PUBLIC COMMENTS SUBMITTED AT
HIGHLANDS COUNCIL MEETING
APRIL 20, 2023

frontiers | Frontiers in Forests and Global Change

TYPE Policy and Practice Reviews
PUBLISHED 09 January 2023
DOI 10.3389/ffgc.2022.1073677

OPEN ACCESS

EDITED BY
Alfredo Di Filippo,
University of Tuscia, Italy

REVIEWED BY
Giuliano Maselli Locosselli,
University of São Paulo, Brazil
Heather Keith,
Griffith University, Australia

*CORRESPONDENCE
Michael J. Kellett
✉ kellett@restore.org

SPECIALTY SECTION
This article was submitted to Forest Management, a section of the journal Frontiers in Forests and Global Change

RECEIVED 18 October 2022
ACCEPTED 12 December 2022
PUBLISHED 09 January 2023

CITATION
Kellett MJ, Maloof JE, Masino SA, Frelich LE, Faison EK, Brosi SL and Foster DR (2023) Forest-clearing to create early-successional habitats: Questionable benefits, significant costs.
Front. For. Glob. Change 5:1073677.
doi: 10.3389/ffgc.2022.1073677

Forest-clearing to create early-successional habitats: Questionable benefits, significant costs

Michael J. Kellett[1]*, Joan E. Maloof[2], Susan A. Masino[3], Lee E. Frelich[4], Edward K. Faison[5], Sunshine L. Brosi[6] and David R. Foster[7]

[1]RESTORE: The North Woods, Lincoln, MA, United States, [2]Department of Biological Sciences, Salisbury University, Salisbury, MD, United States, [3]Trinity College, Hartford, CT, United States, [4]Department of Forest Resources, University of Minnesota Center for Forest Ecology, St. Paul, MN, United States, [5]Highstead Foundation, Redding, CT, United States, [6]Department of Wildland Resources, Utah State University Eastern, Price, UT, United States, [7]Harvard Forest, Harvard University, Petersham, MA, United States

A campaign is underway to clear established forests and expand early-successional habitats—also called young forest, pre-forest, early seral, or open habitats—with the intention of benefitting specific species. Coordinated by federal and state wildlife agencies, and funded with public money, public land managers work closely with hunting and forestry interests, conservation organizations, land trusts, and private landowners toward this goal. While forest-clearing has become a major focus in the Northeast and Upper Great Lakes regions of the U.S., far less attention is given to protecting and recovering old-forest ecosystems, the dominant land cover in these regions before European settlement. Herein we provide a discussion of early-successional habitat programs and policies in terms of their origins, in the context of historical baselines, with respect to species' ranges and abundance, and as they relate to carbon accumulation and ecosystem integrity. Taken together, and in the face of urgent global crises in climate, biodiversity, and human health, we conclude that public land forest and wildlife management programs must be reevaluated to balance the prioritization and funding of early-successional habitat with strong and lasting protection for old-growth and mature forests, and, going forward, must ensure far more robust, unbiased, and ongoing monitoring and evaluation.

KEYWORDS
natural climate solutions, forest carbon, old-growth forests, young forest, clearcutting, biodiversity, ecosystem services, wildlands

10.3389/ffgc.2022.1073677

1. Introduction

In this paper we conduct a wide-ranging and integrated assessment of the campaign to expand early-successional forest habitats in two regions of the United States: (1) the Northeast, i.e., New England states (Connecticut, Maine, Massachusetts, New Hampshire, Rhode Island, Vermont) and mid-Atlantic states (New York, Pennsylvania, New Jersey, Maryland, Delaware); and (2) the Upper Great Lakes areas of Michigan, Wisconsin, and Minnesota north and east of the prairie-forest border [see Cochrane and Iltis (2000), Frelich and Reich (2010), Anderson et al. (2018)]. We review the history of forest disturbance and biodiversity research, the genesis of the forest-clearing campaign and the conservation rationales, the contrasts between natural old-growth forests and intensively managed forests, the impacts of forest-clearing projects, and the current balance of activity between forest management and protection. We conclude that instead of intensive and costly management to create additional early-successional habitats, a new "natural" alternative should be considered which would protect and allow the dynamic growth of established aggrading, mature, and old-growth forests alongside maintaining existing early-successional habitats, where appropriate, for targeted species and cultural values. Although the focus of our analysis is two regions, we believe it offers useful lessons for many other parts of the U.S. and world experiencing similar situations (DellaSala et al., 2022b).

1.1. History of forest development and disturbance

Every place on Earth has a dynamic ecological trajectory based on temperature, rainfall, soils, natural disturbances, and other conditions. In the Northeast and Upper Great Lakes regions of the United States the predominant ecological trajectory of the landscape in the absence of intensive human activity is toward "old-growth" forests: a resilient, diverse, carbon-dense, and self-sustaining "shifting mosaic" of tree ages, microhabitats, and native species above and below ground (Pelley, 2009; Thom et al., 2019; Raiho et al., 2022).

For thousands of years before European settlement, vast "primary" forests were inhabited by a thriving Native human population and harbored many exceptionally large trees, and ecosystems that would be characterized as "old-growth" today (Lorimer, 1977; Whitney, 1994; Lorimer and White, 2003). Up to 90% of the Northeast was covered by such forests, and dominated by shade-tolerant and moderately shade-tolerant species (Foster, 1995; Cogbill, 2000; Cogbill et al., 2002; Shuman et al., 2004; Thompson et al., 2013; Foster et al., 2017; Oswald et al., 2020b). Approximately 50–60% of the Upper Great Lakes landscape, and 40–50% of the Southern Great Lakes landscape, consisted of old-growth forests (Cottam and Loucks, 1965; Frelich, 2002). These percentages in the Great Lakes regions pertain to older even-aged and multi-aged forests (generally more than 120 years old). Old-growth forests in the East include sites with trees more than 380 years old, established in the 1640s and earlier (Lorimer, 1980; McCarthy and Bailey, 1996; Abrams et al., 1998; Abrams and Copenheaver, 1999; Pederson, 2013; Hecter et al., 2019), and studies of remnant old-growth stands indicate they are adapted to long-intervals between catastrophic disturbances. Young trees of late-successional species (e.g., sugar maple, hemlock, beech) released from suppression combined with new seedlings of mid-tolerant tree species (e.g., white pine, yellow birch, American basswood, black cherry, white ash, northern red oak) after windstorms, and high intensity fires in conifer forests or blown down hardwood forests are followed by early-successional shade-intolerant species (e.g., paper birch, quaking, and bigtooth aspen) with some mid-tolerant species as listed above.

The terms "primary forest," "old-growth forest," and "mature forest," are not standardized (Leverett, 1996; Buchwald, 2005; Mackey et al., 2014; DellaSala et al., 2022a). For this analysis, we use the following definitions:

- Primary forest. A forest composed of native species that has never been logged and has developed following natural disturbances and under natural processes, regardless of its age (Kormos et al., 2018; FAO, 2020).
- Old-growth forest. A forest affected primarily by the forces of nature, with dominant canopy tree species at or beyond half their lifespan, and with structural characteristics such as canopy gaps, pit and mounds, large snags, gnarled tree crowns, a thick duff layer, and accumulated large coarse woody debris (Martin, 1992; Frelich, 1995; Dunwiddie and Leverett, 1996; Mosseler et al., 2003b; D'Amato et al., 2006; Mackey et al., 2014; USDA Forest Service and Bureau of Land Management, 2022).
- Mature forest. A forest with trees of intermediate age and lower levels of old-growth structural characteristics, but from which old-growth conditions are likely to develop over time if allowed to continue to grow (Spies and Franklin, 1991, Frelich, 1995; Strittholt et al., 2006; Keeton et al., 2011).

Old-growth forests not only have a high degree of structural diversity, but also contain a wide variety of tree species, herbaceous plants, insects, mosses and fungi, and deep, carbon-rich soil with an associated soil microbiome (Frelich, 1995; Davis, 1996; Lapin, 2005; D'Amato et al., 2009; Maloof, 2023). Permanent and semi-permanent large openings are rare in old-growth forests of these regions, associated mainly with cliffs and scree slopes, ridge tops, wetlands, peat bogs, serpentine barrens, avalanche tracks, river margins, pond and lake margins, and coastal shrublands and bluffs (Whitney, 1994; Foster and Motzkin, 2003; Fraver et al., 2009). Old-growth forests contain

10.3389/ffgc.2022.1073677

natural gaps of different sizes, which can be location-specific (wet, rocky, sandy) or part of a dynamic ecological trajectory due to disturbances, such as fire, windstorms, beaver activity, and insect outbreaks (Whitney, 1994; Boose et al., 2001; Frelich, 2002; Seymour et al., 2002; D'Amato et al., 2017). As a result the forest ecosystem remains intact and resilient, supporting widespread re-sprouting and recovery of trees.

Openland and early-successional habitats were not common before the arrival of Europeans in the Northeast or Upper Great Lakes (Cooper-Ellis et al., 1999; Foster et al., 2002; Faison et al., 2006; Anderson et al., 2018; Oswald et al., 2020b; Frelich et al., 2021). Early-successional habitats characterized about 1–4.5% of the Northeast, with greater amounts in coastal pine barren communities of Cape Cod, Long Island, and New Jersey (Lorimer and White, 2003). About 32% of the entire states of Minnesota, Wisconsin, and Michigan was represented by early-successional habitats, mostly in the savannas and prairies in the southern and western parts of the region. To the north, early-successional habitats were found in tens of thousands of patches of shorelines, marshes, sloughs, bogs, cliffs, and fire-prone sand plains (Veatch, 1928; Curtis, 1959; Marschner, 1975). Thus, the region had both dense forests and permanently open habitats maintained by the physiography of the landscape, including prairies and savannas maintained before European settlement by frequent fires—now almost absent due to agricultural conversion of the land. It is important to note that these open habitats were not early-successional stages for forests.

Native people living in the Great Lakes and the Northeast practiced subsistence hunting, fishing, and plant gathering, as well as burning and small-scale farming. Their population was less than 1% of the current population and largely centered along the coast and in major river valleys, with localized and modest impacts across most of the region (Whitney, 1994; Lorimer and White, 2003; Milner and Chaplin, 2010; Oswald et al., 2020b; Frelich et al., 2021; Tulowiecki et al., 2022).

The arrival of Europeans generated a radical landscape transformation. Upland areas, densely forested for thousands of years, were cleared for agriculture and kept open by crop cultivation, cattle and sheep grazing, increased burning of (dry) cleared land, and intensive use of remaining woodlands (Foster and Motzkin, 2003; Faison et al., 2006; Rhemtulla and Mladenoff, 2007; Scheller et al., 2008; Curtis and Gough, 2018; Oswald et al., 2020b). By the height of deforestation from 1850 to 80, 30% of northern New England and 40–50% of southern New England had been cleared (Foster et al., 2017), and by 1920 more than 90% of the Upper Great Lakes region was cutover (Greeley, 1925; Frelich, 1995).

Widespread deforestation caused a major shift in vegetation from long-lived and interior forest species to generalist and early-successional species (Thompson et al., 2013; Foster et al., 2017). Many of the latter species had been uncommon before European settlement, others migrated to the region, and some plants that had previously grown only on extreme and rare sites expanded their distribution and became common "old field" species (Marks, 1983). Early naturalists recognized that populations of some wildlife species had increased greatly due to this abundance of human-created early-successional habitats (Peabody, 1839). By the late 19th century, New England agriculture was declining, leaving countless abandoned and overgrown fields, grasslands, heathlands, and shrublands, as well as old-field white pine forests, and dense sprout woodlands. By the mid–20th century, significant areas of cutover forests were acquired by the public and allowed to begin growing back on state and federal lands (Titus, 1945; Jones, 2011; Knowlton, 2017). Today, millions of acres of forest are a globally significant example of ecological recovery, and the extent of early-successional habitats has declined accordingly (McKibben, 1995; Foster et al., 2002; Litvaitis, 2003; Foster et al., 2017). Consequently, species that depend on early-successional habitats have been returning to more historic levels, including the Bobolink *(Dolichonyx oryzivorus)*, Eastern Meadowlark *(Sturnella magna)*, Golden-winged Warbler (*Vermivora chrysoptera*), Yellow-breasted Chat (*Icteria virens*), and New England Cottontail (*Sylvilagus transitionalis*) (Figure 1; Litvaitis, 1993; Foster, 2002; Askins, 2011; Foster, 2017).

Although old-growth forests were the predominant ecological condition before European settlement, they are extremely rare today (Frelich, 1995; Dunwiddie et al., 1996; Davis, 2003; D'Amato et al., 2006; DellaSala et al., 2022b), much less common than younger habitats (Figure 2). A few relatively large tracts of old-growth and protected recovering forests survive in New York, Michigan, and Minnesota, but just small fragments remain across vast regions including all of New England. However, many mature forests are poised to transition to old-growth, and some are undergoing this transition (Ducey et al., 2013; Gunn et al., 2014). This can occur through a straightforward process of forest development and maturation.

In the Northeast, forests older than 150 years of age cover only about 0.3% of New England and 0.2% of the Mid-Atlantic region (USDA Forest Service, 2022b). Old-growth forests cover a scant 0.06% of Connecticut (Ruddat, 2022). A Massachusetts survey found a mere 1,100 acres of old-growth forest in 33 small stands, comprising just 0.02% of the land base (D'Amato et al., 2006). Most of the old-growth forest in the Northeast is found in the Adirondack and Catskill parks in New York (Dunwiddie et al., 1996; Davis, 2003; Keeton et al., 2011; New York Department of Environmental Conservation, 2021). In the Upper Great Lakes region, only about 1.9% of the currently forested area remains as primary forest that was never logged. Including secondary forests, approximately 5.5% of the northern hardwood forest type is older than 120 years of age, compared to 89% in the presettlement forest; for red-white pine this is 2.5% versus 55%. For all forest types, about 5.2%

10.3389/ffgc.2022.1073677

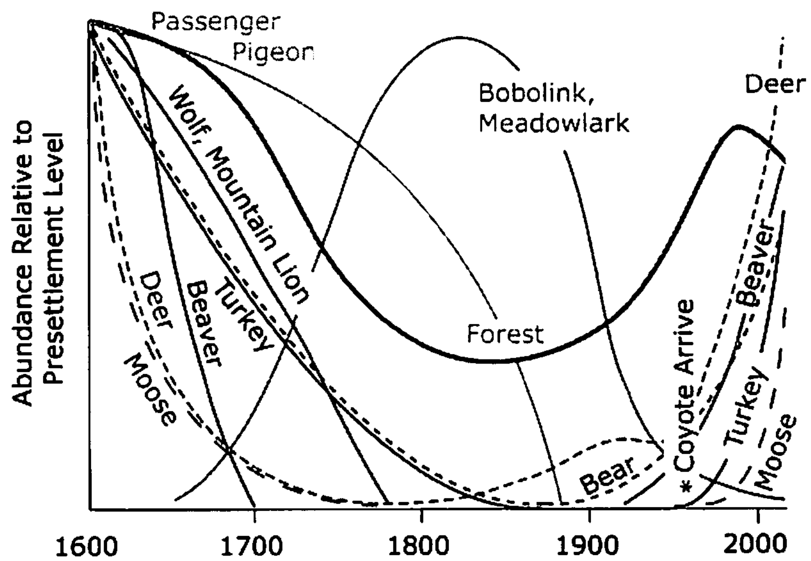

FIGURE 1
Changes in land cover and wildlife dynamics in New England from ~1600–2000. The green line shows the abundance, decline and then recovery of forest in New England, which paralleled the population changes in moose, beaver, and deer. The inverse trend is found in openland (early-successional) species, typified by bobolink and meadowlark. The inverted U shows the low population densities of these and other early-successional species before European settlement, increasing populations of these species as forests were cleared, and a return to lower populations as the forests have grown back. *The coyote is not native to New England. Adapted from Foster et al. (2002); also see **Figure 2**.

is old-growth compared with 68% before European settlement (Frelich, 1995).

1.2. Genesis and rationales of the early-successional habitat campaign

1.2.1. Genesis of the campaign and the "Young Forest Initiative"

A concerted campaign is working to slow and reverse the natural decline in early-successional habitat and species that accompanied the regional reduction in deforestation, intensive forestry, and agriculture. This campaign is promoting early-successional habitat through multiple activities: clearcutting, "group selection," and other forms of patch clearfelling in established forests; intensive "mechanical treatments" such as brushhogging and mowing; and herbicide application and prescribed fire in successional habitats and younger forests, which are often accompanied by other mechanical treatments (DeGraaf and Yamasaki, 2003; Oehler et al., 2006; American Bird Conservancy, 2007; Schlossberg and King, 2007; King et al., 2011; Yamasaki et al., 2014).

These intensive management activities have long been advocated to benefit popular game species that favor early-successional habitats, such as the American Woodcock (*Scolopax minor*), Ruffed Grouse (*Bonasa umbellus*), and White-tailed Deer (*Odocoileus virginianus*) (Lenarz, 1987; Caron, 2009; Derosier et al., 2015). In the last decade, an expanded management campaign has included popular non-game species that also use these habitats (see Section "1.2.2 Rationale for forest-clearing: halt the decline of specific wildlife species" below). This campaign involves an increasing number and diversity of agencies and organizations, and captures rising amounts of public money from state and federal sources. The goal is to maintain the recent historical and degraded condition of the natural forests of the region.

A key milestone in the genesis of this campaign was the 2008 American Woodcock Conservation Plan (AWCP; see Table 1 for Abbreviations), published by the Wildlife Management Institute (WMI) in collaboration with game management agencies and sportsmen's organizations (Kelley et al., 2008). The goal is to increase American Woodcock populations to early 1970s levels by clearcutting 11.2 million acres of forest in the Northeast and Upper Great Lakes regions—an area larger than the state of Maryland. WMI also launched the Upper Great Lakes Woodcock and Young Forest Initiative (YFI) to gain public support for the creation of early-successional habitats in Michigan, Minnesota, and Wisconsin (Wildlife Management Institute, 2009, 2010).

Wildlife Management Institute (WMI) soon began expanding the YFI to a national campaign (Gassett, 2018; Weber and Cooper, 2019). Recognizing the controversial nature of such widespread forest-clearing, the organization hired a marketing firm to "shape an overall communications

10.3389/ffgc.2022.1073677

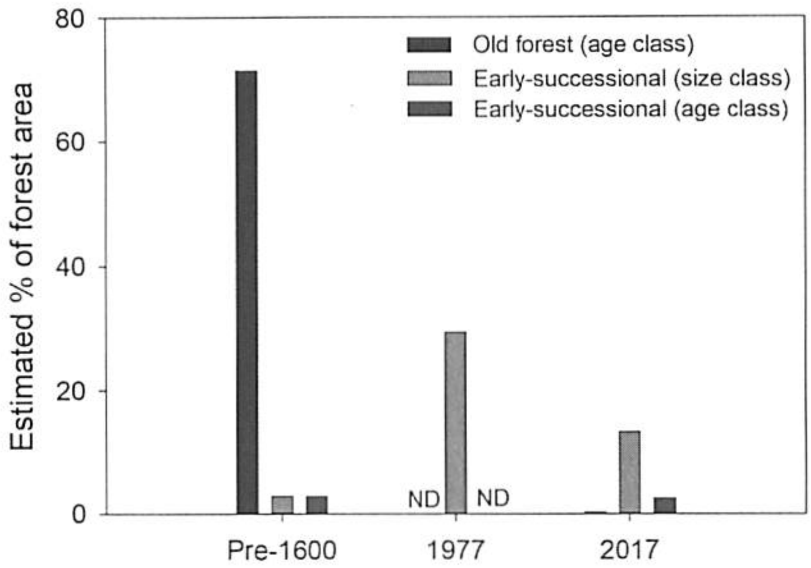

FIGURE 2
Estimated change in average % of early-successional and old forest habitat from pre-European settlement to current times in the Northeast US as extracted from multiple sources. Old forest is defined > 150 years old. The 1600 estimate for early successional forest is based on "seedling-sapling (1–15 years)" age class (Lorimer and White, 2003). The 1977 estimate is based only on "seedling-sapling" size class as reported in Oswalt et al. (2019); age class data were unavailable (ND = no data). Current estimates (2017) reflect two sources: Oswalt et al. (2019) and USDA Forest Service (2022b) wherein early successional forest (size class) reflects "seedling-sapling," the smallest class defined by the USDA Forest Service; and early successional forest (age class) reflects the 1–15 year age class. Note that while early-successional forest declined since 1977, it is similar and perhaps multiple times higher than pre-settlement values; and recent accounting is likely an underestimate: it does not include areas such as highway medians, small patches, or corridors (< 0.4 ha or < 36.5 m wide) that may be found on properties such as golf courses, farms, public and private institutions, and private yards. In contrast, old forest habitat has decreased dramatically (old forest data are barely visible in 2017 on this scale).

strategy" (Seng and Case, 2019). This firm administered opinion surveys and focus groups that showed most forest landowners value beauty, scenery, nature, and biodiversity far more than logging or financial return. A plan was then devised to promote early-successional habitats through an extensive network of partnerships. Terms which focus group participants found unappealing, such as clearcutting, early-successional habitats, shrub, and scrub, were replaced with the more appealing "young forests." Simple and positive language emphasized forest "health," wildlife, habitat diversity, and scientific-sounding outcomes. A pseudo-historical pitch was crafted to emphasize the decline of once common and familiar species without acknowledging the highly artificial and historically anomalous nature of their former abundance (see Table 2 for more details). Numerous publications were produced, such as, "Talking About Young Forests," to help "natural resource professionals…effectively advocate for creating and managing young forest habitat on public and private lands" (Oehler et al., 2013).

In 2012, YFI inaugurated the "youngforest.org" website, aimed at persuading target audiences to support the campaign (Young Forest Project, 2012). Within a decade, the YFI had recruited more than 100 "partners" (Supplementary material 1, Young Forest Project, 2022a). These are primarily traditional forestry and game species management interests, such as timber companies (Lyme Timber Company, 2017; Weyerhaeuser Company, 2020), federal and state forestry agencies (New York Department of Environmental Conservation, 2015; USDA Forest Service, 2018), federal and state wildlife agencies (U.S. Fish and Wildlife Service, 2015c; Connecticut Department of Energy and Environmental Protection, 2021b), and sportsmen's organizations (Russell, 2017; Weber and Cooper, 2019). All of these partners benefit from forest-clearing through increased profits from timber sales, larger agency budgets, more staff, direct payments for creating young forest habitat, or elevated populations of desired game species (see Supplementary material 1 for state-by-state examples of forest-clearing).

The YFI has attracted generous financial support from a wide range of public agencies, private organizations, and large corporations such as Richard King Mellon Foundation, U.S. Forest Service, U.S. Fish and Wildlife Service, American Forest Foundation, and Shell Oil Company [see Connecticut Department of Energy and Environmental Protection (2018); New Jersey Audubon (2018);

10.3389/ffgc.2022.1073677

TABLE 1 Abbreviations.

AWCP	American Woodcock Conservation Plan.
BBS	North American Breeding Bird Survey.
GAP 1	Gap Analysis Project Status 1. An area permanently protected from conversion of natural land cover, where ecosystems are allowed to function and develop predominantly under the influence of natural processes. Examples include National Parks, Wilderness Areas [see U.S. Geological Survey (2022b)].
GAP 2	Gap Analysis Project Status 2. An area permanently protected from conversion of natural land cover, but which may allow management practices that degrade the quality of existing natural communities. Examples include National Wildlife Refuges, State Parks, and Nature Conservancy preserves [see U.S. Geological Survey (2022b)].
GAP 3	Gap Analysis Project Status 3. An area predominantly protected from conversion of natural land cover, but subject to extractive uses. Examples include National Forests, Bureau of Land Management lands, most State Forests, and some State Parks [see U.S. Geological Survey (2022b)].
GAP 4	Gap Analysis Project Status 4. Lands with no mandates to prevent conversion of natural habitat types to unnatural land cover. Examples include agricultural and developed lands [see U.S. Geological Survey (2022b)].
IUCN	International Union for the Conservation of Nature.
SGCN	Species of Greatest Conservation Need.
SWAP	State Wildlife Action Plan.
WMI	Wildlife Management Institute.
YFI	Young Forest Initiative.

National Fish and Wildlife Foundation (2022b)]. In addition to activities on public lands, money is directed to land trusts (New England Cottontail, 2021) and private landowners (Natural Resources Conservation Service, 2018) through numerous state and federal sources. Much of this activity, supported by the significant money available for forest-clearing for early successional habitats (American Bird Conservancy, 2015; Natural Resources Conservation Service, 2019; Ruffed Grouse Society, 2022), engages broad support by well-intentioned landowners and conservationists by portraying this clearing as "restoration" to retain or save declining species (Smith, 2017; Weidensaul, 2018). There is little acknowledgment that, although these species are truly declining, they were artificially elevated in their abundance by colonial and relatively modern land-use practices that were abandoned in 19th and especially the 20th century.

Currently, every state in the Northeast receives substantial funding for early-successional habitat projects, either through direct federal programs or shared stewardship agreements (Fergus, 2014; USDA Forest Service, 2021b, 2022e; National Fish and Wildlife Foundation, 2022a; Sharon, 2022; Young Forest Project, 2022b). Even as forests are naturally recovering and helping to mitigate climate change in the absence of intensive logging, the momentum and money to clear forests and create open habitats is growing. For instance, the Infrastructure Investment and Jobs Act (2021) authorizes billions of dollars to increase logging for "wildfire risk reduction," "ecosystem restoration," and production of "mass timber" buildings (Parajuli, 2022; USDA Forest Service, 2022a). These massive programs will significantly increase early-successional forest habitats across the country, including in the Northeast and Upper Great Lakes regions. In contrast, there appear to be few resources devoted to protecting and expanding mature and old-growth forests.

Meanwhile, forest and wildlife managers–and a surprisingly large number of scientists—contend that the campaign to artificially expand early-successional habitats is vital because: (1) numerous wildlife species that depend on these habitats are declining and potentially endangered (Fergus, 2014), (2) the "restoration" of such habitats is needed to halt and reverse this decline (Young Forest Project, 2022c), and (3) the history of the region includes significant disturbance and presence of early successional habitats (Oehler et al., 2006). However, as noted previously, targeted population increases in specific species are mismatched generally with longer historical trends (Figure 1). Below is a more specific review of the rationales for these

TABLE 2 Marketing and communication strategies used by Young Forest Initiative.

Strategies	Recommendations	Actions and outcomes
Identify public values	Mobilize opinion surveys and host focus groups of landowners and the public to identify values. Set up regional pilot campaigns.	Recognize that forest owners and the public value beauty, scenery, nature, and biodiversity more than logging or financial return. Promote these values as enhanced by young forests.
Change language	Avoid terms with negative or unclear or connotations, i.e., "clearcutting," "early successional," "scrub," or "shrub."	Refocus language to emphasize "young forest" and emphasize that "a diversity of wildlife requires a diversity of habitats."
Create websites	Focus on target audiences such as private landowners, conservation professionals, residents of forested communities, and hunters.	Establish the Young Forest Project website as a central information hub that emphasizes benefits and collaboration to promote campaign goals.
Recruit partners	Identify partners with an interest in "young forest" species (i.e., deer, Ruffed Grouse, Wild Turkey, and Golden-winged Warbler).	Use the Young Forest Project website to build an extensive network of "partners" and include links to their websites (see Supplementary 2).
Persuade the public	Promote timber harvesting and active management to create young forests as a benefit to plants and wildlife.	Avoid and diminish negative impacts of clearcutting and focus on how "ugly [clearcuts] grow quickly into beautiful [habitats]."

10.3389/ffgc.2022.1073677

assertions, along with questions and concerns that have been raised in response.

1.2.2. Rationale for forest-clearing: Halt the decline of specific wildlife species

The primary justification cited for forest-clearing is that populations of many species needing early-successional habitats are declining (King et al., 2001; King and Schlossberg, 2014; Yamasaki et al., 2014; North American Bird Conservation Initiative, 2019; Rosenberg et al., 2019). Monitoring populations of species and preventing decline is a legitimate concern. Failure to take action in the past has allowed many species to become endangered or go extinct. Therefore, if these assertions are true, if losing species is a possibility, and if there are no plausible alternative explanations, a reasonable conclusion is that some species may need additional early-successional habitat to survive and thrive and would therefore justify habitat experiments and intensive habitat management programs to protect these species.

It is important to recognize that documentation of the decline of early-successional species is almost invariably based on a very recent baseline, generally dating to the 1960s or later (DeGraaf and Yamasaki, 2003; Massachusetts Audubon Society, 2013; North American Bird Conservation Initiative, 2014; Rosenberg et al., 2016, 2017, 2019; Connecticut Department of Energy and Environmental Protection, 2019; Sauer et al., 2020; Littlefield and D'Amato, 2022). This time period is a convenient benchmark because it falls within the lived experience of many of today's wildlife and forest managers and the landowners and public that they are trying to reach. It also coincides with the first annual North American Breeding Bird Survey (BBS), which took place in 1966 (Sauer et al., 2013). Prior to this time there was little reliable quantitative information on most bird populations (Foster, 1995; Foster et al., 2002; Dunn et al., 2005).

Although useful in many ways, the BBS is flawed as a truly long-term baseline for bird population trends. An ongoing deficiency is that the BBS is not a representative sampling of the broader landscape: it surveys habitats primarily near secondary roads and leaves out a wide range of habitats (Dunn et al., 2000; Dunn et al., 2005; Sauer et al., 2017). Furthermore, the quality of the data is inconsistent because volunteer observers have varying abilities (Dunn et al., 2000), including age-related declines in bird detection abilities and mobility (Farmer et al., 2014).

Beyond these problems, using a mid–1960s baseline for wildlife populations is fundamentally misguided. Every history of the region shows that at the time of the first BBS the Northeast and Upper Great Lakes regions were (and still are) in transition—with unnaturally high amounts of early-successional habitat such as abandoned farmland and forests recovering from intensive clearing and historically anomalous levels of fire, grazing and other human disturbances (Whitney, 1994; Foster et al., 2002; Mladenoff et al., 2008; Mladenoff and Forrester, 2018). As a result, the 1960s populations of wildlife species that occupied and thrived on such habitats were likely inflated well beyond what they would be in natural forests before European settlement (Litvaitis, 1993). This set the stage for a decades-long dramatic downward population trend due to recovering landscapes that are not yet within their true ecological trajectories (Massachusetts Audubon Society, 2013; Connecticut Department of Energy and Environmental Protection, 2019; Rosenberg et al., 2019).

Wildlife population trends since the 1960s need to be viewed in the context of a much longer timeframe (Schulte et al., 2005a,b), as has been provided by many superb studies of changes in major tree species for the region (Mladenoff et al., 2008; Thompson et al., 2016). For examples, Figure 1 spans the period from 1600 to today, displaying dual juxtaposed bell curves—one with forests (and some forest-associated species) steadily declining until the mid–1800s and then recovering through present day, and the other an inverse curve showing early-successional species populations increasing and then declining during that period (Foster et al., 2002). The recovery of the forested landscape may be causing previously inflated early-successional populations to restabilize closer to their natural baseline prior to the arrival of Europeans and under the conditions in which these species evolved.

Despite these caveats, State Wildlife Action Plans (SWAPs) rely heavily on the erroneous 1960s baseline for gauging the status of early-successional species. A SWAP must be filed with the U.S. Fish and Wildlife Service by each state to qualify for a number of major federal grants (The Wildlife Society, 2017). SWAPs include a list of Species of Greatest Conservation Need (SGCN), encompassing species that appear on federal or state lists as threatened or endangered, as well as those which are deemed rare, declining, or vulnerable to decline within that state (Minnesota Department of Natural Resources, 2016). SWAPs are useful sources of information for wildlife managers, but they are limited in scope, focusing on individual species within one state, rather than regional and national biodiversity (Pellerito and Wisch, 2002; Paskus et al., 2015).

With their mid–1900s baseline, SWAPs skew state-level biodiversity policies and programs toward management for conditions of that era. As noted, this is comfortable for wildlife and land managers who grew up during and recently after that time and appeals to many members of the public. However, this has created a false sense of endangerment for early-successional species that: (1) are common and of "least concern" based on International Union for the Conservation of Nature (IUCN) criteria (IUCN, 2012); (2) were historically uncommon (i.e., naturally rare, and at a natural population level); or (3) are non-native (i.e., did not occur in that state prior to European settlement and contribute to under-estimating populations of mature and old-growth forest species). The supposedly grave state of these species is reinforced further by the YFI. For example, its handbook for wildlife managers includes a list of "89 species of wildlife classified as [SGCN] that require young forest habitat to survive and breed" (Oehler et al., 2013).

10.3389/ffgc.2022.1073677

Although these species use early-successional habitats, only a small number of them are listed under the federal Endangered Species Act (U.S. Fish and Wildlife Service, 2022b), and many of them fall into the following categories:

- They are at the edge of their range in a particular state and were temporarily increased in numbers by past forest-clearing, but are now abundant and widely distributed across their range, such as the Yellow-breasted Chat (*Icteria virens*) in Connecticut or the Prairie Warbler in Massachusetts (Nolan, 1978; Southwell, 2001);
- They were probably rare in, or not native to, a particular state before the arrival of Europeans and moved in as a result of the widespread forest clearing in the 19th century, such as Golden-winged Warbler (Askins, 2011) and Chestnut-sided Warbler (Litvaitis, 1993; Foster et al., 2002) in New England;
- They have declined in population and distribution since the 1960s, but had a limited distribution in the landscape before European settlement, such as the New England Cottontail (*Sylvilagus transitionalis*) (Figure 3; U.S. Fish and Wildlife Service, 2015a);
- They have declined from past unnaturally high mid–20th century populations, but continue to be abundant and widely distributed, such as the American Woodcock (Seamans and Rau, 2018), Northern Bobwhite (*Colinus virginianus*) (Giocomo et al., 2017), Whip-poor-will (*Caprimulgus vociferus*), Bobcat (*Lynx rufus*), Smooth Green Snake *Opheodrys vernalis*), Eastern Buck Moth (*Hemileuca maia*), and Wild Lupine (*Lupinus perennis*) (NatureServe, 2022);
- Their declines can be attributed to other causes besides lack of habitat, such as the impact of West Nile virus on Ruffed Grouse populations (Stauffer et al., 2018);
- They benefit from limited, scientifically-backed habitat management, not forest-clearing, as with restoration of Wild Lupine (*Lupinus perennis*) for the protection of specialist butterflies (Pavlovic and Grundel, 2009; Plenzler and Michaels, 2015).

Including species of questionable "conservation need" on state SGCN lists has helped to validate and encourage forest-clearing and other intensive management to expand early-successional habitats. For instance, a major goal of the Connecticut SWAP is to "keep common species common" (Connecticut Department of Energy and Environmental Protection, 2015), which has been translated into an intensive focus on forest-clearing (Neff, 2017) and is promulgated in agency publications such as "The Clear Cut Advantage" (Connecticut Department of Energy and Environmental Protection, 2013). Many federal and state agencies have goals for significantly expanding early-successional habitats from current levels (USDA Forest Service, 2018; Massachusetts Division of Fisheries and Wildlife, 2022b) without clear plans for monitoring and maintaining the habitat they are creating.

A further problem is that forest-clearing advocates exaggerate the number of species that "require" or "need" early-successional habitat. For instance, the YFI website asserts, without evidence, that, "if we fail to actively create and renew young forest…[m]any songbirds will rarely be seen or heard [and] the New England Cottontail and Appalachian Cottontail could…go extinct (Young Forest Project, 2022c). Another YFI publication claims that, "more than 40…kinds of birds need young forest…" (Fergus, 2014), yet only 12 species of birds in the Northeast are actually considered early-successional forest specialists (Askins, 1993).

Among the species most commonly cited to justify large-scale forest-clearing are the American Woodcock, Ruffed Grouse, Golden-winged Warbler, and New England Cottontail. As discussed in detail in Supplementary 3, whether this strategy is necessary or desirable is open to question for each of these species. For example, the woodcock (Seamans and Rau, 2018), grouse (Wiggins, 2006), and cottontail (Fuller and Tur, 2012) are game species subject to being killed by hunters while the cause and potential solutions to warbler declines are uncertain (Streby et al., 2016).

There is a contention that forest-clearing not only "restores" early-successional species, but also benefits many interior species (Chandler et al., 2012; Stoleson, 2013; King and Schlossberg, 2014; Yamasaki et al., 2014; Schlossberg et al., 2018; New Jersey Department of Environmental Protection, 2018). Yet, these claims are based on a few studies that are limited in their targeted species, timeframe, and geographic scope, and rarely examine alternative hypotheses. For instance, although interior forest bird species may use available early-successional habitats to some extent, there is little evidence that such habitats are favored or necessary for their survival (Vega Rivera et al., 1998; Marshall et al., 2003; Dorazio et al., 2015).

Aside from questions regarding its necessity, long-term effectiveness, and unintended consequences, the intense focus on creating and restoring early-successional habitats diverts resources from exploring strategies to address other factors that are known to impact wildlife populations. These factors include food availability, over-hunting, disease, climate change, environmental toxins, and myriad other reasons that are not connected simply to the areal extent of early-successional habitat.

1.2.3. Rationale for forest-clearing: Halt decline of early-successional habitats

Before European settlement, countless small patches of early-successional habitats were created in the forests of the Northeast and Upper Great Lakes regions on a continuing basis, including by wind and ice storms, insect infestations and disease, drought, floods, fire, and to a lesser extent grazing by large mammals (Runkle, 1982; Peterken, 1996). Contemporary

10.3389/ffgc.2022.1073677

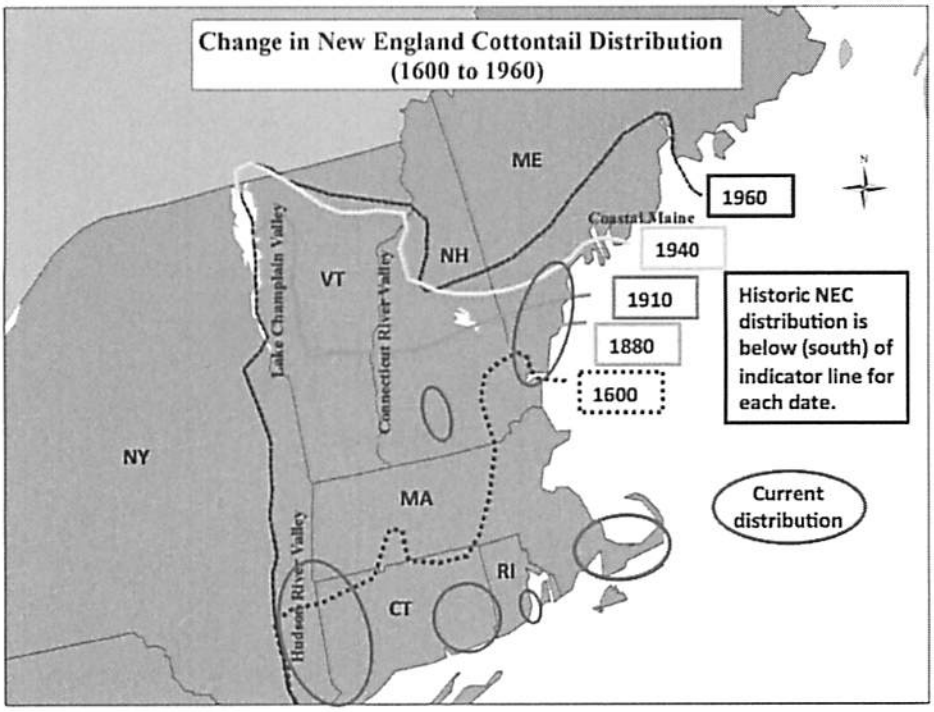

FIGURE 3
Changes in New England Cottontail (NEC) distribution over time. The estimated range of New England Cottontails (NEC) documented circa ~1600 (below the dashed line) included primarily Connecticut (CT) and Rhode Island (RI), and part of Massachusetts (MA). The distribution expanded dramatically northward following European settlement and land use (~1620–1960) to include Vermont (VT), Maine (ME), New Hampshire (NH), and into New York (NY; Hudson River Valley and Lake Champlain Valley). This dramatic expansion was followed by range contraction (~1960–2022) with forest regrowth and urban and suburban development. Green ovals represent the current documented distribution of NEC. Note that parts of current range still extend outside of pre-European settlement bounds, particularly in ME. NEC distribution map adapted from U.S. Fish and Wildlife Service (2015a,b).

studies of old-growth forests in the eastern U.S. suggest such small gaps are less than 0.1 acre in size. Larger openings were created by beaver impoundments and at intervals of hundreds of years by catastrophic windstorms and tornados. While uncommon in the Northeast outside of coastal pine barren communities, fire occurred every few decades and sometimes created large openings in the Upper Great Lakes region (Frelich, 1995; Lorimer and White, 2003). Native people generally caused minimal forest disturbances except around settlements scattered along coasts and river corridors (Motzkin and Foster, 2002; Parshall and Foster, 2002; Munoz and Gajewski, 2010; Oswald et al., 2020b; Frelich et al., 2021).

Advocates of clearing forests for early-successional habitats assert that natural and pre-European disturbances have been greatly attenuated and, therefore, managers must step in to create them (DeGraaf and Yamasaki, 2003; Oehler et al., 2006; Fergus, 2014; King and Schlossberg, 2014; Littlefield and D'Amato, 2022). While these habitats are reduced from their zenith in the 1800s and early 1900s (Foster et al., 2002; Litvaitis, 2003; Lorimer and White, 2003), extensive early-successional habitats still exist and are continuously produced, naturally and by widespread human activity. Natural disturbances such as storms, insect infestations and disease (including many novel non-native types that were not present when Europeans arrived), floods, and beaver impoundments, continue to create forest openings (Whitney, 1994; Askins, 2000; Frelich, 2002; Zlonis and Niemi, 2014; Wilson et al., 2019). Many types of human disturbances including farming, forest harvesting, and the expansion of electrical transmission lines provide additional extensive areas of early-successional habitats.

About 13% of forest area in the Northeastern United States is currently in the smallest (seedling-sapling) size class (Oswalt et al., 2019), a decline of more than 50% over the past 40 years, but several times higher than estimated presettlement values (Lorimer and White, 2003; Figure 2). Early-successional habitats in the Upper Great Lakes regions today are more difficult to quantify, because much of the southern and western portions of the three states are covered by savannas, prairies, and agricultural land. However, a study found that 4.4% of the area of Michigan north of the prairie-hardwood transition

10.3389/ffgc.2022.1073677

is characterized by forests less than 20 years old (Tavernia et al., 2016), and forests less than 20 years old are estimated to cover 12% of all forested lands in Wisconsin and Minnesota, respectively (Kilgore and Ek, 2013; Wisconsin Department of Natural Resources, 2020; USDA Forest Service, 2022b).

Approximately 65% of timber removals in the Northeast detected in U.S. Forest Service Inventory Data (FIA) are commercial clearcuts, shelterwood, high-grade, group selection, or pre-commercial thinning treatments (Belair and Ducey, 2018)—all major sources of early-successional habitats. In the Northeast and Upper Great Lakes, tens of thousands of acres of these habitats are created each year by the clearcutting of public and private timberlands—more than 10,000 acres in the national forests alone (USDA Forest Service, 2003; USDA Forest Service, 2017). Among the nine Northeast states, almost 19 million acres (16%) are farmland, most of which was formerly forested (U.S. Department of Agriculture, 2020), and about one-third of agricultural lands provide a mosaic of early-successional habitats such as grassland, woodland, wetland, and other open habitats (Brady, 2007; Jeswiet and Hermsen, 2015).

Expansive early-successional habitats are also the byproduct of urban and industrial developments. Examples include pipeline and powerline corridors (King et al., 2009; Askins et al., 2012), highway rights of way (Huijser and Clevenger, 2006; Amaral et al., 2016), golf courses (Tanner and Gange, 2005), greenways (Mason et al., 2007), wind and solar power arrays (South Carolina Department of Natural Resources, 2020; Zaplata and Dullau, 2022), military bases (Young Forest Project, 2022d), airports (Cousineau, 2017), and reclaimed strip mines (Bulluck and Buehler, 2006). Most of these development categories are not included in current inventories of early-successional habitats.

Additional factors are expected to increase the inventory of early-successional habitats. The forests of New England, for example, are rated as "above average" in health, but climate change is projected to have widespread impacts that will expand early-successional habitats (Janowiak et al., 2018; USGCRP, 2018). These impacts include major disturbances from storms (Miller-Weeks et al., 1999; Koches, 2019; Seitz, 2019), increased precipitation and flooding (National Wildlife Federation, 2009; Connecticut Department of Energy and Environmental Protection, 2020; Moustakis et al., 2021), periods of extreme heat and drought (Baca et al., 2018), insect and disease outbreaks (Paradis et al., 2008; Massachusetts Department of Conservation and Recreation, 2018), the introduction of new invasive species (Seidl et al., 2017), and shifts of vegetation and habitats northward (Chen et al., 2011; Toot et al., 2020). SWAPs and the YFI do not take into account such climate impacts.

Another potential source of early-successional habitats is the use of intensive forest management to increase climate "adaptation" and "resilience" of forests, which includes clearcutting, thinning, prescribed burning, and "assisted migration" through tree plantings (Foster and Orwig, 2006; USDA Forest Service, 2021a, 2022c; Climate Change Response Network, 2022a,b, Massachusetts Department of Conservation and Recreation, 2022; Northern Institute of Applied Climate Science, 2022; USDA Forest Service, 2022c). Such intensive forest interventions are, to date, mostly conceptual and experimental (Millar et al., 2007, D'Amato et al., 2011; Sheikh, 2011; Schwartz et al., 2012; Park and Talbot, 2018; Aquilué et al., 2020; Palik et al., 2022). Many questions remain regarding their economic, ecological, and legal and administrative feasibility (Handler et al., 2018). A prudent course would be to move cautiously with such novel strategies while expanding protection for mature and old-growth forests, which have a high degree of ecosystem integrity, genetic diversity, and adaptive capacity (Mosseler et al., 2003a; Thompson et al., 2009; Rogers et al., 2022).

An increasingly common rationale for forest-clearing is that it is necessary to recreate the way that Native people lived in relationship with the land. This is based on the extensively criticized hypothesis that long before European settlement, humans were deliberately managing most of the Northeast and Upper Great Lakes landscape using forest burning and clearing to improve habitat for favored plants and animals (Day, 1953; Mann, 2005; Abrams and Nowacki, 2008; Poulos and Roy, 2015). Some accounts take the idea even further, contending that by 1600, North America was "a humanized landscape almost everywhere" (Denevan, 1992), managed by Native people as a "garden" (Pyne, 2000), with virtually no "natural" plant communities (Williams, 2002). According to this view, the cessation of widespread and frequent pre-European burning and the reforestation of large parts of the region (which had been cleared after European settlement) have resulted in a massive loss of early-successional habitats and species, seriously threatened major plant communities, and reduced native biodiversity (Brose et al., 2001; Poulos and Roy, 2015; Abrams and Nowacki, 2020). The assumed loss of management by Native people is also cited as a major cause of the transition now underway of many oak forests to forests dominated by shade-tolerant species (Abrams, 1992; Brose et al., 2001; Abrams, 2005; Nowacki and Abrams, 2008).

Native burning and other subsistence practices, such as hunting, fishing, plant gathering, and small-scale farming had notable ecological impacts in the immediate vicinity of native encampments and settlements in the Northeast and Upper Great Lakes regions (Whitney, 1994; Lorimer and White, 2003; Oswald et al., 2020b; Frelich et al., 2021; Tulowiecki et al., 2022). However, modern land managers seem to be inappropriately misinterpreting a set of novel landscape conditions created by European land use over the last few centuries as having pre-European origins (Chilton, 2002; Oswald et al., 2020b; Cachat-Schilling, 2021). Extrapolating this misinterpretation to a regional scale has led to claims of widespread and intensive Native manipulation for millennia before European settlement. Unfortunately, these sweeping assumptions are

10.3389/ffgc.2022.1073677

being used to justify large-scale clearing and prescribed burning of established and recovering forests (Pyne, 2000; Brose et al., 2001; Williams, 2002; Oehler et al., 2006; Poulos and Roy, 2015; Abrams and Nowacki, 2020). In 2019 alone, 365,306 acres of forest—an area larger than Rocky Mountain National Park—were burned through prescribed fire in the Northeast and Upper Great Lakes, according to state forestry agencies (Melvin, 2020). Examples of major prescribed fire projects are found in Connecticut (Connecticut Department of Energy and Environmental Protection, 2021a), Massachusetts (Clark and Patterson, 2003), Michigan (Michigan Department of Natural Resources, 2022), and Vermont (USDA Forest Service, 2022d). This is in addition to the significant expanses of forest that are cleared under the premise of creating early-successional habitat.

Beyond the greater risks from mechanized modern forest management, there is significant controversy regarding the hypothesis of intensive and extensive management of the pre-European landscape by Native people (cf., Cachat-Schilling, 2021). For example:

- The presumption that the presettlement landscape was dominated by agriculturally based Native people who regularly burned large areas relies primarily on written or oral accounts by European explorers, travelers, and colonists. The vast majority of these narratives were not objective descriptions, but were vague, subjective, biased, or even meant to promote profit-making enterprises (Russell, 1981; Forman and Russell, 1983; Russell, 1983; Vale, 1998; Vale, 2002; Barrett et al., 2005; Munoz et al., 2014; Foster, 2017).
- Maintenance of the envisioned anthropocentric landscape would have required Native communities to move every 10–20 years, thereby creating extensive early-successional habitat and a wide variety of even-aged forest patches. This scenario is not supported by archeological studies of pollen and charcoal (Chilton, 2002; Oswald et al., 2020b).
- Localized burning and other land use did commonly occur in some population centers along the New England coast where maize agriculture had developed, the estuaries of New York, New Jersey, Delaware, and Maryland, around the eastern Great Lakes, and along major rivers (Russell, 1981; Motzkin and Foster, 2002; Milner and Chaplin, 2010; Munoz and Gajewski, 2010). However, throughout much of the rest of the Northeast and Upper Great Lakes regions, there is no evidence of significant land clearing or agriculture (Chilton, 2002; Parshall and Foster, 2002; Lorimer and White, 2003; Faison et al., 2006; Matlack, 2013; Oswald et al., 2020b). Rather, pollen and charcoal studies show that the vast interior of these regions had a dispersed, low-density population that was seasonally mobile and utilized native resources, not agriculture (Milner and Chaplin, 2010; Foster, 2017; Oswald et al., 2020b; Frelich et al., 2021). Archeological evidence indicates that many Native settlements in these regions are a relatively recent phenomenon—for example, Iroquois settlement began during the last millennium (Warrick, 2000; Bruchac, 2004; Jordan, 2013) and New England coastal settlement was likely encouraged by trade with Europeans (Foster, 2017).
- Pollen and charcoal studies as well as fire records indicate that fire activity before the arrival of Europeans tracked climate and vegetation at broad scales, rather than changes in the size of Native populations (Oswald et al., 2020b; Frelich et al., 2021). Indeed, the period of greatest Native population, shortly before the time of European colonization, was one of relatively low fire activity. At smaller spatial scales, particularly near the coast, some pollen records do show relatively high fire activity just prior to European settlement in areas of higher human population densities (Stevens, 1996; Lorimer and White, 2003; Parshall et al., 2003). Sites on steep slopes in the Appalachians have both a pre-history and a historic pattern of frequent crown and ground fires (Delcourt and Delcourt, 1998; Shumway et al., 2001; Buckley, 2010). Overall fire activity spiked after forest-clearing by European settlers created dry and flammable early-successional habitats, spiked again in the late 19th and early 20th centuries with the expansion of fire-prone abandoned farmlands and cutover forests, and has dramatically declined in the last century (Irland, 2013, 2014; Frelich et al., 2021).
- Long before the first colonization of North America 15,000–18,000 years ago, Northeast and Upper Great Lakes ecosystems had evolved and were maintained by climate and natural disturbances (Foster et al., 2002; McEwan et al., 2011; Noss et al., 2014; Pederson et al., 2014; Oswald et al., 2020b). Historical data and pollen studies indicate that before European settlement, forests were mainly characterized by long-lived shade tolerant and moderately shade tolerant species, not fast growing, early-successional and weedy species that would indicate widespread Native burning (Russell, 1983; Foster et al., 2002; Motzkin and Foster, 2002; Parshall and Foster, 2002; Parshall et al., 2003; Faison et al., 2006; Shuman et al., 2019; Oswald et al., 2020b). Oak savannahs along the prairie-forest border in the Upper Great Lakes region were far more widespread than today and likely maintained at least in part by greater frequencies of fire, including burning by Native people (Whitney, 1994; Frelich et al., 2021; Paciorek et al., 2021). However, the current shift of some forests from disturbance-tolerant species to shade-tolerant species can be explained by changes in climate and other factors rather than a lack of human-caused fires (Foster et al., 2002; McEwan et al., 2011; Noss et al., 2014; Pederson et al., 2014; Oswald et al., 2020b).
- Fire-prone ecosystems occupy about 25% of the forested landscapes of northern Minnesota, Wisconsin, and Michigan (Heinselman, 1973; Frelich, 1995;

10.3389/ffgc.2022.1073677

Frelich and Reich, 1995). However, even with the high occurrence of fires, there was still a much higher proportion of old-growth prior to European settlement than today (Frelich, 1995). Approximately 55% of forests were old growth within the 25% of the landscape that is fire prone (pine and oak forests with some aspen birch and spruce). These areas had 100–250 year return times for severe fires, so that only 55% of the stands would reach an age of 120 years or more. There were both natural and human understory burns, which helped maintain the old multi-aged condition in some stands. Elsewhere, for example in northern hardwood forests, where fires were much less common, the proportion of old-growth was much higher and wind storms were the primary disturbance. Severe fires that set succession back to birch and aspen were quite rare in these areas and were confined largely to blowdown areas. Only small proportions of fire-prone forest landscapes in the Boundary Waters Canoe Area Wilderness and Voyageurs National Park had a long history of regular understory burns (Johnson and Kipfmueller, 2016; Kipfmueller et al., 2017).

- In the Northeast, only limited areas are susceptible to fire, such as coastal pine barrens of Massachusetts, New York, and New Jersey, as well as scattered pavement barrens and sandplain communities in upstate New York and the Connecticut Valley (Forman and Boerner, 1981; Motzkin et al., 1999). Climate change and European land use have been the most important agents of change on these landscapes (Motzkin et al., 1999; Parshall et al., 2003).

In summary, current understanding of the role of fire and other disturbances in the Northeast and Upper Great Lakes regions before the arrival of Europeans is based on uneven, area-specific, and often-inconclusive information (Oswald et al., 2020a; Frelich et al., 2021). Available evidence does not support the hypothesis of widespread, intensive, ongoing burning and other land management over millennia by Native people (Cachat-Schilling, 2021). Instead, the evidence points to human use before European colonization limited to areas near settlements and ultimately constrained by a regional human population that is estimated to be less than 1% of the present population (Milner and Chaplin, 2010).

1.2.4. Rationale for forest-clearing: Reduce the prevalence of "mature" forests

Forest-clearing advocates assert that, in parallel with the presumed lack of "young" forests, there is an overabundance of "mature," and "even-aged" forests across the landscape. They contend that these forests do not provide an adequate diversity of habitats, and that "active management" can "restore" forest diversity and resiliency by "mimicking" natural forest disturbances and conditions (National Commission on Science for Sustainable Forestry, 2007; Fergus, 2014; King and Schlossberg, 2014; New Jersey Department of Environmental Protection, 2018; Rohrbaugh et al., 2020; Littlefield and D'Amato, 2022). Prior to evaluating this rationale it is important to note that a forest termed "even-aged" can include ages that vary by about 20% of the dominant age, and may also include young trees/advance regeneration, dead trees, and a mosaic of habitats (for example, due to insect damage or storms). "Even-aged" does not mean "even-sized" and tree growth is highly influenced by local site conditions for that species. The term "even-aged" can evoke images of a tree farm or a plantation, but natural forests do not have such a uniform structure, particularly those older than 60–80 years. Although 60–80 year old trees may be termed "mature," or almost "overmature," they are at far less than half their natural lifespan and likely at far less than 20% of their potential carbon accumulation (Thompson et al., 2009; Leverett et al., 2021). Most important, forests that are relatively even-aged will transition on naturally toward old-growth and uneven-aged condition if simply left alone (Gunn et al., 2014; Catanzaro and D'Amato, 2019).

With these caveats in mind, it is important to determine if and when removing mature or "even-aged" forests has net benefits. In terms of risks, there is considerable evidence that human-created or -maintained habitats do not provide the complexity, resilience, and diversity over long periods of time that are provided by natural forest ecosystems (Nitschke, 2005; North and Keeton, 2008; Thompson et al., 2009; Lindenmayer and Laurance, 2012; Belair and Ducey, 2018; Thom and Keeton, 2020). Moreover, countless interconnected and long-term ecological variables and processes are not well understood or are still simply unknown—and therefore cannot be "replicated" by human intervention with any confidence.

Taken together, long-term monitoring and further research on these issues should be a top priority. After a natural disturbance a forest can be a chaotic jumble of dead and damaged trees, downed wood, and tip-ups—many involving immense old trees and their associated biodiversity above and below ground (Lain et al., 2008; Santoro and D'Amato, 2019). In a natural forest, snags and downed logs and uproot mounds and pits are large and enduring for 100 years or more, there are no large areas of bare ground or scarified soil, and downed wood and vegetation remains on site (Foster et al., 2003). After an extreme event, such as a hurricane, there may be abundant advance regeneration, understory vegetation, and a mix of damaged and undamaged trees. These building blocks help the forest recover and resist the intrusion of invasive species (Plotkin et al., 2013, D'Amato et al., 2017). Even forests with almost no advance regeneration can regenerate rapidly after a major disturbance (Faison et al., 2016).

To summarize, current programs that create new early-successional forest habitats involve clearing established forested areas. These human-made habitats are dramatically different from the old-growth forest habitats with a mosaic of natural disturbances that dominated the landscape of the Northeast and

10.3389/ffgc.2022.1073677

most of the Upper Great Lakes before European settlement. Early-successional habitats have declined since their peak in the 19th and early 20th centuries but they are still widely represented, actively created by natural and human disturbances, likely undercounted, and expected to increase in the future. In light of the concerns discussed above, there is a compelling argument for re-evaluating the assertion that creating more early-successional habitat is essential for the survival and health of ecosystems, habitats, or species.

2. Impacts of forest clearing projects

2.1. Impacts on biodiversity

Advocates contend that widespread and increased forest-clearing will not have significant negative environmental impacts and can even benefit species associated with mature and old-growth forests (Chandler et al., 2012; Schlossberg et al., 2018; Nareff et al., 2019). Yet, there is ample evidence that this will result in the loss of mature forests and future old-growth habitats, reduced connectivity, an increase in edge habitats, the spread of invasive species, and deleterious effects due to mechanical disruption and species isolation (Wilcove et al., 1986; Small and Hunter, 1988; Franklin, 1989; Askins, 1992; Faaborg et al., 1993).

Meanwhile, and perhaps most important, we have insufficient data on many classes of organisms, and vast numbers of species are still undiscovered (Mora et al., 2011). Numerous moss species need older trees with thicker moisture-holding bark to survive droughts (Zhao et al., 2020). Native snails and insects are more abundant in older forests (Jordan and Black, 2012; Maloof, 2023). These forests host vast networks of plant roots and mycorrhizae, which may link trees to each other and allow the transfer of resources between mature trees (Simard et al., 2012). There is evidence that millions of species of fungi and bacteria swap nutrients between soil and the roots of trees in an interconnected "wood-wide web" of organisms (Steidinger et al., 2019). As scientific methodology evolves, so does our ability to detect tiny organisms and new molecules, including those of critical importance for medicine. In 2018, 16 new species were discovered in a teaspoonful of soil in Massachusetts (Schulz et al., 2018). A study of enchytraeids (a type of annelid worms) in maple forests of northern Minnesota found 9 species new to science (Schlaghamerský et al., 2014). Forest maturity increases the presence of groundwater macroinvertebrates and, in particular, uncommon species (Burch et al., 2022).

Unfortunately, few forests are surveyed for all types of life-forms before clearing to create early-successional habitats. "Resetting" a forest to age "zero" by clearing it reduces ecological complexity immediately because it prevents the full expression of structural and ecological diversity as well as myriad ecosystem services. Recovery is uncertain. Although southeastern U.S. forests are some of the most frequently logged forests in the world (Hansen et al., 2013)—resulting in ample early successional habitat—the region has experienced dramatic long-term declines in early-successional birds (Hanberry and Thompson, 2019). Even less-intensive logging activity can diminish or eliminate disturbance-sensitive and slowly dispersing plant and animal species, with recovery potentially taking many decades, if at all (Duffy and Meier, 1992; Petranka et al., 1994; Hocking et al., 2013).

It is instructive to contrast previously cleared forests that are designated as parks or preserves, where forest ecosystems have been allowed to function and develop predominantly under the influence of natural processes (i.e., GAP 1 areas) with forests subject to clearing of established forests to create early-successional habitats (i.e., some GAP 2 areas) or to commercial logging (i.e., GAP 3 or GAP 4 areas). For more detail on GAP classifications, see Table 1 and U.S. Geological Survey (2022b). Forests that are allowed to recover naturally and develop past the stem-exclusion phase steadily gain structural complexity and biodiversity, in part from ongoing low-to-moderate severity disturbances (Zlonis and Niemi, 2014; Miller et al., 2016; Hilmers et al., 2018). Indeed, the accumulated legacy of a mosaic of natural disturbances is greatest in unmanaged old-growth forests (Oliver and Larson, 1996; Askins, 2000; Lorimer and White, 2003). For instance, the 1-million-acre Boundary Waters Canoe Area Wilderness in Minnesota has taller tree canopies, greater tree species richness, and a larger number of bird species than adjacent managed national forest lands (Zlonis and Niemi, 2014). This wilderness also hosts a similar richness of bird species that favor young forests, such as the Chestnut-sided Warbler (Zlonis and Niemi, 2014). In Maine's "forever wild" Baxter State Park, natural insect outbreaks create open habitats that benefit early-successional species (Oliveri, 1993). A survey of Michigan habitats concluded that designated wilderness areas had considerable early-successional habitats, even though they were not open to logging or habitat management (Tavernia et al., 2016). As discussed below, findings were similar in New York's "forever wild" Adirondack and Catskill forest preserves (Widmann et al., 2015).

Numerous rare, threatened, and endangered wildlife species depend upon mature and old-growth forests and their ecosystem services. These species include migratory birds such as the Cerulean Warbler (*Setophaga cerulean*) (U.S. Fish and Wildlife Service, 2006; Dawson et al., 2012) and Wood Thrush (*Hylocichla mustelina*) (Bertin, 1977; Hoover et al., 1995; Rosenberg et al., 2003). They include mammals such as the Eastern Spotted Skunk (*Spilogale putorius interrupta*) (Lombardi et al., 2017; Hassler et al., 2021; Pearce et al., 2021), Appalachian Cottontail (*Sylvilagus obscurus*) (Chapman et al., 1992), Northern Long-eared Bat (*Myotis septentrionalis*) (U.S. Fish and Wildlife Service, 2022a), and Allegheny Woodrat (*Neotoma magister*) (Balcom and Yahner, 1996; Lombardi et al., 2017). They include plants such as Butternut (*Juglans cinerea*), (Schultz, 2003), Canada Yew (*Taxus canadensis*) (Dunwiddie et al., 1996; Windels and Flaspohler, 2011), Frasier Sedge (*Cymophyllus fraserianus*) (Godt et al., 2004), and American Ginseng (*Panax quinquefolius*) (McGraw et al., 2013). Some species reach their highest densities in old-growth forests, including southern and northern flying squirrels, forest interior birds, and spring ephemeral wildflowers.

The fragmentation of forests, particularly with roads and other human intrusion, can result in the decline of forest interior species. This can have significant impacts on the abundance, species richness, and community dynamics of migratory birds (Small and Hunter, 1988; Askins, 1992; Hagan et al., 1996; Zuckerberg and Porter, 2010; Askins, 2015; Betts et al., 2022). Apex predators can be lost, leading to further biodiversity loss as well as altered dynamics of disease, carbon accumulation, invasive species, and biogeochemical cycles (Terborgh et al., 1999; Anderson et al., 2004; Estes et al., 2011; Terborgh, 2015). Even common forest species are subject to major declines due to loss of natural forest habitats. A global report shows a 69% decrease in monitored wildlife populations between 1970 and 2018, in large part due to habitat fragmentation and degradation (WWF, 2022). Fragmentation can increase prevalence of wildlife diseases including Raccoon Roundworm (*Baylisascaris procyonis*) (Wolfkill et al., 2021) and may be a factor in oak decline and loss of ecosystem services (Tallamy, 2021) as well as reduced underground biodiversity—a concern that is less explored in the Northeast and Upper Great Lakes than in western forests (Simard, 2021).

Figure 1 reflects biodiversity impacts of habitat changes and hunting over several hundred years. Habitat loss was a factor in the decline of deer, moose, beaver, turkey, wolf, mountain lion, and bear, but intensive hunting and trapping probably had the greatest impact (Foster et al., 2002). Coyotes migrated eastward following wolf extirpation, interbred with wolves, and partially filled the vacant niche left by wolf extirpation. Deer can thrive in disturbed landscapes, which explains their recovery once hunting pressure was relieved (Michigan Department of Natural Resources, 2016). Forest-clearing is widely used today to boost populations of deer and other game species (Lashley et al., 2011; Dechen Quinn et al., 2013; Michigan Department of Natural Resources, 2017). However, high deer population densities can have significant negative effects on forest regeneration, native herbaceous plants—especially charismatic floristic groups such as orchids—and songbirds and their habitats (Alverson et al., 1988; deCalesta, 1994; Rooney and Waller, 2003; Knapp and Wiegand, 2014; Jirinec et al., 2017). Clearing established forests can also introduce and spread invasive and non-native species that ultimately reduce biodiversity (McDonald et al., 2008; Eschtruth and Battles, 2009; LeDoux and Martin, 2013; Coyle et al., 2017). Managed forests have been found to have as much as three times more invasives than fully protected national parks or wilderness (Riitters et al., 2018). Invasive plants can have a negative impact on native animal populations, including birds, mammals and other vertebrates (Fletcher et al., 2019). Invasive earthworms are a serious concern, particularly the new threat of jumping worms (*Amynthas spp.*) that destroy forest soil very rapidly (Frelich et al., 2019).

2.2. Impacts on the atmosphere

Forests influence water cycles, reduce local and global temperatures, and sequester and accumulate carbon. While carbon receives the most attention, multiple biophysical processes are crucial and interactive (Makarieva et al., 2020; Lawrence et al., 2022). Proponents of forest-clearing assert that carbon emissions are offset by increased sequestration rates of younger forests, by converting trees to wood products, by burning logging "waste" for bioenergy, and by forest carbon accumulation elsewhere—or that the amount of forest removal is so small as to be inconsequential (Hawthorne, 2020; Jenkins and Kroeger, 2020; USDA Forest Service, 2021a). On the contrary, these activities have significant climate costs, including the release of greenhouse gases from the cutting, processing, and transporting of trees for wood products; the disposal of waste and wood products; the release of methane from each log landing; the release of carbon from disturbed soils; and the loss of carbon uptake and accumulation by standing trees (Smith et al., 2006; Nunery and Keeton, 2010; Ingerson, 2011; Mika and Keeton, 2013; Catanzaro and D'Amato, 2019; Cook-Patton et al., 2020; Leturcq, 2020; Vantellingen and Thomas, 2021).

Some studies suggest that younger forests between 30 and 70 years (Catanzaro and D'Amato, 2019) or 40–80 years (Leverett et al., 2021) can sequester carbon at a faster rate than mature or old-growth forests. Other analyses indicate that lands reserved from logging in the Northeast have net carbon sequestration rates that are roughly 33% higher than in logged forests and are projected to sequester more carbon over the next 150 years (Brown et al., 2018). Nevertheless, the climate mitigation value of forest carbon lies not in the sequestration rate but in the total amount that is accumulated and kept out of the atmosphere (Mackey et al., 2013). The power of forests in this process is unparalleled and far greater in old forests than in young forests, both above and below ground; carbon continues to accumulate for centuries (Zhou et al., 2006; Luyssaert et al., 2008; Keeton et al., 2011; Curtis and Gough, 2018; Leverett et al., 2021; Law et al., 2022).

The amount of carbon lost when cutting a mature or old-growth forest is not recovered by fast-growing young forests for many decades to well over a century (Harmon et al., 1990; Aalde et al., 2006; Krebs et al., 2017). One study found almost no net carbon accumulation for 15 years after clearcutting—currently a critical time window for reining in global greenhouse gas emissions (Hamburg et al., 2019). In some cases, older forests are accumulating more carbon as the climate warms (Finzi et al., 2020), they are better able to withstand physiological stress, and they are also more resistant to the stress of climate change than younger forests, particularly regarding carbon storage,

10.3389/ffgc.2022.1073677

timber growth rate, and species richness (Thom et al., 2019). Soil accounts for approximately 50% of total ecosystem carbon storage in the Northeast, with mineral soils comprising the majority (Fahey et al., 2005; Petrenko and Friedland, 2015). Forest-clearing can mobilize and release soil carbon for decades (Nave et al., 2010; Petrenko and Friedland, 2015; Lacroix et al., 2016). It can take from 60 to 100 years for soils on a site to recover from clearcut logging (James and Harrison, 2016).

It is crucial to note that forest carbon stocks in the U.S. are already depleted by about 60% due to past logging and clearing (McKinley et al., 2011) and ongoing timber removals (Gunn et al., 2019). Logging accounts for about 86% of the carbon emitted by U.S. forests each year—far greater than insects, storm damage, fire, development and other uses combined (Harris et al., 2016; Duveneck and Thompson, 2019). Although a small percentage of the carbon in trees that are cut is stored in durable wood products, in the U.S. about 76% of carbon in trees cut for timber is released into the atmosphere each year (Domke et al., 2018), with most of it emitted quickly in processing, waste, and short-lived products (Harmon et al., 1996; Ingerson, 2011; Harmon, 2019; Leturcq, 2020). A logged mature forest stores less than half of the carbon of an uncut mature forest, even if carbon stored in wood products is included in the carbon storage total of the logged areas (Nunery and Keeton, 2010; Law et al., 2022). Impacts are similar for forest-clearing to produce wood bioenergy, which advocates claim is "carbon neutral" (Collins et al., 2015). However, cutting and burning trees releases large amounts of carbon immediately that would take many decades to be recover-if the forest grows back. In addition to other disrupted biophysical processes, this is time we cannot afford in light of the urgent climate crisis (Schulze et al., 2012; Law et al., 2018; Sterman et al., 2022). In short, clearing forests—whether for early-successional habitat or bioenergy—results in serious impacts to the atmosphere. In terms of maximizing carbon accumulation, allowing forests to regrow and remain standing—termed proforestation—is demonstrably preferable to cutting them (Buotte et al., 2019; Moomaw et al., 2019; Mackey et al., 2020; Rogers et al., 2022).

Despite widespread past clearing, the forests of the Northeast and Upper Great Lakes have recovered to the point that they are among the most intact and carbon-dense in the eastern U.S. (Zheng et al., 2008; Zheng et al., 2010; Foster et al., 2017). In addition, because these forests grow vigorously, decay slowly, and are, on average, less than 100 years old, they have centuries of growth ahead and enormous capacity for additional carbon storage (Pan et al., 2011; Williams et al., 2012) and climate stabilization. If allowed to continue growing, these forests can potentially increase *in situ* carbon storage by a factor of 2.3 to 4.2 (Keeton et al., 2011) and perform crucial ecosystem services (Meyer et al., 2022). For these reasons, the New England Acadian region was identified as a Tier 1 stabilization area in the Global Safety Net (Dinerstein et al., 2020). The potential in the Upper Great Lakes region is also significant, where continued forest recovery in existing forests could add substantial amounts of carbon storage (Rhemtulla et al., 2009).

2.3. Impacts on human health and well-being

With more than 50 million acres of U.S. forests projected to be developed over the next 50 years (Thompson, 2006), forest-clearing for early-successional habitats risks further loss of vital natural green space and threatens the stability of regional temperature and water cycles. All of these have impacts on communities. There is an increasing recognition that natural ecosystems offer the public numerous benefits to physical, mental, and spiritual health, as well as social well-being (Karjalainen et al., 2010; Berman et al., 2012; Buttke et al., 2014; Newman and Cragg, 2016; Hansen et al., 2017; Watson et al., 2018; Connecticut Department of Energy and Environmental Protection, 2020). Adolescents may benefit more from natural woodlands than other types of green space in terms of cognitive development and reduced emotional and behavioral problems (Maes et al., 2021). Natural areas are important places to avoid human-related noise and listen to sounds of the natural world, which can decrease pain, lower stress, improve mood, and enhance cognitive performance (Bratman et al., 2015; Buxton et al., 2021).

Protecting intact habitats as refuges for people—even small areas—aligns with the principles of "harm reduction"—practical strategies and ideas aimed at reducing negative consequences. Increasing the well-being of a community, and avoiding or minimizing negative consequences of heat stress, acute physical and mental stress, and a long-term sense of loss can prevent a more serious or chronic condition, particularly in vulnerable populations such as adolescents, pregnant women, seniors, veterans, and those in recovery (Wang et al., 2019; Tiako et al., 2021). The positive impacts of nature on the promotion of mental health has enormous economic benefits (Bratman et al., 2019) and as does preventing mental illness (The Lancet Global Health, 2020).

In addition to social well-being, nature-based outdoor recreation can be an important factor in diversifying and stabilizing local economies (Power, 1996; Power, 2001; Haefele et al., 2016). Studies have shown that recreationists prefer spending time in forests and other landscapes that are natural and free of human manipulation (Vining and Tyler, 1999; Dwyer, 2003; Eriksson et al., 2012). The positive economic effects of robust ecotourism and increased property values can benefit an entire community (Morton, 1998; Lorah and Southwick, 2003; Holmes and Hecox, 2004; Phillips, 2004; Rasker et al., 2013; Fernandez et al., 2018; Cullinane et al., 2022).

In contrast, clearing forests to expand early-successional habitat can threaten human health. For example, it provides optimal habitat for White-tailed Deer and White-footed Mouse

10.3389/ffgc.2022.1073677

(*Peromyscus leucopus*)—the most competent hosts for the vector of Lyme disease, the Eastern Blacklegged Tick (*Ixodes scapularis*) (Allan et al., 2003; LoGiudice et al., 2003; Levi et al., 2012; Telford, 2017; DellaSala et al., 2018; Robertson et al., 2019). There were 185 deaths from auto collisions with animals in 2019 and an estimated 2.1 million animal collision insurance claims in 2020–21, up 7.2 percent from the previous year, with most collisions involving deer (Insurance Information Institute, 2021).

3. Options and alternatives

As discussed above, forest-clearing projects across the Northeast and Upper Great Lakes are proceeding without well-founded consideration of conditions before European settlement, long-term plans for experimental controls and monitoring, or baseline ecological inventories. Assessments made of potential harm to non-target species are cursory, incomplete, or outdated. Quantifiable negative impacts—such as the spread of invasive species, elevated temperatures, increased fire and flood risk, destabilized and decreased climate mitigation and adaptation, degradation of healthy public green spaces, and ongoing expenditures of time and resources—are frequently overlooked. Meanwhile, potentially imperiled interior and old-growth forest species often do not receive adequate attention. Such chronic knowledge gaps render scientific assessment of the impacts of early-successional habitat projects difficult or impossible. Major interdisciplinary reports (Connecticut Department of Energy and Environmental Protection, 2020) offer a strong rationale for addressing these gaps by devoting significant funding to balancing these priorities, to monitoring, comprehensive ecological inventories, and to strengthening management standards and guidelines.

Reassessing the current forest-clearing campaign is an urgent priority: negative impacts are immediate, and once a forest has been cleared or fragmented it takes a century or more to begin to recover a mature or old-growth condition. This is far too late to address the biodiversity, climate, and public health crises that we face in the next critical decades. There are multiple compelling arguments for a new approach that greatly expands wildland preserves while maintaining needed amounts of early-successional habitats and timber production.

3.1. The importance of parks and preserves

There is growing international recognition that the preservation of mature and old-growth forests is essential to address the dual global crises of biodiversity loss and climate change, as well as to promote public health and well-being (Zhou et al., 2006; Luyssaert et al., 2008; Gilhen-Baker et al., 2022; Law et al., 2022). However, in their drive to expand early-successional habitats, land managers have relegated the recovery and protection of old-growth forests to a tiny fraction of their pre-European extent (New Jersey Department of Environmental Protection, 2017; Massachusetts Division of Fisheries and Wildlife, 2022b). The U.S. Forest Service and Bureau of Land Management together administer the largest remaining tracts of mature and old-growth forests in the U.S., yet they are only now beginning a process to inventory these forests (The White House, 2022). Nationally, only about 24% of these forests are protected from logging (DellaSala et al., 2022a).

An extensive system of large, diverse, and connected parks and preserves can help address this challenge (Noss, 1983; Noss et al., 1999; Wuerthner et al., 2015). Studies show that eastern national parks tend to have larger trees, older forests, and more standing deadwood than surrounding managed forests (Miller et al., 2016). They also have greater tree species richness and a higher percentage of rare tree species (Miller et al., 2018). Cool interior forests such as those in parks and other preserves provide shelter for species that are most sensitive to temperature increases (Betts et al., 2017; Betts et al., 2022; Kim et al., 2022; Xu et al., 2022). Protected forests provide important climate benefits in accumulated carbon and avoided greenhouse gas emissions, and the potential to significantly increase carbon storage (Depro et al., 2008; Keeton et al., 2011; Zheng et al., 2013; McGarvey et al., 2015; Brown et al., 2018; Williams et al., 2021; Law et al., 2022). In addition, parks and preserves directly benefit people by producing clean air and water, reducing flooding, preventing soil erosion, cooling surrounding areas, and buffering damage from sea level rise (Luedke, 2019).

Climate scientists and conservation biologists around the world agree that a major expansion of nature preserves is necessary to address the threats of species extinctions and climate change (Di Marco et al., 2019; Yeo et al., 2019; Barber et al., 2020; FAO and UNEP, 2020; Bradshaw et al., 2021). There is a broad consensus that this requires the permanent protection of at least 30% of the Earth by 2030 (Noss et al., 2012; Dinerstein et al., 2019; Rosa and Malcom, 2020; Thompson and Walls, 2021). The U.S. falls far short of meeting this goal. Only about 8% of the U.S. land base now has protection from resource extraction and development equivalent to the U.S. Geological Survey's GAP 1 level and less than 5% meets GAP 2 standards; the vast majority of these lands are in Alaska and the West (Scott et al., 2001; Aycrigg et al., 2013; Jenkins et al., 2015; Lee-Ashley, 2019; Rosa and Malcom, 2020; Thompson and Walls, 2021; U.S. Geological Survey, 2022a,b). As noted previously, most old-growth forests in the U.S. have no formal protection, even on many GAP 2 public lands, leaving their future uncertain (DellaSala et al., 2022b).

The Northeast and Upper Great Lakes regions are deficient in natural area protection (Scott et al., 2001; Anderson and Olivero Sheldon, 2011; Foster et al., 2023). There are a few

10.3389/ffgc.2022.1073677

notable exceptions, such as the Boundary Waters Canoe Area Wilderness, Isle Royale National Park, Adirondack Forest Preserve, and Baxter State Park, which meet GAP 1 standards (U.S. Geological Survey, 2022a,b). However, less than 1% of the Northeast and Upper Great Lakes regions is estimated to meet this strict level of protection U.S. Geological Survey (2022a). This percentage could be greatly increased through an expanded network of parks and preserves on large tracts of federal and state public lands, and could include key undeveloped private lands acquired from willing sellers (Foster et al., 2017; Meyer et al., 2022; Office of Senator Angus King, 2022). This would have numerous outsized benefits; for example, one study estimated that protected forests cover about 5% of the Northeast (including Virginia) yet store 30% of the aboveground carbon in the region (Lu et al., 2013). New wildland preserves would promote the recovery of mature and old-growth forest ecosystems and provide habitats for wide-ranging imperiled wildlife such as the Gray Wolf (*Canis lupus*) and Canada Lynx (*Lynx Canadensis*). They would also offer natural green space to tens of millions of people in major urban communities, reducing pressure on the few existing protected areas (Rhode Island Division of Statewide Planning and Rhode Island Department of Environmental Management, 2019; Reynolds, 2021).

There is ample evidence that expanded wildland preserves governed by natural disturbance regimes would provide early-successional habitats at least equivalent to the natural conditions in which native species evolved. For example, "On reserved forest land in New York [i.e., primarily the "forever wild" Adirondack and Catskill Preserves]... 3 percent [of forest area is] in seedling/sapling and non-stocked stands" (Widmann et al., 2015). Consistent with this, it is estimated that the proportion of the landscape before European settlement "in seedling-sapling forest habitat ranged from 1 to 3% in northern hardwood forests [i.e., beech-birch-maple-hemlock] of the interior upland" (Lorimer and White, 2003).

3.2. Protecting and restoring natural forest ecosystems

The most common strategy for creating early-successional habitats is to clear established forest tracts, purportedly to simulate the continually "shifting mosaic" of patches across a natural landscape (Schlossberg and King, 2007; Smith, 2017; Massachusetts Division of Fisheries and Wildlife, 2022a). However, as discussed above, forest-clearing is not equivalent to natural disturbances; it has significant costs in biodiversity, carbon accumulation, and other ecosystem services; and reduces the possibility of recovering old-growth forest ecosystems dramatically. Moreover, unlike the conservation of mature and old-growth forests, creating and/or maintaining (every 10-12 years) early-successional habitats requires a permanent, resource-consuming commitment of intensive management to replace openings lost to forest succession (DeGraaf and Yamasaki, 2003; Askins, 2011; Bakermans et al., 2011; Yamasaki et al., 2014). This does not take into consideration the mitigation and remediation of unintended environmental side effects: such artificially created "restoration" areas are expensive to maintain (Oehler, 2003; Schlossberg and King, 2007) and there is no assurance that adequate funding will continue to be available. These are serious disadvantages that argue against the current forest-clearing of established natural forest ecosystems.

Among these different perspectives, there is a more balanced alternative: protect and recover mature and old-growth forests wherever possible, quantify the true extent of early-successional habitat and focus maintenance on ecologically suitable lands, including private lands, and encourage efforts to increase protection the full range of natural ecosystems on private lands. At this time there is no indication that this approach is receiving serious consideration from land managers. Yet the likelihood of significant benefits and greatly reduced costs are a compelling argument for such consideration.

4. Discussion

We evaluated peer-reviewed papers, published research, agency reports, and other materials related to a campaign that is focused on expanding early-successional habitats in the Northeast and Upper Great Lakes regions. Each year, this campaign is clearing thousands of acres of established forests. Conversely, the protection of old-growth forests and unmanaged mature forests is currently relegated to a tiny fraction of the land base.

Overall, the forest-clearing campaign is based on two main rationales, which are both open to serious questions and alternative hypotheses:

The primary rationale is that the decline of a number of early-successional species is a pervasive and potentially existential threat. Yet, the baseline for measuring this decline almost invariably begins in the late 1960s, when populations had begun to decrease from abnormally high levels as forests recovered from past clearing. Relying on an artificial baseline that reaches back only 60 years, in an ecosystem where most tree species live for hundreds of years, and during a regional recovery from widespread and intensive land clearing, is fraught with problems. Moreover, it is questionable that any species in these regions needs artificial expansion of early-successional forest habitats to survive and thrive across its multi-state range. Other than limited surveys of birds, game species, and endangered species, there is no reliable information on wildlife populations before the arrival of Europeans, no comprehensive census of forest species even today, and no long-term analysis that

10.3389/ffgc.2022.1073677

systematically estimates wildlife population trends over the last several hundred years.

A second major rationale is that early-successional habitats have dwindled dangerously, have already fallen below the levels that existed before European settlement, and are not being adequately replenished—thereby endangering native biodiversity. However, there is ample evidence that these habitats remain plentiful across these regions (and are likely more prevalent than is accounted for currently), are considerably more abundant than presettlement, and continue to be created by natural and human disturbances—including by mounting climate change impacts. Although early-successional habitats were maintained to some extent by Native people before the arrival of Europeans, these were limited to areas of high population densities near settlements.

Despite its wide-ranging and long-term implications, the campaign for early-successional forest clearing was formulated by a small number of agency, academic, and special interest professionals, with little comprehensive research and analysis, controlled experimentation, strategic planning, monitoring and evaluation, or public involvement and accountability. This organized and aggressive campaign has confused the public and made it challenging for a range of scientists to engage in an open dialogue about an optimal and balanced approach that prioritizes climate stability, ecosystem integrity and public health. Yet, public awareness has grown regarding the evident impacts of forest-clearing projects on biodiversity, climate change, and natural green spaces and, in turn, so has public opposition to these projects (Ketcham, 2022; Potter, 2022; Whitcomb, 2022).

The Gap Analysis Project (GAP) of the U.S. Geological Survey (2022b) can provide the foundation for a balanced alternative to the current costly, intrusive and controversial approach that prioritizes protecting and sustaining natural systems and processes to the greatest extent possible. We suggest the following.

- Establish a significantly expanded system of public parks, wildland preserves, and connecting corridors across the Northeast and Upper Great Lakes with permanent protection under GAP 1 standards. This would preserve old-growth, mature, and recovering forests and allow them to reach their natural maximum ecological potential. Openlands that were deforested in the past, such as grassy areas and farm fields, would be allowed to recover unimpaired, which would provide ample young forest habitats over the next decade. In parallel, new areas of successional habitat would be created by natural disturbance regimes now, and in the future.
- End the clearing of established forests to create early-successional habitats on lands, such as wildlife refuges, under GAP 2 classification. Instead, focus on conserving grassland, shrubland, and savanna habitats where the landform and soil naturally supports their ecological function and species. Examples include coastal landscapes of southern New England and New York, and the Upper Great Lakes prairie-forest transition zone. Re-establish natural disturbance regimes to the extent possible, but allow targeted forest management where appropriate.
- Strengthen the protection of GAP 3 "multiple-use" public lands such as national forests, to maintain natural ecosystems, carbon storage, and public access to green spaces to the extent possible. This includes avoiding intensive resource extraction that destroys or permanently impairs the integrity and productivity of natural systems.
- Regarding public and private lands with no formal protection (GAP 4), encourage the conservation of natural ecosystems and species to the extent possible. This includes agricultural lands and other open space with considerable potential to conserve early-successional habitats. These landowners would continue to determine how they manage their lands, but they would be provided with complete and accurate information on the benefits and costs of habitat management alternatives.

Implementing this "natural" alternative would be prudent, cautious, and low cost, and would permanently sustain the full range of native ecosystems. Allowing deforested lands to recover would accumulate much more carbon and avoid the steep carbon loss associated with cutting established forests (Smith et al., 2006; Cook-Patton et al., 2020).

In the face of many challenges, the people of the Northeast, Upper Great Lakes, and beyond are looking to public lands as a major solution to the loss of biodiversity, the threat of climate change, and the need for healthy public green spaces. We can realize this potential by rebalancing the vision for these lands to ensure the recovery and preservation of the full range of native habitats and the wildlife that depend on them—without ongoing intensive human intervention. There has never been a more appropriate time to make such a transition.

Author contributions

MK, JM, and SM developed the original concept and contributed research, writing, and editing of the manuscript. LF, EF, SB, and DF contributed research, writing, and editing of the manuscript. All authors contributed to its completion and approved the submitted version.

Funding

This work was supported by the Eddy Foundation, Forest Carbon Coalition, and Fund for Wild Nature, and Common Stream.

 10.3389/ffgc.2022.1073677

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

All claims expressed in this article are solely those of the authors and do not necessarily represent those of their affiliated organizations, or those of the publisher, the editors and the reviewers. Any product that may be evaluated in this article, or claim that may be made by its manufacturer, is not guaranteed or endorsed by the publisher.

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2022.1073677/full#supplementary-material

References

Aalde, H., Gonzalez, P., Gytarsky, M., Krug, T., Kurz, W. A., Ogle, et al. (2006). "Chapter 4. Forest land," in *2006 IPCC guidelines for national greenhouse gas inventories: agriculture, forestry and other land use*, Vol. 4, eds H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe (Kanagawa: IGES).

Abrams, M. D. (1992). Fire and the development of oak forests. *Bioscience* 42, 346–353. doi: 10.2307/1311781

Abrams, M. D. (2005). Prescribing fire in eastern oak forests: Is time running out? *Northern J. Appl. For.* 22, 190–196. doi: 10.1093/njaf/22.3.190

Abrams, M. D., and Copenheaver, C. A. (1999). Temporal variation in species recruitment and dendroecology of an old-growth white oak forest in the Virginia Piedmont, USA. *For. Ecol. Manag.* 124, 275–284. doi: 10.1016/S0378-1127(99)00071-7

Abrams, M. D., and Nowacki, G. J. (2008). Native Americans as active and passive promoters of mast and fruit trees in the eastern USA. *Holocene* 18, 1123–1137. doi: 10.1177/0959683608095581

Abrams, M. D., and Nowacki, G. J. (2020). Native American imprint in palaeoecology. *Nat. Sustain.* 3, 896–897.

Abrams, M. D., Ruffner, C. M., and DeMeo, T. E. (1998). Dendroecology and species co-existence in an old-growth Quercus-Acer-Tilia talus slope forest in the central Appalachians, USA. *For. Ecol. Manag.* 106, 9–18. doi: 10.1016/S0378-1127(97)00234-X

Allan, B. F., Keesing, F., and Ostfeld, R. S. (2003). Effect of forest fragmentation on lyme disease risk. *Conserv. Biol.* 17, 267–272. doi: 10.1046/j.1523-1739.2003.01260.x

Alverson, W. S., Waller, D. M., and Solheim, S. L. (1988). Forests too deer: Edge effects in Northern Wisconsin. *Conserv. Biol.* 2, 348–358. doi: 10.1111/j.1523-1739.1988.tb00199.x

Amaral, K. E., Palace, M., O'Brien, K. M., Fenderson, L. E., and Kovach, A. I. (2016). Anthropogenic habitats facilitate dispersal of an early successional obligate: Implications for restoration of an endangered ecosystem. *PLoS One* 11:e0148842. doi: 10.1371/journal.pone.0148842

American Bird Conservancy (2007). *Top 20 most threatened bird habitats in the U.S.* The Plains, VA: American Bird Conservancy.

American Bird Conservancy (2015) *$10 Mil. Forest restoration project will benefit imperiled golden-winged warbler, 14 January.* Available online at: https://abcbirds.org/article/10-mil-forest-restoration-project-will-benefit-imperiled-golden-winged-warbler/ (accessed November 5, 2022).

Anderson, M. G., and Olivero Sheldon, A. (2011). *Conservation status of fish, wildlife, and natural habitats in the northeast landscape: implementation of the northeast monitoring framework.* Arlington, VA: The Nature Conservancy, Eastern Conservation Science. 289.

Anderson, M. G., Clark, M. M., Cornett, M. W., Hall, K. R., Olivero Sheldon, A., and Prince, J. (2018). *Resilient sites for terrestrial conservation in the great lakes and tallgrass prairie.* Arlington, VA: The Nature Conservancy, Eastern Conservation Science and North America Region.

Anderson, M., Bernstein, S., Lowenstein, F., Smith, N., and Pickering, S. (2004). *Determining the size of eastern forest reserves.* Boston, MA: The Nature Conservancy and Sweet Water Trust.

Aquilué, N., Filotas, É, Craven, D., Fortin, M., Brotons, L., and Messier, C. (2020). Evaluating forest resilience to global threats using functional response traits and network properties. *Ecol. Appl.* 30:e02095. doi: 10.1002/eap.2095

Askins, R. A. (1992). Forest fragmentation and the decline of migratory songbirds. *Bird Observer* 20, 13–21.

Askins, R. A. (1993). "Population trends in Grassland, Shrubland, and forest birds in Eastern North America," in *. Current ornithology*, Vol. 11, ed. D. M. Power (Boston, MA: Springer). doi: 10.1007/978-1-4757-9912-5_1

Askins, R. A. (2000). *Restoring North America's wild birds: lessons from landscape ecology.* New Haven, CT: Yale University Press.

Askins, R. A. (2011). The future of blue-winged and golden-winged warblers in Connecticut. *Connecticut Woodlands* 76, 12–15.

Askins, R. A. (2015). The critical importance of large expanses of continuous 1988 forest for bird conservation in connecticut state of the birds: Protecting and connecting large landscapes. *Biol. Faculty Public* 25, 24–28.

Askins, R. A., Folsom-O'Keefe, C. M., and Hardy, M. C. (2012). Effects of vegetation, corridor width and regional land use on early successional birds on powerline corridors. *PLoS One* 7:e31520. doi: 10.1371/journal.pone.0031520

Aycrigg, J. L., Davidson, A., Svancara, L. K., Gergely, K. J., McKerrow, A., and Scott, J. M. (2013). Representation of ecological systems within the protected areas network of the Continental United States. *PLoS One* 8:e54689. doi: 10.1371/journal.pone.0054689

Baca, A., Larsen, J., Treasure, E., Gavazzi, M., and Walker, N. (2018). *Drought impacts in the southern region: a synopsis of presentations and ideas from the drought adaptation workshop in region 8.* Atlanta, GA: USDA Forest Service. doi: 10.32747/2018.7280913.ch

Bakermans, M. H., Larkin, J. L., Smith, B. W., Fearer, T. M., and Jones, B. C. (2011). *Golden-winged warbler habitat best management practices for forestlands in Maryland and Pennsylvania.* The Plains: American Bird Conservancy, 26.

Balcom, B. J., and Yahner, R. H. (1996). Microhabitat and landscape characteristics associated with the threatened allegheny woodrat. *Conserv. Biol.* 10, 515–525.

Barber, C. V., Petersen, R., Young, V., Mackey, B., and Kormos, C. (2020). *Thenexus report: nature based solutions to the biodiversity and climate crisis. F20 foundations, campaign for nature and SEE foundation.* Available online at: https://wild-heritage.org/wp-content/uploads/2021/01/The-Nexus-Report.pdf (accessed January 22, 2021).

Barrett, S. W., Swetnam, T. W., and Baker, W. L. (2005). Indian fire use: Deflating the legend. *Fire Manag. Today* 31–34.

Belair, E. P., and Ducey, M. J. (2018). Patterns in forest harvesting in New England and New York: Using FIA data to evaluate silvicultural outcomes. *J. For.* 116, 273–282. doi: 10.1093/jofore/fvx019

Berman, M. G., Kross, E., Krpan, K. M., Askren, M. K., Aleah Burson, A., Deldin, P. J., et al. (2012). Interacting with nature improves cognition and affect for individuals with depression. *J Affect Disord.* 140, 300–305. doi: 10.1016/j.jad.2012.03.012

Bertin, R. (1977). Breeding habitats of the wood thrush and veery. *Condor* 79, 303–311. doi: 10.1007/s00442-005-0340-9

Betts, M. G., Phalan, B., Frey, S. J. K., Rousseau, J. S., and Yang, Z. (2017). Old-growth forests buffer climate- sensitive bird populations from warming. *Divers Distrib.* 24, 439-447. doi: 10.1111/ddi.12688

Betts, M. G., Yang, Z., Hadley, A. S., Smith, A. C., Rousseau, J. S., Northrup, J. M., et al. (2022). Forest degradation drives widespread avian habitat and population declines. *Nat. Ecol. Evol.* 6, 709-719. doi: 10.1038/s41559-022-01737-8

Boose, E. R., Chamberlin, K. E., and Foster, D. R. (2001). Landscape and regional impacts of hurricanes in New England. *Ecol. Monogr.* 71, 27-48. doi: 10.1890/0012-96152001071[0027:larioh]2.0.co;2

Bradshaw, C. J. A., Ehrlich, P. R., Beattie, A., Ceballos, G., Crist, E., Diamond, J., et al. (2021). Underestimating the challenges of avoiding a ghastly future. *Front. Conserv. Sci* 13:615419. doi: 10.3389/fcosc.2020.615419

Brady, S. J. (2007). "Chapter 1: Effects of cropland conservation practices on fish and wildlife habitat," in *Fish and wildlife response to farm bill conservation practices, in technical review 07-1* (Bethesda, MD: The Wildlife Society).

Bratman, G. N., Anderson, C. B., Berman, J. G., Cochran, B., de Vries, S., Flanders, J., et al. (2019). Nature and mental health: An ecosystem service perspective. *Sci. Adv.* 5:eaax0903. doi: 10.1126/sciadv.aax0903

Bratman, G. N., Hamilton, J. P., Hahn, K. S., Daily, G. C., and Gross, J. J. (2015). Nature experience reduces rumination and subgenual prefrontal cortex activation. *Proc. Natl. Acad. Sci. USA* 112, 8567-8572. doi: 10.1073/pnas.1510459112

Brose, P., Schuler, T., Van Lear, D., and Berst, J. (2001). Bringing fire back the changing regimes of the Appalachian mixed-oak forest. *J. For.* 99, 30-35.

Brown, M. L., Canham, C. D., Murphy, L., and Donovan, T. M. (2018). Timber harvest as the predominant disturbance regime in northeastern U.S. forests: Effects of harvest intensification. *Ecosphere* 9:e02062. doi: 10.1002/ecs2.2062

Bruchac, M. (2004). Native land use and settlements in the Northeastern Woodlands. Raid on deerfield: the many stories of 1704. Philadelphia, PA: University of Pennsylvania Department of Anthropology.

Buchwald, E. (2005). "A hierarchical terminology for more or less natural forests in relation to sustainable management and biodiversity conservation," in *Proceedings of the third expert meeting on harmonizing forest-related definitions*, (Rome), 11-19.

Buckley, G. L. (2010). *America's conservation impulse*. Charlottesville, VA: The University of Virginia Press.

Bulluck, L. P., and Buehler, D. A. (2006). Avian use of early successional habitats: Are regenerating forests, utility right-of-ways and reclaimed surface mines the same? *For. Ecol. Manag.* 236, 76-84. doi: 10.1016/j.foreco.2006.08.337

Buotte, P. C., Law, B. E., Ripple, W. J., and Berner, L. T. (2019). Carbon sequestration and biodiversity co-benefits of preserving forests in the western United States. *Ecol. Appl.* 30:e02039. doi: 10.1002/eap.2039

Burch, E., Culver, D. C., Alonzo, M., and Malloy, E. J. (2022). Landscape features and forest maturity promote the occurrence of macroinvertebrates specialized for seepage springs in urban forests in Washington, DC. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 32, 922-929. doi: 10.1002/aqc.3803

Buttke, D., Allen, D., and Higgins, C. (2014). Benefits of biodiversity to human health and well-being. *Park Sci.* 31, 24-29.

Buxton, R. T., Pearson, A. L., Allou, C., Fristrup, K., and Wittemyer, G. (2021). A synthesis of health benefits of natural sounds and their distribution in national parks. *Proc. Natl. Acad. Sci. USA* 118:e2013097118. doi: 10.1073/pnas.2013097118

Cachat-Schilling, N. (2021). *Fire and myths about northeast native land stewardship. Massachusetts ethical archaeology Society, 31 March.* Available online at: https://www.ethicarch.org/post/fire-and-myths-about-northeast-native-land-stewardship (accessed November 30, 2022).

Caron, S. (2009). *Managing your woodland for white-tailed deer*. St. Paul, MN: Minnesota Department of Natural Resources.

Catanzaro, P., and D'Amato, A. (2019). *Forest carbon: an essential natural solution for climate change*. Amherst, MA: University of Massachusetts Amherst and University of Vermont.

Chandler, C. C., King, D. I., and Chandler, R. B. (2012). Do mature forest birds prefer early-successional habitat during the post-fledging period? *For. Ecol. Manag.* 264, 1-9. doi: 10.1016/j.foreco.2011.09.018

Chapman, J. A., Cramer, K. L., Dippenaar, N. J., and Robinson, T. J. (1992). Systematics and biogeography of the New England cottontail, Sylvilagus transitionalis (Bangs, 1895), with the description of a new species from the Appalachian mountains. *Proc. Biol. Soc. Washington* 105, 841-866.

Chen, I., Hill, J. K., Ohlemüller, R., Roy, D. B., and Thomas, C. D. (2011). Rapid range shifts of species associated with high levels of climate warming. *Science* 333, 1024-1026. doi: 10.1126/science.1206432

Chilton, E. S. (2002). "Towns they have none:" diverse subsistence and settlement strategies in native New England," in *Northeast subsistence-settlement change A.D. 700-1300*, eds J. P. Hart and C. Reith (Albany, NY: New York State Museum), 289-300.

Clark, K. H., and Patterson, W. A. III. (2003). *Fire management plan for montague plain wildlife management area*. Amherst, MA: Department of Natural Resources Conservation, University of Massachusetts, 48.

Climate Change Response Network (2022a). *U.S. fish and wildlife service and university of vermont: Nulhegan Basin, Silvio O. Conte national fish and wildlife refuge adaptation demonstration project*. Houghton, MI: Northern Institute of Applied Climate Science.

Climate Change Response Network (2022b). *Chippewa national forest: adaptive silviculture for climate change (ASCC)*. Houghton, MI: Northern Institute of Applied Climate Science.

Cochrane, T. S., and Iltis, H. H. (2000). *Atlas of the wisconsin prairie and savanna flora. Wisconsin department of natural resources technical bulletin No. 191*. Madison, WI: 226.

Cogbill, C. V. (2000). Vegetation of the presettlement forests of Northern New England and New York. *Rhodora* 102, 250-276.

Cogbill, C. V., Burk, J., and Motzkin, G. (2002). The forests of presettlement New England, USA: Spatial and compositional patterns based on town proprietor surveys. *J. Biogeogr.* 29, 1279-1304. doi: 10.1046/j.1365-2699.2002.00757.x

Collins, S., Merkley, J., Ayotte, K., Baldwin, T., Blunt, A., Brown, S., et al. (2015). *Letter to gina mccarthy, ernest moniz, and tom vilsack supporting biomass energy, 30 June*. Available online at: http://mediad.publicbroadcasting.net/p/mpbn/files/collinsletter.pdf (accessed November 4, 2022).

Connecticut Department of Energy and Environmental Protection (2013). *The clear cut advantage for wildlife and forest health*. Connecticut: Connecticut Department of Energy and Environmental Protection.

Connecticut Department of Energy and Environmental Protection (2015). *From the director's desk. Connecticut wildlife, September/October. Connecticut department of energy and environmental protection*. Available online at: https://portal.ct.gov/-/media/DEEP/wildlife/pdf_files/outreach/connecticut_wildlife_magazine/cwso15pdf.pdf (accessed November 5, 2022).

Connecticut Department of Energy and Environmental Protection (2018). *Return of historic species: young forest initiative. Vol.7, No. 1*. Available online at: https://portal.ct.gov/-/media/DEEP/wildlife/pdf_files/habitat/yfshrubinitiative/ynewsletterissue1pdf.pdf (accessed November 5, 2022).

Connecticut Department of Energy and Environmental Protection (2019). *Shrubland bird monitoring*. Available online at: https://www.ct.gov/deep/cwp/view.asp?a=2723&q=594738&deepNav_GID=1655 (accessed November 5, 2022).

Connecticut Department of Energy and Environmental Protection (2020). *The governor's council on climate change (GC3) science and technology working group final phase 1 report*. Available online at: https://portal.ct.gov/-/media/DEEP/climatechange/GC3/GC3-working-group-reports/GC3-Science-and-Technology-Working-Group-Final-Report-11-19-20.pdf (accessed October 12, 2022).

Connecticut Department of Energy and Environmental Protection (2021b). *Connecticut's young forest habitat initiative*. Available online at: https://portal.ct.gov/DEEP/Wildlife/Habitat/Young-Forest-Habitat-Initiative (accessed November 5, 2022).

Connecticut Department of Energy and Environmental Protection (2021a). *Native American use of prescribed fire*. Available online at: https://portal.ct.gov/DEEP/Forestry/Native-American-Use-of-Prescribed-Fire (accessed November 5, 2022).

Cook-Patton, S. C., Leavitt, S. M., Gibbs, D., Harris, N. L., Lister, K., Anderson-Teixeira, K. J., et al. (2020). Mapping carbon accumulation potential from global natural forest regrowth. *Nature* 585, 545-550. doi: 10.1038/s41586-020-2686-x

Cooper-Ellis, S., Foster, D. R., Carlton, G., and Lezberg, A. (1999). Forest response to catastrophic wind: Results from an experimental hurricane. *Ecology* 80, 2683-2696. doi: 10.1890/0012-9658(1999)080[2683:FRTCWR]2.0.CO;2

Cottam, G., and Loucks, O. L. (1965). *Early vegetation of Wisconsin University of Wisconsin. Extension Geological and natural history survey department of botany. The University of Wisconsin*. Available online at: https://wgnhs.wisc.edu/pubshare/M035.pdf (accessed November 5, 2022).

Cousineau, M. (2017). *NH fish and game to take ownership of cottontail habitat. Union Leader, 2 October*. Available online at: https://newenglandcottontail.org/news/nh-fish-and-game-take-ownership-cottontail-habitat (accessed November 5, 2022).

Coyle, D. R., Nagendra, U. J., Taylor, M. K., Campbell, J. H., Cunard, C. E., Joslin, A. H., et al. (2017). Soil fauna responses to natural disturbances, invasive species, and global climate change: Current state of the science and a call to action. *Soil Biol. Biochem.* 110, 116-133. doi: 10.1016/j.soilbio.2017.03.008

Cullinane, T. C., Flyr, M., and Koontz, L. (2022). *2021 national park visitor spending effects: economic contributions to local communities, states, and the nation. Natural resource report NPS/NRSS/EQD/NRR—2022/2395.* Fort Collins, CO: National Park Service. doi: 10.36967/nrr-2293346

Curtis, J. T. (1959). *The Vegetation of Wisconsin.* Madison, WI: University of Wisconsin press.

Curtis, P. S., and Gough, C. M. (2018). Forest aging, disturbance and the carbon cycle. *N. Phytol.* 219, 1188–1193. doi: 10.1111/nph.15227

D'Amato, A. W., Bradford, J. B., Fraver, S., and Palik, B. J. (2011). Forest management for mitigation and adaptation: Insights from long-term silvicultural experiments. *For. Ecol. Manag.* 262, 803–816. doi: 10.1016/j.foreco.2011.05.014

D'Amato, A. W., Orwig, D. A., and Foster, D. R. (2006). New estimates of Massachusetts old-growth forests: Useful data for regional conservation and forest reserve planning. *Northeastern Naturalist* 13, 495–506. doi: 10.1656/1092-6194200613[495:NEOMOF]2.0.CO;2

D'Amato, A. W., Orwig, D. A., and Foster, D. R. (2009). Understory vegetation in old-growth and second-growth tsuga canadensis forests in Western Massachusetts. *For. Ecol. Manag.* 257, 1043–1052. doi: 10.1016/j.foreco.2008.11.003

D'Amato, A. W., Orwig, D. A., Foster, D. R., Plotkin, A. B., Schoonmaker, P. K., and Wagner, M. R. (2017). Long-term structural and biomass dynamics of virgin Tsuga canadensis-Pinus strobus forests after hurricane disturbance. *Ecology* 98, 721–733. doi: 10.1002/ecy.1684

Davis, M. B. (2003). *Old growth in the east: a survey. Revised edition.* Mount Vernon, KY: Appalachia-Science in the Public Interest, 40456.

Davis, M. B. (ed.) (1996). *Eastern old-growth forests/prospects for rediscovery and recovery.* Washington, DC: Island Press, 383.

Dawson, D. K., Bently Wigley, T., and Keyser, P. D. (2012). Cerulean warbler technical group: Coordinating international research and conservation. *Ornitol. Neotropical* 23, 273–280.

Day, G. M. (1953). The Indian as an ecological factor in the Northeastern forest. *Ecology* 34, 329–346. doi: 10.2307/1930900

deCalesta, D. S. (1994). Effect of white-tailed deer on songbirds within managed forests in Pennsylvania. *J. Wildl. Manag.* 58, 711–718.

Dechen Quinn, A. C., Williams, D. M., and Porter, W. F. (2013). Landscape structure influences space use by white-tailed deer. *J. Mammal.* 94, 398–407. doi: 10.1644/11-MAMM-A-221.1

DeGraaf, R. M., and Yamasaki, M. (2003). Options for managing early-successional forest and shrubland bird habitats in the Northeastern United States. *For. Ecol. Manag.* 185, 179–191. doi: 10.1016/S0378-1127(03)00254-8

Delcourt, P. A., and Delcourt, H. R. (1998). The influence of prehistoric human-set fires on oak- chestnut forests in the Southern Appalachians. *Castanea* 63, 337–345.

DellaSala, D. A., Baker, B. C., Hanson, C. T., Ruediger, L., and Baker, W. (2022b). Have Western USA fire suppression and megafire active management approaches become a contemporary Sisyphus? *Biol. Conserv.* 268:109499. doi: 10.1016/j.biocon.2022.109499

DellaSala, D. A., Mackey, B., Norman, P., Campbell, C., Comer, P. J., Kormos, C. F., et al. (2022a). Mature and old-growth forests contribute to large-scale conservation targets in the conterminous United States. *Front. For. Glob. Change* 5:979528. doi: 10.3389/ffgc.2022.979528

DellaSala, D. A., Middelveen, M. J., Liegner, K. B., and Luché-Thayer, J. (2018). Lyme disease epidemic increasing globally due to climate change and human activities. *Encycl. Anthropocene* 2, 441–451. doi: 10.1016/B978-0-12-809665-9.10516-6

Denevan, W. M. (1992). The pristine myth: The landscape of the Americas in 1492. *Ann. Assoc. Am. Geogr.* 82, 369–385. doi: 10.1111/j.1467-8306.1992.tb01965.x

Depro, B. M., Murray, B. C., Alig, R. J., and Shanks, A. (2008). Public land, timber harvests, and climate mitigation: Quantifying carbon sequestration potential on U.S. Public Timberlands. *For. Ecol. Manag.* 255, 1122–1134. doi: 10.1016/j.foreco.2007.10.036

Derosier, A. L., Hanshue, S. K., Wehrly, K. E., Farkas, J. K., and Nichols, M. J. (2015). *Michigan's wildlife action plan: young forests.* Lansing: Michigan Department of Natural Resources.

Di Marco, M., Ferrier, S., Harwood, T. D., Hoskins, A. J., and Watson, J. E. M. (2019). Wilderness areas halve the extinction risk of terrestrial biodiversity. *Nature* 573, 582–585. doi: 10.1038/s41586-019-1567-7

Dinerstein, E., Joshi, A. R., Vynne, C., Lee, A. T. L., Pharand-Deschênes, F., França, M., et al. (2020). A "Global Safety Net" to reverse biodiversity loss and stabilize Earth's climate. *Sci. Adv.* 6:eabb2824. doi: 10.1126/sciadv.abb2824

Dinerstein, E., Vynne, C., Sala, E., Joshi, A. R., Fernando, S., Lovejoy, T. E., et al. (2019). A global deal for nature: Guiding principles, milestones, and targets. *Sci. Adv.* 5:eaaw2869. doi: 10.1126/sciadv.aaw2869

Domke, G., Williams, C. A., Birdsey, R., Coulston, J., Finzi, A., Gough, C., et al. (2018). "Chapter 9: Forests," in *Second state of the carbon cycle report (SOCCR2): A sustained assessment report,* eds N. Cavallaro, G. Shrestha, R. Birdsey, M. A. Mayes, R. G. Najjar, S. C. Reed, et al. (Washington, DC: U.S. Global Change Research Program), 365–398. doi: 10.7930/SOCCR2.2018.Ch9

Dorazio, R. M., Connor, E. F., and Askins, R. A. (2015). Estimating the effects of habitat and biological interactions in an avian community. *PLoS One* 10:e0135987. doi: 10.1371/journal.pone.0135987

Ducey, M. J., Whitman, A. A., and Gunn, J. (2013). Late-successional and old-growth forests in the northeastern United States: Structure, dynamics, and prospects for restoration. *Forests* 4, 1055–1086. doi: 10.3390/f4041055

Duffy, D. C., and Meier, A. J. (1992). Do appalachian herbaceous understories ever recover from clearcutting? *Conserv. Biol.* 6, 196–201. doi: 10.1046/j.1523-1739.1992.620196.x

Dunn, E. H., Francis, C. M., Blancher, P. J., Drennan, S. R., Howe, M. A., Lepage, D., et al. (2005). Enhancing the scientific value of the christmas bird count. *Auk* 122, 338–346. doi: 10.1642/0004-8038(2005)122[0338:ETSVOT]2.0.CO;2

Dunn, E., Johnson, D. H., Jones, S. L., O'Connor, R. J., Petit, D., Pollock, K., et al. (2000). *A programmatic review of the North American breeding bird survey: Report of a peer review panel to USGS Patuxent.* Laurel, MD: USGS Patuxent Wildlife Research Center.

Dunwiddie, P. W., and Leverett, R. T. (1996). Survey of old-growth forests in Massachusetts. *Rhodora* 98, 419–444.

Dunwiddie, P. W., Foster, D. R., Leopold, D. J., and Leverett, R. T. (1996). "Old-growth forests of southern New England, New York, and Pennsylvania," in *Eastern old-growth forests: prospects for rediscovery and recovery,* ed. M. B. Davis (Washington, D.C: Island Press), 126-143.

Duveneck, M. J., and Thompson, H. R. (2019). Social and biophysical determinants of future forest conditions in New England: Effects of a modern land-use regime. *Glob. Environ. Change* 55, 115–129. doi: 10.1016/j.gloenvcha.2019.01.009

Dwyer, J. F. (2003). "Urban perceptions of national forests: Three examples from the Northern United States," in *Proceedings of the 2002 northeastern recreation research symposium. Gen. Tech. Rep. NE-302,* eds Schuster, Rudy, Comp, (Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station), 159–162.

Eriksson, L., Nordlund, A. M., Olsson, O., and Westin, K. (2012). Recreation in different forest settings: A scene preference study. *Forests* 3, 923–943. doi: 10.3390/f3040923

Eschtruth, A. K., and Battles, J. J. (2009). Assessing the relative importance of disturbance, herbivory, diversity, and propagule pressure in exotic plant invasion. *Ecol. Monogr.* 265–280. doi: 10.1890/08-0221.1

Estes, J. A., Terborgh, J., Brashares, J. S., Power, M. E., Berger, J., Bond, W. J., et al. (2011). Trophic downgrading of planet earth. *Science* 333, 301–306. doi: 10.1126/science.1205106

Faaborg, J., Brittingham, M., Donovan, T., and Blake, J. (1993). "Habitat fragmentation in the temperate zone: A perspective for managers," in *Status and management of neotropical migratory birds: September 21-25, 1992, Estes Park, Colorado. Gen. Tech. Rep. RM-229,* eds Finch, M. Deborah, Stangel, and W. Peter (Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station, U.S. Dept. of Agriculture, Forest Service), 331–338.

Fahey, T. J., Siccama, T. G., Driscoll, C. T., Likens, G. E., Campbell, J., Johnson, C. E., et al. (2005). The biogeochemistry of carbon at hubbard brook. *Biogeochemistry* 75, 109–176. doi: 10.1007/s10533-004-6321-y

Faison, E. K., Foster, D. R., and Oswald, W. W. (2006). Early Holocene openlands in southern New England. *Ecology* 87, 2537–2547. doi: 10.1890/0012-9658200687[2537:EHOISN]2.0.CO;2

Faison, E., DeStefano, S., Foster, D., and Barker Plotkin, A. (2016). Functional response of ungulate browsers in disturbed eastern hemlock forests. *For. Ecol. Manag.* 362, 177–183. doi: 10.1016/j.foreco.2015.12.006

FAO (2020). *Global forest resources assessment 2020: main report.* Rome: FAO. doi: 10.4060/ca9825en

FAO, and UNEP (2020). *The state of the world's forests 2020. Forests, biodiversity and people.* Rome: FAO. doi: 10.4060/ca8642en

Farmer, R. G., Leonard, M. L., Mills Flemming, J. E., and Anderson, S. C. (2014). Observer aging and long-term avian survey data quality. *Ecol Evol.* 4, 2563–2576. doi: 10.1002/ece3.1101

10.3389/ffgc.2022.1073677

Fergus, C. (2014). *The young forest project: helping wildlife through stewardship and science*. Washington, DC: Wildlife Management Institute.

Fernandez, L., Mukherjee, M., and Scott, T. (2018). The effect of conservation policy and varied open space on residential property values: A dynamic hedonic analysis. *Land Use Policy* 73, 480–487. doi: 10.1016/j.landusepol.2017.12.058

Finzi, A. C., Giasson, M., Barker Plotkin, A. A., Aber, J. D., Boose, E. R., Eric, A., et al. (2020). Carbon budget of the harvard forest long-term ecological research site: Pattern, process, and response to global change. *Ecol. Monogr.* 90:e01423. doi: 10.1002/ecm.1423

Fletcher, R. A., Brooks, R. K., Lakoba, V. T., Sharma, G., Heminger, A. R., Dickinson, C. C., et al. (2019). Invasive plants negatively impact native, but not exotic, animals. *Glob Change Biol.* 25, 1–12. doi: 10.1111/gcb.14752

Forman, R. T. T., and Boerner, R. E. (1981). Fire frequency and the pine barrens of New Jersey. *Bull. Torrey Botanical Club* 108, 34–50. doi: 10.2307/2484334

Forman, R. T. T., and Russell, E. W. B. (1983). Evaluation of historical data in ecology. *Bull. Ecol. Soc. Am.* 64, 5–7.

Foster, D. R. (1995). "Land-use history and four hundred years of vegetation change in New England," in *Global land use change: a perspective from the columbian encounter, SCOPE publication*, eds B. L. Turner, A. G. Sal, F. G. Bernaldez, and F. DiCastri (Madrid: Consejo Superior de Investigaciones Cientificas).

Foster, D. R. (2002). Thoreau's country: A historical-ecological perspective on conservation in the New England landscape. *J. Biogeogr.* 29, 1537–1555. doi: 10.1046/j.1365-2699.2002.00786.x

Foster, D. R. (2017). *A meeting of land and sea. Nature and the future of Martha's Vineyard*. New Haven, CT: Yale University Press, 352.

Foster, D. R., and Motzkin, G. (2003). Interpreting and conserving the openland habitats of coastal New England. *For. Ecol. Manag.* 185, 127–150. doi: 10.1016/S0378-1127(03)00251-2

Foster, D. R., and Orwig, D. A. (2006). Preemptive and salvage harvesting of New England forests: When doing nothing is a viable alternative. *Conserv. Biol.* 20, 959–970. doi: 10.1111/j.1523-1739.2006.00495.x

Foster, D. R., Johnson, E., Hall, B., Leibowitz, J., Thompson, E., Donahue, B., et al. (2023). *Wildlands in New England. past, present and future*. Petersham, MA: Harvard Forest, Highstead Foundation and Northeast Wilderness Trust.

Foster, D. R., Lambert, K. F., Kittredge, D., Donahue, B., Hart, C., Labich, W., et al. (2017). *Wildlands and woodlands, farmlands and communities: broadening the vision for New England*. Petersham, MA: Harvard Forest.

Foster, D. R., Motzkin, G., Bernardos, D., and Cardoza, J. (2002). Wildlife dynamics in the changing New England landscape. *J. Biogeogr.* 29, 1337–1357. doi: 10.1046/j.1365-2699.2002.00759.x

Foster, D., Swanson, F., Aber, J., Burke, I., Brokaw, N., Tilman, D., et al. (2003). The importance of land-use legacies to ecology and conservation. *Bioscience* 77–88. doi: 10.1641/0006-3568(2003)053[0077:TIOLUL]2.0.CO;2

Franklin, J. F. (1989). The "new forestry". *J. Soil Water Conserv.* 44:549.

Fraver, S., White, A. S., and Seymour, R. S. (2009). Natural disturbance in an old-growth landscape of Northern Maine, USA. *J. Ecol.* 97, 289–298. doi: 10.1111/j.1365-2745.2008.01474.x

Frelich, L. E. (1995). Old forest in the lake states today and before European settlement. *Natural Areas J.* 15, 157–167.

Frelich, L. E. (2002). *Forest dynamics and disturbance regimes*. Cambridge: Cambridge University Press.

Frelich, L. E., and Reich, P. B. (1995). Spatial patterns and succession in a Minnesota Southern-boreal forest. *Ecol. Monogr.* 65, 325–346. doi: 10.2307/2937063

Frelich, L. E., and Reich, P. B. (2010). Will environmental changes reinforce the impact of global warming on the prairie-forest border of central North America? *Front. Ecol. Environ.* 8:371–378. doi: 10.1890/080191

Frelich, L. E., Blossey, B., Cameron, E. K., Davalos, A., Eisenahuer, N., Fahey, T., et al. (2019). Side swiped: Ecological cascades emanating from earthworm invasion. *Front. Ecol. Environ.* 17:502–510. doi: 10.1002/fee.2099

Frelich, L. E., Lorimer, C. G., and Stambaugh, M. C. (2021). "History and future of fire in hardwood and conifer forests of the Great Lakes-Northeastern forest region, USA," in *Fire ecology and management: past, present, and future of US forested ecosystems*, eds C. H. Greenberg and B. Collins (New York. Springer), 243–285.

Fuller, S., and Tur, A. (2012). *Conservation strategy for the New England Cottontail (Sylvilagus transitionalis). US Fish & wildlife publications. Paper 320*. Washington, DC: U.S. Fish and Wildlife Service.

Gassett, J. (2018). *The young forest initiative - southern appalachian style. Outdoor news bulletin volume 72, Issue 3, 16 Mar*. Available online at: https://wildlifemanagement.institute/outdoor-news-bulletin/march-2018/young-forest-initiative-southern-appalachian-style (accessed November 6, 2022).

Gilhen-Baker, M., Roviello, V., Beresford-Kroeger, D., and Roviello, N. (2022). Old growth forests and large old trees as critical organisms connecting ecosystems and human health. A review. *Environ. Chem. Lett.* 20, 1529–1538. doi: 10.1007/s10311-021-01372-y

Giocomo, J., Vermillion, W., DeMaso, S., and Panjabi, A. (2017). How many are there? Estimating the North American Northern bobwhite population size for conservation planning purposes. *Natl Quail Symposium Proc.* 8:36.

Godt, M. J. W., Hamrick, J. L., and Meier, A. (2004). Genetic diversity in Cymophyllus fraserianus (Cyperaceae), a rare monotypic Genus. *Genetica* 122, 207–215. doi: 10.1023/b:gene.0000041049.91375.8c

Greeley, W. B. (1925). The relation of geography to timber supply. *Econ. Geogr.* 1:1. doi: 10.2307/140095

Gunn, J. S., Ducey, M. J., Andrew, A., and Whitman, A. A. (2014). Late-successional and old-growth forest carbon temporal dynamics in the Northern Forest (Northeastern USA). *For. Ecol. Manag.* 312, 40–46. doi: 10.1016/j.foreco.2013.10.023

Gunn, J. S., Duceya, M. J., and Belair, E. (2019). Evaluating degradation in a North American temperate forest. *For. Ecol. Manag.* 432, 415–426. doi: 10.1016/j.foreco.2018.09.046

Haefele, M., Loomis, J., and Bilmes, L. J. (2016). Total economic value of US national park service estimated to be $92 billion: Implications for policy. *George Wright Forum* 33, 335–345.

Hagan, J. M., Vander Haegen, W. M., and McKinley, P. S. (1996). The early development of forest fragmentation effects on birds. *Conserv. Biol.* 10, 188–202. doi: 10.1046/j.1523-1739.1996.10010188.x

Hamburg, S. P., Vadeboncoeur, M. A., Johnson, C. E., and Sanderman, J. (2019). Losses of mineral soil carbon largely offset biomass accumulation fifteen years after whole-tree harvest in a northern hardwood forest. *Biogeochemistry* 144, 1–14. doi: 10.1007/s10533-019-00568-3

Hanberry, B. B., and Thompson, F. R. III. (2019). Open forest management for early successional birds. *Wildl. Soc. Bull.* 43, 141–151. doi: 10.1002/wsb.957

Handler, S., Pike, C., and St. Clair, B. (2018). *Assisted migration. USDA Forest service climate change resource center*. Available online at: https://www.fs.usda.gov/ccrc/topics/assisted-migration (accessed November 5, 2011).

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, et al. (2013). High-resolution global maps of 21st-century forest cover change. *Science* 342, 850–853. doi: 10.1126/science.1244693

Hansen, M. M., Jones, R., and Tocchini, K. (2017). Shinrin-Yoku (forest bathing) and nature therapy: A state-of-the-art review. *Int. J. Environ. Res. Public Health* 14:851. doi: 10.3390/ijerph14080851

Harmon, M. E. (2019). *Statement From Dr. Mark E. Harmon, Professor Emeritus to the United States house natural resources committee subcommittee on national parks, forests, and public lands concerning the hearing on climate change and public lands: examining impacts and considering adaptation opportunities committee. Testimony date: 21 February*. Available online at: https://docs.house.gov/meetings/II/II10/20190213/108911/HHRG-116-II10-20190213-SD012.pdf (accessed November 5, 2022).

Harmon, M. E., Ferrell, W. K., and Franklin, J. F. (1990). Effects on carbon storage of conversion of old-growth forests to young forests. *Science* 247, 699–702. doi: 10.1126/science.247.4943.699

Harmon, M. E., Harmon, J. M., Ferrell, W. K., and Brooks, D. (1996). Modeling carbon stores in Oregon and Washington forest products: 1900-1992. *Climatic Change* 33, 1996. doi: 10.1007/BF00141703

Harris, N. L., Hagen, S. C., Saatchi, S. S., Pearson, T. R. H., Woodall, C. W., Domke, G. M., et al. (2016). Attribution of net carbon change by disturbance type across forest lands of the conterminous United States. *Carbon Balance Manag.* 11:24. doi: 10.1186/s13021-016-0066-5

Hassler, K. N., Waggy, C. D., Spinola, R. M., Oxenrider, K. J., Rodgers, R. E., Pearce, K. J., et al. (2021). Den-site selection by Eastern Spotted Skunks in the central Appalachian mountains of West Virginia. *Southeastern Naturalist Southeastern Naturalist* 20, 209–224. doi: 10.1656/058.020.0sp1118

Hawthorne, B. (2020). *Overview of masswildlife carbon analysis. Massachusetts*. Available online at: https://www.mass.gov/doc/overview-of-masswildlife-carbon-analysis/download (accessed November 5, 2022).

Heeter, K. J., Brosi, S. L., and Brewer, G. L. (2019). Dendroecological analysis of xeric, upland, Quercus-dominated old-growth forest within the Ridge and Valley Province of Maryland, USA. *Natural Areas J.* 39, 319–332. doi: 10.3375/043.039.0304

Heinselman, M. L. (1973). Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. *Quaternary Res.* 3, 329–382. doi: 10.1016/0033-5894(73)90003-3

Hilmers, T., Friess, N., Bässler, C., Heurich, M., Brandl, R., Pretzsch, H., et al. (2018). Biodiversity along temperate forest succession. *J. Appl. Ecol.* 55, 2756–2766. doi: 10.1111/1365-2664.13238

Hocking, D. J., Babbitt, K. J., and Yamasaki, M. (2013). Comparison of silvicultural and natural disturbance effects on terrestrial salamanders in Northern Hardwood forests. *Biol. Conserv.* 167, 194–202. doi: 10.1016/j.biocon.2013.08.006

Holmes, F. P., and Hecox, W. E. (2004). Does wilderness impoverish rural regions? *Int. J. Wilderness* 10, 34–39.

Hoover, J., Brittingham, M., and Goodrich, L. (1995). Effects of forest patch size on nestling success of wood thrushes. *Auk* 112, 146–155. doi: 10.2307/4088774

Huijser, M. P., and Clevenger, A. P. (2006). "Chapter 11: Habitat and corridor function of rights-of-way," in *The ecology of transportation: managing mobility for the environment*, eds J. Davenport and J. L. Davenport (Dordrecht: Springer), 233–254. doi: 10.1007/1-4020-4504-2_11

Infrastructure Investment and Jobs Act (2021). *Pub. L. No. 117-58 117 135 Stat. 429*. Available online at: https://www.congress.gov/117/plaws/publ58/PLAW-117publ58.pdf (accessed November 5, 2022).

Insurance Information Institute (2021). *Facts + statistics: Deer VEHICLE COLLISIONS*. Available online at: https://www.iii.org/fact-statistic/facts-statistics-deer-vehicle-collisions (accessed July 29, 2022).

Ingerson, A. (2011). Carbon storage potential of harvested wood: Summary and policy implications. *Mitig Adapt Strateg Glob. Change* 16, 307–323. doi: 10.1007/s11027-010-9267-5

Irland, L. (2013). Extreme value analysis of forest fires from New York to Nova Scotia, 1950-2010. *For. Ecol. Manag.* 294, 150–157. doi: 10.1016/j.foreco.2012.09.004

Irland, L. (2014). *What happened to S. New England fires? They virtually disappeared over a few decades in the Mid 20th Century. The northern logger and timber processor*. Old Forge, NY: Northeastern Loggers' Association.

IUCN (2012). *IUCN Red list categories and criteria, version 3.1, Second Edition. As approved by the 51st meeting of the IUCN Council Gland, Switzerland 9 February 2000*. Gland: IUCN, 32.

James, J., and Harrison, R. (2016). The effect of harvest on forest soil carbon: A meta-analysis. *Forests* 7:308. doi: 10.3390/f7120308

Janowiak, M. K., D'Amato, A. W., Swanston, C. W., Iverson, L., Thompson, F. R. III., Dijak, W., et al. (2018). *New England and Northern New York forest ecosystem vulnerability assessment and synthesis: a report from the New England climate change response framework project. Gen. Tech. Rep. NRS-173*. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station, 234. doi: 10.2737/nrs-gtr-173

Jenkins, C. N., Van Houtan, K. S., Pimm, S. L., and Sexton, J. O. (2015). US protected lands mismatch biodiversity priorities. *Proc. Natl. Acad. Sci. USA* 112, 5081–5086. doi: 10.1073/pnas.1418034112

Jenkins, J., and Kroeger, A. (2020). *Seeing the forest: Sustainable wood bioenergy in the Southeast United States*. Enviva White Paper 1, 5/2020. Bethesda, MD: Enviva Inc.

Jeswiet, S., and Hermsen, L. (2015). *Agriculture and wildlife: a two-way relationship. Catalogue no. 16–002–X ISSN 1913-4320*. Ottawa, ON: Statistics Canada.

Jirinec, V., Cristol, D. A., and Leu, M. (2017). Songbird community varies with deer use in a fragmented landscape. *Landsc. Urban Plan.* 161:1. doi: 10.1016/j.landurbplan.2017.01.003

Johnson, L. B., and Kipfmueller, K. F. (2016). A fire history derived from Pinus resinosa Ait. for the Islands of Eastern Lac La Croix, Minnesota, USA. *Ecol. Appl.* 26, 1030–1046. doi: 10.1890/15-1151

Jones, J. J. (2011). *Transforming the cutover: the establishment of national forests in Northern Michigan*. Durham: Forest History Today, Spring/Fall.

Jordan, K. A. (2013). Incorporation and colonization: Postcolumbian iroquois satellite communities and processes of indigenous autonomy. *Am. Anthropol.* 115, 29–43. doi: 10.1111/j.1548-1433.2012.01533.x

Jordan, S. F., and Black, S. H. (2012). *Effects of Forest land management on terrestrial mollusks: a literature review*. Portland: The Xerces Society for Invertebrate Conservation.

Karjalainen, E., Sarjala, T., and Raitio, H. (2010). Promoting human health through forests: Overview and major challenges. *Environ. Health Prev. Med.* 15, 1–8. doi: 10.1007/s12199-008-0069-2

Keeton, W. S., Whitman, A. A., McGee, G. C., and Goodale, C. L. (2011). Late-successional biomass development in northern hardwood-conifer forests of the Northeastern United States. *For. Sci.* 57:2011. doi: 10.1093/forestscience/57.6.489

Kelley, J. R. Jr., Williamson, S., and Cooper, T. R. (2008). *American woodcock conservation plan: a summary of and recommendations for woodcock conservation in North America. U.S. Fish and Wildlife Publications*. Washington. D.C: U.S. Fish and Wildlife Service, 430.

Ketcham, C. (2022). *Is clear-cutting U.S. forests good for wildlife? National geographic, 24 March*. Available online at: https://www.nationalgeographic.com/environment/article/is-clear-cutting-us-forests-good-for-wildlife (accessed November 6, 2022).

Kilgore, M. A., and Ek, A. R. (2013). *Minnesota forest age-class distribution, 2011. Minnesota forestry research notes, No. 295*. St. Paul, MN: University of Minnesota, Forest Resources Department.

Kim, H., McComb, B. C., Frey, S. J. K., Bell, D. M., and Betts, M. G. (2022). Forest microclimate and composition mediate long-term trends of breeding bird populations. *Glob. Change Biol.* 28, 6180–6193. doi: 10.1111/gcb.16353

King, D. I., and Schlossberg, S. (2014). Synthesis of the conservation value of the early-successional stage in forests of Eastern North America. *For. Ecol. Manag.* 324, 186–195. doi: 10.1016/j.foreco.2013.12.001

King, D. I., Chandler, R. B., Collins, J. M., Petersen, W. R., and Lautzenheiser, T. E. (2009). Effects of width, edge and habitat on the abundance and nesting success of scrub-shrub birds in powerline corridors. *Biol. Conserv.* 2672–2680. doi: 10.1016/j.biocon.2009.06.016

King, D. I., DeGraaf, R. M., and Griffin, C. R. (2001). Productivity of early successional shrubland birds in clearcuts and groupcuts in an eastern deciduous forest. *J. Wildl. Manag.* 65, 345–350. doi: 10.2307/3802914

King, D. I., Schlossberg, S., Brooks, R. T., and Akresh, M. E. (2011). Effects of fuel reduction on birds in pitch pine-scrub oak barrens of the United States. *For. Ecol. Manag.* 261, 10–18. doi: 10.1016/j.foreco.2010.08.039

Kipfmueller, K. F., Schneider, E. A., Weyenberg, S. A., and Johnson, L. B. (2017). Historical drivers of a frequent fire regime in the red pine forests of Voyageurs National Park, MN, USA. *For. Ecol. Manag.* 405, 31–43. doi: 10.1016/j.foreco.2017.09.014

Knapp, W. M., and Wiegand, R. (2014). Orchid (Orchidaceae) decline in the Catoctin Mountains, Frederick County, Maryland as documented by a long-term dataset. *Biodivers. Conserv.* 23, 1965–1976. doi: 10.1007/s10531-014-0698-2

Knowlton, J. (2017). *Continuing the conservation legacy: centennial of the weeks act of 1911. USDA forest service*. Available online at: https://www.usda.gov/media/blog/2011/03/02/continuing-conservation-legacy-centennial-weeks-act-1911 (accessed November 6, 2022).

Koches, J. (2019). *Hurricane hugo and the woodpeckers: the silver lining of a monster storm. U.S. Fish and Wildlife Service, 21 February*. Available online at: https://web.archive.org/web/20220207182013/https://www.fws.gov/southeast/articles/hurricane-hugo-and-the-woodpeckers-the-silver-lining-of-a-monster-storm/ (accessed November 6, 2022).

Kormos, C. F., Mackey, B., DellaSala, D. A., Kumpe, N., Jaeger, T., Mittermeier, R. A., et al. (2018). "Primary forests: definition, status and future prospects for global conservation," in *The Encyclopedia of the Anthropocene*, Vol. 2, eds A. Dominick, DellaSala, I. Michael, and Goldstein (Oxford: Elsevier), 31–41.

Krebs, J., Pontius, J., and Schaberg, P. G. (2017). Modeling the impacts of hemlock woolly adelgid infestation and presalvage harvesting on carbon stocks in northern hemlock forests. *Can. J. For. Res.* 47, 727–734. doi: 10.1139/cjfr-2016-0291

Lacroix, E., Petrenko, C. L., and Friedland, A. J. (2016). Evidence for losses from strongly bound SOM pools after clear cutting in a northern hardwood forest. *Soil Sci.* 181, 202–207. doi: 10.1097/SS.0000000000000147

Lain, E. J., Haney, A., Burris, J. M., and Burton, J. (2008). Response of vegetation and birds to severe wind disturbance and salvage logging in a southern boreal forest. *For. Ecol. Manag.* 256, 863–871. doi: 10.1016/j.foreco.2008.05.018

Lapin, M. (2005). *Old-growth forests: a literature review of the characteristics of eastern North American forests*. Montpelier: Vermont Natural Resources Council.

Lashley, M. A., Harper, C. A., Bates, G. E., and Keyser, P. D. (2011). Forage availability for white-tailed deer following silvicultural treatments in hardwood forests. *J. Wildl. Manag.* 75, 1467–1476. doi: 10.1002/jwmg.176

Law, B. E., Hudiburg, T. W., Berner, T., Kent, J. J., Buotte, P. C., and Harmon, M. E. (2018). Land use strategies to mitigate climate change in carbon dense temperate forests. *Proc. Natl. Acad. Sci. USA* 115, 3663–3668. doi: 10.1073/pnas.1720064115

Law, B. E., Moomaw, W. R., Hudiburg, T. W., Schlesinger, W. H., Sterman, J. D., and Woodwell, G. M. (2022). Creating strategic reserves to protect forest carbon and reduce biodiversity losses in the United States. *Land* 11, 721.

Lawrence, D., Coe, M., Walker, W., Verchot, L. L., and Vandecar, K. (2022). The unseen effects of deforestation: Biophysical effects on climate. *Front. For. Glob. Change* 5:756115. doi: 10.3389/ffgc.2022.756115

LeDoux, C. B., and Martin, D. K. (2013). *Proposed BMPs for invasive plant mitigation during timber harvesting operations. General technical report NRS-118.* Washington, DC: USDA Forest Service, Northern Research Station. doi: 10.2737/NRS-GTR-118

Lee-Ashley, M. (2019). *How much nature should America keep? Center for American progress public lands team and oceans team.* Available online at: https://cdn.americanprogress.org/content/uploads/2019/08/05133927/NatureAmerica-report.pdf (accessed November 6, 2022).

Lenarz, M. S. (1987). Economics of forest openings for white-tailed deer. *Wildl. Soc. Bull.* 15, 568–573.

Leturcq, P. (2020). GHG displacement factors of harvested wood products: The myth of substitution. *Sci. Rep.* 10:20752. doi: 10.1038/s41598-020-77527-8

Leverett, R. T. (1996). "Definitions and history," in *Eastern old-growth forests: Prospects for rediscovery and recovery*, ed. M. B. Davis (Washington, DC: Island Press), 3–17.

Leverett, R. T., Masino, S. A., and Moomaw, W. R. (2021). Older eastern white pine trees and stands sequester carbon for many decades and maximize cumulative carbon. *Front. For. Glob. Change* 4:620450. doi: 10.3389/ffgc.2021.620450

Levi, T., Kilpatrick, A. M., Mangel, M., and Wilmers, C. C. (2012). Deer, predators, and the emergence of lyme disease. *Proc Natl Acad Sci U S A* 109, 10942–10947. doi: 10.1073/pnas.1204536109

Lindenmayer, D. B., and Laurance, W. F. (2012). A history of hubris – cautionary lessons in ecologically sustainable forest management. *Biol. Conserv.* 151, 11–16. doi: 10.1016/j.biocon.2011.10.032

Littlefield, C. E., and D'Amato, A. W. (2022). Identifying trade-offs and opportunities for forest carbon and wildlife using a climate change adaptation lens. *Conserv. Sci. Pract.* 2022:e12631. doi: 10.1111/csp2.12631

Litvaitis, J. A. (1993). Response of early successional vertebrates to historic changes in land use. *Conserv. Biol.* 7, 866–873. doi: 10.1046/j.1523-1739.1993.740866.x

Litvaitis, J. A. (2003). Are pre-Columbian conditions relevant baselines for managed forests in the northeastern United States? *For. Ecol. Manag.* 185, 113–126. doi: 10.1016/S0378-1127(03)00250-0

LoGiudice, K., Ostfeld, R. S., Schmidt, K. A., and Keesing, F. (2003). The ecology of infectious disease: Effects of host diversity and community composition on Lyme disease risk. *Proc. Natl. Acad. Sci. U.S.A.* 100, 567–571. doi: 10.1073/pnas.0233733100

Lombardi, J. V., Mengak, M. T., Castleberry, S. B., and Terrell, V. K. (2017). Mammal occurrence in rock outcrops in shenandoah national park: Ecological and anthropogenic factors influencing trap success and co-occurrence. *Natural Areas J.* 37, 507–514. doi: 10.3375/043.037.0407

Lorah, P., and Southwick, R. (2003). Environmental protection, population change, and economic development in the rural Western United States. *Population Environ.* 24, 255–272. doi: 10.1023/A:1021299011243

Lorimer, C. G. (1977). The presettlement forest and natural disturbance cycle of northeastern Maine. *Ecology* 58, 139–148. doi: 10.2307/1935115

Lorimer, C. G. (1980). Age structure and disturbance history of a southern Appalachian virgin forest. *Ecology* 61, 1169–1184. doi: 10.2307/1936836

Lorimer, C. G., and White, A. S. (2003). Scale and frequency of natural disturbances in the Northeastern US: Implications for early successional forest habitats and regional age distributions. *For. Ecol. Manag.* 185, 41–64. doi: 10.1016/S0378-1127(03)00245-7

Lu, X., Kicklighter, D. W., Melillo, J. M., Yang, P., Rosenzweig, B., Vörösmarty, C. J., et al. (2013). A contemporary carbon balance for the northeast region of the United States. *Environ. Sci. Technol.* 47, 13230–13238. doi: 10.1021/es403097z

Luedke, H. (2019). *Fact sheet: nature as resilient infrastructure: an overview of nature-based solutions. environmental and energy study institute.* Available online at: https://www.eesi.org/files/FactSheet_Nature_Based_Solutions_1016.pdf (accessed November 6, 2022).

Luyssaert, S., Schulze, E. D., Börner, A., Knohl, A., Hessenmöller, D., Beverly, E., et al. (2008). Old-growth forests as global carbon sinks. *Nature* 455, 213–221. doi: 10.1038/nature07276

Lyme Timber Company (2017). *Kunjamuk young forest demonstration project.* Available online at: http://lymetimber.com/wp/wp-content/uploads/2017/12/KunjamukYoungForestFactSheet.pdf (accessed November 6, 2022).

Mackey, B., DellaSala, D. A., Kormos, C., Lindenmayer, D., Kumpel, N., Zimmerman, B., et al. (2014). Policy options for the world's primary forests in multilateral environmental agreements. *Conserv. Lett.* 8, 139–147. doi: 10.1111/conl.12120

Mackey, B., Kormos, C. F., Keith, H., Moomaw, W. R., Houghton, R. A., Mittermeier, R. A., et al. (2020). Understanding the importance of primary tropical forest protection as a mitigation strategy. *Mitig. Adapt Strateg Glob Change* 25, 763–787. doi: 10.1007/s11027-019-09891-4

Mackey, B., Prentice, I., Steffen, W., House, J. I., Lindenmayer, D., Keith, H., et al. (2013). Untangling the confusion around land carbon science and climate change mitigation policy. *Nat. Clim. Change* 3, 552–557. doi: 10.1038/nclimate1804

Maes, M. J. A., Pirani, M., Booth, E. R., Shen, C., Milligan, B., Jones, K. E., et al. (2021). Benefit of woodland and other natural environments for adolescents' cognition and mental health. *Nat. Sustain.* 4, 851–858. doi: 10.1038/s41893-021-00751-1

Makarieva, A. M., Nefiodov, A. V., Morozov, V. E., Aleynikov, A. A., and Vasilov, R. G. (2020). Science in the vanguard of rethinking the role of forests in the third millennium: Comments on the draft concept of the federal law "forest code of the russian federation". *For. Clim. Issues* 3, 1–25. doi: 10.31509/2658-607x-2020-3-3-1-25

Maloof, J. (2023). *Nature's temples: a natural history of old-growth forests revised and expanded.* Princeton, NJ: Princeton University Press, 240.

Mann, C. C. (2005). *1491: new revelations of the Americas before Columbus.* New York, NY: Alfred A. Knopf.

Marks, P. L. (1983). On the origin of the field plants of the Northeastern United States. *Am. Naturalist* 122, 210–228.

Marschner, F. J. (1975). *The original vegetation of Minnesota (map).* St. Paul, MN: USDA Forest Service, North Central Forest Experiment Station.

Marshall, M., DeCecco, J. A., Williams, A. B., Gale, G. A., and Cooper, R. J. (2003). Use of regenerating clearcuts by late-successional bird species and their young during the post-fledging period. *For. Ecol. Manag.* 183, 127–135. doi: 10.1016/S0378-1127(03)00101-4

Martin, W. H. (1992). Characteristics of old-growth mixed mesophytic forests. *Natural Areas J.* 12, 127–135.

Mason, J., Moorman, C., Hess, G., and Sinclair, K. (2007). Designing suburban greenways to provide habitat for forest-breeding birds. *Landsc. Urban Plan.* 80, 153–164. doi: 10.1016/j.landurbplan.2006.07.002

Massachusetts Audubon Society (2013). *State of the birds 2013: Massachusetts breeding birds: a closer look.* Massachusetts: Lincoln.

Massachusetts Department of Conservation and Recreation (2018). *Sykes mountain forest management proposal.* Available online at: https://www.mass.gov/files/documents/2018/02/09/Sykes%20Mountain%20final%20posted.pdf (accessed November 6, 2022).

Massachusetts Department of Conservation and Recreation (2022). *Managing our forests . For carbon benefits.* Available online at: https://www.mass.gov/info-details/managing-our-forests-for-carbon-benefits (accessed November 5, 2022).

Massachusetts Division of Fisheries and Wildlife (2022a). *Wood harvest, mowing, and mulching for habitat management.* Available online at: https://www.mass.gov/service-details/wood-harvest-mowing-and-mulching-for-habitat-management (accessed November 6, 2022).

Massachusetts Division of Fisheries and Wildlife (2022b). *masswildlife's habitat goals.* Available online at: https://www.mass.gov/service-details/masswildlifes-habitat-goals (accessed November 6, 2022).

Matlack, G. R. (2013). Reassessment of the use of fire as a management tool in deciduous forests of Eastern North America. *Conserv. Biol.* 27, 916–926. doi: 10.1111/cobi.12121

McCarthy, B. C., and Bailey, D. R. (1996). Composition, structure, and disturbance history of crabtree woods: An old-growth forest of Western Maryland. *Bull. Torrey Bot. Club* 123, 350–365. doi: 10.2307/2996783

McDonald, R. I., Motzkin, G., and Foster, D. R. (2008). Assessing the influence of historical factors, contemporary processes, and environmental conditions on the distribution of invasive species. *J. Torrey Bot. Soc.* 135, 260–271. doi: 10.3159/08-RA-012.1

McEwan, R. W., Dyer, J. M., and Pederson, N. (2011). Multiple interacting ecosystem drivers: Towards an encompassing hypothesis of oak forest dynamics across eastern North America. *Ecography* 34, 244–256. doi: 10.1111/j.1600-0587.2010.06390.x

McGarvey, J. C., Thompson, J. R., Epstein, H. E., and Shugart, H. H. (2015). Carbon storage in old-growth forests of the Mid-Atlantic: Toward better understanding the eastern forest carbon sink. *Ecology* 96, 311–317. doi: 10.1890/14-1154.1

 10.3389/ffgc.2022.1073677

McGraw, J. B., Lubbers, A. E., Van der Voort, M., Mooney, E. H., Furedi, M. A., Souther, S., et al. (2013). Ecology and conservation of ginseng (Panax quinquefolius) in a changing world. *Ann. N.Y. Acad. Sci.* 1286, 62–91. doi: 10.1111/nyas.12032

McKibben, B. (1995). An explosion of green. *Atlantic Monthly* 275, 61–83.

McKinley, D. C., Ryan, M. G., Birdsey, R. A., Giardina, C. P., Harmon, M. E., Heath, L. S., et al. (2011). A synthesis of current knowledge on forests and carbon storage in the United States. *Ecol. Applic.* 21, 1902–1924. doi: 10.1890/10-0697.1

Melvin, M. A. (2020). *National prescribed fire use report. Technical bulletin 04-20.* Washington, DC: National Association of State Foresters and the Coalition of Prescribed Fire Councils.

Meyer, S. R., MacLeod, K. K., Thompson, J., Macleod, K. K., Foster, D. R., Perschel, R., et al. (2022). *New England's climate imperative: our forests as a natural climate solution.* Redding, CT: Highstead Foundation.

Michigan Department of Natural Resources (2016). *Michigan deer management plan, wildlife division report No. 3626.* Lansing, MI: Michigan Department of Natural Resources.

Michigan Department of Natural Resources (2017). *Deer range improvement program (DRIP) report.* Available online at: https://www.michigan.gov/documents/dnr/DRIP_project_report_607688_7.pdf (accessed November 6, 2022).

Michigan Department of Natural Resources (2022). *Prescribed burns.* Lansing, MI: Michigan Department of Natural Resources.

Mika, A. M., and Keeton, W. S. (2013). Factors contributing to carbon fluxes from bioenergy harvests in the US Northeast: An analysis using field data. *GCB Bioenergy* 5, 290–305. doi: 10.1111/j.1757-1707.2012.01183.x

Millar, C., Stephenson, N. L., and Stephens, S. L. (2007). Climate change and forests of the future: Managing in the face of uncertainty. *Ecol. Applic.* 17, 2145–2151. doi: 10.1890/06-1715.1

Miller, K. M., Dieffenbach, F. W., Campbell, J. P., Cass, W. B., Comiskey, J. A., Matthews, E. R., et al. (2016). National parks in the Eastern United States harbor important older forest structure compared with matrix forests. *Ecosphere* 7:e01404. doi: 10.1002/ecs2.1404

Miller, K. M., McGill, B. J., Mitchell, B. R., Comiskey, J., Dieffenbach, F. W., Matthews, E. R., et al. (2018). Eastern national parks protect greater tree species diversity than unprotected matrix forests. *For. Ecol. Manag.* 414, 74–84. doi: 10.1016/j.foreco.2018.02.018

Miller-Weeks, M., Eagar, C., and Petersen, C. M. (1999). *The Northeastern ice storm 1998, a forest damage assessment for New York, Vermont, New Hampshire, and Maine.* Waterbury, VT: North East State Foresters Association, 32.

Milner, G. R., and Chaplin, G. (2010). Eastern North American population at ca. A.D. 1500. *Am. Antiquity* 75, 707–726. doi: 10.7183/0002-7316.75.4.707

Minnesota Department of Natural Resources (2016). *State wildlife action plans: revitalizing conservation in America.* St. Paul, MN: Minnesota Department of Natural Resources.

Mladenoff, D. J., and Forrester, J. A. (2018). "Historical patterns and contemporary processes in northern lake states old-growth landscapes," in *Ecology and recovery of eastern old-growth forests*, eds A. M. Barton and W. S. Keeton (Washington, DC: Island Press). doi: 10.5822/978-1-61091-891-6_7

Mladenoff, D. J., Schulte, L. A., and Bolliger, J. (2008). "Broad-scale changes in the Northern Forests: From past to present," in *The vanishing present: Wisconsin's changing lands, waters, and wildlife*, eds D. Waller and T. Rooney (Chicago, IL: University of Chicago Press). doi: 10.7208/chicago/9780226871745.003.0005

Moomaw, W. R., Masino, S. A., and Faison, E. K. (2019). Intact forests in the United States: Proforestation mitigates climate change and serves the greatest good. *Front. For. Glob. Change* 11:27. doi: 10.3389/ffgc.2019.00027

Mora, C., Tittensor, D. P., Adl, S., Simpson, A. G. B., and Worm, B. (2011). How many species are there on earth and in the ocean? *PLoS Biol.* 9:e1001127. doi: 10.1371/journal.pbio.1001127

Morton, P. (1998). The economic benefits of wilderness: Theory and practice. *Denver. Law. Rev* 76:465.

Mosseler, A., Major, J. E., and Rajora, O. P. (2003a). Old-growth red spruce forests as reservoirs of genetic diversity and reproductive fitness. *Theor. Appl. Genet.* 106, 931–937. doi: 10.1007/s00122-002-1156-1

Mosseler, A., Thompson, I., and Pendrel, B. A. (2003b). Overview of old-growth forests in Canada from a science perspective. *Environ. Rev.* 11, S1–S7. doi: 10.1139/a03-018

Motzkin, G., and Foster, D. R. (2002). Grasslands, heathlands and shrublands in coastal New England: Historical interpretations and approaches to conservation. *J. Biogeogr.* 29, 1569–1590. doi: 10.1046/j.1365-2699.2002.00769.x

Motzkin, G., Patterson, W. A III., and Foster, D. R. (1999). A historical perspective on pitch pine-scrub oak communities in the Connecticut Valley of Massachusetts. *Ecosystems* 2, 255–273. doi: 10.1007/s100219900073

Moustakis, Y., Papalexiou, S. M., Onof, C. J., and Paschalis, A. (2021). Seasonality, intensity, and duration of rainfall extremes change in a warmer climate. *Earths Fut.* 9:e2020EF001824. doi: 10.1029/2020EF001824

Munoz, S. E., and Gajewski, K. (2010). Distinguishing prehistoric human influence on late-Holocene forests in southern Ontario, Canada. *Holocene* 20, 967–981. doi: 10.1177/0959683610362815

Munoz, S. E., Mladenoff, D. J., Schroeder, S., and Williams, J. W. (2014). Defining the spatial patterns of historical land use associated with the indigenous societies of eastern North America. *J. Biogeogr.* 41, 2195–2210. doi: 10.1111/jbi.12386

Nareff, G. E., Wood, P. B., Brown, D. J., Fearer, T., Larkin, J. L., and Ford, W. M. (2019). Cerulean Warbler (*Setophaga cerulea*) response to operational silviculture in the central Appalachian region. *For. Ecol. Manag.* 448, 409–423. doi: 10.1016/j.foreco.2019.05.062

National Commission on Science for Sustainable Forestry (2007). *Conserving biodiversity through sustainable forestry.* Available online at: https://www.ddcf.org/globalassets/news-and-publications/imported-news-and-publications/conserving-biodiversity-through-sustainable-forestry.pdf (accessed November 5, 2022).

National Fish and Wildlife Foundation (2022a). *New England Forests and Rivers Fund.* Available online at: https://www.nfwf.org/programs/new-england-forests-and-rivers-fund (accessed November 5, 2022).

National Fish and Wildlife Foundation (2022b). *Central appalachia habitat stewardship program.* Available online at: https://www.nfwf.org/programs/central-appalachia-habitat-stewardship-program (accessed November 5, 2022).

National Wildlife Federation (2009). *Increased flooding risk: global warming's wake-up call for riverfront communities.* Reston, VA: National Wildlife Federation.

Natural Resources Conservation Service (2018). *Private landowner response to NRCS young forest programs. U.S. department of agriculture.* Available online at: https://www.nrcs.usda.gov/publications/ceap-wildlife-2018-LandownerResponse-YoungForest.pdf (accessed November 5, 2022).

Natural Resources Conservation Service (2019). *The young forest initiative for at-risk species. U.S. department of agriculture.* Available online at: https://web.archive.org/web/20210324212730/https://www.nrcs.usda.gov/wps/portal/nrcs/detail/me/programs/farmbill/rcpp/?cid=nrcseprd1322729 (accessed November 5, 2022).

NatureServe (2022). *NatureServe network biodiversity location data accessed through NatureServe explorer [web application].* Available online at: https://explorer.natureserve.org/ (accessed November 5, 2022).

Nave, L. E., Vance, E. D., Swanston, C. W., and Curtis, P. S. (2010). Harvest impacts on soil carbon storage in temperate forests. *For. Ecol. Manag.* 259, 857–866. doi: 10.1016/j.foreco.2009.12.009

Neff, C. (2017). *Keeping common species common. New Jersey Audubon. 25 May.* Available online at: https://njaudubon.org/keeping-common-species-common/ (accessed November 5, 2022).

New England Cottontail (2021). *Monterey preservation land trust, berkshires, Massachusetts: young forest project delivers multiple benefits.* Available online at: https://newenglandcottontail.org/demo/monterey-preservation-land-trust-berkshires-massachusetts (accessed November 5, 2022).

New Jersey Audubon (2018). *Sparta mountain wildlife management area forest stewardship plan.* Available online at: https://njaudubon.org/wp-content/uploads/2018/04/New-Jersey-Audubon-Sparta-Mountain-WMA_handout.pdf (accessed November 5, 2022).

New Jersey Department of Environmental Protection (2017). *Sparta mountain wildlife management area stewardship plan.* Available online at: https://www.nj.gov/dep/fgw/sparta/smwma_approved_forest_stewardship_plan.pdf (accessed November 5, 2022).

New Jersey Department of Environmental Protection (2018). *New Jersey's wildlife action plan.* Trenton, NJ: New Jersey Department of Environmental Protection.

New York Department of Environmental Conservation (2015). *A DEC strategic plan for implementing the young forest initiative on wildlife management areas 2015-2020.* Albany, NY: New York Department of Environmental Conservation.

New York Department of Environmental Conservation (2021). *Assessing old-growth forests in New York state forests and preserves. New York natural heritage program.* Available online at: https://www.nynhp.org/ogre/ (accessed November 5, 2022).

Newman, D. J., and Cragg, G. M. (2016). Natural products as sources of new drugs from 1981 to 2014. *J. Nat. Prod.* 79, 629–661. doi: 10.1021/acs.jnatprod.5b01055

Nitschke, C. R. (2005). Does forest harvesting emulate fire disturbance? A comparison of effects on selected attributes in coniferous-dominated headwater systems. *For. Ecol. Manag.* 214, 305–319. doi: 10.1016/j.foreco.2005.04.015

Nolan, V. Jr. (1978). *The ecology and behavior of the prairie warbler dendroica discolor. Ornithological monographs. No. 26.* Washington, DC: American Ornithologists' Union.

North American Bird Conservation Initiative (2014). *The state of the birds 2014 report.* Washington, DC: U.S. Department of Interior. 16.

North American Bird Conservation Initiative (2019). *The State of the birds 2019 report: America's birds in crisis.* Gatineau, QC: North American Bird Conservation Initiative.

North, M. P., and Keeton, W. S. (2008). "Emulating natural disturbance regimes: An emerging approach for sustainable forest management," in *Patterns and processes in forest landscapes*, eds R. Lafortezza, G. Sanesi, J. Chen, and T. Crow (Berlin: Springer), 341–372. doi: 10.1007/978-1-4020-8504-8_19

Northern Institute of Applied Climate Science (2022). *U.S. fish and wildlife service and university of vermont: Nulhegan Basin, Silvio O. conte national fish and wildlife refuge adaptation demonstration project.* Available online at: https://forestadaptation.org/adapt/demonstration-projects/us-fish-and-wildlife-service-and-university-vermont-nulhegan-basin (accessed November 5, 2022).

Noss, R. F. (1983). A regional landscape approach to maintain diversity. *Bioscience* 33, 700–706. doi: 10.2307/1309350

Noss, R. F., Dinerstein, E., Gilbert, B., Gilpin, M., Miller, B., Terborgh, J. J., et al. (1999). "Core areas: Where nature reigns," in *Continental conservation: scientific foundations of regional reserve networks*, eds M. E. Soulé and J. Terborgh (Washington, DC: Island Press), 99–128.

Noss, R. F., Dobson, A. P., Baldwin, R., Beier, P., Davis, C. R., Dellasala, D. A., et al. (2012). Bolder thinking for conservation. *Conserv. Biol.* 26, 1–4. doi: 10.1111/j.1523-1739.2011.01738.x

Noss, R. F., Platt, W. J., Sorrie, B. A., Weakley, A. S., Means, B. D., Costanza, J., et al. (2014). How global biodiversity hotspots may go unrecognized: Lessons from the North American Coastal Plain. *Divers. Distributions* 21, 236–244. doi: 10.1111/ddi.12278

Nowacki, G. J., and Abrams, M. D. (2008). The demise of fire and "mesophication" of forests in the eastern United States. *Bioscience* 58, 123–138. doi: 10.1641/b580207

Nunery, J. S., and Keeton, W. S. (2010). Forest Carbon Storage in the Northeastern United States: Effects of harvesting frequency and intensity including wood products. *For. Ecol. Manag.* 259, 13631375. doi: 10.1016/j.foreco.2009.12.029

Oehler, J. D. (2003). State efforts to promote early-successional habitats on public and private lands in the northeastern United States. *For. Ecol. Manag.* 185, 169–177. doi: 10.1016/s0378-1127(03)00253-6

Oehler, J. D., Covell, D. F., Capel, S., and Long, B. (2006). *Managing grasslands, shrublands and young forest habitats for wildlife.* Augusta, ME: Northeast Upland Habitat Technical Committee.

Oehler, J., Gifford, N., Fergus, C., Edwards, T., Racey, M., and Allred, S. (2013). *Talking about young forests: A communication handbook.* Hadley, MA: Northeast Association of Fish and Wildlife Agencies.

Office of Senator Angus King (2022). *King introduces bill to improve access to katahdin woods and waters. newsroom/press releases, 10 August.* Available online at: https://www.king.senate.gov/newsroom/press-releases/king-introduces-bill-to-improve-access-to-katahdin-woods-and-waters (accessed November 22, 2022).

Oliver, C. D., and Larson, B. A. (1996). *Forest Stand dynamics, update edition Yale school of the environment other publications.* New York, NY: John Wiley & Sons.

Oliveri, S. F. (1993). Bird responses to habitat changes in baxter state park, Maine. *Maine Naturalist* 1:145. doi: 10.2307/3858237

Oswald, W. W., Foster, D. R., Shuman, B. N., Chilton, E. S., Doucette, D. L., Deena, L., et al. (2020b). Conservation implications of limited native american impacts in pre-contact New England. *Nat. Sustain.* 3, 241–246. doi: 10.1038/s41893-019-0466-0

Oswald, W. W., Foster, D. R., Shuman, B. N., Chilton, E. S., Doucette, D. L., and Duranleau, D. L. (2020a). W. W. Oswald et al. reply to M. D. Abrams and G. J. Nowacki. Nature Sustainability. (2020). *Nat. Sustain.* 3, 900–903. doi: 10.1038/s41893-020-0580-z

Oswalt, S. N., Smith, W. B., Miles, P. D., and Pugh, S. A. (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, Washington Office, 223.

Paciorek, C. J., Cogbill, C. V., Peters, J. A., Williams, J. W., Mladenoff, D. J., Dawson, A., et al. (2021). The forests of the midwestern United States at Euro-American settlement: Spatial and physical structure based on contemporaneous survey data. *PLoS One* 16:e0246473. doi: 10.1371/journal.pone.0246473

Palik, B. J., Clark, P. W., D'Amato, A. W., Swanston, C., and Nagel, L. (2022). Operationalizing forest-assisted migration in the context of climate change adaptation: Examples from the Eastern USA. *Ecosphere* 13:e4260. doi: 10.1002/ecs2.4260

Pan, Y., Chen, J. M., Birdsey, R., McCullough, K., He, L., and Deng, F. (2011). Age structure and disturbance legacy of North American forests. *Biogeosciences* 8, 715–732. doi: 10.5194/bg-8-715-2011

Paradis, A., Elkinton, J., Hayhoe, K., and Buonaccorsi, J. (2008). Role of winter temperature and climate change on the survival and future range expansion of the hemlock woolly adelgid (*Adelges tsugae*) in eastern North America. *Mitig. Adapt Strateg Glob. Change* 13, 541–554. doi: 10.1007/s11027-007-9127-0

Parajuli, R. (2022). *The infrastructure act and forestry: a brief overview. North Carolina state extension, NC State University.* Available online at: https://forestry.ces.ncsu.edu/2022/01/iija-forestry/ (accessed November 5, 2022).

Park, A., and Talbot, C. (2018). Information underload: Ecological. *Bioscience* 68, 251–263. doi: 10.1093/biosci/biy001

Parshall, T., and Foster, D. R. (2002). Fire on the New England landscape: Regional and temporal variation, cultural and environmental controls. *J. Biogeogr.* 29, 1305–1317. doi: 10.1046/j.1365-2699.2002.00758.x

Parshall, T., Foster, D. R., Faison, E., MacDonald, D., and Hansen, B. C. S. (2003). Long-term history of vegetation and fire in pitch pine–oak forests on cape cod, Massachusetts. *Ecology* 84, 736–748. doi: 10.1890/0012-96582003084[0736:LTHOVA]2.0.CO;2

Paskus, J. J., Pearsall, D. R., and Ross, J. A. (2015). *Facilitating the effectiveness of state wildlife action plans at multiple scales in the upper midwest/great lakes LCC: Findings and recommendations. Report number MNFI 2016-02, 56 pp. + appendices.* East Lansing, MI: US Fish and Wildlife Service, Upper Midwest Great Lakes Landscape Conservation Cooperative.

Pavlovic, N. B., and Grundel, R. (2009). Reintroduction of wild lupine (*Lupinus perennis* L.) depends on variation in canopy, vegetation, and litter cover. *Restorat. Ecol.* 17, 807–817. doi: 10.1111/j.1526-100X.2008.00417.x

Peabody, W. B. O. (1839). *A report on the ornithology of Massachusetts. In reports on the fishes, reptiles and birds of Massachusetts. Published agreeable to an order of the legislature, by the commissioners on the zoological and botanical survey of the state.* Boston, MA: Zoological and botanical survey, 255–404.

Pearce, K. J., Serfass, T. L., McCann, J. M., and Feller, D. J. (2021). Status and distribution of the Eastern Spotted Skunk (*Spilogale putorius*) in Maryland: A historic review and recent assessment. *Southeastern Naturalist* 20, 52–63. doi: 10.1656/058.020.0sp1106

Pederson, N. (2013). *Eastern OLDLIST: a database of maximum ages for Eastern North America.* Available online at: https://www.ldeo.columbia.edu/~adk/oldlisteast/ (accessed October 2, 2022).

Pederson, N., D'Amato, A. W., Dyer, J. M., Foster, D. R., Goldblum, D., Hart, J. L., et al. (2014). Climate remains an important driver of post-European vegetation change in the Eastern United States. *Glob. Chang. Biol.* 21, 2105–2110. doi: 10.1111/gcb.12779

Pellerito, R., and Wisch, R. (2002). *State endangered species chart. Animal legal & historical center, Michigan State University College of Law.* Available online at: https://www.animallaw.info/article/state-endangered-species-chart (accessed November 5, 2022).

Pelley, J. (2009). Old-growth forests store a treasure trove of carbon. *Environ. Sci. Technol.* 43, 7602–7603. doi: 10.1021/es902647k

Peterken, G. F. (1996). *Natural woodland: ecology and conservation in northern temperate regions.* Cambridge: Cambridge University Press.

Petranka, J. W., Brannon, M. P., Hopey, M. E., and Smith, C. K. (1994). Effects of timber harvesting on low elevation populations of southern appalachian salamanders. *For. Ecol. Manag.* 67, 135–147. doi: 10.1016/0378-1127(94)90012-4

Petrenko, C. L. and Friedland, A. J. (2015). Mineral soil carbon pool responses to forest clearing in northeastern hardwood forests. *GCB Bioenergy* 7, 1283–1293. doi: 10.1111/gcbb.12221

10.3389/ffgc.2022.1073677

Phillips, S. (2004). *The economic benefits of wilderness: focus on property value enhancement. The Wilderness Society.* Avalable at: https://web.archive.org/web/20140808201303it_/http://wilderness.org/sites/default/files/The-Economic-Benefits-of-Wilderness-With-a-Focus-on-Land-Value-Enhancement_low res.pdf (accessed November 5, 2022).

Plenzler, M. A., and Michaels, H. J. (2015). Seedling recruitment and establishment of *Lupinus perennis* in a mixed-management landscape. *Natural Areas J.* 35, 224–234. doi: 10.3375/043.035.0203

Plotkin, A. B., Foster, D., Carlson, J., and Magill, A. (2013). Survivors, not invaders, control forest development following simulated hurricane. *Ecology* 94, 414–423. doi: 10.1890/12-0487.1

Potter, C. (2022). *Forest service pressing ahead with logging around lake. Valley News, 25 April.* Available online at: https://www.vnews.com/New-comment-period-opens-on-Lake-Tarleton-proposed-logging-46053192 (accessed November 6, 2022).

Poulos, L. P., and Roy, B. A. (2015). Fire and false brome: How do prescribed fire and invasive Brachypodium sylvaticum affect each other? *Invas. Plant Sci. Manag.* 8, 122–130. doi: 10.1614/IPSM-D-14-00024.1

Power, T. M. (1996). Wilderness economics must look through the windshield, not the rear-view mirror. *Int. J. Wilderness* 2, 5–9.

Power, T. M. (2001). *The economic impact of the proposed Maine woods national park & preserve. RESTORE.* Hallowell, ME: The North Woods.

Pyne, S. J. (2000). *Where have all the fires gone? Fire management today*, Vol. 60. Washington, DC: USDA Forest Service.

Raiho, A. M., Paciorek, C. J., Dawson, A., Jackson, S. T., Mladenoff, D. J., and Williams, J. W. (2022). 8000-year doubling of Midwestern forest biomass driven by population- and biome-scale processes. *Science* 376:1491. doi: 10.1126/science.abk3126

Rasker, R., Gude, P. H., and Delorey, M. (2013). The effect of protected federal lands on economic prosperity in the non-metropolitan west. *J. Regional Anal. Policy* 43, 110–122.

Reynolds, M. T. (2021). *National parks overcrowding. Statement before the senate energy and natural resources subcommittee onn national parks, 28 July.* Available online at: https://www.doi.gov/ocl/national-parks-overcrowding (accessed October 12, 2022).

Rhemtulla, J. M., and Mladenoff, D. J. (2007). Regional land-cover conversion in the U.S. upper Midwest: Magnitude of change and limited recovery (1850-1935-1993). *Landsc. Ecol.* 22, 57–75. doi: 10.1007/s10980-007-9117-3

Rhemtulla, J. M., Mladenoff, D. J., and Clayton, M. K. (2009). Historical forest baselines reveal potential for continued carbon sequestration. *Proc. Natl. Acad. Sci. U.S.A.* 106, 6082–6087. doi: 10.1073/pnas.081007610

Rhode Island Division of Statewide Planning and Rhode Island Department of Environmental Management (2019). *Ocean state outdoors: Rhode Island's comprehensive outdoor recreation plan: State guide plan element 152, Report No. 122.* Providence, RI: Rhode Island Division of Statewide Planning and Rhode Island Department of Environmental Management.

Riitters, K. H., Potter, K. M., Iannone, B. V. III., Oswalt, C., Guo, Q., and Fei, S. (2018). Exposure of protected and unprotected forest to plant invasions in the Eastern United States. *Forests* 9:723. doi: 10.3390/f9110723

Robertson, D. L., Babin, L. M., Krall, J. R., von Fricken, M. E., Baghi, H., and Jacobsen, K. H. (2019). The association between hunter-killed deer and lyme disease in New Jersey, 2000–2014. *Ecohealth* 16, 330–337. doi: 10.1007/s10393-019-01401-

Rogers, B. M., Mackey, B., Shestakova, T. A., Keith, H., Young, V., Kormos, C. F., et al. (2022). Using ecosystem integrity to maximize climate mitigation and minimize risk in international forest policy. *Front. For. Glob. Change* 5:929281. doi: 10.3389/ffgc.2022.929281

Rohrbaugh, R. W., Treyger, S., McGinley, K., and Loucks, K. (2020). *Healthy forests: A bird-based silvicultural guide for forestry professionals.* Audubon, PA: Pennsylvania Chapter, National Audubon Society, 40.

Rooney, T., and Waller, D. (2003). Direct and indirect effects of white-tailed deer in forest ecosystems. *For. Ecol. Manag.* 181, 165–176. doi: 10.1016/S0378-1127(03)00130-0

Rosa, L., and Malcom, J. (2020). *Getting to 30X30: guidelines for decision-makers.* Washington, DC: Defenders of Wildlife.

Rosenberg, K. B., Blancher, P. J., Stanton, J. C., and Panjabi, A. O. (2017). Use of North American breeding bird survey data in avian conservation assessments. *Condor* 119, 594–606. doi: 10.1650/CONDOR-17-57.1

Rosenberg, K. V., Dokter, A. M., Blancher, P. J., Sauer, J. R., Smith, A. C., Smith, P. A., et al. (2019). Decline of the North American Avifauna. *Science* 366, 120–124. doi: 10.1126/science.aaw1313

Rosenberg, K. V., Hames, R. S., Rohrbaugh, R. W., Swarthout, S. B., Lowe, J. D., and Dhondt, A. A. (2003). *A land manager's guide to improving habitat for forest thrushes.* Ithaca, NY: The Cornell Lab of Ornithology.

Rosenberg, K. V., Kennedy, J. A., Dettmers, R., Ford, R. P., Reynolds, D., Alexander, J. D., et al. (2016). *Partners in flight landbird conservation plan: 2016 revision for Canada and Continental United States. Partners in flight science committee.* Washington, DC: North American Bird Conservation Initiative (NABCI), Association of Fish and Wildlife Agencies, 119.

Ruddat, J. (2022). *An inventory of Connecticut's primeval woodlands. Rhodora. No. 995.* Cambridge, MA: New England Botanical Club.

Ruffed Grouse Society (2022). *RGS & AWS and partners awarded forest service landscape scale restoration grant in Massachusetts, 19 July.* Available online at: https://ruffedgrousesociety.org/rgs-aws-and-partners-awarded-forest-service-landscape-scale-restoration-grant-in-massachusetts/ (accessed November 5, 2022).

Runkle, J. R. (1982). Patterns of disturbance in some old-growth mesic forests of eastern North America. *Ecology* 63, 1533–1546. doi: 10.2307/1938878

Russell, E. W. B. (1981). Vegetation of Northern New Jersey before European settlement. *Am. Midland Naturalist* 105, 1–12. doi: 10.2307/2425004

Russell, E. W. B. (1983). Indian-set fires on the forests of the Northeastern United States. *Ecology* 64, 78–88. doi: 10.2307/1937331

Russell, T. (2017). *The 3 biggest obstacles to creating young forest cover. National deer association (aka Quality Deer Management Association), 27 March.* Available online at: https://www.deerassociation.com/3-biggest-obstacles-creating-young-forest-cover/ (accessed November 5, 2022).

Santoro, J. A., and D'Amato, A. W. (2019). Structural, compositional, and functional responses to tornado and salvage logging disturbance in southern New England hemlock-hardwood forests. *For. Ecol. Manag.* 444, 138–150. doi: 10.1016/j.foreco.2019.04.039

Sauer, J. R., Link, W. A., and Hines, J. E. (2020). *The North American breeding bird survey, analysis results 1966 - 2019.* Laurel, MD: U.S. Geological Survey data release. Eastern Ecological Science Center. doi: 10.5066/P96A7675

Sauer, J. R., Link, W. A., Fallon, J. E., Pardieck, K. L., and Ziolkowski, D. J. (2013). The North American breeding bird survey 1966–2011: Summary analysis and species accounts. *North Am. Fauna* 79, 1–32. doi: 10.3996/nafa.79.0001

Sauer, J. R., Pardieck, K. L., Ziolkowski, D. J., Smith, A. C., Hudson, M. R., Rodriguez, et al. (2017). The first 50 years of the North American Breeding Bird Survey. *Condor* 119, 576–593. doi: 10.1650/CONDOR-17-83.1

Scheller, R. M., Van Tuyl, S., Clark, K., Hayden, N. G., Hom, J., and Mladenoff, D. J. (2008). Simulation of forest change in the New Jersey Pine Barrens under current and pre-colonial conditions. *For. Ecol. Manag.* 255, 1489–1500. doi: 10.1016/j.foreco.2007.11.025

Schlaghamerský, J., Eisenhauer, N., and Frelich, L. E. (2014). Earthworm invasion alters enchytraeid community composition and individual biomass in northern hardwood forests of North America. *Appl. Soil Ecol.* 83, 159–169. doi: 10.1016/j.apsoil.2013.09.005

Schlossberg, S., and King, D. I. (2007). *Ecology and management of scrub-shrub birds in New England: A comprehensive review.* Washington, DC: U.S. Department of Agriculture Natural Resource Conservation Service, Resource Inventory and Assessment Division.

Schlossberg, S., King, D. I., Destefano, S., and Hartley, M. (2018). Effects of early-successional shrubland management on breeding wood thrush populations. *J. Wildl. Manag.* 82, 1572–1581. doi: 10.1002/jwmg.21559

Schulte, L. A., Mladenoff, D. J., Burrows, S. N., Sickley, T. A., and Nordheim, E. V. (2005a). Spatial controls of Pre-Euro-American wind and fire in northern Wisconsin (USA) forest landscapes. *Ecosystems* 8, 73–94. doi: 10.1007/s10021-004-0052-8

Schulte, L. A., Pidgeon, A. M., and Mladenoff, D. J. (2005b). One hundred fifty years of change in forest bird breeding habitat: Estimates of species distributions. *Conserv. Biol.* 19, 1944–1956. doi: 10.1111/j.1523-1739.2005.00254.x

Schultz, J. (2003). *Conservation assessment for butternut or white walnut (Juglans cinerea)* L. Milwaukee, WI: USDA Forest Service, Eastern Region.

Schulz, F., Alteio, L., Goudeau, D., Ryan, E. M., Yu, F. B., Malmstrom, R. S., et al. (2018). Hidden diversity of soil giant viruses. *Nat. Commun.* 9:4881. doi: 10.1038/s41467-018-07335-2

Schulze, E. D., Korner, C., Law, B., Haberl, H., and Luyssaert, S. (2012). Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral. *GCB Bioenergy* 10, 1–6. doi: 10.1111/j.1757-1707.2012.01169.x

Schwartz, M. W., Hellmann, J. J., McLachlan, J. M., Sax, D. F., Borevitz, J. O., Brennan, J., et al. (2012). Integrating the scientific, regulatory, and ethical challenges. *Bioscience* 62, 732–743. doi: 10.1525/bio.2012.62.8.6

Scott, J. M., Davis, F. W., McGhie, R. G., Wright, R. G., Groves, C., and Estes, J. (2001). Nature reserves: Do they capture the full range of America's biological diversity? *Ecol. Applic.* 11, 999–1007. doi: 10.1890/1051-0761(2001)011[0999:NRDTCT]2.0.CO;2

Seamans, M. E., and Rau, R. D. (2018). *American woodcock population status, 2018.* Laurel, MD: U.S. Fish and Wildlife Service.

Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., et al. (2017). Forest disturbances under climate change. *Nat. Clim. Change* 7, 395–402. doi: 10.1038/nclimate3303

Seitz, G. (2019). *Fourth of July marks 20th anniversary of boundary waters blowdown. Quetico superior wilderness news, 1 July.* Available online at: https://queticosuperior.org/blog/fourth-of-july-marks-20th-anniversary-of-boundary-waters-blowdown/ (accessed November 6, 2022).

Seng, P. T., and Case, D. J. (2019). "Communicating effectively about young forest management to benefit associated wildlife species," in *Proceedings of the Eleventh American Woodcock Symposium*, eds D. G. Krementz, D. E. Andersen, and T. R. Cooper (Minneapolis, MN: University of Minnesota Libraries Publishing), 67–75. doi: 10.24926/AWS.0109

Seymour, R. S., White, A. S., and deMaynadier, P. G. (2002). Natural disturbance regimes in Northeastern North America — evaluating silvicultural systems using natural scales and frequencies. *For. Ecol. Manag.* 155, 357–367. doi: 10.1016/S0378-1127(01)00572-2

Sharon, S. (2022). *USDA grants $30 million for increased carbon storage in New England forests. Maine Public, 14 September.* Lewiston, ME: Maine Public.

Sheikh, P. A. (2011). *Forest management for resilience and adaptation (CRS Report No. R41691).* Washington, DC: Congressional Research Service.

Shuman, B. N., Marsicek, J., Oswald, J. W., and Foster, D. R. (2019). Predictable hydrological and ecological responses to Holocene North Atlantic variability. *Proc. Natl. Acad. Sci. U.S.A.* 116, 5985–5990. doi: 10.1073/pnas.1814307116

Shuman, B., Newby, P., Huang, Y., and Webb, T. III. (2004). Evidence for the close climatic control of New England vegetation history. *Ecology* 85, 1297–1310. doi: 10.1890/02-0286

Shumway, D. L., Abrams, M. D., and Ruffner, C. M. (2001). A 400-year history of fire and oak recruitment in an old-growth oak forest in Western Maryland, USA. *Can. J. For. Res.* 31, 1437–1443. doi: 10.1139/cjfr-31-8-1437

Simard, S. (2021). *Finding the mother tree: discovering the wisdom of the forest.* New York, NY: Knopf Doubleday Publishing Group, 368.

Simard, S. W., Beiler, K. J., Bingham, M. A., Deslippe, J. R., Philip, L. J., and Teste, F. P. (2012). Mycorrhizal networks: mechanisms, ecology and modelling. *Fungal Biol. Rev.* 26, 39–60. doi: 10.1016/j.fbr.2012.01.001

Small, M. F., and Hunter, M. L. (1988). Forest fragmentation and avian nest predation in forested landscapes. *Oecologia* 76, 62–64. doi: 10.1007/bf00379601

Smith, J. (2017). *The American woodcock, and why we should be cutting more trees. the nature conservancy, 28 March.* Available online at: https://blog.nature.org/science/2017/03/28/american-woodcock-why-cutting-more-trees-logging-forests/ (accessed November 6, 2022).

Smith, J. E., Heath, L. S., Skog, K. E., and Birdsey, R. A. (2006). *Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States. Gen. Tech. Rep. NE-343.* Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station, 216. doi: 10.2737/NE-GTR-343

South Carolina Department of Natural Resources (2020). *Technical guidance for the development of wildlife and pollinator habitat at solar farms: South Carolina solar habitat act.* Columbia: South Carolina Department of Natural Resources.

Southwell, D. K. (2001). *Conservation assessment for prairie warbler (Dendroica discolor).* Washington, DC: USDA Forest Service.

Spies, T. A., and Franklin, J. F. (1991). "The structure of natural young, mature, and old-growth Douglas-fir forests in Oregon and Washington," in *Wildlife and vegetation of unmanaged Douglas-fir forests. General technical report PNW-GTR-85*, eds L. F. Ruggiero, K. B. Aubry, A. B. Carey, and M. Huff (Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station), 91–109.

Stauffer, G. E., Miller, D. A. W., Williams, L. M., and Brown, J. (2018). Ruffed grouse population declines after introduction of West Nile virus. *J. Wild. Mgmt.* 82, 165–172. doi: 10.1002/jwmg.21347

Steidinger, B. S., Crowther, T. W., Liang, J., Van Nuland, M. E., Werner, G. D. A., Reich, P. B., et al. (2019). Climatic controls of decomposition drive the global biogeography of forest-tree symbioses. *Nature* 569, 404–408. doi: 10.1038/s41586-019-1128-0

Sterman, J., Moomaw, W., Rooney-Varga, J. N., and Siegel, L. (2022). Does wood bioenergy help or harm the climate? *Bull. Atomic Sci.* 78, 3,128–138. doi: 10.1080/00963402.2022.2062933

Stevens, A. (1996). *The paleoecology of coastal sandplain grasslands on Martha's Vineyard, Massachusetts.* Doctoral Dissertations. Amherst, MA: University of Massachusetts Amherst.

Stoleson, S. H. (2013). Condition varies with habitat choice in postbreeding forest birds. *Auk* 130, 417–428. doi: 10.1525/auk.2013.12214

Streby, H. M., Peterson, S. M., and Andersen, D. E. (2016). "Survival and habitat use of edgling Golden-winged Warblers in the western Great Lakes region," in *Golden-winged warbler ecology, conservation, and habitat management. Studies in Avian Biology (no. 49)*, eds H. M. Streby, D. E. Andersen, and D. A. Buehler (Boca Raton, FL: CRC Press), 127–140. doi: 10.7717/peerj.4319

Strittholt, J. R., DellaSala, D. A., and Jiang, H. (2006). Status of mature and old-growth forests in the Pacific Northwest. *Conserv. Biol.* 20, 363–374. doi: 10.1111/j.1523-1739.2006.00384.x

Tallamy, D. W. (2021). *The nature of Oaks: the rich ecology of our most essential native trees.* Portland: Timber Press.

Tanner, R. A., and Gange, A. C. (2005). Effects of golf courses on local biodiversity. *Landsc. Urban Plann.* 71, 137–146. doi: 10.1016/j.landurbplan.2004.02.004

Tavernia, B. G., Nelson, M. D., Garner, J. D., and Perry, C. H. (2016). Spatial characteristics of early successional habitat across the upper great lakes states. *For. Ecol. Manag.* 372, 164–174. doi: 10.1016/j.foreco.2016.04.003

Telford, S. R. (2017). Deer reduction is a cornerstone of integrated deer tick management. *J. Integr. Pest Manag.* 8, 25;1–5. doi: 10.1093/jipm/pmx024

Terborgh, J. W. (2015). Toward a trophic theory of species diversity. *Proc. Natl. Acad. Sci. U.S.A.* 11415–11422. doi: 10.1073/pnas.1501070112

Terborgh, J., Estes, J. A., Paquet, P., Ralls, K., Boyd-Heger, D., Miller, B. J., et al. (1999). "The role of top carnivores in regulating terrestrial ecosystems," in *Continental Conservation: Scientific Foundations of Regional Reserve Networks*, eds M. E. Soulé and J. Terborgh (Washington, DC: Island Press), 39–64. doi: 10.1111/j.1469-185X.2011.00203.x

The Lancet Global Health (2020). Editorial: Mental health matters. *Lancet Glob. Health* 8:E1352. doi: 10.1016/S2214-109X(20)30432-0

The White House (2022). *Executive order on strengthening the nation's forests, communities, and local economies. presidential actions, April 22, 2022.* Available online at: https://www.whitehouse.gov/briefing-room/presidential-actions/2022/04/22/executive-order-on-strengthening-the-nations-forests-communities-and-local-economies/ (accessed November 6, 2022).

The Wildlife Society (2017). *State and tribal wildlife grant program.* Available online at: https://wildlife.org/wp-content/uploads/2014/11/Policy-Brief_STWG_FINAL.pdf (accessed November 6, 2022).

Thom, D., and Keeton, W. S. (2020). Disturbance-based silviculture for habitat diversification: Effects on forest structure, dynamics, and carbon storage. *For. Ecol. Manag.* 469:118132. doi: 10.1016/j.foreco.2020.118132

Thom, D., Golivets, M., Edling, L., Meigs, G. W., Gourevitch, J. D., Sonter, L. J., et al. (2019). The climate sensitivity of carbon, timber, and species richness covaries with forest age in boreal–temperate North America. *Glob. Change Biol.* 25, 2446–2458. doi: 10.1111/gcb.14656

Thompson, A., and Walls, M. A. (2021). *Getting to 30x30: important considerations for the biden administration's conservation Agenda. Resources for the future.* Available online at: https://www.resources.org/common-resources/getting-to-30x30-important-considerations-for-the-biden-administrations-conservation-agenda/ (accessed November 6, 2022).

Thompson, I., Mackey, B., McNulty, S., and Mosseler, A. (2009). *Forest resilience, biodiversity, and climate change. A synthesis of the biodiversity/resilience/stability relationship in forest ecosystems. Technical Series no. 43.* Montreal: Secretariat of the Convention on Biological Diversity, 67.

Thompson, J. (2006). *Society's choices: land use changes, forest fragmentation, and conservation. Science Findings 88.* Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, 5.

Thompson, J. R., Carpenter, D. N., Cogbill, C. V., and Foster, D. R. (2013). Four centuries of change in Northeastern United States forests. *PLoS One* 8:e72540. doi: 10.1371/journal.pone.0072540

Thompson, J. R., Lambert, K. F., Foster, D. R., Broadbent, E. N., Blumstein, M., Almeyda Zambrano, A. M., et al. (2016). The consequences of four land-use scenarios for forest ecosystems and the services they provide. *Ecosphere* 7:e01469. doi: 10.1002/ecs2.1469

Tiako, M., Nguemeni, J., McCarthy, C., Meisel, Z. F., Elovitz, M. A., Burris, H. H., et al. (2021). Association between low urban neighborhood greenness and hypertensive disorders of pregnancy. *Am. J. Perinatol.* doi: 10.1055/s-0041-1733786 [Epub ahead of print].

Titus, H. (1945). *The land nobody wanted: the story of Michigan's public domain.* East Lansing, MI: Michigan State College, Agricultural Experiment Station, Section of Conservation.

Toot, R., Frelich, L. E., Butler, E., and Reich, P. B. (2020). Climate-biome envelope shifts create enormous challenges and novel opportunities for conservation. *Forests* 11:1015. doi: 10.3390/f11091015

Tulowiecki, S. J., Ranney, E. R., Keenan, E. M., Neubert, G. M., and Hogan, M. L. (2022). Localized Native American impacts on past forest composition across a regional extent in north-eastern United States. *J. Biogeogr.* 49, 1099–1109. doi: 10.1111/jbi.14369

U.S. Department of Agriculture (2020). *Farms and land in farms: 2019 summary. National agricultural statistics service. ISSN: 1995-2004.* Available online at: https://www.nass.usda.gov/Publications/Todays_Reports/reports/fnlo0220.pdf (accessed November 6, 2022).

U.S. Fish and Wildlife Service (2006). *Species of concern: cerulean warbler (Setophaga cerulean).* Available online at: https://web.archive.org/web/20220127133641/https://www.fws.gov/midwest/es/soc/birds/cerw/pdf/Cerulean_Warbler_Fact_Sheet.pdf (accessed November 5, 2022).

U.S. Fish and Wildlife Service (2015a). *Historical distribution of the New England cottontail (Sylvilagus transitionalis). Supplemental document to the New England cottontail 12-month petition finding, docket number FWS-R5-2015-0136 July 27, 2015.* Available online at: https://web.archive.org/web/20220120213600/https://www.fws.gov/northeast/newenglandcottontail/pdf/20150727_NEC_12M_HistoricalDistributionSupplement.pdf (accessed November 7, 2022).

U.S. Fish and Wildlife Service (2015b). *Endangered and threatened wildlife and plants; 12-month finding on a petition to list the New England cottontail as an endangered or threatened species. Fed. Reg. Vol. 80 No. 178 55286 (proposed September 15, 2015).* Washington, D.C: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service (2015c). *New England cottontail (Sylvilagus transitionalis).* Available online at: https://web.archive.org/web/20170312201433/https://www.fws.gov/northeast/newenglandcottontail/pdf/NEcottontail2015.pdf (accessed November 5, 2022).

U.S. Fish and Wildlife Service (2022a). *Endangered and threatened wildlife and plants; endangered species status for northern long-eared Bat. Fed. Reg. Vol. 87, No. 229 73488 (proposed Wednesday, November 30, 2022).* Washington, D.C: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service (2022b). *FWS-Listed U.S. species by taxonomic group. ECOS environmental conservation online system.* Available online at: https://ecos.fws.gov/ecp/report/species-listings-by-tax-group-totals (accessed November 7, 2022).

U.S. Geological Survey (2022a). *Protected areas database of the United States (PAD-US) 3.0: U.S. Geological Survey data release.* Reston, VA: U.S. Geological Survey. Available online at: https://www.usgs.gov/programs/gap-analysis-project/science/pad-us-data-download (accessed November 7, 2022).

U.S. Geological Survey (2022b). *Gap analysis project (GAP): PAD-US data overview.* Available online at: https://www.usgs.gov/programs/gap-analysis-project/science/pad-us-data-overview (accessed October 12, 2022).

USDA Forest Service (2003). *Major trend data 1760-2000. Forest inventory & analysis.* Available online at: https://web.archive.org/web/20220611180428/https://www.fia.fs.fed.us/slides/major-trends.pdf (accessed October 12, 2022).

USDA Forest Service (2017). *Harvest trends on national forest system lands: 1984 to present.* Available online at: https://web.archive.org/web/20220120151939/https://www.fs.fed.us/forestmanagement/documents/harvest-trends/NFS-HarvestHistory1984-2017.pdf (accessed November 7, 2022).

USDA Forest Service (2018). *Early successional habitat creation project: notice of proposed action and opportunity to comment. Green mountain national forest.* Available online at: https://www.wallingfordvt.com/wp-content/uploads/2018/07/GMNF-Scoping-Details-Habitat-Creation.pdf (accessed October 12, 2022).

US Forest Service (2021a) *Tarleton integrated resource project grafton county, new hampshire draft environmental assessment and preliminary finding of no significant impact. White Mountain National Forest, - Ranger District.* Available online at: https://www.fs.usda.gov/project/?project=56394 (accessed November 6, 2022).

USDA Forest Service (2021b). *Fiscal year 2021 S&PF landscape scale restoration funded projects for the Northeast and Midwest.* Available online at: https://usfs-public.app.box.com/v/FY21FundedList (accessed November 7, 2022).

USDA Forest Service (2022a). *Biden administration announces $32 million to advance climate-smart mass timber construction, expand wood markets, 27 May. [Press release].* Available online at: https://www.usda.gov/media/press-releases/2022/05/27/biden-administration-announces-32-million-advance-climate-smart (accessed November 15, 2022).

USDA Forest Service (2022b). *Forest inventory EVALIDator web-application Version 1.8.0.01., Forest inventory and analysis program.* St. Paul, MN: Northern Research Station.

USDA Forest Service (2022c). *Climate adaptation plan. FS-1196.* Washington, D.C: U.S. Department of Agriculture.

USDA Forest Service (2022d). *Forest service to use prescribed fire to improve wildlife habitat, 26 April.* Available online at: https://www.fs.usda.gov/detail/gmfl/news-events/?cid=FSEPRD1015733 (accessed November 7, 2022).

USDA Forest Service (2022e). *Fiscal year 2022 S&PF landscape scale restoration funded projects for the northeast and midwest.* Available online at: https://usfs-public.app.box.com/s/1v1o17zgofogxnolgf1jmgm8cltaw49e (accessed November 7, 2022).

USDA Forest Service and Bureau of Land Management (2022). *Request for information (RFI) on federal old-growth and mature forests. Fed. Reg. Vol. 87, No. 135, 42493 (proposed Friday, July 15, 2022).* Washington, D.C: USDA Forest Service and Bureau of Land Management.

USGCRP (2018). *Impacts, risks, and adaptation in the United States: fourth national climate assessment, volume II: report-in-brief*, eds D. R. Reidmiller, C. W. Avery, D. R. Easterling, K. E. Kunkel, K. L. M. Lewis, T. K. Maycock, et al. (Washington, DC: U.S. Global Change Research Program), 186. doi: 10.7930/NCA4.2018.RiB

Vale, T. R. (1998). The myth of the humanized landscape: An example from yosemite national park. *Natural Areas J.* 18, 231–236.

Vale, T. R. (2002). "The pre-European landscape of the United States: pristine or humanized?," in *Fire, native peoples and the natural landscape*, ed. T. R. Vale (Washington, D.C: Island Press), 1–39.

Vantellingen, J., and Thomas, S. C. (2021). Log landings are methane emission hotspots in managed forests. *Can. J. For. Res.* 51, 1916–1925. doi: 10.1139/cjfr-2021-0109

Veatch, J. O. (1928). Reconstruction of forest cover based on soil maps. *Q. Bull. Michigan Agric. Exp. Station* 10, 116–126.

Vega Rivera, J. H., Rappole, J. H., Mcshea, W. J., and Haas, C. A. (1998). Wood thrush postfledging movements and habitat use in Northern Virginia. *The Condor* 100, 69–78.

Vining, J., and Tyler, D. E. (1999). Values, emotions and desired outcomes reflected in public responses to forest management plans. *Hum. Ecol. Rev.* 6, 21–34.

Wang, K., Lombard, J., Rundek, T., Dong, C., Marinovic Gutierrez, C., Byrne, M. M., et al. (2019). Relationship of neighborhood greenness to heart disease in 249 405 US medicare beneficiaries. *J. Am. Heart Assoc.* 8:e010258. doi: 10.1161/JAHA.118.010258

Warrick, G. (2000). The precontact iroquoian occupation of Southern Ontario. *J. World Prehistory* 14, 415–466. doi: 10.1023/A:1011137725917

Watson, J. E. M., Evans, T., Venter, O., Williams, B., Tulloch, A., Stewart, C., et al. (2018). The exceptional value of intact forest ecosystems. *Nat. Ecol. Evol.* 2, 599–610. doi: 10.1038/s41559-018-0490-x

Weber, S., and Cooper, T. R. (2019). "Implementing the American woodcock conservation plan: wildlife management institute's young forest initiative," in *Proceedings of the eleventh american woodcock symposium*, eds D. G. Krementz, D. E. Andersen, and T. R. Cooper (Minneapolis, MN: University of Minnesota Libraries Publishing), 5–8. doi: 10.24926/AWS.0102

Weidensaul, S. (2018). *Old-Growth is great, but here's why we need new-growth forests, too. Living bird magazine, 28 March. The cornell lab of ornithology.* Available online at: https://www.allaboutbirds.org/news/old-growth-is-great-but-heres-why-we-need-new-growth-forests-too/ (accessed November 7, 2022).

Weyerhaeuser Company (2020). *How we do it: wildlife habitat.* Available online at: https://www.weyerhaeuser.com/application/files/3715/9363/0295/WY_How_We_Do_It_Wildlife_Habitat_2020.pdf (accessed November 7, 2022).

Whitcomb, K. Jr. (2022). *Vermont logging drawing criticism. The rutland herald, 15 July.* Available online at: https://www.eagletimes.com/vermont-logging-drawing-criticism/article_0b9113c-fad7-54e5-a13d-38d92c3e4303.html (accessed November 7, 2022).

Whitney, G. G. (1994). *From coastal wilderness to fruited plain: a history of environmental change in temperate North America, 1500 to the present.* Cambridge: Cambridge University Press.

Widmann, R. H., Crawford, S., Kurtz, C. M., Nelson, M. D., Miles, P. D., Morin, R. S., et al. (2015). *New York forests, 2012. Resource bulletin NRS-98.* Newtown

Square, PA: U.S Department of Agriculture, Forest Service, Northern Research Station, 128.

Wiggins, D. A. (2006). *Ruffed grouse (Bonasa umbellus): A Technical Conservation Assessment*. Washington, DC: USDA Forest Service, Rocky Mountain Region.

Wilcove, D. S., McClellan, C. H., and Dobson, A. P. (1986). "Habitat fragmentation in the temperate zone," in *Conservation biology: the science of scarcity and diversity*, ed. M. E. Soule (Sunderland, MA: Sinauer Associates), 237–256.

Wildlife Management Institute (2009). *Upper great lakes woodcock and young forest initiative: best management practices for woodcock & associated bird species*. Washington, DC: Wildlife Management Institute.

Wildlife Management Institute (2010). *Implementing the American woodcock conservation plan: progress to date*. Washington, DC: Wildlife Management Institute.

Williams, C. A., Collatz, G. J., Masek, J., and Goward, S. N. (2012). Carbon consequences of forest disturbance and recovery across the conterminous United States. *Glob. Biogeochem. Cycles* 26:GB1005. doi: 10.1029/2010GB003947

Williams, C. A., Hasler, N., and Xi, L. (2021). *Avoided deforestation: a climate mitigation opportunity in New England and New York prepared for the United States climate alliance natural and working lands research program*. Worcester, MA: Clark University, 1–42.

Williams, G. W. (2002). "Aboriginal use of fire: are there any 'natural' plant communities?," in *Wilderness and political ecology: aboriginal land management – myths and reality*, eds C. E. Kay and R. T. Simmons (Salt Lake City, UT: University of Utah Press), 48.

Wilson, D. C., Morin, R., Frelich, L. E., and Ek, A. R. (2019). Monitoring disturbance intervals in forests: A case study of increasing forest disturbance in Minnesota. *Ann. For. Sci.* 76:78. doi: 10.1007/s13595-019-0858-3

Windels, S., and Flaspohler, D. J. (2011). The ecology of Canada Yew (Taxus canadensis Marsh.): A review. *Botany* 89, 1–17. doi: 10.1139/B10-084

Wisconsin Department of Natural Resources (2020). *Wisconsin 2020 Statewide Forest Action Plan*. Madison, WI: Wisconsin Department of Natural Resources.

Wolfkill, J., Bejarano, M. E., Serfass, T. L., Turner, G., Brosi, S., Feller, D., et al. (2021). The prevalence of the raccoon roundworm, Baylisascaris procyonis, in allegheny woodrat habitat in the Mid-Atlantic Region, USA. *Am. Midland Naturalist* 185, 145–147.

Wuerthner, G., Crist, E., and Butler, T. (eds) (2015). *Protecting the Wild: parks and wilderness, the foundation for conservation*. London: Island Press.

WWF (2022). *Living planet report 2022 – Building a nature-positive society*, eds R. E. A. Almond, M. Grooten, D. Juffe Bignoli, and T. Petersen (Gland: WWF).

Xu, X., Huang, A., Belle, E., De Frenne, P., and Jia, G. (2022). Protected areas provide thermal buffer against climate change. *Sci. Adv.* 8:eabo0119. doi: 10.1126/sciadv.abo0119

Yamasaki, M., Costello, C. A., and Leak, W. B. (2014). *Effects of clearcutting, patch cutting, and low-density shelterwoods on breeding birds and tree regeneration in New Hampshire northern hardwoods. Res. Pap. NRS- 26*. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station, 15. doi: 10.2737/NRS-RP-26

Yeo, S., Weber, C., Carvalho, F., Clare, L., Woods, M., Merriman, P., et al. (2019). *Climate, nature and our 1.5°C future: a synthesis of IPCC and IPBES reports*. Gland: WWF International.

Young Forest Project (2012). *The solution: a new understanding*. Available online at: https://web.archive.org/web/20130427082104/http://youngforest.org/the-solution (accessed October 2, 2022).

Young Forest Project (2022a). *Partners*. Available online at: https://youngforest.org/partners (accessed October 2, 2022).

Young Forest Project (2022b). *Wildlife and sport fish restoration program boosts young forest*. Available online at: https://youngforest.org/content/wsfr-funding (accessed November 7, 2022).

Young Forest Project (2022c). *The challenge: we're losing young forest on the land*. Available online at: https://youngforest.org/the-challenge (accessed November 7, 2022).

Young Forest Project (2022d). *Fort Indiantown Gap, Southeastern Pennsylvania: "Training-scape" helps soldiers, wildlife*. Available online at: https://youngforest.org/demo/fort-indiantown-gap-southeastern-pennsylvania (accessed October 2, 2022).

Zaplata, M. K., and Dullau, S. (2022). Applying ecological succession theory to birds in solar parks: An approach to address protection and planning. *Land* 11:718. doi: 10.3390/land11050718

Zhao, D., Sun, Z., Wang, C., Hao, Z., Sun, B., Zuo, Q., et al. (2020). Using count data models to predict epiphytic bryophyte recruitment in *Schima superba* Gardn. et Champ. Plantations in urban forests. *Forests* 11:174. doi: 10.3390/f11020174

Zheng, D., Heath, L. S., and Ducey, M. J. (2008). Spatial distribution of forest aboveground biomass estimated from remote sensing and forest inventory data in New England, USA. *J. Appl. Remote Sens.* 2:021502. doi: 10.1117/1.2940686

Zheng, D., Heath, L. S., and Ducey, M. J. (2013). Carbon benefits from protected areas in the conterminous United States. *Carbon Balance Manag.* 8:4. doi: 10.1186/1750-0680-8-4

Zheng, D., Heath, L. S., Ducey, M. J., and Butler, B. (2010). Relationships between major ownerships, forest aboveground biomass distributions, and landscape dynamics in the New England region of USA. *Environ. Manag.* 45, 377–386. doi: 10.1007/s00267-009-9408-3

Zhou, G., Liu, S., Li, Z., Zhang, D., Tang, X., Zhou, C., et al. (2006). Old-growth forests can accumulate carbon in soils. *Science* 314:1417. doi: 10.1126/science.1130168

Zlonis, E. J., and Niemi, G. J. (2014). Avian communities of managed and wilderness hemiboreal forests. *For. Ecol. Manag.* 328, 26–34. doi: 10.1016/j.foreco.2014.05.017

Zuckerberg, B., and Porter, W. F. (2010). Thresholds in the long-term responses of breeding birds to forest cover and fragmentation. *Biol. Conserv.* 143, 952?962. doi: 10.1016/j.biocon.2010.01.004

frontiers in Forests and Global Change

PERSPECTIVE
published: 11 June 2019
doi: 10.3389/ffgc.2019.00027

Intact Forests in the United States: Proforestation Mitigates Climate Change and Serves the Greatest Good

William R. Moomaw[1], Susan A. Masino[2,3] and Edward K. Faison[4]*

[1] Emeritus Professor, The Fletcher School and Co-director Global Development and Environment Institute, Tufts University, Medford, MA, United States, [2] Vernon Roosa Professor of Applied Science, Trinity College, Hartford, CT, United States, [3] Charles Bullard Fellow in Forest Research, Harvard Forest, Petersham, MA, United States, [4] Senior Ecologist, Highstead Foundation, Redding, CT, United States

Climate change and loss of biodiversity are widely recognized as the foremost environmental challenges of our time. Forests annually sequester large quantities of atmospheric carbon dioxide (CO_2), and store carbon above and below ground for long periods of time. Intact forests—largely free from human intervention except primarily for trails and hazard removals—are the most carbon-dense and biodiverse terrestrial ecosystems, with additional benefits to society and the economy. Internationally, focus has been on preventing loss of tropical forests, yet U.S. temperate and boreal forests remove sufficient atmospheric CO_2 to reduce national annual *net* emissions by 11%. U.S. forests have the potential for much more rapid atmospheric CO_2 removal rates and biological carbon sequestration by intact and/or older forests. The recent *1.5 Degree Warming Report* by the Intergovernmental Panel on Climate Change identifies *reforestation* and *afforestation* as important strategies to increase negative emissions, but they face significant challenges: afforestation requires an enormous amount of additional land, and neither strategy can remove sufficient carbon by growing young trees during the critical next decade(s). In contrast, growing existing forests intact to their ecological potential—termed *proforestation*—is a more effective, immediate, and low-cost approach that could be mobilized across suitable forests of all types. Proforestation serves the greatest public good by maximizing co-benefits such as nature-based biological carbon sequestration and unparalleled ecosystem services such as biodiversity enhancement, water and air quality, flood and erosion control, public health benefits, low impact recreation, and scenic beauty.

Keywords: biodiversity crisis, Pinchot, afforestation, reforestation, forest ecosystem, biological carbon sequestration, old-growth forest, second-growth forest

OPEN ACCESS

Edited by:
Alexandra C. Morel,
University of Oxford, United Kingdom

Reviewed by:
Don Waller,
University of Wisconsin System, United States
Dominick Anthony DellaSala,
Geos Institute, United States

***Correspondence:**
William R. Moomaw
william.moomaw@tufts.edu

Specialty section:
This article was submitted to Tropical Forests, a section of the journal Frontiers in Forests and Global Change

***Received:** 19 January 2019*
***Accepted:** 20 May 2019*
***Published:** 11 June 2019*

Citation:
Moomaw WR, Masino SA and Faison EK (2019) Intact Forests in the United States: Proforestation Mitigates Climate Change and Serves the Greatest Good. Front. For. Glob. Change 2:27. doi: 10.3389/ffgc.2019.00027

INTRODUCTION

Life on Earth as we know it faces unprecedented, intensifying, and urgent imperatives. The two most urgent challenges are (1) mitigating and adapting to climate change (Intergovernmental Panel on Climate Change, 2013, 2014, 2018), and (2) preventing the loss of biodiversity (Wilson, 2016; IPBES, 2019). These are three of the Sustainable Development Goals, Climate, Life on Land and Life under Water (Division for Sustainable Development Goals, 2015), and significant international resources are being expended to address these crises and limit negative impacts on economies, societies and biodiverse natural communities. The recent *1.5 Degree Warming Report* of the Intergovernmental Panel on Climate Change (2018) was dire and direct, stating the need for "rapid, far-reaching and unprecedented changes in all aspects of society." We find that growing additional existing forests as intact ecosystems, termed *proforestation*, is a low-cost approach for immediately increasing atmospheric carbon sequestration to achieve a stable atmospheric carbon dioxide concentration that reduces climate risk. Proforestation also provides long-term benefits for biodiversity, scientific inquiry, climate resilience, and human benefits. This approach could be mobilized across all forest types.

Forests are essential for carbon dioxide removal (CDR), and the CDR rate needs to increase rapidly to remain within the 1.5 or 2.0°C range (Intergovernmental Panel on Climate Change, 2018) specified by the Paris Climate Agreement (2015). Growing existing forests to their biological carbon sequestration potential optimizes CDR while limiting climate change and protecting biodiversity, air, land, and water. Natural forests are by far the most effective (Lewis et al., 2019). Technologies for direct CDR from the atmosphere, and bioenergy with carbon capture and storage (BECCS), are far from being technologically ready or economically viable (Anderson and Peters, 2016). Furthermore, the land area required to supply BECCS power plants with tree plantations is 7.7 million km^2, or approximately the size of Australia (Intergovernmental Panel on Climate Change, 2018). Managed plantations that are harvested periodically store far less carbon because trees are maintained at a young age and size (Harmon et al., 1990; Sterman et al., 2018). Furthermore, plantations are often monocultures, and sequester less carbon more slowly than intact forests with greater tree species diversity and higher rates of biological carbon sequestration (Liu et al., 2018). Recent research in the tropics shows that natural forests hold 40 times more carbon than plantations (Lewis et al., 2019).

Alternative forest-based CDR methods include *afforestation* (planting new forests) and *reforestation* (replacing forests on deforested or recently harvested lands). Afforestation and reforestation can contribute to CDR, but newly planted forests require many decades to a century before they sequester carbon dioxide in substantial quantities. A recent National Academy study titled *Negative Emissions Technologies and Reliable Sequestration: A Research Agenda* discusses afforestation and reforestation and finds their contribution to be modest (National Academies of Sciences, 2019). The study also examines changes in conventional forest management, but neglects proforestation as a strategy for increasing carbon sequestration. Furthermore, afforestation to meet climate goals requires an estimated 10 million km^2-an area slightly larger than Canada (Intergovernmental Panel on Climate Change, 2018). The massive land areas required for afforestation and BECCS (noted above) compete with food production, urban space and other uses (Searchinger et al., 2009; Sterman et al., 2018). More importantly, neither of these two practices is as effective quantitatively as proforestation in the next several decades when it is needed most. For example, Law et al. (2018) reported that extending harvest cycles and reducing cutting on public lands had a larger effect than either afforestation or reforestation on increasing carbon stored in forests in the Northwest United States. In other regions such as New England (discussed below), longer harvest cycles and proforestation are likely to be even more effective. Our assessment on the climate and biodiversity value of natural forests and proforestation aligns directly with a recent report that pinpointed "stable forests" – those not already significantly disturbed or at significant risk – as playing an outsized role as a climate solution due to their carbon sequestration and storage capabilities (Funk et al., 2019).

Globally, terrestrial ecosystems currently remove an amount of atmospheric carbon equal to one-third of what humans emit from burning fossil fuels, which is about 9.4 GtC/y (10^9 metric tons carbon per year). Forests are responsible for the largest share of the removal. Land use changes, i.e., conversion of forest to agriculture, urban centers and transportation corridors, emit ~1.3 GtC/y (Le Quéré et al., 2018). However, forests' potential carbon sequestration and additional ecosystem services, such as high biodiversity unique to intact older forests, are also being degraded significantly by current management practices (Foley et al., 2005; Watson et al., 2018). Houghton and Nassikas (2018) estimated that the "current gross carbon sink in forests recovering from harvests and abandoned agriculture to be −4.4 GtC/y, globally." This is approximately the current gap between anthropogenic emissions and biological carbon and ocean sequestration rates by natural systems. If deforestation were halted, and secondary forests were allowed to continue growing, they would sequester −120 GtC between 2016 and 2100 or ~12 years of current global fossil carbon emissions (Houghton and Nassikas, 2018). Northeast secondary forests have the potential to increase biological carbon sequestration between 2.3 and 4.2-fold (Keeton et al., 2011).

Existing proposals for "Natural Climate Solutions" do not consider explicitly the potential of proforestation (Griscom et al., 2017; Fargione et al., 2018). However, based on a growing body of scientific research, we conclude that protecting and stewarding intact diverse forests and practicing proforestation as a purposeful public policy on a large scale is a highly effective strategy for mitigating the dual crises in climate and biodiversity and ultimately serving the "greatest good" in the United States and the rest of the world. **Table 1** summarizes some of the key literature supporting this point.

A SMALL FRACTION OF U.S. FORESTS IS MANAGED TO REMAIN INTACT

Today, <20% of the world's forests remain intact (i.e., largely free from logging and other forms of extraction and development). Intact forests are largely tropical forests or boreal forests in Canada and Russia (Watson et al., 2018). In the U.S.—a global pioneer in national parks and wildlife preserves—the percentage of intact forest in the contiguous 48 states is only an estimated 6-7% of total forest area (Oswalt et al., 2014), with a higher proportion in the West and a lower proportion in the East. Setting aside a large portion of U.S. forest in Inventoried Roadless Areas (IRAs) was groundbreaking yet only represents 7% of total forest area in the lower 48 states—and, ironically,

TABLE 1 | Comparison of climate and biodiversity benefits of *intact* (either old-growth forest or younger forest managed as Gap 1 or Gap 2, and thus protected from logging and other resource extraction) and traditionally *managed* forests for multiple forest types in the United States.

	Location	Forest type	Forest condition with greater value	References
ECOSYSTEM CHARACTERISTICS				
Density of large trees (>60 cm DBH)	Eastern US	mid-Atlantic oak-hickory forests, northern hemlock-hardwood forests, and boreal spruce-fir forests	Intact (81% greater)	Miller et al., 2016
Proportion of old forest	Eastern US	Same as above	Intact	Miller et al., 2016
Basal area of dead standing trees	Eastern US	Same as above	Intact	Miller et al., 2016
Coarse woody debris volume	Eastern US	Same as above	Intact (135% greater)	Miller et al., 2016
Carbon storage	Pacific Northwest US	Douglas fir and western hemlock;	Intact (75–138% greater)	Harmon et al., 1990
Carbon storage	Northeastern US	Northern hardwood conifer	Intact (39–118% greater)	Nunery and Keeton, 2010
Forest fire burn severity	Western US	Pine and mixed conifer forests	Managed (two SEs greater)	Bradley et al., 2016
BIODIVERSITY				
Tree species richness	Eastern US	mid-Atlantic oak-hickory forests, northern hemlock-hardwood forests, and boreal spruce-fir forests	Intact	Miller et al., 2018
Proportion rare tree species	Eastern US	Same as above	Intact	Miller et al., 2018
Bird species richness and abundance	Northeastern Minnesota	Hemi-boreal	Intact (12–20% greater)	Zlonis and Niemi, 2014
Trunk bryophyte and lichen species richness	Northwestern Montana	Grand-fir	Intact (33% greater)	Lesica et al., 1991
Salamander density	Ozark Mountains, Missouri	Oak-hickory	Intact (395–9,500% greater)	Herbeck and Larsen, 1999
Probability of occurrence of invasive plant species	Eastern US	Deciduous and mixed forest	managed	Riitters et al., 2018

Intact forests range in size and previous disturbance history but they are not under active management and have been allowed to continue growing according to the procedures described for proforestation.

management of some IRAs allows timber harvest and road building (Williams, 2000), a scenario happening currently in the Tongass National Forest in Alaska (Koberstein and Applegate, 2018). These scant percentages worldwide and particularly in the U.S. are insufficient to address pressing national and global issues such as rising CO_2 levels, flooding, and biodiversity loss, as well as provide suitable locations for recreation and associated public health benefits (Cordell, 2012; Watson et al., 2018). In heavily populated and heavily forested sub-regions in the Eastern U.S., such as New England, the total area dedicated as intact (i.e., primary management is for trails and hazard removals) is even more scarce, comprising only ~3% of land area. Just 2% of the region is legally protected from logging and other resource extraction (**Figure 1**). A large portion of forest managed currently as intact or "reserved forest" – and thus functioning as "stable forest" (Funk et al., 2019) – is designated solely by administrative regulations that can be altered at any time.

Intact forests in the U.S. include federal wilderness areas and national parks, some state parks, and some privately-owned holdings and conservation trust lands. Recent studies reveal that intact forests in national parks tend to be older and have larger trees than nearby forests that are not protected from logging (Miller et al., 2016; **Table 1**). Scaling up protection of intact forests and designating and significantly expanding reserved forest areas are public policy imperatives that are compatible with public access and with the country's use of forest products. Identifying suitable forest as intact (for carbon sequestration, native biodiversity, ecosystem function, etc.) can spawn new jobs and industries in forest monitoring, tourism and recreation, as well as create more viable local economies based on wood reuse and recycling. Public lands with significant biodiversity and proforestation potential also provide wildlife corridors for climate migration and resilience for many species.

PROFORESTATION INCREASES BIOLOGICAL CARBON SEQUESTRATION AND LONG-TERM STORAGE IN U.S. FORESTS

Net forest carbon reflects the dynamic between gains and losses. Carbon is lost from forests in several ways: damage from natural disturbances including insects and pathogens ("pests"), fire, drought and wind; forest conversion to development or other non-forest land; and forest harvest/management. Together, fires, drought, wind, and pests account for ~12% of the carbon lost in the U.S.; forest conversion accounts for ~3% of carbon loss; and forest harvesting accounts for 85% of the carbon lost from forests each year (Harris et al., 2016). Forests in the Southern US have the highest percentage of carbon lost to timber harvest (92%) whereas the Western US is notably lower (66%) because of the

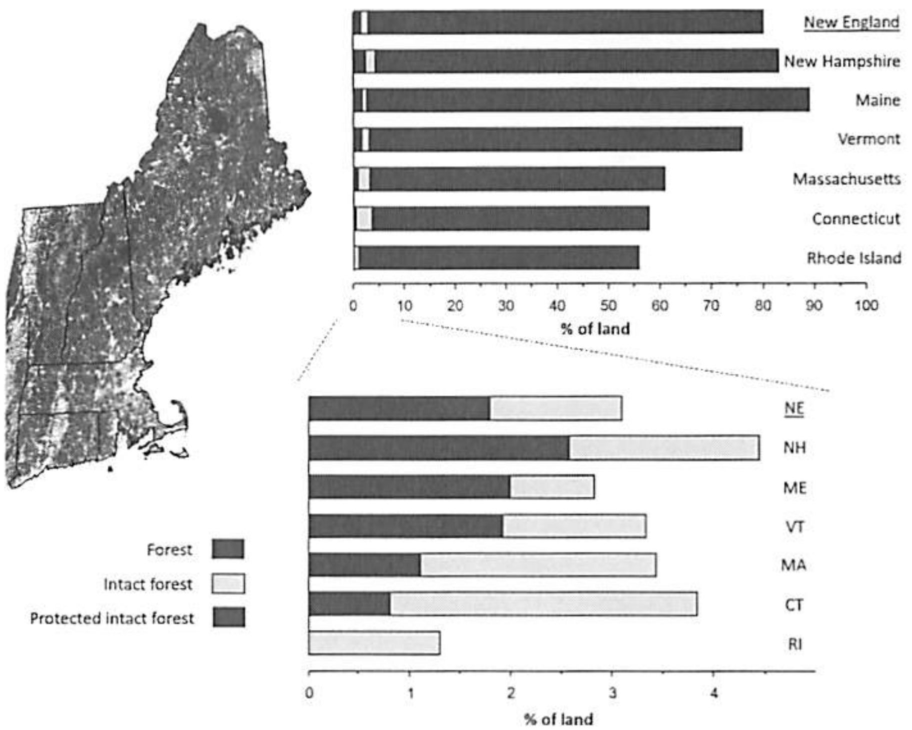

FIGURE 1 | Distribution of forest cover and intact "wildland" forest across six New England states. At left, map of overall forest cover (green) vs. forest protected legally (red) or managed currently (yellow) as intact in New England. At right, regional and state specific % forest cover (green), % managed as intact Gap 1 (limited intervention other than trails and hazard removals) but not protected legally (yellow), and % legally protected as intact forest (red, designated U.S Geological Survey (USGS) Gap 1 or Gap 2 and primarily federal and state wilderness areas, and certain national parks). Adapted and compiled from National Conservation Easement Database (2014); United States Geological Survey (2019a,b), and the University of Montana (2019). USGS Gap level 1 or 2 lands receive the highest level of protection from logging and other resource extraction and generally correspond with IUCN protected categories 1a, 1b, and II (https://gapanalysis.usgs.gov/blog/iucn-definitions/).

greater contribution of fires to carbon removal. The Northern U.S. is roughly equivalent to the national average at 86% (Harris et al., 2016).

Proforestation produces natural forests as maximal carbon sinks of diverse species (while supporting and accruing additional benefits of intact forests) and can reduce significantly and immediately the amount of forest carbon lost to non-essential management. Because existing trees are already growing, storing carbon, and sequestering more carbon more rapidly than newly planted and young trees (Harmon et al., 1990; Stephenson et al., 2014; Law et al., 2018; Leverett and Moomaw, in preparation), proforestation is a near-term approach to sequestering additional atmospheric carbon: a significant increase in "negative emissions" is urgently needed to meet temperature limitation goals.

The carbon significance of proforestation is demonstrated in multiple ways in larger trees and older forests. For example, a study of 48 undisturbed primary or mature secondary forest plots worldwide found, on average, that the largest 1% of trees [considering all stems $\geq$1 cm in diameter at breast height (DBH)] accounted for half of above ground living biomass (The largest 1% accounted for ~30% of the biomass in U.S. forests due to larger average size and fewer stems compared to the tropics) (Lutz et al., 2018). Each year a single tree that is 100 cm in diameter adds the equivalent biomass of an entire 10–20 cm diameter tree, further underscoring the role of large trees (Stephenson et al., 2014). Intact forests also may sequester half or more of their carbon as organic soil carbon or in standing and fallen trees that eventually decay and add to soil carbon (Keith et al., 2009). Some older forests continue to sequester additional soil organic carbon (Zhou et al., 2006) and older forests bind soil organic matter more tightly than younger ones (Lacroix et al., 2016).

If current management practices continue, the world's forests will only achieve half of their biological carbon sequestration potential (Erb et al., 2018); intensifying current management practices will only decrease living biomass carbon and increase soil carbon loss. Forests in temperate zones such as in the Eastern U.S. have a particularly high untapped capacity for carbon storage and sequestration because of high growth and low decay rates (Keith et al., 2009) and because of recent recovery from an extensive history of timber harvesting and land conversion for agriculture in the 18th, 19th, and early 20th centuries (Pan et al., 2011; Duveneck and Thompson, 2019). In New England, median forest age is about 75 years of age (United States Forest Service, 2019), which is only about 25–35% of the lifespan of many of the common tree species in these forests (Thompson et al., 2011). Much of Maine's forests have been harvested continuously for 200 years and have a carbon density less than one-third of the forests of Southern Vermont and New Hampshire, Northwestern Connecticut and Western Massachusetts—a region that has not been significantly harvested over the past 75–150 years (National Council for Air Stream Improvement, 2019). Western Massachusetts in particular has a significant portion classifed as Tier 1 matrix forest, defined as "*large contiguous areas whose size and natural condition allow for the maintenance of ecological processes*" (Databasin, 2019). However, forests managed as intact do not need to be large or old in absolute terms to have ecological value: disturbances create gaps and young habitats, and the official policy of the Commonwealth of Massachusetts Department of Environmental Management (now Department of Conservation and Recreation) considers an old-growth forest of at least 2 hectares ecologically significant (Department of Environmntal Management, 1999).

As shown in **Table 1**, ecosystem services accrue as forests age for centuries. Far from plateauing in terms of carbon sequestration (or added wood) at a relatively young age as was long believed, older forests (e.g., >200 years of age without intervention) contain a variety of habitats, typically continue to sequester additional carbon for many decades or even centuries, and sequester significantly more carbon than younger and managed stands (Luyssaert et al., 2008; Askins, 2014; McGarvey et al., 2015; Keeton, 2018). A recent paper affirmed that letting forests grow is an effective way to sequester carbon—but unlike previous studies it suggested that sequestration is highest in "young" forests (Pugh et al., 2019). This conclusion is problematic for several reasons. One confounding factor is that older forests in the tropics were compared to young forests in temperate and boreal areas; temperate forests in particular have the highest CO_2 removal rates and overall biological carbon sequestration (Keith et al., 2009) but this high rate is not limited to young temperate and boreal forests. The age when sequestration rates decrease is not known, and Pugh et al. defined "young" as up to 140 years. As noted above, Keeton et al. (2011) estimate that secondary forests in the Northeast have the potential to increase their biological carbon sequestration several-fold. More field work is needed across age ranges, species and within biomes, but the inescapable conclusion is that growing forests is beneficial to the climate and maintaining intact forest has additional benefits (**Table 1**). We conclude that proforestation has the potential to provide rapid, additional carbon sequestration to reduce *net* emissions in the U.S. by much more than the 11% that forests provide currently (United States Environmental Protection Agency, 2019). A recent report on natural climate solutions determined that negative emissions could be increased from 11 to 21% even without including proforestation (Fargione et al., 2018). Quantified estimates of increased forest sequestration and ecosystem services were based on re-establishing forests where possible and lengthening rotation times on private land; they explicitly did not account for proforestation potential on public land.

Although biological carbon storage in managed stands, regardless of the silvicultural prescription, is generally lower than in unmanaged intact forests (Harmon et al., 1990; Ford and Keeton, 2017)—even after the carbon stored in wood products is included in the calculation—stands managed with reduced harvest frequency and increased structural retention sequester more carbon than more intensively managed stands (Nunery and Keeton, 2010; Law et al., 2018). Such an approach for production forests, or "working" forests—balancing resource extraction with biological carbon sequestration—is often termed "managing for net carbon" or "managing for climate change" and an approach that should be promoted alongside dedicating significant areas to intact ecosystems. Oliver et al. (2014) acknowledge a balance between intact and managed forest and suggest that long term storage in "efficient" wood products like wood building materials (with the potential for less carbon emissions compared to steel or concrete, termed the "avoidance pathway") can offer a significant carbon benefit. To achieve this, some questionable assumptions are that 70% of the harvested wood is merchantable and stored in a lasting product, all unmerchantable wood is removed and used, harvesting occurs at optimum intervals (100 years) and carbon sequestration tapers off significantly after 100 years. Forestry models underestimate the carbon content of older, larger trees, and it is increasingly clear that trees can continue to remove atmospheric carbon at increasing rates for many decades beyond 100 years (Robert T. Leverett, pers. comm. Stephenson et al., 2014; Lutz et al., 2018; Leverett et al., under review). Because inefficient logging practices result in substantial instant carbon release to the atmosphere, and only a small fraction of wood becomes a lasting product, increasing market forces and investments toward wood buildings that have relatively short lifetimes could increase forest extraction rates significantly and become unsustainable (Oliver et al., 2014).

HABITAT PROTECTION, BIODIVERSITY AND SCIENTIFIC VALUE OF PROFORESTATION

Large trees and intact, older forests are not only effective and cost-effective natural reservoirs of carbon storage, they also provide essential habitat that is often missing from younger, managed forests (Askins, 2014). For example, intact forests in Eastern U.S. national parks have greater tree diversity, live and dead standing basal area, and coarse woody debris, than forests that are managed for timber (Miller et al., 2016, 2018; **Table 1**). The density of cavities in older trees and the spatial and structural heterogeneity of the forest increases with stand age (Ranius et al., 2009; Larson et al., 2014), and large canopy gaps develop as a result of mortality of large trees, which result in dense patches of regeneration (Askins, 2014). These complex structures and habitat features support a greater diversity of lichens and bryophytes (Lesica et al., 1991), a greater density and diversity of salamanders (Petranka et al., 1993; Herbeck and Larsen, 1999), and a greater diversity and abundance of birds in old, intact forests than in nearby managed forests (Askins, 2014; Zlonis and Niemi, 2014; **Table 1**). Forest bird guilds also benefit from small intact forests in urban landscapes relative to unprotected matrix forests (Goodwin and Shriver, 2014). Several bird species in the U.S. that are globally threatened—including the wood thrush, cerulean warbler, marbled murrelet, and spotted owl are, in part, dependent on intact, older forests with large trees (International Union for Conservation of Nature, 2019). Two species that are extinct today—Bachman's warbler and Ivory-billed woodpecker—likely suffered from a loss of habitat features associated with old forests (Askins, 2014).

Today, forest managers often justify management to maintain heterogeneity of age structures to enhance wildlife habitat and maintain "forest health" (Alverson et al., 1994). However, early successional forest species (e.g., chestnut-sided warbler and New England cottontail) that are common targets for forest management may be less dependent on forest management than is commonly believed (cf. Zlonis and Niemi, 2014; Buffum et al., 2015). Management also results in undesirable consequences such as soil erosion, introduction of invasive and non-native species (McDonald et al., 2008; Riitters et al., 2018), loss of carbon—including soil carbon (Lacroix et al., 2016), increased densities of forest ungulates such as white-tailed deer (Whitney, 1990)—a species that can limit forest regeneration (Waller, 2014)—and a loss of a sense of wildness (e.g., Thoreau, 1862).

Forest health is a term often defined by a particular set of forestry values (e.g., tree regeneration levels, stocking, tree growth rates, commercial value of specific species) and a goal of eliminating forest pests. Although appropriate in a commercial forestry context, these values should not be conflated with the ability of intact natural forests to continue to function and even thrive indefinitely and provide a diversity of habitats on their own (e.g., Zlonis and Niemi, 2014). Natural forests, regardless of their initial state, naturally develop diverse structures as they age and require from us only the time and space to self-organize (e.g., Larson et al., 2014; Miller et al., 2016).

Intact forests provide irreplaceable scientific value. In addition to a biodiverse habitat an intact forest provides an area governed by natural ecological processes that serve as important scientific controls against which to compare the effects of human activities and management practices (Boyce, 1998). Areas without resource extraction (i.e., timber harvesting, hunting), pest removal, or fire suppression allow for a full range of natural ecological processes (fire, herbivory, natural forest development) to be expressed (Boyce, 1998). Only if we have sufficient natural areas can we hope to understand the effects of human activities on the rest of our forests. Additional research and monitoring projects that compare ecological attributes between intact and managed forests at a range of spatial scales will also help determine how effective protected intact forests can be at conserving a range of biota, and where additional protected areas may need to be established (e.g., Goodwin and Shriver, 2014; Jenkins et al., 2015).

PROFORESTATION AND FOREST FIRES

Given the increase in forest area burned in the United States over the past 30 years (National Interagency Fire Center, 2019), it is important to address the relationship between forest management and forest fires. There is a widely held perception that the severity and size of recent fires are directly related to the fuels that have accumulated in the understory due to a lack of forest management to reduce these fuels (i.e., pulping, masticating, thinning, raking, and prescribed burning; Reinhardt et al., 2008; Bradley et al., 2016). However, some evidence suggests that proforestation should actually *reduce* fire risk and there are at least three important factors to consider: first, fire is an integral part of forest dynamics in the Western U.S.; second, wildfire occurrence, size, and area burned are generally not preventable even with fuel removal treatments (Reinhardt et al., 2008); and third, the area burned is actually far less today than in the first half of the twentieth century when timber harvesting was more intensive and fires were not actively suppressed (Williams, 1989; National Interagency Fire Center, 2019). Interestingly, in the past 30 years, intact forests in the Western U.S. burned at significantly *lower* intensities than did managed forests (Thompson et al., 2007; Bradley et al., 2016; **Table 1**). Increased potential fuel in intact forests appear to be offset by drier conditions, increased windspeeds, smaller trees, and residual and more combustible fuels inherent in managed areas (Reinhardt et al., 2008; Bradley et al., 2016). Rather than fighting wildfires wherever they occur, the most effective strategy is limiting development in fire-prone areas, creating and defending zones around existing development (the wildland-urban interface), and establishing codes for fire-resistant construction (Cohen, 1999; Reinhardt et al., 2008).

PROFORESTATION AND ECOSYSTEM SERVICES: SERVING THE GREATEST GOOD

In 1905 Gifford Pinchot, Chief of the U.S. Forest Service, summarized his approach to the nation's forests when he wrote "...where conflicting interests must be reconciled, the question will always be decided from the standpoint of the greatest good of the greatest number in the long run." This ethos continues to define the management approach of the U.S. Forest Service from its inception to the present day. Remarkably, however, even in 2018 the five major priorities of the Forest Service do not mention biodiversity, carbon storage, or climate change as major aspects of its work (United States Forest Service, 2018).

Today, the needs of the nation have changed: emerging forest science and the carbon and biodiversity benefits of proforestation demand a focus on growing intact natural public and private forests, including local parks and forest reserves (Jenkins et al., 2015). There is also a growing need across the country, and particularly within reach of highly populated areas, for additional local parks and protected forest reserves that serve and provide the public with solitude, respite, and wild experiences (e.g., Thoreau, 1862). Detailed analysis of over one thousand public comments regarding management of Hoosier National Forest, a public forest near population centers in several states, revealed a strong belief that wilderness contributes to a sense of well-being. Responses with the highest frequency reflected an interest in preservation and protection of forests and wildlife, a recognition of the benefits to human physical and mental health, a sense of ethical responsibility, opposition to damage and destruction, monetary concerns, and a preponderance of sadness, fear and distress over forest loss (Vining and Tyler, 1999).

Quantifiable public health benefits of forests and green spaces continue to emerge, and benefits are highest in populations with chronic and difficult-to-treat conditions like anxiety, depression, pain and post-traumatic stress disorder (Karjalainen et al., 2010; Frumkin et al., 2017; Hansen et al., 2017; Oh et al., 2017). In the United Kingdom "growing forests for health" is the motto of the National Health Service Forest (2019) and there is a recognized need for evidence-based analysis of human health co-benefits alongside nature-based ecosystem services (Frumkin et al., 2017).

POLICY RECOMMENDATIONS

To date, the simplicity of the idea of proforestation has perhaps been stymied by inaccurate or non-existent terminology to describe it. Despite a number of non-binding international forest agreements (United Nations Conference on Environment Development, 1992; United Nations Forum on Forests, 2008; Forest Declaration, 2014) and responsibilities by a major UN organization [Food and Agriculture Organization (FAO)], current climate policies lack science-based definitions that distinguish forest condition—including the major differences between young and old forests across a range of ecosystem services. Lewis et al. (2019) further note that broad definitions and confused terminology have an unfortunate result that policymakers and their advisers mislead the public (Lewis et al., 2019). Most discussions concerning forest loss and forest protection are in terms of percentage of land area that has tree canopy cover (Food and Agriculture Organization, 2019). This lack of specificity significantly hampers efforts to evaluate and protect intact forests, to quantify their value, and to dedicate existing forests as intact forests for the future. For example, the UN Framework Convention on Climate Change and the FAO consider and group tree plantations, production forests, and mature intact forests equally under the general term "forest" (Mackey et al., 2015). In addition, "forest conservation" simply means maintaining "forest cover" and does not address age, species richness or distribution—or the degree that a forest ecosystem is intact and functioning (Mackey et al., 2015). The erroneous assumption is that all forests are equivalently beneficial for a range of ecosystem services—a conclusion that is quantitatively inaccurate in terms of biological carbon sequestration and biodiversity as well as many other ecosystem services.

Practicing proforestation should be emphasized on suitable public lands as is now done in U.S. National Parks and Monuments. Private forest land owners might be compensated to practice proforestation, for sequestering carbon and providing associated co-benefits by letting their forests continue to grow. At this time, we lack national policies that quantify and truly maximize benefits across the landscape. At a regional scale, however, some conservation visions do explicitly recognize and promote the multiple values and services associated with forest reserves or wildlands (e.g., Foster et al., 2010) and climate offset programs can be used explicitly to support proforestation. For example, a recent project by the Nature Conservancy protected 2,185 hectares (5,400 acres) in Vermont as wildland and is expected to yield ~\$2 M over 10 years for assuring long-term biological carbon storage (Nature Conservancy, 2019). Burnt Mountain is now protected by a "forever wild" easement and part of a 4,452 hectare (11,000 acre) preserve. More public education and similar incentives are needed.

CONCLUSIONS

To meet any proposed climate goals of the Paris Climate Agreement (1.5, 2.0° C, targets for reduced emissions) it is essential to simultaneously *reduce greenhouse gas emissions from all sources* including fossil fuels, bioenergy, and land use change, and *increase CDR* by forests, wetlands and soils. Concentrations of these gases are now so high that reducing emissions alone is insufficient to meet these goals. Speculation that untested technologies such as BECCS can achieve the goal while allowing us to continue to emit more carbon has been described as a "moral hazard" (Anderson and Peters, 2016). Furthermore, BECCS is not feasible within the needed timeframe and CDR is urgent. Globally, existing forests only store approximately half of their potential due to past and present management (Erb et al., 2018), and many existing forests are capable of immediate and even more extensive growth for many decades (Lutz et al., 2018). During the timeframe while seedlings planted for afforestation and reforestation are growing (yet will never achieve the carbon density of an intact forest), proforestation is a safe, highly effective, immediate natural solution that does not rely on uncertain discounted future benefits inherent in other options.

Taken together, proforestation is a rapid and essential strategy for achieving climate and biodiversity goals and for serving the greatest good. Stakeholders and policy makers need to recognize that the way to maximize carbon storage and sequestration is to grow intact forest ecosystems where possible. Certainly, all forests have beneficial attributes, and the management focus of some forests is providing wood products that we all use. But until we acknowledge and quantify differences in forest status (Foster et al., 2010), we will be unable to develop policies (and educate landowners, donors, and the public) to support urgent forest-based benefits in the most effective, locally appropriate and cost-effective manner. A differentiation between production forests and natural forest ecosystems would garner public support for a forest industry with higher value products and a renewed focus on reducing natural resource use—and for recycling paper and wood. It could also spur long-overdue local partnerships between farms and forests—responsible regional composting keeps jobs and resources within local communities while improving soil health and increasing soil carbon (Brown and Cotton, 2011). The forest industry as a whole can benefit from proforestation-based jobs that focus on scientific data collection, public education, public health and a full range of ecosystem services.

In sum, proforestation provides the most effective solution to dual global crises—climate change and biodiversity loss. It is the only practical, rapid, economical, and effective means for atmospheric CDR among the multiple options that have been proposed because it removes more atmospheric carbon dioxide in the immediate future and continues to sequester it long-term. Proforestation will increase the diversity of many groups of organisms and provide numerous additional and important ecosystem services (Lutz et al., 2018). While multiple strategies will be needed to address global environmental crises, proforestation is a very low-cost option for increasing carbon sequestration that does not require additional land beyond what is already forested and provides new forest related jobs and opportunities along with a wide array of quantifiable ecosystem services, including human health.

AUTHOR CONTRIBUTIONS

WM, SM, and EF contributed equally to conceiving, writing and editing this manuscript and all agree to its publication.

FUNDING

Supported by Charles Bullard Fellowship in Forest Research, Harvard Forest (SM).

ACKNOWLEDGMENTS

The authors thank the reviewers for improving the manuscript with substantive and thoughtful comments and thank David N. Ruskin, Ph.D. (Trinity College) for feedback and assistance throughout.

REFERENCES

Alverson, W. S., Waller, D., and Kuhlmann, W. (1994). *Wild Forests: Conservation Biology and Public Policy*. Washington, DC: Island Press.

Anderson, K., and Peters, G. (2016). The trouble with negative emissions. *Science* 354, 182–183 doi: 10.1126/science.aah4567

Askins, R. A. (2014). *Saving the World's Deciduous Forests: Ecological Perspectives From East Asia, North America, and Europe*. New Haven, CT: Yale University Press.

Boyce, M. S. (1998). Ecological-process management and ungulates: Yellowstone's conservation paradigm. *Wildlife Soc. Bull.* 26, 391–398.

Bradley, C. M., Hanson, C. T., and DellaSala, D. A. (2016). Does increased forest protection correspond to higher fire severity in frequent-fire forests of the western United States? *Ecosphere* 7:e01492. doi: 10.1002/ecs2.1492

Brown, S., and Cotton, N. (2011). Changes in soil properties and carbon content following compost application: Results of on-farm sampling. *Compost Sci. Util.* 19, 87–96. doi: 10.1080/1065657X.2011.10736983

Buffum, B., McGreevy, T. J. Jr., Gottfried, A. E., Sullivan, M. E., and Husband, T. P. (2015). An analysis of overstory tree canopy cover in sites occupied by native and introduced cottontails in the Northeastern United States with recommendations for habitat management for new England Cottontail. *PloS ONE* 10:e0135067. doi: 10.1371/journal.pone.0135067

Cohen, J. D. (1999). *Reducing the Wildland Fire Threat to Homes: Where and How Much?* U.S.D.A Forest Service Gen.Tech. Rep., PSW-GTR-173, 189-195. Available online at: https://www.fs.fed.us/rm/pubs_other/rmrs_1999_cohen_j001.pdf (accessed April 15, 2019).

Cordell, H. K. (2012). *Outdoor Recreation Trends and Futures: A Technical Document Supporting the Forest Service 2010 RPA Assessment*. Asheville, NC: United States Department of Agriculture, Southern Research Station. Available online at: https://www.srs.fs.usda.gov/pubs/gtr/gtr_srs150.pdf

Databasin (2019). Available online at: https://databasin.org/datasets/68c240fb9dc14fda8ccd965064fb3321 (accessed April 15, 2019).

Department of Environmntal Management (1999). *Old Growth Policy*. State of Massachusetts, Division of Forest and Parks, Bureau of Forestry.

Division for Sustainable Development Goals (2015). *Sustainable Development Goals*. Knowledge Platform, United Nations. Available online at: https://sustainabledevelopment.un.org/?menu=1300 (accessed april 15, 2019).

Duveneck, M. J., and Thompson, J. R. (2019). Social and biophysical determinants of future forest conditions in New England: effects of a modern land-use regime. *Global Environ. Change* 55, 115–129. doi: 10.1016/j.gloenvcha.2019.01.009

Erb, K.-H., Kastner, T., Plutzar, C., Bais, A. L. S., Carvalhais, N., Fetzel, T., et al. (2018). Unexpectedly large impact of forest management and grazing on global vegetation biomass. *Nature* 553, 73–76. doi: 10.1038/nature25138

Fargione, J. E., Bassett, S., Boucher,. T., Bridgham, S. D., Conant, R. T., Cook-Patton, S. C., et al. (2018). Natural climate solutions for the United States. *Sci Adv.* 4:eaat1869. doi: 10.1126/sciadv.aat1869

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., et al. (2005). Global consequences of land use. *Science* 309, 570–574. doi: 10.1126/science.1111772

Food and Agriculture Organization (2019). *Food and Agriculture Organization of the United Nations*. Available online at: http://www.fao.org/home/en/ (accessed April 15, 2019).

Ford, S. E., and Keeton, W. E. (2017). Enhanced carbon storage through management for old-growth characteristics in northern hardwoods. *Ecosphere* 8, 1–20. doi: 10.1002/ecs2.1721

Forest Declaration (2014). *New York Declaration on Forests*. Available online at: Available online at: http://forestdeclaration.org/about/ (accessed April 15, 2019).

Foster, D. R., Donahue, B. M., Kittredge, D. B., Lambert, K. F., Hunter, M. L., Hall, B. R., et al. (2010). *Wildlands and Woodlands: A Vision for the New England Landscape*. Cambridge, MA: Harvard University Press. Available online at: https://www.wildlandsandwoodlands.org/sites/default/files/Wildlands%20and%20Woodlands%20New%20England.pdf

Frumkin, H., Bratman, G. N., Breslow, S. J., Cochran, B., Kahn, P. H. Jr., Lawler, J. J., et al. (2017). Nature contact and human health: a research agenda. *Environ. Health Perspect.* 125:075001. doi: 10.1289/EHP1663

Funk, J. M., Aguilar-Amuchastegui, N., Baldwin-Cantello, W., Busch, J., Chuvasov, E., Evans, T., et al. (2019). Securing the climate benefits of stable forests. *Clim. Policy*. doi: 10.1080/14693062.2019.1598838

Goodwin, S. E., and Shriver, W. G. (2014). Using a bird community index to evaluate national parks in the urbanized national capital region. *Urban Ecosyst.* 17, 979–990. doi: 10.1007/s11252-014-0363-2

Griscom, B. W., Adams, J., Ellis, P. W., Houghton, R. A., Lomax, G., Miteva, D. A., et al. (2017). Natural climate solutions. *Proc. Natl. Acad. Sci. U.S.A* 114, 11645–11650. doi: 10.1073/pnas.1710465114

Hansen, M. M., Jones, R., and Tocchini, K. (2017). Shinrin-Yoku (forest bathing) and nature therapy: a state-of-the-art review. *Int. J. Environ. Res. Public Health* 14:851. doi: 10.3390/ijerph14080851

Harmon, M. E., Ferrell, W. K., and Franklin, J. F. (1990). Effects on carbon storage of conversion of old-growth forests to young forests. *Science* 247, 699–702. doi: 10.1126/science.247.4943.699

Harris, N. L., Hagen, S. C., Saatchi, S. S., Pearson, T. R. H., Woodall, C. W., Domke, G. M., et al. (2016). Attribution of net carbon change by disturbance type across forest lands of the conterminous United States. *Carbon Balance Manag.* 11:24. doi: 10.1186/s13021-016-0066-5

Herbeck, L. A., and Larsen, D. R. (1999). Plethodontid salamander response to silvicultural practices in Missouri Ozark forests. *Conserv. Biol.* 13, 623–632. doi: 10.1046/j.1523-1739.1999.98097.x

Houghton, R. A., and Nassikas, A. A. (2018). Negative emissions from stopping deforestation and forest degradation, globally. *Glob. Change Biol.* 24, 350–359. doi: 10.1111/gcb.13876

Intergovernmental Panel on Climate Change (2013). "Summary for policymakers," 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*, eds T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, and P. M. Midgley (New York, NY: Cambridge University Press). Available online at: https://www.ipcc.ch/report/ar5/wg1/.

Intergovernmental Panel on Climate Change (2014). "Summary for policymakers," in *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, eds C. B. Field, V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. Maccracken, P. R. Mastrandrea, and L. L. White (New York, NY: Cambridge University Press). Available online at: https://www.ipcc.ch/report/ar5/wg2/.

Intergovernmental Panel on Climate Change (2018). "Summary for Policymakers," in *Global warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C Above Pre-Industrial Levels and Related Global Greenhouse Gas Emission Pathways, in the Context of Strengthening the Global Response to the Threat of Climate Change, Sustainable Development, and Efforts to Eradicate Poverty*, eds V. Masson-Delmotte, P. Zhai, H. O. Pörtner, D. Roberts, J. Skea, P. R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M. I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, and T. Waterfield. (Geneva: World Meteorological Society). Available online at: https://www.ipcc.ch/sr15/.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES). (2019). Available online at: https://www.ipbes.net/

International Union for Conservation of Nature (2019). *IUCN Red List of Threatened Species* Available online at: https://www.iucnredlist.org/ (accessed April 15, 2019).

Jenkins, C. N., Van Houtan, K. S., Pimm, S. L., and Sexton, J. O. (2015). US protected lands mismatch biodiversity priorities. *Proc. Natl. Acad. Sci. U.S.A.* 112, 5081–5086. doi: 10.1073/pnas.1418034112

Karjalainen, E., Sarjala, T., and Raitio, H. (2010). Promoting human health through forests: overview and major challenges. *Environ. Health Prev. Med.* 15, 1–8. doi: 10.1007/s12199-008-0069-2

Keeton, W. S. (2018). "Source or sink? Carbon dynamics in old-growth forests and their role in climate change mitigation," in *Ecology and Recovery of Eastern Old-growth Forests*, eds A. Barton and W. S. Keeton (Washington, DC: Island Press), 340.

Keeton, W. S., Whitman, A. A., McGee, G. C., and Goodale, C. L. (2011). Late-successional biomass development in northern hardwood-conifer forests of the Northeastern United States. *Forest Sci.* 57, 489–505. doi: 10.1093/forestscience/57.6.489

Keith, H., Mackey, B. G., and Lindenmayer, D. B. (2009). Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proc. Natl. Acad. Sci. U.S.A.* 106, 11635–11640. doi: 10.1073/pnas.0901970106

Koberstein, P., and Applegate, J. (2018). *Trump's Great American Forest Liquidation Sale*. Cascadia Times. Available online at: https://www.times.org/forest-liquidation-sale (accessed April 15, 2019).

Lacroix, E. M., Petrenko, C. L., and Friedland, A. J. (2016). Evidence for losses from strongly bound SOM pools after clear cutting in a northern hardwood forest. *Soil Sci.* 181, 202–207. doi: 10.1097/SS.0000000000000147

Larson, A. J., Lutz, J. A., Donato, D. C., Freund, J. A., Swanson, M. E., Hillerislambers, J., et al. (2014). Spatial aspects of tree mortality strongly differ between young and old-growth forests. *Ecology* 96, 2855–2861. doi: 10.1890/15-0628.1

Law, B. E., Hudiburg, T. W., Berner, L. T., Kent, J. J., Buotte, P. C., and Harmon, M. E. (2018). Land use strategies to mitigate climate change in carbon dense temperate forests. *Proc. Natl. Acad. Sci. U.S.A.* 115, 3663–3668. doi: 10.1073/pnas.1720064115

Le Quéré, C., Andrew, R. M., Friedlingstein, P., Sitch, S., Pongratz, J., Manning, A. C., et al. (2018). Global carbon budget 2017. *Earth Syst. Sci. Data* 10, 405–448. doi: 10.5194/essd-10-405-2018

Lesica, P., Mccune, B., Cooper, S. V., and Hong, W. S. (1991). Differences in lichen and bryophyte communities between old-growth and managed second-growth forests in the Swan Valley, Montana *Can. J. Botany* 69, 1745–1755. doi: 10.1139/b91-222

Lewis, S. L., Wheeler, C. E., Mitchard, E. T. A., and Koch, A. (2019). Restoring natural forests is the best way to remove atmospheric carbon. *Nature* 568, 25–28. doi: 10.1038/d41586-019-01026-8

Liu, X., Stefan, T., Jin-Sheng, H., Pascal, A. N., Helge, B., Zhiyao, T., et al. (2018). Tree species richness increases ecosystem carbon storage in subtropical forests. *Proc. R. Soc. B* 285:20181240. doi: 10.1098/rspb.2018.1240

Lutz, J. A., Furniss, T. J., Johnson, D. J., Davies, S. J., Allen, D., Alonso, A., et al. (2018). Global importance of large-diameter trees. *Glob. Ecol. Biogeogr.* 27, 849–864. doi: 10.1111/geb.12747

Luyssaert, S., Schulze, E. D., Börner, A., Knohl, A., Hessenmöller, D., Law, B. E., et al. (2008). Old-growth forests as global carbon sinks. *Nature* 455, 213–215. doi: 10.1038/nature07276

Mackey, B., Dellasala, D. A., Kormos, C., Lindenmayer, D., Kumpel, N., Zimmerman, B., et al. (2015). Policy options for the world's primary forests in multilateral environmental agreements. *Conserv. Lett.* 8, 139–147. doi: 10.1111/conl.12120

McDonald, R. I., Motzkin, G., and Foster, D. R. (2008). Assessing the influence of historical factors, contemporary processes, and environmental conditions on the distribution of invasive species. *J. Torrey Bot. Soc.* 135, 260–271. doi: 10.3159/08-RA-012.1

McGarvey, J. C., Thompson, J. R., Epstein, H. E., and Shugart, H. H. Jr. (2015). Carbon storage in old-growth forests of the Mid-Atlantic: toward better understanding the eastern forest carbon sink. *Ecology* 96, 311–317. doi: 10.1890/14-1154.1

Miller, K. M., Dieffenbach, F. W., Campbell, J. P., Cass, W. B., Comiskey, J. A., Matthews, E. R., et al. (2016). National parks in the eastern United States harbor important older forest structure compared with matrix forests. *Ecosphere* 7:e01404. doi: 10.1002/ecs2.1404

Miller, K. M., Mcgill, B. J., Mitchell, B. R., Comiskey, J., Dieffenbach, F. W., Matthews, E. R., et al. (2018). Eastern national parks protect greater tree species diversity than unprotected matrix forests. *Forest Ecol. Manag.* 414, 74–84. doi: 10.1016/j.foreco.2018.02.018

National Academies of Sciences, Engineering, and Medicine (2019). *Negative Emissions Technologies and Reliable Sequestration: A Research Agenda.* Washington, DC: The National Academies Press.

National Conservation Easement Database (2014) Available online at: conservationeasement.us (accessed April 15, 2019).

National Council for Air and Stream Improvement (2019). *United States Department of Agriculture Forest Service. COLE: Carbon on Line Estimator.* Available online at: https://www.fs.usda.gov/ccrc/index.php?q=tools/cole (accessed April 15, 2019).

National Health Service Forest (2019). *National Health Service.* Available online at: http://nhsforest.org/ (accessed April 15, 2019).

National Interagency Fire Center (2019). *Total Wildland Fires and Acres (1926-2017).* Available online at: https://www.nifc.gov/fireInfo/fireInfo_stats_totalFires.html (accessed April 15, 2019).

Nature Conservancy (2019). *The. "Burnt Mountain Beauty: Explore Vermont's Newest Preserve and the State's Largest Carbon Project".* Available online at: https://www.nature.org/en-us/about-us/where-we-work/united-states/vermont/stories-in-vermont/burnt-mountain-beauty/ (accessed April 15, 2019).

Nunery, J. S., and Keeton, W. S. (2010). Forest carbon storage in the northeastern United States: net effects of harvesting frequency, post-harvest retention, and wood products. *Forest Ecol. Manag.* 259, 1363–1375. doi: 10.1016/j.foreco.2009.12.029

Oh, B., Lee, K. J., Zaslawski, C., Yeung, A., Rosenthal, D., Larkey, L., et al. (2017). Health and well-being benefits of spending time in forests: systematic review. *Environ. Health Prev. Med.* 22:71. doi: 10.1186/s12199-017-0677-9

Oliver, C. D., Nassar, N. T., Lippke, B. R., and McCarter, J. B. (2014). Carbon, fossil fuel, and biodiversity mitigation with wood and forests. *J. Sustain. Forest.* 33, 248–275. doi: 10.1080/10549811.2013.839386

Oswalt, S. N., Smith, W. B., Miles, P. D., and Pugh, S. A. (2014). *Forest Resources of the United States, 2012: A Technical Document Supporting the Forest Service*

2010 Update of the RPA Assessment. United States Department of Agriculture, Forest Service, Washington Office. (Washington, DC), Gen. Tech. Rep. WO-91. doi: 10.2737/WO-GTR-91

Pan, Y., Chen, J. M., Birdsey, R., McCullough, K., He, L., and Deng, F. (2011). Age structure and disturbance legacy of North American forests. *Biogeosciences* 8, 715–738. doi: 10.5194/bg-8-715-2011

Paris Climate Agreement (2015). Available online at: https://unfccc.int/sites/default/files/english_paris_agreement.pdf (accessed April 15, 2019).

Petranka, J. W., Eldridge, M. E., and Haley, K. E. (1993). Effects of timber harvesting on southern Appalachian salamanders. *Conserv. Biol.* 7, 363–370. doi: 10.1046/j.1523-1739.1993.07020363.x

Pugh, T. A. M., Lindeskog, M., Smith, B., Poulter, B., Arneth, A., Haverd, V., et al. (2019). Role of forest regrowth in global carbon sink dynamics. *Proc. Natl. Acad. Sci. U.S.A.* 116, 4382–4387. doi: 10.1073/pnas.1810512116

Ranius, T., Niklasson, M., and Berg, N. (2009). Development of tree hollows in pedunculate oak (*Quercus robur*). *Forest Ecol. Manag.* 257, 303–310. doi: 10.1016/j.foreco.2008.09.007

Reinhardt, E. D., Keane, R. E., Calkin, D. E., and Cohen, J. D. (2008). Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. *Forest Ecol. Manag.* 256, 1997–2006. doi: 10.1016/j.foreco.2008.09.016

Riitters, K., Potter, K., Iannone, B., Oswalt, C., Guo, Q., and Fei, S. (2018). Exposure of protected and unprotected forest to plant invasions in the eastern United States. *Forests* 9:723. doi: 10.3390/f9110723

Searchinger, T. D., Hamburg, S. P., Melillo, J., Chameides, W., Havlik, P., Kammen, D. M., et al. (2009). Climate change. Fixing a critical climate accounting error. *Science* 326, 527–528. doi: 10.1126/science.1178797

Stephenson, N. L., Das, A. J., Condit, R., Russo, S. E., Baker, P. J., Beckman, N. G., et al. (2014). Rate of tree carbon accumulation increases continuously with tree size. *Nature* 507, 90–93. doi: 10.1038/nature12914

Sterman, J. D., Siegel, L., and Rooney-Varga, J. N. (2018). Reply to comment on 'Does replacing coal with wood lower CO_2 emissions? Dynamic lifecycle analysis of wood bioenergy'. *Environ. Res. Lett.* 13:128003. doi: 10.1088/1748-9326/aaf354

Thompson, J. R., Foster, D. R., Scheller, R., and Kittredge, D. (2011). The influence of land use and climate change on forest biomass and composition in Massachusetts, USA. *Ecol. Appl.* 21, 2425–2444. doi: 10.1890/10-2383.1

Thompson, J. R., Spies, T. A., and Ganio, L. M. (2007). Reburn severity in managed and unmanaged vegetation in a large wildfire. *Proc. Natl. Acad. Sci. U.S.A.* 104, 10743–10748. doi: 10.1073/pnas.0700229104

Thoreau, H. D. (1862). *Walking. The Atlantic Monthly, A Magazine of Literature, Art, and Politics*. Available online at: https://www.theatlantic.com/magazine/archive/1862/06/walking/304674/.

United Nations Conference on Environment and Development (1992). *Non-Legally Binding Authoritative Statement of Principles for a Global Consensus on the Management, Conservation and Sustainable Development of all Types of Forests*.Available online at: http://www.un-documents.net/for-prin.htm (accessed April 15, 2019).

United Nations Forum on Forests (2008). *Non-Legally Binding Instrument on All Types of Forests*. Available online at: http://www.undocs.org/A/res/62/98 (accessed April 15, 2019).

United States Environmental Protection Agency (2019). *Sources of Greenhouse Gas Emissions* Available online at: https://www.epa.gov/ghgemissions/sources-greenhouse-gas-emissions (accessed April 15, 2019).

United States Forest Service (2019). *Forest Inventory and Analysis National Program, Forest Inventory EVALIDator web-application Version 1.8.0.00*. U.S. Department of Agriculture, Forest Service, Northern Research Station. Available online at: https://apps.fs.usda.gov/Evalidator/evalidator.jsp (accessed April 15, 2019).

United States Forest Service. Leadership Corner. (2018). Available online at: https://www.fs.fed.us/blogs/five-national-priorities-build-legacy-guide-agency-forward-0 (accessed April 15, 2019).

United States Geological Survey (2019a). *Gap Analysis Program. Protected Areas Database of the United States (PADUS), version 1.3 Combined Feature Class*. (accessed April 15, 2019).

United States Geological Survey (2019b). *National Land Cover Database Land Cover Collection* Available online at: https://catalog.data.gov/dataset/national-land-cover-database-nlcd-land-cover-collection (accessed April 15, 2019).

University of Montana (2019). *Wilderness Connect*. Available online at: https://wilderness.net/ (accessed April 15, 2019).

Vining, J., and Tyler, D. E. (1999). Values, emotions and desired outcomes reflected in public responses to forest management lans. *Hum. Ecol. Rev.* 6, 21–34.

Waller, D. M. (2014). "Effects of deer on forest herb layers," in *The Herbaceous Layer in Forests of Eastern North America, Ch. 16, 2nd Edn*, ed F. S. Gilliam (New York, NY: Oxford University Press), 369–399. doi: 10.1093/acprof:osobl/9780199837656.003.0016

Watson, J. E. M., Evans, T., Venter, O., Williams, B., Tulloch, A., Stewart, C., et al. (2018). The exceptional value of intact forest ecosystems. *Nat. Ecol. Evol.* 2, 599–610. doi: 10.1038/s41559-018-0490-x

Whitney, G. G. (1990). The history and status of the hemlock-hardwood forests of the Allegheny Plateau. *J. Ecol.* 1, 443–458. doi: 10.2307/2261123

Williams, M. (1989). *Americans and Their Forests: A Historical Geography*. New York, NY: Cambridge University Press.

Williams, M. (2000). Available online at: https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsm8_035779.pdf (accessed April 15, 2019).

Wilson, E. O. (2016). *Half-Earth: Our Planet's Fight for Life*. New York, NY: Liveright Publishing Corp.

Zhou, G., Liu, S., Li, Z., Zhang, D., Tang, X., Zhou, C., et al. (2006). Old-growth forests can accumulate carbon in soils. *Science*. 314:1417. doi: 10.1126/science.1130168

Zlonis, E. J., and Niemi, G. J. (2014). Avian communities of managed and wilderness hemiboreal forests. *Forest Ecol. Manag.* 328, 26–34. doi: 10.1016/j.foreco.2014.05.017

Conflict of Interest Statement: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

land

MDPI

Perspective

Creating Strategic Reserves to Protect Forest Carbon and Reduce Biodiversity Losses in the United States

Beverly E. Law [1,*], William R. Moomaw [2], Tara W. Hudiburg [3], William H. Schlesinger [4], John D. Sterman [5] and George M. Woodwell [6]

1 Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331, USA
2 The Fletcher School and Global Development and Environment Institute, Tufts University, Medford, MA 02155, USA; william.moomaw@tufts.edu
3 Department of Forest, Rangeland, and Fire Sciences, University of Idaho, Moscow, ID 83844, USA; thudiburg@uidaho.edu
4 Cary Institute of Ecosystems Studies, Millbrook, NY 12545, USA; schlesingerw@caryinstitute.org
5 MIT Sloan School of Management, Massachusetts Institute of Technology, Cambridge, MA 02139, USA; jsterman@mit.edu
6 Woodwell Climate Research Center, Falmouth, MA 02540, USA; gwoodwell@woodwellclimate.org
* Correspondence: bev.law@oregonstate.edu

Abstract: This paper provides a review and comparison of strategies to increase forest carbon, and reduce species losses for climate change mitigation and adaptation in the United States. It compares forest management strategies and actions that are taking place or being proposed to reduce wildfire risk and to increase carbon storage with recent research findings. International agreements state that safeguarding biodiversity and ecosystems is fundamental to climate resilience with respect to climate change impacts on them, and their roles in adaptation and mitigation. The recent Intergovernmental Panel on Climate Change report on impacts, mitigation, and adaptation found, and member countries agreed, that maintaining the resilience of biodiversity and ecosystem services at a global scale is "fundamental" for climate mitigation and adaptation, and requires "effective and equitable conservation of approximately 30 to 50% of Earth's land, freshwater and ocean areas, including current near-natural ecosystems." Our key message is that many of the current and proposed forest management actions in the United States are *not consistent* with climate goals, and that preserving 30 to 50% of lands for their carbon, biodiversity and water is feasible, effective, and necessary for achieving them.

Keywords: carbon dioxide; biodiversity; preservation targets; climate mitigation; climate adaptation; deforestation proforestation

Citation: Law, B.E.; Moomaw, W.R.; Hudiburg, T.W.; Schlesinger, W.H.; Sterman, J.D.; Woodwell, G.M. Creating Strategic Reserves to Protect Forest Carbon and Reduce Biodiversity Losses in the United States. *Land* **2022**, *11*, 721. https://doi.org/10.3390/land11050721

Academic Editor: Edward Morgan

Received: 29 March 2022
Accepted: 10 May 2022
Published: 11 May 2022

Publisher's Note: MDPI stays neutral with regard to jurisdictional claims in published maps and institutional affiliations.

1. Introduction

The climate is changing rapidly at an accelerating rate in every region of the planet. Immediate and sustained actions are needed to reduce dangerous and amplifying warming feedbacks. To avoid catastrophic, irreversible release of heat trapping methane and carbon dioxide, it is essential that natural land and ocean sinks remove and store substantially more atmospheric carbon dioxide to halt Arctic warming that is increasing over 3 times faster than the planetary average [1,2]. The next 10 to 30 years are a critical window for climate action, when severe ecological disruption is expected to accelerate [2–4]. Analysis of country-based pledges to reduce emissions in the nationally determined contributions (NDCs) suggests that emissions reductions should increase by 80% above the combined NDCs to keep temperature increases below the proposed 2 °C limit [5], and even greater reductions are required to remain below 1.5 °C. It is worth noting that these limits are warmer than the current temperature increase of 1.1 °C, meaning that the consequences for all climate-related changes will be more severe if those limits are reached or breached.

Forests play an important role in storing carbon, along with oceans, wetlands, and peatlands. Forests account for 92% of all terrestrial biomass globally, storing approximately 400 gigatons carbon [6]. Despite regional negative effects of climate change on the net amount of carbon removed from the atmosphere annually by land ecosystems, their removal of carbon dioxide from the atmosphere has remained fairly constant over the last 60 years at about 31% of emissions, with forests contributing the most [7]. Forests can play an important role in capturing and storing immense amounts of carbon. Reducing emissions from energy systems, deforestation, forest degradation, and other sources while increasing accumulation of carbon by natural systems are the primary means by which we will control atmospheric carbon dioxide (CO_2).

Here we present the status of science on forest management to mitigate climate change, and protect water and biodiversity in the United States, as well as the importance of Strategic Reserves to accomplish national and international goals of reducing biodiversity losses, and increasing the forest carbon reservoirs using natural climate solutions.

As discussed in more detail below, functionally separating carbon, water, and biodiversity and considering them independently leads to actions that inadvertently reduce the values of each, and can increase carbon emissions. This is why the 2021 report by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services and the Intergovernmental Panel on Climate Change (IPBES-IPCC) [8] stresses that climate change and biodiversity need to be examined together as parts of the same complex problem when developing climate mitigation and adaptation solutions [9,10].

The IPCC Assessment Report 6 confirms the findings of a growing body of research that maintaining ecosystem integrity and its biodiversity are essential to an effective response to a changing climate [1]. The Summary for Policy Makers, which is approved line by line by all IPCC member governments *including the United States*, summarizes current adaptation and mitigation climate science as follows:

"Summary for Policy Makers.D.4 Safeguarding biodiversity and ecosystems is fundamental to climate resilient development, in light of the threats climate change poses to them and their roles in adaptation and mitigation (very high confidence)."

"Summary for Policy Makers.D.4.1 Building the resilience of biodiversity and supporting ecosystem integrity can maintain benefits for people, including livelihoods, human health and well-being and the provision of food, fibre and water, as well as contributing to disaster risk reduction and climate change adaptation and mitigation." The formal definition of ecosystem integrity refers to the "ability of ecosystems to maintain key ecological processes, recover from disturbance, and adapt to new conditions."

Many current U.S. forest management practices that optimize resource extraction are inconsistent with this scientific consensus, are worsening both climate change and biodiversity loss, and decreasing multiple ecosystem services of U.S. forests. Strategies to mitigate and adapt to climate change have been proposed by scientists [8] and policymakers or those implemented by land managers and industries, and recent research has quantified their effectiveness and inadequacies. The strategies include:

- Avoiding deforestation and forest degradation—keeping forests intact;
- Reducing carbon loss by increasing harvest intervals and decreasing harvest intensity;
- Carbon storage in long-lived forest products (e.g., in combination with shorter harvest intervals);
- Burning trees for bioenergy;
- Thinning to reduce fire risk or severity and thus carbon losses.

We provide a synthesis of literature on evaluation of these strategies, as well as the importance of protecting the many values of forests, including carbon accumulation, biodiversity, and water availability. We focus on two regions of the U.S., the Pacific Coast, and southeast regions, which account for about 45% of the total U.S. forests' living biomass and removals by harvest [11].

2. Strategies

2.1. Avoid Deforestation and Forest Degradation, and Decrease Harvest-Related Carbon Losses

Primary forests are defined as forests composed of native species in which there are no clearly visible indications of human activities and ecological processes have not been significantly disturbed [12]. Multiple values are found at higher levels in intact forests of a given type, including habitat for endangered species, water security, and accumulated forest carbon stocks that keep carbon out of the atmosphere, and provide moderation of air and surface temperature through evapotranspiration [13,14]. Only 7% of the forest area in the U.S. is considered intact, with the exception of the nearly 68,000 km^2 Tongass National Forest in southeast Alaska, of which about 20,000 km^2 is defined as productive old-growth. Most of its 900 watersheds are near natural conditions, and its carbon-rich rainforests have similar carbon densities to the Pacific Northwest U.S. rainforests [15–17]. It is the largest intact temperate rainforest in the world, yet logging of old-growth continues while the USDA is in the process of restoring the roadless protections. The 2001 Roadless Rule prohibits road construction and timber harvesting on almost 30 million hectares of inventoried roadless areas (IRAs) on National Forest System lands, and is intended to provide protection for multiple uses.

Federal lands managed by the U.S. Forest Service (FS), the National Forest System (NFS), and the Bureau of Land Management (BLM) are managed under a multiple use—sustained yield model [18,19]. The statute directs the agencies to "balance multiple uses of their lands and ensure a sustained yield of those uses in perpetuity" [20]. The forest management plans describe where timber harvesting may occur as well as measures of sustainable harvest levels. The balance of these uses on federal lands has been an ongoing point of contention with the public [20].

Most timber harvesting occurs on private lands [11], however, there is increasing pressure to allow more timber cutting on federal lands. In the Pacific Northwest (PNW), removals declined on public lands after the peak in the late 1980s [11], partly due to implementation of the Northwest Forest Plan on public lands that aimed to protect endangered species in old-growth forests. The result was a strong increase in forest carbon accumulation on public lands over the next 17 years, while private lands remained near zero carbon accumulation, accounting for losses due to wildfire and harvesting [21].

Most forests in the U.S. have been harvested multiple times, and many managed forests are harvested well before reaching maturity. As of 2014, 51% of timber land in the south was less than 40 years old compared with 20% in the north and 22% in the west. In contrast, 56% of northern timber land was more than 60 years old, compared with 27% in the south and 69% in the west [11]. Since then, harvest ages have decreased in some cases because of changes in forest products (e.g., increasing production of cross-laminated timber, wood for bioenergy), thinning to reduce wildfire risk or severity, or removals after fire or beetle kill. Consequently, forest carbon densities are much lower than their potential, and could accumulate much more carbon and avoid carbon emissions associated with harvest [22].

Evaluation of strategies to mitigate climate change showed that forests can store more carbon if the harvest interval is lengthened on private lands and harvest is reduced on public lands in Oregon (Figure 1) [15]. A comparison of strategies showed that reducing harvest by half on public forests to allow them to continue to accumulate carbon (cumulative net ecosystem carbon balance, NECB) while increasing harvest rotation age from 40 years back to 80 years in forests with relatively low vulnerability to drought and fire under future climate conditions contribute the most to increasing forest carbon and reducing emissions. Far less effective are reforestation—just one-third as much carbon accumulation—and lastly, afforestation—just one-tenth as much carbon accumulation—that can compete with land usage for agriculture and urban development. This finding is supported by a recent National Academy report on "Negative Emissions" or atmospheric CO_2 removal options that finds the potential for afforestation and reforestation in limiting atmospheric CO_2 to be modest [23].

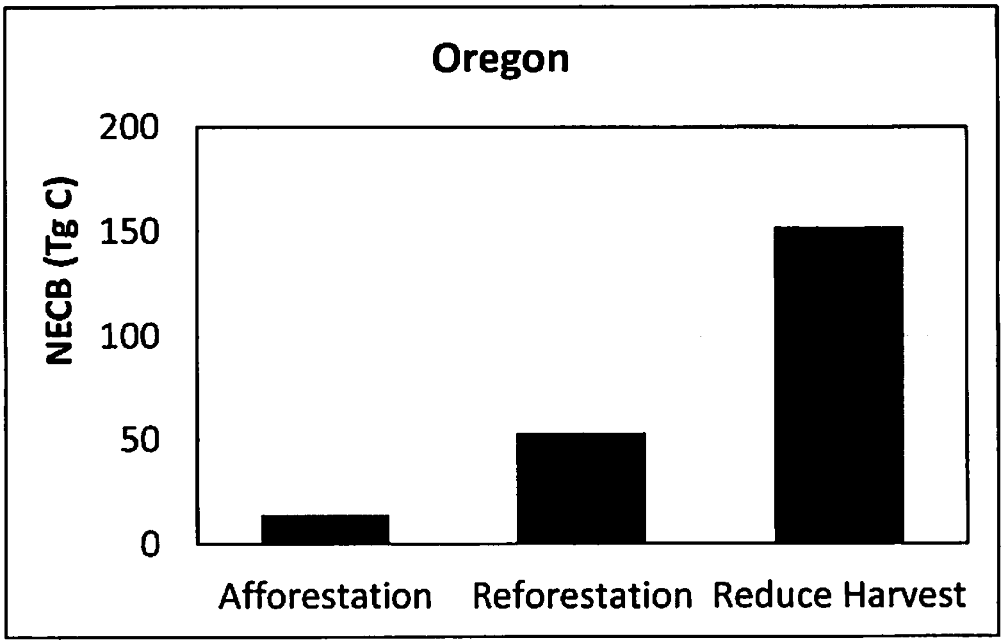

Figure 1. Land-use strategies to mitigate climate change across Oregon. Values on *y*-axis are cumulative change in net ecosystem carbon balance (NECB) from 2015 to 2100. Reduced harvest is a combination of restricted harvest by half on federal lands, and increased harvest intervals to 80 years on private lands. Data are from observation-based modeling [15].

A global study of 48 forests of all types found that among "mature multi-aged forests" half the living aboveground carbon was in the largest diameter 1% of the trees [24]. A study of six National Forests in Oregon found that trees of 53 cm DBH or greater comprised just 3% of the total stems, but held 43% of the aboveground carbon [25]. The U.S. Forest Service decided to drop a restriction on harvesting large trees in this category (Federal Register Document 2021-00804; https://www.govinfo.gov/content/pkg/FR-2021-01-15/pdf/2021-00804.pdf, accessed 20 April 2022), an action at odds with climate and biodiversity goals. Contrary to common belief, older forests continue to accumulate large quantities of carbon in trees and forest soils. Globally, forests older than 200 years continue to accumulate carbon at a rate of 1.6 to 3.2 Mg C ha^{-1} yr^{-1} [26].

Thus, temperate forests with high carbon and lower vulnerability to mortality have substantial additional capacity for climate mitigation. On a global level, it is estimated that forests could hold twice as much carbon as they currently do if managed differently [27]. While planting trees is desirable, that will contribute relatively little to carbon accumulation out of the atmosphere by 2100 compared to reducing harvest (See Figure 1). For example, if the Bonn Challenge of restoring 350 Mha by 2030 is given to natural forests, they would store an additional 42 Pg C by 2100, whereas giving the same area to plantations would store only 1 Pg C [15,28].

The potential for additional carbon accumulation is also being degraded by current management practices [29]. It was estimated that the "current gross carbon sink in forests recovering from harvests and abandoned agriculture to be −4.4 GtC/y, globally" [30]. This is more than the current difference between anthropogenic emissions and land and ocean annual accumulation out of the atmosphere (3.4 GtC/y) [7].

Mature and old forests generally store more carbon in trees and soil than young forests, and continue to accumulate it over decades to centuries [15,16,25] making them the most effective forest-related climate mitigation strategy. For example, restricting harvest by half on federal forests and changing the harvest cycle to 80 years across Oregon would increase forest carbon stocks 118 Tg C by 2100 [15,16,25]. Converting mature and older forests to younger forests results in a significant loss of total carbon stores, even when wood products are considered [31,32]. For example, a comparison of carbon stored in an unharvested versus harvested mature forest using the Forest-GHG life cycle assessment model to track harvested carbon from forest to landfill [31] shows that the unharvested forest has a much higher carbon density 120 years later, even when carbon in wood products is summed with the post-harvest carbon storage (Figure 2).

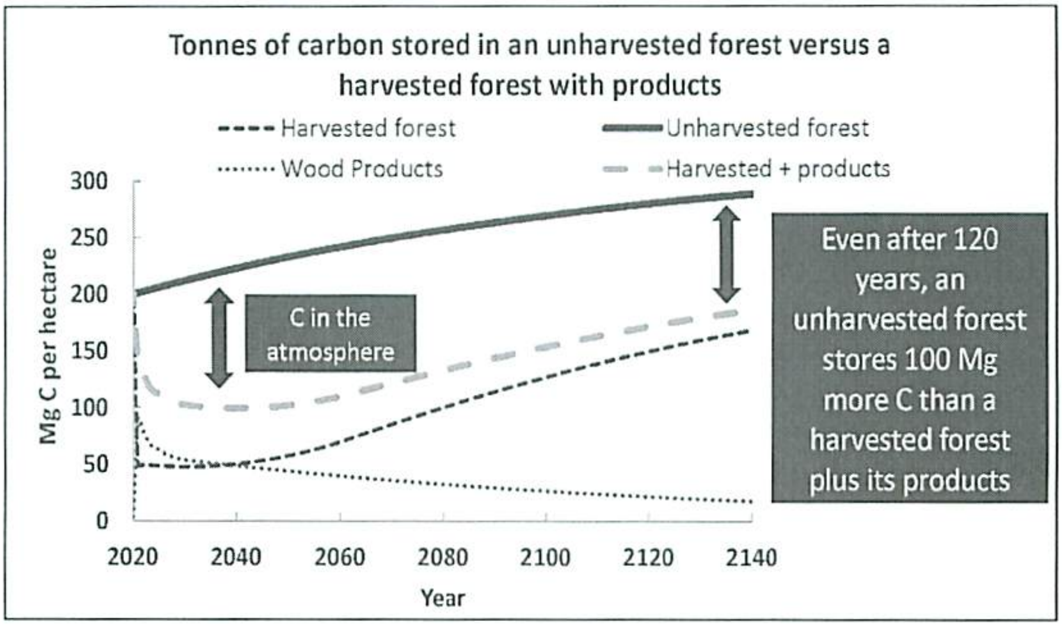

Figure 2. A mature forest with a carbon density of 200 tons of carbon per hectare (green line) is harvested (blue line) in 2020. This results in an immediate reduction of live tree carbon stocks. Approximately half of the aboveground carbon is removed and taken to the mills (as wood) while the other half remains behind in slash piles (leaves, bark, branches, etc.) and in the dead belowground roots. The slash is burned on-site and the carbon is immediately emitted to the atmosphere. The roots decompose over the next few decades, emitting carbon to the atmosphere. The carbon taken to the mill as wood is processed into short- and long-term wood products (red line), that decay over years to centuries, eventually returning the carbon to the atmosphere. Estimates comparing the carbon benefits of wood products to alternative materials have been found to overestimate the benefit by factors of between 2- and 100-fold by not counting the full life cycle carbon and the shorter durability of wood relative to alternative materials [33].

2.2. *Harvesting Forests for Bioenergy Production*

Utilizing wood biomass as a substitute for coal *increases* CO_2 emissions and *worsens* climate change for many decades or more [34]. Meeting U.S. national emissions reduction goals requires net emissions to drop by approximately 50% by 2030, reach net zero by 2050, and be net negative beyond 2100 [2,4].

Although wood and coal release comparable amounts of carbon dioxide per unit of primary energy [35], wood chips and pellets burn less efficiently. For example, a 500-megawatt power plant burning wood pellets emits an estimated 437,300 tons of CO_2-C annually, whereas the same plant burning coal would emit 392,000 tons/year [36]. The situation is worse if wood displaces other fossil fuels: wood releases about 25% more CO_2 per unit of primary energy than fuel oil, and about 75% more CO_2 than fossil (natural) gas [35]. Further, greenhouse gas emissions from the wood supply chain exceed those of the coal supply chain: Approximately 27% of harvested carbon equivalent is used to produce dry pellets [37], while coal processing adds just about 11% to emissions [38]. Therefore, the immediate impact of wood bioenergy is an increase in CO_2 emissions, creating a "carbon debt", even when wood displaces coal, the most carbon intensive fossil fuel. The harvested forests can regrow, repaying the debt, but regrowth is uncertain and takes time.

Regrowth takes time: The time between the combustion of wood and the potential, *eventual* removal of that excess CO_2 by regrowth is known as the carbon debt payback time [39]. For forests in the eastern U.S., which supply much of the wood for pellet production and national and international export, carbon debt payback times range from many decades to a century or more, depending on forest age at harvest, species, and climate zone [38,40].

Carbon debt payback times are longer in the young forests prevalent in the U.S. because harvesting wood from growing forests also prevents the CO_2 removal that would have occurred had trees not been harvested and burned [41]. If a 40-year-old forest was harvested and burned, releasing its carbon immediately to the atmosphere, under ideal conditions, it would take another 40 years to remove the added carbon from the atmosphere and restore the initial carbon stocks in the regrown forest, known as "slow in, fast out" [42–44]. However, if not harvested, the same forests would have continued to accumulate significantly more carbon, thereby further reducing the amount in the atmosphere. Shorter rotation times between harvests for bioenergy leave the greatest amount of CO_2 in the atmosphere [40].

Forests of the southeastern and southcentral U.S. are the largest source of wood for commercial scale bioenergy, mostly for use in Europe. If allowed to continue growing (proforestation), they could remove significant additional atmospheric CO_2 and accumulate the additional carbon in trees and soils [22].

Note that wood bioenergy harvest worsens climate change even if the harvested forests are managed sustainably, because the average total stock of carbon on the land is lower than prior to harvest, and the carbon lost from the land is added to the atmosphere, worsening climate change [38,40]. Moreover, reforestation following harvest of a diverse bottomland hardwood forest that provided habitat for multiple animal species would, in most cases, be converted to a pine monoculture plantation.

Eventual carbon neutrality does not mean *climate neutrality*. The excess CO_2 from wood bioenergy worsens global warming immediately upon entering the atmosphere. The harms caused by that additional warming are not undone even if regrowth eventually removes all the excess CO_2. Global average surface temperatures will not immediately return to previous levels and may persist for a millennium or more [45]. The Greenland and Antarctic ice sheets melt faster, sea level rises higher, accelerated permafrost thaw releases more methane, wildfires become more likely, storms intensify more, and extinction is greater than if the forest had not been harvested and the wood had not been burned [45]. Recent simultaneous temperature spikes of tens of degrees Celsius in the Arctic and Antarctica demonstrate that unprecedented warming signals are already occurring, resulting in some changes, such as sea-level rise, that are irreversible for centuries to millennia [1]. Even eventual full forest recovery and carbon removal will not replace lost ice, lower sea level, undo climate disasters, or bring back communities lost to floods or wildfires.

2.3. *Thinning to Reduce Fire Risk or Severity and Carbon Loss*

2.3.1. Broad-Scale Thinning to Reduce Fire Severity Conflicts with Climate Goals

A reaction to the recent increase in the intensity and frequency of wildfires is to thin forests to reduce the quantity of combustible materials. However, the amount of carbon removed by thinning is much larger than the amount that might be saved from being burned in a fire, and far more area is harvested than would actually burn [42,46–49]. Most analyses of mid- to long-term thinning impacts on forest structure and carbon storage show there is a multi-decadal biomass carbon deficit following moderate to heavy thinning [50]. For example, thinning in a young ponderosa pine plantation showed that removal of 40% of the tree biomass would release about 60% of the carbon over the next 30 years [51]. Regional patchworks of intensive forest management have increased fire severity in adjacent forests [49]. Management actions can create more surface fuels. Broad-scale thinning (e.g., ecoregions, regions) to reduce fire risk or severity [52] results in more carbon emissions than fire, and creates a long-term carbon deficit that undermines climate goals.

As to the effectiveness and likelihood that thinning might have an impact on fire behavior, the area thinned at broad scales to reduce fuels has been found to have little relationship to area burned, which is mostly driven by wind, drought, and warming. A multi-year study of forest treatments such as thinning and prescribed fire across the western U.S. showed that about 1% of U.S. Forest Service treatments experience wildfire each year [53]. The potential effectiveness of treatments lasts only 10–20 years, diminishing annually [53]. Thus, the preemptive actions to reduce fire risk or severity across regions have been largely ineffective.

Effective risk reduction solutions need to be tailored to the specific conditions. In fire-prone dry forests, careful removal of fuel ladders such as saplings and leaving the large fire-resistant trees in the forest may be sufficient and would have lower carbon consequences than broad-scale thinning [54]. The goals of restoring ecosystem processes and/or reducing risk in fire-prone regions can be met by removing small trees and underburning to reduce surface fuels, not by removal of larger trees, which is sometimes done to offset the cost of the thinning. With continued warming and the need to adapt to wildfire, thinning may restore more frequent low-severity fire in some dry forests, but could jeopardize regeneration and trigger a regime change to non-forest ecosystems [53].

While moderate to high severity fire can kill trees, most of the carbon remains in the forest as dead wood that will take decades to centuries to decompose. Less than 10% of ecosystem carbon enters the atmosphere as carbon dioxide in PNW forest fires [21,46]. Recent field studies of combustion rates in California's large megafires show that carbon emissions were very low at the landscape-level (0.6 to 1.8%) because larger trees with low combustion rates were the majority of biomass, and high severity fire patches were less than half of the burn area [55,56]. These findings are consistent with field studies on Oregon's East Cascades wildfires and the large Biscuit Fire in southern Oregon [57,58].

To summarize, harvest-related emissions from thinning are much higher than potential reduction in fire emissions. In west coast states, overall harvest-related emissions were about 5 times fire emissions, and California's fire emissions were a few percent of its fossil fuel emissions [59]. In the conterminous 48 states, harvest-related emissions are 7.5 times those from all natural causes [60]. It is understandable that the public wants action to reduce wildfire threats, but false solutions that make the problem worse and increase global warming are counterproductive.

2.3.2. Change Focus from Broadscale Thinning to the Home Ignition Zone

Over the past century, public agencies have been responsible for managing fire risk and protecting communities, however, their focus has been on suppression, fuel reduction, and prevention. Yet, of all the ignitions that crossed jurisdictional boundaries, more than 60% originated on private property and 28% in national forests [61]. These findings are in stark contrast to the common narrative that wildfires start on remote public land and then move into communities [62].

Hardening home structures in areas with high risk of wildfires such as the wildland-urban interface has been found to be the most effective means to reduce property damage from wildfires [63]. Many rural homes use propane tanks that explode from the intense heat. Safer energy options for homeowners would reduce the spread from house to house and the loss of the structures. Community safety experts and wildfire risk managers indicate that focus should be on addressing the home ignition zone by using fire-resistant designs, more intensive fuel reduction close to buildings, and preventing new developments in high fire-risk areas [64]. Incentives are misaligned because zoning and approval of building locations are functions of local governments, but responding to fires, and shouldering those costs, are the responsibility of state and federal agencies. Additionally, a large number of the most destructive fires have been ignited by poorly maintained powerlines [65]. Buried lines and better maintenance could reduce the frequency of wildfires.

2.3.3. Post-Fire Harvest versus Natural Regeneration

After fires, the remaining live and dead trees in the burn area and those on the periphery provide seed sources for natural regeneration [66]. Fires also provide ash which can act as a natural fertilizer, providing macro- and micronutrients for regrowth. Natural regeneration allows germination of genetic- and species-diverse seeds, and resprouting of shrubs that provide important habitat as forests recover. The diversity of early successional species also increases the resilience of the ecosystem to future disturbance, and accumulates additional carbon [67]. Natural and managed regeneration failures have occurred, particularly in dry regions [67–69], but here we are referring to the diversity of seed stock in natural regeneration compared to planting of less diverse seedling sources. Although there is enthusiasm about participating in reforestation, tree planting must be done carefully to ensure appropriate species selection for specific sites, whereas natural growth has more likelihood of re-establishing local biodiversity [67].

The complex early seral forest habitats that develop after high severity burns are important to a broad range of wildlife [70]. Post-fire harvest and felling of live and dead trees can harm soil integrity, hydrology, natural regeneration, slope stability, and wildlife habitat [71]. Large standing dead, live yet possibly dying, and downed trees help forests recover and provide habitat for more than 150 vertebrates in the PNW [72].

In burned watersheds, post-fire logging worsens conditions that have resulted from a century of human activity [73,74] and impedes the rate of recovery. In sum, post-fire treatments can cause a significant loss of ecosystem services [75].

3. Solutions

To mitigate climate change and avoid additional irreversible changes, we must reduce energy consumption through greater end-use efficiency gains and shift to carbon-free energy sources (e.g., solar and wind) [76], and simultaneously increase removal and accumulation of additional carbon from the atmosphere in forests, wetlands, and soils.

Global studies have identified areas for protection of intact forests that would stem biodiversity loss and prevent land conversion to other uses [77,78]. A recent study suggests assessment of ecosystem integrity represented by faunal intactness (no loss of species), habitat intactness, and functional intactness (no reduction in faunal densities below ecologically functional densities) [1]. However, global analyses can miss important local to regional ecological features that affect species and thus, the potential for protections. A global meta-analysis showed that most vulnerable bird species need large intact forests, although relatively small fragments can still have substantial biodiversity value if protected at the highest levels (IUCN categories I-VI) [79]. To address this issue, the International Union for Conservation of Nature (IUCN) developed a policy [80] for defining forests of conservation value:

"While primary forests of all extents have conservation value, areas of greater extent warrant particular attention where they persist, as they support more biodiversity, contain larger carbon stocks, provide more ecosystem services, encompass larger-scaled natural processes, and are more resilient to external stresses. The significance of large areas of primary forests has been highlighted by the global mapping of Intact Forest Landscapes (IFL) greater than 500 km^2 in extent. While suitable for many purposes, other thresholds may be more suitable at regional and national levels that reflect local ecological factors." (IUCN Policy Statement on Primary Forests, https://www.iucn.org/sites/dev/files/content/documents/iucn_pf-ifl_policy_2020_approved_version.pdf, accessed on 22 April 2020).

Much focus has been on protecting some notable primary forests [81] such as the Amazon, but that should not distract our attention from the need to retain significant intact forests within North America. There is more carbon stored in the world's temperate and boreal forests combined than in all remaining tropical forests [81]. There are ecosystems in many ecoregions that meet the conditions for protecting half of forestlands [82]. Bird populations are good indicators of ecosystem integrity. A net population decline of 2.9 billion birds in North America occurred between 1970 and 2017, of which forest- dependent species accounted for over one-third of the total, indicating a loss of insects and rapid recent degradation of forest ecosystem integrity [83,84].

Areas in the lower 48 states with high concentrations of imperiled forest- and non-forest species with small ranges in the west and east should be considered for protection (Figure 3) [85].

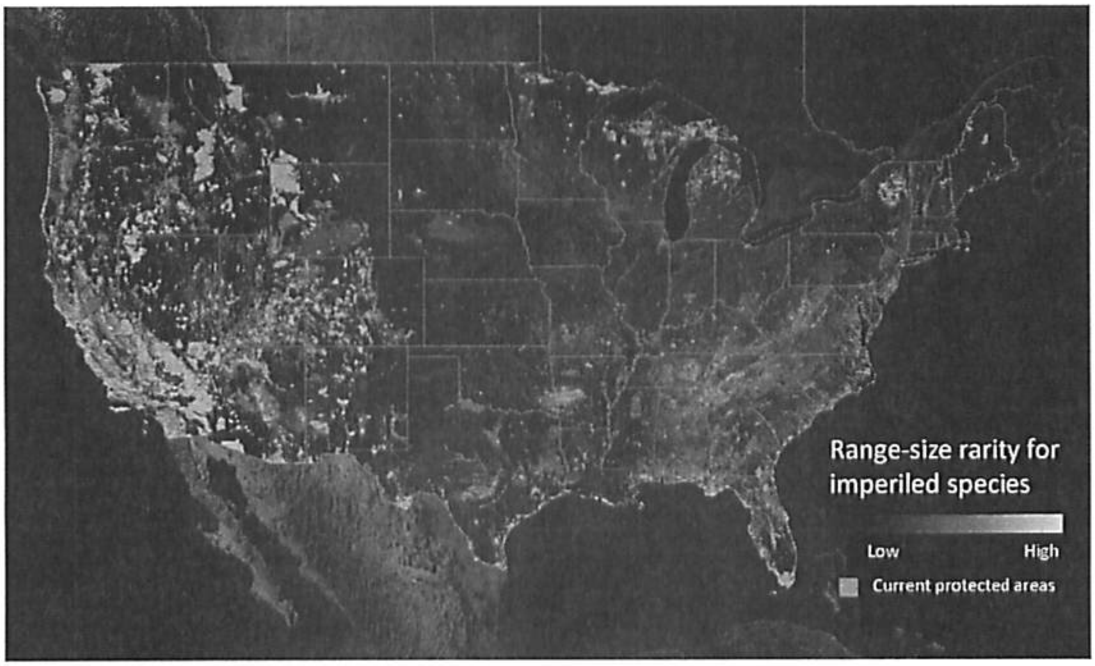

Figure 3. Summed range-size rarity of forest and non-forest species in the lower 48 states that are protected by the Endangered Species Act and/or considered to be in danger of extinction. Species include vertebrates (birds, mammals, amphibians, reptiles, freshwater fishes), freshwater invertebrates, pollinators, and vascular plants. High values (yellow) are areas where species with small ranges (and thus fewer places where they can be conserved) are likely to occur; the presence of multiple imperiled species contributes to higher scores. (Image produced by NatureServe; https://livingatlas.arcgis.com, accessed 21 April 2022).

Instead of regularly harvesting on all of the 70% of U.S. forest land designated as "timberlands" by the U.S. Forest Service, setting aside sufficient areas as Strategic Reserves would significantly increase the amount of carbon accumulated between now, 2050 and 2100, and reestablish greater ecosystem integrity, helping to slow climate change and restore biodiversity. The 2022 IPCC AR6 report stated that "Recent analyses, drawing on a range of lines of evidence, suggest that maintaining the resilience of biodiversity and ecosystem services at a global scale depends on effective and equitable conservation of approximately 30% to 50% of Earth's land, freshwater and ocean areas, including currently near-natural ecosystems (high confidence)." Continuing commercial timber harvest on a portion of the remaining public lands and tens of millions of hectares of private lands would continue to adequately supply a sustainable forestry sector.

Preserving and protecting mature and old forests would not only increase carbon stocks and growing carbon accumulation, they would slow and potentially reverse accelerating species loss and ecosystem deterioration, and provide greater resilience to increasingly severe weather events such as intense precipitation and flooding.

Domestic livestock grazing occurs on 85% of public lands in the western U.S. and is a significant source of greenhouse gas emissions (12.4 Tg CO_2 equivalents per year). Due to overgrazing, it was estimated to decrease aboveground biomass carbon by about 85% when converted from forests and woodlands to grass-dominated ecosystems [86]. Discontinuing or greatly reducing this practice would be an important climate mitigation strategy.

High carbon forests in the western U.S. are highly biodiverse ecosystems that store and provide water to millions of people and to major agricultural regions, and are more resilient to climate change [9]. The PNW and Alaska stand out as having the largest mature and old forests with immense carbon stores and high biodiversity that meet the IPCC criteria of meriting protection to remove significant additional carbon from the atmosphere. A majority of these areas are on public lands with the potential for permanent protection consistent with the highest international standards, and could be complemented with additional protections on private and indigenous lands [87]. These forests are critical for greater future carbon accumulation, and are an essential source of clean drinking water [9]. Forests dominate the drinking water supply in the U.S. that must be protected at the source [88,89]. For example, forests account for almost 60% of the most important areas for surface drinking water in the western U.S., yet only about 19% are protected at the highest levels. Other regions of the U.S. such as the southeast host some of the greatest biodiversity on the continent, and require protection for their forest carbon, biodiversity, and water.

Across the eleven western U.S. states, a framework was applied to prioritize protection of high carbon and biodiversity forest areas to meet the 30 × 30 and 50 × 50 preservation targets (Figure 4). Out of 92.5 Mha of forestland in the region, 14% is currently protected at the level equivalent to wilderness areas, IUCN classification Ia to II, and 5% is protected at IUCN classifications III to VI, which allows practices that degrade existing natural communities, such as road building and suppression of natural disturbances [90]. To achieve 30% protection of forest area by 2030, an additional 10 Mha would need to be protected at these levels. To meet the 50% target by 2050, an increase of 29 Mha is required. The analysis examined, removing from consideration, areas that are at high risk of mortality from wildfire or drought under future climate conditions (Figure 5) [91] to determine if there was sufficient qualifying area to protect. The prioritization used an ecoregion approach [82] to determine relative importance for protection of biodiversity and/or carbon within each ecoregion. Ecoregions are delineated based on similarity of a range of abiotic and biotic characteristics (topography, climate, soils, vegetation), e.g., EPA Level III [92]. Ecoregion-based conservation was evaluated in a range of habitats, and is recognized as a strong basis for the need to conserve about half of each region [82]. A similar framework could be applied in other regions, with additional data such as species endemism, if available.

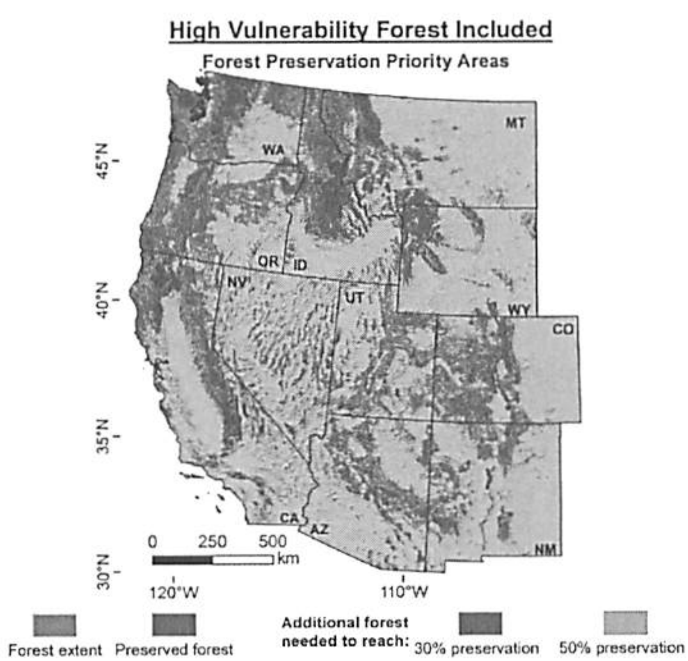

Figure 4. Forestlands that are currently preserved, and additional areas identified as high priority for protection of biodiversity and forest carbon for climate mitigation across the western U.S. Adapted from [5].

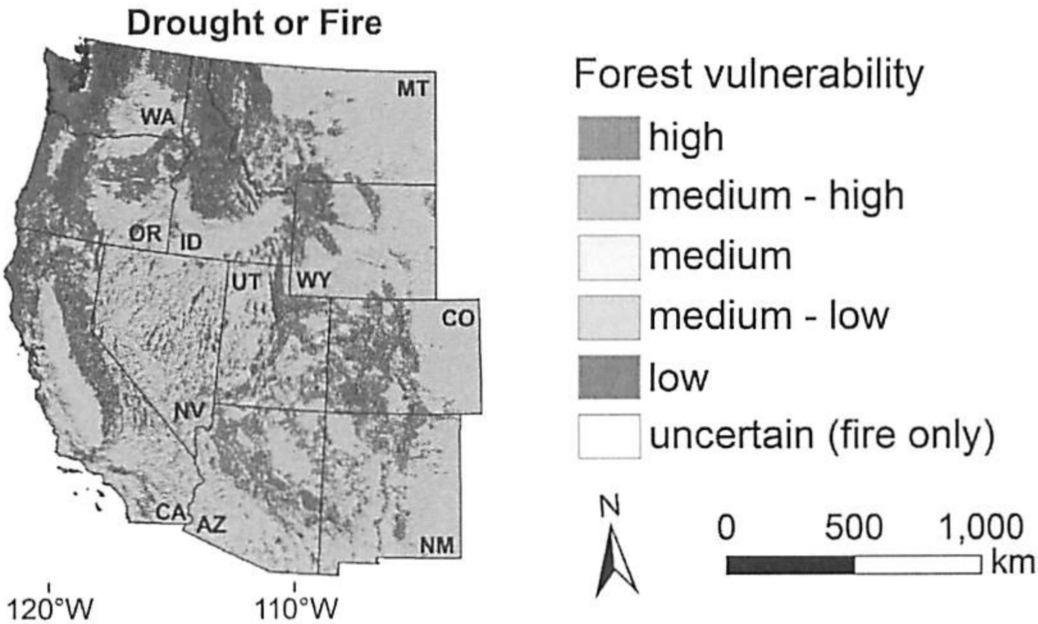

Figure 5. Vulnerability of forestlands to either drought or fire under future climate scenarios to year 2050. Adapted from [83].

The strategic reserves defined within each ecoregion would protect carbon, water, and biodiversity, and recognize the value of forested landscapes that are diverse in structure and function. Across the climate gradient from mesic to drier ecoregions, portions can be impacted by wildfire, but they are still important to protect their biodiversity, allowing species to persist (e.g., in refugia), migrate, and reorganize with a changing climate. An example is the Klamath Mountains ecoregion in Oregon and California, which has high biodiversity partly because of its unique geology. It is one of the top four temperate coniferous forests in species richness globally. Its vulnerability to forest fires should not disqualify it from protecting the rich diversity of plant and animal species from human degradation [70].

4. Conclusions

Maintaining forest ecosystem integrity is *"fundamental" to resilient development and climate mitigation and adaptation.* Current extractive management practices on all forests designated as "timberlands" are inconsistent with slowing, and eventually achieve lower "atmospheric concentrations of greenhouse gases that will avoid dangerous anthropogenic interference with the climate system" [93]. Many of the existing forest management practices allegedly protect forests and homes from wildfire and are having severe adverse effects on forest ecosystem integrity and resilience, and are worsening climate change and diminishing biodiversity. Forest bioenergy adds significantly more CO_2 to the atmosphere than fossil fuels. Its use is based upon a mistaken assumption that it is necessary to shift to renewable energy than to reduce heat-trapping gas emissions such as carbon dioxide, rather than to reduce emissions from all sources including forest bioenergy for electricity.

Climate change mitigation and biodiversity protection is an essential component of forest management decision-making. To avoid dangerous anthropogenic interference with the climate system, provide water security, and stem biodiversity losses, permanent Strategic Climate and Biodiversity Reserves need to be established quickly, and their integrity monitored and maintained.

Author Contributions: Investigation, B.E.L., W.R.M., T.W.H., W.H.S., and J.D.S.; writing—original draft preparation, B.E.L., W.R.M., T.W.H., W.H.S., and J.D.S.; writing—review and editing, B.E.L., W.R.M., T.W.H., W.H.S., J.D.S., and G.M.W. All authors have read and agreed to the published version of the manuscript.

Funding: T.H. was funded by NSF DEB-1553049; B.L. was funded by OSU Agricultural Research Foundation; W.M. was funded by Rockefeller Brothers Fund.

Data Availability Statement: Not applicable.

Conflicts of Interest: The authors declare no conflict of interest.

References

1. IPCC. Summary for Policymakers. In *Climate Change 2022: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*; Pörtner, H.-O., Roberts, D.C., Poloczanska, E.S., Mintenbeck, K., Tignor, M., Alegría, A., Craig, M., Langsdorf, S., Löschke, S., Möller, V., et al., Eds.; Cambridge University Press: Cambridge, UK, 2022.
2. IPCC. Summar for Policymakers. In *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*; Cambridge University Press: Cambridge, UK, 2021.
3. Trisos, C.H.; Merow, C.; Pigot, A.L. The projected timing of abrupt ecological disruption from climate change. *Nature* **2020**, *580*, 496–501. [CrossRef] [PubMed]
4. IPCC. Summary for Policymakers. In *Global Warming of 1.5 °C*; World Meteorological Organization: Geneva, Switzerland, 2018.
5. Liu, P.R.; Raftery, A.E. Country-based rate of emissions reductions should increase by 80% beyond nationally determined contributions to meet the 2 C target. *Commun. Earth Environ.* **2021**, *2*, 1–10. [CrossRef] [PubMed]
6. Pan, Y.; Birdsey, R.A.; Phillips, O.L.; Jackson, R.B. The structure, distribution, and biomass of the world's forests. *Annu. Rev. Ecol. Evol. Syst.* **2013**, *44*, 593–622. [CrossRef]
7. Friedlingstein, P.; Jones, M.W.; O'Sullivan, M.; Andrew, R.M.; Bakker, D.C.E.; Hauck, J.; Le Quéré, C.; Peters, G.P.; Peters, W.; Pongratz, J.; et al. Global Carbon Budget 2021. *Earth Syst. Sci. Data Discuss.* **2021**, *2021*, 1917–2005. [CrossRef]
8. Pandit, R.; Pörtner, H.-O.; Scholes, R.J.; Agard, J.; Archer, E.; Arneth, A.; Bai, X.; Barnes, D.; Burrows, M.; Chan, L. Scientific Outcome of the IPBES-IPCC Co-Sponsored Workshop on Biodiversity and Climate Change. 2021. Available online: https://zenodo.org/record/5101133#.YnqZFYfMLb0 (accessed on 20 April 2022).
9. Law, B.E.; Berner, L.T.; Buotte, P.C.; Mildrexler, D.J.; Ripple, W.J. Strategic Forest Reserves can protect biodiversity in the western United States and mitigate climate change. *Commun. Earth Environ.* **2021**, *2*, 254. [CrossRef]
10. Buotte, P.C.; Law, B.E.; Ripple, W.J.; Berner, L.T. Carbon sequestration and biodiversity co-benefits of preserving forests in the western United States. *Ecol. Appl.* **2020**, *30*, e02039. [CrossRef]
11. Oswalt, S.N.; Smith, W.B.; Miles, P.D.; Pugh, S.A. *Forest Resources of the United States, 2017: A Technical Document Supporting the Forest Service 2020 RPA Assessment*; Gen. Tech. Rep. WO-97; US Department of Agriculture, Forest Service, Washington Office: Washington, DC, USA, 2019; Volume 97.
12. FAO. *Global Forest Resources Assessment 2020–Key Findings*; FAO: Rome, Italy, 2020. [CrossRef]
13. Novick, K.A.; Katul, G.G. The Duality of Reforestation Impacts on Surface and Air Temperature. *J. Geophys. Res. Biogeosci.* **2020**, *125*, e2019JG005543. [CrossRef]
14. Lawrence, D.; Coe, M.; Walker, W.S.; Verchot, L.; Vandecar, K.L. The unseen effects of deforestation: Biophysical effects on climate. *Front. For. Glob. Change* **2022**, *5*, 756115. [CrossRef]
15. Law, B.E.; Hudiburg, T.W.; Berner, L.T.; Kent, J.J.; Buotte, P.C.; Harmon, M.E. Land use strategies to mitigate climate change in carbon dense temperate forests. *Proc. Natl. Acad. Sci. USA* **2018**, *115*, 3663–3668. [CrossRef]
16. Hudiburg, T.; Law, B.; Turner, D.P.; Campbell, J.; Donato, D.; Duane, M. Carbon dynamics of Oregon and Northern California forests and potential land-based carbon storage. *Ecol. Appl.* **2009**, *19*, 163–180. [CrossRef]
17. Vynne, C.; Dovichin, E.; Fresco, N.; Dawson, N.; Joshi, A.; Law, B.E.; Lertzman, K.; Rupp, S.; Schmiegelow, F.; Trammell, E.J. The importance of Alaska for climate stabilization, resilience, and biodiversity conservation. *Front. For. Glob. Change* **2021**, *121*. [CrossRef]
18. US Congress. Multiple Use-Sustained Yield Act. 1. PL 86-517; 74 Stat 1960, 215. Available online: https://www.fs.fed.us/emc/nfma/includes/musya60.pdf (accessed on 20 April 2022).
19. 94th US Congress. Federal Land Management and Policy ACT OF 1976. PL 94–579. Available online: https://www.govinfo.gov/content/pkg/STATUTE-90/pdf/STATUTE-90-Pg2743.pdf#page=1 (accessed on 20 April 2022).
20. Riddle, A.A. *Timber Harvesting on Federal Lands*; Congressional Research Service, R45688. Available online: https://crsreports.congress.gov/product/pdf/R/R45688 (accessed on 20 April 2022).
21. Law, B.E.; Waring, R.H. Carbon implications of current and future effects of drought, fire and management on Pacific Northwest forests. *For. Ecol. Manag.* **2015**, *355*, 4–14. [CrossRef]
22. Moomaw, W.R.; Masino, S.A.; Faison, E.K. Intact Forests in the United States: Proforestation Mitigates Climate Change and Serves the Greatest Good. *Front. For. Glob. Change* **2019**, *2*, 27. [CrossRef]
23. National Academies of Sciences, Engineering, and Medicine. Negative Emissions Technologies and Reliable Sequestration: A Research Agenda. 2018. Available online: https://nap.nationalacademies.org/read/25259/chapter/1 (accessed on 15 April 2022).
24. Lutz, J.A.; Furniss, T.J.; Johnson, D.J.; Davies, S.J.; Allen, D.; Alonso, A.; Anderson-Teixeira, K.J.; Andrade, A.; Baltzer, J.; Becker, K.M.L.; et al. Global importance of large-diameter trees. *Glob. Ecol. Biogeogr.* **2018**, *27*, 849–864. [CrossRef]

25. Mildrexler, D.J.; Berner, L.T.; Law, B.E.; Birdsey, R.A.; Moomaw, W.R. Large Trees Dominate Carbon Storage in Forests East of the Cascade Crest in the United States Pacific Northwest. *Front. For. Glob. Change* **2020**, *3*, 17. [CrossRef]
26. Luyssaert, S.; Schulze, E.D.; Borner, A.; Knohl, A.; Hessenmoller, D.; Law, B.E.; Ciais, P.; Grace, J. Old-growth forests as global carbon sinks. *Nature* **2008**, *455*, 213–215. [CrossRef]
27. Erb, K.-H.; Kastner, T.; Plutzar, C.; Bais, A.L.S.; Carvalhais, N.; Fetzel, T.; Gingrich, S.; Haberl, H.; Lauk, C.; Niedertscheider, M. Unexpectedly large impact of forest management and grazing on global vegetation biomass. *Nature* **2018**, *553*, 73–76. [CrossRef]
28. Lewis, S.L.; Wheeler, C.E.; Mitchard, E.T.; Koch, A. Regenerate natural forests to store carbon. *Nature* **2019**, *568*, 25–28. [CrossRef]
29. Watson, J.E.; Evans, T.; Venter, O.; Williams, B.; Tulloch, A.; Stewart, C.; Thompson, I.; Ray, J.C.; Murray, K.; Salazar, A. The exceptional value of intact forest ecosystems. *Nat. Ecol. Evol.* **2018**, *2*, 599–610. [CrossRef]
30. Houghton, R.A.; Nassikas, A.A. Negative emissions from stopping deforestation and forest degradation, globally. *Glob. Change Biol.* **2018**, *24*, 350–359. [CrossRef]
31. Hudiburg, T.W.; Law, B.E.; Moomaw, W.R.; Harmon, M.E.; Stenzel, J.E. Meeting GHG reduction targets requires accounting for all forest sector emissions. *Environ. Res. Lett.* **2019**, *14*, 095005. [CrossRef]
32. Harmon, M.E.; Marks, B. Effects of silvicultural practices on carbon stores in Douglas-fir western hemlock forests in the Pacific Northwest, USA: Results from a simulation model. *Can. J. For. Res.* **2002**, *32*, 863–877. [CrossRef]
33. Harmon, M.E. Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions. *Environ. Res. Lett.* **2019**, *14*, 065008.
34. Searchinger, T.D.; Beringer, T.; Holtsmark, B.; Kammen, D.M.; Lambin, E.F.; Lucht, W.; Raven, P.; van Ypersele, J.-P. Europe's renewable energy directive poised to harm global forests. *Nat. Commun.* **2018**, *9*, 3741. [CrossRef] [PubMed]
35. EPA. Emissions Factors for Greenhouse Gas Inventories. Available online: https://www.epa.gov/sites/default/files/2018-03/documents/emission-factors_mar_2018_0.pdf (accessed on 23 February 2022).
36. EPA. *Compilation of Air Pollutant Emission Factors, AP-42*; US Environemental Protection Agency: Washington, DC, USA, 1997.
37. Röder, M.; Whittaker, C.; Thornley, P. How certain are greenhouse gas reductions from bioenergy? Life cycle assessment and uncertainty analysis of wood pellet-to-electricity supply chains from forest residues. *Biomass Bioenerg.* **2015**, *79*, 50–63. [CrossRef]
38. Sterman, J.D.; Lori, S.; Juliette, N.R.-V. Does replacing coal with wood lower CO_2 emissions? Dynamic lifecycle analysis of wood bioenergy. *Environ. Res. Lett.* **2018**, *13*, 015007.
39. Mitchell, S.R.; Harmon, M.E.; O'Connell, K.E.B. Carbon debt and carbon sequestration parity in forest bioenergy production. *GCB Bioenergy* **2012**, *4*, 818–827. [CrossRef]
40. Sterman, J.D.; Siegel, L.; Rooney-Varga, J.N. Reply to comment on 'Does replacing coal with wood lower CO_2 emissions? Dynamic lifecycle analysis of wood bioenergy'. *Environ. Res. Lett.* **2018**, *13*, 128003. [CrossRef]
41. Obermeier, W.A.; Nabel, J.E.; Loughran, T.; Hartung, K.; Bastos, A.; Havermann, F.; Anthoni, P.; Arneth, A.; Goll, D.S.; Lienert, S. Modelled land use and land cover change emissions—A spatio-temporal comparison of different approaches. *Earth Syst. Dyn.* **2021**, *12*, 635–670. [CrossRef]
42. Hudiburg, T.W.; Law, B.E.; Wirth, C.; Luyssaert, S. Regional carbon dioxide implications of forest bioenergy production. *Nat. Clim. Change* **2011**, *1*, 419–423. [CrossRef]
43. Schlesinger, W.H. Are wood pellets a green fuel? *Science* **2018**, *359*, 1328–1329. [CrossRef] [PubMed]
44. Körner, C. Slow in, Rapid out–Carbon Flux Studies and Kyoto Targets. *Science* **2003**, *300*, 1242–1243. [CrossRef] [PubMed]
45. Solomon, S.; Plattner, G.-K.; Knutti, R.; Friedlingstein, P. Irreversible climate change due to carbon dioxide emissions. *Proc. Natl. Acad. Sci. USA* **2009**, *106*, 1704–1709. [CrossRef] [PubMed]
46. Campbell, J.L.; Harmon, M.E.; Mitchell, S.R. Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions? *Front. Ecol. Environ.* **2012**, *10*, 83–90. [CrossRef]
47. Mitchell, S.R.; Harmon, M.E.; O'Connel, K.E.B. Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. *Ecol. Appl.* **2009**, *19*, 643–655. [CrossRef]
48. Rhodes, J.J.; Baker, W.L. Fire probability, fuel treatment effectiveness and ecological tradeoffs in western US public forests. *Open For. Sci. J.* **2008**, *1*, 1–7.
49. Hudiburg, T.W.; Luyssaert, S.; Thornton, P.E.; Law, B.E. Interactive Effects of Environmental Change and Management Strategies on Regional Forest Carbon Emissions. *Environ. Sci. Technol.* **2013**, *47*, 13132–13140. [CrossRef]
50. Zhou, D.; Zhao, S.; Liu, S.; Oeding, J. A meta-analysis on the impacts of partial cutting on forest structure and carbon storage. *Biogeosciences* **2013**, *10*, 3691–3703. [CrossRef]
51. Stenzel, J.; Berardi, D.; Walsh, E.; Hudiburg, T. Restoration Thinning in a Drought-Prone Idaho Forest Creates a Persistent Carbon Deficit. *J. Geophys. Res. Biogeosciences* **2021**, *126*, e2020JG005815. [CrossRef]
52. Zald, H.S.; Dunn, C.J. Severe fire weather and intensive forest management increase fire severity in a multi-ownership landscape. *Ecol. Appl.* **2018**, *28*, 1068–1080. [CrossRef]
53. Schoennagel, T.; Balch, J.K.; Brenkert-Smith, H.; Dennison, P.E.; Harvey, B.J.; Krawchuk, M.A.; Mietkiewicz, N.; Morgan, P.; Moritz, M.A.; Rasker, R.; et al. Adapt to more wildfire in western North American forests as climate changes. *Proc. Natl. Acad. Sci. USA* **2017**, *114*, 4582–4590. [CrossRef] [PubMed]
54. Hurteau, M.D.; North, M.P.; Koch, G.W.; Hungate, B.A. Opinion: Managing for disturbance stabilizes forest carbon. *Proc. Natl. Acad. Sci. USA* **2019**, *116*, 10193–10195. [CrossRef] [PubMed]

55. Stenzel, J.E.; Bartowitz, K.J.; Hartman, M.D.; Lutz, J.A.; Kolden, C.A.; Smith, A.M.S.; Law, B.E.; Swanson, M.E.; Larson, A.J.; Parton, W.J.; et al. Fixing a snag in carbon emissions estimates from wildfires. *Glob. Change Biol.* **2019**, *25*, 3985–3994. [CrossRef] [PubMed]
56. Harmon, M.E.; Hanson, C.T.; DellaSala, D.A. Combustion of Aboveground Wood from Live Trees in Megafires, CA, USA. *Forests* **2022**, *13*, 391. [CrossRef]
57. Meigs, G.; Donato, D.; Campbell, J.; Martin, J.; Law, B. Forest fire impacts on carbon uptake, storage, and emission: The role of burn severity in the Eastern Cascades, Oregon. *Ecosystems* **2009**, *12*, 1246–1267. [CrossRef]
58. Campbell, J.; Donato, D.; Azuma, D.; Law, B. Pyrogenic carbon emission from a large wildfire in Oregon, United States. *J. Geophys. Res. Biogeosciences* **2007**, *112*, G04014. [CrossRef]
59. Bartowitz, K.J.; Walsh, E.S.; Stenzel, J.E.; Kolden, C.A.; Hudiburg, T.W. Forest carbon emission sources arenot equal: Putting fire, harvest, and fossil fuel emissions in context. *Front. For. Glob. Change* **2022**, *5*, 867112. [CrossRef]
60. Harris, N.L.; Hagen, S.C.; Saatchi, S.S.; Pearson, T.R.H.; Woodall, C.W.; Domke, G.M.; Braswell, B.H.; Walters, B.F.; Brown, S.; Salas, W.; et al. Attribution of net carbon change by disturbance type across forest lands of the conterminous United States. *Carbon Balance Manag.* **2016**, *11*, 24. [CrossRef]
61. Downing, W.M.; Dunn, C.J.; Thompson, M.P.; Caggiano, M.D.; Short, K.C. Human ignitions on private lands drive USFS cross-boundary wildfire transmission and community impacts in the western US. *Sci. Rep.* **2022**, *12*, 1–14. [CrossRef]
62. Ager, A.A.; Palaiologou, P.; Evers, C.R.; Day, M.A.; Ringo, C.; Short, K. Wildfire exposure to the wildland urban interface in the western US. *Appl. Geogr.* **2019**, *111*, 102059. [CrossRef]
63. Smith, A.M.; Kolden, C.A.; Paveglio, T.B.; Cochrane, M.A.; Bowman, D.M.; Moritz, M.A.; Kliskey, A.D.; Alessa, L.; Hudak, A.T.; Hoffman, C.M. The science of firescapes: Achieving fire-resilient communities. *Bioscience* **2016**, *66*, 130–146. [CrossRef] [PubMed]
64. Syphard, A.D.; Rustigian-Romsos, H.; Mann, M.; Conlisk, E.; Moritz, M.A.; Ackerly, D. The relative influence of climate and housing development on current and projected future fire patterns and structure loss across three California landscapes. *Glob. Environ. Change* **2019**, *56*, 41–55. [CrossRef]
65. Keeley, J.E.; Syphard, A.D. Twenty-first century California, USA, wildfires: Fuel-dominated vs. wind-dominated fires. *Fire Ecol.* **2019**, *15*, 24. [CrossRef]
66. Donato, D.C.; Fontaine, J.B.; Campbell, J.L.; Robinson, W.D.; Kauffman, J.B.; Law, B.E. Conifer regeneration in stand-replacement portions of a large mixed-severity wildfire in the Klamath–Siskiyou Mountains. *Can. J. For. Res.* **2009**, *39*, 823–838. [CrossRef]
67. Cook-Patton, S.C.; Leavitt, S.M.; Gibbs, D.; Harris, N.L.; Lister, K.; Anderson-Teixeira, K.J.; Briggs, R.D.; Chazdon, R.L.; Crowther, T.W.; Ellis, P.W.; et al. Mapping carbon accumulation potential from global natural forest regrowth. *Nature* **2020**, *585*, 545–550. [CrossRef]
68. Stevens-Rumann, C.S.; Kemp, K.B.; Higuera, P.E.; Harvey, B.J.; Rother, M.T.; Donato, D.C.; Morgan, P.; Veblen, T.T. Evidence for declining forest resilience to wildfires under climate change. *Ecol. Lett.* **2018**, *21*, 243–252. [CrossRef]
69. Davis, K.T.; Dobrowski, S.Z.; Higuera, P.E.; Holden, Z.A.; Veblen, T.T.; Rother, M.T.; Parks, S.A.; Sala, A.; Maneta, M.P. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. *Proc. Natl. Acad. Sci. USA* **2019**, *116*, 6193–6198. [CrossRef]
70. Fontaine, J.B.; Donato, D.C.; Robinson, W.D.; Law, B.E.; Kauffman, J.B. Bird communities following high-severity fire: Response to single and repeat fires in a mixed-evergreen forest, Oregon, USA. *For. Ecol. Manag.* **2009**, *257*, 1496–1504. [CrossRef]
71. Beschta, R.L.; Frissell, C.A.; Gresswell, R.; Hauer, R.; Karr, J.R.; Minshall, G.W.; Perry, D.A.; Rhodes, J.J. *Wildfire and Salvage Logging: Recommendations for Ecologically Sound Post-Fire Salvage Management And Other Post-Fire Treatments on Federal Lands in the West*; Oregon State University: Corvallis, OR, USA, 1995.
72. Rose, C.L.; Marcot, B.G.; Mellen, T.K.; Ohmann, J.L.; Waddell, K.L.; Lindley, D.L.; Schreiber, B. Decaying wood in Pacific Northwest forests: Concepts and tools for habitat management. In *Wildlife-Habitat Relationships in Oregon and Washington*; Oregon State University Press: Corvallis, OR, USA, 2001; pp. 580–623.
73. Thorn, S.; Bässler, C.; Brandl, R.; Burton, P.J.; Cahall, R.; Campbell, J.L.; Castro, J.; Choi, C.-Y.; Cobb, T.; Donato, D.C.; et al. Impacts of salvage logging on biodiversity: A meta-analysis. *J. Appl. Ecol.* **2018**, *55*, 279–289. [CrossRef]
74. Karr, J.R.; Rhodes, J.J.; Minshall, G.W.; Hauer, F.R.; Beschta, R.L.; Frissell, C.A.; Perry, D.A. The Effects of Postfire Salvage Logging on Aquatic Ecosystems in the American West. *BioScience* **2004**, *54*, 1029–1033. [CrossRef]
75. Beschta, R.L.; Rhodes, J.J.; Kauffman, J.B.; Gresswell, R.E.; Minshall, G.W.; Karr, J.R.; Perry, D.A.; Hauer, F.R.; Frissell, C.A. Postfire Management on Forested Public Lands of the Western United States. *Conserv. Biol.* **2004**, *18*, 957–967. [CrossRef]
76. Pehl, M.; Arvesen, A.; Humpenöder, F.; Popp, A.; Hertwich, E.G.; Luderer, G. Understanding future emissions from low-carbon power systems by integration of life-cycle assessment and integrated energy modelling. *Nat. Energy* **2017**, *2*, 939–945. [CrossRef]
77. Elsen, P.R.; Monahan, W.B.; Dougherty, E.R.; Merenlender, A.M. Keeping pace with climate change in global terrestrial protected areas. *Sci. Adv.* **2020**, *6*, eaay0814. [CrossRef] [PubMed]
78. Dinerstein, E.; Joshi, A.; Vynne, C.; Lee, A.; Pharand-Deschênes, F.; França, M.; Fernando, S.; Birch, T.; Burkart, K.; Asner, G. A "Global Safety Net" to reverse biodiversity loss and stabilize Earth's climate. *Sci. Adv.* **2020**, *6*, eabb2824. [CrossRef]
79. Timmers, R.; van Kuijk, M.; Verweij, P.A.; Ghazoul, J.; Hautier, Y.; Laurance, W.F.; Arriaga-Weiss, S.L.; Askins, R.A.; Battisti, C.; Berg, Å. Conservation of birds in fragmented landscapes requires protected areas. *Front. Ecol. Environ.* **2022**. [CrossRef]
80. IUCN. The IUCN Red List of Threatened Species. 2020. Available online: https://www.iucnredlist.org/ (accessed on 20 April 2022).

81. Mackey, B.; Kormos, C.F.; Keith, H.; Moomaw, W.R.; Houghton, R.A.; Mittermeier, R.A.; Hole, D.; Hugh, S. Understanding the importance of primary tropical forest protection as a mitigation strategy. *Mitig. Adapt. Strateg. Glob. Change* **2020**, *25*, 763–787. [CrossRef]
82. Dinerstein, E.; Olson, D.; Joshi, A.; Vynne, C.; Burgess, N.D.; Wikramanayake, E.; Hahn, N.; Palminteri, S.; Hedao, P.; Noss, R.; et al. An Ecoregion-Based Approach to Protecting Half the Terrestrial Realm. *BioScience* **2017**, *67*, 534–545. [CrossRef]
83. Rosenberg, K.V.; Dokter, A.M.; Blancher, P.J.; Sauer, J.R.; Smith, A.C.; Smith, P.A.; Stanton, J.C.; Panjabi, A.; Helft, L.; Parr, M. Decline of the North American avifauna. *Science* **2019**, *366*, 120–124. [CrossRef]
84. Wagner, D.L.; Grames, E.M.; Forister, M.L.; Berenbaum, M.R.; Stopak, D. Insect decline in the Anthropocene: Death by a thousand cuts. *Proc. Natl. Acad. Sci. USA* **2021**, *118*, e2023989118. [CrossRef]
85. Hamilton, H.; Smyth, R.L.; Young, B.E.; Howard, T.G.; Tracey, C.; Breyer, S.; Cameron, D.R.; Chazal, A.; Conley, A.K.; Frye, C.; et al. Increasing taxonomic diversity and spatial resolution clarifies opportunities for protecting US imperiled species. *Ecol. Appl.* **2022**, *32*, e2534. [CrossRef]
86. Kauffman, J.B.; Beschta, R.L.; Lacy, P.M.; Liverman, M. Livestock Use on Public Lands in the Western USA Exacerbates Climate Change: Implications for Climate Change Mitigation and Adaptation. *Environ. Manag.* **2022**, *69*, 1137–1152. [CrossRef] [PubMed]
87. Fa, J.E.; Watson, J.E.; Leiper, I.; Potapov, P.; Evans, T.D.; Burgess, N.D.; Molnár, Z.; Fernández-Llamazares, Á.; Duncan, T.; Wang, S. Importance of Indigenous Peoples' lands for the conservation of Intact Forest Landscapes. *Front. Ecol. Environ.* **2020**, *18*, 135–140. [CrossRef]
88. Liu, N.; Caldwell, P.V.; Dobbs, G.R.; Miniat, C.F.; Bolstad, P.V.; Nelson, S.A.; Sun, G. Forested lands dominate drinking water supply in the conterminous United States. *Environ. Res. Lett.* **2021**, *16*, 084008. [CrossRef]
89. USDA. Forests to Faucets 2.0 [Spatial Data Set]. 2019. Available online: https://usfs-public.app.box.com/v/Forests2Faucets (accessed on 5 April 2022).
90. USGS. Protected Areas Database of the United States (PAD-US) 2.1: U.S. Geological Survey Data Release. 2020. Available online: https://www.sciencebase.gov/catalog/item/5f186a2082cef313ed843257 (accessed on 30 March 2022).
91. Cook, B.; Mankin, J.; Marvel, K.; Williams, A.; Smerdon, J.; Anchukaitis, K. Twenty-first century drought projections in the CMIP6 forcing scenarios. *Earth's Future* **2020**, *8*, e2019EF001461. [CrossRef]
92. Omernik, J.M.; Griffith, G.E. Ecoregions of the conterminous United States: Evolution of a hierarchical spatial framework. *Environ. Manag.* **2014**, *54*, 1249–1266. [CrossRef]
93. UNFCCC. *MLA*, 7th ed.; United Nations Framework Convention on Climate Change; General Assembly: New York, NY, USA, 1992.