Dye et al.
Carbon Balance and Management (2024) 19:35
https://doi.org/10.1186/s13021-024-00282-0

Carbon Balance and Management

REVIEW **Open Access**

Carbon, climate, and natural disturbance: a review of mechanisms, challenges, and tools for understanding forest carbon stability in an uncertain future

Alex W. Dye[1*], Rachel M. Houtman[2], Peng Gao[3], William R. L. Anderegg[4], Christopher J. Fettig[5], Jeffrey A. Hicke[6], John B. Kim[7], Christopher J. Still[1], Kevin Young[8] and Karin L. Riley[2]

Abstract

In this review, we discuss current research on forest carbon risk from natural disturbance under climate change for the United States, with emphasis on advancements in analytical mapping and modeling tools that have potential to drive research for managing future long-term stability of forest carbon. As a natural mechanism for carbon storage, forests are a critical component of meeting climate mitigation strategies designed to combat anthropogenic emissions. Forests consist of long-lived organisms (trees) that can store carbon for centuries or more. However, trees have finite lifespans, and disturbances such as wildfire, insect and disease outbreaks, and drought can hasten tree mortality or reduce tree growth, thereby slowing carbon sequestration, driving carbon emissions, and reducing forest carbon storage in stable pools, particularly the live and standing dead portions that are counted in many carbon offset programs. Many forests have natural disturbance regimes, but climate change and human activities disrupt the frequency and severity of disturbances in ways that are likely to have consequences for the long-term stability of forest carbon. To minimize negative effects and maximize resilience of forest carbon, disturbance risks must be accounted for in carbon offset protocols, carbon management practices, and carbon mapping and modeling techniques. This requires detailed mapping and modeling of the quantities and distribution of forest carbon across the United States and hopefully one day globally; the frequency, severity, and timing of disturbances; the mechanisms by which disturbances affect carbon storage; and how climate change may alter each of these elements. Several tools (e.g. fire spread models, imputed forest inventory models, and forest growth simulators) exist to address one or more of the aforementioned items and can help inform management strategies that reduce forest carbon risk, maintain long-term stability of forest carbon, and further explore challenges, uncertainties, and opportunities for evaluating the continued potential of, and threats to, forests as viable mechanisms for forest carbon storage, including carbon offsets. A growing collective body of research and technological improvements have advanced the science, but we highlight and discuss key limitations, uncertainties, and gaps that remain.

Keywords Bark beetles, Carbon risk, Climate change, Disease, Drought, Forest management, Fuels, Future, Insect, Wildfire

*Correspondence:
Alex W. Dye
alex.dye@oregonstate.edu
Full list of author information is available at the end of the article

Background

Forests contain a large proportion of Earth's terrestrial surface carbon storage, with continued potential for carbon sequestration over the coming decades to centuries [1, 2]. Forests cover over a third (~310 million ha) of the United States' land area, of which ~58% are privately owned [3]. Maintaining or increasing forest carbon on these lands is a critical component of some nature-based climate solutions, including forest carbon offset projects. Various carbon offset programs have been developed to credit landowners for managing forests to maintain and increase long-term carbon storage, usually focusing on the live tree and standing dead carbon pools because of the potential to broadly manage these portions for long-term carbon stability [4, 5]. Because forests naturally have continual carbon exchanges with the atmosphere, offset programs try to account for these exchanges to achieve carbon stability on the order of decades and up to a century into the future [6].

Forest carbon storage is a critical ecosystem service, and because of this, offset programs can incentivize carbon stability by issuing credits to landowners to create and maintain additional forest carbon [7–9]. Losses of previously credited carbon can sometimes be compensated either by the project owner (for 'intentional' losses) or an insurance buffer pool of credits held by the issuing agency (for 'unintentional' losses). These guarantees against carbon loss from projects may be compromised by events that cause larger than anticipated carbon emissions. These events, called reversals, are expected to happen periodically as a consequence of natural disturbance regimes endemic to a particular ecosystem [10–13].

Carbon projects exist within a framework wherein there are initial stocks, management that can impact those stocks and the disturbance risks for a finite period of time, and realized disturbances, all which feed back to the initial carbon in the next time step (Fig. 1). Initial carbon stocks represent the carbon before a disturbance or at the start of a year or period of analysis. Potential disturbances, including wildfires, insects and disease outbreaks, and drought, change the composition of initial forest carbon and alter future risk from wildfire [14]. The intensity and likelihood of the disturbances and the susceptibility of forest carbon can be modulated by management actions. Management actions have an immediate, known impact on carbon stocks, and alter the risk of various types of disturbance. Disturbances can be understood through a risk framework, where carbon stocks are impacted based on likelihood of occurrence, the conditional intensity, and conditional severity. More broadly, large enough fluctuations in forest carbon may feedback into climate change, which also impacts disturbance risk.

Wildfire, insect and disease outbreaks, and drought are among the primary natural processes that reduce carbon in forests, but anthropogenic climate change has and will continue to alter the frequency, severity, and timing of disturbances beyond what has occurred historically, affecting carbon stocks at broad spatial and temporal scales (Fig. 1) and challenging carbon storage and policy goals [15–18]. Understanding the evolving

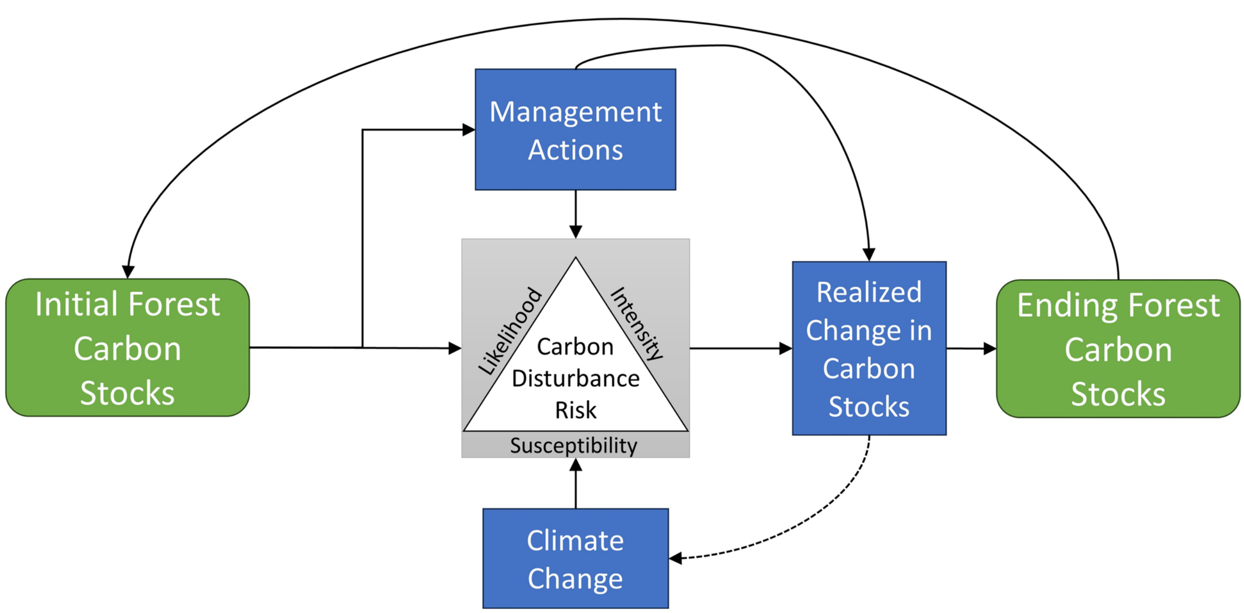

Fig. 1 Conceptual cycle of forest carbon risk as modified by disturbances, management actions, and climate change

Table 1 Natural disturbances and their major effects on forest carbon, expected influences of climate change, and management strategies for carbon risk reduction

	Effects on forest carbon	**Climate change effects**	**Carbon risk reduction**
Wildfire	High-severity fire = high tree mortality and carbon loss; low-severity = low tree mortality and carbon loss; long term impacts dependent on severity	Increased fire severity, size, and frequency in many places	Scale, timing, design, and placement of fuel treatments
Insects & disease	Reduced growth and carbon uptake, often leading to tree mortality	Generally, higher susceptibility of hosts to infestation/infection, and for many agents, more favorable environments	Managing for more resistant (increased host defense) and resilient (increased host adaptability) forest structures and compositions
Drought	Reduced growth and carbon uptake, tree mortality when drought is outside a species' adaptive capacity	Higher intensity and frequency	Managing for more resistant (increased drought defense) and resilient (increased drought adaptability) forest structures and compositions; incorporating changes in species envelopes during management
Heat waves	Reduced growth and carbon uptake, and mortality, when heat exceeds a species' adaptive capacity	Warmer temperatures increase likelihood of heat extremes	Managing for more resistant (increased heat defense) and resilient (increased heat adaptability) forest structures and compositions; incorporating changes in species envelopes during management
Wind	Tree mortality via windthrow or breakage	Higher intensity of hurricanes, but no change in frequency; other storm types may become more frequent and/or severe	Alter forest structure to prevent windthrow; reduce forest edges

dynamics of disturbance regimes is critical for maintaining carbon permanence (e.g. ensuring carbon projects are viable over their lifetimes of 20–100 years) because deviations from natural disturbance regimes influence whether forest carbon can in fact remain a viable offset option. However, many offset standards do not incorporate all risks to forest carbon stocks associated with climate change and disturbance events [19, 20]. Climate change has already introduced new uncertainties by rapidly driving ecosystems toward warmer, and in some places drier, futures with greater moisture stress, which likely will include non-stationarity in disturbance regimes that make it more difficult to assess forest carbon risk [17, 21, 22]. With sufficient knowledge of how climate change affects disturbances and forests, preventative management practices or treatments may be implemented to reduce the likelihood of carbon reversals and loss of forest resilience in the future.

For a forest carbon offset protocol to reasonably account for climate change and disturbance, it is necessary to map and model multiple data layers: the distribution of forest carbon across the country; the frequency, severity, and timing of possible future disturbances; the mechanisms by which disturbances may affect carbon storage; and how climate change may alter each of these elements (Fig. 1). In the United States, there is a growing body of scientific literature, tools, and methods that consider one or more of the aforementioned elements, but rarely all of them, and future climate change effects are often not sufficiently considered.

In this review, we first summarize current literature on the general effects of disturbance on forests and forest carbon in the United States, followed by the general effects of climate change on each disturbance type (Table 1). In ecosystems that have evolved with high-frequency, low-severity disturbance regimes such as wildfire (i.e., fire-adapted forest ecosystems), there are many potential benefits from shifting carbon from ephemeral pools (i.e. duff, litter, and small diameter ladder fuels) into more stable pools (i.e., large aboveground live and dead trees), including reduced temporal variability, reduced wildfire severity, and reduced risk to infrastructure from wildfire. To better understand these interactions, we explore how carbon in different pools affects disturbance risk and carbon stability. Next, we identify and discuss several of the primary tools and datasets that have resulted in significant advances in assessing forest carbon risk to disturbance for the United States. Lastly, we review how data from some of these tools have been utilized to develop management practices for creating ecosystem resilience, with co-benefits of building forest carbon stability to climate change.

Main text

Types of disturbances

Wildfires

How does wildfire affect forest carbon? Wildfires emit existing forest carbon into the atmosphere and affect the capacity to sequester new carbon as well as the rate at which it is sequestered (Fig. 2a). Fire effects on forest carbon may be immediate, through combustion, or may occur over many years through mortality, as dead trees decompose and/or surviving trees exhibit reduced growth rates. This process has occurred for millennia, often tied to human activity [23], and wildfire has been one of the most important contributors to interannual forest carbon variability in North America [10, 24]. Since 1990, wildfire carbon emissions have increased in the United States; however, over the same period, forest growth has offset these emissions, resulting in a net carbon sink in the total forest sector [25]. In Oregon, the 200,000 ha Biscuit Fire in 2002 is estimated to have released 17–22 Mg C ha^{-1}, 16 times the pre-fire annual net ecosystem production (NEP) for this region and negating nearly 50% of the annual total net biome production (NBP) for the entire state [26]. Fire may damage living trees, contributing to delayed mortality from other disturbances [27, 28] or may result in extended periods of post-fire carbon loss due to decomposition of standing or fallen dead trees [29]. By definition, low-severity fires result in less tree mortality than high-severity fires, but still release carbon through combustion of duff, litter, and small trees (e.g. saplings), which are usually not incorporated in carbon accounting protocols but may significantly alter future fire risk and intensity [30].

Large quantities of total aboveground carbon could be lost through combustion and mortality during high-severity wildfire; for example, up to 85% loss was documented following California's 2013 Rim Fire [31]. Additionally, compounding disturbances such as drought, heat, insects, and diseases that result in reduced fuel moisture content or mortality can accelerate carbon loss during subsequent wildfire [32]. During most wildfires, the majority of combustion and carbon loss occurs in smaller trees, brush, shrubs, fine woody debris, litter, and duff, with relatively low percentages of loss in larger tree size classes and coarse woody debris, although these losses increase with fire severity [32, 33] and higher severity fires may be more frequent with climate change [34–37]. For example, the area burned by high-severity wildfires has increased in the western United States by ~8X since 1985 [38], partly due to warmer, drier conditions attributed to climate change. While wildfires themselves are a relatively small proportion of the current total anthropogenic emissions in the United States (between 4–6% during the early 2000s [39]), in

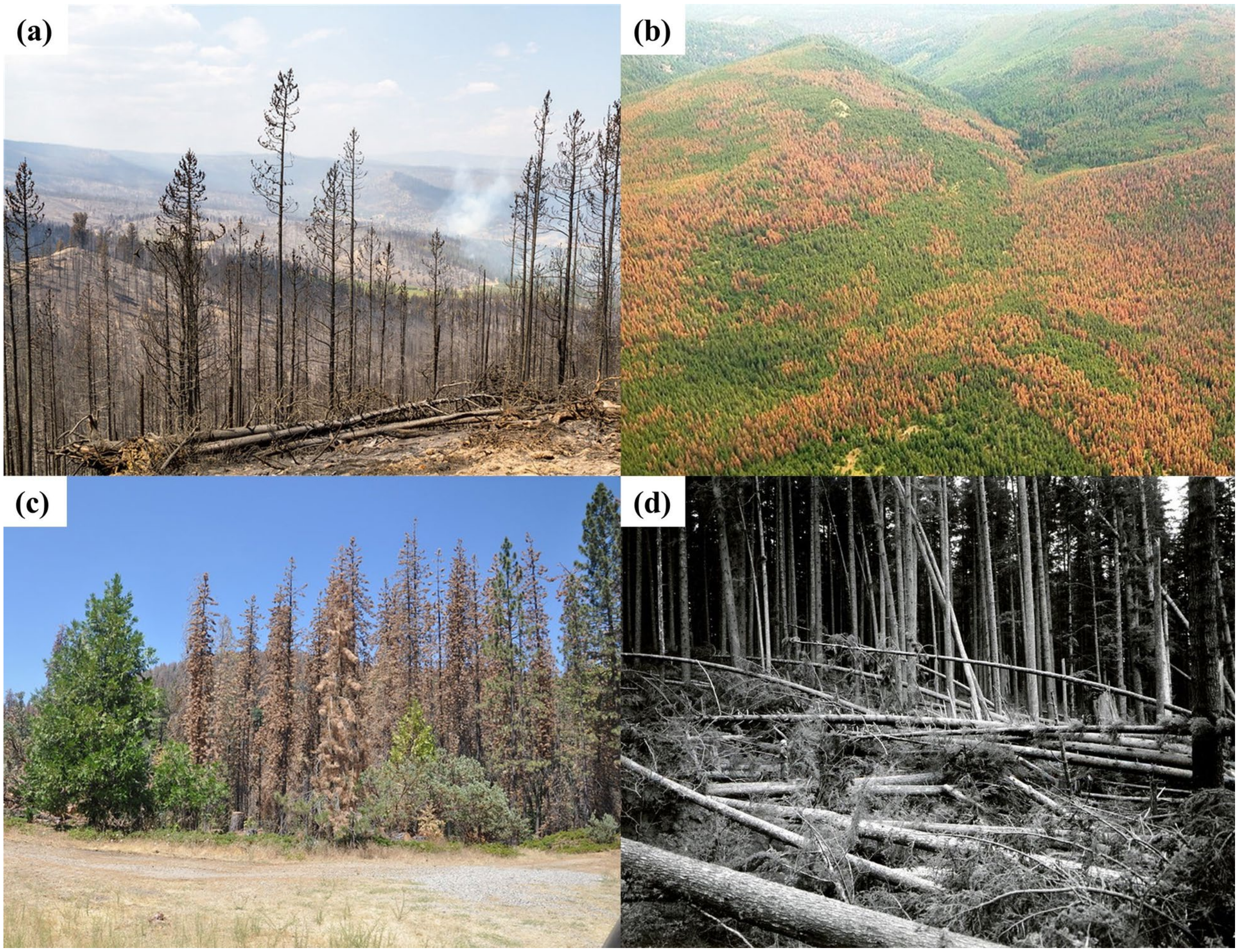

Fig. 2 Examples of four disturbance categories. **a** Wildfire: a post-fire landscape on the Shoshone National Forest, Wyoming [287], **b** insects and disease outbreaks: pine mortality in the Blue Mountains, Oregon [288], **c** drought: dead trees on the Sierra National Forest, California [289], and **d** other localized disturbances, including but not limited to windthrow and hurricanes: windthrow from the 1962 Columbus Day Windstorm in Otis, Oregon [290]

the context of leveraging carbon in forests as a sink, this trend towards increasing high-severity fire is an important piece of the puzzle. In instances with substantial large tree mortality, the remaining dead standing carbon stores of un-consumed large trees take years to decades to decompose and release their carbon, allowing time for partial replacement carbon growth from successional trees in stands where regrowth occurs [40].

Post-fire total carbon recovery generally occurs sooner after low-severity wildfires than high-severity wildfires, but rates vary widely depending on forest type [41–43]. For example, in eastern Oregon, live aboveground carbon lost to high-severity wildfires was over 6X greater than low-severity fires in ponderosa pine (*Pinus ponderosa*) forests, and nearly 3X greater than in mixed-conifer forests [44]. Over the twentieth century, fire suppression policies and other factors have led to increased fuel loads in many fire-adapted forests, increasing susceptibility to high-severity wildfire and subsequent risks to carbon stability [45]. Across the West, tree mortality and carbon sequestration reductions of surviving trees were significantly higher following wildfires in high-severity/low-frequency fire regimes than in low-severity/high-frequency fire regimes [46]. With climate change, some forests will struggle to regenerate trees post-fire, especially following high-severity fires at large spatial scales and in more arid locations [47].

How does climate change affect wildfire? Climate is a primary driver of wildfire activity with cascading consequences for forest carbon [48]. For thousands of years, wildfire has increased in tandem with periods of high temperatures, e.g., during the Medieval Climate Anomaly [49–51]. The role of humans in managing and altering natural fire cycles is compounding the climate change effects [52]. For example, prior to Euro-American colonization, cultural burning by Native Americans was widespread across the United States, creating fire regimes that were not fully dependent on climate [53–57]. Over the past century, fire suppression policies in the United States have led to a buildup of fuels, priming broad expanses of the landscape to burn under extreme weather conditions when fires cannot be suppressed [58]. Beginning in the late twentieth century, increasing temperature and aridity [23, 59–61] have created more frequent extreme fire-weather conditions, leading to more very large wildfires [62, 63] and area burned at high severity [38, 64], although sufficient fuels are necessary for this pattern to continue.

In the western United States, anthropogenic warming between 1984–2015 has been estimated to explain 45% of total forest area burned [65] and contributed to the increase of very large wildfires [66–68]. Eighteen of the 20 largest wildfires in California history have occurred since 2000, with the five largest occurring since 2018 [69]. Many of the largest wildfires in recent history are linked to climate extremes, including those in the infamous 2020 fire season in the West [70–73]. Over the twentieth century, lengthening of the fire season led to increased wildfire activity [74–76], including increased fire severity [35, 77–80].

Climate change is expected to continue to alter fire regimes into the future. The fire season is projected to increase by as many as 58 days in Southern California by the end of the twenty-first century [81], extending much later into the fall [82–84]. In the Northern Rockies, wildfires are projected to occur more frequently in spring and fall, and intensify in the summer by the mid-twenty-first century [85]. In the Southern Rockies, similar trends are projected, increasing the total annual area burned [86]. In the Northwest, future warming and drying is expected to create more severe fire-weather conditions [87–90], and increase the area burned [91–93], an emerging trend already observed in recent decades. These regional trends are confirmed by multiple national-scale studies [62, 63, 83, 89, 94–96].

In Alaska, area burned in the last half of the twentieth century has been strongly tied to climate [97–99]. These linkages are projected to continue in the future [100, 101]. For example, using a process-based ecosystem model driven with future climate, Balshi et al. [102] reported that by the end of the twenty-first century, wildfire in North American boreal forests could increase carbon emissions from these forests by 4.4X the contemporary rate, and using a dynamic global vegetation model, Bachelet et al. [103] found that increases in wildfire could eventually transition Alaska from a net carbon sink to a net carbon source by the end of the twenty-first century.

In the Southeast, projections for the mid-twenty-first century suggest the fire season will lengthen by 2–3 months, and summertime fire danger, as measured by the Keetch–Byran Drought Index (KBDI), will increase by 40% [104]. Although higher KBDI is primarily driven by increasing temperatures in the Southeast [105], other factors may be important in some subregions [106–108]. For the Upper Midwest and Northeast, Kerr et al. [109] suggested that the maximum period of consecutive days exceeding high-fire danger thresholds (95th percentile Canadian Fire Weather Index) will double by 2100, with the onset of peak fire season beginning in early spring. Some of the highest increases in wildfire probability in the United States are projected to occur in the Upper Midwest and Northeast, doubling by 2100, and are consistent with projections of rising burned areas in parts of the East [94, 95]. Determining the effects of climate change on wildfires has been difficult in areas of the United States where wildfires were historically rare (e.g., Northeast), and where forests represent a small portion of the landscape (e.g., agricultural regions).

For the Hawaiian Islands, wildfire has been a rare occurrence historically, moderated primarily by human activity and secondarily by climate [110, 111]. Recent influxes of non-native vegetation, particularly invasive grasses, have altered the islands' natural fire regimes, increasing fire frequency [112]. Linking climate and wildfire in Hawaii is difficult due to the extreme microclimate gradients in the islands, but lack of precipitation is generally the main climatological driver of wildfire in Hawaii [113, 114]. Resolving these climate connections will be critical to identify wildfire risks to forest carbon in Hawaii [115, 116].

Insects and diseases

How do insects and diseases affect forest carbon? Insects and diseases alter forest function, structure, and composition in complex ways, by regulating primary production, nutrient cycling, stand succession, and the abundance of associated plants and animals [117]. Insects and diseases can also affect other disturbances, such as wildfire. As with wildfire, forest carbon loss occurs due to both tree mortality and sublethal infestation (Fig. 2b; [16]). Defoliation can reduce the capacity of trees to sequester carbon for years after infestation [118–122].

Many insects and diseases pose significant risks to forest carbon in the United States (Table 2). In the West, major tree losses have occurred during the twenty-first century, with affected areas sometimes exceeding burned areas [123, 124]. As much as 15% of total forest cover in the United States is affected annually by insects and diseases [125]. Diseases are often more diffuse than insect outbreaks but affect large areas [126, 127]. During the

Table 2 Examples of forest insects with consequences for forest carbon stability in the United States

Region	Agent	Host tree	References
East	Emerald ash borer (*Agrilus planipennis*)	*Fraxinus* spp.	[292]
East	Hemlock woolly adelgid (*Adelges tsugae*)	*Tsuga* spp.	[152, 154, 293, 294]
East	Balsam woolly adelgid (*Adelges piceae*)	*Abies* spp.	[295]
East	Spruce budworm (*Choristoneura* spp.)	*Picea* spp.	[296, 297]
East	Spongy moth (*Lymantria dispar dispar*)	Most hardwoods	[298]
East	Southern pine beetle (*Dendroctonus frontalis*)	*Pinus* spp.	[299, 300]
West	Mountain pine beetle (*Dendroctonus ponderosae*)	*Pinus* spp.	[129, 228, 301–303]
West	Spruce beetle (*Dendroctonus rufipennis*)	*Picea* spp.	[304]
West	Other bark beetles	Most conifers	[124, 130, 137, 305, 306]
West	Western spruce budworm (*Choristoneura occidentalis*)	*Picea* spp.	[120]
West	Douglas-fir tussock moth (*Orgyia pseudotsugata*)	*Pseudotsuga menziesii*	[307]
West	Balsam woolly adelgid (*Adelges piceae*)	*Abies* spp.	[308, 309]
West	Forest tent caterpillar (*Malacosoma disstria*)	Most hardwoods	[145]

Region denotes primary area of impact, east or west of the Rocky Mountains

early twenty-first century in western North America, severe outbreaks of mountain pine beetle (*Dendroctonus ponderosae*) [128] caused some forests to switch from carbon sinks to major carbon sources over just 6 years, with multiple decades predicted before full recovery [129]. Since 2000, > 27 million ha have been impacted by mountain pine beetle, partly driven by climate change [128]. Warming allowed mountain pine beetles to erupt at elevations and latitudes where, previously, cold winters killed most brood within host trees [130]. In the West, the amount of carbon in trees killed by bark beetle outbreaks during 1997–2010 was similar to that in trees killed by wildfire, ~ 4.5% of the total carbon in trees in the region [124]. Across the United States, forests recently impacted by insects and diseases sequestered 69% and 28% less total forest carbon, respectively, than did similar unimpacted forests [122]. Using predictive models based on forest inventory data, Anderegg et al. [96] documented that insect-driven mortality risk is highest in the Rocky Mountains, Southwest, and Southeast, and is comparable to observed mortality from wildfires in these regions. Insect-driven forest carbon risk is projected to continue to increase throughout the twenty-first century, particularly in the Rocky Mountains, Sierra Nevada, and the Upper Midwest, but at a much lower rate than wildfire risk, which increases strongly across the entire United States [96].

Disease impacts have also been significant but more difficult to quantify. Beech bark disease has substantially reduced the growth of American beech (*Fagus grandifolia*) in the Northeast, decreasing live tree carbon production by 11% in Maine alone [131]. Root diseases are persistent in the northern Rocky Mountains, reducing live tree carbon as much as wildfire and more than harvesting or insect outbreaks [132].

The body of literature on the effects of invasive insects and diseases on forest carbon stocks is growing [120]. More than 450 non-native forest insects and pathogens have been introduced into natural areas of the United States, of which > 83 are invasive and cause significant impacts [133]. The rates of new introductions and establishments are high but projecting future impacts is difficult. Under current and projected import patterns, an average of two invasive forest insects are expected to be established in the United States each year, and an economically important forest insect pest is expected to be established every 5–6 years [134]. Historically, impacts of invasive insects and diseases have been much greater in the East than in the West. For example, the chestnut blight and Dutch elm disease are among a few disturbances that have threatened the existence of entire tree genera in the East.

How does climate change affect insects and diseases? Broad generalizations of the effects of climate change on insects and diseases are difficult to make due to the complexity of the life history traits involved among species. Climate change may increase susceptibility to insects and disease by two primary mechanisms: (1) warming driving range expansions in areas historically below temperature thresholds for survival, reducing overwintering mortality, and increasing phenology and voltinism [130, 135–137]; and (2) warming and drought compromising the defense mechanisms of otherwise vigorous trees, leading to higher risks of tree mortality from insects and diseases [16, 138–141]. In the West, warming is expected to expand the range of important insects. For example, future warming is projected to favor the growth of mountain pine beetle populations at higher latitudes and higher elevations [130, 142] but may be tempered by disrupted seasonality and fractional voltinism, both maladaptive to mountain pine beetle [143]. Furthermore, areas heavily impacted by mountain pine beetle outbreaks during the early twenty-first century are unlikely to experience outbreaks for decades because suitable host trees are depleted. In Alaska, warming and late-summer droughts have been positively correlated with spruce beetle (*Dendroctonus rufipennis*) outbreaks [144], suggesting levels of tree mortality attributed to spruce beetle are likely to increase in the future.

Defoliator responses to warming and drought are variable [139] and include important indirect effects mediated through changes in host tree physiology, primarily leaf chemistry and palatability [139]. Some insect fungal pathogens are important regulators of defoliator populations in the United States and are expected to be negatively affected by drought (e.g., *Entomophaga maimaiga*, which causes extensive epizootics in spongy moth (*Lymantria dispar dispar*) in the East). Hotter and drier conditions have been positively correlated with increased levels of tree mortality from defoliators [145, 146].

Fungal pathogens are sensitive to the timing and quantity of precipitation, ambient temperature, relative humidity, and other factors that influence leaf-surface or soil-moisture content. Some tree diseases that require moist conditions are expected to be negatively affected by climate change [147]. For example, hotter and drier conditions in the Southwest are expected to reduce white pine blister rust infections, but infections may increase where conditions become warmer and wetter [148, 149]. Other tree diseases (e.g., Armillaria root disease) are indirectly affected by climate change through increases in host stress, suggesting warming and drought may increase epizootics [147]. Forest diseases are expected to become more frequent and severe with climate change, but the magnitude of that change is both uncertain and varies by disease and ecosystem. One estimate suggests that by the end of the twenty-first century, the rate of tree mortality due to insects and diseases in the United States will increase by as much as 1.7X, which is a fraction of the 4–14X increase in tree mortality projected due to wildfire [96]. One study projects a 2X (insects) and 3x (diseases) increase in tree mortality rates for the West [150]. In the East, insects will remain one of the most impactful disturbances in the future [151], where summer warming and milder winters are expected to facilitate northward range expansions of some cold-limited insects like the hemlock woolly adelgid (*Adelges tsugae*) and southern pine beetle (*Dendroctonus frontalis*) [148, 152, 153]. However, exact climate change effects will be complicated by each insect's tolerances of climate and strategies for propagation, insect management practices, and regeneration dynamics in a warmer, drier climate, and natural selection of hosts for resistance to certain insects. For example, Albani et al. [154] projected that expansion of hemlock woolly adelgid would result in a nearly 13% reduction in carbon sequestration in the first third of the twenty-first century, but by end of the twenty-first century these same forests may experience a nearly 20% gain in carbon sequestration due to forest regeneration of climate-adapted tree species. However, this assumes that the insect outbreak's effects do not exceed the carrying capacity of an ecosystem, and that climate change-adapted trees are able to regenerate and replace trees lost to hemlock woolly adelgid.

Overall, climate change likely creates conditions more favorable for many, but not all, insect and disease and pathogen species. Many complex interactions occur among insects and diseases, their host trees, and other community associates that are directly and indirectly influenced by climate, making robust quantitative projections of future carbon risks to insects and diseases difficult [96, 120, 147, 148]. Relevant human activities are particularly multifaceted and difficult to project—including domestic and international trade, global economic markets and commodity pathways, and human population densities and travel—which all influence insect and pathogen introductions and establishments in the United States [155, 156].

Droughts

How does drought affect forest carbon? Drought reduces tree carbon uptake and in some cases results in large tree mortality events (Fig. 2c; [17, 157–163]). Many tree species have some level of adaptive capacity to drought. Some tree species have deep rooting, stomatal control, and leaf shedding that allow them to tolerate drought better than other tree species. Despite this, drought has been identified as the largest disturbance driving primary productivity declines globally [164]. In the United States, the effects of drought on forest carbon are most pronounced in the West, though effects in the East are substantial [165]. The 2011–2015 drought in California killed an estimated 140 million trees, tipping the carbon balance of the state to a net carbon source of −600 Tg CO_2 during 2001–2015 [166]. The projected carbon stocks of California ponderosa pines may not return to levels observed prior to the 2011–2015 drought due to future warming, droughts, and western pine beetle (*Dendroctonus brevicomis*) outbreaks [18], although those prior levels may be inflated due to widespread fire suppression [167]. Similarly, drought and high vapor pressure deficit are strongly correlated with rising tree mortality levels in Alaska [168]. Based on forest inventory data, drought and warming are key drivers of the decline of half of the most abundant tree species in the western United States over 2001–2018 [169].

How does climate change affect drought? Climate change is projected to bring more frequent and severe droughts in many regions of the world, and the western United States is frequently identified as a hotspot for increasing droughts [170, 171]. Warming exacerbates drought effects (termed 'hot droughts' or 'climate change-type droughts'; [157, 172]), elevating their lethality to trees and reducing forest carbon stocks. However, projecting changes to forest carbon due to drought remains a major challenge [22, 173]. Ultimately, (1) the effects of drought on forest carbon in the United States during the twenty-first century will be large, and (2) many current models likely underestimate drought stress and associated levels of tree mortality due to drought [22, 96].

Drought impacts on forest carbon stocks in the West are substantial [170, 174–177], and likely to increase in the future. Elevated temperatures increase soil evaporation, thereby reducing soil moisture available to plants, and higher vapor pressure deficit can result in greater transpiration in some plants, where stomatal conductance is a non-linear function of temperature [170, 178]. These climate effects have been widely documented, but the complexity of species-specific plant responses to future climates remain challenging to model. Some experts argue the loss of forests to drought could be substantial across parts of the contiguous United States in the future [22, 96].

Other disturbances—windthrow, heat waves, hurricanes

How do these disturbances affect forest carbon? Forests in the United States also experience many other disturbances that affect carbon stocks, notably hurricanes, severe storms, and heat waves [11]. Particularly for the temperate mixed deciduous forests in the northeast and Midwest, natural cycles of localized disturbances such as ice storms [179] and windthrow [180, 181] are historically more common than wildfire [182], and help create the structural complexity and species/age diversity that are important to maintaining stable carbon storage and sequestration over long time frames [183–185]. While these disturbances cause local and regionwide carbon losses (Fig. 2d), they typically have lower U.S.-wide impacts than wildfire, insect and disease outbreaks, and drought [11, 17, 124]. For example, Hurricane Katrina damaged or killed 385 Tg CO_2 equivalent trees in the Southeast [186]. Over many decades, the net impact of hurricanes to forests in the United States is likely a slight loss of live carbon [187]. Severe storms and windthrow may be important disturbance events, but few comprehensive studies document the carbon impacts of these events [11]. One critical recent advancement is a better understanding of how tropical cyclone regimes shape the ecology and evolution of tree species as the intensity and frequency of hurricanes affect forest structure and function [188]. Finally, while heat waves can decrease carbon uptake in forests [189], the most severe consequences occur when heat waves co-occur with severe drought (e.g., 'hot drought'), as described above. One exception is the heat wave in the Northwest in 2021 in which temperatures of > 40 °C caused substantial damage and mortality to trees [190].

How does climate change affect these disturbances? While the overall number of hurricanes is not projected to change substantially with climate change, the intensity of hurricanes is likely to increase [191]. Tree mortality is sensitive to wind speed [187] and more intense hurricanes may increase forest carbon losses, but the overall effect is likely much lower than for wildfire, insect and disease outbreaks, and drought. Of note, a recent meta-analysis found no consistent projected change in wind disturbance in North American forests [16]. A recent review of climate change in the United States indicates that climate change may increase the frequency and/or intensity of multiple storm types that affect forests, including hurricanes, atmospheric rivers, and thunderstorms [192].

Disturbance interactions

While often discussed here and elsewhere as distinct, isolated events, in reality disturbances can have compounding effects. For example, mortality induced by drought, disease, or insect infestation can create elevated quantities of large woody surface fuels with consequences for subsequent wildfire [193, 194]. Fire-induced mortality, which can sometimes be delayed several years post-fire, can affect subsequent fire severity [28] and also increase susceptibility to insect infestation in weakened surviving trees [195]. However, over longer time periods, low severity fires and prescribed fires can reduce tree density and promote resistance to future insect attack [196]. As increased temperatures and aridity widen the opportunity for disturbance-induced mortality at widespread spatial scales, dead trees add surface fuels at rates and quantities that exceed the natural range of variation [197]. Many operational fire behavior models are not yet capable of accurately predicting how these contributions to the large woody surface fuel pool will impact fire behavior [194, 198], but at minimum would likely increase the potential energy release during fires due to an influx of fuels with low fuel moisture content [32], combined with a sharp increase in live-tree density in small size classes as new growth regenerates post-disturbance. In the Sierra

Nevada, areas of higher fire severity and unpredictable fire behavior during the 2022 Creek Fire have been linked to widespread mortality from bark beetles and drought a decade prior [194, 197]. Generalizing disturbance interactions across forest types and conditions is challenging due to uncertainties in the timing, intensity, and spatial scale of mortality [199]. In fire regimes defined by surface fuel spread, mortality may increase fire severity in the short term by creating greater surface loads and fuel continuity [197], whereas in fire regimes defined by spread in canopy fuels, mortality may mimic thinning processes by decreasing the continuity of canopy fuels [199].

Mapping and managing forest carbon and forest carbon risk

As outlined in "Types of disturbances", disturbances have significant effects on forest carbon. Mapping risk to forest carbon requires two basic components: (1) mapping forest carbon quantities, and (2) mapping disturbance risk via likelihood, frequency, and/or intensity.

Mapping forest carbon on the landscape

Existing forest carbon has been mapped in two primary ways: continental synthesis of inventory data and remote sensing. In this section, we review the foremost methods and datasets that are available to map forest carbon in the United States. Since existing carbon offset programs in the U.S. account for carbon nationally, we focus on datasets that exist for the contiguous United States and/or Alaska and Hawaii and do not include regional datasets and models unless they can be readily scaled to national level.

The USDA Forest Service's Forest Inventory and Analysis (FIA) program manages the most comprehensive, large-scale measurements of forested plots in the United States. The program includes sites located across ownership boundaries for every state and includes plot measurements repeated every ~ 10 years. FIA data can be used to explore trends in forest conditions, growth, and disturbance across large areas. FIA divides the United States into 2402.8 ha hexagons [200], and varying numbers of plots within each hexagon are surveyed. FIA records for each tree on each plot include tree species, diameter, height, and status (live or dead). Carbon density can be calculated for each plot and interpolated to obtain estimates of forest carbon stocks (e.g., [201]). FIA has standardized statistical methods for making population estimates from the sparse plot locations in their network. Based on the rich information FIA provides, the Forest Service regularly updates forest carbon stock and flux estimates [25]. In this approach, a system of weights is applied to plot-level data to estimate conditions for the enclosing hexagon [202]. Another recent dataset used FIA data aggregated at the state level to develop maps of state and regional aboveground forest carbon stocks and carbon sequestration over the course of multiple FIA remeasurements [203].

While FIA is the most comprehensive measurement dataset, it does not provide a spatially contiguous ("wall-to-wall") map of forest attributes (including carbon) because only a small number of plots are measured within each hexagon. Computational methods have been devised to synthesize contiguous maps from site measurements (e.g., [204]). Although regional data sets are available (e.g., the LEMMA/GNN product for the West Coast [205]), currently TreeMap has the greatest spatial coverage, spanning the conterminous United States. TreeMap uses a machine learning algorithm called random forests to assign the most similar FIA plot to each pixel, producing a seamless tree-level forest model of the conterminous United States (CONUS) at 30 m resolution [206–208]. Because each pixel in TreeMap is linked to an FIA plot visit, each pixel has a list of trees and their measurements that can be used in allometric equations to estimate carbon stocks [209]. TreeMap is available for the western US for 2008 and for CONUS for 2014 and 2016. Currently TreeMap data sets are being created for Alaska and Hawaii.

The second method of measuring forest carbon uses remotely sensed data, which can result in spatially complete coverage and are often combined with FIA data (see [210] for a comprehensive review). The National Aeronautics and Space Administration (NASA) National Forest Carbon Monitoring System (NFCMS) produces carbon estimates through applications of the remote sensing-based Carnegie-Ames-Stanford Approach (CASA) carbon cycle process model [211] and has used FIA data to validate aboveground forest carbon estimates. NFCMS is available for CONUS at decadal intervals (1990, 2000, and 2010) at 30-m pixel resolution [212]. NFCMS builds upon previous mapping work of the National Biomass and Carbon Dataset from NASA and Woods Hole Research Center, which used similar methods to map a static baseline for the year 2000 [213]. NFCMS and FIA have been combined for additional downstream applications, including The Forest Carbon Map sponsored by the Trust for Public Land and American Forests, which provides a simple viewer for existing aboveground carbon storage and sequestration at larger scales (county, state, and/or watershed) for the purposes of conservation and carbon-cost accounting [214].

Mapping carbon risk to disturbance

Numerous methods and models map forest carbon risk to disturbance, and here we highlight noteworthy datasets that have been created across the United States (Table 3). We selected datasets and tools that are publicly available, peer-reviewed, and have attained widespread use by both the science and forest management communities. While these datasets have significant utility for use in carbon offset programs accounting for climate change, our review is not comprehensive of all tools.

Statistical disturbance models Statistical methods have been used to create future maps of disturbances and carbon stocks while accounting for climate change. For example, Anderegg et al. [96] used statistical models of wildfire, climate stress (primarily drought), and insect disturbance for the contiguous United States from 2000–2100, including the impacts of projected climate change from three different climate scenarios and six earth system models (ESMs). Wildfire effects were based on the Monitoring Trends in Burn Severity (MTBS) dataset [215], which maps burned area based on statistical analyses of observations from 1984–2018 and generally does not include fuel limitations or management effects, and climate stress and insect models of tree mortality from disturbance were constructed from FIA data from 2000–2018 in combination with satellite imagery. The models were cross-validated and tested against independent datasets. These models capture impacts of climate change, several climate scenarios, and multiple ESMs over the full 2000–2100 period (Fig. 3a). These datasets are publicly available, as is all the underlying code that created them, and a viewer and analysis tool was developed to visualize disturbance risks (https://carbonplan.org/research/forest-risks).

Similar to the above models, Wu et al. [22] used several different approaches to project forest carbon stocks across the contiguous United States using multiple ESMs from 2020–2100. Their study examined simplified mechanistic vegetation models, climate niche models of forest biomass by FIA Forest Group, and demographic models of forest growth and climate-sensitive disturbance developed by a previous study [96]. The demographic models showed the strongest agreement with historical biomass and disturbance trends and are expected to provide the most robust information about forest carbon risk. Despite substantial uncertainty and differences across models, there was consistency in some areas with regards to lower or higher climate risk across models [22]. All of these datasets are publicly available (https://wilkescenter.utah.edu/tools/us-forest-carbon-futures/).

Table 3 Major datasets and tools available for evaluating forest carbon risk to disturbances in the United States

Datasets and tools	Disturbance	Includes climate change	Ability to project?	Carbon-relevant metrics	Caveats
United States-wide statistical forest carbon models [22]	Fire, insect, drought	✓	✓	Forest carbon stocks 2020–2100; disturbance probabilities and carbon losses from disturbance	Coarse representation of climate-dependent disturbance and regrowth
United States-wide statistical disturbance models [94]	Fire, insect, drought	✓	✓	Burn area, tree mortality from insect and climate stress/drought	Does not model fire behavior or spread
TreeMap and FuelMap	Fire, insect, disease		✓	Tree diameter, height, species, and status	For fuels, relies on FIA transects in a limited subset of plots
Forest Vegetation Simulator (FVS)	Fire, insect, disease		✓	Tree volumes, biomass, density, canopy cover, harvest yields, fire effects	Does not model fire spread
LANDFIRE	Fire, insect, disease			Current state of landscape, disturbance locations	Does not represent full extent of insect and disease-related mortality
National Insect and Disease Risk and Hazard Mapping (NIDRM)	Insect, disease	✓	✓	Insect and disease risk	Uses outdated climate change scenarios; only near-term
USFS Insect and Disease Aerial Survey	Insect, disease			Area infested and carbon affected	Relies on human eyesight/judgement; low accuracy
FSim	Fire	check mark	✓	Burn probability, flame length/fire intensity	Fire spread equations insufficiently model novel fuel structures; does not project vegetation change
Interagency Fuel Treatment Decision Support System (IFTDSS)	Fire		✓	Conditional burn probability under static weather conditions, risk	Static weather; excludes low severity impacts

TreeMap and FuelMap data can be used with Forest Vegetation Simulator (FVS) to project tree growth and carbon storage

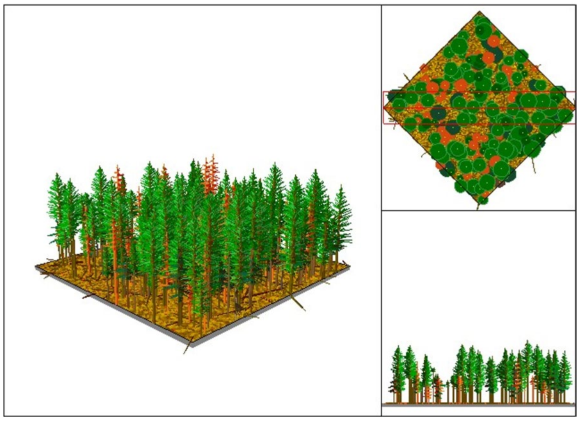

Fig. 3 Modeled visualization of a forest stand from Forest Vegetation Simulator (FVS) using the Stand Visualization Simulator module [217]. FVS is an individual tree growth model for forest stands that can simulate effects of disturbances on tree growth and forest carbon

Forest Vegetation Simulator (FVS) Forest Vegetation Simulator (FVS) is an individual tree growth model for forest stands that began as the growth and yield model Prognosis [216] and has since been expanded to include geographic variants that are specific to every region and many ecologically distinct areas in the continental United States and Alaska [217]. Several model extensions allow users to incorporate economics [218], climate change [219], and disturbances, such as wildfire [220], insects, and disease. The model simulates some stand dynamics (i.e., competition, fire spread, insect and disease spread), but because FVS is designed to be run at the stand level, there are no interactions among stands (Fig. 3).

The Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) estimates potential fire behaviors and simulates fire behavior and effects at a stand under specific weather, wind, and moisture conditions. FFE-FVS does not model landscape fire spread but can be combined with landscape spread models to estimate conditional wildfire effects. FFE-FVS can be used to estimate conditional fire behaviors without causing any impacts to the stand, which may be useful in exploring short-term changes in fire behavior due to treatments. FFE-FVS can also simulate the effect of fires on tree mortality and growth, carbon pools, and produce smoke and carbon emissions estimates. A submodel within FFE-FVS tracks carbon in seven pools: aboveground live trees, belowground live, belowground dead, standing dead, forest dead and downed wood, forest floor (duff and litter), and herbs and shrubs. It also calculates carbon emissions for simulated fires. Long-term carbon emissions from dying and dead trees are computed for subsequent cycles as well. The carbon submodel also tracks carbon stored in harvested wood products, including leakage.

In addition, FVS has nine insect and disease extensions, in which each extension affects tree growth and stand development [217]. Recent software updates have created compatibility problems, so only the dwarf mistletoe and root disease extensions are available today. Updates for the extensions for blister rust, bark beetle, Douglas-fir tussock moth, and western spruce budworm are underway.

FSim (Large Fire Simulator) FSim is a stochastic, spatially explicit fire behavior model that estimates the likelihood and intensity of wildfires across large regions. Representing the landscape as a grid, FSim simulates probability of ignition, fire spread, and behavior using thousands of hypothetical fire seasons (Fig. 4; [221]). FSim results for the United States at 270-m resolution for landscape conditions circa 2014 and 2020 are publicly available [222, 223]. Simulations have been used to evaluate the effects of climate change on wildfire [84, 85, 90] and the effects of an invasive species (annual grass) on wildfire behavior [224], among other applications. FSim can also be used for risk assessment using the Highly Valued Resources and Assets (HVRAs) concept, which combines the probability and intensity of burning with the susceptibility of the valued resource, such as forest carbon, to burn probability and intensity gradients [225]. The effects of fire can be positive or negative depending on the types of HVRAs and their response to different intensity levels (e.g., quantifying the effect on post-fire carbon due to low-, medium-, or high-severity fires). This risk assessment method can be applied at any spatial extent (e.g., national forest, counties, watersheds) by aggregating the relative importance of each HVRA within the area. However, modeled fire behavior in FSim, and other models based on the Rothermel fire spread equation [226], are insufficient to model novel fuel structures that may become more common with climate change. In particular, these models cannot simulate fire behavior under conditions where compounding disturbances create large quantities of dead large woody debris [194], nor under mass fire conditions where fires can generate unique local weather systems [198]. Models to capture these types of spread conditions are an active area of research. However, FSim is calibrated until fire spread produces a number of large fires and mean fire size within targets based on recently observed fires. In this way, despite the above limitation, model results are within observed parameters. FSim also does not project changes in vegetation due to climate change, a major area of uncertainty.

FuelMap FuelMap is a dataset that was built by imputing FIA measurements of litter, duff, and downed woody material (DWM) to a contiguous grid across the contigu-

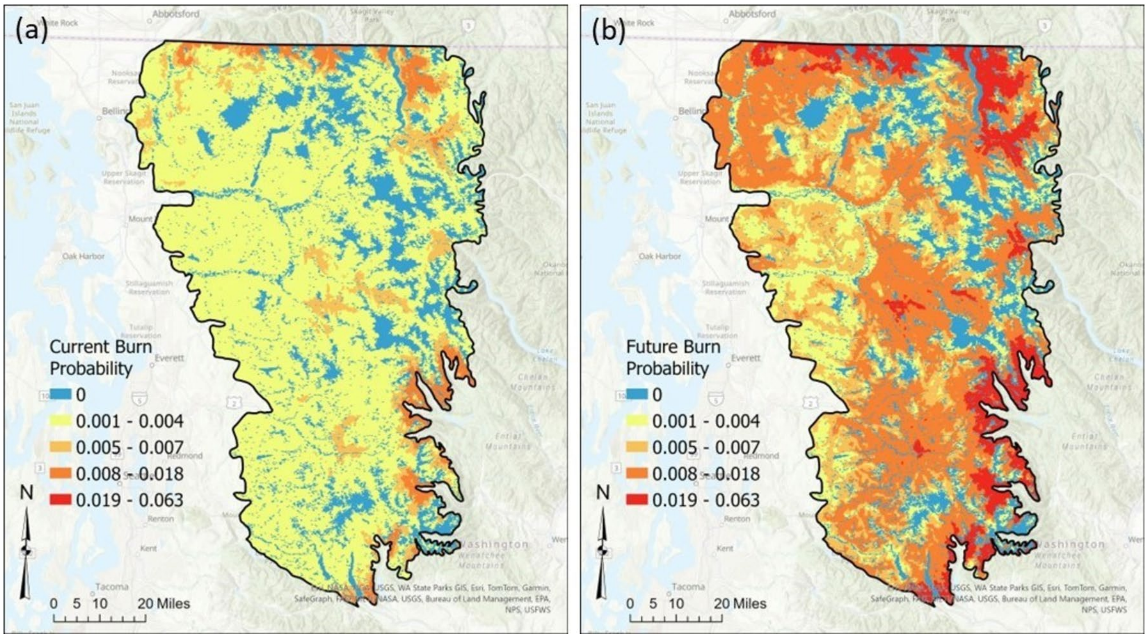

Fig. 4 The FSim Large Fire Simulator can project how future climate change may alter spatial fire probability patterns, which can then affect forest carbon. The two maps show contemporary, 1992–2020 (**a**) and mid-twenty-first century, 2035–2064 (**b**) annual burn probability at 270-m resolution for a landscape in Cascade Mountains of northern Washington State, United States, simulated by FSim [291]

ous United States [227]. FuelMap and TreeMap ("Mapping forest carbon on the landscape") are consistent with each other, both by input data and methods. Forest carbon response functions to fire have been successfully mapped across the contiguous United States at 30-m resolution using FuelMap and Tree Map (see "Mapping forest carbon on the landscape" above). In other words, the amount of carbon retained onsite and emitted during fire have been calculated in FVS across a set of six fire intensities for each FIA plot in FuelMap and TreeMap, meaning that these can now be mapped spatially across CONUS. These functions can be combined with the probability of fire at each of these six intensities that FSim outputs to estimate the risk of levels of tree mortality and carbon emissions (Fig. 5). FuelMap is reliant on FIA transects in a limited number of plots. This method can miss the impact of spatial heterogeneity of fuels on the landscape, so the resolution at which FuelMap should be used is still under investigation.

USFS Insect and Disease Survey The USDA Forest Service Insect and Disease Survey maps insect and disease activity annually across the United States. Surveyors in airplanes record damage to trees in polygons across forests, noting the disturbance agent, host tree species, and damage type and severity. Nationally consistent geospatial data sets are available back to 1997, and data sets for individual regions are available for earlier years. This dataset has been used to map bark beetle-caused tree mortality in the western United States [228] and associated carbon loss [121]. The accuracy of snag counts in this dataset was found to be 3–44% in two recent studies [229, 230].

National Insect and Disease Risk Map (NIDRM) NIDRM is a comprehensive nationwide assessment and database created by the USDA Forest Service of the potential hazard for tree mortality due to major forest insects and diseases. It summarizes landscape-level patterns of potential insect and disease activity and offers a science-based administrative planning tool for allocating pest-management resources. To capture spatial variations in forest health, NIDRM utilizes 186 insect and disease hazard models. The NIDRM products, compiled at a resolution of 240 m, support forest planning and enable forest-health hazard assessments at regional and national scales. These products can be used to identify the potential impacts of insects and pathogens on forests in the United States. The latest version of NIDRM (available at https://usfs.maps.arcgis.com/apps/MapTour/index.html?appid=ade657567ff445d5bb3aaa7d898d9fb9) was completed in 2018 for a 15-year assessment period (2013–2027) [231].

LANDFIRE disturbance layers LANDFIRE (LF) tracks annual landscape changes resulting from natural disturbances starting in 1999 and provides spatial vegetation and fuels layers for FSim, FuelMap, and FVS at 30-m res-

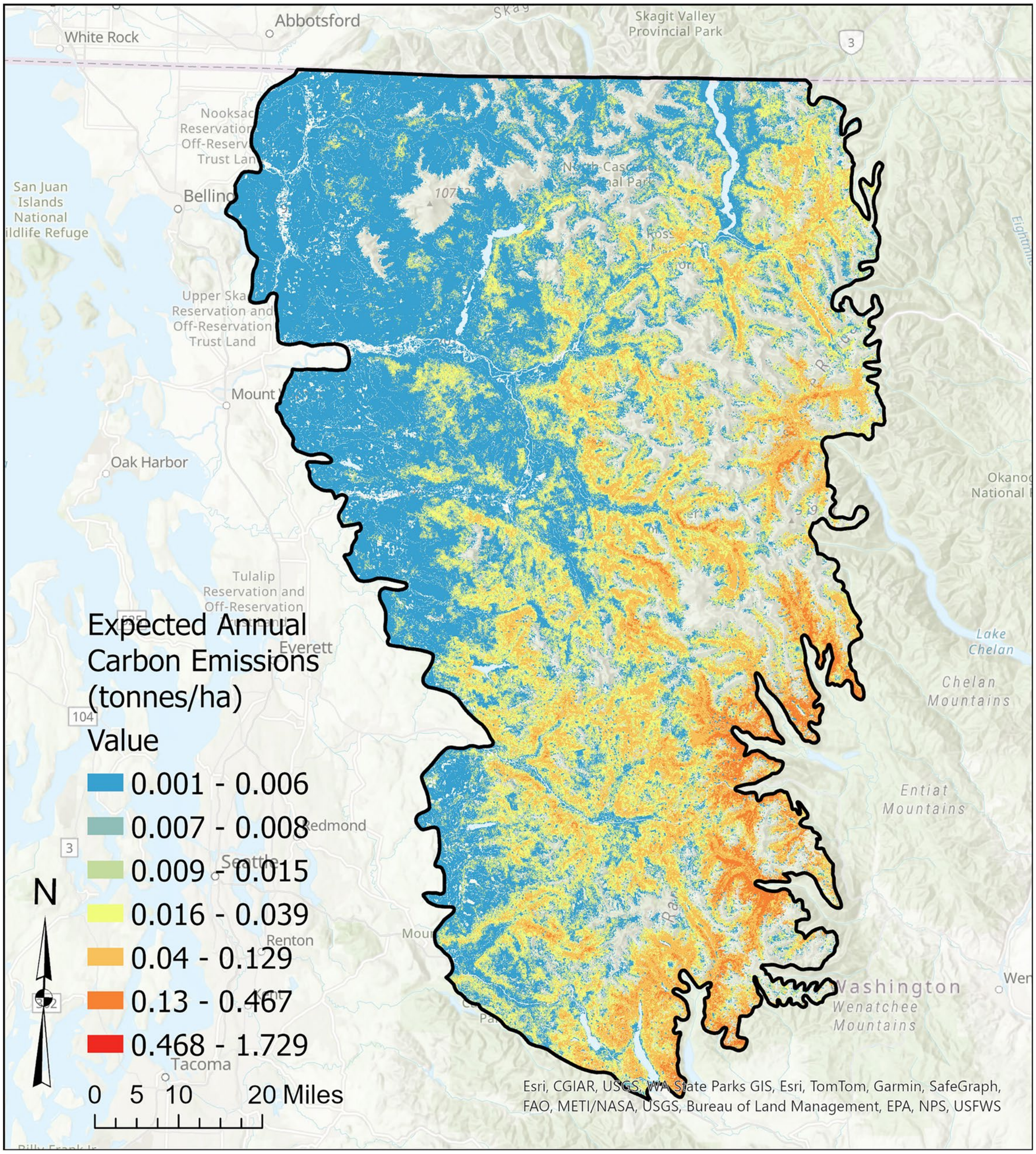

Fig. 5 A risk map derived from integrating TreeMap and FuelMap data with the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (e.g. Fig. 3) and FSim simulations (Fig. 4) for a landscape in the Cascade Mountains of northern Washington, United States [R. Houtman, unpublished data]

olution. The initial version of the dataset identified disturbances using diverse geospatial datasets to detect and categorize changes in vegetation cover, supplemented by Landsat-based composite images to assign disturbance severity between time steps using the Difference Normalized Burn Ratio (dNBR) [232]. In the second generation of the dataset (LF ReMap, or LF 2.0 [233]), multiple teams enhanced the accuracy of disturbance representation by incorporating Monitoring Trends in Burn Severity (MTBS) and the Rapid Assessment of Vegetation Condition after Wildfire (RAVG) datasets to identify burned areas. The disturbance dataset tracks wildfires > 1000 acres (404.7 ha) in the West and 500 acres (202.3 ha) in the East and can be used to monitor tree mortality from insects, diseases, and drought. Prior to 2020, the dataset incorporated only polygons reported through national datasets such as the USDA Forest Service Activity Tracking System (FACTS) database. The 2020 release of LANDFIRE [234] includes polygons reported by Insects and Disease Detection Survey (IDS), which may not capture the full extent of insect and disease-related tree mortality and are coarse resolution with low accuracy.

Interagency Fuel Treatment Decision Support System (IFTDSS) IFTDSS combines fire behavior and spread simulation models with an approachable user interface. IFTDSS models simple fire behavior and landscape burn probability under static weather from a set of random ignitions, and the expected fire spread from a user-defined ignition set. Treatments like thinning, mastication, and prescribed fire are simulated by altering input files representing the fuel model [235], canopy cover, canopy height, crown base height, and/or crown bulk density based on the intended effects of the treatment. The user then specifies a set of static weather and fuel moisture conditions to simulate fires, allowing for exploration of various scenarios. The primary purpose of IFTDSS is to aid land managers in fuel treatment planning by simulating the effects of fuel treatments on the landscape. This led to some simplifications of the model framework to make problems tractable. Because IFTDSS models fire spread under static weather, the risk model excludes impacts from lower severity fires. Fire model outputs are conditional based on the specified weather scenario. The same fire spread caveats apply to IFTDSS as to FSim due to the utilization of the same underlying Rothermel equations. In order to explore impacts to carbon, IFTDSS outputs would need to be combined with a tree-level carbon map such as TreeMap and FuelMap.

Strategies to manage forest carbon

In forests adapted to low- to moderate-severity wildfire, fuel reduction treatments are effective at reducing the intensity and severity of wildfire [236–240]. Fire intensity affects severity, which in turn affects the amount of carbon emitted during the fire (through combustion) and afterward (through decomposition), as well as the rate of carbon sequestration after wildfire [10]. If fuel treatments spatially coincide with a future wildfire, carbon emissions from the fire are often reduced [241]. However, the benefits of individual fuel treatments are uncertain because it is impossible to predict the specific locations of future wildfires. Furthermore, the effects of fuel treatments diminish over time, typically lasting 5–15 years in forest fuels. In one study, ~7% of the area treated was subsequently intersected by a wildfire during the effective life of the fuel treatment [242]. Also, potential reductions in carbon emissions must be balanced with initial losses occurring during fuel treatments, including carbon emissions by machinery and vehicles, and long-term effects on forest carbon fluxes. The choice of treatment type—mechanical, prescribed fire, or a combination of the two—significantly affects how much carbon is initially lost and how much is sequestered after treatment, as well as fire behavior in the event that a fire occurs [243, 244]. Treatments can be strategically applied in some forests, for example where high-severity wildfires may cause type-conversions to shrublands or grasslands, which have lower carbon storage capacities.

In the near-term (< 50 years) following landscape-level fuel reduction treatments, total forest carbon stocks are diminished, even when accounting for avoided carbon losses from subsequent wildfires [239, 245–248]. Rather than tracking just total carbon, an alternative approach tracks carbon stored in large aboveground live trees as the dominant stable carbon pool in the forest [249–251]. In mixed conifer forests in the Sierra Nevada, live tree carbon in both untreated and recently treated stands was substantially lower than that estimated for stands in a historical (1865) landscape, which had frequent fires [249]. Prescribed fire released 14.8 Mg C ha^{-1}, with pre-fire thinning increasing the average release by 70% and contributing 21.9–37.5 Mg C ha^{-1} in milling waste. All fuel treatments increased fire resistance, but treatments that included prescribed fire had lower torching and crowning potentials. Hurteau et al. [250] reported similar findings in ponderosa pine forests in the Southwest. While aboveground carbon was greater under the baseline fire-excluded treatment scenarios, higher potential for torching and crowning in untreated stands exposed the carbon stocks to greater risk.

Management regimes that reduce forest stand density and favor large, fire-resistant trees may increase the amount of stable carbon on the landscape. This equates to a potential risk reduction that can be quantified, or at least qualitatively categorized, by some of the mapping frameworks discussed in "Mapping carbon risk to disturbance". For example, using the LSim model to explore the effects of different treatment and wildfire management approaches over 60 years, Young and Ager [251] found that increasing the treatment area 5X over current treatment rates produced the most stable forest carbon on the landscape over time. Similarly, fuel treatment effects simulated over a 90-year period using LANDIS-II found that the no-treatment scenario has significantly greater overall carbon loss than treatment scenarios in frequent-fire forest types in the Sierra Nevada [252]. Specific ecosystems may attain a net benefit from treatments over decades to centuries. In another study, long-range carbon dynamics were modeled in conjunction with fuel treatments in the Pacific Northwest. The authors concluded that due to the low consumption of the majority of fuels and the limited duration of the effectiveness of fuel treatments, long-term carbon storage potential was reduced, even where high wildfire risk existed [253]. Balancing demands for carbon storage with demands for reducing wildfire severity would require fuel treatments to be implemented strategically throughout landscapes, rather than indiscriminately treating all stands [237].

All of the disturbance risks discussed here are also being impacted by climate change (Fig. 1). In that context, climate change is not only a driver for management designed to sequester carbon in light of disturbances, but also contributes to the design of silviculture treatments to mitigate direct impacts on specific forests. Managing for site-specific stand structures that include features such as high overstory compositional diversity [184], age diversity [254], spatial complexity [255], and trait diversity [256] are strategies that can increase forest resilience in the face of climate change, ideally while also continuing to maintain stable carbon stocks [185]. Treatments can be implemented that create climate-resilient forest structure traits while simultaneously maximizing carbon sequestration rates that create long-term stability. These strategies may also reduce the initial carbon stored onsite [257], demonstrating the value of understanding how treatments impact multiple overlapping priorities.

Although some fuel treatments may catalyze net carbon uptake over a long period, Wiechmann et al. [244] reported three of five types of fuel treatments resulted in a net loss in forest carbon 10 years after treatment in the Sierra Nevada. The two treatment types that recovered carbon over 10 years were a burn-only and understory thin-only. Conversely, over the long run, a combination of prescribed fire and mechanical thinning is most effective at reducing potential fire severity where it has been studied [238, 257]. Across 10 locations in the United States, Boerner et al. [243] found that mechanical treatments do reduce forest carbon significantly more than prescribed fire, although there was a greater increase in carbon sequestration after mechanical treatments.

Spatial relationships of fuel treatments and wildfires are important to consider. At the landscape level, effects from different treatment scenarios can be highly dependent on the spatial configuration of treatments and the percent of the landscape that is treated. Finney [258] and Ager et al. [259] reported that treating 10% of a forest landscape reduces expected losses of large trees by 70%. Treatments strategically placed in relation to how fire spreads are an effective technique for reducing fire exposure beyond the treatment area, including transmission to valued natural and cultural resources [260]. However, the dynamics of fire spread vary widely by forest type, and the design and implementation of fuel treatments that are sensitive to the ecology of an individual forest could help achieve desired objectives in forest resilience [45]. For example, Agee and Skinner [236] outlined both a methodology for restoration of fire-excluded dry forests (thin from below, reduce surface fuels, and retain the largest, oldest trees) and the characteristics of the ecosystem that determine the forests in which the application of these techniques should be prioritized (historically fire-frequent, low-severity, low-density forests). Management of longleaf pine (*Pinus palustris*) forests in the Southeast using the Stoddard-Neel method, a holistic management approach that was derived in the first half of the twentieth century, may require greater initial fuel reductions prior to the reintroduction of repeated prescribed fires [253]. Alternatively, in order to return forests to within the range of natural variation prior to fire suppression, management of forests that tend to burn at high intensity and high severity, such as jack pine (*Pinus banksiana*) and lodgepole pine (*Pinus contorta*) forests, could benefit from a patchy fuel landscape that allows for some amount of self-limiting high-severity fire. Eastern hardwoods may require an entirely different approach, with mechanical thinning being the primary activity used to meet restoration goals [261].

Management practices that lead to an unintended increased risk of high-severity wildfire (e.g., fire suppression and reductions in harvesting) have also led to an increased risk of insect and disease-induced tree mortality in certain forests [262]. Greater homogeneity can decrease overall resistance and resilience of a forest to biotic disturbances, while higher tree densities increase the number of available hosts for transmission of insects and diseases. Because of this, many of the same strategies that have been proposed for preserving forest carbon in the face of wildfire are also proposed for reducing levels of tree mortality attributed to some types of insects and diseases. In the Northern Rockies, Hood et al. [263] reported there was up to 50% host tree mortality after a mountain pine beetle infestation in dense, untreated stands; 39% mortality in stands that had been treated with prescribed fire; and almost no mortality in stands that were treated with mechanical thinning and prescribed fire. Low-severity fire has been shown to induce trees to fortify their resin ducts, thereby increasing resistance to future mountain pine beetle infestation [264]. In addition, thinning is widely regarded as an effective means for increasing resistance and resilience to several notable bark beetles, likely due to reductions in tree competition, increases in tree vigor, increases in tree spacing, and changes in microclimate that disrupt aggregation pheromone plumes [265–268]. The efficacy of thinning to reduce levels of bark beetle-induced tree mortality has even been demonstrated under extreme drought conditions (e.g., [269]). Although sanitation harvesting of infested trees has been proposed to alleviate tree mortality [270], these strategies are not as effective for reducing forest carbon risk since they run counter to the natural disturbance cycle of many forest types [271]. In eastern hemlock (*Tsuga canadensis*) forests, simulations show that allowing hemlock wooly adelgid to progress naturally through a stand (versus salvage scenarios) may result in the least impact to long-term carbon sequestration and net forest carbon [272].

As with fuel treatments for wildfire, forest thinning can sometimes serve as an effective strategy to reduce forest carbon risk from drought, bolstering forest resistance (the ability to sustain growth during drought) and resilience (the capacity to recover growth after drought) [273–277]. This is achieved by reducing tree densities, which is the primary driver of resource competition at the stand level [277], and by enhancing available growing space [278]. While the potential of thinning to reduce drought effects is widely acknowledged, its effectiveness varies substantially by context and ecosystem, and is influenced by various factors. Stands with lower tree densities, achieved through more substantial thinning practices, generally exhibit heightened resistance and resilience [279–281]. Results from a recent meta-analysis of stand density and tree mortality relationships in yellow pine (ponderosa pine and Jeffrey pine, *Pinus jeffreyi*) forests suggest that substantially lower stand densities are required to maintain adequate levels of resistance to bark beetles in contemporary forests compared to recent historic forests, due to the effects of warming and drought on forest structure and composition, including transitions from low-density, open and park-like forests to dense, second-growth forests [268]. Maintaining stands at such low densities may be required to promote high levels of resistance to drought and bark beetles in the future and represents a substantial change from current management prescriptions, of which the carbon consequences over time are largely unknown.

The benefits of thinning on carbon risk reduction, through reduced future disturbances, diminish with time since last treatment [281, 282]. Notably, the effects of thinning differ between broadleaves and conifers, although additional studies are needed [283]. Furthermore, the specific thinning method employed can yield contrasting outcomes. Thinning from above (involving removal of dominant and co-dominant trees) has the potential to minimize drought-induced growth reductions by reducing tree diameter, fostering a more intricate vertical structure that stratifies competition, while thinning from below (involving the removal of smaller diameter trees in lower canopy positions) may result in larger diameters and a monolayered structure, intensifying competition [284]. The type of thinning may also result in either complementary or opposing effects on different disturbances [285]. For example, thinning from below is often used in fuels reduction resulting in forests of large, older trees, which are typically more susceptible to bark beetles (e.g., [231]). Conversely, thinning from above can reduce susceptibility to bark beetles but may increase surface fuels resulting from harvest residuals.

Overall, our understanding of relationships between forest carbon risks and treatments that reduce tree damage and mortality from insects, diseases, and drought has some critical limitations, uncertainties, and gaps. These include: (1) a complete accounting of carbon emissions during treatments, (2) the likelihood of a tree mortality event (e.g., bark beetle outbreak) occurring during the period of time when treatments are effective, recognizing that efficacy declines with time since treatment, (3) the amount of landscape that needs to be treated to impart desired effects, and (4) the post-treatment rate of carbon uptake.

Conclusions

Forest carbon storage is a critical ecosystem service that is facing heightened risks as climate change facilitates larger and more severe wildfires, widespread insect and disease outbreaks, and more intense droughts in many forests of the United States [286]. To minimize negative effects and maximize resilience of forest carbon, these risks must be accounted for in carbon offset protocols, carbon management practices, and carbon mapping and modeling techniques. Many of the example tools discussed in "Mapping carbon risk to disturbance" demonstrate the significant conceptual challenges of combining all the interacting land surface, climate, and ecological processes that are needed to analyze forest carbon risk, a task that becomes increasingly complicated with climate change and over temporal trajectories. Additional challenges lie in scaling up modeling efforts and management techniques across the United States, and it is noteworthy that all the studies on managing forest carbon risk discussed in "Conclusions" have been executed on a local project or regional scale, not at a national scale. The latter requires consistent accounting across variations in forest types, disturbance trajectories, and resource availabilities. However, with the continuous advancement in scientific understanding and computational capabilities, the foundation now exists to scale up analyses that were previously only possible on a local scale. This represents the next critical step towards elevating forest carbon risk science to a level that can facilitate a better understanding of forest carbon risk from climate change and disturbance across the entire United States and create opportunities for strategic forest management directed at reducing those risks.

Acknowledgements
The findings and conclusions in this report are thoseof the author(s) and should not be construed to represent any official USDA or US Government determination or policy. CARB was instrumental in assisting with final editing and organization of the manuscript.

Author contributions
AD, RH, and PG designed, conducted, wrote, and performed final editing on the manuscript. WA, CF, JH, JK, KR, CS, and KY contributed intellectual material to the manuscript and assisted with review and editing throughout the process.

Funding
This research is supported by the State of California Air Resources Board (CARB), under Agreement No. 22ISD003, and the USDA Forest Service Western Wildland Environmental Threat Assessment Center and Rocky Mountain Research Station.

Availability of data and materials
No datasets were generated or analysed during the current study.

Declarations

Competing interests
The authors declare no competing interests.

Author details
[1]Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR, USA. [2]USDA Forest Service Rocky Mountain Research Station, Missoula Fire Sciences Lab, Missoula, MT, USA. [3]Department of Earth & Ocean Sciences, University of North Carolina at Wilmington, Wilmington, NC, USA. [4]Wilkes Center for Climate Science and Policy, University of Utah, Salt Lake City, UT, USA. [5]USDA Forest Service Pacific Southwest Research Station, Davis, CA, USA. [6]Department of Earth & Spatial Sciences, University of Idaho, Moscow, ID, USA. [7]USDA Forest Service Western Wildland Environmental Threat Assessment Center, Corvallis, OR, USA. [8]University of North Carolina at Wilmington, Wilmington, NC, USA.

Received: 12 February 2024 Accepted: 1 October 2024
Published online: 10 October 2024

References

1. Dixon RK, Solomon AM, Brown S, Houghton RA, Trexier MC, Wisniewski J. Carbon pools and flux of global forest ecosystems. Science. 1994;263:185–90.
2. Pan Y, Birdsey RA, Fang J, Houghton R, Kauppi PE, Kurz WA, et al. A large and persistent carbon sink in the world's forests. Science. 2011;333:988–93.
3. Oswalt SN, Smith WB, Miles PD, Pugh SA. Forest resources of the United States, 2017: a Technical Document Supporting the Forest Service 2020 RPA Assessment, WO-GTR-97. US Dep. Agric. For. Serv. Washington, DC; 2019.
4. Gifford L. "You can't value what you can't measure": a critical look at forest carbon accounting. Clim Change. 2020;161:291–306.
5. Haya B, Cullenward D, Strong AL, Grubert E, Heilmayr R, Sivas DA, et al. Managing uncertainty in carbon offsets: insights from California's standardized approach. Clim Policy. 2020;20:1112–26.
6. California Air Resources Board. Compliance offset protocol U.S. forest projects. Sacramento: California Air Resources Board; 2015.
7. Pan C, Shrestha A, Innes JL, Zhou G, Li N, Li J, et al. Key challenges and approaches to addressing barriers in forest carbon offset projects. J For Res. 2022;33:1109–22.
8. Galik CS, Jackson RB. Risks to forest carbon offset projects in a changing climate. For Ecol Manage. 2009;257:2209–16.
9. Hurteau MD, Hungate BA, Koch GW. Accounting for risk in valuing forest carbon offsets. Carbon Balance Manag. 2009;4:1.
10. Loehman RA, Reinhardt E, Riley KL. Wildland fire emissions, carbon, and climate: Seeing the forest and the trees—a cross-scale assessment of wildfire and carbon dynamics in fire-prone, forested ecosystems. For Ecol Manage. 2014;317:9–19.
11. Williams CA, Gu H, MacLean R, Masek JG, Collatz GJ. Disturbance and the carbon balance of US forests: a quantitative review of impacts from harvests, fires, insects, and droughts. Glob Planet Change. 2016;143:66–80.
12. Pugh TAM, Arneth A, Kautz M, Poulter B, Smith B. Important role of forest disturbances in the global biomass turnover and carbon sinks. Nat Geosci. 2019;12:730–5.
13. Carton W, Lund JF, Dooley K. Undoing equivalence: rethinking carbon accounting for just carbon removal. Front Clim. 2021;3: 664130.
14. Thompson MP, Calkin DE. Uncertainty and risk in wildland fire management: a review. J Environ Manage. 2011;92:1895–909.
15. Millar CI, Stephenson NL. Temperate forest health in an era of emerging megadisturbance. Science. 2015;349:823–6.
16. Seidl R, Thom D, Kautz M, Martin-Benito D, Peltoniemi M, Vacchiano G, et al. Forest disturbances under climate change. Nat Clim Chang. 2017;7:395–402.
17. Anderegg WRL, Trugman AT, Badgley G, Anderson CM, Bartuska A, Ciais P, et al. Climate-driven risks to the climate mitigation potential of forests. Science. 2020;368:eaaz7005.
18. Robbins ZJ, Xu CG, Jonko A, Chitra-Tarak R, Fettig CJ, Costanza J, et al. Carbon stored in live ponderosa pines in the Sierra Nevada will not return to pre-drought (2012) levels during the 21st century due to bark beetle outbreaks. Front Environ Sci. 2023;11:1112756.
19. Badgley G, Chay F, Chegwidden OS, Hamman JJ, Freeman J, Cullenward D. California's forest carbon offsets buffer pool is severely undercapitalized. Front For Glob Chang. 2022;5: 930426.
20. Herbert C, Haya BK, Stephens SL, Butsic V. Managing nature-based solutions in fire-prone ecosystems: competing management objectives in California forests evaluated at a landscape scale. Front For Glob Chang. 2022;5: 957189.
21. Hartmann H, Bastos A, Das AJ, Esquivel-Muelbert A, Hammond WM, Martínez-Vilalta J, et al. Climate change risks to global forest health: emergence of unexpected events of elevated tree mortality worldwide. Annu Rev Plant Biol. 2022;73:673–702.
22. Wu C, Coffield SR, Goulden ML, Randerson JT, Trugman AT, Anderegg WRL. Uncertainty in US forest carbon storage potential due to climate risks. Nat Geosci. 2023;16:422–9.
23. Bowman D, Kolden CA, Abatzoglou JT, Johnston FH, van der Werf GR, Flannigan M. Vegetation fires in the Anthropocene. Nat Rev Earth Environ. 2020;1:500–15.
24. Bond-Lamberty B, Peckham SD, Ahl DE, Gower ST. Fire as the dominant driver of central Canadian boreal forest carbon balance. Nature. 2007;450:89–92.
25. Domke GM, Walters BF, Nowak DJ, Greenfield EJ, Smith JE, Nichols MC, et al. Greenhouse gas emissions and removals from forest land, woodlands, urban trees, and harvested wood products in the United States, 1990–2020, Resource Update FS 382. U.S. Dept. Agric. For. Serv. North. Res. Stn., Madison, WI; 2023.
26. Campbell J, Donato D, Azuma D, Law B. Pyrogenic carbon emission from a large wildfire in Oregon. United States J Geophys Res Biogeosciences. 2007;112:G04014.
27. Fettig CJ, Runyon JB, Homicz CS, James PMA, Ulyshen MD. Fire and insect interactions in North American forests. Curr For Reports. 2022;8:301–16.
28. Reilly MJ, Zuspan A, Yang Z. Characterizing post-fire delayed tree mortality with remote sensing: sizing up the elephant in the room. Fire Ecol. 2023;19:64.
29. Johnson DW, Susfalk RB, Caldwell TG, Murphy JD, Miller WW, Walker RF. Fire effects on carbon and nitrogen budgets in forests. In: Wieder R, Novák M, Vile M, editors. Biogeochem Investig Terr Freshwater, Wetl Ecosyst across Globe. Dortrecht: Springer; 2004. p. 263–75.
30. Ghimire B, Williams CA, Collatz GJ, Vanderhoof M. Fire-induced carbon emissions and regrowth uptake in western U.S. forests: Documenting variation across forest types, fire severity, and climate regions. J Geophys Res Biogeosci. 2012;117:G03036.
31. Garcia M, Saatchi S, Casas A, Koltunov A, Ustin S, Ramirez C, et al. Quantifying biomass consumption and carbon release from the California Rim fire by integrating airborne LiDAR and Landsat OLI data. J Geophys Res Biogeosciences. 2017;122:340–53.

32. Goodwin MJ, Zald HSJ, North MP, Hurteau MD. Climate-driven tree mortality and fuel aridity increase wildfire's potential heat flux. Geophys Res Lett. 2021;48:e2021GL04954.
33. Kane VR, Lutz JA, Roberts SL, Smith DF, McGaughey RJ, Povak NA, et al. Landscape-scale effects of fire severity on mixed-conifer and red fir forest structure in Yosemite National Park. For Ecol Manage. 2013;287:17–31.
34. Fried JS, Torn MS, Mills E. The impact of climate change on wildfire severity: a regional forecast for northern California. Clim Change. 2004;64:169–91.
35. Parks SA, Miller C, Abatzoglou JT, Holsinger LM, Parisien MA, Dobrowski SZ. How will climate change affect wildland fire severity in the western US? Environ Res Lett. 2016;11: 035002.
36. Davis KT, Dobrowski SZ, Higuera PE, Holden ZA, Veblen TT, Rother MT, et al. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proc Natl Acad Sci U S A. 2019;116:6193–8.
37. Jones MW, Abatzoglou JT, Veraverbeke S, Andela N, Lasslop G, Forkel M, et al. Global and regional trends and drivers of fire under climate change. Rev Geophys. 2022;60:e2020RG000726.
38. Parks SA, Abatzoglou JT. Warmer and drier fire seasons contribute to increases in area burned at high severity in western US forests from 1985 to 2017. Geophys Res Lett. 2020;47:e2020GL089858.
39. Wiedinmyer C, Neff JC. Estimates of CO_2 from fires in the United States: implications for carbon management. Carbon Balance Manage. 2007;2:10.
40. Harmon ME, Hanson CT, Dellasala DA. Combustion of aboveground wood from live trees in megafires, CA. USA Forests. 2022;13:391.
41. Amiro BD, Barr AG, Barr JG, Black TA, Bracho R, Brown M, et al. Ecosystem carbon dioxide fluxes after disturbance in forests of North America. J Geophys Res. 2010;115:G000K02.
42. Goulden ML, McMillan AMS, Winston GC, Rocha AV, Manies KL, Harden JW, et al. Patterns of NPP, GPP, respiration, and NEP during boreal forest succession. Glob Chang Biol. 2011;17:855–71.
43. Hemes KS, Norlen CA, Wang JA, Goulden ML, Field CB. The magnitude and pace of photosynthetic recovery after wildfire in California ecosystems. Proc Natl Acad Sci U S A. 2023;120: e2201954120.
44. Meigs GW, Donato DC, Campbell JL, Martin JG, Law BE. Forest fire impacts on carbon uptake, storage, and emission: the role of burn severity in the Eastern Cascades. Oregon Ecosystems. 2009;12:1246–67.
45. Hurteau MD, Brooks ML. Short- and long-term effects of fire on carbon in US dry temperate forest systems. Bioscience. 2011;61:139–46.
46. Jiang P, Russell MB, Frelich L, Babcock C, Smith JE. Wildfires correlate with reductions in aboveground tree carbon stocks and sequestration capacity on forest land in the western United States. Sci Total Environ. 2023;893: 164832.
47. Davis KT, Robles MD, Kemp KB, Higuera PE, Chapman T, Metlen KL, et al. Reduced fire severity offers near-term buffer to climate-driven declines in conifer resilience across the western United States. Proc Natl Acad Sci U S A. 2023;120: e2208120120.
48. Sommers WT, Loehman RA, Hardy CC. Wildland fire emissions, carbon, and climate: science overview and knowledge needs. For Ecol Manage. 2014;317:1–8.
49. Clark JS. Fire and climate change during the last 750 yr in Northwestern Minnesota. Ecol Monogr. 1990;60:135–59.
50. Whitlock C, Shafer SL, Marlon J. The role of climate and vegetation change in shaping past and future fire regimes in the northwestern US and the implications for ecosystem management. For Ecol Manage. 2003;178:5–21.
51. Calder WJ, Parker D, Stopka CJ, Jiménez-Moreno G, Shuman BN. Medieval warming initiated exceptionally large wildfire outbreaks in the Rocky Mountains. Proc Natl Acad Sci U S A. 2015;112:13261–6.
52. Syphard AD, Keeley JE, Pfaff AH, Ferschweiler K. Human presence diminishes the importance of climate in driving fire activity across the United States. Proc Natl Acad Sci U S A. 2017;114:13750–5.
53. Loope WL, Anderton JB. Human vs lightning ignition of presettlement surface fires in coastal pine forests of the upper Great Lakes. Am Midl Nat. 1998;140:206–18.
54. Batek MJ, Rebertus AJ, Schroeder WA, Haithcoat TL, Compas E, Guyette RP. Reconstruction of early nineteenth-century vegetation and fire regimes in the Missouri Ozarks. J Biogeogr. 1999;26:397–412.
55. Fowler C, Konopik E. The history of fire in the southern United States. Hum Ecol Rev. 2007;14:165–76.
56. Long JW, Lake FK, Goode RW. The importance of Indigenous cultural burning in forested regions of the Pacific West, USA. For Ecol Manage. 2021;500: 119597.
57. Roos CI, Swetnam TW, Ferguson TJ, Liebmann MJ, Loehman RA, Welch JR, et al. Native American fire management at an ancient wildland-urban interface in the Southwest United States. Proc Natl Acad Sci U S A. 2021;118: e2018733118.
58. Calkin DE, Thompson MP, Finney MA. Negative consequences of positive feedbacks in US wildfire management. Forest Ecosystems. 2015;2:9.
59. Holden ZA, Swanson A, Luce CH, Jolly WM, Maneta M, Oyler JW, et al. Decreasing fire season precipitation increased recent western US forest wildfire activity. Proc Natl Acad Sci U S A. 2018;115:E8349–57.
60. Williams AP, Abatzoglou JT, Gershunov A, Guzman-Morales J, Bishop DA, Balch JK, et al. Observed impacts of anthropogenic climate change on wildfire in California. Earth's Futur. 2019;7:892–910.
61. Zhuang YZ, Fu R, Santer BD, Dickinson RE, Hall A. Quantifying contributions of natural variability and anthropogenic forcings on increased fire weather risk over the western United States. Proc Natl Acad Sci U S A. 2021;118: e2111875118.
62. Barbero R, Abatzoglou JT, Steel EA, Larkin NK. Modeling very large-fire occurrences over the continental United States from weather and climate forcing. Environ Res Lett. 2014;9: 124009.
63. Barbero R, Abatzoglou JT, Larkin NK, Kolden CA, Stocks B. Climate change presents increased potential for very large fires in the contiguous United States. Int J Wildl Fire. 2015;24:892–9.
64. Moritz MA, Parisien M-A, Batllori E, Krawchuk MA, Van Dorn J, Ganz DJ, et al. Climate change and disruptions to global fire activity. Ecosphere. 2012;3:1–22.
65. Williams AP, Abatzoglou JT. Recent advances and remaining uncertainties in resolving past and future climate effects on global fire activity. Curr Clim Chang Rep. 2016;2:1–14.
66. Stavros EN, Abatzoglou J, Larkin NK, Mckenzie D, Steel EA. Climate and very large wildland fires in the contiguous western USA. Int J Wildl Fire. 2014;23:899–914.
67. Stavros EN, Abatzoglou JT, McKenzie D, Larkin NK. Regional projections of the likelihood of very large wildland fires under a changing climate in the contiguous western United States. Clim Change. 2014;126:455–68.
68. Juang CS, Williams AP, Abatzoglou JT, Balch JK, Hurteau MD, Moritz MA. Rapid growth of large forest fires drives the exponential response of annual forest-fire area to aridity in the Western United States. Geophys Res Lett. 2022;49:e2021GL097131.
69. CALFire (California Department of Forestry and Fire Protection). Top 20 Largest California Wildfires. https://www.fire.ca.gov/media/4jandlhh/top20_acres.pdf
70. Keeley JE, Syphard AD. Large California wildfires: 2020 fires in historical context. Fire Ecol. 2021;17:22.
71. Higuera PE, Abatzoglou JT. Record-setting climate enabled the extraordinary 2020 fire season in the western United States. Glob Chang Biol. 2021;27:1–2.
72. Mass CF, Ovens D, Conrick R, Saltenberger J. The September 2020 Wildfires over the Pacific Northwest. Weather Forecast. 2021;36:1843–65.
73. Reilly MJ, Zuspan A, Halofsky JS, Raymond C, McEvoy A, Dye AW, et al. Cascadia Burning: the historic, but not historically unprecedented, 2020 wildfires in the Pacific Northwest. USA Ecosphere. 2022;13: e4070.
74. Westerling ALR. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philos Trans R Soc B Biol Sci. 2016;371:20150178.
75. Goss M, Swain DL, Abatzoglou JT, Sarhadi A, Kolden CA, Williams AP, et al. Climate change is increasing the likelihood of extreme autumn wildfire conditions across California. Environ Res Lett. 2020;15: 094016.
76. Abatzoglou JT, Juang CS, Williams AP, Kolden CA, Westerling ALR. Increasing synchronous fire danger in forests of the western United States. Geophys Res Lett. 2021;48: e2021GL091377.
77. Steel ZL, Safford HD, Viers JH. The fire frequency-severity relationship and the legacy of fire suppression in California forests. Ecosphere. 2015;6:1–23.

78. Safford HD, Paulson AK, Steel ZL, Young DJN, Wayman RB. The 2020 California fire season: A year like no other, a return to the past or a harbinger of the future? Glob Ecol Biogeogr. 2022;31:2005–25.
79. Wasserman TN, Mueller SE. Climate influences on future fire severity: a synthesis of climate-fire interactions and impacts on fire regimes, high-severity fire, and forests in the western United States. Fire Ecol. 2023;19:43.
80. Westerling AL, Bryant BP. Climate change and wildfire in California. Clim Change. 2008;87:231–49.
81. Dong C, Williams AP, Abatzoglou JT, Lin K, Okin GS, Gillespie TW, et al. The season for large fires in Southern California is projected to lengthen in a changing climate. Commun Earth Environ. 2022;3:22.
82. Yue X, Mickley LJ, Logan JA. Projection of wildfire activity in southern California in the mid-twenty-first century. Clim Dyn. 2014;43:1973–91.
83. Heidari H, Arabi M, Warziniack T. Effects of climate change on natural-caused fire activity in Western US national forests. Atmosphere. 2021;12:981.
84. Dye AW, Gao P, Kim JB, Lei T, Riley KL, Yocom L. High-resolution wildfire simulations reveal complexity of climate change impacts on projected burn probability for Southern California. Fire Ecol. 2023;19:20.
85. Riley KL, Loehman RA. Mid-21st-century climate changes increase predicted fire occurrence and fire season length, Northern Rocky Mountains, United States. Ecosphere. 2016;7: e01543.
86. Litschert SE, Brown TC, Theobald DM. Historic and future extent of wildfires in the Southern Rockies Ecoregion, USA. For Ecol Manage. 2012;269:124–33.
87. Davis R, Yang ZQ, Yost A, Belongie C, Cohen W. The normal fire environment Modeling environmental suitability for large forest wildfires using past, present, and future climate normals. For Ecol Manage. 2017;390:173–86.
88. Gergel DR, Nijssen B, Abatzoglou JT, Lettenmaier DP, Stumbaugh MR. Effects of climate change on snowpack and fire potential in the western USA. Clim Change. 2017;141:287–99.
89. Brown EK, Wang JL, Feng Y. US wildfire potential: a historical view and future projection using high-resolution climate data. Environ Res Lett. 2021;16: 034060.
90. Dye AW, Reilly MJ, McEvoy A, Lemons R, Riley KL, Kim JB, et al. Simulated future shifts in wildfire regimes in moist forests of Pacific Northwest, USA. J Geophys Res Biogeosciences. 2024;129: e2023JG007722.
91. Littell JS, Oneil EE, McKenzie D, Hicke JA, Lutz JA, Norheim RA, et al. Forest ecosystems, disturbance, and climatic change in Washington State, USA. Clim Change. 2010;102:129–58.
92. Halofsky JE, Peterson DL, Harvey BJ. Changing wildfire, changing forests: the effects of climate change on fire regimes and vegetation in the Pacific Northwest, USA. Fire Ecol. 2020;16:4.
93. Barros AMG, Day MA, Preisler HK, Abatzoglou JT, Krawchuk MA, Houtman R, et al. Contrasting the role of human- and lightning-caused wildfires on future fire regimes on a Central Oregon landscape. Environ Res Lett. 2021;16: 064081.
94. Liu YQ, Goodrick SL, Stanturf JA. Future US wildfire potential trends projected using a dynamically downscaled climate change scenario. For Ecol Manage. 2013;294:120–35.
95. Gao P, Terando AJ, Kupfer JA, Varner JM, Stambaugh MC, Lei TL, et al. Robust projections of future fire probability for the conterminous United States. Sci Total Environ. 2021;789: 147872.
96. Anderegg WRL, Chegwidden OS, Badgley G, Trugman AT, Cullenward D, Abatzoglou JT, et al. Future climate risks from stress, insects and fire across US forests. Ecol Lett. 2022;25:1510–20.
97. Weber MG, Flannigan MD. Canadian boreal forest ecosystem structure and function in a changing climate: impact on fire regimes. Environ Rev. 1997;5:145–66.
98. Gillett NP, Weaver AJ, Zwiers FW, Flannigan MD. Detecting the effect of climate change on Canadian forest fires. Geophys Res Lett. 2004;31:L18211.
99. Kasischke ES, Turetsky MR. Recent changes in the fire regime across the North American boreal region—spatial and temporal patterns of burning across Canada and Alaska. Geophys Res Lett. 2006. https://doi.org/10.1029/2006GL025677.
100. Flannigan M, Stocks B, Turetsky M, Wotton M. Impacts of climate change on fire activity and fire management in the circumboreal forest. Glob Chang Biol. 2009;15:549–60.
101. Young AM, Higuera PE, Duffy PA, Hu FS. Climatic thresholds shape northern high-latitude fire regimes and imply vulnerability to future climate change. Ecography. 2017;40:606–17.
102. Balshi MS, Mcguire AD, Duffy P, Flannigan M, Kicklighter DW, Melillo J. Vulnerability of carbon storage in North American boreal forests to wildfires during the 21st century. Glob Chang Biol. 2009;15:1491–510.
103. Bachelet D, Lenihan J, Neilson R, Drapek R, Kittel T. Simulating the response of natural ecosystems and their fire regimes to climatic variability in Alaska. Can J For Res. 2005;35:2244–57.
104. Mitchell RJ, Liu YQ, O'Brien JJ, Elliott KJ, Starr G, Miniat CF, et al. Future climate and fire interactions in the southeastern region of the United States. For Ecol Manage. 2014;327:316–26.
105. Bedel AP, Mote TL, Goodrick SL. Climate change and associated fire potential for the southeastern United States in the 21st century. Int J Wildl Fire. 2013;22:1034–43.
106. Barbero R, Abatzoglou JT, Kolden CA, Hegewisch KC, Larkin NK, Podschwit H. Multi-scalar influence of weather and climate on very large-fires in the eastern United States. Int J Climatol. 2015;35:2180–6.
107. Fill JM, Davis CN, Crandall RM. Climate change lengthens southeastern USA lightning-ignited fire seasons. Glob Chang Biol. 2019;25:3562–9.
108. Terando A, Hiers JK, Williams M, Goodrick SL, O'Brien JJ. Is there a dry season in the Southeast US? Glob Chang Biol. 2021;27:713–5.
109. Kerr GH, DeGaetano AT, Stoof CR, Ward D. Climate change effects on wildland fire risk in the Northeastern and Great Lakes states predicted by a downscaled multi-model ensemble. Theor Appl Climatol. 2018;131:625–39.
110. Burney DA, DeCandido RV, Burney LP, Kostel-Hughes FN, Stafford TW, James HF. A Holocene record of climate change, fire ecology, and human activity from montane Flat Top Bog. Maui J Paleolimnol. 1995;13:209–17.
111. Pau S, MacDonald GM, Gillespie T. A dynamic history of climate change and human impact on the environment from Kealia Pond, Maui, Hawaiian Islands. Ann Assoc Am Geogr. 2012;102:748–62.
112. D'Antonio CM, Hughes RF, Tunison JT. Long-term impacts of invasive grasses and subsequent fire in seasonally dry Hawaiian woodlands. Ecol Appl. 2011;21:1617–28.
113. Dolling K, Chu PS, Fujioka F. A climatological study of the Keetch/Byram drought index and fire activity in the Hawaiian Islands. Agric For Meteorol. 2005;133:17–27.
114. Trauernicht C. Vegetation-Rainfall interactions reveal how climate variability and climate change alter spatial patterns of wildland fire probability on Big Island, Hawaii. Sci Total Environ. 2019;650:459–69.
115. Wardell-Johnson GW, Keppel G, Sander J. Climate change impacts on the terrestrial biodiversity and carbon stocks of Oceania. Pacific Conserv Biol. 2011;17:220–40.
116. Selmants PC, Sleeter BM, Liu JN, Wilson TS, Trauernicht C, Frazier AG, et al. Ecosystem carbon balance in the Hawaiian Islands under different scenarios of future climate and land use change. Environ Res Lett. 2021;16: 104020.
117. Mattson WJ. The role of arthropods in forest ecosystems. Springer Science & Business Media; 2012.
118. Ellison AM, Bank MS, Clinton BD, Colburn EA, Elliott K, Ford CR, et al. Loss of foundation species: Consequences for the structure and dynamics of forested ecosystems. Front Ecol Environ. 2005;3:479–86.
119. Boyd IL, Freer-Smith PH, Gilligan CA, Godfray HCJ. The consequence of tree pests and diseases for ecosystem services. Science. 2013;342:3167–81.
120. Hicke JA, Allen CD, Desai AR, Dietze MC, Hall RJ, Hogg EHT, et al. Effects of biotic disturbances on forest carbon cycling in the United States and Canada. Glob Chang Biol. 2012;18:7–34.
121. Lovett GM, Weiss M, Liebhold AM, Holmes TP, Leung B, Lambert KF, et al. Nonnative forest insects and pathogens in the United States: impacts and policy options. Ecol Appl. 2016;26:1437–55.
122. Quirion BR, Domke GM, Walters BF, Lovett GM, Fargione JE, Greenwood L, et al. Insect and disease disturbances correlate with reduced carbon sequestration in forests of the contiguous United States. Front For Glob Chang. 2021;4: 716582.
123. Kurz WA, Apps MJ. A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector. Ecol Appl. 1999;9:526–47.

124. Hicke JA, Meddens AJH, Allen CD, Kolden CA. Carbon stocks of trees killed by bark beetles and wildfire in the western United States. Environ Res Lett. 2013;8: 035032.
125. Dale VH, Joyce LA, McNulty S, Neilson RP, Ayres MP, Flannigan MD, et al. Climate change and forest disturbances: climate change can affect forests by altering the frequency, intensity, duration, and timing of fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, or landslides. Bioscience. 2001;51:723–34.
126. Harausz E, Pimentel D. North American forest losses due to insects and plant pathogens. In: Pimentel D, editor. Encycl Pest Manag. Marcel Dekker Inc; 2002. p. 539–41.
127. Singh P. Research and management strategies for major tree diseases in Canada: synthesis Part 1. For Chron. 1993;69:151–62.
128. Fettig CJ, Asaro C, Nowak JT, Dodds KJ, Gandhi KJK, Moan JE, et al. Trends in bark beetle impacts in North America during a period (2000–2020) of rapid environmental change. J For. 2022;120:693–713.
129. Kurz WA, Dymond CC, Stinson G, Rampley GJ, Neilson ET, Carroll AL, et al. Mountain pine beetle and forest carbon feedback to climate change. Nature. 2008;452:987–90.
130. Bentz BJ, Rgnire J, Fettig CJ, Hansen EM, Hayes JL, Hicke JA, et al. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. Bioscience. 2010;60:602–13.
131. Busby PE, Canham CD. An exotic insect and pathogen disease complex reduces aboveground tree biomass in temperate forests of eastern North America. Can J For Res. 2011;41:401–11.
132. Healey SP, Raymond CL, Lockman IB, Hernandez AJ, Garrard C, Huang C. Root disease can rival fire and harvest in reducing forest carbon storage. Ecosphere. 2016;7: e01569.
133. Aukema JE, McCullough DG, Von Holle B, Liebhold AM, Britton K, Frankel SJ. Historical accumulation of nonindigenous forest pests in the continental United States. Bioscience. 2010;60:886–97.
134. Koch FH, Yemshanov D, Colunga-Garcia M, Magarey RD, Smith WD. Potential establishment of alien-invasive forest insect species in the United States: where and how many? Biol Invas. 2011;13:969–85.
135. Bentz BJ, Logan JA, Amman GD. Temperature-dependent development of the mountain pine beetle (Coleoptera: scolytidae) and simulation of its phenology. Can Entomol. 1991;123:1083–94.
136. Six DL, Bentz BJ. Temperature determines symbiont abundance in a multipartite bark beetle-fungus ectosymbiosis. Microb Ecol. 2007;54:112–8.
137. Robbins ZJ, Xu C, Aukema BH, Buotte PC, Chitra-Tarak R, Fettig CJ, et al. Warming increased bark beetle-induced tree mortality by 30% during an extreme drought in California. Glob Chang Biol. 2022;28:509–23.
138. Bradley BA, Wilcove DS, Oppenheimer M. Climate change increases risk of plant invasion in the eastern United States. Biol Invasions. 2010;12:1855–72.
139. Kolb TE, Fettig CJ, Ayres MP, Bentz BJ, Hicke JA, Mathiasen R, et al. Observed and anticipated impacts of drought on forest insects and diseases in the United States. For Ecol Manage. 2016;380:321–34.
140. Jactel H, Petit J, Desprez-Loustau ML, Delzon S, Piou D, Battisti A, et al. Drought effects on damage by forest insects and pathogens: a meta-analysis. Glob Chang Biol. 2012;18:267–76.
141. Kautz M, Meddens AJH, Hall RJ, Arneth A. Biotic disturbances in Northern Hemisphere forests—a synthesis of recent data, uncertainties and implications for forest monitoring and modelling. Glob Ecol Biogeogr. 2017;26:533–52.
142. Sambaraju KR, Carroll AL, Zhu J, Stahl K, Moore RD, Aukema BH. Climate change could alter the distribution of mountain pine beetle outbreaks in western Canada. Ecography. 2012;35:211–23.
143. Bentz BJ, Hansen EM, Davenport M, Soderberg D. Complexities in predicting mountain pine beetle and spruce beetle response to climate change. In: Bark beetle management, ecology, and climate change. London: Academic Press; 2022. p. 31–54.
144. Sherriff RL, Berg EE, Miller AE. Climate variability and spruce beetle (*Dendroctonus rufipennis*) outbreaks in south-central and southwest Alaska. Ecology. 2011;92:1459–70.
145. Volney WJA, Fleming RA. Climate change and impacts of boreal forest insects. Agric Ecosyst Environ. 2000;82:283–94.
146. Logan JA, Régnière J, Gray DR, Munson AS. Risk assessment in the face of a changing environment: gypsy moth and climate change in Utah. Ecol Appl. 2007;17:101–17.
147. Sturrock RN, Frankel SJ, Brown AV, Hennon PE, Kliejunas JT, Lewis KJ, et al. Climate change and forest diseases. Plant Pathol. 2011;60:133–49.
148. Weed AS, Ayres MP, Hicke JA. Consequences of climate change for biotic disturbances in North American forests. Ecol Monogr. 2013;83:441–70.
149. Dudney J, Willing CE, Das AJ, Latimer AM, Nesmith JCB, Battles JJ. Nonlinear shifts in infectious rust disease due to climate change. Nat Commun. 2021;12:5102.
150. McNellis BE, Smith AMS, Hudak AT, Strand EK. Tree mortality in western US forests forecasted using forest inventory and Random Forest classification. Ecosphere. 2021;12: e03419.
151. Swanston C, Brandt LA, Janowiak MK, Handler SD, Butler-Leopold P, Iverson L, et al. Vulnerability of forests of the Midwest and Northeast United States to climate change. Clim Change. 2018;146:103–16.
152. Ellison AM, Orwig DA, Fitzpatrick MC, Preisser EL. The past, present, and future of the hemlock woolly adelgid (*Adelges tsugae*) and its ecological interactions with eastern hemlock (*Tsuga canadensis*) forests. Insects. 2018;9:172.
153. Rustad LE, Campbell J, Dukes JS, Huntington T, Lambert KF, Mohan J, et al. Changing climate, changing forests: the impacts of climate change on forests of the Northeastern United States and Eastern Canada. NRS-GTR-99. U.S. Dept. Agric. For. Serv. Nor. Res. Stn., Newton Square, PA; 2012.
154. Albani M, Moorcroft PR, Ellison AM, Orwig DA, Foster DR. Predicting the impact of hemlock woolly adelgid on carbon dynamics of eastern United States forests. Can J For Res. 2010;40:119–33.
155. Everett RA. Patterns and pathways of biological invasions. Trends Ecol Evol. 2000;15:177–8.
156. Finch DM, Butler JL, Runyon JB, Fettig CJ, Kilkenny FF, Jose S, et al. Effects of climate change on invasive species. Invasive species for rangelands United States a Compr Sci Synth United States For Sect. 2021. p. 57–83.
157. Breshears DD, Cobb NS, Rich PM, Price KP, Allen CD, Balice RG, et al. Regional vegetation die-off in response to global-change-type drought. Proc Natl Acad Sci U S A. 2005;102:15144–8.
158. Adams HD, Macalady AK, Breshears DD, Allen CD, Stephenson NL, Saleska SR, et al. Climate-induced tree mortality: earth system consequences. Eos Trans Am Geophys Union. 2010;91:153–4.
159. Allen CD, Macalady AK, Chenchouni H, Bachelet D, McDowell N, Vennetier M, et al. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. For Ecol Manage. 2010;259:660–84.
160. Schwalm CR, Williams CA, Schaefer K, Baldocchi D, Black TA, Goldstein AH, et al. Reduction in carbon uptake during turn of the century drought in western North America. Nat Geosci. 2012;5:551–6.
161. Anderegg WRL, Plavcová L, Anderegg LDL, Hacke UG, Berry JA, Field CB. Drought's legacy: Multiyear hydraulic deterioration underlies widespread aspen forest die-off and portends increased future risk. Glob Chang Biol. 2013;19:1188–96.
162. Brodribb TJ, Powers J, Cochard H, Choat B. Hanging by a thread? For Drought Sci. 2020;368:261–6.
163. Humphrey V, Berg A, Ciais P, Gentine P, Jung M, Reichstein M, et al. Soil moisture-atmosphere feedback dominates land carbon uptake variability. Nature. 2021;592:65.
164. Zscheischler J, Mahecha MD, Harmeling S, Reichstein M. Detection and attribution of large spatiotemporal extreme events in Earth observation data. Ecol Inform. 2013;15:66–73.
165. Clark JS, Iverson L, Woodall CW, Allen CD, Bell DM, Bragg DC, et al. The impacts of increasing drought on forest dynamics, structure, and biodiversity in the United States. Glob Chang Biol. 2016;22:2329–52.
166. Sleeter BM, Marvin DC, Cameron DR, Selmants PC, Westerling ALR, Kreitler J, et al. Effects of 21st-century climate, land use, and disturbances on ecosystem carbon balance in California. Glob Chang Biol. 2019;25:3334–53.
167. Harris LB, Scholl AE, Young AB, Estes BL, Taylor AH. Spatial and temporal dynamics of 20th century carbon storage and emissions after wildfire in an old-growth forest landscape. For Ecol Manage. 2019;449: 117461.

168. Trugman AT, Medvigy D, Anderegg WRL, Pacala SW. Differential declines in Alaskan boreal forest vitality related to climate and competition. Glob Chang Biol. 2018;24:1097–107.
169. Stanke H, Finley AO, Domke GM, Weed AS, MacFarlane DW. Over half of western United States' most abundant tree species in decline. Nat Commun. 2021;12:451.
170. Williams AP, Allen CD, Macalady AK, Griffin D, Woodhouse CA, Meko DM, et al. Temperature as a potent driver of regional forest drought stress and tree mortality. Nat Clim Chang. 2013;3:292–7.
171. Cook BI, Mankin JS, Marvel K, Williams AP, Smerdon JE, Anchukaitis KJ. Twenty-first century drought projections in the CMIP6 forcing scenarios. Earth's Futur. 2020;8:e2019EF001461.
172. Anderegg WRL, Berry JA, Smith DD, Sperry JS, Anderegg LDL, Field CB. The roles of hydraulic and carbon stress in a widespread climate-induced forest die-off. Proc Natl Acad Sci U S A. 2012;109:233–7.
173. Trugman AT, Anderegg LDL, Anderegg WRL, Das AJ, Stephenson NL. Why is tree drought mortality so hard to predict? Trends Ecol Evol. 2021;36:520–32.
174. Anderegg WRL, Schwalm C, Biondi F, Camarero JJ, Koch G, Litvak M, et al. Pervasive drought legacies in forest ecosystems and their implications for carbon cycle models. Science. 2015;349:528–32.
175. McDowell NG, Williams AP, Xu C, Pockman WT, Dickman LT, Sevanto S, et al. Multi-scale predictions of massive conifer mortality due to chronic temperature rise. Nat Clim Chang. 2016;6:295–300.
176. Buotte PC, Levis S, Law BE, Hudiburg TW, Rupp DE, Kent JJ. Near-future forest vulnerability to drought and fire varies across the western United States. Glob Chang Biol. 2019;25:290–303.
177. Venturas MD, Todd HN, Trugman AT, Anderegg WRL. Understanding and predicting forest mortality in the western United States using long-term forest inventory data and modeled hydraulic damage. New Phytol. 2021;230:1896–910.
178. Dai AG. Characteristics and trends in various forms of the Palmer Drought Severity Index during 1900–2008. J Geophys Res. 2011;116:D12115.
179. Millward AA, Kraft CE. Physical influences of landscape on a large-extent ecological disturbance: the northeastern North American ice storm of 1998. Landsc Ecol. 2004;19:99–111.
180. Hanson JJ, Lorimer CG. Forest structure and light regimes following moderate wind storms: implications for multi-cohort management. Ecol Appl. 2007;17:1325–40.
181. Meigs GW, Keeton WS. Intermediate-severity wind disturbance in mature temperate forests: legacy structure, carbon storage, and stand dynamics. Ecol Appl. 2018;28:798–815.
182. Schulte LA, Mladenoff DJ. Severe wind and fire regimes in northern forests: historical variability at the regional scale. Ecology. 2005;86:431–45.
183. Keeton WS, Whitman AA, McGee GC, Goodale CL. Late-successional biomass development in northern hardwood-conifer forests of the northeastern United States. Forest Science. 2011;57:489–505.
184. McGarvey JC, Thompson JR, Epstein HE, Shugart HH. Carbon storage in old-growth forests of the Mid-Atlantic: toward better understanding the eastern forest carbon sink. Ecology. 2015;96:311–7.
185. Thom D, Keeton WS. Stand structure drives disparities in carbon storage in northern hardwood-conifer forests. For Ecol Manage. 2019;442:10–20.
186. Chambers JQ, Fisher JI, Zeng H, Chapman EL, Baker DB, Hurtt GC. Hurricane Katrina's carbon footprint on U.S. Gulf Coast forests. Science. 2007;318:1107.
187. Fisk JP, Hurtt GC, Chambers JQ, Zeng H, Dolan KA, Negrón-Juárez RI. The impacts of tropical cyclones on the net carbon balance of eastern US forests (1851–2000). Environ Res Lett. 2013;8: 045017.
188. Cannon JB, Peterson CJ, Godfrey CM, Whelan AW. Hurricane wind regimes for forests of North America. Proc Natl Acad Sci. 2023;120: e2309076120.
189. Sippel S, Reichstein M, Ma XL, Mahecha MD, Lange H, Flach M, et al. Drought, heat, and the carbon cycle. Curr Clim Chang Rep. 2018;4:266–86.
190. Still CJ, Sibley A, DePinte D, Busby PE, Harrington CA, Schulze M, et al. Causes of widespread foliar damage from the June 2021 Pacific Northwest Heat Dome: more heat than drought. Tree Physiol. 2023;43:203–9.
191. Seneviratne SI, Zhang X, Adnan M, Badi W, Dereczynski C, Di Luca A, et al. Weather and climate extreme events in a changing climate. In: Masson-Delmotte Z, Zhai VP, Pirani A, Connors SL, Péan C, Berger S, editors., et al., Clim chang 2021 Phys Sci Basis Contrib Work Gr I to Sixth Assess Rep Intergov Panel Clim Chang. Cambridge University Press; 2021. p. 1513–766.
192. Marvel K, Su W, Delgado R, Aarons S, Chatterjee A, Garcia ME, et al. Chapter 2: climate trends. In: Crimmins AR, Avery CW, Easterling DR, Kunkel KE, Stewart BC, Maycock TK, editors., et al., Fifth Natl Clim Assess. Washington, DC: US Global Change Research Program; 2023.
193. Hicke JA, Johnson MC, Hayes JL, Preisler HK. Effects of bark beetle-caused tree mortality on wildfire. For Ecol Manage. 2012;271:81–90.
194. Stephens SL, Bernal AA, Collins BM, Finney MA, Lautenberger C, Saah D. Mass fire behavior created by extensive tree mortality and high tree density not predicted by operational fire behavior models in the southern Sierra Nevada. For Ecol Manage. 2022;518: 120258.
195. Ray C, Cluck DR, Wilderson RL, Siegel RB, White AM, Tarbill GL, et al. Patterns of woodboring beetle activity following fires and bark beetle outbreaks in montane forests of California, USA. Fire Ecol. 2019;15:21.
196. Hood SM, Baker S, Sala A. Fortifying the forest: thinning and burning increase resistance to a bark beetle outbreak and promote forest resilience. Ecol Appl. 2016;26:1984–2000.
197. Stephens SL, Collins BM, Fettig CJ, Finney MA, Hoffman CM, Knapp EE, et al. Drought, tree mortality, and wildfire in forests adapted to frequent fire. Bioscience. 2018;68:77–88.
198. Finney MA, McAllister SS. A review of fire interactions and mass fires. J Combust. 2011;2011: 548328.
199. Meigs GW, Zald HSJ, Campbell JL, Keeton WS, Kennedy RE. Do insect outbreaks reduce the severity of subsequent forest fires? Environ Res Lett. 2016;11: 045008.
200. Brand G, Nelson M, Wendt D, Nimerfro K. The Hexagon/Panel System for Selecting FIA Plots Under an Annual Inventory. In: McRoberts R, Reams G, Van Deusen P, editors. Proc 1st Annu For Invent Anal Symp. San Antonio: U.S. Dept. Agric. For. Serv. Nor. Cent. Res. Stn; 2000. p. 8–13.
201. Hicke JA, Jenkins JC, Ojima DS, Ducey M. Spatial patterns of forest characteristics in the western United States derived from inventories. Ecol Appl. 2007;17:2387–402.
202. Bechtold WA, Patterson PL. The enhanced forest inventory and analysis program—national sampling design and estimation procedures, SRS-GTR-80. Asheville: U.S. Dept. Agric. For. Serv. South Res. Stn.; 2005.
203. Hoover CM, Smith JE. Current aboveground live tree carbon stocks and annual net change in forests of conterminous United States. Carbon Balance Manag. 2021;16:17.
204. Ohmann JL, Gregory MJ, Roberts HM, Cohen WB, Kennedy RE, Yang Z. Mapping change of older forest with nearest-neighbor imputation and Landsat time-series. For Ecol Manage. 2012;272:13–25.
205. Bell DM, Gregory MJ, Ohmann JL. Imputed forest structure uncertainty varies across elevational and longitudinal gradients in the western Cascade Mountains, Oregon, USA. For Ecol Manage. 2015;358:154–64.
206. Riley KL, Grenfell IC, Finney MA. Mapping forest vegetation for the western United States using modified random forests imputation of FIA forest plots. Ecosphere. 2016;7: e01472.
207. Riley KL, Grenfell IC, Finney MA, Wiener JM, Houtman RM, Riley KL, Grenfell IC, Finney MA, Wiener JM, Houtman RM. Fire lab tree list: a tree-level model of the conterminous United States landscape circa 2014. Fort Collins: U.S. Dep. Agric. For. Serv. Rocky Mtn. Res. Stn.; 2019. https://doi.org/10.2737/RDS-2019-0026.
208. Riley KL, Grenfell IC, Finney MA, Wiener JM. TreeMap, a tree-level model of conterminous US forests circa 2014 produced by imputation of FIA plot data. Sci Data. 2021;8:11.
209. Riley KL, Grenfell IC, Shaw JD, Finney MA. TreeMap 2016 dataset generates CONUS-wide maps of forest characteristics including live basal area, aboveground carbon, and number of trees per acre. J For. 2022;120:607–32.
210. Lister AJ, Andersen H, Frescino T, Gatziolis D, Healey S, Heath LS, et al. Use of remote sensing data to improve the efficiency of national forest inventories: a case study from the United States national forest inventory. Forests. 2020;11:1–41.
211. Potter CS, Randerson JT, Field CB, Matson PA, Vitousek PM, Mooney HA, et al. Terrestrial ecosystem production: a process model based on global satellite and surface data. Global Biogeochem Cycles. 1993;7:811–41.

212. Williams CA, Hasler N, Gu H, Zhou Y. Forest carbon stocks and fluxes from the NFCMS, Conterminous USA, 1990-2010. Oak Ridge, Tennessee; 2020. https://doi.org/10.3334/ORNLDAAC/1829
213. Kellndorfer J, Walker W, Kirsch K, Fiske G, Bishop J, LaPoint L, et al. NACP aboveground biomass and carbon baseline data, V. 2 (NBCD 2000), USA, 2000. ORNL DAAC. Oak Ridge, Tennessee; 2013. https://doi.org/10.3334/ORNLDAAC/1161
214. Carbon Map; 2023. https://site.tplgis.org/carbonmapsecure/viewer/
215. Eidenshink J, Schwind B, Brewer K, Zhu Z-L, Quayle B, Howard S. A project for monitoring trends in burn severity. Fire Ecol. 2007;3:3–21.
216. Stage AR. Prognosis Model for Stand Development. Ogden: U.S. Dept. Agric. For. Serv. Intermountain For. Range Exp. Stn.; 1973.
217. Dixon GE. Essential FVS: A User's Guide To The Forest Vegetation Simulator. Fort Collins: U.S. Dept. Agric. For. Serv. Rocky Mtn. Stn.; 1973.
218. Martin F. User Guide to the Economic Extension (ECON) of the Forest Vegetation Simulator. Fort Collins: U.S. Dept. Agric. For. Serv. Manage Serv. Cent.; 2009.
219. Crookston NL. Climate-FVS version 2: content, users guide, applications, and behavior, RMRS-GTR-319. Fort Commins: U.S. Dept. Agric. For. Serv. Rocky Mtn. Res. Stn.; 2014.
220. Rebain SA, Reinhardt ED, Crookston NL, Beukema SJ, Kurz WA, Greenough JA, et al. The fire and fuels extension to the forest vegetation simulator: updated model documentation. Internal Rep. U.S. Dep. Agric. For. Serv. For. Manag. Serv. Cent.: Fort Collins; 2010. p. 228.
221. Finney MA, McHugh CW, Grenfell IC, Riley KL, Short KC. A simulation of probabilistic wildfire risk components for the continental United States. Stoch Environ Res Risk Assess. 2011;25:973–1000.
222. Short K, Finney M, Vogler K, Scott J, Gilbertson-Day J, Grenfell I. Spatial datasets of probabilistic wildfire risk components for the United States (270 m). Fort Collins, CO; 2020. https://doi.org/10.2737/RDS-2016-0034-2
223. Jaffe MR, Scott JH, Callahan MN, Dillon GK, Karau EC, Lazarz MT. Wildfire risk to communities: spatial datasets of wildfire risk for populated areas in the United States. 2nd edn. Fort Collins, CO; 2024. https://doi.org/10.2737/RDS-2020-0060-2.
224. Tortorelli CM, Kim JB, Vaillant NM, Riley K, Dye A, Nietupski TC, et al. Feeding the fire: annual grass invasion facilitates modeled fire spread across Inland Northwest forest-mosaic landscapes. Ecosphere. 2023;14: e4413.
225. Calkin DE, Ager A, Thompson MP, Finney MA, Lee DC, Quigley TM, et al. A Comparative Risk assessment framework for wildland fire management: the 2010 cohesive strategy science report, RMRS-GTR-262. U.S. Dep. Agric. For. Serv. Rocky Mtn. Res. Stn.: Fort Collins; 2011.
226. Rothermel RC. A mathematical model for predicting fire spread in wildland fuels. USDA Forest Service Research Paper. INT-115; 1972.
227. Riley KL, Grenfell IC, Shaw JD. FuelMap 2014: imputed map of carbon stored in litter, duff, fine woody debris, and coarse woody debris for CONUS forests circa 2014. U.S. Dep. Agric. For. Serv. Rocky Mtn. Res. Stn.: Fort Collins; 2023. https://doi.org/10.2737/RDS-2023-0042.
228. Pfeifer EM, Hicke JA, Meddens AJH. Observations and modeling of aboveground tree carbon stocks and fluxes following a bark beetle outbreak in the western United States. Glob Chang Biol. 2011;17:339–50.
229. Coleman TW, Graves AD, Heath Z, Flowers RW, Hanavan RP, Cluck DR, Ryerson D. Accuracy of aerial detection surveys for mapping insect and disease disturbances in the United States. For Ecol Manage. 2018;430:321–36.
230. Slaton MR, Warren K, Koltunov A, Smith S. Chapter 12–Accuracy assessment of insect and disease survey and eDART for monitoring forest health. In Forest Health Monitoring: National Status, Trends, and Analysis 2020; Gen. Tech. Rep. SRS-261; USDA Forest Service, Southern Research Station: Asheville, NC, USA. 2021; p.187–95.
231. Krist FJ, Romero SA, Ellenwood JR, Woods ME, McMahan AJ, Cowardin JP, et al. National insect and disease risk and hazard mapping. 2012. https://www.fs.usda.gov/foresthealth/applied-sciences/mapping-reporting/national-risk-maps.shtml
232. Reeves MC, Ryan KC, Rollins MG, Thompson TG. Spatial fuel data products of the LANDFIRE Project. Int J Wildl Fire. 2009;18:250–67.
233. LANDFIRE 2016 Remap (LF 2.0.0). LANDFIRE; 2021. https://www.landfire.gov/lf_remap.php
234. LF 2.3.0 [Internet]. LANDFIRE; 2023. https://www.landfire.gov/lf_230.php
235. Scott JH, Burgan RE. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model, RMRS-GTR-153. U.S. Dep. Agric. For. Serv. Rocky Mtn. Res. Stn.: Fort Collins; 2005.
236. Agee JK, Skinner CN. Basic principles of forest fuel reduction treatments. For Ecol Manage. 2005;211:83–96.
237. Stephens SL. Evaluation of the effects of silvicultural and fuels treatments on potential fire behaviour in Sierra Nevada mixed-conifer forests. For Ecol Manage. 1998;105:21–35.
238. Stephens SL, Moghaddas JJ, Hartsough BR, Moghaddas EEY, Clinton NE. Fuel treatment effects on stand-level carbon pools, treatment-related emissions, and fire risk in a Sierra Nevada mixed-conifer forest. Can J For Res. 2009;39:1538–47.
239. Chiono LA, Fry DL, Collins BM, Chatfield AH, Stephens SL. Landscape-scale fuel treatment and wildfire impacts on carbon stocks and fire hazard in California spotted owl habitat. Ecosphere. 2017;8: e01648.
240. Krofcheck DJ, Hurteau MD, Scheller RM, Loudermilk EL. Restoring surface fire stabilizes forest carbon under extreme fire weather in the Sierra Nevada. Ecosphere. 2017;8: e01663.
241. Wiedinmyer C, Hurteau MD. Prescribed fire as a means of reducing forest carbon emissions in the western United States. Environ Sci Technol. 2010;44:1926–32.
242. Barnett K, Parks SA, Miller C, Naughton HT. Beyond fuel treatment effectiveness: characterizing interactions between fire and treatments in the US. Forests. 2016;7:237.
243. Boerner REJ, Huang J, Hart SC. Fire, thinning, and the carbon economy: effects of fire and fire surrogate treatments on estimated carbon storage and sequestration rate. For Ecol Manage. 2008;255:3081–97.
244. Wiechmann ML, Hurteau MD, North MP, Koch GW, Jerabkova L. The carbon balance of reducing wildfire risk and restoring process: an analysis of 10-year post-treatment carbon dynamics in a mixed-conifer forest. Clim Change. 2015;132:709–19.
245. Ager AA, Finney MA, Mcmahan A, Cathcart J. Measuring the effect of fuel treatments on forest carbon using landscape risk analysis. Nat Hazards Earth Syst Sci. 2010;10:2515–26.
246. Campbell JL, Harmon ME, Mitchell SR. Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions? Front Ecol Environ. 2012;10:83–90.
247. Restaino JC, Peterson DL. Wildfire and fuel treatment effects on forest carbon dynamics in the western United States. For Ecol Manage. 2013;303:46–460.
248. Stephens JL, Alexander JD. Effects of fuel reduction on bird density and reproductive success in riparian areas of mixed-conifer forest in southwest Oregon. For Ecol Manage. 2011;261:43–9.
249. North M, Hurteau M, Innes J. Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. Ecol Appl. 2009;19:1385–96.
250. Hurteau MD, Stoddard MT, Fulé PZ. The carbon costs of mitigating high-severity wildfire in southwestern ponderosa pine. Glob Chang Biol. 2011;17:1516–21.
251. Young JD, Ager AA. Resource objective wildfire leveraged to restore old growth forest structure while stabilizing carbon stocks in the southwestern United States. Ecol Modell. 2024;488:110573.
252. Liang S, Hurteau MD, Westerling AL. Large-scale restoration increases carbon stability under projected climate and wildfire regimes. Front Ecol Environ. 2018;16:207–12.
253. Mitchell SR, Harmon ME, O'Connell KEB. Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. Ecol Appl. 2009;19:643–55.
254. Franklin JF, Spies TA, Van PR, Carey AB, Thornburgh DA, Berg DR, et al. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. For Ecol Manage. 2002;155:399–423.
255. Vanderwel MC, Coomes DA, Purves DW. Quantifying variation in forest disturbance, and its effects on aboveground biomass dynamics, across the eastern United States. Glob Chang Biol. 2013;19:1504–17.
256. Anderegg WRL, Konings AG, Trugman AT, et al. Hydraulic diversity of forests regulates ecosystem resilience during drought. Nature. 2018;561:538–41.

257. D'Amato AW, Bradford JB, Fraver S, Palik BJ. Effects of thinning on drought vulnerability and climate response in north temperate forest ecosystems. Ecol Appl. 2013;23:1735–42.
258. Stephens SL, McIver JD, Boerner REJ, Fettig CJ, Fontaine JB, Hartsough BR, et al. The effects of forest fuel-reduction treatments in the United States. Bioscience. 2012;62:549–60.
259. Finney MA. A computational method for optimising fuel treatment locations. Int J Wildl Fire. 2007;16:702–11.
260. Ager AA, Vaillant NM, Finney MA. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. For Ecol Manage. 2010;259:1556–70.
261. Thompson MP, Vogler KC, Scott JH, Miller C. Comparing risk-based fuel treatment prioritization with alternative strategies for enhancing protection and resource management objectives. Fire Ecol. 2022;18:26.
262. Keeton WS. Managing for late-successional/old-growth characteristics in northern hardwood-conifer forests. For Ecol Manage. 2006;235:129–42.
263. Parker TJ, Clancy KM, Mathiasen RL. Interactions among fire, insects and pathogens in coniferous forests of the interior western United States and Canada. Agric For Entomol. 2006;8:167–89.
264. Hood S, Sala A, Heyerdahl EK, Boutin M. Low-severity fire increases tree defense against bark beetle attacks. Ecology. 2015;96:1846–55.
265. Negrón JF, Popp JB. Probability of ponderosa pine infestation by mountain pine beetle in the Colorado Front Range. For Ecol Manage. 2004;191:17–27.
266. Fettig CJ, Klepzig KD, Billings RF, Munson AS, Nebeker TE, Negrón JF, et al. The effectiveness of vegetation management practices for prevention and control of bark beetle infestations in coniferous forests of the western and southern United States. For Ecol Manage. 2007;238:24–53.
267. Fettig CJ, Gibson KE, Munson AS, Negrón JF. Cultural practices for prevention and mitigation of mountain pine beetle infestations. For Sci. 2014;60:450–63.
268. Fettig CJ, Egan JM, Delb H, Hilszczański J, Kautz M, Munson AS, et al. Management tactics to reduce bark beetle impacts in North America and Europe under altered forest and climatic conditions. In: Bark Beetle Manag Ecol Clim Chang. Academic Press; 2022. p. 345–94.
269. Knapp EE, Bernal AA, Kane JM, Fettig CJ, North MP. Variable thinning and prescribed fire influence tree mortality and growth during and after a severe drought. For Ecol Manage. 2021;479: 118595.
270. Waring KM, O'Hara KL. Silvicultural strategies in forest ecosystems affected by introduced pests. For Ecol Manage. 2005;209:27–41.
271. Foster DR, Orwig DA. Preemptive and salvage harvesting of New England forests: when doing nothing is a viable alternative. Conserv Biol. 2006;20:959–70.
272. Krebs J, Pontius J, Schaberg PG. Modeling the impacts of hemlock woolly adelgid infestation and presalvage harvesting on carbon stocks in northern hemlock forests. Can J For Res. 2017;47:727–34.
273. McDowell NG, Adams HD, Bailey JD, Kolb TE. The role of stand density on growth efficiency, leaf area index, and resin flow in southwestern ponderosa pine forests. Can J For Res. 2007;37:343–55.
274. Kohler M, Sohn J, Nägele G, Bauhus J. Can drought tolerance of Norway spruce (*Picea abies* (L.) Karst.) be increased through thinning? Eur J For Res. 2010;129:1109–18.
275. Brooks JR, Mitchell AK. Interpreting tree responses to thinning and fertilization using tree-ring stable isotopes. New Phytol. 2011;190:770–82.
276. Sohn JA, Gebhardt T, Ammer C, Bauhus J, Häberle KH, Matyssek R, et al. Mitigation of drought by thinning: Short-term and long-term effects on growth and physiological performance of Norway spruce (*Picea abies*). For Ecol Manage. 2013;308:188–97.
277. Bottero A, D'Amato AW, Palik BJ, Bradford JB, Fraver S, Battaglia MA, et al. Density-dependent vulnerability of forest ecosystems to drought. J Appl Ecol. 2017;54:1605–14.
278. Aussenac G, Granier A. Effects of thinning on water stress and growth in Douglas-fir. Can J For Res Can Rech For. 1988;18:100–5.
279. Breda N, Granier A, Aussenac G. Effects of thinning on soil and tree water relations, transpiration and growth in an oak forest (*Quercus petraea* (Matt.) Liebl.). Tree Physiol. 1995;15:295–306.
280. Thomas Z, Waring KM. Enhancing resiliency and restoring ecological attributes in second-growth ponderosa pine stands in Northern New Mexico, USA. For Sci. 2015;61:93–104.
281. Elfstrom LM, Powers MD. Effects of thinning on tradeoffs between drought resistance, drought resilience, and wood production in mature Douglas-fir in western Oregon, USA. Can J For Res. 2023;53:605–19.
282. Sohn JA, Saha S, Bauhus J. Potential of forest thinning to mitigate drought stress: a meta-analysis. For Ecol Manage. 2016;380:261–73.
283. Sohn JA, Hartig F, Kohler M, Huss J, Bauhus J. Heavy and frequent thinning promotes drought adaptation in *Pinus sylvestris* forests. Ecol Appl. 2016;26:2190–205.
284. Jones SM, Bottero A, Kastendick DN, Palik BJ. Managing red pine stand structure to mitigate drought impacts. Dendrochronologia. 2019;57: 125623.
285. Elkin C, Giuggiola A, Rigling A, Bugmann H. Short- and long-term efficacy of forest thinning to mitigate drought impacts in mountain forests in the European Alps. Ecol Appl. 2015;25:1083–98.
286. Domke, GM, Fettig CJ, Marsh AS, Baumflek M, Gould WA, Halofsky JE, et al. Forests. Fifth National Climate Assessment. U.S. Global Change Research Program; 2023. p. 7-1–7-42.
287. Honig K. Lava Mountain Fire, Shoshone National Forest, Wyoming. Public Domain; 2016. https://www.flickr.com/photos/usforestservice/43251352122/in/album-72157674964330963/
288. Flowers R. A large area of pine mortality on National Forest lands at the southern end of the Blue Mountains in Oregon. Public Domain; 2015. https://www.flickr.com/photos/151887236@N05/31099086678/in/album-72157687935138872/
289. Unknown. Dead trees on the Sierra National Forest. Public Domain; 2016. https://www.flickr.com/photos/usfsregion5/26878284272/in/album-72157668018976165/
290. Guy WC. Blowdown from Columbus Day storm that happened October 12, 1962. Public Domain; 1962. https://archive.org/details/usdafs-34232113312
291. Dye AW, Reilly MJ, McEvoy A, Lemons R, Riley KL, Kim JB, et al. Mid-21st century simulated burn probability projections for moist temperate forests of the Pacific Northwest. USDA Forest Service Research Data Archive: Fort Collins; 2024. https://doi.org/10.2737/RDS-2023-0061.
292. Flower CE, Knight KS, Gonzalez-Meler MA. Impacts of the emerald ash borer (*Agrilus planipennis* Fairmaire) induced ash (*Fraxinus* spp.) mortality on forest carbon cycling and successional dynamics in the eastern United States. Biol Invasions. 2013;15:931–44.
293. Nuckolls AE, Wurzburger N, Ford CR, Hendrick RL, Vose JM, Kloeppel BD. Hemlock declines rapidly with hemlock woolly adelgid infestation: impacts on the carbon cycle of southern appalachian forests. Ecosystems. 2009;12:179–90.
294. Domec JC, Rivera LN, King JS, Peszlen I, Hain F, Smith B, et al. Hemlock woolly adelgid (*Adelges tsugae*) infestation affects water and carbon relations of eastern hemlock (*Tsuga canadensis*) and Carolina hemlock (*Tsuga caroliniana*). New Phytol. 2013;199:452–63.
295. Hessl A, Liebhold AM, Leef ML. Dendrochronological reconstruction of the historical invasion of balsam woolly adelgid, *Adelges piceae*, feeding on Canaan Fir, *Abies balsamea* subsp. *phanerolepis* in the Central Appalachian Mountains. Castanea. 2022;87:1–11.
296. Dymond CC, Neilson ET, Stinson G, Porter K, MacLean DA, Gray DR, et al. Future spruce budworm outbreak may create a carbon source in eastern Canadian forests. Ecosystems. 2010;13:917–31.
297. Gunn JS, Ducey MJ, Buchholz T, Belair EP. Forest carbon resilience of eastern spruce budworm (*Choristoneura fumiferana*) salvage harvesting in the northeastern United States. Front For Glob Chang. 2020;3:1–13.
298. Davidson CB, Gottschalk KW, Johnson JE. Tree mortality following defoliation by the European gypsy moth (*Lymantria dispar* L.) in the United States: a review. For Sci. 1999;45:74–84.
299. Gan JB. Risk and damage of southern pine beetle outbreaks under global climate change. For Ecol Manage. 2004;191:61–71.
300. Dodds KJ, Aoki CF, Arango-Velez A, Cancelliere J, D'Amato AW, DiGirolomo MF, et al. Expansion of southern pine beetle into northeastern forests: Management and impact of a primary bark beetle in a new region. J For. 2018;116:178–91.
301. Edburg SL, Hicke JA, Lawrence DM, Thornton PE. Simulating coupled carbon and nitrogen dynamics following mountain pine beetle outbreaks in the western United States. J Geophys Res Biogeosci. 2011;116:G04033.

302. Creeden EP, Hicke JA, Buotte PC. Climate, weather, and recent mountain pine beetle outbreaks in the western United States. For Ecol Manage. 2014;312:239–51.
303. Harvey BJ, Donato DC, Turner MG. Recent mountain pine beetle outbreaks, wildfire severity, and postfire tree regeneration in the US Northern Rockies. Proc Natl Acad Sci U S A. 2014;111:15120–5.
304. Berg EE, Henry JD, Fastie CL, De Volder AD, Matsuoka SM. Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: relationship to summer temperatures and regional differences in disturbance regimes. For Ecol Manage. 2006;227:219–32.
305. Meddens AJH, Hicke JA, Ferguson CA. Spatiotemporal patterns of observed bark beetle-caused tree mortality in British Columbia and the western United States. Ecol Appl. 2012;22:1876–91.
306. Hicke JA, Xu BB, Meddens AJH, Egan JM. Characterizing recent bark beetle-caused tree mortality in the western United States from aerial surveys. For Ecol Manage. 2020;475: 118402.
307. Fettig C, Progar R, Paschke J, Sapio F. Forest insects. Disturb sustain For West United States. PNW-GTR-992, U.S. Dept. Agric. For. Serv. Pac. Northwest Res. Stn., Portland, OR; 2021. p. 81–121.
308. Mitchell RG, Buffam PE. Patterns of long-term balsam woolly adelgid infestations and effects in Oregon and Washington. West J Appl For. 2001;16:121–6.
309. Hicke JA, Davis G, Lowrey L, Xu BB, Smirnova E, Kalachev L. An evaluation of climate influences on balsam woolly adelgid infestations in Idaho. For Ecol Manage. 2023;534: 120849.

Publisher's Note