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Multi-scale assessment of wildfire use on carbon stocks in the Sierra Nevada, CA

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Abstract

Background The active use of wildfire to meet forest management objectives is an important tool to increase the scale of forest restoration in dry, historically frequent-fire forests. While there are many benefits of reintroducing fire to these forests, the impact of wildland fire use policies in frequent-fire forests on aboveground carbon stocks has not yet been studied. In this study, we begin to fill this knowledge gap by assessing how fire frequency and severity affected aboveground carbon dynamics in two basins in the Sierra Nevada with a history of wildfire use over the past 20 to 50 years, compared to a nearby basin that has remained largely unburned.

Results Across two spatial and temporal scales, live carbon stocks in wildfire use areas decreased by on average 22–48% (depending on the scale and basin), while dead carbon stocks changed little or decreased in areas managed with wildfire. The unburned basin held higher amounts of carbon stocks than the burned basins, and on average, these stocks remained relatively constant over the study period. Fire severity appeared to exert a stronger influence on carbon change than frequency, with areas that burned at high severity resulting in the largest losses in aboveground carbon, regardless of the number of fires that the area had experienced.

Conclusions We found that greater wildfire use over several decades comes at some costs to carbon stocks as trees—which store the greatest amount of carbon in forested systems—are killed or consumed and forest patches transition to shrublands and meadows. In our study area, these conversions from forest to non-forest types were in relatively small high-severity patches and likely represent areas that were historically maintained as non-forested shrublands, grasslands, or wetlands by an intact fire regime. Additionally, we found that the surviving carbon stocks may be more resistant to future disturbances. Restoration of dry mixed-conifer forests solely by reintroducing a characteristic fire regime is a slow process, but increasing wildfire use in areas where it is feasible is an important tool that managers should consider to address our growing forest restoration needs.

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Resumen

Antecedentes El uso activo del fuego para lograr objetivos de manejo forestal representa una herramienta importante para incrementar su escala en la restauración de ese disturbio en bosques secos, donde éstos han sido históricamente afectados por incendios frecuentes. Aunque hay muchos beneficios con la reintroducción del fuego en estos bosques, el impacto de las políticas de uso de fuegos frecuentes sobre el stock de carbono no ha sido todavía lo suficientemente estudiado. En este trabajo, comenzamos a completar este vacío en el conocimiento mediante la determinación de cómo la frecuencia y severidad de los fuegos afectan la dinámica del carbono en dos cuencas de la Sierra Nevada, con una historia de uso del fuego en los pasados 20 a 50 años, comparados con una cuenca cercana que permaneció sin quemarse por un largo tiempo.

Resultados A través de dos escalas espaciales y temporales, el stock de carbono vivo en áreas quemadas decreció en un promedio de 22–48% (dependiendo de la escala y la cuenca), mientras que el stock de carbono muerto cambió levemente o decreció en esas áreas manejadas con quemas. La cuenca no quemada almacenó mayores stocks de carbono que las cuencas quemadas y, en promedio, esos stocks permanecieron relativamente constantes durante el período de estudio. La severidad del fuego parece ejercer una influencia más significativa que la frecuencia en su ocurrencia/uso sobre los cambios en el stock del carbono, y áreas que se quemaron a una alta severidad resultaron en mayores pérdidas de carbono almacenado en partes aéreas, independientemente del número de fuegos que el área pudiese haber experimentado.

Conclusiones Encontramos que un mayor uso del fuego durante varias décadas tuvo un costo en pérdida de los stocks de carbono, ya que los árboles – que almacenan la mayor cantidad de carbono en sistemas boscosos – se mueren o son consumidos y esas áreas quemadas transicionan hacia parches de arbustos y humedales. En nuestra área de estudios, la conversión de bosques a tipos no forestales fue ocurriendo en parches relativamente pequeños quemados a altas severidades, y parecen representar áreas que fueron históricamente mantenidas como arbustales, pastizales, o humedales bajo un régimen de fuego intacto. Adicionalmente, encontramos que el stock de carbono sobreviviente puede ser más resistente a disturbios futuros. La restauración de bosques secos mixtos de coníferas mediante la reintroducción de un régimen de fuegos característico, puede ser un proceso lento, aunque el incremento en el uso del fuego en áreas donde éste pueda ser factible, se presenta como una herramienta importante que los gestores del manejo debieran considerar para satisfacer las necesidades de restauración para el crecimiento de nuestros bosques.

Keywords Aboveground carbon stocks, Carbon change, Fire history, Forest restoration, Managed wildfire, Sierra Nevada, Wildfire use, Yosemite National Park

Background

Most dry, frequent fire-adapted forests in the western U.S. are highly departed from their historical fire regimes and are currently dominated by uncharacteristically dense stands that are more homogenous in structure and composition across the landscape (Hagmann et al. 2021; Stephenson et al. 2024). In order to effectively reestablish disturbance regimes and restore ecosystem function and resilience in these forests, increasing the scale of forest restoration or fuels treatment is necessary (North et al. 2012; Little Hoover Commission 2018; Vaillant and Reinhardt 2017). One method to meaningfully increase the scale of treatments is through the use of managed wildfire (previously termed Prescribed Natural Fire), which involves allowing lightning-ignited fires to burn to meet resource management objectives. At the same time, maintaining carbon storage or increasing carbon sequestration in forests supports natural climate solutions (Fargione et al. 2018). Wildfire use and carbon storage may seem like antithetical objectives, since in the short term, fire releases carbon into the atmosphere directly via combustion and indirectly via vegetation mortality and subsequent decomposition (Hurteau and Brooks 2011), but the impacts of increased fire use on carbon stocks in the long term and at different spatial scales are not well understood.

For millennia, managing wildfires to maintain open forest canopies that promoted culturally important foods, medicines, and materials, as well as reduced fire hazard and protected human communities, was a fundamental aspect of Indigenous stewardship (Anderson 2006; Goode et al. 2022). Beginning in the late 1700s, the cessation of Indigenous burning, harvesting of large, fire-resistant trees, and an early 1900s policy of fire suppression altered forest structure and composition in dry mixed-conifer forests (Stephens et al. 2015; Taylor et al. 2016). The practice of fire suppression remained the primary goal of federal land management until 1968, when the U.S. National Park Service created a policy to recognize fire as an ecological process in the western U.S. Since then, many land management plans in National Parks and U.S. Forest Service wilderness areas have included wildfire use policies to varying degrees (van Wagtendonk, 2007). In Yosemite National Park—which adopted these policies in 1972—several studies on the effects of allowing some areas of naturally ignited fires to burn have been conducted, the results of which have been overwhelmingly positive (Stephens et al. 2021). For example, the mosaic of overlapping and abutting fires has reduced fuel loads, limited the spread and severity of future wildfires (Collins et al. 2009; Meyer 2015), increased landscape heterogeneity (Boisramé et al. 2017a), and increased water yield (Boisramé et al. 2017b, Rakhmatulina et al. 2021). The ecological impacts of wildfire use policies on carbon stocks and carbon dynamics have not yet been studied.

In the long-term, carbon stock stabilization in forests, rather than carbon storage maximization, may be a more effective goal in mitigating climate change impacts (Hurteau and Brooks 2011; Liang et al. 2018). Increasing carbon stability—the ability of the residual carbon stocks to resist or recover from disturbances—minimizes the potential losses of aboveground carbon in high-severity, stand-replacing fires in dry, temperate forests (Hurteau and Brooks 2011). Before the interruption of historical fire regimes, dry, temperate western U.S. forests typically had low tree density and an open stand structure that was maintained by frequent, largely low-intensity fires (Scholl and Taylor 2010; Bernal et al. 2022). These fires would have released some stored carbon into the atmosphere by combusting mainly surface and ladder fuels and smaller diameter trees, but a majority of the aboveground carbon would have remained stored in the surviving trees, particularly in large-diameter trees, which store the greatest amount of aboveground carbon in forests (Lutz et al. 2012). The initial carbon loss would likely be offset by the subsequent regrowth of understory vegetation and growth of large, fire-resistant trees. That said, historical forests generally had much lower aboveground carbon stocks than those observed in the contemporary period (Collins et al. 2011; Bernal et al. 2022). The frequency of fire in historical forests kept stands at tree density levels well below their maximum site potential (Show and Kotok 1924, North et al. 2021). But, these lower tree densities allowed for persistence of relatively stable aboveground carbon stocks through drought and other climatic shifts over centuries or more (Knight et al. 2022). This is in direct contrast with aboveground carbon stocks in contemporary forests, which greatly exceed their carbon carrying capacity, especially given current climatic conditions (Goodwin et al. 2020; Hurteau et al. 2024). As such, contemporary forests are vulnerable to large-scale carbon losses due to both drought (Goodwin et al. 2020) and wildfire (Hurteau et al. 2011; Taylor et al. 2014; Stephenson et al. 2024).

In addition to the differences in temporal scales (short-term vs. long-term) of the potential impacts of increased wildfire use on carbon stocks, the differences in spatial scales need to be considered. At the individual tree or stand scale, fire reduces carbon by combusting and killing trees and understory vegetation and consuming surface fuels. At the landscape scale, carbon is released and sequestered in patches over time as different areas experience fires, creating a mosaic of fire histories—i.e., pyrodiversity—which may limit the extent of high-severity patches (Stephens et al. 2021) and increase the ability of residual carbon stocks to resist or recover from future fires (Koontz et al. 2020). In broader ecological terms, the effects of pyrodiversity (Martin and Sapsis 1992; Steel et al. 2021, 2024) created through long-term wildfire use may create quasi-steady state conditions, where the proportional composition of vegetation types and carbon stocks do not change at the landscape scale even though individual patches change with any given disturbance (Holling 1973; Bonnicksen and Stone 1982; Boisramé et al. 2017a). The temporal scale at which this might occur in contemporary dry, western U.S. forests requires more investigation.

In this study, we had two overarching research questions regarding the reintroduction of wildfire on aboveground carbon stocks in Sierra Nevada forests. First, we asked how does fire frequency and severity affect aboveground carbon change? We investigated this question using two datasets with different spatial and temporal scales: (1) at the plot-level across 20 years using forest and fuels inventory plots in one basin in Yosemite (Illilouette Creek Basin), and (2) at the landscape-level across 37 years using remote sensing and carbon flux modeling data in three basins in the Sierra Nevada with different fire histories. Second, we asked how resistant is the current aboveground carbon to future disturbance, and are there any relationships between carbon stability and fire history? We evaluated this question by comparing carbon stability to future disturbance among the three basins with different fire histories at the plot-level, using a network of plots that were sampled in 2021 and 2022. Our study furthers our knowledge about the potential carbon impacts of increasing the use of wildfire for forest restoration.

Methods

Study area

Our study was conducted in three areas in the Sierra Nevada: Illilouette Creek Basin, North and Middle Fork watersheds of the San Joaquin River, and the Badger Pass area (hereafter referred to as Illilouette, San Joaquin, and Badger, respectively; Fig. 1). Prior to Euro-American settlement in the mid-1800s, these areas burned frequently

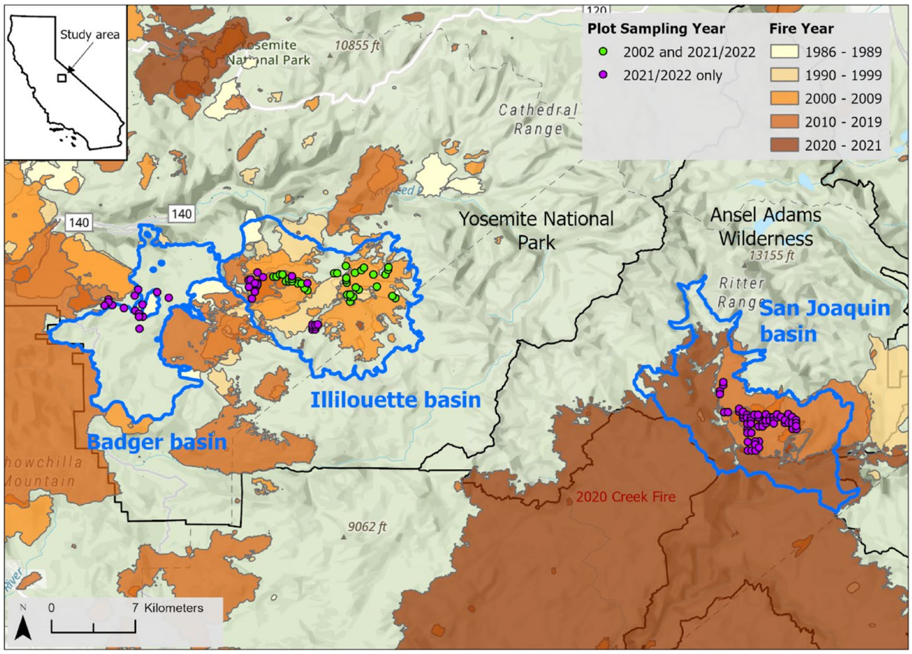

Fig. 1 The three study area basins with different fire histories. Badger is largely unburned (since at least 1930), Illilouette has had many fires since the onset of the wildfire use policy in 1972, and San Joaquin has had a few fires since the onset of the wildfire use policy in 2003. Polygons in gradations of orange are fire perimeters from 1985 to 2021; data is from CAL Fire (2025). Green points are the plot locations in Illilouette basin that were sampled in both 2002 and 2021/2022; data from these plots was used to assess how fire history affected carbon change. The purple points were established in 2021/2022; data from the purple plots in addition to the 2021/2022 data from the green plots were used to assess the carbon stability of the current carbon stocks (i.e., ability of residual carbon stocks to resist or recover from future disturbance)

from both lightning and Indigenous ignitions. After settlement, fire frequency declined due to the cessation of Indigenous burning after their forced removal from the landscape, and beginning in the late 1800s and early 1900s, fires in these areas were essentially excluded due to a federal policy of full suppression (Collins and Stephens 2007). After many decades of fire exclusion, wildfire use was established in Yosemite National Park (within which Illilouette and Badger are located) and the Ansel Adams Wilderness (within which San Joaquin is located). These policies allowed for some lightning-ignited fires to burn to restore the process of fire on the landscape. The policy was established in Yosemite in 1972 with the first widespread fire in Illilouette in 1974 (Starr King Fire, ~ 1600 ha) (van Wagtendonk 2007), and was established in Ansel Adams Wilderness in 2003, with the first widespread fire in San Joaquin in 2018 (Lions Fire, ~ 5400 ha). Since the beginning of their respective programs, 45 fires > 0.5 ha in size have burned in Illilouette and seven fires have burned in San Joaquin. Badger remains largely unburned since 1930 (when fire records in Yosemite began being documented). Even though a few fires > 4 ha have burned in Badger basin from 1930 to 2021, the fire rotation (the amount of time necessary to burn a cumulative area equal to the area of the basin) remains very long (403.5 years) compared to San Joaquin and Illilouette (95.3 and 68.8 years, respectively; fire rotation was calculated using fire polygons from CAL Fire 2025 and Yosemite National Park 2024). It is worth noting that despite the extensive period of wildfire use in Illilouette, suppression actions have taken place on some lightning-ignited fires since the establishment of the natural fire program (Collins et al. 2009). The reasons for these suppression actions include that burning exceeded maximum manageable area, violated air quality standards, or impacted park visitor use, and due to other political concerns (Collins et al. 2009; Stephens et al. 2021).

All three basins in our study had similar environmental conditions. The climate is Mediterranean, characterized by cool, wet winters and warm, dry summers, with precipitation falling primarily as snow in the winter and averaging around 100 cm annually (PRISM Climate Group 2025). Temperatures vary widely throughout the year, ranging from an average January minimum of − 6 °C to an average July maximum of 25 °C (PRISM Climate Group 2025). Elevation ranges from 1400 to 3000 m. Vegetation is dominated by upper-elevation mixed-conifer forests composed of white fir (*Abies concolor* [Gord. & Glend.] Lindl. ex Hildebr.), red fir (*A. magnifica* A. Murray bis), Jeffrey pine (*Pinus jeffreyi*

Balf.), sugar pine (*P. lambertiana* Douglas), lodgepole pine (*P. contorta* var. *murrayana* [Grev. & Balf.] Critchfield), and scattered incense-cedar (*Calocedrus decurrens* [Torr.] Florin) and quaking aspen (*Populus tremuloides* Michx.). Forested areas are interspersed with meadows, shrublands (primarily whitethorn [*Ceanothus cordulatus* Kellogg]), and rock outcrops.

Plot sampling design

To investigate the relationship between the change in aboveground carbon stocks and fire history (fire frequency and severity) at the plot level, we used data from 37 plots that were established in 2002 and remeasured in 2021 and 2022 in Illilouette. We investigated the relationship between total aboveground carbon stocks and fire history, assessed the stability/resistance of aboveground carbon to future wildfire and other disturbance, and compared the differences in carbon dynamics among the three study areas using data from 129 plots that were measured in 2021 and 2022 (57 in Illilouette, 57 in San Joaquin, 15 in Badger). Plot locations were stratified based on fire occurrence since the beginning of the fire use policy and vegetation type, representing a stratified random sample. All of the plots in this study were sampled with the same protocol.

Plots were 500 m^2 circular fixed-radius (radius = 12.62 m), with GPS coordinates and overall site conditions recorded at plot center. For every overstory tree (≥ 10 cm diameter-at-breast-height [DBH]) in the plot, species and status (live or dead) were recorded, and DBH was measured. In the 2002 inventory in Illilouette, height and height to live crown base (HLCB) were measured for every tree; in the 2021/2022 inventory in all three areas, height and HLCB were only measured on a subset of trees. We predicted the heights for trees that were not measured in the field by creating species-specific DBH-height log–log regressions using the data from trees with field-measured heights. Saplings (< 10 cm DBH and ≥ 2 m tall) and seedlings (0.5–2 m tall) were tallied by species in two quadrants (for a total area of 250 m^2) in each plot. Tree canopy cover was measured with a sighting tube at 25 points distributed in a grid across the plot. Shrub cover and average height by species was visually estimated over the entire plot area. Downed woody surface fuels, litter, and duff were sampled along three transects using the planar-intercept method (Brown 1974), with 1-h (0–0.64 cm diameter) and 10-h fuels (0.65–2.54 cm diameter) tallied from 2 to 4 m, 100-h fuels (2.55–7.62 cm diameter) tallied from 2 to 6 m, and 1000-h fuels (> 7.62 cm diameter) sampled from 2 to 12 m on the transect. Litter and duff fuel depths (cm) were measured at two locations along each transect.

Plot-level biomass and carbon calculations

Biomass of overstory trees (live and dead) and load of fine woody fuels (1-h, 10-h, and 100-h), coarse woody fuels (1000-h), litter, and duff were calculated using the *BerkeleyForestsAnalytics* package in R (Rutherford et al. 2024), which contains a set of functions to calculate and report standard metrics from forest inventory data. For overstory tree biomass, we used the *BiomassNSVB* function, which includes equations and parameters from the 2024 national-scale volume and biomass (NSVB) framework (Westfall et al. 2024). For loads of downed dead fuels, we used the *FineFuels*, *CoarseFuels*, and *LitterDuff* functions for fine woody, coarse woody, litter, and duff fuels, respectively. To calculate the biomass of seedlings and saplings, we used the Forest Vegetation Simulator (FVS) because it contains a biomass model for small trees that is based only on height. DBH and height of seedlings and saplings were not measured in the field, so we entered a DBH of 0.25 cm for these trees (to signal to FVS that the tree is on the small tree plot so the height-only model is used) and a height of 2 m for saplings (the minimum height for this size class) and 1.25 m for seedlings (the midway height for this size class). Because we used the minimum height for all saplings, we likely underestimated the biomass for this size class, but the effect of this on the total aboveground biomass for each plot is minimal because the proportion of sapling biomass to total biomass is small (see Results). Shrub biomass was calculated using species-specific coefficients from McGinnis et al. (2010). McGinnis et al.'s (2010) equations are for individual shrubs, so we had to estimate the number of average-sized individuals on each plot by converting percent shrub cover to area (m^2), computing the mean individual crown area (assuming a circular crown; crown diameters from Table 2 in McGinnis et al. 2010), and estimating the number of average-sized individuals on the plot by dividing area cover by the mean individual crown area. Then, we calculated each individual's biomass using Eq. 2 in McGinnis et al. (2010), which has inputs of crown diameter (computed in above step) and height (estimated in field). Finally, we calculated the total biomass per plot by summing the individual species' biomass and converting to Mg ha^{-1}. For the few and uncommon species in our plots that were not in McGinnis et al. (2010), we used the generic coefficients.

Carbon fractions were applied individually to each carbon pool. The carbon fraction for trees (overstory, saplings, and seedlings) was species-specific and different for live vs. dead trees (Table S10a and S10b in Westfall et al. 2024). We used a carbon fraction of 49% for shrubs (Campbell et al. 2009), 50% for fine woody fuels (Penman et al. 2003), 51% for coarse woody fuels (average fraction of logs in Harmon et al. 1987), and 37% for litter and duff

(Smith and Heath 2002). Carbon density (MgC ha^{-1}) for each carbon pool was summarized for each plot.

For each plot, we calculated total tree density, tree species composition by basal area, total basal area, quadratic mean diameter, percent shrub cover, and percent canopy cover. In the 37 plots in Illilouette that were established in 2002 and remeasured in 2021/2022, we also calculated the change in carbon density by pool. In the 129 plots in Illilouette, San Joaquin, and Badger that were measured in 2021 and 2022, we calculated two metrics of carbon stability: the proportion of carbon in a disturbance-resistant pool and relative Stand Density Index (SDI). We defined the "resistant pool" carbon as live trees > 60 cm DBH; studies have shown that trees that are greater than this size are more resistant to wildfire (Hurteau and North 2009). Relative SDI was calculated per North et al. (2022); this metric is a relative measure of tree competition and compares a plot's calculated SDI (using the summation method) with the maximum SDI (max number of trees per hectare based on species composition and local conditions). We classified each plot into three forest types as described in North et al. (2022)—pine-mixed conifer (> 50% basal area (BA) pine), mesic-mixed conifer (> 50% BA fir), and xeric-mixed conifer (< 50% pine or fir)—and used the max SDI from those, which ranged from 902 trees ha^{-1} to 1359 trees ha^{-1}. We compared the mean relative SDI among the study areas and forest types to thresholds that have been established for different degrees of competition (< 25%: no competition, 25–35%: onset of competition, 35–60%: full site occupancy, > 60%: imminent mortality) (North et al. 2022).

Basin-level dataset

The three basin areas (Illilouette, San Joaquin, and Badger) were delineated to include areas that have burned since the onset of wildfire use policies (in the cases of Illilouette and San Joaquin) or have remained largely unburned since the last widespread fire in the area (in the case of Badger). Basin area boundaries were based on Hydrologic Unit Code subwatershed (HUC12; U.S. Geological Survey 2024) boundaries and an upper elevation threshold of 2700 m (above which fire does not spread easily in this region, due to limited fuel and lack of fuel continuity). Illilouette was a 11,540 ha area within the Illilouette Creek watershed, San Joaquin was a 11,092 ha area within parts of the North Fork and Lower Middle Fork San Joaquin River watersheds, and Badger was a 7447 ha area within parts of the Bridalveil Creek, Lower South Fork, and Middle South Fork Merced River watersheds (Fig. 1).

To investigate carbon dynamics and fire history at the basin scale, we used the aboveground carbon data layers developed by the Center for Ecosystem Climate Solutions (CECS) (Potter et al. 1993, Hemes et al. 2023, Wang et al. 2024, data available at: Center for Ecosystem Climate Solutions 2024). The datasets are 30 m resolution maps of live (tree, shrub, and herbaceous) and dead (standing dead trees, downed coarse woody debris, and downed fine woody debris) aboveground carbon density values from remote sensing techniques paired with Random Forest classification and regression models and calculated using the Carnegie-Ames-Stanford Approach model (Potter et al. 1993). The datasets are annual (beginning in 1985), with biomass calculated at the end of the water year (end of September). We used the live and dead aboveground carbon datasets from 1985 and 2021 in this analysis to align with our plot-level analyses, and calculated the change in carbon density from 1985 to 2021 for live and dead pools separately at 30 m resolution for each basin.

Fire history datasets

Fire frequency metrics (number of fires, time since fire, and fire return interval) were derived from the historical fire perimeter dataset that is maintained by the Fire and Resource Assessment Program of the California Department of Forestry and Fire Protection (CAL Fire 2025). The fire severity dataset (30 m resolution) was derived using Landsat TM and OLI sensors, Google Earth Engine (Gorelick et al. 2017), and the Parks model (Parks et al. 2019), which created continuous Composite Burn Index (CBI) values (Key and Benson 2006). These continuous CBI values were used in the conditional inference tree analysis, but for all other analyses, severity values were binned into four categories based on the CBI value (unchanged = 0–0.1, low = 0.1–1.24, moderate = 1.25–2.24, high = 2.25–3.0; Miller and Thode 2007).

We used different fire frequency and severity datasets for each analysis due to the different spatial and temporal scales. For the plot-level change in carbon analysis in Illilouette, only fires that occurred between 2002 and 2021 in the basin were considered, to align with the plot sampling years. No fires occurred in Illilouette between the 2021 and 2022 plot sampling efforts, so we used the same fire frequency dataset regardless of the year the plot was sampled. For the plot-level carbon stability analysis in the three basins, we included fires from 1985 to 2021 (1985 was the earliest year on record for the severity dataset); again, no fires occurred in the plot locations between 2021 and 2022. For each plot, fire frequency (number of fires and time since fire) and severity values (severity of most recent fire and average severity across multiple fires) were extracted based on the plot's spatial intersection with the fire perimeter and fire severity datasets.

For the basin-level analyses, we included fires that occurred in or after 1985 and before September 2021 in

Illilouette and San Joaquin basins, to match the temporal extent of the CECS carbon data. The 2020 Creek Fire burned through the southern part of the San Joaquin basin, so this fire is included in the basin-level analysis, but did not directly affect any plot-level data. Fires that were less than 1 ha in size were removed from analysis because they did not have true boundaries mapped in the fire perimeter dataset but were rather delineated by a point. For the fire frequency analysis, we converted the fire perimeter dataset from vector form to a raster and summed the number of unique fires within each 30 m pixel to calculate the number of fires per pixel. We then merged this with the carbon change dataset at each pixel to investigate the relationship between carbon change and fire frequency; fire frequency values were categorized into groups (0, 1, 2, 3, or 4 fires in Illilouette and 0, 1, 2, or 3 fires in San Joaquin). The fire severity dataset was a raster of CBI values of the most recent fire from 1985 to 2021 for each 30 m pixel encompassing the extents of Illilouette and San Joaquin basins. The fire severity raster was resampled to match the extent and alignment of the carbon change rasters, then both rasters were overlaid, and the CBI and carbon change values were extracted for each pixel. CBI values were categorized into four fire severity classes (unburned, low, moderate, and high).

Analyses

All analyses were conducted in R statistical software (R Core Team 2024). At both the plot- and basin-level scales, we performed bootstrapping analyses using the *boot* package in R (Canty and Ripley 2024); this non-parametric approach is useful for small datasets and non-normal distributions. We assessed differences in the carbon response variable (mean carbon change or proportion of carbon in a resistant pool) among fire frequency groups (0, 1, 2, 3, or 4 fires, depending on the analysis) and fire severity groups (unburned, low, moderate, or high severity) using 1000 resamples. In the plot-level analysis, the low- and moderate-severity groups were combined into a single low-moderate severity class due to low sample size. Statistically significant differences in the carbon response variable among fire frequency and fire severity groups were determined by non-overlapping 95% confidence intervals from the bootstrapping analysis.

Additionally, with the remeasured plot data in Illilouette, we performed a conditional inference tree analysis using the *ctree* function in the *partykit* package in R (Hothorn et al. 2006; see Table S1 for model parameters) to explain the differences in carbon change as it relates to vegetation composition and structure and fire history. The dependent variable was the change in total carbon density from 2002 to 2021/2022. The predictors were vegetation composition and structure (including percent composition of basal area by species, percent composition of basal area by species groups (fir-dominant, yellow pine mixed-conifer, and lodgepole pine), total stems per hectare, total basal area, quadratic mean diameter, percent cover of shrub, canopy cover), fire severity of most recent fire, time since last fire, and number of fires. Vegetation composition and structure variables were from the 2002 data, and fire history variables were from fires that burned between 2002 and 2021 (see Fig. S1 for fire history of plots). A significance threshold of 0.05 was applied to evaluate all splits.

Results

Fire history and carbon change over 20 years in Illilouette (plot-based)

Between 2002 and 2021/2022, ~ 22% of Illilouette basin burned in nine fires (> 1 ha in size). Only two fires overlapped in this time period: the 2004 Meadow Fire and the 2017 Empire Fire. Of the 37 plots that were established in 2002 and remeasured 20 years later, 23 plots experienced no fires, six plots burned one time (in the Meadow Fire), and eight plots burned two times (in the Meadow Fire then Empire Fire) between 2002 and 2021/2022. Despite a low sample size of plots that burned, the distribution of fire severities experienced among the plots was fairly even: of the six plots that burned once, three burned at low, two burned at moderate, and one burned at high severity; of the eight plots that burned twice, three burned at moderate and five burned at high severity.

A majority of plots measured in 2002 were fir-dominated (white or red fir; 62% of plots); just over half of these plots remained fir-dominated in 2021/2022. Most of the other plots that were fir-dominated in 2002 had converted to shrubland, and a single plot changed in composition to yellow pine mixed-conifer-dominant (YPMC; Jeffrey and sugar pine) by 2021/2022 (Table 1). YPMC was the next most common vegetation type measured in 2002 (24% of plots), with all but one of these plots remaining as YPMC; this one plot had changed to fir-dominated by 2021/2022. The remainder of the plots (14%) were lodgepole pine-dominated and non-forest (herbaceous or shrub-dominated) vegetation types in 2002, with all but one plot remaining as such in 2021/2022. The single plot that changed was dominated by herbaceous plants in 2002 but had substantial lodgepole pine ingrowth by 2021/2022.

Between 2002 and 2021/2022, the total aboveground carbon density measured in plots decreased by a mean of 54.4 MgC ha^{-1}, from 244 (SD = 156) to 190 (SD = 151) MgC ha^{-1} (Fig. 2). Carbon dynamics at the individual plot level were highly variable, with total aboveground carbon increasing in 19 plots and decreasing in 18 plots, and the overall average decrease

Table 1 Vegetation type of Illilouette plots sampled in 2002 and 2021/2022

N plots	Vegetation type 2002	Vegetation type 2021/2022
13	White/red fir-dominated	White/red fir-dominated
9	White/red fir-dominated	Shrubfield
1	White/red fir-dominated	YPMC
8	YPMC	YPMC
1	YPMC	White/red fir-dominated
2	Lodgepole pine-dominated	Lodgepole pine or YPMC
2	Dead overstory, herbaceous understory	Lodgepole pine
1	Shrubfield	Shrubfield

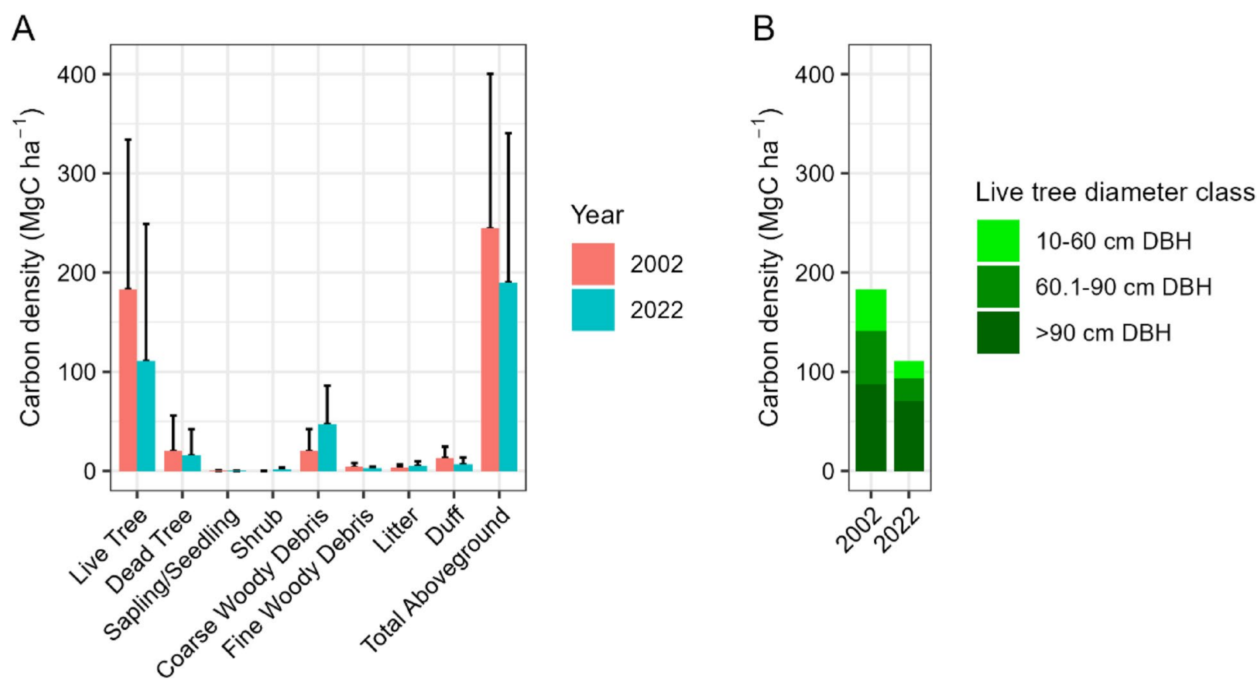

Fig. 2 Changes in plot-level carbon density in Illilouette. **A** Mean carbon density across 37 plots in 2002 and 2021/2022 by carbon pool. Error bar represents 1 standard deviation. **B** Changes in the live tree carbon pool by tree diameter class

in carbon stocks was driven by large losses in live tree carbon in about a third of the plots (Fig. S1). The only carbon pool that increased substantially was downed coarse woody debris, by a mean of 27 MgC ha^{-1}. All other pools changed minimally over the 20 years. In both years, the live tree (trees > 10 cm DBH) pool consisted of the majority of the total aboveground carbon (~ 75% in 2002 and 58% in 2021/2022), particularly in large trees (trees > 60 cm DBH, 58% in 2002 and 49% in 2021/2022; Fig. 2B). A smaller proportion of carbon loss was observed in the largest trees (trees > 90 cm DBH, 20% carbon loss) compared to smaller diameter trees (trees 10-90 cm DBH, ~ 57% carbon loss; Fig. 2B). The coarse dead woody pool, including both standing dead trees (snags) and downed coarse woody debris, made up a majority of the rest of the carbon (combined proportion of 16% in 2002 and 33% in 2021/2022), followed by the fine dead surface fuel pools (duff, litter, and fine woody debris; combined proportion of 8% in both 2002 and 2021/2022). The sapling/seedling and shrub pools consisted of a very minor portion of total aboveground carbon (combined proportion of 0.08% in 2002 and 0.8% in 2021/2022).

On average, aboveground carbon density increased in the plots with no fire (mean of 22.0 MgC ha^{-1}), moderately decreased in plots with one fire (mean of − 55.7 MgC ha^{-1}) or that burned most recently at low-moderate severity (mean of − 86.2 MgC ha^{-1}), and

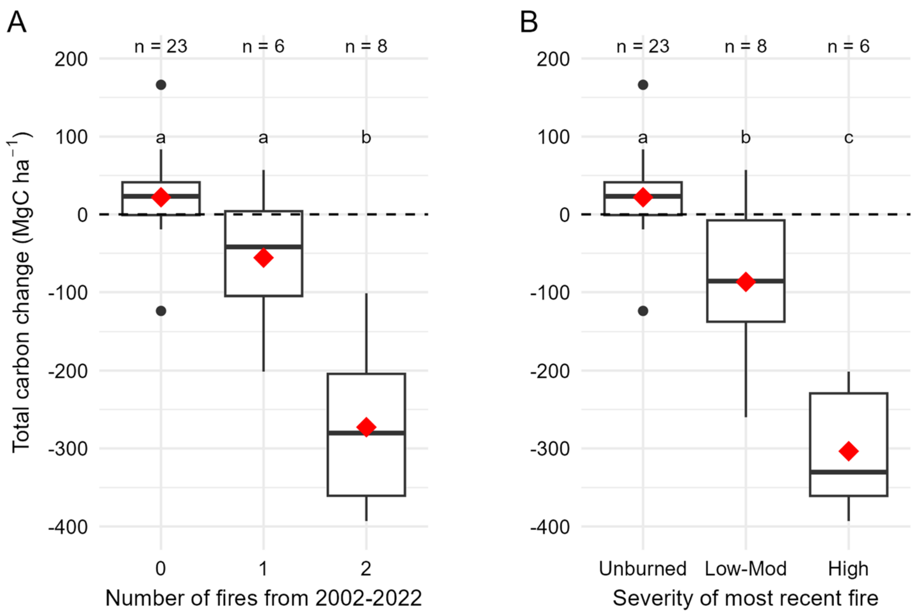

Fig. 3 Relationships between the change in total aboveground carbon and fire history (frequency and severity) at the plot level in Illilouette. **A** Fire frequency is represented as the number of fires experienced between measurement years (2002–2022) at plots. **B** Fire severity is represented as the severity of the most recent fire. Letters above boxplots indicate significant differences in mean carbon change between groups from a bootstrapping analysis

substantially decreased in plots with two fires (mean of − 272.7 MgC ha^{-1}) or that burned most recently at high severity (mean of − 304.2 MgC ha^{-1}; Fig. 3). The ~ 10% increase in carbon density in unburned plots was significant (assessed by non-overlapping 95% confidence intervals) compared to almost all plots that burned, except for the group of plots that burned one time (Fig. 3A). This group—plots that burned one time in the 20-year period—was a small sample size (6 plots) and included two plots that had an increase in carbon density; these factors likely led to this group not quite having a significant difference in mean carbon change compared to unburned plots. Mean carbon change was also significantly different between the two fire severity groups (95% confidence intervals were − 151.9 to − 21.6 and − 365.8 to − 244.4 2 MgC ha^{-1} for low-moderate and high severity groups, respectively).

From the conditional inference tree analysis, fire severity of the most recent fire was indicated as the only significant predictor of change in aboveground carbon stocks (Fig. S2). In the analysis, the two terminal nodes split at a CBI of 1.917, which is a threshold that is at the high end of the moderate severity class. For plots that experienced a CBI < 1.917, the predicted change in carbon density was slightly positive (9.3 MgC ha^{-1}); for plots that experienced a CBI > 1.917, the predicted change in carbon density was substantially negative (− 285.3 MgC ha^{-1}).

Similar to total aboveground carbon stock trends, carbon density decreased with increasing severity in the live tree, fine woody debris, litter, and duff pools (Fig. S3). Carbon density changes in dead standing tree, sapling, and coarse woody debris pools were not different among plots burned at different severities. The shrub pool was the only aboveground pool that increased in carbon density with increasing severity.

Fire history and carbon change over 37 years across all three basins (remotely sensed)

Between 1985 and 2021 (the temporal scale of the remotely sensed carbon data), 64% of Illilouette burned in 29 fires (> 1 ha) with areas experiencing from 0 to 4 overlapping fires, and 86% of San Joaquin burned in seven fires (> 1 ha) with areas experiencing from 0 to 3 overlapping fires. In San Joaquin, the 2020 Creek Fire and 2018 Lions Fire made up a majority of the burned area across the basin, though these two fires only overlapped slightly at their edges. No fires > 1 ha in size burned in Badger during this time period.

At the basin-scale, on average, live carbon density decreased and dead carbon density increased or remained constant from 1985 to 2021 (Table 2, Fig. S4 and S5). Live carbon stocks decreased by a mean of 48% in San Joaquin (from 148.7 to 77.9 MgC ha^{-1}), by 25% in Illilouette (from 147.6 to 111.1 MgC ha^{-1}), and by 9% in Badger (from 233.6 to 213.6 MgC ha^{-1}; Table 2). Dead

Table 2 Mean (± 1 standard deviation) aboveground carbon density (MgC ha^{-1}) and mean percent change of live and dead carbon stocks from remotely sensed data for three basins with different fire histories. Badger is largely unburned, Illilouette had fire reintroduced more than 50 years ago, and San Joaquin had fire reintroduced 20 years ago

	Badger			Illilouette			San Joaquin		
	1985	2021	% change	1985	2021	% change	1985	2021	% change
Live carbon	233.6 (±90.1)	213.6 (±69.2)	−9%	147.6 (±89.9)	111.1 (±64.2)	−25%	148.7 (±88.8)	77.9 (±68.5)	−48%
Dead carbon	205.6 (±142.1)	211.7 (±138.1)	+3%	81.7 (±57.3)	81.8 (±54.0)	< 1%	92.4 (±72.0)	121.8 (±81.9)	+32%

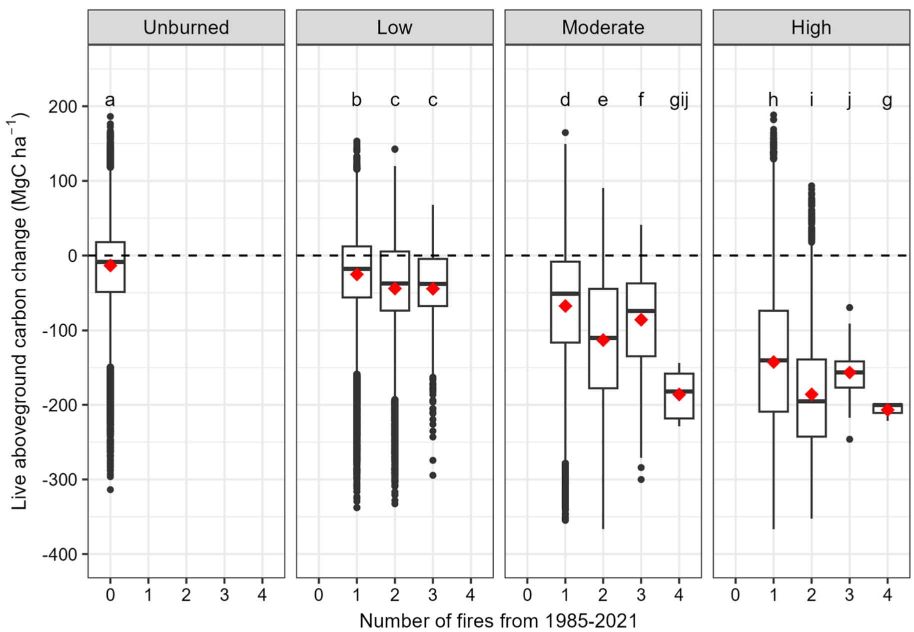

Fig. 4 Relationships between the change in live aboveground carbon density, fire frequency, and fire severity across three basins, using remotely sensed CECS data. Red squares are the mean values for each group. Letters above boxes indicate significantly different means in live carbon change among groups from a bootstrapping analysis

carbon stocks increased by a mean of 32% in San Joaquin (from 92.4 to 121.8 MgC ha^{-1}), stayed the same in Illilouette (~81.7 MgC ha^{-1}), and increased by a mean of 3% in Badger (from 205.6 to 211.7 MgC ha^{-1}; Table 2).

Mean live carbon density decreased both with increasing fire frequency and increasing fire severity, but fire severity exerted a stronger influence on carbon change than fire frequency (Fig. 4). On average, live carbon decreased by 5.4, 29.6, 75.1, and 147.4 MgC ha^{-1} in unburned, low-, moderate-, and high-severity pixels, respectively. Means for each frequency-severity group were significantly different from most other groups, as evidenced by non-overlapping 95% confidence intervals from the bootstrapping analysis (Fig. 4).

There was no clear trend of carbon density change in dead pools in relation to fire history when data from all three basins were combined (Fig. S6). Between Illilouette and San Joaquin basins, however, the pattern of dead carbon change was different in relation to fire severity: on average, dead carbon changed little among the fire severity groups (means ranging from a decrease of 4.5 to an increase of 1.2 MgC ha^{-1}) in Illilouette, while dead carbon increased greatly as fire severity increased, particularly in pixels that burned at high severity (mean of an increase of 5.4 vs. 62.2 MgC ha^{-1} in unburned vs. high-severity pixels) in San Joaquin.

Resistance of carbon to future disturbance across all three basins (plot-based)

There was a mean of 223 (SD = 187) MgC ha^{-1} in the 129 plots that were measured in Badger, Illilouette, and San Joaquin in 2021 and 2022. The carbon pool proportions differed among vegetation types, but of forested plots, the live tree pool was the largest pool, followed by the coarse dead pool (standing snags and coarse woody debris), with the other pools (saplings, shrubs, fine woody debris, litter, and duff) constituting the remainder. On average, the plots in Badger had greater absolute aboveground carbon and a lower proportion of carbon in large, live trees (trees > 60 cm DBH, one measure of carbon stability) than plots in Illilouette and San Joaquin, which were more similar (Fig. S7).

In fir-dominated and YPMC forests, there was a higher proportion of stable carbon in plots that experienced low or moderate severity fire most recently compared to unburned plots (50–53% vs. 37% resistant carbon in fir and 38–55% vs. 27% resistant carbon in YPMC; Fig. 5). From the bootstrapping analysis, the only significant difference was between unburned, low, or moderate severity groups and the high severity group. In lodgepole pine forests, there was a higher proportion of resistant carbon in

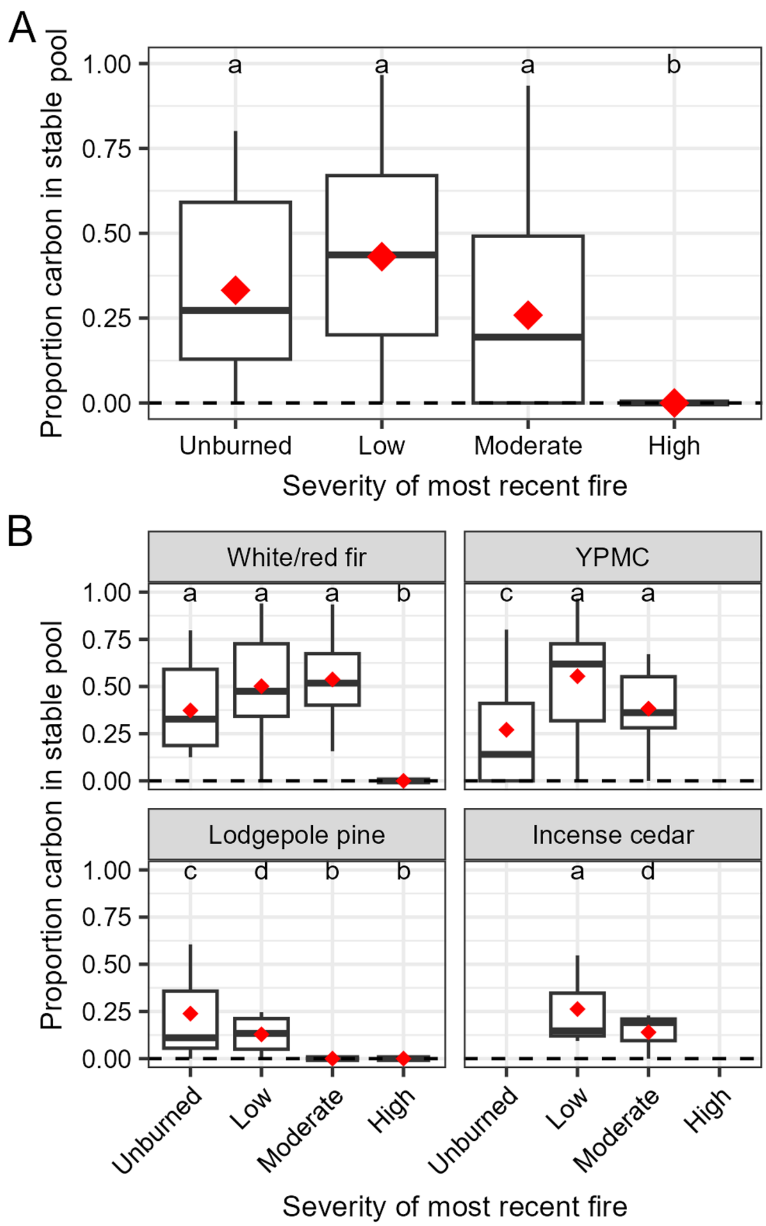

Fig. 5 Plot-level relationship between carbon stability and fire severity at plots measured in three basins in 2021 and 2022. **A** Plots aggregated by severity. **B** Plots split by forest type. Carbon stability was defined as the proportion of total aboveground carbon that was in a resistant pool (live trees > 60 cm DBH). Red squares are mean values for each severity group, and letters above boxes indicate significantly different means in carbon stability among groups from a bootstrapping analysis. Only a few plots in the San Joaquin basin were dominated by incense cedar

unburned plots (24%) compared to low (13%) and moderate (0%) plots; the only significant difference in mean proportion stable carbon was between the unburned and low-severity groups compared to the moderate- and high-severity groups. In all forest types, plots that burned at high severity resulted in complete mortality of live overstory trees, including those of large diameter.

At the plot level, the relative SDI in 2021 and 2022—the other metric we used to assess carbon stability—was on average 39–69% of maximum SDI, depending on forest type (Fig. S8); this corresponds with the full site occupancy (35–60%) and imminent mortality (>60%) zones. Pine mixed-conifer (YPMC and lodgepole pine) plots were on the lowest end of this range (mean of 39% relative SDI), mesic mixed-conifer (fir-dominated) plots were slightly higher with a mean of 46% relative SDI, and xeric mixed-conifer (incense-cedar, only in San Joaquin basin) plots had the highest relative SDI at 69% on average. There were no substantial differences in relative SDI among the three study areas at the plot level.

Discussion

After 50 years of wildfire use in the Illilouette Creek basin of Yosemite National Park (and approximately 20 years in the adjacent San Joaquin basin of Ansel Adams Wilderness), aboveground carbon stocks continue to decrease, and the ecosystem is not yet in a steady state of vegetation and carbon stock change (Crompton et al. 2022). These continued decreases could also indicate that both basins are still above their respective carbon carrying capacities, as they correct for both the long periods of prior fire exclusion and the current climate (Hurteau et al. 2024). In Illilouette Creek Basin, this decrease in carbon stocks was largely driven by patches of forest cover being converted to shrublands and meadows with successive fires; this observation is consistent with previous research in the basin that showed that conifer forests decreased by 24% while shrub and meadow area increased by 35% to 199%, respectively, from 1969 to 2012 (Boisramé et al. 2017a). It is possible that these transitions to shrub-dominated and meadow areas are a "reclaiming" of non-forest vegetation that likely existed in these landscapes under intact fire regimes (Hessburg et al. 2021). Similar fire-caused forest loss, and associated carbon losses, were observed at the statewide level across California from 2001 to 2010 (Gonzalez et al. 2015). Although it should be noted that these losses observed across California tended to be concentrated in large patches (also see Cova et al. 2023), which could hardly be considered a reclaiming of non-forest vegetation communities given the scale of these transitions relative to the natural range of variability (Miller and Safford 2017; Safford and Stevens 2017). Overall, restoration of dry mixed-conifer forests solely by reintroducing lightning-ignited fire appears to be a slow process—models in similar vegetation types in the Sierra Nevada have shown that areas may take 100 to 200 years to recover to a steady state after 100 years of fire suppression (Miller and Urban 2000; Boisramé et al. 2017a; Crompton et al. 2022).

In the absence of restorative fire, as was the case in Badger basin, aboveground carbon stocks were considerably higher than the other two basins, despite the modest decrease in live carbon from 1985 to 2021 (−9% MgC ha^{-1}). This finding is not surprising given the continued removal of fire from fire-dependent forests. However, what is somewhat surprising is that this area has remained largely unburned despite the elevated recent wildfire activity in Sierra Nevada forests (Safford et al. 2022, Cova et al. 2023). This is surprising because, on average, Badger basin has incredibly high levels of dead carbon (>200 MgC ha^{-1}). This amount of dead carbon, i.e., fuel, combined with the regional increases in fuel aridity observed recently, can lead to extreme fire behavior (Goodwin et al. 2021) and ultimately massive contiguous patches of stand-replacing fire effects (Stephens et al. 2022). Because of this vulnerability to wildfire, Badger basin is at high risk of losing much of its stored aboveground carbon very quickly.

In our study, fire severity appeared to be a stronger driver of aboveground carbon change than fire frequency. Areas that burned at high severity experienced the greatest declines in aboveground carbon, regardless of the number of fires that the area had experienced, while areas that burned at low and moderate severity had moderate decreases in aboveground carbon. Due to the nature of our data, we were not able to explicitly examine the relationships between fire severities in successive burns and were not able to include the metric of time since last fire; there likely is a relationship between carbon dynamics and the number of fires, time since fire, and fire severity, but this warrants further investigation. Other studies in the Sierra Nevada have shown that forested areas that burn at high severity are likely to reburn at high severity due to high amounts of large dead wood and shrubs that are created after the first fire (Coppoletta et al. 2016; Lydersen et al. 2019). Much of the stored carbon in these areas would be released during combustion in each successive burn, but eventually the carbon dynamics would stabilize—albeit in a lower state of total carbon. In basins that have had wildfires for several decades—including in one of our study basins, Illilouette—successive burns have been shown to be self-limiting by restricting the extent and severity of subsequent fires (Collins et al. 2009; Parks et al. 2014). In Illilouette, 9 years since last fire was identified as the threshold that limited the extent of a subsequent fire; after 9 years, fuel had accumulated enough in previously burned areas to carry fire again (Collins et al. 2009). While the conversion of forests to shrublands and grasslands due to high-severity fire is concerning when it occurs at broad spatial scales (Fornwalt et al. 2016; Coop et al. 2020), the extents of the high-severity patches in our study areas were relatively small (Collins and Stephens 2010) and thus not concerning.

Despite the decreases we observed in aboveground carbon over the past 20 and 37 years (at the plot-level and landscape-level, respectively), there is still a large amount of carbon being stored in these three basins today: a mean of 223 (SD = 187) MgC ha^{-1} from the plot data and 253 (SD = 169) MgC ha^{-1} from the remotely sensed CECS data. These findings generally support a landscape-scale study of fire impacts on tree carbon density in Yosemite and Sequoia and Kings Canyon National Parks, which found that low- and moderate-severity fire had little to no effect on total tree carbon density, though we were primarily interested in the effects of fire on the *change* in carbon stocks rather than total carbon stocks (Lutz et al. 2017). Lutz et al. (2017) concluded that a mixed-severity fire regime in these National Parks does not result in much overall loss of carbon stocks because mortality of large-diameter trees is low. This may be true at the even broader landscape scale than the plot and basin level scales in our study, but we found that plots that burned at high severity resulted in complete mortality of live trees, including large-diameter ones.

In terms of carbon stability—the ability of carbon stocks to resist or recover from future disturbances—we found that low- and moderate-severity fire may actually increase the stability of carbon in most of the forest types present in the Sierra Nevada relative to forests that remain unburned, as measured by total stable carbon (i.e., the amount of carbon in live, large-diameter trees). These types of fires preferentially consume more fire-sensitive carbon pools like smaller diameter trees, shrubs, and fine dead surface fuels, while sparing the more fire-resistant larger diameter trees. However, it is worth noting that the large variation in fire effects observed in the moderate severity class results in a range of fuel conditions, which, when reburned, can limit or exacerbate the severity of subsequent fires (Collins et al. 2018).

Large-diameter trees store the greatest amount of aboveground carbon in forested systems, so retaining them across the landscape and increasing their resistance to future disturbances by removing surface and ladder fuels and competition is integral to forest management objectives of maintaining stable carbon stocks. In our study area, wildfire use appears to emulate other fuel treatments, such as prescribed fire and mechanical treatment, which have been shown to create forest conditions that are resistant to future disturbances (Stephens et al. 2024). Still, many of the forests in the Sierra Nevada are much denser today than they would have been historically, as measured by relative SDI, indicating lower carbon stability. Historical mixed-conifer stands are estimated to have had a mean relative SDI of 23–28% (North et al. 2022); only about a quarter of our plots measured in 2021 and 2022 had a relative SDI < 28%.

The discrepancy in the aboveground carbon density estimates between the remotely sensed CECS data and plot data (Fig. S9) may be due to several factors, which are common with any comparisons between remotely sensed and plot-based data (Holland et al. 2019). This includes mismatches in spatial scales (500 m^2 area for plots vs. 900 m^2 area for CECS data) and locations (plot centers were precisely geolocated to < 1 m precision vs. CECS plot locations having a precision of at least 20 m), as well as differences in the methods used for biomass calculations and vegetation type characterization. The carbon estimates at the plot level are summed from measurements of individual trees—so we know the species, DBH, and height with high confidence—while the estimates at the landscape level do not have this high a specificity, but characterize the biomass more broadly. This difference in resolution may in part explain the saturation observed in the remotely sensed carbon density datasets: the live aboveground carbon estimates in the CECS data saturate around 200–300 MgC ha^{-1}, while several plots exceeded 300 MgC ha^{-1} (Fig. S9A). These high carbon plots tended to have a few very large diameter trees (> 100 cm DBH) that made up a majority of the carbon on the plot; the CECS data may not characterize these very large diameter trees well. Additionally, while other studies have found high uncertainty in plot-level carbon estimates of very large trees because they exceed the range of existing allometric equations (e.g., Lutz et al. 2017), our plot-based carbon estimates are less uncertain because the 2024 NSVB equations that we used included allometric equations for larger trees (Westfall et al. 2024). Still, while the absolute values of aboveground carbon at both the plot and landscapelevels may not be directly comparable to other studies due to differences in equations, the temporal changes in carbon—and how fire induces these changes—are valuable and comparable between the two datasets. The temporal changes observed in the carbon change analysis from 2002 to 2021 were similar among the two datasets, with a strong correlation (an R^2 value of 0.66), though the remotely sensed CECS data does slightly underestimate the change in live carbon compared to the plot data (Fig. S10).

From our study, with continued wildfire use in mixed-conifer forests, we can expect to see further declines in tree density, and thus aboveground carbon stocks, until the landscape reaches a quasi-steady state of vegetation type change (i.e., a shifting-mosaic steady state; Bormann and Likens 2012). Based on modeling efforts, it may take another 50 or more years for these forests to reach such a state given the extent, frequency, and severity of fires in the past 50 years, and with no additional management actions taken to restore the landscape (Miller and Urban 2000; Boisramé et al. 2017a; Rakhmatulina et al. 2021; Crompton et al. 2022). In general, the relatively small decreases in aboveground carbon stocks (particularly with low- and moderate-severity fire) are outweighed by the many benefits that fire has on these landscapes, including creating greater landscape heterogeneity and resilience and decreasing drought-induced tree mortality (Boisramé et al. 2017a; Stephens et al. 2021, Zhu et al. 2025). Other fuel treatments, such as mechanical treatment or prescribed burning, have been shown to similarly reduce carbon stocks in the short-term while increasing carbon resistance in the long-term (Hurteau and North 2010; Stephens et al. 2012), but in areas where these types of treatments are infeasible or are too expensive, wildfire use may be the appropriate tool (North et al. 2015). Restoration of mixed-conifer forests solely by reintroducing a characteristic fire regime is a slow process, but increasing wildfire use management in areas where it is feasible should be a tool that managers consider to address our growing forest restoration and resilience needs.

Conclusions

In this study, we investigated how reintroducing a characteristic fire regime via wildfire impacted aboveground carbon dynamics at two spatial and temporal scales in Sierra Nevada forests. At both the plot- and basin-levels, live carbon stocks decreased while dead carbon stocks remained the same or slightly increased in areas managed with wildfire over the past 20–50 years. Fire severity exerted a stronger influence on carbon change than fire frequency; areas that burned at high-severity resulted in the largest losses in carbon, regardless of the number of fires that the area had experienced. The extents of these high-severity areas were relatively small (Collins and Stephens 2010) and tended to be in patches of high conifer cover that converted to shrublands or grasslands. While vegetation type conversion is a concern at broader landscapes where large patches or even watersheds burn at uncharacteristically high-severity (Fornwalt et al. 2016; Coop et al. 2020), many of the patches that have burned severely in our study area were small, consistent with the natural range of variation in these forests, and may actually be considered "reclaimed" by shrubs, grass, and wetlands, i.e., they were historically maintained as non-forested by an intact fire regime.

In the two basins that have had fire reintroduced, the increased heterogeneity in vegetation types, species composition, and forest structure creates conditions that are more resistant and resilient to future disturbances (Koontz et al. 2020, Steel et al. 2025). In a nearby basin that has remained largely unburned since at least 1930 (when fire records began being documented), the vegetation is dominated by continuous conifer cover and high tree density; while this basin stores high amounts of carbon at the moment (see Fig. S4), it has a higher risk of burning severely (Miller et al. 2009; Hurteau et al. 2011; Steel et al. 2015) in which case that stored carbon would be lost quickly. Overall, our study indicates that increasing the use of wildfire in our fire-deprived forests comes at some costs in terms of carbon stocks, at least in the short term, but the carbon stocks that survive the fires are more likely to persist even through future disturbances.

Abbreviations

RAWS	Remote automatic weather station
DBH	Diameter at breast height
HLCB	Height to live crown base
SDI	Stand density index
HUC12	Hydrologic Unit Code subwatershed
CECS	Center for Ecosystem Climate Solutions
CBI	Composite Burn Index
YPMC	Yellow pine-mixed conifer

Supplementary Information

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Supplementary Material 1.

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Authors' contributions

BMC, ZLS, JLB, and SLS conceived of the study and secured funding. KAN completed the analysis and prepared the first draft. ZLS curated the fire frequency and severity datasets, and MLG curated the remotely-sensed CECS datasets. JLB developed the BerkeleyForestsAnalytics package and assisted with carbon calculations. All authors read, edited, and approved the final manuscript.

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Data availability

The datasets used and/or analyzed during the current study are available from the corresponding author on reasonable request. The Center for Ecosystems Climate Solutions datasets are available to the public here: https://drive.google.com/drive/folders/1838VouV-Y7MIZY3A-5ZxRiFKIR1EHP7Q.

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Competing interests

The authors declare no competing interests.

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