Wildfire Fuel Treatments and Long-Term Carbon Storage in California Mixed Conifer Forests

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ABSTRACT

Forest carbon storage has been identified as a major component of global climate change mitigation, but fire suppression and climate change has resulted in forests that are extremely vulnerable to catastrophic wildfire events in California. Fuel reduction treatments, like mechanical thinning and prescribed fire, require initial carbon removal, but can effectively mitigate wildfire behavior, resulting in reduced wildfire carbon loss. At the Blodgett Forest Research Station in the central Sierra Nevada, the Fire and Fire Surrogate study has been ongoing since 2001, in which identical units of mixed conifer forests have been randomly assigned one of four treatments: control, mechanical thin only, prescribed burn only, or mechanical thin and prescribed burn. I analyzed observed aboveground carbon stocks from 2001-2020, modeled wildfire mitigation effects of treatment on stable carbon stocks, and net carbon stocks using increasing estimates of annual wildfire probability. According to my results, net carbon stocks of the mechanical thin and prescribed burn treatment should exceed those of the control treatment when annual burn probability reaches 0.055, and all active treatments should exceed the control when annual burn probability reaches 0.075. My results indicate that as wildfire risk increases, the reduction in modeled wildfire emissions from fuel treatments outweighs the upfront carbon costs of treatment. Incentivizing short-term carbon storage in CA forests in the current carbon offset program framework is unsustainable. Forest carbon offset protocols should consider long-term net carbon storage of forests and the net effects of wildfire fuel treatments in offset credit issuance.

KEYWORDS

Forest carbon offsets, net forest carbon storage, prescribed fire, mechanical thinning, annual burn probability

INTRODUCTION

Forest carbon offsets

To mitigate the magnitude of anthropogenic climate change, atmospheric carbon dioxide concentration must be minimized by reducing and/or offsetting global carbon emissions (IPCC 2023). Forests play a major role in the global carbon cycle, as forest vegetation fixes massive amounts of atmospheric CO2 via photosynthesis, which can then be stored for hundreds of years in forest biomass. The potential for forest carbon storage has been identified as a major component of global climate change mitigation. The California mixed conifer forest (MCF) covers approximately 32,000 km^2 and stores over 960 TgC (Christensen et al. 2021), which is equivalent to over 2.5 times California's total greenhouse gas emissions in 2020 (CARB 2022). Forest carbon offsets have been introduced in California as a part of a larger cap and trade program, providing an economic incentive for landowners to increase long-term carbon storage in forest ecosystems. Functionally, forest carbon offsets act as permits for carbon emission, and can be traded within the offset credit market. The California Air Resources Board's Improved Forest Management program has generated 78.8% of carbon offset credits that are circulating in California's compliance market (California Air Resources Board Offset Credit Issuance Table, 2022). The purpose of the cap and trade program is to reduce California's carbon emissions, making the integrity of forest carbon offsets crucial to the real climate change mitigation effects of the program. However, forest carbon offset credits incentivize short-term biomass accumulation, which directly conflicts with the wildfire mitigation objective of generally reducing the density of California's forested ecosystems.

Increasing wildfire risk

Fire suppression has severely altered the disturbance regimes of California's MCF (Stephens 1997), and climate change has increased the probability of wildfire (Westerling et al. 2006), resulting in forests that are extremely vulnerable to catastrophic wildfire events. Over a century of fire suppression in California has increased surface and canopy fuel accumulation and, thus, probability of large and severe wildfires relative to the disturbance regimes (the characteristic type, frequency, and severity of disturbance in a given forested landscape) of pre-settlement forests MCF (Stephens, 1997). Not only have altered forest structures increased vulnerability to wildfire, but also the climatic conditions conducive to large, high-severity wildfires have extended spatially and temporally as a result of anthropogenic climate change (Westerling 2016). California's MCF have experienced longer and more frequent droughts and longer fire seasons in the past decades, contributing to an increase in frequency of large (>9400 ha) wildfires (Westerling et al. 2006). The carbon stored in California's MCF is at significant risk of reversal from wildfire by releasing the carbon back into the atmosphere via combustion and the decay of killed vegetation (Campbell et al. 2007, Earles et al. 2014). The potential for large, high-severity wildfires to cause type conversions to other ecosystem types (namely forest to shrubland) poses a considerable risk to the long-term stability of carbon storage in California MCF (Liang et al. 2017, Stephens et al. 2020). The increasing extent and severity of wildfires and their associated carbon emissions motivate wildfire mitigation strategies in California's MCF ecosystems.

Fuel reduction treatments

Fuel reduction treatments can effectively mitigate wildfire behavior, resulting in reduced wildfire carbon emissions. Treatments such as mechanical thinning and prescribed burning emulate the natural disturbance regime that regulated California's MCF before colonization (Stephens et al. 1997). These treatments directly reduce hazardous fuel loads, while catalyzing restoration of forest structure and species composition to be more resistant to wildfire (Stephens et al. 2012). Fuel reduction treatments have been observed to significantly reduce modeled and observed wildfire emissions (North and Hurteau 2011, Foster et al. 2020). Repeated fuel treatments also have the potential to stabilize California's MCF disturbance regimes, which could make the forest's carbon storage more stable in the long-term (Loudermilk et al. 2013). Yet, the initial carbon loss from the direct biomass removal of fuel treatments is disincentivized by the forest carbon offset market. The net effect of fuel treatments on California's MCF carbon storage requires the comparison of carbon losses from treatment to reduced wildfire emissions weighted by the probability of wildfire. The scientific assessment of the potential for fuel treatments to increase the long-term net carbon storage of forests has not come to consensus.

Daniel Foster et al. conducted a comprehensive study of long-term net carbon storage of CA MCF treated with common fuel reduction treatments (Foster et al. 2020), and this study simplifies their methods, includes more recent data, and projects findings into the future.

METHODS

Study site

Blodgett Forest Research Station (BFRS) is located near Georgetown, CA, USA (38°54045″ N; 120°39027″ W), between 1100 and 1410 meters of elevation. The soils are primarily sandy loam, composed mainly of Ultic Haploxeralfs (Alfisols), which are well developed, well draining, and highly productive. Slopes are generally less than 30%. The local climate is Mediterranean, experiencing long warm-dry seasons and cool-wet winters. Precipitation is experienced mainly in winter and spring, at 160 cm/year on average, and temperatures range from 0-8° C in winter, and 10-29° C in summer. The species composition is generic mixed conifer forest (MCF), including (but not limited to) Abies concolor, Calocedrus decurrens, Pinus lambertiana, Pinus ponderosa, Pseudotsuga menziesii, and Quercus kelloggii. The historical (pre-colonial) disturbance regime of the area was one of frequent low-to-moderate severity wildfires with mean fire return intervals of 8-25 years. The recent disturbance regime includes intensive logging in the early 1900s, and decades of effective fire suppression. The forest structure is consistent with California MCF with this altered disturbance regime: moderate to high canopy cover, heavy fuel loads, high density with more small trees and fewer large trees, and species composition shifting away from fire-resistant species (pines) toward less fire-resistant species (firs).

Treatments

The FFS study at BFRS includes 12 similar experimental units (compartments), which were randomly assigned one of four treatments (3 compartments per treatment): control, mechanical only, burn only, or mechanical + burn. The similarity of compartments was confirmed by pre-treatment measurements, and initial treatments were installed in late 2001 and

2002 (Stephen & Moghaddas 2005). The objective of the treatment design was to reduce fire severity using management practices common in northern Sierra Nevada forests (Agee and Skinner 2005, Schwilk et al. 2009).

Control compartments (40, 240, 590) were left unmanaged for the duration of the study period. Mechanical only compartments (190, 350, 490) were commercially harvested using crown thinning (removing larger merchantable trees) followed by a thinning from below (thinning trees below a threshold diameter) between summer 2001 and summer 2003. The goal of these thinning treatments in the mechanical only compartments was to achieve a target mean basal area of 28-32 m^2/hectare, which is slightly more than half of pre-treatment mean basal area of all compartments (53 m^2/hectare). Following the harvest, small trees with diameter at breast height (DBH, breast height = 1.37m) less than 25 cm were masticated. Mastication occurred following the initial harvest between 2001 and 2003, and new small trees with DBH < 25 cm were masticated again between spring 2017 and spring 2018. Residual activity fuels (tree foliage, limbs, and tops) from the initial harvest, masticated material, and residual small trees were distributed throughout the compartments in ~0.04-hectare clumps.

Burn only compartments (60, 340, 400) were treated with prescribed fires in October/November of 2002, 2009, and 2017 with prescription objectives of reducing surface and ladder fuel loads, while limiting mortality to ≤10% of trees larger than 46 cm DBH. Fire weather conditions were similar for both burns, and are detailed in Kobziar et al. 2006. Fire behavior in all 6 burns (all 3 compartments, both years) consisted of lower than 2 meter flame lengths and occasional torching of live trees.

The mechanical + burn compartments (180, 380, 570) were treated using a combination of the two treatments detailed above. They received the same mechanical treatments as the mechanical only compartments, then the residual fuels were broadcast burned using the same prescription as the burn only compartments, except they were not reburned in 2009 (they were burned only twice in 2002 and 2019). The surface fuels in the mechanical + burn compartments were mainly masticated chips and residual materials from the mechanical treatment, which had cured for a season before being burned, resulting in a longer burn than in the burn only compartments.

Field measurements

Field measurements of all treatment compartments were taken by field crews in permanent 0.04 hectare plots in the summers of 2001, 2003, 2009, 2016, and 2020 as pre-treatment, 1-, 7-, 14-, and 18-year post-treatment measurements, respectively. Twenty circular plots were established within a 10 hectare core area of each compartment (to reduce edge effects), in a regular 60 meter grid formation. Tree species, DBH, height to crown base, and total height were measured and recorded for all trees ≥11.4 cm DBH within each plot. DBH and total height were measured and recorded for all standing dead trees (snags) ≥20.5 cm DBH. Data on snag limb condition, wood hardness, bark coverage, and estimated years since death were recorded in 2016 to determine a live:dead carbon ratio for snags.

Fuels data were measured and collected using Brown's line-intercept method (Brown 1974) on two 11.43 meter transects per 0.04 hectare plot. For each transect, litter and duff depths were measured at two fixed locations. Additionally, 1 hour (0-0.64 cm), 10 hour (0.64-2.54 cm), and 100 hour (2.54-7.62 cm) woody fuel particles that intersected with the transect were tallied along sub-transects of consistent lengths. Diameter and decay class of all 1000 hour fuels (≥7.62 cm) that intersected with either transect were measured and recorded.

Ocular estimates of percent cover of understory vegetation was recorded for each plot by species, and binned into classes of <5%, 5–25%, and 25–100%, which were interpreted as 2.5%, 15%, and 63%, respectively. The species included were the following understory species common in BFRS: Arctostaphylos spp., Ceanothus spp., Chamaebatia foliosa, Chrysolepis spp., Notholithocarpus densiflorus, Ribes roezlii, Rosa gymnocarpa, and Symphorocarpus mollis.

Analysis

Observed carbon stocks

To calculate aboveground live tree biomass, BFRS researchers used regional biomass equations used by the Forest Inventory and Analysis (FIA) program. These equations use tree measurements (species, DBH, and height) to estimate each tree's cubic volume, and use species-specific wood density to estimate biomass of the entire tree stem. Aboveground live tree biomass is the sum of stem, branch, and bark biomass, and separate allometric equations were used to calculate branch and bark biomass.

To calculate snag biomass, the same equations (stem only) were used and adjusted using a live:dead biomass ratio of 0.88, based on the findings of Cousins et al. (2015). Plot-level biomass, in units of Megagrams per hectare (Mg/ha), was estimated as a sum of individual tree/snag biomass scaled by plot size. Conversion of biomass estimates to carbon content was accomplished by assuming a ratio of 0.48 MgC per Mg biomass for live trees, and 0.5145 MgC per Mg biomass for snags, consistent with established literature (IPCC 2003, Cousins et al. 2015, Dore et al. 2016).

Understory carbon stocks were determined using observed percent-cover data and biomass equations from McGinnis et al. (2010). The number of average-sized individuals per species populating each plot was estimated from field data and multiplied by their species-specific per-individual biomass to estimate total observed biomass per species. These estimates were summed and scaled by plot size to determine understory biomass per hectare, then converted to MgC per hectare (MgC/ha) using a carbon density ratio of 0.49 (Chojnacky and Milton 2008).

Table 1. Abbreviations used in the study.

Abbreviation	Term	Meaning
AFC	Aboveground forest carbon	Carbon in aboveground live and dead biomass (trees, understory, and fuels)
LTC	Live tree carbon	Carbon in aboveground stem, bark, and branches of live trees
SLTC	Stable live tree carbon	Carbon in trees predicted to survive a wildfire
ELTC	Expected live tree carbon	Sum of LTC and SLTC, weighted by burn probability

Fuel loads were estimated from transect data using Sierra nevada-specific equations and species-specific coefficients (Van Wagtendonk et al 1998). For each plot, the species-specific coefficient used was determined as an average of coefficients of all species measured in the plot, weighted by species' basal area as a proportion of total plot basal area (Stephens 2001).

Plot-level fuel load estimates were calculated as an average of both transect-level estimates of total fuel load, and were converted to MgC/ha using a 50% carbon concentration for coarse (1000-hour fuels) and fine woody fuels (1-100 hour fuels) and 37% for litter and duff (IPCC 2003, Stephens et al. 2012). I calculated aboveground forest carbon (AFC - Table 1) as the sum of live tree carbon (LTC - Table 1), understory carbon stocks, and fuel carbon stocks.

Wildfire modeling and carbon stability

To assess the treatments' effect on carbon fluxes associated with wildfire, I modeled potential wildfire behavior for each plot in 2003 (following initial treatment) and 2020 (following repeated treatments) with the Fire and Fuels Extension to the Forest Vegetation Simulator (FVS-FFE) (Reinhardt et al. 2003). Fire behavior and crown fire potential are modeled by FVS-FFE using established equations, and user-input tree data. Some research shows that overriding FVS-FFE's fuel model selection can result in more appropriate model outputs (Foster et al. 2020), but I chose to use the default fuel models assigned by FVS-FFE for each plot.

FVS-FFE automatically models wildfire under both moderate and severe weather conditions, but current and projected climatic conditions make severe fire weather conditions more probable in California MCF (Collins et al. 2014, Starrs et. al 2018), so I omitted results from the moderate wildfire. My analysis focused on the predicted mortality output (PMORT) from FVS-FFE, representing the percentage of plot basal area predicted to die within the first three years following the modeled wildfire. This metric incorporates both immediate and delayed mortality factors, using crown length, diameter, tree species, and predicted scorch height. FVS-FFE's estimates of mortality rely on empirical relationships (Reinhardt and Ryan 1988) adjusted by coefficients specific to tree species of the Western Sierras. I defined stable live tree carbon (SLTC - Table 1) as the total LTC (Table 1) expected to remain at least 3 years following a wildfire (note that this does not include the carbon stored in fire-killed snags, making it an underestimation of the total stable carbon stored in standing trees following a wildfire). I calculated SLTC in 2003 and 2020 as

(Equation 1) $SLTC = LTC \times (1 - PMORT)$

Expected carbon stocks

To assess the expected carbon stocks considering direct treatment effects, wildfire-contingent treatment effects, and the probability of treated stands experiencing wildfire, I applied the concept of expected utility to risk-adjust the compartments' carbon stocks (Schoemaker 1982, Finney 2005, Ager et al. 2010). The expected carbon stock is a weighted average, combining both outcomes (observed carbon stocks and predicted post-wildfire stocks), weighted by the respective probabilities of their occurrence. For example, I calculated the Expected Live Tree Carbon (ELTC) for compartment n as

$$\text{(Equation 2) } ELTC_n = [LTC_n \text{ x } (1 - P_{burned)})] + (SLTC_n \text{ x } P_{burned})$$

where P_{burned} represents the cumulative probability that the compartment will have been burned by the given year. I calculated P_{burned} as

$$\text{(Equation 3) } P_{burned} = 1 - (1 - P_{annual})^t$$

where t is the number of years since the most recent data (t = 1 in 2020, t = 11 in 2030), and P_{annual} is the annual burn probability. To account for uncertainty in P_{annual} as climate change and altered disturbance regimes increases the frequency of wildfires in California's Sierra Nevada, I used a continuous range of values for P_{annual}, from 0.01 to 0.1. The P_{annual} value of 0.01 is accepted and used in established literature as the current annual probability of wildfire for a given area of forest in the Sierra Nevada (Foster et al. 2020). I chose to use increasing values of P_{annual} values to assess the magnitude of change in ELTC given increases in wildfire probability, representing changes to long-term stable carbon storage in fuel-treated forests as wildfire frequency is predicted to increase (Stephens 1997, Westerling et al. 2006, Westerling 2016). I used LTC and SLTC values from 2020 to estimate ELTC for all treatments across the given range of P_{annual}. I estimated ELTC in 2030 (t = 11) to assess ELTC trends during the treatment's effective lifetime. In the Fire and Fire Surrogate study, mechanical only and mechanical + burn treatments were repeated after 15 years, and the burn only treatments were conducted every 8 years, so I used an approximate average treatment interval of 10 years to analyze ELTC over the treatments' estimated lifetime.

RESULTS

Observed carbon stocks

I found that the total AFC increased across the entirety of the 18 year study period within the control compartments (Figure 1), with a total increase of 90.38 MgC/ha from 2001 to 2020 (Table A1). The control compartments' AFC measurements had the highest average standard error of 13.45 MgC/ha (Table A1).

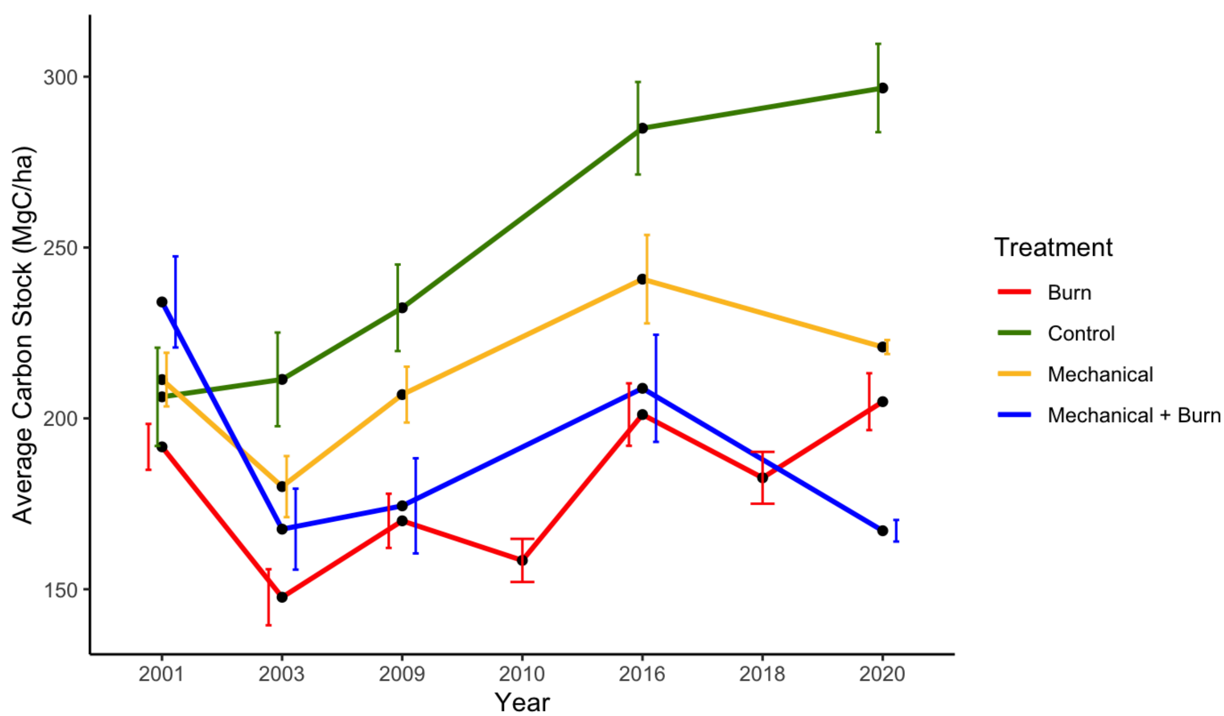

Figure 1. Total AFC over study period for all treatments. Mechanical Only and Mechanical + Burn treatments occurred only in 2001 and 2017, while Burn Only treatments occurred in 2001, 2009, and 2017. Error bars indicate the standard error of the mean AFC of all plots of all compartments of the given treatment.

I found that the mechanical only and mechanical + burn compartments' total AFC dropped following the initial treatment in 2002 (Figure 1), with decreases of -31.32 MgC/ha and -66.49 MgC/ha, respectively (Table A1). They then increased between measurements in 2003 and 2016 by a total of +60.71 MgC/ha and +41.22 MgC/ha, respectively (Table A1). The total AFC of the mechanical only and mechanical + burn compartments then dropped again following the second mastication treatment in 2017/2018 (Figure 1), with total decreases of -19.86 MgC/ha and -41.68 MgC/ha, respectively, between 2016 and 2020 (Table A1). The mechanical only and mechanical + burn compartments' AFC measurements had the third and second highest average standard errors of 7.99 MgC/ha and 11.59 MgC/ha, respectively (Table A1).

I found that the burn only compartments' total AFC decreased slightly following each burn in 2002, 2009, and 2017 (Figure 1), with decreases of -44.01 MgC/ha between 2001 and 2003, -11.59 MgC/ha 2009 and 2010, and -18.51 MgC/ha between 2016 and 2018 (Table A1). Following the third prescription burn in 2017, AFC of the burn only compartments increased by 22.28 MgC/ha from 2018 to 2020 (Table A1). The burn only compartments' AFC measurements had the lowest average standard error of 7.75 MgC/ha (Table A1).

Wildfire modeling and carbon stability

I found that the average percent mortality, or PMORT, values for the high severity wildfire modeled in 2003 was highest in the control compartments (72.35%), followed by the mechanical only compartments (47.97%), then the burn only compartments (42.02%), with the mechanical + burn compartments producing the lowest average PMORT in 2003 (24.08%) (Figure 2, Table B1). Thus, the calculated average SLTC in 2003 was lowest for the control compartments (46.27 MgC/ha), followed by the mechanical only compartments (73.32 MgC/ha), then the burn only compartments (80.25 MgC/ha), with the mechanical + burn compartment producing the highest average SLTC in 2003 (108.65 MgC/ha) (Figure 2, Table B2).

The average PMORT value for the control compartments increased by +20.28% from 2003 to 92.63% in 2020, producing an average SLTC of 17.50 MgC/ha in 2020 (Figure 2, Table B1, Table B2). This reflects that the average SLTC of the control compartments decreased by a total of -28.77 MgC/ha between 2003 and 2020, which is the largest decrease in SLTC from 2003 to 2020 out of all the treatments.

For the mechanical only compartments, I found that the average PMORT value increased by +11.47% from 2003 to 59.44% in 2020, producing an average SLTC of 69.55 MgC/ha in 2020 (Figure 2, Table B1, Table B2). Average SLTC for the mechanical only compartments decreased by a total of -3.77 MgC/ha between 2003 and 2020, which is a decrease that is over 7 times smaller in magnitude than the control compartments.

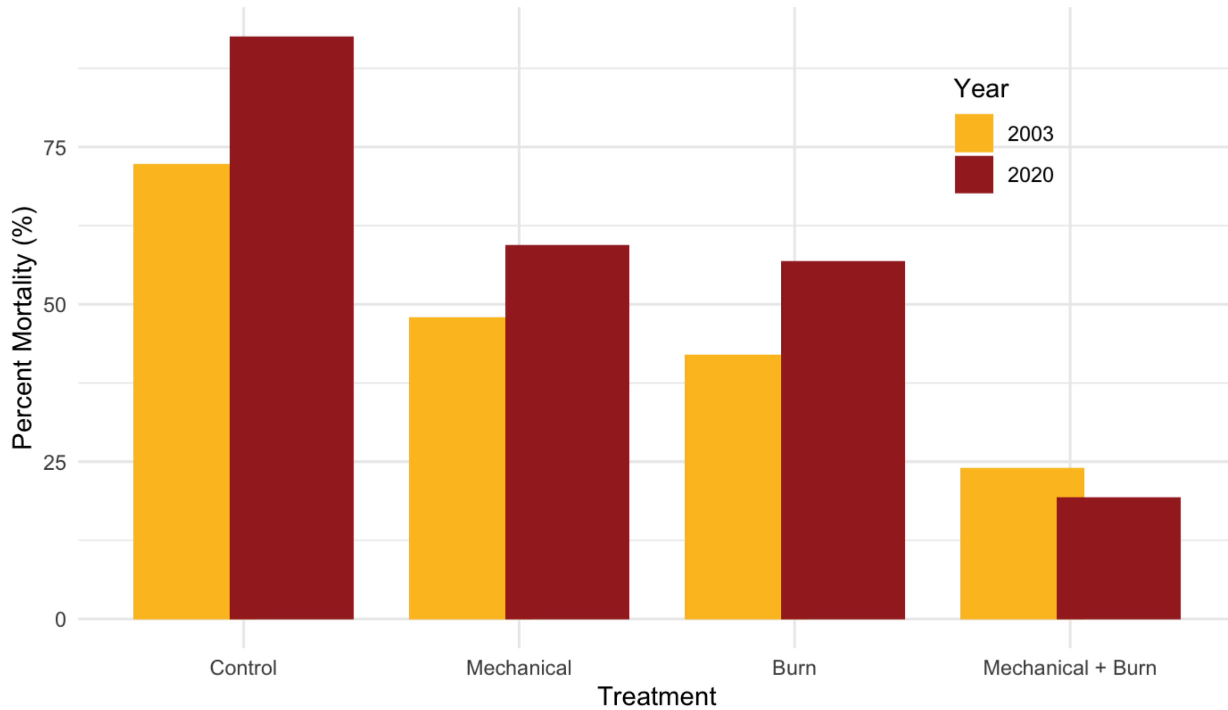

Figure 2. Modeled percent mortality (PMORT) under severe wildfire weather conditions after initial (2003) and repeated treatments (2020). PMORT is an output from FVS that represents the percentage of plot basal area predicted to die within the first three years following the modeled wildfire. This metric incorporates both immediate and delayed mortality factors, using crown length, diameter, tree species, and predicted scorch height.

I found that the average PMORT value for the burn only compartments increased by +14.79% from 2003 to 56.81% in 2020, producing an average SLTC of 76.93 MgC/ha in 2020 (Figure 2, Table B1, Table B2). Between 2003 and 2020, average SLTC of the burn only compartments decreased by a total of -3.32 MgC/ha, which is a decrease that is over 8 times smaller in magnitude than the control compartments.

For the mechanical + burn compartments, I found that the average PMORT value decreased by -4.70% from 2003 to 19.38% in 2020, making it the only treatment in which average PMORT decreased between 2003 and 2020 (Figure 2, Table B1). Between 2003 and 2020, the average SLTC for the mechanical + burn compartments increased by a total of +10.20 MgC/ha to 118.85 MgC/ha in 2020, making it the only treatment in which average SLTC increased between 2003 and 2020, and producing the highest average SLTC value of all treatments in either 2003 or 2020.

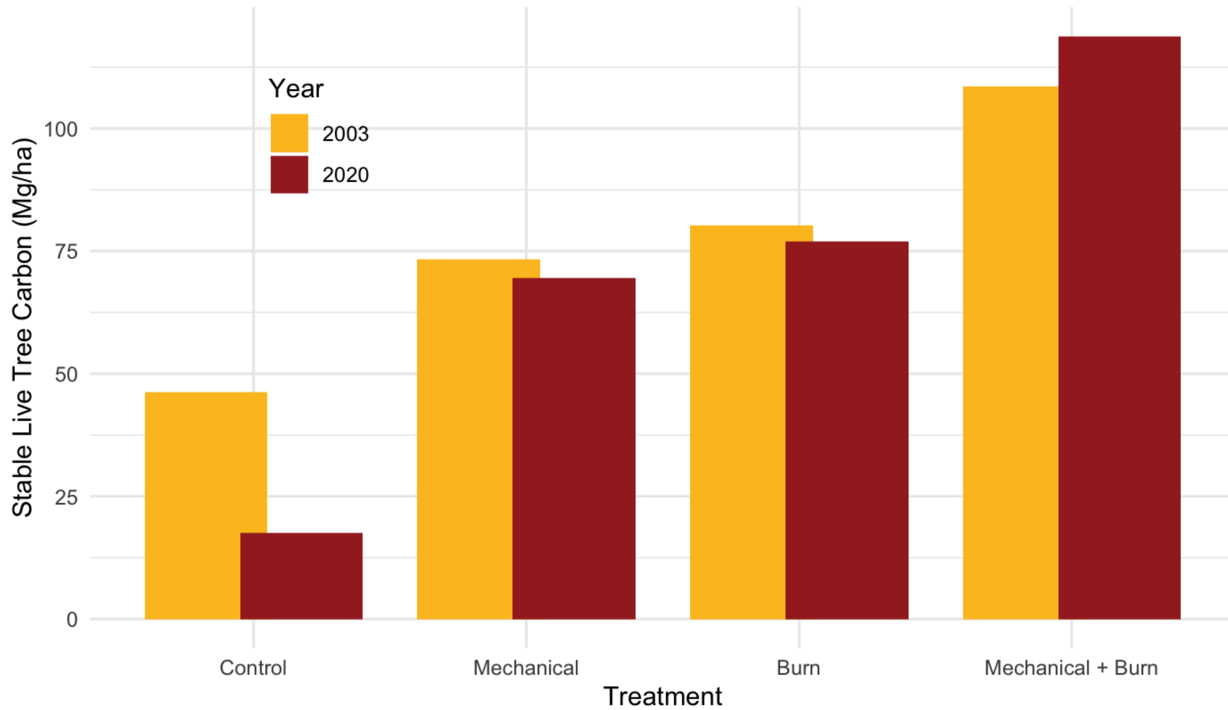

Figure 3. Stable live tree carbon (SLTC – MgC/ha) stocks after initial (2003) and repeated treatments (2020). SLTC is calculated as LTC x (1 - PMORT), and represents the total live tree carbon expected to remain at least 3 years following a wildfire. Note that this does not include the carbon stored in fire-killed snags, making it an underestimation of the total stable carbon stored in standing trees following a wildfire.

Expected carbon stocks

Given the mathematical relationship between P_{annual} and ELTC (Equations 2 & 3), higher P_{annual} values directly correlated with lower calculated values of average ELTC (Figure 3). The average calculated ELTC for the control compartments decreased by a total of -127.98 MgC/ha between P_{annual} values of 0.01 and 0.01, corresponding to an average rate of change of -14.22 MgC/ha per 1% (0.01) increase in P_{annual} (Table C1).

For mechanical only compartments, I calculated average ELTC in 2030 as 160.79 MgC/ha given the current P_{annual} value of 0.01 (Table C1). The calculated ELTC of the mechanical only compartments in 2030 exceeds that of the control compartments at P_{annual} values of ~0.07 and above (Figure 3). Of the active treatments, the mechanical only compartments' calculated ELTC decreased the fastest with increasing P_{annual} values, with a corresponding average rate of change of -6.58 MgC/ha per 1% (0.01) increase in P_{annual} (Table C1).

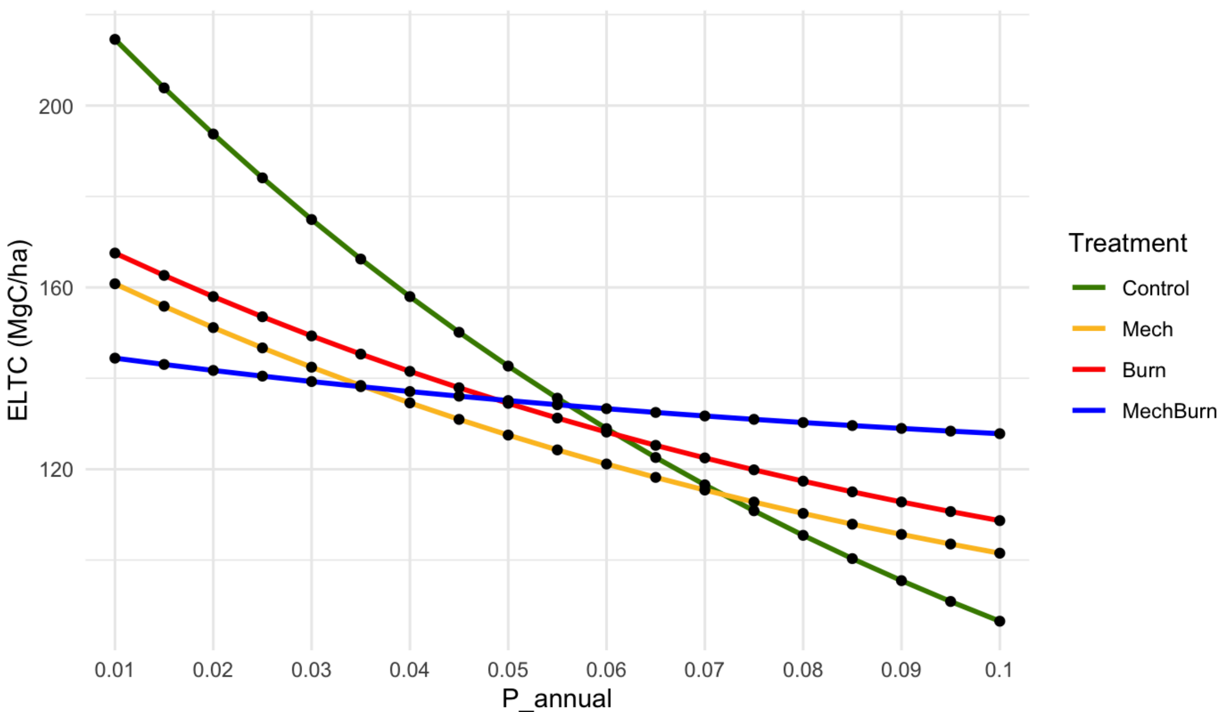

Figure 3. Total average ELTC over in 2030 for all treatments. The current value of P_{annual} is estimated as 0.01.

I calculated the average ELTC of burn only compartments in 2030 as 167.54 MgC/ha given the current P_{annual} value of 0.01 (Table C1). For the burn only compartments, the calculated ELTC in 2030 exceeds that of the control compartments at P_{annual} values of ~0.06 and above (Figure 3). Of the active treatments, the burn only compartments' calculated ELTC decreased the second fastest with increasing P_{annual} values, with a corresponding average rate of change of -6.54 MgC/ha per 1% (0.01) increase in P_{annual} (Table C1).

For the mechanical + burn compartments, I calculated average ELTC in 2030 as 144.43 MgC/ha given the current P_{annual} value of 0.01 (Table C1). The calculated ELTC of the mechanical + burn compartments in 2030 exceeds that of the control compartments at P_{annual} values of ~0.055 and above (Figure 3). Of the active treatments, the mechanical + burn compartments' calculated ELTC decreased the slowest with increasing P_{annual} values, with a corresponding average rate of change of -1.85 MgC/ha per 1% (0.01) increase in P_{annual}, which is over 3 times slower than any of the other treatments (Table C1).

DISCUSSION

The stability of LTC stocks is crucial to carbon management goals, considering that LTC stocks are the largest pool of AFC stocks in most forests (North et al. 2009, Fahey et al. 2010). The direct (carbon removal) and indirect (wildfire mitigation) effects of wildfire fuel treatments on LTC determine the stability of a forest's total LTC stocks, depending on the probability that the forest will burn in a wildfire during the treatment's effective lifetime. Diversity in fuel treatment types results in diversity in LTC stability, and, according to my results, treatments involving both mechanical thinning and prescribed burning results in the most stable LTC stocks (highest predicted values of SLTC). When weighted by the probability of wildfire, the total LTC predicted to result within the treatments' effective lifetimes (ELTC in 2030) decreases with annual wildfire probability, but active treatments result in higher ELTC values relative to the control treatment at higher annual wildfire probabilities. Of the active treatments, the mechanical + burn treatment compartments' average ELTC in 2030 exceeded that of the control treatment at the lowest P_{annual} (~0.055). This reflects the significance of wildfire fuel treatments on forest carbon sink stability as altered forest structures and anthropogenic climate change intensify the probability of large, high severity wildfire in California MCF (Campbell et al. 2007, Earles et al. 2014, Stephens 1997, Westerling 2016).

Observed carbon stocks

Average total AFC stocks in compartments of each of the treatment types in 2020 imply that repeated mechanical + burn treatments should result in the lowest AFC stocks, followed by burn only treatments, then mechanical only treatments should result in the highest AFC stocks, despite differences when comparing single treatment entries of each type. I found that all active treatments reduced total AFC stocks below that of the control treatment for the entire duration of the study. This reflects the upfront carbon cost inherent to wildfire fuel treatments, which is their main disincentive within forest carbon offset programs. It is important to recognize that my methods for estimating total AFC stocks of each treatment do not include carbon stored in harvested materials from the commercial thins, making my estimates of AFC significant underestimates of total carbon storage following the mechanical only and mechanical + burn treatments (Foster et al. 2020). My methods also do not include the direct carbon emissions from treatment operations (equipment, transportation, etc.), making my values of AFC slight overestimates of true carbon costs of the treatments (Foster et al. 2020).

The repetition of treatments within this study provides unique implications regarding the carbon effects of repeated treatments, namely the changing relationship between AFC stocks of each active treatment type as treatments were repeated. After the first entry for each treatment in 2001/2002, total AFC was reduced to similar levels among treatment types (ranging from 147.66 to 211.41 MgC/ha, which is the smallest such range of any year) (Figure 1, Table A1). I found that 7 years after the first entry (in 2009), total AFC was highest in the mechanical only compartments, followed by the mechanical + burn compartments, with the burn only compartments having the lowest total AFC (Figure 1, Table A1). Then, in 2017, following two burn only treatments and still only one mechanical only/mechanical + burn treatment, the relationship remained the same (mechanical only > mechanical + burn > burn only) (Figure 1, Table A1). Finally, in 2020, following three burn only treatments and two mechanical only/mechanical + burn treatments, I found that total AFC was still highest in the mechanical only compartments, but the burn only compartments now had the second highest total AFC, and the mechanical + burn compartments had the lowest total AFC (Figure 1, Table A1). This implies that the second entry for the mechanical + burn compartments was a turning point at which burn only compartments retained more AFC, even after 3 repeated treatments. Looking at the trajectory of the treatment types' total AFC stocks (Figure 1, Table A1), the stability of the trend in total AFC of the burn only compartments suggest that further repetitions could result in the burn only compartments having the highest total AFC stocks, but further treatments and data would be necessary to make that assertion. The shift in relative AFC among treatments as they were repeated in the study demonstrates that assessing differences between repeated treatments is more influential for carbon management implications than comparing single treatments.

Wildfire modeling and carbon stability

Differences in percent mortality from modeled wildfires, and thus SLTC, for each treatment type suggest differences in the carbon benefits of the wildfire mitigation effects of each treatment type, with relative estimated percent mortality values making mechanical + burn treatments have the highest SLTC, followed by burn only treatments, and mechanical only treatments having the lowest total SLTC. It is important to note that my methods for estimating SLTC do not include carbon stored in fire-killed snags, which could be significant during the period before they fall and decompose. It is also important to recognize that the SLTC metric is only significant when assuming that (1) a wildfire will burn the stand, with certainty, and (2) a wildfire will burn the stand under the specific conditions used to estimate AFC, LTC, and SLTC in the given year.

My results show that all active treatments reduced modeled PMORT in both 2003 and 2020 relative to the control compartments, reflecting the effectiveness of their wildfire mitigation goals. Of the active treatments, the mechanical only compartment's produced the highest PMORT and lowest SLTC in both 2003 and 2020, followed by the burn only compartments, with the mechanical + burn compartments producing the lowest PMORT and highest SLTC. This indicates that combining the two treatments may optimize their wildfire mitigation potential.

When assessing changes in PMORT and SLTC between 2003 and 2020, the differences between the active treatments and the control treatment are considerable. As fuel loads and wildfire hazard increased in the control compartments between 2003 and 2020, their modeled PMORT values and consequent SLTC values changed dramatically, with differences between 2003 and 2020 that were significantly larger in magnitude than those of the active treatments (Figure 2, Figure 3, Table B1, Table B2). The wildfire mitigation effects of the mechanical only and burn only treatments caused their increase in PMORT and consequent decrease in SLTC between 2003 and 2020 to be a much smaller change than that of the control compartments (Figure 2, Figure 3, Table B1, Table B2). It is noteworthy that the mechanical + burn treatment's wildfire mitigation effects were strong enough to reverse the direction of the trend between 2003 and 2020, with a decreasing average PMORT value and an increasing average SLTC value (Figure 2, Figure 3, Table B1, Table B2). This also indicates that combining the two treatments may optimize their wildfire mitigation potential, along with their potential for stabilizing aboveground carbon stocks.

Expected carbon stocks

Total ELTC following repeated treatments varied by treatment type, with the control treatment resulting in the highest ELTC at the current annual burn probability of 0.01, followed by burn only treatments, then mechanical only treatments, with mechanical + burn treatments resulting in the lowest ELTC at the current annual burn portability of 0.01. Again, it is important to recognize that the ELTC metric assumes that a wildfire will burn the stand under the specific conditions used to estimate AFC, LTC, SLTC, and ELTC. I used data from 2020 to estimate ELTC in 2030, in order to assess the net AFC of multiple repeated treatments (3 burn-only treatments, and 2 mechanical only/mechanical + burn treatments). The benchmark wildfire probability of 0.01 was asserted by Foster et al. as the current estimation agreed upon by existing literature, and I chose to use an increasing range of annual burn probability to assess ELTC consistently with the common notion that wildfire frequency is predicted to increase (Foster et al. 2020, Stephens 1997, Westerling et al. 2006, Westerling 2016). Although these intensified estimations of wildfire probability may be overestimated for the near future, my analysis and results clearly demonstrate the increasing ELTC of active treatments relative to the control treatment as wildfire probability increases.

I found that under 2020 conditions, when projected to 2030, the rate of decrease in ELTC with increasing annual burn probabilities was highest for the control treatment, followed by the mechanical only treatment, then the burn only treatment, with the mechanical + burn treatment resulting in the slowest rate of decrease in ELTC with increasing annual burn probability (Figure 3, Table C1). This resulted in the mechanical + burn treatment's average ELTC exceeding that of the control treatment at annual burn probability of ~0.055 and above, the burn only treatments exceeding the control at P_{annual} = ~0.06, and the mechanical only treatments exceeding the control at P_{annual} = ~0.07. This indicates that as wildfire probability increases, the wildfire mitigation effects of active treatments outweighs the upfront carbon costs, resulting in higher net aboveground carbon stocks than the control. This suggests that as climate change and altered forest structures increase the frequency of large, high severity wildfires, all three tested treatments provide higher net AFC stocks than if no management is prescribed.

Synthesis

Revisiting the central research question of this study, which is whether fuel-reduction treatments can increase long-term net carbon storage by mitigating wildfire in CA MCF, my results show that they can. Total average ELTC of all active treatment types exceeded that of the control treatment at higher values of annual burn probability. Therefore all active treatments increased the net AFC retained following a high severity wildfire relative to the control at higher values of annual burn probability, considering upfront carbon costs and probability of burning. This suggests that as anthropogenic climate change increases the duration and intensity of weather conditions conducive to large, high severity wildfires in the long-term, all three tested treatments will provide higher net AFC stocks than if no management is prescribed.

Limitations and future directions

Along with the limitations previously mentioned, there are some other constraints to the study design and methods I used. First, although the structure and species composition of Blodgett Research Forest is representative of average California mixed conifer forest conditions (Stephens and Moghaddas 2005), diversity in management, disturbance history, and other confounding factors may make applying my conclusions to other areas of CA MCF unreasonable. The influential factors excluded from this study include those previously mentioned (carbon in harvested wood, carbon emissions from treatment operations, and carbon stored in fire-killed snags), and other significant forest carbon pools, like the underground (root/soil) carbon pool. My model for evaluating the reduction in potential wildfire carbon emissions (i.e. PMORT in 2003 and 2020) is simplified, and does not fully account for all carbon fluxes during and following a potential wildfire.

Some limitations of the wildfire modeling include the ambiguity of the averaged wildfire probabilities I used, and FVS-FFE's limited choices of fuel models. The default fuel models I used are generally considered the most fitting for CA MCF (Collins 2014), but may be too generalized for the unpredictable reality of wildfire behavior and fuel dynamics. Another considerable factor excluded from my study is the changes to potential for future carbon accumulation caused by high severity wildfire. In a high severity patch of wildfire, regeneration potential is limited, and vegetation type conversion to shrubland may significantly limit the future carbon uptake potential of these forests. Given these limitations, further research is needed to address system complexities like harvested wood carbon, treatment operational emissions, carbon in fire-killed snags, and future carbon accumulation potential following high severity wildfire. Additionally, further experimentation with FVS-FFE's full capability to customize wildfire modeling is needed to overcome limitations not addressed in this study.

Management implications

Incentivizing short-term carbon storage in CA forests in the current carbon offset program framework is unsustainable as the probability of high severity wildfire increases. The risk of destabilizing live tree carbon stocks in CA MCF through large high severity wildfires and potential type conversion is of great concern to CA climate change mitigation goals (Campbell et al. 2007, Earles et al. 2014, Liang et al. 2017, Stephens et al. 2020). My estimations of active treatments' ELTC exceeded that of the control treatments as the probability of wildfire increased, meaning wildfire fuel treatments will become more beneficial to long-term carbon accumulation as climate change and altered forest structures increase wildfire probability in the future. Forest carbon offset protocols should consider long-term net carbon storage of forests in offset credit calculation/issuance, along with incentivizing wildfire fuel treatments to prevent carbon leakage via wildfire. Additionally, ample research shows that wildfire fuel treatments like those of the Fire and Fire Surrogate study can have lasting benefits on forest structure, tree vigor, wildlife habitat, water quality, and resilience to disturbances like wildfire, bark beetle outbreaks, and drought, among other virtues (Zald et al. 2022, Stephens and Moghaddas 2005, Collins et al. 2014, Steel et al. 2023, Dobre et al. 2022, Foster et al. 2020). The increase in long-term net carbon storage caused by wildfire fuel treatments is one of many co-benefits of fuel treatments, and the upfront carbon costs and other obstacles to treatment may be outweighed by these various benefits in many management circumstances.

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REFERENCES

Agee, J. K., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211: 83-96 211:83–96.

Ager, A. A., M. A. Finney, A. McMahan, and J. Cathcart. 2010. Measuring the effect of fuel treatments on forest carbon using landscape risk analysis. Natural Hazards and Earth System Sciences 10:2515–2526.

Brown, J. K. 1974. Handbook for inventorying downed woody material. Gen. Tech. Rep. INT-16. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 24 p. 016.

California Air Resources Board Offset Credit Issuance Table, (2022). California Air Resources Board. Available online at: https://ww2.arb.ca.gov/our-work/programs/compliance-offset-program

CARB (California Air Resources Board). 2022. California Greenhouse Gas Emissions for 2000 to 2020 Trends of Emissions and Other Indicators.

Campbell JC, Donato DC, Azuma DA, and Law B. 2007. Pyrogenic carbon emission from a large wildfire in Oregon, United States. J Geophys Res-Atmos 112: G04014.

Chojnacky, D. C., and M. Milton. 2008. Measuring Carbon in Shrubs. Pages 45–72 in C. M. Hoover, editor. Field Measurements for Forest Carbon Monitoring: A Landscape-Scale Approach. Springer Netherlands, Dordrecht.

Christensen, G.A.; Gray, A.N.; Kuegler, O.; Tase, N.A.; Rosenberg. M. 2021. AB 1504 California Forest Ecosystem and Harvested Wood Product Carbon Inventory: 2019

Reporting Period Data update. U.S. Forest Service agreement no. 18-CO-11052021-214, California Department of Forestry and Fire Protection agreement no. 8CA04056. Sacramento, CA: California Department of Forestry and Fire Protection and California Board of Forestry and Fire Protection. 447 p.

Collins, B. M., A. J. Das, J. J. Battles, D. L. Fry, K. D. Krasnow, and S. L. Stephens. 2014. Beyond reducing fire hazard: fuel treatment impacts on overstory tree survival. Ecological Applications. 24(8): 1879-1886 24:1879–1886.

Cousins, S. J. M., J. J. Battles, J. E. Sanders, and R. A. York. 2015. Decay patterns and carbon density of standing dead trees in California mixed conifer forests. Forest Ecology and Management 353:136–147.

Dobre, M., J. W. Long, C. Maxwell, W. J. Elliot, R. Lew, E. S. Brooks, and R. M. Scheller. 2022. Water quality and forest restoration in the Lake Tahoe basin: impacts of future management options. Ecology and Society 27.

Dore, S., D. Fry, B. Collins, R. Vargas, R. York, and S. Stephens. 2016. Management Impacts on Carbon Dynamics in a Sierra Nevada Mixed Conifer Forest. PloS one 11:e0150256.

Earles, J. M., M. P. North, and M. D. Hurteau. 2014. Wildfire and drought dynamics destabilize carbon stores of fire-suppressed forests. Ecological Applications 24:732–740.

Fahey, T., P. Woodbury, J. Battles, C. Goodale, S. Hamburg, S. Ollinger, and C. Woodall. 2010. Forest carbon storage: Ecology, management. Frontiers in Ecology and the Environment 8:245–252.

Finney, M. A. 2005. The challenge of quantitative risk analysis for wildland fire. Forest Ecology and Management 211:97–108.

Foster, D. E., J. J. Battles, B. M. Collins, R. A. York, and S. L. Stephens. 2020. Potential wildfire and carbon stability in frequent-fire forests in the Sierra Nevada: trade-offs from a long-term study. Ecosphere 11:e03198.

Hurteau, M. D., S. Liang, K. L. Martin, M. P. North, G. W. Koch, and B. A. Hungate. 2016. Restoring forest structure and process stabilizes forest carbon in wildfire-prone southwestern ponderosa pine forests. Ecological Applications 26:382–391.

IPCC. 2003. Good practice guidance for land use, land-use change and forestry /The ` Intergovernmental Panel on Climate Change. Ed. by Jim Penman. Page (J. Penman, Ed.). Hayama, Kanagawa.

IPCC, 2023: Climate Change 2023: Synthesis Report. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, H. Lee and J. Romero (eds.)]. IPCC, Geneva, Switzerland, pp. 35-115, doi: 10.59327/IPCC/AR6-9789291691647

Kobziar, L., J. Moghaddas, and S. Stephens. 2006. Tree mortality patterns following prescribed fires in a mixed conifer forest. Canadian Journal of Forest Research-revue Canadienne De Recherche Forestiere - CAN J FOREST RES 36:3222–3238.

Liang, S., M. D. Hurteau, and A. L. Westerling. 2017. Potential decline in carbon carrying capacity under projected climate-wildfire interactions in the Sierra Nevada. Scientific Reports 7:2420.

Loudermilk, E. L., R. M. Scheller, P. J. Weisberg, J. Yang, T. E. Dilts, S. L. Karam, and C. Skinner. 2013. Carbon dynamics in the future forest: the importance of long-term successional legacy and climate-fire interactions. Global Change Biology 19:3502–3515.

Mcginnis, T., C. Shook, and J. Keeley. 2010. Estimating Aboveground Biomass for Broadleaf Woody Plants and Young Conifers in Sierra Nevada, California, Forests. Western Journal of Applied Forestry 25:203–209.

North, M., M. Hurteau, and J. Innes. 2009. Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. Ecological Applications 19:1385–1396.

North, M. P., and M. D. Hurteau. 2011. High-severity wildfire effects on carbon stocks and emissions in fuels treated and untreated forest. Forest Ecology and Management 261:1115–1120.

Reinhardt, E., N. Crookston, S. Beukema, W. Kurz, J. Greenought, D. Robinson, and D. Lutes. 2003. The Fire and Fuels Extension to the Forest Vegetation Simulator (June 2015 Revision).

Ryan, K. C., and E. D. Reinhardt. 1988. Predicting postfire mortality of seven western conifers. Canadian Journal of Forest Research 18:1291–1297.

Schoemaker, P. 1982. The Expected Utility Model: Its Variants, Purposes, Evidence and Limitations. Journal of Economic Literature 20:529–63.

Schwilk, D. W., J. E. Keeley, E. E. Knapp, J. McIver, J. D. Bailey, C. J. Fettig, C. E. Fiedler, R. J. Harrod, J. J. Moghaddas, K. W. Outcalt, C. N. Skinner, S. L. Stephens, T. A. Waldrop, D. A. Yaussy, and A. Youngblood. 2009. The National Fire and Fire Surrogate Study: Effects of Fuel Reduction Methods on Forest Vegetation Structure and Fuels. Ecological Applications 19:285–304.

Starrs, C. F., V. Butsic, C. Stephens, and W. Stewart. 2018. The impact of land ownership, firefighting, and reserve status on fire probability in California. Environmental Research Letters 13:034025.

Steel, Z. L., G. M. Jones, B. M. Collins, R. Green, A. Koltunov, K. L. Purcell, S. C. Sawyer, M. R. Slaton, S. L. Stephens, P. Stine, and C. Thompson. 2023. Mega-disturbances cause rapid decline of mature conifer forest habitat in California. Ecological Applications 33:e2763.

Stephens, C., B. Collins, and J. Rogan. 2020. Land ownership impacts post-wildfire forest regeneration in Sierra Nevada mixed-conifer forests. Forest Ecology and Management 468:118161.

Stephens, S., J. Moghaddas, B. Hartsough, E. Moghaddas, and N. Clinton. 2009. Fuel treatment effects on stand-level carbon pools, treatment-related emissions, and fire risk in a Sierra Nevada mixed-conifer forestPublication No. 143 of the National Fire and Fire Surrogate Project. Canadian Journal of Forest Research 39:1538–1547.

Stephens, S. L. 1997. Evaluation of the effects of silvicultural and fuels treatments on potential fire behaviour in Sierra Nevada mixed-conifer forests - ScienceDirect. Forest Ecology and Management 105:21–35.

Stephens, S. L. 2001. Fire history differences in adjacent Jeffrey pine and upper montane forests in the eastern Sierra Nevada. International Journal of Wildland Fire 10:161.

Stephens, S. L., and J. J. Moghaddas. 2005. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. Forest Ecology and Management 215:21–36.

Stephens, S. L., B. M. Collins, and G. Roller. 2012. Fuel treatment longevity in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 285:204–212.

Stephens, S. L., R. E. J. Boerner, J. J. Moghaddas, E. E. Y. Moghaddas, B. M. Collins, C. B. Dow, C. Edminster, C. E. Fiedler, D. L. Fry, B. R. Hartsough, J. E. Keeley, E. E. Knapp, J. D. McIver, C. N. Skinner, and A. Youngblood. 2012. Fuel treatment impacts on estimated wildfire carbon loss from forests in Montana, Oregon, California, and Arizona. Ecosphere. 3(5): 1-17, Article 38.

Van Wagtendonk, J. W., J. M. Benedict, and W. M. Sydoriak. 1998. Fuel Bed Characteristics of Sierra Nevada Conifers. Western Journal of Applied Forestry 13:73–84.

Westerling, A. L. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B: Biological Sciences 371:20150178.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science (New York, N.Y.) 313:940–943.

Zald, H. S. J., C. C. Callahan, M. D. Hurteau, M. J. Goodwin, and M. P. North. 2022. Tree growth responses to extreme drought after mechanical thinning and prescribed fire in a Sierra Nevada mixed-conifer forest, USA. Forest Ecology and Management 510:120107.

APPENDIX A: Observed Carbon Stocks

Table A1. Observed carbon stocks for each treatment type between 2001-2020. Average aboveground forest carbon (AFC - MgC/ha) was calculated as the sum of plot level average tree, understory, and fuels biomass. Methods for converting biomass to carbon are described in the methods section.

Treatment	Year	Average AFC (MgC/ha)	Standard Error	Difference	Percent change
Control	2001	206.30269	14.38457	N/A	N/A
Control	2003	211.41011	13.71032	5.10742	2.475692392
Control	2009	232.361	12.66232	20.95089	9.910070053
Control	2016	284.91448	13.53918	52.55348	22.61716897
Control	2020	296.68202	12.9314	11.76754	4.130200754
Mechanical Only	2001	211.3527335	7.847869	N/A	N/A
Mechanical Only	2003	180.0337555	8.939965	-31.318978	-14.81834537
Mechanical Only	2009	206.95907	8.17444	26.9253145	14.9557034
Mechanical Only	2016	240.74337	12.939857	33.7843	16.32414564
Mechanical Only	2020	220.88579	2.051015	-19.85758	-8.248443145
Burn Only	2001	191.666124	6.718881	N/A	N/A
Burn Only	2003	147.660411	8.204759	-44.005713	-22.95956744
Burn Only	2009	170.019223	7.933388	22.358812	15.14204914
Burn Only	2010	158.433092	6.309003	-11.586131	-6.814600605
Burn Only	2016	201.119884	9.14062	42.686792	26.94310353
Burn Only	2018	182.611251	7.590096	-18.508633	-9.202786235
Burn Only	2020	204.8868875	8.320342	22.2756365	12.19839215
Mechanical + Burn	2001	234.07683	13.335366	N/A	N/A
Mechanical + Burn	2003	167.58285	11.853069	-66.49398	-28.40690384
Mechanical + Burn	2009	174.40417	13.921829	6.82132	4.070416513
Mechanical + Burn	2016	208.79865	15.676825	34.39448	19.72113396
Mechanical + Burn	2020	167.11801	3.156545	-41.68064	-19.9621214

APPENDIX B: Wildfire Modeling and Carbon Stability

Table B1. Modeled percent mortality (PMORT) under severe wildfire weather conditions after initial (2003) and repeated treatments (2020). PMORT is an output from FVS that represents the percentage of plot basal area predicted to die within the first three years following the modeled wildfire. This metric incorporates both immediate and delayed mortality factors, using crown length, diameter, tree species, and predicted scorch height.

Treatment	Year	Average PMORT	Difference	Percent change
Control	2003	72.34804	N/A	N/A
Control	2020	92.63246	20.28442	28.03727648
Mechanical Only	2003	47.96639	N/A	N/A
Mechanical Only	2020	59.43529	11.4689	23.91028385
Burn Only	2003	42.02193	N/A	N/A
Burn Only	2020	56.8114	14.78947	35.19464718
Mechanical + Burn	2003	24.08009	N/A	N/A
Mechanical + Burn	2020	19.38333	-4.69676	-19.50474438

Table B2. Stable live tree carbon (SLTC – MgC/ha) stocks after initial (2003) and repeated treatments (2020). SLTC was calculated using Equation 1, and represents the total live tree carbon expected to remain at least 3 years following a wildfire. Note that this does not include the carbon stored in fire-killed snags, making it an underestimation of the total stable carbon stored in standing trees following a wildfire.

Treatment	Year	SLTC (MgC/ha)	Difference	Percent change
Control	2003	46.27368	N/A	N/A
Control	2020	17.50431	-28.76937	-62.17221107
Mechanical Only	2003	73.31848	N/A	N/A
Mechanical Only	2020	69.55103	-3.76745	-5.138472592
Burn Only	2003	80.25023	N/A	N/A
Burn Only	2020	76.93483	-3.3154	-4.131327723
Mechanical + Burn	2003	108.65278	N/A	N/A
Mechanical + Burn	2020	118.8492	10.19642	9.384407836

APPENDIX C: Expected Carbon Stocks

Table C1. Expected live tree carbon (ELTC - MgC/ha) stocks in 2030. Expected carbon stocks were calculated using Equation 2 and Equation 3, and is a weighted average that combines both outcomes (observed carbon stocks and predicted post-wildfire stocks), weighted by the respective probabilities of their occurrence. The current annual burn probability is accepted and used in established literature as 0.01, and represents the annual probability of wildfire occurring in a given area of forest in the Sierra Nevada.

Annual burn probability	Control ELTC (MgC/ha)	Mechanical Only ELTC (MgC/ha)	Burn Only ELTC (MgC/ha)	Mechanical + Burn ELTC (MgC/ha)
0.01	214.5527	160.7913	167.5449	144.4343
0.015	203.8778	155.8485	162.6362	143.0483
0.02	193.7313	151.1503	157.9705	141.7308
0.025	184.0895	146.6858	153.5368	140.4789
0.03	174.9297	142.4445	149.3248	139.2896
0.035	166.23	138.4163	145.3244	138.16
0.04	157.9697	134.5914	141.526	137.0875
0.045	150.1285	130.9607	137.9203	136.0694
0.05	142.6873	127.5152	134.4986	135.1032
0.055	135.6276	124.2463	131.2523	134.1865
0.06	128.9317	121.1458	128.1733	133.3171
0.065	122.5827	118.206	125.2537	132.4928
0.07	116.5643	115.4193	122.4862	131.7113
0.075	110.8608	112.7784	119.8636	130.9708
0.08	105.4575	110.2765	117.379	130.2692
0.085	100.34	107.9069	115.0257	129.6047
0.09	95.49468	105.6633	112.7977	128.9756
0.095	90.90836	103.5397	110.6887	128.3801
0.1	86.56856	101.5302	108.6931	127.8166
Average rate of change (MgC/ha/0.01)	-14.22046	-6.584566667	-6.539088889	-1.846411111