and patterns for established seedlings are still being refined on an annual basis. This outcome also warrants further evaluation of site suitability for reforestation of conifers.

Varying spacing to emulate natural ecological patterns may prove to be a useful tactic, but it can be hard to achieve fine-scale variation during implementation, which is commonly achieved on public lands through reforestation contracts. Planting crews are often incentivized for productivity rather than placement, and descriptions of targeted microsites can be difficult to translate into standardized contractual language. However, many contracts do include specifications to space surviving mature trees and oaks, and they may vary spacing based on unit conditions. In the end, variability inevitably results during planting as a result of natural variation in topography, soils, residual slash, and surviving trees. Furthermore, variation results from vagaries associated with how the planting crews operate, the time of day, and whether there is a shortage or surplus of planting stock at the time an area is planted. Although varying spacing and arrangement adds complexity to contract administration, these types of plantations have been demonstrated in production field settings in recent years.

In historical and contemporary reference sites, conifer densities are often inversely related to slope, which may function as a proxy for site potential. Drier, steep ridges on south-facing aspects, often support more shrubs and hardwoods (Bohlman et al. 2021, North et al. 2019, Taylor and Skinner 2003), a topographic pattern that is likely to be reinforced by climate change. Previous synthesis work suggested reducing fuels preferentially on drier, southern and western slopes and areas with shallower, less developed and less productive soils, including ridges (Long et al. 2014a, Moghaddas and Hubbert 2014). Meanwhile, higher tree densities and associated fuels would be expected in more mesic, north-facing slopes and canyon bottoms as well as areas with deeper, more productive soils (North et al. 2014). Areas in upper slope positions have also been shown to burn at higher severities (Estes et al. 2017) and therefore appear less suited for reestablishing high tree densities, even if they have potential for growing such dense stands.

Decisions about planting need to consider where planting is likely to fail and where it is likely to be unnecessary. In a study of natural regeneration in burned areas, Welch et al. (2016) found that plots lacking live-tree basal area in cool aspects below about 40 percent slope were likely to experience successful natural regeneration but that areas in warm aspects or on steeper slopes would likely not. Similarly, findings from a recent study in the Klamath region (Lopez Ortiz et al. 2019) suggested that postfire planting was unnecessary on north-facing slopes where natural regeneration appeared abundant; however, on south-facing aspects, planting aided regeneration of ponderosa pine. These relationships might be informed by climatic water deficit (annual evaporative demand that exceeds available water); one study found that sites in the Klamath Mountains were susceptible to poor recruitment when climatic water deficit values exceeded 300 mm (Tepley et al. 2017).

Considering accessibility for treatment—

Targeting interventions in accessible areas (based on road networks) may reduce costs and increase opportunities for maintenance. North et al. (2019) advised that less-accessible areas could be targeted for treatments to encourage "founder stands," with the recognition that they might not get frequent followup treatments and may be impractical to protect during wildfires. Areas near WUIs and control line features slated for managing fires (fig. 12) might be planted at lower densities and maintained more frequently to keep vegetative fuels from accumulating.

Deferring areas from planting—

Though much attention has been focused on postfire reforestation and where and how to plant, managers also decide where not to plant. North et al. (2019) highlighted examples, including steep, south-facing slopes; areas with shallow soils; and areas on ecotonal edges. Field-based decisions to defer planting often include an intersection of ecology, economics, and organizational capacity. Tools have been developed to predict where natural regeneration is not likely to be sufficient to meet management targets (Meyer et al. 2021, Shive et al. 2018) and the additive effects of planting (Young et al. 2021). In addition to considering whether to plant seedling stock adapted to drier environments, managers may defer replanting in sites that are marginal because of droughty conditions (North et al. 2019). Riparian areas; meadow areas; and areas adapted to shrubs and hardwoods, including some steeper sloped areas, may also be common locations to defer conifer plantings. For example, conifer planting was deferred in part of the Angora Fire footprint on the upper slopes of Angora Ridge because the historical reference conditions were deemed to have been dominated by shrubs; aspen were also planted in riparian areas and edges of meadows within some parts of the burned area (USDA FS 2010). These issues are discussed in more detail in the sections below on ridges, riparian areas, meadows, and hardwoods.

Management of Vegetation Competing With Conifers

Shrubs and resprouting hardwoods often compete with conifers, both naturally regenerating and planted trees, in postfire environments. Research has indicated that postfire interventions can limit the predominance of nonconifer vegetation types and promote overall native plant diversity (Bohlman et al. 2016). Bohlman et al. (2016) and DiTomaso et al. (1997) found that species richness and diversity were greater in herbicide-treated areas, which appeared to increase native forbs and grasses by reducing the dominance of native shrubs (*Ceanothus* spp. and *Arctostaphylos* spp.).

Interventions that include shrub control can include manual (e.g., hand grubbing), chemical (e.g., herbicides), and mechanical (e.g., mastication) treatments.

Interventions that control shrub communities have been demonstrated to accelerate growth of young ponderosa pine trees (McDonald and Fiddler 2010; Ritchie et al. 2019; Zhang et al. 2006, 2008). Over 25 years, McDonald and Fiddler (2010) noted that ponderosa pines were nearly 25 ft (7.6 m) tall in areas without shrub competition, compared to less than 10 ft (3 m) tall in plantations with heavy brush. Reductions in competing vegetation improve tree height growth, and correspondingly, radial growth, which accelerates development of mid-seral stands and fire resistance.

Although initial control of competing vegetation and follow-up release treatments were a fairly standard practice on both public and private lands in California for several decades, such treatments have reportedly declined on national forest lands in recent decades, for example, by 70 percent from 1998–2007 to 2008–2017 (North et al. 2019). Stephens et al. (2020) noted that plantations on national forest lands in the Moonlight Fire were dominated by shrubs because competing vegetation control was lacking; in contrast, on private timberlands, use of intensive chemical control of competing vegetation resulted in a faster transition to young-forest conditions.

Managers may consider managing competing vegetation to improve the resistance of planted stands to fires by promoting tree growth and altering understory and ladder fuels. Mastication of shrubs can increase the separation between understory fuels and tree canopies, which can reduce the probability of transition from surface to crown fire); however, mastication may also increase the amount of surface fuels (Reiner et al. 2009). Machine pulling, piling, and burning of shrubs can also be used to effectively reduce shrub cover (Moore et al. 2021), though mechanized pulling may be feasible only in older shrub components that have enough structure to be effectively pulled.

To achieve goals for moderating future fire severity and overall resilience, nonconifer vegetation and fire-based treatments need to be considered.

As noted throughout this report, nonconifer vegetation needs to be considered not only because of its competitive influence on conifers but also in terms of its value as alternative vegetation types and its role as fuel in fires. To achieve goals for moderating future fire severity and overall resilience, nonconifer vegetation and fire-based treatments need to be considered (Dobre et al. 2022; North et al. 2021, 2022).

Applying Approaches to an Archetypal Landscape

This section illustrates with examples how to craft a strategy to achieve restoration goals across an archetypal burned landscape using the framework described in figure 2. By customizing the application of principles based on local factors and social considerations, one can generate a portfolio of interventions to support restoration and adaptation objectives (Meyer et al. 2021, Swanston et al. 2020). We consider an archetypal landscape that is predominantly low- to mid-elevation, mixed-conifer forest, one of the most common forest types in the mixed-forest ecoregion province M261 (fig. 1). Common tree species include ponderosa pine, Jeffrey pine, sugar pine, white fir, Douglas-fir, incense cedar, and California black oak. This forest type was historically characterized by a frequent (>20 years), low-severity fire regime prior to colonization by Euro-Americans (Metlen et al. 2018, Safford and Stevens 2017). Dry, mixed-conifer forests in the Klamath ecoregion are usually characterized as having a frequent, low-severity regime, although mixed-severity fires were not uncommon, reflecting diverse topography, climate, and substrate (Halofsky et al. 2011). This interplay resulted in many transition zones between vegetation communities and allowed for the fire regime of one type to influence adjacent types. Because of historical fire exclusion and accumulation of fuels, these areas are among the most vulnerable to uncharacteristically severe wildfire outcomes and type conversions.

Table 1 provides an example of the different zones that managers may encounter within an archetypal mixed-conifer landscape as well as some of the considerations and objectives that often help to inform development of a restoration strategy (i.e., a restoration portfolio) in a postfire landscape. We begin by considering variation in fire severity within these upland forest types from large, high-severity burn patches (zone 1) to small, high-severity burn patches (zone 2), to moderate-severity burn patches (zone 3), and then low-severity burn patches (zone 4), and then consider areas with distinctive topography and vegetation such as ridges (zone 5), hardwood groves (zone 6), riparian areas (zone 7), and meadows (zone 8). Within zone 9, we consider higher elevation stands of upper montane forests, dominated by red fir (*Abies magnifica*), lodgepole pine (*Pinus contorta*), mountain hemlock (*Tsuga mertensiana*), and western white pine (*Pinus monticola).* Lastly, we consider lower elevation pine-oak forests (zone 10) that may be particularly vulnerable to type conversion. Note that in addition to these ecologically defined zones, there may be zones defined by social importance, including roads (box 3), campgrounds, and areas with structures or that are adjacent to human communities. We highlight real-world examples of interventions that managers have considered as part of a postfire management strategy. Because actual interventions need to be considered within a broader social and ecological context, these are presented as examples to illustrate how the framework can be applied rather than an exhaustive list of restoration actions.

King Fire Illustration

To illustrate the archetypal landscape, we feature an example from the King Fire (2014) on the Eldorado National Forest (figs. 15 and 16; table 1), which included one of the largest recorded high-severity burn patches up to that point (Coen et al. 2017, Meyer et al. 2021) (larger patches have since occurred in recent wildfires). We enrich this discussion with examples from other fires that fit the archetypal profile, including the Moonlight Fire (2007) on the Plumas National Forest and the Ranch Fire (2018) on the Mendocino National Forest, to which the postfire restoration framework (Meyer et al. 2021) was applied as an initial demonstration.

Box 3: Roads, Trails, and Postfire Interventions

Roads and trails are an important consideration in postfire restoration planning for multiple reasons. First, they are important from the perspective of safety and accessibility for transportation, recreation, public access, access by tribal nations, and fire management (figs. 12 and 14). Roads are particularly important for developing landscape restoration strategies that incorporate use of fire, as recent research has shown that the vast majority (82 percent) of units identified for managing fires align with roads (Thompson et al. 2021b). Roadside contexts will vary greatly across the landscape with topography, vegetation, and burn severity, although roads along ridgelines are particularly conducive to fire management. Both roads and trails may have been established on top of ancestral travel corridors used by American Indians, and those areas may continue to be important subjects of cultural concern. Some of these considerations may help advance broader goals for ecological restoration, although the roadside environment is a novel condition where safety concerns predominate.

Figure 14—Roads, such as this one in the 2018 Carr Fire footprint on the Shasta-Trinity National Forest, are important considerations due to public safety, access for treatments and fire control, and public access. USDA Forest Service photo.

Table 1—Different archetypal zones and associated objectives and interventions to consider as part of a restoration portfolio

No.	Archetypal zone	Burn severity	Note on landscape context	Historical reference composition	Objectives	Potential intervention approach following the postfire flow chart[a]
1	Large, high-severity burn patch	High	Intervention may be most important in areas of uniform topography where seed sources are distant.	Predominantly coniferous	Reduce risk of high-severity reburn, reestablish forest, and accelerate growth of tall forest stands.	II: Consider harvest of dead trees and replanting at appropriate densities where seed source is lacking; manage competing vegetation to promote tree survival and growth; plan for managed fire once trees become established.
2	Small, high-severity burn patch	High	Intervention may be less important in areas where topographic variation reinforces heterogeneous burn patterns.	Predominantly coniferous	Promote heterogeneity and support early successional species.	I. Consider leaving snag patches distributed across the landscape.
3	Moderate-severity burn patch	Mix of low to high	A complex category as conditions could be relatively restored or degraded.	Predominantly coniferous	Reduce risk of high-severity reburn and facilitate regeneration where needed and accessible.	I and II. Consider fuel reduction harvest where needed to prepare for return of fire.
4	Low-severity burn patch within fires and unburned margins	Low to unburned	Areas adjacent to large, high severity burn patches are important to constrain future growth of those patches.	Predominantly coniferous	Reestablish desired stand densities and maintain conditions to limit expansion of high-severity burn areas in future fires.	I and II. Consider further thinning (including green trees) toward desired densities and fuel conditions where needed; use managed fire early.
5	Ridges	Variable	Ridgelines, especially those with road access, can serve as fuel breaks.	Mix of hardwoods, shrubs, and lower density of conifers	Maintain as fuel breaks to reduce risk of high-severity reburn and facilitate future use of managed fire.	II. Consider promoting hardwoods and perhaps planting some conifers at low densities as seed trees, combined with salvage/fuel reduction to reduce reburn risk; use prescribed fire to maintain fuel breaks.
6	Hardwood grove (e.g., black oak and aspen)	Variable	Distinctive edaphic conditions may support hardwoods on sloped or flat areas.	Hardwood dominated	Sustain historical groves that provide diversity of ecosystem services and fuel conditions.	I or II. Encourage low-intensity fire and intervene to limit conifer encroachment, while also tolerating mortality in small patches to encourage younger hardwoods; manage animals to limit impacts of grazing/browsing on hardwood regrowth.

Table 1 (continued)—Different archetypal zones and associated objectives and interventions to consider as part of a restoration portfolio

No.	Archetypal zone	Burn severity	Note on landscape context	Historical reference composition	Objectives	Potential intervention approach following the postfire flow chart[a]
7	Riparian area	Variable	Flatter bottomlands and moist edaphic conditions along streams may support distinctive fire and vegetation patterns and successional processes.	Deciduous trees and shrubs, herbaceous species, some conifers in places	Facilitate recovery of deciduous woody and herbaceous plant species, maintain snags as future wildlife habitat and sources of coarse wood in appropriate amounts.	I or II. Consider generally avoiding interventions (other than continued use of fire) except perhaps limited conifer planting (e.g., on drier benches) and removing or mulching smaller dead trees where their accumulation or risk of debris jams exceeds desirable conditions.
8	Meadow	Variable	Topographic depressions and distinctive hydrogeomorphic conditions.	Herbaceous plants, deciduous trees and shrubs, and few conifers	Facilitate restoration of meadow conditions.	I. Consider encouraging fire, avoiding disruption of recovering vegetation, reducing encroaching conifers, and hydrogeomorphic restoration of degraded meadows to facilitate ecological recovery and sediment retention.
9	Higher elevation forest at upper end of burn area	Variable	Transitions to upper montane and subalpine areas may be interspersed with rocky openings that can break up fire patterns.	Red fir and other upper montane species with subalpine species possible at the highest elevations	Maintain transition zones as fire control areas, especially through use of managed fire.	I. Consider avoiding interventions because burn outcomes tend to be within natural range of variability; however, consider interventions in uncharacteristically large patches and to promote resistant tree composition.
10	Pine-oak forest (or other "trailing edge" forest types)	High	Drier conditions on steeper slopes and in lower elevation areas.	Mix of pines, oaks, and shrubs	Allow for shifts in vegetation communities to support heterogeneity and avoid investment losses.	III. Consider deferral of reforestation where climate conditions suggest that pine are not likely to survive; except consider replanting at low densities to restore important connections between habitats.

[a] Roman numerals refer to the three different restoration opportunities: I. Maintain/promote desired conditions; II: Take management actions to restore desired conditions; III: Reevaluate desired conditions (see fig. 2).

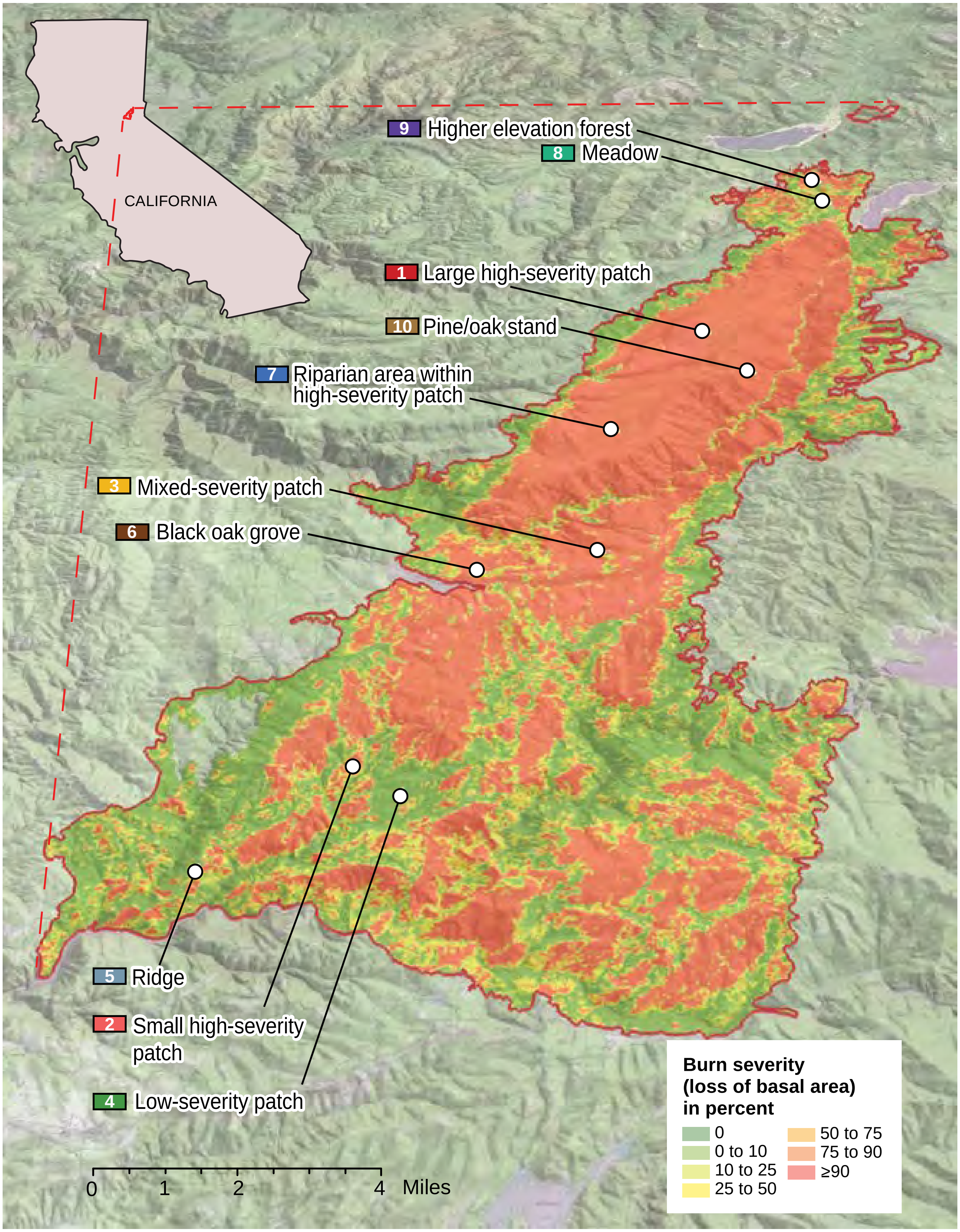

Figure 15—Three-dimensional view of the King Fire (2014) on the Eldorado National Forest overlaid with burn severity (defined by basal area loss), showing examples of archetypal zones where interventions may be considered as part of a postfire restoration strategy. Map by Jonathan Long and Steve Oerding.

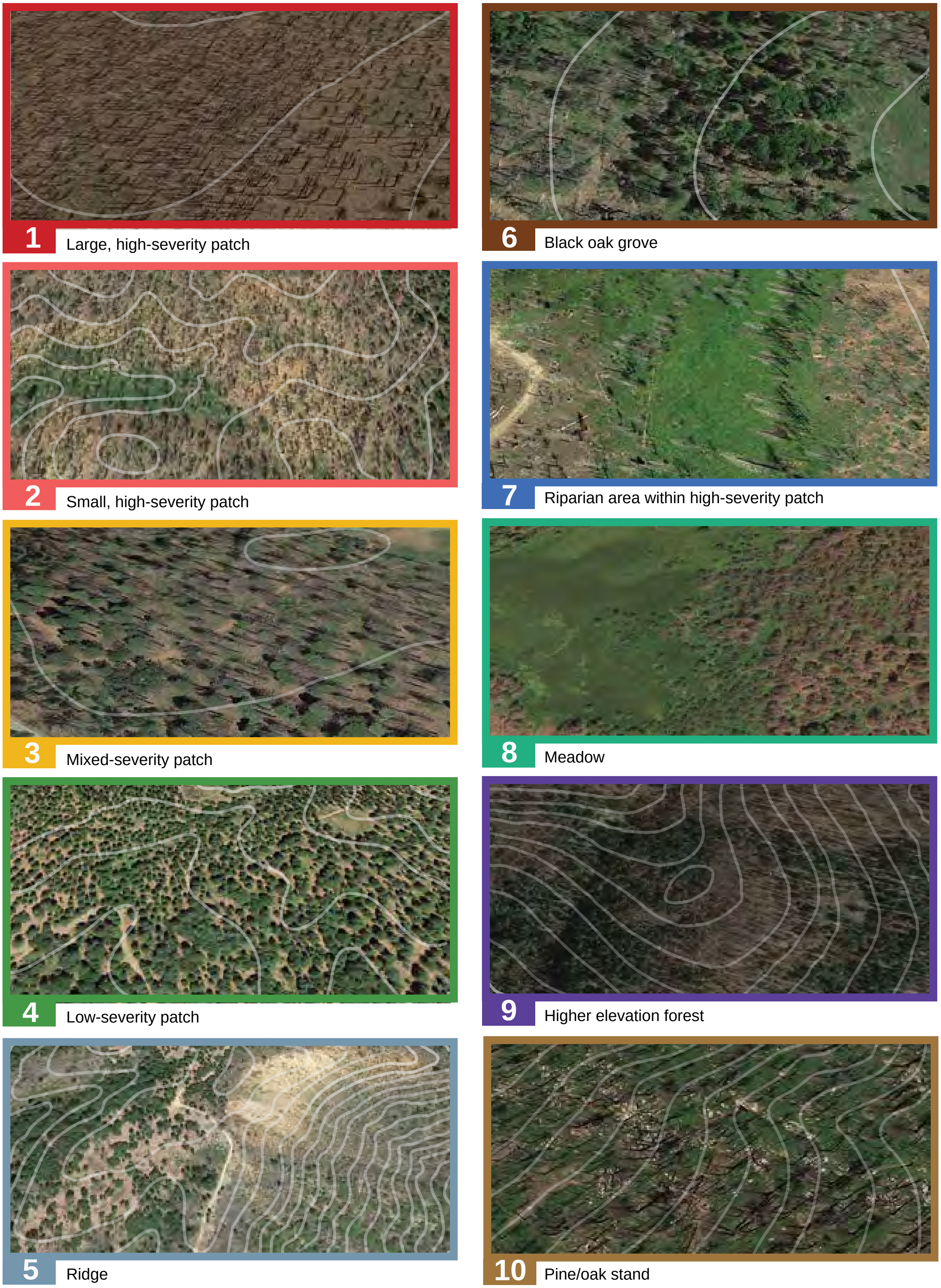

Figure 16—Aerial images overlaid with contour lines (light brown) showing the 10 archetypal zones where interventions may be considered as part of a postfire restoration strategy. Figure by Jonathan Long and Steve Oerding.

For the King Fire, we share examples of how analyses of burn severity (fig. 17) informed postfire restoration plans (fig. 18), although the actual implementation has demonstrated an adaptive process. For example, initial fuel treatments were planned for the entire fire (fig. 18), including areas burned at lower severities. Analysis for prescribed burning as a maintenance treatment was completed for

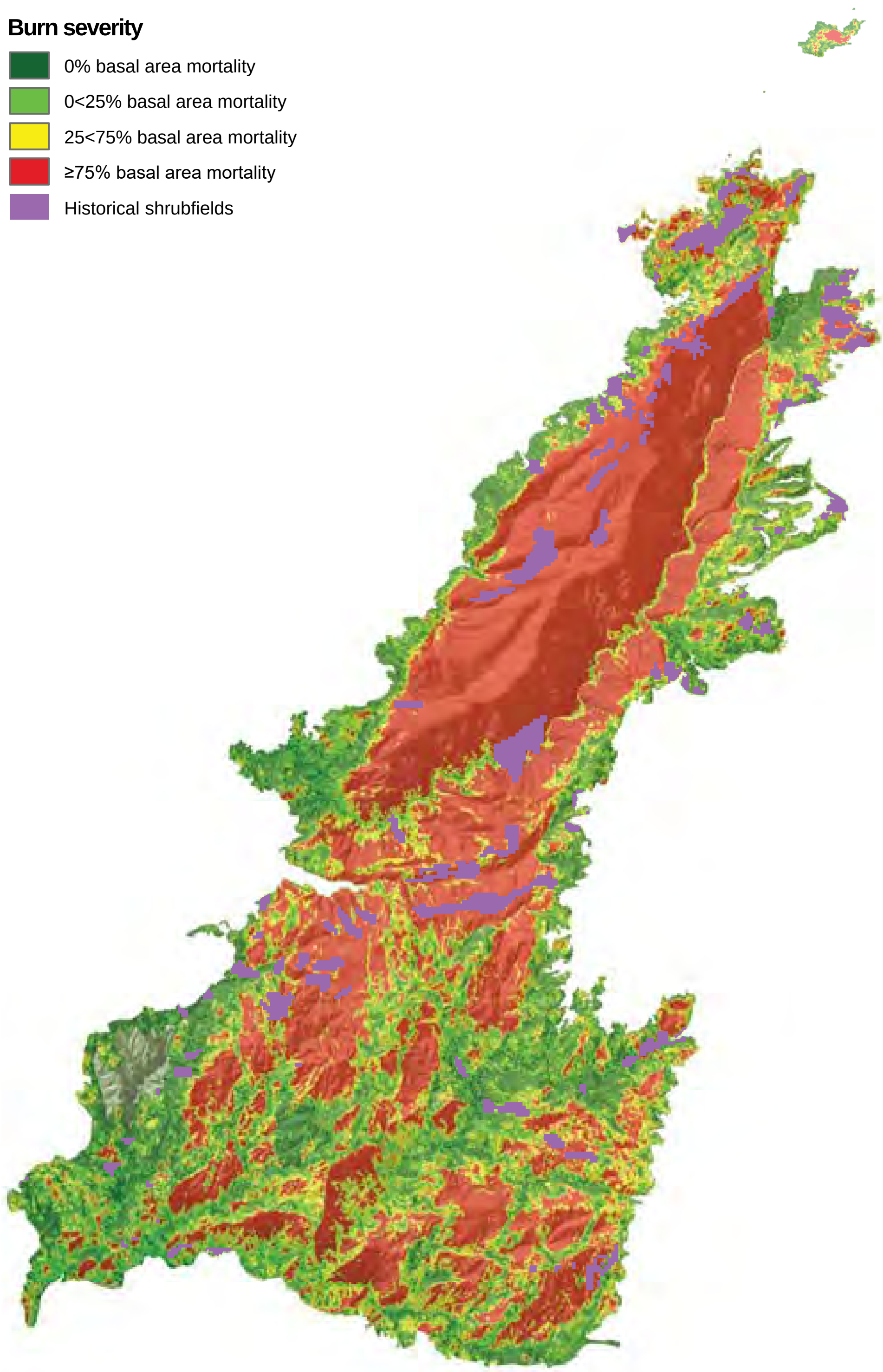

Figure 17—Different levels of fire-related tree mortality, along with historical shrub fields (purple), in the 2014 King Fire footprint on the Eldorado National Forest. Map by Dana Walsh.

most of the reforestation treatment area in the northern portion of the fire, and managers have planned for additional fuels reduction and reforestation within and adjacent to the southern portion of the fire. However, implementation of prescribed burning is often challenging, as will be discussed in later sections.

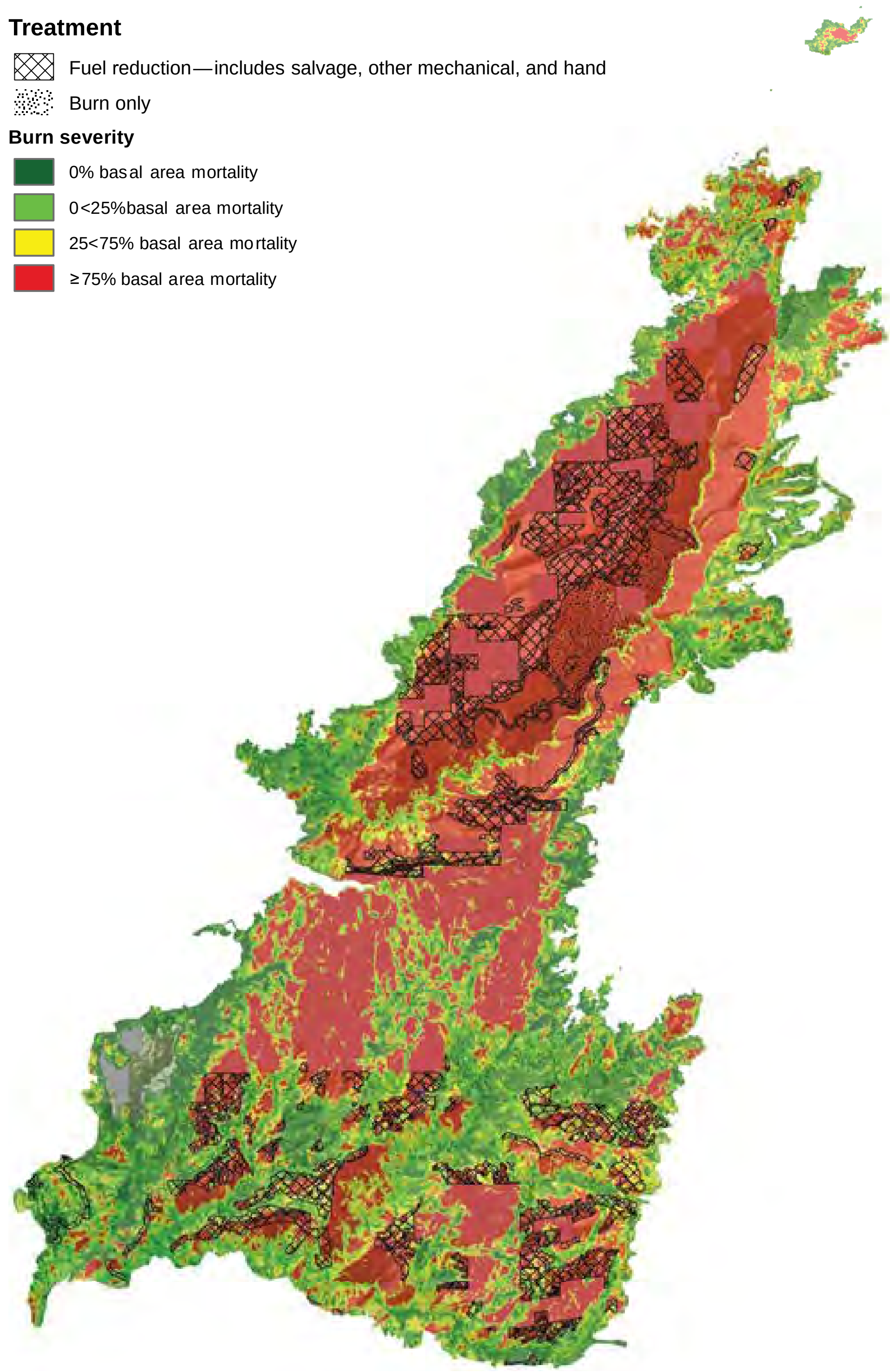

Figure 18—Example of postfire fuels reduction strategy for the 2014 King Fire on the Eldorado National Forest based on burn severity, ownership, slope, and patch configurations. Map by Dana Walsh.

Considerations for Specific Management Areas with Distinctive Vegetation

Before examining the 10 zones that are likely to be important considerations across the region, we also consider inclusions of distinctive vegetation with special relationships to fire as well as areas that have special management designations based on distinctive ecological features (e.g., research natural areas) that may limit some kinds of interventions. Special land management areas are designated on all national forests in California. Land managers need to consider how to promote specific desired conditions and management direction associated with these areas when they burn in large and severe wildfires. Some of these unique areas include national monuments, research natural areas, special interest and management areas, wildlife areas (including wildlife habitat management areas), and experimental forests and ranges. In many cases, these special areas are defined by or encompass a significant portion of unique vegetation or habitat types that are uncommon or rare on the landscape and may possess unique qualities or features for the target species. For example, research natural areas often contain unique vegetation types that have specialized adaptations to fire, such as the Mud Lake Research Natural Area on the Plumas National Forest (box 4) and the Agua Tibia Research Natural Area on the Cleveland National Forest, in which the target vegetation is bigcone Douglas-fir (*Pseudotsuga macrocarpa*), a species that exhibits postfire resprouting. The Giant Sequoia National Monument is another example of a specific land management area that was created to protect specific “objects of interest” on the Sequoia National Forest, most prominently, giant sequoia groves. Giant sequoias are another serotinous conifer that requires moderate- to high-severity fire effects in small patches to facilitate seed dispersal and seedling recruitment (Meyer and Safford 2011); but on the whole, giant sequoia ecosystems are characterized by a frequent, low-severity natural fire regime that maintains the health and resilience of developing, mature, and old giant sequoias (Stephenson 1998). Many research natural areas, national monuments, and other specific management areas or designated areas in California are affected by altered fire regimes (e.g., fires burning too frequently or infrequently compared to historical reference condition) (Coppoletta et al. 2019). Consequently, many of these areas may require more detailed postfire (and prefire) management considerations that account for the area-specific management direction (e.g., forest plan or monument plan direction), unique species adaptations (e.g., serotiny), and site-specific ecological conditions (e.g., low- to moderate-severity fire effects are currently lacking, or the risk of loss to future stand-replacing fire is high).

Zone 1: Uncharacteristically Large, High-Severity Burn Patches

Previous synthesis research has explained why postfire interventions are likely to be ecologically most important in uncharacteristically large, high-severity burn patches (Long et al. 2014b, Meyer et al. 2021, Spies et al. 2018). A common restoration objective after wildfire is to facilitate recovery of conifer vegetation and associated fauna. Harvest of fire-killed trees and replanting could help reduce reburn risk and accelerate growth of tall forest stands, which can help sustain habitat for old-forest-associated wildlife over the long term.

Not all high-severity burn patches are ecologically appropriate to target for replanting with conifer trees. Soil ecological site guides in addition to historical data are important to inform such evaluations (Meyer et al. 2021). It is important to consider whether these areas have long been dominated by conifers or whether they were formerly occupied by hardwoods, shrubs, or grassy vegetation (White and Long 2019). As noted previously, fire exclusion has allowed conifers to encroach into many of those forest openings or alternative vegetative communities in the past century (Airey-Lauvaux et al. 2016, Jones et al. 2005, Skinner 1995). Replanting can reduce the extent of those nonconifer patches, which are an important source of biological diversity, fuel heterogeneity, and other ecosystem services. It is important to consider topography given its relationship to soils, productivity, and fire behavior. Wildfire severity and sapling mortality from wildfires are typically higher on steep, south-facing slopes and above middle slope positions, so planted stands are more likely to succeed in lower slopes and benches in mesic, productive areas, including north-facing aspects (Jain et al. 2021, Lesmeister et al. 2018, Lydersen and North 2012). In the Klamath Mountains, Estes et al. (2017) found higher burn severities within upper and middle slopes than in lower slopes and within east- and southeast-facing aspect burns than on other aspects. In Lassen National Park in the southern Cascades ecoregion, fire severity increased as fuels accumulate on ridgelines and steep slope (Airey-Lauvaux et al. 2022, Estes et al. 2017). Consequently, on steeper and upper slopes, replanting might be deferred altogether (category I), as in the Angora Fire (USDA FS 2010). Alternatively, planting strategies in such areas may be adjusted to rely on more drought- and fire-tolerant trees (Jain et al. 2021) or reducing planting densities (North et al. 2019, USDA FS 2019). As discussed in the concluding section of this report, evaluating and refining such strategies will be a key research need given the increasing impacts of wildfires and drought.

Size and arrangement of patches also need to be considered. The need for interventions is generally much greater in large, high-severity burn patches because the interior of such openings are much larger than what seed from adjacent trees can reach, reducing the likelihood of natural regeneration (Bohlman and Safford 2014, Bonnet et al. 2005, Greene and Johnson 2000). Suggesting a potential rule of thumb, Stevens et al. (2021) used a threshold of 100 ha to distinguish “large” from “small” treeless patches. Although many wildlife species use high-severity burns, especially for foraging, uncharacteristically large patches may be used less frequently. For example, Eyes et al. (2017) and Jones et al. (2020) found that spotted owls avoided the interiors of severely burned patches, especially areas more than 100 m from the edge, whereas Stillman et al. (2019) noted a similar relationship

Box 4: Managing Baker Cypress in the Moonlight Fire Footprint

Baker cypress (*Hesperocyparis bakeri*) is a rare, fire-dependent conifer that is currently limited to 11 disjunct populations in northern California and southern Oregon. This species is adapted to infrequent (i.e., >30-year intervals), stand-replacing fire, which facilitates seed dispersal by opening the serotinous cones and creates suitable conditions for seed germination and establishment. Long periods of fire exclusion in Baker cypress stands can increase the dominance of associated shade-tolerant conifers, significantly reducing cypress regeneration and increasing mortality (Rentz and Merriam 2009). In contrast, repeat, short-interval fires can also increase the risk of localized extirpation if these obligate seeding species burn prior to reaching cone-bearing age.

In 2007, the Moonlight Fire burned the northern portion of a Baker cypress stand in the Mud Lake Research Natural Area on the Plumas National Forest. Prior to the fire, this stand was characterized by high densities of white fir (70 percent of prefire stand density), very little cypress regeneration, and more than 87 percent of cypress were dead or dying (Merriam and Rentz 2010). However, postfire monitoring documented substantial cypress regeneration in response to the Moonlight Fire (e.g., up to 85 individuals per square meter in some plots), with the highest

Figure 19—Postfire fuel loads in the regenerating Baker cypress stand increased the risk of local extirpation when the Dixie Fire reburned the Mud Lake Research Natural Area before trees had matured and produced cones. Photo by Kyle Merriam.

seedling densities recorded in plots that experienced high-severity fire effects (Merriam and Rentz 2010).

Following the Moonlight Fire, managers were concerned that another fire in the developing stand could extirpate the Baker cypress population if it were to burn before the cypress reached maturity. The high density of white fir in the cypress stands prior to the fire were represented by high densities of snags throughout most of the stand, creating a potential fire hazard that would likely increase over time as snags fell and decayed (Coppoletta et al. 2016) (fig. 19). To address this concern, forest managers implemented a 235-ac (95-ha) fuel reduction project around the developing stand, with the intent of excluding fire from the immature Baker cypress cohort in the short term (USDA FS 2015c). Treatments, which included grapple piling and burning of small snags (fig. 20) were designed to break up fuel continuity and promote stand heterogeneity, while also creating a strategic fuel break that could be used to reintroduce high-intensity fire into the stand in the future when conditions were suitable. Unfortunately, severe fire weather and competing fire suppression priorities resulted in the 2021 Dixie Fire reburning the regenerating Baker cypress stand at high severity. With the short interval between the two fires, the several million seedlings that had established after the 2007 Moonlight Fire had not reached cone-bearing age. As a result, this population has been almost completely extirpated from the research natural area.

Figure 20—Grapple piling of ground fuels in the Mud Lake Research Natural Area after the Moonlight Fire was designed to reduce fire risk within the developing stand of Baker cypress. Photo by Kyle Merriam.

for black-backed woodpeckers for interiors more than 500 m from the edge. By targeting the interiors of large patches of severely burned, historically conifer-dominated forest, the total area prioritized for interventions through planting could be reduced. For example, in the 2015 Rough Fire, although 20 percent of the total area burned at high severity, only 3 percent was in a high-severity burn patch that was formerly dominated by conifer forest and more than 120 m from an unburned forest edge (Meyer et al. 2021). However, it is important to recognize that potential for natural regeneration does not guarantee successful reestablishment of species composition or successful reestablishment of desired conditions within a specific timeframe, and thus, additional interventions may still be warranted and needed.

Interventions in these areas are likely to involve multiple tactics, including fuels reduction, replanting, control of competing vegetation, and use of managed fire in and around planted trees when they are sufficiently resistant (table 1). In addition to the potential to offset costs of restoration treatments, salvage may be important for ecological reasons, including reducing fuels and creating safer conditions for crews to manage planted stands (Spies et al. 2018) (box 5). Provisions may be made for early prescribed burning once the young trees become established (North et al. 2019). Managers will need to weigh how much to buffer those stands to promote survivorship in fires or to connect them to green forests to promote habitat connectivity; thinning and burning the connectors might help to achieve multiple objectives. Managers have also occasionally proposed or implemented prescribed burns as a tool to reduce fuels in high-severity burn patches (fig. 11), yet such treatments can be challenging to undertake (box 6).

Box 5: Treating a Large High-Severity Burn Patch in the King Fire Footprint

Within the King Fire footprint, areas of high severity (represented by basal mortality ≥90 percent) were prioritized for salvage and planting efforts, with a particular emphasis on the large, central, high-severity burn patch (fig. 17). That 30,000-ac (12 140 ha) patch of lost conifer forest was one of the largest observed at the time (Meyer et al. 2021) (fig. 17). Postfire interventions included salvage of dead trees; falling and piling or mastication of dead trees and surface fuels (fig. 21); management of competing vegetation using manual methods, such as scalping during planting as well as herbicides applied to resprouting shrubs to maintain shrub cover to less than 30 percent and to grasses within 5 ft of the seedlings; natural regeneration of resprouting and seeding tree species; and planting of conifer seedings. Snag patches and early-seral shrub vegetation were retained in dispersed patches throughout the treated areas, representing a minimum of 10 percent of the area within harvestable units. These patches ranged from ¼ ac to more than 5 ac and were centered on prefire snags and pockets of the largest snags within the treatment units. In addition to those distributed snag retention areas, areas excluded from treatment included stream zones, archaeological sites, and inaccessible areas, resulting in a patchy treatment pattern (fig. 22). This approach was designed both to meet the forest plan requirements for maintaining the largest snags and to provide heterogeneity through the units. Managers supported this approach of maintaining the snags in distributed pockets rather than more evenly across the units, which was more efficient to implement and provided for safety of planting crews. These treated areas have been proposed for prescribed fire as a future treatment to manage accumulating fuels.

Figure 21—Treatment in a large, high-severity burn patch (zone 1) in the 2014 King Fire on the Eldorado National Forest involved removing some dead trees and piling the remainder for burning. Photo by Dana Walsh.

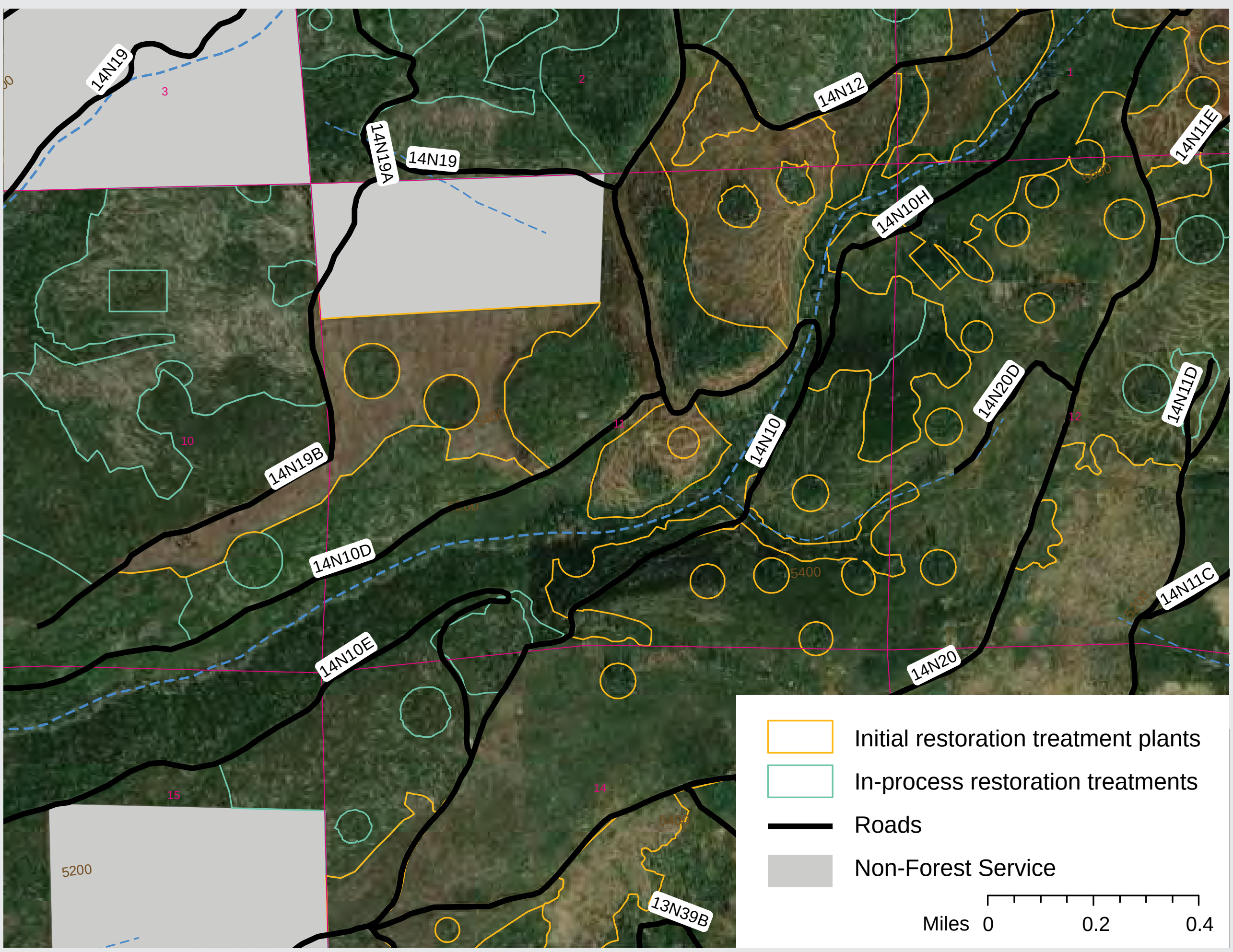

Figure 22— Example of salvage treatments in an area burned at high-severity in the King Fire area, showing exclusion areas, including drainages (blue dashed lines), inaccessible areas, and circular snag retention patches. Map by Dana Walsh.

Box 6: Treating Older, Large, High-Severity Burn Patches in the Moonlight Fire Footprint

As noted in the previous example, organizations face significant challenges in treating large, high-severity burn patches. In the 2007 Moonlight Fire, large areas that were initially planted but not salvaged (because of time constraints and an overwhelming area needing treatment) did not have followup release treatments because deteriorating snags posed overhead hazards to workers. Nearly a decade after the postfire planting, these sites became dominated by 1-m-high shrubs growing in an intermix of jack-strawed, down woody material and standing snags (fig. 23); these conditions could facilitate reburn at high severity and burn up the investment in replanting. Treatments were then designed to reduce overhead snag hazards by reducing accumulations of small standing snags, reducing cover of shrubs, and reducing fuels by piling dead wood for burning. Silvicultural prescriptions included retaining patches of shrub and hardwood cover, especially willows; retaining large snags and patches of snags with high habitat value; and retaining ecologically appropriate levels of large, down woody debris. Operators used excavators to pull up shrubs; fall snags; and then pile the shrubs, slash, and activity fuels for burning (fig. 23), while also protecting established conifer saplings. The slash piles were burned to prepare the site for replanting and interplanting. The new plantings were spaced among existing conifer regeneration or seed sources. Competing vegetation was reduced by applying herbicides in radial patterns around newly planted and existing trees. Managers designed these treatments to manage fuel development and promote tree growth with the intention of being able to return fire to these areas with less risk of stand replacement.

Figure 23—(A) Pretreatment conditions in a stand with many snags. (B) Mechanical piling of shrubs and snags to manage the development of fuel profiles in older patches of high-severity fire. Slash piles are then burned to prepare the site for planting. (C) Posttreatment in which overhead snag hazards and competing vegetation are reduced through piling, while larger, high-habitat-value snags, willows, and surviving trees are retained. Photos by Linda Smith.

Zone 2: Small High-Severity Burn Patches

Some areas that burn at high severity may not warrant intervention if the resulting fuel, stand densities, and stand composition are not far departed from reference conditions, especially if those patches are relatively small and otherwise consistent with desired conditions or the natural range of variation. Instead, they may be priorities for snag retention and early-seral vegetation, including shrubs (fig. 24).

For example, in the King Fire, managers used maps of historical shrublands (fig. 17) to inform their decisions about where not to plant. Another example from

Box 7: Challenges to Prescribed Burning in Large, High-Severity Burn Patches in the King Fire Footprint

The King Fire postfire treatment strategy prescribed fire as a followup treatment to mechanical treatments and as a standalone treatment in an area where managers identified that burning would be the only reasonably foreseeable treatment within the 10-year planning window. They proposed to reintroduce fire 5 years after the King Fire to break up fuel continuity within the steep canyon of the Rubicon River. Because this area had experienced intense, high-severity fire (Coen et al. 2017), large amounts of dead, fallen trees intermixed within shrubs were expected to develop, and mechanical treatments were not practical. They recognized that waiting too long to burn could hinder their ability to manage future wildfires in the area. Unfortunately, this proposed treatment encountered two major challenges: (1) some proposed ridgetop treatments, which were needed to facilitate the prescribed burn, were not completed within the initially planned timeframe; and (2) this approach was considered both risky and complicated. Managers have been challenged to accomplish even easier burns as a result of limited periods of favorable weather, other wildfires, lack of crew availability, and other constraints. This example illustrates how some postfire interventions, such as prescribed fire, especially in areas that are not effectively pretreated using other methods, are likely to become even more challenging in the future.

the Ranch Fire illustrated how measures of departure based on the natural range of variation for high-severity burn patch size, distance from green trees, and prefire vegetation types can influence estimates of areas that may be priorities for reforestation (box 8). Relatively small, high-severity burn patches may have a high likelihood of naturally regenerating, and monitoring may confirm that natural regeneration is sufficient to meet restoration objectives. However, treatments to encourage resilience of stands to fire may still be needed. Furthermore, on higher productivity sites, smaller openings provided by high-severity fire may provide opportunities to introduce desired species compositions and genetics to an area.

Figure 24—Small, high-severity burn patch (zone 2) without intervention in the Rough Fire footprint on the Sequoia National Forest. Photo byMarc Meyer.

Box 8: Identifying Priority Areas for Reforestation in the Ranch Fire Footprint

Identification of priority areas for intervention following the Ranch Fire included an analysis of departure in terms of high-severity burn patch size and likelihood of natural regeneration. The first step was to assess departure from the natural range of variation in terms of fire severity and high-severity burn patch size for yellow pine and mixed-conifer forests (fig. 25). High-severity burn patch size was classified into four groups: (1) patches <10 ac (<4 ha), (2) patches between 10 and <100 ac (4 and <40 ha), (3) patches between 100 and 250 ac (40 and 100 ha), and (4) patches >250 ac (>100 ha) in size. Classes 3 and 4 were considered moderately (displayed in yellow) and extremely (displayed in red) departed from the natural range of variation and desired conditions. Areas in green either burned at low or moderate severity or contained high-severity burn patches that were less than 100

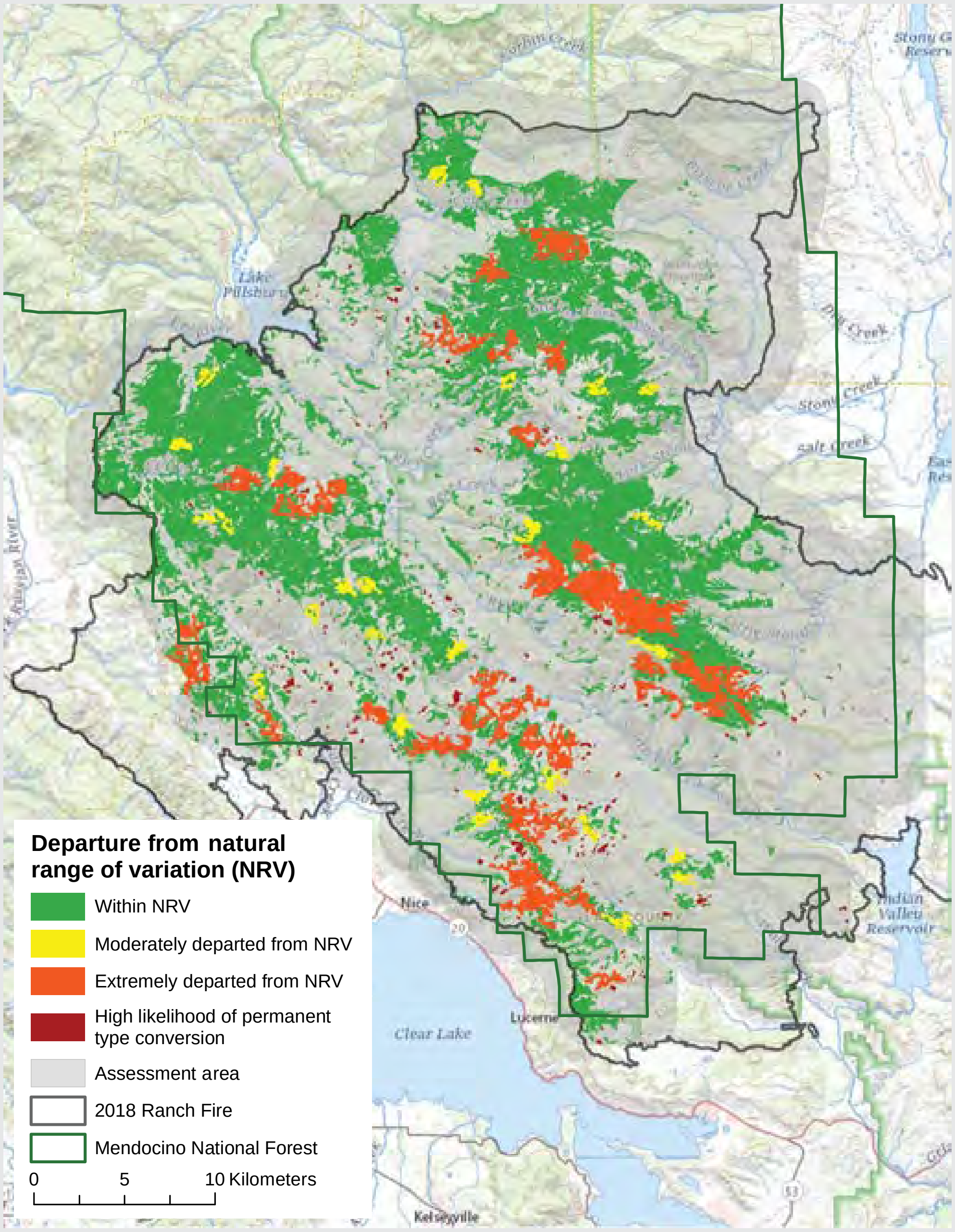

Figure 25—Departure from the natural range of variation, in terms of fire severity and high-severity burn patch size for yellow pine/mixed-conifer forests in the Ranch Fire footprint on the Mendocino National Forest. Map by Gabrielle Bohlman.

ac (40 ha) in extent. Orange areas contained high-severity burn patches that were within the natural range of variation for patch size but unlikely to reforest naturally because they were surrounded by non-yellow pine/mixed-conifer forest and therefore too far from surviving seed trees. A second component of the analysis was to analyze outputs from a spatial postfire conifer regeneration prediction tool (POSCRPT) (Shive et al. 2018).

The POSCRPT analysis found that 22 percent of areas that were yellow pine/mixed-conifer forest prior to the Ranch Fire had <60 percent probability of natural regeneration 5 years after the fire (fig. 26). Areas that were moderately or extremely departed from natural range of variation (in terms of fire severity) and had a low probability of natural regeneration were identified as high priorities for reforestation.

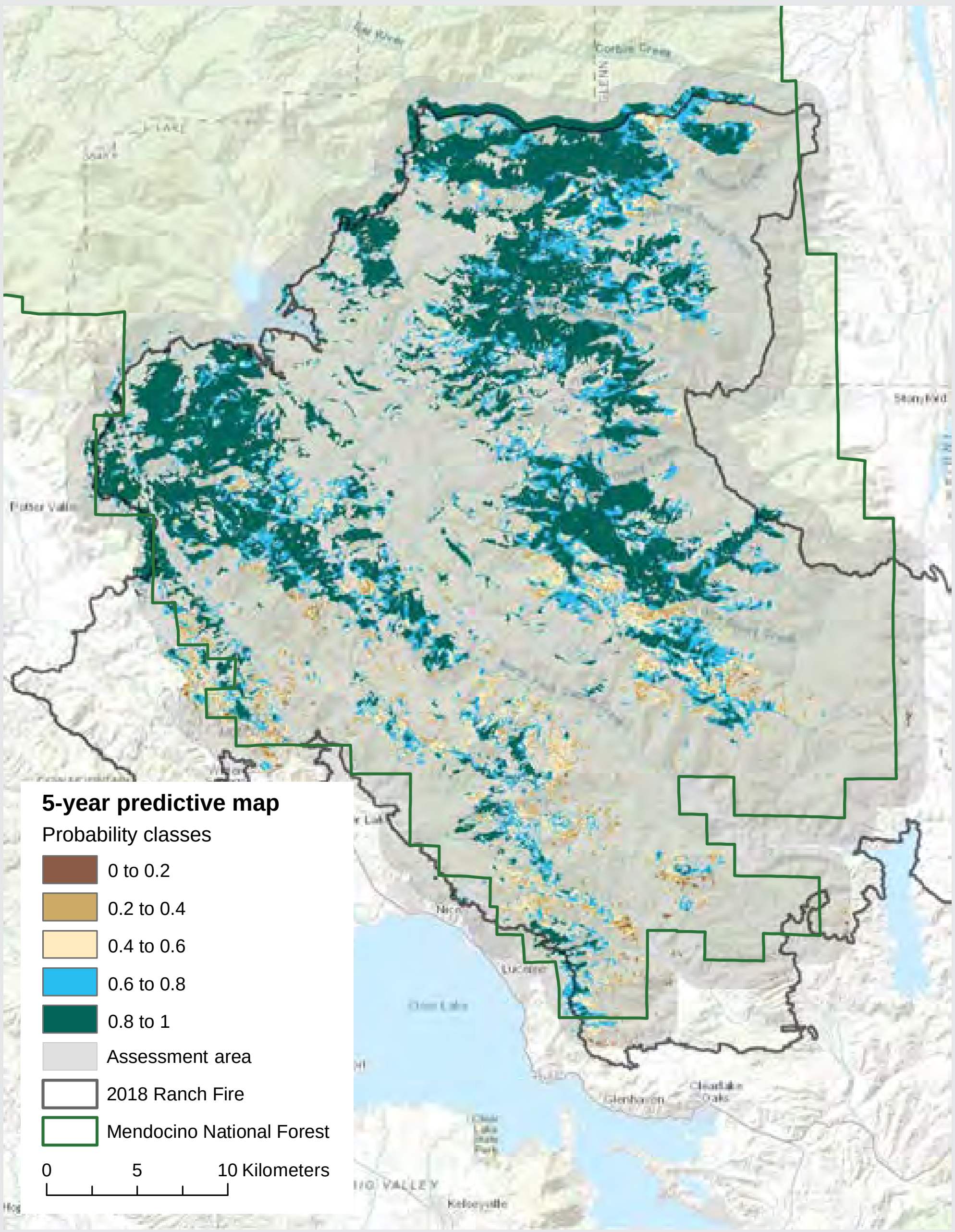

Figure 26—Probability of conifer regeneration (finding at least one regenerating conifer 5 years after fire in a 60-m² area) for yellow pine/mixed-conifer forests in the Ranch Fire analysis area on the Mendocino National Forest. Map by Gabrielle Bohlman.

Zone 3: Moderate-Severity Burn Patches

Although managers often focus on high-severity burn patches for interventions, restoration objectives may be advanced by targeting areas with unnaturally high fuel loads and tree densities, regardless of burn severity (Stephens et al. 2022). Moderate-severity burn patches may occur where many smaller trees have been killed amidst surviving larger trees (fig. 27). These areas therefore may constitute important progress toward restoration of more desirable, heterogenous forest conditions (representing the “maintain” category I). However, such

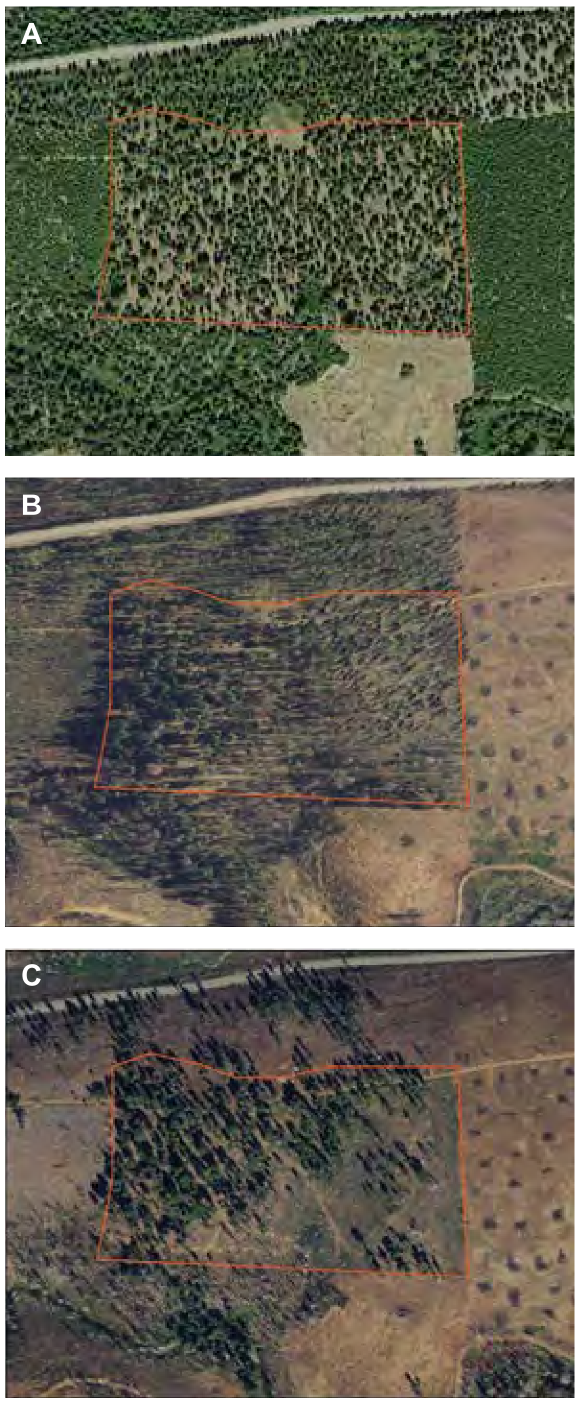

Figure 27—(A) 2014 prefire conditions of an area burned at moderate severity on the Eldorado National Forest; (B) 2016 variable mortality of overstory trees after the fire, but before salvage harvest; and (C) 2018 post-salvage harvest that reduced fuels from dead trees. Figure by Michelle Coppoletta.

Box 9: Restoring Moderate-Severity Burn Patches in Wildfires on the Plumas National Forest

In combination, the 2000 Storrie Fire and 2008 Rich Fire burned more than 2400 ha of serpentine habitat on the Plumas and Lassen National Forests, affecting some of the most extensive ultramafic terrain in the northern Sierra Nevada. These unique ecosystems have long attracted the attention of scientists and land managers because of their distinctive geology, harsh soils, and high plant diversity and endemism (Brooks 1987, Kruckeberg 1985, Roberts and Proctor 2012).

In 2013, the Chips Fire reburned about 2,580 ac (1045 ha) of serpentine habitat within the Storrie Fire footprint. A comparison of fire severity in these two fires suggests that serpentine areas that burned at lower severities in the Storrie Fire tended to reburn at lower severities in the Chips Fire; areas that burned at higher severities in the Storrie Fire also tended to reburn at higher severities in the Chips Fire. This pattern, which has also been observed by others in nonserpentine systems

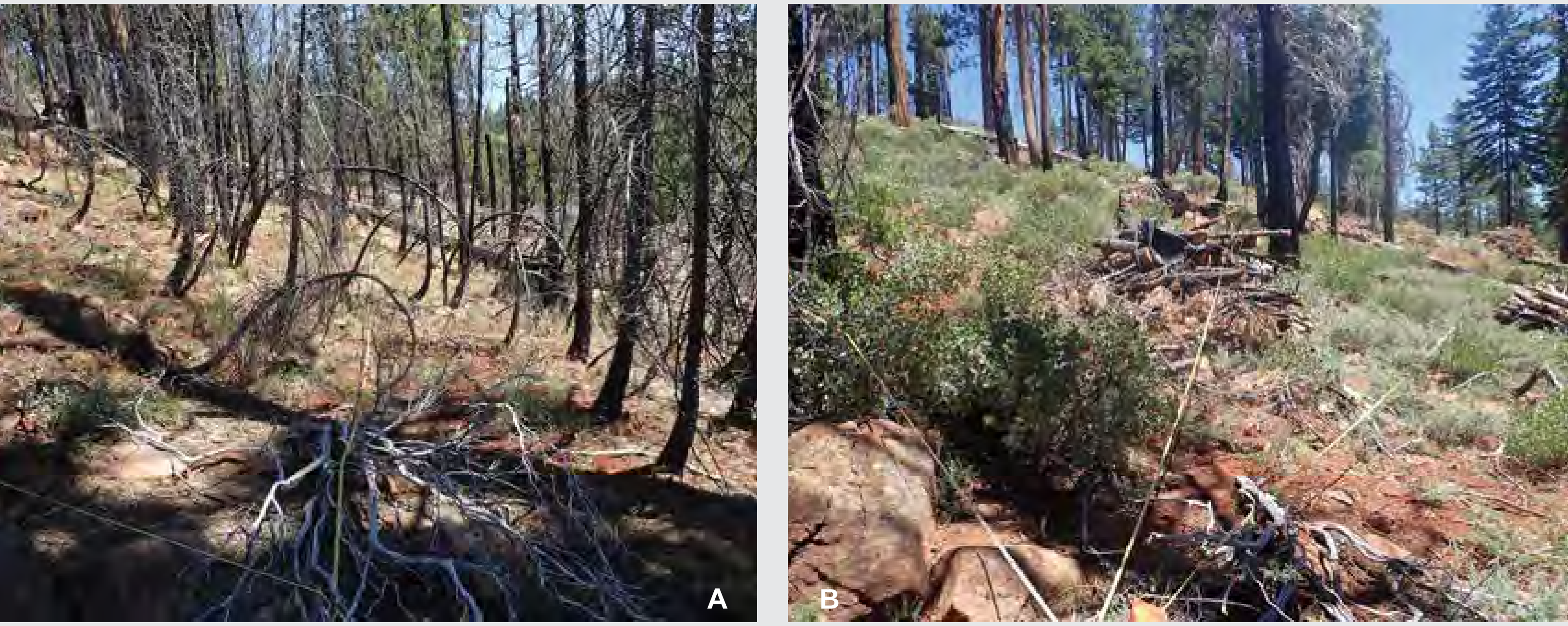

Figure 28—(A) Pretreatment and (B) posttreatment of a serpentine unit that had burned at moderate severity in the Storrie Fire where thinning of live and dead trees was implemented to reduce fuel loads. Photos by Michelle Coppoletta.

(Coppoletta et al. 2016), suggests that the risk of future high-severity fire may be higher in some serpentine areas that have already burned at high to moderate severity.

In 2017, the Plumas National Forest developed a project to increase the resilience of serpentine plant communities affected by the Storrie, Rich, and Chips Fires. They focused on areas that burned at low to moderate severity, which comprised 90 percent of the total project area (figs. 28 and 29). In 2020, hand thinning of both live and dead trees was implemented to reduce the risk of future high-severity fire by reducing fuel loads and increasing the amount of suitable habitat for serpentine plant species by opening the overstory canopy, reducing the duff layer, and thinning dense clusters of trees. Shortly after, the 2021 Dixie Fire burned through the treatment units. Preliminary observations suggest that units that had piled material on site at the time of the fire burned with high intensity; however, future monitoring efforts are needed to assess the impact of the fire on the serpentine rare plant community.

Figure 29—(A) Pretreatment and (B) posttreatment following prescribed fire in the Storrie Fire footprint in a stand that had burned at low severity. Photos by Michelle Coppoletta.

moderate-severity burn patches may also result in unnaturally high fuel levels that could lead to severe reburns and therefore could warrant active intervention (category II). For that reason, limiting salvage only to high-severity burn areas might not effectively promote landscape restoration. Prescribed burning under moderate weather might also be a tactic for reducing fuel loads in such areas. Intervention might not be needed where fire regime departure is low to moderate and fuel loads are within desired conditions. An example of where intervention would likely have been beneficial is one of the southernmost late-successional reserves (a special management area under the Northwest Forest Plan) that had burned at moderate severity in a 1996 wildfire, leaving large, living trees as well as many snags and subsequent downed wood that fueled a high-severity reburn in 2018 (fig. 30) (Bohlman et al. 2021).

Moderate-severity burn patches may also result in unnaturally high fuel levels that could lead to severe reburns and therefore could warrant active intervention.

Figure 30—A late-successional reserve on the Mendocino National Forest that had burned at moderate severity in the 1996 Fork Fire and reburned at high severity during the Ranch Fire of 2018. Photo by Gary Urdahl.

Zone 4: Low-Severity Burn Patches and Green Forests Surrounding High-Severity Burn Patches

Areas surrounding high-severity burns are important to consider because their condition may influence whether future fire, including reburns, will reinforce or induce an expansion of large, high-severity burn patches. These areas of mature trees are critical sources of conifer seed for adjacent high-severity burn patches. Treatments in this zone may facilitate using these areas as control points for future managed fire within higher severity burn patches as well as for maintaining high-quality habitat for wildlife species that require green trees for nesting, denning, and foraging. Consequently, thinning to achieve desired tree densities and using managed fire to reduce fuel loads may help to realign trajectories for this portion of the postfire landscape.

Interventions can build on restorative work done by wildfires in low-severity burn patches and green forests.

Low-severity burn patches may not intuitively register as immediate priorities based on an assumption that such effects indicate that the area proved resilient and may already be on a desirable trajectory (and regarded as category I). However, for several reasons, low-severity burns may not produce a desired multiaged structure or reference composition. Low-severity fire may have resulted from favorable shifts in wind, nighttime burning, or suppression efforts. As such, burning effects may not necessarily realign stand densities, composition, fuel levels, or other restoration objectives, especially when the prefire conditions were altered because of past activities (e.g., fire suppression, timber harvest.). In many cases, more severe fire effects (e.g., moderate-severity fire) or multiple burn entries are required to restore forest stand structure and composition (Collins et al. 2011, Levine et al. 2020). Consequently, in some cases, interventions can build on restorative work done by the fire in low-severity burn patches and green forests; such interventions may include both thinning (box 10) and use of fire (box 11). Managing for the growth and vigor of these "green" early- to mid-seral forests provides an important foundation for resilience by promoting disturbance-resistant structure and future seed sources and by accelerating the development of mid- to late-seral stands. Silvicultural prescriptions can be designed to enhance heterogeneity by promoting clumps, gaps, and thinned matrices of individually spaced trees; retain a diversity of age classes (where present); and promote species and age class diversity. These goals are important particularly for landscapes with high proportions of stand-replacing fire that are vulnerable to being homogenized by future fires.

Box 10: Treating Low-Severity Burn Patches in the Moonlight Fire Footprint

The 2007 Moonlight Fire resulted in large, high-severity burn patches that were targeted for reforestation activities (Collins and Stephens in North et al. 2012); however, a postfire restoration strategy also identified opportunities in younger stands that burned at low and moderate severities immediately adjacent to those stand-replacement patches. For example, within the fire perimeter, 40-year-old plantations that were established after the 1970 Big Fire burned at low severity. Despite the fire, these planted stands remained very homogeneous and densely stocked with about 20- to 40-ft-tall trees at 16-ft spacing between individuals (fig. 31). The postfire restoration strategy identified the opportunity to thin these plantations not only to lower densities to improve tree vigor and growth but also to promote spatial heterogeneity and mid-seral conditions. The additional growing space created by thinning could also stimulate production of seeds, which could benefit adjacent large, high-severity burn areas.

The postfire restoration strategy also highlighted opportunities in naturally established, mid-seral stands that burned at low severities, despite having high densities of ladder fuels and codominant trees (fig. 32). The strategy proposed thinning these stands to reduce ladder fuels and stand densities, accelerate development of late-seral conditions, and improve resilience to insects, disease, and drought. Thinning prescriptions included promoting within-stand heterogeneity by retaining clumps of large, dominant overstory trees; reducing densities in the intervening forest matrix; and creating openings to facilitate regeneration of another young cohort. This approach was designed to promote greater structural and age class diversity.

Figure 31—This 40-year-old plantation burned at low severity in the 2007 Moonlight Fire. The restoration strategy prescribed postfire tree harvest to promote within-stand heterogeneity and development of later seral forest conditions. Photo by Ryan Tompkins.

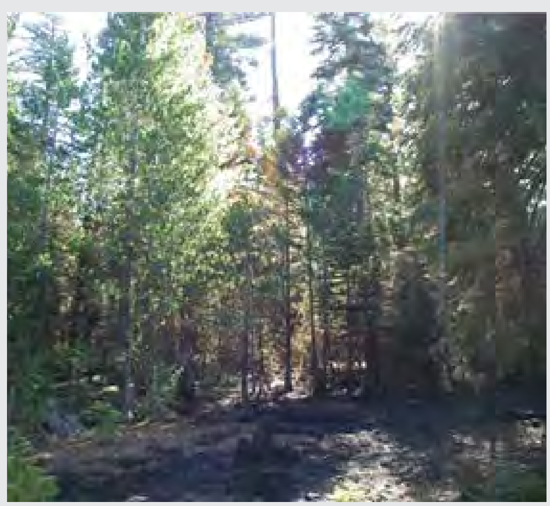

Figure 32—This mid-seral stand burned at low severity in the 2007 Moonlight Fire. High densities of ladder fuels and codominant trees remain far from ideal conditions. Photo by Ryan Tompkins.

Box 11: Burning Low-Severity Burn Patches in the Power Fire Footprint

Researchers collaborated with managers to conduct experimental prescribed burns in previously burned areas of the 2004 Power Fire footprint (fig. 33). Goals of the project included protecting overstory trees; reducing fuels generated by the Power Fire; facilitating future wildfire management; connecting and reinforcing existing fuels treatments; and protecting, maintaining, and improving habitat in and adjacent to California spotted owl protected activity centers (PACs). The treatments targeted areas that burned at low severity in the Power Fire. Managers expressed trepidation about using prescribed fire in areas of moderate burn severity because of high fuel levels. The project, "which included 4,000 ac (1619 ha) of understory burning within and adjacent to the Power Fire footprint, was designed to reinforce fuel breaks within the larger landscape by factoring in features such as plantations (fig. 34) and rock outcrops (fig. 35).

Figure 33—Experimental prescribed burn in an area burned 14 years earlier by the Power Fire on the Eldorado National Forest. Photo by Jesse Plummer.

Figure 34—Unit 4 in the Power Fire fuels maintenance study included 852 ac where treatments were designed to facilitate future fire management, connect multi-ownership fuel breaks, and avoid plantations that were established shortly after the Power Fire. Photo by Jesse Plummer.

Figure 35—Unit 6 in the Power Fire fuels maintenance study contained 597 ac that were anchored into the dam and granite escarpments of Bear River Reservoir and designed to create a fuel break between the North Fork Mokelumne River canyon and the Bear River Cabin tract. Photo by Jesse Plummer.

Zone 5: Ridges

Ridges are important parts of the landscape because they can serve as fuel breaks and other strategic fuel treatments to help divide an area into more topographically relevant burn units. This practical consideration may also be consistent with the historical condition of ridgetop forests when they had active fire regimes. Research of old-growth, mixed-conifer forests with restored fire regimes (mostly in Yosemite National Park and Sequoia and Kings Canyon National Parks) found ridgetops to have very low densities of pine trees and open canopy cover (Jeronimo et al. 2019, Lydersen and North 2012, Ng et al. 2020), which are similar to conditions that managers often produce in treated fuel breaks. Fuels treatments on ridges can also be strategically positioned to facilitate seed dispersal to the slopes below. Roads that follow ridgelines are often particularly important because they afford access for treatment and fire management (fig. 36).

Figure 36—Road accessing a ridgeline in a burned area with steep topography. Photo by Clint Isbell.

Objectives for treatments on ridges may include harvesting trees to reduce postfire fuels and maintain more open stands with lower tree densities and reduced shrubs that can serve as fuel breaks (fig. 37) (Swanston et al. 2020) as well as planting conifers to act as future seed trees. However, conifer plantings are likely to be at comparatively lower densities, reflecting lower productivity of ridges compared to flatter areas. Ridges may be further subdivided based on topography. For example, broad, flat ridges lend themselves well to a mixed stand of conifers and hardwoods. Such ridges may be a good place to promote the growth and development of mature conifer trees, which can act as a source for future seed

Figure 37—Example of a ridge (zone 5) where powerlines are protected with novel fuel break conditions. Photo by Dana Walsh.

dispersal. In contrast, steeper ridges may be more appropriate for lower tree density targets or even devoid of conifers because these areas are more likely to experience more frequent and severe fire.

Various approaches have been applied to treating ridgelines in ways that might afford more resilience to future fires. Some past approaches have focused heavily on fuel reduction and fire control rather than broader restoration. For example, some efforts have promoted even spacing of trees to avoid continuous crowns, in patterns that are unlikely to reflect natural disturbances (North and Keeton 2008). Furthermore, some stakeholders have expressed concern that ridgeline treatments have affected cultural resources important to American Indian tribes, including medicinal and food plants and sacred sites (Norgaard 2019). Alternative treatments along ridgelines have been proposed to better emulate natural conditions, including reestablishing natural structural patterns, such as clumps and gaps (North and Keeton 2008), as well as dominance of sugar pines and hardwoods. Rather than constructing unnaturally hard containment lines to bound fire management units, a restoration focus might reinforce natural vegetation patterns by stitching such thinning treatments along natural and other boundaries to serve as anchor points to help keep fires within more manageable areas (North et al. 2021). Recent approaches on the Klamath National Forest have shown how oaks can serve as a natural fuel break that provides important cultural ecosystem services while also facilitating use of prescribed or naturally ignited fire (fig. 38). Oak-dominated fuel breaks have also proved effective in moderating wildfire effects while also favoring conditions that would support acorn harvest by local tribal nations (fig. 39).

Restoration might stitch treatments along ridges and other boundary features to help keep fires within more manageable areas.

Figure 38—Ridgetop treatments on the Klamath National Forest that were designed to promote oaks and serve as a fuel break. Photo by Kevin Osborne.

Figure 39—Mature California black oaks border a road that has been identified as a fire control line in the potential operational delineations framework on the Mendocino National Forest; it was used to control the 2018 Ranch Fire. Photo by Jonathan Long.

On the King Fire footprint, some ridges were planted at lower density with conifers to encourage a shaded fuel break where firefighters could operate without having to deal with high amounts of volatile shrubs. An objective was to get these planted stands into conditions amenable to burning as quickly as possible and then to apply fire early and often. Managers did not plant every fuel break that they identified, and they retained oaks where they occurred. Reburns of these landscapes during the 2020 Fork and Point Fires and the 2022 Mosquito Fire has reinforced the strategic importance of these ridgelines to landscape fire management.

Zone 6: Hardwood Groves

Hardwood ecosystems represent a substantial proportion of the areas burned in large wildfires, representing 17 percent of area burned in California from 2000 to 2020 (Calhoun et al. 2022). As noted previously, it is important to consider their importance for ecological restoration given changing climate and fire regimes. Groves of oak, aspen, tanoak, madrone, giant chinquapin *(Chrysolepis chrysophylla),* and other hardwoods within the mixed-conifer forests of California are an important concern for long-term ecological restoration for many reasons, including their high value for wildlife and traditional American Indian uses, as well the opportunities they can provide for managing fires (Long et al. 2018a). Many of the more shade-intolerant species, California black oak, Oregon white oak (*Quercus garryana*), and aspen form groves (fig. 40), with California black oak and aspen groves being common in the Sierra Nevada and Oregon white oak

Figure 40—Grove of California black oaks (zone 6) in the Rubicon Canyon within the King Fire footprint, Eldorado National Forest. Photo by Blake Englehardt.

more important to the Northwest. Large trees of the shade-intolerant hardwoods have declined as a result of conifer encroachment and shifts in fire regimes from relatively frequent and low-severity fires to higher severity events that kill the tops of mature hardwoods (Devine and Harrington 2013, Jones et al. 2005, Krasnow and Stephens 2015, Long et al. 2018a, Schriver et al. 2018). Hardwood stands within mixed-conifer forests are important concerns because their mature trees provide important ecological services (Calhoun et al. 2022, Long et al. 2018a). California black oak appears particularly vulnerable to declines under current trajectories (Kralicek et al. 2022, Long et al. 2018a, McCord et al. 2020).

Given their ability to resprout following fires (fig. 41), including repeated short-interval wildfires (Nemens et al. 2018), and general drought resistance, most oak species are expected to prosper in a warmer California that experiences more fire (Dolanc et al. 2014, McIntyre et al. 2015). Hardwood stump sprouts have larger, deeper root systems than young conifers and shrubs, which affords them the upper hand in competition for the first few decades. Oaks can exist in clumps amidst conifers if there is enough growing space and light. Consequently, effects to hardwoods as a result of conifer encroachment are most serious under closed canopy conditions. It may be possible to mitigate the potential for overtopping by conifers in future decades through both thinning and wildland fires managed for resource objectives. Promoting oaks across the entire landscape, including in more productive sites (valley bottoms and deeper soils), can help sustain their natural diversity and ecological services, including acorn gathering sites near meadows and water sources that support other wild foods (Long et al. 2016). The Western Klamath Restoration Partnership has promoted shaded fuel breaks in tanoak groves to promote tribal values while reducing wildfire risk (USDA FS 2018), and similar approaches are being used in black oak groves on the Mendocino National Forest in ancestral Pomo lands (fig. 39).

Balancing conifer planting by maintaining historical hardwood groves— Postfire treatments need to consider opportunities to enhance important hardwood groves, while also maintaining conifer seed sources within the landscape (box 12). The Sierra Nevada Forest Plan Amendment required that conifer trees not be planted within fixed buffers around oaks (Long et al. 2017). In oak-dominated groves, especially those that have been historically tended by American Indians, managers might consider avoiding planting or applying wide buffers (a radius of more than one mature crown height, as suggested in the "full-release" treatment evaluated by Devine and Harrington (2013) around oaks that are shade intolerant (such as California black oak and Oregon white oak), relatively rare in an area (e.g., large trees, top-surviving oaks, oaks at higher elevations), and located in accessible areas and valley bottoms/flats. Although buffers help to control the density immediately around the resource of concern, they could potentially make it harder to promote natural heterogeneity and the complex structures that are historically found in stands with codominant oaks and conifers. Thoughtful restoration may require careful consideration of reference conditions and approaches to promote a mixture of open mixed stands, open oak-dominated stands, and closed canopy conifer-dominated stands.

Figure 41—California black oak and other hardwood species are often resilient to stand-replacing fire, such as this patch in the Rim Fire footprint, because they typically resprout. USDA Forest Service photo.

Box 12: Postfire Interventions in Aspen Stands on the Plumas National Forest

In 2007, the Moonlight Fire burned more than 140 stands (809 ha) of quaking aspen on the Plumas National Forest in northeastern California. Aspen are shade intolerant, making them dependent on disturbance, such as wildfire, to remove encroaching conifers, create suitable conditions for growth, and stimulate vegetative regeneration. Because of this, some of the aspen stands (about 11 percent) in the Moonlight Fire benefited from high-severity fire effects that eliminated competing conifers and stimulated aspen regeneration as observed in other severely burned aspen stands in the Sierra Nevada (Krasnow and Stephens 2015). However, more than three-quarters of the aspen in the Moonlight Fire burned at moderate or low severity, and postfire monitoring determined that more than 80 percent were still considered to be at risk of loss due to conifer encroachment and deer or livestock browsing after the fire (fig. 42). Consequently, managers designed a large-scale aspen restoration project that focused on providing suitable growing conditions with high levels of sunlight. In aspen stands that burned at low to moderate severity, treatments were focused on conifer removal within and adjacent to aspen to increase light availability and reduce the potential for future conifer recruitment. The number of conifers retained varied by stand and was dependent on the tradeoff between maximizing light to aspen and maintaining mature live conifers, which were considered important seed sources for natural reforestation in the adjacent severely burned landscape. As a result, aspen stands that were surrounded by large patches of high-severity fire retained a greater number of live conifers (within the 100-foot buffer surrounding the aspen) than stands characterized by smaller patches of low- to moderate-severity fire effects. Treatments were proposed to extend beyond the individual aspen stand, with variable-density thinning treatment units located between stands to provide continuity and resilience at larger scales (fig. 43).

Figure 42—Quaking aspen stands in the Moonlight Fire footprint on the Plumas National Forest. Photo by Michelle Coppoletta.

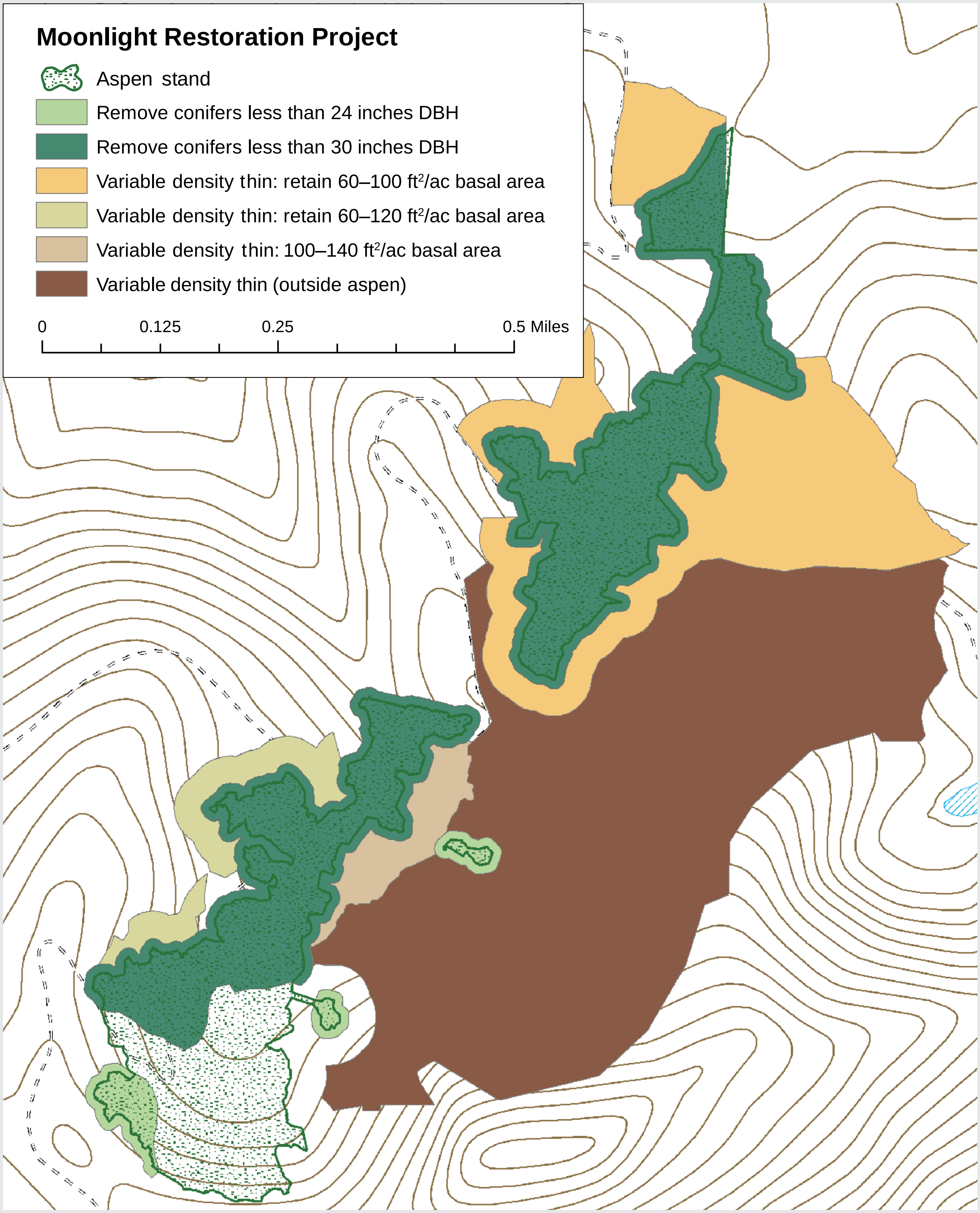

Figure 43—Proposed quaking aspen restoration treatments within the 2007 Moonlight Fire footprint included conifer removal within aspen stands, with variable levels of basal area retention adjacent to (<100 feet) and between individual aspen stands. DBH = diameter at breast height. Map by Michelle Coppoletta.

Thinning oak resprouts after fire—
Foresters have considered thinning multistemmed oaks several years after a fire to concentrate growth into larger stems and accelerate acorn production, based on local knowledge and some research that suggested there could be some benefits to such treatments. Studies in California found limited benefits in nonwildfire contexts (McDonald and Vaughn 2007); however, a study in Spain suggested leaving three resprouts was better than one to encourage development of taller stems (Espelta et al. 2003). The effectiveness of this treatment is currently being evaluated on the Power Fire footprint and other burned areas in California (fig. 44) (Long 2017).

Figure 44—(A) Before and (B) after pruning a resprouted California black oak to leave three stems for a study to evaluate potential to accelerate the development of mature trees in the Power Fire footprint. Photos by Pascal Berrill.

Zone 7: Riparian Zones

Riparian areas pose challenges for postfire management because they constitute only a small percentage of the landscape, yet they are often some of the most productive, dynamic, and habitat-rich areas. Their importance reflects the diversity of species and ecological services that they support, including habitat connectivity (Fremier et al. 2015) and maintaining climate refugia (Wilkin et al. 2016). Riparian areas have distinctive policy frameworks, such as the Aquatic Conservation Strategy of the Northwest Forest Plan, that recognize their importance; however, scientific research has suggested that some conservation practices have resulted in benign neglect (Hunsaker and Long 2014, Reeves et al. 2018). The diversity of riparian types has important implications for fire and postfire interventions. As streams grow larger, they can change microclimate conditions enough to dampen wildfire effects. Riparian areas along large, perennial streams are often dominated by hardwood trees, shrubs, and herbaceous vegetation; they promote heterogeneity among and connectivity between terrestrial systems dominated by upland conifers and aquatic systems. However, riparian areas along smaller, intermittent streams often differ little from adjacent upland areas in terms of fire return interval or historical forest structure (Van De Water and North 2011). Riparian areas are often managed with a hands-off approach, even to the point of limiting prescribed burning because of concerns over possible impacts to water quality and habitat for sensitive species. However, a lack of treatment in these areas may heighten potential for leaving problematic wicks in the landscape, which can allow explosive spread of wildfires, as reported from the Angora Fire (Safford et al. 2009).

By funneling and aggregating water and nutrients, higher order riparian systems support fast regrowth of vegetation, including herbaceous vegetation, shrubs, and deciduous riparian hardwood trees, including cottonwoods (*Populus* spp.) and willows (*Salix* spp.) that can resprout (fig. 45). That rapid vegetation growth can quickly stabilize sediment deposited by floods that often intensify in a postfire setting (fig. 42). Consequently, postfire interventions in higher order riparian areas have often been considered unnecessary, with the notable exception of degraded meadow systems (see next section). A passive approach to managing

Figure 45—Streamside vegetation growth along Big Grizzly Creek (riparian area, zone 7) that burned at high-severity in the 2014 King Fire. Photo by Blake Englehardt.

riparian areas has often been adopted before and after fires to limit the potential for additional erosion resulting from disturbance (Hunsaker and Long 2014). Lack of intervention may also reflect the importance of conserving large snags in riparian areas both as wildlife habitat and as future sources of large wood into channels (fig. 46). The importance of disturbances, including erosion and reconfiguration of in-stream habitats, has been recognized in recent science syntheses (Spies et al. 2019), so discontinuous and variable patches of high-severity burns across riparian zones may be desirable (fig. 47).

Figure 46— Large snags left for retention within a riparian area of the Salt timber sale, Klamath National Forest. Photo by Clint Isbell.

Figure 47—High-severity burn patches, such as this one in the wake of the 2011 managed Lion Fire on the Sequoia National Forest, may promote desirable heterogeneity in riparian systems as long as those patches are not exceptionally large and contiguous. Photo by Brent Skaggs.

However, as with upland areas, heavy accumulations of dead trees could elevate reburn severity (fig. 48). Managers face a challenge in weighing these considerations and determining which factors are most relevant for postfire landscapes. Retaining large, dead trees is important in riparian areas because of their special value as a source of structure when they fall into stream zones. However, research suggests that reference levels of coarse wood in the Sierra Nevada are likely much lower than for moister forest types in the Pacific Northwest, and large woody debris is likely less limiting in small headwater streams than in larger (often dammed) rivers (Long et al. 2014). Nevertheless, fires can substantially reduce the amount of large wood in streams until the dead trees begin to fall (Berg et al. 2002). Leaving large snags in a longitudinally well-distributed arrangement may provide for long-term stream inputs at different locations along channels. Recent research has considered potential benefits of adding wood to stream channels, through "hinging" or "tree tipping" (Reeves et al. 2018). Managers may also consider the potential for dead wood to form hazardous piles above road-stream crossings when evaluating target snag densities in riparian zones. There may be opportunities to integrate fuel management (processing small snags) with erosion control, especially in riparian areas that may be vulnerable to concentrated flows from slopes or roads into streams. One approach could include felling and masticating fire-killed trees to disrupt flow paths and increased ground cover, which can also reduce shrub regrowth (Kocher and Wade 2022). Recent studies indicate that wood mulch treatments can help to reduce soil erosion in sensitive areas (Robichaud et al. 2019).

Desired conditions may include promoting large conifers that may depend on moderate riparian environments to thrive under future climate conditions. Dense regrowth of shrubs is likely to limit recovery of riparian conifer trees without interventions (box 13). In such cases, it may be appropriate to plant selected conifer species at low densities on benches or other areas near enough to influence channels while limiting displacement of deciduous riparian trees. Planting conifers in wetland parts of riparian areas would be discouraged following this approach. Such interventions should be guided by evidence regarding species composition under reference conditions. Slower decomposing species might be particularly desirable as a means of restoring future large wood.

Figure 48—Dense stands of fire-killed trees in the King Fire footprint. As snags decay and fall over time, heavy accumulations of dead and down fuels can increase the risk of future high severity fires. Treatments in these areas need to balance concerns for reducing fuels with the need for coarse wood inputs. Photo by Jonathan Long.

Box 13: Corral Creek in the Rim Fire Footprint

Corral Creek passes through a large patch in which high-severity fire burned during the 2013 Rim Fire where soil erosion was deemed a high concern. Prior to the Rim Fire, this area contained old, mixed-conifer forest dominated by Douglas-fir with nesting spotted owls and goshawks. After the fire, there were scattered seedlings of Douglas-fir that established from residual seeds, but as a result of shrub competition and climate change, managers expect that it will be long dominated by shrub vegetation (fig. 49). Because roads are along the stream corridor, some of the dead trees were harvested as part of roadside hazardous tree removal (fig. 50).

A B

Figure 49—(A) Dead trees in a riparian area in a large, high-severity burn patch three months after the Rim Fire, November 2013; photo by Jonathan Long; (B) Shrub regrowth 7 years after the Rim Fire where some dead trees were removed in September 2020; photo by Adam Rich.

A B

Figure 50—Corral Creek in a high-severity burn patch of the Rim Fire footprint with soil erosion (A) in October 2013; photo by Jonathan Long; (B) in September 2020; photo by Adam Rich.

Zone 8: Meadows

Meadows support important ecosystem services, including regulating wildfires, sustaining biodiversity and favorable water flows, and supporting cultural values of tribes and recreational users (Long and Pope 2014). However, throughout our region, fire exclusion has facilitated encroachment of conifers, especially lodgepole pine (fig. 51) and Douglas-fir, into meadow areas. Moderate- to high-severity fire is an important mechanism for reversing such trends (Boisrame et al. 2017), and restoring more frequent fire regimes can help to maintain these meadow communities into the future. Because of those relationships, wildfires have potential to enhance meadow vegetative conditions, resulting in these areas falling into category I condition. Wildfires may create important opportunities to advance more holistic restoration efforts. For example, the legal settlement of the Moonlight Fire set up dedicated funding for ecosystem restoration in the burned area. Postfire interventions present an efficient opportunity to further reduce live-tree densities, as well as dead wood volumes, where they exceed reference conditions (fig. 52). Such activities may facilitate the use of managed fire to maintain meadow systems.

Wildfires have potential to enhance meadow vegetation conditions and to create opportunities to advance holistic meadow restoration efforts.

Despite the potential of wildfires to reverse conifer encroachment, meadow ecosystems are vulnerable to shifts in hydrogeomorphic conditions following uncharacteristically large and severe fires. For example, meadows may be eroded and desiccated because of intense postfire runoff (Long and Davis 2016), and meadow soils can be damaged by consumption or oxidation of organic matter and loss of associated plants (Long and Pope 2014) (fig. 53). Through such changes,

Figure 51—Young lodgepole pine encroaching on a meadow area in the Lake Tahoe Basin. Photo by La’akea Low.

Figure 52—(A) Before and (B) after cutting small conifer trees that had encroached in a meadow within a low- to moderate-severity burn patch burned in the Moonlight Fire footprint. Photos by Kelby Gardiner.

Figure 53—Severely burned soil at Spotted Fawn fen on October 13, 2021, 3 months after the Dixie Fire. Photo by Dave Immeker.

Figure 54—A beaver dam analog structure installed on Lone Rock Creek in the Moonlight Fire footprint as part of watershed restoration efforts. Photo by Kelby Gardiner.

wildfires can shift vegetation toward shrub-dominated communities (Sulwiński et al. 2020). Meadows facing such risks are likely to benefit from active interventions to control erosion, address grazing pressures, and reestablish native obligate vegetation. In-stream restorative treatments, such as beaver dam analogs below burned areas, may help inhibit or reverse channel degradation while retaining postfire flushes of sediment in floodplains (fig. 54).

Zone 9: High-Elevation Forests

Large and severe wildfires that originate in low- and moderate-elevation forest areas often burn into high-elevation areas (fig. 55). Fires may become easier to control as they burn into these high-elevation areas, where they often encounter reduced fuels and rock outcrops (figs. 56 and 57) (Long et al. 2018b). Consequently, they may burn at reduced severity, and resulting mortality is more likely to fall within the natural range of variation than in lower elevation forests that have more departed fire regimes (Lydersen et al. 2014, Meyer and North 2019). However, it is possible that as high-severity fires increasingly burn into the high-elevation forests areas, the resulting effects will depart from the natural range of variation or desired conditions (Turner et al. 2019). High-elevation forest ecosystems of the Sierra Nevada include upper montane and subalpine forests dominated by red fir, lodgepole pine, mountain hemlock, and high-elevation white pines (e.g., western white pine, whitebark pine [*Pinus albicaulis*]). Natural fire intervals in high-elevation forests are substantially longer than in the low- to mid-elevation

Figure 55—Upper montane, red fir forest stand burned in a large (about 750-ac [304-ha]), high-severity burn patch 27 years after the 1992 Rainbow Fire on the Inyo National Forest. Photo by Marc Meyer.

mixed conifer and yellow pine forests that are the focus of this report (Coppoletta et al. 2021, Meyer and North 2019). Nevertheless, several factors could warrant interventions in these systems. Like lower elevation coniferous forests, wildfires in high-elevation forests of the region are increasing in frequency (although not necessarily in severity), a trend associated with warming climate conditions and increased forest densification and fuels (Schwartz et al. 2015). High-elevation forests are also experiencing increasing tree mortality rates from insects, pathogens,

Figure 56—High-elevation forest in the Power Fire area with many openings and rocky areas. Photo by Jonathan Long.

Figure 57—High-elevation, whitebark pine forest near the 1992 Rainbow Fire footprint on the Inyo National Forest, where rock outcrops and discontinuous fuels can limit fire spread. Photo by Marc Meyer.

and moisture stress associated with climate change (Coppoletta et al. 2021, Meyer and North 2019), which may exacerbate the effects of future wildfires (van Mantgem et al. 2013).

Consistent with the postfire flow chart, a "maintain and promote" strategy may be appropriate for most of these high-elevation landscapes, with a heavy reliance on monitoring to determine if natural regeneration is sufficient for potential stand replacement (Meyer et al. 2019, 2023). Monitoring after 4 years may allow for high-elevation conifer seeds to disperse, germinate, and grow in response to increased seed-caching activity (especially high-elevation white pines) and resource availability such as soil moisture (Tomback et al. 2001). Priority areas for monitoring can be identified based on climate vulnerability assessments, regional conservation strategies, and other information sources. If monitoring suggests that postfire natural conifer regeneration is absent within large, high-severity burn patches, managers may consider planting treatments to promote founder stands, especially western white and whitebark pines that are resistant to white pine blister rust. Reliance on managed fires to promote a diversity of seral and structural classes, but particularly open stand conditions, is important to increase forest age and structural diversity at a landscape scale (Meyer et al. 2019) and to buffer high-elevation forests from the impacts of mountain pine beetle and other stressors following decades of fire exclusion (Meyer and North 2019).

Zone 10: Pine-Oak Areas

Relatively dry, low-elevation areas codominated by ponderosa pine and oak are among the most vulnerable forest types to postfire conversion. These concerns are particularly acute in the Sierra Nevada and in southern California (Lenihan et al. 2008). Pine-oak transition zones have some of the highest fire frequencies because they are moist enough to grow trees but dry enough to burn (Parks et al. 2019). Such trailing edge forests are expected to experience range contractions under a changing climate as they are replaced by grasslands, shrublands, or woodlands (Coop et al. 2020). In coastal ranges and the Klamath ecoregion, shifts toward hardwood-dominated woodland forests are also expected (Lenihan et al. 2008), although Douglas-fir-dominated forests may not be declining overall (Kralicek et al. 2022), and they appear likely to recover so long as severe fires do not become much more frequent (McCord et al. 2020). In the zone where conifer forests are on the drier end of a suitable climate envelope, it may be particularly challenging to determine when to intervene and when to accept a shift to an alternative (e.g., hardwood-dominated) community as part of realigning with the climate.

Analysis of areas with marginal climates may indicate where pine planting may have low probabilities of success. As an alternative, managers may reevaluate desired conditions by relying on oak regeneration (box 14). This approach aligns with a suggestion by Nemens et al. (2018) to design treatments to facilitate a frequent, low-severity fire regime and oak dominance.

Box 14: Conversion to Oak-Dominated Vegetation

There are several examples of wildfires in which managers have largely accepted a shift from a mixed pine-oak forest to oak dominance. The Ranch Fire analysis combined data on high-severity burn patch size and projected climatic water deficit to suggest where high conifer cover might be difficult to achieve and where encouragement of oak-dominated vegetation might be more appropriate. As another example, planting was deferred on the steep, south-facing hillslope in the North Fork of the Feather River canyon that burned in the 2000 Storrie and 2008 Rich Fires (fig. 58). This decision was motivated in part by limited access, steep terrain, and safety concerns, which not only affect the immediate planting challenges but also the ability to manage any planted stands long into the future. Managers also recognized that the area would likely experience repeat fire, based on the canyon topography, aspect, and predominance of smaller stature multistemmed oak in those stands. Therefore, any planted stands would be difficult to protect from future wildfires. In that case, managers contended that resprouting oaks met National Forest Management Act requirements to maintain appropriate forest cover, stocking and productivity of native tree species.

Figure 58—Resprouting hardwoods (oak and maple) in an area that was conifer forest prior to the 2000 Storrie Fire on the Plumas National Forest. Photo by Michelle Coppoletta.

Directions for Adaptive Management and Research

Intervention strategies will undoubtedly evolve as managers face new challenges and learn how ecosystems respond to uncharacteristically large and severe wildfires and interacting disturbances. An adaptive management framework, supported by monitoring, experiments, and other research, will be crucial for evaluating whether postfire interventions effectively meet restoration goals (Chen et al. 2013). Increasing likelihood of reburns and other disturbances are likely to alter system trajectories and increase the need for interventions. Further research will help to better distinguish when interventions are unnecessary, where they are needed to inhibit type conversions and likely to succeed, and where they are likely to fail, thereby informing the postfire flow chart (fig. 2). With so many of these tactics and approaches, we are still in the early phase of adaptive management, so long-term outcomes have not been measured and evaluated.

Research is especially needed to better understand the effects of postfire interventions:

- Operational approaches to reduce fuel in planted stands while avoiding damage to the planted trees in ways that facilitate stand resilience
- Effects of replanting with different spatial patterns on growth rates of regenerating trees, water balance, mortality from insects, and mortality from fire over extended periods (Fertel et al. 2022, North et al. 2019)
- Relationships between fine-scale site conditions and tree density to guide reforestation efforts (North et al. 2019)
- Impacts of nonconifer planting buffers around oaks (Long et al. 2016)
- Effect of prescribed burning on direct mortality in developing stands, including juvenile plantations and naturally regenerated stands, and on their subsequent capacity to withstand wildfire (North et al. 2019)
- Ecological and economic effects of planting at different initial densities and levels of maintenance (North et al. 2019)
- Costs and benefits of salvage/reforestation interventions on long-term forest carbon dynamics and other ecosystem services
- The effectiveness of fuel breaks along ridgetops, roads, and other strategic locations, in facilitating greater use of managed fire, as well as their effects on tribal cultural values and biological diversity, including habitat connectivity for species of conservation concern, such as Pacific fisher
- Effects of interventions in promoting important tribal values, including culturally important species associated with hardwood groves and meadows
- Benefits and risks of postfire treatment in riparian areas for sensitive taxa (e.g., amphibians), effects on water quality (including temperature), riparian vegetation, and large woody debris loading (Chan et al. 2004, Hunsaker and Long 2014, Long et al. 2014b)
- Techniques for reestablishing native understory plants that have special value for biodiversity, conservation concern, and tribal cultural significance

- Postfire management interventions appropriate for distinctive forest types, such as cypress groves, giant sequoia groves, aspen stands, and areas dominated by knobcone pine
- Continuing evaluation of potential benefits and risks of assisted relocation of adaptive genetic stock and possibly even assisted relocation of nonlocal species to support desired ecological services (Schwartz et al. 2012)
- Studies of wildlife relationships to size and arrangement of patches of remnant forest, burned stands, and planted trees

Adaptive management may be aided by mapping and archiving the design and implementation of postfire interventions to facilitate future study of outcomes. Such evaluations might need to be reviewed decades in the future by people who were not involved in their implementation. Treatment areas have been tracked through the Forest Activity Tracking System database for many years. However, retention or "leave" areas, which may be targeted for more passive restoration approaches, might not be explicitly tracked. Systematically recording more landscape-level information about treatment implementation, including deliberately untreated areas, could aid subsequent research and monitoring.

This report may help to explain ecological rationales for interventions, facilitate the pursuit of consensus on high-priority interventions, and winnow topics of discussion down to smaller parts of a landscape.

Finally, this report has not considered ways to engage different communities that have interests in postfire restoration. Postfire restoration involves the kinds of complex issues that have been considered in collaborative landscape restoration projects (Meyer et al. 2015). Planning for likely interventions before wildfires occur might make it easier to implement such measures in a timely fashion (Charnley et al. 2014, Dumroese and Moser 2020, Dunn et al. 2020) or at least to identify topics that need further examination to garner public support. However, postfire interventions are likely to face many of the barriers that have been identified in collaborative restoration, including social controversy over vegetation treatments, challenges in planning large prescribed fires that may need to cross boundaries, different views on reference conditions, unclear objectives for socioeconomic values, and community mistrust concerning agency motivations for decisions (Urgenson et al. 2017). Collaboratives have used a number of approaches to address these challenges, including issue-based recommendations, field visits, landscape-level analysis, engaging facilitators, working in smaller groups, and engaging respected scientists (Urgenson et al. 2017). We hope that this report may help to explain ecological rationales for interventions and facilitate the pursuit of consensus on high-priority interventions as well as help to winnow topics of discussion down to smaller parts of a landscape.

Metric Equivalents

When you know:	Multiply by:	To get:
Acre (ac)	2.54	Hectare
Foot (ft)	0.3048	Meter
Inches (in)	2.54	Centimeters
Trees per acre	2.47	Trees per hectare

Species Referenced in This Report

Common name	Scientific name
Quaking aspen	*Populus tremuloides* Michx.
Baker/Modoc cypress	*Hesperocyparis bakeri* (Jeps.) Bartel
Black-backed woodpecker	*Picoides arcticus*
Bigcone Douglas-fir	*Pseudotsuga macrocarpa* (Vasey) Mayr
Brown creeper	*Certhia americana*
California black oak	*Quercus kelloggii* Newb.
Cheatgrass	*Bromus tectorum* L.
Cottonwood	*Populus* L. spp.
Ceanothus	*Ceanothus* L. spp.
Douglas-fir	*Pseudotsuga menziesii* (Mirb.) Franco
Fringed myotis bat	*Myotis thysansodes*
Giant chinquapin	*Chrysolepis chrysophylla* (Douglas ex Hook.) Hjelmqvist
Giant sequoia	*Sequoia giganteum* (Lindl.) J. Buchholz
Hairy woodpecker	*Dryobates villosus*
Incense cedar	*Calocedrus decurrens* (Torr.) Florin
Jeffrey pine	*Pinus jeffreyi* Balf.
Knobcone pine	*Pinus attenuata* Lemmon
Lodgepole pine	*Pinus contorta* Douglas
Manzanita	*Arctostaphylos* spp.
Mountain bluebird	*Sialia currucoides*
Mountain hemlock	*Tsuga mertensiana* (Bong.) Carr.
Olive-sided flycatcher	*Contopus cooperi*
Oregon white oak	*Quercus garryana* Dougl. ex Hook
Pacific fisher	*Pekania pennanti*
Pallid bat	*Antrozous pallidus*
Ponderosa pine	*Pinus ponderosa* Lawson & C. Lawson
Red fir	*Abies magnifica* A. Murray bis
Redwood	*Sequoia sempervirens* (D.Don) Endl.
Sugar pine	*Pinus lambertiana* Douglas
Tanoak	*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh
Townsend's solitaire	*Myadestes townsendi*
Western cypress	*Hesperocyparis* Bartel & R.A. Price spp.
Western white pine	*Pinus monticola* Douglas ex D. Don
White fir	*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.
Whitebark pine	*Pinus albicaulis* Engelm.
Willow	*Salix* L. spp.

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