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

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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

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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

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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%

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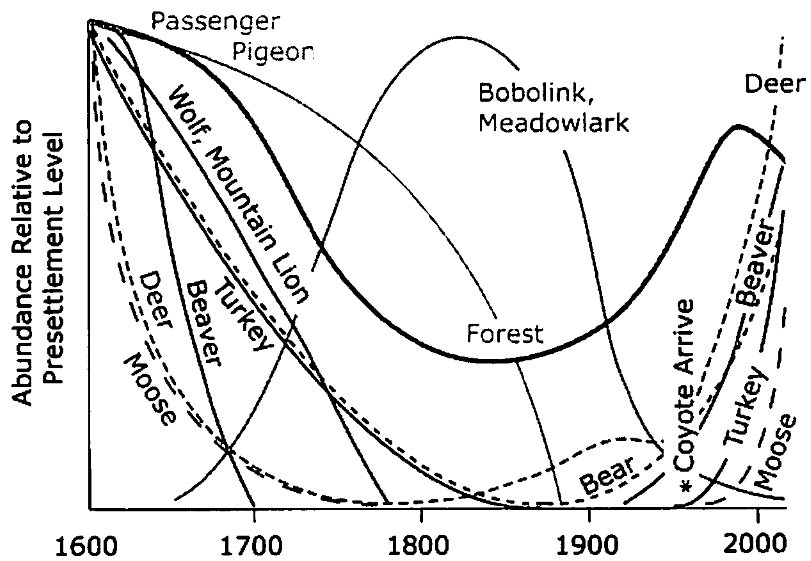

FIGURE 1
Changes in land cover and wildlife dynamics in New England from ~1600–2000. The green line shows the abundance, decline and then recovery of forest in New England, which paralleled the population changes in moose, beaver, and deer. The inverse trend is found in openland (early-successional) 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

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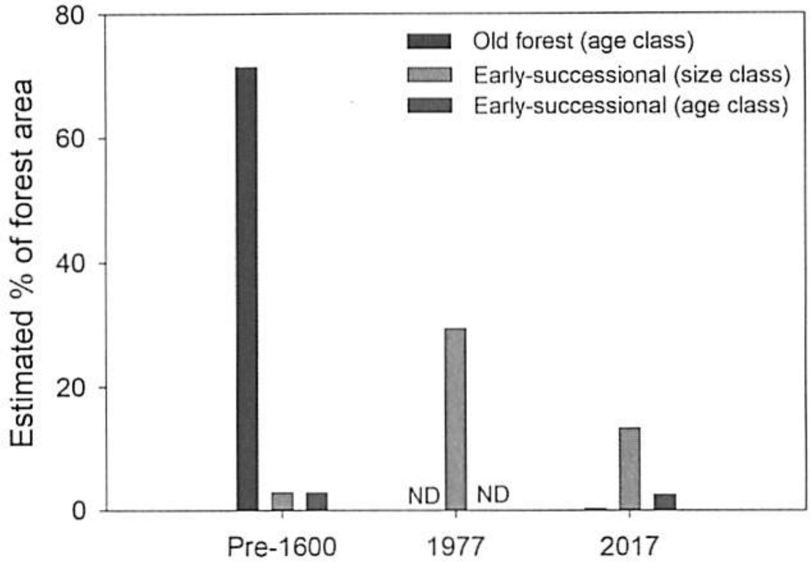

FIGURE 2
Estimated change in average % of early-successional and old forest habitat from pre-European settlement to current times in the Northeast US as extracted from multiple sources. Old forest is defined > 150 years old. The 1600 estimate for early successional forest is based on "seedling-sapling (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);

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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]."

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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).

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

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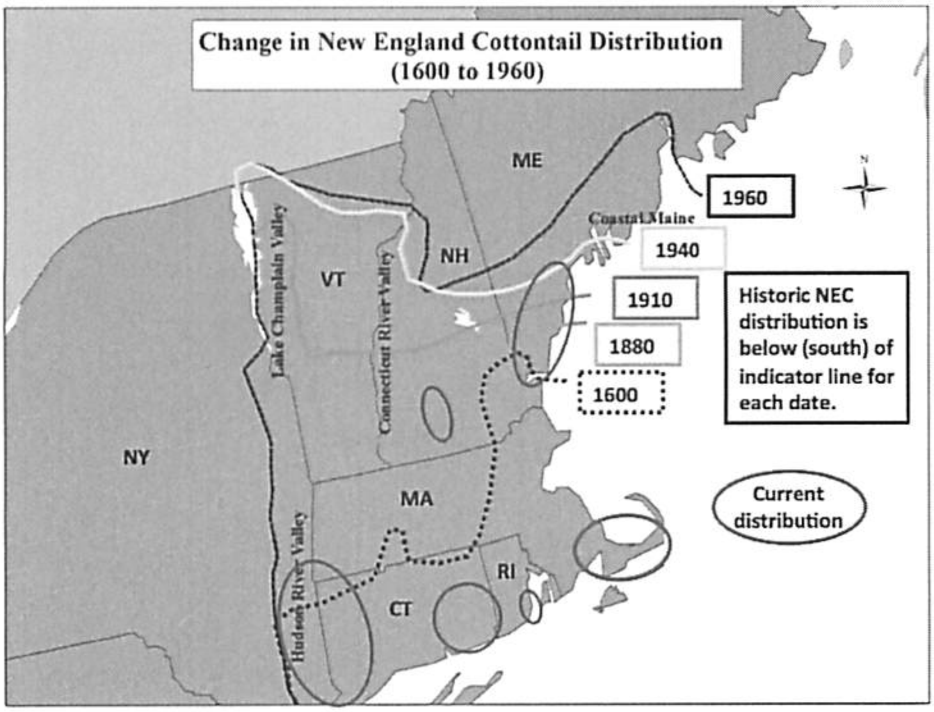

FIGURE 3
Changes in New England Cottontail (NEC) distribution over time. The estimated range of New England Cottontails (NEC) documented circa ~1600 (below the dashed line) included primarily Connecticut (CT) and Rhode Island (RI), and part of Massachusetts (MA). The distribution expanded dramatically 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

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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

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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;

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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

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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*),

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(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,

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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

(*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

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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 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.

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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.

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Supplementary material

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

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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

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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

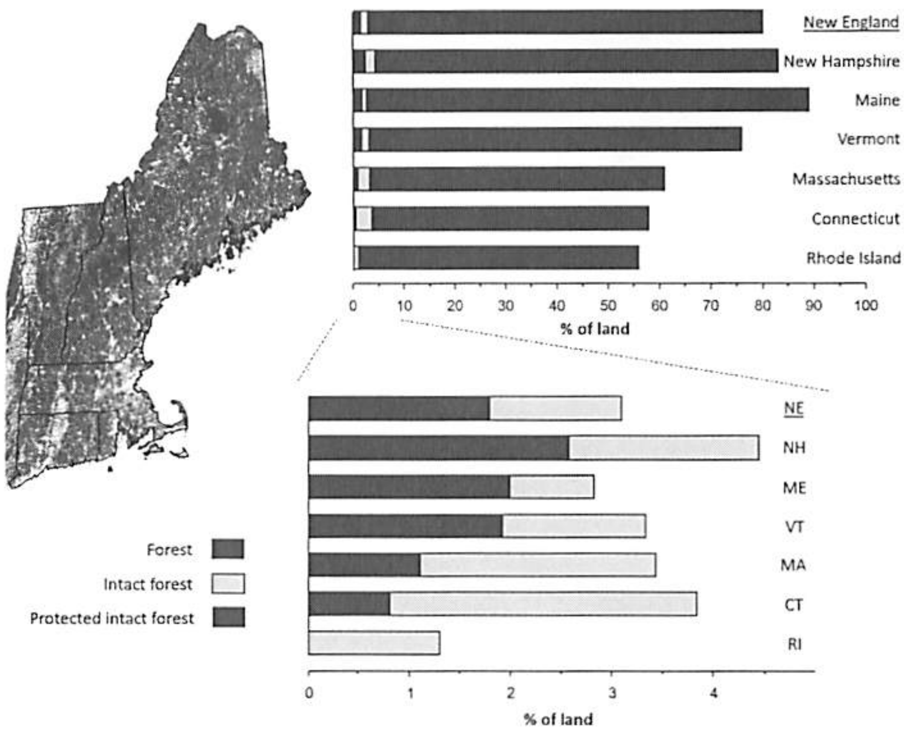

FIGURE 1 | Distribution of forest cover and intact "wildland" forest across six New England states. At left, map of overall forest cover (green) vs. forest protected legally (red) or managed currently (yellow) as intact in New England. At right, regional and state specific % forest cover (green), % managed 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 ≥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.

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Perspective

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

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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

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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].

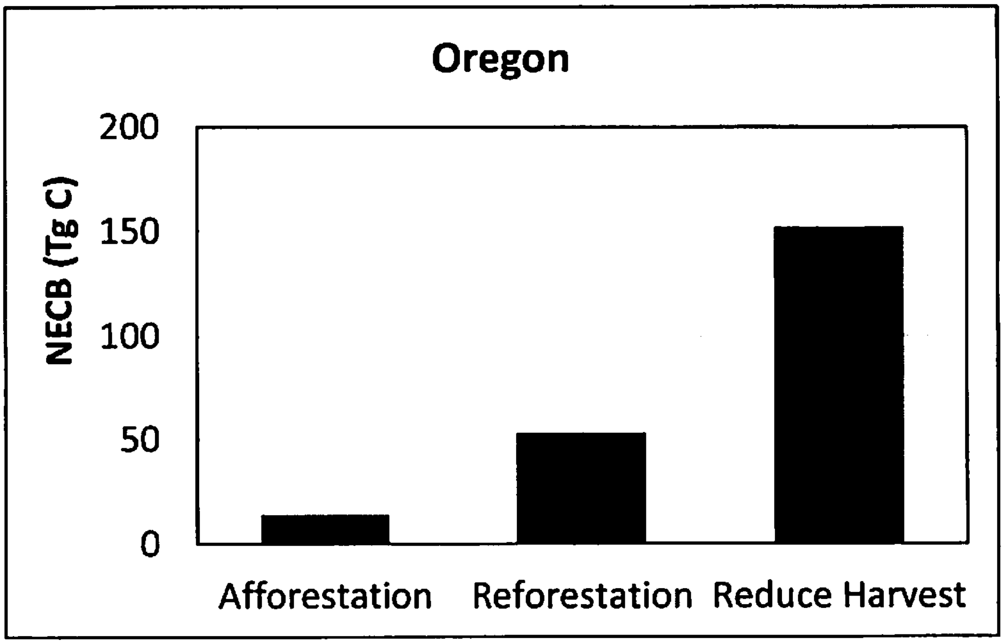

Figure 1. Land-use strategies to mitigate climate change across Oregon. Values on *y*-axis are cumulative change in net ecosystem carbon balance (NECB) from 2015 to 2100. Reduced harvest is a combination of restricted harvest by half on federal lands, and increased harvest intervals to 80 years on private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

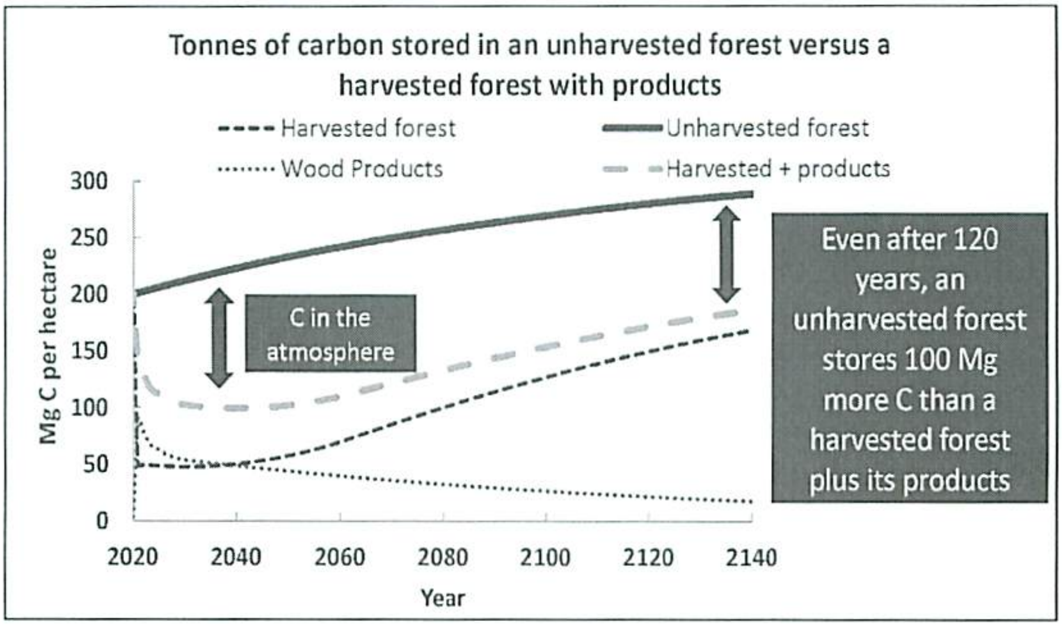

Figure 2. A mature forest with a carbon density of 200 tons of carbon per hectare (green line) is harvested (blue line) in 2020. This results in an immediate reduction of live tree carbon stocks. Approximately half of the aboveground carbon is removed and taken to the mills (as wood) while the other 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].

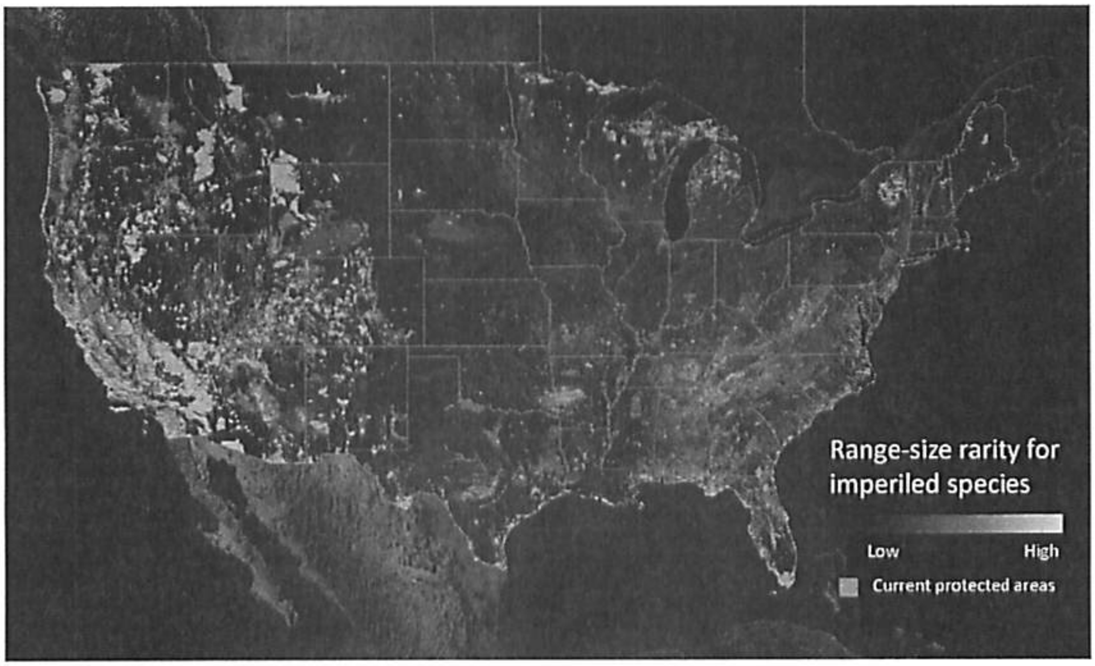

Figure 3. Summed range-size rarity of forest and non-forest species in the lower 48 states that are protected by the Endangered Species Act and/or considered to be in danger of extinction. Species include vertebrates (birds, mammals, amphibians, reptiles, freshwater fishes), freshwater invertebrates, 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.

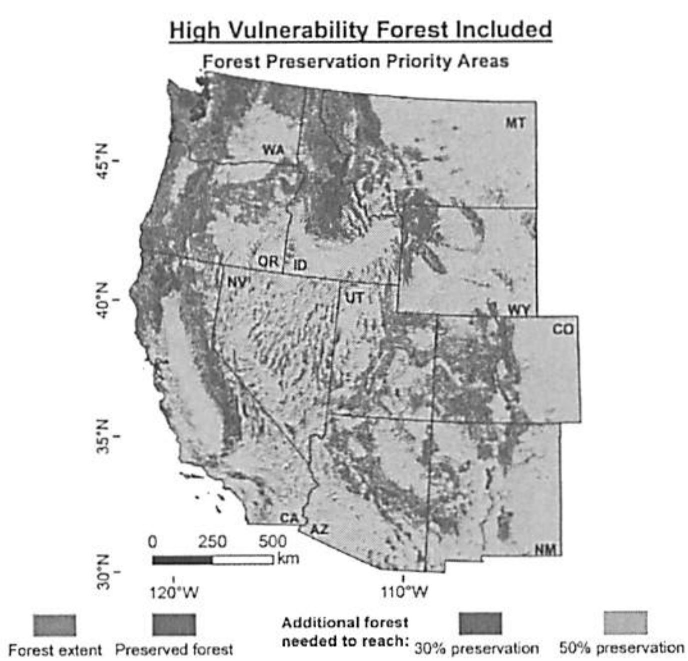

Figure 4. Forestlands that are currently preserved, and additional areas identified as high priority for protection of biodiversity and forest carbon for climate mitigation across the western U.S. Adapted from [5].

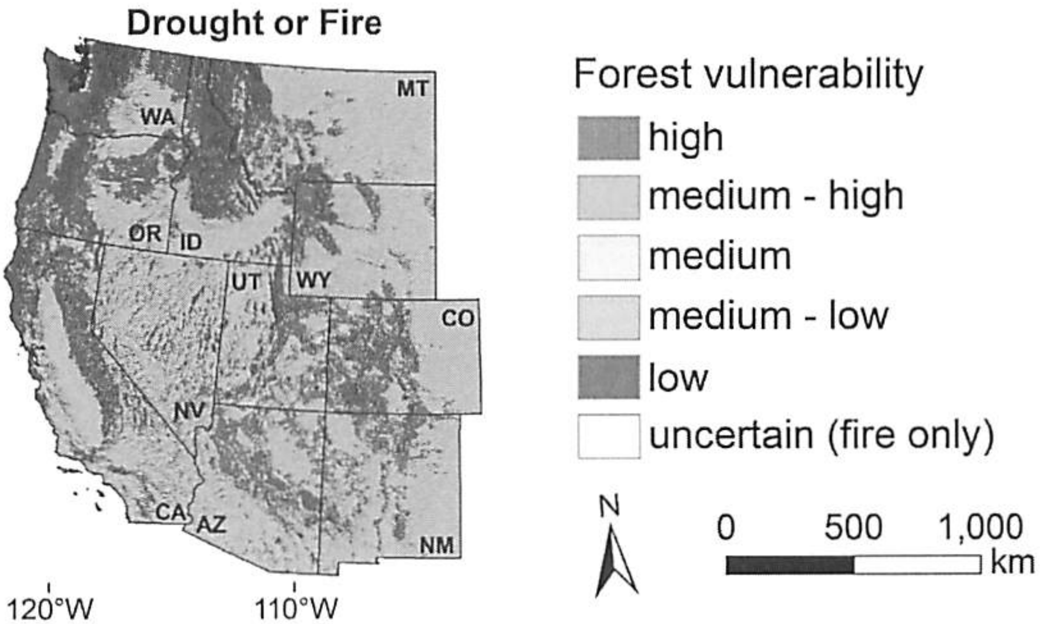

Figure 5. Vulnerability of forestlands to either drought or fire under future climate scenarios to year 2050. Adapted from [83].

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.

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