Dynamics of forest carbon stocks, fires, and harvest under changing climatic conditions in the U.S. during the 20th century

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Abstract

Wildfires and land use play a central role in the long-term carbon (C) dynamics of forested ecosystems of the United States. Important processes include fire suppression during the 20^{th} century and a recent increase in fire activity, partly due to climatic extreme events. Although the historical fire narrative in the U.S. is well understood, its links to changes in forest biomass, resource use and consumption remain understudied. We reconstruct long-term trends in biomass burned, and biomass use by humans, integrating various data sources at different scales (national scale 1926-2017, regional level 1941-2017). We investigate the linear correlation of wildfires and forest biomass C stocks in comparison to forest uses, i.e., the extraction of woody biomass and forest grazing, and potential net primary production (NPP_{pot}). During the 20^{th} century, the reduction in burned biomass and increase in NPP_{pot} coincide with forest regrowth in the Eastern U.S., allowing for increased wood harvest. Only in the Western U.S. these dynamics are less pronounced, indicating that forest fires and biomass harvest were less decisive factors for forest C stock developments in this section. In recent decades, linkages between forest change and wildfires are less straightforward in all regions, indicating that past fire suppression levels are less efficient in present-day forests. Instead, the reduction of harvest in 3 of 4 regions was correlated to stock increase. We conclude that under changing climate, present-day fire and forest management practices might be unsuitable for ensuring both additional forest C sink potential and expanded wood use.

Introduction

Wildfires have been a key feature of the Earth system for millions of years, affecting a large variety of biogeochemical processes, ecosystems and organisms (Bowman et al., 2020; Pausas & Keeley, 2009). Fire is caused either naturally (primarily by lightning) or by humans. In most populated places humans have been the main cause of ignitions for thousands of years (Pyne, 1995) and have fundamentally altered natural fire regimes on Earth (Bowman et al., 2011; Pyne, 1991) with land use being an integral objective and driver for the emergence of human-influenced fire regimes (Bowman et al., 2020; O'Connor et al., 2011). The interconnections between burned area, climate, human, and other biotic processes is highly complex (Teckentrup et al., 2019). Indeed, many applications of landscape fire are beneficial to human wellbeing, or even necessary for regional subsistence and survival. In the context of anthropogenic climate change, fire activity and its far-reaching impact on atmospheric and biotic processes have gained increased attention (Keywood et al., 2013; Moritz et al., 2012; Pausas & Ribeiro, 2017). Still, the extent, as well as the quantitative relationships between land use, biomass use, and fire impacts are related to large uncertainties (Bowman et al., 2020; Chuvieco et al., 2019; Erb et al., 2017; Lauk & Erb, 2016; Pechony & Shindell, 2010; van Marle et al., 2017). With changing climate and enhanced concentration of atmospheric CO_2, temperatures and droughts are expected to rise, possibly triggering more frequent fire occurrence and higher fire severity in the future (Flannigan et al., 2009; Xu et al., 2020). Changing climate has diverse impacts on vegetation, including increased biomass productivity and thus more fuel available for burning (Teckentrup et al., 2019). Whether or not fires will increase in a given place may largely depend on climate, combinations of fire characteristics, geographic location, and human activities in these locations (Archibald et al., 2013).

In the light of projected changes in the size, location and severity of wildfires, on the one hand, and forest C sequestration capacities, on the other hand, it is vitally important to better understand the interlinkages between changing socio-metabolic activities and fire patterns. Land use – fire interrelations can consist of fire suppression for management and protection of forests, crops, livestock, people, and infrastructure, or fire ignition by humans, either unintended or for specific purposes, including biome conversion, burning of agricultural residues, land-use change or shifting cultivation (Chuvieco et al., 2019; Malamud et al., 2005; Marlon et al., 2013; Parisien et al., 2016). In both cases, consequences for ecosystem functioning e.g., biodiversity, ecosystem service provision or changes in carbon cycles, occur (Haugo et al., 2010; Hurteau & Brooks, 2011; Kelly et al., 2020; Parks et al., 2015). Fire regimes describe the pattern of fire occurrence in a specific place, including the size, intensity, frequency, ignition source, fuel types affected, and seasonal timing of landscape conflagrations within geographic units that may be local, regional, ecosystem types, or species communities, and time frames that are typically annual but which may change over the course of decades, centuries, or millennia (Pyne, 2019, Conedera et al., 2009; Krebs et al., 2010). Our study aims to increase knowledge about the annual amount of biomass burned by wildfires on national and regional scales in forests and woodlands of the United States of America in connection to human land use, and forest biomass development to help improve forest management options and models of future fires, which can then be used for climate change mitigation and adaptation strategies (Ford et al., 2021; Rogers et al., 2020)

Due to the well-researched land-use history and the abundance of detailed and long-term data sources for wildfires, and forest use, as well as forest inventories, the U.S. offers a unique opportunity to investigate the interrelation between wildfires, socio-metabolic activities connected to land use, and carbon dynamics in forested ecosystems. Land use and the management of forest ecosystems in the U.S. are inextricably linked to landscape fires (Pyne, 1982). Fire has been used by Native Americans for thousands of years to transform ecosystems, for hunting, habitat enhancement, and small-scale agriculture (Bowman et al., 2011; Frost & Sweeney, 2000; Vale, 2002). European settlers adopted and expanded native fire practices and reduced forests largely in favour of agricultural land between 1600 and 1900 (Courtwright, 2011; Gregg, 2010; Hessburg & Agee, 2003; Liebmann et al., 2016). Especially during the 18th and 19th centuries, large-scale land use changes due to agricultural expansion, as well as clear-cutting of forests, led to an increase in burned area, particularly in the Southern region of the United States (Fowler & Konopik, 2007; Pyne, 1997). In the early 20th century, large-scale fire suppression was introduced to decrease 'catastrophic' wildfires, often connected to timber-harvest activities like slash and debris burning, railways spark ignition, and other factors. Together with changing forest management, afforestation, natural resource conservation efforts, and the modernization of local subsistence-based economies (Fedkiw, 1989; Gregg, 2010), fire prevention and suppression, as well as prescribed burning, drastically reduced fire activity in the U.S. after the 1930s (MacCleery, 1993; Steen, 2004). Consequently, since the early 20th century, U.S. forests have been in a phase of recovery in terms of area and C-density from depletions in the past (Magerl et al., 2019), a process denoted as forest transition (Mather, 1992): During industrialization, forest productivity to provide construction materials to the rapidly expanding economy superseded subsistence farming (Gregg, 2010). Thus, forest recovery was driven mainly by commercial timber forests in the East, while in the West less pronounced regrowth occurred in more diverse ecosystems, including newly reserved national forests and low-productive shrub- or woodlands (Magerl et al., 2019; Steen, 2004).

Gingrich et al., (2019, 2022) showed that the causes and drivers of forest transitions are complex and include, among others, agricultural abandonment, trade of wood, economic development, government policy adjustment, and changes in fuelwood substitution, and vary by country and region. Although the impact of naturally or human-induced fires on forest biomass, especially at local scales, is well documented (Frost & Sweeney, 2000; Wilson et al., 2021), the role of fire management in the context of forest transition research is surprisingly limited (Iriarte-Goñi & Ayuda, 2018) and has not yet been empirically investigated for the US. Many studies have analysed different aspects of U.S. wildfires, including the influence of humans on contemporary fire regimes or recent increases in size and severity of wildfires often in connection to climate change, often with a particular focus on the West (Abatzoglou & Williams, 2016; Barbero et al., 2015; Dennison et al., 2014; Gedalof et al., 2005; Malamud et al., 2005; Mitchell et al., 2014; Singleton et al., 2019). Other works investigated trends over longer time-periods, including burned area, emissions or other processes connected to changing fire activity, for example fuel accumulation due to fire suppression policies (Wuerthner, 2006). While satellites provide comprehensive data from the 1980s onwards for the continental U.S., long-term studies have mostly focused on the regional to local scales (Grala & Cooke, 2010; Guyette et al., 2003; Syphard & Keeley, 2016), with a particular focus on the American West (Higuera et al., 2015; Littell et al., 2009; Marlon et al., 2012). Only a few studies have quantified long-term changes in biomass burned for the total continental U.S. on the national level or on sub-national scales (Houghton et al., 2000; Leenhouts, 1998). The interlinkage between past fire management and land-use activities is often mentioned implicitly, however, rarely explored explicitly or quantified. National, or comprehensive multi-regional, long-term research linking land-use, socio-metabolic activities, and wildfire trajectories are still rare (Balch et al., 2017; Hawbaker et al., 2013; O'Connor et al., 2011).

This study makes those linkages and quantifies forest biomass burned by wildfires in the U.S. across nearly a century of significant change. We analyse the relative role of developments in forest fires for the C-stock dynamics in comparison to harvested biomass (wood and forest grazing), as well as climatic conditions, at the national level from 1926, and at the regional and state level from 1940 to 2017. We synthesise historical statistics and contemporary satellite data on forest wildfire area, and derive fuel loading and combustion completeness factors from field measurements and a fire-enabled dynamic global vegetation model (DGVM). We address the following questions: How did wildfires in comparison to biomass extraction contribute to observed changes in biomass C stocks in U.S. forests? In which regions and on which timeframes were wildfires particularly significant?

1.

Methods and data

(a)

Input data

We reconstructed burned area for the continental U.S. by synthesizing and integrating information from historical agency reports and contemporary satellite data, adinge burned area dataset and field measurements. We collected and compared the most complete and widely used data-sources for burned area available for the U.S. (Table 1). Burned area data were available for different spatial levels and timescales: on the national scale from 1926-1984 (United States Bureau of the Census, 1975; 1984), on the scale of broad geographic regions (North, South, Rocky Mountains, and Pacific Coast) from 1938-1979 by Ciesla & Mason, (2005), on the state level from the United States Forest Service's "forest fire statistics" for several years between 1941-1964, from the "national forest fire reports" and "annual fire report for the national forests" for 1971-1984. For the period 1985-2017, remote sensing data were available at high spatial resolution (Hawbaker et al., 2020).

USFS agency reports

Aggregated national level "forest fire" data by the United States Bureau of the Census (1975; 1984) for 1926-1984 are reported in the "Historical Statistics of the United States – Colonial times to 1970" and the "Statistical Abstract of the United States" which are both compiled from USFS agency reports. These original reports were available in digital format for the years 1941, 1942, 1945-1951, 1954-1956, 1958, 1964, and from 1971-1985 (Table 1). They report burned area on the state level for "federal", and "state & private" forests, but also a small share of ecosystems with sparse or no tree cover (wood-, shrub-, scrublands), called "other forests" or "other land inside national forest boundaries". Houghton et al. (2000) assumed in their study all these reported burned areas referred to forests. However, although other forests represent only a relatively small share of the reported total burned area, they also contain

Table 1: Data sources used to reconstruct continental U.S. burned area 1926-2017. Notes: shaded areas indicate data availability and completeness. Dark green = Reconstruction, data available for all years of decade. Light green = Reconstruction, data available for some years of decade. Blue = Comparison/Validation, Complete data

Source	Spatial level	Min. fire size (ha)	Type of data	Ownership	Land Cover Type	Protection Status	1926	1930s	1940s	1950s	1960s	1970s	1980s	1990s	2000s	2010s
United States Bureau of the Census (1975), (1985), 1926-1979	National	<0.4	Agency reports	Federal	Forest	protected										

Source	**Spatial level**	**Min. fire size (ha)**	**Type of data**	**Ownership**	**Land Cover Type**	**Protection Status**	**1920s**	**1930s**	**1940s**	**1950s**	**1960s**	**1970s**	**1980s**	**1990s**	**2000s**	**2010s**
					Other forest (assume wood/shrubland)	protected										
				State and private	Forest	protected										
						unprotected										

Source	Spatial level	Min. fire size (ha)	Type of data	Ownership	Land Cover Type	Protection Status	1920s	1930s	1940s	1950s	1960s	1970s	1980s	1990s	2000s	2010s
United States Forest Service, Forest Fire Statistics (1941, 1942, 1945-1951, 1954-1956, 1958, 1964)**, Annual Fire Reports for the National Forests** (1960-1966, 1969)**, National Forest Fire Report** (1971-1985)	State	<0.4	Agency reports	Federal	Forest	protected										

Source	Spatial level	Min. fire size (ha)	Type of data	Ownership	Land Cover Type	Protection Status	1920s	1930s	1940s	1950s	1960s	1970s	1980s	1990s	2000s	2010s
						unprotected										
					Other land inside national forest boundaries (assume wood/shrubland)	protected										
				State and private	Forest	protected										
						unprotected										
					Other forest (assume wood/shrubland)	protected										
					Unknown/Other land (assume wood/shrubland)	unprotected										
Ciesla & Mason (2005), 1938-1979	Regional	<0.4	Agency reports	Aggregated	Forest	not specified										

Source	Spatial level	Min. fire size (ha)	Type of data	Ownership	Land Cover Type	Protection Status	1920s	1930s	1940s	1950s	1960s	1970s	1980s	1990s	2000s	2010s
USGS Landsat Burned Area (Hawbaker et al., 2020); 1985-2017	Spatially explicit data (30m) intersected with state and ownership boundaries		Satellite data		Forest (Deciduous, Evergreen, Mixed)	not specified										
					Shrub/scrubland											
					Herbaceous											

woody perennial vegetation storing carbon over more than one year. Thus, they also contribute to total C stocks in above- and belowground biomass, however, to a much lesser extent than productive forests. To account for this differences, we therefore consider three different land-cover types, based on the information in the USFS statistics and from USFS forest categories definitions (Oswalt et al., 2014): "State & Private" forests are almost entirely commercially used productive timber forests, "Federal forests" subsume commercial and non-commercial forests under federal administration, including national and reserved forests.

We assume "Other forests and other land inside national forest boundaries", hereafter simply termed "Other forests", to be areas with sparse woody vegetation (wood-, shrub- and scrubland). For state & private forests, the time series stopped after 1960, whereas statistics for burnt area in federal forests were reported for the whole time-period.

USGS Landsat Burned Area

The USGS Landsat Burned Area product is based on data acquired since 1985 by the Landsat 5, 7 and 8 satellites (Hawbaker et al., 2020), reported by land-cover type as defined by the U.S. Geological Survey (USGS) National Landcover Database (NLCD, Homer et al., 2012) for the period 1985-2017. Each of the Landsat satellites has a revisit cycle of 16 days, which is too long for tracking active fires. Hence, their burned area estimates are based on spectral indices known to be sensitive to recently burned vegetation. These indices are, however, subject to uncertainties due to other factors relating to vegetation dynamics or environmental conditions affecting the observation, for example shadowing by clouds or topography (e.g., Escuin et al., 2008). Cloud cover can further add to the uncertainty and completeness of the record of wildfire events (Hawbaker et al., 2017). Data are provided as date-based burned area classified rasters with a spatial resolution of 30 m and as yearly composites in vector format. The latter provides polygons of contiguous patches burned during a year with derived information, such as date of first detection, burn probability and the number of pixels of a burn patch belonging to each NLCD class. The burned forest and shrubland area used in this study was aggregated by year and state.

Reconstruction of burned area

We combined the national aggregated Bureau of the Census and Statistical abstract data for the years 1926-1984 with the Landsat data (1985-2017) to produce a continuous burnt area reconstruction for the years 1926-2017 on the national scale. Furthermore, we used the state-wise USFS agency reports 1941-1985 as a starting point for the sub-national reconstruction, since they represent the most detailed historic data in terms of land-cover types, ownership, use, and spatial disaggregation. However, as mentioned above, from 1960 onwards, burned area reports for protected and unprotected "state and private forests" were discontinued in the USFS statistics. Therefore, we used data by Ciesla and Mason (2005) to derive this "missing" portion. This publication provides burned area in "total forest ecosystems" on regional scales (aggregated Eastern regions Northern, Southern, and Western regions Rocky Mountain, and Pacific Coast states, see Supplementary Information, SI, Figure S 1) from 1938-1979. It contains only aggregated burned forest area and does not provide detailed information about the forest categories burned. Nevertheless, according to William Ciesla (personal communication), the numbers reported in these publication stem from the original USFS forest fire statistics as well. Hence, we aggregated the USFS state-wise data to the four regions and subtracted them from the data by Ciesla and Mason for the period 1938-1979 for the same regions, yielding the missing fraction of state & private forest area burnt. For the period 1985-2017, we used the spatially explicit Landsat data to split the burnt area into the same regions. Specifically, to match the forest categories reported in the historical agency reports 1941-1985 with the more recent satellite-based data from 1985-2017, we aggregated the respective "forest", and "shrubland" NLCD categories (see Table S 1) in the Landsat burned area. Next, we intersected these data with state-area boundaries and administrative units ("state & private", "federal") from the USGS using ArcGIS. This way we reconstructed regional burnt area for the years 1941-1979 and 1985-2017. The period 1980-1984 was excluded from the regional analysis. Comparison of total aggregated burned area for the overlapping years for USFS agency reports, Bureau of the Census (1975), Statistical Abstract (1985), and Ciesla and Mason (2005) showed excellent agreement

(96-100%). Hence, we consider the latter data source complete and useful for reconstructing total burned area.

Validation data

Besides the main data sources used for reconstruction and validation (Table 1), we collected additional burned area products for the U.S.: The National Interagency Fire Center (NIFC), the USGS Federal Wildland Fire Occurrence database (USGS) (Brown et al., 2002), a study on federal forest fires by Malamud et al. (2005), and the Global Fire Atlas (GFA) (Andela et al., 2019). The GFA is another remote sensing product based on gridded data with 500 m spatial resolution, whereas all other sources contain field observations from various agency reports. The NIFC contains information for "wildland fires", which refers to all land-cover types except agricultural and built-up land, according to Houghton et al., (2000). The USGS database (1980-2015) combines fire records from five federal agencies, Bureau of Land Management (BLM), U.S. Forest Service (USFS), Bureau of Indian affairs (BIA), Fish and Wildlife Service (FWS), and National Park Service (NPS). They monitor wildfires on federal land of various land-cover types, such as forests, rangelands, and deserts across the continental U.S. but excluding all fires on non-federal lands. The study by Malamud et al. uses processed data by the USGS for federal forests. For comparison with the historical sources, we excluded agricultural and built-up infrastructure from the GFA databases.

Estimation of burned biomass

We estimate the amount of burned biomass by wildfires by multiplying the reconstructed burned area with fuel loads, and combustion completeness factors. Fuel loads refer to the share of total biomass within a land-cover type susceptible to fire, whereas combustion completeness factors are coefficients used to estimate the share of the respective fuel loads that actually burns in a fire event. Stenzel et al. (2019) argued that default combustion completeness factors are often prone to severely overestimating biomass burned, thus using case-study specific field-measurements for combustion completeness as well as for fuel loadings (van Leeuwen et al., 2014) should be preferred.

We obtained average contemporary forest fuel loads in tons dry matter per hectare and year [t dry matter /ha/yr] based on field measurements from 2003-2015 from Urbanski et al. (2018), specified by compartments: a) surface fuels consisting of duff/litter and deadwood, delineated by fuel moisture time-lag categories (1hr, 10hr, 100hr, 1000hr). The time-lag represents the fuels' response to changing weather conditions and thus their relative dryness (i.e., a 1hr fuel has dried out at an order of one hour after rain events. b) live fuels, consisting of grasses/herbs, as well as canopy components (leaves, live branches, stems). Using a forest type map by Ruefenacht et al. (2008), we allocated the fuel loadings to the four aggregated regions. To account for the different biomass characteristics of the ecosystems under investigation (low-productivity wood/scrub/shrublands vs productive timber and federal forests), we allocated fuel types by forest categories and region as proxies for the available total fuel loadings (see Table S 2). We assumed that all types of fuels for all strata (duff to canopy) would be present in productive forests ("federal", "state & private" forests). For "other forests", we assumed that duff/litter, 1hr deadwood (small branches from dead shrubs), and grasses/herb fuels would be present.

Based on the reconstructed burned area, we used the fire-enabled dynamic vegetation model LPJ-GUESS-SPITFIRE to model annual fuel characteristics (Figure S 2) by fuel component and region for the period 1941-2017. Next, we applied the yearly change coefficients estimated by LPJ-GUESS-SPITFIRE for this period to the contemporary fuel loadings from Urbanski et al. (2018) by region to derive a modelled fuel-characteristics timeline. This way, instead of assuming static values, we accounted for changing climatic conditions influencing fuel loads. For the regional estimations of burned biomass we used the respective combustion completeness factors, as well modelled by LPJ-GUESS-SPITFIRE (see Figure S3). For national level burned biomass from 1926 to 1940, we extrapolated average fuel-loadings and combustion completeness factors from 1941. Finally, dry matter of burned biomass was converted to tons carbon (C) using an average conversion factor of 0.5 (Schlesinger, 1997).

Forest biomass removals, C stocks and NPP_{pot}

To assess the relative role of forest fires and other factors influencing biomass C-stocks, such as changes in land use and climate conditions, we additionally collected and integrated data on biomass harvest, NPP_{pot}, and C-stocks in our database. Due to the lack of consistent data, in this assessment we did not include information on extreme events, such as windthrows or bark beetle infestation. Data for biomass removals, i.e., wood harvest (timber, fuelwood) and forest grazing by livestock were obtained from Magerl et al. (2022) and converted to tons C by using a C-content factor of dry matter biomass of 50%. In this study, total wood harvest was collected from national statistical sources (USFS timber inventories approximately every 10 years, from 1953-2017) which provide yearly average harvest levels between inventory points plus additional historical publications by the USFS for the years 1907, 1920, 1932 and 1945. Harvest data were expanded for biomass lost during harvest (bark, branches, leaves, below-ground biomass) using factors from Krausmann et al., (2013). Forest grazing was estimated using statistical data for livestock numbers and available feed supply (feed crops, pasture, hay) from the USDA Agricultural Census, and species-specific feed demand (Krausmann et al., 2013). See Magerl et al. (2022) for detailed methods and results.

Total forest area and biomass growing stock by state, ownership, tree species and forest category were obtained from the decadal USFS timber inventories by Magerl et al. (2019) for the period 1907-2012. Forest biomass stocks were expanded for above- and belowground biomass, including deadwood and litter stocks, and C-content, using species and ecoregion-specific IPCC factors (Eggleston et al., 2006). See Magerl et al. (2019) for additional details. All results were updated to 2017 using the same approaches. To account for the large yearly variation in burned biomass, we calculated the 5-years moving average for comparison with the decadal average biomass removals and C stock values. For better comparability of pressures on forest ecosystems inflicted by the studied removals at various spatial scales, all biomass fluxes (harvest, fires, NPP_{pot}) and C stocks were expressed per unit area [tC/ha].

To analyse the relative effect of changing climatic conditions on forest C stock trajectories, we included information on the potential Net Primary Productivity (NPP_{pot}) of forests in our assessment. NPP_{pot} data calculated with LPJ-GUESS version 4.0.1., was taken from Kastner et al. (2022). NPP_{pot} is defined as the NPP of the potential vegetation of ecosystems that would prevail under the hypothetical absence of human land use but with current climatic conditions. We removed NPP_{pot} values below zero and calculated 5-year averages from the yearly NPP_{pot} model output. To combine the land-use data at 5 arcmin resolution from Kastner et al. (2021) with NPP_{pot} data at 30 arcmin resolution, the NPP_{pot} value in gC/m^2 per 30 arcmin grid cell was used for all 5 arcmin cells contained per 30 arcmin grid cell. Based on visual inspection, wilderness areas from Kastner et al. (2021) in the U.S. were found to also cover sparsely forested regions in the Rocky Mountains and Pacific Coast regions that are included in national forest statistics (i.e., "other forests" in this study) and thus were also considered in the average forest NPP_{pot} per ha calculation.

To identify periods and regions where fire or the other studied factors were important for the observed forest stock trajectory, we investigated the relation of per-area burned biomass, forest grazing, harvest, and NPP_{pot} and the observed developments in forest C stock densities, by means of time-varying linear correlations: We considered different time-periods and starting points, on the national scale in the period 1926-2017, and on the regional scale 1941-2017, defined by the available data points for forest C stocks, at roughly 10 years intervals (See SI, Section 5). Linear correlations with coefficients of determination $r^2 > 0.5$ at $= 0.05$ were considered for the analysis (Table S 3).

Sensitivity Analysis

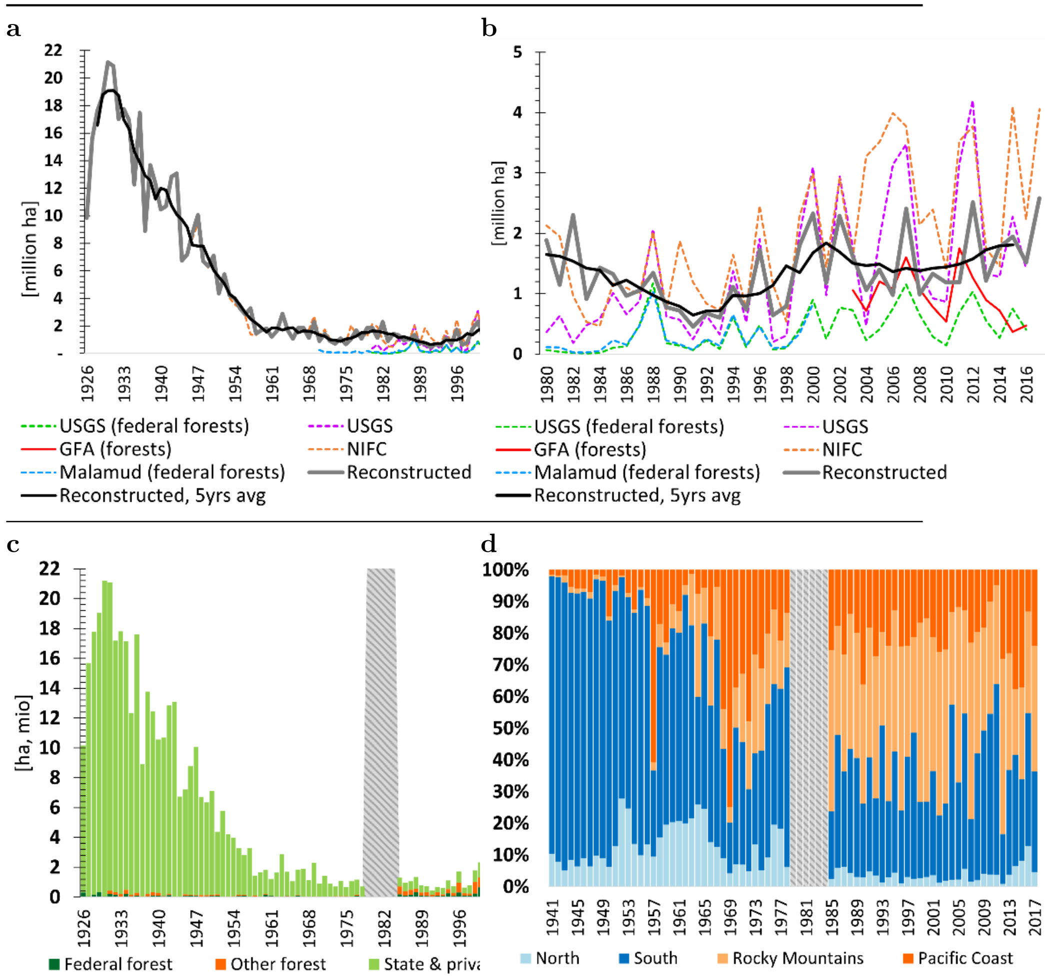
We calculated 48 variations for biomass burned to assess the sensitivity of our results. We used different combinations of factors for fuel loadings (modelled, field-measurements, static vs dynamic over time) and combustion completeness factors (static low, moderate, high severity) from the literature (Yang et al., 2015). Additionally, we re-estimated burned biomass using average fuel loadings for 1.) separate forest categories burned and 2.) aggregated forest area burned. This way, we tested our results for regional variations, temporal evolution of fuel loads, and scaling effects, and assessed the influence of modelled vs field-measurement derived variables. Full results and details for the variants can be found in the SI, Section 7.

1.

Results

(a)

Burned Area



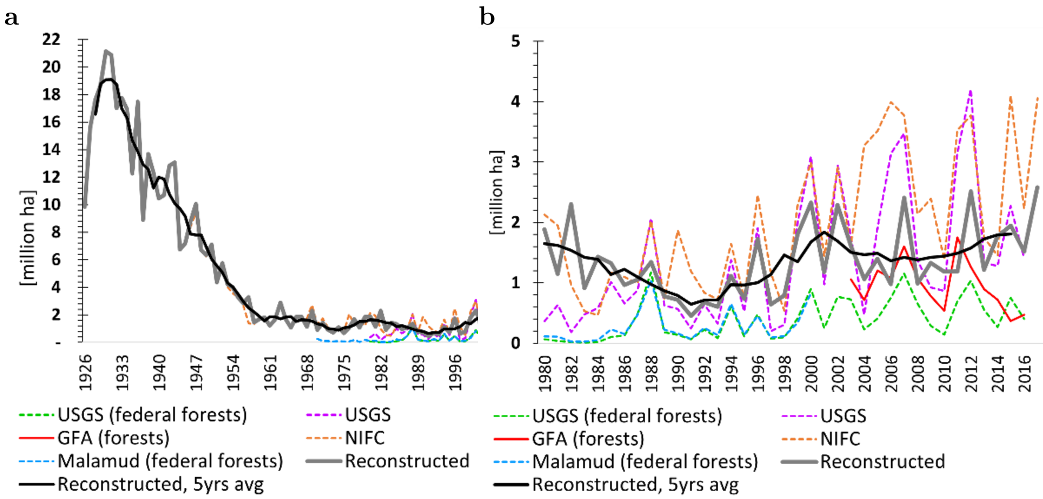


Figure 1: Reconstructed burned area for the continental U.S. total (black line) and five years moving average (grey line) [Combined data by Bureau of the Census, USFS agency reports, Ciesla & Mason, Landsat, see text and Table 1 for details] compared to other burned area products for the U.S. in million ha, **a**) 1926-2017 **b**) 1980-2017. Reconstructed U.S. burned forest area, national total **c**) 1926-2017 by forest category, million ha **d**) 1941-2017 by regions, shares of total burned forest area. Notes: In **a**) and **b**), agency reports or field observations appear as dashed lines; remote sensing, and reconstructed series appear as solid lines. In **c**) and **d**) the grey striped bars represent the 1980-1984 data gap. No data available for 'Federal' and 'Other forest' on the regional scale in 1943, 44, 52, 53, 57, 59, 67, 68, 70 See text, table 1 and SI section 2 for additional details.

Figure 1a and b present the reconstructed total absolute (solid grey line) and five years moving averages (solid black line) burned forest area for the continental U.S. in comparison to other available burned area data sources: Trajectories in burned area are similar, with our reconstructed timeline reporting the highest values throughout the period for forests (Figure 1a). Compared to our reconstruction, average burned area of the GFA 2003-2016 was lower by 43%, USGS (only forests) 1980-2015, and Malamud et al. (2005) 1980-2000 were lower by 52%. Total USGS burned area was higher because it includes other land-cover types such as grasslands, rangelands, croplands under federal administration, and area burned by prescribed fires. USGS forest burned area includes only federal forests and was thus lower than our reconstruction. Until the 1970s the NIFC, Bureau of the Census (1975) and our reconstruction based on USFS agency reports show almost the same numbers (Figure 1a). After the 1970s NIFC data diverge from our reconstruction, but its original sources for the timeline 1983-2017 were not referenced in the online source (https://www.nifc.gov/fire-information/statistics/wildfires). However, the differences can be explained by the fact that it also includes burned area for Alaska and Hawaii, and similar to the USGS, other land-cover types such as grasslands (Houghton et al., 2000), whereas we limited our reconstruction to forests, wood-, shrub-, and scrublands in the continental U.S. Due to the higher spatial resolution of the Landsat data (30m) compared to the MODIS data on which GFA is based (500m), our reconstructed burned forest area is higher, capturing also small fires (minimum fire size 0.9ha in Landsat vs 21ha in GFA). While the reconstructed historical burned area shows almost perfect agreement with the NIFC (complete overlap between 1926-1945, Figure 1a), for 1985-2017 Landsat yielded partly diverging values compared to the USGS (only forests), GFA, and Malamud data, due to different land-cover and spatial coverage, as explained above. Nevertheless, the national trends of our reconstruction, NIFC, and the USGS, albeit on different levels, follow similar patterns (Figure 1b).

Reconstructed burned forest area (Figure 1c) by forest category on the national scale decreased significantly between 1926 and 2017, and shifted from the North and South (East) to Rocky Mountains and Pacific Coast (West) (Figure 1d). In total, burned area shrunk from a maximum of 21 Mha (Megahectare = million hectare) in 1931, to 0.4 Mha in 1991 but increased again to 2 Mha in 2017. On average, between 1926 and 1978, 7 Mha of forests burned annually across the continental U.S. Burnt area fraction during this timeframe amounted to a minimum of 0.27% in 1975 and a maximum of 8% in 1930 and 1931. For a large part of the observed period, state & private forest was the most fire-prone forest category (Figure 1c) and accounted on average for 81% of total aggregated burned area (minimum 35%, maximum 99% of the total). Burnt area fraction in total state & private forests between 1926 and 1978 was on average 4% (maximum of 11% in 1930, minimum of 0.32% in 1975). This fraction decreased to a maximum of 0.62% 1985-2017. Federal and other forests seem to have burned significantly less in the beginning of the observed timeframe but their relative share in total forest area burned, compared to state & private forests, increased significantly in the last decades of the analysed period (Figure 1c, Figure S 5). This is probably due to more accurate reporting of burned area via remote sensing data (Chuvieco et al., 2019; Short, 2015) compared to early field observations, which did not report some fires in woodlands (Loehle, 2020).

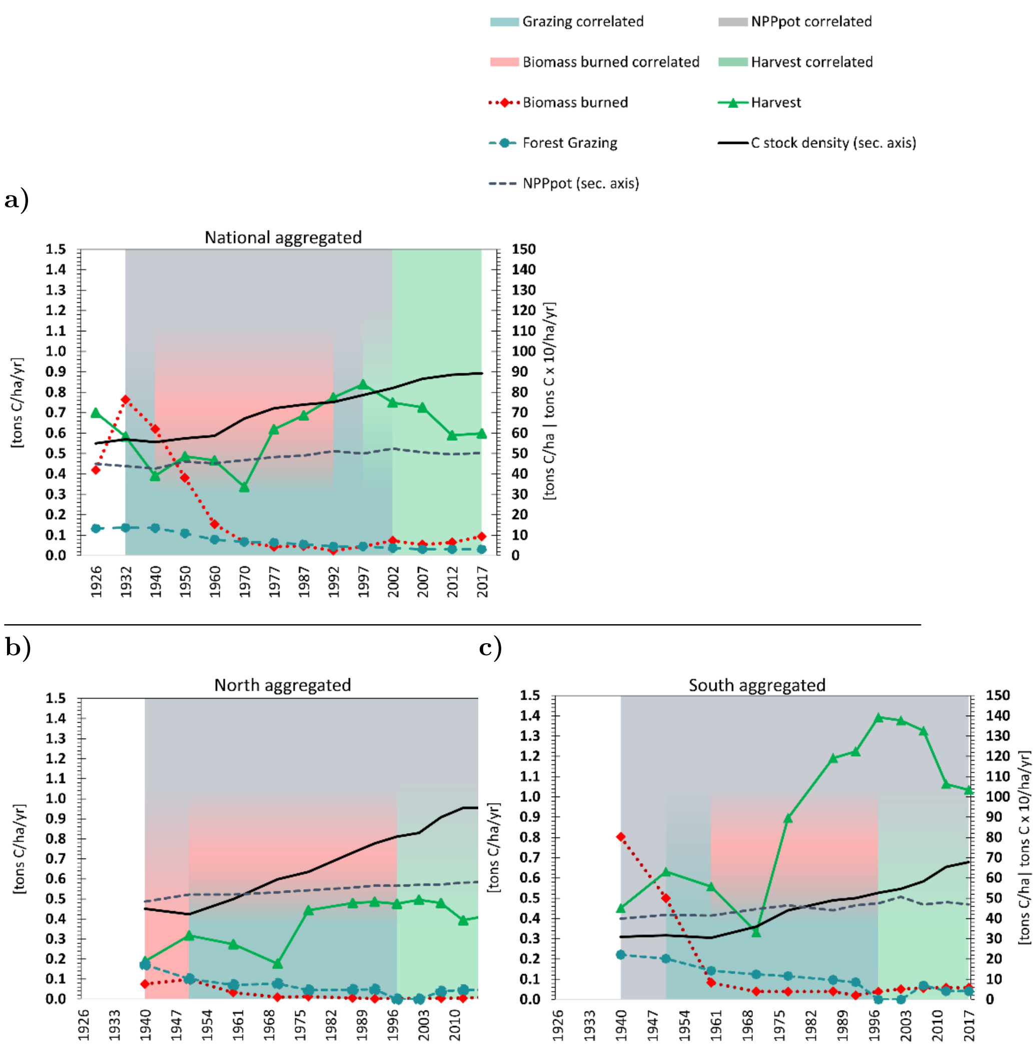
The regional disaggregation of total burned forest area (Figure 1d) varied throughout the observed period but reveal a general pattern of strongly decreasing fire activity in the Eastern and less strongly decreasing fire activity in the Western regions. The South was the major driver of total burned area for most of the period from 1941 until the 1970s, accounting for 16-90% (average 65%) of total burnt forest area in this timeframe. Its share in total burned forest area decreased significantly, from 80% in the 1940s to 58% between 1955 and 1967, further to 38% between 1967 and 1985. Between 1985 and 2017 it remained around 34%. However, the USFS state-level data, starting in 1941, reveal that an average of 54% of total burned forest area between 1941 and 1960 stems from just three Southern states (Florida, Georgia, and Mississippi, Figure S 4).

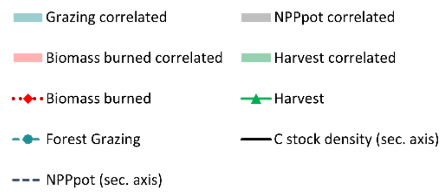
Most burned area in the South and North occurred in state & private forests, in the former representing on average 99% of total burnt area fraction between 1941 and 1978, and still 81% on average between 1985 and 2017 (Figure S 5). In the North, these shares were similar. However, since the 1980s, the relative shares of federal forests in burnt area fraction in these regions have risen to around 25% in the South in 2017 and 30% in the North, respectively. In contrast, forest area burned was more heterogeneously distributed across forest categories in the Western regions throughout the entire time-period.

The role of Wildfires, Wood Harvest, Forest Grazing, and Environmental change for Carbon stock dynamics

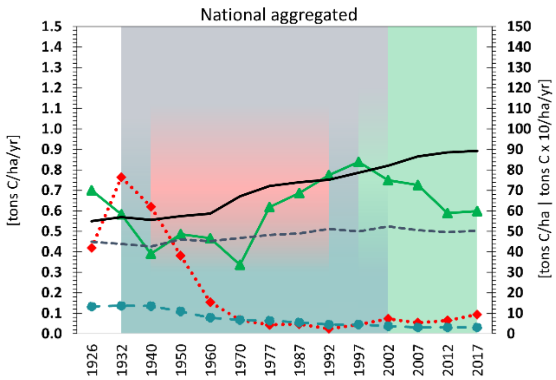
Figure 2 displays trends in burned biomass in wildfires, harvest, forest grazing, and NPP_{pot} per total forest area in tons C/ha/year, as well as forest C stock density trajectories (tons C/ha), on the national (Figure 2a) and regional scales (Figure 2b-e). We investigated linear correlations ($r^2>0.5$, $=0.05$) to identify time-periods in which observed trajectories in removals and NPP_{pot} coincided with C stock density development, on all scales. Shaded areas in Figure 2 indicate which factors in which periods had the highest coefficients of determination and were negative (removals) or positive (NPP_{pot}) linearly correlated with trajectories of C stock densities, compared to the other factors (see Table S 3 for full results).

On the national scale, albeit in different time periods, the trajectories of all C fluxes were correlated to stock density, which grew 39% over the entire period (Figure 2a). The decrease of burned biomass by more than 80% from 1940 to 1970 was negatively correlated with stock density increase. Because of a more consistent decline, forest grazing was negatively correlated to stock density development for a longer period than biomass burned, from 1932 until the 2000s. The increase of NPP_{pot} between 1932-2002 was positively correlated to stock density growth during this period. Wood harvest fluctuated strongly over the observed period and was negatively correlated to stock density increase only between 1997-2017. Burned biomass and harvest per unit area were the largest removals until the 1970s. While burned biomass remained below 0.2 t C/ha/yr from the 1970s until 2017, harvest oscillated between 0.4 and 0.9 t C/ha/yr throughout the observed period. Thus, of all removals, harvest exerted the highest per area pressure on forests after the 1950s. In comparison to the national scale, the regional analyses (Figure 2b-e) in the period 1941-2017 show similarities in C stock densities and biomass removal trajectories in the North and the South but also considerably different patterns in the Rocky Mountains and the Pacific Coast.

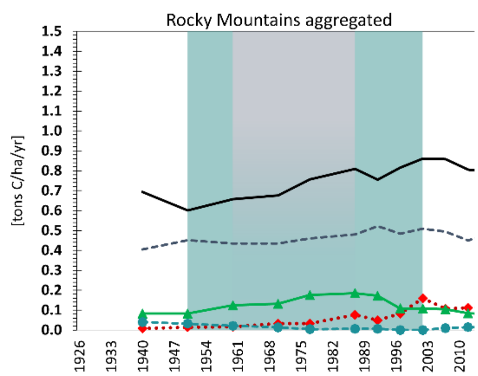



Grazing correlated
NPPpot correlated
Biomass burned correlated
Harvest correlated
Biomass burned
Harvest
Forest Grazing
C stock density (sec. axis)
NPPpot (sec. axis)

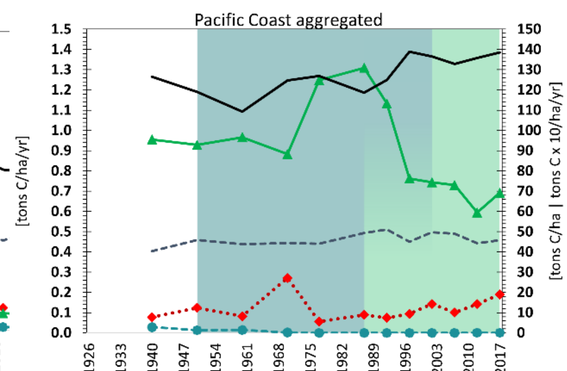
a)


National aggregated
[tons C/ha/yr]
[tons C/ha | tons C x 10/ha/yr]

d)


Rocky Mountains aggregated
[tons C/ha/yr]

e)


Pacific Coast aggregated
[tons C/ha/yr]
[tons C/ha | tons C x 10/ha/yr]

a)

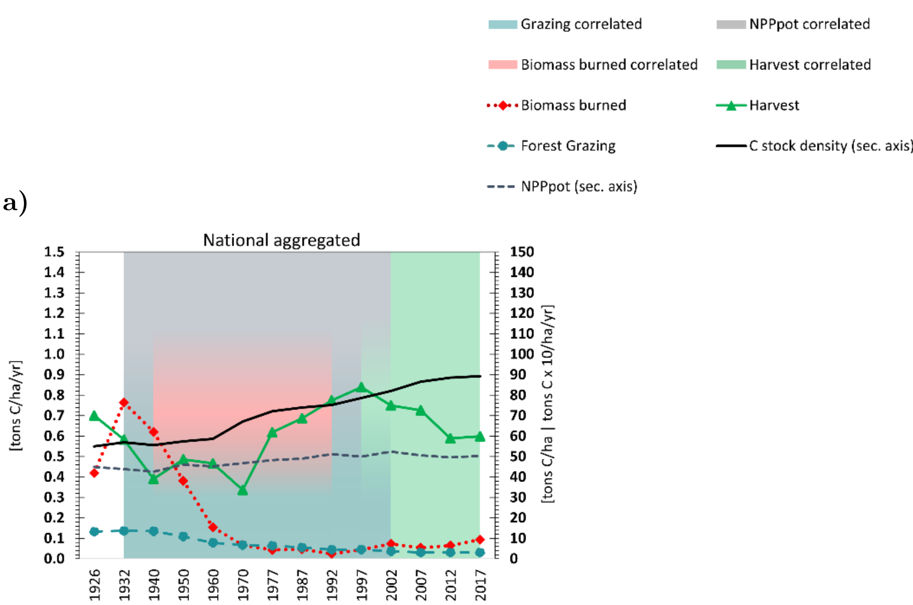
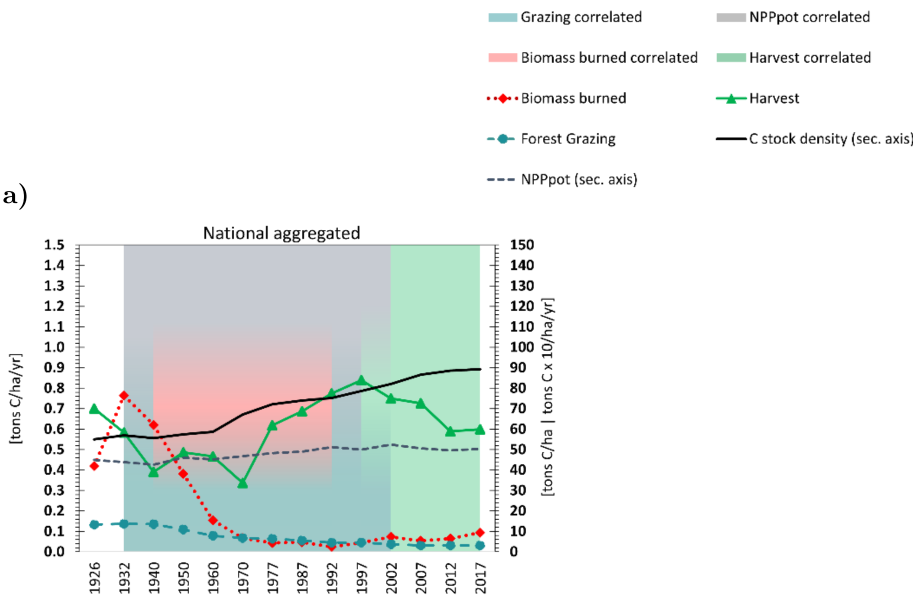


Figure 2: Forest biomass removals, Potential Net Primary Production and C stock density change per total forest area. Primary axis: Burned biomass, 5 years moving average; wood harvest; forest grazing [tons C/ha/yr]; Secondary axis: NPP_{pot}, 5 years moving average [tons C x 10/ha/year]; C stock density [tons C/ha]. **a**) National total, 1926-2017 **b**) North **c**) South **d**) Rocky Mountains **e**) Pacific Coast. 1941-2017. Shadings indicate linear negative correlations between change in C stock density and removals, and positive correlations between NPP_{pot} and C stock density ($r^2 > 0.5$, =0.05) and express which of the studied factors compared to all other factors had the highest coefficient of determination **Notes**: National aggregated data available from 1926, sub-national data from 1941. No data for federal and other forests for 1943, 44, 52, 53, 57, 59, 67, 68, 70. Data for forest grazing, harvest, and C-stock density obtained from (Magerl et al., 2022) and updated for 2017 using data from (Oswalt et al., 2018). NPP_{pot} values multiplied by 10 for better visibility.

a)



In the North (Figure 2b), between 1940 and 1997 the trend of fire was negatively correlated to the observed development of C stock density, which almost doubled during the observed period. Forest grazing declined as well and was negatively correlated with stock density increase 1950-1997. Contrary to the national scale, NPP_{pot} was significantly correlated with stock density increase throughout the observed period, even after 2002. Similar to the national scale, declines in harvest were linearly correlated with stock density trends only between 1997-2017. However, harvest exerted the largest pressures on forests throughout the observed period, although strongly fluctuating, followed by forest grazing, while biomass burned was the smallest removal in this region.

In the South, (Figure 2c) strong fire suppression and stock increase coincided only between 1960 and 1997. Forest grazing was negatively correlated to stock density development in the period 1950-1997. In this region, as in the North, NPP_{pot} was positively correlated to increasing stocks throughout the observed period, while the decline in harvest again coincided with increasing stocks after 1997. In the South stock density was the lowest, and the per-area removals were the highest of all regions. Burnt biomass was the lowest removal between 1960 and 1997 while harvest, after declining between 1950 and 1970, increased dramatically until 1997. Similar to the national trend, grazing declined steadily and remained between 0.04 and 0.2 t C/ha/yr throughout the entire period.

Trajectories in the Rocky Mountains and Pacific Coast regions were different than in the North and South. In the two Western regions, we find no linear correlations between biomass burned and the observed trajectories in stock density. However, the per area pressure of fire in these regions in the last 30 years of this analysis were the largest among all regions.

The development of biomass stock density in the Rocky Mountains (Figure 2d), was less pronounced than in the Eastern regions, increasing by only 17% until 2007, and even declining again by 10% thereafter. The development of NPP_{pot} and forest grazing coincided with the increase in stock density. The reduction of grazing from 1950 until the early 2000s was negatively correlated with stocks, while NPP_{pot} was positively correlated between the 1960s and the 1920s. After 2002, we find no linear correlations between C stock densities and removals or NPP_{pot}. Biomass removals were at much lower levels than in the East over the observed time-period. Neither burned biomass nor harvest exceeded 0.2 t C/ha/yr and neither was correlated with observed trends in stock density. However, in the Rocky Mountains biomass burned increased steadily from 1977 to 2002 and remained at a higher level after 2002 than before, even overtaking harvest, which decreased in this region from 1987 onwards.

In the Pacific Coast (Figure 2e), neither biomass burned nor NPP_{pot} were linearly correlated with C stock density, both fluctuating over the entire period. The consistent decline of forest grazing in this region was negatively correlated with C stock density increase between 1950 and 2002, albeit the per-area pressure of this removal was the lowest of all regions. In this region, the decline of harvest 1987-2017 was negatively correlated with stock density change, similar to the national scale and the Eastern regions. The Pacific Coast region had the highest C stock density in the country throughout the period, 80-140 t C/ha, and biomass burned in this region increased almost linearly between the mid-1970s and 2017. Harvest remained relatively constant between 1940 and 1977 but increased between 1970 and 1987. Between 1987 and 2017 the decline of harvest coincided with increasing C stock density.

1.

Discussion

(a)

Comparison of results

Our study demonstrates how historical statistical reports, contemporary satellite data, field observations, and dynamic vegetation models can be used to reconstruct dynamics of wildfires, harvest, grazing, environmental change, and C stocks in forests of the continental U.S. for 91 years on the national, and 76 years on the regional level. Comparisons reveal that our results are well in line with other published estimates of biomass burned for the U.S. for varying temporal and spatial scales (Table 2), thus confirming the robustness of our estimation method.

Table 2: Comparison of published biomass burning estimates with this study. Range represents minimum and maximum values

Biomass burned cumulative/annual			
Study	**Scale**	**Land-Cover**	**Fuels**
(Hicke et al., 2013)	Western U.S.	Forest	Total Live Biomass
(Urbanski et al., 2018)	CONUS	Forest	Duff Litter, Deadwoods (1hr-
(French et al., 2014)	CONUS	Forest, Rangeland	Canopy, Shrubs, Herbs, Dowr

Although our results largely agree with published estimates, we underline that this study presents only an approximation of long-term fire impacts on forest ecosystems in the continental U.S. We conclude that our estimate may be conservative in its calculation of the total fire-induced C fluxes: While comparing agency reports and satellite data (Figure 1a and b) revealed good agreement between data sources for the period 1985-2017 we must also note that the historical statistics may have underestimated burned area, in particular in federal and other forests (Figure S 5). Nevertheless, despite possible underreporting and inconsistencies in the historical USFS records (Short, 2015), we argue that the data coverage allows drawing reasonable conclusions:

The total area monitored and protected by the USFS (i.e., "burnable land") increased significantly during our study period and reached full coverage only by 1990. However, the major parts of the poorly covered reporting area before 1990 were non-forested ecosystems such as western native rangelands, as discussed by Houghton et al. (2000) and Short (2015). The interior U.S., and especially the Great Plains sub-region, were seriously underreported in the USFS records between 1950-1969. However, since there have been almost no forests and thus comparatively low biomass C stocks in this sub-region (0.7-1% of total continental U.S. forest area) we argue that they are negligible for our analysis of forest fires (Magerl et al., 2019; Oswalt et al., 2018; Reynolds & Pierson, 1941; USDA Forest Service, 1958). A representative portion (around 80%) of total forests in the continental U.S. were already covered in the USFS fire statistics by 1926. By the 1940s, full coverage of national forest area had been achieved. Especially states of the North and South with large shares in total forest area were relatively well covered (between 70-90%) during this period. Poorly covered Eastern states (below <50% coverage) represented only 5% of the total continental forest area between 1950-1960.

The USFS agency records include wildfires as well as prescribed burning and harvest slash burning on unprotected land but do not report these processes separately (Short, 2015). The same applies to the Landsat data (Hawbaker et al., 2020). Although it is not possible to quantitatively assess the share of total biomass burning attributable to human action, Kolden (2019) showed that between 1998-2018 on average 1 Mha (or >60% of our reconstructed burned area) per year can be associated with prescribed burning, over 90% of which occurred in the South. According to Short (2015), the actual number of prescribed fires could have been even larger, at least within the last 20 years of our analysis. If human-induced fires in the contemporary controlled anthropogenic fire regime contributed such a large share in recent years, it is likely that there were even more of these fires during the decades of uncontrolled, private burning in the early 20th century (Hart, 1977; Otto, 1983; Otto & Anderson, 1982).

Limitations

Note that we did not report total C-fluxes from wildfires in all land-cover types in the US. We do only include forested areas and shrub/scrublands, which account for 15-40% (average 27%) of the total burned area (see section S 8). All other land cover types (e.g., agricultural land, infrastructure, wetlands, unused/barren land) are also subject to burning. We did not estimate biomass burned in these land-cover types because they are negligible for our analysis of biomass C stock dynamics, since neither of these ecosystems contain considerable quantities of perennial aboveground biomass (trees). However, for analyses which assess for example annual or seasonal total emissions from biomass burning or include deep soil organic C, they might be of particular interest. 2000 and 2014 were the first years which were covered entirely in length by the Landsat 7 and Landsat 8 missions, respectively. This coincides with an increase in average burned area since 2002 identified by Landsat. However, overall temporal trends in burned area for the whole time-period covered by Landsat were validated by Hawbaker et al (2020) by means of linear regression with good fit, hence making the data usable for our purpose. Additionally, the increase in burned area had already started in the 1980s. The agency records assessed in this study show similar increases in total burned area (Figure 1b).

Despite possible sources of bias and uncertainty the data sources used represent, to our knowledge, the most widely utilised, comprehensive data products for analysing long-term trends in national and sub-national burned area in the United States (Houghton et al., 2000; Littell et al., 2010; Marlon et al., 2012; Syphard & Keeley, 2016; Westerling et al., 2003). Although not perfect, our estimates are robust approximations of actual wildfire activities in U.S. forests for the period 1926-2017, with even higher confidence for the period after 1985. Due to data limitation (especially for wildfires), we could not comprehensively and empirically assess the period of large scale timber harvest and declines thereof prior to 1926 (Magerl et al., 2022). However, consulting historical qualitative sources and other long-term land-use analysis for the U.S., in section 4.4.1. we discuss the different historical land-use contexts of the regions analysed in section 3.4. Besides the studied removals, biomass losses due to windfall, insects, and similar disturbances, as well as other factors like afforestation, intentional and incidental reforestation on abandoned agricultural land, or nutrient deposition represent important components of forest change. Due to the scope of this study, we did not account for these factors, but want to highlight their importance for future research.

Sensitivity analysis

The sensitivity analyses (see SI Section 7 for details) revealed a large range of uncertainty for our estimate of biomass burned (Figure 3). We used different combinations of fuel load factors (modelled by SPITFIRE vs static field measurements) and combustion completeness factors (published static low, moderate and high severity from Yang et al. (2015) vs dynamic modelled by SPITFIRE) to re-estimate 4 main variants including 3 sub-variants each, yielding in total 48 alternative estimates for burned biomass on the regional scale for 1941-2017. Additionally, we tested the effect of using forest-category specific fuel loads vs average forest fuel loads. In general, using low-to-high combustion completeness factors from the literature, and excluding duff fuel loads exerted the largest impact on our results. Using average fuel loads instead of specific factors for each forest category and static instead of dynamic fuel loadings caused less variation in overall results. The largest alternative estimate was on average 4 times higher (+220 TgC/yr) than the study result (Fig. 3b, variant 3c) while the lowest estimation (Fig. 3b, variant 2) was on average 11 times smaller (-75 TgC/yr) than our best guess.

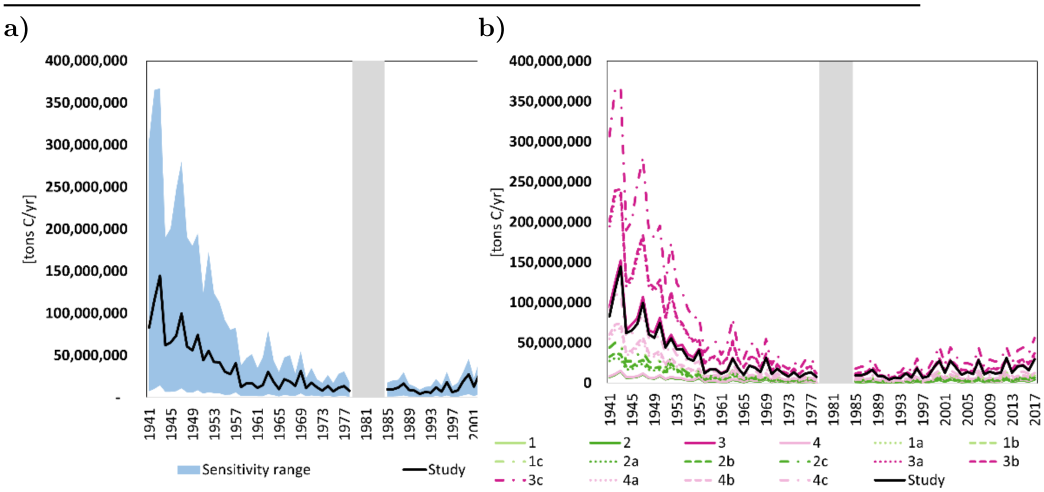


Figure 3: Results of the sensitivity analysis on the national level **a**) combined sensitivity range of all variants and best-guess study estimate **b**) individual results for variants 1-4 [**1**: dynamic average modelled fuel loads; modelled CC; **2**: dynamic modelled fuel loads for separate forest categories; modelled CC;

3: static field measurement fuel loads for separate forest categories; modelled CC **4**: static field measurement fuel loads (excluding duff) for separate forest categories; modelled CC. Sub-variants 1-4**a**: average fuel loads; static CC low 1-4**b**: average fuel loads; static CC moderate. 1-4**c**: average fuel loads; static CC high.]

These lower and upper boundaries represent hypothetical constant low and high fire intensities. For example, variant 3c in Figure 3b assumes that in each fire event, depending on fuel category, between 55-99% of all fuels are destroyed. These estimates can be considered extreme and unrealistic outliers, as in reality usually only few live trees are killed in a wildfire event (Ito, 2004; Stenzel et al., 2019). Nevertheless, our sensitivity analysis underlines the high level of uncertainty connected with burned biomass calculations, as discussed in the literature (Reid et al., 2005; Robinson, 1989). However, comparison with other published estimates (Table 3) confirms that our estimation is within a reasonable range. We thus consider our results as robust.

Regional dynamics and historical context

The national level (Figure 2a) trajectories of C stocks and fluxes largely reflect the change in USFS conservation policies (Steen, 2004); aggregated burned biomass as well as forest grazing and harvest decreased between 1926 and the 1970s. Consequently, wood harvest levels could be maintained and even increase during the rest of the studied period, simultaneous to the observed increase in biomass C stock density. However, the role of wildfires in comparison to the other observed factors as well as their correlations to C stock density trajectories showed strong spatial and temporal variability. These dynamics need to be discussed in the context of the regionally different land-use histories.

North and South: Stock Recovery and Fire Suppression

As described in historical literature, low C densities in the North and South in the first years of our analysis were the result of strong pressures on forests during the 19^{th} and early 20^{th} century, including extensive clearcutting of mountain slopes (Davis, 2000): Native-American fire practices, as a tool for biome conversion, hunting, and pest control, as well as land-use change and agricultural conversion, were adapted by Euro-American settlers (Fowler & Konopik, 2007; Pyne, 1982). Such cultural fire practices, in combination with large-scale wood harvest and other disturbances like invasive species and pests (e.g., chestnut blight in the early 20^{th} century) had contributed to largely depleted forests, especially in the South (Gregg, 2010). The introduction of fire policies in the early 1900s led to a large decline in aggregated burned area. In the South this decline indicates a change from human ignition of many small fires to extended human fire suppression, rather than a decline of large, severe fires (Pyne, 1982). Additionally, the emergence of commercial timber utilization and the modernization of local subsistence-based economies was also connected to fire suppression efforts and declines in forest grazing (Grelen, 1978; Hansbrough, 1961; Jurgelski, 2008; Steen, 2004). Our results reflect how these altered use patterns together with enhanced growing conditions, arguably due to human induced climate change, persisted through much of the 20th century and coincided with recovering stocks in the North and South. These regions show the largest per-area reduction in fire and grazing until the 1970s, in parallel to the largest nationwide increases in C stock densities (Figure 2b, c). Stock density increases happened mostly on highly productive commercially used timber forest under state & private ownership (Magerl et al., 2019). However, in the last decades of the observed period, we find no linear correlations between fire or grazing and the development of C stock densities. Instead, the correlation of increasing stock densities, and declining harvest, as well as increasing (North) or stable (South) NPP_{pot} indicates that a) the contribution of contemporary fire suppression to forest recovery might be less important than in the past, especially under changing climate (Singleton et al., 2019) and b) simultaneous growth of harvest and C stock density may be feasible only temporarily for few decades (Gingrich et al., 2022). Forests in the South are less resilient to wildfires, mainly due to stand age, and structure (Mitchell et al., 2014; Nowacki & Abrams, 2008; White et al., 1985) and wildfires have already increased throughout the East in recent decades, although on relatively low levels compared to the West. The expected further increase of occurrence and severity of wildfires with changing climate in the continental U.S. (Barbero et al., 2015) indicates that additional prescribed burning for fuel reduction rather than fire suppression may be necessary, if contemporary harvest levels in the East are to be maintained alongside continued C sequestration and biodiversity conservation (Kolden, 2019; Parks et al., 2015).

Rocky Mountains and Pacific Coast: Fluctuating dynamics and increasing fires

In contrast, in Western states biomass burned was not linearly correlated to C stock density trajectories. Especially in the Rocky Mountains, there is less commercial timberland than in the East but much more low-productive shrubland, as well as reserved forests and protected national parks (Magerl et al., 2019). Land-use change as well as harvest in both Western regions was less extensive than in their Eastern counterparts prior to the 1940s (Houghton & Hackler, 2000). Consequently, in both regions, C stock densities were much higher in the 1940s than in the East. In the Rocky Mountains, stock density increase was linearly correlated with growing NPP_{pot} during the second half of the 20th century, and per-area removal rates were the lowest of all regions. Harvest could increase in parallel to stock regrowth for approximately 35 years, before it declined in the 1980s. Declining stock density since the mid-2000s was not correlated with any of the investigated removals or NPP_{pot}, which fluctuated since the early 1990s in this region. The dense forests of the Pacific Coast enabled for high per area timber harvest, however, these rates never exceeded 1% of stock density (compared to 2-3% in the South), thus exerting lower relative pressure on forests than in the East (Figure 2c,e). The pronounced decline of harvest in this region between 1986 and 2012 coincided with growing stock density.

Wildfires, although increasing, were not linearly correlated with stock density change in the West, but are arguably connected to climate change (e.g., Abatzoglou & Williams, 2016; Dennison et al., 2014; Kolden, 2019; Westerling, 2006). Extended fire seasons, and higher temperatures support increased occurrence, size and severity of fires, especially in the Rocky Mountains region (Balch et al., 2017; Barbero et al., 2015; Singleton et al., 2019; Yang et al., 2015). Concomitant effects like drought stress and bark beetle infestations exert additional pressures on forests and could explain the observed C density decline in this region (Figure 2d; Anderegg et al., 2022; Hicke et al., 2016; Kolb et al., 2016). Vice versa, increased tree mortality due to bark beetle outbreaks could be related to increasing wildfires in the Rocky Mountains (Hicke et al., 2013, 2020). Hence, we argue that compared to changes in biomass burned or wood harvest, the relative effects of other processes contributable to environmental change might have been more important for the observed stock density development in the Rocky Mountains, although the relative effect of wildfires on C stocks in the future might increase. In both the Pacific Coast and Rocky Mountains, forest grazing pressures were linearly correlated with stock density change, but per area rates were lower than in the East. However, how grazing affects forest biomass is still poorly understood (Öllerer et al., 2019) and the interlinkage between livestock, ecosystems, and wildfires remains an important research frontier (Erb et al., 2018; Foster et al., 2020).

Implications for future forest C dynamics

This analysis highlights the importance of understanding the tight and complex linkage between wildfire management, climate change and forest use, and the associated effects impacting reforestation and terrestrial carbon sinks (Gingrich et al., 2019; Loudermilk et al., 2013; Magerl et al., 2022). Current levels of harvest and C sequestration in the U.S., mostly determined by dynamics in the East, are largely the results of twentieth-century reductions of forest pressures and enhanced climatic growing conditions, enabling regrowth of regionally depleted forests. The time-series reconstruction indicates that in recent decades, changing climate and associated interrelated disturbances (wildfire, bark beetles, droughts, mostly in the Rocky Mountains) on forests might be counteracting possible increases in future harvest and C-sink rates, let alone maintaining current levels thereof. These findings indicate that the anthropogenic fire suppression regime established over the 20th century, which played a decisive role for the contemporary observed stock densities (especially in the Eastern U.S.) could, under changing climate, less effectively add to future C stock gains than in the past. As shown in several studies, climate conditions and vegetation always were strong controls influencing wildfire regimes, but during the twentieth century fire regimes have become altered due to human influence and changing climate (Higuera et al., 2015; Littell et al., 2009; Marlon et al., 2012). While more severe wildfires in recent decades have already posed serious threats to wildlife, human health, infrastructure, and property (Baijnath-Rodino et al., 2021; Palinkas, 2020), from a carbon sequestration perspective, the current aggregated wildfire impacts in the U.S. are - compared to historical levels - relatively low. As shown in this study, current levels of biomass burning due to wildfires are not significantly reducing the overall potential for additional C sequestration in forests on an aggregated national level. However, research suggests that future impacts on forested ecosystems on regional scales might be severe (Mitchell et al., 2014; Singleton et al., 2019).

In the light of predicted increasing wildfires in the U.S., this suggests that further suppression efforts alone will not be a suitable measure for stabilising forest C dynamics (Abatzoglou et al., 2021; Anderegg et al., 2022). These developments call for better coordination of regionally adapted forest use and fire protection policies, integrating assessments of potential fires, i.e., theoretical fire occurrence and severity without human influence, and regional climate projections. Although we did not address legacy effects empirically in this study, the observed lower levels of harvest and C stocks in the early 20th century in the East, as well as the strong stock regrowth in this region compared to the other regions and later decades point to the importance of legacy effects of past forest use, in line with historical literature (e.g., MacCleery, 1993; Maxwell, 1973) and previous studies on past U.S. land use and forest change (e.g., Houghton, 1999; Raiho et al., 2022). In this context, we argue that time-lags involved with forest recovery from disturbances (decades to centuries) should be better included into future management strategies of fires and forests. However, despite these developments, strategies for increased use of forest biomass, e.g. harvested wood products and biofuels, in the context of climate change mitigation have been proposed in the U.S. and other places (European Union, 2009; United States Environmental Protection Agency (USEPA), 2018). These measures are linked to additional reforestation and afforestation as a means of C storage and decarbonisation of the energy system. While reforestation and fossil fuel substitution undoubtedly play an important role in climate mitigation strategies, we argue that in the light of increasing uncertainty of forest disturbances due to climate change, forest protection should be preferred over increased biomass use (Erb & Gingrich, 2022).

Conclusions

This study investigated the impact of wildfires on forest biomass Carbon stocks in comparison to wood harvest, forest grazing and potential net primary production. We developed a robust reconstruction of burnt biomass in the U.S. by combining historical statistics and satellite observations. Our analyses were based on a consistent dataset, harmonizing different data sources over the 20th century on various spatial and temporal scales for the continental U.S.

The overall trends in forest carbon dynamics were influenced more by the North and South than by the Rocky Mountains and Pacific Coast. A wildfire suppression regime, aimed to eliminate human-ignited wildfires in few states was established in the East in largely depleted forests. In combination with enhanced growing conditions due to climate change this enabled regrowth of commercial timber forests used for intensive harvest in the second half of the twentieth century. In contrast, in the West we did not find such pronounced correlations between wildfires, forest stocks, changing climate and harvest. Possibly due to the absence of extensive past harvest and land-clearing legacy effects, C stock densities in these regions were higher than in the East. Here, wildfire pressures were more dynamic and increased in both regions during the last decade of the observed period. Linear correlations indicate that recent reductions in timber harvest were connected to growing stocks in all regions except the Rocky Mountains. Hence, environmental change, past fire management and forest use patterns in the East have contributed significantly to reforestation in the U.S. and reflected the current state of forests, while in the West the links to changes in fire patterns is less evident.

From this reasoning we conclude that the contribution of further fire suppression on forest (re-)growth in the U.S. might be limited, while a major determinant is and will be forest harvest. Additionally, increased forest fires in recent decades imply that the fire suppression regime established in the past may not be sufficient to ensure for future uses simultaneously with increasing forest stock density and C sinks. We argue that harvest is in contrast the key driver, and its dynamics will be decisive for future C-stock developments. Increasing harvest, for example for bioeconomy-approaches that aim at substitution for fossil fuels might halt or reverse the trend of C-stock increases.

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