Co-designed management scenarios shape the responses of seasonally dry forests to changing climate and fire regimes

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Abstract
1. Climate change is altering disturbance regimes and recovery rates of forests globally. The future of these forests will depend on how climate change interacts with management activities. Forest managers are in critical need of strategies to manage the effects of climate change.
2. We co-designed forest management scenarios with forest managers and stakeholders in the Klamath ecoregion of Oregon and California, a seasonally dry forest in the Western US subject to fire disturbances. The resultant scenarios span a broad range of forest and fire management strategies. Using a mechanistic forest landscape model, we simulated the scenarios as they interacted with forest growth, succession, wildfire disturbances and climate change. We analysed the simulations to (a) understand how scenarios affected the fire regime and (b) estimate how each scenario altered potential forest composition.
3. Within the simulation timeframe (85 years), the scenarios had a large influence on fire regimes, with fire rotation periods ranging from 60 years in a minimal management scenario to 180 years with an industrial forestry style management scenario. Regardless of management strategy, mega-fires (>100,000 ha) are expected to increase in frequency, driven by stronger climate forcing and extreme fire weather.
4. High elevation conifers declined across all climate and management scenarios, reflecting an imbalance between forest types, climate and disturbance. At lower elevations (<1,800 m), most scenarios maintained forest cover levels; however, the minimal intervention scenario triggered $5 \times 10^5$ ha of mixed conifer loss by the end of the century in favour of shrublands, whereas the maximal intervention scenario added an equivalent amount of mixed conifer.
5. Policy implications. Forest management scenarios that expand beyond current policies—including privatization and aggressive climate adaptation—can strongly influence forest trajectories despite a climate-enhanced fire regime. Forest management can alter forest trajectories by increasing the pace and scale of actions taken, such as fuel reduction treatments, or by limiting other actions, such as fire suppression.
1 | INTRODUCTION

Climate change is altering plant communities globally (Franklin, Serra-Díaz, Syphard, & Regan, 2016), and it presents substantial challenges for the management of forests in particular (Keenan, 2015; Millar, Stephenson, & Stephens, 2007). The aggregate effects of climate change—the complex interactions created by precipitation variability and seasonality, temperature change, and along with changes to disturbance and forest regeneration—are widely expected to drive fast ecosystem transitions (Grimm et al., 2013). Most landscapes are managed, but the consequences of climate change are dependent on past and present human influences and pressures. Fire suppression and changing forest management practices in western US forests have led to widespread densification (Hessburg & Agee, 2003) and, in turn, to increased vulnerability to drought, insects (Fettig, Mortenson, Bulaon, & Foulk, 2019) and fire (Kolb et al., 2016). However, management actions such as mechanical thinning, prescribed fire and allowing wildfires to burn are all approaches that can restore forests and improve their resilience to disturbance and climate change (North et al., 2015).

Nevertheless, management actions with respect to forests and wildland fires can be contentious. In question is whether current fires are within the normal range of variation for these landscapes given the increase in recent fire sizes (Barbero, Abatzoglou, Larkin, Kolden, & Stocks, 2015), and the overall effectiveness of fuel treatments in mitigating fire events (Baker, 2014). Prichard and Kennedy (2017) found that fuel treatments were unlikely to impact the area burned, but they were effective in reducing fire severity and increasing forest resilience in very large fires. Managing fire severity has important consequences, as increasing high severity patch size has been linked to declining forest regeneration as limits on dispersal preclude recolonization (Donato, Fontaine, Robinson, Kauffman, & Law, 2009).

With its history of extreme fire events (fires greater than 100,000 ha) and the potential for higher levels of management interventions on publicly owned forest lands, the Klamath–Siskiyou region of SW Oregon and NW California (‘Klamath region’) is an ideal area to develop and test ideas about the role of forest management under climate change. The Klamath forests are a seasonally dry, fire adapted, Mediterranean forest type, but are at risk as climate change is projected to increase temperatures resulting in higher evaporative demand and greater drought stress (Adams et al., 2009). The region also has a history of extreme fire events (e.g. the Biscuit fire which was 200,000 ha in 2002) but also has had challenges in implementing management at scale on public lands due to litigation by environmental and forest industry groups.

We evaluated how forest management can change the trajectories of these forests under climate change and changing disturbance regimes. We asked the following questions: (a) Can management shift the future fire regime, given that climate change at the end of the century may increase fire size and frequency? (b) Can management maintain the forested landscape, and more specifically, the mixed conifer community? To answer these questions, we co–designed six management scenarios with a team of management experts and stakeholders representing diverse interests and agendas. Next, we translated these scenarios into a forest landscape model that incorporates forest dynamics, management actions, climate change and disturbances. We analysed the outputs to estimate the effects of climate and management on the fire regime, forested area and the spatial distribution of extant tree species.

2 | MATERIALS AND METHODS

2.1 | Study area

The Klamath–Siskiyou region, as we have defined it in this study, includes approximately 3.2 million hectares of SW Oregon and NW California (Figure 1). The elevation ranges from sea level at the Pacific coast to 2,754 m.a.s.l. (https://viewer.nationalmap.gov/basic/). The climate is classified as warm-summer Mediterranean with cool, wet winters and warm, dry summers. Given the topographic complexity, local climate can be highly variable. There is a strong west to east temperature and moisture gradient, with precipitation ranging from as little as 400 mm to over 4,000 mm per year (range derived from Maurer, Wood, Adam, Lettenmaier, & Nießens, 2002 dataset). The number of growing days per year can range from 60 to 250 (Agee, Skinner, & Taylor, 2006). The vegetation cover follows both the elevational and climatic gradients: with low elevation oak woodlands featuring California black oak Quercus kelloggii (Newberry) and Oregon white oak Quercus garryana (Douglas ex Hook.); mid-elevation mixed conifer dominated by Douglas-fir Pseudotsuga menziesii (Mirb.); transitioning to high elevation mixed conifer which includes true firs like white fir Abies concolor (Gord. & Glend.) and red fir Abies magnifica × procer a. Tanoak Notholithocarpus densiflorus (Hook. & Arn.; P.S. Manos, C.H. Cannon & S.H. Oh) is generally found by the Pacific coast, while ponderosa pine Pinus ponderosa (Lawson & C. Lawson) can be found in the drier interior. The complete list of species modelled and their assigned cover type are included in Table 1. Land ownership in the study area was classified into federal (public) lands (64%); private lands (34%) that can be broken down further into private industrial forest holdings (20%), private non–industrial forests (9%) and other private landholdings (5%); tribal lands (2%); and state lands (<1%). Approximately 20% of the area is under some sort of protected status (e.g. designated Wilderness).
2.2 | Forest management scenarios

We convened two workshops, one in April and one in November 2016 in Yreka, CA with federal land managers, Karuk tribal members and others representing not-for-profit groups (e.g. watershed and restoration councils) in the region. There were 20 plus attendees per meeting. We used a scenario co-design workshop approach that included small group brainstorming sessions, group discussions to highlight and resolve differences, additional small group sessions to identify methods and voting to finalize the choice of strategies at the April workshop. These scenarios were then integrated into LANDIS-II forest landscape model (see below)
and further refined by adding specific treatment targets and locations in the November 2016 workshop. The strategies considered multiple objectives: reducing mega-fires; promoting timber extraction; adapting to climate change and alternative approaches to landscape restoration.

The workshops resulted in six scenarios: business-as-usual (BAU), privatization (PVT), let-it-burn (LIB), strategic fuels treatments (SFT), eco-restoration (Rx) and climate change adaptation (CCA; BAU, PVT and LIB detailed below, the remaining three in Appendix S2: Supporting Information Methods). They broadly range in management intensity from continuing existing management levels (BAU) to intensive harvesting and fuel treatments (Privatization) to minimal management for public lands in the region (LIB; Table 2). The other scenarios focused on changing the methods used in management using prescribed fire (eco-restoration) to strategically placed fuels treatments (SFT) to an all of the above approach (CCA). We developed ownership and management zone maps that controlled what sort of management actions were possible at each location (Figure 2; Table 2; Appendix S1: Figure S1, Tables S4 and S5).

The BAU scenario simulated recent harvest rates and areas (2000–2014) for the forests after the implementation of the Northwest Forest Plan (see Figure S2). Harvest levels were based on county-level timber receipts and direct reports from National Forests with 2012 as the baseline. Current practices vary significantly among the different ownership groups. Most harvesting occurs on private industrial lands and mostly in the form of clear cuts, the size of which is limited by state regulations. Very little harvesting occurs on Federal lands and typically in the form of thinning for the promotion of old-growth characteristics. The general emphasis was promoting stands with a diversity of age and structural characteristics containing trees greater than 200 years old, and to achieve that objective within the model, prescriptions included proportionally thinning trees from 1 to 120 years old.

Under the PVT scenario, the management objective for the region would maximize the extractive resource value from the landscape. Therefore, this scenario was dominated by industrial forest practices. Management actions included the following: short rotation monoculture forestry with planting and salvage logging; full fire suppression (supported by an intensive road network to allow harvest) and no prescribed fire. Timber harvest levels were based on historical averages of production before the implementation of the Northwest Forest Plan (Gale et al., 2012).

On the other end of the management intensity spectrum, the LIB scenario, fire suppression was limited to community protection assuming that the onus of fire protection for private industrial forestry lands was on industrial landowners and that they would develop treated (mechanically thinned) buffers around their holdings. Suppression on public lands was not simulated. No active management occurred on federal lands.

### 2.3 Model overview

LANDIS-II simulates forests as tree or shrub species-by-age cohorts within raster cells and incorporates spatial interactions across the landscape and among processes (e.g. management, growth and succession, and disturbance) over many decades. Individual cohorts compete for resources (e.g. soil moisture, nitrogen and growing space) among different species and age cohorts within each cell and all cohorts are explicitly located on the landscape. Simulations were run on a 7.29 ha grid (270 m cell side) over 85 years (2015–2100).

Details of this parameterization, calibration and validation of LANDIS-II in this landscape were previously published by Serra-Diaz et al. (2018). All model inputs and parameters necessary to run the model were based on an extensive suite of forest inventory, satellite data, fieldwork and literature sources. Data sources included the following: Forest Inventory and Analysis (FIA) from the USDA Forest Service; forest community maps based on remote sensing and statistical models (Ohmann, Gregory, Henderson, & Roberts, 2011); fire distribution and severity based on remote sensing data (Monitoring

### Table 2 Scenario abbreviation, name and description, along with the general trends in forest landscape management and their changes with respect to harvesting, fuel treatments and fire suppression

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Name</th>
<th>Description</th>
<th>Harvesting</th>
<th>Fuel treatments</th>
<th>Fire suppression</th>
</tr>
</thead>
<tbody>
<tr>
<td>BAU</td>
<td>Business-as-usual</td>
<td>Continuation of present-day policies</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>CCA</td>
<td>Climate change adaptation</td>
<td>Promote oak/pine habitat, treat plantations on federal land, fuels reduction treatments along roads and ridgelines</td>
<td>-</td>
<td>++</td>
<td>-</td>
</tr>
<tr>
<td>LIB</td>
<td>Let-it-burn</td>
<td>No federal management or suppression, fires left to burn</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>PRIV</td>
<td>Privatization</td>
<td>No federal lands, private industrial methods</td>
<td>++</td>
<td>+</td>
<td>++</td>
</tr>
<tr>
<td>Rx</td>
<td>Eco-restoration</td>
<td>Prescribed burning everywhere, protection of legacy trees</td>
<td>-</td>
<td>++</td>
<td>-</td>
</tr>
<tr>
<td>SFT</td>
<td>Strategic Fuels Treatment</td>
<td>Fuels reduction treatments along roads and ridgelines</td>
<td>-</td>
<td>+</td>
<td>-</td>
</tr>
</tbody>
</table>

Abbreviations: -, less; --, none; +, more; ++, substantial increase; 0, status quo.
Trends in Burn Severity—MTBS; Eidenshink et al., 2007); soils data based on national-level soil maps (STATSGO); relationships to mortality derived from specific fieldwork campaigns to determine regeneration dynamics (Tepley, Thompson, Epstein, & Anderson-Teixeira, 2017); and extensive literature on plant functional parameters related to species ecology (Serra-Diaz et al., 2018). See Appendix S2: Supporting Information Methods section for additional details.

FIGURE 2 Three different scenarios represented by the different ownership areas where management activities could occur. Each management zone had its own set of silvicultural prescriptions, harvest targets and prescribed fire targets (e.g. withdrawn areas had no management actions while private industrial forestry practiced clear cut harvesting and pre-commercial thinning). Such practices generally followed existing regulations of forest practices, except for scenarios like the Let-it-burn and Privatization. For a full breakdown of the different management activities by area, see Table 2: Appendix S1: Table S5, or associated GitHub material for description of activities. For the remaining management scenarios, see Appendix S1: Figure S1.

2.3.1 | Forest growth and succession

We used the NECN extension to simulate tree and shrubs growth and reproduction based on limitations from temperature, water, nitrogen and leaf area index based on species and functional group parameters (Scheller, Hua, Bolstad, Birdsey, & Mladenoff, 2011). NECN tracks carbon and nitrogen across multiple pools of live and dead biomass (including leaf, wood, roots, coarse woody debris, litter and surface residue) and soils (including fast decomposing, slow decomposing and passive pools of soil organic matter) that influence species cohort growth. Mortality is caused by age (the mortality rate increases as a species approaches its specified longevity) and by disturbances (harvest and fire, see next section). We did not include drought-related mortality (Williams et al., 2013). However, we did test the model’s sensitivity to water-use efficiency (WUE), given that increased atmospheric carbon dioxide ($CO_2$), which has been found to increased WUE in trees (Peruelas, Canadell, & Ogaya, 2011). Increasing WUE increased landscape level biomass but did not have a substantial effect on the fire regime, and was therefore not presented in this paper (see Appendix S2).

Species establishment probability is determined by species-specific traits, including seed dispersal distances, sexual maturity,
post-fire behaviour (e.g. serotiny, resprouting), light and water availability taken from the literature (Serra-Diaz et al., 2018). Dispersal distance is determined by a two-part negative exponential probability distribution, with the majority of the seed falling within the effective dispersal distance (Ward, Scheller, & Mladenoff, 2004) and those values were also taken from the literature from sources such as USFS Silvics Manual (https://www.srs.fs.usda.gov/pubs/misc/ag_654/table_of_contents.htm) and the USFS Fire Effects Information System (https://www.feis-crs.org/feis/; Serra-Diaz et al., 2018). Regeneration is dependent on several factors: (a) growing degree days, (b) soil moisture and (c) minimum January temperature (Scheller et al., 2011). The soil moisture requirement is a function of soil depth, available water capacity and precipitation. There are also functional group parameters based on growth forms—hardwood, conifer, shrub—the ability to fix nitrogen and evergreen versus deciduous. Species and functional group parameters determined how individual species responded to climate, see Appendix S1: Tables S1 and S2.

Model calibration and validation was performed by comparing the distribution of biomass across the landscape against forest inventory (FIA) data and remote sensing (Wilson, Woodall, & Griffith, 2013). We calibrated growth response curves across a wide gradient of environments and biotic interactions and validated current biomass per species in adjacent cells against their nearest FIA plot. Full details of calibration procedure are found in Serra-Diaz et al., 2018; the model was able to match biomass for each modelled species within 10% across a diverse range of stands, see Appendix S1: Figure S3.

2.3.2 Forest management

Management was implemented in the simulations using the Biomass Harvest extension (Gustafson, Shifley, Mladenoff, Nimerfro, & He, 2000), following similar approaches for Western US forests as in Creutzburg, Scheller, Lucasch, LeDuc, and Johnson (2017) and Loudermilk, Scheller, Weisberg, and Kretchun (2017). Harvest removal ranged in intensities, from partial thinning of cohorts and patch-cutting to clear-cutting and planting after harvest depending on the scenario.

2.3.3 Forest fire

Fire was simulated as a function of ignition, initiation, fuels, topography and fire weather (Sturtevant, Scheller, Miranda, Shinneman, & Syphard, 2009; Syphard, Scheller, Ward, Spencer, & Strittholt, 2011). Fire severity ranges from 1 to 5, with 1 affecting only the youngest of cohorts and 5 affecting all cohorts (Sturtevant et al., 2009), with different species and ages having different responses to severity. The region was divided up into three areas based on historical ignition rates (Short, 2013) and fuel moisture. Fire size was calibrated using historical fire perimeter data from MTBS++ (Eidenshink et al., 2007). For the calibration period of 2000–2010, the median fire size in MTBS was 1,206 ha versus 1,177 ha in LANDIS under the contemporary climate. Fire severity was calibrated between the difference normalized burned index from MTBS and the relationship to crown damage estimated by Thompson, Spies, and Ganio (2007). The per cent of crown damage calculated from MTBS was 55%, while LANDIS overestimated 68% crown damage (Serra-Diaz et al., 2018). The interaction between fire and management was incorporated through changing fuel types (see Appendix S1: Table S3) in response to management activities, like fuels treatments and prescribed fires.

2.4 Experimental design

We simulated forest dynamics and management scenarios with four climate projections from four global circulation models (GCMs) and three different representative concentration pathways (RCPs) as well as contemporary climate. These four future climate projections were chosen because they ‘pinned the corners’ of future potential climate space (Figure S4). To assess stochasticity of the simulated disturbances, each run was replicated nine times, resulting in 270 model runs (6 management scenarios × 5 climate projections × 9 replicates). Simulated fire regimes were summarized as the fire rotation period (FRP—the length of time required for fires to burn an area equal to the size of the study area) and the mean fire rotation interval (MFRI—the frequency at which a particular cell is burned). Simulated tree communities were aggregated into cover types (see Table 1 for more information). Microsoft’s cloud computing platform Azure was utilized. R (version 3.4.1) and ArcGIS (version 10.4.1) were used for analysing model outputs.

3 RESULTS

3.1 Consequences of management and climate change on fire regimes

Management and climate change had comparable effects on the future fire regime of the area across a number of metrics—fire rotation period (Figure 3a), number of large fires (Figure 3b) and mean fire return interval (MFRI; Figure 4). The MFRI was calculated for each cell for each management scenario for the hottest and driest (average change of +3°C, −10% precipitation over contemporary climate) and the coolest and wettest (+1°C, +10%) climate projections, and the metric varied more by management scenario than by climate change projection (Figure 4). Minimum MFRI values ranged between 19 and 22 years for the BAU scenario across the different climate replicates, whereas the smallest minimum MFRI across the scenarios was 13 years under the LIB scenario and largest minimum MFRI was 28 years. While fire size was affected by
the spatial arrangement of fuel treatments, there was a level of similarity among scenarios with respect to the spatial location of the fires, which followed a strong west to east pattern, reflecting the higher climatic water deficit in the east (Appendix S1: Figure S5; Littell & Gwozdz, 2011).

Virtually, all replicates from all scenarios and climate projections had at least one mega-fire (size >100,000 ha) over the course of 85 years of the simulation run (Figure 3b). The range in the number of these events reflects the interaction of management and climate; under the LIB scenario, the lack of management provided more opportunities for fires to grow and their final size becomes more climate-dependent and more variable (Table S6). On the other hand, even the assumed increased fire suppression efforts under Privatization could not prevent such mega-fires, though the associated variability in likelihood was lower (Figure 3b). However, the frequency of such extreme fire events ranged from once every 10 years in the LIB and eco-restoration scenarios to the privatization scenario with once every 40 years (Figure 3b).

3.2 Forest cover and community changes under climate change and management scenarios

The management scenario that maintained the most forest cover was privatization (Figures 5 and 6; Appendix S1: Table S8, Figures S7 and S8). It was the only scenario in which forest cover was stable.
throughout the study period as any losses in other cover classes were offset by increases in the mixed conifer type in that scenario; BAU and CCA supported the next closest levels (Appendix S1: Table S8, Figure S7). Regardless of management scenario, most of the forest cover that was lost was in the high elevation conifer category, approximately $2.5 \times 10^5$ hectares, which was replaced by other shrubs and grasses. The LIB scenario maintained the least forest cover with a 25% decline by the year 2,100, including losses in the high elevation and Klamath mixed conifer categories. The remaining management scenarios fell in a range between the BAU and LIB scenarios.

Overall, there was limited change in the total amount of area occupied by the Klamath Mixed Conifer forest type (Figures 5 and 6).
**FIGURE 5** Vegetation type map for the Business-as-usual scenario, under the Access rcp 8.5 projection in year 2015 (top row) and year 2095 (bottom row). The cover type with the most biomass is highlighted in black. Changes in biomass dominance (i.e. expansion or contraction of the cover type) at the end of the century are highlighted in blue (gain) or red (loss). See Table 2 for more information on the management scenarios.

**FIGURE 6** Area occupied in hectares by cover type and year, averaged across all climate change projections and replicates for each management scenarios. Error bars represent ±1 SD. See Appendix S1: Figure S9 for a breakdown of this figure across all scenarios and climate projections. See Table 2 for more information on the management scenarios.
The area of mixed conifer that converted to hardwood shrub chaparral was offset by expansion into the high elevation conifer area (Figures 5 and 6). There was a substantial decrease in the area occupied by high elevation mixed conifers (ranging from 50% to 80%), driven by higher climate forcing which resulted in regeneration failure of those species (Figure S10). In total, management scenarios may determine more variation in forest communities than different climate change projections (Figure 7; Figures S8 and S9). Agreement in vegetation types across projections highlighted huge uncertainty at the transition between warm-dry conditions in the east and wetter cooler conditions in the coast (Figure 7).

4 | DISCUSSION

Our simulations enabled us to explore a controversial and unresolved topic: the ability of management to shape disturbance regimes under climate change. In the case of fire, this has been focused in fuel management. Some researchers point to larger and more severe fires in the historical record for this region, and advocate that fuels treatments are misguided for restoration in areas with a climate-mediated fire regime and fire suppression is only contributing to the problem (Baker, 2014; DellaSala & Hanson, 2015; DellaSala et al., 2017). The LIB scenario suggests that such a strategy would result in significant compositional shifts throughout this landscape as climate change interacts with the legacy of fire suppression and logging and converts forests to shrub cover, which, in turn, reduces landscape carbon carrying capacity (Liang, Hurteau, & Westerling, 2017). Duane et al. (2019) found similar undesired impacts for carbon storage in a relaxed fire suppression scenario for the Catalonia region of Spain. However, increased fire may not reduce regional biodiversity; Donato et al. (2009) found that areas that burned under short interval high severity fires had the highest levels of shrub and grass richness. Regardless, any scenario would have to be analysed comprehensively to identify potential trade-offs, whether in terms of ecosystem services (Langner et al., 2017) or in fostering adaptive capacity (Seidl & Lexer, 2013), and we would expect that there would be large trade-offs in ecosystem services with the scenarios presented here.

Using wildfire to reduce accumulated fuels under managed conditions has been discussed as an alternative to fuel treatments (Quinn-Davidson & Varner, 2012), although political and logistical constraints have prevented any sort of widespread application of wildland fire use (North et al., 2015). While such ‘managed wildfires’ are not explicitly represented in our simulations, three of the scenarios incorporate a version of fire use (CCA, eco-restoration and SFT), with similar outcomes in terms of changes of forest cover. Further analysis is necessary to assess fire risk to human communities and WUI areas associated with a fire use approach: Price and Bradstock (2013) found that the risk of property destruction increased in Victoria, Australia with increasing slope and proximity to forest edge, and so forest treatments should target the WUI to reduce risk before using the fire-use approach. Given consistent direction and rate of change in forest cover among all but the privatization and LIB scenarios, using wildland fire (as simulated) was a viable alternative to widespread forest thinning in terms of reducing fuel accumulation, especially in remote areas (Stevens-Rumann & Morgan, 2016).

While the privatization scenario approximated a return to harvest rates prior to the NWFP, and a management intensity comparable to the predominately private industrial forestry-owned Oregon Coast Range (Creutzburg et al., 2017), the risk of extreme fire events persists, especially with higher climate forcing. More intensive management did not remove the risk of fires over 100,000 ha, even with the assumed increased road network density to support the higher level
of harvesting. This contrasts with the Mediterranean Basin, where intensive land-use histories result in disruptions in fuel continuity, thereby limiting fire spread (Pausas & Fernández-Muñoz, 2012).

Despite our ability to influence fire regimes, positive fire feedbacks and climate change can also trigger alternative stable states (Fletcher, Wood, & Haberle, 2014). A warmer and drier climate and changing disturbance dynamics can result in regeneration failure of trees, which, in turn, results in establishment of a shrubland stable state that is reinforced by frequent or high severity fires (Kitzberger et al., 2016; Tepley et al., 2017, 2018). Such a process has been observed in the conversion of forests to shrubland in Patagonia (Paritis, Veblen, & Holz, 2015). In the western US, this conversion may largely reflect that the system has already reached an inflection point with respect to climate change and management; areas that either were or became forested under high levels of fire suppression—due to intense suppression policies during the 1970–1980s—cannot maintain themselves as forest into the future under climate change (Crawford, Mensing, Lake, & Zimmerman, 2015; Serra-Díaz et al., 2018). Our projections suggest that montane and subalpine forests are most at risk under climate change (Figure 5; Figure S10) and will remain only until disturbance removes them (Johnstone et al., 2016), which is a trend observed both in the Klamath (DeSiervo, Jules, Bost, De Stigter, & Butz, 2018) in other areas of the United States (Andrus, Harvey, Rodman, Hart, & Veblen, 2018; Harvey, Donato, & Turner, 2016).

While our results generally are in line with other studies in the region, there are limitations to any modelling approach, including the one taken here. There are many sources of uncertainty operating at multiple scales that constrain not only our capacity to understand but also forecast future forests, including parameter uncertainty (how well do parameters represent real landscape values), process uncertainty (how well the simulation represents the underlying mechanisms and processes) and inherent uncertainty (unresolvable uncertainty—novel events or unanticipated feedback loops; Higgins et al., 2003; Reyer, Fleischig, Lasch-Born, & Van Oijen, 2016).

Ultimately, our results support the idea that management of ecosystems will largely shape vegetation in the 21st century, despite the influence of altered disturbance regimes. This idea will be applicable beyond wildfires and to any area experiencing unprecedented disturbance regimes, for example, hurricanes, windstorms, insect outbreaks (Johnstone et al., 2016). This study exemplifies how co-designing management scenarios across actors with different goals is especially useful to achieve meaningful forest management strategies moving forward under climate change, which will, in turn, encourage trust between the public and land management agencies (Acosta & Corral, 2017).

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AUTHORS’ CONTRIBUTIONS

C.J.M. and R.M.S. conceived the ideas and designed methodology; C.J.M., R.M.S. and J.R.T. collected the data; C.J.M. analysed the data; C.J.M. and J.M.S.-D. led the writing of the manuscript. All authors contributed to the drafts and gave final approval for publication.

DATA AVAILABILITY STATEMENT

All model inputs, parameters and R code used are available through GitHub and Zenodo https://doi.org/10.5281/zenodo.3726321 (Maxwell, Serra-Díaz, Scheller, & Thompson, 2020).

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**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section.

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