Ecological Modelling 488 (2024) 110573

Contents lists available at ScienceDirect

Ecological Modelling

journal homepage: www.elsevier.com/locate/ecolmodel

Resource objective wildfire leveraged to restore old growth forest structure while stabilizing carbon stocks in the southwestern United States

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A R T I C L E I N F O

Keywords:
Aboveground carbon
Carbon carrying capacity
Forest management
Forest landscape model
Old growth forest
Resource objective wildfire

A B S T R A C T

Wildfire futures and aboveground carbon (C) dynamics associated with forest restoration programs that integrate resource objective wildfire as part of a larger treatment strategy are not well understood. Using simulation modeling, we examined alternative forest and fuel management strategies on a 237,218-ha study area within a 778,000-ha landscape that is a high priority target for federal restoration programs. We simulated two wildfire management scenarios combined with three levels of conventional forest restoration treatments over 64 years using a detailed landscape disturbance and succession model developed in prior work. We found accelerated forest restoration used in concert with resource objective wildfire was the most effective at returning old growth forest structure, while stabilizing aboveground C stocks and restoring the fire return interval to its historic range of variation. In scenarios without forest restoration, the continued practice of resource objective wildfires during shoulder fire seasons reduced summer emissions in a negative feedback loop. In the short term, scenarios without forest restoration increased live tree C, but also increased the likelihood of C loss during wildfire activity driven by extreme fire weather. We found scenarios most effective at restoring fire-excluded pine forests to their historical old growth conditions came at a short-term cost of lost C, but with the long-term benefit of substantially increasing fire-resistant live tree C. Our results inform how local decision making can best balance competing goals of sequestering C, and stabilizing C stocks in frequent-fire pine forests using the principles of local fire ecology to restore and maintain old growth forest structure.

1. Introduction[1]

High severity wildfires threaten the stability of carbon (C) stored in frequent-fire forests in the western United States by triggering vegetation type conversions and impeding post-fire regeneration (Cansler and McKenzie, 2014; Chambers et al., 2016; Coop, 2023; Coop et al., 2020; Kemp et al., 2016). These forests have experienced a history of harvesting large diameter trees and a prolonged policy of fire suppression resulting in the conversion of old growth structures to dense stands of small diameter trees (Bauhus et al., 2009; Busenberg, 2004; Covington et al., 1997; Covington and Moore, 1994; Naficy et al., 2010) elevating risk of high severity fire and C loss (Peeler et al., 2023). Forest restoration and fuel management programs offer a potential solution to reduce fire size and severity, enhancing forest resilience and stabilizing forest C over the long term (Ager et al., 2020; Davis and al., 2023; Foster et al., 2020; Hurteau et al., 2019; Sorensen et al., 2011). Over time, as the post restoration forest structure matures (Allen et al., 2002; Bauhus et al., 2009), yearly C uptake of individual trees can be expected to increase with size (Hudiburg et al., 2009; Mildrexler et al., 2020) until potentially entering a new dynamic equilibrium where yearly C uptake equates yearly C emissions across aboveground C pools (Odum, 1969; Riebeek, 2011; Sousa, 1984).

Across many ecologies and geographic regions, studies have suggested that restoring an old growth forest structure stabilizes forest C

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[1] 0x = no treatment; 1x = current treatment level; 5x = five-times current treatment level; d.b.h. = diameter at breast height; DMT = dwarf mistletoe; FFRI = Four Forest Restoration Initiative; FSim = Large Fire Simulator; FVS = Forest Vegetation Simulator; LSim = Landscape Simulator; PRO = protection objective; RO = resource objective; SDI = stand density index; WUI = wildland-urban interface

https://doi.org/10.1016/j.ecolmodel.2023.110573
Received 8 August 2023; Received in revised form 20 October 2023; Accepted 22 November 2023
Available online 8 December 2023
0304-3800/Published by Elsevier B.V.

(Barros et al., 2018; Foster et al., 2020; Grace et al., 2014; Hurteau et al., 2016; McCauley et al., 2019) while acting as an important C sink (Luyssaert et al., 2008; Pugh et al., 2019) so long as the aboveground forest structure retains a resilient condition with respect to natural disturbance and successional processes (Liang et al., 2018; Liu et al., 2019). While C pools can be relatively stable over broad scales, C pools in localities increase or decrease depending on local perturbations and disturbances (Grace et al., 2014; Kashian et al., 2006; Mitchard, 2018). When ecologies are in a pseudo-equilibrium, biomass growth is balanced by biomass turnover (e.g., mortality, needle casting), heterotrophic respiration, and wildfire removing C from above and belowground C pools (Ameray et al., 2021; Harmon et al., 2011). External perturbations of the equilibrium can increase biomass turnover rates (Brienen and al., 2015); or dampen or stimulate biomass growth relative to heterotrophic respiration, with wildfire playing a key role in fire adapted ecosystems (Harmon et al., 2011).

Though the underling empirical and modeling science support the goal of restoring forests of the Western US to stabilize forest C, these programs have encountered practical limitations including low economic viability of removed material and a lack of processing facilities (Ager et al., 2017; Chang et al., 2022; Hjerpe et al., 2009; Taylor et al., 2015), as well as legal, administrative, and operational constraints to mechanical thinning and broadcast burning (Bone et al., 2023; Carney et al., 2023; Hartsough et al., 2008; Hjerpe et al., 2009; Kolden, 2019; Nicholls et al., 2018; North et al., 2021; Swain et al., 2023). Moreover, starting from degraded conditions, mechanical thinning operation and prescribed fire designed to expand old growth forest structures that are resistant to wildfire (Bauhus et al., 2009; He et al., 2016) will release C that has accumulated from a prolonged policy of fire suppression (Ager et al., 2022; Busenberg, 2004).

An alternative approach to mechanical thinning and broadcast burning is returning natural fire regimes to the landscape by reducing wildfire suppression operations during weather conditions that generate ecological benefits aligned with restoration goals (Huffman et al., 2020). In the southwestern US, land managers have been expanding the implementation of what is commonly called resource objective (RO) wildfire (Huffman et al., 2018; Hunter et al., 2014) to achieve multiple ecological and management objectives. Empirical studies have shown that RO wildfire can be an effective tool in restoring frequent-fire forests to resilient, pre-settlement conditions through multiple low to moderate severity fire events (Huffman et al., 2018). Simulation studies support these findings, indicating that RO wildfire can help expand and maintain forest restoration projects (Greenler et al., 2023; Young et al., 2022). However, planning for RO wildfire is challenged by the need to align weather and fuel conditions with restoration objectives, while being mindful of protection goals (Ager et al., 2017; Young et al., 2019b) and firefighter safety (Pietruszka et al., 2023). Typically, RO wildfire is practiced in remote areas with minimal risk to infrastructure and private property (Miller and Aplet, 2016; Scott et al., 2012), and where mechanical thinning and prescribed fire may not be feasible (Doane et al., 2006; North et al., 2015). In other instances, RO benefits are not the primary drivers of management decisions, but rather wildfires are allowed to burn naturally to avoid firefighter exposure to hazardous work conditions in remote areas (Pietruszka et al., 2023). However, the long-term impacts of RO wildfire on forest restoration, future fire regimes and C dynamics are largely unknown and in need of additional investigations (Hessburg et al., 2021; Huffman et al., 2020).

In this study we used a landscape simulation framework to analyze the independent and combined effects of RO wildfire and forest restoration treatments on long term C dynamics. We hypothesized that in the short term, a rapid increase in treatment rates (i.e., mechanical thinning and prescribed fire) will rapidly decrease live tree C stores and rapidly increase C emissions as the result of removing small diameter trees (Covington and Moore, 1994). However, with time an equivalent level of emitted C will be absorbed by the residual forest structure (Hudiburg et al., 2009; Mildrexler et al., 2020). We further hypothesize, that the reduction in tree density and crown connectivity associated with restoration activities would reduce potential crown fire activity thereby improving the overall resistance of the forested landscape to wildfire (Foster et al., 2020; Peeler et al., 2023). We expect forested landscapes that are returned to an old growth structure alongside an ecologically appropriate disturbance regime (frequent, low-severity fire in our case) will have aboveground C pools that are in equilibrium (Harmon et al., 2011; Kashian et al., 2006; Odum, 1969; Sousa, 1984). Study results advance the understanding of how alternative forest restoration practices affect short-term C loss in exchange for the long-term C stability in fire adapted forests.

2. Methods

2.1. Study area

The 237,218-ha study area is managed by the Kaibab and Coconino National Forests of the USDA Forest Service (USFS) and sits along the Mogollon Rim on the southwestern extent of the Colorado Plateau in northern Arizona, inclusive of the Coconino Plateau along the northern extent (Fig. 1). The Forest Service is currently restoring ponderosa pine (*Pinus ponderosa*) forests in the study area under the Four Forest Restoration Initiative (FFRI; USDA, 2015) using treatments of mechanical thinning and prescribed fire. The overall objective is to increase the health and function of the forest by restoring old growth forest structures that are more resistant to wildfire. Local restoration management objectives include restoring goshawk (*Accipiter gentilis*) and Mexican spotted owl (*Strix occidentalis lucida*) habitat, reducing widespread dwarf mistletoe infestation, restoring ponderosa pine savannas and grasslands to pre-settlement conditions, and reducing hazardous fuels with prescribed fire alone (Fig. 1A, Table 1) (USDA, 2015).

The wildfire simulation domain enclosed 1488,900 ha of public, private, and tribal lands spanning from deserts, grasslands and pinyon-juniper vegetation types at lower ecotones, to ponderosa pine forest in the mid-range ecotones and mixed conifer in the upper ecotones. Dry conifer forests throughout this modeling domain and the wider southwestern US, have a history of using RO wildfire to augment restoration treatments (Young et al., 2019b) (Fig. 1B), with increased fuels reduction following multiple wildfires that burned at low to moderate severity (Huffman et al., 2018, 2017; Hunter et al., 2014, 2011). The expanded use of RO wildfire has also shown the potential to promote floristic diversity (Laughlin et al., 2005). Despite local success, the task of restoring the landscape is a vast undertaking complicated by an extensive backlog of fuel reduction needs (Marlon et al., 2012; North et al., 2015) in an area with a recent history of high-severity wildfires (e.g., 2000 Pumpkin Fire, 3545 ha; 2010 Shultz Fire, 6101 ha) that contained large patches of sporadic and inconsistent tree regeneration (e.g., Owen et al., 2017). The model domain also contains 7425 ha of urban development or WUI where the use of RO wildfire to augment restoration treatments is not practiced. WUI classification was determined by wildland-urban interface designation in the SILVIS lab's 2010 WUI database (Radeloff et al., 2017) and was then expanded based on data provided by the FFRI management team at the United States Forest Service (Fig. 1A).

Combined the forested area has 26,944 tree stands over 778,378 ha that have been delineated into 38 planning areas (PAs) used to prioritize treatments in the study area (Fig. 1). Ponderosa pine forests targeted for treatments in each PA ranged from 1113 to 12,315 ha. In total the PAs are 49 % ponderosa pine forests and savannas with a 3 % median slope and altitude ranging from 1657 to 2828 m above sea level (masl). Mixed-conifer, aspen (*Populus* spp.), and spruce-fir forests comprise 2 % of the PAs with a 17 % median slope and altitude ranging from 1576 to 3783 masl. Pinyon-juniper ecological types comprise 26 % with a median slope of 3 %, and an altitude ranging from 1019 to 2756 masl.

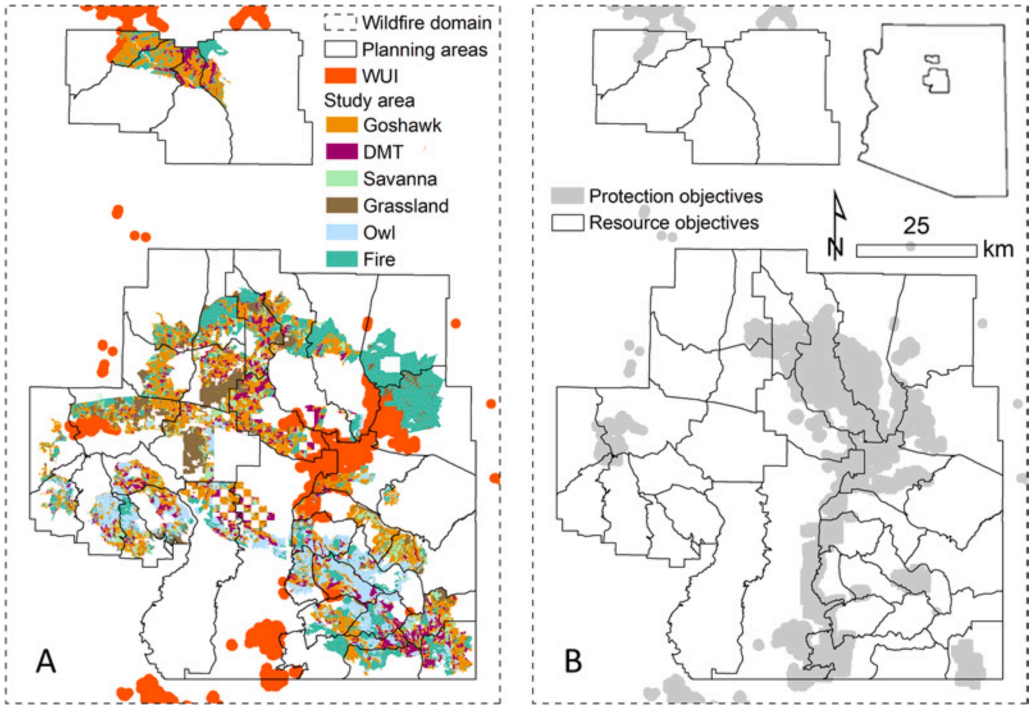

Fig. 1. (A) Wildfire domain (dashed line) in the northern Arizona on the Kaibab and Coconino National Forests, and study area treatments (Table 1). The wildfire domain defined the extent for wildfire simulations. Management activities in the study area were prioritized and implemented within planning areas ($N = 38$) based on potential fire mortality as a percentage of total basal area as described in Section 2.4. DMT = dwarf mistletoe; WUI = wildland-urban interface. (B) Boundaries for wildfire management objectives. Protection objectives are located around WUI, mixed-conifer and spruce–fir ecoregions, national monuments, and steep interior chaparral canyons leading to the WUI. The potential for resource objective wildfires is located within ponderosa pine and pinyon–juniper forests.

Table 1

Treatment type, percentage of study area, and associated active management actions. Each tree stand is assigned an active management action that in addition to the action described could remove no more than half of the ponderosa pine with a diameter at breast height (d.b.h.) greater than 40.6 cm in order to retain old growth forest structure (FFRI, 2011).

Treatment type	Acronym	Percentage of study area†	Active management action
Goshawk habitat restoration	Goshawk	32 %	Thin throughout to SDI = 130 followed by prescribed fire
Prescribed fire only	Fire	27 %	Prescribed fire
Mexican spotted owl habitat restoration	Owl	14 %	Thin throughout to SDI = 115 followed by prescribed fire
Dwarf mistletoe mitigation	DMT	10 %	Thin from below to SDI = 135 followed by prescribed fire
Ponderosa pine grassland restoration	Grassland	10 %	Thin to 5 % canopy closure followed by prescribed fire
Ponderosa pine savanna restoration	Savanna	7 %	Thin throughout to SDI = 14 followed by prescribed fire

Note: SDI = Stand Density Index; DMT = dwarf mistletoe.
† Study area = 237,218 ha.

2.2. Overview of LSim

LSim is a forest landscape model that was built by linking the Parallel Processing version of the Forest Vegetation Simulator (FVS; Crookston and Dixon, 2005) with the large fire simulator (FSim, Finney et al., 2011). Both FVS and FSim models are extensively described in decades of foundational research (Ager et al., 2006; Finney et al., 2007), and they were coupled into LSim by Ager et al. (2020). The modeling components are described in brief below with detailed descriptions in Ager et al. (2020) and Young et al. (2022).

2.2.1. Forest vegetation modeling system

We model forest succession and management for National Forest System lands with FVS (Crookston and Dixon, 2005), a nonspatial, individual tree growth and yield model that simulates stand development and mortality based on empirical relationships. Homogenous polygons of trees (i.e., tree stands) characterized by a record of tree density by species and size, along with elevation, aspect of the slope, and ecotype (Crookston and Dixon, 2005) are the simulation inputs. The USFS has developed over 20 FVS variants, each covering a unique grouping of ecologies across National Forest System lands of the United States (Dixon, 2002). In this study we use the Central Rockies (CR) variant capable of modeling 36 unique tree species and 2 species groups across the United States of Wyoming, South Dakota, Colorado, New Mexico and Arizona (Keyser and Dixon, 2008). Using the FVS-Parallel Process Extension (FVS-PPE; Crookston and Stage, 1991) described in Ager et al. (2020) and Seli et al. (2008), we integrated thousands of stands into an ecologically diverse landscape where dynamic forest modeling is paused at user-defined time intervals to reprioritize management activities, apply landscape-scale constraints, and generate input files for wildfire and other simulation programs. The use of FVS-PPE within the LSim model has the potential to study wildlife habitat, C dynamics, risk to WUI, the effect of accelerated management scenarios (Ager et al., 2020, 2007, 2010), and the spatial optimization of fuel treatments (Finney et al., 2007).

2.2.2. Wildfire modeling

We model wildfires with the large fire simulator (FSim; Finney et al., 2011) (Fig. 2) on a 90-m resolution grid covering the wildfire domain, inclusive of a 5- to 35-km buffer around the study area to reduce edge effects (Fig. 1). Due to the Grand Canyon National Park to the north and sparse vegetation surrounding National Forest System lands, the origin of an edge effect would be limited to neighboring ponderosa pine forests

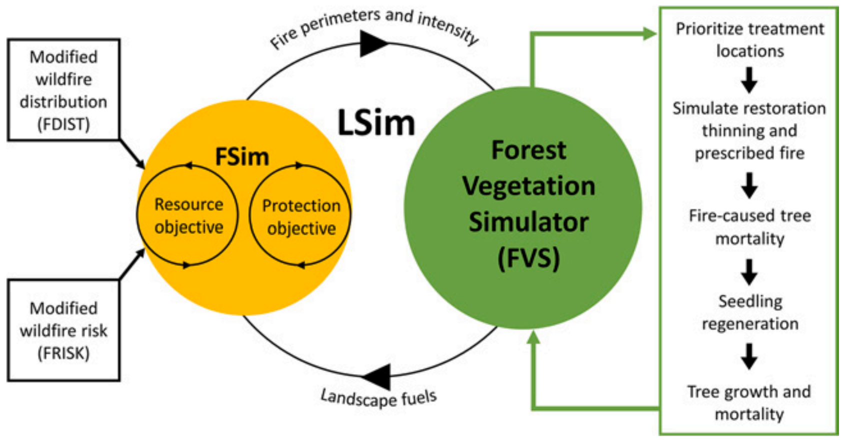

Fig. 2. Diagram of landscape simulator (LSim) model components. The simulation of wildfire was performed with the large fire simulator (FSim), which was modified to enable both protection and resource objective wildfire management dependent on ignition location and weather conditions (black boxes and circles). FDIST, wildfire distribution file used to guide the daily occurrence of wildfire ignitions based on daily energy release component (ERC) values; FRISK, fire risk file used to model wildfire occurrence and spread based on fire weather metrics including percentile fuel moisture and daily ERC values; FSim, large fire simulator. Figure modified from Ager et al. (2020).

in the southeastern extent of the wildfire domain. The FSim model iterates through each day of the calendar year to model fire ignition and spread. We determined large fire ignitions based on their historic distribution (FDIST file) as a function of the energy release component (ERC via the fire risk [FRISK] file) in combination with the distribution of days with multiple fires. If an ignition occurs, fire spread is determined using fire weather conditions in the FRISK file, landscape fuels (surface fuel model, canopy cover, canopy height, crown base height, and crown bulk density) and topography data (elevation, slope, and aspect) contained in a binary raster stack (Young et al., 2022, Appendix S1). A fire is contained as a function of the wildfire's rate of spread and the occurrence of timber fuels, where slower spreading fires in non-timber fuels are more likely to be contained (Finney et al., 2009). We used additional perimeter trimming to simulate progressive fire containment that limited a fire's growth on portions of the perimeter with low fireline intensity (Scott et al., 2018). Output files used by LSim each cycle included the FSim perimeter shapefile and a fireline intensity grid that we converted to flame length using Byram's equation (Byram, 1959) with the Wilson (1980) modification. Each raster pixel of the model domain could only be burned by one wildfire within each cycle (i.e., FSim switch BurnCellsOnlyOnce was engaged).

2.2.3. Integrating FVS and FSim

We integrated FVS and FSim into LSim using a wrapper that sequences their execution and transfers data each 4-year cycle (Fig. 2). Following each cycle, vegetation modeling (i.e., FVS) ispaused to translate forest inventory data into a binary raster stack (Young et al., 2022, Appendix S1) required to simulate the four wildfire seasons (i.e., FSim; Fig. 2). A raster grid of the wildfire perimeters is then transferred back to FVS where it is overlaid with the tree stand layer. If more than 50 % of a stand's pixels burn during a cycle, the stand is determined to have burned, and the fire line intensity grid from FSim is converted to average flame length. The average flame length is then used alongside the tree stand's average crown base height to compute the percent of the stand that burned in a crown fire (Ager et al., 2020; Young et al., 2022, Appendix S1). Together, flame length, scorch height, and percent crown fire serve as parameters in the FLAMEADJust keyword via the FVS Fire and Fuels Extension (FVS-FFE; Rebain et al., 2010). We then simulate a wildfire with the SIMFIRE keyword, and tree mortality is predicted with the first-order fire effects model (FOFEM; Reinhardt et al., 1997). Tree mortality is then used to adjust the forest inventory before FVS resumes for the next cycle. Additional details on wildfire-induced tree mortality are described in Young et al. (2022, Appendix S1). This process is similar to methods in Ager et al. (2020); Ager et al. (2010), and Young et al. (2022).

2.3. Model parameters

2.3.1. Forest and fuel management

We developed stand treatments (Table 1) based on management objectives developed by the FFRI management team of the USFS (USDA, 2015) and implemented them using FVS keywords (Young et al., 2022, Appendix S2). The overarching management objective was to restore low-severity fire regimes by returning old-growth forest structure to the ponderosa pine forest followed with periodic prescribed fire (FFRI, 2011; USDA, 2014, 2015, 2018). Local management objectives are described in Table 1. Each treatment was supported by empirical studies as effective for reducing potential fire behavior (Fulé et al., 2012; Parks et al., 2018; e.g., Stephens et al., 2009). Simulated treatment actions included restoration thinning followed by prescribed fire, pre-commercial thinning followed by prescribed fire, or prescribed fire alone (Table 1; Young et al., 2022, Appendix S2). We determined the appropriate treatment action based on habitat-specific objectives, thresholds of merchantable timber, trees ha^{-1}, time since fire, and fuel loads. Prescribed fire fuel moistures and wind speed were chosen to replicate typical prescribed burning conditions in both the spring and the fall on the Coconino and Kaibab National Forests (Young et al., 2022, Appendix S2) including a blend of fixed and predicted mortality estimated with FOFEM via FVS-FFE (Rebain et al., 2010).

2.3.2. Forest growth and mortality

The tree list data populating our stands and serving as inputs to forest growth models were provided by the FFRI management team of the USFS (USDA, 2015) and then supplemented by a forest inventory analysis (FIA)-imputed tree list developed by Riley et al. (2016). Tree lists were then adjusted to reflect recent mechanical thinning (2009–2019; USDA Forest Service, 2020b) and wildfire activity (2009–2017; MTBS, 2020). The landscape data were then vetted by land managers of the Coconino and Kaibab National Forests to ensure species compositions were properly distributed. Non-wildfire-related tree mortality is based on species-specific models when SDI is $<$ 55 % of the maximum, and on density-dependent models when SDI is $\geq$55 % of the maximum (Keyser and Dixon, 2008).

2.3.3. Forest regeneration

Regeneration models were developed for Coconino County of northern Arizona (Young et al., 2019a) and then calibrated based on prior research (e.g., Singleton et al., 2021) and expert opinion (Young et al., 2022, Appendix S3). Models include nine tree species (blue spruce [*Picea pungens*], corkbark fir [*Abies lasiocarpa* var. *arizonica*], Douglas-fir [*Pseudotsuga menziesii*], Engelmann spruce [*Picea engelmannii*], Gambel oak [*Quercus gambelii*], limber pine [*Pinus flexilis* var. *flexilis*], ponderosa pine, quaking aspen [*Populus tremuloides*], and white fir [*Abies concolor*]) and three species groups (pinyon pine [e.g., common needle pinyon (*Pinus edulis*)], sprouting juniper [e.g., alligator juniper (*Juniperus deppeana*)], and non-sprouting juniper [e.g., Utah juniper (*Juniperus osteosperma*)]) across three ecological zones: pinyon–juniper woodlands, ponderosa pine, and mixed-conifer forests.

To account for regeneration failures in undisturbed forests due to seed predation, frost heaving, etc., each tree species was calibrated to randomly engage regeneration models based on expert opinion (e.g., undisturbed stands will engage ponderosa pine regeneration models in 25 % of the simulated cycles, i.e., 16-year average increments [Randy Fuller, USFS, Apache-Sitgreaves National Forest, personal communication]). Then, to account for failed recruitment to the upper canopy, we calibrated the median predicted regeneration values for each regeneration model based on expert opinion and existing literature (e.g., Ouzts et al., 2015; Strom and Fulé, 2007). When responding to wildfire, we also programed regeneration models to consider site-specific potential for forest growth captured in the site index (SI; FFRI, 2011; USDA Forest Service, 2020a), alongside underlying mechanisms of type conversion (e.g., disturbed ponderosa pine forest returning as alligator juniper or Gambel oak woodland) based on revegetation potential (Coop et al., 2020; Winthers et al., 2005) and the presence of sprouting trees species (e.g., Johnson et al., 1962; Kaufmann et al., 2016). For example, when compared to a moderate-intensity wildfire on a ponderosa pine stand with a low SI (<80), regeneration following a high-intensity fire was reduced by 80 % for 16 years when revegetation potential was moderate, and by 90 % for 40 years when revegetation potential was low (Randy Fuller, USFS, personal communication; Savage and Mast, 2005; Singleton et al., 2021; Young et al., 2020, Appendix S3). Low revegetation potential coincided with high erosion potential, occurring on steep slopes (Coop et al., 2020; Winthers et al., 2005). When compared to low-intensity wildfires, the regeneration of alligator juniper, a tree species known to sprout post-fire was enhanced 3x and 6x following a moderate- and high-intensity wildfire respectively (Johnson et al., 1962). Regeneration models for numerous other sprouting (e.g., Gambel oak, quaking aspen) and mixed-conifer species (e.g., Douglas-fir, white fir) were only initiated during the cycle that followed wildfires or mechanical thin and prescribed fire operations. Regeneration for each species–ecological zone combination was predicted using an FVS written ASCII file containing current stand information (topographic information and species-specific trees ha^{-1} and basal area). Regeneration counts were then written to an ASCII file used to update the inventory record for each tree stand by FVS.

2.3.4. Landscape fuel dynamics

To account for temporal variability in live fuels and shrubby lifeforms on portions of the study area with tree list data, we built a rule set for surface fuel models that deviates from FVS-FFE (Young et al., 2022, Appendix S4). Every LSim cycle, fuel models were selected based on live woody fuel load, fuel bed depth, plant association (USDA Forest Service, 1997), canopy attributes (canopy cover, canopy base height, canopy bulk density, and stand height), and time since disturbance to account for the time needed for the maximum shrub height. Other landscape inputs obtained from LANDFIRE for fire modeling included elevation, slope, and aspect (Krasnow et al., 2009; Rollins, 2009). Portions of the study area without tree list data were assigned 2012 surface fuel models and canopy fuels from LANDFIRE (2013).

2.3.5. Wildfire calibration

We calibrated FSim to replicate historical fire occurrence and size on the Kaibab and Coconino NFs from 2000 to 2014 (≥4 ha)(KCFAST, 2020) using corresponding fire weather data from the Flagstaff Remote Automated Weather Station (RAWS) located 4 km south of Flagstaff, Arizona. Using these data, we developed a modified fire distribution file (FDIST file) using a kinked logistic regression approach that allowed us to adequately calibrate the probability of high ERC fire ignitions, while retaining the predictive value of numerous RO wildfires in the study area (Young et al., 2022). We also artificially lowered the 80th percentile within the LSim FRISK file to allow resource objective wildfire ignitions during shoulder seasons and developed an ignition probability grid to direct fire ignitions based on the historical occurrence of 9444 wildfires (0.004 to 9666 ha) from 1992 to 2015. Refer to Young et al. (2022, Appendix S5) for more information regarding fire calibration, including seasonal distributions of fire ignitions and area burned.

2.4. Forest and wildfire management scenarios

Using a 3 × 2 factorial design (Table 2), we examined potential long-term tradeoffs and synergies between restoration treatments (i.e., mechanical thinning followed by prescribed fire) and wildfire management. Specifically, we simulated three treatment scenarios: no treatments (0x), the current rate of treatments of roughly 8000 ha $year^{-1}$ (1x), and five-times the current rate of treatments (5x). These three treatment scenarios were combined with two wildfire management scenarios: all wildfires managed for protection objectives (PRO) with full suppression strategies, and the option to manage wildfires for resource objectives (RO) based on ignition location and weather conditions (Table 2, Fig. 1B). The business-as-usual scenario is 1xRO. Each of the six scenarios was simulated for 64 years (16 cycles) with 10 replicates to account for stochastic processes within LSim (total = 60 simulations). In total, each scenario simulated 640 wildfire seasons, which adequately reflects the inter-replicate variance of 10,000 seasons (Ager et al., 2021). The sequence of activities in each cycle, including management, growth and mortality, regeneration, and wildfire effects are depicted in Fig. 2. In each cycle, four years of management and wildfire were simulated.

The RO wildfire management scenarios were simulated with a feature in FSim that allows for suppression decisions to be made for each ignition based on location and month of year. Ignition locations assigned to protection objectives (Fig. 1B) and managed with a full suppression strategy were either WUI, within 1 km of WUI or contained wet mixed-conifer forests, steep canyons of interior chaparral, or national monuments and other protected sites. The timing for RO wildfire management was designed to only allow RO wildfires during periods of mild weather within the shoulder fire season (first four months [January through April] and last five months [August through December] of the year). During each month of the shoulder fire season, we adjusted fire

Table 2
Wildfire management, treatment levels and scenarios. The yearly rate of treatment was simulated at three levels, which include no treatment (0x), current treatment of 8000 ha $year^{-1}$ (1x), and five-times current treatment level (5x) on the Kaibab and Coconino National Forests. The combination of wildfire management strategy and treatment level corresponds to a simulation scenario.

Wildfire management	Acronym	Treatment level	Scenario
Fire managed for protection objectives (PRO)	PRO	0x	0xPRO
with full suppression strategies		1x	1xPRO
		5x	5xPRO
Inclusive of fire managed for multiple resource	RO	0x	0xRO
objectives (RO) based on ignition location,		1x	1xRO †
seasonality, and weather conditions		5x	5xRO

† Scenario 1xRO represents business-as-usual (BAU) on the Kaibab and Coconino National Forests.

management response using FSim calibration parameters. This included modifying perimeter trimming to reduce suppression activity, increasing the burn period to resemble fire growth and burnout operations, and precluding fires that spread beyond 6070 ha to simulate the measured response of quickly suppressing fire during weather conditions likely to lead to extreme fire behavior if managed for an extended period (details in Young et al., 2022, Appendix S5). The upper threshold of 6070 was chosen to reflect the largest wildfire used to meet resource objectives in the study area (e.g., 2009 Wildhorse Fire, 5581 ha). The strategic objective of RO wildfire management was to opportunistically help restore and maintain an old growth forest structure.

Planning areas were prioritized for treatment based on potential basal area losses associated with severe fire (70°F with 20-foot windspeed of 20 mph, Rebain et al., 2010) estimated with FVS-FFE. Active management goals are not always met for each 4-year LSim cycle if there was insufficient area to treat based on minimum thresholds of merchantable timber, or the minimum time allowed since the last treatment had not been met (Young et al., 2022, Appendix S2, Fig. S1). Other constraints to management include the exclusion of stands that burned with low to moderate intensity in the last 10 years, which includes prescribed fire, and in the last 20 years for stands that burned with high intensity (flame length >3.66 m). Five-times the current yearly rate of treatments (5x) was the proposed rate of implementation by the USFS (USDA Forest Service, 2015).

2.5. Simulation sequence and response metrics

We first examined future landscapes (2082) that result from each scenario to determine which best restored the ponderosa pine forest to an old growth structure. Then we examined the dynamics of wildfire severity based on conditional wildfire-induced mortality of trees $\geq$ 13 cm d.b.h. (Barrett et al., 2010). We classified severity as low (< 25 % mortality), moderate (25 % $\leq$ mortality $\leq$ 75 %) or high (> 75 % mortality). As depicted in Fig. 3, we assessed C dynamics across aboveground fuel types including shrubs and herbs, coarse woody debris, litter and duff, standing snags, and live trees. Live tree C was further scrutinized to estimate its resistance to severe fire (i.e., resistant live tree C). Resistant live tree C was estimated by reducing live tree C by the proportion of expected total basal area loss resulting from a severe wildfire (70°F with 20-foot windspeed of 20 mph, Rebain et al., 2010) as estimated by FVS-FFE. Then, we assessed C emissions from treatments and wildfire by season and severity class. Finally, the aboveground net C was calculated for each scenario. We define aboveground net C as the accumulation in aboveground C from one cycle to the next, minus the accumulation in aboveground C emissions from one cycle to the next, including C additions to harvested wood products (i.e., ecological-cycle assessment). Specifically, across our study area, we test the hypothesis that the accumulation of aboveground C should roughly equate aboveground C released from disturbance when old growth structures have been restored and the historic fire return interval has been returned (i.e., net zero aboveground C emissions) (Harmon et al., 2011; Kashian et al., 2006; Odum, 1969; Sousa, 1984). We assume half of whole tree C removed from logging operations is emitted during the production of sawlogs. The remaining saw log C is then split equally between processing residuals that are emitted, and long-term storage in harvest wood products (25 % of whole tree C removed) (Steele, 1984). Forest C stocks and emissions were calculated using the Fire and Fuels Extension (FVS-FFE). Belowground C stocks were not considered exhaustively, but generally an increase in aboveground C was found to be associated with an increase in belowground C. See Appendix A, Fig. A1 for estimates of belowground C. Our results inform how to best balance competing goals of sequestering aboveground C, and stabilizing C stocks in fire-excluded, dry forests.

3. Results

3.1. Forest structure

The ability of each scenario to return the ponderosa pine forest in the study area to an old growth forest structure by 2082 (end of simulation) varied widely (Fig. 4). Broadly, as the yearly treatment rate increased so did the prevalence of old growth forest structure, with resource

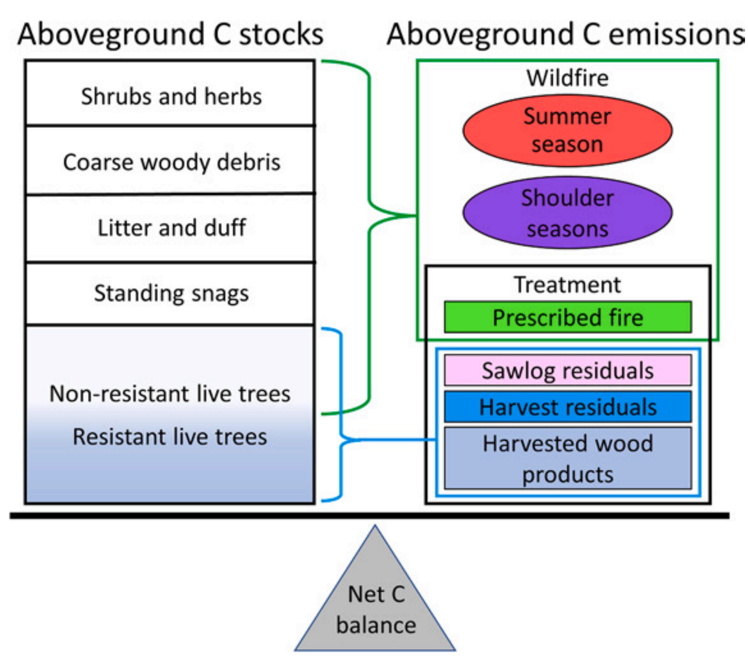

Fig. 3. Aboveground C stocks and C emissions of a forested ecosystem. Aboveground net C of a forested ecosystem is in balance when the accumulation of aboveground C from one cycle to the next equates the accumulation in aboveground C emissions from one cycle to the next, including C additions to harvested wood products (i.e., ecological-cycle assessment). An alternative approach could designate harvested wood products as an aboveground C stock in a life-cycle assessment of the forest products sector. Coarse woody debris includes all dead surface fuels. Litter and duff are fuels that compose the forest floor. Snags are standing dead. Resistant live tree C is C stored in trees expected to survive a severe wildfire (70°F with 20-foot windspeed of 20 mph, Rebain et al., 2010).

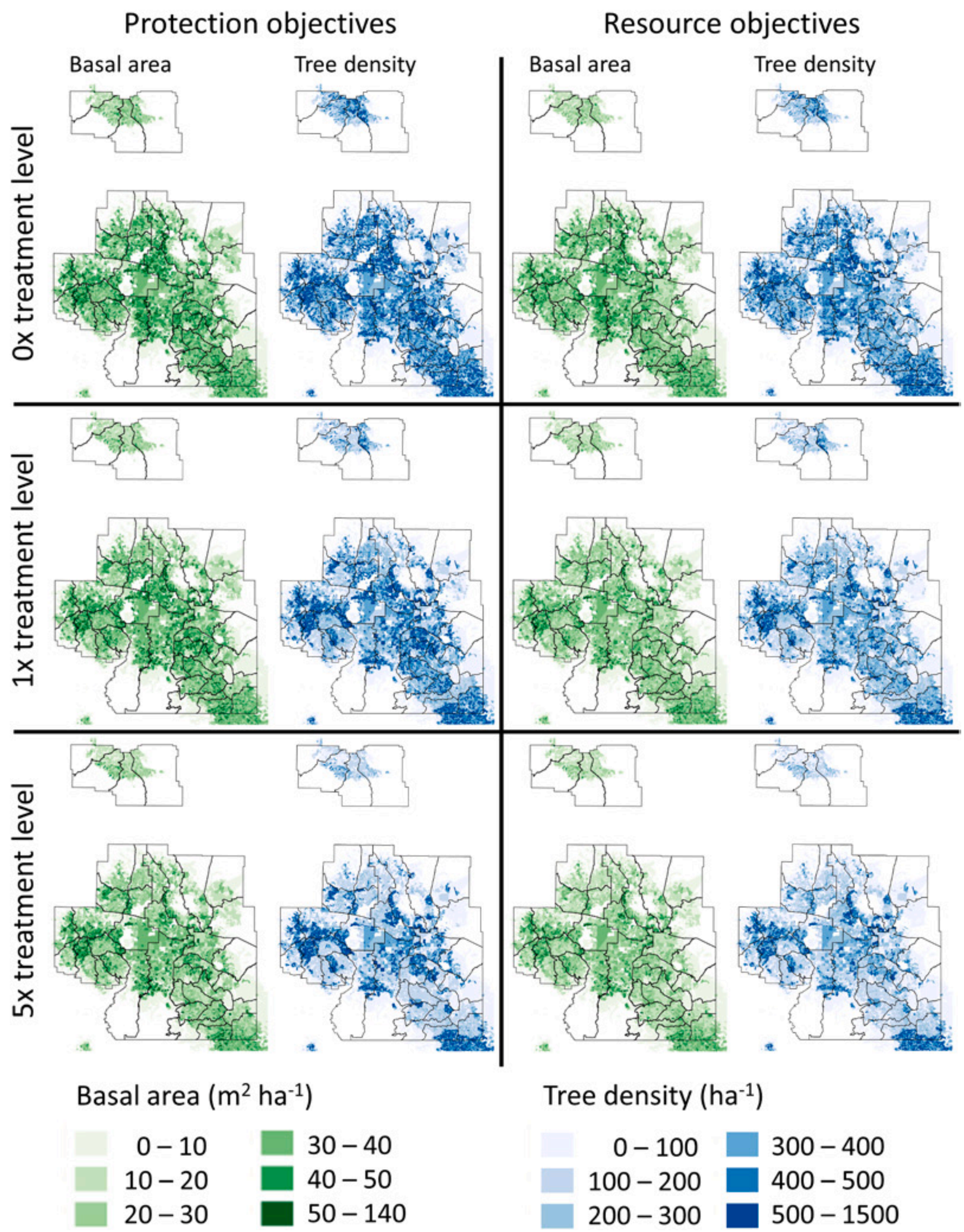

Fig. 4. Mean ponderosa pine basal area (m^2 ha^{-1}) and tree density (ha^{-1}) in 2082 for each scenario. The 5xPRO and 5xRO scenarios best returned the historical stocking of ponderosa pine to the study area (interquartile range: 56 to 138 trees ha^{-1} with 9 to 17 m^2 ha^{-1}) in an old growth forest structure containing large ponderosa pine (Reynolds et al., 2013). Refer to Table 2 for scenario details.

objective (RO) wildfire management accelerating the transition. When treatments and RO wildfire were not applied, the average stocking of ponderosa pine in the study area remained high at 235 trees ha^{-1} (95 trees ac^{-1}) with a basal area of 20 m^2 ha^{-1} (89 ft^2 ac^{-1}) and an average quadratic mean diameter of 36 cm d.b.h (14.2 in d.b.h). Little change was observed when using RO wildfires alone, while 1x treatment reduced average ponderosa pine stocking to 180 trees ha^{-1} (73 trees ac^{-1}), with a basal area of 18 m^2 ha^{-1} (79 ft^2 ac^{-1}) and an average quadratic mean diameter of 39 cm d.b.h (15.4 in d.b.h). When used in combination, 1xRO reduced average ponderosa pine stocking to 148 trees ha^{-1} (60 trees ac^{-1}), with a basal area of 15 m^2 ha^{-1} (64 ft^2 ac^{-1}) and an average quadratic mean diameter of 38 cm d.b.h (14.8 in d.b.h). When increasing the treatment level in the study area to 5xPRO the average stocking was reduced to 101 trees ha^{-1} (41 trees ac^{-1}) with a basal area of 14 m^2 ha^{-1} (63 ft^2 ac^{-1}) and a QMD of 44 cm d.b.h (17.4 in d.b.h). The addition of RO wildfire management (5xRO) further reduced forest stocking to an average of 94 trees ha^{-1} (38 trees ac^{-1}) with a basal area of 12 m^2 ha^{-1} (53 ft^2 ac^{-1}) and a QMD of 41 cm d.b.h (16.2 in d.b.h). The 5xPRO and 5xRO scenarios were most effective at returning the historical stocking and structure of ponderosa pine to the study area (Reynolds et al., 2013, interquartile range: 56 to 138 trees ha^{-1} with 9 to 17 m^2 ha^{-1}) (Allen et al., 2002; Bauhus et al., 2009). Notably, at the end of the simulation, RO wildfire management had the added benefit of reducing ponderosa pine stocking in stands adjacent to restoration treatments and thus provided indirect restoration benefits.

3.2. Wildfire mortality and area burned

Mean conditional wildfire mortality (mortality given a wildfire) at the end of the simulation also revealed differences among the scenarios related to area burned by severity and the presence of old growth forest conditions (Fig. 5). In the absence of treatment and RO management (0xPRO) future burned area was limited and infrequent but had a greater relative occurrence of high severity wildfire. Increasing treatment to 1x (1xPRO) reduced wildfire activity and severity overall, with an increase in burned area in stands that had been previously treated (see Fig. 1 for areas available for treatment). On the other hand, implementing RO wildfire management (0xRO) more evenly distributed wildfire activity in and around treatment areas, expanding heterogeneous fire effects across the landscape, while disaggregating large, continuous tracks of dense forests with the potential to support high severity fire. When 1x treatment levels and RO wildfire management were used in concert (business-as-usual) future wildfire activity was widespread with large areas of low to moderate fire severity, both in and around treatment areas, with occurrence of high severity patches throughout. The 5xPRO scenario resembled 1xRO, except that the PRO wildfire management increased fire activity in treated areas relative to untreated areas. Conversely the restored landscape of the 5xRO scenario had the greatest burned area, expanding heterogeneous fire effects in and around treatment areas.

Notable differences in wildfire activity emerged between total area burned and area burned with high severity (Fig. 6). Though higher treatment levels and RO wildfire management both increased total area burned, the area burned with high severity remained relatively low.

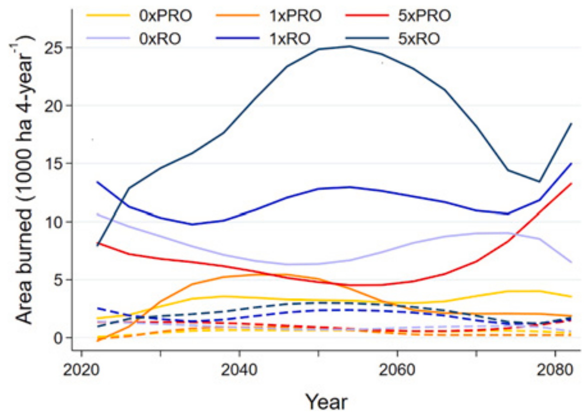

Fig. 6. Mean total area burned (solid line) and mean area burned with high-severity wildfire (dashed lines) in the study area by scenario and year. Area burned with high severity are those with > 75 % mortality of trees ≥ 13 cm d.b.h. as the result of wildfire activity. Data are displayed using locally weighted regressions. 0x, 1x, and 5x represent treatment levels. PRO represents protection objective wildfire management and RO is inclusive of resource objective wildfire management. Refer to Table 2 for scenario details.

Focusing on the transition from 1xRO to 5xRO, the increase in high-severity wildfire (Fig. 6, dark blue dashed line – medium blue dashed line) was relatively small in comparison to the increase in low to moderate severity wildfire (Fig. 6, [dark blue line – dark blue dashed line] – [medium blue line – medium blue dashed line]). Focusing on the

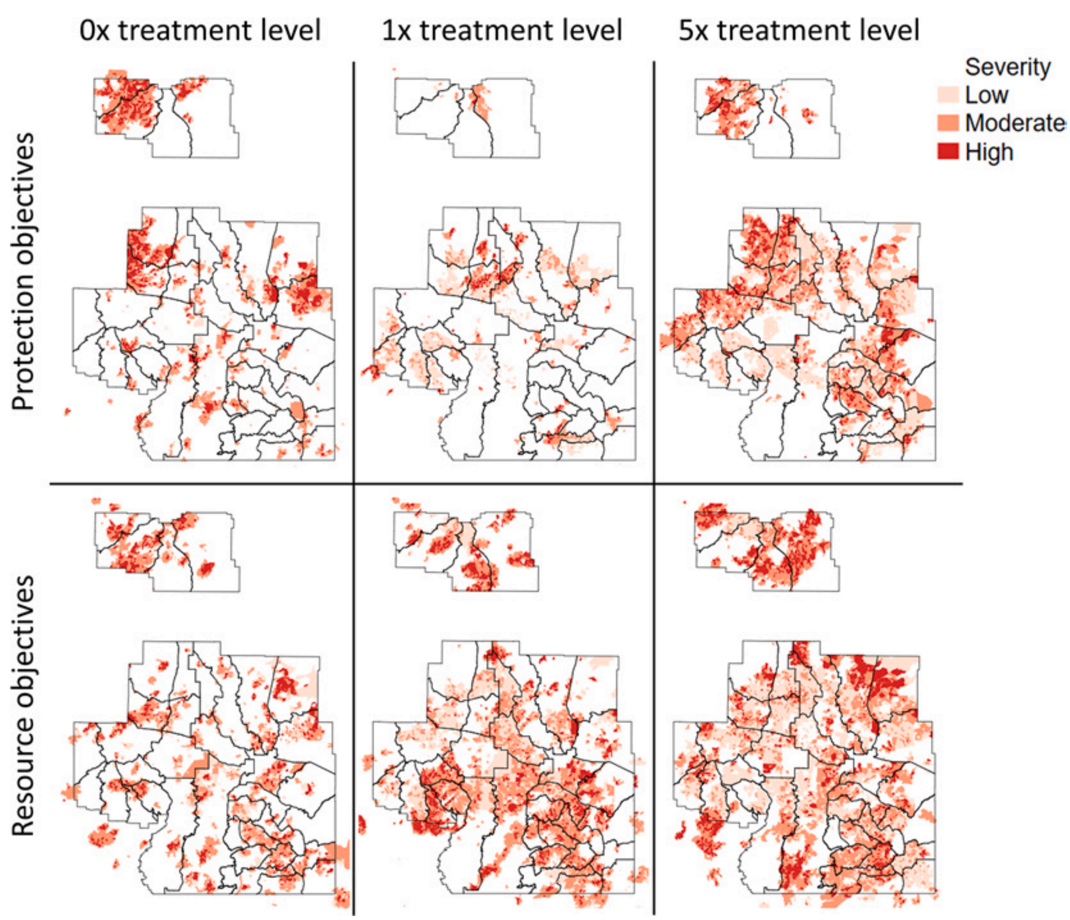

Fig. 5. Mean conditional wildfire mortality by severity class from 2079 to 2082 for each scenario. Severity classes were averaged when a stand burned in multiple replicates. Severity was based on wildfire-induced mortality of trees ≥ 13 cm d.b.h. (Barrett et al., 2010). Low: 〈 25 % mortality; Moderate: ≥ 25 % and ≤ 75 %; High: 〉 75 %. Burned stands without trees ≥ 13 cm d.b.h. at the time of the wildfire are not included. Refer to Table 2 for scenario details.

transition from 1xPRO to 5xPRO, total area burned increased dramatically early in the simulation, with a similar but muted increase in high-severity wildfire, both of which decreased over time before increasing again. Conversely, 1xPRO had lower wildfire activity early in the simulation, with both total area burned and area burned with high severity increasing overtime until declining again. Rapidly restoring the landscape with the 5xRO scenario led to a rapid increase in low- to moderate-severity wildfire in the midterm that began to moderate as old growth characteristics were restored (Allen et al., 2002; Bauhus et al., 2009; Reynolds et al., 2013). The business-as-usual scenario (1xRO) maintained a steady, high level of low- to moderate-severity wildfire, with the landscape beginning to approach an old growth structure by simulation's end.

When examining wildfire mortality (Figs. 5 and 6), several important factors should be considered. First, forest restoration can open the forest canopy and increase the production of fine fuel, while also increasing wind intrusion, resulting in increased fire spread (Young et al., 2022) and an increased prevalence of high severity fire during arid summer seasons. Conversely, during docile weather conditions unrestored forests with closed canopies tend to resist the incursion of fire due to dense canopies hindering wind intrusion and maintaining higher humidity levels along the forest floor (Harrison et al., 2021; He et al., 2016; Hoffman et al., 2011). Second, restored frequent-fire forests and savannas in our simulations contain scattered pockets of advanced regeneration (i.e., small to medium-sized trees). Pockets of advanced regeneration periodically experience high mortality rates, while large trees with fire-adapted traits, such as thick insulating bark, have a higher likelihood of survival, leaving the old growth forest structure intact (He et al., 2016; Pausas and Keeley, 2023).

3.3. Carbon stocks

Aboveground mean C stocks were affected by treatment levels and wildfire management to varying degrees (Fig. 7A). An increase in the treatment level (1x, 5x) decreased C stocks across both wildfire management regimes (PRO, RO), with the decrease being dramatic within 8 years for the 5x treatment level. However, following 5x treatment, aboveground C stocks returned to pre-treatment levels within 20 years (2038).

Individual components of aboveground C in the study area were also affected by treatment level and wildfire management (Fig. 7B). By 2082, live tree C was reduced equally by the 1x treatment level (1xPRO) and RO wildfire management (0xRO) when compared to the no treatment scenario (0xPRO). The C in coarse woody debris was reduced most by the 5x treatment level and least by RO wildfire management. Both 5x management scenarios (5xPRO, 5xRO) along with the 1xRO scenario increased snags, providing habitat for bats and other wildlife (Johnson and Chambers, 2017). The C stored in shrubs and herbs did not substantially change when compared to C stored in other forest components.

3.4. Fire-resistant live tree carbon

Fire-resistant live tree C in the study area was affected by each scenario in distinct ways reflecting the complexity and tradeoffs associated with restoration treatments and wildfire (Fig. 8). Resistant live tree C is live tree C expected to survive severe wildfire (70°F with 20-foot windspeed of 20 mph, Rebain et al., 2010). Though both 5x treatment scenarios reduced live tree C greatly in the short term, it is notable that residual live trees reached pre-treatment C levels in 16 years (2034), which was not reached with 1x and 0x treatment levels for 32 (2050) and 42 years (2060). Additionally, after the 5x treatment level completed the first treatment cycle (2030), the proportion of resistant live tree C (Appendix A, Fig. A2) approached 1.00, which remained high following regular maintenance that was triggered every 20 years (primarily prescribed fire). With 0xPRO and 1xPRO scenarios, the proportion of resistant live tree C approached 0.65 and 0.78, with RO wildfire management making a considerable contribution in both cases, increasing these values to 0.70 and 0.85 with 0xRO and 1xRO. Though the 0xPRO scenario maximized live tree C (Fig. 7B), 5x treatment levels

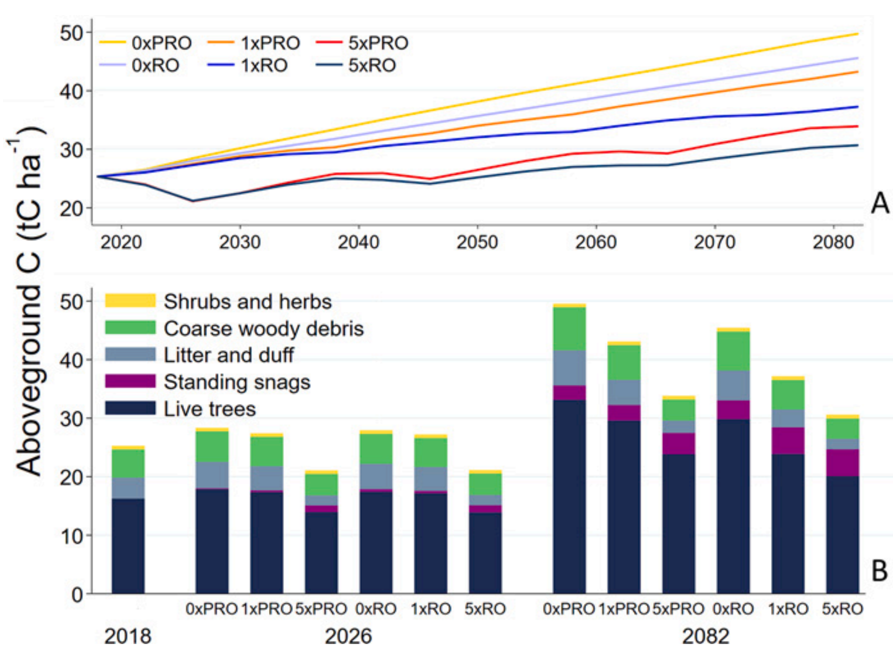

Fig. 7. (A) Aboveground mean C stocks in the study area by scenario and year. Treatments each cycle were sequenced on the landscape based on expected proportion of basal area loss in a severe wildfire (70°F with 20-foot windspeed of 20 mph, Rebain et al., 2010). Implementation was constrained by yearly treatment levels or quotas and tree stands available for treatment based on treatment criteria (Young et al., 2022, Appendix S2). (B) Total aboveground mean C stocks by scenario and C stock type at the initiation of the simulation (2018), and after 8 (2026) and 64 years (2082). Coarse woody debris includes all dead surface fuels. Litter and duff are fuels that compose the forest floor. Snags are standing dead. 0x, 1x, and 5x represent treatment levels. PRO represents protection objective wildfire management and RO is inclusive of resource objective wildfire management. Refer to Table 2 for scenario details.

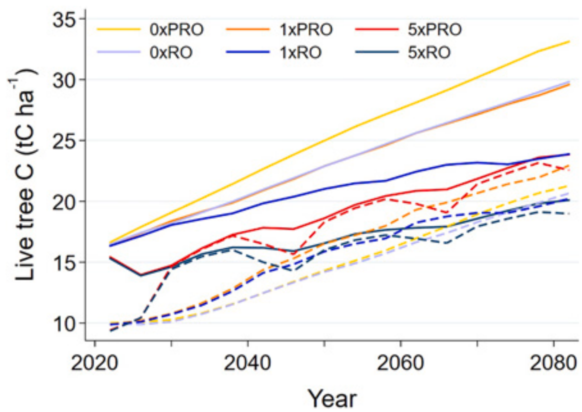

Fig. 8. Aboveground mean live tree C (solid lines) and aboveground mean resistant live tree C (dashed lines) in the study area by scenario. Graph shows that scenarios with high levels of restoration management (5x) have the largest reduction in live tree C but also have the highest level of resistant live tree C. 0x, 1x, and 5x represent treatment levels. PRO represents protection objective wildfire management and RO is inclusive of resource objective wildfire management. Refer to Table 2 for scenario details.

most reliably increased the proportion of resistant live tree C (Appendix A, Fig. A2).

3.5. Carbon emissions

The C released from fire was highly dependent on the yearly rate of treatments with wildfire management playing an increasing role when treatment levels declined (Fig. 9A). Emissions were greatest with 5x treatments and lowest with 0x treatments. Reviewing the components of emissions (Fig. 9B), prescribed fires accompanying 5x treatments led to the greatest accumulation of C emissions following a 20-year cyclical pattern of maintenance burning. By simulations' end (2082), prescribed fire emissions accompanying the 1x treatment level exceeded wildfire emissions regardless of the wildfire management strategy. Uniquely, the 5xRO scenario experienced a decline in prescribed fire emissions when compared to 5xPRO, signifying a possible decrease in the need for maintenance burning when RO wildfire management was used alongside accelerated restoration thinning and unconstrained prescribed fire operations.

The use of RO wildfire management also affected the seasonality of C emissions. Wildfires that ignited during the spring (prior to May) and fall seasons (after July) had emissions that exceeded other wildfires across all scenarios (Fig. 9B, 2082). The RO scenarios had similar levels of high-severity wilfire emissions in shoulder (i.e., spring and fall) and summer seasons (Fig. 10, 2082). At the same time, cumulative emissions from low and moderate severity wildfire were much greater during the shoulder season in RO scenarios. Notably, in the last 20 years of simulated wildfire activity (Fig. 10, 2063 to 2082), the use of RO wildfire management reduced summer emissions by 17 % and 2 %, respectively, when combined with 0x and 5x treatment. However, in the case of the 1x treatment level, the use of RO wildfire management (1xRO) doubled summer wildfire emissions with a notable increase in high-severity emissions. Under 1xRO, the rate of maintenance burning failed to limit the accumulation of surface and ladder fuels, while increasing wildfire activity from RO wildfire management continued.

Examining C from saw log and harvest residuals, there was little difference between 1x and 5x treatment levels (Fig. 9B). By simulations' end, the 5x treatment level only produced 25 % more emissions than the 1x treatment level, despite enacting management at 5x the rate. This is reflective of the limited quantity of merchantable timber being removed from the system, which is by design based on a diameter cap for harvested trees (i.e., in order to retain old growth forest structure, forest thinning operations achieving desired conditions could remove no more than half of the ponderosa pine with a d.b.h. greater than 40.6 cm (FFRI, 2011)). In both cases, forest-based emissions from industrial activities were substantially lower than C released from other sources. Recall, we assume accumulated emissions from sawlog residuals roughly equate

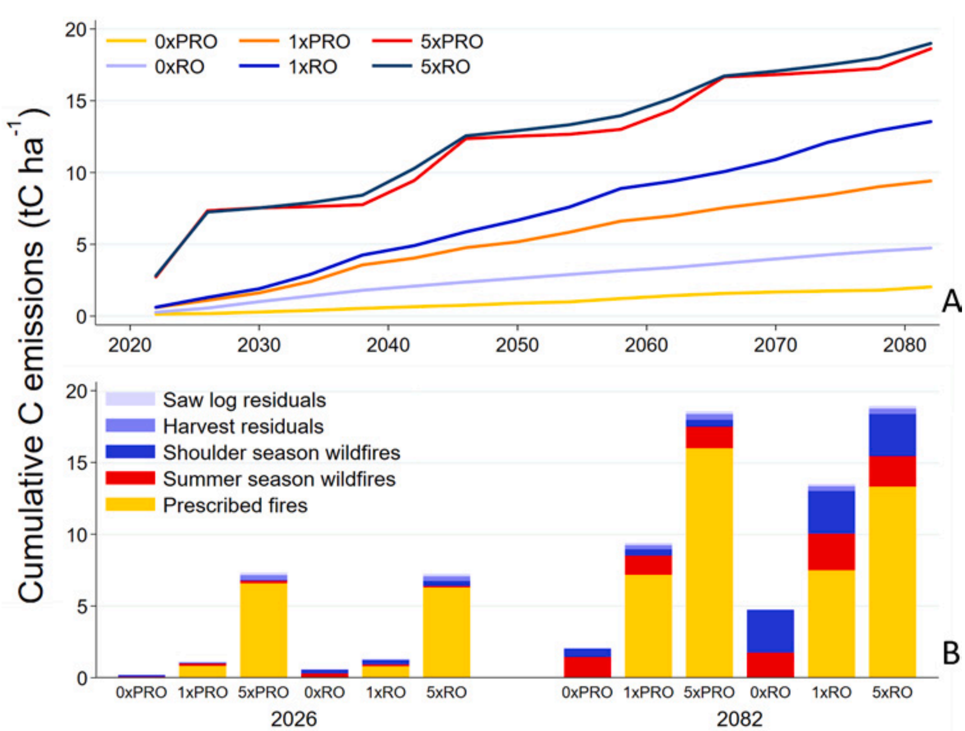

Fig. 9. (A) Mean cumulative C emissions in the study area by scenario. (B) Mean cumulative C emissions in the study area by scenario and emissions type after 8 (2026) and 64 years (2082). See Fig. 10 for seasonal wildfire emissions by severity type. 0x, 1x, and 5x represent treatment levels. PRO represents protection objective wildfire management and RO is inclusive of resource objective wildfire management. Refer to Table 2 for scenario details.

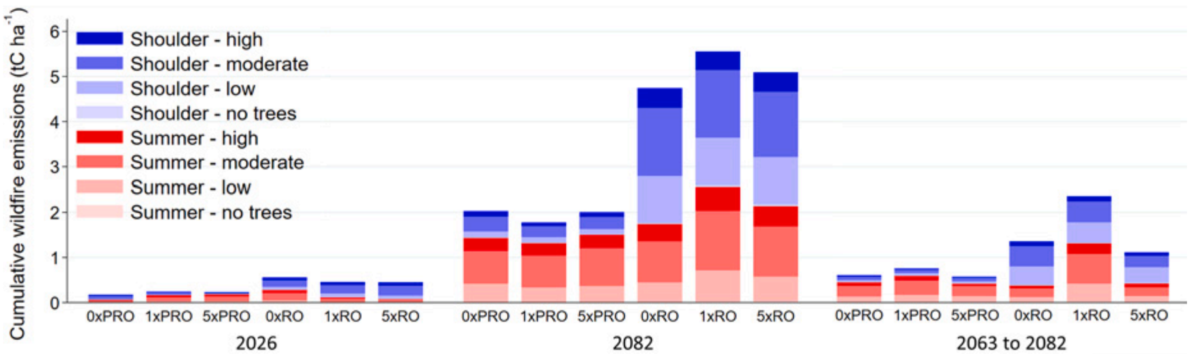

Fig. 10. Mean cumulative wildfire C emissions in the study area by scenario, season, and severity after 8 (2026), 64 years (2082), and from 2063 to 2082. Shoulder season wildfires ignited prior to May or after July. Summer season wildfires ignited from May to July. Severity was based on wildfire-induced mortality of trees ≥ 13 cm d.b.h. (Barrett et al., 2010). No trees: no trees ≥ 13 cm d.b.h.; Low: 〈 25 % mortality; Moderate: ≥ 25 % and ≤ 75 %; High: 〉 75 %. 0x, 1x, and 5x represent treatment levels. PRO represents protection objective wildfire management and RO is inclusive of resource objective wildfire management. Refer to Table 2 for scenario details.

the C stored in harvested wood products (Steele, 1984).

3.6. Net carbon

The aboveground net C diverged widely across scenarios with treatment level being the primary influence and wildfire management being a secondary but important factor (Fig. 11). The 1x treatment level represented a tipping point, where PRO wildfire management resulted in an increase in aboveground net C over time, while RO wildfire management resulted in a decrease. Similarly, the 0x treatment level resulted in an increase in aboveground net C, while the 5x treatment level resulted in a substantial decrease early in the simulation with cyclical recoveries. In both instances, RO wildfire management resulted in reduced levels in aboveground net C. Following 2046, the accumulation in aboveground C from one cycle to the next in the 5xPRO and 5xRO scenarios began to roughly equate C emissions that accumulated from one cycle to the next, including C additions to harvested wood products. Following 2066, the 5xRO scenario had net C from cycle-to-cycle that was near zero. Management approaching the 5xRO scenario best reflected the pace and scale of wildland fire needed to achieve net zero aboveground C emissions (burning 6 % of the landscape year^{-1}), which replicates a return to the historic fire return interval (FRI) of frequent-fire ponderosa pine forests (FRI: 5xRO = 16.8 years; historic = 2.1 to 17.1 years) in an old growth forest structure (Sneed et al., 2002; Swetnam and Baisan, 1996).

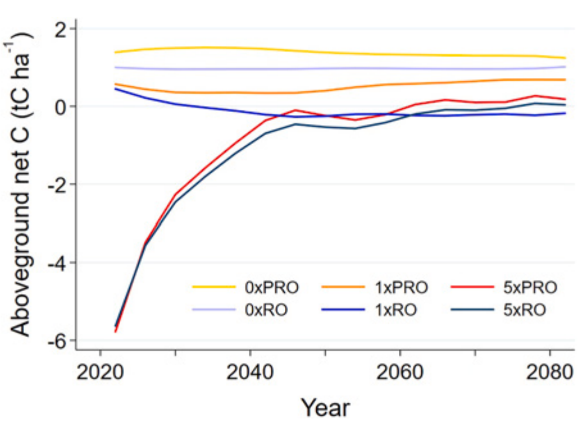

Fig. 11. Mean aboveground net C in the study area by scenario. Aboveground net C of a forested ecosystem is in balance when the accumulation of aboveground C from one cycle to the next equates the accumulation in aboveground C emissions from one cycle to the next, including C additions to harvested wood products (i.e., ecological-cycle assessment). Data are displayed using a locally weighted regression of aboveground net C. Positive values represent net additions to aboveground C stocks (i.e., net C sink). Negative values represent net reductions to aboveground C stocks (i.e., net C emitter). A value of zero represents a net neutral aboveground C flux. 0x, 1x, and 5x represent treatment levels. PRO represents protection objective wildfire management and RO is inclusive of resource objective wildfire management. Refer to Table 2 for scenario details.

4. Discussion

This study examined wildfire futures and C dynamics for dry, fire-excluded pine forests in the southwestern US. Initial aboveground C stocks were consistent with previous studies (20.1 tC/ha, Hoover and Smith, 2021), and an old growth forest structure was most efficiently obtained when an aggressive treatment level (5x) was combined with resource objective (RO) wildfire management (Fig. 4). Negative impacts were also identified including RO wildfire spreading to neighboring untreated landscapes with a high risk of severe fire during summer months. Aggressive 5x treatments led to faster fire spread rates by increasing production of fine surface fuels and opening the forest canopy, increasing mid-flame wind speeds and easily spreading fire to untreated dense forest canopies (Young et al., 2022). However, the areas burned by wildfire in the 5xRO scenario had a higher proportion of low severity wildfire and a positive spillover effect on untreated lands. The 5xRO scenario also improved landscape heterogeneity in forest structure and standing tree snags, improving forest health and increasing wildlife habitat (Allen et al., 2002; Johnson and Chambers, 2017; Jones et al., 2021; Reynolds et al., 1992; 2013; Seidl et al., 2018; USFWS, 2012).

Consistent with prior research (e.g., Foster et al., 2020), we also observed an increase in fire-resistant live tree C when hazardous fuels were reduced and old growth forest structures were restored. Most dramatically, the accelerated application of mechanical thinning and prescribed fire (5x) maximized the proportion of fire-resistant live tree C, with RO wildfire management playing a more substantial role as the treatment level was reduced. Conversely, in the absence of mechanical thinning and prescribed fire (0x) live tree C was maximized (Fig. 8), while the proportion of specifically fire-resistant live tree C was at its lowest (Appendix A, Fig. A2). This dichotomy suggests that in the short- to medium-term, the yearly treatment rate in frequent fire forests may have a greater influence on future outcomes than high severity wildfire activity, with the latter being dependent on the former. Healthy coniferous forests are more likely to be both resist and be resilient to the deleterious effects of drought, pests and increasing wildfire activity (Kolb et al., 2020; Křivan et al., 2016; Stoddard et al., 2021).

Leveraging natural disturbances in the form of RO wildfire management during periods of mild weather also affected the timing and quantity of fire induced C emissions. Principally, RO wildfire management in the spring and fall more than doubled yearly C emission from smoke production, highlighting the need to consider how extended smoke exposure may impact human health when optimizing forest management strategies (D'Evelyn and al., 2022; Williamson et al., 2016). However, summer wildfire emissions were reduced by 17 % in a negative feedback loop over the last 20 years of the simulation (Fig. 10, 2063 to 2082) when RO wildfire management was sustained in the absence of mechanical thinning and prescribed fire (0x). Conversely, the 1x treatment level alongside RO wildfire management (1xRO), led to an increase in summer wildfire emissions, despite full suppression strategies being applied. By comparison, the 1x treatment level combined with PRO wildfire management maintained a smaller future fire footprint (Fig. 5) but failed to restore an old growth forest structure to the extent of 1xRO (Fig. 4).

The simulation results suggest differing potentials for each scenario to reach an aboveground net C equilibrium (Fig. 11) (Canelles et al., 2019). The current level of restoration treatments and prescribed fire (1x) combined with RO wildfire management reflected a tipping point between a continuation of increasing C stores in unhealthy forest structures and decreasing C stores in the near- to mid-term through the intentional act of restoration. Accelerating restoration treatment levels (5x) initially led to a dramatic reduction in aboveground C stores as an excess of small diameter trees were removed, as observed in other studies (Ager et al., 2022; McCauley et al., 2019; Young et al., 2022). Sixteen years later (2034), live tree C in the residual forest structure reached pre-treatment levels (Fig. 8), driven in part by large trees accumulating C at a faster pace than small trees in a dense forest structure (Hudiburg et al., 2009; Mildrexler et al., 2020). After twelve additional years (2046), the aboveground net C began to stabilize (Fig. 11) with the convergence of an old growth forest structure returning to the study area, alongside maintenance treatments with an emergent need-to-burn rate of every 20 years (Fig. 7A) (Allen et al., 2002; Bauhus et al., 2009). By simulation's end, the addition of RO wildfire (5xRO) best replicated the pace and scale of wildland fire needed to achieve a neutral aboveground C flux – burning 6 % of the landscape year^{-1} with a fire return interval of 16.8 years, replicating the historic fire return interval of frequent-fire ponderosa pine systems (2.1 to 17.1 years) (Sneed et al., 2002; Swetnam and Baisan, 1996). The landscape's C carrying capacity began to reach a maximum as the landscape approached a diverse, old growth equilibrium structure sequestering 20 tC ha^{-1} in fire-resistant live tree C (a 31 % increase compared to the simulations start) (Fig. 7A, 5xRO) while minimizing the potential for fire-driven conversions of forested areas (Walker et al., 2018).

It should be noted that prior modeling studies on the long term effects of management on C pools have led to highly variable findings (James et al., 2018), resulting from a diversity of approaches and key assumptions (Ager et al., 2022). This includes assumptions on the potential effects of climate and fire on forest succession, especially tree mortality and seedling establishment. Even minor adjustments in modeling parameters can reverberate through simulation studies as interactions between parameters playout over time. Further diversifying scenarios to explore parameter interactions between climate induced deviations can further enhance sensitivity analysis of C stocks (Campbell and Ager, 2013) including dynamics between above and belowground C pools. Though we do not consider belowground C stocks in detail, we found them to be positively associated with aboveground C with limited differences between scenarios (Appendix A, Fig. A1), which is consistent with previous studies (Mayer et al., 2020). To the extent that long-term forest restoration and fuel management programs can decrease high severity fire and perpetuate an old growth forest structure (Bauhus et al., 2009; Kolb et al., 2007; Luyssaert et al., 2008), belowground C storage will remain resilient (Liang et al., 2018) and continue to serve as a vital and complex C sink (Liu et al., 2019).

In general, this study contributes to a clearer understanding of how management interacts with wildfire to achieve landscape-scale equilibriums, findings that are relevant to a wide range of ecologies and geographic regions (Grace et al., 2014; Kashian et al., 2006; Mitchard, 2018; Odum, 1969; Sousa, 1984). This study sets the stage for further investigations into the restoration of fire-excluded pine forests in terms of climate-driven deviations and their effects on wildfire activity, tree growth and mortality, and seedling establishment (Bradford et al., 2020; Kolb et al., 2020; Petrie et al., 2023). We believe the key findings reported here will be sustained in future modeling studies related to restoring old growth forest structures and historic fire regimes, and improving the resilience of dry, temperate forests (Bradford et al., 2020; McCauley et al., 2022; Montwé et al., 2022; Stoddard et al., 2021; Walker et al., 2018).

5. Conclusions

The results of this study suggest that the expanded use of RO wildfire management as part of restoration programs in dry pine forests can have both positive and negative outcomes that are co-dependent on the intensity of forest and fuel management treatments, especially with respect to the stabilization of C stored in resilient old growth forest structures. The potential for negative outcomes under management practices that include RO wildfire are numerous, including increased C emissions, and the potential for escaped fires and resulting damage. Adopting a long-term management strategy that depends on uncertain fire events as a treatment poses additional challenges to long-term restoration programs. On the other hand, there is significant risk to long-term C storage in overstocked forests susceptible to high-severity fire and long-term vegetation type conversion, a process that is increasingly observed in pine forests of the western United States. Further refinements of forest landscape models, including carefully calibrated parameters for future fire regimes and post-fire regeneration under changing climate, will be required to further understand the long term positive and negative aspects of RO wildfire management as part of ongoing restoration programs in the western United States and elsewhere.

Disclaimer

The findings and conclusions in this report are those of the author(s) and should not be construed to represent any official USDA or U.S. Government determination or policy. This research was supported by the U.S. Department of Agriculture, Forest Service. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. government.

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Availability of data and materials

Simulation data (Young, 2023) used to develop figures are available from the United States Forest Service, Forest Service Research Data Archive: https://doi.org/10.2737/RDS-2023-0065.

Credit author statement

Study conception and design: JDY, AAA; data collection: JDY; data analysis: JDY; interpretation of results: JDY, AAA; manuscript preparation: JDY; manuscript review: JDY, AAA

Funding

This research was supported by the United States Forest Service, Rocky Mountain Research Station, National Fire Decision Support Center (RMRS-NFDSC), a Joint Venture Agreement between RMRS-NFDSC and Northern Arizona University (17-JV-11221,637-123), and the McIntire–Stennis Cooperative Forestry Research Program (G1003488).

Declaration of Competing Interest

The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

Acknowledgments

We thank Luke Sheivlin, GreenWave Professional, LLC, Missoula, MT, for programming support; Andrea E. Thode, Robert C. Seli, Rachel Houtman, and Ching-Hsun Huang for their important input during model development; the land and fire management professionals of northern Arizona; and Michelle Day and anonymous reviewers for their time and insightful comments.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.ecolmodel.2023.110573.

References

Ager, A.A., Barros, A., Preisler, H.K., Day, M.A., Spies, T., Bailey, J., Bolte, J., 2017. Effects of accelerated wildfire on future fire regimes and implications for the United States federal fire policy. Ecol. Soc. 22, 12.
Ager, A.A., Barros, A.M., Houtman, R., Seli, R., Day, M.A., 2020. Modelling the effect of accelerated forest management on long-term wildfire activity. Ecol. Modell. 421, 108962.
Ager, A.A., Barros, A.M.G., Day, M.A., 2022. Contrasting effects of future wildfire and forest management scenarios on a fire excluded western US landscape. Landsc. Ecol. 37, 1091–1112.
Ager, A.A., Evers, C.R., Day, M.A., Alcasena, F.J., Houtman, R., 2021. Planning for Future fire: Scenario Analysis of an Accelerated Fuel Reduction Plan for the Western United States 215.
Ager, A.A., Finney, M.A., Kerns, B.K., Maffei, H., 2007. Modeling wildfire risk to northern spotted owl (*Strix occidentalis* caurina) habitat in Central Oregon. USA. For. Ecol. Manag. 246, 45–56.
Ager, A.A., Finney, M.A., McMahan, D., 2006. A wildfire risk modeling system for evaluating landscape fuel treatment strategies. In: Andrews, P.L., Butler, B. (Eds.), Fuels Management-How to Measure Success. Proceedings RMRS-P-41. USDA Forest Service, Rocky Mountain Research Station, pp. 149–162.
Ager, A.A., Vaillant, N.M., Finney, M.A., 2010. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. For. Ecol. Manage. 259, 1556–1570.
D.A., F. Allen, C.D., Savage, M., Suckling, K.F., Swetnam, T.W., Schulke, T., Stacey, P.B., Morgan, P., Hoffman, M., Klingel, J.T., 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective Ecol. Appl. 12, 1418–1433.
Ameray, A., Bergeron, Y., Valeria, O., Girona, M., Cavard, X., 2021. Forest carbon management: a review of silvicultural practices and management strategies across boreal, temperate and tropical forests. Curr. For. Rep. 7.
Barrett, S., Havlina, D., Jones, J., Hann, W., Frame, C., Hamilton, D., Schon, K., Demeo, T.E., Hutter, L., Menakis, J., 2010. Interagency Fire Regime Condition Class guidebook. Version 3.0. USDA Forest Service, US Department of Interior, and The Nature Conservancy, Homepage of the Interagency Fire Regime Condition Class website.
Barros, A.M.G., Ager, A.A., Day, M.A., Krawchuk, M., Spies, T.A., 2018. Wildfires managed for restoration enhance ecological resilience. Ecosphere 9, e02161. https://doi.org/10.1002/ecs2.2161.
Bauhus, J., Puettmann, K., Messier, C., 2009. Silviculture for old-growth attributes. For. Ecol. Manage. 258, 525–537.
Bone, C., Shultz, C., Huber-Stearns, H., Kelley, J., Cunnin, E., 2023. Evaluating the potential role of federal air quality standards in constraining applications of prescribed fire in the western United States. Appl. Geogr. 157.
Bradford, J.B., Andrews, C., Robles, M.D., McCauley, L.A., Woolley, T., Marshall, R.M., 2020. Landscape-scale restoration minimizes tree growth vulnerability to 21st century drought in a dry forest. Ecol. Appl. 0.
Brienen, R.J.W., 2015. Long-term decline of the Amazon carbon sink. Nature 519.
Busenberg, G., 2004. Wildfire management in the United States: the evolution of a policy failure. Rev. Policy Res. 21, 145–156.
Byram, G.M., 1959. Forest fire behaviour. In: Davis, K. (Ed.), Forest Fire Control and Use. McGraw-Hill, New York, pp. 90–123.
Campbell, J.L., Ager, A.A., 2013. Forest wildfire, fuel reduction treatments, and landscape carbon stocks: a sensitivity analysis. J. Environ. Manage. 121, 124–132.
Canelles, Q., Aquilué, N., Duane, A., Brotons, L., 2019. From stand to landscape: modelling post-fire regeneration and species growth. Ecol. Modell.
Cansler, C.A., McKenzie, D., 2014. Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. Ecol. Appl. 24, 1037–1056.
Carney, T.J., Williams, R.J., Kapellusch, T., Crandall, R., Susaeta, A., Adams, D., 2023. Fanning the flames of cooperation: a collective action approach to prescribed fire agreements. Environ. Challenges 10.
Chambers, M.E., Fornwalt, P.J., Malone, S.L., Battaglia, M.A., 2016. Patterns of conifer regeneration following high severity wildfire in ponderosa pine–dominated forests of the Colorado Front Range. For. Ecol. Manage. 378, 57–67.
Chang, H., Han, H.-S., Anderson, N., Kim, Y.S., Han, S.-K., 2022. The cost of forest thinning operations in the western united states: a systematic literature review and new thinning cost model. J. For.
Coop, J., 2023. Postfire futures in southwestern forests: climate and landscape influences on trajectories of recovery and conversion. Ecol. Appl.
Coop, J.D., Parks, S.A., Stevens-Rumann, C.S., Crausbay, S.D., Higuera, P.E., Hurteau, M. D., Tepley, A., Whitman, E., Assal, T., Collins, B.M., Davis, K.T., Dobrowski, S., Falk, D.A., Fornwalt, P.J., Fule, P.Z., Harvey, B.J., Kane, V.R., Littlefield, C.E., Margolis, E.Q., North, M., Parisien, M.A., Prichard, S., Rodman, K.C., 2020. Wildfire-driven forest conversion in Western North American landscapes. Bioscience 70, 659–673.
Covington, W.W., Fule, P.Z., Moore, M.M., Hart, S.C., Kolb, T.E., Mast, J.N., Sackett, S.S., Wagner, M.R., 1997. Restoring ecosystem health in ponderosa pine forests of the southwest. J. For. 95, 23–29.
Covington, W.W., Moore, M.M., 1994. Postsettlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. J. Sustain. For. 2, 153–181.
Crookston, N.L., Dixon, G.E., 2005. The forest vegetation simulator: a review of its structure, content, and applications. Comput. Electron. Agric. 49, 60–80.
Crookston, N.L., Stage, A.R., 1991. User's guide to the parallel processing extension of the prognosis model. General Technical Report INT-281. USDA Forest Service, Intermountain Research Station, Ogden, UT, p. 93 pp.
D'Evelyn, S.M., 2022. Wildfire, Smoke exposure, Human Health, and Environmental Justice Need to be Integrated into Forest Restoration and Management. Current Environmental Health Reports.
Davis, K.T., 2023. Reduced fire severity offers near-term buffer to climate-driven declines in conifer resilience across the western United States. Proc. Natl Acad. Sci. 120.
Dixon, G.E., 2002. Essential FVS: A User's Guide to the Forest Vegetation Simulator. USDA Forest Service, Forest Management Service Center, Fort Collins, CO, p. 226. Internal Report.
Doane, D., O'Laughin, J., Morgan, P., Miller, C., 2006. Barriers to wildland fire use: a preliminary problem analysis. Int. J. Wilderness 12, 36–38.
FFRI, 2011. Old Growth Protection & Large Tree Retention Strategy. Four Forst Restoration Initiative.
Finney, M.A., Grenfell, I.C., McHugh, C.W., 2009. Modeling containment of large wildfires using generalized linear mixed model analysis. For. Sci. 55, 249–255.
Finney, M.A., McHugh, C.W., Grenfell, I.C., Riley, K.L., Short, K.C., 2011. A simulation of probabilistic wildfire risk components for the continental United States. Stochast. Environ. Res. Risk Assess. 25, 973–1000.
Finney, M.A., Seli, R.C., McHugh, C.W., Ager, A.A., Bahro, B., Agee, J.K., 2007. Simulation of long-term landscape-level fuel treatment effects on large wildfires. Int. J. Wildland Fire 16, 712–727.
Foster, D.E., Battles, J.J., Collins, B.M., York, R.A., Stephens, S.L., 2020. Potential wildfire and carbon stability in frequent-fire forests in the Sierra Nevada: trade-offs from a long-term study. Ecosphere 11, e03198.
Fulé, P.Z., Crouse, J.E., Roccaforte, J.P., Kalies, E.L., 2012. Do thinning and/or burning treatments in western USA ponderosa or Jeffrey pine-dominated forests help restore natural fire behavior? For. Ecol. Manage. 269, 68–81.
Grace, J.B., Mitchard, E., Gloor, E., 2014. Perturbations in the carbon budget of the tropics. Glob. Chang. Biol. 20.
Greenler, S.M., Dunn, C.J., Johnston, J., Reilly, M.J., Merschel, A.G., Hagmann, K., Bailey, J.D., 2023. Too hot, too cold, or just right: can wildfire restore dry forests of the interior Pacific Northwest? PLoS One.
Harmon, M.E., Bond-Lamberty, B., Tang, J., Vargas, R., 2011. Heterotrophic respiration in disturbed forests: a review with examples from North America. J. Geophys. Res. 116.
Harrison, S.P., Prentice, I.C., Bloomfield, K.J., Dong, N., Forkel, M., Forrest, M., Ningthoujam, R.K., Pellegrini, A., Shen, Y., Baudena, M., 2021. Understanding and modelling wildfire regimes: an ecological perspective. Environ. Res. Lett. 16, 125008 https://doi.org/10.1088/1748-9326/ac39be.
Hartsough, B.R., Abrams, S., Barbour, R.J., Drews, E.S., McIver, J.D., Moghaddas, J.J., Schwilk, D.W., Stephens, S.L., 2008. The economics of alternative fuel reduction treatments in western United States dry forests: financial and policy implications from the National Fire and Fire Surrogate Study. For. Policy Econ. 10, 344–354.
He, T., Belcher, C.M., Lamont, B.B., Lim, S.L., 2016. A 350-million-year legacy of fire adaptation among conifers. J. Ecol. 104, 352–363.

Hessburg, P., Prichard, S., Hagmann, K., Povak, N.A., Lake, F.K., 2021. Wildfire and climate change adaptation of western North American forests: a case for intentional management. Ecol. Appl. 31.
Hjerpe, E., Abrams, J., Becker, D.R., 2009. Socioeconomic barriers and the role of biomass utilization in Southwestern ponderosa pine restoration. Ecol. Restorat. 27, 169–177.
Hoffman, C.M., Sieg, C.H., McMillin, J.D., Fulé, P.Z., 2011. Fuel loadings 5 years after a bark beetle outbreak in south-western USA ponderosa pine forests. Int. J. Wildland Fire.
Hoover, C.M., Smith, J.E., 2021. Current aboveground live tree carbon stocks and annual net change in forests of conterminous United States. Carbon Balance Manag. 16, 1–12.
Hudiburg, T., Law, B., Turner, D.P., Campbell, J., Donato, D., Duane, M., 2009. Carbon dynamics of Oregon and Northern California forests and potential land-based carbon storage. Ecol. Appl. 19, 163–180.
Huffman, D.W., Crouse, J.E., Sánchez Meador, A.J., Springer, J.D., Stoddard, M.T., 2018. Restoration benefits of re-entry with resource objective wildfire on a ponderosa pine landscape in northern Arizona. USA. For. Ecol. Manag. 408, 16–24.
Huffman, D.W., Meador, A.J.S., Stoddard, M.T., Crouse, J.E., Roccaforte, J.P., 2017. Efficacy of resource objective wildfires for restoration of ponderosa pine (*Pinus ponderosa*) forests in northern Arizona. For. Ecol. Manage. 389, 395–403.
Huffman, D.W., Roccaforte, J.P., Springer, J.D., Crouse, J.E., 2020. Restoration applications of resource objective wildfires in western US forests: a status of knowledge review. Fire Ecol. 16, 18. https://doi.org/10.1186/s42408-020-00077-x.
Hunter, M.E., Iniguez, J.M., Farris, C.A., 2014. Historical and Current Fire Management Practices in Two Wilderness Areas in the Southwestern United States: The Saguaro Wilderness Area and the Gila-Aldo Leopold Wilderness Complex. Gen. Tech. Rep. RMRSGTR-325, Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
Hunter, M.E., Iniguez, J.M., Lentile, L.B., 2011. Short- and long-term effects on fuels, forest structure, and wildfire potential from prescribed fire and resource benefit fire in southwestern forests. USA. Fire Ecol. 7, 108–121.
Hurteau, M.D., Liang, S., Martin, K.L., North, M.P., Koch, G.W., Hungate, B.A., 2016. Restoring forest structure and process stabilizes forest carbon in wildfire-prone southwestern ponderosa pine forests. Ecol. Appl. 26, 382–391.
Hurteau, M.D., Liang, S., Westerling, A.L., Wiedinmyer, C., 2019. Vegetation-fire feedback reduces projected area burned under climate change. Sci. Rep. 9, 2838.
James, J.N., Kates, N., Kuhn, C.D., Littlefield, C.E., Miller, C.W., Bakker, J.D., Butman, D. E., Haugo, R.D., 2018. The effects of forest restoration on ecosystem carbon in western North America: a systematic review. For. Ecol. Manag. 429, 625–641.
Johnson, D.E., Mukhtar, H.A., Mapston, R., Humphrey, R., 1962. The mortality of oak-juniper woodland species following a wild fire. J. Range Manage. 15, 201–205.
Johnson, S.A., Chambers, C.L., 2017. Effects of ponderosa pine forest restroation on habitat for bats. Western North Am. Nat. 77.
Jones, G.M., Keyser, A.R., Westerling, A.L., Baldwin, W.J., Keane, J.J., Sawyer, S.C., Clare, J.D.J., Gutiérrez, R.J., Peery, M.Z., 2021. Forest restoration limits megafires and supports species conservation under climate change. Front. Ecol. Environ.
Kashian, D.M., Romme, W.H., Tinker, D.B., Turner, M.G., Ryan, M.G., 2006. Carbon storage on coniferous landscapes with stand-replacing fires. Bioscience 56, 598–606.
Kaufmann, M.R., Huisjen, D.W., Kitchen, S., Babler, M., Abella, S.R., Gardiner, T.S., McAvoy, D., Howie, J., Page, D.H., 2016. Gambel Oak Ecology and Management in the Southern Rockies: the Status of Our knowledge. SRFSN Publication 2016-1. Colorado State University, Southern Rockies Fire Sciences Network, Fort Collins, CO, p. 16.
KCFAST, 2020. Kansas City Fire Access Software. Fire Weather Information Extraction. https://famit.nwcg.gov/applications/KCFast.
Kemp, K.B., Higuera, P.E., Morgan, P., 2016. Fire legacies impact conifer regeneration across environmental gradients in the US northern Rockies. Landsc. Ecol. 31, 619–636.
Keyser, C.E., Dixon, G.E., 2008. Central Rockies (CR) Variant Overview – Forest Vegetation Simulator, in: States, U., of, D., Agriculture, Service, F., Management, F., Center, S. (Eds.), Fort Collins, CO.
Kolb, T., Agee, J., Fulé, P., McDowell, N., Pearson, K., Sala, A., Waring, R., 2007. Perpetuating old ponderosa pine. For. Ecol. Manage. 249, 141–157.
Kolb, T.E., Flathers, K., Bradford, J.B., Andrews, C., Asherin, L.A., Moser, W.K., 2020. Stand density, drought, and herbivory constrain ponderosa pine regeneration pulse. Can. J. For. Res. 50.
Kolden, C.A., 2019. We're not doing enough prescribed fire in the Western United States to mitigate wildfire risk. Fire 2, 30.
Krasnow, K., Schoennagel, T., Veblen, T.T., 2009. Forest fuel mapping and evaluation of LANDFIRE fuel maps in Boulder County, Colorado, USA. For. Ecol. Manage. 257, 1603–1612.
Křivan, V., Lewis, M., Bentz, B.J., Bewick, S., Lenhart, S.M., Liebhold, A., 2016. A dynamical model for bark beetle outbreaks. J. Theor. Biol. 407.
LANDFIRE, 2013. LANDFIRE 40 Scott and Burgan fire behavior fuel models. http://www.landfire.gov/NationalProductDescriptions2.php (30 August 2016).
Laughlin, D.C., Bakker, J.D., Fulé, P.Z., 2005. Understorey plant community structure in lower montane and subalpine forests, Grand Canyon National Park, USA. J. Biogeogr. 32, 2083–2102.
Liang, S., Hurteau, M.D., Westerling, A.L., 2018. Large-scale restoration increases carbon stability under projected climate and wildfire regimes. Front. Ecol. Environ. 16, 207–212.
Liu, Y., Piao, S., Gasser, T., Ciais, P., Yang, H., Wang, H., Keenan, T.F., Huang, M., Wan, S., Song, J., 2019. Field-experiment constraints on the enhancement of the terrestrial carbon sink by CO2 fertilization. Nat. Geosci. 12, 809–814.
Luyssaert, S., -Detlef Schulze, E., Börner, A., Knohl, A., Hessenmöller, D., Law, B.E., Ciais, P., Grace, J.B., 2008. Old-growth forests as global carbon sinks. Nature 455.
Marlon, J.R., Bartlein, P.J., Gavin, D.G., Long, C.J., Anderson, R.S., Briles, C.E., Brown, K. J., Colombaroli, D., Hallett, D.J., Power, M.J., Scharf, E.A., Walsh, M.K., 2012. Long-term perspective on wildfires in the western USA. Proc. Natl. Acad. Sci. U.S.A. 109. E535-E543.
Mayer, M., Prescott, C.E., Abaker, W.E., Augusto, L., Cécillon, L., Ferreira, G.W., James, J., Jandl, R., Katzensteiner, K., Laclau, J.-P., 2020. Influence of forest management activities on soil organic carbon stocks: a knowledge synthesis. For. Ecol. Manage. 466, 118127.
McCauley, L.A., Bradford, J.B., Robles, M.D., Shriver, R.K., Woolley, T.J., Andrews, C.A., 2022. Landscape-scale forest restoration decreases vulnerability to drought mortality under climate change in southwest USA ponderosa pine forest. For. Ecol. Manage. 509, 120088 https://doi.org/10.1016/j.foreco.2022.120088.
McCauley, L.A., Robles, M.D., Woolley, T., Marshall, R.M., Kretchun, A., Gori, D.F., 2019. Large-scale forest restoration stabilizes carbon under climate change in Southwest United States. Ecol. Appl. 29, e01979.
Mildrexler, D.J., Berner, L.T., Law, B.E., Birdsey, R.A., Moomaw, W.R., 2020. Large trees dominate carbon storage in forests east of the cascade crest in the United States Pacific Northwest. Front. For. Glob. Change 3, 594274. https://doi.org/10.3389/ffgc.2020.594274.
Miller, C., Aplet, G.H., 2016. Progress in wilderness fire science: embracing complexity. J. For. 114, 373–383.
Mitchard, E., 2018. The tropical forest carbon cycle and climate change. Nature 559.
Montwé, D., Isaac-Renton, M., Standish, A., Axelson, J., 2022. Partial cutting in a dry temperate forest ecosystem alleviates growth loss under drought. Front. For. Glob. Change.
MTBS, 2020. MTBS data access: burned areas boundaries, 1984-2018. https://www.mtbs.gov/index.php/direct-download (6 November 2020).
Naficy, C., Sala, A., Keeling, E.G., Graham, J.D., DeLuca, T.H., 2010. Interactive effects of historical logging and fire exclusion on ponderosa pine forest structure in the northern Rockies. Ecol. Appl. 20, 1851–1864.
Nicholls, D., Halbrook, J., Benedum, M., Han, H.-S., Lowell, E., Becker, D., Barbour, R., 2018. Socioeconomic constraints to biomass removal from forest lands for fire risk reduction in the Western U.S. Forests 9, 264.
North, M., Brough, A., Long, J., Collins, B., Bowden, P., Yasuda, D., Miller, J., Sugihara, N., 2015. Constraints on mechanized treatment significantly limit mechanical fuels reduction extent in the Sierra Nevada. J. For. 113, 40–48.
North, M., York, R., Collins, B., Hurteau, M., Jones, G., Knapp, E., Kobziar, L., McCann, H., Meyer, M., Stephens, S., 2021. Pyrosilviculture needed for landscape resilience of dry Western United States forests. J. For.
Odum, E.P., 1969. The strategy of ecosystem development. Science 164.
Ouzts, J., Kolb, T., Huffman, D., Meador, A.S., 2015. Post-fire ponderosa pine regeneration with and without planting in Arizona and New Mexico. For. Ecol. Manage. 354, 281–290.
Owen, S.M., Sieg, C.H., Sánchez Meador, A.J., Fulé, P.Z., Iniguez, J.M., Baggett, L.S., Fornwalt, P.J., Battaglia, M.A., 2017. Spatial patterns of ponderosa pine regeneration in high-severity burn patches. For. Ecol. Manage. 405, 134–149.
Parks, S.A., Holsinger, L.M., Panunto, M.H., Jolly, W.M., Dobrowski, S.Z., Dillon, G.K., 2018. High-severity fire: evaluating its key drivers and mapping its probability across western US forests. Environ. Res. Lett. 13, 044037.
Pausas, J.G., Keeley, J.E., 2023. Evolutionary fire ecology: an historical account and future directions. Bioscience 0.
Peeler, J.L., McCauley, L.A., Metlen, K., Woolley, T., Davis, K.T., Robles, M.D., Haugo, R., Riley, K., Higuera, P.E., Fargione, J., Addington, R.N., Bassett, S., Blankenship, K., Case, M.J., Chapman, T.B., Smith, E., Swaty, R., Welch, N., 2023. Identifying opportunity hot spots for reducing the risk of wildfire-caused carbon loss in western US conifer forests. Environ. Res. Lett., 094040
Petrie, M.D., Hubbard, R.M., Bradford, J.B., Kolb, T.E., Noel, A., Schlaepfer, D.R., Bowen, M.A., Fuller, L.R., Moser, W.K., 2023. Widespread regeneration failure in ponderosa pine forests of the Southwestern United States. For. Ecol. Manag.
Pietruszka, B.M., Young, J.D., Short, K.C., St Denis, L.A., Thompson, M., Calkin, D., 2023. Consequential lightning-caused wildfires and the "let burn" narrative. Fire Ecol. 19.
Pugh, T.A.M., Lindeskog, M., Smith, B., Poulter, B., Arneth, A., Haverd, V., Calle, L., 2019. Role of forest regrowth in global carbon sink dynamics. Proc. Natl Acad. Sci. 116.
Radeloff, V.C., Helmers, D.P., Kramer, H.A., Mockrin, M.H., Alexandre, P.M., Bar Massada, A., Butsic, V., Hawbaker, T.J., Martinuzzi, S., Syphard, A.D., Stewart, S.I., 2017. The 1990-2010 Wildland-Urban Interface of the Conterminous United States - geospatial data, 2nd Edition. Forest Service Research Data Archive, Fort Collins, CO.
Rebain, S.A., Reinhardt, E., Crookston, N.L., Beukema, S.J., Werner, A.K., Greenough, J. A., 2010. The Fire and Fuels Extension to the Forest Vegetation Simulator: Updated Model Documentation. USDA Forest Service, Forest Management Service Center, Fort Collins, CO, p. 403. Internal Report.
Reinhardt, E.D., Keane, R.E., Brown, J.K., 1997. First Order Fire Effects Model: FOFEM 4.0, User's Guide. Gen. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT. Tech. Rep. INT-GTR-344.
Reynolds, R.T., Graham, R.T., Reiser, M.H., Bassett, R.L., Kennedy, P.L., Boyce, D.A., Goodwin, G., Smith, R., Fisher, E.L., 1992. Management Recommendations for the Northern Goshawk in the Southwestern United States. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, p. 90. Gen. Tech. Rep. RMRS-GTR-217.
Reynolds, R.T., Meador, A.J.S., Youtz, J.A., Nicolet, T., Matonis, M.S., Jackson, P.L., DeLorenzo, D.G., Graves, A.D., 2013. Restoring Composition and Structure in Southwestern Frequent-Fire Forests: a Science-Based Framework for Improving

Ecosystem Resiliency. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO, p. 76. Gen. Tech. Rep. RMRS-GTR-310.
Riebeek, H., 2011. The carbon cycle.
Riley, K.L., Grenfell, I.C., Finney, M.A., 2016. Mapping forest vegetation for the western United States using modified random forests imputation of FIA forest plots. Ecosphere 7, e01472. https://doi.org/10.1002/ecs2.1472.
Rollins, M.G., 2009. LANDFIRE: a nationally consistent vegetation, wildland fire, and fuel assessment. Int. J. Wildland Fire 18, 235–249.
Savage, M., Mast, J.N., 2005. How resilient are southwestern ponderosa pine forests after crown fires? Can. J. For. Res. 35, 967–977.
Scott, J.H., Helmbrecht, D.J., Parks, S.A., Miller, C., 2012. Quantifying the threat of unsuppressed wildfires reaching the adjacent wildland-urban interface on the Bridger-Teton National Forest, Wyoming, USA. Fire Ecol. 8, 125–142.
Scott, J.H., Short, K.C., Finney, M., Gilbertson-Day, J., Vogler, K.C., 2018. FSim: the large-fire simulator — guide to Best Practices version 0.3.1.
Seidl, R., Albrich, K., Thom, D., Rammer, W., 2018. Harnessing landscape heterogeneity for managing future disturbance risks in forest ecosystems. J. Environ. Manage. 209.
Seli, R.C., Ager, A.A., Crookston, N.L., Finney, M.A., Bahro, B., Agee, J., McHugh, C.W., 2008. Incorporating landscape fuel treatment modeling into the forest vegetation simulator. In: Havis, R.N., Crookston, N.L. (Eds.), Third Forest Vegetation Simulator Conference. Proceedings RMRS-P-54. Rocky Mountain Research Station, Fort Collins, CO. USDA Forest Service, pp. 27–39.
Singleton, M.P., Thode, A.E., Sánchez Meador, A.J., Iniguez, J.M., 2021. Moisture and vegetation cover limit ponderosa pine regeneration in high-severity burn patches in the southwestern US. Fire Ecol. 17, 14. https://doi.org/10.1186/s42408-021-00095-3.
Sneed, P., Floyd-Hana, L., Hanna, D., 2002. Prescott Basin Fire History Project. Final Report, Prescott College, Prescott, AZ.
Sorensen, C.D., Finkral, A.J., Kolb, T.E., Huang, C.H., 2011. Short- and long-term effects of thinning and prescribed fire on carbon stocks in ponderosa pine stands in northern Arizona. For. Ecol. Manage. 261, 460–472.
Sousa, W.P., 1984. The role of disturbance in natural communites. Ann. Rev. Ecol. Syst. 15.
Steele, P.H., 1984. Factors Determining Lumber Recovery in Sawmilling. USDA, Forest Service, Forest Products Laboratory. Gen. Tech. Rep. FPL-39.
Stephens, S.L., Moghaddas, J.J., Edminster, C., Fiedler, C.E., Haase, S., Harrington, M., Keeley, J.E., Knapp, E.E., McIver, J.D., Metlen, K., Skinner, C.N., Youngblood, A., 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecol. Appl. 19, 305–320.
Stoddard, M.T., Roccaforte, J.P., Meador, A.J.S., Huffman, D.W., Fulé, P.Z., Waltz, A.E., Covington, W.W., 2021. Ecological restoration guided by historical reference conditions can increase resilience to climate change of southwestern US ponderosa pine forests. For. Ecol. Manage. 493, 119256 https://doi.org/10.1016/j.foreco.2021.119256.
Strom, B.A., Fulé, P.Z., 2007. Pre-wildfire fuel treatments affect long-term ponderosa pine forest dynamics. Int. J. Wildland Fire 16, 128–138.
Swain, D.L., Abatzoglou, J., Kolden, C.A., Shive, K.L., Kalashnikova, D.A., Singh, D., Smith, E., 2023. Climate change is narrowing and shifting prescribed fire windows in western United States. Commun. Earth Environ.
Swetnam, T.W., Baisan, C.H., 1996. Historical fire regime patterns in the southwestern United States since AD 1700. in: Allen, C.D. (Ed.). In: Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, pp. 11–32. Gen. Tech. Rep. RM-GTR-286.
Taylor, M.H., Sanchez Meador, A.J., Kim, Y.S., Rollins, K., Will, H., 2015. The Economics of Ecological Restoration and Hazardous Fuel Reduction Treatments in the Ponderosa Pine Forest Ecosystem. Forest Science.
USDA, 2014. land and resource management plan for the Kaibab national forest Coconino, Yavapai, and Mojave Counties, Arizona, in: United States Department of Agriculture, F.S., Southwestern Region, MB-R3-07-17 (Ed.), p. 210.
USDA, 2015. Record of decision for the four-forest restoration initiative: Coconino and Kaibab National Forests Coconino County, Arizona. Forest Service Southwestern Region. M.-R.–A. (Ed.).
USDA, 2018. Land and resource management plan for the coconino national forest Coconino, Gila, and Yavapai Counties, Arizona. United States Department of Agriculture, F.S., Southwestern Region, p. 303. MB-R3-04-31 (Ed.).
USDA Forest Service, 2015. Record of decision for the four-forest restoration initiative: Coconino and Kaibab National forests, Coconino County, Arizona. USDA Forest Service, Southwestern Region, Washington, D.C.
USDA Forest Service, 2020a. FIA DataMart, Forest inventory and analysis database. https://apps.fs.usda.gov/fia/datamart/datamart.html.
USDA Forest Service, 2020b. Forest service activity tracking system (FACTS). https://www.fs.usda.gov/managing-land/natural-resource-manager#facts.
USDA Forest Service, Southwestern Region, 1997. Plant associations of Arizona and New Mexico, Volume 1: forests, Volume 2: woodlands. Habitat Typing Guides Edition 3, USDA Forest Service, Southwestern Region, Albuquerque, NM.
USFWS, 2012. Mexican Spotted Owl Recovery plan, First Revision (*Strix Occidentalis Lucida*). US Fish and Wildlife Service, Southwest Region, Albuquerque, New Mexico.
Walker, R.B., Coop, J.D., Parks, S.A., Trader, L., 2018. Fire regimes approaching historic norms reduce wildfire-facilitated conversion from forest to non-forest. Ecosphere 9, e02182.
Williamson, G.J., Bowman, D.M.J.S., Price, O.F., Henderson, S.B., Johnston, F.H., 2016. A transdisciplinary approach to understanding the health effects of wildfire and prescribed fire smoke regimes. Environ. Res. Lett. 11.
Wilson, R.A., 1980. Reformulation for Forest Fire Spread Equations in SI units. Res. Note INT-292. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, 5 p.
Winthers, E., Fallon, D., Haglund, J., DeMeo, T., Nowacki, G., Tart, D., Ferwerda, M., Robertson, G., Gallegos, A., Rorick, A., Cleland, D.T., Robbie, W., 2005. Terrestrial ecological unit inventory technical guide: landscape and land unit scales. Gen. Tech. Rep. WO-68, USDA Forest Service, Washington, D.C.
Young, D.J., Werner, C.M., Welch, K.R., Young, T.P., Safford, H.D., Latimer, A.M., 2019a. Post-fire forest regeneration shows limited climate tracking and potential for drought-induced type conversion. Ecology 100, e02571. https://doi.org/10.1002/ecy.2571.
Young, J.D., Ager, A.A., Thode, A.E., 2022. Using wildfire as a management strategy to restore resiliency to ponderosa pine forests in the southwestern United States. Ecosphere 13, e4040. https://doi.org/10.1002/ecs2.4040.
Young, J.D., Evans, A.M., Iniguez, J.M., Thode, A., Meyer, M.D., Hedwall, S.J., McCaffrey, S., Shin, P., Huang, C.-H., 2020. Effects of policy change on wildland fire management strategies: evidence for a paradigm shift in the western US? Int. J. Wildland Fire 29.
Young, J.D., Thode, A.E., Huang, C.-H., Ager, A.A., Fule, P.Z., 2019b. Strategic application of wildland fire suppression in the southwestern United States. J. Environ. Manage. 245, 504–518.
Young, Jesse D., Ager, Alan A., 2023. Landscape Simulator (LSim) replicate data resulting from leveraging wildfire as a management strategy to restore old growth forest structure while stabilizing carbon stocks in the southwest United States. Fort Collins, CO: Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2023-0065.