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Vegetation type conversion in the US Southwest: frontline observations and management responses

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Abstract

Background: Forest and nonforest ecosystems of the western United States are experiencing major transformations in response to land-use change, climate warming, and their interactive effects with wildland fire. Some ecosystems are transitioning to persistent alternative types, hereafter called “vegetation type conversion” (VTC). VTC is one of the most pressing management issues in the southwestern US, yet current strategies to intervene and address change often use trial-and-error approaches devised after the fact. To better understand how to manage VTC, we gathered managers, scientists, and practitioners from across the southwestern US to collect their experiences with VTC challenges, management responses, and outcomes.

Results: Participants in two workshops provided 11 descriptive case studies and 61 examples of VTC from their own field observations. These experiences demonstrate the extent and complexity of ecological reorganization across the region. High-severity fire was the predominant driver of VTC in semi-arid coniferous forests. By a large margin, these forests converted to shrubland, with fewer conversions to native or non-native herbaceous communities. Chaparral and sagebrush areas nearly always converted to non-native grasses through interactions among land use, climate, and fire. Management interventions in VTC areas most often attempted to reverse changes, although we found that these efforts cover only a small portion of high-severity burn areas undergoing VTC. Some areas incurred long (>10 years) observational periods prior to initiating interventions. Efforts to facilitate VTC were rare, but could cover large spatial areas.

Conclusions: Our findings underscore that type conversion is a common outcome of high-severity wildland fire in the southwestern US. Ecosystem managers are frontline observers of these far-reaching and potentially persistent changes, making their experiences valuable in further developing intervention strategies and research agendas. As its drivers increase with climate change, VTC appears increasingly likely in many ecological contexts and may require

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management paradigms to transition as well. Approaches to VTC potentially include developing new models of desired conditions, the use of experimentation by managers, and broader implementation of adaptive management strategies. Continuing to support and develop science-manager partnerships and peer learning groups will help to shape our response to ongoing rapid ecological transformations.

Keywords: Adaptive management, Alternative stable states, Forest management, High-severity fire, Post-fire recovery, Resilience, Vegetation type conversion, Community reorganization, Wildland fire

Resumen

Antecedentes: Los ecosistemas boscosos y no boscosos en el oeste de los EE.UU. están experimentando grandes transformaciones en respuesta al cambio de uso de la tierra, el calentamiento del clima y sus efectos interactivos con los incendios naturales. Algunos ecosistemas están en transición hacia tipos alternativos persistentes, a partir del ahora denominado “conversión del tipo de vegetación” VTC, por sus siglas en inglés. VTC es uno de los temas que más presión ejerce en cuestiones de manejo en el sudoeste de los EE.UU, aunque las estrategias actuales para intervenir y abordar el cambio usan frecuentemente acercamientos de prueba y error ideados después del evento. Para entender mejor cómo manejar el VTC, reunimos gestores, científicos y practicantes de todo el sudoeste de los EE.UU para recolectar sus experiencias con desafíos de la VTC, respuestas de manejo, y resultados.

Resultados: Los participantes en dos talleres proveyeron 11 casos descriptivos y 61 ejemplos de VTC de sus propios campos de observación. Estas experiencias demostraron la amplitud y la complejidad de la reorganización ecológica a través de la región. Los incendios de alta severidad fueron los conductores predominantes del VTC en bosques semiáridos de coníferas. Por un amplio margen, estos bosques se convirtieron en arbustales, con algunas conversiones a comunidades herbáceas nativas y no nativas. Áreas de chaparral y de artemisia casi siempre se convirtieron en pastizales no nativos a través de interacciones como el uso de la tierra, el clima y el fuego. Las intervenciones de manejo en áreas de VTC intentaron más frecuentemente revertir cambios, a pesar de que encontramos que estos esfuerzos cubrieron solamente una pequeña porción de áreas quemadas con alta severidad que experimentaron VTC. Algunas áreas tuvieron largos períodos de observación (>10 años), previos a iniciarse las intervenciones. Los esfuerzos para facilitar el VTC fueron raros, pero pudieron cubrir áreas amplias.

Conclusiones: Nuestros resultados ponen en relieve que este tipo de conversión es una consecuencia común de fuegos de alta severidad en el sudoeste de los EE.UU. Los que manejan los ecosistemas son observadores de primera línea de estos cambios de largo alcance y potencialmente persistentes, haciendo que sus experiencias sean además valiosas para desarrollar estrategias de intervención y en agendas de investigación. A medida que las causas se incrementan con el cambio climático, los VTC aparecen cada vez más probables en varios contextos ecológicos, y pueden requerir también paradigmas de manejo hacia la transición. Acercamientos al VTC incluyen potencialmente nuevos modelos de desarrollo con condiciones deseadas, el uso de la experimentación por parte de los gestores, y una amplia implementación de estrategias de manejo adaptativas. El continuo apoyo y desarrollo a las asociaciones científicas y de gestión y de grupos de aprendizaje entre colegas ayudará a formar nuestra respuesta a las transformaciones ecológicas rápidas que están ocurriendo.

Introduction

When disturbances overwhelm resilience mechanisms, vegetative communities change in composition, structure, and trajectory (Beisner et al. 2003; Millar and Stephenson 2015; Coop et al. 2020; Falk et al. 2022). If the new state is persistent and resilient to, or reinforced by, further disturbance, it can be considered a vegetative type conversion (VTC, Syphard et al. 2019; van Mantgem et al. 2020). Key drivers of VTC in the southwestern US are associated with climatic warming, land-use change, introductions of non-native species,

and anthropogenically-altered fire regimes. Throughout semi-arid forests of the region, the widespread disruption of historical fire regimes in the late 19th century has led to increased stand densities (Covington and Moore 1994), increasingly large and severe fires (Miller et al. 2009; Singleton et al. 2019), and accelerating fire frequencies in shrub-dominated landscapes subject to high numbers of anthropogenic ignitions (Balch et al. 2017). Simultaneously, climate change facilitates VTC by producing “hotter droughts” that stress existing vegetation (Williams et al. 2013; Allen et al. 2015), increase

fire severity (Mueller et al. 2020; Parks and Abatzoglou 2020), and limit the success of ecosystem re-establishment and recovery (Keeley 1991; Keeley et al. 2019; Stevens-Rumann and Morgan 2019; Davis et al. 2019). Novel drought effects are now emerging as a consequence of interactions between climate change, land-use change, and human-induced declines in water availability, particularly in arid environments with growing human populations (Crausbay et al. 2020). Acute moisture deficits are increasingly recognized as a driver of ecological transformation that may be irreversible (Crausbay et al. 2017; Batllori et al. 2020). As anthropogenic climate change continues to amplify these trends (Nolan et al. 2018; Williams et al. 2020), transitions to novel ecosystem types can be expected to become increasingly common.

Conifer-dominated, historically frequent-fire forests in the southwestern US are particularly vulnerable to VTC. Here, we focus on Arizona, California, Colorado, and New Mexico, but many events and trends we discuss are relevant elsewhere in western North America (Hessburg et al. 2019). Southwestern dry-conifer forests are defined as those dominated by ponderosa (*Pinus ponderosa*) or Jeffrey pine (*P. jeffreyi*) and often include associated species such as Douglas-fir (*Pseudotsuga menziesii*), red fir (*Abies magnifica*), southwestern white pine (*P. strobiformis*), limber pine (*P. flexilis*), and white fir (*A. concolor*). Over the last century or more, these forests have undergone significant changes in structure and function, mainly due to the lack of recurrent fire activity (Allen et al. 2002; Hagmann et al. 2021). Throughout the region, loss of Native American burning practices, industrial logging, livestock grazing, and active fire suppression disrupted historical fire regimes (Swetnam et al. 2016). With climate warming, recent fires often include large areas of high-severity (stand-replacing) fire effects that can result in rapid post-fire transitions to hardwood-, shrub-, herb-, or grass-dominated ecosystems (Savage and Mast 2005; Airey Lauvaux et al. 2016; Tepley et al. 2017; Coop et al. 2020). Post-fire recovery depends largely on the extent of parent tree survival, understory composition, and local- to micro-scale temperature and soil moisture conditions. Recovery is most challenged in uncharacteristically large high-severity burn patches that include spatially extensive mortality of parent trees and potentially severe and long-lasting impacts to the soil (Shive et al. 2018; Safford and Vallejo 2019; Dove et al. 2020). In warm and semi-arid regions, higher elevation and north-facing localities within a species distribution tend to be more favorable for post-fire recovery (Collins and Roller 2013; Korb et al. 2019; Stevens-Rumann and Morgan 2019). Fire-catalyzed VTC may be most common at warm/dry ecotones or in areas experiencing drought events, where low moisture availability had already stressed or killed overstory trees

prior to burning (Allen et al. 2015) and subsequently reduced post-fire regeneration rates (Rother and Veblen 2016; Young et al. 2019; Davis et al. 2019; Rodman et al. 2020). However, these same ecotonal forests are often resilient to recurrent low-severity fire, even with climate warming (Harris and Taylor 2020).

Recovery following stand-replacing disturbances in dry conifer forests can include successional pathways through aspen (*Populus tremuloides*), hardwood, or shrub-dominated stages, but current climatic and fire regime trends are enhancing the likelihood of permanent conversion and the spatial extent of hardwood and shrub dominance in many parts of the southwestern US. In portions of the Colorado Plateau and southern Rockies, ponderosa pine and mixed-conifer forests are converting to shrublands of Gambel oak (*Quercus gambelii*) and New Mexico locust (*Robinia neomexicana*) (Guiterman et al. 2015, 2018; Coop et al. 2016; Rodman et al. 2020). In the Sky Island ecosystems of southern Arizona and New Mexico, Madrean oak woodland species (e.g., *Q. arizonica* and *Q. hypoleucoides*) and *Ceanothus* shrubs are replacing conifers, even where a resprouting pine species (*P. leiophylla*) is common (Minor et al. 2017; Barton and Poulos 2018). In parts of southern Oregon and northern California, repeated high severity fires are helping to expand the colonization of knobcone pine (*Pinus attenuata*), a serotinous-cone species that is highly adapted to such a fire regime (Reilly et al. 2019). Elsewhere in California, severe fires typically induce a strong shrub response, often from *Ceanothus* or *Arctostaphylos* species, which compete intensively with conifer regeneration (Helms and Tappeiner 1996). Because they resprout, hardwoods—especially oaks—can benefit from conifer mortality, and their density has been generally increasing in California montane forests for decades due to interactions between forest disturbance and climate warming (Dolanc et al. 2014; McIntyre et al. 2015). Subsequent burning tends to reinforce hardwood and shrub response (Coppoletta et al. 2016; Haffey et al. 2018; Keyser et al. 2020), especially where other factors including sparsity of parent trees already inhibit conifer recovery. Reburning at low- to mixed-severity within decades of the initial high-severity fire may explain centuries-long persistence of shrublands in which fire was historically frequent (Iniguez et al. 2009; Guiterman et al. 2018; Roos and Guiterman 2021). As these examples illustrate, there is no intrinsic, single time scale that can be used to define when a type conversion has occurred without imposing an arbitrary standard. The distinction between transient and persistent reorganization depends more on the mechanisms at work, in particular, if the converted state is reinforced by altered climate or disturbance regimes (Falk et al. 2022).

The spread of non-native grasses and forbs (e.g., *Bromus* spp., *Avena* spp., *Erodium* spp.) due to interactions among land uses, climate, and changing fire regimes is generating substantial change in chaparral and sagebrush areas. These herbaceous species can support uncharacteristically frequent fire relative to historical intervals, resulting in positive feedback with fire that is driving extensive VTC (Balch et al. 2013; Syphard et al. 2019). The mechanism for woody decline and conversion is the relatively long period of recovery required to regenerate post-fire. Chaparral requires 10–15 years for recovery (Keeley et al. 2011; Keeley and Brennan 2012; Lippitt et al. 2013), while sagebrush may require several decades under favorable conditions (Shriver et al. 2018). These lapse periods are outpaced by the spread of non-native species such as cheatgrass (*B. tectorum*) that invade under and throughout shrub ecosystems, increase flammability, and set the stage for post-fire community reorganization (D'Antonio and Vitousek 1992).

Prevention of VTC is emphasized in forest and shrubland management in the southwestern US through measures that promote species or community resistance or recovery (e.g., Franklin et al. 2018). Current intervention strategies that include fuel reduction and repeated low-severity fire have a strong scientific foundation (Allen et al. 2002; Prichard et al. 2021) and are effective (Stoddard et al. 2021). These strategies often accord with the cultural burning activities of many Indigenous groups across the southwestern US (Kimmerer and Lake 2001; Roos et al. 2021), and, where they are conducted in diverse collaborations with tribes and other stakeholders, can have benefits to social systems that extend beyond ecosystem resilience (Lake et al. 2017).

Management after extensive high-severity fires is more challenging than prevention because we simply have not obtained adequate knowledge or experience. Research on VTC is relatively new, and we have yet to capture the scale of the phenomenon in space and time, including how many areas are undergoing VTC and how many areas might not experience VTC despite major post-fire changes. Studies on both natural and managed recovery following fires have yet to answer how future climate and disturbances interact with treatments to either promote recovery or reorganization.

To better understand the challenge of managing ongoing VTC, we held two multi-day workshops in 2019 that brought together managers, scientists, and practitioners to discuss their observations of, perspectives on, and experiences with VTC events (Gregg and Marshall 2020a, 2020b). Participants voiced a need for greater clarity on the regional extent of VTC and responses to it, felt that focusing on their own management units (though many are quite extensive) limited their understanding of others'

experiences with similar challenges, and found limited resources in the scientific literature to help answer questions. In this paper, we address these concerns by presenting the firsthand experiences of the workshop participants through a series of 11 case studies and a summary of 61 VTC examples (Fig. 1). During the workshops and throughout this paper, we categorized management responses to VTC as (i) *Reverse change*: restore pre-fire conditions or manage recovery such that the affected ecosystem is brought to a recognizable (perhaps pre-fire exclusion) and ideally more resilient composition and structure; (ii) *Observe change*: exercise patience and monitor the system and its post-disturbance trajectory; and (iii) *Facilitate change*: push the system along a new, potentially novel, trajectory (Table 1). We recognize that these responses generally align with the resist-accept-direct (RAD) framework (Schuurman et al. 2020) and chose to maintain our classifications because many of the VTC examples lack a specific management response, which may or may not constitute intentional selection of "accept" as the desired future condition. Below, we summarize the VTC case studies and the individual examples, then synthesize these in the context of pressing management challenges and opportunities. The full case study descriptions and details regarding our approach are provided in the online Supplemental Information that accompanies this article.

Case studies

Participant-provided case studies of VTC demonstrate the profound complexity of ecological reorganization in the region. For example, the conversion of forests by high-severity wildfire illustrates that history and land-use changes are important. In each case, processes that led to VTC started a century or more earlier with the disruption of historical fire regimes and associated changes to composition and structure. This slow but profound change set the stage for multiple disturbance agents often acting in conjunction to fundamentally shift the ecosystem type or its dominant species. Management responses have been similarly diverse, reflecting individual situations, constraints, and goals. We note that in several case studies, more than one category of management response is described, representing the evolving nature of VTC management and its trial-and-error approach.

Reversing change

One possible management response to VTC is to actively attempt to reverse changes. Such responses are highlighted by recovery efforts on the *Klamath Reservation in southern Oregon* (case study #1) where long-term fire exclusion allowed tree encroachment into important

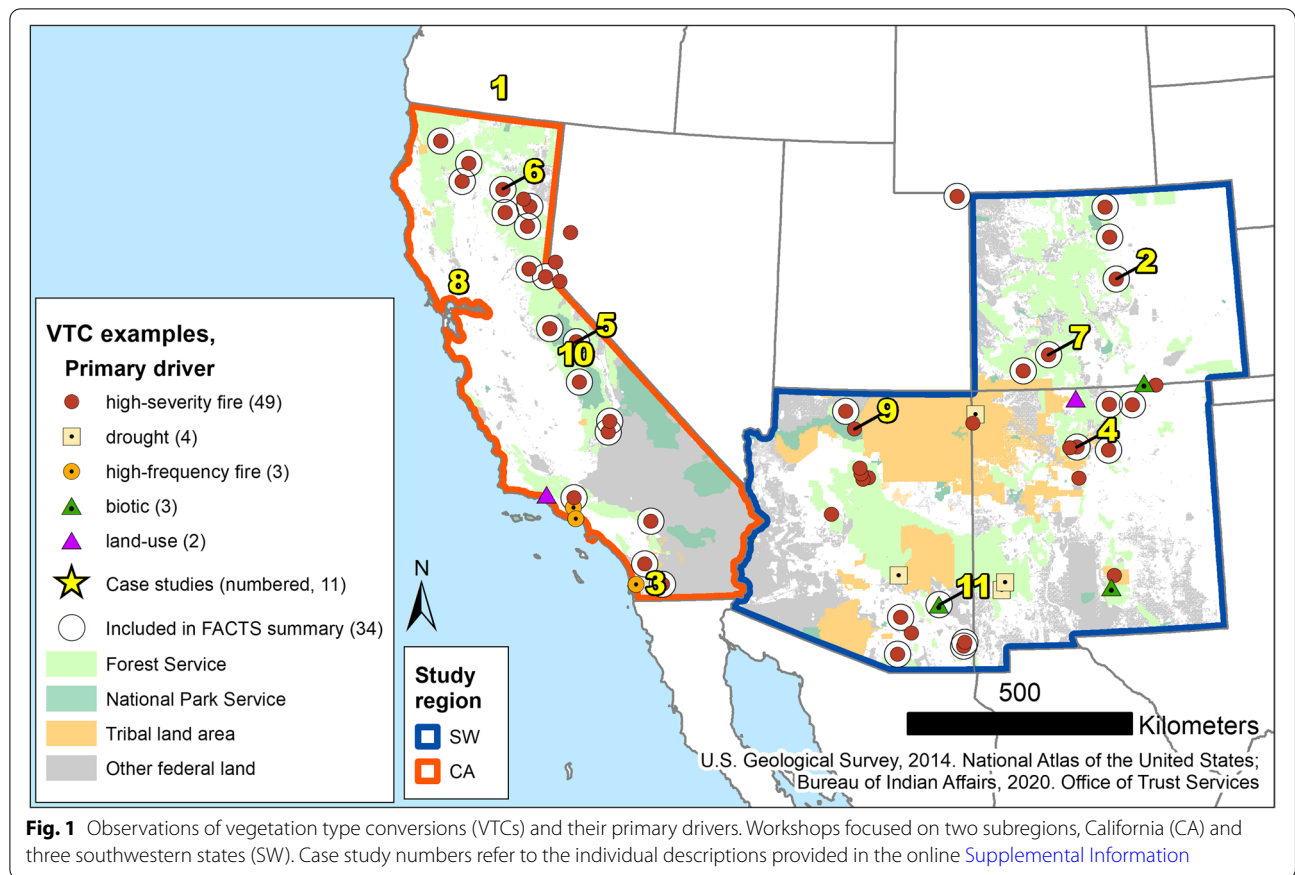


Fig. 1 Observations of vegetation type conversions (VTCs) and their primary drivers. Workshops focused on two subregions, California (CA) and three southwestern states (SW). Case study numbers refer to the individual descriptions provided in the online [Supplemental Information](#)

Table 1 Descriptions of management responses to VTC from workshop participants along with case study examples

Management response	Description	Case study examples
Reverse change	<p>Actively try to reverse change via:</p> <ul style="list-style-type: none"> • Coupled thinning and prescribed fire treatments to reduce fuel loads and fire severity and promote fire-dependent species and ecosystem recovery (Stephens et al. 2009) • Planting or seeding pre-VTC species • Removing or managing new or undesirable species (e.g., non-native grasses and shrubs that may increase fire frequency and/or severity) • Fire suppression to reduce fire extent and allow for recovery time • Preventing post-disturbance soil loss to sustain ecological functions 	<ol style="list-style-type: none"> 1. Klamath Reservation, southern Oregon 2. Southern Front Range, Colorado 3. Laguna Mountain, California
Observe change	<p>Take no active intervention measures and adopt monitoring to assess ecosystem trajectory over time. This approach may be most appropriate where there is:</p> <ul style="list-style-type: none"> • Limited management capacity (e.g., high upfront and maintenance costs of active intervention, limitations to access in sites such as those in wilderness or roadless lands) (Rother et al. 2015; Aplet and Mckinley 2017) • High uncertainty of unintended consequences of active intervention (e.g., one workshop participant noted that “sometimes doing something is worse than doing nothing”) (Landres 2010). This approach is consistent with restoration paradigms emphasizing a spectrum of approaches to spread risk (Aplet and Mckinley 2017). 	<ol style="list-style-type: none"> 4. Eastern Jemez Mountains, New Mexico 5. Devils Postpile National Monument, California 6. Lassen Volcanic National Park, California 7. San Juan Mountains, Colorado 8. Inner Coast Range, northern California
Facilitate change	<p>Actively direct system toward alternative and/or novel acceptable conditions by:</p> <ul style="list-style-type: none"> • Planting or seeding with focus on more drought- and fire-tolerant species compared to pre-disturbance species (e.g., assisted gene flow; Young et al. 2020) • Follow-up wildfires with ecologically-credible fuel reduction activities 	<ol style="list-style-type: none"> 9. North Rim of the Grand Canyon, Arizona 10. Southern Sierra Nevada, California 11. Pinaleno Mountains, Arizona

wetland and moist forest areas, altering the hydrology of the ecosystem and triggering the loss of culturally-important plants and environments. Tribal forest managers are working to restore forest structure and composition, improve wetland habitats, and recover the historical forest resilience and ecosystem services of the area. These efforts will hopefully stave off the kind of high-severity fires that are affecting areas of the *southern Front Range in Colorado* (#2). There, managers are achieving relatively high survival of planted ponderosa pine and Douglas-fir seedlings in the footprint of the 2002 Hayman Fire, despite years of drought since the planting operations (Fig. 2A). The success to date is credited to early spring planting operations targeted to the most productive sites, often at higher elevations and on northerly slopes, and using coarse-woody debris or other objects for additional shade. On *Laguna Mountain in southern California* (#3), however, a series of droughts, fires, and bark beetles have slowed or stopped post-fire recovery efforts in Jeffrey pine forests (Fig. 2B). Years of drought following the 2003 Cedar Fire prevented any tree recruitment and all planting operations failed. As managers were accepting the conversion to shrubland and hermland with scattered black oak (*Q. kelloggii*) and Coulter pine (*P. coulteri*), the newly established non-native goldspotted oak borer (*Agrilus auroguttatus*) decimated mature oaks (Safford and Vallejo 2019).

Observing change

The complexity of compounding disturbances including fire, insects, and climate warming can incapacitate recovery efforts. In many cases, observing changes is necessary

to gauge ecological trajectories, decide whether and how far outside of the natural range of variation the system has moved (Jackson 2012), and plan future management actions. In the *eastern Jemez Mountains of New Mexico* (#4), a series of high-severity fires culminating in the 2011 Las Conchas Fire left tens of thousands of hectares depleted of living conifers (Fig. 3A). Nearly 10 years post-fire, a coalition of stakeholders emerged with diverse plans to employ a variety of actions across the RAD framework based on variability in post-fire environments, community needs, tribal resources, and the risks of floods and debris flows originating from the burned area. Managers at the *Devils Postpile National Monument in California* (#5) found an array of post-fire trajectories in the decades following a mixed-severity fire. The pre-fire forest was recovering in lower-severity burn areas, but extensive shrublands were developing following complete overstory mortality in high-severity patches. Similar findings come from *Lassen Volcanic National Park in California* (#6) where mixed-conifer forests were widely transformed into shrublands, except where earlier prescribed fires reduced the intensity and severity of wildfire. In lodgepole pine (*P. contorta*) forests, low to moderate fire severity in 1984 generated legacy effects in a 2012 fire in which recent post-fire regeneration is abundant everywhere except for areas twice-burned at high-severity. The trajectory of these un-regenerated lodgepole pine forests is uncertain in light of warming temperatures, and may not return to pre-fire conditions. The same is true for subalpine forests in the *San Juan Mountains of southern Colorado* (#7) where a severe bark beetle outbreak and subsequent high-severity fire resulted in high aspen

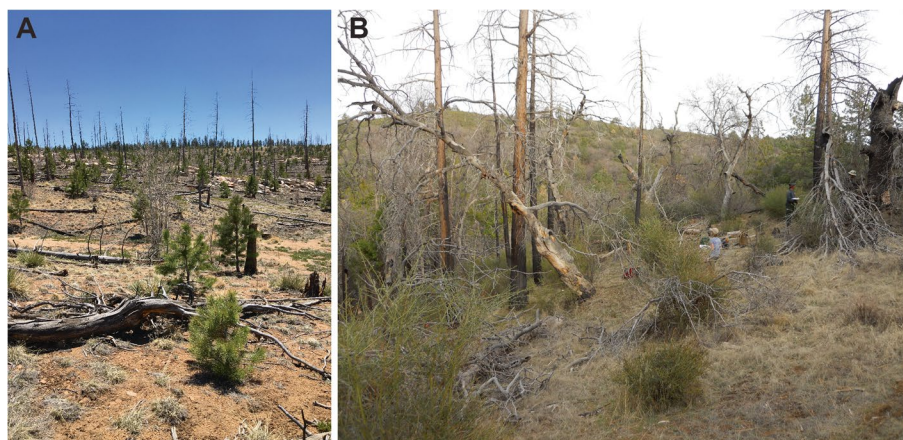


Fig. 2 Examples of reversing change. **A** The distribution of coarse woody debris around planted ponderosa pine seedlings following the 2002 Hayman Fire in Colorado is credited with helping to mitigate drought effects on the developing seedlings (credit: Paula Fornwalt). **B** Forest Service staff inventory stand conditions in a former Jeffrey pine–black oak forest on Laguna Mountain, Cleveland National Forest, eastern San Diego County, California (**B**). This site was impacted by multiyear drought, then severe wildfire, then drought again, Jeffrey pine beetle mortality, and most recently by an oak borer outbreak (credit: Hugh Safford)

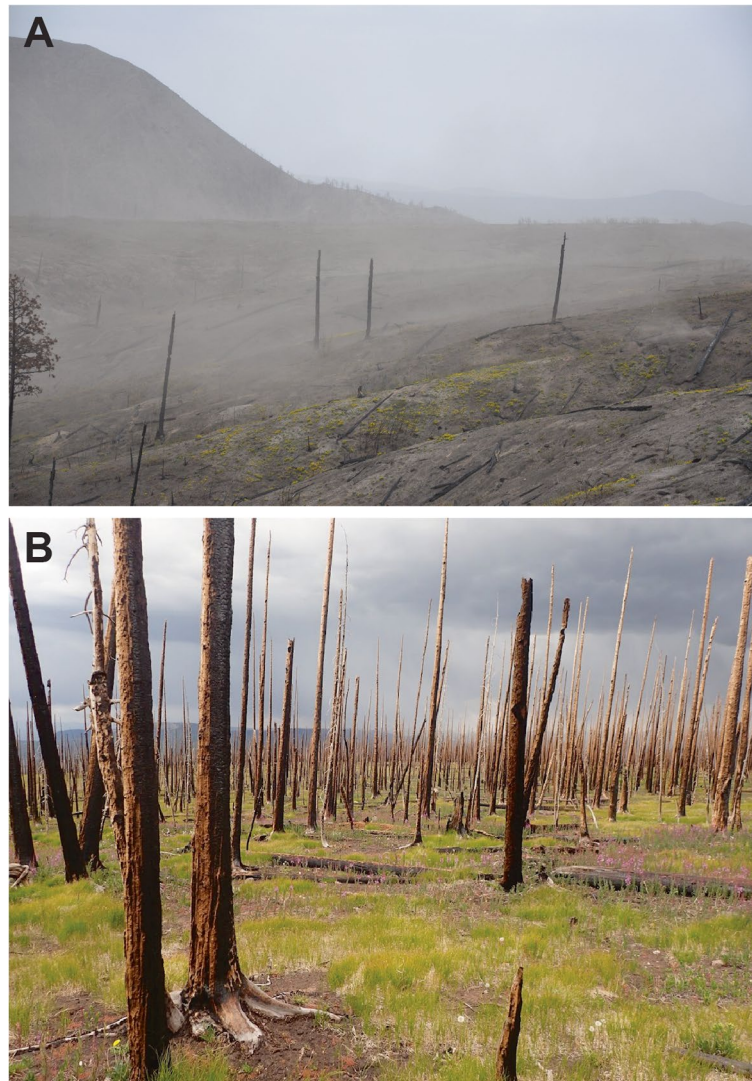


Fig. 3 Examples of observing change. **A** Light wind mobilizes ash and dried soil in a high-severity burn patch of the 2011 Las Conchas Fire, where it reburned an earlier high-severity patch. This photo was taken on April 26, 2012, nearly 1 year after the fire when only some herbaceous plants were growing (credit: Chris Guiterman). **B** Former Engelmann spruce-dominated forest impacted by spruce beetle and fire within the 2013 West Fork Complex Burn, Colorado. Matchstick-like snags are indicative that the trees were killed by beetles prior to the fire (credit: Jonathan Coop)

reproduction in some areas and a variety of herbaceous vegetation in others (Fig. 3B). That these VTC events occur in designated wilderness areas can limit management including fire suppression, prescribed fire, and tree planting. In one of the largest wildland-urban interface regions of the United States, the *Inner Coast Range of California* (#8), VTC has only recently emerged following the disruption of historical fire regimes and associated reduction in the spatial diversity of the grassland-woodland-forest mosaic. The devastating “wine country” wildfires in 2017 marked the return of fire to this coupled human-natural ecosystem. Some areas have now

experienced four fires in the last 5 years. Beyond losses to human life and property, the entire ecological mosaic has been affected, with major loss of chaparral communities, fundamentally changing the landscape to non-native grasslands and leaving human infrastructure vulnerable to flooding and debris flows.

Facilitating change

Facilitation of VTC is the least common management response documented in our study, though ideas of when, where, and how to direct changes are becoming clearer (Millar and Stephenson 2015). The facilitation

case studies we present include management actions that direct change knowingly but perhaps without the explicit intention of promoting type change. In the case of the *North Rim of the Grand Canyon in Arizona* (#9), fire managers successfully reintroduced fire in ponderosa pine forests following many decades of fire exclusion. However, with more recurrent fire activity, they noted higher-than-expected conifer mortality in surface fires, which is benefiting Gambel oak and slowly converting the forests to shrubby woodlands (Fig. 4A). Some of the small shrubland patches that are established in high-severity burn areas are expanding as large, downed fire-killed trees burn in subsequent fires with enough intensity to expand the shrubland gaps, sometimes merging into large patches. Frequent fire may be more in line with projected climate conditions but also threatens large, old trees. The management goal to maintain

fire as an ecological process (https://www.nps.gov/grca/learn/management/upload/grca_fmp.pdf) is promoting this ecological transition. In the *southern Sierra Nevada of California* (#10), a decade of drought and recurrent fires is rapidly removing conifers from commercial forest areas where thinning has reduced relative mortality but progressed the transition from conifer-dominated forests to oak- and hardwood-dominated woodlands (Fig. 4B). Now, unthinned areas are vulnerable to fire due to their composition of dense fire-intolerant tree species and heavy loading of drought-killed trees, but thinned stands dominated by oak trees are vulnerable to the advance of goldspotted oak borers. Finding a balance between these options is challenging, so managers are utilizing new decision support tools to guide post-fire recovery efforts and the facilitation of VTC in some areas to be used as fuel breaks in generating a landscape mosaic. Along the



Fig. 4 Examples of facilitating change. **A** Tree mortality of ponderosa pines following two high-severity fire events on the North Rim of the Grand Canyon, AZ. This expanding gap is now dominated by forbs and New Mexico locust with no pine regeneration (credit: Chris Marks). **B** Tree mortality following a multi-year drought in a pre-drought thinned ponderosa pine and black oak stand on the Sierra National Forest, southern Sierra Nevada, California. The foreground illustrates the current open stand conditions dominated by black oak and canyon live oak with an understory of mountain misery (*Chamaebatia foliolosa*) following the cutting and piling of dead conifers (mostly ponderosa pine and sugar pine). The background shows post-drought stand conditions prior to conifer removal (credit: Marc Meyer)

high summit of *Pinaleño Mountains in Arizona* (#11) spruce-fir (*Picea engelmannii* and *Abies lasiocarpa* var. *arizonica*) forests are critical habitat for the endangered Mount Graham red squirrel (*Tamiasciurus fremonti grahamensis*) (USFWS 2011) but were decimated by two fires in 2004 and 2016 (Merrick et al. 2021). Managers recognize that re-planting a spruce-fir forest will neither rapidly re-establish habitat nor be resilient and productive given the changing climate. They have therefore opted to plant a native, but more drought- and insect-resilient, mix of conifer species (including spruce and fir) that could, once mature, potentially aid in the return of the spruce-fir type. The key idea here is to help push the system in a trajectory of conifer forest, rather than shrub or grassland conditions.

VTC examples

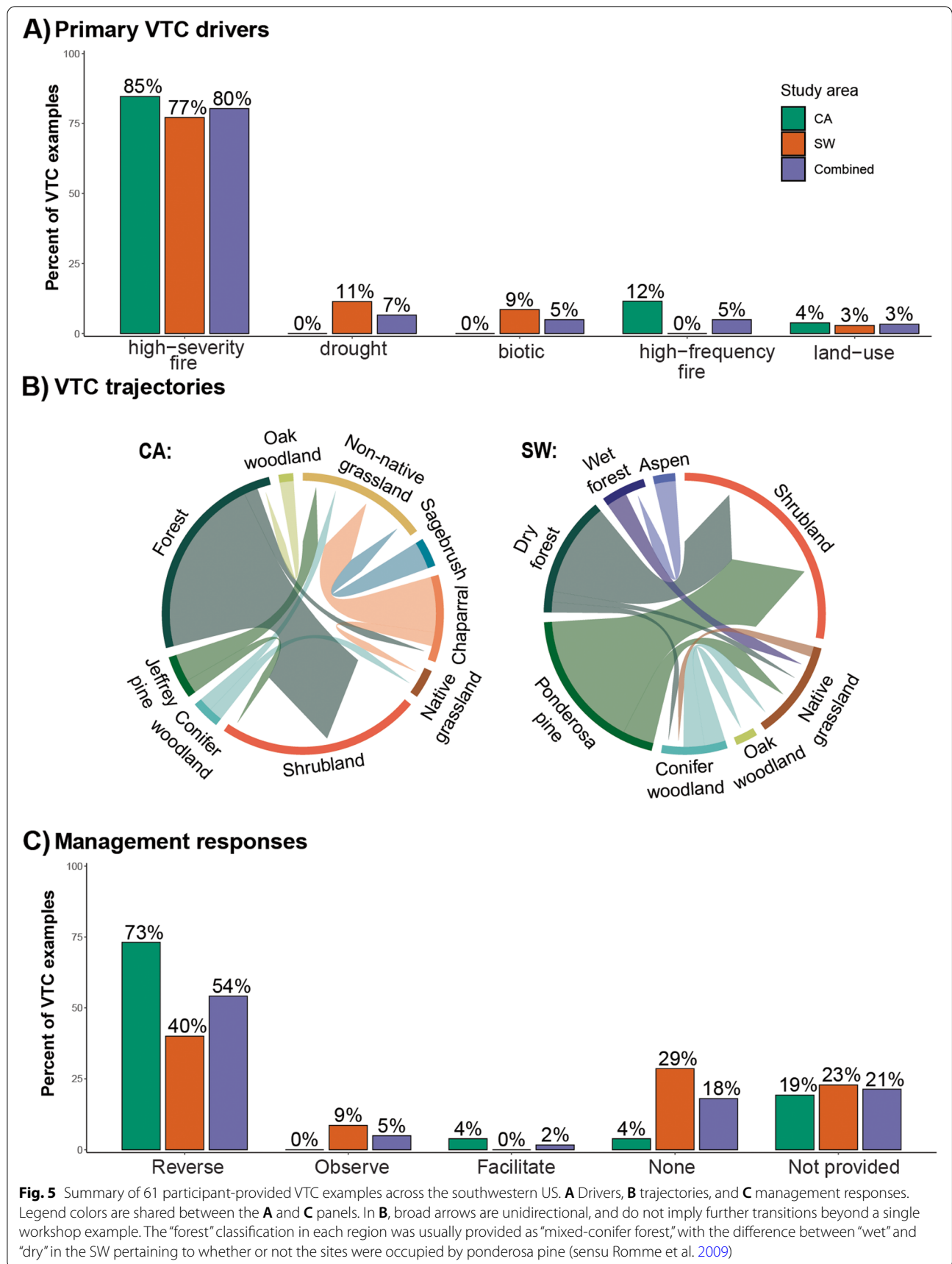
In order to capture the regional scope and diversity of VTC, workshop participants identified sites undergoing VTC on printed maps that we later geolocated in a geographic information system. Each workshop had a subregional focus (Fig. 1). The workshop in Tucson, AZ (March 2019) focused mainly on Arizona, New Mexico, and Colorado (Southwest (SW) study region). The workshop in Sacramento, CA (December 2019) focused on California and adjacent environments (CA study region). For each location they marked, participants described their observations on paper forms that included the (1) location of the VTC, (2) land ownership of the area, (3) ecosystem types before and after the VTC, (4) year of any precipitating event(s), (5) driving mechanism(s) of change, (6) species of interest in the area, and (7) management actions, if any, taken to address the VTC. We emphasize that these examples of VTC represent the site-specific knowledge and expert opinion of scientists and practitioners who attended the workshops and are not an attempt to identify or quantify the true extent of regional VTC. The examples were summarized in the context of two large-scale spatially explicit data sets, Monitoring Trends in Burn Severity (MTBS, Eidenshink et al. 2007) and the US Forest Service Activity Tracking System (FACTS) (<https://data.fs.usda.gov/geodata/edw/datasets.php>), to describe broad patterns in the VTC observations (see online [supplemental information](#) for details).

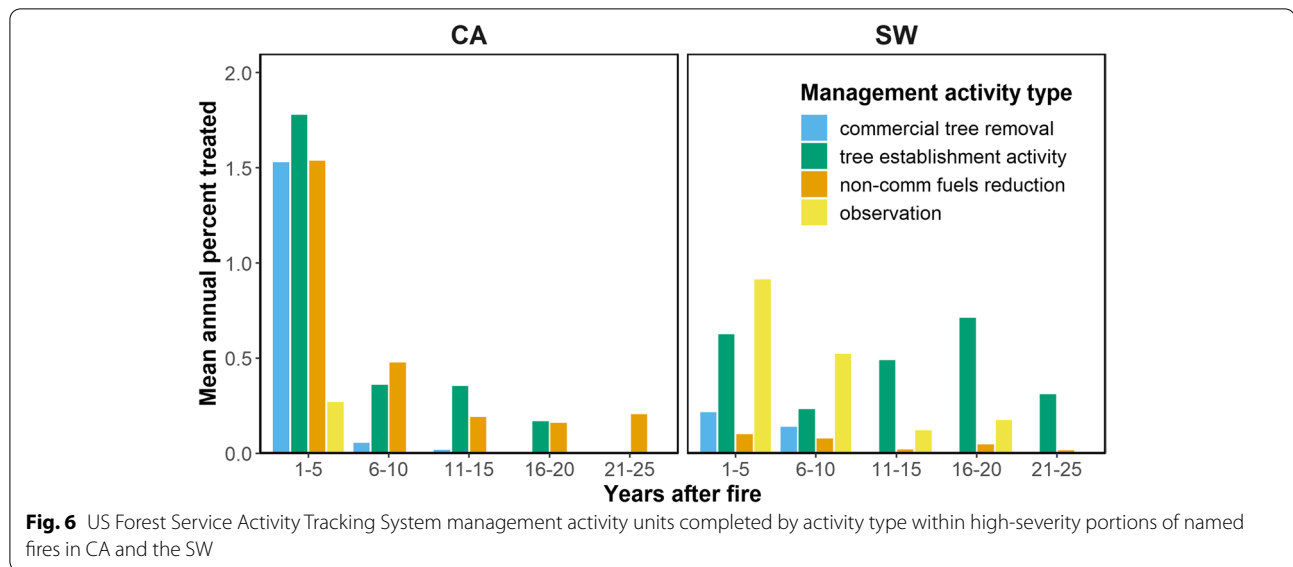
Workshop participants provided 61 examples of VTC across six southwestern US states (Fig. 1), with 26 in the CA study area and 35 in the SW (each example is provided in the online [Supplemental Table](#)). The vast majority (80%) of these examples related to high-severity fire (Fig. 5A). Drought, biotic agents, high-frequency fire, and land use each account for <10% of the identified VTC drivers. Some examples represent changes across vast areas that could not be accurately portrayed by our

approach. For example, within the land-use category, only a single record in southern CA describes widespread fuel breaks in which repeated disturbances including bulldozing, prescribed fire, herbicide applications, and mastication of vegetation have converted chaparral within the fire lines to herbaceous dominance, predominantly non-native grasses. Although these actions were intentional, they were not necessarily intended for the establishment of non-native vegetation.

Trajectories of VTC underscore the commonality of forest-to-shrubland transitions (Fig. 5B). In total, 59% of the examples include conversion to shrubland. In the SW, both ponderosa pine and dry mixed-conifer forests (which often include ponderosa pine, Romme et al. 2009), are seen to almost always transition to shrublands. In CA, 54% of the examples include the shrubland trajectory, predominantly resulting from fire-driven conversions of mixed-conifer and Jeffrey pine forests. Grasslands dominated by mostly native herbaceous vegetation are the next most common post-VTC type, with non-native grass making up 15% of the examples, all of which were reported in CA. This latter group includes a variety of pre-VTC vegetation communities such as chaparral, Jeffrey pine forest, and sagebrush.

Reversing change was the most common management response to VTC (Fig. 5C). The second most common response was either no management (often written as “none”) or was not provided. If we could not supplement the participant’s entry with information from FACTS, we report what the participants provided, leaving 13 examples in which a management action was not provided. There were three examples that included observing change, and one example (the fuel breaks described above) of facilitate change. These examples show that interventions to reverse change were more common in CA than in the SW, and by contrast, observing change was more common in the SW than in CA. These subregional differences were notable in our analysis of the FACTS data (Fig. 6), in which we explored 34 examples of VTC that were within patches of high-severity fire, as recorded in MTBS. We identified 55 high-severity burn areas over the 34 individual sites, suggesting that repeated high-severity fire may have been a factor in some examples of VTC. FACTS data show that in CA, most post-fire management interventions occur within 5 years of the fire and aim to reverse change (commercial tree removal, fuel reduction, and tree establishment). Little observation of change was recorded for CA, and none occurred after 5 years, whereas in the SW, observation was more common than tree removal or fuel reduction, and could last as long as 20 years post-fire. The rate of tree establishment dwindled in CA after 15 years post-fire, while it only increased in the SW through 20 years





post-fire. Across all of these management responses, however, the spatial coverage of treatments recorded in FACTS shows that less than 25% of individual high-severity burn areas saw any treatment.

Synthesis

Across the breadth of ecosystems represented in our case studies and VTC examples, we found that forests typically convert to shrubland, and chaparral or sagebrush communities convert to herblands, often dominated by non-native grasses. The post-fire types represent transitions to vegetative states that are shorter in height, better adapted to disturbance and drought, and, as more areas are affected, reduce landscape-scale diversity in ecological structure. Our findings emphasize that altered fire regime characteristics, including frequency and severity, are likely to generate novel transitions. In general, these processes increase overstory mortality among trees and chaparral, which is the key trigger of a state transition, especially in larger patches (Chambers et al. 2016; Falk et al. 2022). Other mortality agents, such as insect outbreaks, often in combination with fire, further promote transitions. Recovery to the initial state is likely to be inhibited by a hotter and drier climate (Davis et al. 2019; Stewart et al. 2020). When all of these factors align, as they have in recent decades across most of the Southwest, VTC is the likely outcome.

Once converted, new vegetative states are highly persistent. This underscores the need for management to consider undertaking preventive strategies that capitalize on the persistence mechanisms of intact vegetative types (Falk et al. 2022), if these are the desired long-term communities (see Matonis and Binkley 2018). Effective prevention strategies often include fuel reduction

and re-introduction of recurrent low-severity fire (Stoddard et al. 2021), which can be accomplished in diverse partnerships that promote important ecocultural products and values along with a suite of ecosystem services (Hessburg et al. 2021; case study #1). Treatments are ideally conducted at landscape scales, but smaller, targeted actions can be undertaken to promote refugia areas following future wildfires that would help recovery efforts by providing seed sources (Krawchuk et al. 2020).

While some prevention strategies are effective, they do not address all concerns regarding VTC. Participants in our workshops are frontline observers to ecological changes rarely witnessed until recent decades. As the case study descriptions echo, there is a palpable sense of futility when confronting the scale and uncertain ecological trajectories of VTC. Indeed, in many cases, little can be done to reverse changes wrought by multiple compounding disturbances and long-term drivers. The rapid and stubborn spread of non-native species further frustrates recovery and intervention strategies. This emphasizes the importance of management frameworks that have an option to accept rapid and profound change (Lynch et al. 2021) and calls on increasing research to evaluate a variety of approaches (Crausbay et al. 2021).

Reversing change is often resource intensive. To expand recovery efforts and maximize often limited resources, it may be critical for managers to prioritize particular sites. Recovery via planting conifers has received mixed success (Ouzts et al. 2015; case studies #2, 3, 11), and thus more focus is currently being placed on targeted planting operations that have the highest potential for survival through drought and subsequent fire (Dumroese et al. 2016; North et al. 2019). Recovery efforts will have to rely on appropriate seed sources and planting stock, but the

necessary infrastructure has declined in recent decades (Fargione et al. 2021), as has the availability of appropriate species. Opting to plant more drought-tolerant or more commercially-desired species could represent a choice to facilitate change rather than resist it (case study #3). Federal support and local efforts are needed to re-establish nursery production capacity, and doing so could present an opportunity to invest in underrepresented groups such as Native American communities and tribal forestry programs that have the capacity but may lack market demand to re-establish their nurseries. Open Source tools are also emerging that help to identify potential seed sources for planting operations (e.g., <https://seedlotselectiontool.org/sst/>, <https://climaterestorationtool.org/csrt/>) as well as where natural regeneration after disturbance may be insufficient (https://code.usgs.gov/werc/redwood_field_station/poscrptr) and when and where planting operations may be most efficacious (e.g., <https://reforestation.shinyapps.io/preset/>).

The option of observing change may be determined by a desire to “wait and see,” a lack of the resources needed to take more deliberate intervention measures to reverse change or by constraints in land designations, such as in wilderness areas. Uncertainties regarding unintended consequences of active intervention (e.g., moving towards “undesired” conditions, “sometimes doing something is worse than doing nothing”) may also delay or prevent other actions. Allowing managers time to observe change is a valid approach to informed adaptive management (Sagarin and Pauchard 2010; Halofsky et al. 2018; Chazdon et al. 2021), especially given highly variable seasonal climates of recent years. Observing an ecosystem’s trajectory and understanding the dynamics of the developing community will help managers gain a general sense of the probability of type conversion, and whether the site risks invasion by problematic non-native species. However, institutional constraints may limit the ability to experiment with different approaches, particularly with wildfire management (e.g., Abrams et al. 2021). For example, most agency mandates and funding streams are directed toward fire suppression rather than prevention or recovery, leading to a mismatch between policy directives and ecological needs in some cases. In other cases, the number of agency staff available to support fire prevention or recovery may be limited by budgetary constraints.

Choosing to facilitate or direct change depends on agency mandates, site objectives, individual managers’ risk tolerance, and values. While examples of and research on intentional on-the-ground facilitation of VTC are generally lacking to date, more flexibility in management directives would allow for opportunities to better understand the dynamics of novel systems (Millar

and Stephenson 2015). Findings from other efforts to facilitate change (e.g., assisted gene flow, assisted range expansion), while not specific to fire-driven VTC, may be useful for inspiration and lessons learned (McLane and Aitken 2012; McPherson et al. 2017; Richardson and Chaney 2018; Crotteau et al. 2019).

Trepidation in confronting the scale of VTC stems in part from the uncertainty of its trajectory given slow and variable recovery processes. Insights from Indigenous knowledge can aid in understanding the degree of a possible departure from historical ranges of variability, whether changes are undesirable from an ecological perspective, and options for management that proved effective in the past (Lake et al. 2017). Paleocological and historical studies are helpful in gauging the long-term dynamics and persistence of various ecological communities (Jackson 2012). Our understanding of the mechanisms and drivers of VTC is improving apace, with critical reviews on resilience and its properties (Falk et al. 2019; Syphard et al. 2019; Coop et al. 2020; Falk et al. 2022) that provide a basis for comparison among events, and a focused language by which managers can compare events and areas (Stevens et al. 2021). Efforts are also underway to estimate landscape resilience or lack thereof, and thus the probability of VTC ahead of disturbance (Walker et al. 2018; Marshall and Falk 2020).

As management paradigms shift to accommodate impending change (e.g., Truitt et al. 2015; Schuurman et al. 2020), decisions around whether and how to accept or direct change will require new datasets and detailed models of plausible future ecological scenarios. Defining “desired conditions” may necessitate new models of collaboration that deeply engage stakeholders including local communities, tribes, and the broader public to better incorporate social and economic considerations in ecological management discussions. Manager-scientist collaborations such as the Fire Science Exchange Networks (https://www.firescience.gov/JFSP_exchanges.cfm) provide opportunities for workshops and field gatherings, peer-to-peer efforts such as the Burned Area Learning Network (<https://www.conservationgateway.org/ConservationPractices/FireLandscapes/FireLearningNetwork/RegionalNetworks/Pages/BALN.aspx>), and regional and place-based nongovernmental group initiatives help to promote awareness and readiness for VTC events. These efforts are changing the perceptions of managers, scientists, and the public, helping to incorporate VTC into the planning and decision making of agencies and land managers as they strive for “desired conditions” in a changing climate. Developing and assessing the capacity for management to achieve these conditions will require abundant experimentation within a co-production framework and social license for less-than-certain success.

Opening the door to accepting and directing VTC has potentially far-reaching and long-lasting implications for species, ecosystems, and society. Managing for change represents a potentially dramatic departure from traditional land management philosophy, especially in areas designated as natural areas or wilderness. Engaging with VTC may require more intensive intervention in ecosystem processes in many cases, but foundational principles for how to do this do not exist as yet. New and shared ethical frameworks drawing on science, Indigenous knowledge, and social consensus will be needed to guide this transition.

Future directions

VTC is among the most pressing issues for ecosystem management in the southwestern United States. Although the phenomenon eludes a simple definition (van Mantgem et al. 2020), land managers “know it when they see it,” and there is a strong sense of alarm at what they have been witnessing in recent years. The experiences and stories captured in 11 case studies presented here underscore that VTC is occurring at broad spatial and temporal scales (e.g., large patches to regional ecological ranges, from decadal land-use changes to rapid post-fire transitions) across most southwestern forest and woodland types to grasslands, shrublands, and chaparral. The rising sentiment among many managers appears to be that VTC at some scales and across many sites is a foregone conclusion following many high-severity fires in the study region. As VTC areas grow larger and more common, managers will increasingly need to shift their focus from persistence measures to recovery efforts in type-converted areas (Falk 2016). And as our collective understanding of VTC drivers, trajectories, and persistence mechanisms grow, options for its management will expand. Some may prove to be ineffective, such as traditional plantation layouts in large patches far from parent trees, while others may emerge that provide multiple benefits but might be considered acceptance or facilitation of VTC by current standards. More systematic collection and analyses of observations and on-the-ground experiences will be important to provide clarity and direction for research efforts that will help guide management. Land managers, practitioners, and scientists share many of the same trepidations regarding VTC, and the pace at which land management agencies are adapting to current conditions, but may also find strength in the collective experience and freedom to discuss experiences. Future adaptive management of VTC-prone areas and areas that are undergoing VTC depends on co-production and collaboration among managers, scientists, and stakeholders, particularly as we contend with rapid environmental changes.

Abbreviations

VTC: Vegetative type conversion; RAD: Resist-adapt-direct management framework; SW: Southwestern US study area, encompassing Arizona, Colorado, and New Mexico; CA: California and adjacent ecosystems study area; MTBS: Monitoring Trends in Burn Severity data set; FACTS: US Forest Service Activity Tracking System.

Supplementary Information

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Additional file 1. Supplemental text describing 11 case studies and methods used to evaluate the VTC examples.

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Authors' contributions

PvM, RMG, JEK, and DAF acquired funding. CHG, DAF, PvM, RMG, and LAEM designed the study. CHG, JJB, RMG, and LAEM carried out the research. CHG, DAF, PvM, JEK, JJB, and RMG interpreted the findings and wrote the initial draft. CHG, ACC, JDC, PJF, CH, RKH, AML, CM, MDM, HS, and AHT wrote the case studies. All authors validated the findings, contributed to revisions, and read and approved the manuscript.

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Availability of data and materials

All data used in this manuscript was derived from publicly available sources. Materials consisting of observations from the workshops are provided in SI Table 1.

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

Consent for publication not applicable as we did not use data from individual people.

Competing interests

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

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Supplemental Information for:

Vegetation type conversion in the US Southwest: Frontline observations and management responses

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Case studies

Reversing Change

Managers may act to reverse change by restoring pre-fire conditions or managing recovery such that the affected ecosystem is brought to a recognizable – and ideally more resilient – composition and structure.

1. Klamath Reservation, southern Oregon

Exclusion of the once abundant influence of low- and moderate-severity fire contributed to the loss of meadow and woodland ecosystems (Reilly et al. 2017; Matonis and Binkley 2018; Hessburg et al. 2019). In the central Oregon Pumice plateau ecoregion (Omernik and Griffith 2014), lodgepole pine (*Pinus contorta*) cover expanded during fire exclusion and now dominates areas historically maintained by frequent fire as wet meadows and aspen (*Populus tremuloides*) stands (Fig. S1; Seager et al. 2013b; Haggmann et al. 2019). Biodiversity hot spots like these can provide more habitat value than the surrounding conifer forests (DeByle and Winokur 1985). They also have high ecocultural value and contribute substantially to food security (Long and Lake 2018; Sowerwine et al. 2019). Aspen, for example, provides a protein-rich food source for ungulates (Seager et al. 2013a).

On the former Klamath Reservation, frequent low-severity fire historically maintained low-density forests dominated by large and old fire- and drought-tolerant conifers upslope of the drainage areas that supported wet meadows and aspen stands (Haggmann et al. 2019). Over more than a century of fire exclusion, increasing tree cover contributed to changing hydrological regimes and enabled lodgepole pine establishment on sites that had been too wet to support them prior to fire exclusion. Trees with high water demands, young trees and shade-tolerant species with deep crowns, now dominate forests that once had open-canopies composed of fire- and drought tolerant trees (Haggmann et al. 2013, 2019).

Climate- and wildfire-adaptation strategies on this dry forest landscape require restoration of frequent low-severity fire as a process that restores and maintains resistance to severe fire and drought (Merschel et al. 2021). Reducing forest cover with fire and silvicultural treatments can make more water available to remaining over- and understory vegetation as well as downslope meadow ecosystems (Boisramé et al. 2017; Saksa et al. 2017; Rakhmatulina et al. 2021). Despite substantial alteration of landscape hydrology through the drainage of marshes and building of dams and roads, the removal of young conifers from the periphery of aspen stands results in vigorous growth and expansion of aspen suckers and overstory trees (Seager 2017).

Increasing drought places additional stresses on social and ecological systems competing for scarce water resources (Crausbay et al. 2020). On-going restoration projects in the Klamath reservation forest are recovering characteristic spatial patterns of forest structure and composition and reversing type conversions associated with fire exclusion. These efforts aid restoration of the hydrology and biodiversity of meadow and riparian ecosystems. The Klamath Tribes are committed to the development of ecological research and tribal resources to inform and support stewardship of ecocultural values and their community (Hatcher et al. 2017). They are the senior partner in a stewardship agreement that covers most of the Fremont-Winema NF and is focused on implementing the Klamath Tribes' forest restoration strategy (Hatcher et al. 2017).



Fig. S1. Dense lodgepole pine cover encroaches on nonforest ecosystems historically maintained by frequent fire. After more than a century of fire exclusion, forests upslope of these drainage areas are also denser and strongly dominated by young and shade-tolerant trees with lower water use efficiency than the old, fire- and drought-tolerant ponderosa that historically dominated this landscape. Both fire exclusion and diversion of water from downslope ecosystems facilitate lodgepole encroachment on wet meadows and aspen stands. Photos by Trent Seager and Andrew Merschel.

2. Southern Front Range, Colorado

Until the devastating fire season of 2020 spawned the Pine Gulch, Cameron Peak, and East Troublesome Fires, the 2002 Hayman Fire had the dubious distinction of being the largest and most severe wildfire known to burn in Colorado. Driven by high winds, severe drought, and overly-abundant fuels, the Hayman Fire burned across more than 52,000 ha of predominately ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) forest (Graham 2003). Around 70% of the fire footprint burned with high severity, greatly compromising natural forest recovery due to a lack of surviving seed trees (Chambers et al. 2016). Although naturally-regenerating trees are sparse in high-severity patches, grasses, sedges, forbs, and shrubs are abundant and have formed diverse, productive, and predominately native plant communities (Fig. S2; Fornwalt and Kaufmann 2014; Abella and Fornwalt 2015).

Much of the area impacted by the Hayman Fire is managed by the Pike National Forest, which ranks third among US National Forests for recreational visits and supplies water to more than

60% of Denver's residents. Tree planting to promote the recovery of ponderosa pine and Douglas-fir forests, and in turn sustain water supply and other valued ecosystem services became a top management priority. Together with partners including the National Forest Foundation (<https://www.nationalforests.org>), Denver Water (<https://www.denverwater.org>), Vail Resorts (<http://www.vailresorts.com/Corp/index.aspx>), the National Arbor Day Foundation (<https://www.arborday.org>), and the Coalition for the Upper South Platte (<https://cusp.ws>), the Pike National Forest began planting tree seedlings in the footprint of the Hayman Fire in 2004. More than two million tree seedlings have been planted across more than 6,000 ha to date.

Despite several years of severe drought during the post-fire period, three-year survival rates of planted seedlings have been high, averaging around 75% (Pike National Forest, *unpublished data*) (Fig. 2A in main text). These survival rates stand in stark contrast to those observed in post-fire planting units elsewhere in the Southwest (Ouzts et al. 2015). Managers have anecdotally credited the high survival rates to a host of factors. For example, they have prioritized the most productive sites for planting, such as those at higher elevations and on more northerly aspects. They have also conducted most planting operations in April, when soil moisture is relatively high, and they have relied heavily on the use of coarse wood or other objects to shade tree seedlings. Research is underway to quantitatively evaluate the role these and other factors play in the survival of tree seedlings planted in the Hayman Fire and other southwestern wildfires.



Fig. S2. Recovery of native plant communities following the 2002 Hayman Fire (see Fig. 2A in the main text for a photo of the recovery of planted pine). Photo by Paula Fornwalt.

3. Laguna Mountain, California

Southern California experienced a major drought between 1999 and 2002 with only 50% of normal precipitation falling in eastern San Diego County. This event, which killed about 25% of the Jeffrey pines (*P. jeffreyi*) on Laguna Mountain in the Cleveland National Forest (Freeman et al. 2017), was followed by the massive 2003 Cedar Fire, which burned over 109,000 ha in San Diego County, including northern portions of Laguna Mountain as well as 98% of the mixed-conifer forest in nearby Cuyamaca Rancho State Park (Franklin and Bergman 2011). The Cedar Fire was followed by a nearly complete conifer regeneration failure. In response, Forest Service managers attempted multiple replantings of Jeffrey pine, but efforts failed due to a series of hot and dry years that caused high seedling mortality (Safford and Vallejo 2019). Most of the mature Jeffrey pine-black oak (*Quercus kelloggii*) forest burned by the Cedar Fire has since transitioned to open shrubland and grassland with scattered black oak and Coulter pine (*P. coulteri*), a lower-elevation pine that is more drought-tolerant than Jeffrey pine and adapted to reproduction after severe fire. Since 2000, mean annual temperatures in the mountains of eastern San Diego County have increased by 1.2°C and annual precipitation has decreased by 25%. As a result, outbreaks of Jeffrey pine beetle (*Dendroctonus jeffreyi*) have continued to occur in the remaining forest, especially after very dry years in 2013 and 2014. In 2012, the very severe Chariot Fire burned another portion of the Laguna Mountain forest. Given climate change trends and the progressive loss of Jeffrey pine on the mountain, managers and the public were becoming resigned to the eventual replacement of the conifer-dominated forest by a black oak woodland. However, in an example of compounding disturbance events, the arrival of the goldspotted oak borer (*Agrilus auroguttatus*) in Southern California decimated mature oaks in this portion of the Peninsular Range and the species is rapidly moving northward (Safford and Vallejo 2019). Recent Forest Service aerial surveys document tens-of-thousands of dead oaks and ongoing Jeffrey pine mortality (Fig. 2B in main text). Forest management on Laguna Mountain continues to focus on reducing stand density through targeted thinning and prescribed fire, which has reduced both pine mortality (Freeman et al. 2017) and fuel loads. The failed planting efforts made it clear that climatic conditions on the mountain are no longer favorable for juvenile Jeffrey pine, and the increasing mortality of adult pines and oaks is casting serious doubt on the long-term resilience of montane forest in the area. In response, the U.S. Forest Service and partners at the U.S. Geological Survey Southwest Climate Adaptation Science Center, San Diego State University (<https://www.sdsu.edu>), and the Climate Science Alliance (<https://www.climatesciencealliance.org>) have begun the development of a montane forest conservation strategy for southern California (<https://www.climatesciencealliance.org/southern-forests>) whose goals are to better understand the interacting threats facing montane forests; identify opportunities and strategies for increasing forest resilience; and develop, prioritize, and promote collaborative, multi-partner solutions.

Observing change

Managers may choose to observe change by exercising patience and monitoring the system and its post-disturbance trajectory.

4. Eastern Jemez Mountains, New Mexico

The 2011 Las Conchas Fire burned 63,250 ha on the eastern side of Jemez Mountains near Los Alamos, New Mexico. Extreme, wind-driven fire behavior reburned many moderate to high-

severity patches from prior fires dating back to the late 1970s (Coop et al. 2016), and through areas that had seen high tree mortality in the droughts of the 1950s and the 1990s through the 2000s (Breshears et al. 2005; Allen et al. 2015). These prior disturbances created a patchy matrix of remnant stands of ponderosa pine and other dry mixed conifer forests with numerous patches of Gambel oak (*Q. gambelii*) shrubland. Over large areas, the Las Conchas Fire caused complete mortality of aboveground vegetation and entirely consumed standing and down dead trees. The combination of fires created two shrubland patches, dominated by New Mexico locust (*Robinia neomexicana*) and Gambel oak, each roughly 10,000 ha in size, in areas once occupied by dry-conifer forests (Fig. 3A in main text). This tripled the area of shrubland in the Jemez Mountains compared to historical estimates (Guiterman et al. 2018). Fire managers in the region consider recovery efforts in the Las Conchas footprint to be highly challenging, and in many areas the fire appears to have reinforced earlier forest to shrublands and grassland VTCs (Coop et al. 2016). The scale of the burn area, recurring warmer droughts, planting materials supply limitations, and concern regarding the timing and severity of the next fire are all obstacles to large-scale reforestation.

Effective restoration treatments may need to coincide with a period of a year or longer of high precipitation and the potential for future fires to reduce heavy fuels that would otherwise threaten regeneration. Work by the US Forest Service is ongoing to protect existing conifer stands that survived these fires from future severe fire. Subsequent fires may add additional heterogeneity to the landscape in a way that is desirable to long-term management goals.

In 2016, the East Jemez Landscape Futures (<https://www.nmconservation.org/field-notes/2018/12/6/east-jemez-landscape-futures>) as launched to aid in organizing stakeholders, local communities, and land management agencies in managing the area. Through interviews and meetings, the organization completed a needs assessment (Stortz et al. 2017) that outlines the impacts, challenges, needs, and opportunities of the eastern Jemez Mountains as well as a diversity of perspectives regarding the forests and recent alterations by fire, drought, and post-fire floods. In 2019, the group received funding to complete a collaborative restoration strategy across the burned areas. For part of the strategy, the group is utilizing the RAD framework (Schuurman et al. 2020) to determine where and how to intervene with management action and when to allow post-disturbance processes to continue with little human influence. Because of the participation of multiple Native American tribes, federal agencies, researchers, and local communities, the place-based restoration strategies are rooted in sustaining vibrant culture and adapting ecosystems to a warming climate.

5. Devils Postpile National Monument, California

On August 20, 1992, lightning ignited the Rainbow Fire that burned 3,378 ha in the central Sierra Nevada, including 84% of Devils Postpile National Monument (DEPO). Large areas of the burn within and outside of the monument resulted in high-severity patches with 90-100% tree mortality (across 25% of the monument). Twenty-five years later, many of the high-severity burn patches, once conifer forest, are still shrublands with little or no conifer recovery. In contrast, areas that burned at low to moderate fire severity, or where there is a local seed source, have considerable post-fire tree regeneration (Caprio and Webster 2009). Studies of fire effects and fire history within the monument indicate the current fire regime is outside its historic range (NPS 2017). Fire was excluded for over 100 years prior to the Rainbow Fire with pre-exclusion mean fire intervals (MFI) of 14 to 18 years (Caprio et al. 2006).

Although the extent of the Rainbow Fire is relatively small compared to the massive fires that have occurred in the southern Sierra Nevada over the last 10 years (including the Rim 2013, Creek 2020, and Castle 2020), it provides a longer-term perspective on post-fire recovery (Fig. S3). Pre-fire tree densities (pole and overstory), generally around 1,000 to 1,300 trees ha⁻¹, were reduced to 930 trees ha⁻¹ at low-severity sites, 140 trees ha⁻¹ at moderate-severity, and 5 trees ha⁻¹ in high-severity sites (Caprio et al. 2006). Dense shrub cover (primarily *Ceanothus cordulatus*) dominates high severity sites, increasing from 34% cover in 2004 to 48% in 2013, compared to <5% in low/moderate burn severity areas. There is plentiful conifer regeneration in areas under or adjacent to surviving overstory trees, but it is limited in areas >100 m from surviving trees, most likely due to a limited seed source. Seedling and sapling densities post-fire averaged 2,080 ha⁻¹ at unburned sites, 3,048 ha⁻¹ at low/moderate severity sites, 934 ha⁻¹ at high-severity sites <100 m from surviving trees, and 54 ha⁻¹ at high-severity sites >100 m from surviving trees. Nearly all tree regeneration in high-severity areas is Jeffrey pine that germinated one- or two-years post-fire from seed caches. These trees could eventually be a future seed source, except they are at risk to a subsequent fire (such as the Creek Fire that threatened the monument in 2020), which would no doubt result in further conifer exclusion. The potential effects of repeated burns within these high-severity patches and how they affect vegetation long-term may need to be considered in the management of these areas.

The current DEPO Fire Management Plan recognizes the need to restore fire to areas where fire has been excluded and to maintain fire in areas previously treated or burned by low/moderate fire during the Rainbow Fire. Additionally, it proposes the potential use of experimental burns in high-severity patches dominated by shrubs to determine whether a low-intensity prescribed fire could break up fuels and shrub cover to reduce the severity of a unwanted wildfire while also promoting forest regeneration.



Fig. S3. High-severity patch of the Rainbow Fire in the southern portion of Devils Postpile National Monument and adjacent Inyo National Forest (top left). Shrub-dominated high-severity patch with little tree regeneration in 2018, twenty-six-years post-fire (bottom left). Moderate tree regeneration in high-severity patch with a nearby seed source (right). Photos by Anthony Caprio.

6. Lassen Volcanic National Park, California

Fire management in Lassen Volcanic National Park emphasized suppression until the 1980s when prescribed fire and wildfires burning under moderate conditions began to be used at larger scales to restore fire as an ecosystem process (Taylor 2000). However, fuel buildup from a century of fire exclusion has eroded the historical resistance of forests to high-intensity fire behavior, and as the landscape transitions back to an active fire regime, the behavior and associated fire severity of each new fire strongly affects subsequent fire severity (Harris et al. 2021).

Increased surface and canopy fuels in mixed conifer forests that historically burned frequently at low-moderate severity can now burn at high severity (Fig. S4). Areas where prescribed fire has been applied reduce severity of wildfires that intercept them even if burns occur under extreme

conditions (Harris et al. 2021). In severely burned areas, heat from intense fire and loss of the forest canopy triggers germination of shrub seeds stored in the soil, and a dense shrub canopy can develop in a few years (Fig. S4). Competition from shrubs greatly reduces tree regeneration and re-establishment of forest, especially in large high-severity patches. Thus, montane chaparral may emerge as an alternative stable state to forest if the time needed for forests to develop exceeds the fire return interval (Airey Lauvaux et al. 2016).

In lodgepole pine forests, which dominate wet lowlands with cold air drainage, the drivers of type conversion are more complex. Lodgepole stands historically burned less frequently than mixed conifer forests and with more variable severity. Following wildfire in 1984, the system seemed to be operating normally, with regeneration abundance being controlled mainly by distance to surviving parent trees and low snowpack in years following the event (Pierce and Taylor 2011). These legacy effects were amplified in a subsequent high-severity fire in 2012 (Fig. S5). Regeneration was abundant in areas initially burned in 1984 at low severity, but regeneration was very low in areas twice burned at high-severity. These areas could convert to woodland or non-forest with subsequent fires (Harris et al. 2020).

These fire-vegetation interactions underscore the importance of legacy effects on the drivers and trajectory of VTC. They also underscore the advantage of low- to moderate-severity fire in maintaining forest resilience as landscapes transition back to an active fire regime, especially as temperatures warm. To reduce the potential for VTC from recurrent high-severity fires, managers are developing plans to use prescribed fires (via ground and aerial ignitions) to reduce the continuity and cover of shrub fuels and large downed fuels in high-severity burn patches. The goal is to increase heterogeneity of the fuel mosaic to reduce potential for severe fire effects in a subsequent fire that would eliminate the bulk of tree regeneration and surviving mature trees.



Fig. S4. Repeat photographs of a mixed Jeffrey pine-white fir forest in Lassen Volcanic National Park showing the effects of fire suppression and a recent high severity fire. Fires burned frequently in this forest type until the early 1900s (Taylor, 2000). Vegetation changes since 1925 increased surface and canopy fuels and the forest burned at high severity in 2012 killing the forest canopy. Montane chaparral now dominates the site. The 1925 photo is from the A.E. Wieslander collection (<https://guides.lib.berkeley.edu/Wieslander>), with subsequent photographs by Alan Taylor.

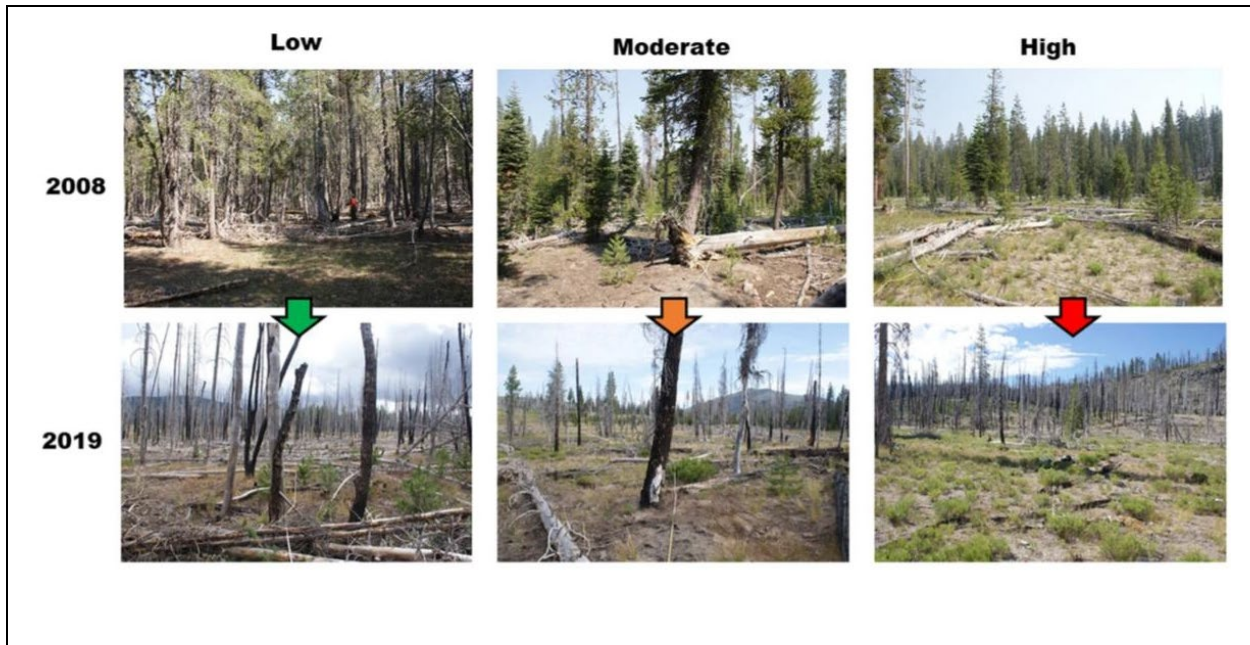


Fig. S5. Paired photographs taken in 2008 and 2019 in three plots representative of low, moderate and high burn severity in the 1984 Badger Fire which all reburned at high severity in the 2012 Reading Fire. Photo credits, Alan Taylor.

7. San Juan Mountains, Colorado

Compound disturbance may enhance the likelihood of persistent forest conversion to alternate states by weakening different mechanisms of resilience. In the early 2000s, an outbreak of spruce beetle (*D. rufipennis*) rapidly expanded across subalpine forests of the San Juan Mountains of southern Colorado under warm and dry conditions. Beetle mortality of mature Engelmann spruce (*Picea engelmannii*) was unusually severe, exceeding 99% at many locations. However, abundant advanced regeneration by spruce, subalpine fir (*Abies lasiocarpa*) and aspen occupied the understories of impacted forests (Savage et al. 2017).

In 2013, the 44,000-ha West Fork Complex Fire burned across beetle-altered landscapes in the San Juan Mountains, entirely removing tree regeneration and surviving trees from extensive patches (Fig. 3B in main text). Under normal circumstances, unburned spruce stands at forest edges and refugia within the burn could be expected to serve as seed sources that promote abundant spruce regeneration within two to three decades, up to and well beyond 100 m into burned patches (Coop et al. 2010). However, refugia are sparse within West Fork Complex and lack mature, seed-bearing spruce trees, which were killed by beetles. Where aspens occurred prior to these disturbances, resprouting is abundant and landscapes appear poised to shift to a substantial period of aspen dominance (Andrus et al. 2021). Aspen regeneration from seed is also occurring, albeit sparsely, in some severely burned areas, including sites above its former upper elevational limit (Nigro et al. 2021), illustrating the capacity for severe disturbance to catalyze

vegetation shifts aligned with warming. However, many large patches (>>1,000 ha) currently lack any tree regeneration, are distant from surviving tree seed sources, and are currently only sparsely covered by a depauperate flora consisting of few resprouting or weedy understory herbs and shrubs such as common dandelion (*Taraxacum officinale*), dryspike sedge (*Carex siccata*), and whortleberry (*Vaccinium myrtilis*) (J.D. Coop, *unpublished data*). The future of these landscapes is highly uncertain, but a return to pre-fire conditions appears unlikely for centuries.

Because much of the West Fork Complex occurs in designated roadless areas in the Weminuche Wilderness, management options are few and raise important questions about the role of wilderness as changing conditions increasingly imperil natural ecological function. For example, would extensive reforestation efforts (e.g., via drone seeding) be an appropriate use of limited resources outside of the commercial timber base? Further, what level of fire management is appropriate in designated wilderness where natural processes are given precedence? Currently, fire suppression in wilderness is accepted, yet prescribed fire is controversial. Could patches of prescribed mixed- or high-severity fire under moderate conditions create heterogenous wilderness landscape mosaics more resistant and resilient to climate change and attendant amplification of disturbance regimes? These and other questions could benefit from informed deliberation by managers, researchers, and the public in a future that appears increasingly prone to disturbance-driven type conversion even in deeply revered and protected landscapes.

8. Inner Coast Range, northern California

Over the last decade the center of mass for wildfire and wildfire-driven structure loss and human fatalities in California has shifted from southern to northern California. One of the most affected areas has been the Coast Ranges north and northeast of the San Francisco Bay Area. The rapid increase in fire frequency and burned area in the region has accompanied massive expansion of exurban housing into landscapes under high fire risk and the contemporaneous expansion of the high voltage power grid, a major ignition source (Syphard et al. 2019). Vegetation in the region is a mosaic of oak-dominated forest and woodland, chaparral, and grassland. The climate is moist and cool in the winter and spring, and hot and dry in the summer and fall — perfect conditions to grow fuel and subsequently burn it.

Human fire use was a major driver of vegetation distribution in the region before Euroamerican arrival. The loss of Native American populations and cultural fire use, the cessation of late 19th and early 20th century livestock grazing, and the imposition of fire suppression resulted in the expansion of woody vegetation and contraction of open grass-dominated habitats (Russell and McBride 2003). Since the late 1800s most of the region has been developed or farmed, and wildfires became a rare and almost forgotten part of the landscape. Recently, fire has returned to the region with a vengeance. In the summer of 2015, four separate fires burned 72,000 ha and destroyed more than 2,000 homes and killed four people when they burned through multiple towns (<https://www.fire.ca.gov/incidents/2015>). The “wine country” wildfires of October 2017 killed 40 people and destroyed more than 8,000 structures, catching the bulk of the population by surprise even though large fires had occurred in the same landscapes — when human population and housing density was much lower — in earlier decades (<https://www.fire.ca.gov/incidents/2017>). In 2020 multiple large fires affected the region again, burning 146,000 ha and resulting in six deaths and 1,490 destroyed structures (<https://www.fire.ca.gov/incidents/2020>).

The largest of the 2020 events was the Hennessy Fire, which burned more than 123,000 ha between August and October. Almost 40% of the landscape burned by the Hennessy Fire had burned in the previous 10 years, including two areas that have become foci of high-frequency burning, the canyons of Cache and Putah Creeks, which drain the inner Coast Ranges to the Sacramento River. Both drainages support perennial streams, reservoirs, and lakes that are a major draw for dry-season recreation, and both canyons include busy state highways that link the Central Valley to Napa and Lake Counties. As a result, the canyons experience multiple human fire ignitions every year (<https://www.fire.ca.gov/incidents>). The climate is also drying rapidly in the region, with 20-50% losses in mean annual precipitation since 1995 reported at most meteorological stations, and mean monthly temperatures have risen by an average of about 0.6° over the same period (WRCC 2021). Figures S6 and S7 focus on the Putah Creek Canyon, just below the Berryessa Dam. Until the early 2000s, most of the map area had experienced no fire since at least the 1950s, and in many cases since the beginning of the 20th century. The area denoted by the arrow in Fig. S6 is shown in aerial imagery in Fig. S7. This area burned in 1959 for the first recorded time in the 20th century, then subsequently in 2007, 2014, 2016, 2018 and 2020. The imagery highlights how five fires in 13 years (or four fires in six years) can change a landscape. The extent of chaparral in Fig. S7 has decreased by more than 60% since 1993, and the density of trees in the areas of oak-dominated forest and savanna has noticeably diminished. Field sampling in the remaining patches of chaparral in the spring of 2021 showed that populations of obligate seeding shrub species such as *C. cuneatus*, *C. oliganthus*, and *Arctostaphylos manzanita* — common in nearby areas where burning has been less frequent — have been locally extirpated (Hugh Safford, Personal observation). In addition, densities of disturbance-tolerating low-statured resprouting shrubs that are relatively rare in mature chaparral (e.g., *Lepechinia calycina*, *Eriodictyon californicum*) have greatly increased and these species now dominate much of the remaining area of chaparral (Hugh Safford, Personal observation).

The loss of woody cover on the steep slopes above Putah and Cache Creeks threatens two important state highways with soil ravel and debris flows, but low precipitation after the Hennessy Fire was a lucky break in this respect. However, 2020 was the second driest since 1943 and during field sampling it was noticed that many resprouts from burned chaparral shrubs, especially *Adenostoma fasciculatum*, were dying. The ongoing type conversion of native-dominated chaparral to alien-dominated annual grassland will represent a major ecosystem change in the region, but there is little consensus on what to do aside from redoubling fire prevention and suppression efforts and periodically repairing roads. Interactions between expanding non-native grassland, increasing exurban housing and ignitions, warming temperatures, and drying winters will likely greatly change the California landscape in coming years.

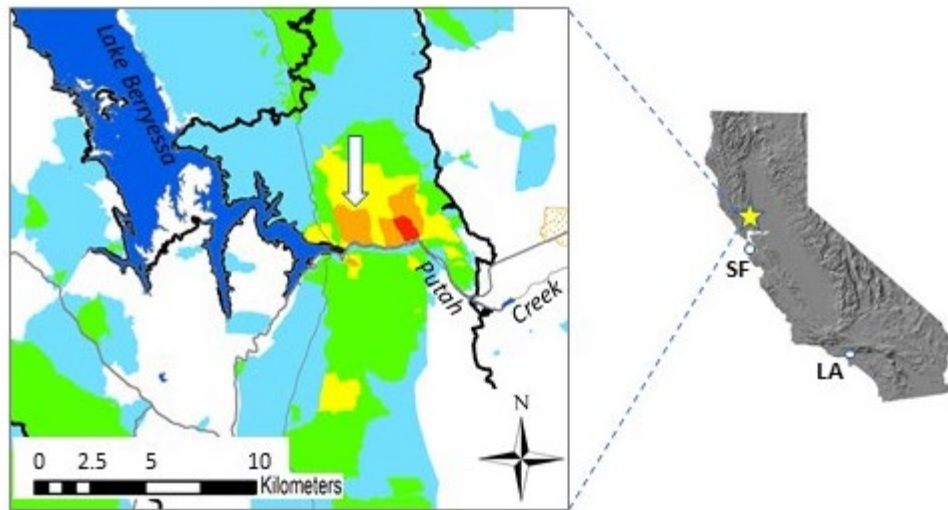


Fig. S6. Recent fire history of middle reaches of Putah Creek watershed, inner Coast Ranges, California. Easternmost embayment of Lake Berryessa is 85 km NE of San Francisco and 53 km W of Sacramento. Black line represents outer perimeter of 2020 Hennessy Fire (i.e., entire map was burned except for area east of NE shore of Lake Berryessa and eastern 1/5 of map). Colors represent fire history over the 40 years before the occurrence of the Hennessy Fire (i.e., Hennessy Fire is not counted in the fire history). Blue = 1 fire, Green = 2, yellow = 3, brown = 4, red = 5. Almost all of the burning in the figure has occurred since 2005. Arrow denotes area of Figure S9. Data from California Fire Return Interval Departure Database v. 2020.

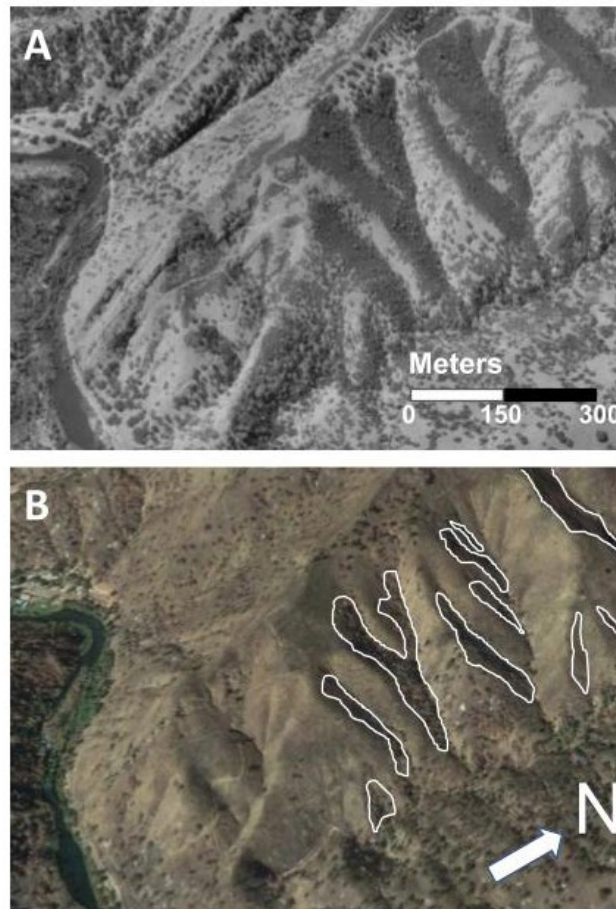


Fig. S7. Impacts of frequent burning on vegetation in the Putah Creek Canyon, Yolo County, California. See Figure S8 for geographic location. Bray Creek flows from right to left across the bottom-right of the figures. A: June 1993, area of photo was last burned in 1959; B: October 2020, a few months after passage of the Hennessy Fire. White polygons in B portray approximate remaining extent of chaparral (>60% loss of area between 1993 and 2020); field visits were made in April and May 2021 to confirm areas of remaining chaparral. Note also decreases in density of oaks. Area of photo was burned in 1959, 2007, 2014, 2016, 2018, and 2020. Imagery from Google Earth™.

Facilitating change

Managers may facilitate change by pushing the system along an alternative new and potential novel trajectory.

9. North Rim of the Grand Canyon, Arizona

The Grand Canyon National Park has an active and successful fire management program (https://www.nps.gov/grca/learn/management/upload/grca_fmp.pdf). One of the directives of the National Park system is to preserve and maintain natural processes, and for the park, that includes wildfire activity approaching historical ranges of variability. In some areas of the park's North Rim, however, wildfires are affecting older, supposedly fire-resistant ponderosa pine trees with higher than expected mortality. Reburning by wildfire in former high-severity patches with heavy fuel loads from trees killed previously allowed for intense fire activity and expansion of the high-severity patch. Thus, with each new fire, stand densities were beginning to fall well below historical norms, often in clumps or in moderate to large patches (Fig. 4A in main text). Gambel oak and New Mexico locust, which had historically been held in check by fire, were now benefiting from conifer-depleted, high-severity patches. In some areas, shrubland patches have begun to merge into large patches. The maintenance of fire on this landscape seems to be facilitating a stand structure and compositional transition that is in accord with experimental treatments in areas with heavy Gambel oak cover (Korb et al. 2020) and might be expected with climate change (Batllori et al. 2020). From the manager's perspective, this seems to be the natural process of adaptation, albeit aided by managing wildfires for resource benefit objectives. Preserving and maintaining natural processes on the North Rim may lead to more VTC and the development of new natural fire regimes. Managers continue to work on finding a balance between preserving natural processes by managing wildfires, and protecting old growth trees by implementing prescribed fire, even in proposed wilderness areas where fire regimes and forest conditions are generally within a range of natural variability.

10. Southern Sierra Nevada, California

In 2012 to 2016, southern and central California experienced an extreme drought that was likely unprecedented in the past 1,200 years or more (Robeson 2015). This event, coupled with widespread bark beetle outbreaks and warming climate trends, resulted in elevated tree mortality throughout the state, with the epicenter of mortality occurring in the southern Sierra Nevada (Preisler et al. 2017; Young et al. 2017). On the Sierra National Forest, mortality of medium to large diameter pines and other conifers was often very high in many sites (Fettig et al. 2019; Stephenson et al. 2019), particularly in relatively drier sites with high pre-drought tree densities (Restaino et al. 2019). Ponderosa pine and dry mixed conifer stands thinned with prescribed burning or mechanical harvest prior to the drought experienced lower levels of drought-related tree mortality (Restaino et al. 2019), and post-drought stand structure and composition (considering trees and tree regeneration) were more aligned with the natural range of variation than neighboring untreated stands (Young et al. 2020). Untreated stands with high levels of post-drought tree mortality became heavily dominated by shade-tolerant, drought-sensitive, and fire-intolerant white fir (*Abies concolor*) and incense cedar (*Calocedrus decurrens*), suggesting low resilience of these stands to future fires and climate change. In contrast, the composition of thinned stands experiencing moderate to high levels of tree mortality shifted in dominance from pines to oaks (particularly black oak and canyon live oak; *Q. chrysolepis*) (Young et al. 2020),

which are more resilient to fire, drought, and insect attack than most Sierra Nevada conifers (Safford and Stevens 2017) assuming the continued absence of goldspotted oak borer from the region (see case study #3, *Laguna Mountain, California*). Following the drought, forest managers focused on the removal of residual dead conifers and retention of most oaks along roads and within strategic fuel treatment units (e.g., fuel breaks), resulting in the directed facilitation of moderately closed canopy pine to open canopy oak stands (Fig. 4B in main text). In other previously thinned stands heavily impacted by drought, forest managers are considering reforestation actions to restore stand composition using ponderosa pine and sugar pine (*P. lambertiana*) seedlings (USDA 2018; Steel et al. 2020). The aim would be to improve habitat suitability and connectivity for species of conservation concern, such as the endangered southern Sierra Nevada population of fisher (*Pekania pennanti*). In the greater Sierra Nevada bioregion, large and severe wildfires prior to or during the drought have resulted in additional VTCs, such as the conversion of pine forest to fir-cedar-pine forest (Wayman and Safford 2021) and conifer forest to shrublands (Shive et al. 2018; Young et al. 2019). Forest managers and collaborative partners in the Sierra Nevada are using reforestation decision-support tools (e.g., USDA Climate Hub 2021; <https://www.climatehubs.usda.gov/hubs/california/topic/reforestation-decision-support-tools>) and post-fire restoration frameworks (e.g., Meyer et al. 2021) to guide post-drought and post-fire vegetation management efforts in the region, including the facilitation of VTCs where desirable.

11. Pinaleno Mountains, Arizona

The Pinaleno spruce-fir forest burned with high severity in the 2004 Nuttall Complex and 2016 Frye Fires, with large areas burned twice (Merrick et al. 2021). Drastic change from mature spruce-fir to effectively bare ground raised concerns regarding type conversion. However, the same forest burned with high severity in 1685 (Margolis et al. 2011; O'Connor et al. 2014) and did not incur a type conversion, though it took a long time for the spruce-fir species assemblage to re-establish (O'Connor et al. 2015, 2017; Lynch 2018). Tree-ring records indicate that ca. 50 years passed without substantial conifer recruitment. Colonization of the ca. 1100 ha spruce-fir vegetation community by Engelmann spruce and corkbark fir (*Abies lasiocarpa* var. *arizonica* (Hook.) Nutt.) took 175 years or more, with trees establishing from refugia in ciénegas (freshwater wetlands particular to the American Southwest) and north-facing canyons (O'Connor et al. 2015). Spruce and fir survived both recent fires in the same ciénegas (Fig. S8) and in a small area surrounding an astrophysical observatory, indicating the potential for a spruce-fir forest to re-establish. This demonstrates the importance of refugia, however small, to a forest vegetation community's ability to resist permanent type conversion, as well as the temporal scale of successional processes in ecosystems with low-frequency high-severity disturbances.

The temporal scale of Pinaleno spruce-fir succession is too long, and the uncertainties regarding future climate too great, for a wait-and-see management approach, and managers recognize a need to facilitate more rapid change. The Pinaleno spruce-fir forest is designated critical habitat for the endangered Mt. Graham red squirrel (*Tamiasciurus fremonti grahamensis*) (USFWS 2011) and the Nuttall and Frye Fires damaged 60 and 95%, respectively, of the habitat (Merrick et al. 2021). The primary management goal for the Pinaleno spruce-fir is to accelerate establishment of a healthy and resilient forest that has the potential to return to a spruce-fir forest. Management actions and plans include establishing a mix of native conifer species present pre-fire in both the spruce-fir and mixed-conifer forest that borders it at lower elevations (AM

Lynch, *personal observations*), and promoting fire- and insect-resilient conditions in lower-elevation forests (USFS 2010). Managing forest structure and species composition in the mixed-conifer forest are critical components of spruce-fir management, as the overstocked, multi-storied, and closed-canopy nature of the mixed-conifer forest, dominated by shade-tolerant species, contributed greatly to both fires spreading from the mixed-conifer into the spruce-fir as canopy fires, and to the severity of insect outbreaks that preceded them (Lynch 2018; O'Connor et al. 2014, 2015; USFS 2018, 2022). Planting Engelmann spruce, corkbark fir, Douglas-fir and southwestern white pine (*P. strobiformis* Engelmann) should accelerate the establishment of a conifer forest. Given climate comparable to the 1685-2003 period, planting Douglas-fir and southwestern white pine should not prevent a return to a spruce-fir vegetation type, as both species were early recruiters post-1685 but ceased recruiting after 145-185 years, while Engelmann spruce and corkbark fir were dominant and consistent recruiters (O'Connor et al. 2015). Given warmer and/or dryer conditions, planting a mix of species would presumably accelerate type conversion to a mesic mixed-conifer forest, which would be desirable if the spruce-fir community is not attainable.

Planting in the spruce-fir footprint and promoting fire- and insect-resiliency in the lower elevation forests have both met with challenges. The Frye Fire reburned much of the area planted after the Nuttall Fire (Merrick et al. 2021), seed availability is very limited, and the available Engelmann spruce seed was collected from a population highly vulnerable to spruce aphid (*Elatobium abietinum* Walker), an exotic predicted to diminish Engelmann spruce representation regardless of other disturbances (Lynch 2004, 2009). Steep inoperable terrain and administrative constraints hinder thinning and fuel-reduction operations across the mountain range. The administrative situation is complicated by resource limitations, habitat needs of threatened and endangered species, multiple and often conflicting socioecological values, and uncertainties regarding vegetation trajectories (USFS 2010, 2022; Lynch 2018).



Fig. S8. Engelmann spruce and corkbark fir in a Pinaleño cienega, survivors of both the 2004 Nuttall Complex Fire and the 2017 Frye Fire. Photo by Ann Lynch, 10 October 2017.

Methods and Materials

Processing VTC examples

Following the workshops, we geolocated the examples from the paper maps into a GIS and digitized the paper forms into a spreadsheet (SI tables). We used a combination of QGIS (v3.8) (<https://www.qgis.org/en/site>) and Google Earth Pro™ (v7.3.4.8248) to verify vegetation structure from aerial imagery. The participant descriptions were distilled into standardized terms for vegetation type, time scale, spatial scale, driver, and management response. Finally, we computed summaries for the various descriptive factor levels in R (v4.1.0, R Core Team 2020) with help from tidyverse packages (v1.3.0, Wickham et al. 2019), and developed visualizations with the graphics packages ggplot2 (v3.3.5, Wickham 2016) and circlize (v0.4.11, Gu et al. 2014).

We excluded several examples that did not broadly fit the collective concept of VTC that emerged from the workshops, as defined above. Two examples included extensive but gradual in-filling or replacement of dominant woodland tree species (e.g., piñon-juniper or oak) by fire-intolerant mixed conifer species due to the lack of recurrent fire activity. Another example included dense post-fire reproduction of aspen in mesic mixed-conifer forests, which is generally viewed as a decadal to centennial succession dynamic leading back to conifer dominance (Dick-Peddie 1993; but see Morris et al. 2019). We also excluded examples of VTC that were beyond the vegetative types we focus on here, including the invasion of non-native buffelgrass (*Pennisetum ciliare*) into Sonoran Desert environments that is generating new fire regimes and causing high mortality of fire-intolerant species such as saguaro (*Carnegiea gigantea*) and other cacti (Stevens and Falk 2009; McDonald and McPherson 2011).

FACTS (US Forest Service Activity Tracking System)

We mapped the VTC examples onto a fire atlas compiled from the Monitoring Trends in Burn Severity (MTBS) database (Eidenshink et al. 2007), finding that 34 examples were spatially and temporally associated with high-severity fire. There were 55 individual fires events, with nearly equal numbers of fires between our two study regions (27 in CA, 28 in SW), although total burn areas and the percentage of high-severity fire were greater in CA than in the SW. In CA, the fire sizes ranged from 450 ha to 108,602 ha, with 46% of the burn areas in high-severity. In the SW, fire sizes ranged from 1,156 ha to 91,490 ha, with 36% burning in high-severity.

Using the high-severity burn patches enabled extraction of reported management actions in the US Forest Service Forest Activity Tracking System (FACTS, <https://data.fs.usda.gov/geodata/edw/datasets.php>), which we used to supplement participant knowledge. We downloaded the FACTS database from the FSGeodata Clearinghouse on December 7, 2020, including the following four feature classes: “Activity Silviculture Reforestation,” “Timber Harvests,” “Hazardous Fuel Treatment Reduction: Polygon,” and “Integrated Resource Restoration (IRR): Polygon.” Polygons of completed activity with more than one hectare or >10% area within an identified VTC were selected and categorized according to five activity types relevant to management response: “Tree Establishment,” “Commercial Tree Removal,” “Non-commercial Fuels Reduction,” “Observation,” and “Invasive Removal” (SI data). To minimize overestimation of the area treated, FACTS data were summarized by year completed and activity type, eliminating overlapping polygons and duplicate counting of related

treatments. The resultant “units completed” (in ha) was the smaller of: a) the intersection of high-severity patch area and treatment area, or b) the sum of Forest Service “number of units accomplished” for intersecting and overlapping treatment polygons. High-severity “patch area” included MTBS dNBR-derived high-severity burn area with an added 31m diameter buffer to reduce pixelated patch geometry. Finally, we summarized the management response data by five-year post-fire intervals and normalized units completed by the high-severity patch area available for management within each calendar year. Normalization provided relative potential treatment areas as a basis for comparison of treatments by omitting land allocations that preclude management activity (e.g., wilderness or roadless areas: “USFS - Other”), and also by accounting for the varying age of fire events (median year burned: 2007, range: 1984-2018) relative to the time of FACTS download.

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