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A review of PNAS paper entitled:

Land use strategies to mitigate climate change in carbon dense temperate forests *by Beverly E. Law, Tara W. Hudiburg, Logan T. Berner, Jeffrey J. Kent, Polly C. Buotte, and Mark E. Harmon, www.pnas.org/cgi/doi/10.1073/pnas.1720064115 with supporting documents found at: Law et al. 10.1073/pnas.1720064115*

By:

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In summary:

The [Law et al 2018 PNAS paper] is nearly opaque with respect to its underlying assumptions so assessing the validity and accuracy of the simulation modeling effort is onerous. What little is shared in the article regarding inputs to the simulation model ignores the latest developments in wood life cycle assessment and sustainable building design rendering the results at best inaccurate and most likely incorrect. Notwithstanding input errors, the simulation model arrives at unrealistic alternatives for maximizing carbon stored on the land base including the conversion of crop land, failure to acknowledge the risk of increased wildfires while assuming a growth benefit from CO2 fertilization as components of climate change, ignoring the realities of forest land economics as they pertain to reforestation and management effort. The methodology clearly ignores the reality of continued demand for infrastructure that supports our burgeoning population. It also ignores the economic benefits of Oregon's forest sector which produces over 22% of the total US softwood products and nearly 34% of the total US solid softwood products, which means that stopping harvest in Oregon would create tremendous leakage as wood demand would be fulfilled elsewhere where forest productivity is much lower than it is in Oregon.

In detail:

First and foremost, this is a very opaque paper. The assumptions that underlay the assertions of impacts attributable to fire, harvest, and even to forest growth, rely on material buried in cited documents rather than in the text and/or supplementary materials. Supplementary materials provide only the model outputs which are based on methods in the cited papers, or indeed borrowed from the cited papers. This fact makes it onerous to assess the accuracy of the simulation modeling effort, and thus its relevance as a tool for policy makers who are examining what role the forest sector might play in a carbon mitigation strategy. That said, to the extent that I did look to the underlying model structure there were several cautionary elements that should be taken into consideration.

The paper indicates that they conducted a lifecycle assessment to arrive at wood product and harvested carbon impacts. There is no indication in either the paper or the supporting documents that this is so. According to international standards (ISO 14040 and 14044) a life cycle assessment must at a minimum include four elements: goal and scope which identifies the functional unit, system boundaries, assumptions and limitations, allocation methods and impact categories; a life cycle inventory analysis; an impact assessment; and an interpretation. LCA's that do not explicitly address these elements are not life cycle assessments regardless of any assertion to the contrary. Given the scientific credentials of the authors, I am surprised that they would even entertain the idea that they are doing a life cycle assessment when that is clearly not the case.

Within that context they also seem to be missing some key facts regarding service life of buildings (eg even cross laminated timber buildings are only expected to last 30 years), decay rates of buildings (exponential at 2% per year – there are no economic models that would suggest that 2% of buildings are torn down within a year), and reforestation potential.

Data on how they arrived at forest sector emissions is missing so there is no way to evaluate if it is modeled correctly. I.e. supplementary tables S3, S4, and S5 have no information on how they were derived (supplementary materials say that is in the text and the text says it is in the supplementary materials and provides no overview of inputs and outputs as required by an LCA (which authors assert this is)). As near as I can surmise the emissions include cutting all the trees, not replanting them (according to future projections and modeling description) which is not defensible under Oregon law, assuming a cut tree is an immediate emission, that most wood in service goes to the landfill within 30 years, and that substitution is not a relevant part of the wood use story. They cite a 2009 paper as their source for asserting that substitution is not a relevant factor, but that paper does not really address how they incorporate substitution at all. That same paper is referenced for how it accounts for harvested wood products, but there is no real accounting for harvested wood product longevity in the description of the simulation model used. In fact that simulation model indicates that it does not track what happens to the wood once it leaves the forest so somewhere there is either a missing reference or lack of explanation.

On regeneration, there are substantial differences in the model predictions for growth in two regions (Fig S2 with an over prediction of growth by 75% of median value in the coast range and 100% of median value in Blue Mountains) that when propagated over a century will create increasing error rates with time. These two regions together account for nearly ½ the total carbon stocks in the state at the beginning of the simulation and since that median value is used in the monte carlo simulation it will propagate an overestimate of growth and long term carbon storage potential under a do nothing alternative which is the outcome that is predicted to provide the greatest carbon storage over the long term.

There is also a mix of flux data and stock data reported as outputs and it isn't clear (due to lack of transparency in the paper and supplemental materials) if those two are not confounded within the simulation model. The simulation model used is a climate model that uses as inputs the outputs from an ecosystem process model (I presume – it does not say how the inputs are processed) to calculate potential outcomes for future times under a number of different alternative scenarios. What goes into the ecosystem process model determines what comes out and is therefore what is fed into the climate model. This is an elegant method to examine ecosystem processes, make predictions, and understand inflexion points and uncertainty. Unfortunately outcomes often fail to stand up to scrutiny when compared to real world outcomes, particularly when it comes to using them to predict measurable outputs of products as the outputs are totally dependent on model specifications not piece size, volume, diameter distribution, or species - all of which drive harvest decision making processes.

I am struck by the lack of articulating the economic likelihood of these simulation outcomes and how they stack up against recent (as in the past 50 years recent) forest harvest activities. The suggestion that harvest should be stopped on public lands is largely a moot point as harvest has been substantially reduced on public lands in Oregon since about 1993. A review of harvest trends over the past nearly 50 years shows a remarkably consistent harvest level on private lands over the entire period suggesting a sustainable harvest level as the period in question is longer than a rotation length. In fact growth, harvest and mortality data suggest that even when maintaining significant harvest rates these forests are accumulating more carbon stores than are removed. A comparison of recent trends for the USFS lands and private lands shows an increase in volume accumulated on public lands but with 3x more mortality than on private lands, based on plot remeasurements for the 1st 5 years of the 10 year FIA remeasurement period.

Annual Change between 2001-2005 and 2011-2015	NFS	Private
Net Removals	3.035 mmt C	27.909 mmt C
Gross Growth	32.975 MMT C	36.838
Mortality	15.847	4.785

- Private lands harvested 9 times as much as national forest between 2001 and 2010 AND grew more on fewer acres.
- Mortality 3 times as much on national forest than private lands in that same time period.

Since 1993, the harvest on private land amounts to about 82% of total volume removed from about 53% of the total commercially available forested acres based on recent spatial analysis using land ownership data and harvest statistics for western Oregon forests. The volume grown on 10.7 million acres of private land is more than the volume grown on 19 million acres of public land according to the latest forest inventory and analysis data (2017 RPA). That same RPA data shows that conifer harvests in the Pacific coast supply 33.9% of all saw and veneer logs produced in the entire USA and 22.6% of all HWP (including pulp) in the entire USA. Oregon alone produces 5.8% of the world's softwood lumber. To blithely suggest that that amount of harvested wood products would be available elsewhere without substantial leakage is not a defensible assumption.

The analysis on wildfire impacts eliminates the data from the Biscuit fire citing that it is an anomalous occurrence (Fig S3). That model assumption will dramatically affect the outcomes of the simulation and is not substantiated given that the model runs are for a very extended time frame and large westside fires are becoming less anomalous given the doubling of the nationwide fire rate since 2000. This looks like an instance of an a priori assumption driving an outcome as the results report no increase in fire impacts but do report increases in CO2 fertilization growth effects. It isn't clear why the potential benefits of climate change such as CO2 fertilization are included whereas the risks are not. Therefore it would make sense to include the Biscuit given the expectation of additional climate impacts including CO2 fertilization that are reported in the results.

Overall this paper lacks the transparency needed to understand where assumptions drive the predicted model outcomes and how robust they are when real data from surveys and inventory modified to reflect current land conditions provides a completely different picture than the one presented in this paper. For that reason, it should be approached with caution as a tool that can answer complex policy questions about the role of the forest and forest sector in carbon mitigation strategies.

USDA

United States Department of Agriculture

Assessment of the Influence of Disturbance, Management Activities, and Environmental Factors on Carbon Stocks of United States National Forests

Richard Birdsey, Alexa Dugan, Sean Healey, Karen Dante-Wood, Fangmin Zhang, Jing Chen, Alexander Hernandez, Crystal Raymond, James McCarter

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Abstract

This report assesses how carbon stocks at regional scales and in individual national forests are affected by factors such as timber harvesting, natural disturbances, climate variability, increasing atmospheric carbon dioxide concentrations, and nitrogen deposition. Previous baseline assessments of carbon stocks (https://www.fs.fed.us/managing-land/sc/carbon) evaluated observed trends based on forest inventory data but were limited in ability to reveal detailed causes of these trends. The expanded assessments reported here are based on an extensive disturbance and climate history for each national forest, and two forest carbon models, to estimate the relative impacts of disturbance (e.g., fires, harvests, insect outbreaks, disease) and nondisturbance factors (climate, carbon dioxide concentration, nitrogen deposition). Results are summarized for each region of the National Forest System in the main document. A set of regional appendices to this report provides more detailed information about individual national forests within each region. Results are highly variable across the United States. Generally, carbon stocks are increasing in forests of the eastern United States as these forests continue to recover and grow older after higher historical harvesting rates and periods of non-forest land use. In contrast, carbon stocks in forests of the western United States may be either increasing or decreasing, depending on recent effects of natural disturbances and climate change. The information supports national forest units in assessing carbon stocks, quantifying carbon outcomes of broad forest management strategies and planning, and meeting carbon assessment requirements of the 2012 Planning Rule and directives. Results of these expanded assessments will provide context for project-level decisions, separated from the effects of factors that are beyond land managers' control.

Keywords: Forest carbon stock, national forest, land management, natural disturbance, climate change

Cover: Examples of disturbance to forests are, clockwise from top left, wildfire in Columbia River Gorge, Oregon (photo: ©Christian Roberts-Olsen via Shutterstock), harvested pine and spruce logs (photo: ©AVN Photo Lab via Shutterstock), broken trees after a powerful hurricane (photo: ©Victor Lauer via Shutterstock), and damage by spruce bark beetles (photo: © Klaus Reitmeier via Shutterstock). Image in top right corner: aspen leaves (photo: ©Labrador Photo Video via Shutterstock).

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Executive Summary

This report assesses how carbon (C) stocks on forest land of the Forest Service, U.S. Department of Agriculture, National Forest System are affected by timber harvesting, natural disturbances, aging, climate variability, increasing atmospheric carbon dioxide (CO_2) concentrations, and nitrogen (N) deposition. Carbon assessments were developed for individual national forests or small groups of national forests which have been administratively combined; assessments were then aggregated to regional scales. The information supports national forest units in making measurable progress toward meeting C assessment requirements of the 2012 Planning Rule and directives. Results of these assessments will help forest managers quantify C outcomes of broad forest management strategies and plan alternatives, and provide context for project-level decisions, separated from the effects of factors that are beyond land managers' control (e.g., climate).

Previously released baseline assessments of C stocks reported observed trends on forest land in each national forest based on Forest Service Forest Inventory and Analysis (FIA) data. These long-term changes represent the aggregate effects of all factors, reflecting the history of land management, disturbance, and environmental variability (e.g., climate). The C stocks in wood products harvested from the national forests at the regional scale were assessed using a harvested wood products (HWP) model, which applied a production accounting approach (Stockmann et al. 2012). These initial baseline assessments (e.g., USDA FS 2015g) were limited in ability to reveal causes of observed trends, which is the main purpose of this report.

The expanded assessments reported here build on the baseline assessments by incorporating detailed disturbance, climate, and atmospheric histories of each national forest, and two additional forest C models: the Forest Carbon Management Framework (ForCaMF) and Integrated Terrestrial Ecosystem Carbon (InTEC) model. The combination of extensive data compilation and modeling allows estimation of the relative impacts of disturbance factors (fires, harvests, insect outbreaks, wind, disease, and recovery and aging) and nondisturbance factors (climate, N deposition, CO_2 concentrations) on C stocks. Disturbance histories account for the main categories of disturbance—fire, harvesting, insects, disease, and abiotic (wind) - compiled by intensity of impact on tree canopy cover. Manually verified disturbance maps were created for the period 1990 through 2011 by integrating satellite imagery with agency records of harvests, the multiagency Monitoring Trends in Burn Severity (MTBS) database of fires, and annual aerial detection surveys of insect and abiotic disturbances. For years prior to 1990, forest stand ages derived from forest inventory data were used as a proxy for historical stand-replacing disturbance events. Additional spatial datasets such as climate records and measurements of N deposition and CO_2 concentrations were integrated to model the effects of environmental factors on forest C stocks and trends.

The ForCaMF model incorporates FIA data, satellite-based disturbance histories, and the Forest Vegetation Simulator (FVS) to provide regional and forest-level assessments of the impact of different kinds of disturbance on ecosystem C storage. These assessments take the form of estimating how much more C would be stored on each national forest if disturbances that took place from 1990 through 2011 had not occurred.

The InTEC model is a process-based biogeochemical model driven by satellite-based disturbance histories and stand-age information, monthly climate and atmospheric data, and productivity parameters to estimate the relative effects of disturbance and nondisturbance factors on forest C accumulation and annual C fluxes. Within the model, disturbance factors alter terrestrial C balances and influence stand-age structures, which in part drive the rate of C uptake. Nondisturbance factors influence photosynthesis, respiration, and other variables related to growth rates that determine C production in the model. Historical C dynamics are estimated progressively through time from 1950 through 2011, and the results at any point in time are the accumulated effects of all factors since the starting year.

In this main report, model results are summarized for the combination of all national forests in each National Forest System region. The available regional appendices provides more detailed information about individual national forests within each region.

The C stock trends (e.g., USDA FS 2015g) and the relative influence of disturbance and environmental factors affecting C stocks in each region between approximately 1990 and 2011 are summarized in table 1 and figure 1. Results indicate that forest C trends and the relative impacts of disturbance and environmental factors are highly variable across the United States. Generally, C stocks are increasing but at a declining rate in forests of the eastern United States as stands continue to grow older after historical harvesting rates and periods of nonforest land use. Carbon stocks in forests of the western United States may be either increasing or decreasing. These changes depend on recent effects of natural disturbances and climate change, which have caused many areas of forest in the West to switch from a C sink to a C source during the last two decades. Climate has had variable effects, largely depending on temperature trends and drought effects. Increasing atmospheric CO_2 concentrations and N deposition have had consistently positive effects on C stocks, partially offsetting C losses from natural disturbances, particularly fire and insects. Changes in the stock of C in wood products and solid waste disposal are relatively small or negative because harvest rates have declined significantly over many decades. Emissions of C from previously harvested wood are about the same as the C that is stored in newly harvested wood.

Results of the various models contained in this assessment and the previous baseline assessments (e.g., USDA FS 2015g) may vary due to differences in datasets used and modeling approaches applied. For instance, changes in forest C stocks reported in the baseline assessments rely completely on empirical FIA data that track long-term changes representing the aggregate effects of all factors. These assessments indicate that C stocks in forests of most regions have been increasing but at variable rates. However, because the inventory remeasurement period is 5 to 10 years, inventories may not fully account for the effect of more recent disturbances, which have been increasing over the last decade, particularly in the West. Both the ForCaMF and InTEC models make use of more contemporary satellite observations of disturbances, so they may better capture the C impacts of recent disturbance events.

Eastern Region Results

As part of the first region to be widely settled in the United States, the forests of the Eastern Region have a long history of unregulated harvesting and conversion to agriculture. Harvesting has been the most common and consistent disturbance type in the region since 1990, although the annual area harvested generally did not exceed 0.25 percent of the landscape. Harvests between 1990 and 2011 resulted in the removal of approximately 1.6 percent of nonsoil C stocks from the forest (fig. 1). Wind and fire also have been significant in the region, accounting

Table 1 - Cumulative changes in carbon (C) stocks (Tg) for regional and national scales, from C assessments for individual national forests, 1990–2011.

Region	Change in forest C stocks[a]	Disturbance impacts[b]	Climate and atmospheric impacts[c]	Change of C stock in wood products[d]	Summary ranking of main factors affecting C stocks[e]
Eastern	93	-9	63	3.5	• Harvesting • Climate and atmosphere • Fire
Southern	193	-21	16	2.8	• Harvesting • Fire • Climate and atmosphere • Insects • Wind
Northern	123	-37	23	0.3	• Fire • Disease • Climate and atmosphere • Harvesting
Rocky Mountain	32	-44	49	1.3	• Insects • Fire • Climate and atmosphere • Harvesting
Intermountain	43	-32	32	0.6	• Fire • Insects • Climate and atmosphere • Harvesting
Pacific Northwest	117	-42	67	-0.8	• Fire • Harvesting • Climate and atmosphere • Insects
Southwestern	-31	-9	4	-0.3	• Fire • Climate and atmosphere • Harvesting
Pacific Southwest	63	-26	42	0.1	• Fire • Climate and atmosphere • Harvesting
Alaska	17	-2	-	0.4	• Harvesting
All regions	650	-225	296	7.9	• Fire • Harvesting • Climate and atmosphere • Insects

[a] Observed changes in C stocks from Carbon Calculation Tool (CCT), which is based on forest inventory data reported in USDA Forest Service (2015a–i). This column represents an independent estimate of net biome production, and is not the sum of other columns.
[b] Harvesting and natural disturbances from Forest Carbon Management Framework (ForCaMF). Results do not include the soil pool.
[c] Net effects of nondisturbance factors—temperature, precipitation, carbon dioxide concentration, and nitrogen deposition - from the Integrated Terrestrial Ecosystem Carbon (InTEC) model. Results for the Northern and Southern Regions are for 1990 through 2010.
[d] Change of C in wood products and landfills based on the IPCC production accounting approach to harvested wood products (IPCC 2006; Stockmann et al. 2012).
[e] Factors ranked by approximate magnitude of effect in descending order of importance.
[f] The effects of forest aging as modeled by InTEC are not explicitly reported as a separate column in this table. However, all factors affecting C stocks, including aging, are inherently included in the *Change in forest C stocks* column.

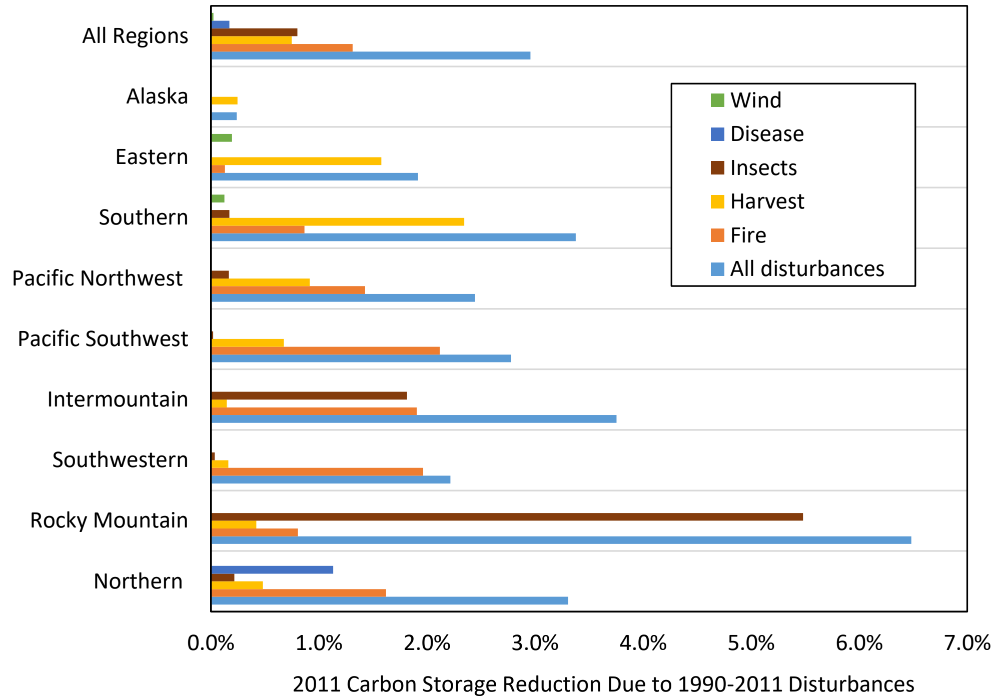

Figure 1. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by National Forest System region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates. Results from the Northern and Southern Regions reflect disturbances occurring from 1990 through 2010.

for the loss of 0.2 percent and 0.1 percent of nonsoil C stocks, respectively. The regionwide stand-age distribution shows that about 50 percent of the forests are greater than 80 years old and indicates a period of elevated stand establishment occurring about 70 to 100 years before the end of the study period or from approximately 1910 to 1940. Both temperature and precipitation have increased slightly over the past few decades, and have had a mostly positive effect on C stocks. Nitrogen deposition has been declining while CO_2 levels have been increasing, with both continuing to have positive effects on C stocks. Considering all factors, the national forests in the Eastern Region have mostly maintained a C sink from 1950 through 2011. The overall trend shows that this C sink has been declining due to disturbance and aging effects but may have stabilized over the past two decades due to the positive effects of nondisturbance factors.

Southern Region Results

In the late 1800s and early 1900s, agricultural expansion and large-scale timber extraction became the dominant driving forces of forest change in the South, followed by the establishment of national forests from marginal lands and a focus on forest restoration beginning around the 1920s. Timber harvesting from 1990 through 2011 accounted for the removal of about 2.4 percent of the nonsoil carbon stocks from the forest ecosystem (fig. 1), though harvests never affected more than 0.5 percent of the landscape in any single year. Fire has also played an important role in the forest C trends in the Southern Region, resulting in the loss of about 0.9 percent of nonsoil C stocks by 2011, while insects and abiotic factors had relatively small effects on C storage across the region (fig. 1). The regionwide stand-age distribution shows that most stands in the Southern Region are older than 70 years, with a pulse of stands that established from about 1910 to 1940. Temperature in the region has been stable and precipitation has increased slightly since 1950, although variability in weather has been significant. Consequently, climate trends have had a moderately positive effect on C stocks, but effects of droughts and heat waves have been periodically sizable. Both atmospheric CO_2 concentrations and N deposition have increased over the past few decades, resulting in a positive effect on C stocks. Overall, national forests in the Southern Region were mostly a C sink because the positive effects of climate, atmospheric CO_2, and N deposition helped to offset the C losses due to disturbances and aging forests.

Northern Region Results

In the 1880s, large areas of forest land were cleared for agriculture, settlements, and railroad expansion. This history is reflected in the stand-age structure and continues to influence C trends. It was not until the 1940s that the Northern Region experienced large-scale logging operations on national forest lands. In addition to timber harvesting, the forest C legacy of the Northern Region is tied to its history of fires and fire suppression. Recently, fires and root disease have been the dominant disturbances affecting carbon stocks, resulting in the loss of about 1.6 percent and 1.1 percent of nonsoil carbon stocks, respectively, between 1990 and 2011 (fig. 1). The regionwide stand-age distribution in 2010 shows that most stands in the Northern Region are older (>80 years old) with a distinct pulse of stands which established between approximately 80 and 110 years before the end of the study period, or from about 1900 to 1930. Consequently, accumulated C in forests has declined, largely as a result of disturbance and aging effects, as most stands have already reached older, less productive ages. Exacerbating these disturbances and aging effects, climate variability and the recent warming trend have had a mostly negative effect on C stocks. Increases in N deposition and atmospheric CO_2 concentrations have both had positive effects, thus helping to partially offset the negative effects of recent disturbances and aging.

Rocky Mountain Region Results

Like many regions of the western United States, the forests of the Rocky Mountain Region were subject to logging, livestock grazing, mining, and clearing of forest for agriculture and human settlements by the latter half of the 19th century. Forest regrowth following historical land use and forest management activities such as fire suppression caused an increase in C accumulation in the mid-20th century, but stocks have declined in recent decades as these stands have aged and become less productive. Timber harvest volumes greatly declined in the early 1990s and have remained relatively low since. The regionwide stand-age distribution shows that most of the stands in the Rocky Mountain Region are older (>80 years old) with a notable pulse of stands which established from about 1880 to 1930. During the latter part of the study period, the Rockies experienced severe bark beetle outbreaks that caused widespread tree mortality and resulted in the loss of around 5.5 percent of forest C stocks between 1990 and 2011 (fig. 1). Over the past few decades, several forests in the region experienced large and severe wildfires. Fires caused the loss of approximately 0.8 percent of nonsoil C stocks in the region, while harvests resulted in the removal of about 0.4 percent of nonsoil C stocks. The climate in the Rocky Mountain Region has on average grown warmer and slightly wetter, though severe droughts are not uncommon. Though the baseline assessments indicate that across the region forests have maintained a C sink, forests are likely to be switching to a C source due to negative disturbance and aging effects. Nondisturbance factors had a positive effect, thus helping to maintain the C sink.

Intermountain Region Results

The forest C legacy of the Intermountain Region is tied to its history of land use and fire management, as well as more recent, severe natural disturbances. Annual timber output declined precipitously in the early 1990s and has remained low ever since. Recently, severe bark beetle outbreaks have resulted in extensive tree mortality in parts of the Intermountain Region. Despite regeneration after recent disturbances, the regionwide stand-age distribution in 2011 shows that about half of the forests are greater than 100 years old and therefore undergoing declines in productivity and C accumulation. From 1990 through 2011, fires significantly affected C storage in the region, reducing nonsoil C stocks by about 1.9 percent (fig. 1). However, extensive tree mortality due to increased insect activity in the latter part of this period substantially altered the regional disturbance signature upon C stocks, reducing nonsoil C stocks by about 1.8 percent. Increasing levels of CO_2 and N deposition caused forests to accumulate more C and helped to counteract the C declines due to disturbances and stand aging. Climate factors, in particular, increasing temperatures, have also caused a decline of forest C since about 2000. Accounting for all effects, the forests have generally had a very low rate of increase in C stocks, and may already be switching from a C sink to a C source.

Pacific Northwest Region Results

In the late 1800s, increased Euro-American settlement and the arrival of the transcontinental railroad transformed the Pacific Northwest into one of the highest timber-producing regions of the country. Annual timber output in the Pacific Northwest increased in the mid-20th century, but has declined since the 1990s and remained low ever since. Due to this history of land use and timber harvesting, the stand-age distribution is shifting toward older ages, causing recent declines in productivity and C accumulation. Of all disturbance factors, wildfires occurring from 1990 to 2011 have had the most significant effect on C storage, causing the loss of about 1.4 percent of nonsoil C stocks by 2011 (fig. 1). Timber harvests reduced nonsoil C storage by approximately 0.9 percent since 1990, while insect outbreaks have also resulted in tree mortality and C losses in parts of the region. Although there has been significant interannual variability in both temperature and precipitation, between the 1950s and 2011 the climate has on average become much warmer and somewhat drier, but has had little effect on C stocks in recent years. Increasing levels of atmospheric CO_2 and N deposition caused forests to accumulate more C and helped to counteract the C declines from disturbances and the aging stands. The national forests in the Pacific Northwest Region have generally increased in C stocks.

Southwestern Region Results

Though Euro-American settlement brought livestock grazing, mining, and timber harvesting to the region, the timber industry of the Southwest has remained relatively modest compared to other regions of the United States. This history of land use as well as wildfires followed by nearly a century of fire suppression policies has resulted in a relatively older age structure, characterized by lower productivity and rates of C accumulation. More recently, fire has also been the primary disturbance shaping forests of the Southwestern Region, and today prescribed burning is commonly used to manage fuel loads. The acceleration of fire activity after 2000 within the region resulted in a surge in fire's impact on C stocks. Between 1990 and 2011, fires resulted in the loss of approximately 2 percent of nonsoil carbon storage (fig. 1). Insects and harvesting each accounted for C losses that were less than 0.5 percent of total C nonsoil storage. Climate has on average become much warmer, while the region has experienced periodic droughts. The national forests in the Southwestern Region were a C source from 1990 through 2011. In the early 2000s, drought conditions combined with elevated temperatures enhanced the C source. The positive effects of increasing N deposition and CO_2 concentrations partially offset the negative disturbance and aging effects and the more recent negative climate effects.

Pacific Southwest Region Results

National forests in California were heavily logged in the late 1800s, and timber harvest volumes increased steadily into the early 1900s to support mining operations, immigration, and development. The regionwide stand-age distribution indicates a pulse of stands establishing from around 1900 to 1940, reflecting recovery after these historical disturbances and land use changes and leading to age-induced declines in C accumulation. Both land use history and recent changes in climate have led to large and severe disturbances in the Pacific Southwest Region over the past few decades. Fires were particularly large in 2000 and 2008, when they affected more than 1 percent of the region's forest land. Fires from 1990 to 2011 resulted in the loss of about 2.1 percent of nonsoil forest C stocks by 2011 (fig. 1). The C impact of harvest within the region was relatively stable from 1990 through 2011, causing the removal of about 0.7 percent of nonsoil C stocks. Forests have had very modest increases in C stocks since 1990, and may already be switching from a C sink to a C source because of disturbance and aging effects, as well as increasing temperatures and droughts. Increases in N deposition and atmospheric CO_2 concentrations have consistently enhanced the C sink and helped to counteract the negative disturbance and aging effects and climate effects.

Alaska Region Results

The forest C legacy of the Alaska Region is tied to the history of timber harvesting and natural disturbances. Extensive timber harvests began in the 1950s, largely in high volume, old-growth stands. Timber harvests steeply declined in the 1990s and since 2000 have remained at low levels, resulting in the removal of approximately 0.2 percent of the nonsoil C stocks (fig. 1). Unlike much of the western United States, fires are relatively minor in south-eastern Alaska as a result of the cool, moist, temperate-rainforest conditions. Harvest activity was the only disturbance process to have a significant effect on C stocks in the region; the detected fire, wind, and insect activity on the two national forests had negligible impact on C storage. Due to its high latitude, Alaska has seen a more rapid increase in temperatures over the last century than any other region in the United States. National forests in Alaska have shown a very modest increase in C stocks since 1990.

1. Introduction

Greenhouse gas (GHG) concentrations have increased significantly since 1750 and have greatly exceeded preindustrial values (IPCC 2007). Human activities such as fossil fuel burning, industrial production, land use change, and agriculture are responsible for releasing large amounts of carbon dioxide (CO_2) and other GHGs into the atmosphere. Globally, about half of the emitted GHGs are absorbed by oceans and land, with the remainder staying in the atmosphere for long periods of time (Le Quéré et al. 2018). Greenhouse gases trap energy in the atmosphere and cause it to warm. This phenomenon, called the "greenhouse effect," is necessary to sustain life on Earth. However, the large amounts of GHGs that humans are releasing to the atmosphere are causing the surface temperature of the Earth to increase with a number of associated large-scale changes (USGCRP 2017), many of which are detrimental to human health and ecosystems.

The Forest Service, U.S. Department of Agriculture (hereafter, Forest Service) recognizes the vital role that our Nation's forests and grasslands play in carbon (C) sequestration. Box 1 defines this and other important terms that are commonly used to depict the C cycle and C management activities. Carbon sequestration by forests is one way to mitigate GHG emissions by offsetting a portion of those emissions through removal and storage of C from the atmosphere. Carbon dioxide uptake by forests in the conterminous United States, and storage in live and dead organic matter and harvested wood products, offset approximately 11 percent of the national total CO_2 emissions annually over the last decade (USEPA 2018). Forests and other ecosystems generally act as C sinks because, through photosynthesis, growing plants remove CO_2 from the atmosphere and store it. However, forests may become sources of CO_2 during and after disturbances such as wildfire and timber harvesting. This is usually a temporary effect before the disturbed forests begin to regrow and resume their function as C sinks. Recent estimates of net annual storage indicate that globally and in the United States, forests are an important C sink, removing more C from the atmosphere than they emit (Pan et al. 2011).

The Forest Service also recognizes that C sequestration and storage is one of many ecosystem services provided by forests and grasslands; these services also include clean water, clean air, biodiversity, wood products, wildlife habitat, food, and recreation. Changes in weather patterns and extreme weather events place forest and grassland ecosystems and their services at risk. The National Forest System (NFS) constitutes one-fifth (22 percent) of the Nation's total forest land area and contains one-fourth (24 percent) of the total C stored in all U.S. forests, excluding interior Alaska. Thus, management of these lands and disturbances influence sequestration and storage of C and mitigation of GHG emissions. The future trajectory of C stocks on the national forests will be influenced by the variability of conditions and disturbance regimes such as wildfire, insect outbreak, and extreme weather across the United States.

Sequestration of C in U.S. forests is projected to decline over the next 25 years primarily due to land use change (net loss of forest area) and forest aging with corresponding slower growth (Wear and Coulston 2015). There are strong regional differences. For example, the forest C sink of the Rocky Mountain Region is projected to decline rapidly or change to a net source due to fire, insect outbreaks, and aging. Eastern forests, which are more intensively managed and lack such widespread disturbances, are expected to continue to be C sinks for

Box 1

Terminology

Accumulation - The increase of carbon in a location over a period of time.

Carbon units - Megagram (Mg): 1,000 kilograms (2,204.6 pounds) = 1 tonne; Teragram (Tg): 1,000,000 tonnes

Mitigation - Measures to reduce the amount and rate of future climate change by reducing emissions of heat-trapping gases or removing carbon dioxide from the atmosphere.

Sequestration - Storage of carbon through natural, deliberate, or technological processes in which carbon dioxide is diverted from emission sources or removed from the atmosphere and stored biologically in oceans and terrestrial environments (vegetation, soils, and sediment), or in geologic formations.

Sink - A physical location where carbon is removed from the atmosphere and stored, either through natural or technological processes. Entire ecosystems, specific ecosystem components (e.g., forest, soil), or political boundaries may be characterized as a sink.

Source - A physical location where carbon is released to the atmosphere, either through natural or technological processes. Entire ecosystems, specific ecosystem components (e.g., forest, soil), or political boundaries may be characterized as a source.

Stock - A term referring to the mass of carbon contained within a particular compartment, or pool, within the Earth system.

Storage - The action of putting carbon in a location that prevents its release to the atmosphere for a period of time.

Uptake - The action of taking up carbon dioxide from the atmosphere by plants.

several decades.

Forests are highly dynamic systems that are continuously repeating the natural progression of establishment, growth, death, and recovery, while cycling C throughout the ecosystem and the atmosphere. This cycle, which drives overall forest C dynamics, varies geographically and by forest type, and by the frequency, magnitude, and type of disturbance events. Natural and anthropogenic disturbances can cause both immediate and gradual changes in forest structure, which in turn affect forest C dynamics by transferring C between the different ecosystem and atmospheric C pools (fig. 2). While disturbances may be the predominant drivers of forest C dynamics (Pan et al. 2011), environmental factors (e.g., the concentration of CO_2 in the atmosphere), the availability of key forest nutrients, such as nitrogen (N), and climate variability influence forest growth rates and consequently the cycling of C through a forest ecosystem (Hyvönen et al. 2007; Pan et al. 2009). Thus, an accurate and comprehensive assessment of forest C stocks and trends and the drivers that influence them must include the effects of both disturbances and environmental factors.

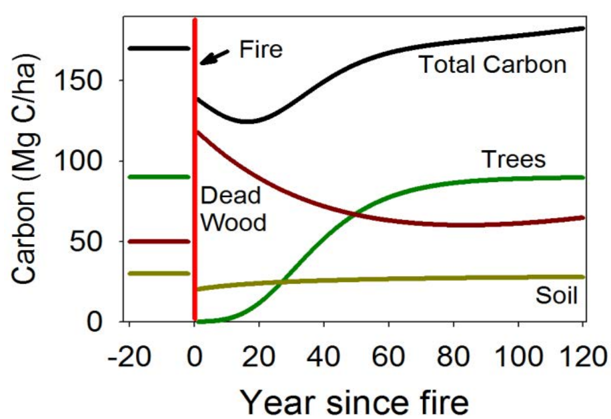

Figure 2. The effect of a stand-replacing fire on forest ecosystem carbon pools and total ecosystem carbon (source: McKinley et al. 2011).

The long-term capacity of forest ecosystems and harvested wood products to sequester and store C depends, in large part, on their health, resilience, adaptive capacity, and utilization of timber (McKinley et al. 2011). Under a changing climate, forests are increasingly affected by many factors such as multiyear droughts, insect and disease epidemics, wildfires, and catastrophic storms (Cohen et al. 2016; Westerling et al. 2006). Maintaining healthy forest structure and composition may not eliminate disturbance, and may in fact entail additional low-magnitude disturbance, but is likely to reduce the risk of large and long-term C losses through catastrophic disturbance (Millar and Stephenson 2015). Forest ecosystems capable of adapting to changing conditions will sequester C and store it more securely over the long term, while also furnishing woody materials to help reduce fossil fuel use. For forests managed for timber products, it is important to account for the C that is retained in harvested wood as well as substitution effects of using wood instead of other energy-intensive materials, because these quantities may be large and should not be considered as emitted CO_2 (Perez-Garcia 2005).

A first step toward managing for healthy forests is understanding how past patterns of disturbance and climate have affected their ecosystem functions such as C storage. A nationally consistent C assessment framework has been developed for the NFS to deliver forest C disturbance information for every region and individual national forest. This report builds on the baseline C storage assessments produced for each region (https://www.fs.fed.us/managing-land/sc/carbon). It focuses on disturbance and environmental effects on ecosystem C, setting aside temporary storage of harvested C in product pools. The storage value of C in harvested wood products is quantified in the existing regional baseline assessments, and related analyses have appeared elsewhere (Healey et al. 2009; Stockman et al. 2012). Ongoing work to integrate ecosystem and product life cycle dynamics will allow the sideboards of future Forest Service assessments to expand beyond the ecosystem boundary.

Box 1 cont.

Processes that Exchange Carbon with the Atmosphere

Combustion - Process of burning something; occurs when a substance such as wood, coal, or natural gas reacts with oxygen to produce carbon dioxide, water vapor, heat and energy.

Decomposition - Natural process of dead organisms being rotted or broken down into smaller bits; decomposers respire carbon dioxide to the surrounding soil and air.

Net biome production - The difference between the amount of organic carbon fixed by photosynthesis in an ecosystem and the loss of carbon from autotrophic and heterotrophic respiration and disturbances.

Net ecosystem production - The difference between the amount of organic carbon fixed by photosynthesis in an ecosystem and the loss of carbon from autotrophic and heterotrophic respiration.

Net primary production - The net uptake of carbon dioxide by plants through gross primary productivity in excess of losses from plant, or autotrophic, respiration.

Photosynthesis - The process by which green plants, algae, and other organisms use sunlight to synthesize energy fromcarbon dioxide and water. Photosynthesis in plants generally involves the green pigment chlorophyll, consumes carbon dioxide and water, and generates oxygen as a byproduct.

Respiration - Metabolic pathways that break down complex molecules to release chemically stored energy for maintenance, growth, and reproduction, and that result in the release of waste products such as carbon dioxide, nitrous oxide, or methane.

2. Purpose of the Assessment

This report expands on previous assessments of baseline C stocks across individual national forests and at the regional scale by assessing how stocks at those scales are affected by factors such as timber harvesting, natural disturbances, land use change, climate variability, increasing atmospheric CO_2 concentrations, and N deposition. The likelihood of management activities affecting future disturbance rates is not assessed. Such assessments are pursued through stand- and landscape-level risk analysis (e.g., Ager and Vaillant 2010), and are beyond the scope of this report. The goal of this report is to assess the causes of changes in C stocks as quantified by C monitoring programs such as forest inventories.

Existing guidelines for considering forest C are found in various Forest Service policies, programs, and activities such as

- the 2012 Planning Rule and directives ,
- National Roadmap for Responding to Climate Change (PDF, 3.3MB),
- Climate Change Performance Scorecard (PDF, 2.2MB),
- Ecosystem Restoration Policy and Directive (FSH 2020),
- other internal guidance for planning (Climate Change - Land Management & Project Planning),
- project-level decisions, and
- several State and Private Forestry programs.

The information in this C assessment directly supports NFS units in making measurable progress on Climate Change Performance Scorecard Element 9 (Carbon Assessment and Stewardship), while helping forests meet C assessment requirements of the 2012 Planning Rule and directives. Scorecard Element 9 poses the following questions:

> *Does the Unit have a baseline assessment of carbon stocks and an assessment of the influence of disturbance and management activities on these stocks? Is the Unit integrating carbon stewardship with the management of other benefits being provided by the Unit?*

Initial baseline assessments of C stocks were produced in early 2015 using the Carbon Calculation Tool (CCT), which summarizes the data collected by the Forest Service's Forest Inventory and Analysis program, and data about harvested wood products from regional Forest Service reports. The baseline assessments (USDA FS 2015a–i) meet the intent of assessing observed trends but are limited in ability to reveal detailed causes.

The expanded assessments reported here evaluate the influence of disturbance, management, and environment by integrating two additional forest carbon models - the Forest Carbon Management Framework (ForCaMF) (Healey et al. 2014, 2016) and the Integrated Terrestrial Ecosystem Carbon (InTEC) model (Chen et al. 2000a; Zhang et al. 2012) - to calculate the relative impacts of disturbance (e.g., fires, harvests, insect outbreaks, disease) and nondisturbance factors (climate, N deposition, CO_2 concentrations). Results of these expanded assessments will help forest managers quantify C outcomes of broad forest management strategies and plan-level decisions, separated from the effects of factors that are beyond their control. See Dugan et al. (2017) for further details on how these C models (CCT, ForCaMF, and InTEC) may be integrated to provide useful information for characterizing C

Box 2

Forest Service Forest Carbon Principles

Although carbon is a relatively new consideration in land management, including it is consistent with sustaining the health, diversity, and productivity of the Nation's forests and grasslands to meet the needs of present and future generations. Land management actions on public and private forests and grasslands can be designed to achieve carbon outcomes while meeting other sustainable resource management objectives. Forests are important in capturing and storing carbon, both onsite and in products, and management of these lands can contribute to mitigating climate change. The following principles should be at the forefront when considering carbon along with other management objectives.

1. Emphasize ecosystem function and resilience. (Function First). Carbon sequestration capacity depends on sustaining and enhancing ecosystem function. Long-term sequestration should be planned in the context of changing climate and other environmental drivers. Management actions that help maintain resilient forests or transition vulnerable forests to a fully functioning and resilient state are more likely to store sequestered carbon over the long run.
2. Recognize carbon sequestration as one of many ecosystem services. (One of Many Services.) Carbon sequestration is one of many ecosystem services provided by forests and grasslands. Strategies for including carbon in forest and grassland management must consider the suite of resources and outcomes desired from management actions. A balanced and comprehensive program of sustainable management will consider many ecosystem services, including carbon sequestered in biomass, soils, and wood products.
3. Support diversity of approach (Diverse Approaches). Recognize that decisions about carbon in America's forests are influenced by ownership goals, policy, ecology, geography, socioeconomic concerns, and other factors that vary widely. The Forest Service supports a variety of approaches to managing carbon and deriving value from carbon that are compatible with the objectives of different owners. A wide diversity in approaches can also foster more rapid learning about forest, grassland, and carbon management in the context of climate adaptation.
4. Consider system dynamics and scale in decisionmaking. (Scale and Timeframe). Different ecosystems sequester carbon in different ways, at different rates, and within differing mosaics of landscape plans and trends. The carbon effects of forest and grassland management options should be evaluated within the carbon dynamics of long timeframes and landscape scales, with explicit consideration of uncertainties and assumptions. Where practical, system dynamics should be broadened beyond the ecosystem to consider full life-cycle impacts of decisions, including carbon storage in forest products and substituting wood-based options for fossil fuel-intensive applications.
5. Use the best information and analysis methods. (Decision Quality). Base forest management and policy decisions on the best available science-based knowledge and information about system response and carbon cycling in forests, grasslands, and wood products. Use this information wisely by dealing directly with uncertainties, risks, opportunities, and tradeoffs through sound and transparent risk management practices. Forest plan revision, project-level implementation, and other decision processes should consider tools and approaches that explicitly address uncertainty, risks, and opportunities about climate impacts and forest carbon response.

Including carbon in land management planning activities through these considerations supports the widespread approaches of ecosystem and watershed management and does not require significant alteration of management strategies and approaches. A balanced approach to including carbon in management activities may, however, complement existing objectives and priorities even as it helps mitigate the Nation's GHG emissions and promote climate adaptation.

dynamics and the relative contributions of driving factors, while also fulfilling Forest Service guidance.

Using the baseline and expanded assessments, and the draft Forest Carbon Principles outlined in box 2, NFS units can begin to integrate C stewardship thinking and practices into management activities, thus helping to address the final question of Scorecard Element 9: Is the Unit integrating carbon stewardship with the management of other benefits being provided by the Unit? This requirement could be accomplished through use and application of available information related to C through the land management planning process, program guidance, forest plans, project plans and analyses, or other strategic program planning. Units are encouraged to review the Scorecard Guidance, including Scorecard Appendix F (PDF, 471KB).

Units can apply these data in their forest management planning and practices by asking questions such as:

- Has the land management plan area sequestered and stored C in the recent past, or has it emitted stored C?
- How have disturbances, projects, and activities influenced C stocks (including harvested wood products) in the past and how may they affect C stocks in the future?
- Are existing conditions and trends of forest vegetation and soils indicating that the plan area is a C sink or source?
- Under existing plan guidance and alternatives being considered, what is the likely future trend of the plan area in sequestering and storing C, including in harvested wood products?
- What is the risk of loss to C storage due to potential disturbance factors?
- Are there opportunities to change plan components to influence these trends?

These expanded assessments are produced following methods described in several cited source documents.

3. Forest Carbon Management and Stewardship

The Forest Service is leading government agencies in the national conversation and action on forest C. The basic approach involves managing C through managing the health and productivity of the Nation's forests. The approach focuses on managing risks to the health, productivity, and ability of the resource to provide the goods and services called for in management plans. Management actions have C outcomes and those are considered among the benefits being managed. Forest systems are dynamic and emit and capture C regardless of human intervention. The Forest Service C strategy is embedded in a larger adaptation strategy for managing the resource that considers multiple impacts of natural and anthropogenic stressors.

Carbon management is an aspect of sustainable land management (Janowiak et al. 2017) targeted to maintain the long-term health and productivity of forests and grasslands, and to maintain a flow of all of their benefits. Carbon adds another dimension to the work. There will be both tradeoffs and synergies between C flows and other services, and balancing these interactions will continue to be part of the mission in managing these resource (box 2).

The Forest Service's goal is to manage system vulnerability to multiple stressors through adaptation and mitigation activities. Management principles consider C and other benefits flowing from forests, integrate climate adaptation and mitigation, and balance C uptake and storage among a wide range of ecosystem services. The Forest Service also strives to provide for social, economic, and ecological sustainability.

Forest management strategies include retaining and protecting forest land from conversion to nonforest uses; restoring, maintaining, and enhancing resilient forests that are better adapted to a changing climate and more resistant to catastrophic wildfires and other stressors; and reforesting lands impacted by catastrophic wildfires and other disturbances. Other strategic management options for reducing GHG emissions include changes in land management, afforestation (and other land use changes), avoiding loss of forest land, adding to the harvested wood product pool, and bioenergy (Scorecard Appendix F (PDF, 471KB)).

4. Methods and Uncertainty

The expanded assessments reported here are based on a detailed disturbance, climate, and atmospheric history of each national forest, and two additional forest C models: the Forest Carbon Management Framework (ForCaMF) and Integrated Terrestrial Ecosystem Carbon (InTEC) model. The ForCaMF model provides forest-level assessments of the impact of different kinds of disturbance (e.g., fires, harvests, insect outbreaks, abiotic, disease) on ecosystem C storage. The InTEC model is a process-based biogeochemical model driven by monthly climate data, vegetation parameters, and forest disturbance information to estimate the relative effect of disturbance and aging and nondisturbance factors (climate, N deposition, CO_2 concentrations) on forest-level C accumulation and fluxes. The combination of extensive data compilation and modeling provides an approach to estimate the relative impacts of disturbance (fires, harvests, insect outbreaks, abiotic, disease, aging) and nondisturbance factors (climate, N deposition, CO_2 concentrations) on C stocks.

This report follows the international reporting requirements established under the umbrella of the Intergovernmental Panel on Climate Change (IPCC) as implemented for the forestry sector of the U.S. GHG inventory compiled by the U.S. Environmental Protection Agency (USEPA 2018). The seven main forest sector carbon pools, generally defined by IPCC and adapted for reporting in the United States, are:

Live trees—Live trees with diameter at breast height (d.b.h.) of at least 2.5 cm (1 inch), including carbon mass of coarse roots (greater than 0.2 to 0.5 cm [0.08 to 0.2 inch]; published distinctions between fine and coarse roots are not always clear), stems, branches, and foliage.

Standing dead trees—Standing dead trees with d.b.h. of at least 2.5 cm, including carbon mass of coarse roots, stems, and branches.

Understory vegetation—Live vegetation that includes the roots, stems, branches, and foliage of seedlings (trees less than 2.5 cm d.b.h.), shrubs, and bushes.

Down dead wood—Woody material that includes logging residue and other coarse dead wood on the ground and larger than 7.5 cm (3 inches) d.b.h., and stumps and coarse roots of stumps.

Forest floor—Organic material on the floor of the forest that includes fine woody debris up to 7.5 cm d.b.h., tree litter, humus, and fine roots in the organic forest floor layer above mineral soil.

Soil organic carbon—Belowground carbon without coarse roots, but including fine roots and all other organic carbon not included in other pools, to a depth of 1 meter (39 inches).

Carbon in harvested wood—Includes products in use and in landfills. "Products in use" includes end-use products that have not been discarded or otherwise destroyed. Examples are residential and nonresidential construction, wooden containers, and paper products. "Products in landfills" includes discarded wood and paper placed in landfills, where most carbon is stored long-term and only a small portion of the material is assumed to degrade, at a slow rate.

Carbon is transferred among these seven pools and the atmosphere. The amount of C in each pool is commonly called a stock, and the transfers may be called fluxes or changes in C stocks (box 1). Forest sector C pools and fluxes following disturbances are shown in detail in figure 3. The different C models described in this report, and those in the previously published baseline assessments, include subsets (or expansions) of the seven main reporting C pools (table 2).

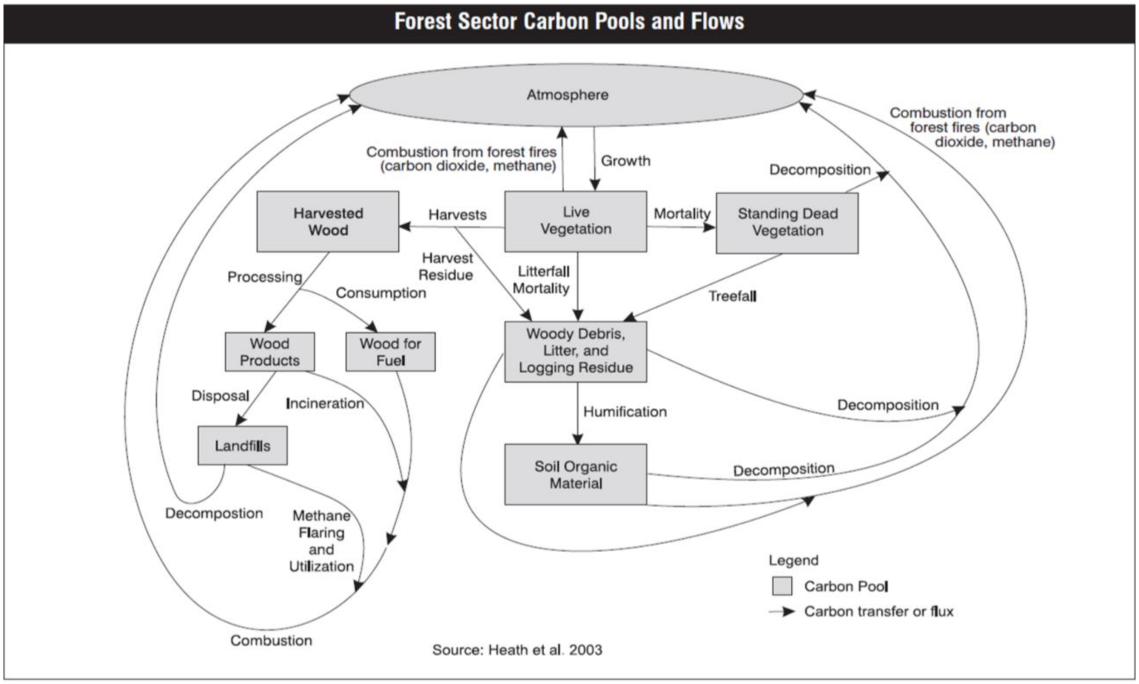

Figure 3. Forest sector carbon pools and flows (source: Heath et al. 2003).

Table 2. The seven main forest carbon pools and their representation in carbon models[a] referenced in this report.

	Carbon models				
Carbon pool	**Carbon Calculation Tool (CCT)**	**USFS Harvested Wood Products[a]**	**Forest Vegetation Simulator (FVS)**	**Forest Carbon Management Framework (ForCaMF)**	**Integrated Terrestrial Ecosystem Carbon (InTEC)**
Live trees	X		X	X	X
Standing dead trees	X		X	X	X
Understory vegetation	X		X	X	X
Down dead wood	X		X	X	X
Forest floor	X		X	X	X
Soil organic carbon	X				X[b]
Carbon in harvested wood		X			

[a] Source: Stockmann et al. (2012). USFS = USDA Forest Service.
[b] Soil carbon pools expanded to facilitate representation of key ecosystem processes: soil structural and soil metabolic detritus, soil microbes, surface microbes, slow organic matter, and passive organic matter.

4.1 Disturbance and Management Activities

4.1.1 Disturbance Data

The starting point for this C assessment is mapped records of disturbance and harvest activity across each national forest from 1990 through 2011. These disturbance maps were created by manually editing initial maps that were based on satellite data and created by the automated Vegetation Change Tracker algorithm (Huang et al. 2010). Editing was conducted at the pixel level to align mapped disturbances with several independent data sources: the multiagency Monitoring Trends in Burn Severity (MTBS) database of fires over 1,000 acres (405 hectares) in size (Schwind et al. 2010); multitemporal composites of Landsat data transformed to one band per year with the Disturbance Index (Healey et al. 2005); high-resolution time series of aerial imagery served through Google Earth; a combination of a tabular database (Forest Activity Tracking System; FACTS) of historical harvest activities and a spatial database (FACTS spatial) that provides an associated spatial representation of activity locations; and annual aerial forest detection survey (ADS) data (Johnson and Wittwer 2008). In the Rocky Mountain Region (NFS Region 2) and the Intermountain Region (NFS Region 4), where insect disturbance was most pronounced, ADS data were enhanced with Landsat-based disturbance maps generated from an ensemble of automated change detection algorithms (Healey et al. 2018). Methods used to produce the disturbance maps in this report were documented by Hernandez et al. (2018).

4.1.2 Uncertainty

Maps of disturbance year and type are not considered to be a significant source of error. To develop these maps, manual methods were used in consultation with independent error records. These records mimic methods used to develop “truth” data in other studies involving disturbance maps created with more automated methods (e.g., Cohen et al. 2010; Schroeder et al. 2014; Thomas et al. 2011).

4.2 Forest Carbon Management Framework

4.2.1 Methods

The Forest Carbon Management Framework takes advantage of corporate Forest Service monitoring data, management records, and management tools, complemented with forest change information from the Landsat series of satellites, to provide forest-level assessments of the impact of different kinds of disturbance on ecosystem C storage. These assessments estimate how much more C would be stored on each national forest if disturbances that took place from 1990 to 2011 had not occurred. Specifically, the impact of disturbance factor *F* (e.g., harvest, fire, insects, disease) is estimated as the difference, D_F, in the landscape’s nonsoil C stocks between an “undisturbed” scenario (*U*) where no disturbances occur during the study period and a scenario where only factor *F* occurs, in each simulation unit (*i*)(eq. 1).

$$D_F = \sum_{(i=0)}^{n} \left(C_{i\,(U)} - C_{i\,(F)} \right)$$ **Equation 1.**

Each scenario includes simulation of normal density-dependent tree mortality that is not attributable to disturbance. An estimate of D_F is produced for every year, considering the impact of all disturbances that occurred from 1990 to that date. D_F is a function of mapped starting conditions and disturbance history across 10-hectare (25-acre) sections of the landscape (i) that share the same starting conditions and disturbance patterns. These sections are called "simulation units" because they are the base units of error simulations described in Section 4.4.2, but they are also ForCaMF's finest level of carbon estimation. D_F is summed across all simulation units to produce a national forest-scale assessment of disturbance impact (eq. 1).

The C storage associated with mapped stand dynamics is obtained by combining the representative field sample measured by the FIA program with the Forest Vegetation Simulator (FVS) (Crookston and Dixon 2005; Hoover and Rebain 2011), a growth model that allows projection of nonsoil C stocks (Rebain 2010; Reinhardt and Crookston 2003) under a variety of disturbance scenarios. The FIA program maintains a plot network across the country consisting of one randomly located plot per approximately 6,000 acres (2,428 hectares). The tree list from each of the FIA plots with at least one forest land condition on NFS land (within the NFS region) is entered into FVS. For each scenario, a generalized C storage model is developed by combining results of simulations with similar starting tree lists. Each group of plots is subjected to a range of simulated disturbances to develop C storage models applicable to the gamut of observed disturbance patterns. Uncertainty measures from the fitting process are stored for later uncertainty analysis, as described more fully by Raymond et al. (2015).

Software in ForCaMF simply applies regionally averaged C dynamics described on the right side of figure 4 to the remotely sensed vegetation and disturbance history information summarized on the left side of the figure. When disturbance affects a simulation unit (as indicated by the disturbance map), it is moved from an FVS-derived "undisturbed" C accumulation function to the appropriate postdisturbance function. If maps show the simulation unit to be disturbed again, it is moved to the new postdisturbance C accumulation function at the appropriate time (Healey et al. 2014). Disturbance impacts on C storage (D_F) are calculated by comparing C storage under an "undisturbed" scenario to a scenario with disturbance factor *F*. In this report, results are reported for *F* = all observed disturbances and for *F* = each individual disturbance factor (e.g., fire, harvest, insects, disease). Where *F* = all, the effects of successive disturbances (e.g., fires following insect disturbance) on C storage were realistically represented by the previously mentioned switching from one postdisturbance C accumulation function to another. Where *F* = only a single disturbance factor, all non-*F* disturbances were zeroed out and this switching did not occur (unless successive disturbances were of the same type). In both cases, calculation of D_F in each year from 1990 to 2011 allowed temporally precise evaluation of how the impact of disturbance evolved over time, particularly in light of events such as large fire years or emerging insect problems that change overall C storage patterns.

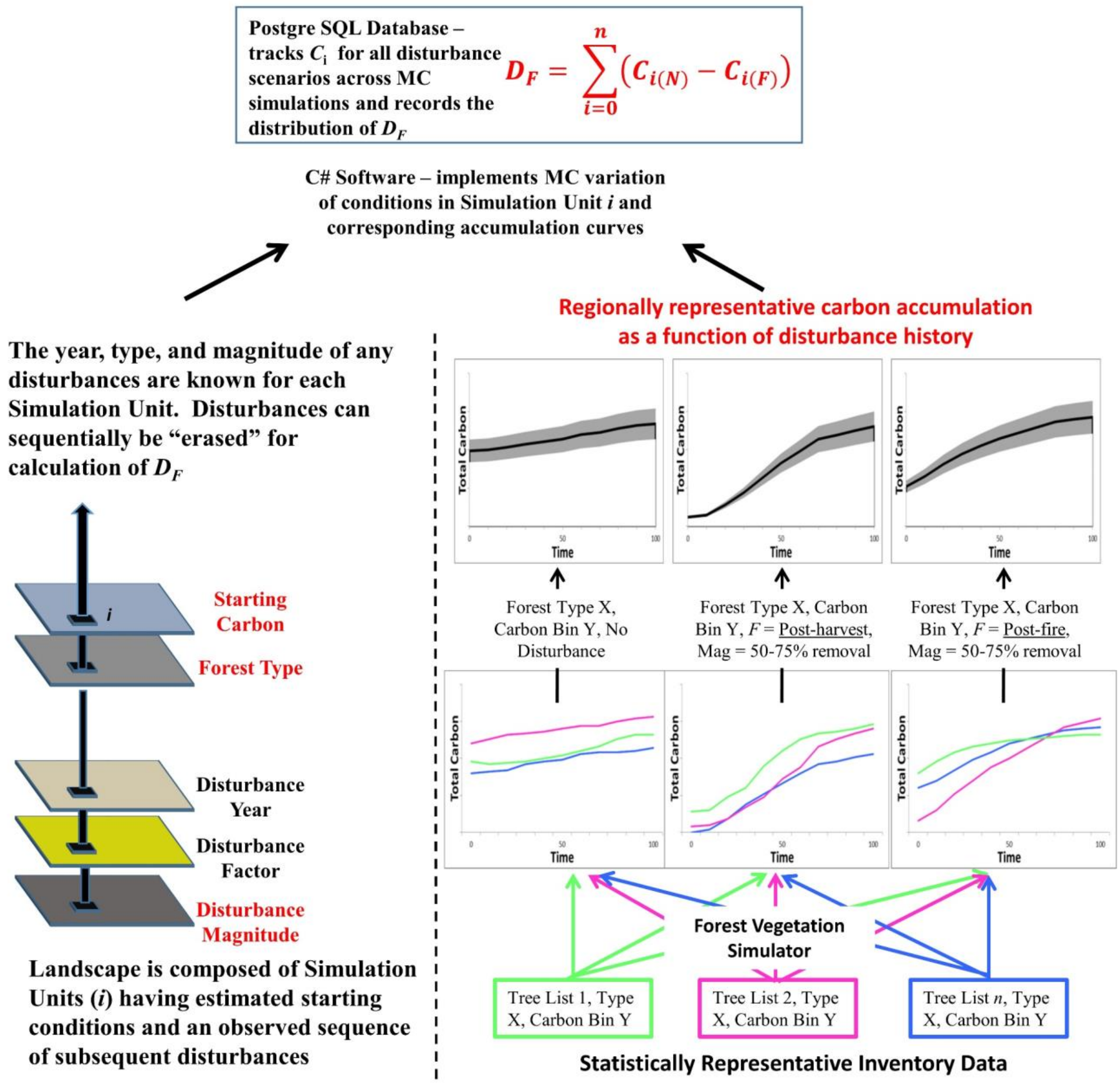

Figure 4. Flowchart of how ForCaMF calculates the impact of each type of disturbance. Carbon (C) storage associated with mapped forest conditions and disturbances is determined by applying regionally generalized C dynamics derived from the combination of USDA Forest Service, Forest Inventory and Analysis data and Forest Vegetation Simulator data. Disturbance-specific, stand-level C scenarios are shown on the right side of figure. Mapped stand dynamics are designated on the left side of the figure.

4.2.2 Uncertainty

There are several potential sources of uncertainty in the ForCaMF workflow described, including (designated in red in fig. 4):

- mapped starting conditions (initial C storage and forest type);
- modeled disturbance magnitude; and
- modeled C accumulation estimates, as obtained from FIA and FVS.

The complexity and interaction of ForCaMF's potential error sources preclude solving for error using an analytical approach. Instead, a Monte Carlo approach is used, where randomly selected alternative values for each input are substituted into the calculation of DF over a large number of error simulations. The variance of D_F in response to simulation of possible error patterns produces an empirical estimate of the system's integrated uncertainty. Healey et al. (2014) described an innovative method of carrying out the Monte Carlo method with mapped inputs such as those used here. This process, called Probability Density Function (PDF) Weaving, uses FIA data to calibrate and constrain error simulations related to mapped forest type and starting-condition maps as well as maps of disturbance magnitude. As a result, ForCaMF analyses are aligned with FIA estimates of historical C storage, distribution of forest type, and forest cover change.

In addition to uncertainty in input map products, ForCaMF simulates uncertainty associated with each of the disturbance-specific, stand-level C scenarios (fig. 4 right side) that ForCaMF links to mapped stand dynamics (fig. 4 left side). Error functions are obtained from fitting each scenario via a process called quantile regression, as previously mentioned and described by Raymond et al. (2015). These models are used to calibrate how C storage patterns are allowed to vary in the ForCaMF Monte Carlo process. The net result of these error simulation processes is that an empirical 95-percent confidence interval can be associated with estimates of D_F for each year.

4.3 Integrated Terrestrial Ecosystem Carbon Model

4.3.1 Methods

The InTEC model builds on ForCaMF by attributing the observed changes in C stocks to a full suite of both natural and anthropogenic factors. InTEC is a process-based biogeochemical model driven by monthly climate data, vegetation parameters, and forest disturbance information to estimate annual forest C and fluxes in C pools at regional and local scales (fig. 5) (Chen et al. 2000a,b,c; Ju et al. 2007). InTEC relies on empirical FIA datasets (USDA FS 2016) containing variables such as stand age, forest (or dominance) type, and net growth, resulting in a hybrid approach which combines a process-based biogeochemical model with empirical models that are comparable to the results from the baseline assessment (CCT) and ForCaMF. Specifically, the FIA-based stand age, dominance (or forest) types, and net primary productivity (NPP)-stand age relationships determine when stands were initially disturbed and, depending on forest (or dominance) type, how the productivity changes with stand age over time.

The C dynamics of a forest region are a function of multiple factors including disturbance, stand age, climate, and atmospheric composition (Chen et al. 2000a). These are grouped into

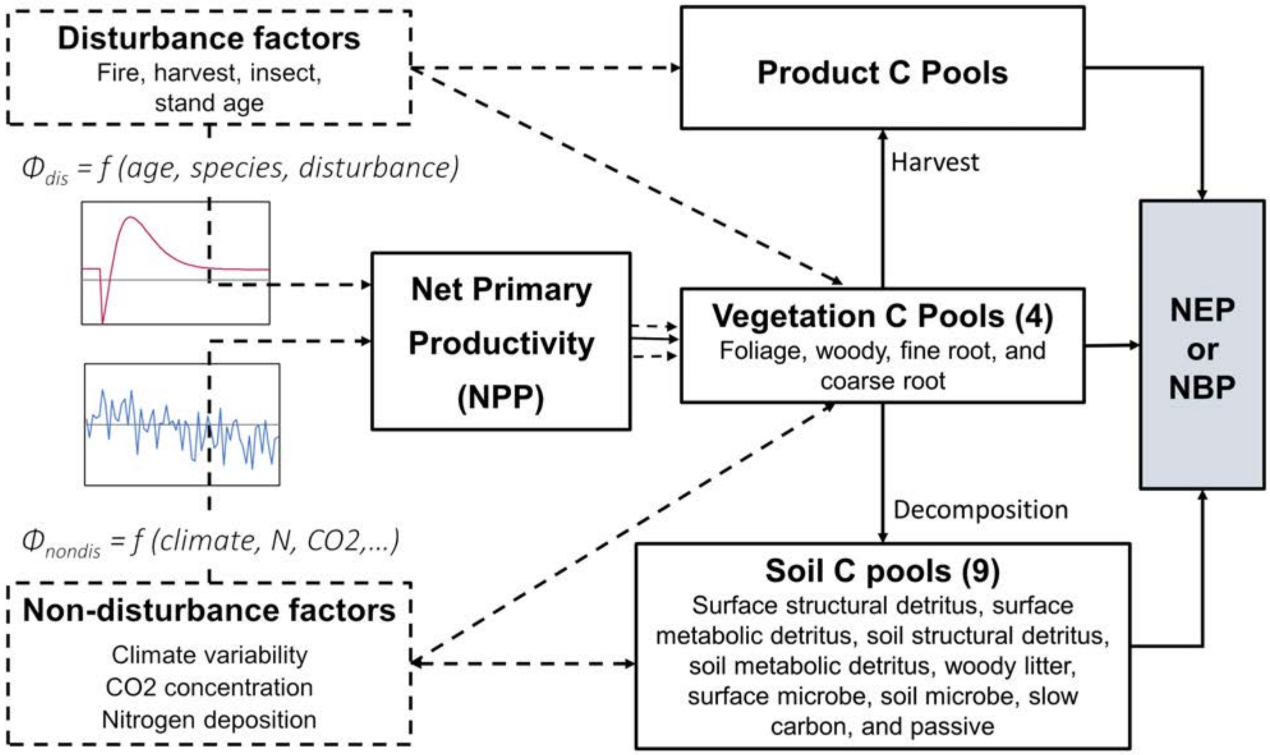

Figure 5. Conceptual scheme of the carbon (C) cycle in the Integrated Terrestrial Carbon Cycle (InTEC) model. Solid arrows indicate C flow and dashed arrows indicate influences. $\phi_{dis}(i)$: disturbance function; $\phi_{nondis}(i)$: nondisturbance function; NEP: net ecosystem productivity; NBP: net biome productivity. NPP is the net production of organic matter by plants, which is equal to the difference between the total amount of C fixed in photosynthesis (gross primary productivity) and C losses from respiration of plants (autotrophic respiration). NEP is equal to the sum of NPP and the C loss to the atmosphere via heterotrophic respiration. NBP is equal to the sum of NEP and C fluxes associated with nonrespiratory losses due to disturbances such as combustion from fire or export to external pools following harvest. If no disturbances occurred in a given year, NBP is equal to NEP (source: Zhang et al. 2012).

disturbance and nondisturbance factors. Disturbance factors include fire, harvest, insects, and forest stand age or time since stand-replacing disturbance, which can include disturbances that are not specifically identified or occurred prior to the satellite-based disturbance maps (pre-1990). Nondisturbance factors include climate (temperature and precipitation), atmospheric CO_2 concentrations, and N deposition. The InTEC model integrates the effects of nondisturbance and disturbance factors since the initial modeling year (1900 in this study). The historical C dynamics are estimated progressively from 1950 through 2011, and the results at any point in time are the accumulated effects of all factors since the starting year (Chen et al. 2003).

The InTEC model is run pixel-by-pixel, with a 90-m (300-foot) pixel size, in each individual national forest. Summary results are calculated by summing or averaging all pixels in a forest, depending on the measure being reported. Carbon pools include aboveground live (wood and foliage), belowground live (coarse roots), forest floor (fine roots, and surface structural and surface metabolic detritus), dead wood (standing and down dead, foliage, and coarse roots), and soil (soil structural and soil metabolic detritus, soil microbes, surface microbes, slow organic matter, and passive organic matter).

4.3.1.1 Disturbance, Regrowth, and Aging Effects

Disturbances are explicitly considered as processes that release C into the atmosphere, modify the terrestrial C balance, initiate regrowth, and subsequently transfer C from one pool to another (e.g., live trees to standing dead) in the disturbance year and thereafter. For the period 1990 to 2011, Landsat-obtained disturbance maps (described in Section 4.1.1) were used to determine the year, location, type (fire, harvest, insects), and magnitude of disturbance events. For years prior to 1990, stand-age maps were obtained from forest inventory data to act as proxies for the timing of the last stand-replacing disturbance. The time since disturbance influences the rate and accumulation of biomass and C during regrowth after disturbance. Each pre-1990 disturbance is considered to be a stand-replacing fire because it is difficult to assign a disturbance type from stand age alone, and any disturbance that reset the stand age to zero would have been stand replacing—most likely a high-severity fire or clearcut harvest. The type and magnitude of each disturbance determine the amount of C released directly to the atmosphere as well as the transfer of C from live to dead pools as a result of mortality, and the changes in C pools during regrowth.

For instance, if a fire has a mortality rate of 100 percent, a portion of the biomass C is immediately released to the atmosphere via combustion, the entire stand dies, and C is transferred from the live to the dead pool. Stand age is set to zero, and the stand regrows beginning the next year following the NPP-stand age relationships specific to each forest type. If a fire has a mortality rate of 50 percent, a portion of the biomass C is immediately released to the atmosphere via combustion, and 50 percent of the live trees die, transferring C to the dead pool. The stand age remains the same, and the forest continues to grow following the NPP-stand age relationship.

For harvests, the C in the harvested wood is transferred to the harvested wood C pool, thus leaving the forest ecosystem. Wood that is not removed from the forest during harvest, such as branches and roots, is transferred to the appropriate ecosystem C pool, such as woody debris. The removed C may be stored in the harvested wood product pools for several decades. Refer to the baseline assessments (USDA FS 2015a–i) for a detailed analysis of C in harvested wood product pools.

Relationships between NPP and stand age vary regionally, by forest types and environmental conditions (He et al. 2012; Zhang et al. 2012). For U.S. forests, NPP typically increases rapidly at a young age, reaches a maximum at middle age at about the time the canopy closes, and then gradually declines and stabilizes with older ages. Consequently, middle-aged forests have a greater capacity for C uptake than young and old forests. In InTEC, the aging effects reflect these changing rates of C sequestration with stand age. After a disturbance, C changes may initially be negative (C source), but later become positive (C sink) and reach a peak as vegetation regrows and decomposition declines (Pregitzer and Euskirchen 2004). Depending on stand age or the number of years since the last disturbance, the disturbance and aging effects on changing C stocks can be positive (causing a C sink) or negative (promoting a C source).

4.3.1.2 Nondisturbance Effects

Both long-term climatic trends and interannual climate variability can impact forest C dynamics by affecting growth, productivity, and decomposition. The InTEC model integrates climate data from the Parameter-Elevation Regressions on Independent Slopes Model (PRISM), which combines empirical measurements from a network of weather stations with elevation and other topographic factors to map precipitation and temperature at monthly timescales across the United States (PRISM Climate Group 2004). Anthropogenic climate change has led to increases in average temperatures in most regions across the United States since 1900, as well as regional shifts in precipitation; some regions have become drier and others wetter (Walsh et al. 2014). In some regions, warmer temperatures can cause moisture stress and more rapid decomposition of surface and soil C (Ju et al. 2007), thus increasing C emissions. In high-latitude or -altitude locations, warmer temperatures can enhance tree growth (Way and Oren 2010). Drought conditions can reduce tree growth both during the drought and up to several years later, in turn making forests less able to act as C sinks (Anderegg et al. 2015). On the other hand, increased precipitation and humidity can promote tree growth and C uptake (Dale et al. 2001; Nemani et al. 2002).

Like climate, atmospheric composition is known to impact plant growth rates and C dynamics (e.g., Hyvönen et al. 2007; Law 2013; Pan et al. 2009). Data collected at field monitoring stations of the National Atmospheric Deposition Program's National Trends Network established coverage maps of N deposition (Pan et al. 2009). The increased N deposition due to human activities can stimulate C sequestration in forests, through increased production of biomass, surface litter, and soil organic matter.

The InTEC model also incorporates annual concentrations of atmospheric CO_2 measured at the Mauna Loa Observatory in Hawaii (Keeling et al. 2009). Atmospheric CO_2 levels have increased steadily from 280 ppm in 1901 to 390 ppm in 2010 due to industrialization and other human activities. Like N deposition, elevated atmospheric CO_2 concentration acts as a fertilizer, stimulating photosynthesis and biomass production (Keenan et al. 2013). Additional details about the datasets and their use by InTEC are available (Dugan et al. 2017).

4.3.1.3 Simulation Scenarios

To model the effects of individual and combined disturbance and nondisturbance factors, a series of simulation scenarios were developed (Zhang et al. 2012, 2015). Six overall scenarios were simulated by InTEC: 1) all factors including all disturbance and aging and nondisturbance effects, 2) disturbance and aging effects only, 3) combined nondisturbance effects only, 4) climate effects only, 5) CO_2 effects only, and 6) N deposition effects only.

4.3.2 Uncertainty

Quantifying model uncertainty using a Monte Carlo approach and multiple simulations is impractical with InTEC because of the time it takes to perform model runs given the numerous forests, high-resolution datasets, long temporal scales, and multiple simulation scenarios analyzed. The uncertainty and validation of the InTEC model for analyses across the contiguous United States were studied in Zhang et al. (2012, 2015). Uncertainties also depend on the quality and quantity of data available for each forest. The climate and atmospheric chemistry data used in InTEC are derived from a sufficiently dense network of observation stations to be accurately representative of regional conditions, although errors may be significant at local scales. As discussed earlier, InTEC is calibrated to FIA data and satellite imagery observations of disturbance and productivity and uses disturbance estimates and age maps, so that the previously assessed uncertainties in these datasets are propagated into the InTEC model results. Given the lack of stand-age data in the early part of the 20th century, modeled results are more uncertain for that period (Zhang et al. 2015); thus, results prior to 1950 are omitted in this report. For additional information regarding InTEC, including calibration, parameterization, and model inputs, see Zhang et al. (2012, 2015).

5. Integration of Modeled Results

These NFS forest C assessments are based on a variety of datasets and several models. Although we have attempted to reach a high level of consistency, appropriate interpretation and use of the results should reflect inherent differences between datasets and models that cannot be fully reconciled (Dugan et al. 2017). Generally, the FIA data represent the most accurate analysis of trends in C stocks as summarized by CCT in the baseline assessments (USDA FS 2015a–i), the effects of disturbances are best captured by ForCaMF, and the effects of environmental variables are reflected in the results from InTEC. However, despite the extensive use of FIA data by all three models (CCT, ForCaMF, and InTEC), results are not completely compatible because ForCaMF and InTEC use additional datasets with different properties and timing of observations and the models include different C pools (table 2). The models themselves are also different. CCT and ForCaMF are empirical models, whereas InTEC is a hybrid empirical and process model. These different approaches are necessary to provide a complete analysis of the main drivers of change, because none of the approaches can represent all of the important drivers. In a broader context, a land manager may be faced with interpreting conflicting results from studies conducted by different parties; therefore, the goal of this section is to provide some guidance for dealing with this situation in the future.

The three models (CCT, ForCaMF, and InTEC) that were integrated for these forest C assessments use common data sources to enhance agreement between model results. For instance, all three models use FIA data as major inputs so that all results are well grounded in observations. FIA data are the primary data source of CCT, as the model calculates C stocks and stock changes from tree-level data from at least two inventories using allometric models (Woodall et al. 2011). ForCaMF also relies on FIA data to simulate forest C trajectories given different disturbance scenarios (Raymond et al. 2015). Last, InTEC is driven by stand-age maps, dominance type maps, and NPP-age relationships, all obtained in part from FIA plot data.

Along with FIA datasets, ForCaMF and InTEC incorporate high-resolution Landsat-obtained disturbance data (Healey et al. 2014). Although FIA captures disturbance events by periodically remeasuring the same sample plots, it may lack the temporal sensitivity to detect the effects of more recent disturbances. This is because FIA field observations are made on a cycle that may be 10 years or even longer. In some cases older observations are based on periodic inventories and newer observations are based on annual inventories of subsets of sample plots, so that the average age of "current" FIA data can be 5 years (typical in the East) or 10 years (typical in the West), or more. The CCT model interpolates and extrapolates these observations to produce an annual time series since 1990, but if the extrapolation is from older data and there have been more recent significant changes such as an increase in disturbances, the extrapolation represents trends that no longer characterize what is occurring in the landscape. In contrast, ForCaMF and InTEC both use more contemporary satellite-based observations of disturbances as a major input, which reflect landscape changes as they occur.

The FIA data may also lack spatial resolution to detect smaller disturbances outside FIA plots, especially in areas where plot density is sparse. On the other hand, the Landsat satellite captures any recent disturbance greater than pixel resolution (30 m; 100 ft) that alters forest canopies, even those occurring outside FIA plot locations. ForCaMF utilizes all identified disturbance types including fire, harvest, insects, disease, and abiotic disturbances (e.g., wind, ice). InTEC excludes disease and abiotic disturbances due to the complexity and uncertainty in including their effects in a process model. However, stand-replacing disturbances that are not attributed to a specific cause or occurred prior to the mapped satellite record, may be captured in the inventory-based stand-age data. In this case, these disturbances are included in the model and treated as stand-replacing fires (Zhang et al. 2012). Except for the Northern Region, which has had significant disease impacts as indicated by ForCaMF, both diseases and abiotic disturbances affect a relatively small percentage of national forests across the United States compared to other disturbance types.

Although InTEC and ForCaMF both evaluate the effects of disturbances and management on C stocks, there are several key differences between the models that make direct comparison of their results difficult. ForCaMF models only the effects of disturbances and management on nonsoil C stocks, while InTEC tracks the complex C cycling through several soil C pools (fig. 5, table 2). Though soil C is often one of the largest single C pools in forest ecosystems, it is very stable and not significantly affected by disturbances. Additionally, ForCaMF is primarily tracking potential lost C storage as a result of disturbances and management, while InTEC is mostly focused on C stock changes and accumulations due to both disturbance and nondisturbance factors. Therefore, while these models complement one another, direct comparisons between the two should take into account these differences.

Model results may also vary due to inherent differences in modeling approaches. For instance, CCT relies on allometric models of volume, species, and tree dimensions to convert tree measurements to biomass and to C (Woodall et al. 2011). ForCaMF is similar in that it also uses individual tree measurements and site characteristics within the FVS growth and yield model to simulate C stocks and trends. InTEC is fundamentally different as it is driven by a mathematical representation of biogeochemical cycles such as photosynthesis, N mineralization, and nutrient dynamics (Zhang et al. 2012, 2015). Thus, it does not rely only on biometrics. These distinctions in modeling techniques are likely to result in some discrepancies between forest C outputs from the three models.

Last, the area of forest land in each national forest may differ by model. Both CCT and ForCaMF use the FIA definition of forest land as areas at least 120 feet (36.6 meters) wide and 1 acre (0.4 hectare) in size with at least 10 percent cover (or equivalent stocking) by live trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated (O'Connell et al. 2014). Thus, the amount of forest land can change over survey years given deforestation or afforestation activities or administrative boundary changes. However, the changes in FIA sampling design, protocols, and definitions in the late 1990s can introduce discontinuity in attributes such as total forest land area and consequently C estimates over time (Goeking 2015; Woodall et al. 2011). InTEC uses a single forest (or dominance) type map based in part on FIA data (Ruefenacht et al. 2008) and, where available, data sampled by the NFS (e.g., USDA FS 2015j). The model therefore assumes there is no change in area of forest land over the study period. While the extent of the effects of such modeling disparities on forest C estimates has not been evaluated, it is important to consider that model results for individual forests may differ, sometimes considerably, for a variety of reasons.

6. Regional Results

6.1 Eastern Region

6.1.1 Description of Region

The Eastern Region (also referred to as Region 9) in the National Forest System (NFS) consists of 14 national forests from Maine to Minnesota in the north and Maryland to Missouri in the south (fig. 6). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these national forests, so the Midewin National Tallgrass Prairie in Illinois was not included. To restrict the analysis to lands managed by the Forest Service, U.S. Department of Agriculture, any private inholdings within national forest boundaries were excluded.

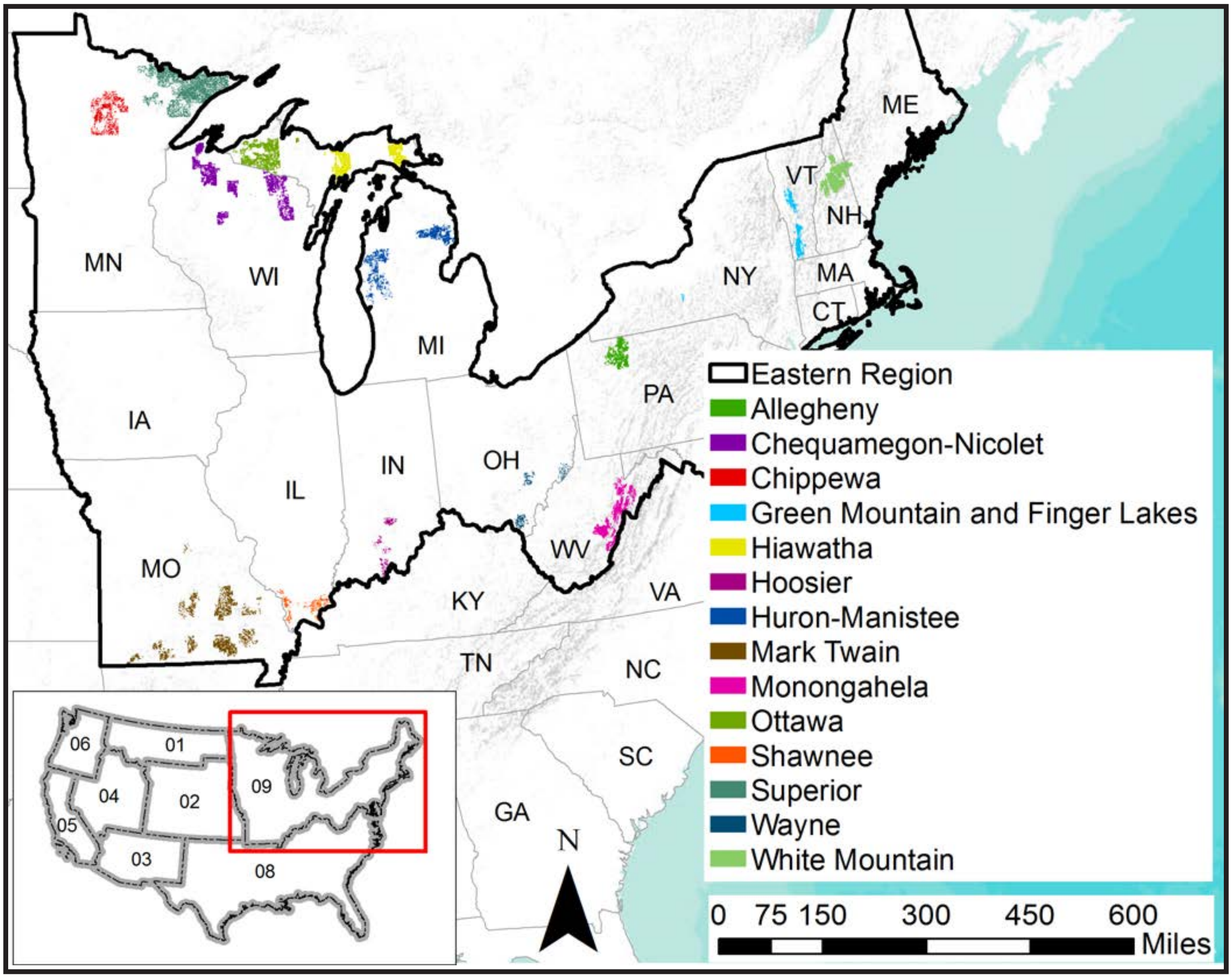

Figure 6. Locations of the national forests in the Eastern Region.

The history of Euro-American settlement, land use, and policies provides useful context for understanding the regional forest C trends that are indicated by the inventory and assessment results. The Eastern Region was the first region to be widely settled in the United States, so its forests have a long history of unregulated harvesting and conversion to agriculture. For much of the 19th century, the U.S. timber industry was centered in the Northeast, largely in the valuable white pine forests of the Lake States region, until the 1890s (see Appendix 1 for scientific names of species mentioned in this report). However, the depletion of merchantable timber products and the settlement of the West caused the logging industry to move westward. In addition to timber harvesting, fire was regularly employed to clear land for agriculture, but fires occasionally spread, causing the destruction of valuable timber resources. In the late 1800s, several large wildfires destroyed millions of acres of forested land in Michigan, Wisconsin, and Minnesota. By 1900, some 300 years after Euro-Americans first settled the region, the need for forest restoration and protection became evident (Conrad 1997).

After passage of the Weeks Act in 1911 (PDF, 37KB), the Forest Service began buying large areas of these heavily cut-over and submarginal lands, referred to as “the lands nobody wanted” (Shands 1992), throughout the eastern United States. By 1920 approximately 2 million acres (800,000 hectares) of excessively logged and degraded land was purchased as national forest land. Soon forest restoration and recovery became the new goal. With help from the Civilian Conservation Corps, millions of trees were planted, erosion was controlled, and forest fires were fought (Williams 2003). The suspension of the use of fire for clearing land, the increased effort to suppress fires, and the forest restoration efforts allowed forests to regrow and increased stocking of trees. In the 1940s the timber industry in the Eastern Region started to pick up steam again. Timber production increased throughout much of the mid-1900s, peaked in the late 1980s, and then declined rapidly in the 1990s and 2000s (Loeffler et al. 2014b). This history of timber harvesting and forest restoration in the Eastern Region played an important role in shaping forest C dynamics over time.

In addition to timber harvesting and fires, natural disturbances including storms and insects have impacted forests and C stocks in this region. For example, in 1999 a massive wind and rain storm known as the “Independence Day Windstorm” caused an unprecedented blow-down of trees across Superior and Chippewa National Forests (Mattson and Shriner 2001; Nelson et al. 2009). In 2009 a historic derecho (storm with straight-line winds) plowed down some 80,000 to 90,000 acres (32,000 to 36,000 hectares) of forest in the Mark Twain National Forest (Vaughn 2013). Though insect outbreaks in the Eastern Region have been relatively small and resulted in lower mortality or removal compared to harvests and wind-storms, the spread and severity of insect outbreaks including the hemlock woolly adelgid, eastern spruce budworm, forest tent caterpillar, and emerald ash borer are projected to intensify with continued climate change (Dukes et al. 2009).

6.1.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Eastern Region are displayed in figure 7. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. Harvest was the most common and consistent type of forest disturbance in the Eastern Region, although harvest rates never greatly exceeded 0.25 percent of the landscape, and there was a general downward trend from the early 1990s to 2011 (fig. 7). Occasional fires and windstorms (labeled "abiotic" in figure 7) caused notable mortality, and localized insect outbreaks were detected in several years. Over this period, disturbances exhibited a relatively even mix of low to high intensities.

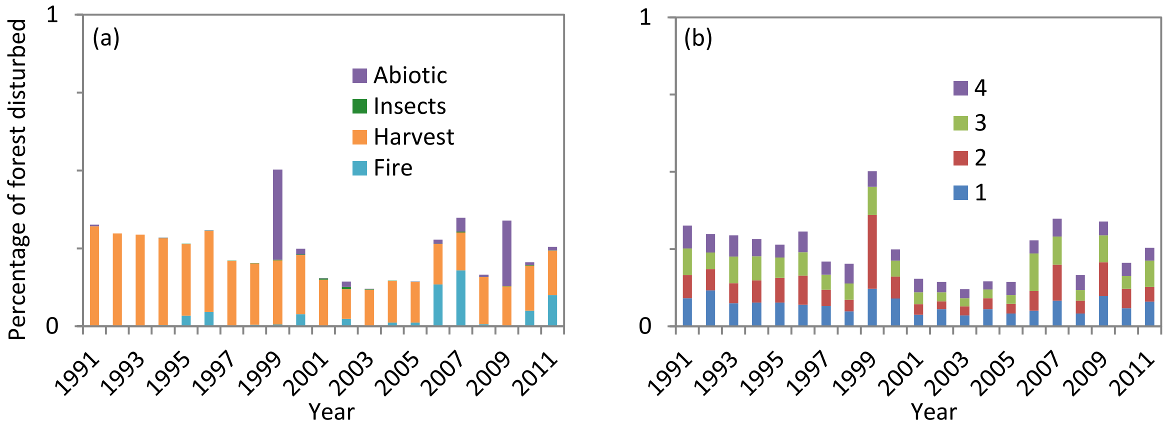

Figure 7. Annual rates of disturbance (0 to 1 percent) in the Eastern Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

6.1.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to estimate the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Eastern Region in figure 8. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires first occur in 1997, for example, the line for fire in figure 8 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an "undisturbed" scenario. Thus, figure 8 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

As mentioned, harvest activity was the dominant type of disturbance within the region during the study period, and had the largest impact on C stocks. Harvests occurring from 1990 to 2011 reduced 2011 storage by approximately 154 g/m^2 or 1.54 Mg/ha (fig. 8), or about 81 percent of the disturbance impact on stocks (fig. 9). By 2011, this represented a reduction of about 1.6 percent of the regional nonsoil C stocks (fig. 10) that were reported in the NFS baseline assessment derived from FIA data (USDA FS 2015b). These patterns do not account for offsite storage of C in wood products. Temporary storage of forest C in products and disposal sites can last for decades, significantly delaying climate impacts resulting from atmospheric emissions. Information about the region's storage of C in wood products can be found in the baseline C assessment (USDA FS 2015b).

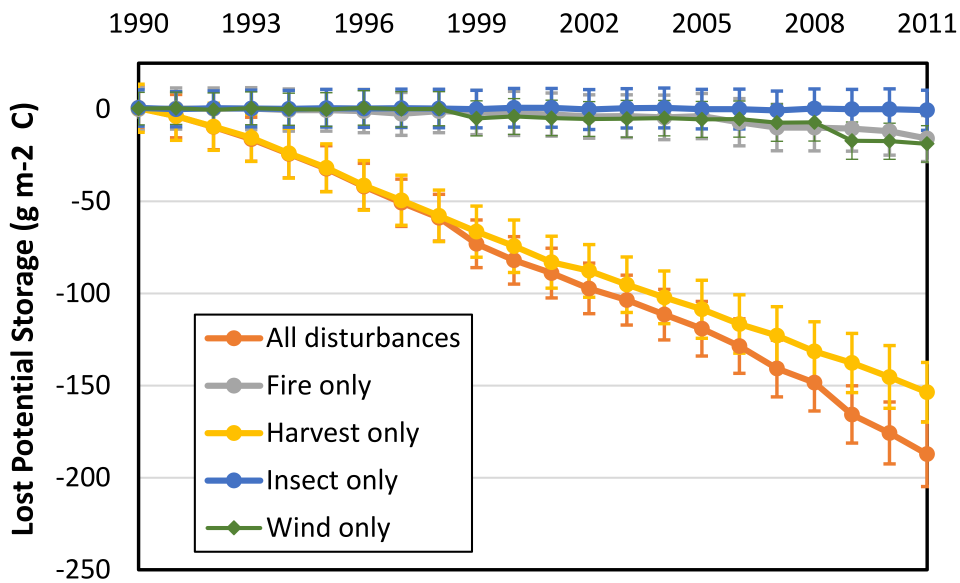

Figure 8. The impact of different kinds of disturbance, occurring from 1990 through 2011, on carbon (C) stores in the Eastern Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

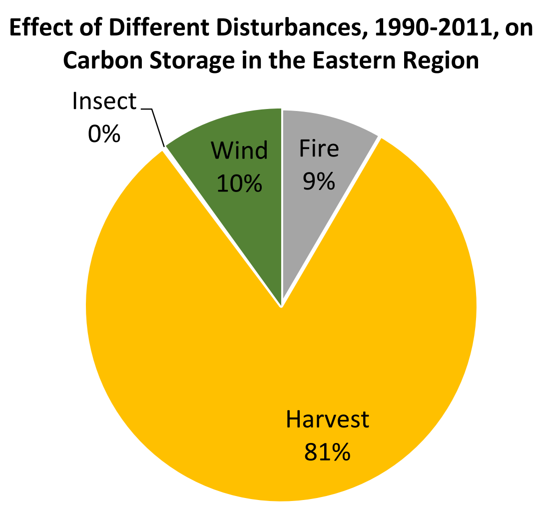

Figure 9. Proportional effect of different kinds of disturbance on carbon storage in all national forests in the Eastern Region for the period 1990 through 2011.

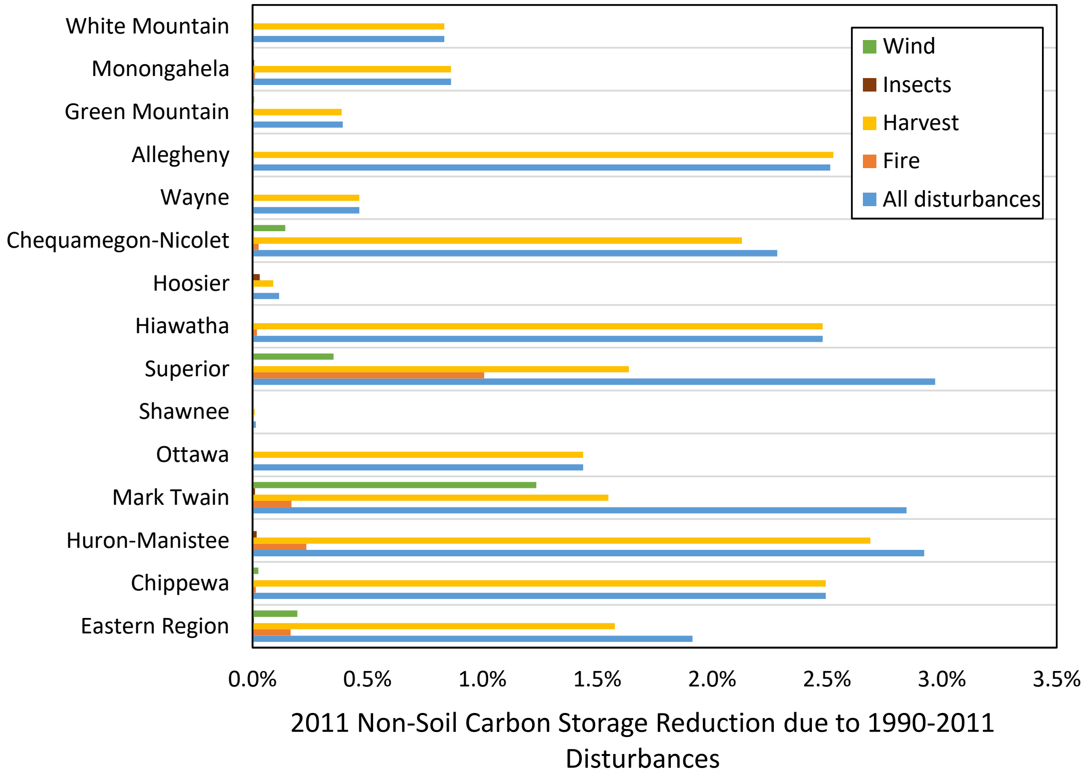

Figure 10. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by each national forest and for all national forests combined in the Eastern Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

Wind and fire also had significant impacts on C storage in the region, making up 10 percent and 9 percent of the total effects of disturbance, respectively (fig. 9). Wind resulted in the loss of 0.2 percent of nonsoil C stocks, and fires accounted for a 0.1-percent loss. These disturbances were concentrated in a few years (fig. 7), although figure 8 illustrates how residual effects of discrete events (such as slowly decomposing dead material) can cause ongoing divergence of the amount of C that is stored versus the amount of C that could be stored. The highly localized nature of the region's fire and wind events and their consequent C impacts is discussed for individual forests (Appendix 3: Eastern Region (PDF, 3.4 MB)). The low level of detected insect activity was noteworthy given the relatively high profile of insect events in the region. One reason for this discrepancy may simply be that by the end of the period (2011), the spread of emerald ash borer was still fairly localized. Further, there was a requirement built into the disturbance-mapping process that canopy mortality must be sustained. Seasonal or temporary defoliation due to gypsy moth activity was widely noted, for example, but imagery from subsequent dates often showed no permanent loss of canopy, so these areas were not included in maps of disturbance. This rule was not to imply that these events were not disturbances; it was instead based on the assumption that loss of a single year's foliage had little impact on C storage or stand dynamics. In addition to affecting estimates of insect activity in the Eastern Region, this rule also significantly reduced the amount of area mapped as "disturbed" due to hurricanes in the Southeast.

One caveat must be stated about the confidence intervals depicted in figure 8. The unit of analysis for ForCaMF was the national forest (Appendix 3: Eastern Region (PDF, 3.4 MB)); rates of simulated error were constrained by FIA estimates at that level. Regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in other forests. This assumption very likely overstated certainty at the regional level; if initial biomass estimates in one forest were high, for example, the same estimates were also likely to be biased high in other units. This caveat does not change the overall trends seen at the aggregated regional level, nor does it affect confidence intervals assessed at the national forest level. Harvests were clearly the most important disturbance process with respect to C storage, although fire and wind were locally important.

6.1.4 Effects of Disturbance, Management, and Environmental Factors

This section provides a regional summary of the predicted (modeled) effects of natural disturbances, land management, and environmental factors on forest C dynamics depicted in a series of figures (figs. 11–14) generated from InTEC model inputs and simulations for individual national forests and summed results across all national forests in the Eastern Region. These regional-scale outputs were generated only from the national forest-specific datasets; therefore, these outputs do not represent lands within the region that are outside of the national forest boundaries. The Appendix 3: Eastern Region (PDF, 3.4 MB) shows the effects of disturbance and nondisturbance factors on forest C dynamics within each of the 14 national forests as modeled by InTEC with this same series of figures.

According to the historical data and model results, forest C trends across the national forests in the Eastern Region have been strongly influenced by the history of land use and policies as well as climate change and atmospheric CO_2 concentrations. Despite variation among national forests, the regionwide stand-age distribution in 2011 shows that about 50 percent of the forests are greater than 80 years old (fig. 11). The stand-age distribution also indicates a period of elevated stand establishment occurring about 70 to 100 years before 2011, or from 1910 to 1940 (fig. 11). During the 1940s and thereafter, the rate of stand establishment sharply declined. This early-1900s pulse of stand establishment reflects forest recovery and regeneration after decades of heavy logging and clearing of land for agriculture. Depending on the forest-type group, which is mostly oak/hickory and maple/beech/birch (fig. 11), the stands making up this pulse of establishment would have been growing at maximum productivity when they were about 30 to 50 years old (fig. 12), or around 1940 through most of the mid-20th century. By the 1980s, most of the forests in the Eastern Region had aged beyond their peak productivity, according to the NPP-age relationships. Thus, productivity has since declined although the forests have not yet reached their potential C stocks (Hoover et al. 2012).

Both temperature and precipitation have increased slightly over the past few decades in the Eastern Region (figs. 13a,b). Warmer temperatures can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity. Consequently, more C is released to the atmosphere. On the other hand, the wetter conditions enhance productivity and growth rates. Although long-term climate trends may be subtle, interannual weather variability has been high with several years of notable climatic extremes.

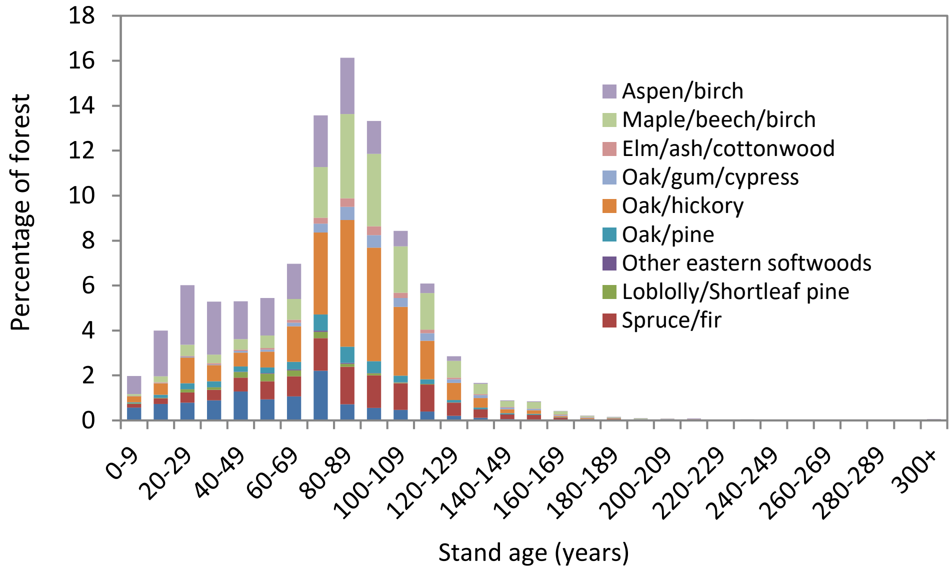

Figure 11. Age-class distribution in 2011 shown as the percentage of forest land in each forest-type group in 10-year age classes summed across the national forests in the Eastern Region (see Appendix 1 for scientific names of trees).

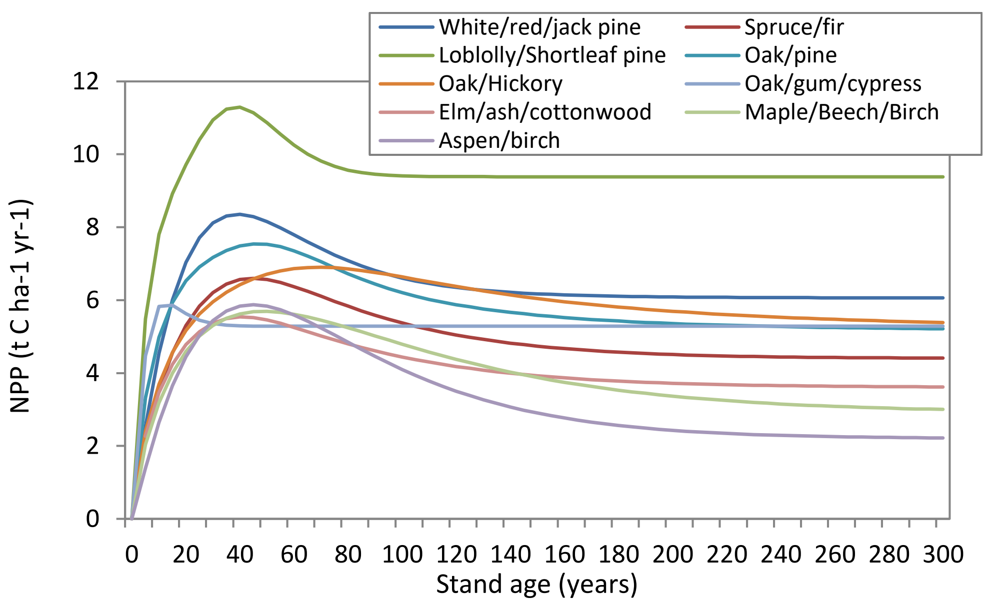

Figure 12. Relationship between net primary productivity (NPP) and stand age for each forest-type group averaged across all national forests in the Eastern Region (see Appendix 1 for scientific names of trees). Due to a small sample of other eastern softwood plots to derive NPP-age curves, the loblolly/shortleaf pine curve was used for these stands.

For instance, much of the Eastern Region experienced a severe drought in the mid-1960s, as well as harsh summer heat waves in 1953 and 1998 (figs. 13a,b). Furthermore, the forests of theEastern Region have experienced some of the highest amounts of N deposition in the United States from the coal industry and automobile emissions. Nitrogen deposition has declined in the past decade due to regulations targeting emissions from power plants and automobiles (fig. 13c). Atmospheric CO_2 concentrations have increased globally over the past few decades and are expected to continue to rise as a result of human activities.

The C stock changes across the national forests in the Eastern Region show that forests have mostly maintained a C sink from 1950 to 2011 (fig. 14c). Modeled results indicate that this C sink has been declining but may have stabilized over the past two decades. This decline in the C sink is a result of forests getting older on average and therefore growing more slowly, highlighted in the disturbance and aging effect (fig. 14b). The pulse of stands 70 to 100 years old (fig. 11) was most productive from the 1940s through approximately the 1970s, but as forests continued to age, their productivity declined (fig. 12). This decline was coupled with lower rates of stand establishment starting in the 1940s (fig. 11). Aside from a few large disturbance events, such as the windstorms in 1999 and 2009, recent disturbances in the Eastern Region have been small and of low magnitude (fig. 7); thus, C emissions from disturbances have also remained low (fig. 14e). As a result, low rates of stand establishment have been persistent since the 1940s and the forest continues to be dominated by these aging stands (fig. 11).

In the early 1950s climate had a mostly negative effect on changing C stocks, causing a lower rate of C accumulation than in more recent years (fig. 14d), which have been characterized by elevated temperatures (fig. 13b). Aside from this early period and a few notably warmer years such as 1998, climate has mostly enhanced the C sink, but overall had a small effect on total C accumulation (fig. 14d). Both the increases in N deposition (fig. 13c) and atmospheric CO_2 concentrations have had consistently positive effects on changing C stocks and C accumulation across all forests in this region (figs. 14a,d). The N deposition effect has mostly stabilized due to declining N deposition rates (fig. 13c) and the potential saturation of N in the forests. However, the CO_2 effect has steadily increased, causing significant C accumulation (fig. 14d). Overall, the positive effects of climate, atmospheric CO_2, and N deposition helped to offset the decline in C stocks due to disturbance and aging effects such that the C stock change and C accumulation due to all effects has stabilized (figs. 14b–d). National forests in the Eastern Region had a combined net gain of 300 Tg of total ecosystem C, including soil C, between 1950 and 2011 (fig. 14d).

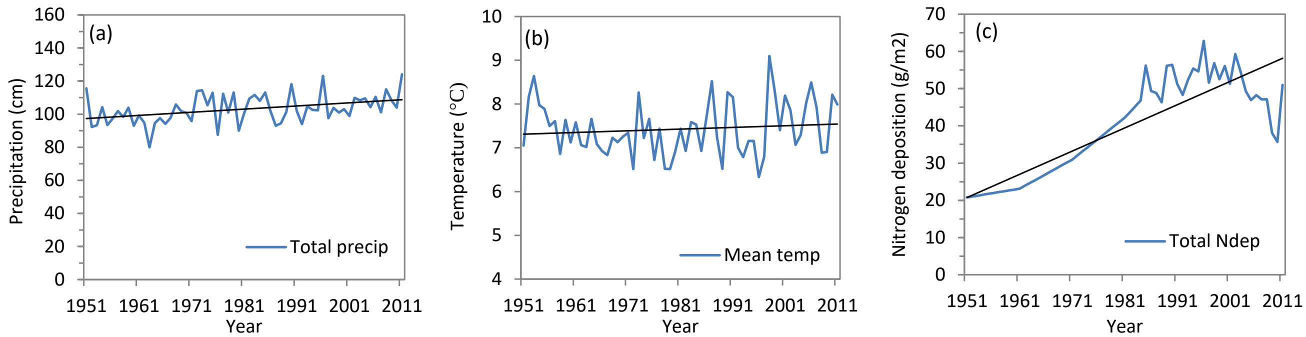

Figure 13a-c. (a) Total annual precipitation, (b) average annual temperature, and (c) total annual nitrogen (N) deposition from 1951 through 2011 averaged across all national forests in the Eastern Region. Linear trend lines shown in black.

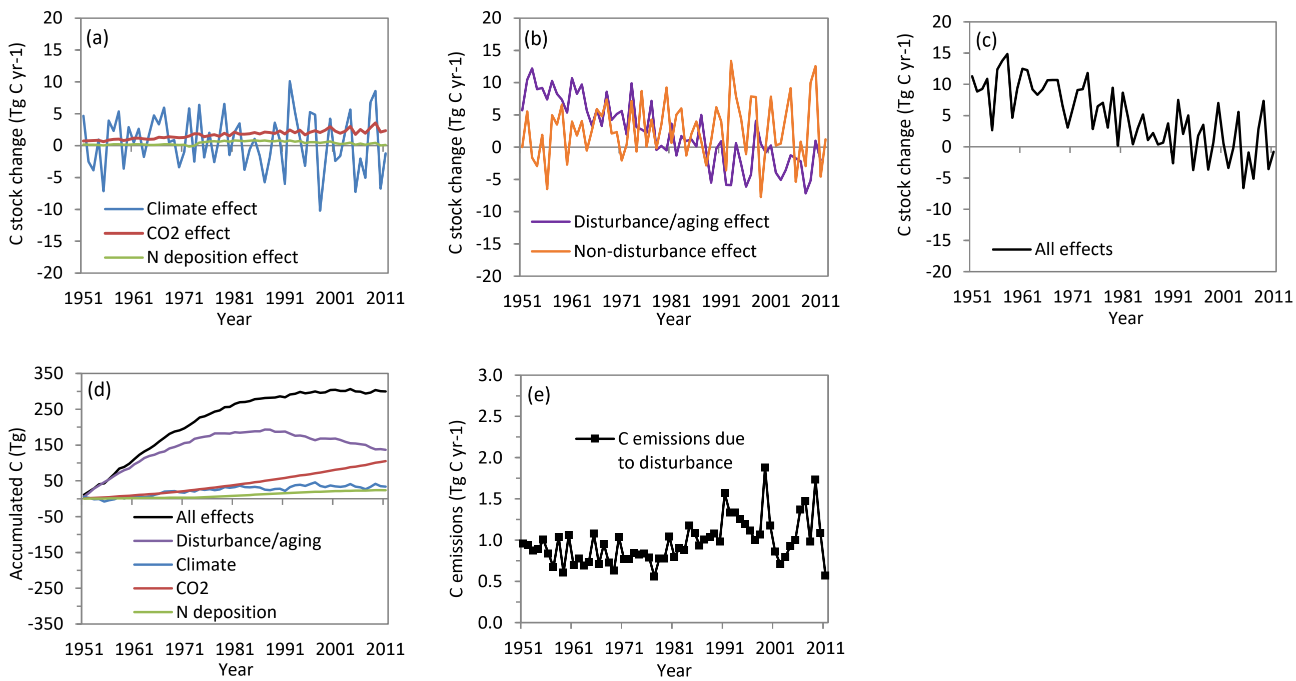

Figure 14a-e. Estimated forest carbon (C) changes and accumulations summed across the national forests in the Eastern Region. Changes in C stocks are attributed to: (a) individual nondisturbance factors alone, including climate variability, atmospheric carbon dioxide (CO_2) concentration, and nitrogen deposition; (b) all disturbance and aging factors including fire, harvest, insects, and regrowth and aging, and the sum of all nondisturbance factors combined; and (c) all factors combined, which is the sum of disturbance and aging and nondisturbance effects. (d) Accumulated C due to individual disturbance and aging and nondisturbance factors and all factors combined from 1950 through 2010 excluding C accumulated pre-1950. (e) Direct C emissions to the atmosphere due to disturbance events only. Positive values in graphs (a) through (c) represent net C sinks from the atmosphere, or increases in C stocks, whereas negative values represent net C sources to the atmosphere, or decreases in C stocks.

6.2 Southern Region

6.2.1 Description of Region

The Southern Region (also referred to as Region 8) in the National Forest System (NFS) contains 14 national forests or groups of national forests which have been administratively combined, spread across 13 States in the southeastern United States from Virginia to Florida and as far west as Texas and Oklahoma, as well as a national forest in Puerto Rico (fig. 15). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these national forests, so national grasslands were not included. To restrict the analysis to lands managed by the Forest Service, any private inholdings within national forest boundaries were excluded. Due to a lack of key datasets, El Yunque National Forest in Puerto Rico was not included in this assessment.

The history of development, land use, and policies provides useful context for understanding the regional forest C trends that are indicated by the inventory and assessment results. After the Civil War in the 1860s, agricultural expansion and large-scale timber extraction became the dominant driving forces of forest change in the South. The

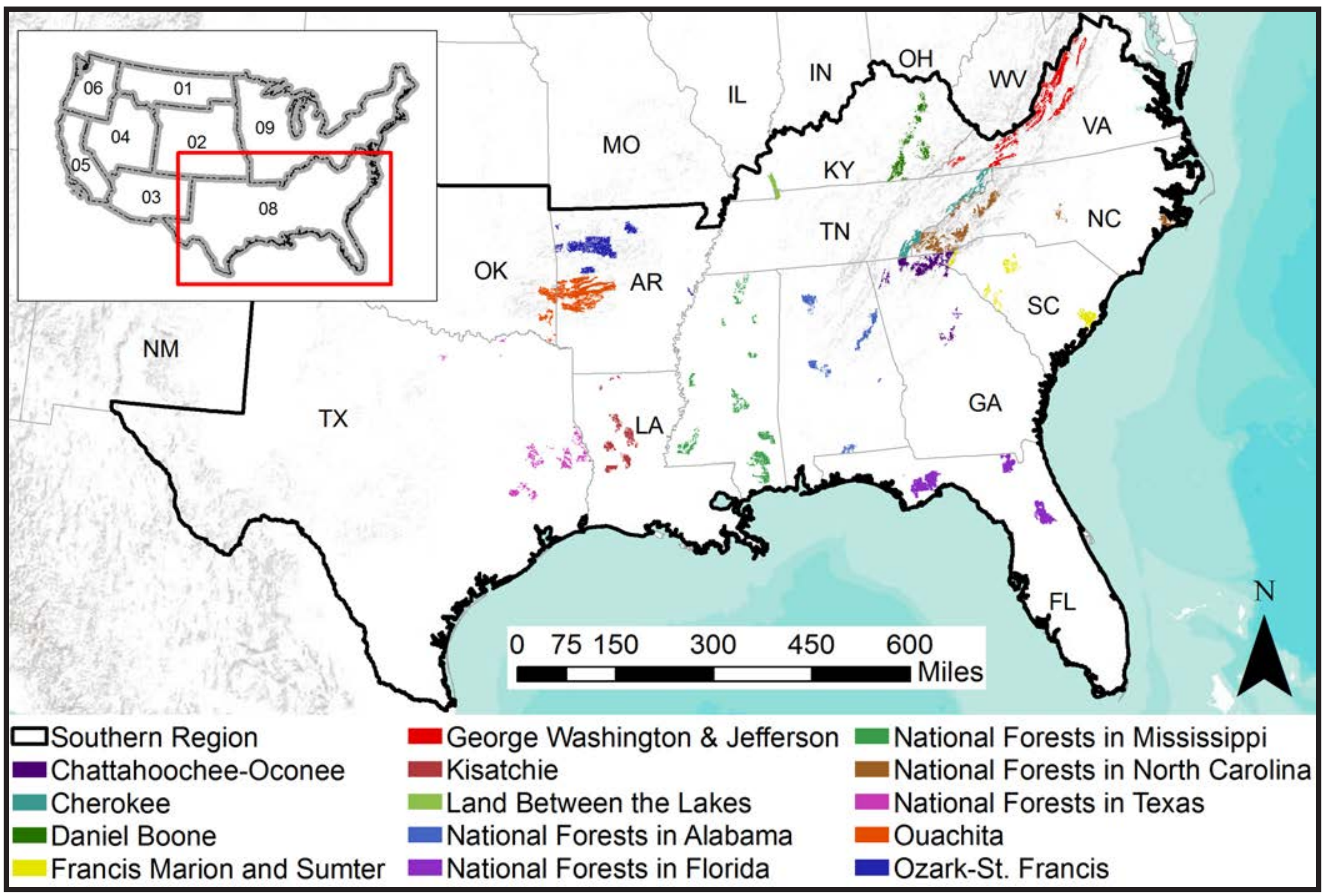

Figure 15. Locations of the national forests in the Southern Region. Not shown: El Yunque National Forest in Puerto Rico.

timber industry soon became centered in the South, and by 1919 the region was producing 37 percent of U.S. lumber (Williams 1989). During this period, most of the remaining primary forests of the South were harvested, and much of the previous forest land was replaced with agriculture and grazing.

After passage of the Weeks Act in 1911, the Forest Service began buying large areas of these heavily cut-over and submarginal lands, referred to as "the lands nobody wanted" (Shands 1992), throughout the eastern United States. By 1920 approximately 2 million acres (800,000 hectares) of excessively logged and degraded land was purchased as national forest land. Soon forest restoration and recovery became the new goal. With help from the Civilian Conservation Corps, millions of trees were planted, erosion was controlled, and forest fires were fought (Williams 2003). The increased effort to suppress fires allowed forests to regrow, which increased stocking of trees. As these forests were restored, the timber industry was also revived. From 1936 to the mid-1950s, timber harvests increased steadily and peaked in the mid-1980s before declining steeply in the 1990s into the 2000s (Loeffler et al. 2014c). The restoration of southern national forests not only is a conservation success story but also shaped the legacy of forest C dynamics in the region.

In addition to land use and policies, natural disturbances have played an important role in the forest C trends in the Southern Region. Except for timber harvesting, fire is the most common disturbance affecting southern forests. Major wildfires occurred during the droughts of the 1930s and 1950s. As research and experience increased understanding of the important role of fire in forest ecosystems, prescribed fire was again used in the South in recent decades to reduce hazardous fuels. Despite prescribed burning to keep hazardous fuels in check, wildfires—most of them human caused (Stanturf et al. 2002)—are still common in the South. The Southern Region also experiences tropical storms and hurricanes that can significantly affect forests. For instance, in 1989 Hurricane Hugo devastated some 3.5 million acres (1.4 million hectares) of the Francis Marion National Forest (Sheffield and Thompson 1992). Outbreaks of insects such as the southern pine beetle and gypsy moth have also impacted forest productivity, C cycling, and overall ecosystem health.

6.2.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Southern Region are displayed in figure 16. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. Regional disturbance patterns are marked by declining (and low) harvest levels, periodic large fire years (including 2000, 2004, and 2007), and a recent increase in insect activity. Of all disturbance types, harvests affected the greatest amount of forested area from 1990 to 2011, although annually harvests never exceeded 0.5 percent of the forested area. From 1990 to 2011, disturbances on about two-thirds of the forest land area were low to moderate intensity (<50 percent change in canopy cover). Validation activities for the disturbance maps in this region showed that the effects of hurricanes were underrepresented in the maps, and were thus underrepresented in subsequent ForCaMF assessments of C impact.

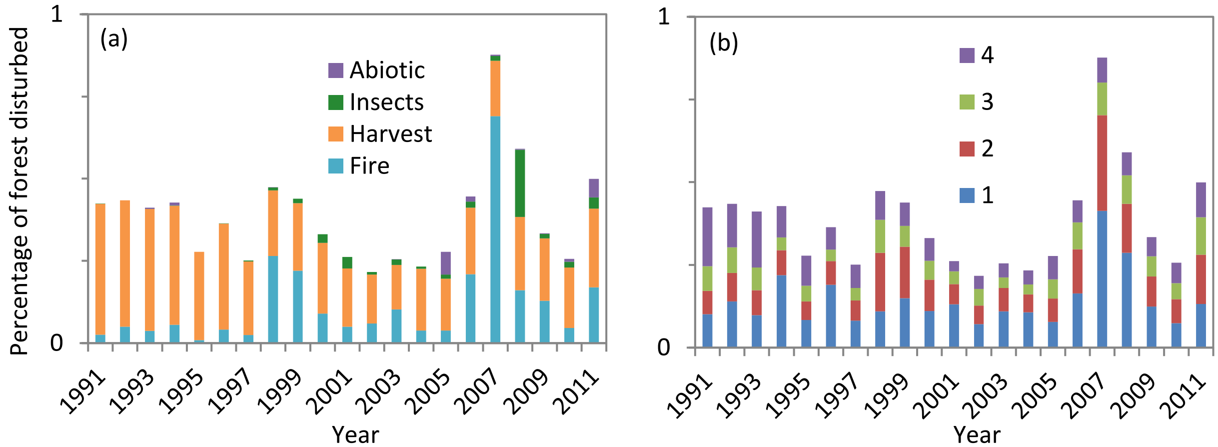

Figure 16. Annual rates of disturbance (0 to 1 percent) in the Southern Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

6.2.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to estimate the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Southern Region in figure 16. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires occur only in 2006, for example, the line for fire in figure 17 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an "undisturbed" scenario. Thus, figure 17 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

Forest harvest from 1990 to 2011 accounted for about two-thirds of the impact of disturbance on 2011 C stocks (figs. 17, 18). By 2011, this represented a 2.4-percent reduction in the regional nonsoil C stocks (fig. 19) that were reported in the NFS baseline C assessment derived from FIA data (USDA FS 2015h). While harvest rates have been relatively steady in terms of area, and were consistently higher than in other NFS regions, they never affected more than 0.5 percent of the landscape in a single year. These patterns do not account for offsite storage of C in wood products. Information about the region's product C storage can be found in the baseline C assessment (USDA FS 2015h).

Fire was responsible for approximately one-fourth of the region's observed disturbance impact (fig. 18), although that impact was not uniformly distributed (fig. 19). As described in Appendix 4: Southern Region (PDF, 4.1MB), the impact of fire per unit area in the National Forests in Florida was greater thanthe impact of all combined disturbance types in any national forest elsewhere in the region. Fires represented 59 percent of the total 1990 to 2011

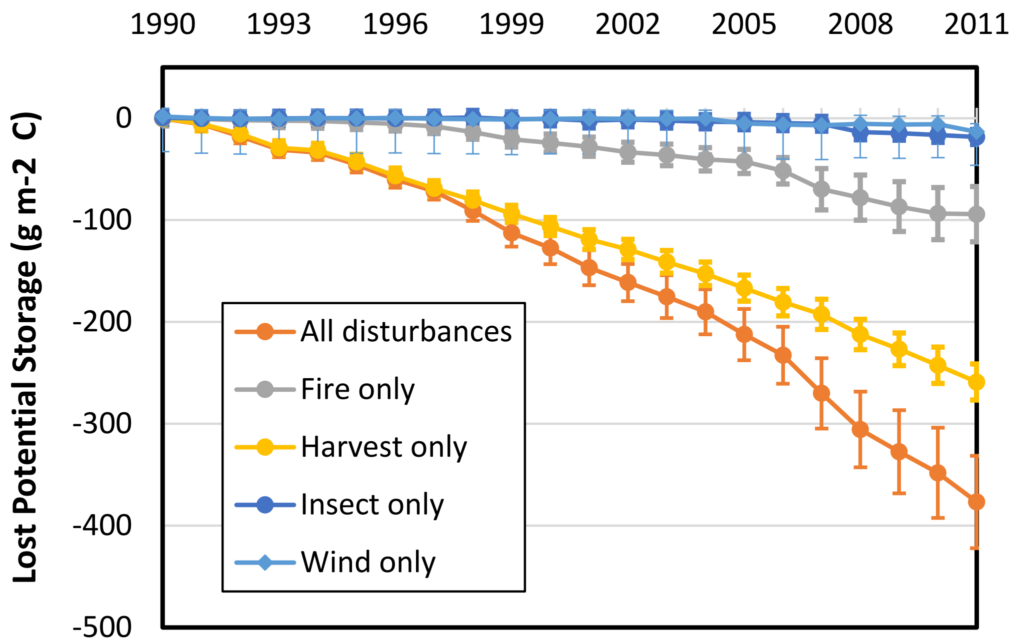

Figure 17. The impact of different kinds of disturbance, occurring from 1990 through 2011, on carbon (C) stores in the Southern Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

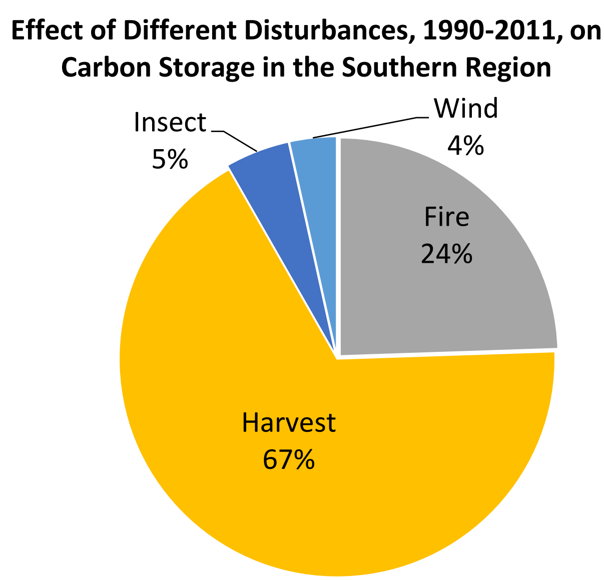

Figure 18. Proportional effect of different kinds of disturbance on carbon storage in all national forests in the Southern Region for the period 1990 through 2011.

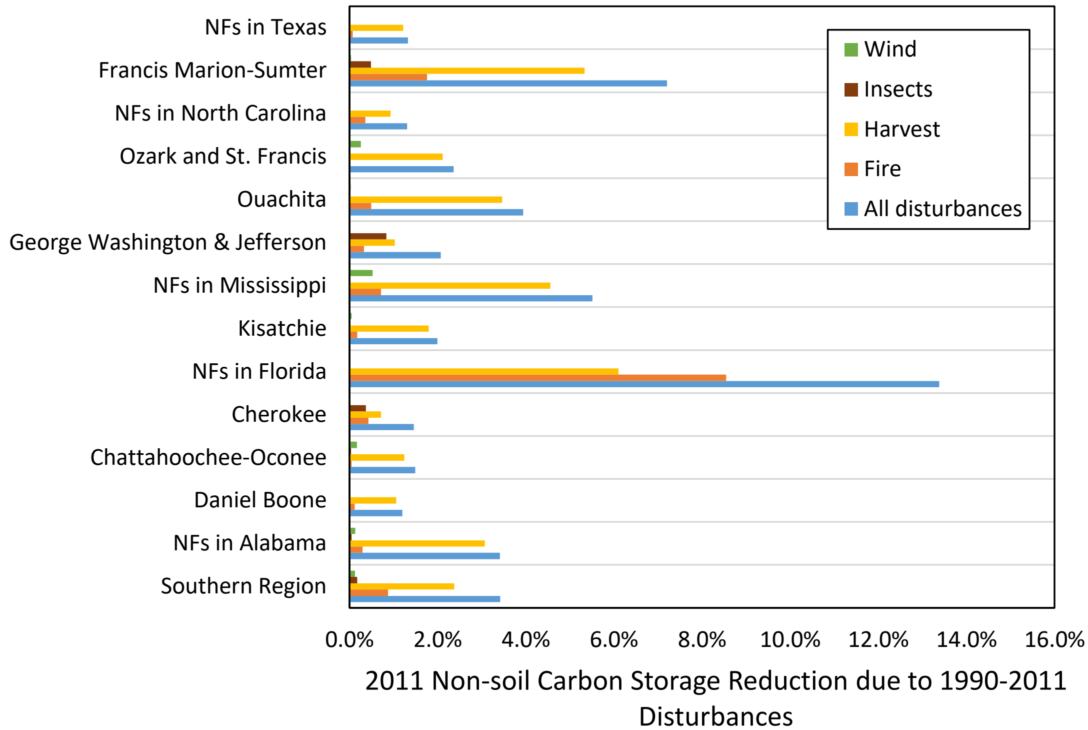

Figure 19. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by each national forest and for all national forests combined in the Southern Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

sequestration potential lost to disturbance in Florida, while that percentage was 28 percent for the Cherokee and 23 percent for the Francis Marion and Sumter National Forests. Fire was a more minor component of disturbance for the rest of the region, resulting in the loss of 0.9 percent of nonsoil C stocks by 2011 (fig. 19).

The effect of insects on C storage accounted for approximately 5 percent of the disturbance impact in the region during the study period (fig. 18), and a 0.2-percent decline in total nonsoil C stocks by 2011 (fig. 19). Losses were centered on the George Washington and Jefferson National Forests (38 percent of losses) and the Cherokee National Forest (25 percent of losses). The timing of the outbreaks responsible for these effects can be seen in figure 16, keeping in mind that release of C associated with mortality in any one year is spread out over many subsequent years.

By 2011, wind resulted in a 0.1-percent reduction in total nonsoil C storage in the Southern Region (fig. 19). Wind damage made up only 4 percent of disturbance-related losses in the period (fig. 18). This is a surprisingly low percentage, given the high profile of hurricane activity in the region and the observed impact of tornado events in forests largely unaffected by hurricanes (e.g., the Ozark St. Francis National Forest). In addition to the omission of some storm effects in the disturbance map, mentioned earlier, several factors may have contributed to the difference between perceived impact and what was observed. First, because the study period began in 1990, it missed the extensive effects of Hurricane Hugo in 1989. More subtly, the disturbance records used in this assessment intentionally avoided areas where ephemeral processes temporarily reduced foliage cover but did not result in permanent forest structure change. This distinction was made using subsequent high- and Landsat-resolution imagery. For C accounting purposes ForCaMF therefore ignored temporary defoliation events caused by storms (or insects), based on the assumption that relatively little C is affected in the loss of a single flush of foliage. Last, because there is no “wind” keyword in FVS, the C effects of mapped hurricanes were approximated by implementing harvest dynamics that left all material on the site. This workaround may have underestimated the C release rate.

One caveat must be stated about the confidence intervals depicted in figure 17. The unit of analysis for ForCaMF was the national forest (Appendix 4: Southern Region (PDF, 4.1MB)); rates of simulated error were constrained by FIA estimates at that level. Regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in other forests. This assumption very likely overstated certainty at the regional level; if initial biomass esti-mates in one forest were high, for example, the same estimates were also likely to be biased high in other regions. This caveat does not affect overall trends seen at the aggregated regional level, nor does it affect confidence intervals assessed at the national forest level. Harvest was the dominant disturbance process with respect to C cycling, and fire was important in some parts of the region, particularly Florida.

6.2.4 Effects of Disturbance, Management, and Environmental Factors

This section provides a regional summary of the predicted (modeled) effects of natural disturbances, land management, and environmental factors on forest C dynamics depicted in a series of figures (figs. 20–23) generated from InTEC model inputs and simulations for individual national forests and summed results across all national forests in the Southern Region. These regional-scale outputs were generated only from the national forest-specific datasets; thus, these outputs do not represent lands within the region that are outside of the national forest boundaries. The Appendix 4: Southern Region (PDF, 4.1MB) shows the effects of disturbance and nondisturbance factors on forest C dynamics within each of the 14 national forests as modeled by InTEC with this same series of figures.

According to the historical data and model results, forest C trends across the national forests in the Southern Region have been strongly influenced by the history of land use and land management policies. Although there is significant variability among national forests, the regionwide stand-age distribution in 2010 shows that most stands in the Southern Region are older (>70 years old) with a distinctive pulse of stands which established about 70 to 100 years earlier, or from approximately 1910 to 1940 (fig. 20). After 1940, the rate of stand establishment dramatically declined. This pulse of stand establishment during the early to mid-1900s reflects forest recovery and regeneration after decades of heavy logging and clearing of land for agriculture. Depending on the forest-type group, which was mostly oak/hickory, oak/pine, or loblolly/shortleaf pine (fig. 20), the stands making up this pulse of establishment would have been growing at maximum productivity when they were 30 to 45 years old (fig. 21), or around 1940 through the 1960s (see Appendix 1 for scientific names of species mentioned in this report). Forests in the Southern Region show another much smaller pulse of stands 20 to 30 years old, which established from 1980 to 1990. This second pulse reflects recovery after a range of natural and anthropogenic disturbances such as the Kisatchie Hills Fire in Louisiana in 1987 (Kulhavy and Ross 1988), Hurricane Hugo in 1989,

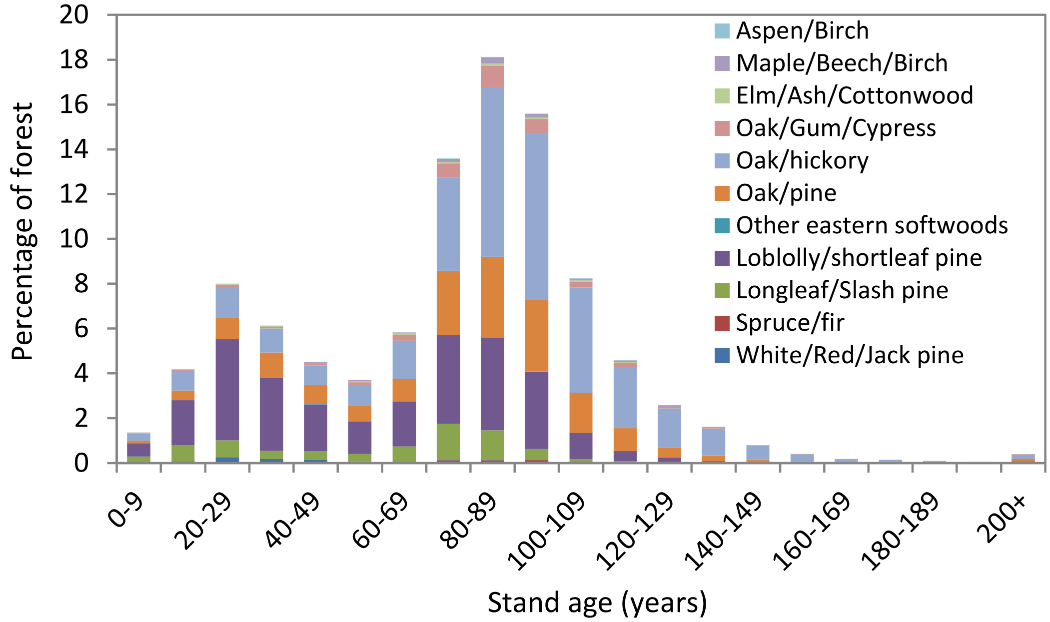

Figure 20. Age-class distribution in 2010 shown as the percentage of forest land in each forest-type group in 10-year age classes summed across the national forests in the Southern Region (see Appendix 1 for scientific names of trees).

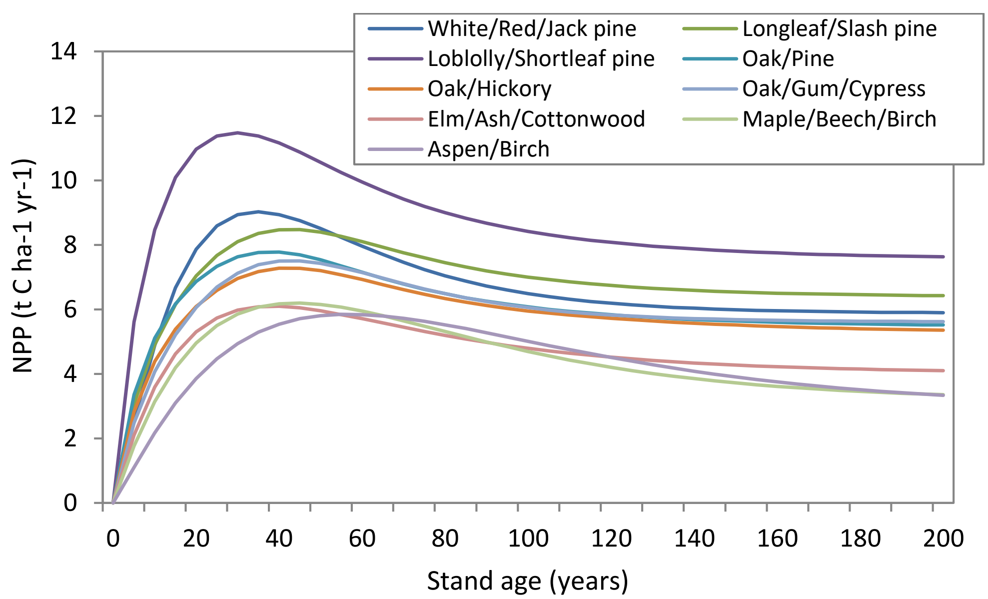

Figure 21. Relationship between net primary productivity (NPP) and stand age for each forest-type group averaged across all national forests in the Southern Region (see Appendix 1 for scientific names of trees).

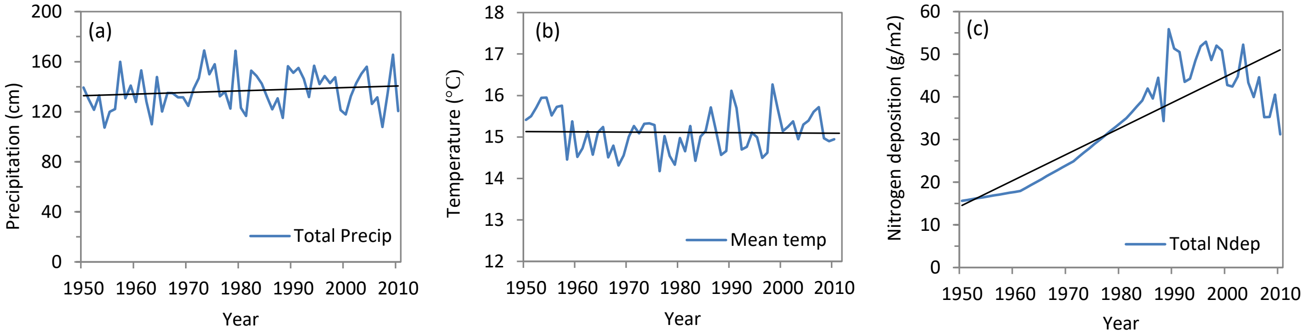

Figure 22a-c. (a) Total annual precipitation, (b) average annual temperature, and (c) total annual nitrogen deposition from 1950 through 2011 averaged across all national forests in the Southern Region. Linear trend lines shown in black.

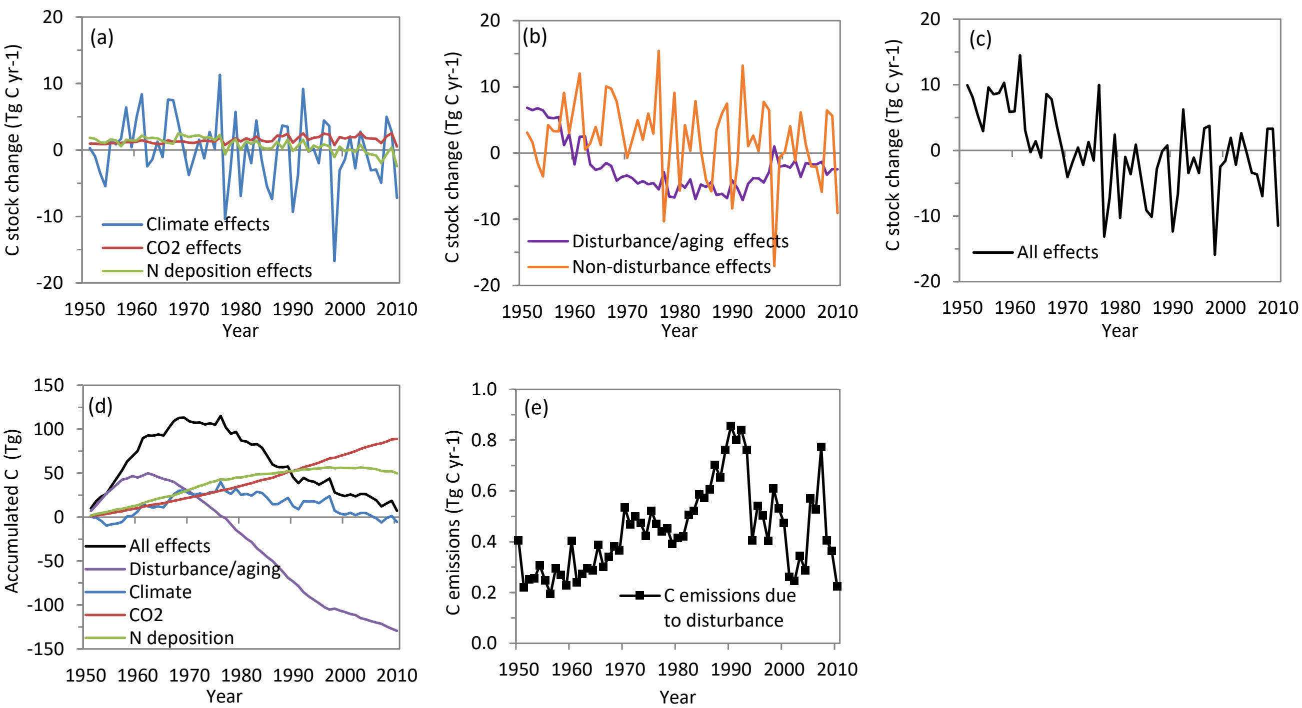

Figure 23a-e. Estimated forest carbon (C) changes and accumulations summed across the national forests in the Southern Region. Changes in C stocks are attributed to: (a) individual nondisturbance factors alone, including climate variability, atmospheric carbon dioxide (CO_2) concentration, and nitrogen deposition; (b) all disturbance and aging factors including fire, harvest, insects, and regrowth and aging, and the sum of all nondisturbance factors combined; and (c) all factors combined, which is the sum of disturbance and aging and nondisturbance effects. (d) Accumulated C due to individual disturbance and aging and nondisturbance factors and all factors combined from 1950 through 2010 excluding C accumulated pre-1950. (e) Direct C emissions to the atmosphere due to disturbance events only. Positive values in graphs (a) through (c) represent net C sinks from the atmosphere, or increases in C stocks, whereas negative values represent net C sources to the atmosphere, or decreases in C stocks.

and a spike in timber harvesting in the mid-1980s (Loeffler et al. 2014c).

The climate in the Southern Region has become slightly wetter over the past few decades. On the other hand, temperature trends have varied over time, with warmer temperatures in the 1950s, a decline in temperatures from the 1960s to 1980s, followed by another warming period through 2010 (figs. 22a,b). Though change may be minimal over the long term, interannual climate variability has been high with several years of notable climatic extremes. For instance, the Southern Region experienced a prolonged drought compounded with above-average temperatures for much of the 1950s. A prominent summer heat wave in 1998 as well as severe drought conditions in 2007 occurred across much of the southern United States. Warmer temperatures and a drier climate can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity. Consequently, more C is released to the atmosphere. Both atmospheric CO_2 concentrations and N deposition (fig. 22c) have increased over the past few decades as a result of human activities. Nitrogen deposition has been slowly declining since the early 2000s.

The C stock changes across the national forests in the Southern Region show that forests were mostly a C sink from the 1950s through the 1970s, then switched to mostly a C source (fig. 23c) according to the modeled results. This shift from a sink to a source is primarily a result of forests getting older on average and therefore growing more slowly, although climate effects have also played an important though highly variable role (figs. 23a,b). The decline in C stocks due to disturbance and aging can largely be explained by the stand-age distribution. The pulse of stands 70 to 100 years old (fig. 20) was growing at maximum productivity for much of the 1950s and 1960s, but as forests continued to age, their productivity declined (fig. 21) and C losses from decomposition became larger than C gains. This decline was coupled with lower rates of stand establishment in the mid-1900s.

Since the late 1990s, the decline in C accumulation has become less steep, most likely due to the recovery after disturbances in the 1980s and 1990s. These more recent disturbances caused increases in direct C emissions (fig. 23e), but also promoted regrowth and recovery (fig. 20). In about a decade, these stands will be highly productive (fig. 21) and may be able to offset the C losses due to the mostly older, less productive stands.

In the 1950s climate had a mostly negative effect on C accumulation, most likely due to the prolonged drought and warmer temperature (figs. 22a,b). Aside from several notably warmer years such as 1990 and 1998, climate has had a mostly positive effect on changing C stocks from 1950 through 2010 (figs. 23a,d). The increases in N deposition (fig. 22c) and atmospheric CO2 concentrations have both had positive effects on changing C stocks and C accumulation across all forests in this region (figs. 23a,d). The N deposition effect started to stabilize and decline in the 2000s due to declining N deposition rates (fig. 22c). Overall, the positive effects of climate, atmospheric CO_2, and N deposition helped to offset the C losses due to disturbances and aging such that when combined the national forests in the Southern Region experienced a slight net gain of 7.2 Tg of total ecosystem C, including soil C (fig. 23d) between 1950 and 2010.

6.3 Northern Region

6.3.1 Description of Region

The Northern Region (also referred to as Region 1) in the National Forest System (NFS) consists of 11 national forests spread across northwestern South Dakota, Montana, northern Idaho, and northeastern Washington (fig. 24). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these national forests, so national grasslands such as the Dakota Prairie Grasslands in North Dakota were not included. To restrict the analysis to lands managed by the Forest Service, any private inholdings within national forest boundaries were excluded.

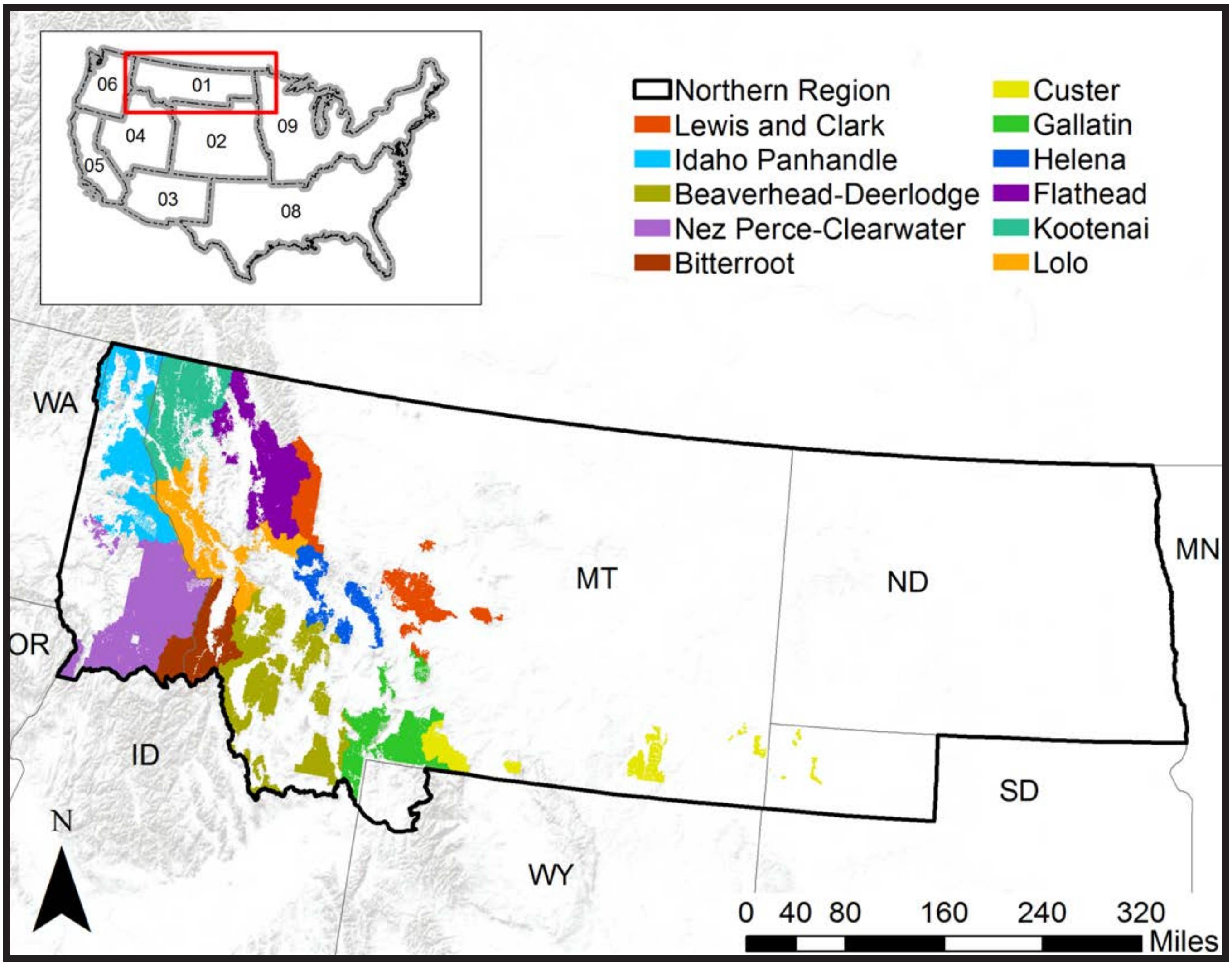

Figure 24. Locations of the national forests in the Northern Region.

The history of Euro-American settlement, land use, and policies provides useful context for understanding the regional forest C trends that are indicated by the inventory and assessment results. Euro-American settlement first expanded into the Northern Region in the early 1800s, but it was not until the mid- to late 1800s that the major influx of permanent settlers arrived and began altering the landscape. In the 1880s, large areas of land, typically in valleys, were cleared for agriculture, settlements, and railroad expansion. By the 1890s, the completion of the railroad opened up new, national markets that propelled the growth of the region's forest products industry. The railroad also brought more settlers to the region. Although timber harvesting began on a small scale in the late 1800s and continued to expand and intensify through the early 1900s (Baker et al. 1993), it was not until the 1940s that the Northern Region experienced large-scale logging operations on national forest lands. Harvest volumes in the Northern Region greatly declined by the 1990s (Stockmann et al. 2014b).

In addition to timber harvesting, the forest C legacy of the Northern Region is tied to its history of fires and fire management. For instance, thousands of acres of the region's forests were burned by large fires in 1889, prior to the organization of a firefighting system and look-out towers. After the devastating 1910 fire season, which left some 3 million acres (1.2 million hectares) of forest land scorched in Idaho and Montana, fire protection came to the forefront of Forest Service policies. The Forest Service introduced a national fire suppression policy, which entailed intensive efforts to prevent, detect, and suppress all wildfires (Baker et al. 1993; Pyne 1982). A policy shift from fire control to fire management in the 1970s allowed natural fires to burn across landscapes where considered safe and appropriate, in an attempt to restore historical fire regimes (Pyne 1982).

Both land use history and recent changes in climate have led to large and intense disturbances in the Northern Region over the past few decades. The Northern Rockies have experienced increases in large wildfires that burn longer, as well as longer fire seasons; much of this change is due to warming temperatures and droughts (Westerling et al. 2006). Over the past few decades, severe bark beetle outbreaks affecting several dominance types (see Appendix 1 for description of dominance types) caused widespread tree mortality, reducing forest C uptake and increasing future emissions from the decomposition of killed trees (Kurz et al. 2008).

6.3.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Northern Region are displayed in figure 25. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. Regional disturbance patterns have been marked by declining (and low) harvest levels, periodic large fire years (including 2000, 2004, and 2007), and a recent increase in insect activity. Fire has been the dominant disturbance type, accounting for about two-thirds of all disturbances and affecting a total of about 5 percent of the forests between 1990 and 2011. Over this period, disturbances exhibited a relatively even mix of low and high intensities.

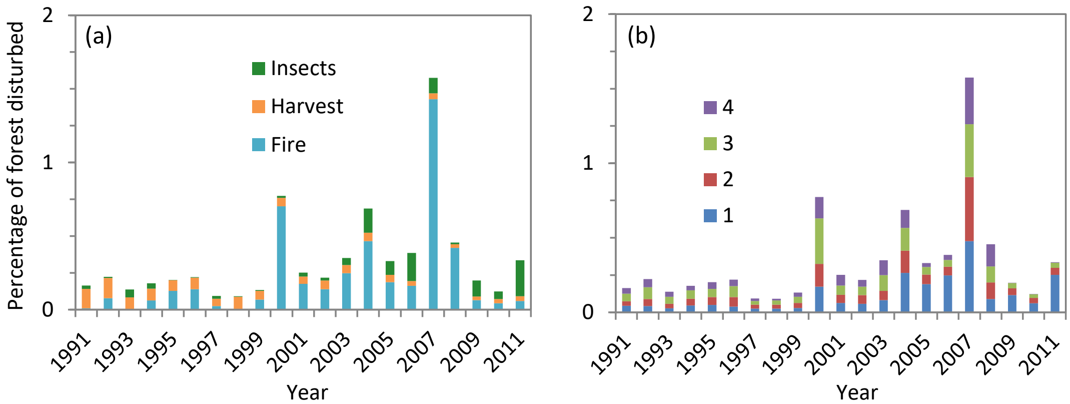

Figure 25. Annual rates of disturbance (0 to 2 percent) in the Northern Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

In addition to these processes, root disease is known to substantially affect stand dynamics in parts of the region (Byler and Hagle 2000). Much of the impact of root disease is manifested in reduced growth rates and suppression of regeneration, which can be difficult to detect using remote sensing. However, FIA maintains a regional root disease severity variable (over nine classes), and at the time of this analysis, results had been processed by Northern Region staff for an area covering six of the region's forests: Lolo, Bitterroot, Nez Perce Clearwater, Flathead, Idaho Panhandle, and Kootenai. Twenty-nine percent of the subplots in those forests had detectable signs of root disease in the last available measurement. This assessment assumed this rate of infection to be constant throughout the study period, although infection is dynamic and generally grows over time. While root disease can remain on a site for decades and infection in the region was well established before 1990 (Byler et al. 1990), the assumption may have resulted in some degree of overestimation of disease prevalence early in the study period. On the other hand, no root disease was accounted for in forests not covered by the disease severity variable, certainly causing some omission of disease effects.

6.3.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to estimate the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Northern Region in figure 26. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires first occur in 2000, for example, the line for fire in figure 26 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an "undisturbed" scenario. Thus, figure 26 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

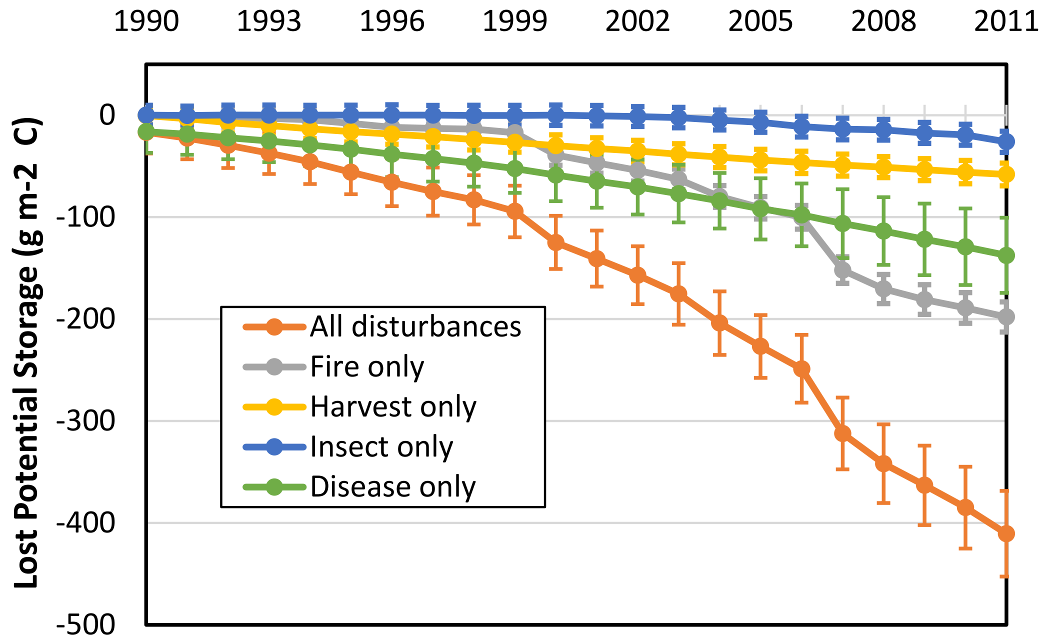

Figure 26. The impact of different kinds of disturbance, occurring from 1990 through 2010, on carbon (C) stores in the Northern Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

The long-term impact is especially evident in the estimated effects of harvest. Effects continue to increase, albeit very slowly (relative to C that would have been stored in the absence of harvest) even though harvest activity virtually ceased across the region during the 1990s. These patterns do not account for offsite storage of C in wood products. Information about the region's product C storage can be found in the NFS baseline C assessment (USDA FS 2015d). Few fires were detected during the early 1990s, so the line for fire starts near zero. However, following an increase in large fires that began around 2000, the impact of fire on C stocks greatly increased. The residual effects of these fires are likely to persist (and perhaps increase) through future decades both because C added through recovery may not equal C that would have been added through continued growth, and because decomposing material killed by the fire will offset C added through recovery. By 2011, fires resulted in a 1.6-percent reduction in the regional nonsoil C stocks (fig. 28) that were reported in the baseline C assessment (USDA FS 2015d).

Recent increases in the area affected by insect activity (fig. 25) are translated into recent increases in the impact of insects on C storage (fig. 26). Root disease was a dominant disturbance factor (fig. 27) because it affects a large portion of the region, and because its steady suppression of growth and regeneration chronically limits the ability of affected stands to add C. Disease impacts from 1990 through 2011 resulted in the loss of 1.0 percent of nonsoil C storage (fig. 28).

One caveat must be stated about the confidence intervals depicted in figure 26. The unit of analysis for ForCaMF was the national forest; rates of simulated error were constrained by FIA estimates at that level. The regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in other forests. This assumption very likely overstated certainty at the regional level; if initial biomass estimates in one forest were high, for example, the same estimates were also likely to be biased high in other units. This caveat does not change the overall trends seen at the regional level, nor does it affect confidence intervals assessed at the national forest level. Fire is clearly the dominant process, but root disease also strongly affects

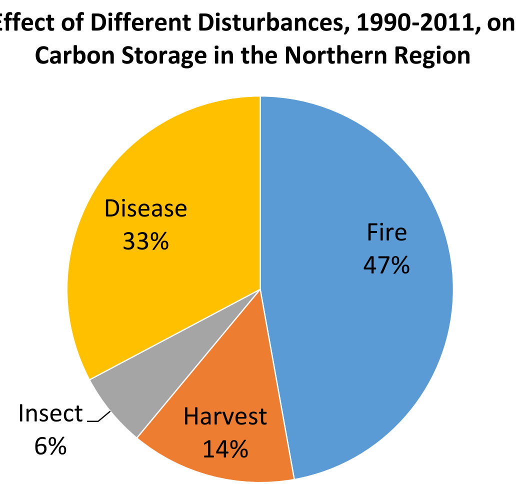

Figure 27. Proportional effect of different kinds of disturbance on carbon storage in all national forests in the Northern Region for the period 1990 through 2011.

C storage in the region (figs. 27, 28). The importance of insect activity, while currently relatively small, was virtually undetectable through the 1990s. Its growing importance parallels recent increases in the area affected (fig. 25).

6.3.4 Effects of Disturbance, Management, and Environmental Factors

This section provides a regional summary of the predicted (modeled) effects of natural disturbances, land management, and environmental factors on forest C dynamics depicted in a series of figures (figs. 29–32) generated from InTEC model inputs and simulations for individual forests and summed results across all national forests in the Northern Region. These regional-scale outputs were generated only from the national forest-specific datasets; thus, these outputs do not represent lands within the region that are outside of the national forest boundaries. The Appendix 5: Northern Region (PDF, 2.9MB) shows the effects of disturbance and nondisturbance factors on forest C dynamics within each of the 11 national forests as modeled by InTEC using this same series of figures.

According to the historical data and model results, forest C trends across the national forests in the Northern Region have been strongly influenced by the history of land use and policies as well as climate change and natural disturbances. Despite variation among these national forests, the regionwide stand-age distribution in 2010 shows that most stands are

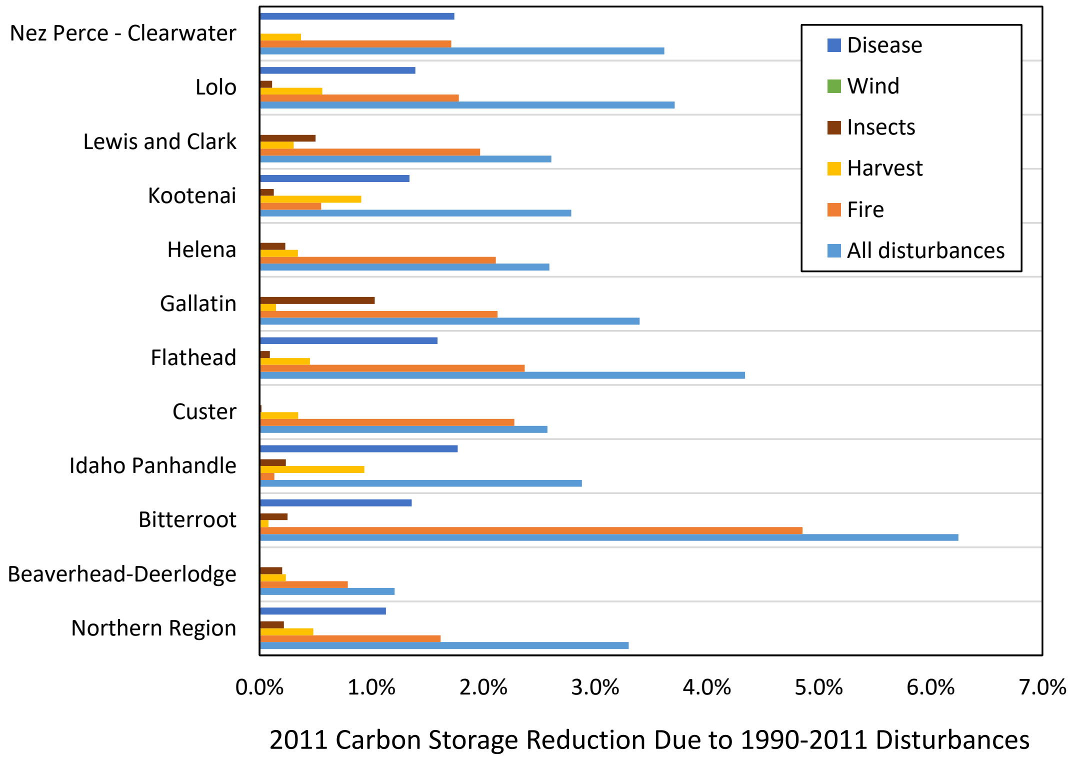

Figure 28. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2010, by each national forest and for all national forests in the Northern Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

older (>80 years old) with a distinctive pulse of stands which established about 80 to 110 years earlier, or from approximately 1900 to 1930 (fig. 29). After 1930 the rate of establishment dramatically declined. This early-1900s pulse of stand establishment may be a result of regeneration after the last major fires before fire suppression, such as those in 1889 or 1910, or after timber harvests, which intensified in the early 1900s (Baker et al. 1993). Fire suppression, which began in the early 1900s, would have allowed more of these young, regenerating stands to survive and continue regrowing rather than being disturbed at the more natural rate of fires (Pyne 1982). Depending on the forest dominance type, which is mostly Douglas-fir and subalpine fir, stands making up this pulse of establishment would have reached maximum productivity between 30 and 60 years of age (fig. 30), or throughout the mid- to late 20th century. Forests in the Northern Region show another pulse of young stands (<20 years old) (fig. 29) that established between 1990 and 2010, suggesting regeneration after recent large and often severe disturbances—mostly fires.

Climate has on average become warmer and slightly drier in this region (figs. 31a,b). Warmer temperatures and a drier climate can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity. Consequently, more C is released to the atmosphere. As climate continues to change, this region is expected to get even warmer and potentially drier into the future. Both atmospheric CO_2 concentrations and N deposition (fig. 31c) have increased over the past few decades as a result of human activities.

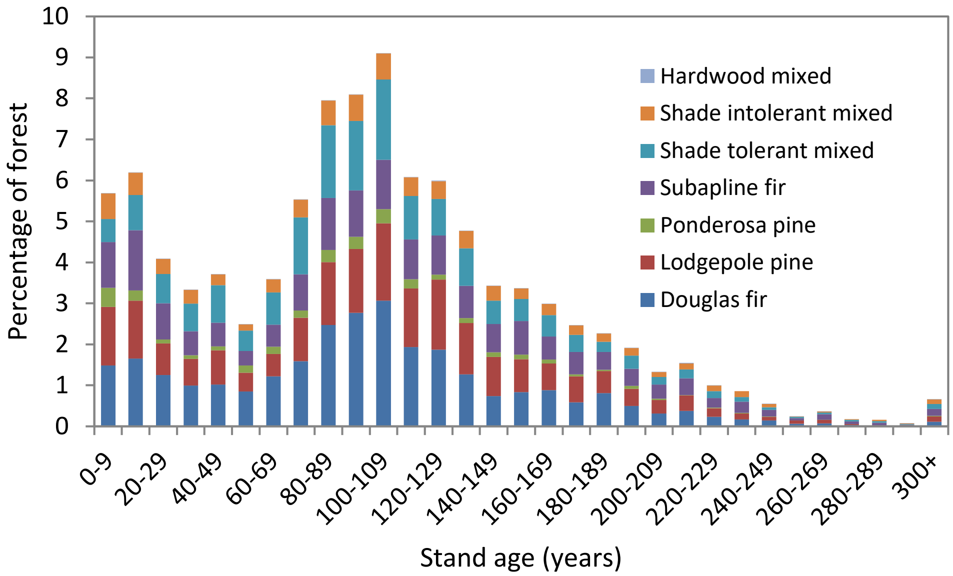

Figure 29. Age-class distribution in 2011 shown as the percentage of forest land in each forest dominance group in 10-year age classes summed across the national forests in the Northern Region (see Appendix 1 for description of dominance types).

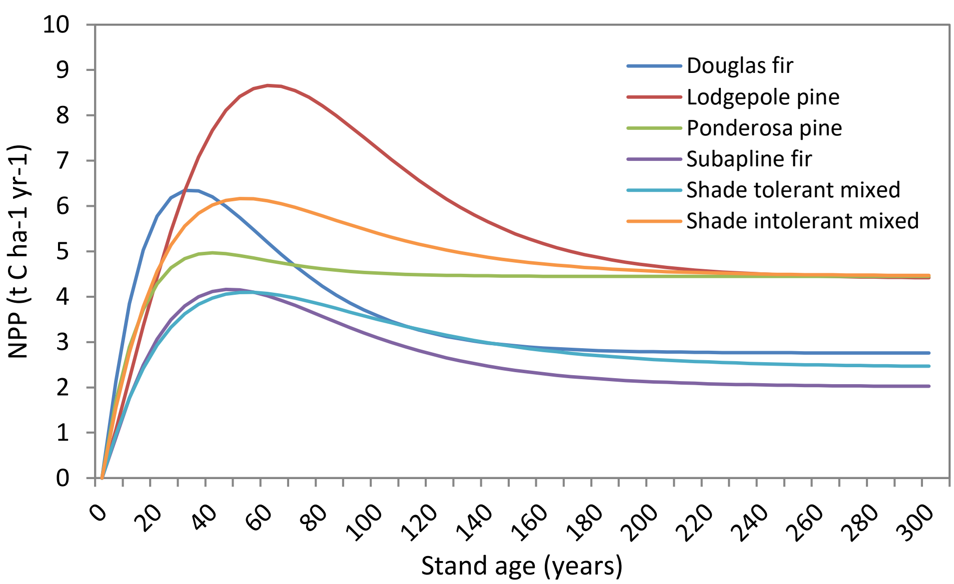

Figure 30. Relationship between net primary productivity (NPP) and stand age for each forest dominance type averaged across all national forests in the Northern Region. Dominance types are Douglas-fir (PSME), subalpine fir (ABLA), lodgepole pine (PICO), ponderosa pine (PIPO), shade-tolerant mixed conifer (TMIX), and shade-intolerant mixed conifer (IMIX) (see Appendix 1 for description of dominance types). Due to a small sample of hardwood mixed (HMIX) plots to derive NPP-age curves, the IMIX curve was used for the HMIX stands.

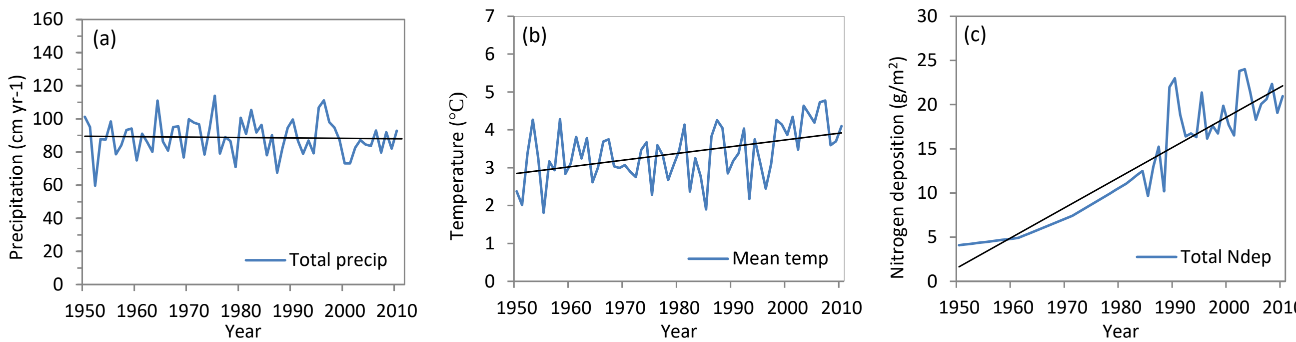

Figure 31a-c. (a) Total annual precipitation, (b) average annual temperature, and (c) total annual nitrogen deposition from 1950 through 2011 averaged across all national forests in the Northern Region. Linear trend lines shown in black.

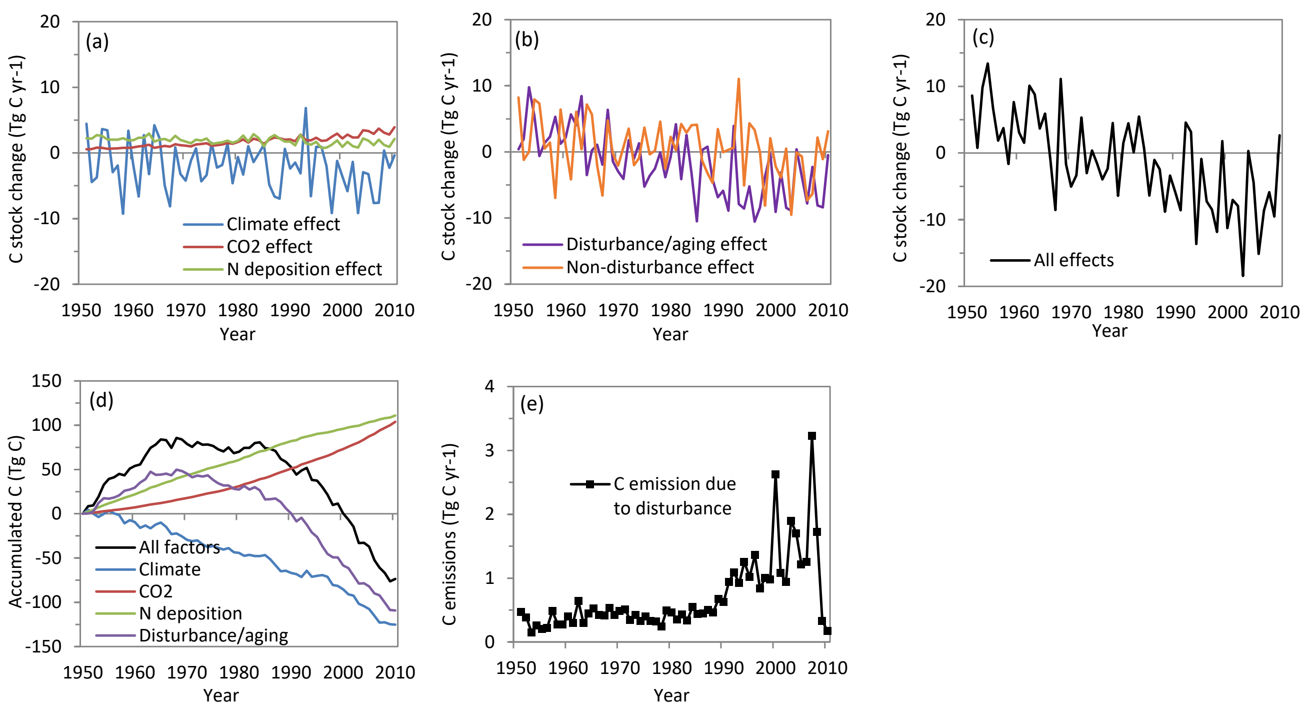

Figure 32a-e. Estimated forest carbon (C) changes and accumulations summed across the national forests in the Northern Region. Changes in C stocks are attributed to: (a) individual nondisturbance factors alone, including climate variability, atmospheric carbon dioxide (CO_2) concentration, and nitrogen deposition; (b) all disturbance and aging factors including fire, harvest, insects, and regrowth and aging, and the sum of all nondisturbance factors combined; and (c) all factors combined, which is the sum of disturbance and aging and nondisturbance effects. (d) Accumulated C due to individual disturbance and aging and nondisturbance factors and all factors combined from 1950 through 2010 excluding C accumulated pre-1950. (e) Direct C emissions to the atmosphere due to disturbance events only. Positive values in graphs (a) through (c) represent net C sinks from the atmosphere, or increases in C stocks, whereas negative values represent net C sources to the atmosphere, or decreases in C stocks.

The C stock changes across the national forests in the Northern Region show that together the forests have generally undergone a switch from a C sink to a C source (figs. 32 a–c) and a decline in accumulated C (fig. 32d). Disturbance and aging effects have been mostly responsible for declining C stocks (fig. 32b), and correspond to disturbances that shaped forest structure (fig. 29), the NPP-stand age relationships (fig. 30), and increasing disturbances in recent years (fig. 25). From 1950 to around 1980, the forests were mostly a C sink due to positive disturbance and aging effects, as the early-1900s pulse of stands was middle-aged and therefore growing at peak productivity. As these forests further aged, productivity declined, causing the rate of C accumulation to decline (fig. 32d). Meanwhile, the effects of disturbance were increasing to the point where C emissions due to decomposition and disturbances exceeded C gains, and as a result the forests became a C source (figs. 32b,c). This decline was coupled with lower rates of stand establishment in the mid-1900s.

The ForCaMF model results indicate that disease caused a significant loss of potential C storage over the past two decades (fig. 26). The InTEC model did not explicitly consider the effects of disease disturbances. But if disease caused stand-replacing mortality, it would be reflected in the current age structure, and thus included in InTEC. However, any lower-severity disease disturbances were not directly modeled by InTEC. Therefore, it is likely that disturbance and aging caused the C source to be even greater in recent decades than the InTEC results suggest.

Although recent disturbances initially caused increases in C emissions during the year of the disturbance events (fig. 32e), they also promoted regrowth and recovery, as shown by the recent pulse of stands less than 20 years old (fig. 29). As these young stands recover and reach middle-age in the coming decades, they will be growing at higher productivity rates (fig. 30). Thus, forests have the potential to accumulate more C and become C sinks again.

Climate variability and the recent warming trend have had a mostly negative effect on C stocks, also contributing to the switch to a C source (fig. 32a) and loss of C (fig. 32d). Future warming may result in an intensification of these already negative climate effects. The increases in N deposition (fig. 31c) and atmospheric CO_2 concentrations have both had positive effects on changing C stocks and C accumulation across all forests in this region (figs. 32a,d). However, the gains from CO_2 fertilization and N deposition were generally overshadowed by C losses due to negative disturbance and aging effects and climate effects (fig. 32d). Atmospheric CO_2 concentrations are expected to continue increasing for the foreseeable future, potentially counteracting the projected negative effects of climate.

Although a few national forests in the Northern Region had a net gain of total forest C, most forests had a loss from 1950 to 2010 (Appendix 5: Northern Region (PDF, 2.9MB)), resulting in a regionwide loss of approximately 74 Tg of total ecosystem C, including soil C, between 1950 and 2010 (fig. 32d).

6.4 Rocky Mountain Region

6.4.1 Description of Region

The Rocky Mountain Region (also referred to as Region 2) in the National Forest System (NFS) consists of 11 national forests within Colorado, Nebraska, Kansas, and most of Wyoming and South Dakota (fig. 33). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these national forests, so Thunder Basin, Nebraska, Pawnee, and Cimarron and Comanche National Grasslands were not included. To restrict the analysis to lands managed by the Forest Service, any private inholdings within national forest boundaries were excluded.

The history of Euro-American settlement, land use, and policies provides useful context for understanding the regional forest C trends that are indicated by the inventory and assessment results. Though Euro-American settlement first expanded into the Rocky Mountain Region in

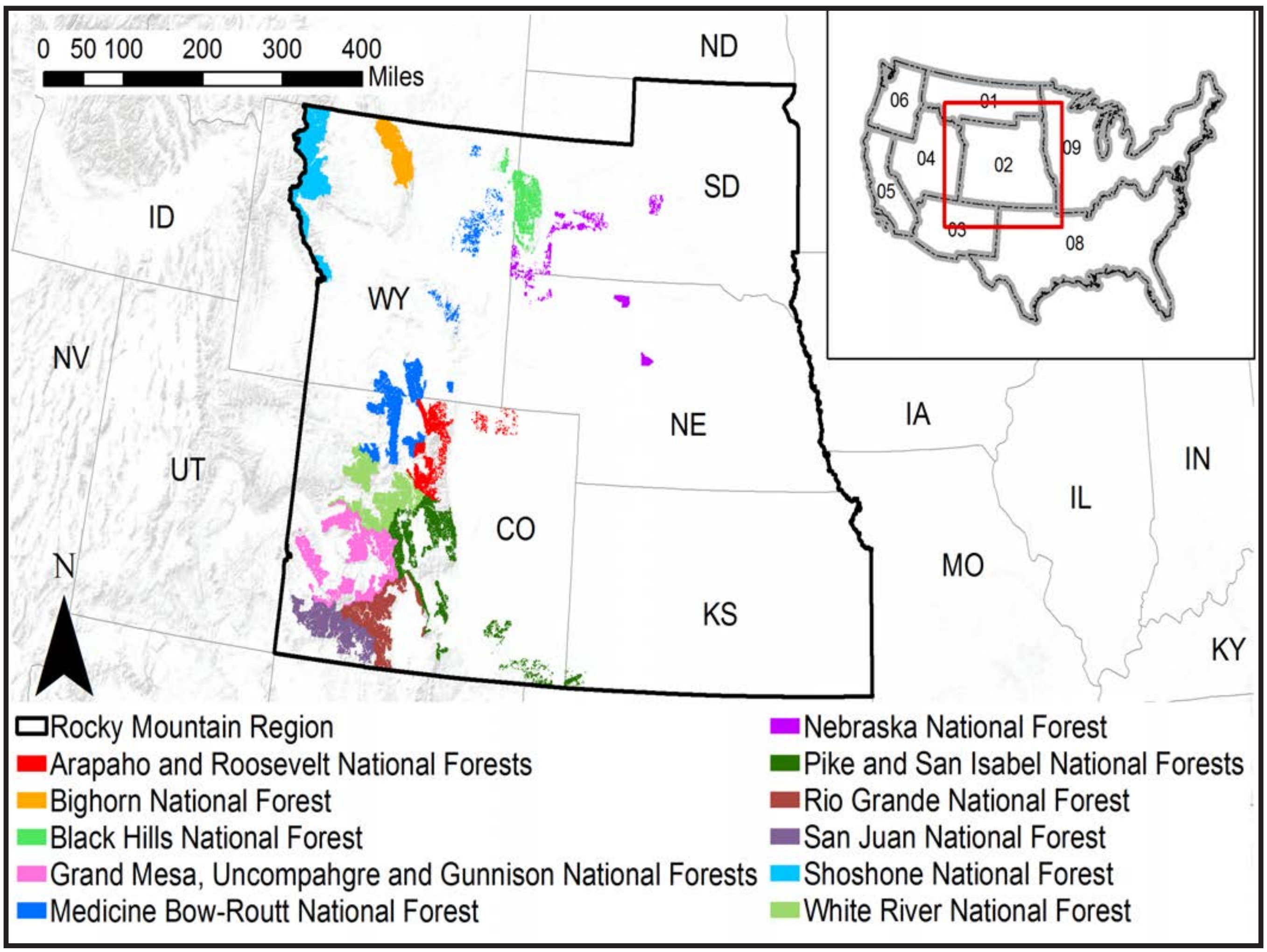

Figure 33. Locations of the national forests in the Rocky Mountain Region.

pursuit of fur trading in the early 1800s, it was not until the mid- to late 1800s that the major influx of permanent settlers arrived and began altering the landscape (Romme et al. 2009). Settlers' activities included logging, livestock grazing, mining, and clearing of forest for agriculture and human settlements. Logging began as early as 1875 in some areas of the region (Romme et al. 2009; Shepperd and Battaglia 2002), but was mostly small-scale and local until the 1890s, when logging greatly intensified with the advent of the railroad in the region. However, timber harvesting in the Rocky Mountain Region remained relatively low until the 1930s, after which it steadily increased before peaking in the late 1980s. Harvest volumes declined sharply in the early 1990s and have since remained relatively low (Stockmann et al. 2014d).

In addition to timber harvesting, the forest C legacy of the Rocky Mountain Region is tied to its history of fires and fire management. Before Euro-American settlement, fire was the dominant disturbance type influencing forest structure and C dynamics in the region. After settlement, extensive livestock grazing throughout the region greatly reduced surface fuels, limiting fire spread and altering fire regimes. Then after the devastating 1910 fire season, which scorched millions of acres of forest across the western United States, fire protection came into the forefront of Forest Service policies. With passage of the Weeks Act in 1911, the Forest Service introduced a national fire suppression policy, which entailed intensive efforts to prevent, detect, and suppress all wildfires (Agee 1998; Pyne 1982). A policy shift from fire control to fire management in the 1970s allowed natural fires to burn across landscapes where considered safe and appropriate, in an attempt to restore historical fire regimes (Pyne 1982).

Both land use history and recent changes in climate have led to large and intense disturbances in the Rocky Mountain Region over the past few decades. The Rockies have experienced increases in large wildfires, as well as longer fire seasons. Much of this change is due to warming temperatures and droughts (Westerling et al. 2006). Over the past few decades, severe outbreaks of bark beetles such as the mountain pine beetle have caused widespread tree mortality, reducing forest C uptake and increasing future emissions from the decomposition of killed trees (Kurz et al. 2008) (see Appendix 1 for scientific names of species mentioned in this report).

6.4.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Rocky Mountain Region are displayed in figure 34. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. Regional disturbance patterns were dominated by an outbreak of mountain pine beetle, which began killing large numbers of trees in the national forests around 2005. Though comparatively low levels of insect activity were noted in the region throughout the beginning of the study period, disturbance rates (indicating newly observed mortality) remained high after 2005, approaching or exceeding 3 percent of the region's forest land in 3 different years (fig. 34a). However, most of these insect outbreaks were characterized as low to moderate intensity (<50 percent change in canopy cover) (fig. 34b). Harvest rates were relatively consistent at the regional level, affecting approximately 0.25 percent of forest land in each year. Large wildfires occurred in several years after 2000 but impacted a much lower percentage of the forest land than fires in neighboring NFS regions.

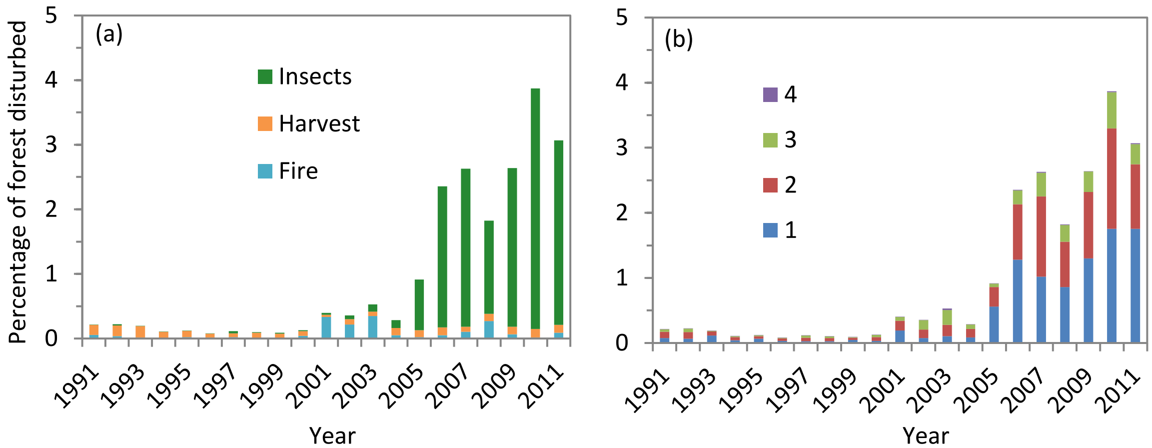

Figure 34a-b. Annual rates of disturbance (0 to 5 percent) in the Rocky Mountain Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

6.4.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to estimate the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Rocky Mountain Region in figure 35. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires first occur in 2002, for example, the line of fire in figure 35 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an "undisturbed" scenario. Thus, figure 35 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

Insect activity accounted for 82 percent of the disturbance impact on 2011 C stocks (fig. 36). The acceleration of insect activity within the region after 2005 (fig. 34) resulted in a surge in insect impact on C stocks (fig. 35). By 2011, insect activity from 1990 to 2011 caused a 5.5-percent reduction in the regional nonsoil C stocks (fig. 37) that were reported in the NFS baseline C assessment derived from FIA data (USDA FS 2010g). From the beginning of the study period (1990) to 2000, harvest was the disturbance process with the highest impact on C stocks (fig. 35), although that impact was significantly lower than in other regions, such as the Southern Region, where harvest activity was more common. Harvests in the region removed approximately 0.4 percent of nonsoil C stocks by 2011 (fig. 37). These patterns do not account for offsite storage of C in wood products. Information about the region's product C storage can be found in the NFS baseline C assessment (USDA FS 2015g).

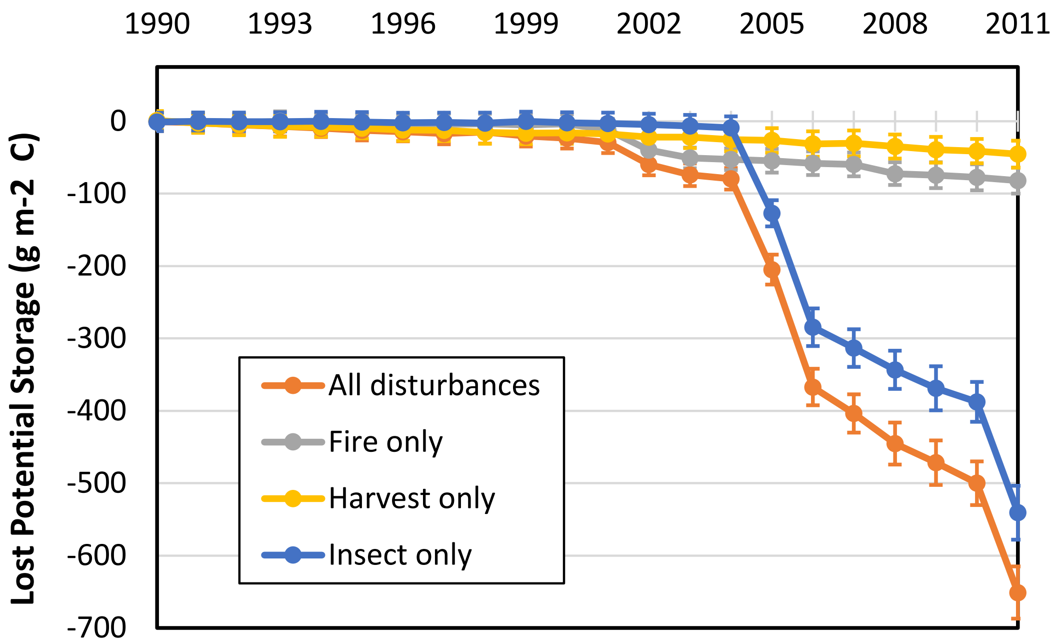

Figure 35. The impact of different kinds of disturbance, occurring from 1990 through 2011, on carbon (C) stores in the Rocky Mountain Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

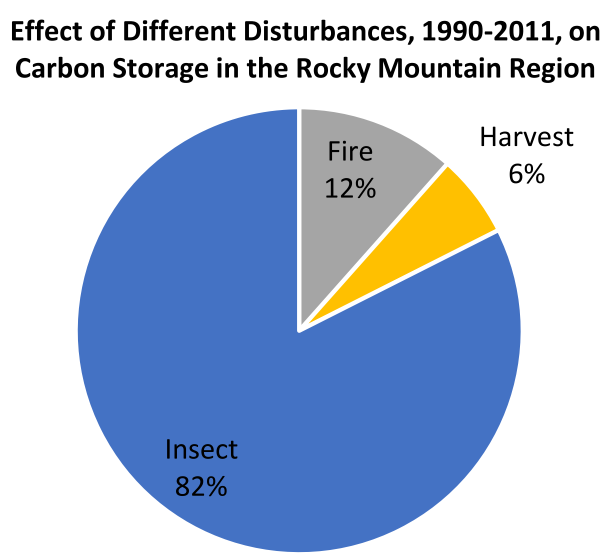

Figure 36. Proportional effect of different kinds of disturbance on carbon storage in all national forests in the Rocky Mountain Region for the period 1990 through 2011.

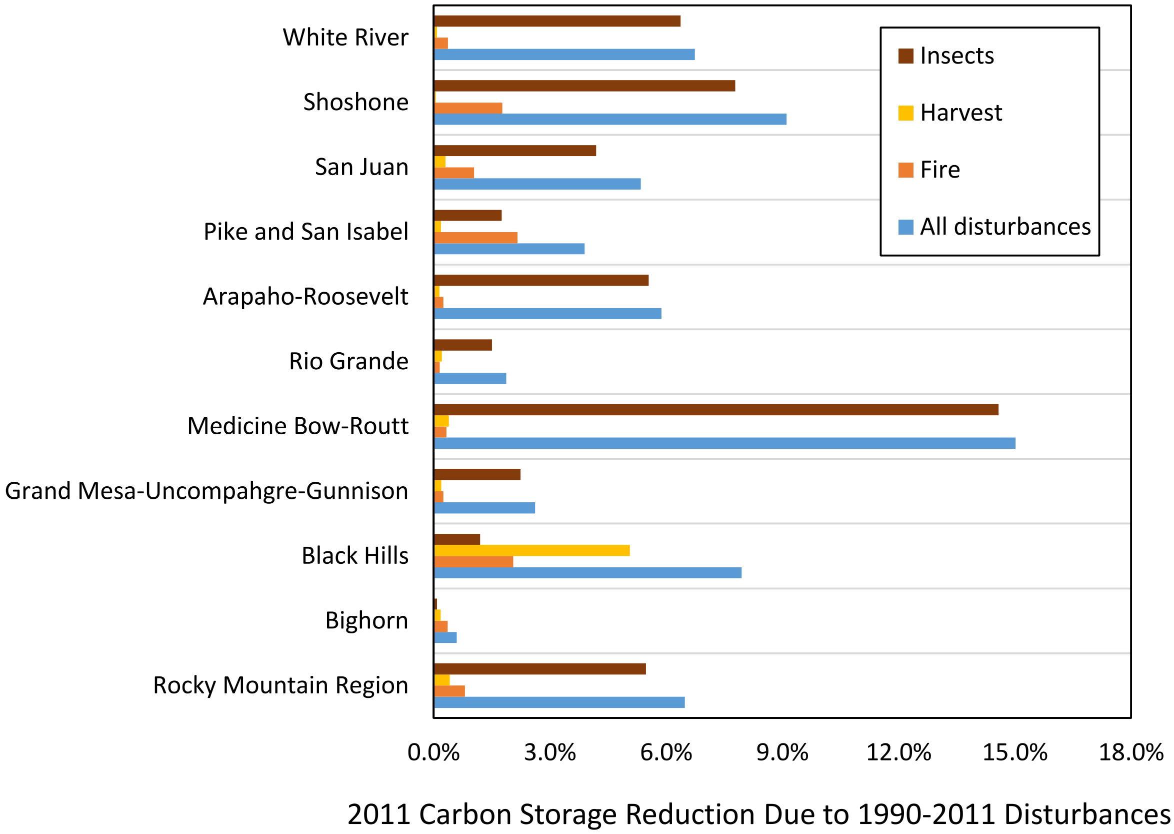

Figure 37. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by each national forest and for all national forests combined in the Rocky Mountain Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

After an increase in fire activity in 2000, fire was the dominant disturbance process affecting C storage until the mountain pine beetle started killing large areas of trees in 2005 (fig. 34). Fires resulted in the loss of about 0.8 percent of nonsoil C stocks by 2011 (fig. 37). Because ForCaMF accounts for emissions of C in a biologically realistic way (using dynamics built into FVS), only a fraction of the full C effects of the beetle outbreak is represented by 2011; most C in beetle-killed trees is still stored in dead pools and will be released over several decades.

One caveat must be stated about the confidence intervals depicted in figure 35. The unit of analysis for ForCaMF was the national forest (Appendix 6: Rocky Mountain, (PDF, 2.7MB)); rates of simulated error were constrained by FIA estimates at that level. Regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in other forests. This assumption very likely overstated certainty at the regional level; if initial biomass estimates in one forest were high, for example, the same estimates were also likely to be biased high in other units. This caveat does not change the overall trends seen at the aggregated regional level, nor does it affect confidence intervals assessed at the national forest level. Insect activity was the dominant disturbance affecting C storage; although large amounts of insect mortality were observed only after 2005, the C impact of those outbreaks quickly overwhelmed effects of harvest and fire.

6.4.4 Effects of Disturbance, Management, and Environmental Factors

This section provides a regional summary of the predicted (modeled) effects of natural disturbances, land management, and environmental factors on forest C dynamics depicted in a series of figures (figs. 38–41) generated from InTEC model inputs and simulations for individual national forests and summed results across all national forests in the Rocky Mountain Region. These regional-scale outputs were generated only from the national forest-specific datasets; thus, these outputs do not represent lands within the region that are outside of the national forest boundaries. The Appendix 6: Rocky Mountain, (PDF, 2.7MB) shows the effects of disturbance and nondisturbance factors on forest C dynamics within each of the national forests as modeled by InTEC using this same series of figures. The forested area of the Nebraska National Forest was too small to reliably model C dynamics with InTEC, so it was excluded from these analyses.

According to the historical data and model results, forest C trends across the national forests in the Rocky Mountain Region have been strongly influenced by the history of land use and policies as well as climate change and natural disturbances. Despite variation among these national forests, the regionwide stand-age distribution in 2011 shows that most stands in the Rocky Mountain Region are older (>80 years old) with a distinctive pulse of stands which established about 80 to 129 years earlier, or from approximately 1880 to 1930 (fig. 38). During the 1930s and thereafter, the rate of stand establishment greatly declined. This pulse of stand establishment between the late 1800s and early 1900s reflects recovery after a range of disturbances, such as timber harvesting and grazing associated with settlement or the last major fires in the region before fire suppression. Fire suppression, which began in the early 1900s, would have allowed more of these young, regenerating stands to survive and continue regrowing rather than being disturbed at a more typical historical rate of fires (Pyne 1982). Depending on the forest type, this pulse of establishment would have reached maximum rates of productivity between 30 and 60 years of age (fig. 39), or throughout the mid- to late 20th century. Forests in the Rocky Mountain Region show another pulse of young stands (<10 years old) (fig. 38) that were established in the early 2000s and represent regeneration after recent disturbances—mostly extensive insect outbreaks, but also large and intense wildfires (fig. 34).

Though there is significant interannual variability in both temperature and precipitation, between 1951 and 2011 climate has on average become warmer and slightly wetter in the Rocky Mountain Region. Although the region has become slightly wetter, severe droughts like the one in the early 2000s are not uncommon. Warmer temperatures and drought conditions can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity. Consequently, more C is released to the atmosphere. As climate continues to change, this region is expected to get even warmer and the vital winter snowfalls are more likely to come as rain in the future. In addition to climate change and variability, both atmospheric CO_2 concentrations and N deposition have increased over the past few decades (fig. 40c) as a result of human activities, and these two factors are known to increase forest productivity in most circumstances.

The C stock changes across the national forests in the Rocky Mountain Region show that together the forests generally underwent a switch from a C sink to a C source (figs. 41a–c) in the late 1980s and early 1990s, causing a decline in accumulated C (fig. 41d). Disturbance and aging effects have been mostly responsible for declining C stocks (fig. 41b), which correspond

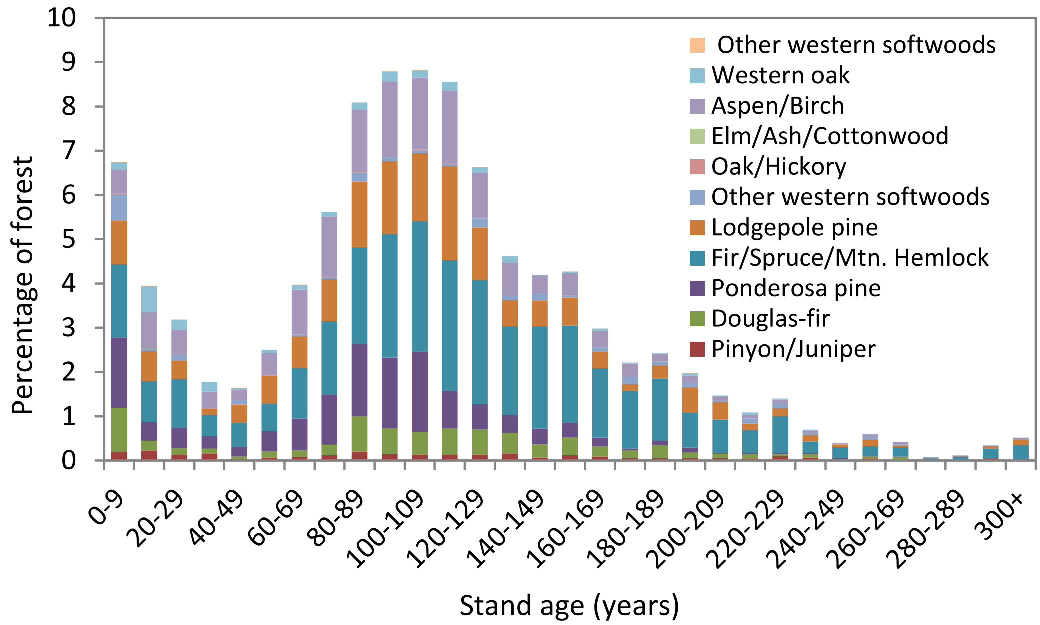

Figure 38. Age-class distribution in 2011 shown as the percentage of forest land in each forest-type group in 10-year age classes summed across the national forests in the Rocky Mountain Region (see Appendix 1 for scientific names of trees).

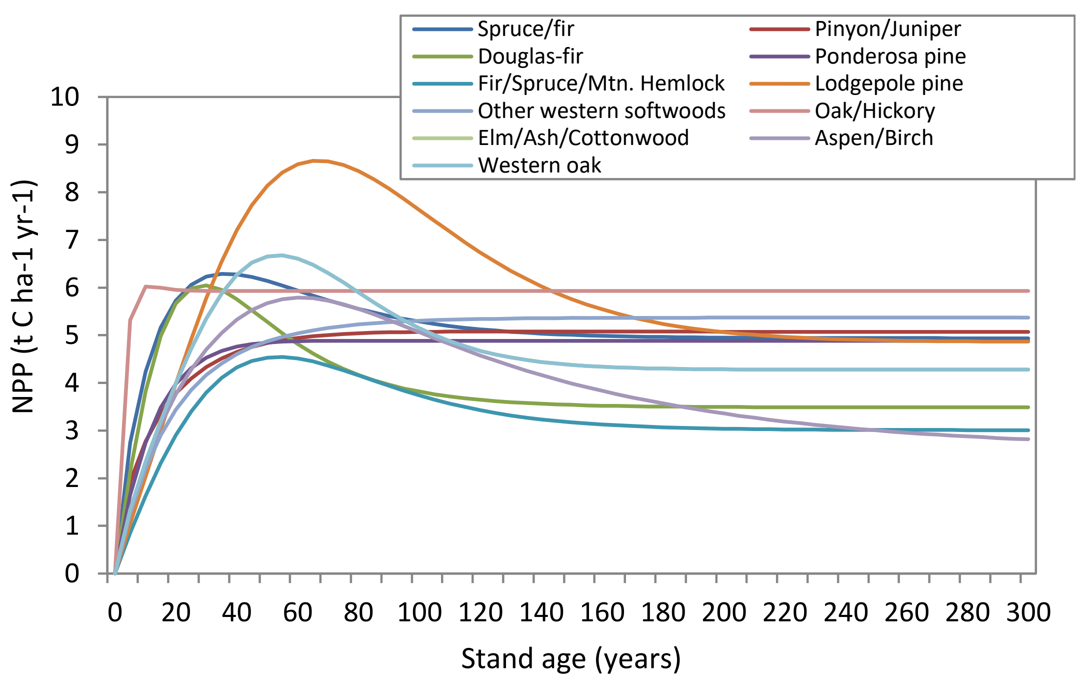

Figure 39. Relationship between net primary productivity (NPP) and stand age for each forest-type group averaged across all national forests in the Rocky Mountain Region (see Appendix 1 for scientific names of trees). Due to a small sample of other western hardwood plots to derive NPP-age curves, the western oak curve was used for these stands.

to disturbances that shaped forest structure (fig. 38), the NPP-stand age relationships (fig. 39), and increasing disturbances in recent years (fig. 34). From 1950 to the 1970s, the forests were a C sink due to disturbance and aging effects, as the 1880-to-1930 pulse of stands that reflect regrowth and recovery after major disturbances was middle-aged and therefore growing at maximum rates of productivity. As these forests further aged, productivity declined, causing the rate of C accumulation to decline (fig. 41d). Meanwhile the effects of disturbance were increasing to the point where C emissions due to decomposition and disturbances exceeded C gains, and as a result the forests became a C source (figs. 41b,c). This decline was coupled with lower rates of stand establishment in the mid-1900s.

Although recent disturbances initially caused increases in C emissions during the year of the disturbance events (fig. 41e), they also promoted regrowth and recovery, as indicated by the recent pulse of stands less than 10 years (fig. 38). As these young stands recover and reach middle-age in the coming decades, they will be growing at maximum productivity (fig. 39). Thus, forests have the potential to accumulate more C and become C sinks again.

As national forests in the Rocky Mountain Region shifted to a C source, nondisturbance effects had a positive effect, thus helping to maintain the C sink for a few more decades (figs. 41b,c). Carbon dioxide levels and N deposition have had a significant positive effect on forest C, perhaps because these forests are normally more nutrient limited. However, by the early 1990s, the C gains from CO_2 concentrations and N deposition were surpassed by the C losses as negative disturbance and aging effects caused the forests to shift to a C sink (fig. 41d). Atmospheric CO_2 concentrations are expected to continue increasing for the foreseeable future, potentially counteracting a portion of the C declines due to disturbance and aging. Despite climate variability and the recent warming trends, climate has not had a significant effect on C stocks compared to other factors (fig. 41d). In the 1950s and early 1960s, climate produced a C source (fig. 41a) due to a period of above-average temperatures (fig. 40b). Thus, projected warming could result in a C source in the future.

Although a few national forests in the Rocky Mountain Region had a net gain of total forest C, most forests had a loss from 1950 to 2011, resulting in a regionwide loss of approximately 15 Tg of total ecosystem C, including soil C, between 1950 and 2011 (fig. 41d).

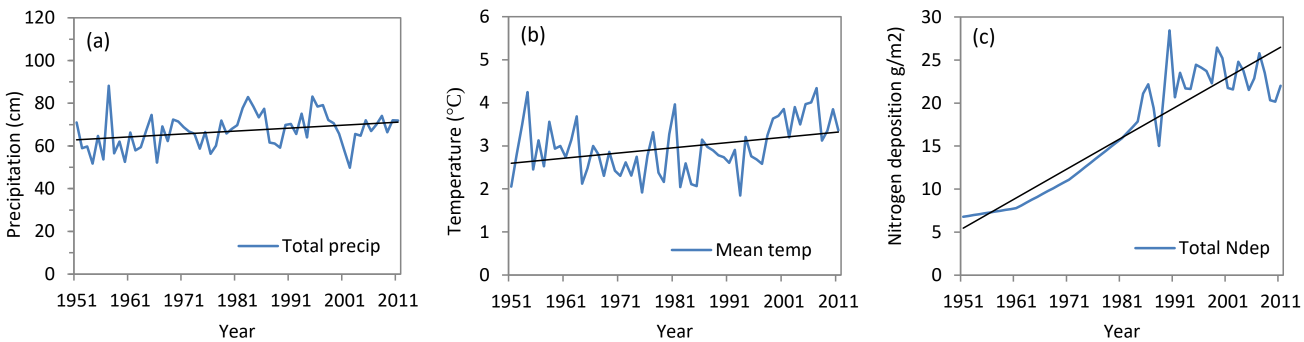

Figure 40a-c. (a) Total annual precipitation, (b) average annual temperature, and (c) total annual nitrogen deposition from 1951 through 2011 averaged across all national forests in the Rocky Mountain Region. Linear trend lines shown in black.

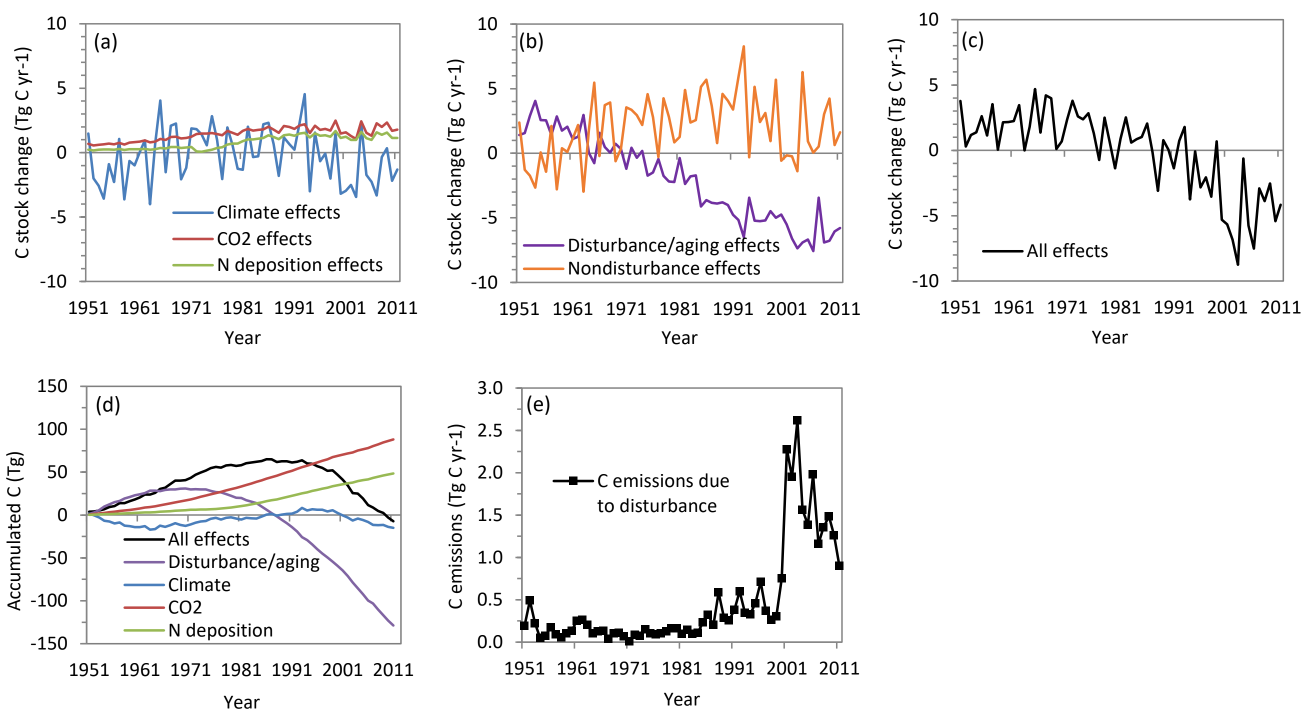

Figure 41a-e. Estimated forest carbon (C) changes and accumulations summed across the national forests in the Rocky Mountain Region. Changes in C stocks are attributed to: (a) individual nondisturbance factors alone, including climate variability, atmospheric carbon dioxide (CO_2) concentration, and nitrogen deposition; (b) all disturbance and aging factors including fire, harvest, insects, and regrowth and aging, and the sum of all nondisturbance factors combined; and (c) all factors combined, which is the sum of disturbance and aging and nondisturbance effects. (d) Accumulated C due to individual disturbance and aging and nondisturbance factors and all factors combined from 1950 through 2011 excluding C accumulated pre-1950. (e) Direct C emissions to the atmosphere due to disturbance events only. Positive values in graphs (a) through (c) represent net C sinks from the atmosphere, or increases in C stocks, whereas negative values represent net C sources to the atmosphere, or decreases in C stocks.

6.5 Intermountain Region

6.5.1 Description of Region

The Intermountain Region (also referred to as Region 4) in the National Forest System (NFS) consists of 12 national forests spread across Nevada, Utah, western Wyoming, and southern and central Idaho (fig. 42). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these national forests, so the Curlew National Grassland in southern Idaho was not included. To restrict the analysis to lands managed by the Forest Service, any private inholdings within national forest boundaries were excluded.

Euro-American settlers came to the Intermountain Region in the mid-1800s mostly for fur trapping. Mormon settlers followed, bringing cattle and oxen grazing to the region (see Appendix 1 for scientific names of species mentioned in this report). The real boom in settlement began in the 1890s, when Euro-Americans came to the region for mining, livestock grazing, and timber. Lumber operations using destructive methods became common

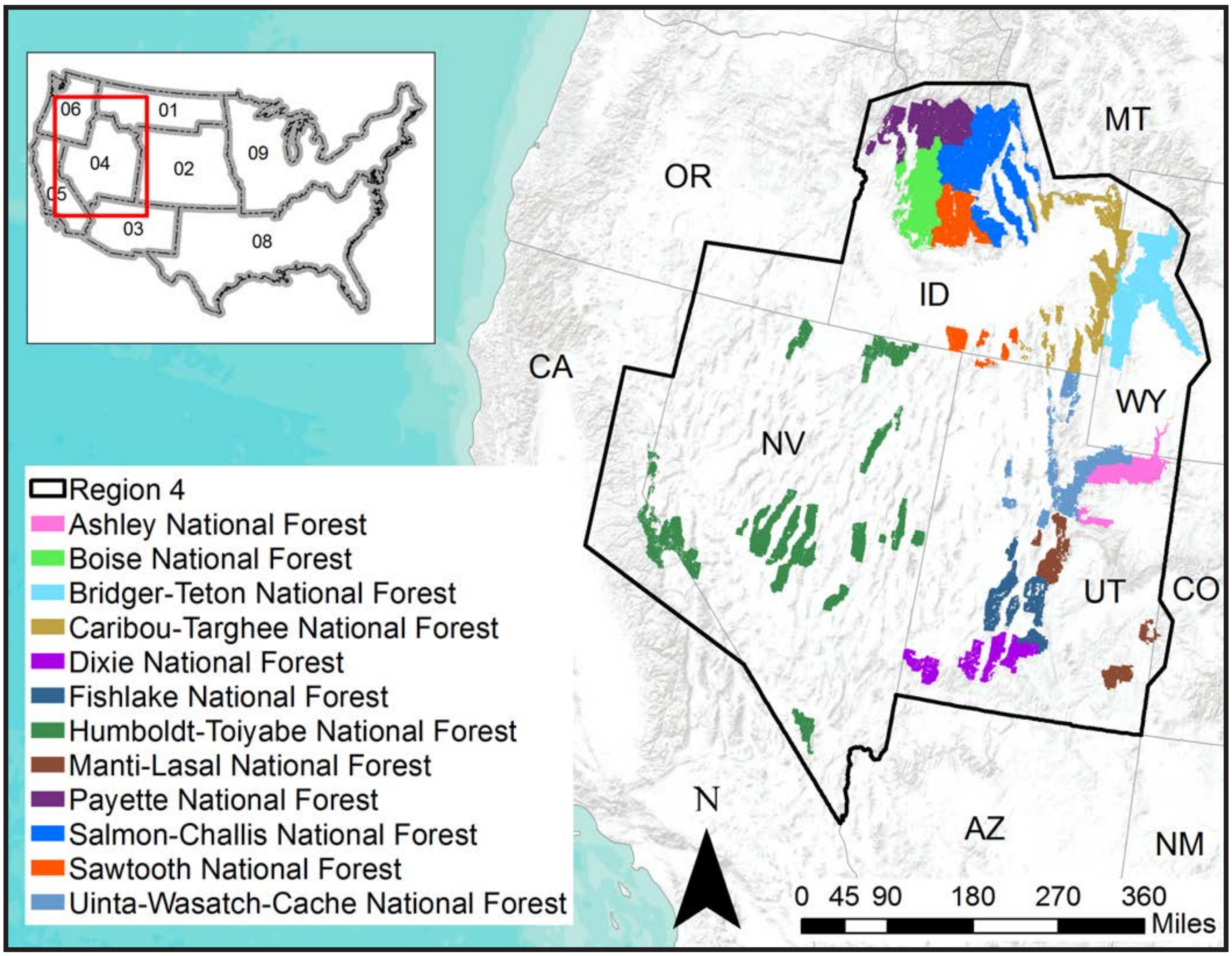

Figure 42. Locations of the national forests in the Intermountain Region.

practice, leaving forests as devastated fire hazards covered in slash, litter, and wasted wood. In the 1890s forests were set aside as forest reserves to protect valuable timber and watersheds from these destructive practices. The land was eventually put under Forest Service control in 1905 and more sustainable timber practices became an objective (USDA FS 1930). Annual timber output was low in the early 1900s, then increased after the Great Depression in the 1930s and peaked in 1972; it declined precipitously in the early 1990s and has remained low ever since (Stockmann et al. 2014a). In addition to timber harvesting, the forest C legacy of the Intermountain Region is tied to its history of fires and fire management. Historically fire was a major component of the disturbance regimes in the Intermountain forests prior to Euro-American settlement. Devastating fire seasons of 1889, 1910, and 1919 burned millions of acres of valuable forest and timberland in this region and across much of the West. Congress began appropriating funds to support fire suppression activities including improvements to communication and transportation facilities, lookout towers, and firefighting tools led by the Forest Service (Barrett et al. 1997). This national fire suppression policy entailed intensive efforts to prevent, detect, and suppress all wildfires, which greatly reduced the area burned and the frequency of fires and consequently altered the ecological system. A policy shift from fire control to fire management in the 1970s allowed natural fires to burn across landscapes where considered safe and appropriate, in an attempt to restore historical fire regimes (Pyne 1982).

Recent changes in climate have also played a role in forest disturbance regimes and C dynamics. The Intermountain Region has experienced an increase in average annual temperature while precipitation has been highly variable. Land use history and climate change have led to large and intense disturbances in the region over the past few decades. Extended fire seasons and large wildfires that burn for longer durations, such as the Clear Creek Complex Fire of 2000 and the Murphy Complex Fire of 2007, both in Idaho, have increased in the Intermountain Region. Much of this change is due to warmer summer temperatures and earlier spring snowmelt, which are projected to continue into the future (Westerling et al. 2006). Over the past few decades, severe bark beetle outbreaks have resulted in tree mortality in parts of the Intermountain Region (Bentz et al. 2010).

6.5.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Intermountain Region are displayed in figure 43. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. For most of the study period (1990 to 2011), regional disturbance patterns were dominated by fire, although the forest area burned in any one year never significantly exceeded 1 percent of the total (fig. 43). Though low levels of insect mortality were observed in several years in the 1990s, insect impacts greatly increased after approximately 2003. In 2011, about 2.5 percent of the region's forest land was affected by insect disturbance. By comparison, insect disturbance (as indicated by newly observed mortality) affected 2.5 percent or more of forest land in the neighboring Rocky Mountain Region in 3 different years during the same period (see Section 6.4.2). Harvest rates have been low throughout the study period, although there was slightly more harvest activity in the early

1990s. Over this period, about 70 percent of the area disturbed was affected by low- to moderate-intensity disturbances (<50 percent change in canopy cover).

6.5.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to understand the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Intermountain Region in figure 44. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires occur only in 1992, for example, the line for fire in figure 44 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an "undisturbed" scenario. Thus, figure 44 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

Throughout most of the study period, fire was the disturbance process that most affected C storage in the region (fig. 45). Larger fire years (fig. 43) corresponded to increased fire impacts on C storage (fig. 44). By 2011, fires from 1990 through 2011 resulted in a reduction of 1.9 percent of the regional nonsoil C stocks (fig. 46) that were reported in the NFS baseline C assessment derived from FIA data (USDA FS 2015c). However, increased insect activity in the latter part of the period, particularly after 2007, substantially altered the regional disturbance signature upon C stocks. By 2011, insect activity made up 47 percent of the disturbance impact on C storage (fig. 45), and caused a 1.8-percent reduction in nonsoil C storage (fig. 46). Because ForCaMF accounts for emissions of C in a biologically realistic way (using dynamics built into FVS), only a fraction of the full C effects of the insect and fire activity

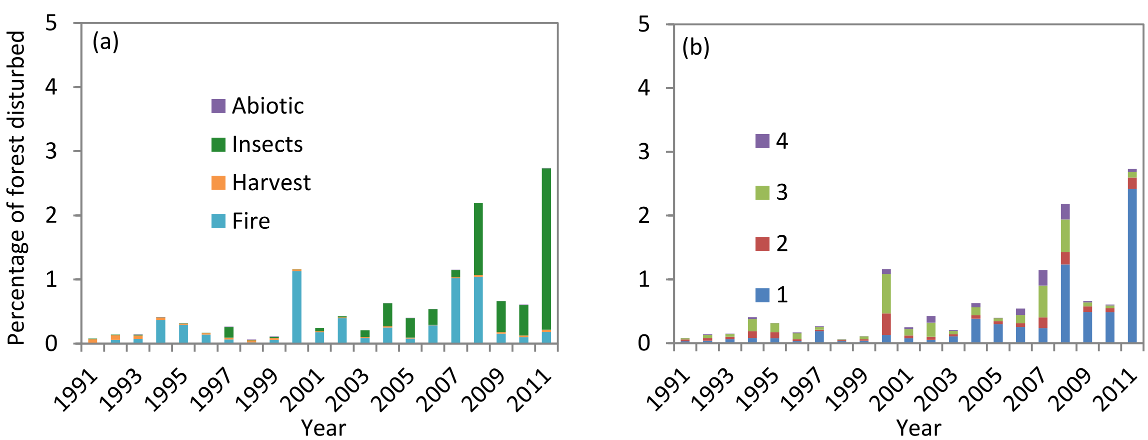

Figure 43. Annual rates of disturbance (0 to 5 percent) in the Intermountain Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

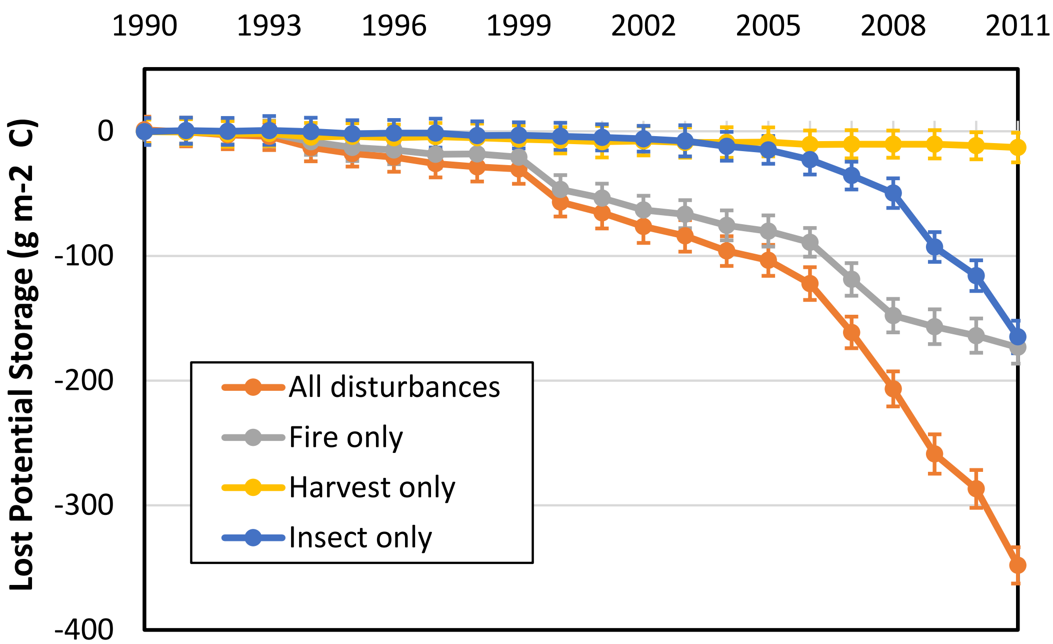

Figure 44. The impact of different kinds of disturbance, occurring from 1990 through 2011, on carbon (C) stores in the Intermountain Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

observed from 1990 through 2011 are represented by 2011; most C in fire- or beetle-killed trees is still stored in dead pools and will be released over several decades.

In interpreting the steady, but small, C impact of harvest throughout the period, it is important to reiterate that these patterns do not account for offsite storage of C in wood products. Information about the region's product C storage can be found in the baseline C assessment (USDA FS 2015c). One caveat must be stated about the confidence intervals depicted in figure 44. The unit of analysis for ForCaMF was the national forest; rates of simulated error were constrained by FIA estimates at that level. Regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in other forests. This assumption very likely overstated certainty at the regional level; if initial biomass estimates in one forest were high, for example, the same estimates were also likely to be biased high in other units. This caveat does not change the overall trends seen at the aggregated regional level, nor does it affect confidence intervals assessed at the national forest level. Insect activity has recently joined fire as an important factor in the amount of C stored by national forests in the Intermountain Region.

6.5.4 Effects of Disturbance, Management, and Environmental Factors

This section provides a regional summary of the predicted (modeled) effects of natural disturbances, land management, and environmental factors on forest C dynamics depicted in a series of figures (figs. 47–50) generated from InTEC model inputs and simulations for individual national forests and summed results across all national forests in the Intermountain Region. These regional-scale outputs were generated only from the national forest-specific datasets; thus, these outputs do not represent lands within the region that are outside of the national forest boundaries. The Appendix 7: Intermountain Region (PDF, 3.1MB)shows the effects of disturbance and nondisturbance factors on forest C dynamics within each of the 12 national forests as modeled by InTEC using this same series of figures.

According to the historical data and model results, forest C trends across the national forests in the Intermountain Region have been strongly influenced by the history of land use and policies as well as natural disturbances. Despite variation among these national forests, the regionwide stand-age distribution in 2011 shows that approximately 50 percent of the forests are greater than 100 years old (fig. 47). The stand-age distribution also shows a pulse of stands that established about 80 to 119 years before this study, or from approximately 1890 to 1930 (fig. 47). Thereafter, the rate of stand establishment steadily declined until 1982, when it increased once again. This early-1900s pulse of stand establishment reflects recovery after a range of disturbances occurring around the time of Euro-American settlement, such as intensive timber harvesting and large wildfires. Although some fires continued into the mid-1900s, fire suppression, which began in the early 1900s, allowed more of these young, regenerating stands to survive and continue regrowing rather than being disturbed at a more typical historical rate of fires (Pyne 1982). Depending on the forest type, which is mostly Douglas-fir, lodgepole pine, or fir/spruce/mountain hemlock, the numerous stands that were established between 1890 and 1930 reached maximum rates of productivity between 30 and 60 years of age (fig. 48). Therefore, for most of the mid-1900s, more than 25 percent of the forest area was highly productive. Recent fires have had extensive high-severity, stand-replacing components, while insect outbreaks have been widespread and unrelenting (fig. 43). These recent disturbances promoted another pulse of stands that were established over the last two decades. These stands will soon be growing at maximum productivity, potentially accumulating more C (fig. 48).

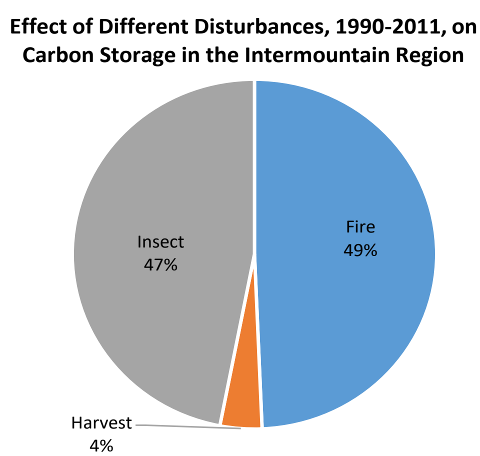

Figure 45. Proportional effect of different kinds of disturbance on carbon storage in all national forests in the Intermountain Region for the period 1990 through 2011.

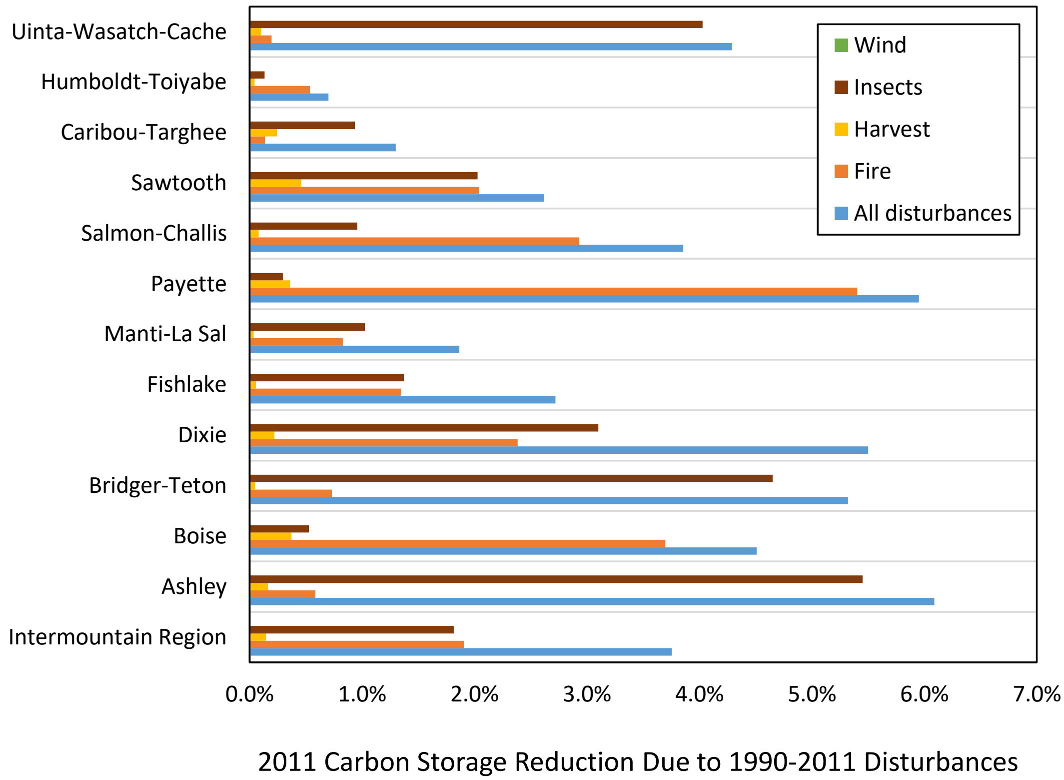

Figure 46. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by each national forest and for all national forests combined in the Intermountain Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

Though there is significant interannual variability in both temperature and precipitation, between 1951 and 2011 climate has on average become much warmer (figs. 49a,b). Precipitation does not show a clear increasing or decreasing trend. However, several droughts such as the one from 1999 through 2002 have been evident through this region. For much of the late 1990s and 2000s, temperatures were consistently higher than average. Warmer temperatures and drought conditions can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity. Consequently, more C is released to the atmosphere. As climate continues to change, this region is expected to get even warmer, with greater variability in precipitation, and snowfall shifting to rainfall. In addition to climate change and variability, both atmospheric CO_2 concentrations and N deposition (fig. 49c) have increased over the past few decades as a result of human activities. Nitrogen deposition in this region peaked around 1995 but has since remained elevated (fig. 49c).

Though there is significant interannual variability in both temperature and precipitation, between 1951 and 2011 climate has on average become much warmer (figs. 49a,b). Precipitation does not show a clear increasing or decreasing trend. However, several droughts such as the one from 1999 through 2002 have been evident through this region. For much of the late 1990s and 2000s, temperatures were consistently higher than average. Warmer temperatures and drought conditions can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity. Consequently, more C is released to the atmosphere. As climate continues to change, this region is expected to get even warmer, with greater variability in precipitation, and snowfall shifting to rainfall. In addition to

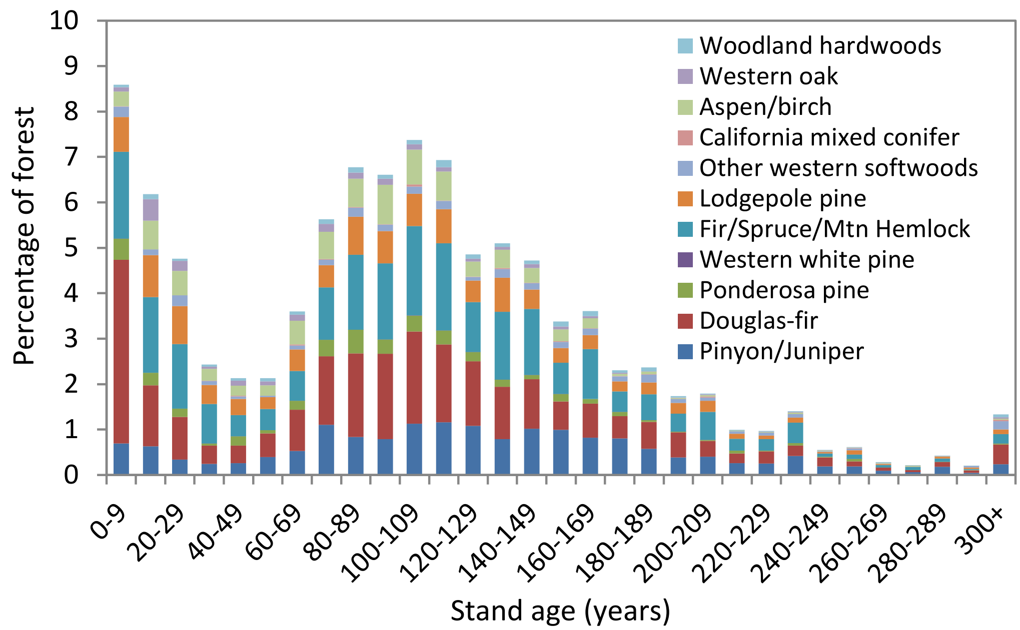

Figure 47. Age-class distribution in 2011 shown as the percentage of forest land in each forest-type group in 10-year age classes summed across the national forests in the Intermountain Region (see Appendix 1 for scientific names of trees).

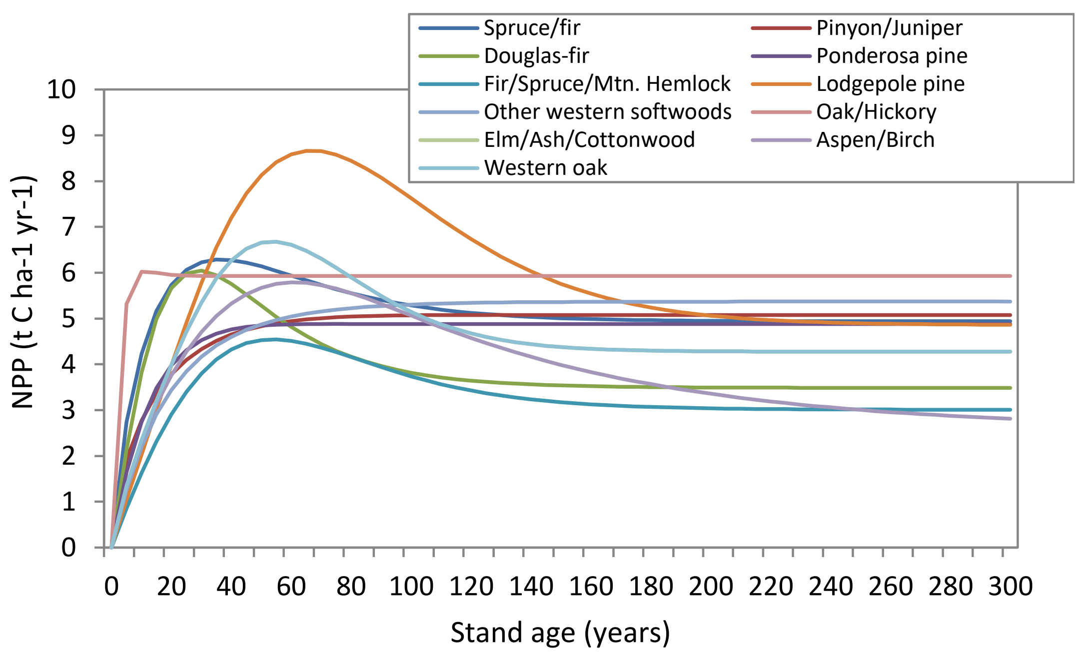

Figure 48. Relationship between net primary productivity (NPP) and stand age for each forest-type group averaged across all national forests in the Intermountain Region (see Appendix 1 for scientific names of trees). Due to small sample sizes, the NPP-age curve for the other western softwoods forest-type group was applied to California mixed conifer stands, and the curve for the ponderosa pine forest-type group was applied to western white pine stands.

climate change and variability, both atmospheric CO_2 concentrations and N deposition (fig. 49c) have increased over the past few decades as a result of human activities. Nitrogen deposition in this region peaked around 1995 but has since remained elevated (fig. 49c).

The C stock changes across the national forests in the Intermountain Region show that together the forests underwent a switch from a C sink to a C source (figs. 50a–c) in the late 1980s to early 1990s. Disturbance and aging effects have been mostly responsible for declining C stocks (fig. 50b), which correspond to disturbances that shaped forest structure (fig. 47), the NPP-stand age relationships (fig. 48), and increasing disturbances in recent years (fig. 43). From 1950 to the early 1980s, the forests were a C sink as forests representing regrowth from previous disturbances were at productive ages. For instance, the 1890 to 1930 pulse of stands was middle-aged and therefore growing at maximum rates of productivity. As these forests further aged, productivity declined, causing the rate of C accumulation to decline (fig. 50d). Meanwhile the effects of disturbance were increasing to the point where C emissions from decomposition and disturbance events exceeded C gains, and the forests consequently became a C source (figs. 50b,c). This decline was coupled with lower rates of stand establishment and increasing rates of timber harvesting after the 1930s.

Recent disturbances such as the large, moderate- to high-severity fires in 2000, 2007, and 2009 and an increase in insect disturbances during the 2000s caused spikes in C emissions (fig. 50e). These disturbances coincided with a drought and high temperatures, which together caused forests to remain a C source from 1996 through 2011 (figs. 50a–c). As forests continue to recover from these disturbance events, as seen in the regional stand-age distribution (fig. 47), and reach middle-age with the highest rates of productivity (fig. 48), regional forests have the potential to become a C sink again in a few decades.

Though disturbance and aging effects initially caused a C sink and then a C source, nondisturbance factors had a mostly positive effect on C stocks in the 1950s and thereafter (figs. 50a,b). Increasing levels of CO_2 and N deposition caused forests to accumulate more C and helped to counteract the C declines due to disturbances and the aging stands. However, by the late 1980s the C gains from CO_2 and N deposition were surpassed by the C losses as negative disturbance and aging effects caused the forests to shift to a C source (fig. 50d). Atmospheric CO_2 concentrations are expected to continue increasing for the foreseeable future, potentially counteracting the C declines from disturbances and aging. Despite increasing temperatures and occasional droughts, climate has had a small effect on total C accumulation for most of the past few decades. Since the early 2000s, however, climate effects have been mostly negative, causing a loss of forest C (fig. 50d). Future projected warming and shifts in precipitation may cause the C source in the Intermountain Region to persist.

Although a few national forests in the Intermountain Region had a net gain of total forest C, most forests had a loss from 1950 to 2011, resulting in a regionwide loss of approximately 89.4 Tg of total ecosystem C, including soil C, between 1950 and 2011 (fig. 50d).

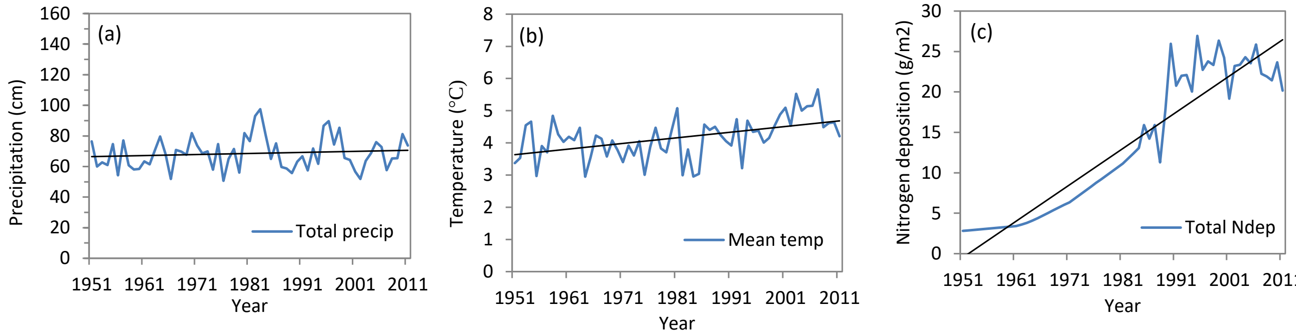

Figure 49a-b. (a) Total annual precipitation, (b) average annual temperature, and (c) total annual nitrogen deposition from 1951 through 2011 averaged across all national forests in the Intermountain Region. Linear trend lines shown in black.

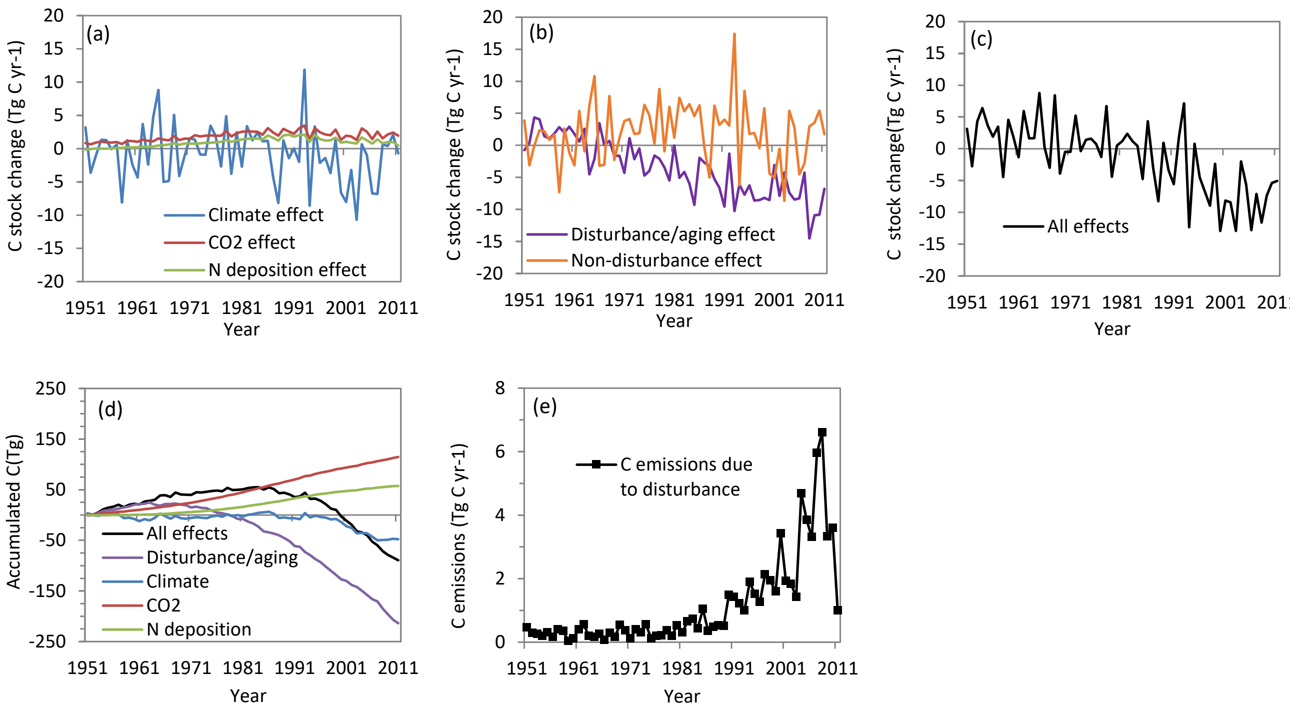

Figure 50a-e. Estimated forest carbon (C) changes and accumulations summed across the national forests in the Intermountain Region. Changes in C stocks are attributed to: (a) individual nondisturbance factors alone, including climate variability, atmospheric carbon dioxide (CO2) concentration, and nitrogen deposition; (b) all disturbance and aging factors including fire, harvest, insects, and regrowth and aging, and the sum of all nondisturbance factors combined; and (c) all factors combined, which is the sum of disturbance and aging and nondisturbance effects. (d) Accumulated C due to individual disturbance and aging and nondisturbance factors and all factors combined from 1950 through 2011 excluding C accumulated pre-1950. (e) Direct C emissions to the atmosphere due to disturbance events only. Positive values in graphs (a) through (c) represent net C sinks from the atmosphere, or increases in C stocks, whereas negative values represent net C sources to the atmosphere, or decreases in C stocks.

6.6 Pacific Northwest Region

6.6.1 Description of Region

The Pacific Northwest Region (also referred to as Region 6) in the National Forest System (NFS) consists of 16 national forests and a national scenic area spread across Washington and Oregon (fig. 51). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these administrative units, so national grasslands such as the Crooked River National Grassland in central Oregon were not included. To restrict the analysis to lands managed by the Forest Service, any private inholdings within national forest boundaries were excluded.

The history of Euro-American settlement, land use, and policies provides useful context for understanding the regional forest C trends that are indicated by the inventory and assessment results. The first Euro-American settlers came to the Pacific Northwest Region in the late 1860s after the Civil War, treaties with Native Americans, and the discovery of gold (Bach 1990). The region was largely uninhabited, however, until the arrival of the transcontinental railroad lines during the 1880s. The railroad helped transform the small-scale timber operations that had

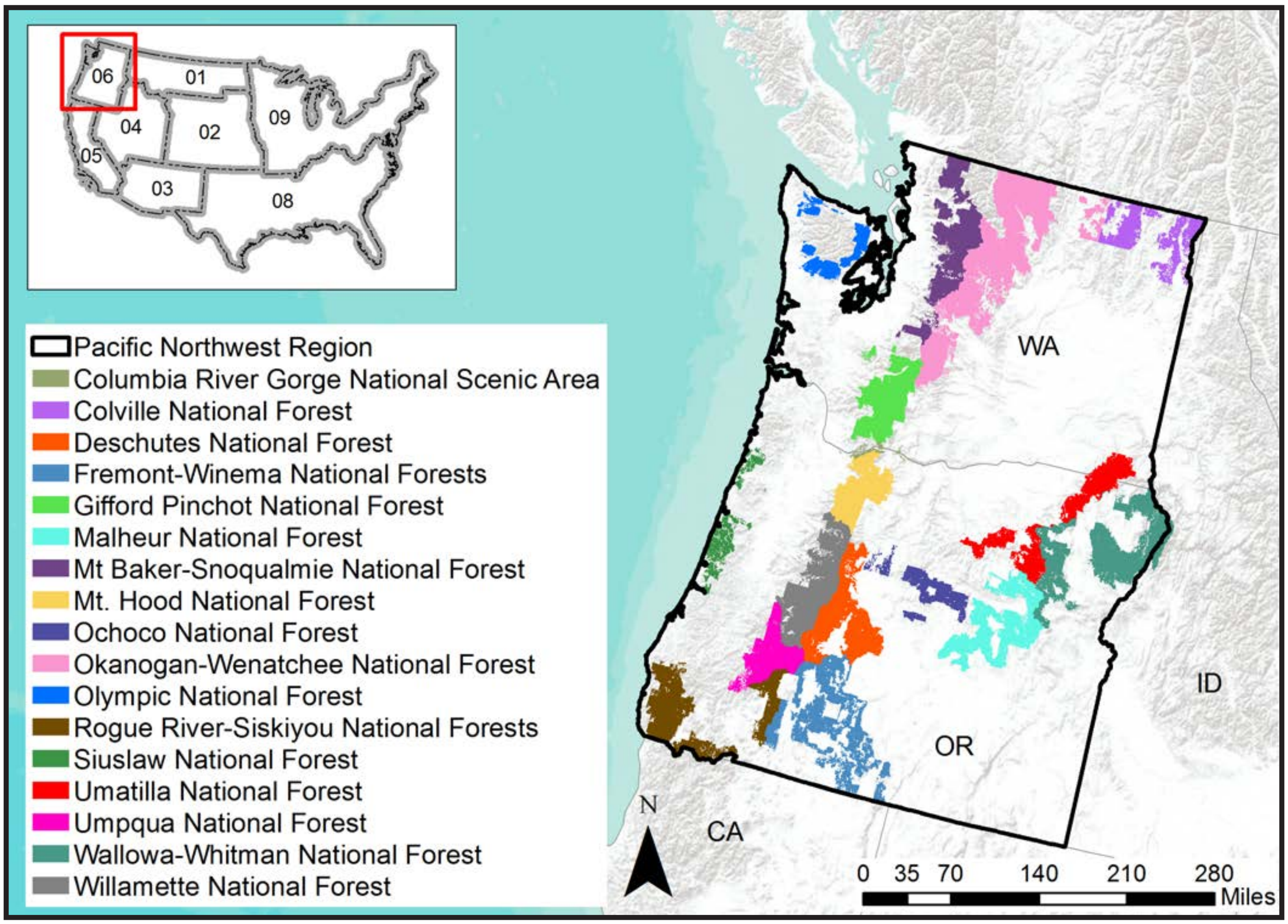

Figure 51. Locations of the national forests and national scenic area in the Pacific Northwest Region.

started in the 1870s (Bach 1990) into one of the most productive and rapidly growing industries in the Pacific Northwest. By 1905, the State of Washington ranked first in the Nation in timber production. In these early settlement years, forests were recklessly clear cut and fires were often accidentally ignited, burning valuable timberland. Annual timber output steadily increased starting in the 1940s, peaked in 1973, but declined precipitously in the early 1990s and has remained low ever since (Butler et al. 2014a). This recent decline in timber harvesting was largely the result of the adoption of the Northwest Forest Plan in 1994, which placed greater restrictions on harvesting in order to protect the northern spotted owl and other species dependent on old-growth forest habitat (Thomas et al. 2006) (see Appendix 1 for scientific names of species mentioned in this report).

In addition to timber harvesting, the forest C legacy of the region is tied to its history of fires and fire management. Historically fire was a major component of the disturbance regimes in the Pacific Northwest forests prior to Euro-American settlement (Agee 1993). However, after the devastating 1910 fire season, which burned millions of acres of valuable forest and timberland in the West, Congress began appropriating money for fire suppression activities led by the Forest Service. This national fire suppression policy entailed intensive efforts to prevent, detect, and suppress all wildfires, greatly altering the ecological system (Agee 1993; Pyne 1982). Despite suppression efforts, major logging-induced fires such as the Tillamook Burn of 1933 in Oregon continued to occur across forests in this region (Morris 1936). A policy shift from fire control to fire management in the 1970s allowed natural fires to burn across landscapes where considered safe and appropriate, in an attempt to restore historical fire regimes (Pyne 1982).

Recent changes in climate have also played a role in forest disturbance regimes and C dynamics. The Pacific Northwest has experienced an increase in average annual temperature while precipitation has declined both in the amount of total snowfall and the proportion of precipitation falling as snow. Land use history and climate change have led to large and intense disturbances in the region over the past few decades. Wildfires that are widespread and burn for longer durations and extended fire seasons have increased in the Pacific Northwest. Much of this change is due to warmer summer temperatures and earlier spring snowmelt, which are projected to continue into the future (Climate Impacts Group 2009; Westerling et al. 2006). Over the past few decades, severe bark beetle outbreaks have resulted in tree mortality in parts of the Pacific Northwest (Bentz et al. 2010).

6.6.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Pacific Northwest Region are displayed in figure 52. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. Regional disturbance patterns exhibited effects of both wildfire and harvests. Periodic large fire years occurred throughout the period, with approximately 1.2 percent of the region's forests affected by fire in 2002. Harvest decreased through the middle of the study period, though there were slight increases toward the end of the period in 2011. These rates, which did not reach 0.5 percent of the landscape in any single year, were lower than those in the 1970s and 1980s, before enactment of the Northwest Forest Plan in 1994 (Healey et al. 2008). In the last 2 years of the period, insect activity became an important regional disturbance factor, with insects affecting approximately 0.5 percent of the region's forests in 2011 (fig. 52a). From 1990 through 2011, about 60 percent of disturbances were low to moderate intensity (<50 percent change in canopy cover) (fig 52b).

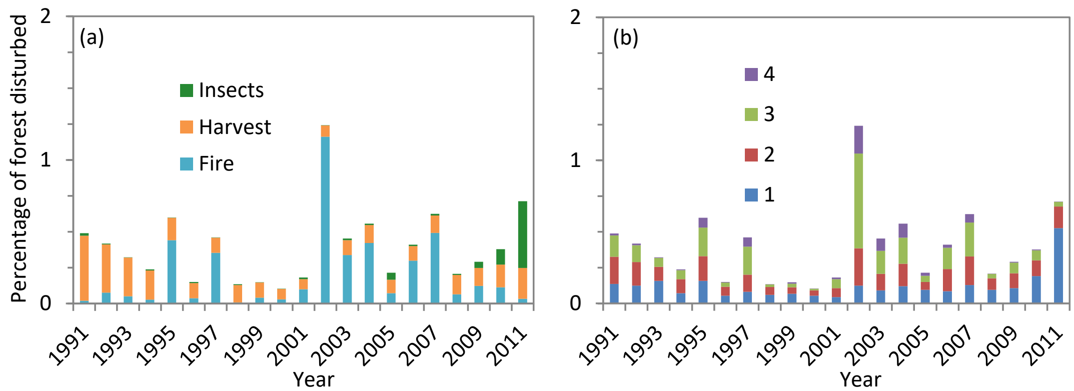

Figure 52a-b. Annual rates of disturbance (0 to 2 percent) in the Pacific Northwest Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

6.6.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to estimate the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Pacific Northwest Region in figure 53. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires first occur in 1992, for example, the line for fire in figure 53 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an "undisturbed" scenario. Thus, figure 53 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

Harvest activity, particularly in the early 1990s, had a significant effect on the region's C stocks; harvests occurring from 1990 to 2011 reduced 2011 C storage by approximately 167 g/m^2 (or 167 Mg/ha) (fig. 53), and was responsible for 36 percent of the total disturbance effect (fig. 54). By 2011, harvesting resulted in a reduction of approximately 0.9 percent of the regional nonsoil C stocks (fig. 55) that were reported in the NFS baseline C assessment (USDA FS 2015e). These patterns do not account for offsite storage of C in wood products. Information about the region's product C storage can be found in the baseline C assessment (USDA FS 2015e).

Fire activity accelerated in 2002 (fig. 53), and fire thereafter became the disturbance factor with the greatest effect on C storage, reducing carbon storage by approximately 1.4 percent by 2011 (fig. 55). By 2011, fires since 1990 had caused 57 percent of the disturbance impact on C stocks, while insect activity had caused 7 percent (fig. 54). Disturbance data (fig. 52) suggest that insects became a significant factor only in 2010 to 2011, and the relative influence of disturbances on C storage (fig. 53) reflects this late emergence of insect activity. Minor wind events were mapped throughout the region, but these events affected relatively small areas and were not considered in this assessment. The unit most affected by wind was the Olympic National Forest, where a total of 0.19 percent of the forest land was affected by wind during the study period, followed by the Gifford Pinchot National Forest (0.10 percent).

One caveat must be stated about the confidence intervals depicted in figure 53. The unit of analysis for ForCaMF was the national forest; rates of simulated error were constrained by FIA estimates at that level. Regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in other forests. This assumption very likely overstated certainty at the regional level; if initial biomass estimates in one forest were high, for example, the same estimates were also likely to be biased high in other units. This caveat does not change the overall trends seen at the aggregated regional level, nor does it affect confidence intervals assessed at the national forest level. Although harvests have impacted the C cycle in this region, fire and, more recently, insect activity have also played important roles.

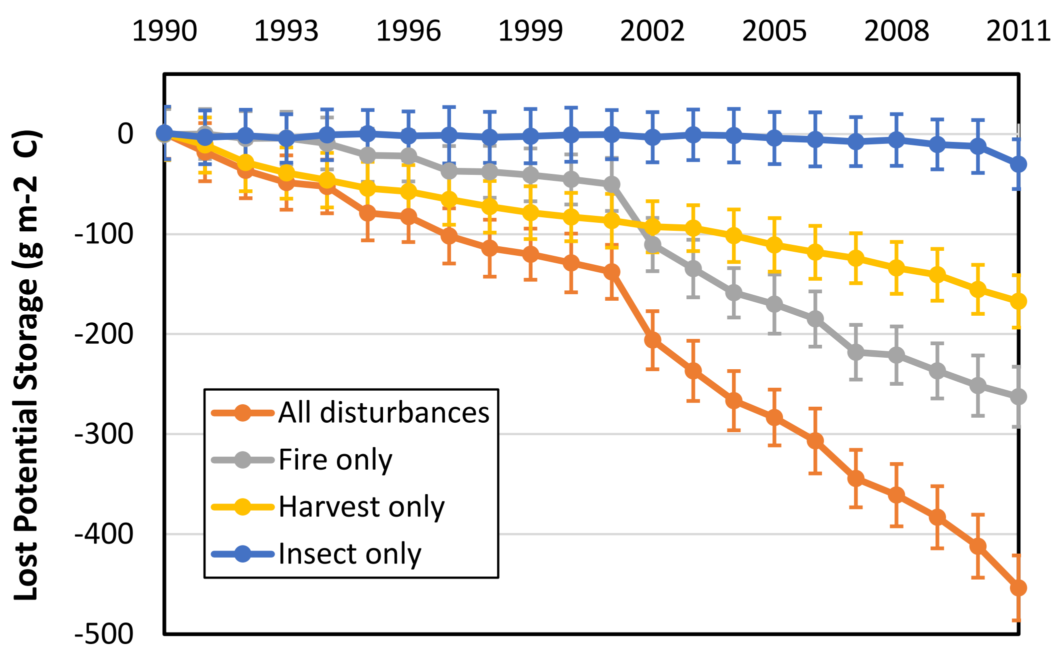

Figure 53. The impact of different kinds of disturbance, occurring from 1990 through 2011, on carbon (C) stores in the Pacific Northwest Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

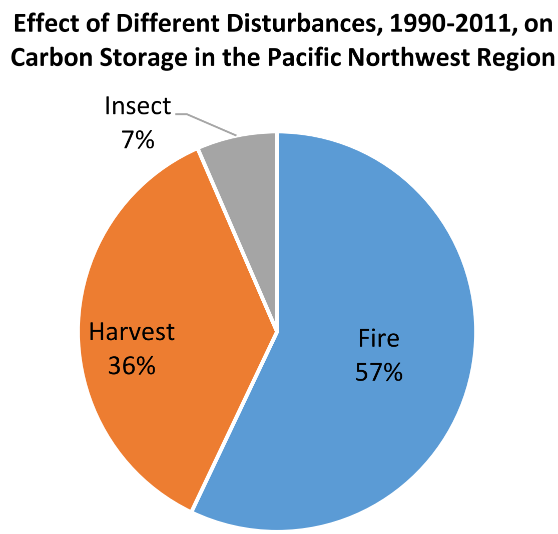

Figure 54. Proportional effect of different kinds of disturbance on carbon storage in all national forests in the Pacific Northwest Region for the period 1990 through 2011.

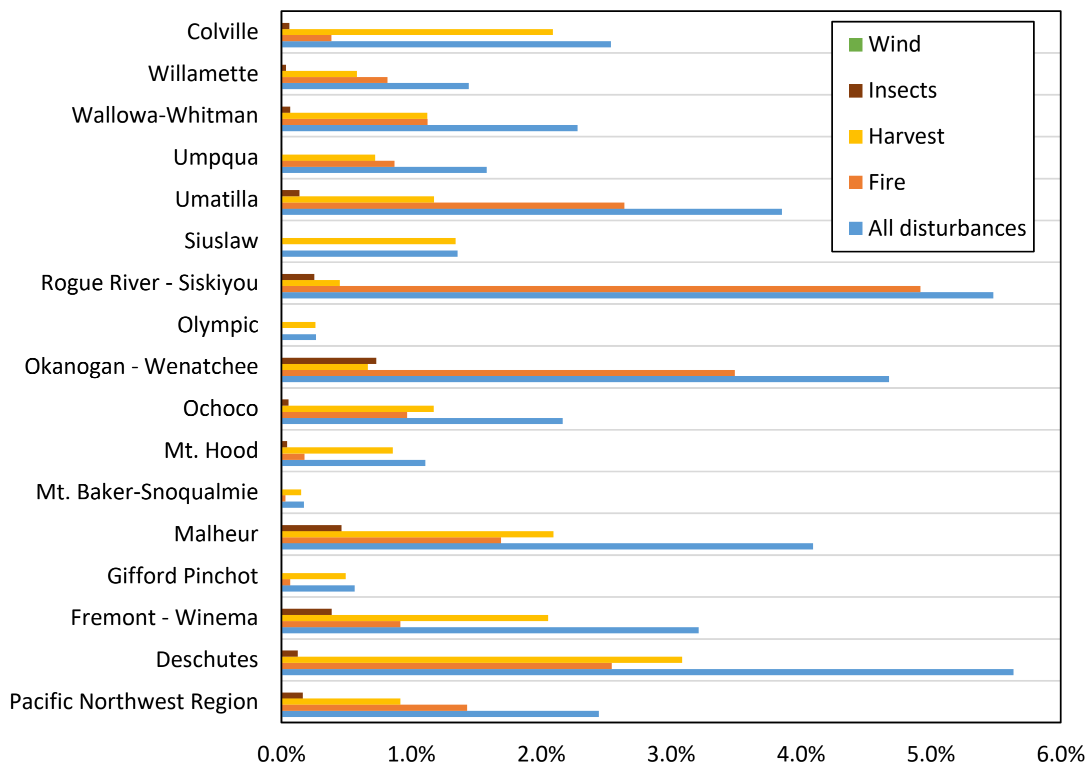

Figure 55. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by each national forest and for all national forests combined in the Pacific Northwest Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

6.6.4 Effects of Disturbance, Management, and Environmental Factors

This section provides a regional summary of the predicted (modeled) effects of natural disturbances, land management, and environmental factors on forest C dynamics depicted in a series of figures (figs. 56–59) generated from InTEC model inputs and simulations for individual national forests and summed results across all the national forests in the Pacific Northwest Region. These regional-scale outputs were generated only from the national forest-specific datasets; thus, these outputs do not represent lands within the region that are outside of the national forest boundaries. The Appendix 8: Pacific Northwest Region (PDF, 4.2MB) shows the effects of disturbance and nondisturbance factors on forest C dynamics within each of the 16 national forests and the national scenic area as modeled by InTEC with this same series of figures.

According to the historical data and model results, forest C trends across the national forests in the Pacific Northwest Region have been strongly influenced by the history of land use and policies as well as climate change and natural disturbances. Despite variation among these national forests, the regionwide stand-age distribution in 2011 shows that approximately 50 percent of the forests are greater than 100 years old, with about 10 percent of the forests more than 250 years old (fig. 56). The stand-age distribution also shows a pulse of stands that established approximately 70 to 109 years before the end of the study period, or between about 1900 and 1940 (fig. 56). During the 1940s and thereafter, the rate of stand establishment steadily declined. This early-1900s pulse of stand establishment reflects recovery after a range of disturbances occurring around the time of Euro-American settlement, such as intensive timber harvesting and large fires. Although some fires continued into the mid-1900s, fire suppression, which began in the early 1900s, allowed more of these young, regenerating stands to survive and continue regrowing rather than being disturbed at a more typical historical rate of fires (Pyne 1982). Depending on the forest type, this pulse of establishment would have reached maximum rates of productivity at 35 to 45 years of age (fig. 57), or throughout the mid- to late 20th century. Recent disturbances, mostly fires and an increased occurrence of insect outbreaks, have been extensive in some forests and included high-severity, stand-replacing components (fig. 52). In several forests with recent large wildfires, stand establishment has increased, reflecting regrowth and recovery.

Though there was significant interannual variability in both temperature and precipitation, between 1951 and 2011 climate has on average become much warmer and somewhat drier in the Pacific Northwest Region (figs. 58a,b). Severe droughts like the one in the early 2000s have occurred periodically (fig. 58b). Warmer temperatures and drought conditions can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity. Consequently, more C is released to the atmosphere. As climate continues to change, this region is expected to get even warmer and drier, with snowfall shifting to rainfall. In addition to climate change and variability, both atmospheric CO_2 concentrations and N deposition (fig. 58c) have increased over the past few decades as a result of human activities. Nitrogen deposition in this region peaked around 1990 and has since declined steadily (fig. 58c).

The C stock changes across the national forests in the Pacific Northwest Region show that together the forests generally underwent a switch from a C sink to a C source (figs. 59a–c) in the mid-1980s. Disturbance and aging effects have been mostly responsible for declining C stocks (fig. 59b), which correspond to disturbances that shaped forest structure (fig. 56), the NPP-stand age relationships (fig. 57), and increasing disturbances in recent years (fig. 52). From 1950 to the early 1980s, the forests were a C sink due to positive disturbance and aging effects, as the pulse of stands that were established between 1900 and 1940 was middle-aged and therefore growing at maximum rates of productivity. As these forests further aged, productivity declined, causing the rate of C accumulation to decline (fig. 59d). Meanwhile the effects of disturbance were increasing to the point where C emissions due to decomposition and disturbance events exceeded C gains, so the forests became a C source (figs. 59b,c). The decline in productivity was accompanied by lower rates of stand establishment and increasing rates of timber harvesting from the 1940s until the 1990s.

Recent disturbances such as the large, moderate- to high-severity fires in 2002 (fig. 52) caused spikes in C emissions (fig. 59e). The 2002 fires coincided with an intense drought, which together produced a significant C source in 2002 and 2003 (figs. 59a–c). As forests begin to recover from these disturbance events, as seen in the stand-age distributions for several forests in the region (Appendix 8: Pacific Northwest Region (PDF, 4.2MB)), and reach middle-age with the highest rates of productivity (fig. 57), forests have the potential to rebound and become a C sink again in a few decades.

While disturbance and aging effects initially caused a C sink and then shifted to creating a C source, nondisturbance factors had a mostly positive effect on C stocks in the 1950s and later (figs. 59a,b). Increasing levels of atmospheric CO_2 and N deposition caused forests to accumulate more C and helped to counteract the C declines due to disturbances and the aging stands. By the early 1980s, however, the C gains from CO_2 and N deposition were surpassed by the C losses due to negative disturbance and aging effects, causing the forests to shift to a C source (fig. 59d). Atmospheric CO_2 concentrations are expected to continue increasing for the foreseeable future, potentially offsetting some of the C declines due to disturbances and aging. Despite increasing temperature and decreasing precipitation trends (figs. 58a,b), climate has had a small positive effect on total C accumulation for much of the past few decades. However, since the early 2000s, climate effects have been mostly negative, causing a loss of forest C (fig. 59d). Future projected warming and declines in precipitation may cause the C source in the Pacific Northwest Region to persist.

Although a few national forests in the Pacific Northwest Region had a net loss of total forest C, most forests had C gains between 1950 and 2011 resulting in a regionwide gain of approximately 93.3 Tg of total ecosystem C, including soil C, between 1950 and 2011 (fig. 59d).

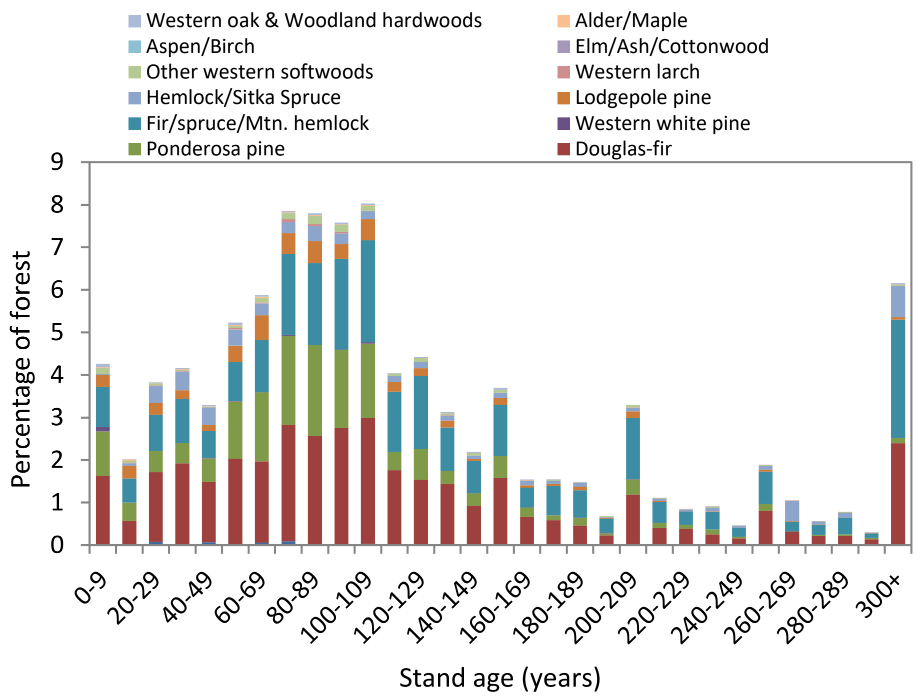

Figure 56. Age-class distribution in 2011 shown as the percentage of forest land in each forest-type group in 10-year age classes summed across the national forests in the Pacific Northwest Region (see Appendix 1 for scientific names of trees).

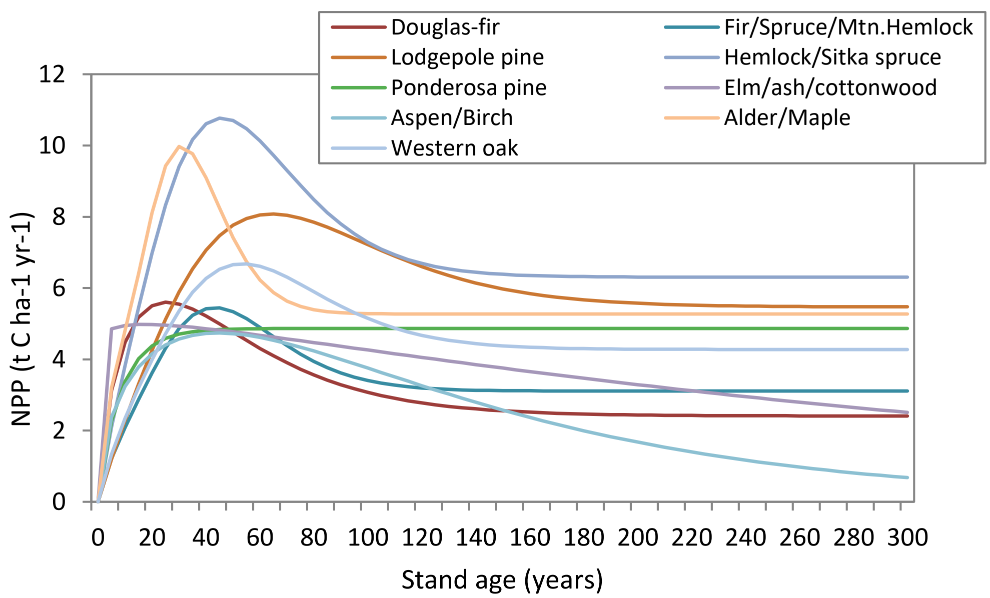

Figure 57. Relationship between net primary productivity (NPP) and stand age for each forest-type group averaged across all national forests in the Pacific Northwest Region (see Appendix 1 for scientific names of trees). Curves were developed by He et al. (2012). The evergreen needleleaf curve was applied to Douglas-fir and western larch stands. The western oak curve was applied to woodland hardwood and other hardwood stands; the ponderosa pine curve was applied to pinyon/juniper, other western softwoods, and western white pine stands.

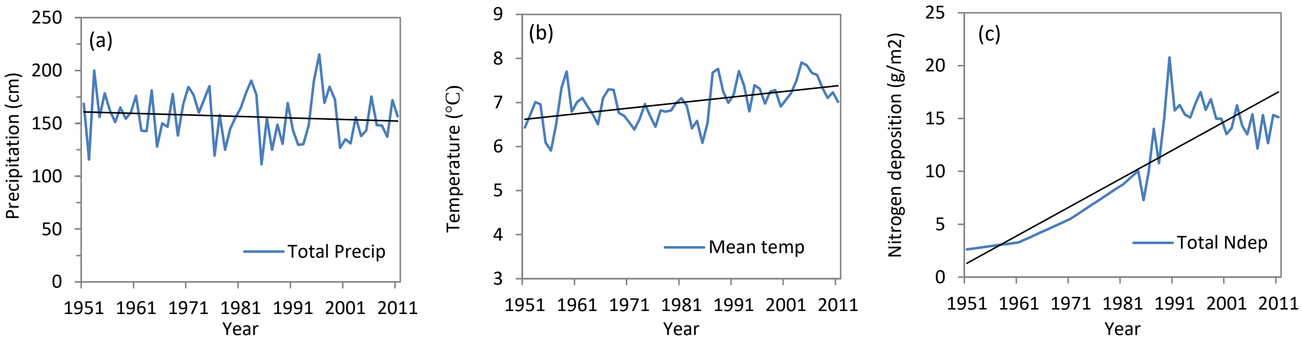

Figure 58a-c. (a) Total annual precipitation, (b) average annual temperature, and (c) total annual nitrogen deposition from 1951 through 2011 averaged across all national forests in the Pacific Northwest Region. Linear trend lines shown in black.

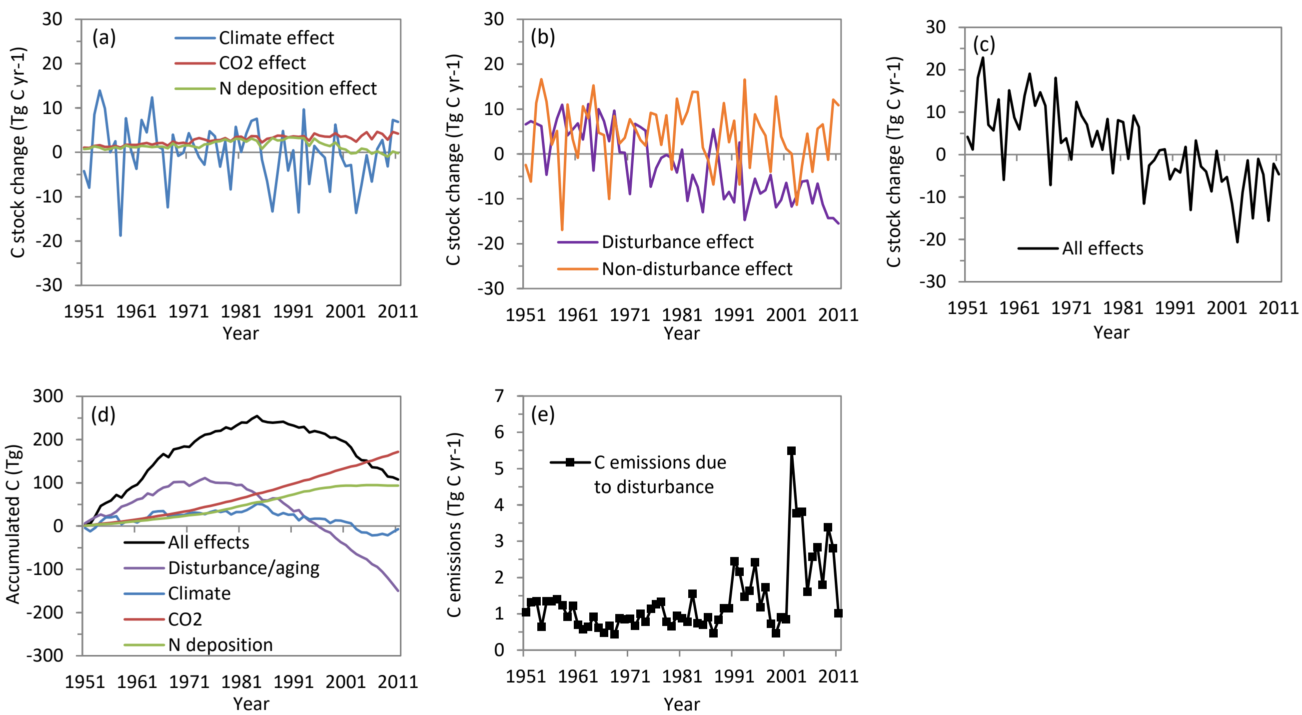

Figure 59a-e. Estimated forest carbon (C) changes and accumulations summed across the national forests in the Pacific Northwest Region. Changes in C stocks are attributed to: (a) individual nondisturbance factors alone, including climate variability, atmospheric carbon dioxide (CO_2) concentration, and nitrogen deposition; (b) all disturbance and aging factors including fire, harvest, insects, and regrowth and aging, and the sum of all nondisturbance factors combined; and (c) all factors combined, which is the sum of disturbance and aging and nondisturbance effects. (d) Accumulated C due to individual disturbance and aging and nondisturbance factors and all factors combined from 1950 through 2011 excluding C accumulated pre-1950. (e) Direct C emissions to the atmosphere due to disturbance events only. Positive values in graphs (a) through (c) represent net C sinks from the atmosphere, or increases in C stocks, whereas negative values represent net C sources to the atmosphere, or decreases in C stocks.

6.7 Southwestern Region

6.7.1 Description of Region

The Southwestern Region (also referred to as Region 3) in the National Forest System (NFS) consists of 11 national forests across Arizona and New Mexico (fig. 60). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these national forests, so national grasslands such as the Cibola National Grassland in Oklahoma were not included. To restrict the analysis to lands managed by the Forest Service, any private inholdings within national forest boundaries were excluded.

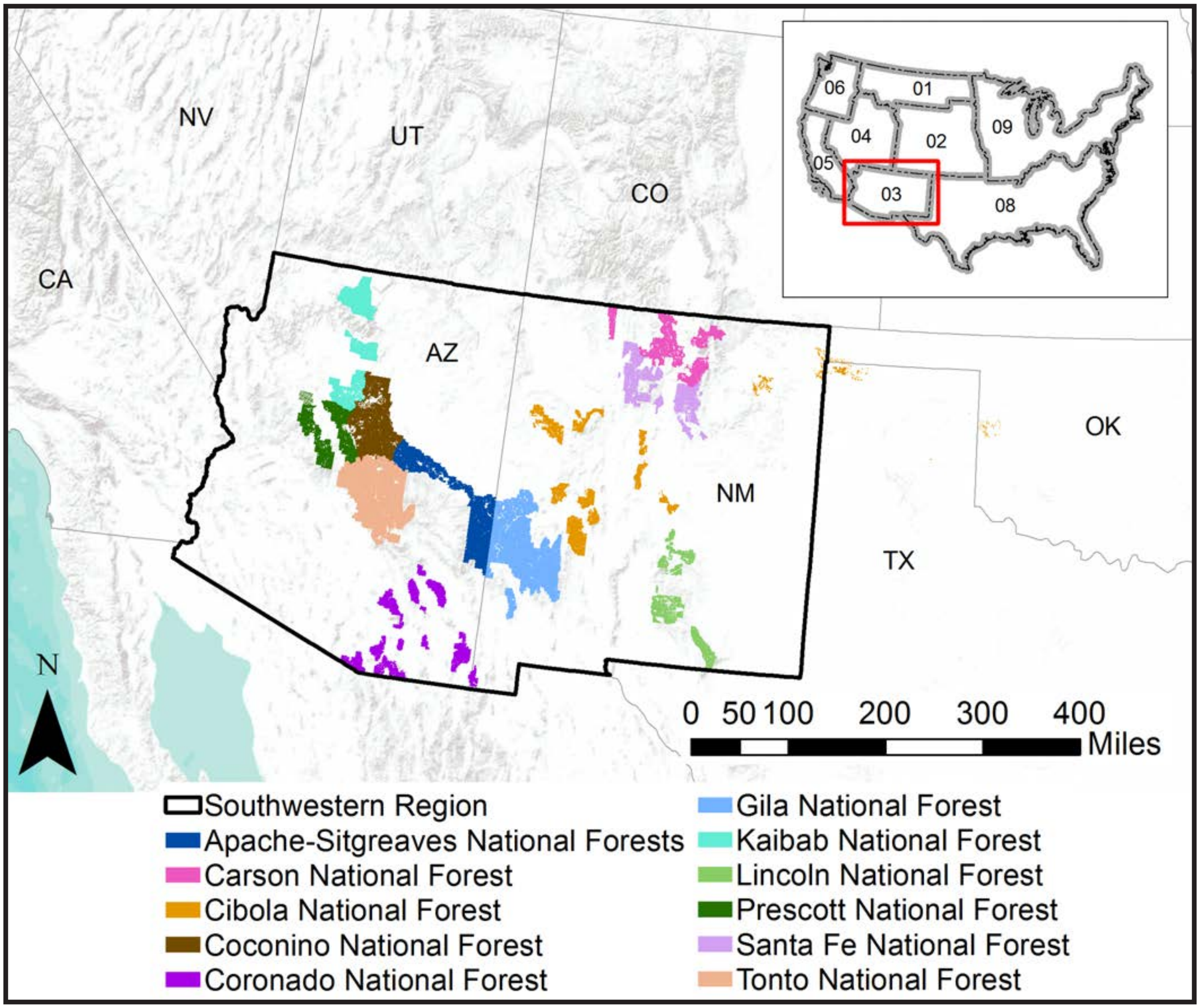

Figure 60. Locations of the national forests in the Southwestern Region.

The history of development, land use, and policies provides useful context for understanding the regional forest C trends that are indicated by the forest inventory and model results. Euro-American settlement in the Southwestern Region had a profound effect on forest structure, disturbance regimes, and subsequently forest C dynamics. Settlement began in the mid-1800s and accelerated in the 1880s and 1890s with the completion of key railroad lines. Railroad access in the Southwest also opened up commercial timber opportunities, though the timber industry remained modest compared to other regions in the United States (Baker et al. 1988). Timber output increased slowly from the early 1900s, peaked in 1994, and has since declined to about the levels of the 1920s (Butler et al. 2014b).

Euro-American settlers also brought intensive livestock grazing to the region. Grazing depleted the range and reduced herbaceous fuels, disrupting natural fire regimes and increasing tree density (Covington and Moore 1994). In 1905 when the Forest Service took control of the forest reserves, the era of free use and unrestrained grazing came to an end (Baker et al. 1988).

Although grazing substantially altered these forests, fire has been the primary disturbance shaping forests of the Southwestern Region for hundreds of years. Prior to Euro-American settlement, fires were more frequent and less severe, maintaining lower forest densities (e.g., Covington and Moore 1994), though infrequent, mixed- and high-severity fires also played an important role in shaping forest structure (e.g., Williams and Baker 2012). A policy of fire suppression began in the early 1900s, disrupting natural fire regimes and altering forest structure (Pyne 1982). In the 1970s, fires were once again allowed to burn, and today prescribed burning is commonly used to manage fuel loads and restore historical disturbance regimes. More recently, the region has experienced several very large fires such as the Wallow Fire on the Apache-Sitgreaves National Forests in 2011.

Climate has also played an important role in historical forest structure and subsequent C trends. Historically, periods of increased precipitation triggered pulses of pine establishment (Savage et al. 1996). The Southwestern Region has a unique monsoonal climate. Up to 50 percent of the annual rainfall occurs from July through September (Sheppard et al. 2002), thus enhancing forest growth rates during the growing season. Over the past few decades, climate change has caused unprecedented temperature increases and has increased the risk of future multidecadal megadroughts (e.g., Ault et al. 2014).

6.7.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Southwestern Region are displayed in figure 61. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. Regional disturbance patterns were dominated by fire, with at least some of the region burning in each of the monitored years (fig. 61a). There were several years in which the area affected by fire approached 1 percent of the forest land, with most of those years occurring after 2000. Though harvest affected more area than fire in the first 3 years of the study period, harvest rates generally decreased afterward, while fire became more common. In several years, disturbance due to insect activity was measurable, although not high. As with fire, years with higher levels of insect damage were generally

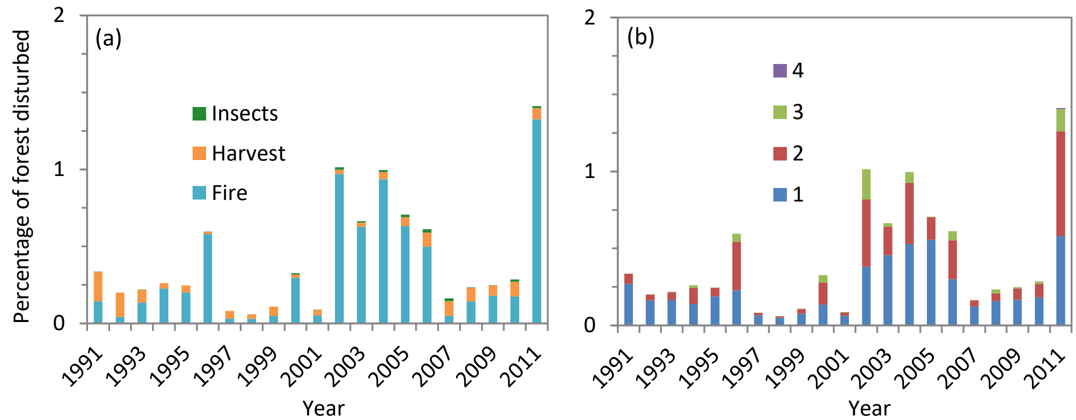

Figure 61a-b. Annual rates of disturbance (0 to 2 percent) in the Southwestern Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

concentrated after 2000. Over 90 percent of the disturbed area from 1990 through 2011 was characterized by low- to moderate- intensity impacts (<50 percent change in canopy cover) (fig. 61b).

6.7.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to estimate the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Southwestern Region in figure 62. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires occur only in 1992, for example, the line of fire in figure 62 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an undisturbed scenario. Thus, figure 62 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

Fire accounted for more than 90 percent of the disturbance impact from 1990 through 2011 on 2011 C stocks (fig. 63). By 2011 fire was responsible for a 2.0-percent reduction in total nonsoil C stocks (fig. 64). The acceleration of fire activity after 2000 within the region (fig. 61) resulted in a surge in fire's impact on C stocks (fig. 62). The stable harvest rate in the region had a relatively low impact on C stocks (fig. 63), by 2011 resulting in removal of 0.2 percent of the regional nonsoil C stocks (fig. 64) that were reported in the NFS baseline C assessment derived from FIA data (USDA FS 2015i). These patterns do not account for offsite storage of C

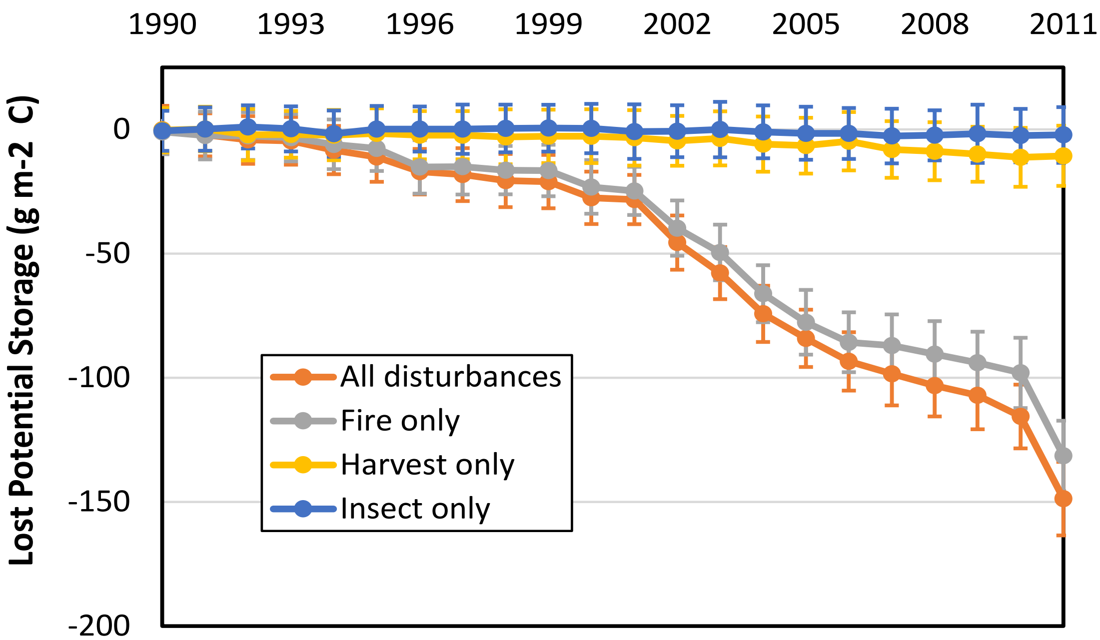

Figure 62. The impact of different kinds of disturbance, occurring from 1990 through 2011, on carbon (C) stores in the Southwestern Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

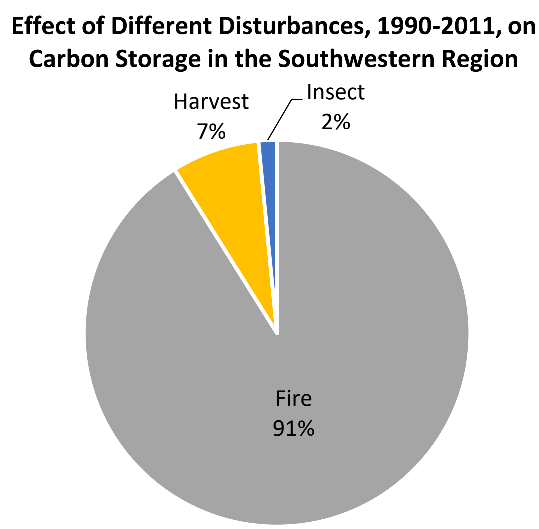

Figure 63. Proportional effect of different kinds of disturbance on carbon storage in all national forests in the Southwestern Region for the period 1990 through 2011. Insect impacts made up less than 1 percent of the regional total.

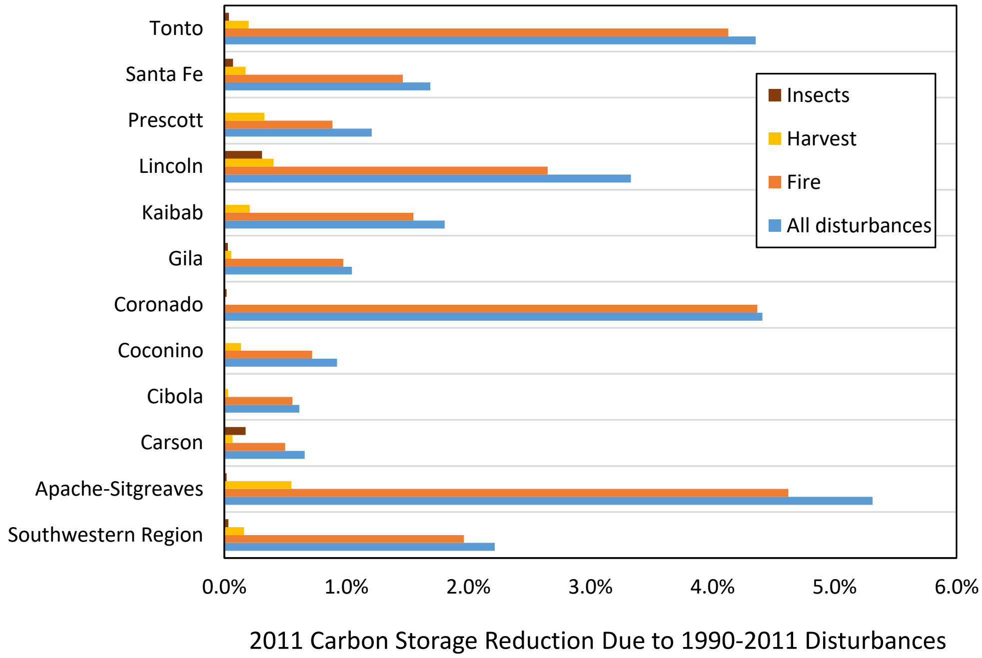

Figure 64. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by each national forest and for all national forests in the Southwestern Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

in wood products. Information about the region's product C storage can be found in the baseline C assessment (USDA FS 2015i).

The effect of insects on C storage accounted for less than 1 percent of the potential C storage lost in the region during the study period (fig. 63). Most observed insect activity occurred at the end of the study period (2008 or later) (fig. 62), and it is unclear whether this disturbance process will continue to increase in importance.

One caveat must be stated about the confidence intervals depicted in figure 62. The unit of analysis for ForCaMF was the national forest (Appendix 9: Southwestern Region, (PDF, 2.9MB)); rates of simulated error were constrained by FIA estimates at that level. Regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in other forests. This assumption very likely overstated certainty at the regional level; if initial biomass estimates in one forest were high, for example, the same estimates were also likely to be biased high in other units. This caveat does not change the overall trends seen at the aggregated regional level, nor does it affect confidence intervals assessed at the national forest level. Fire is by far the dominant disturbance affecting C storage in the region, and its influence is growing. Despite relatively frequent fire activity, the C storage impact of disturbance within the region on a per unit area basis was relatively low, in part because the landscape generally stores less C than national forests in other regions. Another partial explanation may be the relatively slow decay factors built into regional variants of FVS, which generated the C dynamics used here. Despite relatively low absolute impacts of disturbance, the Southwestern Region is notable for the pronounced dominance of one disturbance process (fire).

6.7.4 Effects of Disturbance, Management, and Environmental Factors

This section provides a regional summary of the predicted (modeled) effects of natural disturbances, land management, and environmental factors on forest C dynamics depicted in a series of figures (figs. 65–68) generated from InTEC model inputs and simulations for individual national forests and summed results across all national forests in the Southwestern Region. These regional-scale outputs were generated only from the national forest-specific datasets; thus, these outputs do not represent lands within the region that are outside of the national forest boundaries. The Appendix 9: Southwestern Region, (PDF, 2.9MB) shows the effects of disturbance and nondisturbance factors on forest C dynamics within each of the 11 national forests as modeled by InTEC using this same series of figures.

According to the historical data and model results, forest C trends across the national forests in the Southwestern Region have been strongly influenced by the history of land use and policies as well as climate variability. Despite variation among these national forests, the regionwide stand-age distribution in 2011 shows that most stands are older (>80 years old) with a pulse of stand establishment which occurred about 80 to 119 years earlier, or from approximately 1891 to 1930 (fig. 65). This spike in establishment may be a result of several factors or a sequence of factors, such as regeneration after intensive settlement activities, recovery after the last major fires before fire suppression, lower fire frequency because of livestock grazing, or favorable climate conditions. Fire suppression, which began in the early 1900s, allowed more of these young, regenerating stands to survive and continue regrowing in the absence of fire. Depending on the forest type, which is mostly pinyon-juniper and ponderosa pine, forests would have reached maximum productivity around 45 years old; then productivity would have stabilized and remained relatively constant (fig. 66). However, after reaching peak productivity, all the other forest types would have shown a decline as forests continued to age. Furthermore, after the 1930s, stand establishment dramatically declined, especially between the 1950s and 1991. Stand establishment increased again in the 1990s and 2000s (fig. 65), likely due to recovery after increases in fire disturbances (fig. 61).

Climate has on average grown much warmer in the region, while precipitation has been variable. Though lacking a clear regional trend, precipitation has increased for some individual forests and decreased for others in the second half of the period (figs. 67a,b). Warmer temperatures can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity in the absence of increasing rainfall. Consequently, more C is released to the atmosphere. As climate continues to change, this region is expected to become warmer with higher risks of severe, multidecadal droughts (e.g., Ault et al. 2011). Nitrogen deposition increased substantially between the 1950s and 2000s, but stabilized and declined in 2008 and 2009 before increasing again (fig. 67c). Atmospheric CO_2 concentrations have increased across the United States and globally as a result of human activities since the onset of industrialization. Elevated CO_2 levels and N deposition can enhance forest growth and increase water-use efficiency, resulting in a positive effect on C stocks.

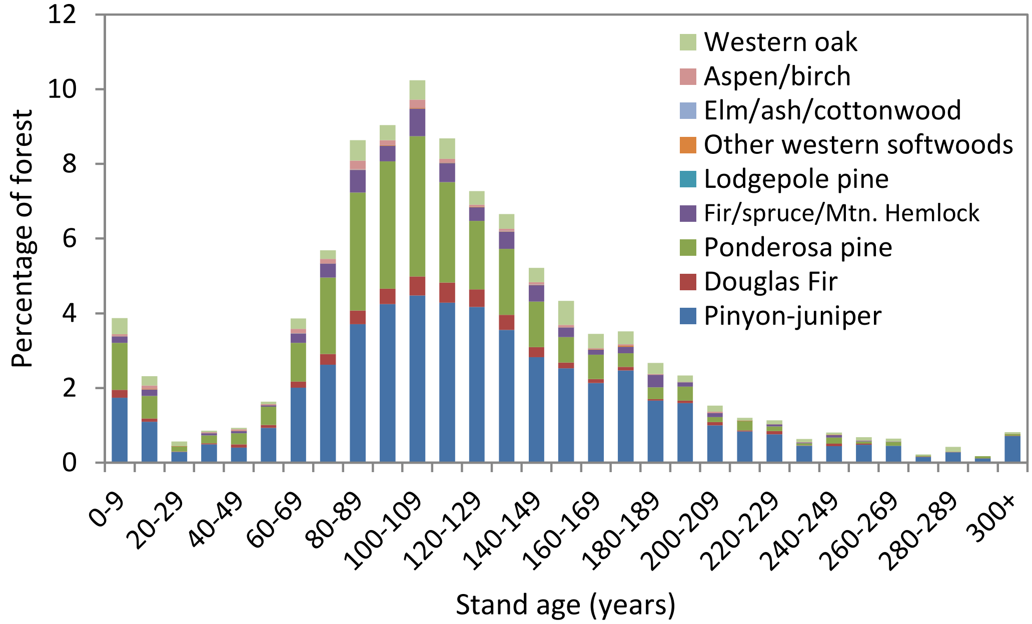

Figure 65. Age-class distribution in 2011 shown as the percentage of forest land in each forest-type group in 10-year age classes summed across the national forests in the Southwestern Region (see Appendix 1 for scientific names of trees).

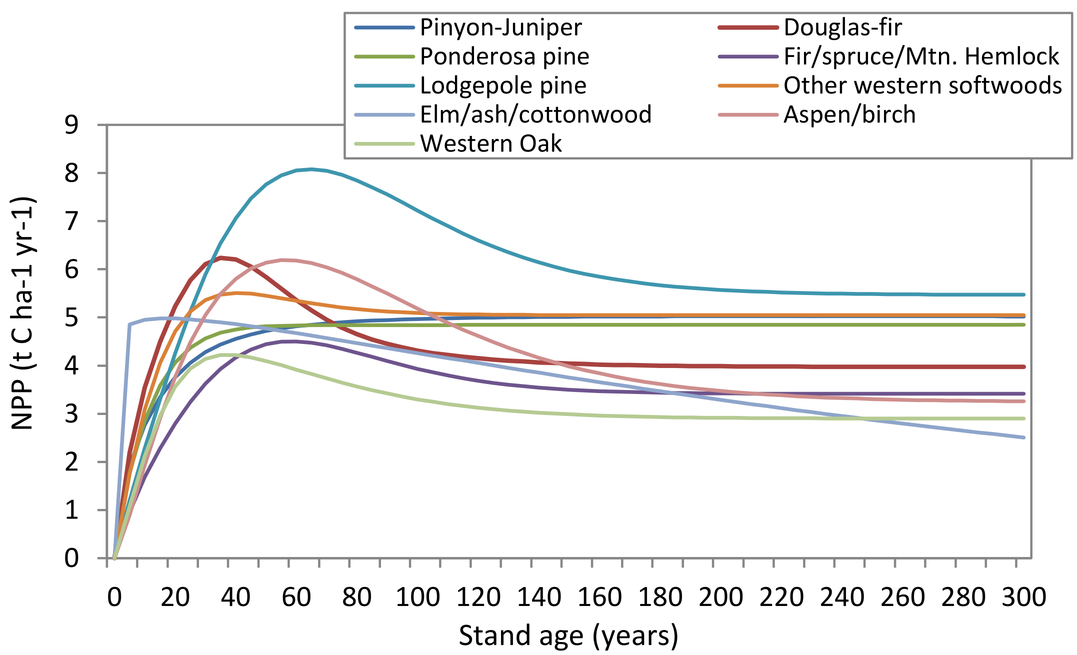

Figure 66. Relationship between net primary productivity (NPP) and stand age for each forest-type group averaged across all national forests in the Southwestern Region (see Appendix 1 for scientific names of trees).

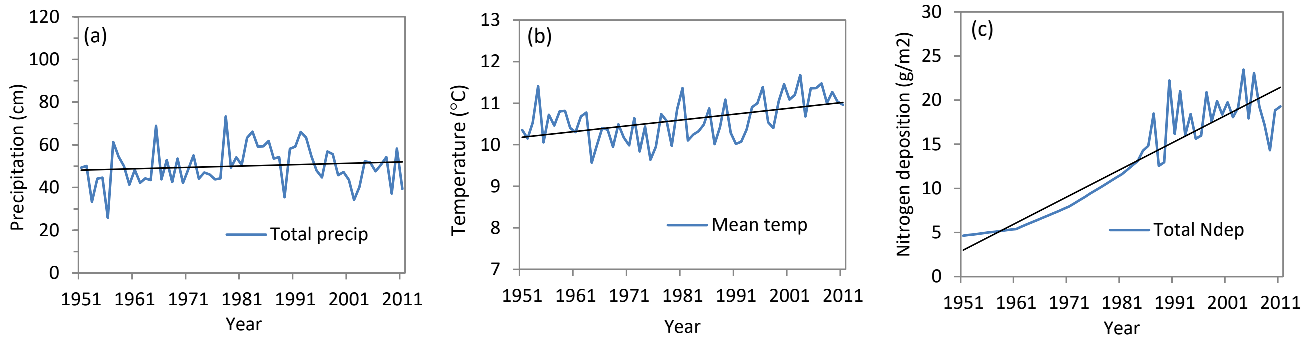

Figure 67a-c. (a) Total annual precipitation, (b) average annual temperature, and (c) total annual nitrogen deposition from 1950 through 2011 averaged across all national forests in the Southwestern Region. Linear trend lines shown in black.

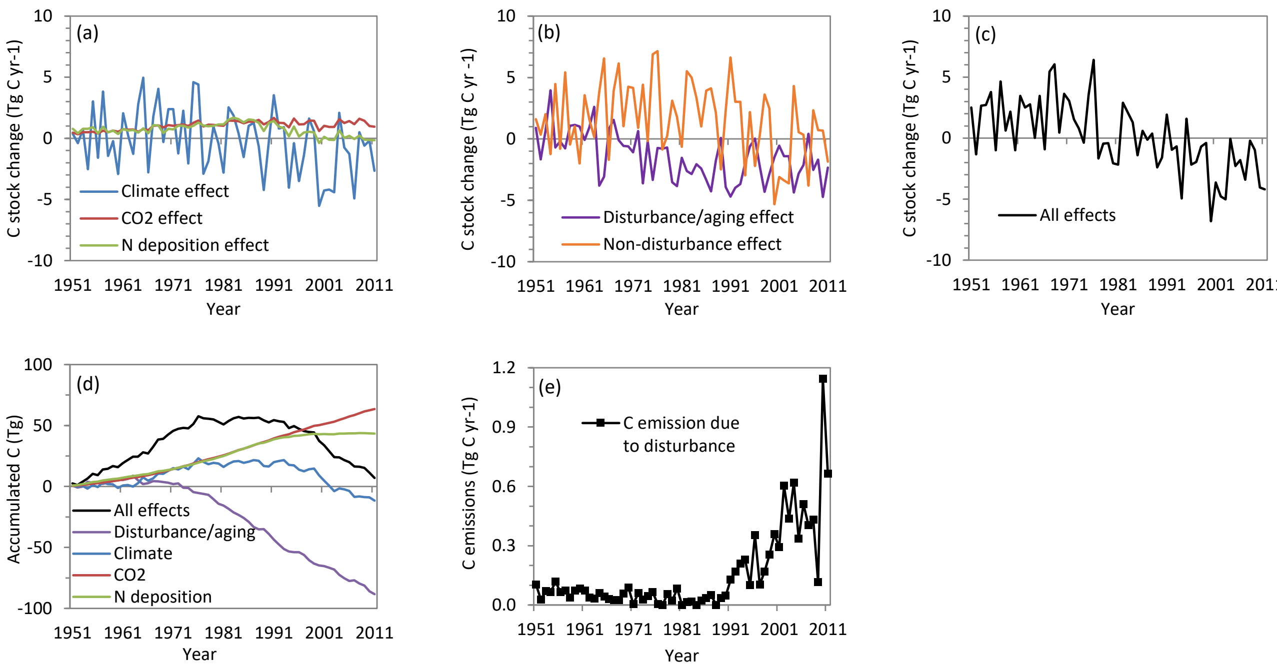

Figure 68a-e. Estimated forest carbon (C) changes and accumulations summed across the national forests in the Southwestern Region. Changes in C stocks are attributed to: (a) individual nondisturbance factors alone, including climate variability, atmospheric carbon dioxide (CO_2) concentration, and nitrogen deposition; (b) all disturbance and aging factors including fire, harvest, insects, and regrowth and aging, and the sum of all nondisturbance factors combined; and (c) all factors combined, which is the sum of disturbance and aging and nondisturbance effects. (d) Accumulated C due to individual disturbance and aging and nondisturbance factors and all factors combined from 1950 through 2011 excluding C accumulated pre-1950. (e) Direct C emissions to the atmosphere due to disturbance events only. Positive values in graphs (a) through (c) represent net C sinks from the atmosphere, or increases in C stocks, whereas negative values represent net C sources to the atmosphere, or decreases in C stocks.

The C stock changes across the national forests in the Southwestern Region show that forests were mostly a net C sink from the 1950s through the mid-1970s, then gradually shifted to a C source in the 1980s and 1990s, before becoming a consistent C source in the 2000s (fig. 68c). Changing C stocks and C accumulation have been influenced strongly by disturbance and aging effects and, to a lesser extent, climate effects (figs. 68a–d). Together the forests were a C sink when the stands making up the pulse of establishment between 1890 and 1930 had reached maximum productivity around 45 years old (1930s–1970s). During this period, climate effects were mostly positive, especially during the 1960s and 1970s, when temperatures were slightly lower than average (fig. 67b) and promoted further accumulation of C in forests (fig. 68d). The forests shifted to a C source in the mid-1970s as stand establishment significantly declined and temperatures started increasing more steadily. The C losses from decomposition and respiration started to exceed C gains, causing forests to become a C source.

In the early 2000s, drought conditions, combined with high temperatures (figs. 67a,b), further exacerbated the C source (fig. 68a). At the same time, low- to moderate-severity fires (fig. 61) also increased, causing a spike in disturbance-induced C emissions (fig. 68e). However, these recent disturbances also promoted regrowth and recovery as shown by the small pulse of stands less than 20 years old (fig. 65). If these young stands are able to survive and reach middle-age in coming decades, they will be growing at a higher productivity rate. Forests could potentially accumulate C faster and become a C sink again.

From the 1950s on, atmospheric CO_2 concentrations and N deposition consistently had positive effects on forest C, enhancing the C sink and C accumulation (figs. 68a,d). In fact, the positive effects of N deposition and CO_2 were able to at least partially offset the negative disturbance effects since the 1970s and the more recent negative climate effects since the early 2000s, such that national forests in the Southwestern Region (Appendix 9: Southwestern Region, (PDF, 2.9MB)) experienced a small net gain of 7 Tg of total ecosystem C, including soil C, between 1950 and 2011 (fig. 68d).

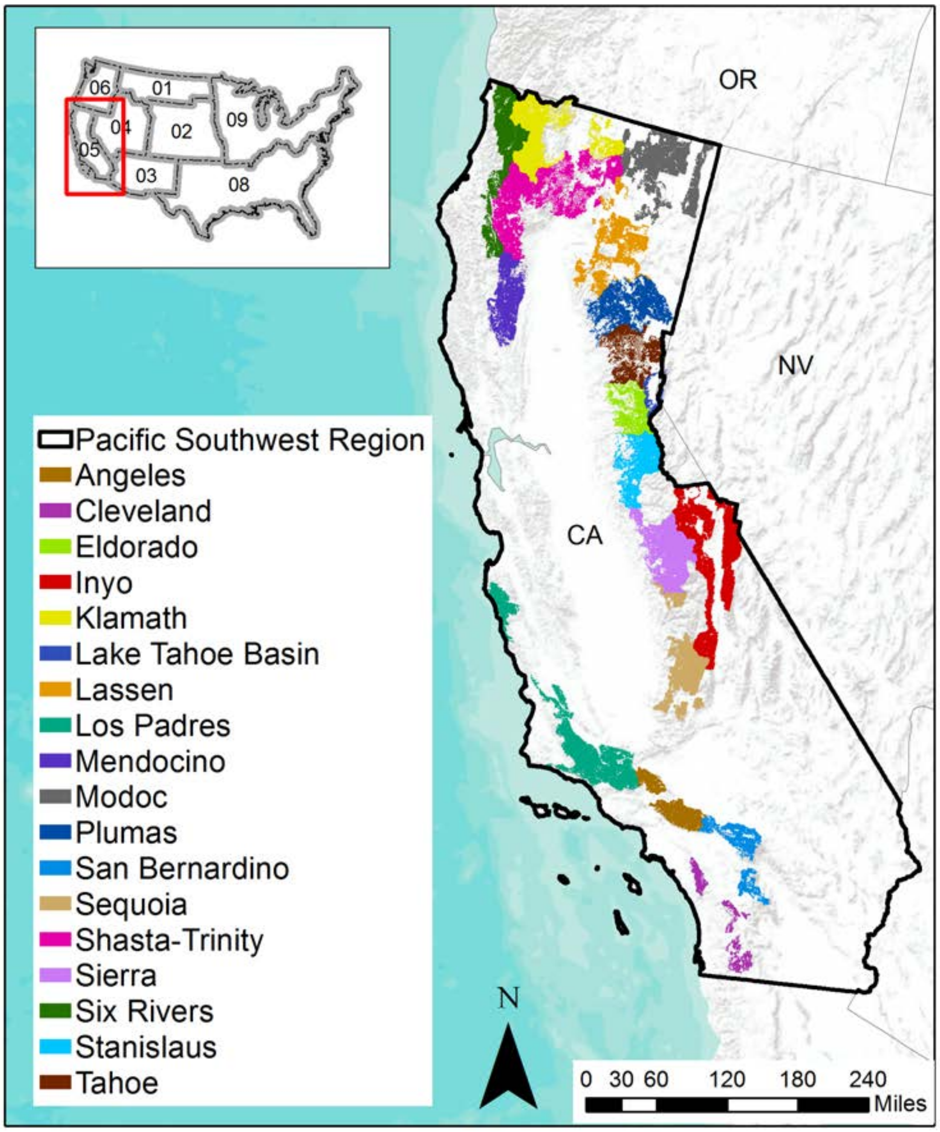

Figure 69. Locations of the national forests in the Pacific Southwest Region.

6.8 Pacific Southwest Region

6.8.1 Description of Region

The Pacific Southwest Region (also referred to as Region 5) in the National Forest System (NFS) consists of the Lake Tahoe Basin Management Unit and 18 national forests spread throughout California and into Oregon and Nevada (fig. 69). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these units. To restrict the analysis to only lands managed by the Forest Service, any private inholdings within national forest boundaries were excluded.

The history of Euro-American settlement, land use, and policies provides useful context for understanding the regional forest C trends that are indicated by the inventory and assessment results. Despite colonization by the Spanish in the mid-1700s and a trickle of immigrants over the Sierra Nevadas in the 1820s, it was the discovery of gold in 1849 that brought a flood of immigrants to California. The development of gold mining operations triggered a population explosion and great demand for wood products. As a result, the forests were cleared to support mining operations, railroad construction, and the spread of towns and cities through the Sierra Nevada region, along the Pacific coast north of San Francisco, and in the pine forests of Southern California (USDA FS 1927) (see Appendix 1 for scientific names of species mentioned in this report). Timber harvesting continued to increase steadily into the early 1900s (Stockman et al. 2014c). Settlers also brought herds of grazing cattle and sheep which trampled, foraged, and eroded forest lands. While early Indians used fire to clear land for hunting and protection, the advent of Euro-American settlers led to more extensive, frequent, and uncontrolled fires in the forests of California. However, many of these gold seekers and settlers were forward-thinking and pressed for forest conservation (USDA FS 1927). Their advocacy inspired the first instance in which the U.S. Government set aside park land for public use and recreation. Lands set aside in 1864 under the Yosemite Grant were initially managed by the State of California, and administration of this grant paved the way for the creation of the first national park in Yellowstone.

By the 1890s, the first forest reserves were designated and more sustainable timber, livestock grazing, and fire management became an objective. The California lumber industry declined during the Great Depression in the 1930s before rapidly increasing again in the 1950s and 1960s (Stockman et al. 2014c). A national policy of fire suppression, which began after the devastating 1910 fire season, entailed intensive efforts to prevent, detect, and suppress all wildfires (Baker et al. 1993; Pyne 1982). This policy caused significant declines in fire frequency and area burned throughout the Cascades, Klamath Mountains, Tahoe Basin, and Sierra Nevada region, allowing forests to regenerate and survive in the absence of fire (Beaty and Taylor 2008). A policy shift from fire control to fire management in the 1970s enabled natural fires to burn across landscapes where safe and appropriate, in an attempt to restore historical fire regimes (Pyne 1982).

Both land use history and recent changes in climate have led to large and severe disturbances as well as more frequent and severe droughts in the Pacific Southwest Region over the past few decades. Between 1984 and 2006 the extent of high-severity, stand-replacing fires has dramatically increased in California and fires are burning at generally higher severities than before Euro-American settlement (Miller et al. 2009). Since the 1970s the frequency of wildfires has also increased in much of the western United States due to warmer and drier conditions as well as the build-up of surface fuels resulting from a history of fire suppression (Westerling et al. 2006). Elevated tree mortality in California has been linked to multiyear droughts characterized by low winter to spring snowpack, followed by high spring and summer temperatures (Guarín and Taylor 2005; van Mantgem et al. 2009). The super El Niño of 1982 to 1983 brought damaging storms with strong winds, heavy snowfall, and flooding rains across all of California.

Recent tree mortality is one of the most urgent issues facing the region and California. This unprecedented event is crossing all land ownerships and affecting entire landscapes and ecosystems. With California in its fifth consecutive year of severe drought, lack of moisture and elevated bark beetle activity have killed an estimated 66 million trees throughout the State since 2010. The west slope of the southern Sierra Nevadas, including the Stanislaus, Sierra, and Sequoia National Forests, has been the hardest hit area.

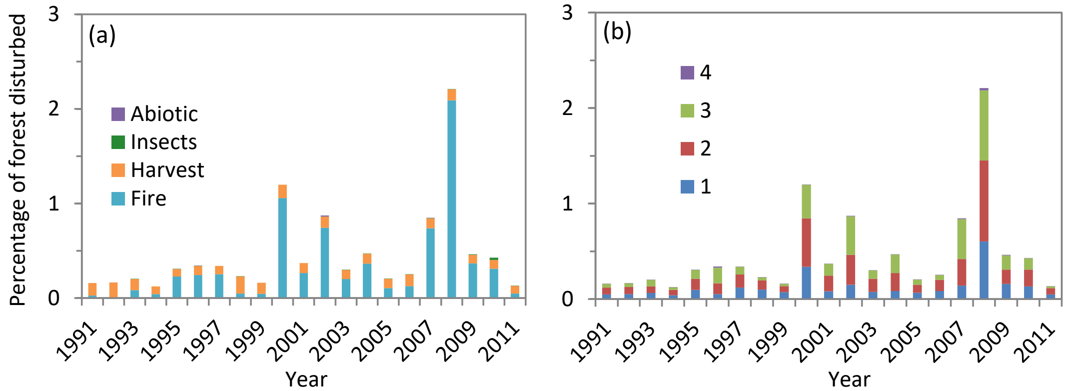

Figure 70. Annual rates of disturbance (0 to 3 percent) in the Pacific Southwest Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

6.8.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Pacific Southwest Region are displayed in figure 70. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. Regional disturbance patterns are marked by relatively low, but stable, harvest rates, and periodic large fire years that have occurred since 2000 (fig. 70a). Fires were largest in 2000 and 2008, disturbing about 1 percent and 2 percent of the forested area, respectively. Disturbances were mostly moderate intensity (25 to 75 percent change in canopy cover), with very little area affected by high-intensity, stand-replacing disturbances (76 to 100 percent change in canopy cover) (fig. 70b).

6.8.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to estimate the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Pacific Southwest Region in figure 71. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires first occur in 2006, for example, the line of fire in figure 71 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an undisturbed scenario. Thus, figure 71 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

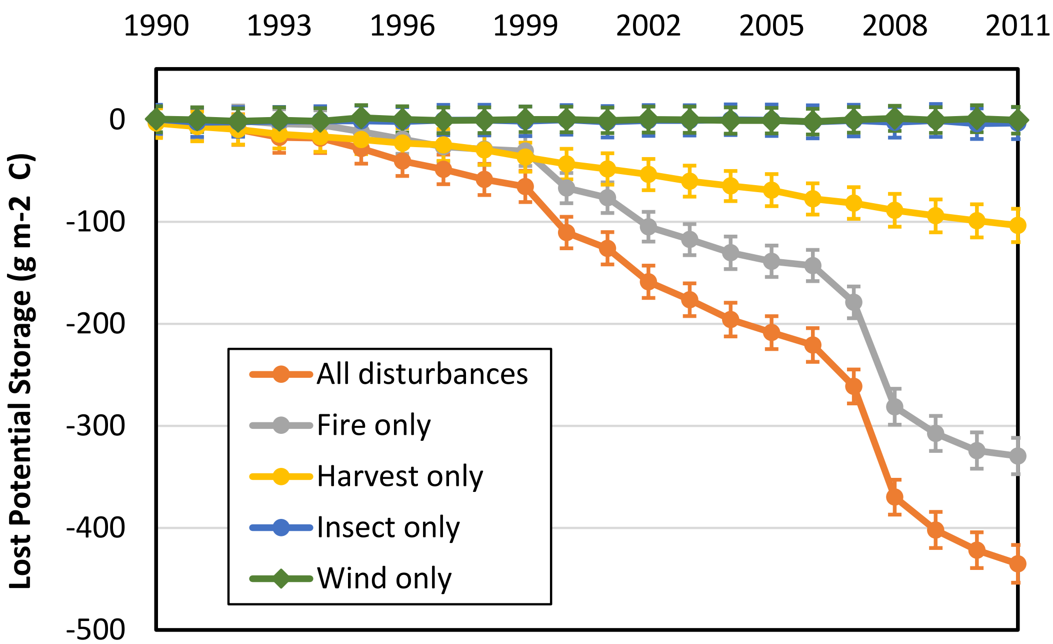

Figure 71. The impact of different kinds of disturbance, occurring from 1990 through 2011, on carbon (C) stores in the Pacific Southwest Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

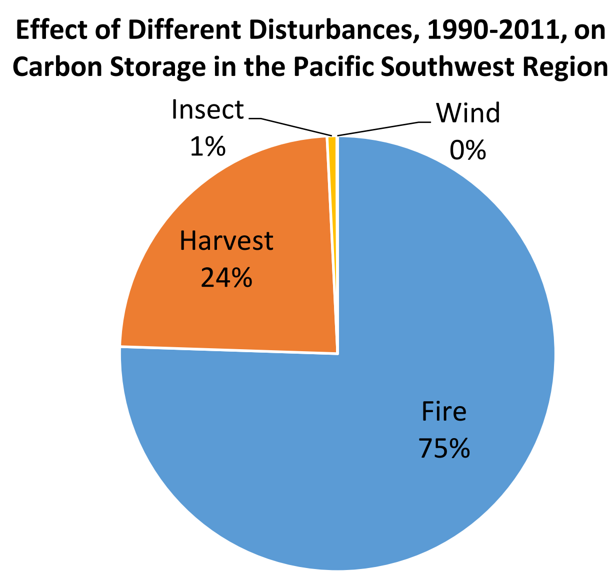

Figure 72. Proportional effect of different kinds of disturbance on carbon storage in all national forests in the Pacific Southwest Region for the period 1990 through 2011.

Fire accounted for approximately three-quarters of the disturbance impact on 2011 C stocks (fig. 72). The area of forests affected by fires started to increase in the 2000s, which was approximately the point during the study period when the impact of fire exceeded the impact of harvest in the region (fig. 71). Fires were particularly large in 2000 and 2008, when they affected more than 1 percent of the region's forest land. In 2008, about 2 percent of the region's forests was affected by fire, and local rates on some forests were much higher. A substantial increase in the impact of fire on the region's C storage may be seen in 2008 (fig. 71). By 2011, fires accounted for a 2.1-percent reduction in nonsoil C storage in the region (fig. 73).

The C impact of harvest within the region was relatively stable from 1990 through 2011 (fig. 71), because harvest levels, while low, were consistent throughout the period. Harvest represented about one-quarter of the disturbance impact on 2011 landscape C storage (fig. 72). By 2011, this represented a reduction of about 0.7 percent of the regional nonsoil C stocks (fig. 73) that were reported in the NFS baseline C assessment derived from FIA data (USDA FS 2015e). These patterns do not account for offsite storage of C in wood products. Information about the region's product C storage can be found in the baseline C assessment (USDA FS 2015e).

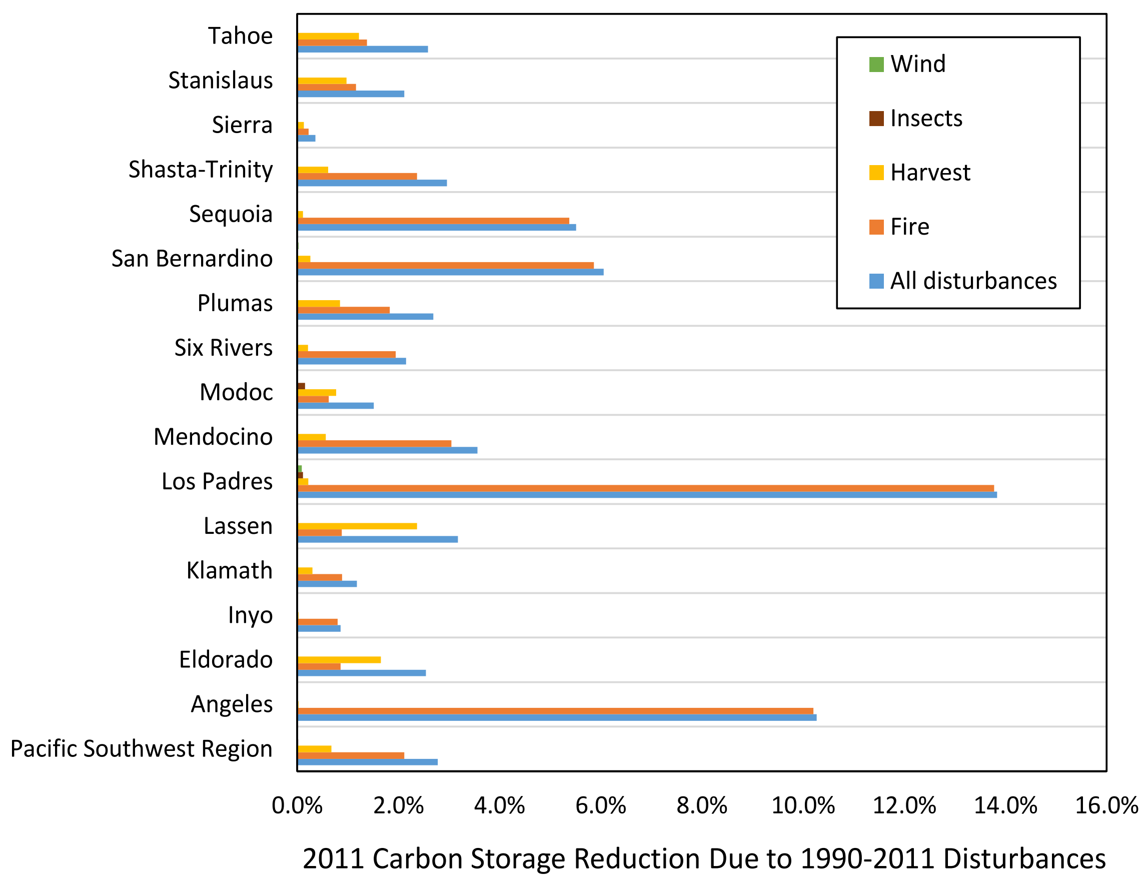

Figure 73. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by each national forest and for all national forests combined in the Pacific Southwest Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

The effect of insects on C storage represented approximately 1 percent of the disturbance impact on the region during the study period (fig. 72), with relatively higher impacts only in the Modoc National Forest (approximately 10 percent; Appendix 10: Pacific Southwest Region (PDF, 4.2MB)). Insect activity caused a small reduction (0.02 percent) in nonsoil C storage by 2011 (fig. 73). Most observed insect activity occurred at the end of the study period (2008 or later).

One caveat must be stated about the confidence intervals depicted in figure 71. The unit of analysis for ForCaMF was the national forest; rates of simulated error were constrained by FIA estimates at that level. Regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in other forests. This assumption very likely overstated certainty at the regional level; if initial biomass estimates in one forest were high, for example, the same estimates were also likely to be biased high in other units. This caveat does not change the overall trends seen at the aggregated regional level, nor does it affect confidence intervals assessed at the national forest level. Fire has been the dominant disturbance affecting C storage in the region, and its influence is growing. On a per unit area basis, the C storage impact of fire in the Pacific Southwest was by far the greatest of all the regions.

6.8.4 Effects of Disturbance, Management, and Environmental factors

This section provides a regional summary of the predicted (modeled) effects of natural disturbances, land management, and environmental factors on forest C dynamics depicted in a series of figures (figs. 74–77) generated by InTEC simulations for individual national forests and summed results across all national forests in the Pacific Southwest Region. These regional-scale outputs were generated only from the national forest-specific datasets; thus, these outputs do not represent lands within the region that are outside of the national forest boundaries. Appendix 10: Pacific Southwest Region (PDF, 4.2MB) shows the effects of disturbance and nondisturbance factors on forest C dynamics within each of the 18 national forests as modeled by InTEC using this same series of figures.

According to the historical data and model results, forest C trends across the national forests in the Pacific Southwest Region have been strongly influenced by the history of land use and policies as well as climate change. Despite variation among these national forests, the regionwide stand-age distribution in 2010 shows a distinct pulse of stands established approximately 70 to 109 years earlier, or from about 1901 to 1940 (fig. 74). This pulse of stand establishment reflects forest recovery from disturbances after the onset of rapid settlement of California following the discovery of gold in the mid-1800s. In the early 1900s, California's national forests were designated and more sustainable land use became a focus, helping forests to regenerate. Another important factor during the regeneration phase was fire suppression, which may have allowed more stands to survive and continue regrowing rather than being disturbed at a more typical historical rate of fires (Pyne et al. 1982). Depending on the forest type, which is mostly California mixed conifer followed by Douglas-fir and fir/spruce/mountain hemlock (fig. 74), these pulses of stand establishment would have reached their maximum productivity at 35 to 45 years old (fig. 75), or from the 1930 through 1980s.

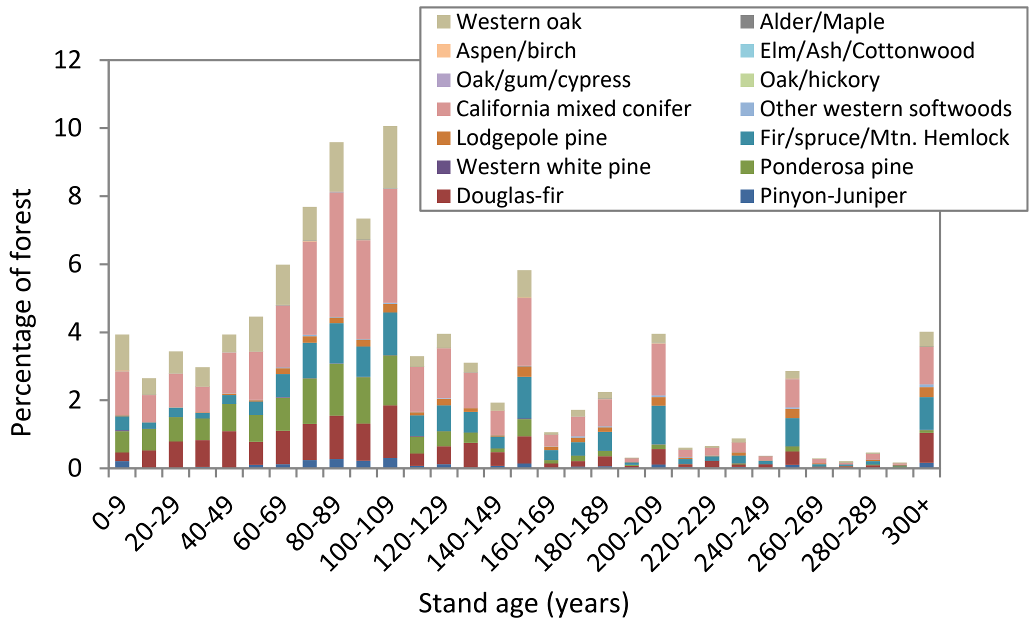

Figure 74. Age-class distribution in 2011 shown as the percentage of forest land in each forest-type group in 10-year age classes summed across the national forests in the Pacific Southwest Region (see Appendix 1 for scientific names of trees).

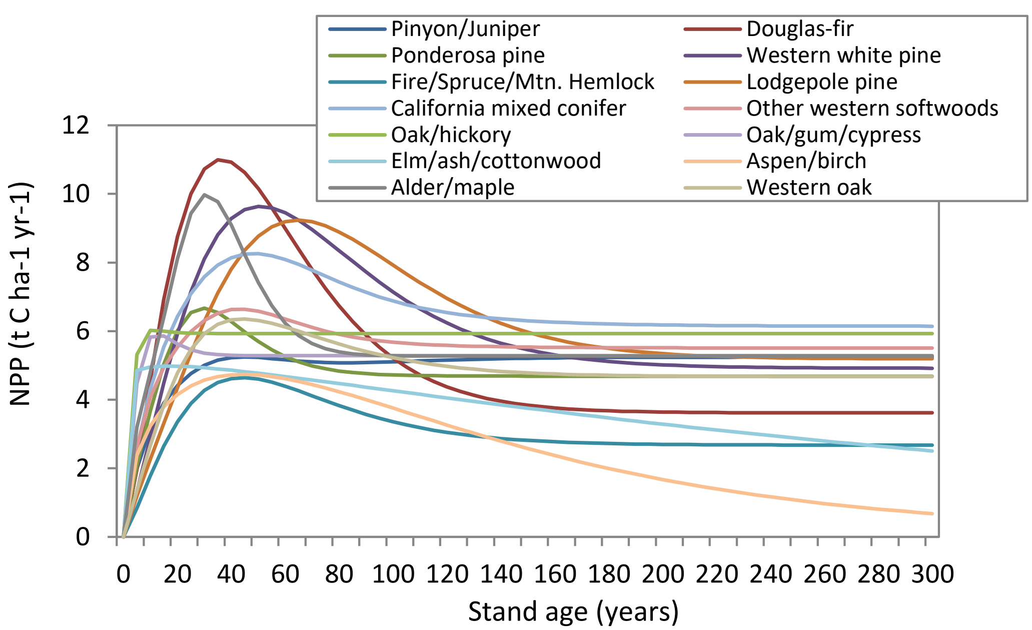

Figure 75. Relationship between net primary productivity (NPP) and stand age for each forest-type group averaged across all national forests in the Pacific Southwest Region (see Appendix 1 for scientific names of trees).

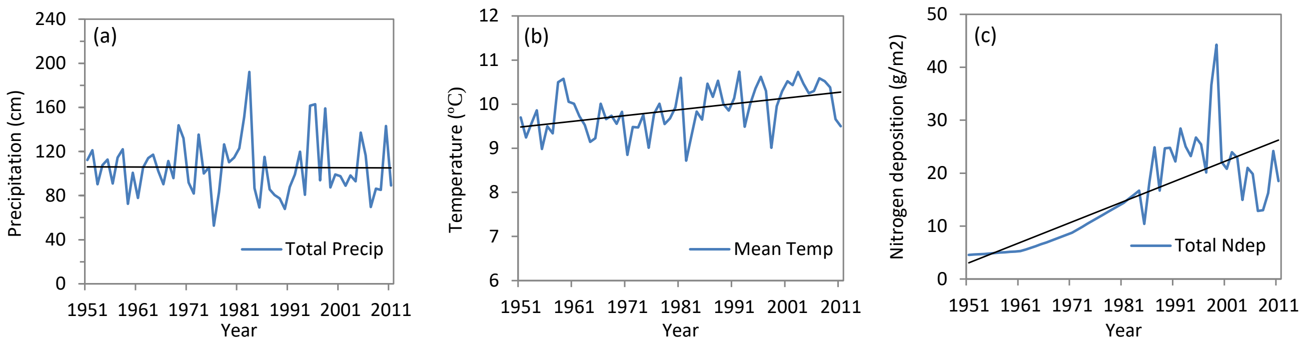

Figure 76a-c. (a) Total annual precipitation, (b) average annual temperature, and (c) total annual nitrogen deposition from 1951 through 2011 averaged across all national forests in the Pacific Southwest Region. Linear trend lines shown in black.

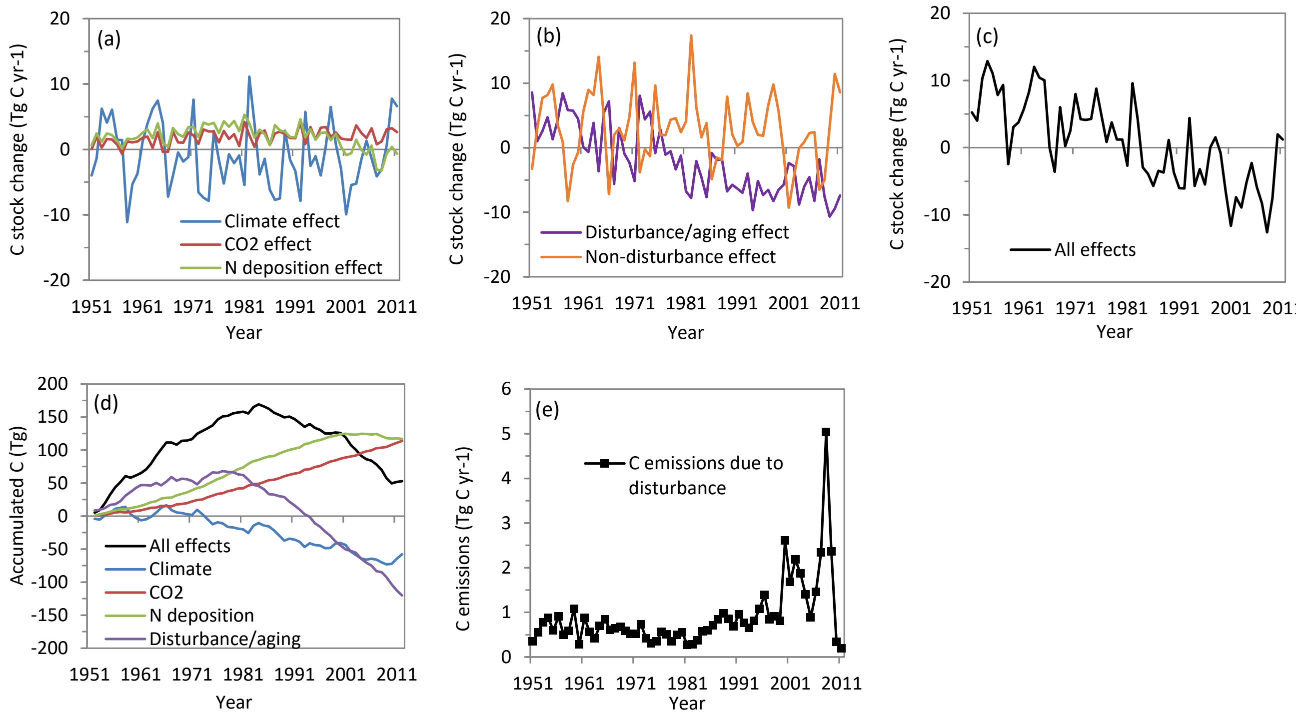

Figure 77a-e. Estimated forest carbon (C) changes and accumulations summed across the national forests in the Pacific Southwest Region. Changes in C stocks are attributed to: (a) individual nondisturbance factors alone, including climate variability, atmospheric carbon dioxide (CO2) concentration, and nitrogen deposition; (b) all disturbance and aging factors including fire, harvest, insects, and regrowth and aging, and the sum of all nondisturbance factors combined; and (c) all factors combined, which is the sum of disturbance and aging and nondisturbance effects. (d) Accumulated C due to individual disturbance and aging and nondisturbance factors and all factors combined from 1950 through 2011 excluding C accumulated pre-1950. (e) Direct C emissions to the atmosphere due to disturbance events only. Positive values in graphs (a) through (c) represent net C sinks from the atmosphere, or increases in C stocks, whereas negative values represent net C sources to the atmosphere, or decreases in C stocks.

The forest C dynamics in the Pacific Southwest Region have also been influenced by a recent increase in disturbances. Although the percentage of forest land harvested remained around 0.1 percent annually from 1991 to 2011, the percentage of forests burned by moderate-severity fires has greatly increased since 2000. In 2008 fires affected more than 2 percent of all forested areas in these national forests (fig. 70a). Despite this increase in disturbances, the stand-age distribution does not yet reflect an increase in regeneration (fig. 74).

While precipitation from 1951 to 2011 did not show a consistent trend, the 1982 to 1983 "super" El Niño event stands out as a period of significantly elevated precipitation (fig. 76a). Wetter conditions can enhance forest growth and C uptake. On the other hand, temperatures increased substantially from 1951 to 2011 (fig. 76b). Warmer temperatures can increase soil respiration and evaporative demands, leading to water stress and declines in net ecosystem productivity. Consequently, more C is released to the atmosphere. As climate continues to change, this region along with most of the United States is expected to get warmer while precipitation will continue to vary annually. Nitrogen deposition increased steadily from 1951 through the mid-1990s before spiking in 1998 to 1999 (fig. 76c). Nitrogen deposition then declined in the 2000s, possibly as the result of tighter environmental regulations or drought conditions decreasing the amount of wet deposition. Atmospheric CO_2 concentrations have risen dramatically and are expected to continue to rise for the foreseeable future.

The C stock changes and accumulations across national forests in this region show that forests generally experienced a switch from a C sink to a C source around the early 1980s (figs. 77a–c). As a result of this shift, C accumulation has been steadily declining since the mid-1980s (fig. 77d). Disturbance and aging effects have been mostly responsible for declining C stocks (fig. 77b), which can be attributed to the timing of Euro-American settlement, land use change, or potentially large fires that shaped forest structure (fig. 74). From the 1950s to the early 1980s, the forests were mostly a C sink due to positive disturbance and aging effects, as the early-1900s pulse of stands was middle-aged and therefore growing at maximum productivity. As these forests further aged, productivity declined and subsequently the rate of C accumulation declined. Meanwhile the effects of disturbance were increasing to the point where C emissions due to decomposition and disturbances exceeded C gains, and as a result the forests became a C source (figs. 77b,c). This decline was coupled with lower rates of stand establishment in the mid-1900s. Additionally, recent large fires caused pulses of C emissions to the atmosphere (fig. 77e), and disturbance and aging effects maintained a C source through 2011 (fig. 77b).

Climate has had a highly variable effect on changing C stocks following interannual variability in temperature and precipitation (figs. 76a,b). Generally climate had a positive effect on C stock change during cooler or wetter years such as 1971 and 2010, and especially during the El Niño of 1982 to 1983 (fig. 77a). On the other hand, climate effects caused a C source during warmer or drier years such as 1958 to 1959 and 2001. The increasing trend in temperatures has had a negative effect on C stocks and contributed to the switch to a C source (fig. 77a) and decline in C accumulation in recent decades (fig. 77d).

Increases in N deposition (fig. 76c) and atmospheric CO2 concentrations have consistently enhanced the C sink and helped to counteract the negative disturbance and aging and climate effects (figs. 77a,d). Without these positive effects from N deposition and CO_2 fertilization, forests would have experienced a net loss of C between 1950 and 2011. However, the N deposition effect decreased in the 2000s (fig. 77a) as N deposition levels were significantly reduced (fig. 76c). The increasing atmospheric CO2 concentrations have the potential to continue to counteract the negative disturbance and aging and climate effects into the future.

Although a few national forests in the Pacific Southwest Region had a net loss of forest C, most forests had C gains from 1950 to 2011, resulting in a regionwide gain of approximately 53 Tg of ecosystem C, including soil C, between 1950 and 2011 (fig. 77d).

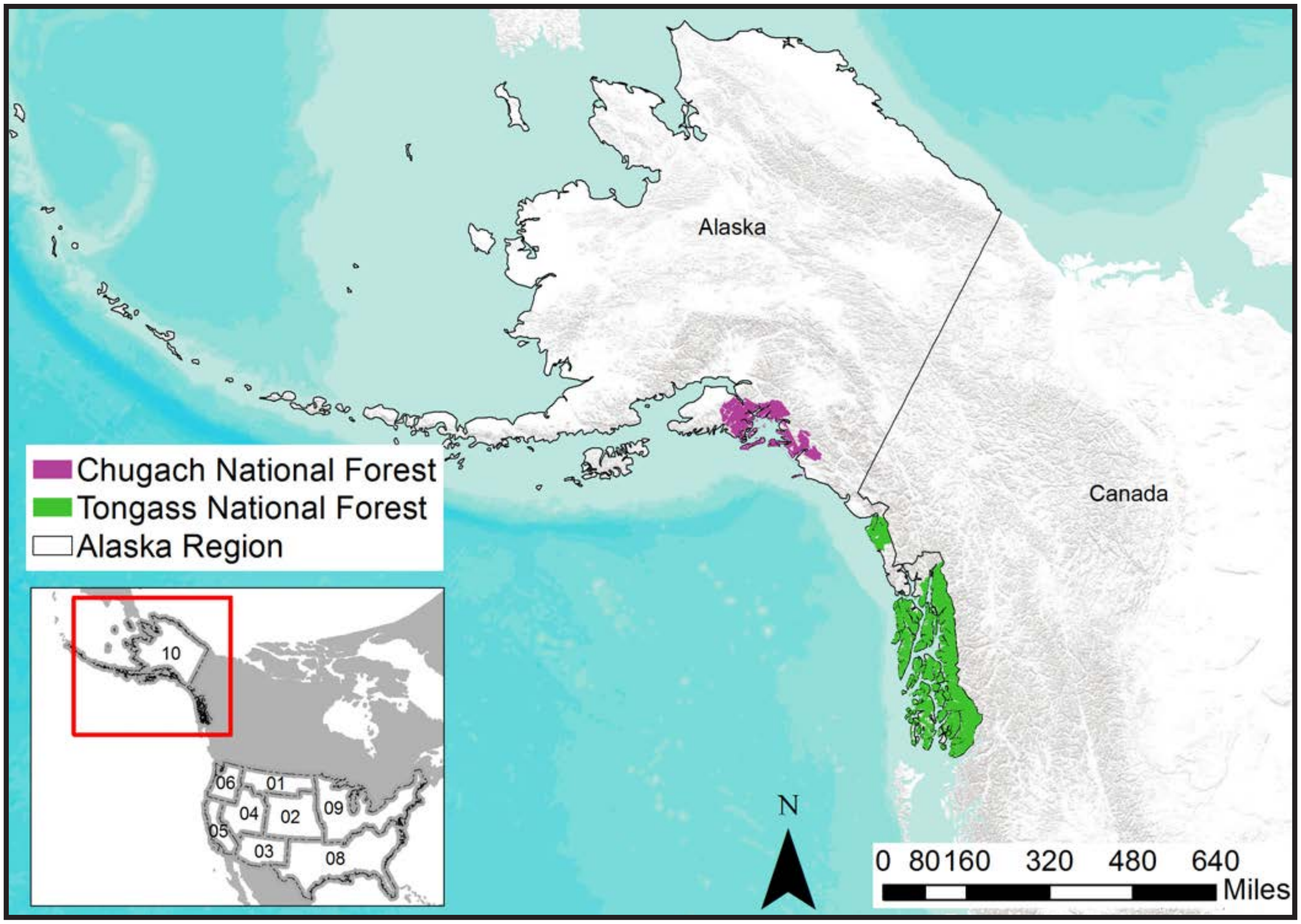

Figure 78. Locations of the national forests in the Alaska Region.

6.9 Alaska Region

6.9.1 Description of Region

The Alaska Region (also referred to as Region 10) in the National Forest System consists of two of the largest national forests in the United States. These units are located on the mainland and islands of southeastern Alaska (fig. 78). The analysis of factors contributing to C stocks and trends was conducted only on the forested areas of these national forests. To restrict the analysis to lands managed by the Forest Service, any private inholdings within national forest boundaries were excluded.

The history of Euro-American settlement, land use, and policies provides useful context for understanding the regional forest C trends that are indicated by the inventory and assessment results. The first Euro-American settlers came to Alaska for the fur trade, the salmon industry, and mining operations. Not until the Klondike gold discovery in 1888, however, was a major boom in settlement spawned in southeastern Alaska. Early timber operations generally supported local development and the needs of the fishing and mining industries but remained small-scale as the distance to prime markets hindered expansion. Despite the influence of the conservation movement in the early 1900s, which led to the establishment of forest reserves in Alaska (e.g., the Tongass and Chugach National Forests in 1907), the forest products industry grew steadily in southeastern Alaska (Haycox 2002).

Both World Wars caused an increase in harvesting operations to support aircraft manufacturing. In 1947, Congress passed the Tongass Timber Act, which authorized long-term timber contracts to support pulp mills in southern Alaska. Annual timber harvests consequently quadrupled in the 1950s. Most of the commercial timber harvests in southeastern Alaska had historically occurred in high-volume old-growth stands. Ecological concerns over the value of intact old-growth forest led to a policy shift from mandated timber harvesting to multiple resource use management of the national forests in the Alaska Region (Sisk 2007). Overall, annual timber harvests in the Alaska Region peaked in the 1970s and late 1980s, but declined precipitously in the 1990s. Since 2000 they have remained at the low levels of the early 1900s (Loeffler et al. 2014a).

Aside from the timber industry, the forest C legacy of the Alaska Region is tied to the history of natural disturbances. Unlike much of the western United States, fires are relatively minor. Since 1900, few fires in southeastern Alaska have burned more than 100 acres (40 hectares), as a result of the cool, moist, temperate-rainforest conditions. In contrast, wind is a common and highly destructive disturbance agent within these coastal forests. The most damaging windstorm was the 1968 Thanksgiving Day storm, which was characterized by up to 100 mile per hour (160 km/hr) winds and left some 1 billion board feet (2 million m^3) of timber in a single concentrated blowdown as well as several smaller scattered blowdowns (Harris and Farr 1974).

In addition to wind, insects such as the spruce beetle are a key disturbance agent in the Alaska Region (Bentz et al. 2010). From 1990 to 2000 an extensive spruce beetle outbreak resulted in the mortality of about 4.7 million acres (1.9 million hectares) of forests in south-central Alaska. Warming temperatures increase the susceptibility to spruce beetle outbreaks while also altering the life-cycle of spruce beetles, thus increasing their numbers

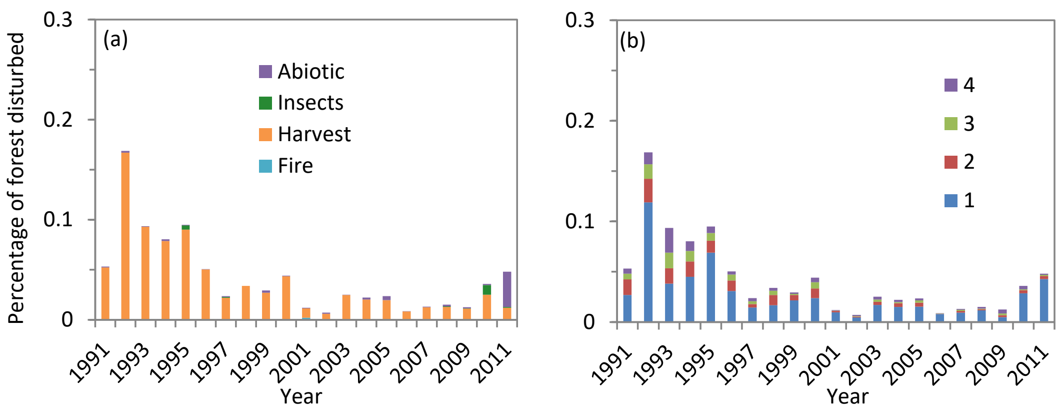

Figure 79a-b. Annual rates of disturbance (0 to 0.3 percent) in the Alaska Region, mapped using visual interpretation of several independent datasets and summarized as the percentage of the forested area disturbed from 1991 through 2011 by (a) disturbance types including fire, harvests, insects, and abiotic; and b) magnitude classes, characterized by percentage change in canopy cover (CC) and categorized as follows: (1) 0 to 25 percent CC, (2) 25 to 50 percent CC, (3) 50 to 75 percent CC, and (4) 75 to 100 percent CC.

(Werner et al. 2006). Due to its high latitude, Alaska has seen a more rapid increase in temperatures during the last century than any other U.S. region, so climate change impacts on the region's forests are especially intense (Haufler et al. 2010).

6.9.2 Disturbance Trends

Mapped rates of fire, harvest, and insect activity and the intensity of these disturbances (change in canopy cover) for the Alaska Region are displayed in figure 79. The disturbance maps were derived from Landsat satellite imagery; a systematic process of manual editing and attribution of disturbance type using spatial records of harvests, fire, and insects; and high- and medium-resolution imagery. Harvest was the most common and consistent type of forest disturbance in the region although harvest rates exceeded 0.1 percent of the landscape in only 1 year during the study period and there was a general downward trend from the early 1990s to 2011 (fig. 79a). Low-level insect disturbance occurred throughout the period, and storm damage was relatively extensive in 2011. During this period, more than 60 percent of the disturbed area was characterized as low intensity (<25 percent change in canopy cover) (fig. 79b).

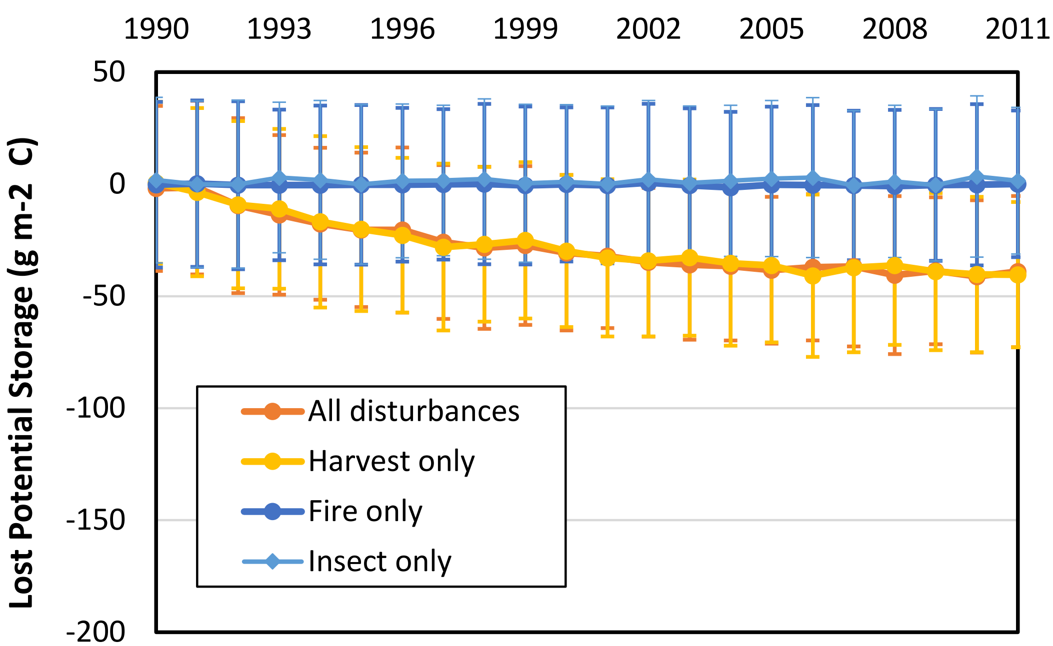

Figure 80. The impact of different kinds of disturbance, occurring from 1990 through 2011, on carbon (C) stores in the Alaska Region. The difference in storage for each year is shown between an "undisturbed" scenario and a scenario that includes only observed amounts of the specified type of disturbance. Error bars represent a 95-percent confidence interval; 100 g/m^2 equals 1 metric tonne (or Mg)/ha.

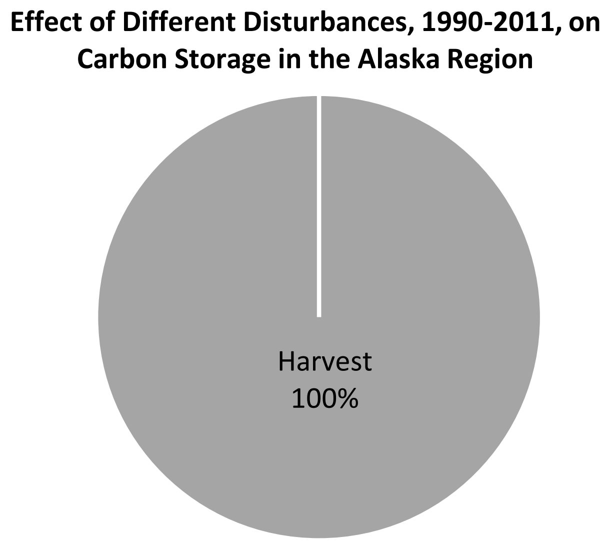

Figure 81. Proportional effect of different kinds of disturbance on carbon storage in both national forests in the Alaska Region for the period 1990 through 2011.

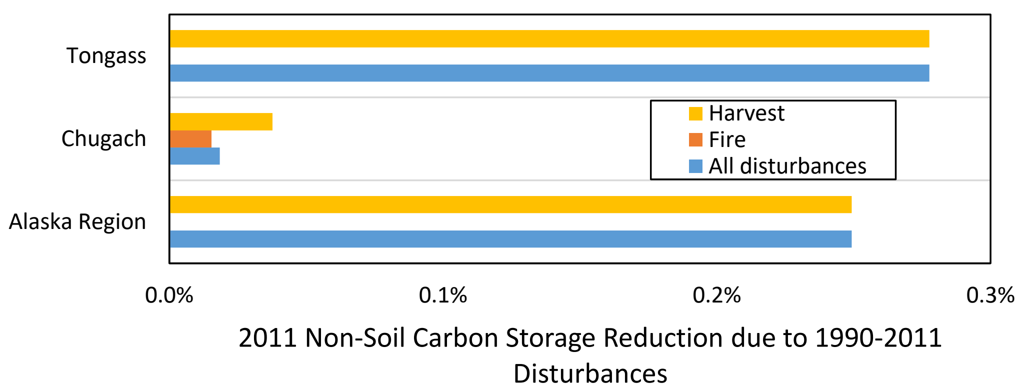

Figure 82. Carbon stock reduction in 2011 due to disturbances occurring from 1990 through 2011, by each national forest and for all national forests combined in the Alaska Region. Percent reduction represents how much nonsoil carbon was lost from the baseline forest inventory carbon stock estimates.

6.9.3 Effects of Disturbance and Management Activities

The primary purpose of ForCaMF is to estimate the relative impacts of different kinds of disturbance in the last two decades on current C stocks, displayed for the Alaska Region in figure 80. The impact of disturbance (D_F, as defined in eq. 1) is expressed in relation to the amount of C that would have been stored in the absence of the particular disturbance process. This means that the impact of a disturbance is felt beyond the year it happens. If fires first occur in 1992, for example, the line of fire in figure 80 will only start to diverge from zero in that year. The fire line may continue to diverge because ForCaMF's accounting of postfire C stocks will (realistically) reflect gradual release of fire-killed material, partially offsetting C gained through regrowth and persistently increasing the difference between postfire C storage and storage likely under an undisturbed scenario. Thus, figure 80 reflects the real-time impact of disturbance on the forest's ability to store C in relation to storage likely in the absence of disturbance.

Harvest activity was effectively the only disturbance process to have an impact on C stocks in the region (figs. 80, 81); the detected fire and insect activity on the two national forests had negligible impact on C stocks at the regional level. Harvest impact, though larger than the other types of disturbance, was among the smallest of any of the regions because cutting generally affected less than 0.05 percent of forest land per year (fig 79). Harvests occurring between 1900 and 2011 resulted in a 0.2-percent reduction in 2011 nonsoil C storage (fig. 82).

One caveat must be stated about the confidence intervals depicted in figure 80. The unit of analysis for ForCaMF was the national forest (Appendix 11: Alaska Region, (PDF, 1.0MB)); rates of simulated error were constrained by FIA estimates at that level. Regional aggregation used a secondary Monte Carlo analysis based on the uncertainty of each individual forest result, and it assumed that the errors in one forest were independent of errors in the other forest. This assumption very likely overstated certainty slightly at the regional level; if initial biomass estimates in one forest were high, for example, the same estimates were also likely to be biased high in the other unit. This caveat does not change the overall trends seen at the aggregated regional level, nor does it affect confidence intervals assessed at the national forest level. Harvests were clearly the most important disturbance process with respect to C storage, although fire and wind were locally important.

6.9.4 Effects of Disturbance, Management, and Environmental Factors

The InTEC model was not applied to the national forests in Alaska.

7. References

Agee, J.K. 1998. The landscape ecology of western forest fire regimes. Northwest Science. 72(spec. issue): 24–34.

Ager, A.; Vaillant, N.F.; Finney, M.A. 2010. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. Forest Ecology and Management. 259(8): 1556–1570.

Anderegg, W.R.L.; Schwalm, C.; Biondi, F.; [et al.]. 2015. Pervasive drought legacies in forest ecosystems and their implications for carbon cycle models. Science. 349(6247): 528–532.

Ault, T.R.; Cole, J.E.; Overpeck, J.T.; [et al.]. 2014. Assessing the risk of persistent drought using climate model simulations and paleoclimate data. Journal of Climate. 27(20): 7529–7549.

Bach, M. 1990. History of the Fremont National Forest. [Tonsfeldt, W., ed.]. U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Fremont National Forest. 309 p.

Baker, R.D.; Burt, L.; Maxwell, R.S.; [et al.]. 1993. The national forests of the Northern Region: Living legacy. College Station, TX: Intaglio Inc. 348 p.

Baker, R.D.; Maxwell, R.S.; Treat, V.H.; Dethloff, H.C. 1988. Timeless heritage: A history of the Forest Service in the Southwest. U.S. Department of Agriculture, Forest Service. FS-409. College Station, TX: Intaglio, Inc. 271 p.

Barrett, S.W.; Arno, S.F.; Menakas, J.P. 1997. Fire episodes in the inland Northwest (1540–1940) based on fire history data. Gen. Tech. Rep. GTR-INT-370. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 17 p.

Beaty, M.R.; Taylor, A.H. 2008. Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. Forest Ecology and Management. 255(3–4): 707–719.

Bentz, B.J.; Regniere, J.; Fettig, C.J.; [et al.]. 2010. Climate change and bark beetles of the western United States and Canada: Direct and indirect effects. BioScience. 60(8): 602–613.

Butler, E.; Stockmann, K.; Anderson, N.; [et al.]. 2014a. Estimates of carbon stored in harvested wood products from United States Forest Service Pacific Northwest Region, 1909–2012. Unpublished report. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 28 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_butler_e001.pdf [Accessed August 9, 2019].

Butler, E.; Stockmann, K.; Anderson, N.; [et al.]. 2014b. Estimates of carbon stored in harvested wood products from United States Forest Service Southwestern Region, 1909–2012. Unpublished report. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 27 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_butler_e002.pdf [Accessed September 6, 2019].

Byler, J.W.; Hagle, S.K. 2000. Succession functions of forest pathogens and insects: Ecosections M332a and M333d in Northern Idaho and Western Montana. FHP Report 00-09. U.S. Department of Agriculture, Forest Service, State and Private Forestry, Forest Health Protection. 43 p.

Byler, J.W.; Marsden, M.A.; Hagle, S.K. 1990. The probability of root disease on the Lolo National Forest, Montana. Canadian Journal of Forest Research. 20(7): 987–994.

Chen, J.M.; Chen, W.; Cihlar, J. 2000a. Integrated terrestrial ecosystem carbon-budget model based on changes in disturbance, climate, and atmospheric chemistry. Ecological Modelling. 135(1): 55–79.

Chen, J.M.; Chen, W.; Liu, J.; Cihlar, J. 2000b. Annual carbon balance of Canada's forests during 1895–1996. Global Biogeochemical Cycles. 14(3): 839–850.

Chen, J.M.; Chen, W.; Liu, J.; Cihlar, J. 2000c. Approaches for reducing uncertainties in regional forest carbon balance. Global Biogeochemical Cycles. 14(3): 827–838.

Chen, J.M.; Ju, W.; Cihlar, J.; [et al.]. 2003. Spatial distribution of carbon sources and sinks in Canada's forests. Tellus. 55B: 622–641.

Climate Impacts Group. 2009. The Washington climate change impacts assessment: Evaluating Washington's future in a changing climate. [McGuire Elsner, M.; Littell, J.; Whitely Binder, L., eds.]. Seattle, WA: University of Washington, Joint Institute for the Study of the Atmosphere and Oceans, Center for Science in the Earth System. 407 p.

Cohen, W.B.; Yang, Z.; Kennedy, R. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 2. TimeSync—Tools for calibration and validation. Remote Sensing of Environment. 114: 2911–2924.

Cohen, W.B.; Yang, Z.; Stehman, S.V.; [et al.]. 2016. Forest disturbance across the conterminous United States from 1985–2012: The emerging dominance of forest decline. Forest Ecology and Management. 360: 242–252.

Conrad, D.E. 1997. The land we cared for... A history of the Forest Service's Eastern Region. 1st ed. [Cravens, J.H., ed.]. Milwaukee, WI: U.S. Department of Agriculture, Forest Service, Region 9. 312 p.

Covington, W.W.; Moore, M.M. 1994. Southwestern ponderosa forest structure: Changes since Euro-American settlement. Journal of Forestry. 92(1): 39–47.

Crookston, N.L.; Dixon, G.E. 2005. The forest vegetation simulator: A review of its structure, content, and applications. Computers and Electronics in Agriculture. 49: 60–80.

Dale, V.H.; Joyce, L.A.; McNulty, S.; [et al.]. 2001. Climate change and forest disturbances. BioScience. 51(9): 723–734.

Dugan, A.J.; Birdsey, R.; Healey, S.P.; [et al.]. 2017. Forest sector carbon analyses support land management planning and projects: Assessing the influence of anthropogenic and natural factors. Climatic Change. 144(2): 207–220.

Dukes, J.S.; Pontius, J.; Orwig, D. [et al.]. 2009. Responses of insect pests, pathogens, and invasive plant species to climate change in the forests of northeastern North America: What can we predict? Canadian Journal of Forest Research. 39(2): 231–248.

Goeking, S.A. 2015. Disentangling forest change from forest inventory change: A case study from the US Interior West. Journal of Forestry. 113(5): 475–483.

Guarín, A.; Taylor, A.H. 2005. Drought triggered tree mortality in mixed conifer forests in Yosemite National Park, California, USA. Forest Ecology and Management. 218(1–3): 229–244.

Harris, A.S.; Farr, W.A. 1974. The forest ecosystem of southeast Alaska. 7. Forest ecology and timber management. Gen. Tech. Rep. PNW-025. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 116 p.

Haufler, J.B.; Mehl, C.A.; Yeats, S. 2010. Climate change: Anticipated effects on ecosystem services and potential actions by the Alaska Region, U.S. Forest Service. Seeley Lake, MT: U.S. Department of Agriculture, Forest Service, Ecosystem Management Research Institute. 53 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_038171.pdf [Accessed August 9, 2019].

Haycox, S. 2002. Alaska: An American colony. Seattle, WA: University of Washington Press. 392 p.

He, L.; Chen, J.M.; Pan, Y.; Birdsey, R.A. 2012. Relationships between net primary productivity and forest stand age derived from Forest Inventory and Analysis data and remote sensing imagery. Global Biogeochemical Cycles. 26(3): GB3009.

Healey, S.P.; Cohen, W.B.; Spies, T.A.; [et al.]. 2008. The relative impact of harvest and fire upon landscape-level dynamics of older forests: Lessons from the Northwest Forest Plan. Ecosystems. 11(7): 1106–1119.

Healey, S.P.; Cohen, W.B.; Yang, Z.; [et al.]. 2018. Mapping forest change using stacked generalization: An ensemble approach. Remote Sensing of Environment. 204: 717–728.

Healey, S.P.; Cohen, W.B.; Yang, Z.Q.; Krankina, O.N. 2005. Comparison of tasseled cap-based Landsat data structures for use in forest disturbance detection. Remote Sensing of Environment. 97(3): 301–310.

Healey, S.P.; Morgan, T.A.; Songster, J.; Brandt, J. 2009. Determining landscape-level carbon emissions from historically harvested forest products. In: McWilliams, W.; Moisen, G.G.; Czaplewski, R., comps. Proceedings of the 2009 FIA symposium; October 21–23, 2008; Park City, UT. RMRS-P-56CD. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. [1 CD]. 11 p.

Healey, S.P.; Raymond, C.L.; Lockman, I.B.; [et al.]. 2016. Root disease can rival fire and harvest in reducing forest carbon storage. Ecosphere. 7(11): 1–16.

Healey, S.P.; Urbanski, S.P.; Patterson, P.L.; Garrard, C. 2014. A framework for simulating map error in ecosystem models. Remote Sensing of Environment. 150: 207–217.

Heath, L.S.; Smith, J.E.; Birdsey, R.A. 2003. Carbon trends in US forest lands: A context for the role of soils in forest carbon sequestration. In: Kimble, J.M.; Heath, L.S.; Birdsey, R.A.; Lal, R.; eds. The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. Boca Raton, FL: CRC Press: 35–45.

Hernandez, A.; Healey, S.; Huang, H.; Ramsey, R. 2018. Improved prediction of stream flow based on updating land cover maps with remotely sensed forest change detection. Forests. 9(6): 1–19.

Hoover, C.M.; Leak, W.B.; Keel, B.G. 2012. Benchmark carbon stocks from old-growth forests in northern New England, USA. Forest Ecology and Management. 266: 108–114.

Hoover, C.M.; Rebain, S.A. 2011. Forest carbon estimation using the Forest Vegetation Simulator: Seven things you need to know. Gen. Tech. Rep. NRS-77. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 16 p.

Huang, C.; Goward, S.N.; Masek, J.G.; [et al.]. 2010. An automated approach for reconstructing recent forest disturbance history using dense Landsat time series stacks. Remote Sensing of Environment. 114(1): 183–198.

Hyvönen, R.; Ågren, G.I.; Linder, S.; [et al.]. 2007. The likely impact of elevated [CO2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: A literature review. New Phytologist. 173(3): 463–480.

Intergovernmental Panel on Climate Change [IPCC]. 2006. 2006 IPCC guidelines for national greenhouse gas inventories. [Prepared by the National Greenhouse Gas Inventories Programme; Eggleston, H.S.; Buendia L.; Miwa K.; [et al.], eds.]. Hayana, Japan: Institute for Global Environmental Strategies. https://www.ipcc-nggip.iges.or.jp/public/2006gl/ [Accessed August 29, 2019].

Intergovernmental Panel on Climate Change [IPCC]. 2007. Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. [Solomon, S.; Qin, D.; Manning, M.; [et al.], eds.]. Cambridge, UK and New York, NY: Cambridge University Press. 996 p.

Janowiak, M.; Connelly, W.J.; Dante-Wood, K.; [et al.]. 2017. Considering forest and grassland carbon in land management. Gen. Tech. Rep. WO-95. Washington, DC: U.S. Department of Agriculture, Forest Service. 68 p.

Johnson, E.W.; Wittwer, D. 2008. Aerial detection surveys in the United States. Australian Forestry. 71(3): 212–215.

Ju, W.M.; Chen, J.M.; Harvey, D.; Wang, S. 2007. Future carbon balance of China's forests under climate change and increasing CO2. Journal of Environmental Management. 85(3): 538–562.

Keeling, R.F.; Piper, S.C.; Bollenbacher, A.F.; Walker, S.J. 2009. Atmospheric CO2 records from sites in the SIO air sampling network. In: Trends: A compendium of data on global change. Oak Ridge, TN: U.S. Department of Energy, Oak Ridge National Laboratory, Carbon Dioxide Information Analysis Center.

Keenan, T.F.; Hollinger, D.Y.; Bohrer, G.; [et al.]. 2013. Increase in forest water-use efficiency as atmospheric carbon dioxide concentrations rise. Nature. 499(7458): 324–327.

Kulhavy, D.; Ross, W.G. 1988. Southern pine beetle and fire in wilderness areas: The Kisatchie Hills Wilderness, Kisatchie National Forest. Faculty Publications. 397. https://scholarworks.sfasu.edu/forestry/397 [Accessed September 6, 2019].

Kurz, W.A.; Dymond, C.C.; Stinson, G.; [et al.]. 2008. Mountain pine beetle and forest carbon feedback to climate change. Nature. 452(7190): 987–990.

Law, B. 2013. Biogeochemistry: Nitrogen deposition and forest carbon. Nature. 496(7445): 307–308.

Le Quéré, C.; Andrew, R.M.; Friedlingstein, P.; [et al.]. 2018. Global climate budget 2017. Earth System Science Data. 10: 405–448. https://www.doi.org/10.5194/essd-10-405-2018.

Loeffler, D.; Anderson, N.; Stockmann, K.; [et al.]. 2014a. Estimates of carbon stored in harvested wood products from United States Forest Service Alaska Region, 1911–2012. Unpublished report. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 26 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_loeffler_d002.pdf [Accessed September 6, 2019].

Loeffler, D.; Anderson, N.; Stockmann, K.; [et al.]. 2014b. Estimates of carbon stored in harvested wood products from United States Forest Service Eastern Region, 1911–2012. Unpublished report. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 27 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_loeffler_d003.pdf [Accessed September 6, 2019].

Loeffler, D.; Anderson, N.; Stockmann, K.; [et al.]. 2014c. Estimates of carbon stored in harvested wood products from United States Forest Service Southern Region, 1911–2012. Unpublished report. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 27 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_loeffler_d004.pdf [Accessed September 6, 2019].

Mattson, W.J.; Shriner, D.S. 2001. Northern Minnesota Independence Day storm: A research needs assessment. Gen. Tech. Rep. NC-216. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Research Station.

McKinley, D.; Ryan, M.; Birdsey, R.; [et al.]. 2011. A synthesis of current knowledge on forests and carbon storage in the United States. Ecological Applications. 21(6): 1902–1924.

Millar, C.I.; Stephenson, N.L. 2015. Temperate forest health in an era of emerging megadisturbance. Science. 349(6250): 823–826.

Miller, J.D.; Safford, H.D.; Crimmins, M.; Thode, A.E. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. Ecosystems. 12(1): 16–32.

Morris, W.G. 1936. The Tillamook Burn—Its area and timber volume. Forest Research Notes 18. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest Experimental Station. 11 p.

Nelson, M.D.; Healey, S.P.; Moser, W.K.; Hansen, M.H. 2009. Combining satellite imagery with forest inventory data to assess damage severity following a major blowdown event in northern Minnesota, USA. International Journal of Remote Sensing. 30(19): 5089–5108.

Nemani, R.; White, M.; Thornton, P.; [et al.]. 2002. Recent trends in hydrologic balance have enhanced the terrestrial carbon sink in the United States. Geophysical Research Letters. 29(10): 106-1 to 106-4.

O'Connell, B.M.; LaPoint, E.B.; Turner, J.A.; [et al.]. 2014. The Forest Inventory and Analysis Database: Database description and user guide, ver. 6.0.1 for Phase 2. Washington, DC: U.S. Department of Agriculture, Forest Service. 748 p. https://www.fia.fs.fed.us/library/database-documentation/historic/ver6/FIADB User Guide P2_6-0-1_final.pdf [Accessed August 29, 2019].

Pan, Y.; Birdsey, R.; Hom, J.; McCullough, K. 2009. Separating effects of changes in atmospheric composition, climate and land-use on carbon sequestration of U.S. Mid-Atlantic temperate forests. Forest Ecology and Management. 259(2): 151–164.

Pan, Y.; Birdsey, R.A.; Fang, J.; [et al.]. 2011. A large and persistent carbon sink in the world's forests. Science. 333(6045): 988–993.

Perez-Garcia, J.; Lippke, B.; Comnick, J.; Manriquez, C. 2005. An assessment of carbon pools, storage, and wood products market substitution using life-cycle analysis results. Wood and Fiber Science. 37(CORRIM Special Issue): 140–148.

Pregitzer, K.; Euskirchen, E. 2004. Carbon cycling and storage in world forests: Biome patterns related to forest age. Global Change Biology. 10(12): 2052–2077.

PRISM Climate Group. 2004. PRISM climate data. Oregon State University. http://prism.oregonstate.edu [Accessed March 1, 2013].

Pyne, S.J. 1982. Fire in America: A cultural history of wildland and rural fire. Princeton, NJ: Princeton University Press. 654 p.

Raymond, C.L.; Healey, S.; Peduzzi, A.; Patterson, P. 2015. Representative regional models of post-disturbance forest carbon accumulation: Integrating inventory data and a growth and yield model. Forest Ecology and Management. 336: 21–34.

Rebain, S.A., comp. 2010 (revised March 23, 2015). The Fire and Fuels Extension to the Forest Vegetation Simulator: Updated model documentation. Internal Rep. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Forest Management Service Center. 403 p.

Reinhardt, E.; Crookston, N.L., tech. eds. 2003. The Fire and Fuels Extension to the Forest Vegetation Simulator. Gen. Tech. Rep. RMRS-GTR-116. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 209 p.

Romme, W.H.; Floyd, L.M.; Hanna, D. 2009. Historical range of variability and current landscape condition analysis: South Central Highlands Section, Southwestern Colorado & Northwestern New Mexico. Colorado State University, Colorado Forest Restoration Institute and U.S. Forest Service, Region 2. 256 p.

Ruefenacht, B.; Finco, M.V.; Nelson, M.D.; [et al.]. 2008. Conterminous US and Alaska forest type mapping using forest inventory and analysis data. Photogrammetric Engineering & Remote Sensing. 74(11): 1379–1388.

Schroeder, T.A.; Healey, S.P.; Moisen, G.G. [et al.]. 2014. Improving estimates of forest disturbance by combining observations from Landsat time series with U.S. Forest Service Forest Inventory and Analysis data. Remote Sensing of Environment. 154: 61–73.

Schwind, B.; Brewer, K.; Quayle, B.; Eidenshink, J.C. 2010. Establishing a nationwide baseline of historical burn-severity data to support monitoring of trends in wildfire effects and national fire policies. In: Pye, J.M.; Rauscher, H.; Sands, Y. [et al.], eds. Advances in threat assessment and their application to forest and rangeland management. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 381–396.

Shands, W.E. 1992. The lands nobody wanted: The legacy of the eastern national forests. In: Steen, H.K., ed. The origins of the national forests: A centennial symposium. Durham, NC: Forest History Society: 19–44.

Sheffield, R.M.; Thompson, M.T. 1992. Hurricane Hugo: Effects on South Carolina's forest resource. Res. Pap. SE-284. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 60 p.

Shepperd, W.D.; Battaglia, M.A. 2002. Ecology, silviculture, and management of Black Hills ponderosa pine. Gen. Tech. Rep. RMRS-GTR-97. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 112 p.

Sisk, J. 2007. The southeastern Alaska timber industry: Historical overview and current status. In: Schoen, J.; Dovichin, E., eds. 2007. The coastal forests and mountain ecoregion of southeastern Alaska and the Tongass National Forest. Anchorage, AK: Audubon Alaska and The Nature Conservancy: Ch. 9.6: 1–20.

Stanturf, J.A.; Wade, D.D.; Waldrop, T.A.; [et al.]. (Background Paper FIRE): Fire in southern forest landscapes. In: Wear, D.N.; Greis, J.G., eds. 2002. Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 607–630. Chapter 25.

Stockmann, K.D.; Anderson, N.M.; Skog, K.E.; [et al.]. 2012. Estimates of carbon stored in harvested wood products from the United States Forest Service northern region. Carbon Balance and Management. 7: 1–16.

Stockmann, K.; Anderson, N.; Young, J.; [et al.]. 2014a. Estimates of carbon stored in harvested wood products from United States Forest Service Intermountain Region, 1911–2012. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 28 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_stockmann_k001.pdf [Accessed September 6, 2019].

Stockmann, K.; Anderson, N.; Young, J.; [et al.]. 2014b. Estimates of carbon stored in harvested wood products from United States Forest Service Northern Region, 1906–2012. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 27 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_stockmann_k002.pdf [Accessed September 6, 2019].

Stockmann, K.; Anderson, N.; Young, J.; [et al.]. 2014c. Estimates of carbon stored in harvested wood products from United States Forest Service Pacific Southwest Region, 1906–2012. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 28 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_stockmann_k003.pdf [Accessed September 6, 2019].

Stockmann, K.; Anderson, N.; Young, J.; [et al.]. 2014d. Estimates of carbon stored in harvested wood products from United States Forest Service Rocky Mountain Region, 1906–2012. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory. 27 p. https://www.fs.fed.us/rm/pubs_other/rmrs_2014_stockmann_k004.pdf [Accessed September 6, 2019].

Thomas, J.W.; Franklin, J.F.; Gordon, J.; Johnson, K.N. 2006. The Northwest Forest Plan: Origins, components, implementation experience, and suggestions for change. Conservation Biology. 20(2): 277–287.

Thomas, N.E.; Huang, C.; Goward, S.N.; [et al.]. 2011. Validation of North American forest disturbance dynamics derived from Landsat time series stacks. Remote Sensing of Environment. 115(1): 19–32.

Tongass Timber Act. 1947. Public Law 80-385. 61 Stat. 920 (not codified).

USDA Forest Service [USDA FS]. 1927. The national forests of California. Misc. Circular 94. Washington, DC. https://foresthistory.org/wp-content/uploads/2017/01/TheNationalForestsOfCalifornia-1927.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 1930. What the national forests mean to the Intermountain Region. USDA Misc. Circular 47. Washington, DC. 22 p.

USDA Forest Service [USDA FS]. 2015a. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Alaska Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 41 p. https://www.fs.fed.us/climatechange/documents/AlaskaRegionCarbonAssessmentTwoBaselines.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015b. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Eastern Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 58 p. https://www.fs.fed.us/climatechange/documents/EasternRegionCarbonAssessmentTwoBaselines.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015c. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Intermountain Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 55 p. https://www.fs.fed.us/climatechange/documents/Intermountain RegionCarbonAssessmentTwoBaselines.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015d. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Northern Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 56 p. https://www.fs.fed.us/climatechange/documents/NorthernRegionCarbonAssessment TwoBaselines.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015e. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Pacific Northwest Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 66 p. https://www.fs.fed.us/climatechange/documents/Pacific Northwest RegionCarbonAssessmentTwoBaselines.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015f. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Pacific Southwest Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 61 p. https://www.fs.fed.us/climatechange/documents/PacificSouthwest RegionCarbonAssessmentTwoBaselines.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015g. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Rocky Mountain Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 54 p. https://www.fs.fed.us/climatechange/documents/Rocky MountainRegion-CarbonAssessmentTwoBaselines.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015h. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Southern Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 60 p. https://www.fs.fed.us/climatechange/documents/SouthernRegionCarbonAssessment TwoBaselines.pdf [Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015i. Baseline estimates of carbon stocks in forests and harvested wood Products for National Forest System units; Alaska Region. Washington, DC: U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis. 53 p. https://www.fs.fed.us/managing-land/sc/carbon[Accessed August 8, 2019].

USDA Forest Service [USDA FS]. 2015j. FSVeg common stand exam user guide ver. 2.12.6. Washington, DC: U.S. Department of Agriculture, Forest Service, Natural Resource Manager. http://www.fs.fed.us/nrm/fsveg/ [Accessed September 1, 2019].

USDA Forest Service [USDA FS]. 2016. Forest Inventory and Analysis Database. St. Paul, MN: U.S. Department of Agriculture, Forest Service, Northern Research Station https://

apps.fs.usda.gov/fia/datamart/datamart.html [Accessed February 15, 2016].

U.S. Environmental Protection Agency [USEPA]. 2018. Inventory of U.S. greenhouse gas emissions and sinks: 1990–2016. EPA 430-R-18-003. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks [Accessed August 8, 2019].

U.S. Global Change Research Program [USGCRP]. 2017. Climate science special report: Fourth National Climate Assessment, vol. I. [Wuebbles, D.J.; Fahey, D.W.; Hibbard, K.A.; [et al.], eds.]. Washington, DC: U.S. Global Change Research Program. 666 p. https://science2017.globalchange.gov [Accessed August 8, 2019].

van Mantgem, P.J.; Stephenson, N.L.; Byrne, J.C. [et al.]. 2009. Widespread increase of tree mortality rates in the western United States. Science. 323: 521–524.

Vaughn, D.H. 2013. Derecho! The forgotten windstorm that changed the Ozarks. Forest History Today. Spring/Fall: 4–12.

Walsh, J.; Wuebbles, D.; Hayhoe, K.; [et al.]. 2014. Our changing climate. In: USGCRP. Climate change impacts in the United States: The Third National Climate Assessment. [Melillo, J.M.; Richmond, T.C.; Yohe, G.W., eds.]. Washington, DC: U.S. Global Change Research Program: 19–67. Chapter 2.

Way, D.A.; Oren, R. 2010. Differential responses to changes in growth temperature between trees from different functional groups and biomes: A review and synthesis of data. Tree Physiology. 30(6): 669–688.

Wear, D.N.; Coulston, J.W. 2015. From sink to source: Regional variation in U.S. forest carbon futures. Scientific Reports. 5: Article number 16518. https://doi.org/10.1038/srep16518.

Werner, R.A.; Holsten, E.H.; Matsuoka, S.M.; Burnside, R.E. 2006. Spruce beetles and forest ecosystems in south-central Alaska: A review of 30 years of research. Forest Ecology and Management. 227(3): 195–206.

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science. 313(5789): 940–943.

Williams, G.W. 2003. The beginnings of the national forests in the South: Protection of watersheds. Paper presented at the14th Annual Environment Virginia Conference; April 29–May 1, 2003; Lexington, VA. Revised version available at https://foresthistory.org/wp-content/uploads/2017/01/ProtectionofWatersheds_Willimas.pdf [Accessed August 8, 2019].

Williams, M. 1989. Americans and their forests: A historical geography. New York, NY: Cambridge University Press. 599 p.

Williams, M.A.; Baker, W.L. 2012. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests.

Global Ecology and Biogeography. 21(10): 1042–1052.

Woodall, C.W.; Heath, L.S.; Domke, G.M.; Nichols, M.C. 2011. Methods and equations for estimating aboveground volume, biomass, and carbon for trees in the U.S. forest inventory, 2010. Gen. Tech. Rep. NRS-88. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 30 p.

Zhang, F., Chen, J.M.; Pan, Y.; [et al.]. 2012. Attributing carbon changes in conterminous U.S. forests to disturbance and non-disturbance factors from 1901–2010. Journal of Geophysical Research. 117: G02021. https://doi.org/10.1029/2011jg001930.

Zhang, F.; Chen, J.M.; Pan, Y.; [et al.]. 2015. Impacts of inadequate historical disturbance data in the 20th century on modeling recent carbon dynamics (1951–2010) in conterminous US forests. Journal of Geophysical Research: Biogeosciences. 120: 549–569.

Appendix 1: Description of Forest-Type Group and Forest Dominance-Type Aggregations, and Common and Scientific Names of Flora and Fauna Mentioned in this Report Forest Inventory and Analysis Forest-Type Groups

Following are forest types and forest-type groups as defined by the Forest Inventory and Analysis (FIA) program of the Forest Service, Department of Agriculture (Burrill et al. 2018). An FIA database code is also associated with each forest type or forest-type group. Carbon modeling in all national forest system regions, except for the Northern Region, used this FIA forest-type group classification. Scientific names accompany each genus or species.

100 White/red/jack pine group
- 101 Jack pine (*Pinus banksiana*)
- 102 Red pine (*Pinus resinosa*)
- 103 Eastern white pine (*Pinus strobus*)
- 104 Eastern white pine/eastern hemlock (*Pinus strobus/Tsuga canadensis*)
- 105 Eastern hemlock (*Tsuga canadensis*)

120 Spruce/fir group
- 121 Balsam fir (*Abies balsamea*)
- 122 White spruce (*Picea glauca*)
- 123 Red spruce (*Picea rubens*)
- 124 Red spruce/balsam fir (*Picea rubens/Abies balsamea*)
- 125 Black spruce (*Picea mariana*)
- 126 Tamarack (*Larix laricina*)
- 127 Northern white-cedar (*Thuja occidentalis*)
- 128 Fraser fir (*Abies fraseri*)
- 129 Red spruce/Fraser fir (*Picea rubens/Abies fraseri*)

140 Longleaf/slash pine group
- 141 Longleaf pine (*Pinus palustris*)
- 142 Slash pine (*Pinus elliotti*)

150 Tropical softwoods group
- 151 Tropical pines (*Pinus* spp.)

160 Loblolly/shortleaf pine group
- 161 Loblolly pine (*Pinus taeda*)
- 162 Shortleaf pine (*Pinus echinata*)
- 163 Virginia pine (*Pinus virginiana*)
- 164 Sand pine (*Pinus clausa*)
- 165 Table mountain pine (*Pinus pungens*)
- 166 Pond pine (*Pinus serotina*)

167 Pitch pine (*Pinus rigida*)
168 Spruce pine (*Pinus glabra*)

170 Other eastern softwoods group
171 Eastern redcedar (*Juniperus virginiana*)
172 Florida softwoods

180 Pinyon/juniper group
182 Rocky Mountain juniper (*Juniperus scopulorum*)
184 Juniper woodland (*Juniperus* spp.)
185 Pinyon/juniper woodland (*Pinus* spp./*Juniperus* spp.)

200 Douglas-fir group
201 Douglas-fir (*Pseudotsuga menziesii*)
202 Port-Orford-cedar (*Chamaecyparis lawsoniana*)
203 Bigcone Douglas-fir (*Pseudotsuga macrocarpa*)

220 Ponderosa pine group
221 Ponderosa pine (*Pinus ponderosa*)
222 Incense-cedar (*Calocedrus decurrens*)
224 Sugar pine (*Pinus lambertiana*)
225 Jeffrey pine (*Pinus jeffreyi*)
226 Coulter pine (*Pinus coulteri*)

240 Western white pine group
241 Western white pine (*Pinus monticola*)

260 Fir/spruce/mountain hemlock group
261 White fir (*Abies concolor*)
262 Red fir (*Abies magnifica*)
263 Noble fir (*Abies procera*)
264 Pacific silver fir (*Abies amabilis*)
265 Engelmann spruce (*Picea engelmannii*)
266 Engelmann spruce/subalpine fir (*Picea engelmannii/Abies lasiocarpa*)
267 Grand fir (*Abies grandis*)
268 Subalpine fir (*Abies lasiocarpa*)
269 Blue spruce (*Picea pungens*)
270 Mountain hemlock (*Tsuga mertensiana*)
271 Alaska-yellow-cedar (*Chaemaecyparis nootkatensis*)

280 Lodgepole pine group
281 Lodgepole pine (*Pinus contorta*)

300 Hemlock/Sitka spruce group
301 Western hemlock (*Tsuga heterophylla*)
304 Western redcedar (*Thuja plicata*)
305 Sitka spruce (*Picea sitchensis*)

320 Western larch group
321 Western larch (*Larix laricina*)

340 Redwood group
341 Redwood (*Sequoia sempervirens*)
342 Giant sequoia (*Sequoiadendron giganteum*)

360 Other western softwoods group
361 Knobcone pine (*Pinus attenuata*)
362 Southwestern white pine (*Pinus strobiformis*)
363 Bishop pine (*Pinus muricata*)
364 Monterey pine (*Pinus radiata*)
365 Foxtail pine/bristlecone pine (*Pinus balfouriana/Pinus aristata*)
366 Limber pine (*Pinus flexilis*)
367 Whitebark pine (*Pinus albicaulus*)
368 Miscellaneous western softwoods
369 Western juniper (*Juniperus occidentalis*)

370 California mixed conifer group
371 California mixed conifer

380 Exotic softwoods group
381 Scotch pine (*Pinus sylvestris*)
383 Other exotic softwoods
384 Norway spruce (*Pinus abies*)
385 Introduced larch (*Larix* spp.)

390 Other softwoods group
391 Other softwoods

400 Oak/pine group
401 Eastern white pine/northern red oak/white ash (*Pinus strobus/Quercus rubra/ Fraxinus americana*)
402 Eastern redcedar/hardwood (*Juniperus virginiana*/hardwood)
403 Longleaf pine/oak (*Pinus palustris/Quercus* spp.)
404 Shortleaf pine/oak (*Pinus echinata/Quercus* spp.)
405 Virginia pine/southern red oak (*Pinus virginiana/Quercus falcata*)
406 Loblolly pine/hardwood (*Pinus taeda*/hardwood)
407 Slash pine/hardwood (*Pinus elliottii*)

409 Other pine/hardwood (*Pinus* spp./hardwood)

500 Oak/hickory group

501 Post oak/blackjack oak (*Quercus stellata/Quercus marilandica*)
502 Chestnut oak (*Quercus prinus*)
503 White oak/red oak/hickory (*Quercus alba/Quercus* spp./*Carya* spp.)
504 White oak (*Quercus alba*)
505 Northern red oak (*Quercus rubra*)
506 Yellow-poplar/white oak/northern red oak (*Liriodendron tulipifera/Quercus alba/Quercus rubra*)
507 Sassafras/persimmon (*Sassafras albidum/Diospyros* spp.)
508 Sweetgum/yellow-poplar (*Liquidambar styraciflua/Liriodendron tulipifera*)
509 Bur oak (*Quercus macrocarpa*)
510 Scarlet oak (*Quercus coccinea*)
511 Yellow-poplar (*Liriodendron tulipifera*)
512 Black walnut (*Juglans nigra*)
513 Black locust (*Robinia pseudoacacia*)
514 Southern scrub oak (*Quercus ilicifolia*)
515 Chestnut oak/black oak/scarlet oak (*Quercus* spp./*Quercus velutina/Quercus coccinea*)
516 Cherry/white ash/yellow-poplar (*Prunus* spp./*Fraxinus americana/Liriodendron tulipifera*)
517 Elm/ash/black locust (*Ulmus* spp./*Fraxinus* spp./*Robinia pseudoacacia*)
519 Red maple/oak (*Acer rubrum/Quercus* spp.)
520 Mixed upland hardwoods

600 Oak/gum/cypress group

601 Swamp chestnut oak/cherrybark oak (*Quercus michauxii/Quercus pagoda*)
602 Sweetgum/Nuttall oak/willow oak (*Liquidambar styraciflua/Quercus texana/Quercus phellos*)
605 Overcup oak/water hickory (*Quercus lyrata/Carya aquatica*)
606 Atlantic white-cedar (*Chamaecyparis nootkatensis*)
607 Baldcypress/water tupelo (*Taxodium* spp./*Nyssa aquatica*)
608 Sweetbay/swamp tupelo/red maple (*Magnolia virginiana/Nyssa biflora/Acer rubrum*)
609 Baldcypress/pondcypress (*Taxodium* spp./*Taxodium ascendens*)

700 Elm/ash/cottonwood group

701 Black ash/American elm/red maple (*Fraxinus nigra/Ulmus americana/Acer rubrum*)
702 River birch/sycamore (*Betula nigra/Platanus* spp.)
703 Cottonwood (*Populus* spp.)
704 Willow (*Salix* spp.)
705 Sycamore/pecan/American elm (*Platanus* spp./*Carya illinoiensis*)
706 Sugarberry/hackberry/elm/green ash (*Celtis laevigata/Celtis* spp./*Ulmus* spp./

Fraxinus pennsylvanica)
707 Silver maple/American elm (*Acer saccharinum/Ulmus americana*)
708 Red maple/lowland (*Acer rubrum*/lowland)
709 Cottonwood/willow (*Populus* spp./*Salix* spp.)
722 Oregon ash (*Fraxinus texensis*)

800 Maple/beech/birch group

801 Sugar maple/beech/yellow birch (*Acer saccharum/Fagus* spp./*Betula alleghaniensis*)
802 Black cherry (*Prunus serotina*)
805 Hard maple/basswood (*Acer* spp./*Tilia* spp.)
809 Red maple/upland (*Acer rubrum*/upland)

900 Aspen/birch group

901 Aspen (*Populus* spp.)
902 Paper birch (*Betula papyrifera*)
903 Gray birch (*Petula populifolia*)
904 Balsam poplar (*Populus balsamifera*)
905 Pin cherry (*Prunus pensylvanica*)

910 Alder/maple group

911 Red alder (*Alnus rubra*)
912 Bigleaf maple (*Acer macrophyllum*)

920 Western oak group

921 Gray pine (*Pinus sabiniana*)
922 California black oak (*Quercus kelloggii*)
923 Oregon white oak (*Quercus garryana*)
924 Blue oak (*Quercus douglasii*)
931 Coast live oak (*Quercus agrifolia*)
933 Canyon live oak (*Quercus chrysolepis*)
934 Interior live oak (*Quercus wislizeni*)
935 California white oak (valley oak) (*Quercus lobaba*)

940 Tanoak/laurel group

941 Tanoak (*Lithocarpus densiflorus*)
942 California laurel (*Umbellaria californica*)
943 Giant chinkapin (*Chrysolepis chrysophylla* var. *chrysophylla*)

960 Other hardwoods group

961 Pacific madrone (*Arbutus menziesii*)
962 Other hardwoods

970 Woodland hardwoods group

971 Deciduous oak woodland

972 Evergreen oak woodland
973 Mesquite woodland (*Prosopis* spp.)
974 Cercocarpus (mountain brush) woodland
975 Intermountain maple woodland
976 Miscellaneous woodland hardwoods

980 Tropical hardwoods group
982 Mangrove (*Avicennia germinans, Conocarpus erectus, Laguncularia racemosa, Rhizophora mangle*)
983 Palms (family Arecaceae)
984 Dry forest
985 Moist forest
986 Wet and rain forest
987 Lower montane wet and rain forest
989 Other tropical hardwoods

990 Exotic hardwoods group
991 Paulownia (*Paulownia tomentosa*)
992 Melaleuca (*Melaleuca quinquenervia*)
993 Eucalyptus (*Eucalyptus* spp.)
995 Other exotic hardwoods

988 Cloud forest

999 Nonstocked

Dominance Types in the Northern Region

Carbon modeling for the Northern Region made use of forest dominance types instead of the FIA forest-type groups shown previously. Dominance types are defined by the species with the greatest abundance of canopy cover, basal area, or trees per acre (hectare) within a setting. The species that define the dominance type are always of the same lifeform; therefore, it is first necessary to identify the dominant lifeform and subclass. The following dominance types have been identified in the Northern Region (Brown and Barber 2012):

Douglas-fir (PSME): Douglas-fir (*Pseudotsuga menziesii*) composes 60 percent or more of total relative tree abundance.

Lodgepole pine (PICO): Lodgepole pine (*Pinus contorta*) composes 60 percent or more of total relative tree abundance.

Subalpine fir (ABLA): Subalpine fir (*Abies lasiocarpa*) composes 60 percent or more of total relative tree abundance.

Ponderosa pine (PIPO): Ponderosa pine (*Pinus ponderosa*) composes 60 percent or more of total relative tree abundance.

Hardwood mixed (HMIX): Abundance of all hardwood trees exceeds 40 percent of total relative tree abundance.

Shade-intolerant mixed (IMIX): Abundance of all hardwood and shade-intolerant conifer trees exceeds 50 percent of total relative tree abundance.

Shade-tolerant mixed (TMIX): Abundance of all hardwood and shade-intolerant conifer trees is less than 50 percent of total relative tree abundance.

Common and Scientific Names of Fauna Mentioned in this Report

hemlock woolly adelgid (*Adelges tsugae*)

emerald ash borer (*Agrilus planipennis*)

cattle, oxen (*Bos taurus*)

eastern spruce budworm (*Choristoneura fumiferana*)

bark beetle (family Curculionidae, subfamily Scolytinae)

southern pine beetle (*Dendroctonus frontalis*)

mountain pine beetle (*Dendroctonus ponderosae*)

spruce beetle (*Dendroctonus rufipennis*)

gypsy moth (*Lymantria dispar*)

forest tent caterpillar (*Malacosoma disstria*)

sheep (*Ovis aries*)

northern spotted owl (*Strix occidentalis caurina*)

References

Brown, S.; Barber, J. 2012. The Region 1 Existing Vegetation Mapping program (VMap) Flathead National Forest Overview; version 12. Numbered Report 12-34. U.S. Department of Agriculture, Forest Service, Northern Region. 5 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5366381.pdf [Accessed September 6, 2019].

Burrill, E.A.; Wilson, A.M.; Turner, J.A.; [et al.]. 2018. The Forest Inventory and Analysis Database: Database description and user guide version 8.0 for Phase 2. U.S.

Department of Agriculture, Forest Service. 946 p. [Online]. https://www.fia.fs.fed.us/library/database-documentation/current/ver80/FIADB%20User%20Guide%20P2_8-0.pdf [Accessed August 5, 2019].

Appendix 2: Online Resources

Appendices 3-11: Regional Disturbance Carbon Assessments

Appendix 3: Eastern Region
Appendix 4: Southern Region
Appendix 5: Northern Region
Appendix 6: Rocky Mountain Region
Appendix 7: Intermountain Region
Appendix 8: Pacific Northwest Region
Appendix 9: Southwestern Region
Appendix 10: Pacific Southwest Region
Appendix 11: Alaska Region

Baseline Carbon Reports - https://www.fs.fed.us/managing-land/sc/carbon

Northern Region (R1) Report
Rocky Mountain Region (R2) Report
Southwestern Region (R3) Report
Intermountain Region (R4) Report
Pacific Southwest Region (R5) Report
Pacific Northwest Region (R6) Report
Southern Region (R8) Report
Eastern Region (R9) Report
Alaska Region (R10) Report
Regional Baseline Rational

Climate Change - Land Management & Project Planning - https://www.fs.fed.us/emc/nepa/climate_change/index.shtml

Climate Change Performance Scorecard - https://www.fs.fed.us/climatechange/advisor/scorecard/scorecard-guidance-08-2011.pdf

Ecosystem Restoration Policy and Directive (FSH 2020) - https://www.federalregister.gov/documents/2016/04/27/2016-09750/ecosystem-restoration-policy

National Roadmap for Responding to Climate Change - https://www.fs.fed.us/climatechange/pdf/Roadmapfinal.pdf

Research Data Archive - https://www.fs.usda.gov/rds/archive/catalog/RMRS-GTR-402-s1

Scorecard Appendix F: Carbon Asessment Technical Guidance - https://www.fs.fed.us/climatechange/advisor/scorecard/appendix/F%201-21-2011.pdf

U.S. Forest Planning Rule - https://www.fs.usda.gov/planningrule

Weeks Act - https://www.fs.fed.us/land/staff/Documents/Weeks%20Law.pdf

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Greenhouse Gas Emissions and Removals from Forest Land, Woodlands, and Urban Trees in the United States, 1990-2018

Introduction

As a signatory to the United Nations Framework Convention on Climate Change (UNFCCC), the United States has been reporting an economy-wide Inventory of greenhouse gas (GHG) emissions and removals since the mid-1990s (US EPA 2020). Forest land, harvested wood products (HWPs), and urban trees within the land sector collectively represent the largest net carbon (C) sink in the United States, offsetting more than 11 percent of total GHG emissions annually (US EPA 2020). Estimates of GHG emissions and removals are compiled by U.S. Department of Agriculture (USDA) Forest Service researchers and are based primarily on National Forest Inventory (NFI) data collected and maintained by the Forest Inventory and Analysis (FIA) program within the USDA Forest Service. This report—the second in a new series of annual updates—provides an overview of the status and trends of GHG emissions and removals from forest land, woodlands in the grassland category, HWPs, and urban trees in settlements in the United States from 1990 to 2018. The estimates for the United States summarized here are based on the compilation reported in the *Land Use, Land-Use Change, and Forestry* chapter of the US EPA (2020) submission to the UNFCCC. New in this report, most of the national scale estimates are also reported by individual U.S. state (Fig. 1) and are available online for the entire 1990-2018 time series (see appendix).

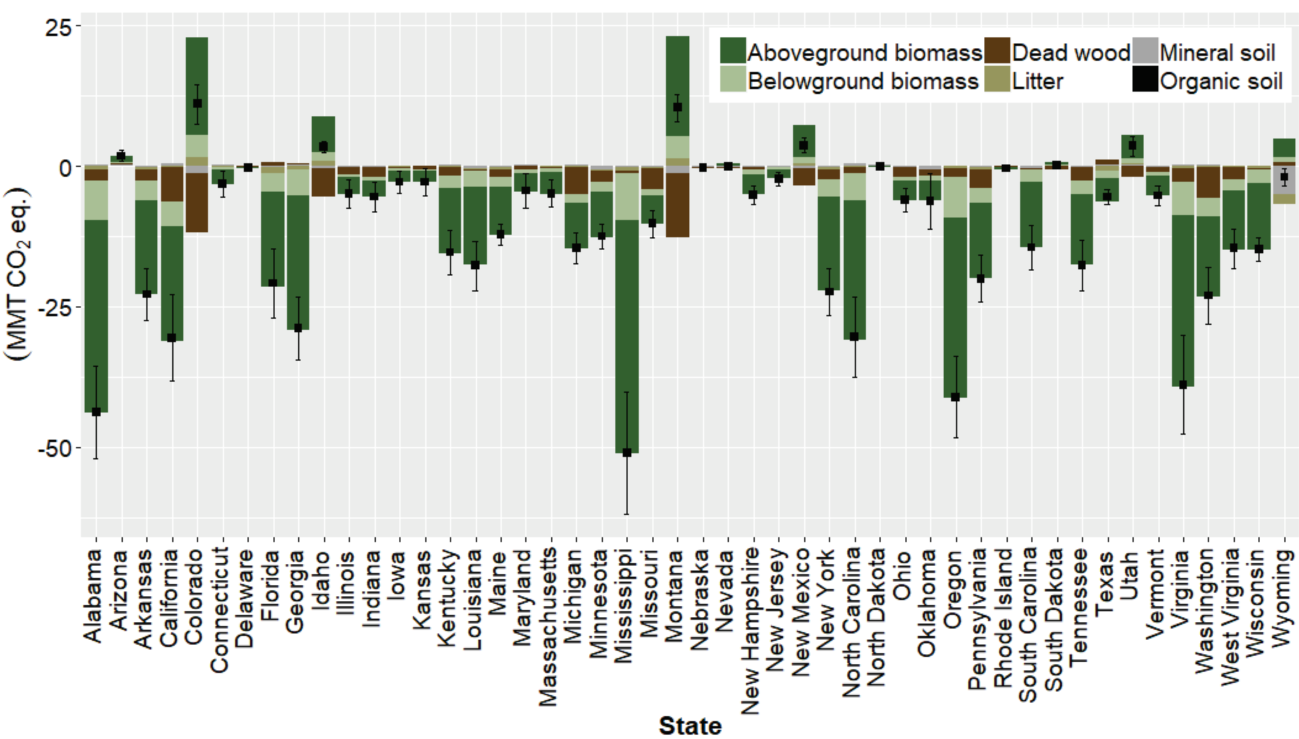

Figure 1.—Estimated annual emissions and removals from forest land remaining forest land by carbon pool for each of the conterminous 48 states in 2018 (MMT CO_2 Eq.). Note that points and confidence intervals (95 percent) reflect net flux for all carbon pools in each state. Negative estimates indicate net C uptake (i.e., a net removal of C from the atmosphere).

Forest Carbon Cycle

Carbon is continuously cycled among ecosystem pools and the atmosphere as a result of biogeochemical processes in forests (e.g., photosynthesis, respiration, decomposition, and disturbances such as fires or pest outbreaks) and anthropogenic activities (e.g., harvesting, thinning, and replanting). As trees photosynthesize and grow, C is removed from the atmosphere and stored in living tree biomass. As trees die and otherwise deposit litter and debris on the forest floor, C is released to the atmosphere and is also transferred to the litter, dead wood, and soil pools by organisms that facilitate decomposition.

The net change in forest C is not equivalent to the net flux between forests and the atmosphere because timber harvests do not result in an immediate flux of all harvested biomass C to the atmosphere. Instead, following harvesting a portion of the C stored in wood is transferred to a "product pool." Once in a product pool, the C is emitted over time as carbon dioxide (CO_2) from decomposition and as CO_2, methane (CH_4), nitrous oxide (N_2O), carbon monoxide (CO), and other nitrogen oxides (NO_x) when the wood product combusts. The rate of emission varies considerably among different product pools.

Carbon Pools

For estimating C stocks or stock change (flux), C in forest ecosystems can be divided into the following five storage pools (IPCC 2006):

- Aboveground biomass—all living biomass above the soil including stem, stump, branches, bark, seeds, and foliage. This pool includes live understory.
- Belowground biomass—all living biomass of coarse living roots greater than 2 millimeters (mm) diameter.
- Dead wood—all nonliving woody biomass either standing, lying on the ground (but not including litter), or in the soil.
- Litter—the litter, fumic, and humic layers, and all nonliving biomass with a diameter less than 7.5 centimeters (cm) at transect intersection, lying on the ground.
- Soil organic C (SOC)—all organic material in soil to a depth of 1 meter but excluding the coarse roots of the belowground pools.

In addition, two harvested wood pools are included when estimating C flux:

- Harvested wood products (HWP) in use.
- HWP in solid waste disposal sites (SWDS).

Total Emissions and Removals

Forest land, HWPs, woodlands, and urban trees in settlements collectively represent a net GHG sink over the UNFCCC reporting period, with interannual variability driven, in large part, by natural and anthropogenic disturbances (e.g., wildfire, harvesting), land conversions, and changes in HWPs in use (Table 1.; US EPA 2020). In 2018, forest land, HWPs, woodlands, and urban trees in settlements collectively represented an estimated net uptake of 752.9 million metric tons of carbon dioxide equivalent (MMT CO_2 Eq.). The category "forest land remaining forest land" was the largest net sink in the land sector, with an estimated uptake of 564.5 MMT CO_2 Eq. Conversions from forest land were the largest source of emissions within the categories included in this report, with estimated emissions of 127.4 MMT CO_2 Eq. (Table 1; US EPA 2020).

Table 1.—Emissions and removals (net flux) from land use, land-use change, and forestry (MMT CO_2 Eq.)

Emissions and Removals Category[a]	1990	1995	2000	2005	2010	2016	2017	2018
Forest land remaining forest land[b]	(610.1)	(598.7)	(572.1)	(572.6)	(556.2)	(565.5)	(552.0)	(564.5)
Non-CO_2 emissions from fire	1.5	0.6	2.9	8.2	4.6	5.6	18.8	18.8
N_2O emissions from forest soils	0.1	0.3	0.5	0.5	0.5	0.5	0.5	0.5
Non-CO_2 emissions from drained organic soils	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Forest land converted to non-forest land[b]	119.1	120.8	122.5	124.4	126.0	127.4	127.4	127.4
Non-forest land converted to forest land[b]	(109.4)	(109.7)	(109.9)	(110.2)	(110.4)	(110.6)	(110.6)	(110.6)
Harvested wood products	(123.8)	(112.2)	(93.4)	(106.0)	(69.1)	(92.4)	(95.7)	(98.8)
Woodlands remaining woodlands[c]	5.0	4.9	4.8	4.6	4.4	4.1	4.0	4.0
Urban trees in settlements[d]	(96.4)	(103.3)	(110.4)	(117.4)	(124.6)	(129.8)	(129.8)	(129.8)
Total Emissions and Removals	**(813.9)**	**(797.2)**	**(755.0)**	**(768.4)**	**(724.7)**	**(760.6)**	**(737.3)**	**(752.9)**

[a] For details on how estimates were compiled see US EPA 2020.
[b] Estimated emissions and removals include the net changes to C stocks stored in all ecosystem pools.
[c] Estimates for woodlands, which are included in the grassland land use category, were compiled using the same methods and models as those in the forest land category.
[d] Estimates of emissions and removals from urban trees in settlements were compiled using percentage tree cover in carbon sequestration density per unit of tree cover.
Notes: Totals may not sum due to independent rounding. Parentheses indicate net C uptake (i.e., a net removal of C from the atmosphere).

Forest Land Remaining Forest Land and Harvested Wood Products

Within the "forest land remaining forest land" category, aboveground live biomass is the largest contributor to the net uptake over the reporting period, followed by belowground live biomass and dead wood (Table 2). Harvested wood products in use and in solid waste disposal sites (SWDS) are also an important contributor to the net sink in the land sector, and 2018 estimates for both pools increased from previous years.

Table 2.—Emissions and removals (net flux) from forest land remaining forest land and harvested wood pools (MMT CO_2 Eq.)

Carbon Pool[a]	1990	1995	2000	2005	2010	2016	2017	2018
Forest ecosystem	**(610.1)**	**(598.7)**	**(572.1)**	**(572.6)**	**(556.2)**	**(565.5)**	**(552.0)**	**(564.5)**
Aboveground biomass	(425.1)	(416.1)	(392.7)	(391.3)	(391.3)	(397.0)	(381.2)	(385.2)
Belowground biomass	(98.6)	(96.6)	(91.5)	(90.8)	(90.3)	(91.1)	(87.6)	(88.6)
Dead wood	(81.9)	(82.8)	(82.7)	(84.1)	(83.4)	(87.6)	(83.1)	(86.4)
Litter	(5.0)	(3.5)	(4.5)	(5.2)	(1.4)	(0.9)	(3.5)	(3.1)
Soil (mineral)	0.3	(0.1)	(1.0)	(1.8)	4.6	8.2	1.4	(3.3)
Soil (organic)	(0.6)	(0.5)	(0.3)	(0.1)	4.9	2.3	1.4	1.4
Drained organic soil	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8
Harvested wood	**(123.8)**	**(112.2)**	**(93.4)**	**(106.0)**	**(69.1)**	**(92.4)**	**(95.7)**	**(98.8)**
Products in use	(54.8)	(51.7)	(31.9)	(42.6)	(7.4)	(27.8)	(30.3)	(31.5)
SWDS	(69.0)	(60.5)	(61.5)	(63.4)	(61.7)	(64.6)	(65.5)	(67.2)
Total Net Flux	**(733.9)**	**(710.9)**	**(665.5)**	**(678.6)**	**(625.3)**	**(657.9)**	**(647.7)**	**(663.2)**

[a] For details on these estimates and how they were compiled see US EPA 2020.
Notes: Totals may not sum due to independent rounding. Parentheses indicate net C uptake (i.e., a net removal of C from the atmosphere).

Carbon stock estimates for forest ecosystem and harvested wood C storage pools are presented in Table 3. Together, the estimated aboveground biomass and soil C pools account for a large proportion of total forest ecosystem C stocks. By maintaining current harvesting practices and regeneration activities on these forested lands, along with continued input of harvested products into the HWP pool, C stocks in forests are likely to continue to increase in the near term, though possibly at a lower rate. Because most of the timber harvested from U.S. forest land is used in wood products and many discarded wood products are disposed of in SWDS rather than by incineration, significant quantities of C in harvested wood are transferred to these long-term storage pools rather than being released rapidly to the atmosphere (Skog 2008).

Table 3.—Carbon stocks in forest land remaining forest land and harvested wood pools (MMT C)

Carbon Pool[a]	1990	1995	2000	2005	2010	2017	2018	2019
Forest	**51,527**	**52,358**	**53,161**	**53,886**	**54,663**	**55,746**	**55,897**	**56,051**
Aboveground biomass	11,833	12,408	12,962	13,484	14,020	14,780	14,884	14,989
Belowground biomass	2,350	2,483	2,612	2,734	2,858	3,033	3,056	3,081
Dead wood	2,120	2,233	2,346	2,454	2,568	2,731	2,753	2,777
Litter	3,662	3,670	3,676	3,647	3,646	3,639	3,640	3,641
Soil (mineral)	25,636	25,636	25,637	25,639	25,641	25,637	25,637	25,638
Soil (organic)	5,927	5,928	5,928	5,929	5,929	5,926	5,926	5,926
Harvested wood	**1,895**	**2,061**	**2,218**	**2,353**	**2,462**	**2,616**	**2,642**	**2,669**
Products in use	1,249	1,326	1,395	1,447	1,471	1,505	1,513	1,521
SWDS	646	735	823	906	991	1,112	1,129	1,148
Total stocks	**53,423**	**54,419**	**55,380**	**56,239**	**57,124**	**58,362**	**58,539**	**58,720**

[a] For details on these estimates and how they were compiled see US EPA 2020.
Notes: Totals may not sum due to independent rounding. Forest C stock estimates include all forest land remaining forest land in the conterminous 48 states and Alaska. Forest ecosystem C stocks do not include U.S. Territories because managed forest land for U.S. Territories is not currently included in Section 6.1 Representation of the U.S. Land Base. Forest ecosystem C stocks also do not include Hawaii because there is not sufficient NFI data to support inclusion at this time. Forest ecosystem C stocks on managed forest land in Alaska were compiled using the gain-loss method as described in Annex 3.13. Harvested wood product stocks include exports, even if the logs are processed in other countries, and excludes imports. Harvested wood estimates are based on results from annual surveys and models. Totals may not sum due to independent rounding. Population estimates compiled using FIA data are assumed to represent stocks as of January 1 of the inventory year. Flux is the net annual change in stock. Thus, flux estimates for 2018 require C stocks for 2018 and 2019.

Forest Land Conversions

Land use conversions to and from forest land result in substantial emissions and removals each year. In this section all emissions and removals included for land conversions to and from forest land, as reported in US EPA (2020), are included in Table 4. Forest land conversion to settlements was the largest source of emissions in the conversion categories while cropland conversion to forest land resulted in the largest annual uptake. Considering all forest land conversions included in the US EPA (2020) report, over the reporting period there have been net emissions each year, with estimated net emissions of 16.7 MMT CO_2 Eq. for the most recent year.

Table 4.—Emissions and removals (net flux) from conversions to and from forest land (MMT CO_2 Eq.)

Land Conversions[a]	1990	1995	2000	2005	2010	2016	2017	2018
Forest land converted to cropland	48.6	48.7	48.5	48.4	48.3	48.7	48.7	48.7
Forest land converted to grassland	15.9	15.8	16.0	16.0	16.0	15.9	15.9	15.9
Forest land converted to settlements	54.6	56.3	58.0	59.9	61.6	62.9	62.9	62.9
Cropland converted to forest land	(45.9)	(45.9)	(46.0)	(46.1)	(46.2)	(46.3)	(46.3)	(46.3)
Grassland converted to forest land	(9.8)	(9.7)	(9.7)	(9.6)	(9.6)	(9.7)	(9.7)	(9.7)
Other land converted to forest land	(14.3)	(14.5)	(14.6)	(14.8)	(14.9)	(14.9)	(14.9)	(14.9)
Settlements converted to forest land	(38.6)	(38.6)	(38.7)	(38.7)	(38.8)	(38.9)	(38.9)	(38.9)
Wetlands converted to forest land	(0.9)	(0.9)	(0.9)	(0.9)	(0.9)	(0.9)	(0.9)	(0.9)
Net Emissions and Removals	**9.6**	**11.2**	**12.6**	**14.3**	**15.6**	**16.7**	**16.7**	**16.7**

[a] For details on these estimates and how they were compiled see US EPA 2020.

Notes: Totals may not sum due to independent rounding. Parentheses indicate net C uptake (i.e., a net removal of C from the atmosphere). Emissions and removals from forest land converted to other lands are currently not included in US EPA (2020). Forest land converted to wetlands estimates were not compiled by the Forest Service.

Land Area

The land area included in the US EPA (2020) report includes lands directly influenced by human intervention. Direct intervention occurs mostly in areas accessible to human activity and includes altering or maintaining the condition of the land to produce commercial or noncommercial products or services; to serve as transportation corridors or locations for buildings, landfills, or other developed areas for commercial or noncommercial purposes; to extract resources or facilitate acquisition of resources; or to provide social functions for personal, community, or societal objectives where these areas are readily accessible to society. Forest Inventory and Analysis data from each of the conterminous 48 states and Alaska comprise an estimated 280 million hectares (ha) of forest land that are considered managed and are included in this report along with an additional 10 million ha of non-forest land converted to forest land. Some differences exist in forest land area estimates in the latest update to the Resources Planning Act Assessment (Oswalt et al. 2019) and the forest land area estimates included in the US EPA (2020) report, which are based on annual FIA data through 2018 for all states (USDA Forest Service 2019). These differences are due, in large part, to the separation of land categories and the managed land definition used in the US EPA (2020) report (Nelson et al. 2020). Sufficient annual inventory data are not yet available for Hawaii, but estimates of these areas are included in Oswalt et al. (2019). Even though Hawaii and U.S. Territories have relatively small areas of forest land that may not substantially influence the overall C budget for forest land, these regions will be added to the forest C estimates as sufficient data become available. Agroforestry systems that meet the definition of forest land are also not currently included in the US EPA (2020) report since they are not explicitly inventoried (i.e., they are classified as agroforestry system) by either the FIA program or the Natural Resources Inventory of the USDA Natural Resources Conservation Service. Woodland area is included in the "grassland remaining grassland" and "land converted to grassland" categories and is not explicitly separated in the US EPA (2020) report as a subcategory of grasslands. Combined, forest land and woodland area accounts for more than 311 million ha (Table 5).

Table 5.—Annual estimates of forest land and woodland area (1000 ha)

Land Area Category[a]	1990	1995	2000	2005	2010	2018	2019
Forest land remaining forest land	279,748	279,840	280,025	279,749	279,918	279,787	279,682
Non-forest land converted to forest land	9,622	9,654	9,689	9,725	9,761	9,796	9,796
Woodland remaining woodland[b]	19,891	19,669	19,255	18,630	17,733	16,000	15,776
Non-woodland converted to woodland[b]	5,782	5,702	5,552	5,322	4,994	4,607	4,607
Total Area	**315,043**	**314,865**	**314,521**	**313,426**	**312,405**	**312,209**	**311,880**

[a] For details on these estimates and how they were compiled see US EPA 2020.

[b]Woodland area is included in the "remaining grassland" and "land converted to grassland" categories and is not explicitly separated in the US EPA (2020) report.

Notes: Totals may not sum due to independent rounding. The estimates reported here may differ from the Land Representation section of US EPA (2020) but are consistent with estimates used to compile emissions and removals in these categories. See Annex 3.13 in US EPA (2020) for more details.

Planned Improvements

Planned improvements to estimation and reporting include the following general topics: development of a more robust estimation and reporting system, individual C pool estimation, coordination with other land-use categories, and annual inventory data incorporation. Research is underway to leverage auxiliary information (i.e., remotely sensed information) to operate at finer spatial and temporal scales. As in past submissions, emissions and removals associated with natural (e.g., wildfire, insects, and disease) and human (e.g., harvesting) disturbances are implicitly included in the report given the design of the annual NFI, but are not explicitly estimated. In addition to integrating auxiliary information into the estimation framework, alternative estimators are also being evaluated that will eliminate latency in population estimates from the NFI, improve annual estimation and characterization of interannual variability, facilitate attribution of fluxes to particular activities, and allow for easier harmonization of NFI data with auxiliary data products. There are also investments being made to leverage state-level wood products and harvest information to allow for the disaggregation of HWPs estimates at the state level. Collectively these improvements are expected to reduce uncertainties in the estimates at the national and state scales and facilitate entity-level estimation and reporting.

2020 Estimates at a Glance

Below are summary statistics from the compilation of the forest land, woodlands, HWPs, and urban trees in settlements in the US EPA (2020) report.

- Forest land, HWPs, and urban trees in settlements collectively offset more than 11 percent (752.9 MMT CO_2 Eq.) of total GHG emissions annually, or 14 percent of CO_2 emissions.
- Forest land accounts for more than 95 percent of the net C sink within the land sector.
- Live vegetation in forests and urban trees account for nearly 80 percent of the C sink strength.
- Land conversions to and from forest land continue to result in net emissions (16.7 MMT CO_2 Eq.).
- More than 56 percent of all carbon in forest ecosystems is stored in the soil with small stock changes annually.
- Carbon storage in HWPs continues to increase annually since the Great Recession.
- Forests uptake averages 0.6 metric tons of C per hectare per year (MT C ha^{-1} yr^{-1}) with live vegetation accounting for more than 85 percent (0.5 MT C ha^{-1} yr^{-1}) of the uptake.

Literature Cited

Intergovernmental Panel on Climate Change [IPCC]. 2006 IPCC guidelines for national greenhouse gas inventories. Hayma, Japan: Institute for Global Environmental Strategies. https://www.ipcc-nggip.iges.or.jp/public/2006gl/ (accessed 9 April 2020).

Nelson, M.D.; Riitters, K.H.; Coulston, J.W.; Domke, G.M.; Greenfield, E.J.; Langner, L.L.; Nowak, D.J.; O'Dea, C.B.; Oswalt, S.N.; Reeves, M.C.; Wear, D.N. 2020. Defining the United States land base: a technical document supporting the USDA Forest Service 2020 RPA assessment. Gen. Tech. Rep. NRS-191. Madison, WI: U.S. Department of Agriculture, Forest Service, Northern Research Station. 70 p. https://doi.org/10.2737/NRS-GTR-191.

Oswalt, S.N.; Smith, W. B.; Miles, P.D.; Pugh, S.A., coords. 2019. Forest resources of the United States, 2017: a technical document supporting the Forest Service 2020 RPA Assessment. Gen. Tech. Rep. WO-97. Washington, DC: U.S. Department of Agriculture, Forest Service. 223 p. https://doi.org/10.2737/WO-GTR-97.

Skog, K.E. 2008. Sequestration of carbon in harvested wood products for the United States. Forest Products Journal. 58(6): 56-72.

USDA Forest Service. 2020. Forest inventory and analysis national program: FIA data mart. Washington, DC: U.S. Department of Agriculture, Forest Service. https://apps.fs.usda.gov/fia/datamart/datamart.html (accessed 1 November 2020).

U.S. Environmental Protection Agency [US EPA]. 2020. Inventory of U.S. greenhouse gas emissions and sinks: 1990-2018. EPA 430-R-20-002. Washington, DC: U.S. Environmental Protection Agency. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2018 (accessed 14 April 2020).

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COMPARATIVE LIFE-CYCLE ASSESSMENT OF A MASS TIMBER BUILDING AND CONCRETE ALTERNATIVE[1]

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Abstract. The US housing construction market consumes vast amounts of resources, with most structural elements derived from wood, a renewable and sustainable resource. The same cannot be said for all nonresidential or high-rise buildings, which are primarily made of concrete and steel. As part of continuous environmental improvement processes, building life-cycle assessment (LCA) is a useful tool to compare the environmental footprint of building structures. This study is a comparative LCA of an 8360-m^2, 12-story mixed-use apartment/office building designed for Portland, OR, and constructed from mainly mass timber. The designed mass timber building had a relatively lightweight structural frame that used 1782 m^3 of cross-laminated timber (CLT) and 557 m^3 of glue-laminated timber (glulam) and associated materials, which replaced approximately 58% of concrete and 72% of rebar that would have been used in a conventional building. Compared with a similar concrete building, the mass timber building had 18%, 1%, and 47% reduction in the impact categories of global warming, ozone depletion, and eutrophication, respectively, for the A1-A5 building LCA. The use of CLT and glulam materials substantially decreased the carbon footprint of the building, although it consumed more primary energy compared with a similar concrete building. The impacts for the mass timber building were affected by large amounts of gypsum board, which accounted for 16% of total building mass. Both lowering the amount of gypsum and keeping the mass timber production close to the construction site could lower the overall environmental footprint of the mass timber building.

Keywords: Cross-laminated timber, environmental assessment, life-cycle analysis, tall wood building.

INTRODUCTION

The building industry is a heavy user of resources and is responsible for more than 30% of total global energy consumption and for about 40% of global

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carbon dioxide emissions (Jones et al 2016; Berardi 2017). Developing energy-efficient and low-impact buildings has become increasingly important. The use of wood as a building material can provide substantial economic and environmental benefits (Ritter et al 2011). Mass timber products, including cross-laminated timber (CLT), glue-laminated timber (glulam), and nail-laminated timber, have been demonstrated to be green building materials with a lower carbon footprint than their concrete and steel alternatives (Perez-Garcia et al 2005; Karacabeyli and Douglas 2013; Bowers at al. 2017; Gu and Bergman 2018). Also, CLT and other mass timber products offer additional advantages such as faster erection times, easier material handling, a high level of prefabrication at the material manufacturing site, and less waste generation and noise pollution during the construction stage (Kremer and Symmons 2015; Connolly et al 2018; Smith et al 2018). In particular, the construction phase of a mass timber building can result in substantial savings with quicker erection times, more than 50% faster than other alternative materials (APA 2019).

CLT is a massive structural composite panel fabricated with kiln-dried dimensional lumber stacked in three to nine layers arranged perpendicular to each other (APA 2012). The production of similar perpendicular engineered wood products dates back to the early 20th century in the United States (Walch and Watts 1923). The use of CLT in mid- to high-rise buildings began to appear several decades ago in European countries (FII 2016; Espinoza and Buehlmann 2018). More recently, CLT and other mass timber technologies have captured the interest of designers, developers, property owners, industry, and governments in North America (Podesto and Berneman 2016; Williamson and Ross 2016). The revised 2021 International Building Code includes provisions for new construction Type IV-A/B/C for up to 18 stories for business and residential buildings using mass timber (Breneman and Richardson 2019). Using mass timber in building systems could be a boost to the wood industry sector, but to gain the support of green building advocates, rigorous scientific analysis on the environmental impacts is required. The whole-building life-cycle assessment (LCA) is a method to analyze building environmental impacts based on ASTM E2921 (ASTM 2016) and EN 15978 (EN 2011) standards. However, only few LCA studies on CLT and other mass timber buildings are publicly available (Cadorel and Crawford 2018). These studies all agree that mass timber buildings have better environmental performance such as lower greenhouse gas (GHG) emissions compared with alternative concrete buildings, although different study periods or system boundaries were applied (Robertson et al 2012; Durlinger et al 2013; Grann 2013; Bowick 2015, 2018). LCA case studies for mass timber buildings in the United States are very limited (Gu and Bergman 2018; Pierobon et al 2019) because few buildings have gone beyond the concept stage. More importantly, there are very few studies that directly compare the LCA implications for mass timber and concrete steel buildings, with similar size, function, and operational energy performance. It is critical to conduct more studies to assess the environmental impacts of mass timber buildings and to further analyze the impacts on local communities, forest health, and the regional economy.

This study conducted a building LCA for a 12-story mixed-use tall wood building in Portland, OR, that comprises CLT and glulam as the main structural building materials. The LCA of this mass timber building was compared with a functionally equivalent concrete building system, with no wood structural elements. The environmental impacts of the two buildings were categorized using the Tool for the Reduction and Assessment of Chemical and other environmental Impacts (TRACI) 2.1 impact method (Bare et al 2012), and a detailed contribution analysis and carbon accounting metrics were performed. This research is part of a more comprehensive project investigating the CLT supply chain along with potential economic contributions and environmental implications of increased CLT and other mass timber building construction (Kelley and Bergman 2017). Results generated from this study will provide solid, transparent evidence on the environmental performance metrics for CLT and other mass timber buildings that can inform the public, building developers and owners, and policy makers.

MATERIALS AND METHODS

Goal and Scope

The goal of this study was to quantify the environmental impacts of a tall wood building built primarily with CLT and glulam structural elements and compare those impacts with a functionally equivalent building with traditional concrete materials. The target building for this cradle-to-site LCA study is an 8360-m^2, 12-story, mixed-use office and apartment complex. The building was designed to be built in Portland, OR, and to have the same fire-proofing performance, insulation, and energy consumption outcomes as a functionally equivalent concrete building design. Both building designs were completed by LEVER Architecture (Portland, OR) with additional structural design and analysis from their partner, KPFF Engineering (Seattle, WA). Both buildings comply with Type 1B fire-resistant construction code with noncombustible capacities of 2-h exterior walls, 2-h structural frame, 2-h ceiling/floor separation, and 1-h ceiling/roof assembly (Heppner 2019). The building structure components to compare include ceiling–roof, floors, foundation, postbeams, and walls.

Functional Unit and System Boundary

The functional unit for this study is defined as 1 m^2 of floor area of the whole building. The system boundary of this building LCA is defined as cradle-to-site, as illustrated in Fig 1, and includes the modules A1-A3 *Product Stage* and modules A4 and A5 *Construction Process Stage*. More specifically, in this study, module A1 *Raw materials supply* covers raw material acquisition (eg from tree seeding to log harvest, or cement and aggregate mining and production); module A2 *Transport* covers the transportation of raw materials to the manufacture plant (eg truck loading logs and transporting to primary and secondary wood products manufacturers, or transportation of cement and aggregate and rebar); module A3 *Manufacturing* covers the gate-to-gate production of secondary products (eg mass timber products processing and packaging at the plant, or formulated concrete); module A4 *Transport* covers the transportation of materials and products from the factory gate to the building site (eg trucking all construction materials to the building site), whereas the transportation of construction equipment to and from the site was excluded; and module A5 *Construction installation process* only covers the energy consumption to install building materials into the building (eg diesel usage by crane to lift CLT and pour/pump concrete), whereas ground works, labor assembly, land use, and other things were excluded.

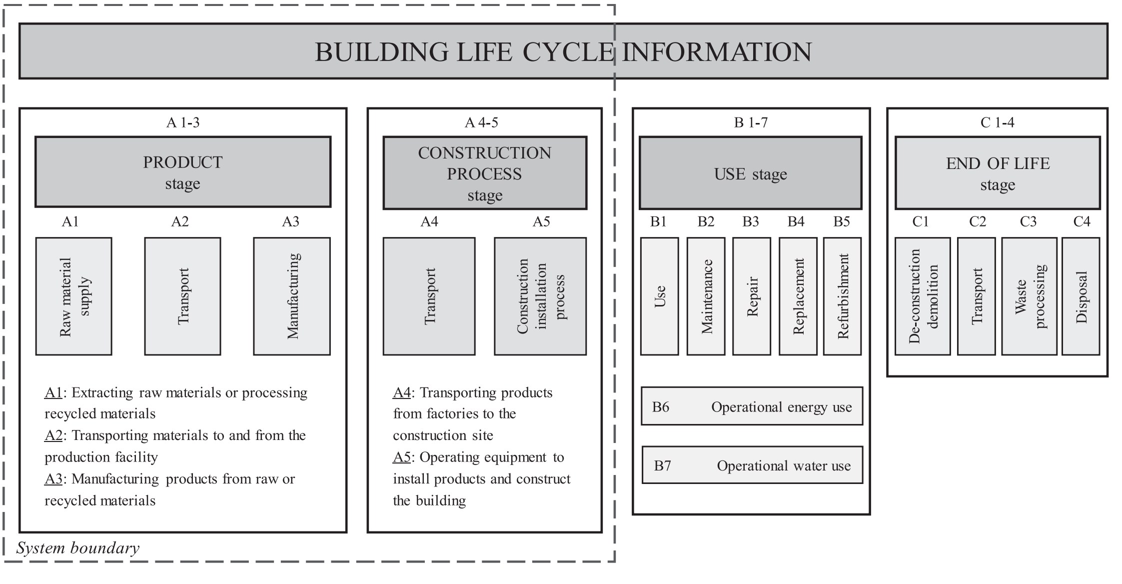

Figure 1. Cradle-to-site life-cycle assessment system boundary for the mass timber and concrete buildings.

Life-Cycle Inventory (LCI) and Impact Assessment

The LCI phase in this study measures the materials use (modules A1-A3), transportation (module A4), and direct energy (module A5) inputs for the construction process of the proposed CLT and concrete buildings. Table 1 summarizes the quantities of building materials used for the two buildings, which were designed with the same U-value. These quantities were provided by the building designer (LEVER Architecture). The building materials transport distances to the

Table 1. Quantities of materials and life-cycle inventory data sources for mass timber and concrete buildings.

Building section/material	Unit	Mass timber building	Concrete building	Database/source
Ceiling and roof				
Hollow structural steel	kg	11,449	7415	AIE
CLT	m^3	0.95		Chen et al 2019
1-inch mineral wool	m^2	285	285	DataSmart
Acrylic latex paint	L	3096	1548	DataSmart
1-inch polystyrene board	m^2	144	144	DataSmart
Steel sheet	kg	5693	5693	DataSmart
5/8-inch gypsum board, fire-resistant	m^2	14,907	5945	AIE
1/2-inch gypsum board, regular	m^2	4154	3337	AIE
Floors				
Acrylic adhesive	L	117	117	
CLT	m^3	1279		Chen et al 2019
Coated steel deck	kg	110	110	AIE
Hollow structural steel	kg	400	400	AIE
Concrete	m^3	932	1878	DataSmart
Mortar	kg	4737	4737	DataSmart
3/8-inch plywood	m^2	661	661	DataSmart/USLCI
1-inch polystyrene board	m^2	4067	4067	DataSmart
Rebar	kg	53,177	170,348	DataSmart
Steel sheet	kg	4193	875	DataSmart
Steel welded wire mesh	kg	110	110	AIE
60-mil TPO membrane	m^2	351	351	AIE
Foundation				
Concrete	m^3	125	149	DataSmart
Rebar	kg	38,590	57,884	DataSmart
Post and beam				
Hollow structural steel	kg	43,527	39,230	AIE
Composite wood I-joist	kg	60	60	DataSmart/USLCI
Concrete	m^3		162	DataSmart
Glulam	m^3	557		DataSmart/USLCI
Rebar	kg		22,089	DataSmart
Steel sheet	kg	830	823	DataSmart
Walls				
Aluminum extrusion	kg	31,039	31,051	DataSmart
CLT	m^3	502		Chen et al 2019
Hollow structural steel	kg	31,947	30,026	AIE
Concrete	m^3	48	438	DataSmart
Concrete masonry unit	kg	71,031	70,908	DataSmart
3/8-inch plywood	m^2	3230	3230	DataSmart/USLCI
Mortar	kg	90,113	89,824	DataSmart
Acrylic latex paint	L	9100	5143	DataSmart
1-inch polystyrene board	m^2	7643	7644	DataSmart
Silicone sealant	L	503	503	DataSmart
Rebar	kg	12,078	125,951	DataSmart
5/8-inch gypsum board, fire-resistant	m^2	57,330	47,097	AIE

construction site were provided by LEVER Architecture (Heppner 2019). It is noteworthy that concrete used a transportation distance of 24 km, and the CLT and glulam were sourced with a transportation distance of 320 km. Direct energy inputs for the construction installation were estimated as diesel usage using an empirical equation provided by Athena Sustainable Materials Institute (Finlayson 2019). The LCI of the CLT production process model was provided by the University of Washington (Chen et al 2019). Other building materials, transportation, and energy LCI data were sourced from the USLCI and US Ecoinvent 2.2 (DATASMART 2019) and Athena Impact Estimator (AIE) databases and from the Pacific Northwest forest resources, as listed in Table 1.

The life-cycle impact assessment (LCIA) integrates the LCI data of each building stage (modules A1-A5) to quantify the total life-cycle environmental impacts, following the ISO 14040 (ISO 2006a) and ISO 14044 (ISO 2006b) environmental management standards. The total environmental impacts were modeled using data sources in SimaPro 8.5 software (PRé Sustainability, Amersfoort, the Netherlands) and AIE for Building 5.6 software (Athena Sustainable Materials Institute, Ottawa, Canada), and the TRACI 2.1 impact method (Bare et al 2012) was used in SimaPro and AIE. The primary energy consumption, categorized as nonrenewable (fossil and nuclear) and renewable (biomass, solar, wind, and hydropower), was calculated using the embedded cumulative energy demand (CED) method v1.10 in SimaPro and AIE. The impact indicators calculated for each building material from the two software were extracted into Microsoft Excel spreadsheets and then integrated for further analysis.

Sensitivity Analysis

Although the United States has significant lumber manufacturing capacity, there is limited CLT manufacturing capacity. However, with increasing interest in this emerging product, new capacity is developing across the United States. Thus, CLT transportation distances to the construction site were varied in determining the portion of this specific variable on the whole-building environmental impact. The current distance was 320 km, which assumes transport from a nearby local facility. Sensitivity analyses included *middle distance* by truck (768 km) from Inland Northwest (INW), *long distance* by truck (465 km) and rail (4189 km) from eastern United States; and *oversea distance* by sea (21,333 km) from Europe, rail (1103 km), and truck (335 km).

RESULTS AND DISCUSSION

Comparison of Building Materials

The quantities of building materials of each building section (ceilings-roof, floor, foundation, postbeam, and wall) for the mass timber and concrete buildings are shown in Table 1. Generally, the mass timber building uses a total of 2376 m^3 of wood products with 98% CLT and glulam. In this study, mass timber usage is about 0.28 m^3 per m^2 of floor area. Other CLT and mass timber building designs have used between 0.1 and 0.45 m^3 per m^2 of floor area (Gustavsson et al 2010; Oregon BEST 2017; Gu and Bergman 2018). Specifically, the walls use 502 m^3 of 7- and 9-ply CLT, the floors use 1279 m^3 of 5-ply CLT, and the postbeams use 557 m^3 of glulam for columns and beams. In addition, the mass timber building also uses a significant amount of concrete and steel, eg 1104 m^3 concrete and 103,845 kg rebar. The concrete and steel are used on the foundation and also to stiffen the CLT floor elements. The concrete building uses no mass timber but uses 2627 m^3 of concrete and 376,272 kg of rebar. In addition, to comply with Type 1B fire-resistant construction code, approximately 36% more gypsum board is applied to the mass timber building than the concrete building.

As shown in Fig 2, the total mass of the mass timber building is about 68% of the functionally equivalent concrete building. It is important to keep in mind that the density of concrete (2400 kg/m^3) is much higher than the density of the two mass timber products and wood building products in general (550 kg/m^3). The total mass of the concrete building is 7.5 million kg for all

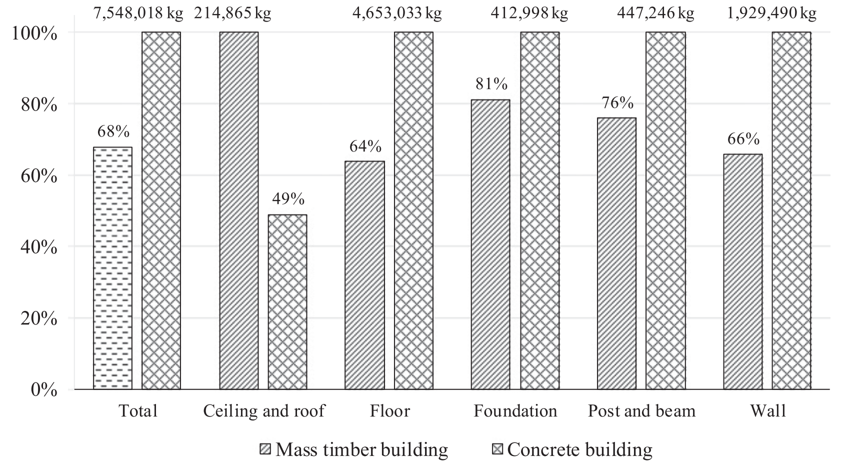

Figure 2. Total mass for the mass timber and concrete buildings.

five structural sections, whereas the total mass of the mass timber building is only 5.1 million kg. In the mass timber building, except for the ceilings-roof which uses more gypsum board and attached materials for fire resistance purposes, the mass of other building sections is about 64%, 81%, 76%, and 66% of the concrete building for floors, foundation, postbeams, and walls, respectively. Light-weight mass timber buildings tend to have lower carbon footprints and lower costs than heavier concrete buildings (Connolly et al 2018).

Environmental Impact Analysis

The comparative cradle-to-site whole-building LCIA results for 1-m^2 floor area of the mass timber and concrete buildings are shown in Table 2. Global warming contribution of the mass timber building was found to be 18% lower (193 kg CO_2-eq/m^2) than that of the concrete building (237 kg CO_2-eq/m^2). The mass timber building performs better in eutrophication than the concrete building (47% lower), whereas the concrete building has better performance in the impact categories of smog and acidification (3% and 16% lower than the mass timber building, respectively). In addition, the two buildings are essentially the same for ozone depletion (1% difference), which is dominated by the use of polystyrene insulation boards (XPS). The two buildings used about the same amount of XPS.

Primary energy consumption, also called CED, which describes the direct and indirect energy use throughout the life cycle of products, is an

Table 2. Life-cycle impact assessment results for 1-m^2 floor area by building types.

Impact category	Unit	Mass timber building	Concrete building	Percentage difference (%)
Global warming	kg CO_2 eq	193	237	18
Ozone depletion	kg CFC-11 eq	1.91E-04	1.93E-04	1
Smog	kg O_3 eq	15.74	15.22	3
Acidification	kg SO_2 eq	1.03	0.89	16
Eutrophication	kg N eq	0.19	0.36	47
Total primary energy	MJ	2868	2673	7
Nonrenewable, fossil	MJ	2344	2371	1
Nonrenewable, nuclear	MJ	198	242	18
Renewable	MJ	326	61	439

important driver of environmental impacts and is indicative for many environmental problems (Huijbregts et al 2006). As shown in Table 2, the mass timber building has 7% higher CED than the concrete building, which is mainly caused by the relatively higher unit CED of 2629 MJ/m^3 for CLT than 1540 MJ/m^3 for concrete. In the mass timber building, the large mass of CLT results in CLT having the highest CED (23% of building CED), followed by 22% for glulam. In the alternative concrete building, rebar has the greatest CED, accounting for 47% of building CED, followed by 21% for concrete. In addition, fossil fuel accounts for 82-90% of building CED for both buildings, and renewable energy accounts for 12% and 2% of building CED for the mass timber and concrete buildings, respectively. As expected, the mass timber building also uses substantially higher renewable energy than its concrete alternative building. This was mainly because of the wood products manufacturer using mill residue as an alternative heating source to dry the lumber before CLT and glulam production (Bowers et al 2017; Chen et al 2019).

The normalized environmental impacts and energy demand for the mass timber and concrete buildings at different life-cycle phases are illustrated in Fig 3. The product phase (modules A1-A3) is the dominant contributor, accounting for 88-98% in all impact categories for the two buildings. The transport phase (module A4) contributes 3-8% to the impact categories of global warming, smog, acidification, eutrophication, total primary energy, and fossil fuel consumption for the mass timber building. Because of the longer transportation distance of wood building materials from the manufacturer to construction site, eg 320 vs 24 km for materials for the concrete building, the transportation impacts are greater in the mass timber case than in the all-concrete case.

The construction installation phase (module A5) contributes a small fraction (1-5%) in all impact categories except for ozone depletion, although the impacts are consistently lower for the less dense mass timber materials than for concrete. The assessment for this phase, module A5, is based on the estimation of diesel consumption used for lifting all the building materials (Finlayson 2019). When more empirical data on diesel consumption by construction equipment and time for mass timber building construction are collected, more precise LCIA results can be reported for this phase.

Figure 4 compares the product phase (modules A1-A3) environmental impacts from each building section for the mass timber and concrete buildings. Floor is the largest contributor for both

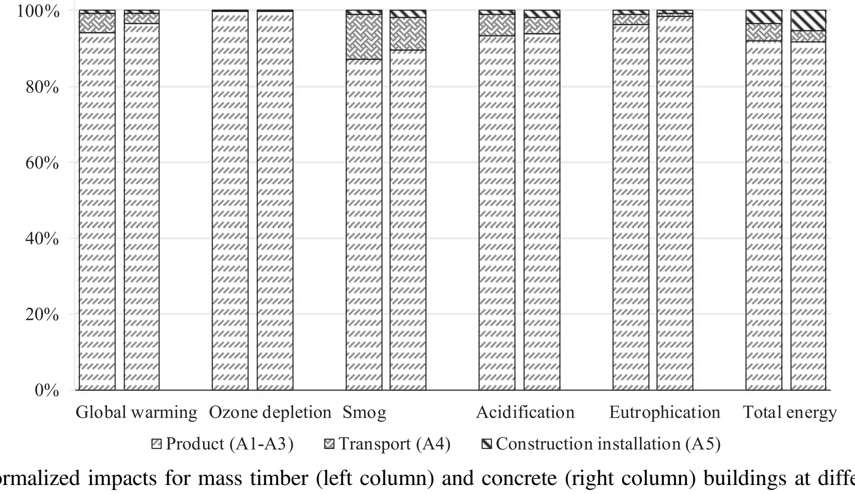

Figure 3. Normalized impacts for mass timber (left column) and concrete (right column) buildings at different life-cycle phases.

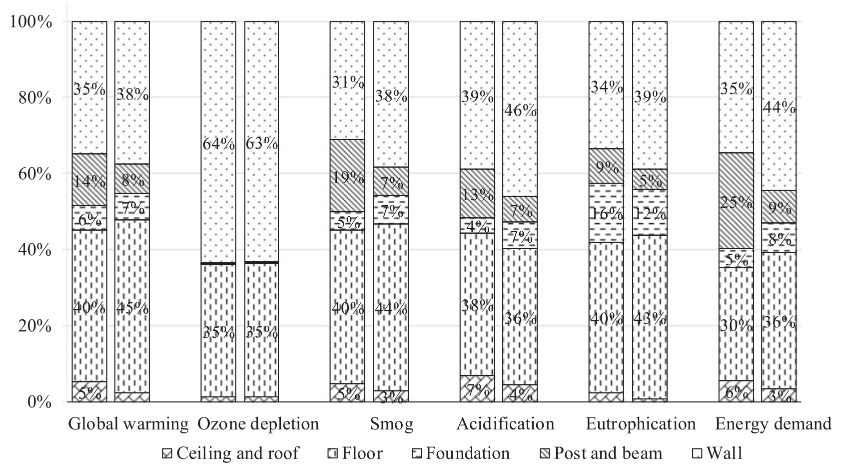

Figure 4. Normalized impacts (A1-A3) from different building sections for mass timber (left column) and concrete (right column) buildings.

buildings and accounts for 40-50% in the impact categories of global warming, smog, and eutrophication, followed by the wall sections at 31-39%. Specifically, the building materials of CLT and concrete/rebar in the mass timber building, as well as concrete/rebar in the concrete building, dominate the impacts of the floor component (93-99%). For acidification, wall is the largest contributor, accounting for 39-48% of impacts for the two buildings. This is caused by the use of gypsum board. Wall and floor together contribute 98-99% of the ozone depletion for both buildings, which is because of the insulation material, eg polystyrene boards (XPS). Although XPS accounts for only 0.4% of the total mass quantities in wall components in the mass timber building, it contributed more than 99% of total ozone depletion impacts. We strongly suggest that an alternative product be considered. The ceilings-roof, foundation, and postbeams together contribute less than 27% to all impact categories for the two buildings, although the mass timber building contributes a higher fraction than the concrete building in these building sections. These differences are caused by the greater use of gypsum board in the ceilings-roof, rebar in the foundation, and glulam in the postbeams.

MasterFormat is a standard for organizing specifications and other written information for commercial and institutional building projects in the United States and Canada. It provides a structured hierarchy for building construction requirements and associated activities (Tecchio et al 2018). To further analyze the environmental impact contribution, the building materials' LCIs were grouped into individual construction divisions based on MasterFormat (CSI 2016), eg *Division 03: Concrete*; *Division 04: Masonry*; *Division 05: Metals*; *Division 06: Wood, Plastics, Composites*; *Division 07: Thermal and Moisture Protection*; and *Division 09: Finishes*. Figure 5 shows the comparison in the product phase (modules A1-A3) of the environmental impacts from individual construction divisions for the mass timber and concrete buildings. The two buildings have significant differences in all impact categories except for ozone depletion, which was dominated by *Division 07* for both buildings, eg XPS. Because the insulation packages are essentially the same, materials such as XPS, aluminum panels, mineral wool, sealants, adhesives, and TPO membranes dominate the impacts. The total environmental impact of the concrete building is dominated by *Division 03,* consisting of concrete and rebar, which accounts for 72%, 72%, 61%, and 86% in the impact categories of global warming, smog, acidification, and eutrophication, respectively. These numbers range

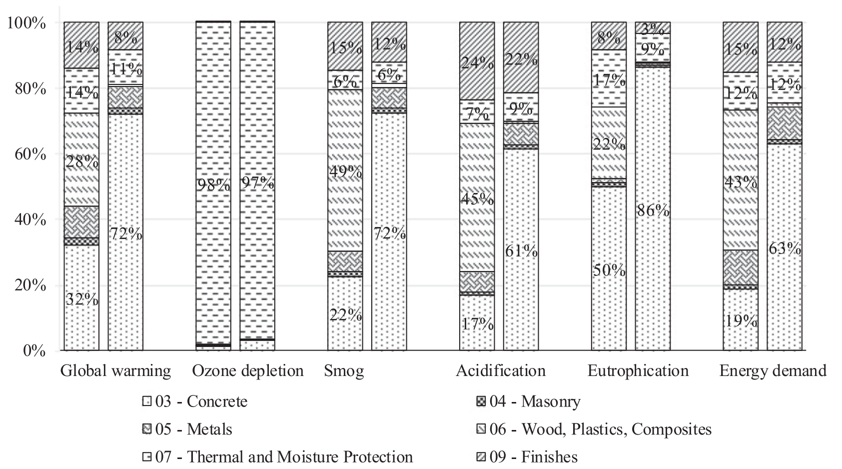

Figure 5. Normalized impacts (A1-A3) of different construction divisions for mass timber (left column) and concrete (right column) buildings.

from 17% to 50% for the mass timber building. Approximately 58% concrete and 72% rebar in *Division 03* of the concrete building are substituted by 2339 m^3 CLT and glulam in *Division 06* for the mass timber building, which resulted in *Division 06* in the mass timber building contributing 28%, 49%, 45%, and 22% to the impact categories of global warming, smog, acidification, and eutrophication, respectively. The two buildings have no significant differences in *Division 07*. In addition, in the mass timber building, *Division 09* contributes relatively more to all impact categories than it does in the concrete building, which is largely caused by the use of 36% more gypsum board in the mass timber building compared with the concrete building.

Carbon Analysis

Biogenic carbon refers to CO_2 emissions that originate from biological sources such as plants, trees, and soil (Harris et al 2018). In LCA studies of durable wood products, it is assumed that harvested timber products will be replaced sustainably by new growth in managed forest land, and therefore, the biogenic CO_2 emissions are considered to be carbon neutral from the climate change prospective. The Intergovernmental Panel on Climate Change (IPCC 2006) also supports the carbon neutral hypothesis, which considers the CO_2 emissions from biomass as part of the natural carbon cycle. The carbon in wood is accounted for as stored CO_2 during the lifetime of the product or building. Such stored carbon is estimated by average carbon content of 50% of dry mass of wood products. This analysis assumed a service life of 100 yr for the mass timber building, which equates to the same time frame used for accounting for GHG emissions in this study. As shown by this LCI analysis and Fig 6, biogenic CO_2 emissions from the mass timber and concrete buildings were 81 and 3.4 kg/m^2 of floor area, respectively. These results were closely aligned with the consumption of renewable energy in the two building types. The

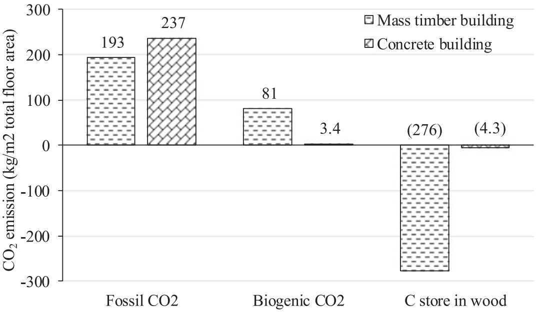

Figure 6. Comparison of CO_2 emissions for mass timber and concrete buildings under different carbon accountings.

sequestered CO_2 values in wood products were about 276 and 4.3 kg/m^2 of floor area for the mass timber and concrete buildings, respectively. As previously calculated, the GHG emissions for the CLT and concrete buildings were 193 and 237 kg CO_2 eq/m^2 of floor area, respectively, which is about 18% different. More significant differences were observed when biogenic carbon and sequestered carbon were combined, resulting in values of CO_2 emissions for the mass timber and concrete buildings of 2.7 and 236 kg/m^2 of floor area, respectively.

Sensitivity Analysis

Transportation accounts for about 5% of the total global warming impact emissions for the mass timber building. But given that there is a limited production infrastructure for CLT, it is worth considering the effects of different production/transportation alternatives on the overall global warming impacts of the building. CLT transportation has a global warming impact of 3.4 kg CO_2 eq/m^2 of floor area when assuming local production and transportation (320 km). Thus, with limited CLT production in the United States, it is important to understand the implication of longer transportation distances. Figure 7 shows the global warming impacts of different CLT transport distances and, specifically, the effects of sourcing CLT from INW, the eastern United States, or Europe. This analysis shows that the global warming impact of CLT transportation increased from 3.4 to 8.2, to 16, to 47 kg CO_2 eq/m^2 of floor area with the increasing levels of distance, respectively (Fig 7), which also accounts for 59%, 73%, and 89% of total impacts in the transport phase (module A4) for the three projected regions. Meanwhile, as shown in Fig 7, the transport phase (module A4) contributes less than 5% to the global warming impact compared with the module A1-A5 impacts under the local (within 320 km) assumption, and this ratio would increase to 7%, 10%, and 22%, respectively, if sourcing the CLT material from the three other locations.

The CLT transport distance revealed substantial changes to the differences between the mass timber and concrete buildings in environmental impacts. The mass timber building outperformed the concrete building in global warming with 18%, 16%, and 13% lower impacts when sourcing from local and projected INW and eastern US regions. However, the mass timber building had 0.2% higher global warming impact than the concrete

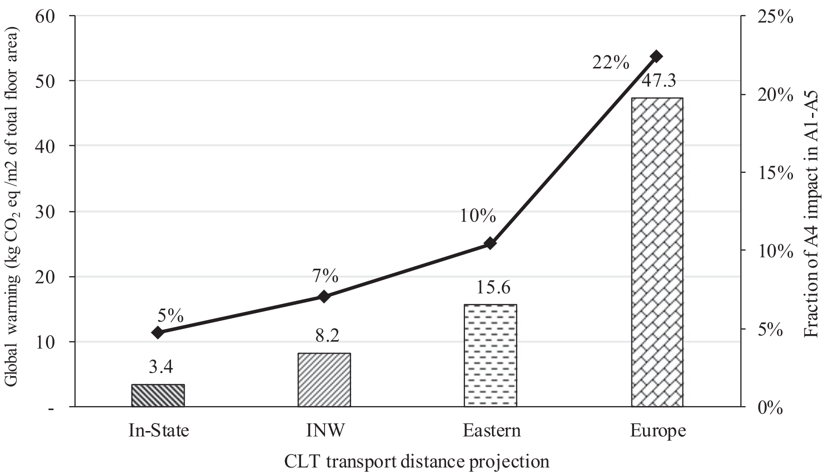

Figure 7. Global warming impacts from transportation of cross-laminated timber (CLT) materials to the building site (assuming CLT manufacturer in-state with a distance of 375 miles; INW: Inland Northwest region in the United States, Eastern: eastern United States, and Oversea: Europe) and the fraction of A4 impact in total A1-A5 for the mass timber building.

building if CLT was sourced from Europe because of the long transportation distance.

Study Strengths and Limitations

A strength of this study was the detailed building design information for two equivalent buildings by our highly qualified architectural and structural engineering partners. The study evaluated the structural elements, modules A1-A5, with the expectations that the bill of the materials for the two buildings incorporated the same U-value. Although mineral wool was incorporated into the analysis at the same quantities, its impacts were less than 0.3%. A limitation of this study was the lack of detail on the CLT manufacturing process. This lack of detail includes energy consumption in the manufacturing, proprietary details such as resin use, and the yield of CLT from dimensional lumber. In this study, the building components were limited to elements that were most different, eg the ceilings-roof, floors, foundation, post-beams, and walls. Other equivalent components such as windows, doors, plumbing, and electricity were excluded in the scope of this study. Construction site data including equipment use, electricity use, and labor count were not available for this study. Therefore, an empirical equation provided by the Athena Sustainable Materials Institute and based on building height was used. As with any LCA, additional data on specific processes, eg CLT manufacturing to concrete transportation, will improve the value of the analysis.

CONCLUSION

In this study, a comparative cradle-to-site LCA of a mass timber, tall wood building, and a functionally equivalent concrete building was conducted. This study shows that the mass timber building outperformed the concrete building on a number of environmental impact categories, eg global warming, ozone depletion, and eutrophication, whereas the concrete building showed better performance on smog and acidification, as well as total primary energy demand. The product phase (modules A1-A3) contributed more than 66% of total cradle-to-site (modules A1-A5) impacts in all impact categories for both buildings. The mass timber building was much lighter, 68% of the total weight of the concrete building. Even with the use of concrete and steel in the foundation and CLT floor systems, the mass timber building used 58% of the concrete and 72% of the rebar of the concrete building. Floors and walls were major environmental contributors in building sections. CLT and concrete were the hotspot for the tall wood building, and concrete and rebar were the hotspot for the concrete building. The required use of more gypsum board for mid- to high-rise mass timber buildings to comply with building codes increased global warming, smog, acidification, and eutrophication. The CLT building had lower CO_2 emissions than the concrete building when biogenic carbon and sequestered carbon were included. Sensitivity analysis showed that the environmental impacts for CLT transport distance, including sourcing from European countries, could reverse the advantages of all impact categories for the mass timber building in this study. Further work will focus on environmental impacts at the use and end-of-life phases.

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REFERENCES

APA (2012) ANSI/APA PRG 320-2012 Standard for performance-rated cross-laminated timber. APA – The Engineered Wood Association, Tacoma, WA.

APA (2019) Case study: Mass timber has banks seeing green. https://www.apawood.org/publication-search?q=Mass+

Timber+has+Banks+Seeing+Green&tid=1 (2 September 2019).

ASTM (2016) ASTM E2911: Practice for minimum criteria for comparing whole building life cycle assessments for use with building codes, standards, and rating systems. American Society for Testing and Materials, West Conshohocken, PA.

Bare J, Young D, Hopton M (2012) Tool for the reduction and assessment of chemical and other environmental impacts 2.1. STD Standard Operating Procedure (SOP) SOP No. S-10637-OP-1-0.

Berardi U (2017) A cross-country comparison of the building energy consumptions and their trends. Resour Conserv Recycl. 123:230-241.

Bowers T, Puettmann ME, Ganguly I, Eastin I (2017) Cradle-to-gate life-cycle impact analysis of glued-laminated (glulam) timber: Environmental impacts from glulam produced in the US Pacific northwest and southeast. For Prod J 67(5-6):368-380.

Bowick M (2015) Wood innovation and design centre Prince George, BC: An environmental building declaration according to EN 15978 standard. Athena Sustainable Materials Institute, Ottawa, ON, Canada. pp. 1-54.

Bowick M (2018) Athena Brock Commons Tallwood House, University of British Columbia: An environmental building declaration according to EN 15978 standard. Athena Sustainable Materials Institute, Ottawa, ON, Canada. pp. 1-55.

Breneman S, Richardson D (2019) Tall wood buildings and the 2021 IBC: Up to 18 stories of mass timber. WW-WSP-12. Wood Works. pp. 1-11.

Cadorel X, Crawford R (2018) Life cycle analysis of cross laminated timber in buildings: A review. Pages 107-114 *in* Engaging Architectural Science: Meeting the Challenges of Higher Density: 52nd International Conference of the Architectural Science Association and RMIT University, Melbourne, VIC, Australia.

Chen CX, Pierobon F, Ganguly I (2019) Life cycle assessment (LCA) of cross-laminated timber (CLT) produced in western Washington: The role of logistics and wood species mix. Sustainability 11:1278.

Connolly T, Loss C, Iqbal A, Tannert T (2018) Feasibility study of mass-timber cores for the UBC tall wood building. Buildings 8:98.

CSI (2016) MasterFormat 2016. Construction Specifications Institute (CSI), Alexandria, VA.

DATASMART 2019. LCI package (US-EI SimaPro® Library). https://ltsexperts.com/services/software/datasmart-life-cycle-inventory/ (26 March 2020).

Durlinger B, Crossin E, Wong J (2013) Life cycle assessment of a cross laminated timber building. Forest & Wood Products Australia, Melbourne, VIC, Australia. ISBN: 978-1-921763-63-2, pp. 1-110.

EN (2011) EN 15978: Sustainability of construction works - Assessment of environmental performance of buildings - Calculation method. European Committee for Standardization, Brussels, Belgium.

Espinoza O, Buehlmann U (2018) Cross-laminated timber in the USA: Opportunity for hardwoods? Curr For Rep 4: 1-12.

FII (2016) Brock Commons Tallwood House. Forestry innovation investment (FII), naturally: Wood. https://www.naturallywood.com/emerging-trends/tall-wood/brock-commons-tallwood-house (17 July 2019).

Finlayson G (2019) Personal communication. Athena Institute.

Grann B (2013) A comparative life cycle assessment of two multistory residential buildings: Cross-laminated timber vs. concrete slab and column with light gauge steel walls. FPInnovations Report, Vancouver, BC, Canada. pp. 1-121.

Gu H, Bergman R (2018) Life cycle assessment and environmental building declaration for the design building at the University of Massachusetts. General Technical Report FPL-GTR-255. U.S. Department of Agriculture, Forest Service, Forest Products Laboratory, Madison, WI. 71 pp.

Gustavsson L, Joelsson A, Sathre R (2010) Life cycle primary energy use and carbon emission of an eight-storey wood-framed apartment building. Energy Build 42(2): 230-242.

Harris ZM, Milner S, Taylor G (2018) Biogenic carbon – Capture and sequestration. *In* Greenhouse Gas Balances of Bioenergy Systems, Chapter 5. Academic Press, Cambridge, MA. pp. 55-76.

Heppner J (2019) Personal communication. LEVER Architecture.

Huijbregts MA, Rombouts LJ, Hellweg S, Frischknecht R, Hendriks AJ, van de Meent D, Ragas AM, Reijnders L, Struijs J (2006) Is cumulative fossil energy demand a useful indicator for the environmental performance of products? Environ Sci Technol 40(3):641-648.

IPCC (2006) Guidelines for national greenhouse gas inventories. Institute for Global Environmental Strategies (IGES) for the IPCC, Kanagawa, Japan.

ISO (2006a) ISO 14040: Environmental management – Life cycle assessment – Principles and framework. International Organization for Standardization, Geneva, Switzerland.

ISO (2006b) ISO 14044: Environmental management – Life cycle assessment – Requirement and guidelines. International Organization for Standardization, Geneva, Switzerland.

Jones K, Stegemann J, Sykes J, Winslow P (2016) Adoption of unconventional approaches in construction: The case of cross-laminated timber. Constr Build Mater 125:690-702.

Karacabeyli B, Douglas B (2013) CLT Handbook, US edition. https://www.fpl.fs.fed.us/documnts/pdf2013/fpl_2013_gagnon001.pdf (17 July 2019).

Kelley S, Bergman R (2017) Potential for tall wood buildings to sequester carbon, support forest communities, and create new options for forest management. Forest Products Laboratory Research in Progress. https://www.fpl.fs.fed.us/documnts/rips/fplrip-4851-018-NCSU-TallBldgs-Bergman-Kelley.pdf (26 March 2020).

Kremer PD, Symmons MA (2015) Mass timber construction as an alternative to concrete and steel in the Australia building industry: A PESTEL evaluation of the potential. Int Wood Prod J 6(3):138-147.

Oregon BEST (2017) Advanced wood product manufacturing study for cross-laminated timber acceleration in Oregon & SW Washington. Technical Report. Oregon BEST, Portland, OR. pp. 1-111.

Perez-Garcia J, Lippke B, Briggs D, Wilson JB, Bowyer J, Meil J (2005) The environmental performance of renewable building materials in the context of residential construction. Wood Fiber Sci 37:3-17.

Pierobon F, Huang M, Simonen K, Ganguly I (2019) Environmental benefits of using hybrid CLT structure in midrise non-residential construction: An LCA based comparative case study in the US PNW. J Build Eng 26: 100862.

Podesto L, Berneman S (2016) CLT research: Available and accessible to North American building designers. Wood Design Focus 26(1):3-7.

Ritter M, Skog K, Bergman R (2011) Science supporting the economic and environmental benefits of using wood and wood products in green building construction. General Technical Report FPL-GTR-206. U.S. Department of Agriculture, Forest Service, Forest Products Laboratory, Madison, WI. 9 pp.

Robertson AB, Lam FC, Cole R (2012) A comparative cradle-to-gate life cycle assessment of mid-rise office building construction alternatives: Laminated timber or reinforced concrete. Buildings 2:245-270.

Smith RE, Grifin G, Rice T, Hagehofer-Daniell B (2018) Mass timber: Evaluating construction performance. Architectural Engineering and Design Management 14(1-2): 127-138.

Tecchio P, Gregory J, Olivetti E, Ghattas R, Kirchain R (2018) Streamlining the life cycle assessment of buildings by structured under-specification and probabilistic triage. J Ind Ecol 23(1):268-279.

Walch F, Watts R (1923) Composite lumber. U.S. Patent 1,465,383.

Williamson T, Ross R (2016) Proceedings: Mass timber research workshop 2015. General Technical Report FPL-GTR-241. U.S. Department of Agriculture, Forest Service, Forest Products Laboratory, Madison, WI. 364 pp.

ENVIRONMENTAL STUDIES

Natural climate solutions for the United States

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Limiting climate warming to <2°C requires increased mitigation efforts, including land stewardship, whose potential in the United States is poorly understood. We quantified the potential of natural climate solutions (NCS)—21 conservation, restoration, and improved land management interventions on natural and agricultural lands—to increase carbon storage and avoid greenhouse gas emissions in the United States. We found a maximum potential of 1.2 (0.9 to 1.6) Pg CO_2e year^{-1}, the equivalent of 21% of current net annual emissions of the United States. At current carbon market prices (USD 10 per Mg CO_2e), 299 Tg CO_2e year^{-1} could be achieved. NCS would also provide air and water filtration, flood control, soil health, wildlife habitat, and climate resilience benefits.

INTRODUCTION

Limiting global warming below the 2°C threshold set by the Paris Climate Agreement is contingent upon both reducing emissions and removing greenhouse gases (GHGs) from the atmosphere (*1*, *2*). Natural climate solutions (NCS), a portfolio of discrete land stewardship options (*3*), are the most mature approaches available for carbon conservation and uptake compared to nascent carbon capture technologies (*4*) and could complement increases in zero-carbon energy production and energy efficiency to achieve needed climate change mitigation. Within the United States, the maximum and economically viable mitigation potentials from NCS are unclear.

Here, we quantify the maximum potential for NCS in the United States and the portion of this maximum that could be achieved at several price points. We consider 21 distinct NCS to provide a consistent and comprehensive exploration of the mitigation potential of conservation, restoration, and improved management in forests, grasslands, agricultural lands, and wetlands (Fig. 1), carefully defined to avoid double counting (details in the Supplementary Materials). We estimate the potential for NCS in the year 2025, which is the target year for the United States' Nationally Determined Contribution (NDC) under the Paris Agreement to reduce GHG emissions by 26 to 28% from 2005 levels. Our work refines a coarser-resolution global analysis (*3*) and updates and expands the range of options considered in previous analyses for the United States (*5*–*8*).

For each NCS opportunity (Fig. 1 and the Supplementary Materials), we estimate the maximum mitigation potential of GHGs measured in CO_2 equivalents (CO_2e), given the below constraints. We then estimate the reductions obtainable for less than USD 10, 50, and 100 per Mg CO_2e. Current carbon markets pay around USD 10 (*9*). The social cost of carbon in 2025 is approximately USD 50, using a 3% discount rate (*10*). However, a price of at least USD 100 is thought to be needed to keep the 100-year average temperature from warming more than 2.5°C (*11*), and an even higher price may be needed to meet the Paris Agreement <2°C target. Many NCS also generate co-benefits, which, even without a price on carbon, provide incentives to invest in NCS implementation. We identified co-benefits generated by each NCS in four categories of ecosystem services: air, biodiversity, water, and soil (Fig. 1 and table S2).

To avoid conflicts with other important societal goals for land use, we constrain our maximum estimate to be compatible with human needs for food and fiber (Supplementary Materials). Within these constraints, 5.1 Mha of cropland can be restored to grasslands, forests, and wetlands, equal to the area that has left the Conservation Reserve Program (CRP) since 2007 (*8*) and less than half the land currently dedicated to corn ethanol. We also estimate that 1.3 Mha of pasture could be reforested without affecting livestock production, assuming recent improvements in efficiency continue (see the Supplementary Materials). We assume that timber production can temporarily decrease by 10%, which maintains timber production

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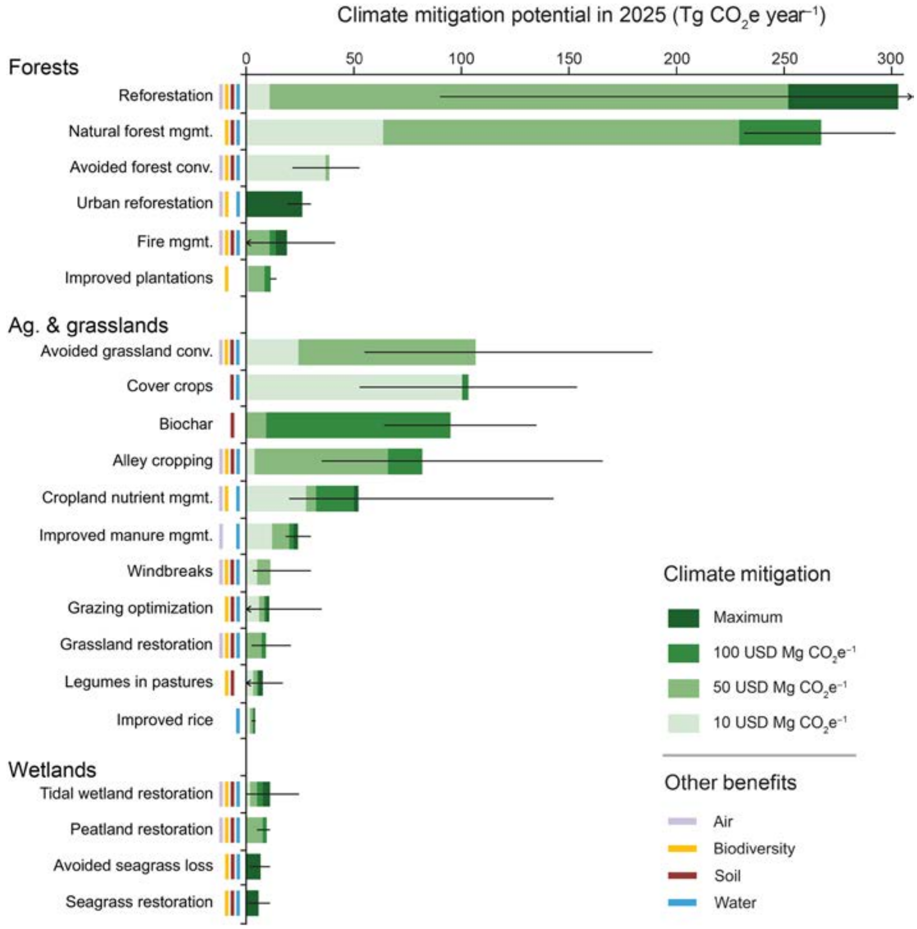

Fig. 1. Climate mitigation potential of 21 NCS in the United States. Black lines indicate the 95% CI or reported range (see table S1). Ecosystem service benefits linked with each NCS are indicated by colored bars for air (filtration), biodiversity (habitat protection or restoration), soil (enrichment), and water (filtration and flood control). See the Supplementary Materials for detailed findings and sources.

levels within the historic range of variation and enables managed forests and plantations to transition to longer harvest rotations (see the Supplementary Materials). We assume that extensive natural forests on private lands can all undergo harvest extension, with the temporary loss of timber supply replaced by reforestation and thinning for fire risk reduction (*12*) or with thinning or select harvest practices that still provide timber but maintain carbon levels (Supplementary Materials) (*13*, *14*). We further constrain our analysis to avoid impacts on biodiversity. This biodiversity constraint precludes both the conversion of natural habitat to energy crops and the afforestation of native grasslands.

RESULTS

We find a maximum additional NCS mitigation potential of 1.2 Pg CO_2e year^{-1} [95% confidence interval (CI), 0.9 to 1.6 Pg CO_2e year^{-1}] in the year 2025 (Fig. 1 and table S1). This is 21% of the 5794.5 Tg CO_2e of net emissions in 2016 (*15*). The majority (63%) of this potential comes from increased carbon sequestration in plant biomass, with 29% coming from increased carbon sequestration in soil and 7% coming from avoided emissions of CH_4 and N_2O. At the USD 10, 50, and 100 price points, 25, 76, and 91%, respectively, of the maximum mitigation would be achieved. This means that 1.1 Pg CO_2e year^{-1} are available at USD 100 per Mg CO_2e, which equals the emission reductions needed to meet the U.S. NDC under the Paris Agreement (see the Supplementary Materials). If NCS were pursued in combination with additional mitigation in the energy sector, then it would therefore enable the United States to exceed its current NDC ambition. This is important because, globally, current NDCs (7 to 9 Pg CO_2e year^{-1}) would need to be dramatically increased (by an additional 10 to 16 Pg CO_2e year^{-1}) to limit warming below 2°C (*16*).

This estimate of maximum NCS potential is similar to or higher than several previous syntheses of mitigation opportunities in the land sector. For example, the United States Mid-Century Strategy for Deep Decarbonization estimated a potential land sink of 912 Tg CO_2e year^{-1}, 30% lower than our estimate (*5*). While other efforts have focused on the forest sector (*7*) or the agricultural sector (*6*), this analysis presents a comprehensive and up-to-date synthesis of NCS opportunities in the United States. For example, this analysis considers potential additional mitigation from tidal wetlands and seagrass ("blue carbon"), which has been comprehensively analyzed for its current status in the United States (*17*), but not its potential for additional mitigation.

Reforestation has the single largest maximum mitigation potential (307 Tg CO_2e year^{-1}). The majority of this potential occurs in the northeast (35%) and south central (31%) areas of the United States (fig. S1). This mitigation potential increases to 381 Tg CO_2e year^{-1} if all pastures in historically forested areas are reforested. Previous estimates of reforestation potential range widely from 208 to 1290 Tg CO_2e year^{-1} (*7*). Higher estimates than ours can be obtained by reforesting or afforesting areas that we excluded (e.g., productive crop and pasture lands and natural grasslands) and/or by using rates of carbon sequestration from plantation systems rather than from natural regenerating forests [e.g., (*7*)].

Natural forest management of privately held forests has the second largest maximum mitigation potential (267 Tg CO_2e year^{-1}). This maximum mitigation is achieved by extending harvest cycles. Mitigation can also be achieved through forest management practices such as reduced impact logging and improved silvicultural practices that release suppressed forest growth (*18*–*20*), although often at lower sequestration rates than extending harvest cycles. These management practices can be implemented at low or no net cost (*21*, *22*) and do not require a change in business-as-usual (BAU) land use or ownership rights.

Another promising opportunity associated with forests is fire management (18 Tg CO_2e year^{-1}; fig. S6). Fire management entails restoring frequent, low-intensity, understory fires in fire-prone forest ecosystems to reduce the potential for high-severity wildfires (*23*). The primary carbon benefit from fire management is avoiding decreased net ecosystem production from tree-killing wildfire. In the absence of improved fire management, climate change is expected to continue to increase the frequency of high-severity fires and compromise the ability of forests to regenerate following these fires (*24*). The high uncertainty associated with the climate mitigation benefits of fire management would be reduced by additional research to quantify the carbon storage benefits of prescribed fire across a diversity of forest types, including the length of time that prescribed fire reduces the risk of subsequent high-severity fires.

Avoided conversion protects carbon stored in extant forests and grasslands from ongoing losses. More than two-thirds of the avoided forest conversion potential (38 Tg CO_2e year^{-1}) occurs in the Southern and Pacific Northwest regions (table S14 and fig. S9). Many of the most intensive areas of rapid forest conversion were located near urban zones, with additional hot spots in recent agricultural expansion zones (such as California's Central Valley) and semi-arid regions of the West. Avoided conversion of grassland to cropland prevents emissions from soils and root biomass (107 Tg CO_2e year^{-1}; fig. S12). The emissions from grassland conversion exceed the emissions from forest conversion because both the rate of conversion and the per hectare emissions are higher (table S1). Cropland expansion is a major cause of conversion that affects grasslands much more than forests (*25*). The higher rate of emissions occurs because the conversion of grasslands to croplands results in a 28% loss of soil carbon from the top meter of soil (*26*). This generates 125 Mg CO_2e ha^{-1} in emissions, comprising 81% of the emissions from grassland conversion (see the Supplementary Materials). Because research shows conflicting conclusions regarding the impact of forest conversion in the United States on soil carbon, we do not include the soil carbon pool in our estimate of emissions from forest conversion (see the Supplementary Materials).

Carbon sequestration opportunities in croplands include the use of cover crops and improved cropland nutrient management. Cover crops, grown when fields are normally bare, provide additional carbon inputs to soils. Growing cover crops on the 88 Mha of the five primary crops in United States not already using cover crops presents a substantial opportunity for mitigation (103 Tg CO_2e year^{-1}). Cover crops are increasingly used by U.S. farmers to improve soil health, yields, and yield consistency (*27*). Improved management of nitrogen fertilizers reduces N_2O emissions and avoids fossil fuel emissions associated with fertilizer production (52 Tg CO_2e year^{-1}). Fertilizer rates can be reduced while maintaining yields by using precision agriculture to apply only the amount required in each part of the field and by splitting fertilizer applications to match the timing and supply of fertilizer with crop demand (see the Supplementary Materials). Emissions can also be reduced by switching from anhydrous fertilizer to urea, which has lower N_2O emission (*6*).

The agronomic practices of biochar incorporation (95 Tg CO_2e year^{-1}) and alley cropping (planting widely spaced trees interspersed with a row crop; 82 Tg CO_2e year^{-1}) also have high maximum potential. However, current adoption is negligible due to a variety of cultural, technological, and cost barriers that would need to be overcome if these practices were to achieve their mitigation potential (*28*, *29*).

Tidal wetland restoration is the largest wetland NCS (12 Tg CO_2e year^{-1}). Roughly 27% of U.S. salt marshes are disconnected from the ocean and subject to freshwater inundation. This results in a large increase in CH_4 emissions from these "freshened" salt marshes. Reconnecting salt marshes with the ocean, such as via culverts under roads or other barriers, can avoid these CH_4 emissions (*30*).

The 10 opportunities described above account for 90% (1082 Tg CO_2e year^{-1}) of the maximum NCS mitigation potential across all 21 opportunities. An additional 11 opportunities, which sum to 122 Tg CO_2e year^{-1}, account for just 10% of the maximum potential. However, these NCS may offer optimal ecological and economic opportunities at local scales (Fig. 1 and Supplementary Materials). For example, peatland restoration offers a high per hectare mitigation benefit, especially in regions of the United States with warm temperate climates (8.2 Mg CO_2e ha^{-1} year^{-1}).

Lower-cost opportunities represent particularly promising areas for increased near-term investment. We identified 299 Tg CO_2e year^{-1} of NCS opportunities that could be realized for USD 10 Mg CO_2e^{-1} or less (table S1), a price that is in line with many current carbon markets (*9*). The two largest lower-cost opportunities are improved management practices: cover crops (100 Tg CO_2e year^{-1}) and improved natural forest management (64 Tg CO_2e year^{-1}). Both of these practices, along with planting windbreaks (5 Tg CO_2e year^{-1}) and legumes in pastures (3 Tg CO_2e year^{-1}), have the potential to increase yields (*21*, *22*, *27*) and therefore to generate additional revenue for landowners. Improved manure management can also provide low-cost mitigation (12 Tg CO_2e year^{-1}) (*8*). In addition, lower-cost NCS include increased efficiencies (cropland nutrient management, 28 Tg CO_2e year^{-1}; grazing optimization, 6 Tg CO_2e year^{-1}) and avoided conversion (avoided forest conversion, 37 Tg CO_2e year^{-1}; avoided grassland conversion, 24 Tg CO_2e year^{-1}).

By itself, the marginal abatement cost gives an incomplete picture of the potential for implementation of NCS, in part because NCS provide a variety of co-benefits (Fig. 1 and table S2). The values of these co-benefits are not captured in our marginal abatement costs yet may drive NCS implementation. For example, investments in fire management are needed to avoid impacts on air quality and drinking water provision; urban forestry provides human health, aesthetic, and direct temperature reduction benefits; nutrient management is needed to improve water quality and avoid toxic algal blooms (table S2). Further, NCS can help provide resilience to climate change impacts on nature and people. For example, building soil carbon increases the resilience of cropland (*31*); protecting coastal wetlands can provide coastal defense against storms (*32*); and fire management can help avoid damaging wildfires (*23*).

We have restricted our analysis to those opportunities where the literature conclusively demonstrates the potential for mitigation. This suggests that new research may reveal additional opportunities for NCS, which would increase the potential identified here. At the same time, substantial uncertainties exist in some NCS opportunities (Fig. 1 and table S1), highlighting the need for implementation to be coupled with monitoring and assessment of NCS.

DISCUSSION

The United States is the largest cumulative emitter of carbon dioxide from fossil fuels (*33*). Despite the immense size of U.S. GHG emissions from fossil fuel use, we find that NCS have the potential to generate mitigation equivalent to 21% of net annual emissions. This reveals the important contribution to climate mitigation that the land sector can make, even in developed countries such as the United States.

Globally, current NCS efforts receive only 0.8% of public and private climate financing (*34*), despite offering roughly 37% of potential mitigation needed through 2030 (*3*). One concern that may have limited the adoption of NCS to date includes competition with other land uses such as food and bioenergy production. A growing body of literature suggests that future global food demand can be met via investments in yield increases, closing yield gaps, diet shifts, aquaculture, and biofuel policy, without the need to further expand cropland into natural areas (*35*, *36*). In the United States, marginal cropland, much of which is unprofitable (*37*), could be restored to grassland or forests with net societal benefits (*38*). Similarly, NCS may compete with bioenergy production. However, this conflict can be reduced or avoided depending on the form of bioenergy production or NCS. Some forms of biomass production, such as residues and wastes, or high-yielding methods, such as algae, do not require productive land (*39*). Our grassland restoration pathway could produce a limited amount of additional biomass while maintaining carbon sequestration in soils if low-productivity croplands are converted to perennial energy grasses (*40*). Further, NCS based on improved management of existing land uses do not create land use conflict and can even increase productivity within that land use (e.g., fire management or cover crops). However, aggressive expansion of dedicated bioenergy crops, given the large land requirement of both first- and second-generation bioenergy crops (*41*), would be likely to reduce the mitigation potential available through NCS, notably via reforestation, avoided grassland conversion, and natural forest management.

A second concern is that ecosystems have a limited ability to store additional carbon. For each pathway, we quantified the duration of time for which mitigation is expected to occur at the rates we estimate, before saturation effects decrease this rate (table S1). We note that carbon can continue to accumulate in forests for hundreds of years and in soils for centuries or millennia (table S1 and the Supplementary Materials). Further, four of our NCS opportunities (cropland nutrient management, tidal wetland restoration, manure management, and improved rice management) are based on avoided emissions of CH_4 and N_2O, which are benefits that do not saturate. The mitigation potential of avoided conversion of habitat is limited by the total carbon contained in the habitat. Our analysis assumes that rates of conversion persist at current levels in a BAU scenario, which would represent a continuing source of emissions for at least 67 years for each habitat considered here before reaching "saturation" when the total area has been lost. However, the long-term benefit of avoided conversion depends on assumed future BAU conversion rates.

The permanence of the ~2270 Pg C currently stored globally in biomass (*42*) and soils to 1 m (*26*) is a significant concern, because unmitigated climate change is likely to cause feedbacks that may increase disturbances such as fire or pest outbreaks (*43*) or limit net ecosystem productivity or forest regeneration (*24*). While NCS would marginally increase this large carbon pool, putting some additional carbon at risk, rapid and widespread implementation of NCS would reduce the overall risk of impermanence to the terrestrial biosphere that unmitigated climate change is likely to cause.

Another challenge is that avoiding conversion in one area can cause conversion to shift to other areas, often referred to as "leakage." Large-scale sectoral and landscape approaches to land use planning and policies will be needed to realize the NCS opportunities identified here. These approaches can and should be designed to buffer risks of leakage associated with individual projects (*44*).

Reducing carbon-intensive energy consumption is necessary but insufficient to meet the ambitious goals of the Paris Agreement. Comprehensive mitigation efforts that include fossil fuel emission reductions coupled with NCS hold promise for keeping warming below 2°C. Beyond providing meaningful climate mitigation, NCS investment can increase other important ecosystem services. The conservation, restoration, and improved management of lands in the United States represent a necessary and urgent component of efforts to stabilize the climate.

MATERIALS AND METHODS

Below, we provide a brief overview of methods for each of the 21 NCS that we quantified. Full methodological details are provided in the Supplementary Materials.

Reforestation: Additional carbon sequestration in above- and belowground biomass and soils gained by converting nonforest (<25% tree cover) to forest [>25% tree cover (*45*)] in areas of the conterminous United States where forests are the native cover type. We excluded areas with intensive human development, including all major roads (*46*), impervious surfaces (*47*), and urban areas (*48*). To eliminate double counting with the peatland restoration pathway, we removed Histosol soils (*49*). To safeguard food production, we removed most cropland and pasture. We discounted the carbon sequestration mitigation benefit in conifer-dominated forests to account for albedo effects.

Natural forest management: Additional carbon sequestration in above- and belowground biomass gained through improved management in forests on private lands under nonintensive timber management. The maximum mitigation potential was quantified on the basis of a "harvest hiatus" scenario starting in 2025, in which natural forests are shifted to longer harvest rotations. This could be accomplished with less than 10% reduction in timber supply with new timber supply from thinning treatments for fuel risk reduction until new timber from reforestation is available in 2030.

Fire management: Use of prescribed fire to reduce the risk of high-intensity wildfire. We considered fire-prone forests in the western United States. We assume that treatment eliminates the risk of subsequent wildfire for 20 years, but only on the land that was directly treated. We assume that 5% of lands are treated each year, and we calculated the benefits that accrue over 20 years, finding that the initial increase in emissions associated with prescribed fire treatment is more than offset over time by the avoided impacts of wildfires. We report the average annual benefit across these 20 years. The impact of wildfires includes both direct emissions from combustion and suppression of net ecosystem productivity following wildfires.

Avoided forest conversion: Emissions of CO_2 avoided by avoiding anthropogenic forest conversion. Most forest clearing is followed by forest regeneration rather than conversion to another land use. To estimate the rate of persistent conversion (i.e., to another land use), we first calculated forest clearing in the conterminous United States from 2000 to 2010 and then used the proportion of forest clearing that historically was converted to another land use to estimate conversion rates in 2000 to 2010. We used estimates of avoided carbon emissions from above- and belowground biomass that are specific to each region and forest type. We did not count forest loss due to fire to avoid double counting with the improved fire management opportunity. We did not count forest loss due to pests because it is unclear whether this loss can be avoided. We reduced the benefit of avoided conversion in conifer-dominated forests to account for their albedo effects.

Urban reforestation: Additional carbon sequestration in above- and belowground biomass gained by increasing urban tree cover. We considered the potential to increase urban tree cover in 3535 cities in the conterminous United States. We considered the potential for additional street trees, and for those cities not in deserts, we also considered the potential for park and yard tree plantings. The potential percent increase in tree cover was estimated on the basis of high-resolution analysis of 27 cities, which excluded sports fields, golf courses, and lawns (*50*).

Improved plantations: Additional carbon sequestration gained in above- and belowground tree biomass by extending rotation lengths for a limited time in even-aged, intensively managed wood production forests. Rotation lengths were extended from current economic optimal rotation length to a biological optimal rotation length in which harvest occurs when stands reach their maximum annual growth.

Cover crops: Additional soil carbon sequestration gained by growing a cover crop in the fallow season between main crops. We quantified the benefit of using cover crops on all of the five major crops in the United States (corn, soy, wheat, rice, and cotton) that are not already growing cover crops (*27*), using the mean sequestration rate quantified in a recent meta-analysis (*51*).

Avoided conversion of grassland: Emissions of CO_2 avoided by avoiding conversion of grassland and shrubland to cropland. We quantified avoided emissions from soil and roots (for shrubs, we also considered aboveground biomass) based on the spatial pattern of conversion from 2008 to 2012. We used spatial information on location of recent conversion and variation in soil carbon and root biomass to estimate mean annual emission rate from historic conversion. We estimated a 28% loss of soil carbon down to 1 m (*26*). We modeled spatial variation in root biomass based on mean annual temperature and mean annual precipitation using data from (*52*).

Biochar: Increased soil carbon sequestration by amending agricultural soils with biochar, which converts nonrecalcitrant carbon (crop residue biomass) to recalcitrant carbon (charcoal) through pyrolysis. We limited the source of biochar production to crop residue that can be sustainably harvested. We assumed that 79.6% of biochar carbon persists on a time scale of >100 years (*53*, *54*) and that there are no effects of biochar on emissions of N_2O or CH_4 (*55*, *56*).

Alley cropping: Additional carbon sequestration gained by planting wide rows of trees with a companion crop grown in the alleyways between the rows. We estimated a maximum potential of alley cropping on 10% of U.S. cropland (15.4 Mha) (*57*).

Cropland nutrient management: Avoided N_2O emissions due to more efficient use of nitrogen fertilizers and avoided upstream emissions from fertilizer manufacture. We considered four improved management practices: (i) reduced whole-field application rate, (ii) switching from anhydrous ammonia to urea, (iii) improved timing of fertilizer application, and (iv) variable application rate within field. We projected a 4.6% BAU growth in fertilizer use in the United States by 2025. On the basis of these four practices, we found a maximum potential of 22% reduction in nitrogen use, which leads to a 33% reduction in field emissions and a 29% reduction including upstream emissions.

Improved manure management: Avoided CH_4 emissions from dairy and hog manure. We estimated the potential for emission reductions from improved manure management on dairy farms with over 300 cows and hog farms with over 825 hogs. Our calculations are based on improved management practices described by Pape *et al.* (*8*).

Windbreaks: Additional sequestration in above- and belowground biomass and soils from planting windbreaks adjacent to croplands that would benefit from reduced wind erosion. We estimated that windbreaks could be planted on 0.88 Mha, based on an estimated 17.6 Mha that would benefit from windbreaks, and that windbreaks would be planted on ~5% of that cropland (*8*).

Grazing optimization: Additional soil carbon sequestration due to grazing optimization on rangeland and planted pastures, derived directly from a recent study by Henderson *et al.* (*58*). Grazing optimization prescribes a decrease in stocking rates in areas that are overgrazed and an increase in stocking rates in areas that are undergrazed, but with the net result of increased forage offtake and livestock production.

Grassland restoration: Additional carbon sequestration in soils and root biomass gained by restoring 2.1 Mha of cropland to grassland, equivalent to returning to the 2007 peak in CRP enrollment. Grassland restoration does not include restoration of shrubland.

Legumes in pastures: Additional soil carbon sequestration due to sowing legumes in planted pastures, derived directly from a recent global study by Henderson *et al.* (*58*). Restricted to planted pastures and to where sowing legumes would result in net sequestration after taking into account potential increases in N_2O emissions from the planted legumes.

Improved rice management: Avoided emissions of CH_4 and N_2O through improved practices in flooded rice cultivation. Practices including mid-season drainage, alternate wetting and drying, and residue removal can reduce these emissions. We used a U.S. Environmental Protection Agency (EPA) analysis that projects the potential for improvement across U.S. rice fields, in comparison with current agricultural practices (*59*).

Tidal wetland restoration: In the United States, 27% of tidal wetlands (salt marshes and mangroves) have limited tidal connection with the sea, causing their salinity to decline to the point where CH_4 emissions increase (*30*). We estimated the potential for reconnecting these tidal wetlands to the ocean to increase salinity and reduce CH_4 emissions.

Peatland restoration: Avoided carbon emissions from rewetting and restoring drained peatlands. To estimate the extent of restorable peatlands, we quantified the difference between historic peatland extent [based on the extent of Histosols in soil maps (*60*)] and current peatland extent. Our estimate of mitigation potential accounted for changes in soil carbon, biomass, and CH_4 emissions, considering regional differences, the type of land use of the converted peatland, and whether the peatland was originally forested.

Avoided seagrass loss: Avoided CO_2 emissions from avoiding seagrass loss. An estimated 1.5% of seagrass extent is lost every year (*61*). We assumed that half of the carbon contained in biomass and sediment from disappearing seagrass beds is lost to the atmosphere (*62*).

Seagrass restoration: Increased sequestration from restoring the estimated 29 to 52% of historic seagrass extent that has been lost and could be restored (*61*). We estimated the average carbon sequestration rate in the sediment of seagrass restorations based on data from six seagrass restoration sites in the United States (*63*).

SUPPLEMENTARY MATERIALS

Supplementary material for this article is available at http://advances.sciencemag.org/cgi/content/full/4/11/eaat1869/DC1

Supplementary Materials and Methods
Fig. S1. Mapped reforestation opportunity areas in the lower 48 states.
Fig. S2. Conceptual framework for improved forest management carbon accounting.
Fig. S3. MAC for carbon sequestration through forest management and aging, after Golub *et al*. (*99*).
Fig. S4. MAC for natural forest management after Latta *et al*. (*98*) and best-fit functions.
Fig. S5. MAC curves for improved plantations.
Fig. S6. Fire management analysis area.
Fig. S7. Regions used for reporting avoided forest conversion results.
Fig. S8. Forest conversion from 1986 to 2000.
Fig. S9. Potential carbon emissions from areas at high risk of forest conversion.
Fig. S10. Cities included in the urban reforestation analysis.
Fig. S11. Calibration of remote sensing data for forest cover estimation in urban areas.
Fig. S12. Avoided grassland conversion map.
Fig. S13. MAC curve for avoided grassland conversion.
Fig. S14. Nitrogen fertilizer use in the United States.
Fig. S15. Marginal abatement cost curve for reducing N fertilizer rate.
Fig. S16. Marginal abatement cost curve for applying variable rate technology fertilizer application.
Fig. S17. Grazing optimization map.
Fig. S18. Legumes in pastures map.
Fig. S19. Grassland restoration map.
Fig. S20. MAC curve for grassland restoration.
Fig. S21. Break-even prices for GHG abatement from rice production.
Fig. S22. MAC curve for salt marsh restoration.
Fig. S23. MAC of avoided GHG emissions from seagrass.
Table S1. Mitigation potential of NCS in 2025.
Table S2. Co-benefits of NCS.
Table S3. Literature MAC estimates for reforestation of agricultural lands.
Table S4. Literature estimates of reforestation costs used to estimate MAC of reforesting natural ecosystems.
Table S5. Estimated marginal abatement cost of fire management by major forest region.
Table S6. Forest disturbance rates by source.
Table S7. Mean annual forest hectares cleared per year from 1986 to 2000.
Table S8. Mean annual forest hectares cleared per year from 2001 to 2010.
Table S9. Mean annual forest hectares converted per year from 1986 to 2000.
Table S10. Proportion of areas cleared from 1986 to 2000 that had not regenerated to forest by 2010.
Table S11. Mean predisturbance dry biomass (kg m^{-2}) in forest areas converted from 1986 to 2000.
Table S12. Mean predisturbance dry biomass (kg m^{-2}) in forest areas converted from 2001 to 2010.
Table S13. Carbon emissions (Mg C $year^{-1}$) from estimated forest conversion from 2001 to 2010.
Table S14. Albedo-adjusted carbon emissions equivalent (Mg C_e $year^{-1}$) from estimated forest conversion from 2001 to 2010.
Table S15. Urban reforestation maximum potential annual net C sequestration in 2025.
Table S16. Uncertainty in urban reforestation average annual abatement (Tg CO_2) by 2025 at a cost of USD 100 per Mg CO_2.
Table S17. Profitability impacts of cover crops for selected crops.
Table S18. Marginal abatement costs of cover crops in the five primary crops.
Table S19. Maximum feasible N_2O reduction for multiple nitrogen fertilizer practices.
Table S20. Results from the literature of the potential for reducing N fertilizer rate using within-field management.
Table S21. Current and projected GHG emissions from nitrogen fertilizer manufacturing in the United States.
Table S22. Mitigation potential for grazing optimization and legumes in pasture NCS at different marginal abatement costs.
Table S23. Areas and carbon fluxes for Histosols in the conterminous United States.
Table S24. Peatland restoration mitigation calculations for climate zones within the United States.
Table S25. 95% CIs for Histosol calculations.
References (*64*–*398*)

REFERENCES AND NOTES

1. United Nations, *United Nations Framework Convention on Climate Change: Adoption of the Paris Agreement* (United Nations, 2015).
2. P. Smith, S. J. Davis, F. Creutzig, S. Fuss, J. Minx, B. Gabrielle, E. Kato, R. B. Jackson, A. Cowie, E. Kriegler, D. P. van Vuuren, J. Rogelj, P. Ciais, J. Milne, J. G. Canadell, D. McCollum, G. Peters, R. Andrew, V. Krey, G. Shrestha, P. Friedlingstein, T. Gasser, A. Grübler, W. K. Heidug, M. Jonas, C. D. Jones, F. Kraxner, E. Littleton, J. Lowe, J. Roberto Moreira, N. Nakicenovic, M. Obersteiner, A. Patwardhan, M. Rogner, E. Rubin, A. Sharifi, A. Torvanger, Y. Yamagata, J. Edmonds, C. Yongsung, Biophysical and economic limits to negative CO_2 emissions. *Nat. Clim. Change* **6**, 42–50 (2016).
3. B. W. Griscom, J. Adams, P. W. Ellis, R. A. Houghton, G. Lomax, D. A. Miteva, W. H. Schlesinger, D. Shoch, J. V. Siikamäki, P. Smith, P. Woodbury, C. Zganjar, A. Blackman, J. Campari, R. T. Conant, C. Delgado, P. Elias, T. Gopalakrishna, M. R. Hamsik, M. Herrero, J. Kiesecker, E. Landis, L. Laestadius, S. M. Leavitt, S. Minnemeyer, S. Polasky, P. Potapov, F. E. Putz, J. Sanderman, M. Silvius, E. Wollenberg, J. Fargione, Natural climate solutions. *Proc. Natl. Acad. Sci. U.S.A.* **114**, 11645–11650 (2017).
4. C. B. Field, K. J. Mach, Rightsizing carbon dioxide removal. *Science* **356**, 706–707 (2017).
5. The White House, *United States Mid-Century Strategy for Deep Decarbonization* (The White House, 2016).
6. A. J. Eagle, L. R. Henry, L. P. Olander, K. Haugen-Kozyra, N. Millar, G. P. Robertson, *Greenhouse Gas Mitigation Potential of Agricultural Land Management in the United States: A Synthesis of the Literature* (Nicholas Institute, Duke University, 2012).
7. C. Van Winkle, J.S. Baker, D. Lapidus, S. Ohrel, J. Steller, G. Latta, D. Birur, *US Forest Sector Greenhouse Mitigation Potential and Implications for Nationally Determined Contributions* (RTI Press, 2017).
8. D. Pape, J. Lewandrowski, R. Steele, D. Man, M. Riley-Gilbert, K. Moffroid, S. Kolansky, *Managing Agricultural Land for Greenhouse Gas Mitigation Within the United States* (U.S. Department of Agriculture, 2016); www.usda.gov/oce/climate_change/mitigation.htm.
9. World Bank; Ecofys; Vivid Economics, *State and Trends of Carbon Pricing 2017* (World Bank, 2017).
10. National Academies of Science Engineering and Medicine, *Valuing Climate Damages: Updating Estimation of the Social Cost of Carbon Dioxide* (The National Academies Press, 2017).
11. W. D. Nordhaus, Evolution of modeling of the economics of global warming: Changes in the DICE model, 1992–2017. *Clim. Change* **148**, 623–640 (2018).
12. K. E. Skog, P. J. Ince, H. Spelter, A. Kramp, R. J. Barbour, in *Woody Biomass Utilization: Challenges and Opportunities* (Forest Products Society, 2008), pp. 3–14.
13. S. C. Davis, A. E. Hessl, C. J. Scott, M. B. Adams, R. B. Thomas, Forest carbon sequestration changes in response to timber harvest. *For. Ecol. Manage.* **258**, 2101–2109 (2009).
14. C. Hoover, S. Stout, The carbon consequences of thinning techniques: Stand structure makes a difference. *J. For.* **105**, 266–270 (2007).
15. U.S. Environmental Protection Agency, *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2016* (U.S. Environmental Protection Agency, 2018); www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2016.
16. J. Rogelj, M. den Elzen, N. Höhne, T. Fransen, H. Fekete, H. Winkler, R. Schaeffer, F. Sha, K. Riahi, M. Meinshausen, Paris Agreement climate proposals need boost to keep warming well below 2°C. *Nat. Clim. Change* **534**, 631–639 (2016).
17. Commission for Environmental Cooperation, *North America's Blue Carbon: Assessing Seagrass, Salt Marsh and Mangrove Distribution and Carbon Sinks* (Commission for Environmental Cooperation, 2016).

18. T. D. Lee, S. E. Eisenhaure, I. P. Gaudreau, Pre-logging treatment of invasive glossy buckthorn (*Frangula alnus* Mill.) promotes regeneration of eastern white pine (*Pinus strobus* L.). *Forests* **8**, 16 (2017).
19. T. M. Schuler, M. Thomas-Van Gundy, J. P. Brown, J. K. Wiedenbeck, Managing Appalachian hardwood stands using four management practices: 60-year results. *For. Ecol. Manage.* **387**, 3–11 (2017).
20. S. A. Moss, E. Heitzman, The economic impact of timber harvesting practices on NIPF properties in West Virginia, in *Proceedings of the 18th Central Hardwood Forest Conference*, G. W. Miller, T. M. Schuler, K. W. Gottschalk, J. R. Brooks, S. T. Grushecky, B. D. Spong, J. S. Rentch, Eds. (U.S. Department of Agriculture, Forest Service, 2013), pp. 129–141.
21. Ruslandi, C. Romero, F. E. Putz, Financial viability and carbon payment potential of large-scale silvicultural intensification in logged dipterocarp forests in Indonesia. *For. Policy Econ.* **85**, 95–102 (2017).
22. V. P. Medjibe, F. E. Putz, Cost comparisons of reduced-impact and conventional logging in the tropics. *J. For. Econ.* **18**, 242–256 (2012).
23. J. Williams, Exploring the onset of high-impact mega-fires through a forest land management prism. *For. Ecol. Manage.* **294**, 4–10 (2013).
24. C. S. Stevens-Rumann, K. B. Kemp, P. E. Higuera, B. J. Harvey, M. T. Rother, D. C. Donato, P. Morgan, T. T. Veblen, Evidence for declining forest resilience to wildfires under climate change. *Ecol. Lett.* **21**, 243–252 (2018).
25. T. J. Lark, J. Meghan Salmon, H. K. Gibbs, Cropland expansion outpaces agricultural and biofuel policies in the United States. *Environ. Res. Lett.* **10**, 044003 (2015).
26. J. Sanderman, T. Hengl, G. J. Fiske, Soil carbon debt of 12,000 years of human land use. *Proc. Natl. Acad. Sci. U.S.A.* **114**, 9575–9580 (2017).
27. Conservation Technology Information Center, Sustainable Agriculture Research and Education, American Seed Trade Association, *Annual Report 2016-2017 Cover Crop Survey* (Conservation Technology Information Center, Sustainable Agriculture Research and Education, American Seed Trade Association, 2017).
28. D. Knowler, B. Bradshaw, Farmers' adoption of conservation agriculture: A review and synthesis of recent research. *Food Policy* **32**, 25–48 (2007).
29. S. Shackley, G. Ruysschaert, K. Zwart, B. Glaser, *Biochar in European Soils and Agriculture: Science and Practice* (Routledge, 2016).
30. K. D. Kroeger, S. Crooks, S. Moseman-valtierra, J. Tang, Restoring tides to avoid methane emissions in impounded wetlands: A new and potent Blue Carbon climate change intervention. *Sci. Rep.* **7162**, 1–23 (2017).
31. S. Banwart, S. Banwart, H. Black, Z. Cai, P. Gicheru, H. Joosten, R. Victoria, E. Milne, E. Noellemeyer, U. Pascual, G. Nziguheba, R. Vargas, A. Bationo, D. Buschiazzo, D. de-Brogniez, J. Melillo, D. Richter, M. Termansen, M. van Noordwijk, T. Goverse, C. Ballabio, T. Bhattacharyya, M. Goldhaber, N. Nikolaidis, Y. Zhao, R. Funk, C. Duffy, G. Pan, N. la Scala, P. Gottschalk, N. Batjes, J. Six, B. van Wesemael, M. Stocking, F. Bampa, M. Bernoux, C. Feller, P. Lemanceau, L. Montanarella, Benefits of soil carbon: Report on the outcomes of an international scientific committee on problems of the environment rapid assessment workshop. *Carbon Manage.* **5**, 185–192 (2014).
32. S. Narayan, M. W. Beck, P. Wilson, C. J. Thomas, A. Guerrero, C. C. Shepard, B. G. Reguero, G. Franco, J. Carter Ingram, D. Trespalacios, The value of coastal wetlands for flood damage reduction in the northeastern USA. *Sci. Rep.* **7**, 9463 (2017).
33. T. A. Boden, R. J. Andres, G. Marland, *Global, Regional, and National Fossil-Fuel CO_2 Emissions (1751-2014) (V. 2017)* (U.S. Department of Energy, 2017).
34. B. K. Buchner, C. Trabacchi, F. Mazza, D. Abramskiehn, D. Wang, *Global Landscape of Climate Finance 2015* (Climate Policy Initiative, 2015); www.climatepolicyinitiative.org.
35. K.-H. Erb, C. Lauk, T. Kastner, A. Mayer, M. C. Theurl, H. Haberl, Exploring the biophysical option space for feeding the world without deforestation. *Nat. Commun.* **7**, 11382 (2016).
36. P. Smith, H. Haberl, A. Popp, K. H. Erb, C. Lauk, R. Harper, F. N. Tubiello, A. de Siqueira Pinto, M. Jafari, S. Sohi, O. Masera, H. Böttcher, G. Berndes, M. Bustamante, H. Ahammad, H. Clark, H. Dong, E. A. Elsiddig, C. Mbow, N. H. Ravindranath, C. W. Rice, C. Robledo Abad, A. Romanovskaya, F. Sperling, M. Herrero, J. I. House, S. Rose, How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? *Glob. Chang. Biol.* **19**, 2285–2302 (2013).
37. E. Brandes, G. S. McNunn, L. A. Schulte, I. J. Bonner, D. J. Muth, B. A. Babcock, B. Sharma, E. A. Heaton, Subfield profitability analysis reveals an economic case for cropland diversification. *Environ. Res. Lett.* **11**, 014009 (2016).
38. K. A. Johnson, B. J. Dalzell, M. Donahue, J. Gourevitch, D. L. Johnson, G. S. Karlovits, B. Keeler, J. T. Smith, Conservation Reserve Program (CRP) lands provide ecosystem service benefits that exceed land rental payment costs. *Ecosyst. Serv.* **18**, 175–185 (2016).
39. D. Tilman, R. Socolow, J. A. Foley, J. Hill, E. Larson, L. Lynd, S. Pacala, J. Reilly, T. Searchinger, C. Somerville, R. Williams, Beneficial biofuels—The food, energy, and environment trilemma. *Science* **325**, 270–271 (2009).
40. D. Tilman, J. Hill, C. Lehman, Carbon-negative biofuels from low-input high-diversity grassland biomass. *Science* **314**, 1598–1600 (2006).
41. J. E. Fargione, R. J. Plevin, J. D. Hill, The ecological impact of biofuels. *Annu. Rev. Ecol. Evol. Syst.* **41**, 351–377 (2010).
42. K. H. Erb, T. Kastner, C. Plutzar, A. L. S. Bais, N. Carvalhais, T. Fetzel, S. Gingrich, H. Haberl, C. Lauk, M. Niedertscheider, J. Pongratz, M. Thurner, S. Luyssaert, Unexpectedly large impact of forest management and grazing on global vegetation biomass. *Nature* **553**, 73–76 (2018).
43. J. T. Abatzoglou, A. P. Williams, Impact of anthropogenic climate change on wildfire across western US forests. *Proc. Natl. Acad. Sci. U.S.A.* **113**, 11770–11775 (2016).
44. J. Sayer, T. Sunderland, J. Ghazoul, J.-L. Pfund, D. Sheil, E. Meijaard, M. Venter, A. Klintuni Boedhihartono, M. Day, C. Garcia, C. van Oosten, L. E. Buck, Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. *Proc. Natl. Acad. Sci. U.S.A.* **110**, 8349–8356 (2013).
45. M. C. Hansen, P. V. Potapov, R. Moore, M. Hancher, S. A. Turubanova, A. Tyukavina, D. Thau, S. V. Stehman, S. J. Goetz, T. R. Loveland, A. Kommareddy, A. Egorov, L. Chini, C. O. Justice, J. R. G. Townshend, High-resolution global maps of 21st-century forest cover change. *Science* **342**, 850–853 (2013).
46. Open Street Map, Osm2Shp (2016); http://osm2shp.ru/.
47. G. Xian, C. G. Homer, J. Dewitz, J. Fry, N. Hossain, J. Wickham, The change of impervious surface area between 2001 and 2006 in the conterminous United States. *Photogramm. Eng. Remote Sens.* **77**, 758–762 (2011).
48. U.S. Census Bureau, *Cartographic Boundary File, Urban Area for United States* (U.S. Census Bureau, 2015); www.census.gov/geo/maps-data/data/cbf/cbf_ua.html.
49. Soil Survey Staff, *U.S. General Soil Map (STATSGO2)* (U.S. Department of Agriculture, 2017); https://sdmdataaccess.sc.egov.usda.gov.
50. T. Kroeger, R. I. McDonald, T. Boucher, P. Zhang, L. Wang, Where the people are: Current trends and future potential targeted investments in urban trees for PM_{10} and temperature mitigation in 27 U.S. cities. *Landsc. Urban Plan.* **177**, 277–240 (2018).
51. C. Poeplau, A. Don, Carbon sequestration in agricultural soils via cultivation of cover crops—A meta-analysis. *Agric. Ecosyst. Environ.* **200**, 33–41 (2015).
52. K. Mokany, R. J. Raison, A. S. Prokushkin, Critical analysis of root: Shoot ratios in terrestrial biomes. *Glob. Chang. Biol.* **12**, 84–96 (2006).
53. R. S. Dharmakeerthi, K. Hanley, T. Whitman, D. Woolf, J. Lehmann, Organic carbon dynamics in soils with pyrogenic organic matter that received plant residue additions over seven years. *Soil Biol. Biochem.* **88**, 268–274 (2015).
54. B. Liang, J. Lehmann, D. Solomon, S. Sohi, J. E. Thies, J. O. Skjemstad, F. J. Luizão, M. H. Engelhard, E. G. Neves, S. Wirick, Stability of biomass-derived black carbon in soils. *Geochim. Cosmochim. Acta* **72**, 6069–6078 (2008).
55. X. Song, G. Pan, C. Zhang, L. Zhang, H. Wang, Effects of biochar application on fluxes of three biogenic greenhouse gases: A meta-analysis. *Ecosyst. Health Sustain.* **2**, e01202 (2016).
56. J. Wang, Z. Xiong, Y. Kuzyakov, Biochar stability in soil: Meta-analysis of decomposition and priming effects. *Glob. Change Biol. Bioenergy* **8**, 512–523 (2016).
57. R. P. Udawatta, S. Jose, Carbon sequestration potential of agroforestry practices in temperate North America, in *Carbon Sequestration Potential of Agroforestry Systems*, vol. 8 of *Advances in Agroforestry*, B. M. Kumar, P. K. R. Nair, Eds. (Springer Netherlands, 2011), pp. 17–42.
58. B. B. Henderson, P. J. Gerber, T. E. Hilinski, A. Falcucci, D. S. Ojima, M. Salvatore, R. T. Conant, Greenhouse gas mitigation potential of the world's grazing lands: Modeling soil carbon and nitrogen fluxes of mitigation practices. *Agric. Ecosyst. Environ.* **207**, 91–100 (2015).
59. U.S. Environmental Protection Agency, *Global Mitigation of Non-CO_2 Greenhouse Gases: 2010-2030* (U.S. Environmental Protection Agency, 2013).
60. Soil Survey Staff, *Gridded Soil Survey Geographic (gSSURGO) Database for the Conterminous United States* (U.S. Department of Agriculture, 2016); https://gdg.sc.egov.usda.gov/.
61. M. Waycott, C. M. Duarte, T. J. Carruthers, R. J. Orth, W. C. Dennison, S. Olyarnik, A. Calladine, J. W. Fourqurean, K. L. Heck Jr., A. R. Hughes, G. A. Kendrick, W. J. Kenworthy, F. T. Short, S. L. Williams, Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proc. Natl. Acad. Sci. U.S.A.* **106**, 12377–12381 (2009).
62. L. Pendleton, D. C. Donato, B. C. Murray, S. Crooks, W. A. Jenkins, S. Sifleet, C. Craft, J. W. Fourqurean, J. B. Kauffman, N. Marbà, P. Megonigal, E. Pidgeon, D. Herr, D. Gordon, A. Baldera, Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. *PLOS ONE* **7**, e43542 (2012).
63. A. Thorhaug, H. M. Poulos, J. López-Portillo, T. C. W. Ku, G. P. Berlyn, Seagrass blue carbon dynamics in the Gulf of Mexico: Stocks, losses from anthropogenic disturbance, and gains through seagrass restoration. *Sci. Total Environ.* **605–606**, 626–636 (2017).
64. J. W. Veldman, G. E. Overbeck, D. Negreiros, G. Mahy, S. Le Stradic, G. W. Fernandes, G. Durigan, E. Buisson, F. E. Putz, W. J. Bond, Where tree planting and forest expansion are bad for biodiversity and ecosystem services. *Bioscience* **65**, 1011–1018 (2015).
65. S. Luyssaert, E. D. Schulze, A. Börner, A. Knohl, D. Hessenmöller, B. E. Law, P. Ciais, J. Grace, Old-growth forests as global carbon sinks. *Nature* **455**, 213–215 (2008).
66. B. E. Law, O. J. Sun, J. Campbell, S. Van Tuyl, P. E. Thornton, Changes in carbon storage and fluxes in a chronoseuence of ponderosa pine. *Glob. Chang. Biol.* **4**, 510–524 (2003).
67. D. J. Nowak, J. C. Stevens, S. M. Sisinni, C. J. Luley, Effects of urban tree management and species selection on atmospheric carbon dioxide. *J. Arboric.* **28**, 113–122 (2002).

68. K. K. McLauchlan, S. E. Hobbie, W. M. Post, Conversion from agriculture to grassland builds soil organic matter on decadal timescales. *Ecol. Appl.* **16**, 143–153 (2006).
69. E. Mcleod, G. L. Chmura, S. Bouillon, R. Salm, M. Björk, C. M. Duarte, C. E. Lovelock, W. H. Schlesinger, B. R. Silliman, A blueprint for blue carbon: Toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Front. Ecol. Environ.* **9**, 552–560 (2011).
70. J. W. Fourqurean, C. M. Duarte, H. Kennedy, N. Marbà, M. Holmer, M. Angel Mateo, E. T. Apostolaki, G. A. Kendrick, D. Krause-Jensen, K. J. McGlathery, O. Serrano, Seagrass ecosystems as a globally significant carbon stock. *Nat. Geosci.* **5**, 505–509 (2012).
71. J. Loisel, J. Loisel, Z. Yu, D. W. Beilman, P. Camill, J. Alm, M. J. Amesbury, D. Anderson, S. Andersson, C. Bochicchio, K. Barber, L. R. Belyea, J. Bunbury, F. M. Chambers, D. J. Charman, F. De Vleeschouwer, B. Fiałkiewicz-Kozieł, S. A. Finkelstein, M. Gałka, M. Garneau, D. Hammarlund, W. Hinchcliffe, J. Holmquist, P. Hughes, M. C. Jones, E. S. Klein, U. Kokfelt, A. Korhola, P. Kuhry, A. Lamarre, M. Lamentowicz, D. Large, M. Lavoie, G. MacDonald, G. Magnan, M. Mäkilä, G. Mallon, P. Mathijssen, D. Mauquoy, J. McCarroll, T. R. Moore, J. Nichols, B. O'Reilly, P. Oksanen, M. Packalen, D. Peteet, P. J. H. Richard, S. Robinson, T. Ronkainen, M. Rundgren, A. B. K. Sannel, C. Tarnocai, T. Thom, E.-S. Tuittila, M. Turetsky, M. Väliranta, M. van der Linden, B. van Geel, S. van Bellen, D. Vitt, Y. Zhao, W. Zhou, A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation. *Holocene* **24**, 1028–1042 (2014).
72. C. Frey, J. Penman, L. Hanle, S. Monni, S. Ogle, Chapter 3: Uncertainties, in *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (Intergovernmental Panel on Climate Change, 2006), p. 3.1–3.66; www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/1_Volume1/V1_3_Ch3_Uncertainties.pdf.
73. Climate Policy Initiative, California Carbon Dashboard (2017); http://calcarbondash.org/.
74. K. Hamrick, M. Gallant, *Unlocking Potential State of the Voluntary Carbon Markets 2017* (Ecosystem Marketplace, 2017).
75. U.S. Bureau of Labor Statistics, Consumer Price Index Inflation Calculator (2017); www.bls.gov/data/inflation_calculator.htm.
76. United Nations, *Convention on Biological Diversity* (United Nations, 1992); www.cbd.int/doc/legal/cbd-en.pdf.
77. Millennium Ecosystem Assessment, *Ecosystems and Human Well-being: Synthesis* (Island Press, 2005); www.millenniumassessment.org/en/Synthesis.html.
78. U.S. Environmental Protection Agency, *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2015* (U.S. Environmental Protection Agency, 2017).
79. E. G. Brockerhoff, H. Jactel,J. A. Parrotta, C. P. Quine, J. Sayer, Plantation forests and biodiversity: Oxymoron or opportunity? *Biodivers. Conserv.* **17**, 925–951 (2008).
80. L. L. Bremer, K. A. Farley, Does plantation forestry restore biodiversity or create green deserts? A synthesis of the effects of land-use transitions on plant species richness. *Biodivers. Conserv.* **19**, 3893–3915 (2010).
81. LANDFIRE, Biophysical Settings (2014); www.landfire.gov/NationalProductDescriptions20.php.
82. S. N. Goward, C. Huang, F. Zhao, K. Schleeweis, K. Rishmawi, M. Lindsey, J. L. Dungan, A. Michaelis, *NACP NAFD Project: Forest Disturbance History from Landsat, 1986-2010* (Oak Ridge National Laboratory Distributed Active Archive Center, 2015); http://dx.doi.org/10.3334/ORNLDAAC/1290.
83. U.S. Department of Agriculture National Agricultural Statistics Service, *2015 Cultivated Layer* (U.S. Department of Agriculture, 2015); www.nass.usda.gov/Research_and_Science/Cropland/Release/.
84. S. Jin, L. Yang, P. Danielson, C. Homer, J. Fry, G. Xian, A comprehensive change detection method for updating the National Land Cover Database to circa 2011. *Remote Sens. Environ.* **132**, 159–175 (2013).
85. U.S. Department of Agriculture National Agricultural Statistics Service, *Quick Stats* (U.S. Department of Agriculture, 2017); https://quickstats.nass.usda.gov.
86. J. Smith, L. Heath, K. Skog, R. Birdsey, *Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States* (U.S. Department of Agriculture Forest Service, 2006); www.actrees.org/files/Research/ne_gtr343.pdf.
87. T. O. West, G. Marland, A. W. King, W. M. Post, A. K. Jain, K. Andrasko, Carbon management response curves: Estimates of temporal soil carbon dynamics. *Environ. Manage.* **33**, 507–518 (2004).
88. B. Ruefenacht, M. V. Finco, M. D. Nelson, R. Czaplewski, E. H. Helmer, J. A. Blackard, G. R. Holden, A. J. Lister, D. Salajanu, D. Weyermann, K. Winterberger, Coterminous U.S. and Alaska forest type mapping using forest inventory and analysis data. *Photogramm. Eng. Remote Sens.* **11**, 1379–1388 (2008).
89. K. Naudts, Y. Chen, M. J. McGrath, J. Ryder, A. Valade, J. Otto, S. Luyssaert, Europe's forest management did not mitigate climate warming. *Science* **351**, 597–600 (2016).
90. L. S. Heath, P. E. Kauppi, P. Burschel, H.-D. Gregor, R. Guderian, G. H. Kohlmaier, S. Lorenz, D. Overdieck, F. Scholz, H. Thomasius, M. Weber, Contribution of temperate forests to the world's carbon budget. *Water Air Soil Pollut.* **70**, 55–69 (1993).
91. J. Creyts, A. Derkach, S. Nyquist, K. Ostrowski, J. Stephenson, *Reducing U.S. Greenhouse Gas Emissions: How Much at What Cost?* (McKinsey & Company, 2007).
92. R. W. Gorte, "U.S. tree planting for carbon sequestration" (Technical Report R40562, Congressional Research Service, 2009).
93. P. Potapov, L. Laestadius, S. Minnemeyer, *Global Map of Potential Forest Cover* (World Resources Institute, 2011); www.wri.org/resources/maps/atlas-forest-and-landscape-restoration-opportunities/data-info.
94. V. A. Sample, Potential for Additional carbon sequestration through regeneration of nonstocked forest land in the United States. *J. For.* **115**, 309–318 (2016).
95. U.S. Environmental Protection Agency, *Greenhouse Gas Mitigation Potential in U.S. Forestry and Agriculture* (U.S. Environmental Protection Agency, 2005).
96. R. Alig, G. Latta, D. Adams, B. McCarl, Mitigating greenhouse gases: The importance of land base interactions between forests, agriculture, and residential development in the face of changes in bioenergy and carbon prices. *For. Policy Econ.* **12**, 67–75 (2010).
97. D. Haim, E. M. White, R. J. Alig, Agriculture afforestation for carbon sequestration under carbon markets in the United States: Leakage behavior from regional allowance programs. *Appl. Econ. Perspect. Policy* **38**, 132–151 (2015).
98. G. Latta, D. M. Adams, R. J. Alig, E. White, Simulated effects of mandatory versus voluntary participation in private forest carbon offset markets in the United States. *J. For. Econ.* **17**, 127–141 (2011).
99. A. Golub, T. Hertel, H.-L. Lee, S. Rose, B. Sohngen, The opportunity cost of land use and the global potential for greenhouse gas mitigation in agriculture and forestry. *Resour. Energy Econ.* **31**, 299–319 (2009).
100. M. M. Atkinson, S. A. Fitzgerald, *Successful Reforestation: An Overview* (Oregon State University, 2002); https://archive.extension.oregonstate.edu/sorec/sites/default/files/sucsessful.pdf.
101. T. Kroeger, F. J. Escobedo, J. L. Hernandez, S. Varela, S. Delphin, J. R. B. Fisher, J. Waldron, Reforestation as a novel abatement and compliance measure for ground-level ozone. *Proc. Natl. Acad. Sci. U.S.A.* **111**, E4204–E4213 (2014).
102. J. Sessions, P. Bettinger, R. Buckman, M. Newton, J. Hamann, Hastening the return of complex forests following fire: The consequences of delay. *J. For.* **102**, 38–45 (2004).
103. J. A. Stanturf, S. H. Schoenholtz, C. J. Schweitzer, J. P. Shepard, Achieving restoration success: Myths in bottomland hardwood forests. *Restor. Ecol.* **9**, 189–200 (2001).
104. H. E. Garrett, W. D. Walter, L. D. Godsey, Alley Cropping: A relic from the past or a bridge to the future? *Inside Agrofor.* **19**, 1–12 (2011).
105. Board of Governors of the Federal Reserve System, *10-Year Treasury Constant Maturity Rate (DGS10)* (Federal Reserve System, 2017); https://fred.stlouisfed.org/series/DGS10.
106. S. N. Oswalt, W. B. Smith, P. D. Miles, S. A. Pugh, "Forest Resources of the United States, 2012: A technical document supporting the Forest Service 2015 update of the RPA Assessment" (General Technical Report WO-91, U.S. Department of Agriculture, Forest Service, 2014).
107. B. Zeide, Thinning and growth: A full turnaround. *J. For.* **99**, 20–25 (2001).
108. M. J. Schelhaas, K. Kramer, H. Peltola, D. C. van der Werf, S. M. J. Wijdeven, Introducing tree interactions in wind damage simulation. *Ecol. Modell.* **207**, 197–209 (2007).
109. R. Goodnow, J. Sullivan, G. S. Amacher, Ice damage and forest stand management. *J. For. Econ.* **14**, 268–288 (2008).
110. C. Kuehne, A. R. Weiskittel, S. Fraver, K. J. Puettmann, Effects of thinning-induced changes in structural heterogeneity on growth, ingrowth, and mortality in secondary coastal Douglas-fir forests. *Can. J. For. Res.* **45**, 1448–1461 (2015).
111. S. M. Hood, S. Baker, A. Sala, Fortifying the forest: Thinning and burning increase resistance to a bark beetle outbreak and promote forest resilience. *Ecol. Appl.* **26**, 1984–2000 (2016).
112. M. G. Ryan, D. Binkley, J. H. Fownes, Age-related decline in forest productivity: Pattern and Process. *Adv. Ecol. Res.* **27**, 213–262 (1997).
113. U.S. Environmental Protection Agency, *Inventory of Greenhouse Gas Emissions and Sinks: 1990–2006* (U.S. Environmental Protection Agency, 2008).
114. U.S. Environmental Protection Agency, *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2013* (U.S. Environmental Protection Agency, 2015).
115. L. S. Heath, J. E. Smith, K. E. Skog, D. J. Nowak, C. W. Woodall, Managed forest carbon estimates for the US greenhouse gas inventory, 1990-2008. *J. For.* **109**, 167–173 (2011).
116. P. B. Woodbury, J. E. Smith, L. S. Heath, Carbon sequestration in the U.S. forest sector from 1990 to 2010. *For. Ecol. Manage.* **241**, 14–27 (2007).
117. R. A. Birdsey, G. M. Lewis, "Carbon in U.S. forests and wood products, 1987-1997: State-by-state estimates" (General Technical Report NE-310, U.S. Department of Agriculture Forest Service, 2003).
118. Y. Pan, R. A. Birdsey, J. Fang, R. Houghton, P. E. Kauppi, W. A. Kurz, O. L. Phillips, A. Shvidenko, S. L. Lewis, J. G. Canadell, P. Ciais, R. B. Jackson, S. W. Pacala, A. D. McGuire, S. Piao, A. Rautiainen, S. Sitch, D. Hayes, A large and persistent carbon sink in the world's forests. *Science* **333**, 988–993 (2011).
119. C. D. Oliver, B. C. Larson, *Forest Stand Dynamics: Updated Edition* (CAB Direct, 1996); www.cabdirect.org/cabdirect/abstract/19980604521.

120. J. S. Nunery, W. S. Keeton, Forest carbon storage in the northeastern United States: Net effects of harvesting frequency, post-harvest retention, and wood products. *For. Ecol. Manage.* **259**, 1363–1375 (2010).
121. A. W. D'Amato, J. B. Bradford, S. Fraver, B. J. Palik, Forest management for mitigation and adaptation to climate change: Insights from long-term silviculture experiments. *For. Ecol. Manage.* **262**, 803–816 (2011).
122. S. Pugh, "RPA Data Wiz users guide, version 1.0" (General Technical Report NC-242, U.S. Department of Agriculture Forest Service, 2012); www.nrs.fs.fed.us/pubs/1950.
123. T. G. Johnson, J. W. Bentley, M. Howell, T. G. Johnson, J. W. Bentley, *The South's Timber Industry—An Assessment of Timber Product Output and Use, 2009* (U.S. Department of Agriculture Forest Service, 2011); www.srs.fs.usda.gov/pubs/39409.
124. M. E. Harmon, B. Marks, Effects of silvicultural practices on carbon stores in Douglas-fir western hemlock forests in the Pacific Northwest, U.S.A.: Results from a simulation model. *Can. J. For. Res.* **32**, 863–877 (2002).
125. B. Griscom, P. Ellis, F. E. Putz, Carbon emissions performance of commercial logging in East Kalimantan, Indonesia. *Glob. Chang. Biol.* **20**, 923–937 (2014).
126. D. Lussetti, E. P. Axelsson, U. Ilstedt, J. Falck, A. Karlsson, Supervised logging and climber cutting improves stand development: 18 years of post-logging data in a tropical rain forest in Borneo. *For. Ecol. Manage.* **381**, 335–346 (2016).
127. C. D. Oliver, N. T. Nassar, B. R. Lippke, J. B. McCarter, Carbon, fossil fuel, and biodiversity mitigation with wood and forests. *J. Sustain. For.* **33**, 248–275 (2014).
128. U.S. Department of Agriculture Forest Service, *Increasing the Pace of Restoration and Job Creation on Our National Forests* (U.S. Department of Agriculture Forest Service, 2012); www.fs.fed.us/publications/restoration/restoration.pdf.
129. M. A. Cairns, S. Brown, E. H. Helmer, G. A. Baumgardner, Root biomass allocation in the world's upland forests. *Oecologia* **111**, 1–11 (1997).
130. J. C. Jenkins, D. C. Chojnacky, L. S. Heath, R. A. Birdsey, "Comprehensive database of diameter-based biomass regressions for North American tree species" (General Technical Report NE-319, U.S. Department of Agriculture Forest Service, 2004).
131. Intergovernmental Panel on Climate Change, *IPCC Guidelines for National Greenhouse Gas Inventories. Chapter 4 Forest Land* (Intergovernmental Panel on Climate Change, 2006).
132. J. Q. Chambers, N. Higuchi, J. P. Schimel, L. V. Ferreira, J. M. Melack, Decomposition and carbon cycling of dead trees in tropical forests of the central Amazon. *Oecologia* **122**, 380–388 (2000).
133. C. A. Williams, H. Gu, R. MacLean, J. G. Masek, G. J. Collatz, Disturbance and the carbon balance of US forests: A quantitative review of impacts from harvests, fires, insects, and droughts. *Glob. Planet. Change* **143**, 66–80 (2016).
134. B. Sohngen, S. Brown, Extending timber rotations: Carbon and cost implications. *Clim. Policy* **8**, 435–451 (2008).
135. U.S. Forest Service, *National Fire Plan Operations and Reporting System* (U.S. Forest Service, 2016); https://cohesivefire.nemac.org/node/251.
136. D. C. Lee, A. A. Ager, D. E. Calkin, M. A. Finney, M. P. Thompson, T. M. Quigley, C. W. McHugh, *A National Cohesive Wildland Fire Management Strategy* (U.S. Department of Agriculture Forest Service, 2011), pp. 1–44.
137. M. G. Rollins, LANDFIRE: A nationally consistent vegetation, wildland fire, and fuel assessment. *Int. J. Wildland Fire* **18**, 235–249 (2009).
138. C. Wiedinmyer, M. D. Hurteau, Prescribed fire as a means of reducing forest carbon emissions in the western United States. *Environ. Sci. Technol.* **44**, 1926–1932 (2010).
139. A. L. Westerling, Increasing western US forest wildfire activity: Sensitivity to changes in the timing of spring. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* **371**, 20150178 (2016).
140. J. Kellndorfer, W. Walker, K. Kirsch, G. Fiske, J. Bishop, L. LaPoint, M. Hoppus, J. Westfall, *NACP Aboveground Biomass and Carbon Baseline Data, V.2. (NBCD 2000), U.S.A., 2000* (Oak Ridge National Laboratory Distributed Active Archive Center, 2013); http://dx.doi.org/10.3334/ORNLDAAC/1161.
141. Intergovermental Panel on Climate Change, *Good Practice Guidance for Land Use, Land-Use Change and Forestry* (Intergovermental Panel on Climate Change, 2003); www.ipcc-nggip.iges.or.jp.
142. LANDFIRE, Fuel Loading Models, LANDFIRE 1.1.0 (2008); http://landfire.cr.usgs.gov/viewer/.
143. J. E. Smith, L. S. Heath, *A Model of Forest Floor Carbon Mass for United States Forest Types* (U.S. Department of Agriculture Forest Service, 2002); http://purl.access.gpo.gov/GPO/LPS25756.
144. G. W. Meigs, D. C. Donato, J. L. Campbell, J. G. Martin, B. E. Law, Forest fire impacts on carbon uptake, storage, and emission: The role of burn severity in the Eastern Cascades, Oregon. *Ecosystems* **12**, 1246–1267 (2009).
145. G. Collatz, C. Williams, B. Ghimire, S. Goward, J. Masek, CMS: Forest Biomass and Productivity, 1-degree and 5-km, Conterminous US, 2005 (Oak Ridge National Laboratory Distributed Active Archive Center, 2014); http://daac.ornl.gov/cgi-bin/dsviewer.pl?ds_id=1221.
146. S. Dore, M. Montes-Helu, S. C. Hart, B. A. Hungate, G. W. Koch, J. B. Moon, A. J. Finkral, T. E. Kolb, Recovery of ponderosa pine ecosystem carbon and water fluxes from thinning and stand-replacing fire. *Glob. Chang. Biol.* **18**, 3171–3185 (2012).
147. J. Eidenshink, B. Schwind, K. Brewer, Z.-L. Zhu, B. Quayle, S. M. Howard, A project for monitoring trends in burn severity. *Fire Ecol.* **3**, 3–21 (2007).
148. Monitoring Trends in Burn Severity, Annual Burn Severity Mosaics (2016); www.mtbs.gov/direct-download.
149. O. F. Price, R. A. Bradstock, J. E. Keeley, A. D. Syphard, The impact of antecedent fire area on burned area in southern California coastal ecosystems. *J. Environ. Manage.* **113**, 301–307 (2012).
150. B. R. Hartsough, S. Abrams, R. James Barbour, E. S. Drews, J. D. McIver, J. J. Moghaddas, D. W. Schwilk, S. L. Stephense, The economics of alternative fuel reduction treatments in western United States dry forests: Financial and policy implications from the National Fire and Fire Surrogate Study. *For. Policy Econ.* **10**, 344–354 (2008).
151. E. D. Reinhardt, R. E. Keane, D. E. Calkin, J. D. Cohen, Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. *For. Ecol. Manage.* **256**, 1997–2006 (2008).
152. LANDFIRE, Biophysical Settings, LANDFIRE 1.3.0 (2012); http://landfire.cr.usgs.gov/viewer/.
153. U.S. Department of Agriculture Forest Service Automated Lands Program (ALP), Forest Service Regional Boundaries. *S_USA.AdministrativeRegion* (2016); http://data.fs.usda.gov/geodata/edw/datasets.php.
154. J. G. Masek, W. B. Cohen, D. Leckie, M. A. Wulder, R. Vargas, B. de Jong, S. Healey, B. Law, R. Birdsey, R. A. Houghton, D. Mildrexler, S. Gowardm, W. B. Smith, Recent rates of forest harvest and conversion in North America. *J. Geophys. Res.* **116**, G00K03 (2011).
155. S. M. Nusser, J. J. Goebel, The National Resources Inventory: A long-term multi-resource monitoring programme. *Environ. Ecol. Stat.* **4**, 181–204 (1997).
156. U.S. Department of Agriculture, *Summary Report: 2007 National Resources Inventory* (Natural Resources Conservation Service, 2009).
157. F. Achard, H. D. Eva, P. Mayaux, H.-J. Stibig, A. Belward, Improved estimates of net carbon emissions from land cover change in the tropics for the 1990s. *Global Biogeochem. Cycles* **18**, GB2008 (2004).
158. Y. Li, M. Zhao, S. Motesharrei, Q. Mu, E. Kalnay, S. Li, Local cooling and warming effects of forests based on satellite observations. *Nat. Commun.* **6**, 6603 (2015).
159. C. Milesi, S. W. Running, C. D. Elvidge, J. B. Dietz, B. T. Tuttle, R. R. Nemani, Mapping and modeling the biogeochemical cycling of turf grasses in the United States. *Environ. Manage.* **36**, 426–438 (2005).
160. C. D. Campbell, J. R. Seiler, P. Eric Wiseman, B. D. Strahm, J. F. Munsell, Soil carbon dynamics in residential lawns converted from Appalachian mixed oak stands. *Forests* **5**, 425–438 (2014).
161. U.S. Geological Survey Gap Analysis Program, *Protected Areas Database of the United States (PAD-US), Version 1.4 Combined Feature Class* (U.S. Geological Survey, 2016); https://gapanalysis.usgs.gov/padus/.
162. N. L. Harris, S. C. Hagen, S. S. Saatchi, T. R. H. Pearson, C. W. Woodall, G. M. Domke, B. H. Braswell, B. F. Walters, S. Brown, W. Salas, A. Fore, Y. Yu, Attribution of net carbon change by disturbance type across forest lands of the conterminous United States. *Carbon Balance Manage.* **11**, 24 (2016).
163. D. Zheng, L. S. Heath, M. J. Ducey, J. E. Smith, Carbon changes in conterminous US forests associated with growth and major disturbances: 1992–2001. *Environ. Res. Lett.* **6**, 019502 (2011).
164. N. E. Thomas, C. Huang, S. N. Goward, S. Powell, K. Rishmawi, K. Schleeweis, A. Hinds, Validation of North American Forest Disturbance dynamics derived from Landsat time series stacks. *Remote Sens. Environ.* **115**, 19–32 (2011).
165. P. Olofsson, G. M. Foody, M. Herold, S. V. Stehman, C. E. Woodcock, M. A. Wulder, Good practices for estimating area and assessing accuracy of land change. *Remote Sens. Environ.* **148**, 42–57 (2014).
166. S. Hagen, N. Harris, S. S. Saatchi, T. Pearson, C. W. Woodall, S. Ganguly, G. M. Domke, B. H. Braswell, B. F. Walters, J. C. Jenkins, S. Brown, W. A. Salas, A. Fore, Y. Yu, R. R. Nemani, C. Ipsan, K. R. Brown, *CMS: Forest Carbon Stocks, Emissions, and Net Flux for the Conterminous US: 2005-2010* (ORNL DAAC, 2016); http://dx.doi.org/10.3334/ORNLDAAC/1313.
167. J. A. Blackard, M. V. Finco, E. H. Helmer, G. R. Holden, M. L. Hoppus, D. M. Jacobs, A. J. Lister, G. G. Moisen, M. D. Nelson, R. Riemann, B. Ruefenacht, D. Salajanu, D. L. Weyermann, K. C. Winterberger, T. J. Brandeis, R. L. Czaplewski, R. E. McRoberts, P. L. Patterson, R. P. Tymcio, Mapping U.S. forest biomass using nationwide forest inventory data and moderate resolution information. *Remote Sens. Environ.* **112**, 1658–1677 (2008).
168. U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis National Program; www.fia.fs.fed.us/tools-data.
169. B. T. Wilson, C. W. Woodall, D. M. Griffith, Imputing forest carbon stock estimates from inventory plots to a nationally continuous coverage. *Carbon Balance Manage.* **8**, 1 (2013).
170. N. Neeti, R. Kennedy, Comparison of national level biomass maps for conterminous US: Understanding pattern and causes of differences. *Carbon Balance Manage.* **11**, 19 (2016).

171. R. N. Lubowski, A. J. Plantinga, R. N. Stavins, Land-use change and carbon sinks: Econometric estimation of the carbon sequestration supply function. *J. Environ. Econ. Manage.* **51**, 135–152 (2006).
172. J. O. Sexton, J. O. Sexton, X.-P. Song, M. Feng, P. Noojipady, A. Anand, C. Huang, D.-H. Kim, K. M. Collins, S. Channan, C. DiMiceli, J. R. Townshend, Global, 30-m resolution continuous fields of tree cover: Landsat-based rescaling of MODIS vegetation continuous fields with lidar-based estimates of error. *Int. J. Digit. Earth* **6**, 427–448 (2013).
173. E. G. Mcpherson, J. R. Simpson, P. J. Peper, S. E. Maco, Q. Xiao, Municipal forest benefits and costs in five US cities. *J. For.* **103**, 411–416 (2005).
174. U.S. Department of Agriculture, Farm Service Agency, National Agricultural Inventory Program (2015); www.fsa.usda.gov/programs-and-services/aerial-photography/imagery-programs/naip-imagery/.
175. D. J. Nowak, E. J. Greenfield, Tree and impervious cover change in U.S. cities. *Urban For. Urban Green.* **11**, 21–30 (2012).
176. D. M. Olson, E. Dinerstein, E. D. Wikramanayake, N. D. Burgess, G. V. N. Powell, E. C. Underwood, J. A. D'amico, I. Itoua, H. E. Strand, J. C. Morrison, C. J. Loucks, T. F. Allnutt, T. H. Ricketts, Y. Kura, J. F. Lamoreux, W. W. Wettengel, P. Hedao, K. R. Kassem, Terrestrial ecoregions of the world: A new map of life on Earth: A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. *Bioscience* **51**, 933–938 (2001).
177. N. Bassuk, D. Curtis, B. Marranca, B. Neal, *Recommended Urban Trees: Site Assessment and Tree Selection for Stress Tolerance* (Cornell University, 2009); www.hort.cornell.edu/uhi/outreach/recurbtree/pdfs/~recurbtrees.pdf.
178. L. Bounoua, J. Nigro, P. Zhang, K. Thome, Mapping impact of urbanization in the continental U.S. from 2001–2020, in *2016 IEEE International Geoscience and Remote Sensing Symposium (IGARSS)* (IEEE, 2016), pp. 6750–6753.
179. E. G. McPherson, J. R. Simpson, P. J. Peper, K. I. Scott, Q. Xiao, *Tree Guidelines for Coastal Southern California Communities* (Local Government Commission, 2000); www.lgc.org/wordpress/docs/freepub/energy/guides/socal_tree_guidelines.pdf.
180. E. G. McPherson, S. E. Maco, J. R. Simpson, P. J. Peper, Q. Xiao, A. Van Der Zanden, N. Bell, *Western Washington and Oregon Community Tree Guide: Benefits, Costs and Strategic Planting* (U.S. Department of Agriculture Forest Service, 2002); www.treesearch.fs.fed.us/pubs/45962).
181. E. G. McPherson, J. R. Simpson, P. J. Peper, Q. Xiao, S. E. Maco, P. J. Hoefer, *Northern Mountain and Prairie Community Tree Guide: Benefits, Costs and Strategic Planting* (U.S. Department of Agriculture Forest Service, 2003); www.fs.fed.us/psw/topics/urban_forestry/products/cufr_258.pdf.
182. H. Pretzsch, P. Biber, E. Uhl, J. Dahlhausen, T. Rötzer, J. Caldentey, T. Koike, T. van Con, A. Chavanne, T. Seifert, B. du Toit, C. Farnden, S. Pauleit, Crown size and growing space requirement of common tree species in urban centres, parks, and forests. *Urban For. Urban Green.* **14**, 466–479 (2015).
183. D. J. Nowak, E. J. Greenfield, R. E. Hoehn, E. Lapoint, Carbon storage and sequestration by trees in urban and community areas of the United States. *Environ. Pollut.* **178**, 229–236 (2013).
184. S. N. Oswalt, W. B. Smith, *U.S. Forest Resource Facts and Historical Trends* (U.S. Department of Agriculture, 2014).
185. R. Harper, G. Hernandez, J. Arseneault, M. Bryntesen, S. Enebak, R. Overton, Forest nursery seedling production in the United States—Fiscal year 2012. *Tree Plant. Notes* **56**, 72–75 (2013).
186. J. Bond, *The Inclusion of Large-Scale Tree Planting in a State Implementation Plan: A Feasibility Study* (Davey Research Group, 2006); www.urbanforestanalytics.com/sites/default/files/pdf/TreesInSIP.pdf.
187. U.S. Census Bureau, Table 1. Annual Estimates of the Resident Population for the United States, Regions, States, and Puerto Rico: April 1, 2010 to July 1, 2015 (NST-EST2015-01); https://factfinder.census.gov/faces/tableservices/jsf/pages/productview.xhtml?src=bkmk.
188. M. W. Strohbach, E. Arnold, D. Haase, The carbon footprint of urban green space—A life cycle approach. *Landsc. Urban Plan.* **104**, 220–229 (2012).
189. G. H. Donovan, D. T. Butry, The value of shade: Estimating the effect of urban trees on summertime electricity use. *Energy Build.* **41**, 662–668 (2009).
190. H. Akbari, Shade trees reduce building energy use and CO_2 emissions from power plants. *Environ. Pollut.* **116**, S119–S126 (2002).
191. M. R. McHale, E. G. McPherson, I. C. Burke, The potential of urban tree plantings to be cost effective in carbon credit markets. *Urban For. Urban Green.* **6**, 49–60 (2007).
192. L. A. Roman, J. J. Battles, J. R. McBride, *Urban Tree Mortality: A Primer on Demographic Approaches* (U.S. Department of Agriculture Forest Service, 2016); www.treesearch.fs.fed.us/pubs/50688.
193. J. B. Bradford, D. N. Kastendick, Age-related patterns of forest complexity and carbon storage in pine and aspen–birch ecosystems of northern Minnesota, USA. *Can. J. For. Res.* **40**, 401–409 (2010).
194. WM Financial Strategies, Rates Over Time - Interest Rate Trends; www.munibondadvisor.com/market.htm.
195. T. Hengl, J. M. de Jesus, G. B. M. Heuvelink, M. R. Gonzalez, M. Kilibarda, A. Blagotić, W. Shangguan, M. N. Wright, X. Geng, B. Bauer-Marschallinger, M. A. Guevara, R. Vargas, R. A. MacMillan, N. H. Batjes, J. G. B. Leenaars, E. Ribeiro, I. Wheeler, S. Mantel, B. Kempen, SoilGrids250m: Global gridded soil information based on machine learning. *PLOS ONE* **12**, e0169748 (2017).
196. European Space Agency, *Land Cover CCI Product User Guide Version 2.0* (European Space Agency, 2017), pp. 1–105.
197. P. L. Sims, J. S. Singh, The structure and function of ten Western North American grasslands: II. Intra-seasonal dynamics in primary producer compartments. *J. Ecol.* **66**, 547–572 (1978).
198. P. L. Sims, J. S. Singh, W. K. Lauenroth, The structure and function of ten Western North American Grasslands: I. Abiotic and vegetational characteristics. *J. Ecol.* **66**, 251–285 (1978).
199. M. B. Cleary, E. Pendall, B. E. Ewers, Aboveground and belowground carbon pools after fire in mountain big sagebrush steppe. *Rangel. Ecol. Manag.* **63**, 187–196 (2010).
200. B. A. Bradley, R. A. Houghton, J. F. Mustard, S. P. Hamburg, Invasive grass reduces aboveground carbon stocks in shrublands of the Western US. *Glob. Chang. Biol.* **12**, 1815–1822 (2006).
201. Conservation Reserve Program Average Payments by County (2017); https://catalog.data.gov/dataset/conservation-reserve-program-average-payments-by-county.
202. U.S. Department of Agriculture National Agricultural Statistics Service, *Crop Production 2016 Summary* (U.S. Department of Agriculture National Agricultural Statistics Service, 2017); www.nass.usda.gov/.
203. U.S. Department of Agriculture National Agricultural Statistics Service, *2012 Census of Agriculture* (U.S. Department of Agriculture National Agricultural Statistics Service, 2014); www.nass.usda.gov/Publications/AgCensus/2012/.
204. K. D. Belfry, L. L. Van Eerd, Establishment and impact of cover crops intersown into corn. *Crop. Sci.* **56**, 1245–1256 (2016).
205. P. Smith, D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, O. Sirotenko, M. Howden, T. McAllister, G. Pan, V. Romanenkov, U. Schneider, S. Towprayoon, M. Wattenbach, J. Smith, Greenhouse gas mitigation in agriculture. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* **363**, 789–813 (2008).
206. J. M. Baker, T. E. Ochsner, R. T. Venterea, T. J. Griffis, Tillage and soil carbon sequestration—What do we really know? *Agric. Ecosyst. Environ.* **118**, 1–5 (2007).
207. Z. Luo, E. Wang, O. J. Sun, Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. *Agric. Ecosyst. Environ.* **139**, 224–231 (2010).
208. C. Palm, H. Blanco-Canqui, F. DeClerck, L. Gatere, P. Grace, Conservation agriculture and ecosystem services: An overview. *Agric. Ecosyst. Environ.* **187**, 87–105 (2014).
209. D. S. Powlson, C. M. Stirling, M. L. Jat, B. G. Gerard, C. A. Palm, P. A. Sanchez, K. G. Cassman, Limited potential of no-till agriculture for climate change mitigation. *Nat. Clim. Change* **4**, 678–683 (2014).
210. A. J. VandenBygaart, The myth that no-till can mitigate global climate change. *Agric. Ecosyst. Environ.* **216**, 98–99 (2016).
211. J. Six, S. M. Ogle, F. J. Breidt, R. T. Conant, A. R. Mosier, K. Paustian, The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. *Glob. Chang. Biol.* **10**, 155–160 (2004).
212. B. A. Linquist, M. A. Adviento-Borbe, C. M. Pittelkow, C. van Kessel, K. J. van Groenigen, Fertilizer management practices and greenhouse gas emissions from rice systems: A quantitative review and analysis. *Field Crops Res.* **135**, 10–21 (2012).
213. P. R. Hill, Use of continuous no-till and rotational tillage systems in the central and northern Corn Belt. *J. Soil Water Conserv.* **56**, 286–290 (2001).
214. S. S. Snapp, S. M. Swinton, R. Labarta, D. Mutch, J. R. Black, R. Leep, J. Nyiraneza, K. O'Neil, Evaluating cover crops for benefits, costs and performance within cropping system niches. *Agron. J.* **97**, 322–332 (2005).
215. M. Liebig, A. J. Franzluebbers, R. F. Follett, *Managing Agricultural Greenhouse Gases: Coordinated Agricultural Research Through GRACEnet to Address Our Changing Climate* (Elsevier, 2012).
216. ICF International, *Greenhouse Gas Mitigation Options and Costs for Agricultural Land and Animal Production Within the United States* (U.S. Department of Agriculture, 2013).
217. A. Clark, *Managing Cover Crops Profitably* (Sustainable Agriculture Network, ed. 3, 2007), vol. 9.
218. R. L. Cochran, R. K. Roberts, J. A. Larson, D. D. Tyler, Cotton profitability with alternative lime application rates, cover crops, nitrogen rates, and tillage methods. *Agron. J.* **99**, 1085–1092 (2007).
219. P. Smith, M. Bustamante, H. Ahammad, H. Clark, H. Dong, E. A. Elsiddig, H. Haberl, R. Harper, J. House, M. Jafari, O. Masera, C. Mbow, N. H. Ravindranath, C. W. Rice, C. Robledo Abad, A. Romanovskaya, F. Sperling, F. Tubiello, *Agriculture, Forestry and Other Land Use (AFOLU)* (Cambridge Univ. Press, 2014).
220. C. L. Keene, W. S. Curran, Optimizing high-residue cultivation timing and frequency in reduced-tillage soybean and corn. *Agron. J.* **108**, 1897–1906 (2016).

221. K. A. O'Reilly, J. D. Lauzon, R. J. Vyn, L. L. Van Eerd, Nitrogen cycling, profit margins and sweet corn yield under fall cover crop systems. *Can. J. Soil Sci.* **92**, 353–365 (2012).
222. R. K. Roberts, J. A. Larson, D. D. Tyler, B. N. Duck, K. D. Dillivan, Economic analysis of the effects of winter cover crops on no-tillage corn yield response to applied nitrogen. *J. Soil Water Conserv.* **53**, 280–284 (1998).
223. CTIC, SARE, ASTA, *Annual Report 2015-2016: Cover Crop Survey* (CTIC, SARE, ASTA, 2016).
224. S. Jeffery, F. G. A. Verheijen, M. van der Velde, A. C. Bastos, A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. *Agric. Ecosyst. Environ.* **144**, 175–187 (2011).
225. U.S. Department of Energy, *2016 Billion-Ton Report: Advancing Domestic Resources for a Thriving Bioeconomy, Volume 1: Economic Availability of Feedstocks* (Oak Ridge National Laboratory, 2016).
226. K. A. Spokas, Review of the stability of biochar in soils: Predictability of O:C molar ratios. *Carbon Manage.* **1**, 289–303 (2010).
227. D. Woolf, J. E. Amonette, F. A. Street-Perrott, J. Lehmann, S. Joseph, Sustainable biochar to mitigate global climate change. *Nat. Commun.* **1**, 56 (2010).
228. P. Gallagher, M. Dikeman, J. Fritz, E. Wailes, W. Gauther, H. Shapouri, "Biomass from crop residues: Cost and supply estimates" (Agricultural Economic Report No. 819, U.S. Department of Agriculture, 2003).
229. A. Milbrandt, "A geographic perspective on the current biomass resource availability in the United States" (Technical Report NREL/TP-560-39181, National Renewable Energy Laboratory, 2005).
230. S. Kumarappan, S. Joshi, H. L. MacLean, Biomass supply for biofuel production: Estimates for the United States and Canada. *BioResources* **4**, 1070–1087 (2009).
231. J. S. Gregg, S. J. Smith, Global and regional potential for bioenergy from agricultural and forestry residue biomass. *Mitigation Adapt. Strategies Glob. Change* **15**, 241–262 (2010).
232. A. Chatterjee, Annual crop residue production and nutrient replacement costs for Bioenergy feedstock production in United States. *Agron. J.* **105**, 685–692 (2013).
233. A. Demirbas, Effects of temperature and particle size on bio-char yield from pyrolysis of agricultural residues. *J. Anal. Appl. Pyrolysis* **72**, 243–248 (2004).
234. H. Sun, W. C. Hockaday, C. A. Masiello, K. Zygourakis, Multiple controls on the chemical and physical structure of biochars. *Ind. Eng. Chem. Res.* **51**, 3587–3597 (2012).
235. Y. Sun, B. Gao, Y. Yao, J. Fang, M. Zhang, Y. Zhou, H. Chen, L. Yang, Effects of feedstock type, production method, and pyrolysis temperature on biochar and hydrochar properties. *Chem. Eng. J.* **240**, 574–578 (2014).
236. Y. Lee, J. Park, C. Ryu, K. S. Gang, W. Yang, Y.-K. Park, J. Jung, S. Hyun, Comparison of biochar properties from biomass residues produced by slow pyrolysis at 500°C. *Bioresour. Technol.* **148**, 196–201 (2013).
237. L. Zhao, X. Cao, O. Mašek, A. Zimmerman, Heterogeneity of biochar properties as a function of feedstock sources and production temperatures. *J. Hazard. Mater.* **256–257**, 1–9 (2013).
238. R. P. Udawatta, S. Jose, Agroforestry strategies to sequester carbon in temperate North America. *Agroforest. Syst.* **86**, 225–242 (2012).
239. P. K. R. Nair, B. M. Kumar, V. D. Nair, Agroforestry as a strategy for carbon sequestration. *J. Plant Nutr. Soil Sci.* **172**, 10–23 (2009).
240. A. D. Bambrick, J. K. Whalen, R. L. Bradley, A. Cogliastro, A. M. Gordon, A. Olivier, N. V. Thevathasan, Spatial heterogeneity of soil organic carbon in tree-based intercropping systems in Quebec and Ontario, Canada. *Agroforest. Syst.* **79**, 343–353 (2010).
241. R. Cardinael, T. Chevallier, B. G. Barthès, N. P. A. Saby, T. Parent, C. Dupraz, M. Bernoux, C. Chenu, Impact of alley cropping agroforestry on stocks, forms and spatial distribution of soil organic carbon—A case study in a Mediterranean context. *Geoderma* **259–260**, 288–299 (2015).
242. P. Tsonkova, C. Böhm, A. Quinkenstein, D. Freese, Ecological benefits provided by alley cropping systems for production of woody biomass in the temperate region: A review. *Agroforest. Syst.* **85**, 133–152 (2012).
243. M. Oelbermann, R. P. Voroney, N. V. Thevathasan, A. M. Gordon, D. C. L. Kass, A. M. Schlönvoigt, Soil carbon dynamics and residue stabilization in a Costa Rican and southern Canadian alley cropping system. *Agroforest. Syst.* **68**, 27–36 (2006).
244. M. Peichl, N. V. Thevathasan, A. M. Gordon, J. Huss, R. A. Abohassan, Carbon sequestration potentials in temperate tree-based intercropping systems, Southern Ontario, Canada. *Agroforest. Syst.* **66**, 243–257 (2006).
245. S. Lu, P. Meng, J. Zhang, C. Yin, S. Sun, Changes in soil organic carbon and total nitrogen in croplands converted to walnut-based agroforestry systems and orchards in southeastern Loess Plateau of China. *Environ. Monit. Assess.* **187**, 688 (2015).
246. F. Montagnini, P. K. R. Nair, Carbon sequestration: An underexploited environmental benefit of agroforestry systems. *Agroforest. Syst.* **61–62**, 281–295 (2004).
247. J. R. Thiessen Martens, M. H. Entz, M. D. Wonneck, Review: Redesigning Canadian prairie cropping systems for profitability, sustainability, and resilience. *Can. J. Plant Sci.* **95**, 1049–1072 (2015).
248. G. M. M. M. A. Senaviratne, R. P. Udawatta, K. A. Nelson, K. Shannon, S. Jose, Temporal and spatial influence of perennial upland buffers on corn and soybean yields. *Agron. J.* **104**, 1356–1362 (2012).
249. M. A. Cary, G. E. Frey, D. E. Mercer, The value of versatile alley cropping in the Southeast US: A Monte Carlo simulation, in *Proceedings of the Inaugural Symposium of the International Society of Forest Resource Economics 2014* (U.S. Department of Agriculture Forest Service, 2014), pp. 1–6; www.srs.fs.usda.gov/pubs/ja/2014/ja_2014_frey_001.pdf.
250. G. Garrett, W. Walter, L. D. Godsey, Alley cropping: Farming between the trees. *Green Horizons.* **19**, 1 (2015).
251. L. Harper, W. Kurtz, Economics of Eastern black walnut agroforestry systems, in *Nut Production Handbook for Eastern Black Walnut* (Southwest Missouri Resources, Conservation & Development, 1998), pp. 32–36.
252. W. T. Stamps, R. L. McGraw, L. Godsey, T. L. Woods, The ecology and economics of insect pest management in nut tree alley cropping systems in the Midwestern United States. *Agric. Ecosyst. Environ.* **131**, 4–8 (2009).
253. U.S. Department of Agriculture National Agroforestry Center, Alley cropping. A relic from the past or a bridge to the future? *Inside Agroforest.* **19**, 1–12 (2009).
254. G. E. Frey, D. E. Mercer, F. W. Cubbage, R. C. Abt, Economic potential of agroforestry and forestry in the lower Mississippi alluvial valley with incentive programs and carbon payments. *South. J. Appl. For.* **34**, 176–185 (2011).
255. D. E. Mercer, G. E. Cubbage, F. W. Frey, Economics of agroforestry, in *Handbook of Forest Resource Economics*, S. Kant, J. R. R. Alavalapati, Eds. (Earthscan from Routledge, 2014), pp. 188–209.
256. U.S. Department of Agriculture, *Summary Report: 2012 National Resources Inventory* (U.S. Department of Agriculture, 2015); www.nrcs.usda.gov/technical/nri/12summary.
257. A. Wotherspoon, N. V. Thevathasan, A. M. Gordon, R. P. Voroney, Carbon sequestration potential of five tree species in a 25-year-old temperate tree-based intercropping system in southern Ontario, Canada. *Agroforest. Syst.* **88**, 631–643 (2014).
258. E. A. Davidson, The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. *Nat. Geosci.* **2**, 659–662 (2009).
259. C. S. Snyder, T. W. Bruulsema, T. L. Jensen, P. E. Fixen, Review of greenhouse gas emissions from crop production systems and fertilizer management effects. *Agric. Ecosyst. Environ.* **133**, 247–266 (2009).
260. J. Hill, S. Polasky, E. Nelson, D. Tilman, H. Huo, L. Ludwig, J. Neumann, H. Zheng, D. Bonta, Climate change and health costs of air emissions from biofuels and gasoline. *Proc. Natl. Acad. Sci. U.S.A.* **106**, 2077–2082 (2009).
261. J. Lelieveld, J. S. Evans, M. Fnais, D. Giannadaki, A. Pozzer, The contribution of outdoor air pollution sources to premature mortality on a global scale. *Nature* **525**, 367–371 (2015).
262. G. Myhre, D. Shindell, F.-M. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, J.-F. Lamarque, D. Lee, B. Mendoza, T. Nakajima, A. Robock, G. Stephens, T. Takemura, H. Zhang, Anthropogenic and Natural Radiative Forcing, in *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, P. M. Midgley, Eds. (Cambridge Univ. Press, 2013), vol. 423, pp. 659–740.
263. S. Sela, H. M. van Es, B. N. Moebius-Clune, R. Marjerison, J. Melkonian, D. Moebius-Clune, R. Schindelbeck, S. Gomes, Adapt-N outperforms grower-selected nitrogen rates in northeast and midwestern united states strip trials. *Agron. J.* **108**, 1726–1734 (2016).
264. Food and Agriculture Organization of the United Nations, *FAOSTAT Online Statistical Service* (Food and Agriculture Organization of the United Nations, 2014); www.fao.org/faostat/en/#data.
265. E. Stehfest, L. Bouwman, N_2O and NO emission from agricultural fields and soils under natural vegetation: Summarizing available measurement data and modeling of global annual emissions. *Nutr. Cycling Agroecosyst.* **74**, 207–228 (2006).
266. D. L. Burton, X. Li, C. A. Grant, Influence of fertilizer nitrogen source and management practice on N_2O emissions from two Black Chernozemic soils. *Can. J. Soil Sci.* **88**, 219–227 (2008).
267. R. T. Venterea, M. S. Dolan, T. E. Ochsner, Urea decreases nitrous oxide emissions compared with anhydrous ammonia in a Minnesota corn cropping system. *Soil Sci. Soc. Am. J.* **74**, 407–418 (2010).
268. M. J. Bell, J. M. Cloy, C. F. E. Topp, B. C. Ball, A. Bagnall, R. M. Rees, D. R. Chadwick, Quantifying N_2O emissions from intensive grassland production: The role of synthetic fertilizer type, application rate, timing and nitrification inhibitors. *J. Agric. Sci.* **154**, 812–827 (2016).
269. X. Hao, C. Chang, J. M. Carefoot, H. H. Janzen, B. H. Ellert, Nitrous oxide emissions from an irrigated soil as affected by fertilizer and straw management. *Nutr. Cycling Agroecosyst.* **60**, 1–8 (2001).
270. C. F. Drury, W. D. Reynolds, X. M. Yang, N. B. McLaughlin, T. W. Welacky, W. Calder, C. A. Grant, Nitrogen source, application time, and tillage effects on soil nitrous oxide emissions and corn grain yields. *Soil Sci. Soc. Am. J.* **76**, 1268–1279 (2011).

271. J. P. Burzaco, D. R. Smith, T. J. Vyn, Nitrous oxide emissions in Midwest US maize production vary widely with band-injected N fertilizer rates, timing and nitrapyrin presence. *Environ. Res. Lett.* **8**, 035031 (2013).
272. R. T. Venterea, J. A. Coulter, M. S. Dolan, Evaluation of intensive "4R" strategies for decreasing nitrous oxide emissions and nitrogen surplus in rainfed corn. *J. Environ. Qual.* **45**, 1186–1195 (2016).
273. R. H. Beach, B. J. DeAngelo, S. Rose, C. Li, W. Salas, S. J. DelGrosso, Mitigation potential and costs for global agricultural greenhouse gas emissions. *Agric. Econ.* **38**, 109–115 (2008).
274. D. S. Reay, E. A. Davidson, K. A. Smith, P. Smith, J. M. Melillo, F. Dentener, P. J. Crutzen, Global agriculture and nitrous oxide emissions. *Nat. Clim. Change* **2**, 410–416 (2012).
275. O. Oenema, X. Ju, C. Klein, M. Alfaro, A. Prado, J. P. Lesschen, X. Zheng, G. Velthof, L. Ma, B. Gao, C. Kroeze, M. Sutton, Reducing nitrous oxide emissions from the global food system. *Curr. Opin. Environ. Sustain.* **9–10**, 55–64 (2014).
276. P. C. West, J. S. Gerber, P. M. Engstrom, N. D. Mueller, K. A. Brauman, K. M. Carlson, E. S. Cassidy, M. Johnston, Graham K. MacDonald, D. K. Ray, S. Siebert, Leverage points for improving global food security and the environment. *Science* **345**, 325–328 (2014).
277. M. Ribaudo, J. Delgado, L. Hansen, M. Livingston, R. Mosheim, J. Williamson, "Nitrogen in agricultural systems: Implications for conservation policy" (Economic Research Report No. ERR-127, U.S. Department of Agriculture Economic Research Service, 2011).
278. N. Millar, G. P. Robertson, P. R. Grace, R. J. Gehl, J. P. Hoben, Nitrogen fertilizer management for nitrous oxide (N_2O) mitigation in intensive corn (Maize) production: An emissions reduction protocol for US Midwest agriculture. *Mitigation Adapt. Strategies Glob. Change* **15**, 185–204 (2010).
279. U. Sehy, R. Ruser, J. C. Munch, Nitrous oxide fluxes from maize fields: Relationship to yield, site-specific fertilization, and soil conditions. *Agric. Ecosyst. Environ.* **99**, 97–111 (2003).
280. P. C. Scharf, D. K. Shannon, H. L. Palm, K. A. Sudduth, S. T. Drummond, N. R. Kitchen, L. J. Mueller, V. C. Hubbard, L. F. Oliveira, Sensor-based nitrogen applications out-performed producer-chosen rates for corn in on-farm demonstrations. *Agron. J.* **103**, 1683–1691 (2011).
281. D. Schimmelpfenning, *Farm Profits and Adoption of Precision Agriculture* (United States Department of Agriculture Economic Research Service, 2016); www.ers.usda.gov/webdocs/publications/80326/err-217.pdf?v=4266.
282. U.S. Department of Agriculture National Agricultural Statistics Service, *Agricultural Prices* (U.S. Department of Agriculture National Agricultural Statistics Service, 2017); http://usda.mannlib.cornell.edu/usda/nass/AgriPric//2010s/2017/AgriPric-12-28-2017.pdf.
283. D. B. Mengel, *Types and Uses of Nitrogen Fertilizers for Crop Production* (Purdue University, 1986).
284. B. G. Bareja, General information and practices in using urea fertilizer (2013); www.cropsreview.com.
285. G. A. Helmers, J. Brandle, Optimum windbreak spacing in great plains agriculture. *Great Plains Res.* **15**, 179–198 (2005).
286. Y. G. Chendev, L. L. Novykh, T. J. Sauer, C. L. Petin, A. N. Zazdravnykh, E. A. Burras, in *Soil Carbon Progress in Soil Science*, A. E. Hartemink, K. McSweeney, Eds. (Springer International Publishing, 2014), pp. 475–482.
287. F. Wang, X. Xu, B. Zou, Z. Guo, Z. Li, W. Zhu Biomass accumulation and carbon sequestration in four different aged *Casuarina equisetifolia* coastal shelterbelt plantations in South China. *PLOS ONE* **8**, e77449 (2013).
288. T. J. Sauer, C. A. Cambardella, J. R. Brandle, Soil carbon and tree litter dynamics in a red cedar–scotch pine shelterbelt. *Agroforest. Syst.* **71**, 163–174 (2007).
289. J. Kort, R. Turnock, Carbon reservoir and biomass in Canadian prairie shelterbelts. *Agroforest. Syst.* **44**, 175–186.
290. M. M. Schoeneberger, Agroforestry: Working trees for sequestering carbon on agricultural lands. *Agroforest. Syst.* **75**, 27–37 (2008).
291. U.S. Department of Agriculture Farm Service Agency, *Conservation Reserve Program: Annual Summary and Enrollment Statistics-FY2010* (U.S. Department of Agriculture Farm Service Agency, 2010).
292. B. Henderson, A. Falcucci, A. Mottet, L. Early, B. Werner, H. Steinfeld, P. Gerber, Marginal costs of abating greenhouse gases in the global ruminant livestock sector. *Mitigation Adapt. Strategies Glob. Change* **22**, 199–224 (2017).
293. L. M. Porensky, E. A. Leger, J. Davison, W. W. Miller, E. M. Goergen, E. K. Espeland, E. M. Carroll-Moore, Arid old-field restoration: Native perennial grasses suppress weeds and erosion, but also suppress native shrubs. *Agric. Ecosyst. Environ.* **184**, 135–144 (2014).
294. I. Kämpf, N. Hölzel, M. Störrle, G. Broll, K. Kiehl, Potential of temperate agricultural soils for carbon sequestration: A meta-analysis of land-use effects. *Sci. Total Environ.* **566–567**, 428–435 (2016).
295. D. L. Gebhart, H. B. Johnson, H. S. Mayeux, H. W. Polley, The CRP increases soil organic carbon. *J. Soil Water Conserv.* **49**, 488–492 (1994).
296. J. Fargione, J. Hill, D. Tilman, S. Polasky, P. Hawthorne, Land clearing and the biofuel carbon debt. *Science* **319**, 1235–1238 (2008).
297. B. O. Sander, R. Wassmann, D. L. C. Siopongco, *Mitigating Greenhouse Gas Emissions from Rice Production Through Water-Saving Techniques: Potential, Adoption and Empirical Evidence* (International Rice Research Institute, 2015).
298. X. Yan, H. Akiyama, K. Yagi, H. Akimoto, Global estimations of the inventory and mitigation potential of methane emissions from rice cultivation conducted using the 2006 Intergovernmental Panel on Climate Change guidelines. *Global Biogeochem. Cycles* **23**, GB2002 (2009).
299. U.S. Department of Agriculture Foreign Agricultural Service, "World agricultural production" (Circular Series WAP 05-17, U.S. Department of Agriculture Foreign Agricultural Service, 2017).
300. P. Fazli, H. C. Man, Comparison of methane emission from conventional and modified paddy cultivation in Malaysia. *Agric. Agric. Sci. Proc.* **2**, 272–279 (2014).
301. M. Peyron, C. Bertora, S. Pelissetti, D. Said-Pullicino, L. Celi, E. Miniotti, M. Romani, D. Sacco, Greenhouse gas emissions as affected by different water management practices in temperate rice paddies. *Agric. Ecosyst. Environ.* **232**, 17–28 (2016).
302. C. M. Pittelkow, Y. Assa, M. Burger, R. G. Mutters, C. A. Greer, L. A. Espino, J. E. Hill, W. R. Horwath, C. van Kessel, B. A. Linquist, Nitrogen management and methane emissions in direct-seeded rice systems. *Agron. J.* **106**, 968–980 (2014).
303. A. L. Hinson, R. A. Feagin, M. Eriksson, R. G. Najjar, M. Herrmann, T. S. Bianchi, M. Kemp, J. A. Hutchings, S. Crooks, T. Boutton, The spatial distribution of soil organic carbon in tidal wetland soils of the continental United States. *Glob. Chang. Biol.* **23**, 5468–5480 (2017).
304. H. J. Poffenbarger, B. A. Needelman, J. P. Megonigal, Salinity influence on methane emissions from tidal marshes. *Wetlands.* **31**, 831–842 (2011).
305. Intergovernmental Panel on Climate Change, *2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands* (Intergovernmental Panel on Climate Change, 2013).
306. S. C. Neubauer, J. P. Megonigal, Moving beyond global warming potentials to quantify the climatic role of ecosystems. *Ecosystems* **18**, 1000–1013 (2015).
307. F. E. Anderson, B. Bergamaschi, C. Sturtevant, S. Knox, L. Hastings, L. Windham-Myers, M. Detto, E. L. Hestir, J. Drexler, R. L. Miller, J. H. Matthes, J. Verfaillie, D. Baldocchi, R. L. Snyder, R. Fujii, Variation of energy and carbon fluxes from a restored temperate freshwater wetland and implications for carbon market verification protocols. *J. Geophys. Res. Biogeosci.* **121**, 777–795 (2016).
308. S. D. Bridgham, J. P. Megonigal, J. K. Keller, N. B. Bliss, C. Trettin, The carbon balance of North American wetlands. *Wetlands* **26**, 889–916 (2006).
309. U.S. Environmental Protection Agency, *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2014* (Environmental Protection Agency, 2016).
310. C. Richardson, R. Evans, D. Carr, in *Pocosin Wetlands: An Integrated Analysis of Coastal Plain Freshwater Bogs in North Carolina* (Hutchinson Ross Publishing Company, 1981), pp. 3–19.
311. S. D. Bridgham, C. J. Richardson, Mechanisms controlling soil respiration (CO_2 and CH_4) in southern peatlands. *Soil Biol. Biochem.* **24**, 1089–1099 (1992).
312. J. P. Megonigal, W. H. Schlesinger, Enhanced CH_4 emission from a wetland soil exposed to elevated CO_2. *Biogeochemistry* **37**, 77–88 (1997).
313. H. Wang, C. J. Richardson, M. Ho, Dual controls on carbon loss during drought in peatlands. *Nat. Clim. Change* **5**, 584–587 (2015).
314. U.S. Department of Agriculture Natural Resources Conservation Service, *Web Soil Survey (SSURGO)* (U.S. Department of Agriculture Natural Resources Conservation Service, 2016); http://websoilsurvey.nrcs.usda.gov/app/.
315. S. D. Bridgham, C. L. Ping, J. L. Richardson, K. Updegraff, in *Wetland Soils: Genesis, Hydrology, Landscapes, and Classification*, J. L. Richardson, M. J. Vepraskas, Eds. (CRC Press, 2001), pp. 343–370.
316. U.S. Fish & Wildlife Service, *National Wetlands Inventory* (U.S. Fish & Wildlife Service, 2017); www.fws.gov/wetlands/Data/Data-Download.html.
317. C. Homer, J. Dewitz, L. Yang, S. Jin, P. Danielson, G. Xian, J. Coulston, N. Herold, J. Wickham, K. Megown, Completion of the 2011 national land cover database for the conterminous United States – representing a decade of land cover change information. *Photogramm. Eng. Remote Sens.* **81**, 345–354 (2015).
318. C. J. Richardson, N. Flanagan, H. Wang, M. Ho, "Impacts of peatland ditching and draining on water quality and carbon sequestration benefits of peatland restoration" (Duke University for the Eastern North Carolina/Southeastern Virginia Strategic Habitat Conservation Team, U.S. Fish and Wildlife Service, Region 4 and The Nature Conservancy North Carolina Chapter, Final Project, 2014).
319. L. Hansen, D. M. Hellerstein, M. O. Ribaudo, J. Williamson, D. Nulph, C. Loesch, W. Crumpton, *Targeting Investments to Cost Effectively Restore and Protect Wetland Ecosystems: Some Economic Insights* (United States Department of Agriculture Economic Research Service, 2015); https://ageconsearch.umn.edu/bitstream/199283/2/ERR183.pdf.
320. J. Howard, A. Sutton-Grier, D. Herr, J. Kleypas, E. Landis, E. Mcleod, E. Pidgeon, S. Simpson, Clarifying the role of coastal and marine systems in climate mitigation. *Front. Ecol. Environ.* **15**, 42–50 (2017).

321. CEC, *North American Blue Carbon Scoping Study* (Commission for Environmental Cooperation, 2013).
322. B. C. Murray, L. Pendleton, W. A. Jenkins, S. Sifleet, "Green payments for blue carbon: Economic incentives for protecting threatened coastal habitats" (NI R 11-04, Nicholas Institute, Duke University, 2011).
323. G. Morrison, H. Greening, in *Integrating Science and Resource Management in Tampa Bay, Florida: U.S. Geological Survey Circular 1348*, K. K. Yates, H. Greening, G. Morrison, Eds. (U.S. Geological Survey Circular 1348, 2011) pp. 105–156.
324. Tampa Bay Regional Planning Council, *Integrating Nitrogen Management with Planning* (2013), Final Technical Report #07-13 of the Tampa Bay Estuary Program (available at https://tbeptech.org/34mkf/6bb2b_ngbz/vxoi13.dol).
325. H. Greening, A. Janicki, Toward reversal of eutrophic conditions in a subtropical estuary: Water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. *Environ. Manage.* **38**, 163–178 (2006).
326. Janicki Environmental Inc., *Estimates of Total Nitrogen, Total Phosphorus, Total Suspended Solids, and Biochemical Oxygen Demand Loadings To Tampa Bay, Florida: 2007-2011* (Tampa Bay Estuary Program, 2013); www.tbeptech.org/TBEP_TECH_PUBS/2013/TBEP_03_13_FINAL_TBEP_Loads_2007-2011 19Mar2013.pdf.
327. S. B. Bricker, Nutrient pollution in US Estuaries: NOAA's National Estuarine Eutrophication Assessment informs nutrient management; https://water.usgs.gov/nawqa/headlines/nut_pest/NOAA-Estuaries-Bricker.pdf.
328. S. Cooper, "Integrating nitrogen management with planning" (Technical Report 07-13, Tampa Bay Estuary Program, 2012).
329. E. T. Sherwood, *2016 Tampa Bay Water Quality Assessment* (Tampa Bay Estuary Program, 2017); www.tbeptech.org/TBEP_TECH_PUBS/2017/TBEP_01_17_2016_Decision_Matrix_Results_Update.pdf.
330. S. S. Rabotyagov, T. D. Campbell, M. White, J. G. Arnold, J. Atwood, M. L. Norfleet, C. L. Kling, P. W. Gassman, A. Valcu, J. Richardson, R. E. Turner, N. N. Rabalais, Cost-effective targeting of conservation investments to reduce the northern Gulf of Mexico hypoxic zone. *Proc. Natl. Acad. Sci. U.S.A.* **111**, 18530–18535 (2014).
331. S. S. Rabotyagov, C. L. Kling, P. W. Gassman, N. N. Rabalais, R. E. Turner, The economics of dead zones: Causes, impacts, policy challenges, and a model of the gulf of Mexico Hypoxic Zone. *Rev. Environ. Econ. Policy* **8**, 58–79 (2014).
332. C. M. Duarte, N. Marbà, D. Krause-Jensen, M. Sánchez-Camacho, Testing the predictive power of seagrass depth limit models. *Estuaries Coast.* **30**, 652–656 (2007).
333. C. M. Duarte, Seagrass depth limits. *Aquat. Bot.* **40**, 363–377 (1991).
334. J. T. Greiner, K. J. McGlathery, J. Gunnell, B. A. McKee, Seagrass restoration enhances "blue carbon" sequestration in coastal waters. *PLOS ONE* **8**, e72469 (2013).
335. E. Bayraktarov, M. I. Saunders, S. Abdullah, M. Mills, J. Beher, H. P. Possingham, P. J. Mumby, C. E. Lovelock, The cost and feasibility of marine coastal restoration. *Ecol. Appl.* **26**, 1055–1074 (2016).
336. U.S. Environmental Protection Agency, *Guidelines for Preparing Economic Analyses* (U.S. Environmental Protection Agency, 2010); www.epa.gov/environmental-economics/guidelines-preparing-economic-analyses.
337. World Bank, *Lending Interest Rate (%)* (World Bank, 2017); https://data.worldbank.org/indicator/FR.INR.LEND?locations=US.
338. U.S. Department of Agriculture Economic Research Service, *Fertilizer Use and Price* (U.S. Department of Agriculture Economic Research Service, 2013); www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx.
339. D. J. Nowak, S. Hirabayashi, A. Bodine, E. Greenfield, Tree and forest effects on air quality and human health in the United States. *Environ. Pollut.* **193**, 119–129 (2014).
340. R. Harrison, G. Wardell-Johnson, C. McAlpine, Rainforest reforestation and biodiversity benefits: A case study from the Australian wet tropics. *Ann. Trop. Res.* **25**, 65–76 (2003).
341. K. Niijima, A. Yamane, Effects of reforestation on soil fauna in the Philippines. *Philipp. J. Sci.* **120**, 1–20 (1991).
342. P. J. Ferraro, K. Lawlor, K. L. Mullan, S. K. Pattanayak, Forest figures: Ecosystem services valuation and policy evaluation in developing countries. *Rev. Environ. Econ. Policy* **6**, 20–44 (2012).
343. Z. Burivalova, Ç. H. Şekercioğlu, L. P. Koh, Thresholds of logging intensity to maintain tropical forest biodiversity. *Curr. Biol.* **24**, 1893–1898 (2014).
344. M. F. Jurgensen, A. E. Harvey, R. T. Graham, D. S. Page-Dumroese, J. R. Tonn, M. J. Larsen, T. B. Jain, Impacts of Timber harvesting on soil organic matter, nitrogen, productivity, and health of inland northwest forests. *For. Sci.* **43**, 234–251 (1997).
345. T. A. Burton, Effects of basin-scale Timber harvest on water yield and peak streamflow. *J. Am. Water Resour. Assoc.* **33**, 1187–1196 (1997).
346. S. Vedal, S. J. Dutton, Wildfire air pollution and daily mortality in a large urban area. *Environ. Res.* **102**, 29–35 (2006).
347. J. Bengtsson, S. G. Nilsson, A. Franc, P. Menozzi, Biodiversity, disturbances, ecosystem function and management of European forests. *For. Ecol. Manage.* **132**, 39–50 (2000).
348. K. T. Takano, M. Nakagawa, T. Itioka, K. Kishimoto-Yamada, S. Yamashita, H. O. Tanaka, D. Fukuda, H. Nagamasu, M. Ichikawa, Y. Kato, K. Momose, T. Nakashizuka, S. Sakai, *Social-Ecological Systems in Transition* (Global Environmental Studies, 2014); http://link.springer.com/chapter/10.1007/978-4-431-54910-9_2/fulltext.html.
349. E. Gómez-Baggethun, D. N. Barton, Classifying and valuing ecosystem services for urban planning. *Ecol. Econ.* **86**, 235–245 (2013).
350. L. Chaparro, J. Terrasdas, Ecological services of urban forest in Barcelona. *Shengtai Xuebao/Acta Ecol. Sin.* **29**, 103 (2009).
351. U. G. Sandström, P. Angelstam, G. Mikusiński, Ecological diversity of birds in relation to the structure of urban green space. *Landsc. Urban Plan.* **77**, 39–53 (2006).
352. Y. Depietri, F. G. Renaud, G. Kallis, Heat waves and floods in urban areas: A policy-oriented review of ecosystem services. *Sustain. Sci.* **7**, 95–107 (2012).
353. M. J. Hartley, Rationale and methods for conserving biodiversity in plantation forests. *For. Ecol. Manage.* **155**, 81–95 (2002).
354. M. Ausden, W. J. Sutherland, R. James, The effects of flooding lowland wet grassland on soil macroinvertebrate prey of breeding wading birds. *J. Appl. Ecol.* **38**, 320–338 (2001).
355. G. W. Randall, D. R. Huggins, M. P. Russelle, D. J. Fuchs, W. W. Nelson, J. L. Anderson, Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. *J. Environ. Qual.* **26**, 1240–1247 (1997).
356. M. J. Helmers, X. Zhou, H. Asbjornsen, R. Kolka, M. D. Tomer, R. M. Cruse, Sediment removal by prairie filter strips in row-cropped ephemeral watersheds. *J. Environ. Qual.* **41**, 1531–1539 (2012).
357. H. Jankowska-Huflejt, The function of permanent grasslands in water resources protection. *J. Water Land Dev.* **10**, 55–65 (2006).
358. P. Smith, M. R. Ashmore, H. I. J. Black, P. J. Burgess, C. D. Evans, T. A. Quine, A. M. Thomson, K. Hicks, H. G. Orr, The role of ecosystems and their management in regulating climate, and soil, water and air quality. *J. Appl. Ecol.* **50**, 812–829 (2013).
359. R. Derpsch, T. Friedrich, A. Kassam, H. Li, Current status of adoption of no-till farming in the world and some of its main benefits. *Int. J. Agric. Biol. Eng.* **3**, 1–25 (2010).
360. M. J. Bell, F. Worrall, Charcoal addition to soils in NE England: A carbon sink with environmental co-benefits? *Sci. Total Environ.* **409**, 1704–1714 (2011).
361. S. Jose, Agroforestry for ecosystem services and environmental benefits: An overview. *Agrofor. Syst.* **76**, 1–10 (2009).
362. S. Pattanayak, D. E. Mercer, Valuing soil conservation benefits of agroforestry: Contour hedgerows in the Eastern Visayas, Philippines. *Agric. Econ.* **18**, 31–46 (1998).
363. D. W. Bussink, Relationships between ammonia volatilization and nitrogen fertilizer application rate, intake and excretion of herbage nitrogen by cattle on grazed swards. *Fertil. Res.* **38**, 111–121 (1994).
364. M. D. Einheuser, A. P. Nejadhashemi, S. P. Sowa, L. Wang, Y. A. Hamaamin, S. A. Woznicki, Modeling the effects of conservation practices on stream health. *Sci. Total Environ.* **435–436**, 380–391 (2012).
365. G. Woodward, M. O. Gessner, P. S. Giller, V. Gulis, S. Hladyz, A. Lecerf, B. Malmqvist, B. G. McKie, S. D. Tiegs, H. Cariss, M. Dobson, A. Elosegi, V. Ferreira, M. A. S. Graça, T. Fleituch, J. O. Lacoursière, M. Nistorescu, J. Pozo, G. Risnoveanu, M. Schindler, A. Vadineanu, L. B.-M. Vought, E. Chauvet, Continental-scale effects of nutrient pollution on stream ecosystem functioning. *Science* **336**, 1438–1440 (2012).
366. M. Quemada, M. Baranski, M. N. J. Nobel-de Lange, A. Vallejo, J. M. Cooper, Meta-analysis of strategies to control nitrate leaching in irrigated agricultural systems and their effects on crop yield. *Agric. Ecosyst. Environ.* **174**, 1–10 (2013).
367. S. R. Carpenter, N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, V. H. Smith, Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecol. Appl.* **8**, 559–568 (1998).
368. M. Bustamante, C. Robledo-Abad, R. Harper, C. Mbow, N. H. Ravindranat, F. Sperling, H. Haberl, A. de Siqueira Pinto, P. Smith, Co-benefits, trade-offs, barriers and policies for greenhouse gas mitigation in the agriculture, forestry and other land use (AFOLU) sector. *Glob. Chang. Biol.* **20**, 3270–3290 (2014).
369. D. Malakoff, Death by suffocation in the Gulf of Mexico. *Science* **281**, 190–192 (1998).
370. P. S. Hooda, A. C. Edwards, H. A. Anderson, A. Miller, A review of water quality concerns in livestock farming areas. *Sci. Total Environ.* **250**, 143–167 (2000).
371. A. Kruess, T. Tscharntke, grazing intensity and the diversity of grasshoppers, butterflies, and trap-nesting bees and wasps. *Conserv. Biol.* **16**, 1570–1580 (2002).
372. C. A. Rotz, S. Asem-Hiablie, J. Dillon, H. Bonifacio, Cradle-to-farm gate environmental footprints of beef cattle production in Kansas, Oklahoma, and Texas. *J. Anim. Sci.* **93**, 2509–2519 (2015).
373. N. M. Haddad, G. M. Crutsinger, K. Gross, J. Haarstad, J. M. H. Knops, D. Tilman, Plant species loss decreases arthropod diversity and shifts trophic structure. *Ecol. Lett.* **12**, 1029–1039 (2009).
374. E. S. Jensen, H. Hauggaard-Nielsen, How can increased use of biological N_2 fixation in agriculture benefit the environment? *Plant Soil* **252**, 177–186 (2003).
375. S. Toze, Reuse of effluent water—Benefits and risks. *Agric. Water Manage.* **80**, 147–159 (2006).
376. B. W. Heumann, Satellite remote sensing of mangrove forests: Recent advances and future opportunities. *Prog. Phys. Geogr.* **35**, 87–108 (2011).
377. B. A. Polidoro, K. E. Carpenter, L. Collins, N. C. Duke, A. M. Ellison, J. C. Ellison, E. J. Farnsworth, E. S. Fernando, K. Kathiresan, N. E. Koedam, S. R. Livingstone, T. Miyagi,

G. E. Moore, V. N. Nam, J. E. Ong, J. H. Primavera, S. G. Salmo III, J. C. Sanciangco, S. Sukardjo, Y. Wang, J. W. H. Yong, The loss of species: Mangrove extinction risk and geographic areas of global concern. *PLOS ONE* **5**, e10095 (2010).
378. J. B. Zedler, Wetlands at your service: Reducing impacts of agriculture at the watershed scale. *Front. Ecol. Environ.* **1**, 65–72 (2003).
379. M. D. Correll, W. A. Wiest, T. P. Hodgman, W. G. Shriver, C. S. Elphick, B. J. McGill, K. M. O'Brien, B. J. Olsen, Predictors of specialist avifaunal decline in coastal marshes. *Conserv. Biol.* **31**, 172–182 (2017).
380. E. B. Barbier, S. D. Hacker, C. Kennedy, E. W. Koch, A. C. Stier, B. R. Silliman, The value of estuarine and coastal ecosystem services. *Ecol. Monogr.* **81**, 169–193 (2011).
381. A. G. Rappold, S. L. Stone, W. E. Cascio, L. M. Neas, V. J. Kilaru, M. S. Carraway, J. J. Szykman, A. Ising, W. E. Cleve, J. T. Meredith, H. Vaughan-Batten, L. Deyneka, R. B. Devli, Peat bog wildfire smoke exposure in rural North Carolina is associated with cardiopulmonary emergency department visits assessed through syndromic surveillance. *Environ. Health Perspect.* **119**, 1415–1420 (2011).
382. S. Page, A. Hosciło, H. Wösten, J. Jauhiainen, M. Silvius, J. Rieley, H. Ritzema, K. Tansey, L. Graham, H. Vasander, S. Limin, Restoration ecology of lowland tropical peatlands in Southeast Asia: Current knowledge and future research directions. *Ecosystems* **12**, 888–905 (2009).
383. S. Chapman, A. Buttler, A.-J. Francez, F. Laggoun-Défarge, H. Vasander, M. Schloter, J. Combe, P. Grosvernier, H. Harms, D. Epron, D. Gilbert, E. Mitchell, Exploitation of northern peatlands and biodiversity maintenance: A conflict between economy and ecology. *Front. Ecol. Environ.* **1**, 525–532 (2003).
384. J. P. Curry, J. A. Good, *Soil Restoration* (Springer New York, 1992); http://link.springer.com/chapter/10.1007/978-1-4612-2820-2_7.
385. D. P. L. Rousseau, E. Lesage, A. Story, P. A. Vanrolleghem, N. De Pauw, Constructed wetlands for water reclamation. *Desalination* **218**, 181–189 (2008).
386. R. Unsworth, L. C. Cullen-Unsworth, in *Coastal Conservation* (Cambridge Univ. Press, 2014), pp. 95–130.
387. U.S. Department of Agriculture Forest Service, *Fiscal Year 2009 President's Budget: Budget Justification* (U.S. Department of Agriculture Forest Service, 2008), pp. 1–426.
388. E. Lichtenberg, J. C. Hanson, A. M. Decker, A. J. Clark, Profitability of legume cover crops in the mid-Atlantic region. *J. Soil Water Conserv.* **49**, 562–565 (1994).
389. M. R. Pratt, W. E. Tyner, D. J. Muth, E. J. Kladivko, Synergies between cover crops and corn stover removal. *Agric. Syst.* **130**, 67–76 (2014).
390. U.S. Department of Agriculture Natural Resources Conservation Service, *Adding Cover Crops for Seed Production to a Corn/Soybean Rotation* (U.S. Department of Agriculture Natural Resources Conservation Service, 2015); https://efotg.sc.egov.usda.gov/references/public/MO/Seed_Production_CaseStudy3.pdf).
391. A. R. Smith, R. S. Tubbs, W. D. Shurley, M. D. Toews, G. D. Collins, G. H. Harris, *Economics of Cover Crop and Supplemental Fertilizer in Strip-Tillage Cotton* (The University of Georgia, College of Agricultural and Environmental Sciences, 2014); https://secure.caes.uga.edu/extension/publications/files/pdf/AP 108-2_1.PDF.
392. D. F. Roberts, N. R. Kitchen, K. A. Sudduth, S. T. Drummond, P. C. Scharf, Economic and environmental implications of sensor-based nitrogen management. *Better Crops* **94**, 4–6 (2010).
393. P. C. Scharf, N. R. Kitchen, K. A. Sudduth, J. G. Davis, V. C. Hubbard, J. A. Lory, Field-scale variability in optimal nitrogen fertilizer rate for corn. *Agron. J.* **97**, 452–461 (2005).
394. N. Hong, P. C. Scharf, J. G. Davis, N. R. Kitchen, K. A. Sudduth, Economically optimal nitrogen rate reduces soil residual nitrate. *J. Environ. Qual.* **36**, 354–362 (2007).
395. C. Snyder, E. Davidson, P. Smith, R. Venterea, Agriculture: Sustainable crop and animal production to help mitigate nitrous oxide emissions. *Curr. Opin. Environ. Sustain.* **9–10**, 46–54 (2014).
396. Association of American Plant Food Control Officials, The Fertilizer Institute, *Commercial Fertilizers 2013* (Association of American Plant Food Control Officials, The Fertilizer Institute, Fertilizer/Ag Lime Control Service, University of Missouri, 2014).
397. U.S. Department of Agriculture, *USDA Agriculture and Forestry Greenhouse Gas Inventory: 1990–2008* (U.S. Department of Agriculture, 2011).
398. M. J. Brown, G. M. Smith, J. McCollum, *Wetland Forest Statistics for the South Atlantic States* (U.S. Department of Agriculture Forest Service, 2001); www.srs.fs.usda.gov/pubs/rb/rb_srs062.pdf).

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Is fire "for the birds"? How two rare species influence fire management across the US

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Is fire "for the birds"? How two rare species influence fire management across the US

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The US Endangered Species Act has enabled species conservation but has differentially impacted fire management and rare bird conservation in the southern and western US. In the South, prescribed fire and restoration-based forest thinning are commonly used to conserve the endangered red-cockaded woodpecker (*Picoides borealis*; RCW), whereas in the West, land managers continue to suppress fire across the diverse habitats of the northern, Californian, and Mexican spotted owls (*Strix occidentalis* subspecies; SO). Although the habitat needs of the RCW and SO are not identical, substantial portions of both species' ranges have historically been exposed to relatively frequent, low- to moderate-intensity fires. Active management with fire and thinning has benefited the RCW but proves challenging in the western US. We suggest the western US could benefit from the adoption of a similar innovative approach through policy, public–private partnerships, and complementarity of endangered species management with multiple objectives. These changes would likely balance long-term goals of SO conservation and enhance forest resilience.

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The US Endangered Species Act (ESA) influences forest and fire management across the US. In both the southern and the western US, several species of conservation concern exist in forests with historically frequent, low- to moderate-intensity fire regimes. In the southern US, the use of prescribed burning to restore historical fire regimes has strong institutional support; it is ubiquitously applied for a broad range of resource benefits (Kobziar *et al*. 2015; Melvin 2015), which are familiar to both the general public and land management personnel. Across the western US, however, wildfire suppression is favored over prescribed burning, which is carried out relatively infrequently due to a number of factors, including limited institutional capacity (Quinn-Davidson and Varner 2011; Schultz *et al*. 2018), air quality concerns (Schultz *et al*. 2018), and the lack of a large trained workforce.

In a nutshell:

- The US Endangered Species Act has influenced fire management differently across regions, even with species that are adapted to similar fire regimes
- Frequent prescribed fire and restoration thinning have greatly improved red-cockaded woodpecker (*Picoides borealis*) recovery in the southern US, while in the western US, large severe wildfires are a major threat to spotted owl (*Strix occidentalis*) recovery
- Complementary benefits from habitat restoration and fuels reduction using fire and thinning help explain the red-cockaded woodpecker's recovery in the South
- Integrating the beneficial roles of fire and restoration thinning into spotted owl conservation in the West may be critical for maintaining its habitat, especially with a changing climate

Given the nationwide requirements for species conservation across the US, the differences between approaches in the two regions may be linked to how the ESA differentially affects fire management. Here, we focus on two emblematic species – the red-cockaded woodpecker (*Picoides borealis*; RCW) of the southern US and the spotted owl (*Strix occidentalis*, including three subspecies; SO) of the western US (Figure 1) – to explore the relationship between the ESA and fire management. RCW habitat and population recovery programs are widely hailed as a success, whereas conservation of SO habitat has been less effective.

The ESA and fire management intersect in three important ways. First, conservation goals are driven by the ecology of the species at risk, how fire affects its preferred habitat, and the scale at which those effects are relevant. Second, even where the relationships between fire regimes and threatened species habitat factors are well understood, the ESA can be used to justify fire suppression and to limit active management. Finally, when considering practices directly aimed at species conservation, policy makers may view fire manage-

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Figure 1. (a) Red-cockaded woodpecker (*Picoides borealis*; RCW) in Florida; (b) California spotted owl (*Strix occidentalis occidentalis*; CSO) in the Sierra Nevada.

ment objectives as being of equal or greater importance. For this reason, fire management may be driving endangered species management decisions rather than the reverse.

We recognize that there are differences between the life histories and habitat needs of the RCW and SO (WebTable 1). However, our goal is to compare and contrast two distinct but related fire paradigms affecting these iconic birds in the US. We explore how the intersection of fire management and rare species protection differs between the two regions in order to better understand what promotes species conservation success in the context of wildland fire management.

Red-cockaded woodpecker in longleaf pine

The RCW is endemic to the pine–grasslands, woodlands, and savannas of 11 US states, and was federally designated as "Endangered" in 1968, prior to passage of the ESA. The species was once common throughout the southern US, coinciding primarily with the distribution of fire-maintained southern pines. Current RCW populations are scattered, and many are isolated, with less than 3% of the bird's pre-colonial population remaining (USFWS 2003). RCWs prefer low-density, pine-dominated habitats, which in the South were historically maintained by very frequent (1–5-year intervals), low-intensity fires. The drastic reductions in RCW abundance that have occurred over the past several decades are primarily due to habitat loss caused by land conversion, development, logging, loss of mature living pines (>75 years), and fire exclusion. Collectively, these changes have resulted in the proliferation of hardwoods and reduced herbaceous understories. The RCW was initially considered an old-growth obligate fire-dependent species because these birds created roost cavities exclusively in large, fire-maintained living pines. Today, RCWs utilize most southern pine tree species even at younger ages, partially as a result of artificial nest cavity installations (Conner *et al.* 2001).

Specific fire regime attributes are critically important for promoting and conserving longleaf pine (*Pinus palustris*) habitat for the RCW, with the most important factor being consistent application of frequent fire. Ultimately, the formula for RCW recovery in longleaf pine has been straightforward ever since the bird was listed: sustaining prescribed fire management explicitly supported by policy, strategic creation and preservation of nest cavities, RCW introduction and translocation programs, and reduced regulatory burden of recovery on private lands through "Safe Harbor" programs (Trainor *et al.* 2013). These measures have likely prevented socioeconomic obstacles from hindering RCW restoration (eg forest thinning can continue as long as RCW habitat is maintained) and have contributed to the widespread support for fire restoration across land ownerships. Also critical is the compatibility of RCW habitat restoration actions with broader forest management goals. Potential wildfire severity reduction associated with prescribed burning has been compatible with the land management needs of government agencies, especially the US Department of Defense (DoD), as well as those of private landowners. Although current RCW population numbers are well below historical estimates, results indicate a clear positive trajectory (Figure 2).

However, this trajectory has not been without costs. Although no official figures have been published regarding the cost of recovery, the DoD – which manages all or part of six of 13 core RCW populations – has provided more than $10 million annually for management for nearly 25 years, with an additional estimated $67 million spent on research and development for RCW recovery between 1991 and 2004 (K Hiers pers comm). This investment often goes toward what some see as excessive use of expensive techniques for managing overly precise habitat parameters (Hiers *et al.* 2014, 2016), which may jeopardize habitats for other rare taxa (Hiers *et al.* 2014). Recent research shows that in relatively well-restored habitat, RCWs may be more tolerant of a wider range of modest hardwood densities than was previously seen in degraded habitat, and therefore "one-size-fits-all" guidelines are not appropriate across the species' range (McKellar *et al.* 2014, 2016). A focus on narrow habitat metrics can lead to ecosystem homogenization, which can compromise long-term resilience of longleaf pine ecosystems. Such considerations are particularly critical to climate resilience strategies that call for RCW habitat expansion outside of the former range of longleaf pine.

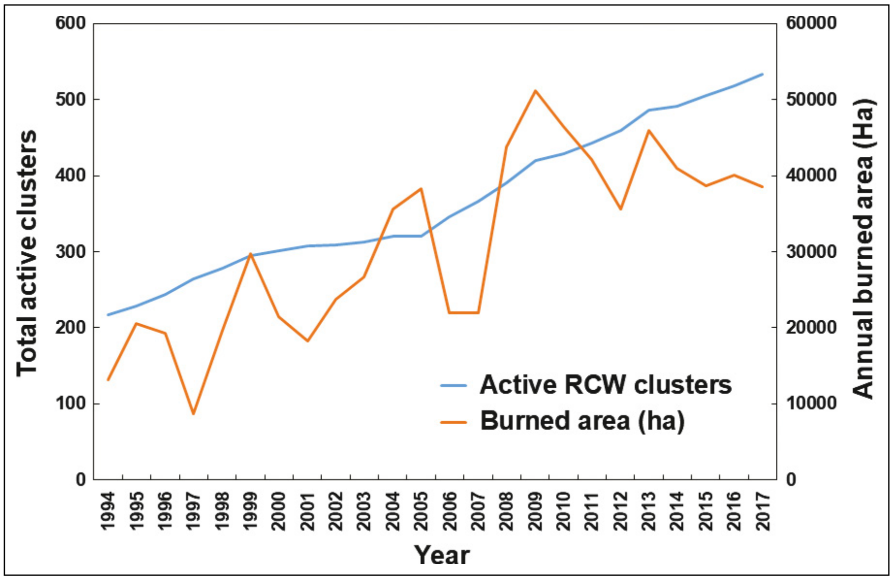

Figure 2. Red-cockaded woodpecker (*Picoides borealis*; RCW) recovery at the US Department of Defense Eglin Air Force Base in northern Florida shows population increases (as indicated by active use of cavity trees by RCW) correspond with increasing area of prescribed burning over the past 23 years.

Red-cockaded woodpecker in shortleaf pine

Farther north, the RCW was once common in fire-dependent shortleaf pine (*Pinus echinata*)–bluestem (*Andropogon* spp) ecosystems, and is at present most abundant in the Ouachita Highlands of Arkansas and Oklahoma. In these areas, prior to the mid-1920s, low tree density and pine dominance were maintained with low-intensity surface fires (every 1–12 years, mean interval = 4.6 years) that limited hardwood establishment (Masters *et al.* 1995). By 1970, this community type, along with the RCW, had all but disappeared because of logging and fire exclusion (Hedrick *et al.* 2007). Following a Forest Plan revision in 2005, the Ouachita National Forest committed to restoring this ecosystem for RCWs and associated species on over 100,000 ha in Arkansas and Oklahoma. Restoration efforts consist of thinning to modify stand density, basal area, and composition, and reintroducing frequent (3-year return interval) prescribed fire (Figure 3; Bukenhofer *et al.* 1994; Guldin 2007). Restored stands are characterized by a three- to seven-fold increase in grass and forb productivity, depending on time since fire (Sparks *et al.* 1998).

The benefits of this ecosystem management approach extend well beyond a single species. Research has demonstrated marked increases in a suite of at least 16 woodland–grassland obligate songbirds, most of which are of special management concern in Arkansas and Oklahoma (Figure 4; Wilson *et al.* 1995). The small mammal community and game species of local interest have benefited (Masters *et al.* 1993, 1998; Cram *et al.* 2002; Masters 2007), and plant and invertebrate communities, particularly lepidopterans, have also experienced substantial population growth (Sparks *et al.* 1998; Hedrick *et al.* 2007).

Treatments are funded by commercial timber sales. Sale receipts are disbursed to the US Treasury and to local governments, with a portion of the timber sale proceeds reinvested in treatments to improve habitat under provisions of the Knutson-Vandenberg Act (Guldin 2007). As such, the presence of viable local markets for timber has been an instrumental element of the success of the RCW recovery in shortleaf pine ecosystems. RCW population growth was further accelerated by artificial management tactics, such as translocations and augmentations, snake exclusion devices, cavity inserts and restrictor installations, and the control of southern flying squirrels (*Glaucomys volans*), which compete with the RCW for cavity occupancy.

Another unique feature of this restoration program is the introduction of prescribed fire during the tree regeneration phase (stand age 3–5 years), which is crucial for the establishment of open stand structures conducive to other flora and fauna associated with this ecosystem. Fire use in the early stages of stand development also serves to favor native shortleaf pine over loblolly pine (*Pinus taeda*) and should eliminate the need for future treatments in middle-aged stands. If sufficient landscape-scale area is to be restored and maintained, increased use of prescribed burning is imperative.

As a result of improved understanding of RCW ecology and dedicated, ecologically informed fire management, most RCW populations across the South are now either stable or increasing. Recent range-wide population assessments show that the number of RCW breeding groups has increased from an estimated 4694 in the 1990s to 7800 at present (ie >49,000 individuals; W McDearman pers comm). This trend suggests that down-listing or even delisting of the species could occur 20 years ahead of previous expectations (USFWS 2003).

Spotted owls

Northern spotted owl

The northern spotted owl (*Strix occidentalis caurina*; NSO), designated as "Federally Threatened" in 1990, is found from

Figure 3. The wildlife stand improvement (WSI) process on the Ouachita National Forest in Arkansas and Oklahoma consists of: (a) hardwoods and smaller shortleaf pine (*Pinus echinata*) trees thinned from below, (b) prescribed burning on a 3-year cycle, (c) hardwood sprouts restricted to <1-m height, and (d) the cumulative effects of 3–4 burns, which promote grass understory. The frequent burn cycle is then continued.

southwestern British Columbia, through Washington and Oregon, and into California. Although habitat loss from logging remains a concern on state and private lands, the primary threats to the NSO on federal lands are competition from the invasive barred owl (*Strix varia*; Singleton *et al*. 2010; Wiens *et al*. 2014) and loss of habitat due to large, high-severity wildfires (WebTable 1; Davis *et al*. 2016). Efforts to conserve the NSO on federal lands have included the establishment of a system of large Late-Successional Reserves, designated under the 1994 Northwest Forest Plan, and additional areas designated as Critical Habitat (Lesmeister *et al*. 2018).

In contrast to habitats for the RCW and other SO subspecies, the 23 million-ha range of the NSO encompasses a wide variety of fire regimes (WebFigure 1). In moist areas where fire occurred infrequently, the NSO's preferred habitat consists of large patches of older, structurally complex, closed canopy conifer forests. In the eastern and southern portions of the NSO's range (~40% of total range), where fire regimes promoted open canopy pine and mixed-conifer forests, nesting and roosting patches were far less common, occurring in locally moist or topographically protected areas (Hagmann *et al*. 2017). In the southern part of the NSO's range, consisting of relatively mesic sites where mixed-severity fire was once common, fires created a patchy landscape mosaic of shrubby vegetation, and open and closed canopy forests that favored the NSO and its prey (Figure 5).

In contrast to that of the RCW, portions of the NSO's range were historically supported by relatively frequent wildfires (return interval <20 years) occurring across large landscapes (10^4–10^5 ha). Spatial heterogeneity resulting from an intact fire regime allowed for the varied structures needed for viable nesting, roosting, and foraging. However, fire exclusion has resulted in fuels accumulation, greater tree density, and a higher degree of canopy layering, which have increased the abundance of NSO nesting and roosting habitat coincident with the likelihood of uncharacteristically large, high-severity fires. Fire exclusion in historically frequent-fire areas of the NSO's range allowed for widespread development of habitat in areas where it had not previously occurred (Hessburg *et al*. 2005). Now, competition from the barred owl may be displacing NSOs in these fire-excluded forests, which are on the margin of the NSO's range (Dugger *et al*. 2016).

Management actions in dry and mesic mixed-conifer forests that promote ecological resilience to fire and climate change at the landscape scale include allowing some wildfires to burn, reducing fuel continuity to enable safe reintroduction of fire, implementing restoration thinning and prescribed burning to restore the dominance of medium- and large-sized fire-resistant trees, restoring the patchwork of open and closed canopy forests and non-forest patches, and tailoring conditions to the topography (Hessburg *et al*. 2015). These actions would reduce the short-term availability of suitable NSO forest cover, but the remaining and future suitable forest cover would be more likely to persist in the face of wildfire and climate change. Meeting dual goals for conservation of fire-dependent ecosystems and NSO populations will require the development of dynamic, landscape-level habitat plans based on difficult trade-offs (Gaines *et al*. 2010; Hessburg *et al*. 2015).

California spotted owl

The California spotted owl (*Strix occidentalis occidentalis*; CSO) is found throughout the Sierra Nevada and the mountains of central coastal California, and in the Transverse and Peninsula ranges of southern California. A recent assessment found that CSO populations have declined in three of four long-term study areas over the past 20 years (Gutiérrez *et al*. 2017). All three areas of CSO decline are on US National Forest lands interspersed with private industrial timberland, whereas the study area in which there is a stable to increasing population is entirely on US National Park Service land. The amount of forest composed of large trees (>60 cm diameter at breast height [DBH]) and high canopy cover (>70%) within an owl territory was determined to be the best predictor of territory extinction rates (Jones *et al*. 2018). The invasive barred owl is also a serious threat to the CSO (Keane *et al*. 2017).

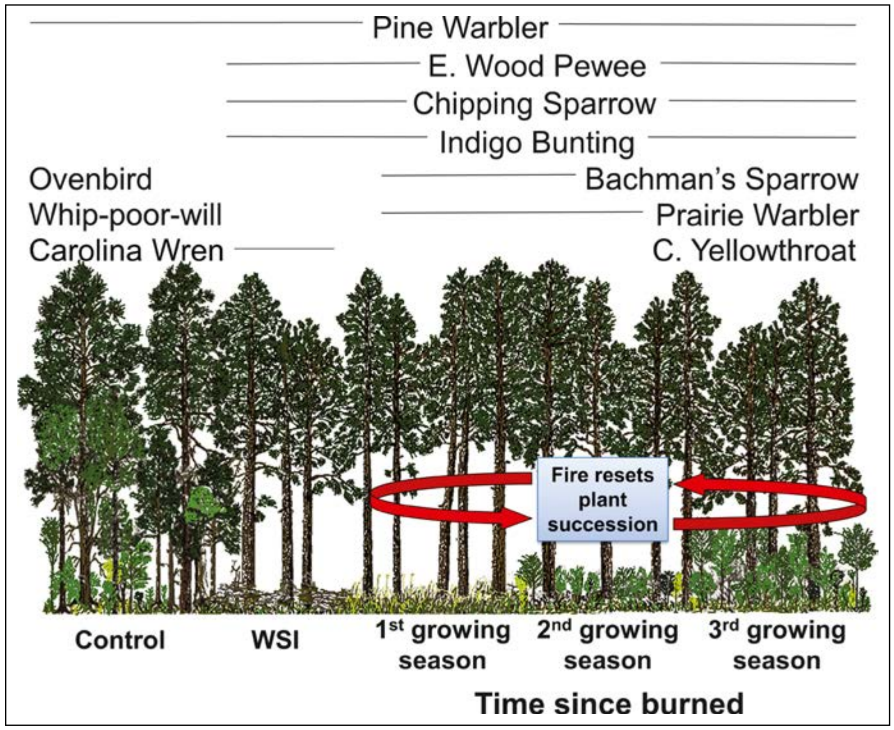

Figure 4. Community response of important bird species to a gradient of forest structures created in shortleaf pine–RCW restoration process that includes wildlife stand improvement (WSI) (thinning from below) and 1-, 2-, and 3-year post-burn conditions in contrast to controls. Bird names along with adjacent lines indicate species presence in the below structural stage. Bird occurrence in diagram is based on study data (1992–1993 and 1999–2000) from the Ouachita National Forest, Arkansas, in Wilson *et al.* (1995) and Masters *et al.* (2002). RCW habitat preferences consist of fire-maintained, thinned forests, which also promote increased songbird, amphibian, and small mammal diversity.

Most CSO habitat is mid-elevation (600–2200 m) mixed-conifer forest that historically experienced frequent (5–25 years) low- to moderate-intensity fires. Historical data and stand reconstruction studies suggest that prior to fire exclusion and large-tree logging, mixed-conifer forests averaged about 160 stems ha^{-1} and ~32% canopy cover, and included areas of denser forest (Safford and Stevens 2017). Since fire exclusion and large-tree logging became commonplace in the early 1900s, many Sierra Nevada mixed-conifer forests now have high surface and ladder fuel loads, with canopy cover exceeding 65% (Safford and Stevens 2017). Most modern-era burns result from wildfires escaping suppression (Miller *et al.* 2009). These wildfires commonly burn large areas with 25–35% stand-replacing effects and, perhaps more importantly, include large and homogenous areas characterized by stand-replacing severity (Stevens *et al*. 2017).

In response to the loss of large trees from past timber harvesting, mechanical fuel reduction treatments are now limited to trees <75 cm DBH on US Forest Service lands. Current CSO management maintains high canopy cover (>70%) within 120-ha protected activity centers (PACs) surrounding nest locations. As a result, PACs are often untreated and vulnerable to stand-replacing fire. For example, the 2014 King Fire severely burned one of the CSO long-term study areas, resulting in the largest owl population reduction over the study's 24-year history (Jones *et al*. 2016). On the basis of observed trends over the past several decades, Stephens *et al*. (2016a) projected that severe wildfire will reduce all potential CSO nesting habitat to <25% canopy cover within 75 years.

Given wildfire and drought trends, a new approach to managing CSO habitat is warranted. Conventional methods focus on maintaining relatively static "reserves" with high canopy cover. Much of the CSO's association with high canopy cover, however, comes from rough model estimates and limited sampling with imprecise field measurements. A study using LiDAR and a complete landscape census found total canopy cover was not as important to the CSO as tall (>32-m) tree cover (Figure 6; North *et al*. 2017). Furthermore, owls avoided cover in the 2–16-m height strata, suggesting that ladder fuel reduction may not be detrimental to CSOs. This more detailed understanding of canopy structure could be used to promote highly heterogeneous forest conditions, providing foraging habitat for prey species while maintaining dense local patches of tall tree cover in moister areas (Gutiérrez *et al*. 2017). Ecologically beneficial fire could also help sustain forests critical to CSO populations (Roberts *et al*. 2011), although such management has attendant risks and less precision than resto-

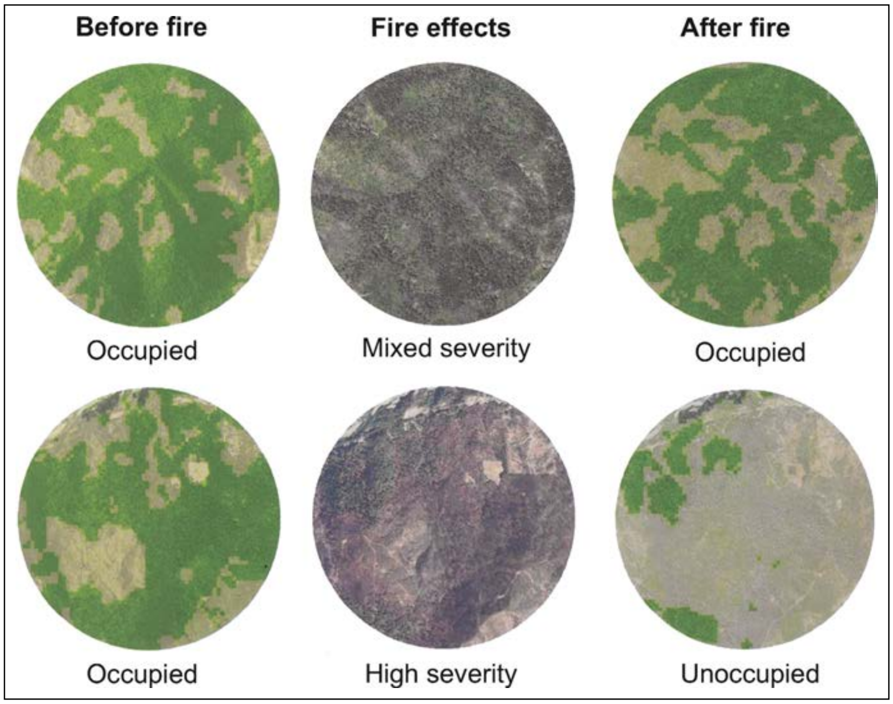

Figure 5. Aerial images of nesting and roosting (NR) forest cover (green shading) before and one year after wildfire in northwest California. The top row shows northern spotted owl (*Strix occidentalis caurina*) response to a mixed-severity fire that did not appreciably alter the pattern of NR forest cover; the territory was not abandoned. In contrast, the bottom row shows high-severity fire, which removed enough NR cover to cause territory abandonment. Occupancy data from Rockweit *et al.* (2017).

ration thinning. Without change, current policies will leave CSO habitat susceptible to severe wildfire, drought, and bark beetle mortality.

Mexican spotted owl

The Mexican spotted owl (*Strix occidentalis lucida*; MSO) inhabits forests and canyonlands in the southwestern US and the highlands of Mexico. Its distribution is driven largely by availability of suitable nesting habitat, which includes mixed-conifer or pine–oak forests with large trees and high canopy cover, and rocky canyons with shaded caves and cliff ledges (USFWS 2012). With recent reductions in timber harvests, high-severity wildfire is now considered by many researchers to be the primary threat to MSO nesting habitat. However, proposed restoration thinning treatments to alleviate risks of high-severity fire can also modify nesting habitat, and the effects of such modifications on MSOs remain largely unknown.

Some habitats used by MSOs in the southwestern US were historically among the most frequently burned forests in western North America (Swetnam and Baisan 2003). Repeated fires maintained relatively open canopy forests with thick-barked, dominant overstory species and rich understory diversity. A century of fire exclusion has led to substantial forest alteration, including increased tree density, canopy cover, and ladder fuels, and a shift in composition to shade-tolerant species in mixed-conifer forests (Margolis and Balmat 2009). A warming climate interacts with heavy fuels in modern forests to shift the fire regime toward infrequent, large, high-severity fires. Because trees in these forests lack adaptations that facilitate regeneration after large, severe fires, such as fire-induced sprouting and cone serotiny (which are also uncommon features in other SO habitats), recovery of MSO habitat can be slow or absent in some areas (Roccaforte *et al.* 2012). Management aimed at restoration of historical ecosystem attributes, including frequent low- to moderate-intensity fire regimes, has proven successful on some Native American lands (Stan *et al.* 2014), and tribal mandates of sustained-yield timber harvesting practices may be consistent with managing for the MSO (Hoagland *et al.* 2017). However, future warming increases uncertainty about maintaining these habitats even if uncharacteristically severe fires can be curtailed (Loehman *et al.* 2018).

Since 1989, management recommendations for MSOs on federal lands have emphasized protection of "Management Territories" or, more recently, PACs (USFWS 2012). Each of these PACs protects a minimum of 243 ha surrounding occupied nests. Secondary recommendations focus on protecting and developing "Recovery Habitat" to provide additional areas similar to nesting habitats, to facilitate population expansion. Desired forest habitat characteristics include large old trees and high canopy cover (USFWS 2012). These areas often also feature high fuel loads, and are therefore susceptible to high-severity wildfire. Thinning and burning is allowed in both PACs and Recovery Habitats, but requires considerable additional justification, analysis, and monitoring (USFWS 2012). Consequently, many managers focus fire-risk reduction activities outside of these areas, leaving occupied nesting sites vulnerable to high-severity fires. As with the CSO, integrating nesting habitat retention with plans to reduce landscape-scale fire risk and restore resilient forests remains one of the greatest challenges facing land managers in the southwestern US (Ganey *et al.* 2017; Wan *et al.* 2018).

Management implications

Although US federal law applies to both the southern and western US, the ability to implement effective, large-scale fire management and fuel treatment options varies greatly between the two regions, with differing consequences for the conservation of the RCW and SO. The considerable uncertainty regarding how the SO responds to fire (Ganey *et al.* 2017) likely contributes to a precautionary principle–driven management response (ie limited active fuels management and fire exclusion). There is ample evidence that this approach is likely to be counterproductive over the long term (Stephens *et al*. 2016a). It may be that fire suppression is exerting an equal or greater influence on SO habitat planning with regard to the ESA, rather than the reverse.

In the southern US, managing RCW habitat is a major driving force for fire use, in part because it complements conservation of other important species and improves long-term forest resilience. This is likely because the scale of treatments applied is consistent with RCW nesting and foraging needs. Moreover, despite substantial costs and a risk of homogeneous conditions resulting from overly precise management, prescribed fire, restoration thinning, cavity creation, and translocations have successfully expanded habitat for the RCW. This approach has profited economies and associated species while reducing fire severity, representing critical complementary benefits. Without such complementary benefits, it is unlikely that this management program would be exceeding projected recovery rates.

In the southern US, institutionally prioritized prescribed burning programs and state laws have facilitated the use of fire. RCW populations began to increase soon after the passage of the 1990 Florida Prescribed Fire Act and similar legislation across many southern states. In shortleaf pine forests, restoration thinning is also supported and, when combined with prescribed burning, provides timber resources for local governments and produces positive results for the RCW and associated species.

History and cultural context greatly influence fire management paradigms in each region: the complexity and nuance of ecological fire knowledge integration with Anglo-American forest management took four centuries to evolve in the South (Ryan *et al*. 2013). The integration in the West is comparatively new and is impeded by groups with polarizing views. There are several reasons for this, but the two key issues are lack of trust in agency motives for restoration and lack of willingness on the part of the interested public to prioritize large-scale forest resilience over potential impacts to local SO populations (Stephens *et al*. 2016b). Proactive fire use (prescribed and wildfire) and restoration thinning may be key to the long-term conservation of SO habitat; not only are these treatments appropriate for the majority of SO habitat types, they could improve resilience in forests adapted to frequent fire (Hessburg *et al*. 2015; Stephens *et al*. 2018). SO management could be shifted from a focus on habitat preservation to the dynamism associated with more intact fire regimes, as has occurred in the southern US. Such a shift may result in short- and long-term reductions in total amounts of SO nesting and roosting habitat but will accrue longer term benefits to the remaining habitat in the face of severe wildfires and climate change.

Fire management and ESA habitat requirements are powerful drivers of forest management. We recognize that the habitat needs of the RCW and SO are not strictly analogous (WebTable 1), but in a large portion of the range of both birds, their habitats are adapted to relatively frequent, low- to moderate-intensity fire regimes. A policy of active management with fire and restoration-based thinning has benefited the RCW and overall ecosystem resilience in the southern US; management of the SO in the western US could also benefit from adopting those actions. By providing late-seral habitats in patch sizes and locations where they would historically occur, and by actively restoring fire regimes over large landscapes with successional conditions to support them, it is likely that more SOs, and more of their habitat, would be conserved over the long term. Importantly, the ESA requires that conservation planning be based on the best available scientific data, but in the case of the SO, recent science on SO vulnerability to wildfire has not translated to management. Implementing conservation plans informed by science would entail revising policies, establishing public–private partnerships, and designing endangered species management plans that complement other forest objectives that have enabled RCW recovery. The co-benefits of improved conservation habitat for multiple species, reduced fire severity, and timber revenues have resulted from decades of experience and an active management culture to include more, not less, fire and restoration thinning. Finding ways to

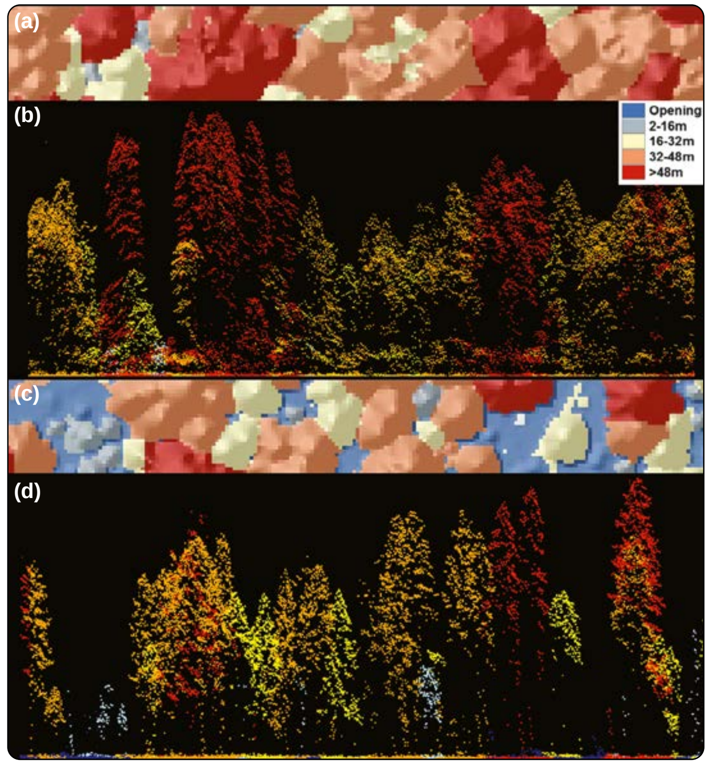

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Figure 6. LiDAR (light detection and ranging) transect images in two CSO nesting areas, showing (a and c) top-down and (b and d) profile views. LiDAR returns are classified into tree approximate objects (TAOs) and in all four images are color-coded by height strata (ranges in meters). The top pair of images from Eldorado National Forest (a and b) show a common nesting condition of fire-suppressed, dense forest with 75% canopy cover. The bottom pair of images (c and d), from a fire-restored location in Sequoia and Kings Canyon National Parks, show that an owl nesting area can have more open forest conditions with 40% canopy cover, if large trees are present.

accelerate these efforts while ensuring SO conservation is a complex task, but such integration would likely improve long-term conservation of the SO and ultimately improve western US forest resilience.

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References

Bukenhofer GA, Neal JC, and Montague WG. 1994. Renewal and recovery: shortleaf pine/bluestem grass ecosystem and red-cockaded woodpeckers. *J Arkansas Acad Sci* **48**: 243–45.

Conner RN, Rudolph DC, and Walters JR. 2001. The red-cockaded woodpecker: surviving in a fire-maintained ecosystem. Austin, TX: University of Texas Press.

Cram DS, Masters RE, Guthery FS, *et al*. 2002. Northern bobwhite population and habitat response to pine–grassland restoration. *J Wildlife Manage* **66**: 1031–39.

Davis RJ, Hollen B, Hobson J, *et al*. 2016. Northwest Forest Plan – the first 20 years (1994–2013): status and trends of northern spotted owl habitats. Portland, OR: US Department of Agriculture Forest Service, Pacific Northwest Research Station. General Technical Report PNW-GTR-929.

Dugger KM, Forsman ED, Franklin AB, *et al*. 2016. The effects of habitat, climate, and barred owls on long-term demography of northern spotted owls. *Condor* **118**: 57–116.

Gaines WL, Harrod RJ, Dickinson J, *et al*. 2010. Integration of northern spotted owl habitat and fuels treatments in the eastern Cascades, Washington, USA. *Forest Ecol Manag* **260**: 2045–52.

Ganey JL, Wan HY, Cushman SA, and Vojta CD. 2017. Conflicting perspectives on spotted owls, wildfire, and forest restoration. *Fire Ecol* **13**: 146–65.

Guldin JM. 2007. Restoration and management of shortleaf pine in pure and mixed stands – science, empirical observation, and the wishful application of generalities. In: Kabrick JM, Dey DC, and Gwaze D (Eds). Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Newtown Square, PA: US Department of Agriculture Forest Service, Northern Research Station. General Technical Report NRS-P-15.

Gutiérrez RJ, Manley PN, and Stine PA (Eds). 2017. The California spotted owl: current state of knowledge. Albany, CA: US Department of Agriculture Forest Service, Pacific Southwest Research Station. General Technical Report PSW-GTR-254.

Hagmann K, Johnson DL, and Johnson KN. 2017. Historical and current forest conditions in the range of the northern spotted owl in south central Oregon, USA. *Forest Ecol Manag* **389**: 374–85.

Hedrick LD, Bukenhofer GA, Montague WG, *et al*. 2007. Shortleaf-pine restoration in the Ouachita National Forest. In: Kabrick JM, Dey DC, and Gwaze D (Eds). Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Newtown Square, PA: US Department of Agriculture Forest Service, Northern Research Station. General Technical Report NRS-P-15.

Hessburg PF, Agee JK, and Franklin JF. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. *Forest Ecol Manag* **211**: 117–39.

Hessburg PF, Churchill DJ, Larson AJ, *et al*. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. *Landscape Ecol* **30**: 1805–35.

Hiers JK, Jackson ST, Hobbs RJ, *et al*. 2016. The precision problem in conservation and restoration. *Trends Ecol Evol* **31**: 820–30.

Hiers JK, Walters JR, Mitchell RJ, *et al*. 2014. Ecological value of retaining pyrophytic oaks in longleaf pine ecosystems. *J Wildlife Manage* **78**: 383–93.

Hoagland SJ, Beier P, and Lee D. 2018. Using MODIS NDVI phenoclasses and phenoclusters to characterize wildlife habitat: Mexican spotted owl as a case study. *Forest Ecol Manag* **412**: 80–93.

Jones GM, Gutiérrez RJ, Tempel DJ, *et al*. 2016. Megafires: an emerging threat to old-forest species. *Front Ecol Environ* **14**: 300–06.

Jones GM, Keane JJ, Gutiérrez RJ, and Peery MZ. 2018. Declining old forest species as a legacy of large trees lost. *Divers Distrib* **24**: 341–51.

Keane JJ. 2017. Threats to the viability of California spotted owls. In: Gutiérrez RJ, Manley PN, and Stine PA (Eds). The California spotted owl: current state of knowledge. Albany, CA: US Department of Agriculture Forest Service, Pacific Southwest Research Station. General Technical Report PSW-GTR-254.

Kobziar LN, Godwin D, Taylor L, and Watts AC. 2015. Perspectives on trends, effectiveness, and impediments to prescribed burning in the southern US. *Forests* **6**: 561–80.

Lesmeister DB, Davis RJ, Singletown PH, and Wiens JD. 2018. Northern spotted owl habitat and populations. In: Spies TA, Stine PA, Gravenmier R, *et al.* (Eds). Science synthesis to inform land management within the Northwest Forest Plan Area, vol 1. Portland, OR: US Department of Agriculture Forest Service, Pacific Northwest Research Station. General Technical Report PNW-GTR-966.

Loehman R, Flatley W, Holsinger L, and Thode A. 2018. Can land management buffer impacts of climate changes and altered fire regimes on ecosystems of the southwestern United States? *Forests* **9**: 192.

Margolis EQ and Balmat J. 2009. Fire history and fire–climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA. *Forest Ecol Manag* **258**: 2416–30.

Masters RE. 2007. The importance of shortleaf pine for wildlife and diversity in mixed oak–pine forests and in pine–grassland woodlands. In: Kabrick JM, Dey DC, and Gwaze D (Eds). Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium. Newtown Square, PA: US Department of Agriculture Forest Service, Northern Research Station. General Technical Report NRS-P-15.

Masters RE, Lochmiller RL, and Engle DM. 1993. Effects of timber harvest and prescribed fire on white-tailed deer forage production. *Wildlife Soc B* **21**: 401–11.

Masters RE, Skeen JE, and Whitehead J. 1995. Preliminary fire history of McCurtain County Wilderness Area and implications for red-cockaded woodpecker management. In: Kulhavy DL, Hooper RG, and Costa R (Eds). Red-cockaded woodpecker: species recovery, ecology and management. Nacogdoches, TX: Stephen F Austin University.

Masters RE, Lochmiller RL, McMurry ST, and Bukenhofer GA. 1998. Small mammal response to pine–grassland restoration for red-cockaded woodpeckers. *Wildlife Soc B* **26**: 148–58.

Masters RE, Wilson CW, Cram DS, *et al*. 2002. Influence of ecosystem restoration for red-cockaded woodpeckers on breeding bird and small mammal communities. In: Ford WM, Russell KR, and Moorman CE (Eds). The role of fire in non-game wildlife management and community restoration: traditional uses and new directions. Proceedings of a special workshop. Annual Meeting of the Wildlife Society, Nashville, TN: US Department of Agriculture Forest Service, Northeast Research Station. General Technical Report NE-288.

McKellar AE, Kesler DC, and Walters JR. 2016. Resource selection reflects fitness associations for an endangered bird in restored habitat. *Anim Conserv* **19**: 131–38.

McKellar AE, Kesler DC, Mitchell RJ, *et al*. 2014. Geographic variation in fitness and foraging habitat quality in an endangered bird. *Biol Conserv* **175**: 52–64.

Melvin M. 2015. 2015 National prescribed fire use survey report. Washington, DC: Coalition of Prescribed Fire Councils Inc. Technical Report 02-15.

Miller JD, Safford HD, Crimmins M, and Thode AE. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. *Ecosystems* **12**: 16–32.

North MP, Kane JT, Kane VR, *et al*. 2017. Cover of tall trees best predicts California spotted owl habitat. *Forest Ecol Manag* **405**: 166–78.

Quinn-Davidson LN and Varner JM. 2011. Impediments to prescribed fire across agency, landscape and manager: an example from northern California. *Int J Wildland Fire* **21**: 210–18.

Roberts SL, van Wagtendonk JW, Miles AK, and Kelt DA. 2011. Effects of fire on spotted owl site occupancy in a late-successional forest. *Biol Conserv* **144**: 610–19.

Roccaforte JP, Fulé PZ, Chancellor WW, and Laughlin DC. 2012. Woody debris and tree regeneration dynamics following severe wildfires in Arizona ponderosa pine forests. *Can J Forest Res* **42**: 593–604.

Rockweit JT, Franklin AB, and Carlson PC. 2017. Differential impacts of wildfire on the population dynamics of an old-forest species. *Ecology* **98**: 1574–82.

Ryan KC, Knapp EE, and Varner JM. 2013. Prescribed fire in North American forests and woodlands: history, current practice, and challenges. *Front Ecol Environ* **11**: e15–24.

Safford H and Stevens J. 2017. Natural range of variation for yellow pine and mixed conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. Albany, CA: US Department of Agriculture Forest Service, Pacific Southwest Research Station. General Technical Report PSW-GTR-256.

Schultz C, Huber-Stearns H, McCaffrey S, *et al*. 2018. Prescribed fire policy barriers and opportunities: a diversity of challenges and strategies across the West. Fort Collins, CO: Colorado State University. Public Lands Policy Group Practitioner Paper 2.

Singleton PH, Lehmkuhl JF, Gaines WL, and Graham SA. 2010. Barred owl space use and habitat selection in the eastern Cascades, Washington. *J Wildlife Manage* **74**: 285–94.

Sparks JC, Masters RE, Engle DM, *et al*. 1998. Effects of late growing-season and late dormant season prescribed fire on herbaceous vegetation in restored pine-grassland communities. *J Veg Sci* **9**: 133–42.

Stan AB, Fulé PZ, Ireland K, *et al*. 2014. Modern fire regime resembles historical fire regime in a ponderosa pine forest on Native American lands. *Int J Wildland Fire* **23**: 686–97.

Stephens SL, Collins BM, Biber E, and Fulé PZ. 2016b. US federal fire and forest policy: emphasizing resilience in dry forests. *Ecosphere* **7**: e01584.

Stephens SL, Collins BM, Fettig CJ, *et al*. 2018. Drought, tree mortality and wildfire in forests adapted to frequent fire. *BioScience* **68**: 77–88.

Stephens SL, Miller JD, Collins BM, *et al*. 2016a. Wildfire impacts on California spotted owl nesting habitat in the Sierra Nevada. *Ecosphere* **7**: e01478.

Stevens JT, Collins BM, Miller JD, *et al*. 2017. Changing spatial patterns of stand-replacing fire in California mixed-conifer forests. *Forest Ecol Manag* **406**: 28–36.

Swetnam TW and Baisan CH. 2003. Tree-ring reconstructions of fire and climate history in the Sierra Nevada and southwestern United States. In: Veblen TT, Baker WL, Montenegro G, *et al*. (Eds). Fire and climate change in temperate ecosystems of the western Americas. New York, NY: Springer Books.

Trainor AM, Walters JR, Urban DL, *et al*. 2013. Evaluating the effectiveness of a Safe Harbor Program for connecting wildlife populations. *Anim Conserv* **16**: 610–20.

USFWS (US Fish and Wildlife Service). 2003. Recovery plan for the red-cockaded woodpecker (*Picoides borealis*). Atlanta, GA: USFWS.

USFWS (US Fish and Wildlife Service). 2012. Mexican spotted owl recovery plan, first revision (*Strix occidentalis lucida*). Albuquerque, NM: USFWS.

Wan HY, Cushman SA, Ganey JL, *et al*. 2018. Managing for emerging threats to spotted owls. *J Wildlife Manage* **82**: 682–97.

Wiens JD, Anthony RG, and Forsman ED. 2014. Competitive interactions and resource partitioning between northern spotted owls and barred owls in western Oregon. *Wildlife Monogr* **185**: 1–50.

Wilson CW, Masters RE, and Bukenhofer GE. 1995. Breeding bird response to pine–grassland community restoration for red-cockaded woodpeckers. *J Wildlife Manage* **59**: 56–67.

Supporting Information

Additional, web-only material may be found in the online version of this article at http://onlinelibrary.wiley.com/doi/10.1002/fee.2076/suppinfo

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Wood buildings as a climate solution

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ABSTRACT

We conducted a systematic literature search and meta-analysis of studies with side-by-side life cycle analysis comparisons of mid-rise buildings using mass timber and conventional, concrete and steel, building materials. Based on 18 comparisons across four continents, we found that substituting conventional building materials for mass timber reduces construction phase emissions by 69%, an average reduction of 216 $kgCO_2e/m^2$ of floor area. Studies included in our analysis were unanimous in showing emissions reductions when building with mass timber compared to conventional materials. Scaling-up low-carbon construction, assuming mass timber is substituted for conventional building materials in half of expected new urban construction, could provide as much as 9% of global emissions reduction needed to meet 2030 targets for keeping global warming below 1.5 °C. Realizing the climate mitigation potential of mass timber building could be accelerated by policy and private investment. Policy actions such as changing building codes, including mass timber in carbon offset crediting programs and setting building-sector-specific emissions reduction goals will remove barriers to and incentivize the adoption of mass timber. Private capital, as debt or equity investment, is poised to play a crucial role in financing mass timber building.

1. Introduction

The buildings sector is a major source of global greenhouse gas emissions. Construction related CO_2 emissions equaled 5.7 billion tons, accounting for 23% of the emissions resulting from global economic activities in 2009 (Huang et al., 2018). More recently, in 2018, building construction and operations accounted for the largest share of both global final energy use (36%) and energy-related CO_2 emissions (39%) (UNEP, 2019). The building sector takes on even greater significance when considering the approaching wave of 2.3 billion new urban residents (United Nations, 2019) who are expected to drive a doubling the global building stock by 2060 (World Green Building Council, 2019). Over the next decade, through 2030, the greatest increases in housing and infrastructure to accommodate growing urban populations are expected in China, North America and Europe (UNEP, 2017).

Despite its significance, most national plans for reducing greenhouse gas emissions (Nationally Determined Contributions) only mention buildings, falling short of setting sector-specific targets (UNEP, 2019). Lacking national plans, many cities, businesses, organizations, states and regions have developed their own building sector emissions targets. For example, with the aim of limiting global warming to under 2 °C, the World Green Business Council's Net Zero Carbon Buildings Commitment "challenges companies, cities, states and regions to reach net-zero operating emissions in their portfolios by 2030, and to advocate for all buildings to be net-zero carbon by 2050 (The Net Zero Carbon Buildings Commitment, n.d)."

Changes in all aspects of building operations—powering lighting, heating, cooling, etc.— will be required to achieve net-zero carbon buildings. This will mean increasing both energy efficiency and the generation and procurement of renewable energy (Wiik et al., 2018). As buildings become more energy efficient, embodied energy—that is, the energy used in the process of building material production, transportation, construction, maintenance, and demolition/re-use—assumes greater relative importance (Cabeza et al., 2014; Huang et al., 2018; Lolli et al., 2019). A building's embodied energy can be reduced by replacing carbon-intensive materials, like concrete and steel, with wood (Gustavsson et al., 2017; Sathre & O'Connor, 2010).

Until recently, wood was primarily used in the construction of single family or small multi-unit wood framed buildings (Brandner et al., 2016), limiting its potential in urban areas where new, low-carbon construction should prioritize larger mid-rise buildings (Churkina et al., 2020). However, the development of mass timber technology in recent decades

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has paved the way for constructing mid- and high-rise buildings with wood to meet the built environment demands of a rapidly growing global urban population (Brandner et al., 2016; Harte, 2017).

Mass timber is a category of wood-based framing characterized by the use of large solid wood panels for wall, floor and roof construction (Harte, 2017). In the construction of multifamily residential and commercial multistory buildings, cross-laminated timber (CLT) is the most widely used mass timber product (Brandner et al., 2016; Harte, 2017). Unlike concrete and steel, which emit CO_2 when produced, trees used to make mass timber products naturally absorb and store CO_2 as they grow. In sustainably managed forests, new trees are regenerated to replace trees that are harvested so that there is no net loss of forest carbon (Gustavsson et al., 2017). Increasing the amount of wood used in buildings, as a substitute for more carbon intensive materials, has the potential to decrease total emissions from the building sector (Hill, 2019).

At the global scale, estimates of the potential for wood buildings to serve as a climate solution range widely. Oliver et al. (2014) estimated that a 14%–31% reduction in global CO_2 emissions—or about 4.7–10.3 $GtCO_2$ $year^{-1}$ based on 2019 emissions (IEA, 2020)—could be realized by substituting wood for concrete and steel in building and bridge construction. The range reported by Oliver et al. was largely dependent on the efficiency of wood use inherent in different products (i.e., wood products where a larger proportion of harvested trees ends up in final product and bi-products are used to produce energy resulted in greater CO_2 emission reductions). The greatest reductions were achieved in flooring applications with engineered wood I-joists covered with plywood that was dried using wood energy, but CLT was not considered in the study (Oliver et al., 2014).

In a more recent study on the potential for wood buildings to serve as a climate solution, Churkina et al. (2020) found that construction of wood building for new urban dwellers could provide long-term carbon storage of 0.01–0.68 GtC/year in the buildings themselves. The variation in estimates demonstrates the inherent complexity of these studies and emphasizes the importance of consistent and systematic methods so that results can be compared (Chastas et al., 2018). The life-cycle assessment (life-cycle analysis or LCA) approach is widely viewed as best-suited for analyzing cradle-to-grave environmental impacts of buildings and has gained global support because the methods are relatively standardized (Lolli et al., 2019; Rønning and Brekke, 2014).

This study compares structure-level estimates of greenhouse gas emissions from building with wood to emissions from conventional materials like reinforced concrete. We systematically searched the literature to find side-by-side LCA comparisons with the two types of building materials and conducted statistical analysis of the pooled results to derive more generalizable conclusions about the climate mitigation potential of wood buildings (Binkley and Menyailo, 2005; Shelby and Vaske, 2008). Results from 18 comparisons were compiled and the differences in emissions between wood and conventional building materials during construction phase were analyzed using a linear mixed model. We estimated that, on average, building with wood can reduce greenhouse gas emissions by 216 $kgCO_2e/m^2$ of floor area, a 69% reduction compared to the estimated average construction phase emissions when conventional materials are used (95% confidence interval bounded by 146 and 287 $kgCO_2e/m^2$). By extension, we estimate that substituting mass timber for conventional building materials in half of expected new urban construction could account for 9% of the annual global emissions reductions needed between 2020 and 2030 in order to keep global temperature increases under 1.5 °C.

The remainder of the paper is organized as follows. In section two, we describe the methods for the systematic literature review, statistical analysis, and scaling-up low-carbon construction. In section three, we describe our results. Section four includes a discussion of the results and potential implications in the context of global climate as well as highlighting potential pathways, through policy instruments and private investment for realizing emissions reductions from wood-based construction. Finally, we conclude by summarizing the key points of the paper.

2. Methods

2.1. Systematic literature review

We conducted a systematic search of the literature for all the following combinations of terms: “mass timber” AND “life cycle analysis”; “mass timber” AND “life cycle assessment”; “mass timber” AND “LCA”; “cross laminated timber” AND “life cycle analysis”; “cross laminated timber” AND “life cycle assessment”; “cross laminated timber” AND “LCA”; “CLT” AND “life cycle analysis”; “CLT” AND “life cycle assessment”; “CLT” AND “LCA”. We used alternative forms, e.g. “CLT” and “cross laminated timber”, in an attempt to capture all relevant articles in the search. We used both “life cycle analysis” and “life cycle assessment” as these terms have been used to describe very similar analysis or used interchangeably. To ensure an adequate survey of literature we conducted the search using two different databases, Web of Science and Google Scholar. The search was carried out in October of 2019 and terms could appear anywhere in the text. Web of science yielded 55 returns and Google Scholar an additional 305 returns for a total of 360. We reviewed all items to isolate peer reviewed studies using LCA methods with side-by-side comparison of mass timber buildings to conventional building materials in structures with three or more stories. While differences in greenhouse gas emissions between mass timber and conventional materials has been documented during operations and end-of-life phases of buildings (Durlinger et al., 2013; Guo et al., 2017; Liu et al., 2016), we focused on studies for which construction phase data could be isolated. We limited the study to construction phase comparisons because: (1) it is most consistently reported in the literature; (2) it is the phase most affected by choice of building material; (3) it is the most relevant phase to achieve near and mid-term emissions goals; and (4) it is not subject to speculation about patterns in future consumption, building life-span, or material disposition (e.g., landfill, recycle, burn, burn for energy).

In the present study, we consider the construction phase to include emissions from building material production, all supply chain related transportation and actual erecting of the building. This definition of construction phase corresponds to modules A1-A5 as defined in the European standard EN 15804 (EN, 2012). However, the European standard is not utilized universally in the literature or in all parts of the world. The construction phase is also sometimes broken down into the “production” (of raw materials) and “construction” (building erection) phases (Tettey et al., 2019) and in other cases they are collectively referred to as the manufacturing phase (Ramesh et al., 2010). In other cases, the term “cradle-to-gate” is used to describe the complete production phase as described above without reference to the European standard modules (Sandanayake et al., 2018; Padilla-Rivera et al., 2018) while others exclude emissions from transportation to the building site and on-site building erection from “cradle-to-gate,” with (Skullestad et al., 2016) or without (Robertson et al., 2012; Zeitz et al., 2019) explicit reference to modules A1-A3. The inconsistent use of standards and terminology in the literature created some difficulty in setting common system boundaries to be included in our analysis. Ultimately, we included data from studies where we could isolate emissions on a $kgCO_2e/m^2$ of floor area basis with or without the inclusion of transportation to the building site and construction activities (Fig. 1). We included studies that did not consider transportation of building materials to the construction site or construction actives (A4&A5) because the relatively small proportion of onsite emissions associated with building erection as a share of total construction phase emissions. Further, there is evidence that the difference in transportation emissions between timber and non-timber building materials was expected to be minor. For example, Cole (1998) found that emissions during actual building construction ranged from 0.8 to 2.5 $kgCO_2eq/m^2$ for wood buildings using glulam structural frames overlaid with wood decking and 5 to 20 $kgCO_2eq/m^2$ for cast-in-place reinforced concrete walls. Dodoo (2019) estimated building construction to account for 5–6 $kgCO_2eq/m^2$ depending on the materials. Further, emissions from

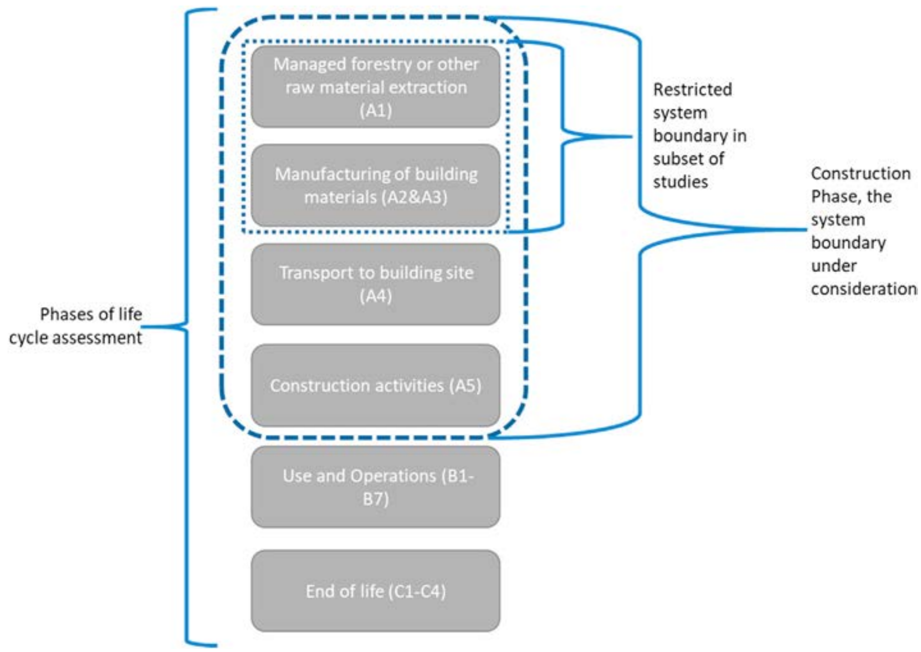

Fig. 1. Phases of a life cycle assessment and corresponding modules following EN 15978:2011 (2011). We analyzed studies reporting construction phase (within the system boundary) comparisons between wood and conventional materials in mid-rise buildings. A subset of the studies (Robertson et al., 2012; Skullestad et al., 2016; Zeitz et al., 2019) did not include emissions from transporting building materials to the construction site or emissions associated with on-site construction activities.

transporting materials to the construction site and erecting the building were often not reported separately from other construction phase emissions in studies that included them. The potential impact of this inconsistency was mitigated by our analytical approach of analyzing differences between within-in study comparisons of different materials rather than differences between the average emission for different building materials.

In total, 11 peer-reviewed publications detailing 18 comparisons met the criteria above. Construction phase emissions in units of $kgCO_2eq/m^2$ for the two types of building materials were compiled from the 18 comparisons along with supporting information like the total floor area, number of stories, type of building and region where the study was conducted (Table 1). Buildings used in the analysis ranged from 3 to 21 stories and from 1,140–29,100 m^2 of floor area. Building types included commercial, mixed-use, multi-unit residential and parking garages. The sample included one building from Australia, 6 from China, 7 from Europe, and 4 from North America (Table 1).

2.2. Statistical analysis

We specify a linear mixed model (LMM) to estimate the average difference in construction phase emissions between buildings constructed using wood and conventional building materials, β_1 below. The corresponding 95% confidence interval around the estimated coefficient β_1 provides a range of plausible values for the parameter. The LMM was estimated from 18 sample pairs of buildings drawn from the literature and specified as follows:

$$Y_t = \beta_{0+} + \beta_1 I.tim_t + (b_r)_t + c_t + \varepsilon_t$$

where

Y_t is the estimated CO_2/m^2 emissions of the *t*th building from the *r*th study; r = 1, …,11; t = 1, …,36
β_0 is the mean emissions of constructing buildings with conventional materials
β_1 is the incremental effect on emissions of constructing buildings with mass timber.
I.tim is a dummy variable equal to 1 for buildings constructed with timber and 0 otherwise
b_r is the random effect of the *r*th study
c_t is the random effect of the building size of the *t*th building
ε_t is the random error associated with the estimate of emissions of the *t*th building.

The residuals are assumed to be independent, normally distributed and with constant variance. Visual inspection of residual plots indicated that the assumptions of the LMM were adequately met except for normality. Despite the exception of the normality assumption, we opted to use the model because linear models are robust against assumptions of normality.

Typically, meta-analyses assign weights to results from different studies based on sample size and variance (Shelby and Vaske, 2008). Because each building comparison in our analysis is a single case study with no variance, there is no need to assign weights. While all studies employ a basic LCA framework, the assumptions underlying each study vary. Furthermore, even though each study is focused on the construction phase of building LCA there are still methodological differences in delineating system boundaries driven by researcher choices and variation in LCA protocol standards. The specific location of each study and the size of building considered could also impact emissions from transportation of raw materials and construction efficiencies. Assigning study and building size as random effects in an LMM allow us to account for some of the variability across studies. All statistical analyses were carried out using R Statistical Software (Team, 2017).

2.3. Scaling-up low-carbon construction

To illustrate the potential of mass timber as a climate solution, we put our results in the context of 2030 global emission targets to limit global warming to 1.5 °C. We begin by assuming 50% of new urban construction substituted mass timber for conventional building materials from 2020 to 2030. We select 50% of new urban construction as a benchmark not because we believe it is the most likely scenario but because we believe it is a feasible upper bound on the potential for mass timber as a substitute for conventional materials in the next ten years. Further, Churkina et al.

Table 1
Detailed list of studies included in analysis.

Paper	Region	Comparison(s)	Building size (m^2)	Number of floors
Dodoo (2019)	Europe	Reinforced concrete residential building with CLT redesign.	1140	4
Guo et al. (2017)	China	Four residential reinforced concrete buildings to CLT re-designs.	2636 4564 7134 10,990◊	4 7 11 17
Liu et al., 2016 [b]	China	Concrete residential building to CLT re-design in two different parts of China.	2799	7
Padilla-Rivera et al., 2018 [bd]	North America	CLT residential building to comparable concrete frame.	1512	4
Peñaloza et al., 2016 [b]	Europe	Simulated comparison of CLT residential building to concrete.	6078	12
Pierobon et al. (2019)	North America	Concrete office building to hybrid CLT re-design.	10,702	11
Robertson et al. (2012)	North America	Concrete-frame office building to CLT redesign.	14,233	5
Sandanayake et al. (2018)	Australia	Reinforced concrete commercial building to comparable CLT mixed use building.	Concrete was 17,104 CLT was 11,960	15 11
Skullestad et al. (2016) [c]	Europe	Four reinforced concrete buildings to CLT redesigns.	2613 6076 10,542 11,823	3 7 12 21
Tettey et al. (2019) [a]	Europe	Prefabricated concrete frame apartment building to CLT redesign.	1686	6
Zeitz et al. (2019)	North America	Post-tension concrete parking garage to comparable CLT parking garage.	Concrete was 29,100 CLT was 19,900	4 4

◊Details on building size from personal communication with the corresponding author.
┘Study included operational and end-of-life phases also.
[a] Study used both coal and fossil gas as a basis for substitution when calculating emissions, we used fossil gas number which were more conservative.
[b] Emissions numbers for construction phase were extrapolated from figures.
[c] Study included reference, best-case and worst-case scenarios, we used emissions from the "reference scenario" for analysis.
[d] Authors derived emissions for concrete comparison from another study.

(2020) suggest even higher levels of substitution may be feasible when considering slightly longer time-frame (30-years) and could be met sustainably from unexploited forest harvest potential. We estimate annual demand for new buildings over the next decade by multiplying the projected annual increase in global urban populations by the mean floor area per/capita of cities (30 m^2) (Churkina et al., 2020; United Nations, 2019). This assumes that the building space requirements for new urban dwellers will remain constant, which is reasonable given the relatively short projection period of 10 years. We then multiply 50% of annual demand by our estimate of annual emissions reductions for each year of the next decade to calculate total global annual emissions reductions from the substitution of mass timber in half of new urban construction. Finally, we compare annual emissions reductions from substituting conventional building materials with wood to the total annual emissions reductions needed to meet global 2030 emissions targets for limiting temperature rise to 1.5 °C, i.e. annual emissions decrease of 7.6% from 2020 to 2030 (see Supplementary Materials) (UNEP, 2019).

3. Results

The average difference in construction phase emissions between wood and conventional building materials was (−)216 $kgCO_2e/m^2$ of floor area ($p < 0.01$) and the 95% confidence interval was (−)146 to (−) 287 $kgCO_2e/m^2$ (Fig. 2). In other words, mass timber buildings emitted on average 216 kg CO_2e/m^2 less in construction phase than their counter parts constructed using conventional building materials. The estimate of the intercept, i.e. the average emissions using conventional materials after accounting for the random effects of study and building size, was 312 $kgCO_2e/m^2$ of floor area ($p < 0.01$). Thus, substituting conventional building materials with timber represents a 69% reduction in CO_2e emissions in the construction phase. The conditional R^2, an indicator of the proportion of variance in the response explained by the both fixed and random effects included in the model, was 0.78 (Lefcheck et al., 2016).

These results are consistent with Cadorel and Crawford's (2018) review of CLT LCAs which found substituting wood for conventional building materials results in reduces greenhouse gas emissions. In a review of global emissions from the construction sector, Huang et al. (2018) also conclude that substituting conventional building materials for those with low embodied carbon, like wood, is a key strategy to reducing global greenhouse gas emissions. Based on this estimate substituting wood for conventional building materials in 50% of new construction globally could meet 9% of annual emission reductions needed by 2030 to prevent temperatures from rising more than 1.5 °C. This result is similar in scale to estimates of potential stored carbon in buildings if 50% of new urban construction used substituted mass timber for conventional buildings materials (Churkina et al., 2020).

4. Discussion

All studies in our analysis found that mass timber buildings produce less emissions compared to reinforced concrete buildings in the construction phase. Some comparisons in our analysis included all building LCA phases—construction, operations (emissions during the operating

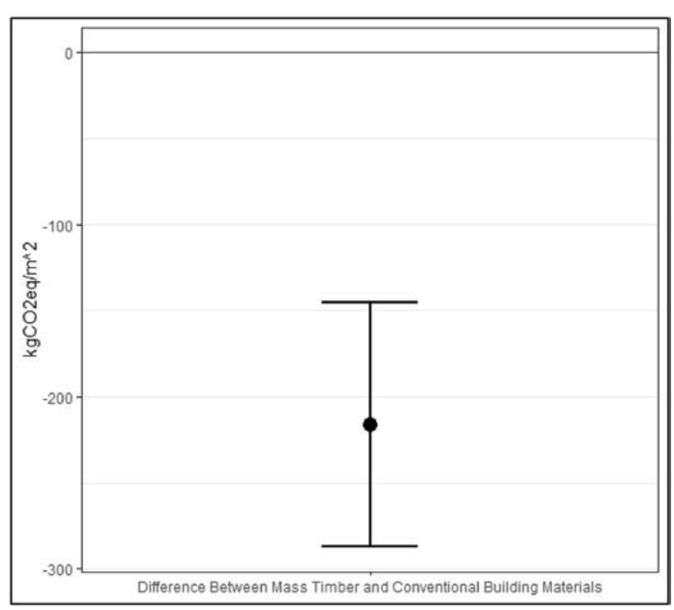

Fig. 2. Average difference in global warming potential during construction phase from building with mass timber compared to conventional materials (mass timber emissions – conventional building material emissions). The average difference equals a 216 $kgCO_2/m^2$ reduction, 69% of the estimated average construction phase emissions using conventional materials after accounting for random effects of study and building size which equaled 312 $kgCO_2/m^2$ (the zero-reference line above).

life the building, typically 50-years), and end-of life (emissions from demolition and material disposition)—and each concluded total lifecycle emissions were lower when building with mass timber compared to conventional materials. Further, all studies found that mass timber has the largest proportional benefit over conventional building materials in the construction phase (Tettey et al., 2019; Dodoo, 2019; Guo et al., 2017; Liu et al., 2016; Peñaloza et al., 2016).

Although the operational phase typically dominates total lifetime energetic and climactic impacts of buildings, these impacts are over many decades and as energy efficiency improves, construction becomes proportionally more significance (Cabeza et al., 2014; Thormark, 2002). In a meta-analysis, Ramesh et al. (2010) noted the construction phase accounts for 10–20% of the buildings' total lifecycle energy use. However, for a low-energy building Thormark (2002) showed that the share of the total lifecycle energy used in the construction phase may be as high as 45% of total lifecycle energy use. Further, the operation phase is typically based on a 50-year or longer building use cycle, a timescale which becomes less relevant in light of potentially catastrophic consequences from global warming if emissions are not curbed in the next 10 years (Global Warming of 1.5 °C —, n.d.). On the other hand, our scaled-up results suggest that substituting conventional building materials with mass timber could make substantial contribution toward achieving the near-term emission reductions necessary in order to prevent global warming from exceeding 1.5 °C.

Methods, including LCA boundary selection, varied somewhat between some studies highlighting the need in future research to further systematize protocol to facilitate easy comparisons. A great amount of variability can arise due to assumptions about carbon stored in wood building materials and whether it is treated as a credit or ignored based on the assumption that any stored carbon will be emitted at the end of life (Hill, 2019). To date, there is no consensus on methods for accounting of carbon stored in wood (Tellnes et al., 2017).

Reflecting this lack of consensus, the studies included in our analysis use a range of approaches to address stored carbon in wood building materials. Dodoo (2019), Padilla-Rivera et al. (2018), Pierobon et al. (2019), Sandanayake et al. (2018), (Ayikoe Tettey et al., 2019), and Zeitz et al. (2019) assumed no net biogenic carbon stored or released, i.e. bioproducts used in the construction phase were treated as carbon neutral and no credit was given for long term storage of carbon in building materials. In contrast, several studies assumed net carbon storage in wood building materials. Guo et al. (2017), Liu et al. (2016), and Robertson et al. (2012) all assume net carbon storage in wood building materials equal to 753–800 kg CO_2e/m^3. Extending this analysis, Peñaloza et al. (2016) and Skullestad et al. (2016) examined multiple scenarios where net carbon storage deviated from a carbon neutral baseline. For example, Peñaloza et al. (2016) included LCA scenarios where forests were considered in the LCA system boundary and they accounted for the timing of biogenic carbon sequestration by trees, duration of carbon storage in wood products and emissions. Skullestad et al. (2016) included approaches suggested by Guest et al. (2013) for incorporating biogenic carbon emissions and storage into LCA which resulted in the CLT building having negative emissions (i.e., they were a net sink) when biofuel from wood production substituted natural gas. In our analysis, if multiple scenarios were presented we used the data reported for scenarios where wood building materials were assumed to be carbon neutral, which were always the most conservative.

The treatment of carbon stored in long-lasting wood products like mass timber in LCA is not trivial. Churkina et al. (2020) estimated that potential for timber buildings to store as much as 0.68 Gt of carbon per year which would create a substantial carbon sink. However, carbon being stored in buildings is being transferred from forests, which also act as a carbon sink. Whether transferring carbon from the forest to the built environment will reduce atmospheric greenhouse gas concentrations depends on how the forest is managed and how long the building is in service (Guest et al., 2013; Gustavsson et al., 2017). For example, if the wood used to make mass timber is taken from a forest that is replaced with agriculture after harvest (deforestation) then the land-use change related emissions and emissions from manufacturing the timber products will likely more than offset the long-term storage benefits. The use of mass timber that is certified sustainable by a reputable certifying body with a chain of custody standard, such as Forest Stewardship Council or Sustainable Forestry Initiative, virtually eliminates that possibility.

The relative lifespan of a mass timber building and the length of the forest rotation (number of years between planting and harvesting) will also influence the potential climate benefits from building with wood. If the lifespan of the building is shorter than the average rotation in the forest and the carbon in the wood is re-emitted to the atmosphere when the building is demolished, then total carbon storage would be less than if the forest was not harvested and simply left to grow (Guest et al., 2013; Skullestad et al., 2016). However, if the lifespan of the building is longer than the average rotation in the forest providing the wood, then total carbon storage between the two sources should be treated as a sink (Guest et al., 2013; Skullestad et al., 2016). This implies that increasing the lifespan of mass-timber buildings and simultaneously accelerating the growth of the forests generating the raw material will increase the importance of wood building's carbon storage potential.

When considering long-term carbon storage potential, it is also important to account for the size of the respective carbon pools, in both the forest and built environment. Law et al. (2018) point out that some forest systems have the capacity to store much more carbon than they do under sustainable short-rotation production forestry but Churkina et al. (2020) challenge that the carbon storage density of mass timber buildings is comparable or even larger than some of the most carbon dense forest systems in the world. Another important consideration is the growth rate of the forest pool. Moving carbon from the forest pool to the built environment pool is more likely to have a global warming benefit if the harvested trees are growing slowly and harvesting and regenerating the forest will lead to more rapid carbon accumulation in the forest (Gustavsson et al., 2017). These complicated interactions between the carbon pools in timber buildings and the forest suggest incorporating forests into LCA system boundaries and utilizing dynamic methods that track sequestration and emissions over time could provide important information for decision makers (Peñaloza et al., 2016).

Some authors have suggested that LCA analysis be viewed cautiously. Harmon (2019) conducted a sensitivity analysis around LCA assumptions for mass timber construction and concluded that GHG benefits compared to concrete and steel may be overestimated 2–100 fold. Harmon's (2019) conclusions are based on challenging the assumptions implicit in many LCA substitution studies that carbon displacement value remains constant, the displacement is permanent without leakage and that there is no relationship between building longevity and substitution longevity. However, even if these assumptions do not hold, the over-estimates Harmon projects take 300-years to accumulate and have negligible impact on estimates of construction phase substitution benefits in the first ten years, which is the critical period considered here. Zeitz et al. (2019) also downplay the benefits of mass timber, demonstrating that the composition of concrete and steel had a large impact on analysis and when best practices for both conventional and mass timber materials are used, differences in global warming potential between the two become very small. Zeitz et al.'s results are important but only represent a single case which has been included in our metanalysis. The impacts of building with mass timber can fluctuate regionally and depend on many factors like the availability of materials, transportation, manufacturing and regional energy sources. Thus, when considering potential global impacts, meta-analysis like this study represent pooled knowledge and the highest level of expert confidence (Binkley and Menyailo, 2005).

Co-benefits and trade-offs of building with mass timber should be considered in conjunction with climate impacts. Mass-timber outperformed re-enforced concrete with respect to other LCA environmental indicators including reduced smog potential, reduced ecological toxicity, reduced acidification potential, reduced ozone depletion potential, and improved human health effect potential (Durlinger et al., 2013; Pierobon et al., 2019; Robertson et al., 2012). Mass-timber has comparable constructions costs to conventional building materials but can reduce construction time for mid- and high-rise buildings, which may make it an attractive alternative from an investment perspective (Cazemier, 2017; Mallo and Espinoza, 2016). The aesthetic and acoustic properties of buildings with interior exposed wood surfaces are also desirable (Espinoza et al., 2016; Laguarda Mallo and Espinoza, 2015; Mallo and Espinoza, 2016). Building with wood has desirable seismic properties (Brandner et al., 2016) but concerns still remain regarding perceived durability and fire risks relative to conventional building materials (Kremer and Symmons, 2018). There are also likely trade-offs in other ecosystem services when forests are managed to maximize carbon benefits from wood utilization (Himes et al., 2020).

Finally, we recognize the important role of both policy and private investment in order to scale-up low-carbon construction. First, removing policy barriers to the deployment of mass timber, such as building standards codes and site permitting, would increase industry adoption and familiarity among architects, developers and construction companies in turn (Espinoza et al., 2016; Kremer and Symmons, 2018; Laguarda Mallo and Espinoza, 2015). Second, the introduction of both regulatory and market-based policies could be used to further unlock the climate mitigation potential of mass timber building. Within a regulatory framework, building-sector-specific targets that include long-term carbon storage in wood could be incorporated into national carbon plans (NDCs). In addition, incorporating mass timber into existing offset crediting programs (voluntary or compliance) would immediately incentivize the use of wood building materials.

Alongside policy changes, private capital, as equity or debt investment, may also play a critical role in unlocking the climate mitigation potential of substituting wood for conventional building materials. For example, the development of institutional scale investment opportunities in sustainable, low-carbon buildings could help investors achieve both financial and climate targets along with the portfolio benefits of real estate. On the debt side, green bonds could provide an effective way to finance large-scale wood buildings, similar to green bonds for energy efficiency upgrades in buildings, which totaled $47 billion (USD) in 2017 (IAE and UNEP, 2018). Further, for investors with climate or portfolio decarbonization targets, both equity and debt investment in mass timber building could provide assurances that their investment will contribute to measurable emissions reductions.

5. Concluding remarks

- According to our meta-analysis, substituting conventional building materials with mass timber in mid-sized urban construction reduces construction phase emissions by an average 216 $kgCO_2e/m^2$ of floor area, a 69% reduction. Studies included in our analysis were unanimous in showing emissions reductions when building with mass timber compared to conventional materials. The importance of this reduction increases as buildings become more energy efficient and the need to reduce emissions becomes more eminent.
- Although the LCA methodologies are relatively standardized, variation in approaches taken by researcher still exist which have substantial impact on individual study results. The variation between individual studies can also arise from context specific parameters, like resource availability and transportation distance. By incorporating the results from multiple individual studies in a single analysis we provide a more generalizable estimate of potential global benefits of substituting conventional building materials with mass timber.
- We estimate that substituting conventional building materials with wood in half of new urban construction could provide 9% of global emissions reduction needed to meet 2030 targets for keeping global warming below 1.5 °C. This could be a substantial piece of the emission reduction portfolio needed to meet the ambitious 2030 target without substantial changes to consumption, life-style or policy. Almost a tenth of necessary emissions reductions could come from choosing a different material for expected urban construction that will occur one way or the other.
- Realizing the climate mitigation potential of mass timber building may be supported by both policy and private investment. Allowing for the incorporation of mass timber in building codes, carbon offset crediting programs and building sector emissions reduction goals will remove barriers and incentivize greater adoption of mass timber. Private capital in the form of institutional scale investment or green bonds could play a crucial role in financing mass timber building.

Author contributions

Austin Himes roles: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Visualization; Writing- original draft; Writing- review & editing.

Gwen Busby roles: Conceptualization; Investigation; Project administration; Supervision; Writing- original draft; Writing- review & editing.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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References

Ayikoe Tettey, U.Y., Dodoo, A., Gustavsson, L., 2019. Carbon balances for a low energy apartment building with different structural frame materials. Energy Procedia 158, 4254–4261. https://doi.org/10.1016/j.egypro.2019.01.801.

Binkley, D., Menyailo, O., 2005. Gaining insights on the effects of tree species on soils. In: Tree Species Effects on Soils: Implications for Global Change. Springer, pp. 1–16.

Brandner, R., Flatscher, G., Ringhofer, A., Schickhofer, G., Thiel, A., 2016. Cross laminated timber (CLT): overview and development. Eur. J. Wood Wood Prod. 74 (3), 331–351. https://doi.org/10.1007/s00107-015-0999-5.

Cabeza, L.F., Rincón, L., Vilariño, V., Pérez, G., Castell, A., 2014. Life cycle assessment (LCA) and life cycle energy analysis (LCEA) of buildings and the building sector: a review. Renew. Sustain. Energy Rev. 29, 394–416. https://doi.org/10.1016/j.rser.2013.08.037.

Cadorel, X., Crawford, R., 2018. Life cycle analysis of cross laminated timber in buildings: a review. International Conference of the Architectural Science Association 107–114.

Cazemier, D.S., 2017. Comparing cross laminated timber with concrete and steel: a financial analysis of two buildings in Australia. Modular and Offsite Construction (MOC) Summit Proceedings 1 (1).

Chastas, Panagiotis, Theodosiou, Theodoros, Kontoleon, Karolos, Bikas, Dimitrios, 2018. Normalising and assessing carbon emissions in teh building sector: A review on the embodied CO2 emissions of residential buildings. Build. Environ. 130, 212–216. https://doi.org/10.1016/j.buildenv.2017.12.032.

Churkina, G., Organschi, A., Reyer, C.P.O., Ruff, A., Vinke, K., Liu, Z., Reck, B.K., Graedel, T.E., Schellnhuber, H.J., 2020. Buildings as a global carbon sink. Nat. Sustain. 1–8. https://doi.org/10.1038/s41893-019-0462-4.

Cole, Raymond, 1998. Energy and greenhouse gas emissions associated with the construction of alternative structural systems. Build. Environ. 34 (3), 335–348. https://doi.org/10.1016/S0360-1323(98)00020-1.
Dodoo, A., 2019. Lifecycle impacts of structural frame materials for multi-storey building systems. J. Sustain. Architect. Civ. Eng. 24 (1), 17–28.
Durlinger, B., Crossin, E., Wong, J., 2013. Life Cycle Assessment of a Cross Laminated Timber Building. Forest & Wood Products Australia. Project no PRA282-1112.
EN, 2012. Sustainability of Construction Works-Environmental Product Declarations-Core Rules for the Product Category of Construction Products. EN, 15804.
Espinoza, O., Trujillo, V.R., Mallo, M.F.L., Buehlmann, U., 2016. Cross-laminated timber: status and research needs in Europe. BioResources 11 (1), 281–295.
Global Warming of 1.5 °C — (n.d.), from. https://www.ipcc.ch/sr15/. (Accessed 19 December 2019).
Guest, G., Cherubini, F., Strømman, A.H., 2013. Global warming potential of carbon dioxide emissions from biomass stored in the anthroposphere and used for bioenergy at end of life. J. Ind. Ecol. 17 (1), 20–30. https://doi.org/10.1111/j.1530-9290.2012.00507.x.
Guo, H., Liu, Y., Meng, Y., Huang, H., Sun, C., Shao, Y., 2017. A comparison of the energy saving and carbon reduction performance between reinforced concrete and cross-laminated timber structures in residential buildings in the severe cold region of China. Sustainability 9 (8), 1426. https://doi.org/10.3390/su9081426.
Gustavsson, L., Haus, S., Lundblad, M., Lundström, A., Ortiz, C.A., Sathre, R., Truong, N.L., Wikberg, P.-E., 2017. Climate change effects of forestry and substitution of carbon-intensive materials and fossil fuels. Renew. Sustain. Energy Rev. 67, 612–624. https://doi.org/10.1016/j.rser.2016.09.056.
Harmon, M.E., 2019. Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions. Environ. Res. Lett. 14 (6), 065008 https://doi.org/10.1088/1748-9326/ab1e95.
Harte, A.M., 2017. Mass timber – the emergence of a modern construction material. J. Struct. Integr. Mainten. 2 (3), 121–132. https://doi.org/10.1080/24705314.2017.1354156.
Hill, C.A.S., 2019. The environmental consequences concerning the use of timber in the built environment. Front. Built Environ. 5 https://doi.org/10.3389/fbuil.2019.00129.
Himes, Austin, Puettmann, Klaus, Muraca, Barbara, 2020. Trade-offs between ecosystem services along gradients of tree species diversity and values. Ecosystem Services 44. https://doi.org/10.1016/j.ecoser.2020.101133.
Huang, L., Krigsvoll, G., Johansen, F., Liu, Y., Zhang, X., 2018. Carbon emission of global construction sector. Renew. Sustain. Energy Rev. 81, 1906–1916. https://doi.org/10.1016/j.rser.2017.06.001.
IEA, 2020. Global CO2 Emissions in 2019. IEA, Paris. https://www.iea.org/articles/global-co2-emissions-in-2019.
IAE and UNEP, International Energy Agency and the United Nations Environment Programme, 2018. 2018 Global Status Report: towards a Zero-Emission, Efficient and Resilient Buildings and Construction Sector.
Kremer, P.D., Symmons, M.A., 2018. Perceived barriers to the widespread adoption of Mass Timber Construction: an Australian construction industry case study. Mass Timber Construct. J. 1 (1), 1–8.
Laguarda Mallo, M.F., Espinoza, O., 2015. Awareness, perceptions and willingness to adopt Cross-Laminated Timber by the architecture community in the United States. J. Clean. Prod. 94, 198–210. https://doi.org/10.1016/j.jclepro.2015.01.090.
Law, B.E., Hudiburg, T.W., Berner, L.T., Kent, J.J., Buotte, P.C., Harmon, M.E., 2018. Land use strategies to mitigate climate change in carbon dense temperate forests. Proc. Natl. Acad. Sci. Unit. States Am. 115 (14), 3663–3668. https://doi.org/10.1073/pnas.1720064115.
Lefcheck, J., Byrnes, J., Grace, J., 2016. Package 'piecewiseSEM'. R package version 1 (1).
Liu, Y., Guo, H., Sun, C., Chang, W.-S., 2016. Assessing cross laminated timber (CLT) as an alternative material for mid-rise residential buildings in cold regions in China—a life-cycle assessment approach. Sustainability 8 (10), 1047. https://doi.org/10.3390/su8101047.
Lolli, N., Fufa, S.M., Wiik, M.K., 2019. An assessment of greenhouse gas emissions from CLT and glulam in two residential nearly zero energy buildings. Wood Mater. Sci. Eng. 14 (5), 342–354. https://doi.org/10.1080/17480272.2019.1655792.
Mallo, M.F.L., Espinoza, O., 2016. Cross-laminated timber vs. concrete/steel: cost comparison using a case study. In: WCTE 2016–World Conference on Timber Engineering.
Oliver, C.D., Nassar, N.T., Lippke, B.R., McCarter, J.B., 2014. Carbon, fossil fuel, and biodiversity mitigation with wood and forests. J. Sustain. For. 33 (3), 248–275.
Padilla-Rivera, A., Amor, B., Blanchet, P., 2018. Evaluating the link between low carbon reductions strategies and its performance in the context of climate change: a carbon footprint of a wood-frame residential building in Quebec, Canada. Sustainability 10 (8), 2715. https://doi.org/10.3390/su10082715.
Peñaloza, D., Erlandsson, M., Falk, A., 2016. Exploring the climate impact effects of increased use of bio-based materials in buildings. Construct. Build. Mater. 125, 219–226. https://doi.org/10.1016/j.conbuildmat.2016.08.041.
Pierobon, F., Huang, M., Simonen, K., Ganguly, I., 2019. Environmental benefits of using hybrid CLT structure in midrise non-residential construction: an LCA based comparative case study in the U.S. Pacific Northwest. J. Build. Eng. 26, 100862. https://doi.org/10.1016/j.jobe.2019.100862.
Ramesh, T., Prakash, R., Shukla, K.K., 2010. Life cycle energy analysis of buildings: an overview. Energy Build. 42 (10), 1592–1600. https://doi.org/10.1016/j.enbuild.2010.05.007.
Robertson, A.B., Lam, F.C.F., Cole, R.J., 2012. A comparative cradle-to-gate life cycle assessment of mid-rise office building construction alternatives: laminated timber or reinforced concrete. Buildings 2 (3), 245–270. https://doi.org/10.3390/buildings2030245.
Rønning, A., Brekke, A., 2014. 4 - life cycle assessment (LCA) of the building sector: strengths and weaknesses. In: Pacheco-Torgal, F., Cabeza, L.F., Labrincha, J., de Magalhães, A. (Eds.), Eco-efficient Construction and Building Materials. Woodhead Publishing, pp. 63–83. https://doi.org/10.1533/978085709772.63.
Sandanayake, M., Lokuge, W., Zhang, G., Setunge, S., Thushar, Q., 2018. Greenhouse gas emissions during timber and concrete building construction—a scenario based comparative case study. Sustain. Cities Soc. 38, 91–97.
Sathre, R., O'Connor, J., 2010. Meta-analysis of greenhouse gas displacement factors of wood product substitution. Environ. Sci. Pol. 13 (2), 104–114. https://doi.org/10.1016/j.envsci.2009.12.005.
Shelby, L.B., Vaske, J.J., 2008. Understanding meta-analysis: a review of the methodological literature. Leisure Sci. 30 (2), 96–110. https://doi.org/10.1080/01490400701881366.
Skullestad, J.L., Bohne, R.A., Lohne, J., 2016. High-rise timber buildings as a climate change mitigation measure – a comparative LCA of structural system Alternatives. Energy Procedia 96, 112–123. https://doi.org/10.1016/j.egypro.2016.09.112.
Team, R.C., 2017. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna. https.Www.R-ProjeCt.Org.
Tellnes, L., Ganne-Chedeville, C., Dias, A., Dolezal, F., Hill, C., Zea Escamilla, E., 2017. Comparative assessment for biogenic carbon accounting methods in carbon footprint of products: a review study for construction materials based on forest products. iFor. Biogeosci. For. 10 (5), 815–823. https://doi.org/10.3832/ifor2386-010.
Tettey, U.Y.A., Dodoo, A., Gustavsson, L., 2019. Effect of different frame materials on the primary energy use of a multi storey residential building in a life cycle perspective. Energy Build. 185, 259–271. https://doi.org/10.1016/j.enbuild.2018.12.017.
The Net Zero Carbon Buildings Commitment. (n.d.). World Green Building Council. Retrieved April 22, 2020, from https://www.worldgbc.org/.
Thormark, C., 2002. A low energy building in a life cycle—its embodied energy, energy need for operation and recycling potential. Build. Environ. 37 (4), 429–435. https://doi.org/10.1016/S0360-1323(01)00033-6.
UNEP, United Nations Environment Programme, 2017. The emissions gap report 2017: a UN environment synthesis report. UN. https://doi.org/10.18356/1cf881fb-en.
UNEP, United Nations Environment Programme, 2019. The emissions gap report 2019. https://www.unenvironment.org/resources/emissions-gap-report-2019.
Wiik, M.K., Fufa, S.M., Kristjansdottir, T., Andresen, I., 2018. Lessons learnt from embodied GHG emission calculations in zero emission buildings (ZEBs) from the Norwegian ZEB research centre. Energy Build. 165, 25–34. https://doi.org/10.1016/j.enbuild.2018.01.025.
World Green Building Council, 2019. World green building Council annual report 2018/19. Retrieved May 30, 2020, from. https://www.worldgbc.org/.
Zeitz, A., Griffin, C.T., Dusicka, P., 2019. Comparing the embodied carbon and energy of a mass timber structure system to typical steel and concrete alternatives for parking garages. Energy Build. 199, 126–133. https://doi.org/10.1016/j.enbuild.2019.06.047.

Further reading

United Nations, Department of Economic and Social Affairs, Population Division, 2019. World Urbanization Prospects: 2018 Revision (ST/ESA/SER.A/420). United Nations, New York.