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Ecologist's Notebook
a biological exploration of western Pennsylvania

Signs of Summer 2: Whites' Woods

Posted on May 25, 2017 by hw7

Photo by D. Sillman

Last Monday Deborah and I went over to Indiana, Pennsylvania and went for a hike in a small, recreational area called Whites' Woods. We read about the trails on a blog site (http://whiteswoodsindianapa.blogspot.com/) and were excited to go and see some new trails and scenery.

Whites' Wood is a 250 acre recreational park situated in White Township just north of Indiana, PA. The land was once owned by a railroad company (Pennsylvania Railroad?) and then in the 1950's was sold to a real estate company (that was owned by someone named "White," I think). Finally, in the 1960's this 250 acre piece of land was donated by the White heirs to the township for use as a recreational park.

Whites' Woods is a publicly owned land resource that has been beset over its 50 plus years of existence by many of the problems facing much larger, more resource rich sites throughout the United States. Competing and often conflicting interests and goals (quiet, wooded trails for hiking vs. income from timber harvesting or natural gas development, areas set aside for wildlife vs. areas opened up for hunting, etc.) openly contend with each other for the use and fate of Whites' Woods. Township supervisors proposed as recently as 2007 to open significant areas of

White's Woods up for logging. This proposal was resisted by a "Friends of Whites' Woods" (FWW) organization. The initial timber removal plan was recognized as flawed by the Pennsylvania Department of Conservation and Natural Resources (DCNR) for both environmental and also possible legal reasons (based on the terms of the original land donation), but several areas within the woods were logged anyway

Photo by D. Sillman

The trails were broad and clear but not well marked. Blazes were few and far between and often the colors of the trail markers did not match up with the trail color code of the maps. There was, though, very little chance of getting lost in this small area (even for us!).

The trees were primarily yellow poplar, red maple, yellow birch and black cherry. The black cherry trees were in states of some age-related decay (and extensive woodpecker damage). This forest is, then, a predictable mix of sun-loving, fast growing trees that quickly grow in a site after clear cutting. Many of the poplars, strictly based on size, were between 60 and 80 years old (which would fit the recent history of the site). This could be a secondary growth forest but it is more likely to be tertiary. Heavy use of timber for building construction and for the railroad (fuel and track ties) in the early to mid Nineteenth Century probably took down first the virgin forest and then the secondary re-growth 60 or 70 years later. What we see is the probably the next re-growth stage. Interestingly, there were also some relatively large big-toothed aspens (probably 40 or 50 years old) planted in fairly regular intervals along one side of the trail. Big-toothed aspens are often planted on reclaimed or intentionally re-forested sites. They may represent some human involvement in the reforestation of Whites' Woods..

Photo by D. Sillman

As we walked up the curving trail we came across an increasing number of red oaks growing in the “double trunk” configuration suggestive of stump sprouting following logging. I estimated that the largest red oaks, based on size, were between 80 and 100 years old. There was also a great deal of downed wood throughout the surrounding forests. Fallen trees, broken trunks, and scattered limbs littered the spaces between the standing trees and suggested an actively re-sculpting forest that was thinning and pruning itself.

In the under-story New York fern, hay-scented fern, Christmas fern, garlic mustard and wild geranium (also called “crane’s bill”) were abundant. Along the edge of the trail were milkweed plants and small patches of multifloral rose.

Photo by D. Sillman

Off of the trail were several areas with thick stands of yellow birch and sugar maple saplings. These dense copses suggest relatively recent removal of the older, established trees and may be the sites of the 2007 logging referred to above. Throughout the poplar/birch/red maple forest there were also abundant sugar maple saplings growing. There had been some discussion on the Whites’ Woods blog about tree damage caused by excessive numbers of white-tailed deer, but our observations were that these woods are a robustly regenerating forest with a rich population of potentially long-lived sugar maples steadily growing up into the canopy.

We saw large numbers of robins noisily digging through the leaf litter, we heard (but did not see) wood thrushes all along our hike. We also heard northern flickers and saw abundant evidence of pileated woodpecker activity (large, rectangular holes in the black cherry trunks). I also saw a pair of flycatchers vigorously interacting up in the branches of some middle canopy trees.

We walked around the “Old Quarry” and then took a trail that led down a shallow, wooded ravine. The multifora rose was very abundant along the small stream that the trail followed. Maidenhair fern and interrupted fern were also along this part of the trail.

Photo by D. Sillman

Laying on the trail were several, golf ball sized, green balls that were incredibly, almost insubstantially light in weight. We opened one of them and saw that it was filled with an array of white fibers that converged on its center. These were “empty oak apple galls” made by the parasitic wasp *Amphibolips quercusinanis*. The female wasp lays her eggs in the leaf buds of oak trees (usually scarlet or red oaks) and hormones associated with the eggs drive the growing tissues of the emerging leaf to make this spherical chamber for the wasp larva. Deborah put one of the galls in her pocket, but by the time we got home it had dried out and collapsed.

Photo by D. Sillman

We also saw squawroot (*Conopholis americana*) (also called “cancer root” or “bear cone” growing in both large and relatively small clusters all over White’s Woods. Squawroot is more common in older forests, and its presence and relative abundance in a site may be significant indicators of forest age and stability. In areas where oak forests are being replaced by secondary forests that are dominated by maples or other non-oak tree species, squawroot is an increasingly uncommon and possibly threatened plant. Here is Whites’ Woods, though, the density of red oaks seems adequate for its sustained existence.

It is not clear in the literature if squawroot seriously compromises the health of its host tree. It is likely that it, by itself, may exist in a very stable parasite host symbiosis with its much larger and longer lived host oak or beech tree. But, if other stresses combine with squawroot’s presence, the health and vitality of the host tree may be reduced.

Photo by D. Sillman

Up on the top of the Whites’ Woods hill we came across a stunning flower blooming at the top of a spindly, eight foot tall, woody trunk. Based on the extreme length of the flower stamens, we tentatively identified it as pink azalea (*Rhododendron periclymenoides*). It is supposed to have very little fragrance, but it was too high up to check!

Photo by D. Sillman

And finally, we found a plant that we have been looking for over the past few weeks: fire pink *(Silene virginica).* Fire pink has a stunningly intense, five petalled, red flower and almost always grows in the crumbling soil of an eroding soil bank. It blooms in late spring/early summer. We followed its blooming season very closely on our northward hike on the Baker Trail back in the spring of 2010. Fire pink's flowers are long and tubular with nectaries and ova housed deep inside. Only organisms with long tongues (like hummingbirds and large butterflies) are able to reach its sweet nectar and, thus, deliver pollen to the ova. Ruby-throated hummingbirds are the principle pollinators of fire pink. June is very near if fire pink is blooming!

About the authors

This site is a synergistic effort of two retired Penn State University biologists, William Hamilton and Deborah Sillman. Hamilton and Sillman spent 34 years exploring the ecosystems of Western Pennsylvania and have collaborated not only on this blog site but also on an on-line representation of the [Penn State New Kensington Campus Nature Trail] ("The Virtual Nature Trail") and on a site describing some of the hiking trails of Western Pennsylvania ("Between Stones and Trees"). In 2020, after retiring from Penn State, Hamilton and Sillman moved west to be closer to family and changed the focus of this site to the descriptions and discussions of the ecosystems of Colorado. For the most part, Hamilton is responsible for the written content of these pages and Sillman is responsible for the photographs, editing, and technical support. Feel free to contact Dr. Hamilton (hw7@psu.edu) or Ms. Sillman (dys100@psu.edu) if you have any questions about our sites or about our topics.

https://sites.psu.edu/ecologistsnotebook/2017/05/25/signs-of-summer-2-whites-woods/

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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

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; Heeter 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 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%

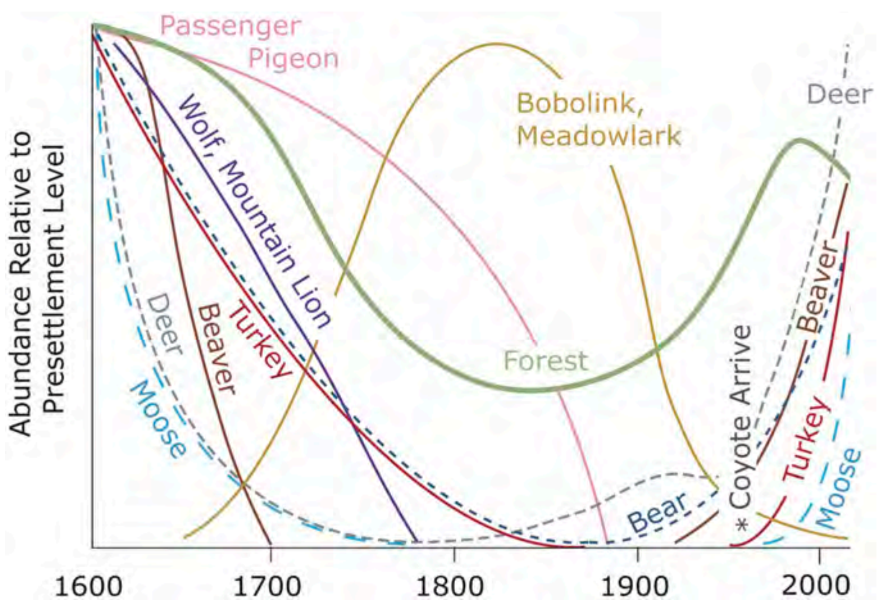

FIGURE 1
Changes in land cover and wildlife dynamics in New England from ~1600–2000. The green line shows the abundance, decline and then recovery of forest in New England, which paralleled the population changes in moose, beaver, and deer. The inverse trend is found in openland (early-successional)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

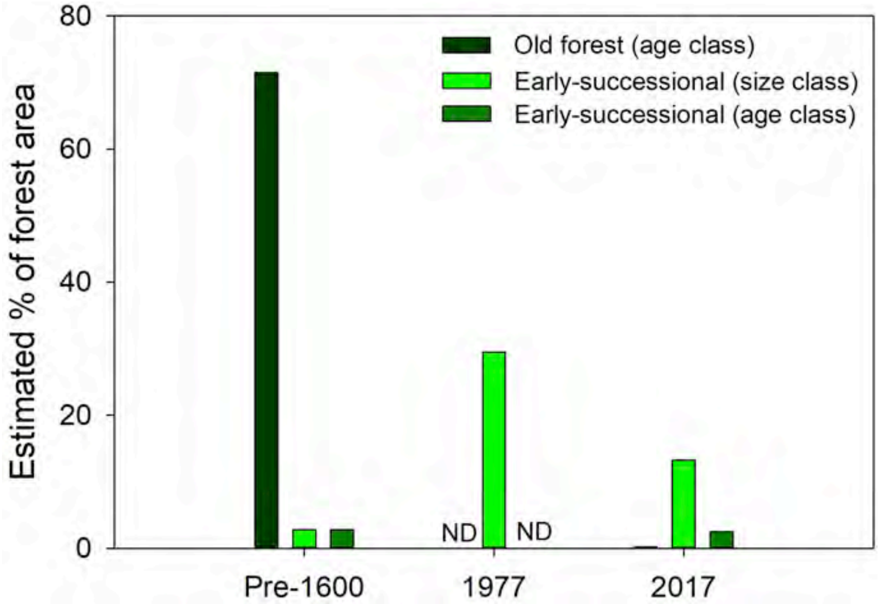

FIGURE 2
Estimated change in average % of early-successional and old forest habitat from pre-European settlement to current times in the Northeast US as extracted from multiple sources. Old forest is defined > 150 years old. The 1600 estimate for early successional forest is based on "seedling-sapling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

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]."

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

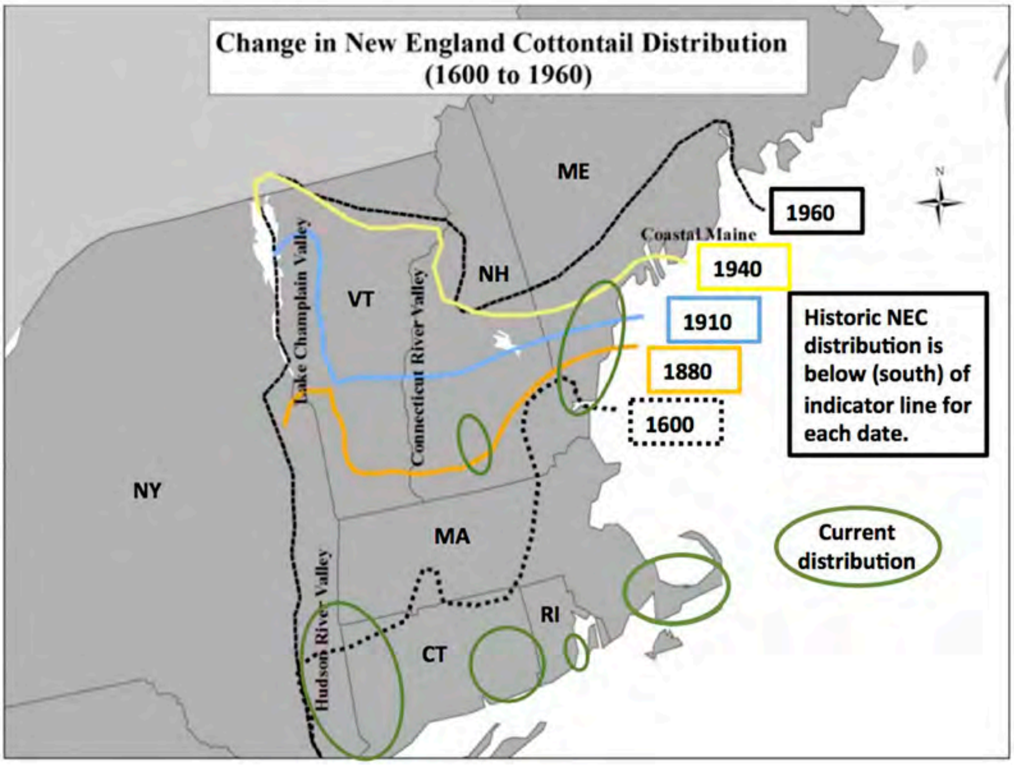

FIGURE 3
Changes in New England Cottontail (NEC) distribution over time. The estimated range of New England Cottontails (NEC) documented circa ~1600 (below the dashed line) included primarily Connecticut (CT) and Rhode Island (RI), and part of Massachusetts (MA). The distribution expanded dramatically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

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

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

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

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

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

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

2.2. Impacts on the atmosphere

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

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

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

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Abstract

Not all urban greenspace is the same. Natural area forests can provision more benefits than designed landscapes, and healthy natural area forests can provide more benefits than degraded and invaded forests. Yet there is little information about the scale of natural areas in cities and their management systems. We used data sets on city parkland from across the United States and surveyed practitioners to understand urban natural area forest extent and management. We find that urban natural areas are a dominant greenspace landcover, accounting for 68% of total city parkland across 96 of the most populous cities in the United States in 2019. In the same cities over a five-year period (2014–2019), natural area parkland decreased by 4% (15,264 hectares). At municipal scales, most cities are managing forested natural areas to conserve native species. Across the 108 organizations and 92 cities that responded to our online survey, many different management interventions are being used to steer forest structure and composition. These activities and their outcomes are being tracked nearly 70% of the time by the managing organizations, suggesting a strong data basis for adaptive management. However, challenges exist: 94% of organizations cite invasive species and limited funding as primary challenges. Lack of data and low public awareness of the value of natural areas are also considered primary challenges by more than 70% of the organizations surveyed. As cities embark on efforts to expand and improve greenspace, protecting natural area parkland from development and addressing the challenges managers of these ecosystems face are two very important goals.

Study Implications: Urban forested natural areas contribute to improving the livability and sustainability of cities. However, urbanization has environmental consequences that can lead to declines in tree canopy, introduced species, and the degradation of forest condition. Because urban forested natural areas are both vulnerable and valuable, ambiguity orbits around appropriate policies and management priorities. We provide the first national assessment of urban forested natural area coverage in cities and their management systems. This baseline data can be used by cities as a point of reference to begin to understand and contextualize natural area forests and common management challenges. This study highlights an emerging field of common forest management strategies adapted to dealing with urban situations that could lead to best-management practices for complex human-impacted forest ecosystems.

Keywords: urban forestry, urban tree canopy, invasive species, urban ecology, forest restoration, urbanization, land development, urban planning, forest management

Expanding the amount and improving the quality of greenspace is a common strategy to make cities more livable (Livesley et al. 2016). As cities look to greenspaces to mitigate heat, absorb stormwater, and provide areas for recreation, the type of greenspace influences the magnitude of benefits provided (Kondo et al. 2015, Mexia et al. 2018, Vieira et al. 2018). Natural area forests can provision some benefits at disproportionately higher rates per area compared with designed parkland. For example, forests can have a greater cooling effect on cities than designed greenspaces, and the bigger the forest, the greater the effect (Jaganmohan et al. 2016). In addition, forested natural areas provide critical habitat for native plants and animals safeguarding and connecting local biodiversity in a fragmented landscape (Ives et al. 2016). The provision of ecosystem services and protection of biodiversity are two commonly reported metrics in city sustainability goals (Nilon et al. 2017), with natural areas therefore having potential to contribute to these goals. However, natural areas are not featured prominently as nature-based solutions in city plans (e.g., policy reporting, climate action plans, and city resiliency plans) (Nilon et al. 2017). This lack of representation may arise from the lack of common descriptive data for urban natural areas, which could contribute to a lack of awareness and hence incorporation of natural area forests into actionable greenspace planning.

Common measurements of any natural resource are important to raise awareness, shape policy, contextualize patterns and processes, and allow for comparisons among management outcomes. Existing methodologies to assess and value urban forests across cities include remote sensing of urban tree canopies (Nowak and Greenfield 2010, Alonzo et al. 2016) and field-based, plot-level sampling to measure forest structure and composition (Nowak et al. 2008). Such approaches show the value of urban forests (Nowak et al. 2007), have been the basis for broadscale justifications and planning of urban tree planting programs (Locke et al. 2010), and inform urbanization and climate models (Lin et al. 2019). However, assessments that have focused on measuring the entire urban forest or city tree canopy (e.g., all trees in the city) typically do not distinguish between natural area forests and trees growing in designed environments (e.g., street and yard trees). The distinction between these canopy types is important for accounting, policy, management, and assessment of greenspace. For example, an assessment in New York City that stratified and measured natural areas apart from the rest of the tree canopy showed that the majority of trees and forest biomass occur in natural area forests; albeit they are a minority (25%) of the total tree canopy area (Pregitzer et al. 2019). Furthermore, the management required and challenges faced are different in natural areas compared with trees growing in designed landscapes. Street and yard trees are managed on a more individual basis (e.g., planted and removed upon mortality), drawing on arboriculture principals. Trees growing in these highly constructed environments can experience challenges to their growing environment such as confined rooting zones, which can lead to tree mortality (Roman and Scatena 2011). Natural area forests, in contrast, are managed as a stand of trees or a collection of stands, where trees can naturally regenerate and upon mortality are typically left in place to decompose. The management of forested natural areas applies silvicultural and ecological restoration principals. These differences make characterizing the condition, types, and management systems of urban natural areas an important part of urban greenspace management and policy.

Communicating priorities, understanding conditions, and implementing management and monitoring are a part of adaptive management strategies foundational to forest ecology and management. Forest management principals have been adapted to the urban context and have documented successful outcomes—for example, management of exotic invasive species and tree planting (Oldfield et al. 2014, Johnson and Handel 2016, Simmons et al. 2016)—but these outcomes are not well summarized beyond an individual site or project. The full breadth of management actions taken to deal with urban conditions has not, as far as we are aware, been assembled at a national level, and many questions and challenges about the applications of local findings to other city contexts remain (Oldfield et al. 2013). Management efforts are usually embedded within the structure of city parks and recreation departments, but little is understood about the challenges that cities face on a collective national basis. To characterize natural area parkland across the United States, we asked a series of questions about natural areas in cities and their management. To first understand the basic composition of urban natural area parkland, we asked: How commonly occurring are, and what is the area of, urban natural area forest in major cities of the United States? Is the amount of urban natural area changing over time? Then, to understand the management of urban forested natural areas, we asked natural resource practitioners who specifically manage these areas: (1) What are your primary factors (goals) considered in management? (2) What management interventions do you conduct? (3) What are the main challenges you face? (4) What management plans, policy reporting, and data do you use?

Methods

We drew on two data sets to describe trends in urban natural area parkland and its management across the United States.

Characterizing Natural Area Parkland Nationally

To characterize the amount of natural area parkland nationally, we used existing data compiled as part of the "city parkland survey" by the Trust for Public Land, Center for City Park Excellence (www.tpl.org). On a biannual basis, the 100 most populous cities in the United States self-report attributes about their city's parkland. The attributes we used were hectares of total parkland, natural area parkland, and designed parkland in each city. Their working definition of natural area parkland is as follows: "Natural and undeveloped areas are pristine or reclaimed lands that are left largely undisturbed and managed for their ecological value (i.e., wetlands, forests, deserts). While they may have trails and occasional benches, they are not developed for any recreation activities beyond walking, running, and cycling." Their working definition of developed parkland is the following: "Designed areas are parklands that have been created, constructed, planted, and managed primarily for human use. They include playgrounds, neighborhood parks, sports fields, plazas, boulevards, municipal golf courses, municipal cemeteries, and all areas served by roadways, parking lots, and service buildings." Using data from 2014 and 2019, we calculated the total hectares of natural area parkland, designed parkland, and total parkland. We then calculated the proportion of natural area parkland of the total, the total change between 2014 and 2019, and the percent change of natural area parkland in each city. Three cities did not report values for both years, so we excluded them from our analysis (Richmond, VA; Ft. Wayne, IN; and Indianapolis, IN). Anchorage, AK, reported having >283,400 hectares of city parkland (Supplementary Table 1), close to the collective amount of the rest of the cities combined; because this skewed the overall results substantively, we chose to not include it in our analysis. Because estimates of natural area parkland include more than just forested natural areas (e.g., also open grasslands and marshes), we treat those numbers we report as the maximum (and presumably an overestimate) of forested natural area. However, because forest can be the dominant landcover type historically in many cities, we expect the results do reflect general trends in forested natural area cover, as a type of natural area parkland across cities in the United States.

Urban Forested Natural Area Survey Development and Deployment

To understand the goals, activities, and challenges of managing urban forested natural areas, we developed a survey and solicited responses to a questionnaire from practitioners that specifically work in urban forested natural areas in cities having more than 50,000 people across the entire United States. The survey creation was an iterative process, with questions developed and then revised with input from external advisors and potential respondents (see "Acknowledgments"). The survey was administered online using Qualtrics Survey Software (Qualtrics, Seattle, WA, USA) under site license to Yale University. Because there is no active network for this specific type of land manager, we relied on existing networks of urban park professionals and urban forestry professionals to broadly distribute the questionnaire. We provided the following operational definition of urban forested natural area: "Forested natural areas refer to woodlands and remnant forests which occur as a forest stand, or a collection of stands. Forested natural areas are often managed at the stand level, with trees considered collectively as a forest, rather than on an individual basis. Street trees or park trees are not part of forested natural areas, and are often managed individually. Forested natural areas can be different ages and sizes but typically are >0.25 acre and can be young developing stands or mature remnant forests."

During the spring of 2018, the survey was distributed in partnership with the Trust for Public Land to the 100 most populous cities across the United States, the same network that completed the city parkland survey. Then, to reach a broader audience, the survey was further distributed to urban forest managers in cities that had >50,000 people. In our solicitation, we asked that if the recipient's organization did not own or manage urban forested natural areas, for the recipient to forward the survey to appropriate urban forest managers in their city or network. In each case, we sent the original solicitation and two follow up requests. The survey respondents were asked to represent their organizational views, not their personal views, as most questions were focused on land management activities and approaches of the organization at large, rather than the individual's role or experience. Only completed surveys were used. Any respondents that completed the survey but explicitly did not manage forested natural areas were identified by a filter question and removed from analysis. In total, 1,314 individuals received the survey over e-mail. One hundred sixty-six people started the survey, and 108 responses qualified for analysis. Therefore, we estimate a response rate of 8.2%. Just under half of responses (48) were from the 100 most populous cities—meaning that a response rate of 8.2% translated to responses from approximately half of those eligible cities—and the remaining responses (60) were from less populous cities (but still with >50,000 people). For 10 cities, we received more than one response, each representing a different organization. In all cases, either the land ownership, management jurisdictions, or scale at which the organization worked were different. We therefore included these responses, treating them as independent because they represented management in urban forested natural areas for an organization's distinct mission and goals.

The survey questions included both qualitative and quantitative questions. Questions were focused on the care of forested natural areas through management activities, reporting and planning, data and information available for decisionmaking, organization size, education of staff, and challenges for management. Our quantitative (both ordinal and categorical), closed-end questions used a predefined set of response categories facilitating direct comparison across all respondents. Qualitative, open-ended questions, by contrast, provided respondents the opportunity to develop their own answers. Organizational demographic data were also collected to determine organization size and education of both field and managerial staff. In this article, we report on a subset of the questions. The full questionnaire is in the Supplementary Materials.

Data Analysis

Closed-ended questions were primarily analyzed by calculating the proportion of the total number of respondents to that question (n = 108). In a handful of cases (n = 3), a respondent did not populate answers to each field in a multipart question. Because the majority of that question was answered, we kept these responses and reduced the sample size for that field to the total completed responses (n = 105–107). In cases where a range was given as a multiple choice (e.g., 1–10), we used the median value in totaling responses (e.g., total hectares). In this article, we specifically focus on the subset of the questions related to forested natural area management themes and challenges. The questions included in this survey are indicated in the Supplementary Materials. Summary statistics were calculated using the open-source statistical software R (version 3.6.2; R Core Team 2020).

Results

Characterizing Natural Area Parkland across the United States

The majority of city parkland is natural, rather than designed (68% in 2019). The total amount of natural area parkland reported across 96 US cities was 317,465 hectares in 2014 and 302,201 hectares in 2019 (Table 1). The mean percent of total city area for natural areas parkland was 7% in 2019. In total, natural area parkland declined by 4% (15,264 hectares) over the five-year period. The amount of natural area parkland per city ranged from 0 (Newark, NJ;

Table 1. Total hectares of designed and natural area parkland in 96 of the most populous cities in the United States.

	2014	2019
Natural area parkland		
Total hectares reported	317,465.6	302,201.2
Mean (± standard deviation [SD]), median hectares per city	3,306.93 (±4,885.0), 1,005.9	3,147.93 (±4,635.3), 1,130.97
Designed parkland		
Total hectares reported	125,436.4	141,515.0
Mean (±SD), median hectares per city	1,306.63 (±1,261.6), 981.4	1,474.11 (±1,442.6), 1,013.2
Total parkland		
Total hectares reported	443,455.1	443,716.2
Mean (±SD), median hectares per city	4,619.3 (±5,560.5), 2,145.3	4,622.04 (±5,282.7), 2,309.7

and Helali, HI) to 24,114 hectares (Jacksonville, FL), and designed parkland ranged from 110 (San Diego, CA) to 20,224 (New York, NY) hectares in 2019. Just over half (51 cities) lost natural area parkland cover, whereas just under half (45 cities) saw an increase in natural area parkland cover over the five-year period. The percent change in natural areas parkland ranged from –100% to a +3797% per city (Supplementary Materials). Of the 51 cities that saw a decrease, the mean percent change was –22.9%, and of the 45 cities that saw an increase, the mean change was 169%, but the median was +10%. During the same period of time, overall parkland increased by 261 hectares and designed parkland increased by 16,078 hectares (Table 1), which suggested then that designed parkland is at least in part replacing natural area parkland.

Land Manager Survey Demographics

Results represent 108 survey responses from 36 states in 92 cities across the United States that actively manage urban forested natural areas. The majority of respondents (66%) were from municipal governments, 16% were from nonprofits, 8% were from state and local governments, and 10% of the respondents listed "other," which often included unique governance structures of private-public partnerships. The total hectares of forested natural areas represented by the respondents include an estimated 124,936 hectares. Most of the organizations (84%) are the primary landowner, whereas 8% manage but did not own the land, and 7% did not know the number of hectares owned or managed by their organization. Responding organizations have been managing forested natural areas for different amounts of time, with 28% managing forested natural areas for less than 20 years, 34% between 20 and 50 years, 31% for more than 50 years, and 8% "did not know." Forty-three percent of field staff, and 74% of senior management had a college degree in some field of natural resources.

Primary Factors Considered in Management Decisions

Conservation of native species was a primary factor in decisionmaking with a majority of respondents (61%) listing it as one of their top three factors. Plant biodiversity was the second most common factor considered with 40% of respondents listing it in the top three (Figure 1). Urban heat island, climate change, public access, and proximity to low-income neighborhoods had the lowest number of respondents (<10%) listing them as primary factors in their decisionmaking. This could be a signal of general lack of consideration in decisionmaking for these same factors, rather than them being secondary to another factor, because the majority of respondents (>50%) listed these factors as something they did not consider (Figure 1).

Types of Management Interventions

Invasive understory species removal is the most commonly conducted management activity, with 91% of respondents conducting this activity (Figure 2). Most respondents were conducting all listed management activities except for release thinning[1] of native trees (Figure 3). Although the focus is on conservation of native trees, release thinning as a type of forest stand improvement is reported as a rare type of management activity for urban natural area forests. Invasive tree removal is, however, commonly conducted (75% of respondents; Figure 2). When management activities are implemented, in all cases the outcomes were monitored nearly 70% of the time for all types of management (Figure 2).

Challenges to Natural Area Forest Management

All the challenges listed were considered important or very important by the majority of respondents (Figure 3). Limited funding or staff and invasive species were ranked jointly as the top challenge, with 94% of respondents listing them as very important or important. Limited data was ranked as an important challenge with 77% of organizations listing it as important or very important to achieving their goals. Uncertainty in management approach was considered to be the least important of the listed challenges, yet 56% of all respondents still considered it important or very important (Figure 3).

Data Available and Used for Management

Maps of conservation zones are the most common type of data available, with 68% of respondents having and using these maps. Half of respondents (50%) reported having ecological baseline data on measures of groundcover composition and cover (Figure 4), and 43% reported having and using measures of forest structure and composition in their decisionmaking. Climate change projection data were used for management decisions by only 26% of respondents, despite most organizations (68%; Figure 3) listing climate change as a challenge that natural area forests face. Less than a quarter (23%; Figure 4) reported having data on understory tree regeneration. Some data on social measures such as the number and

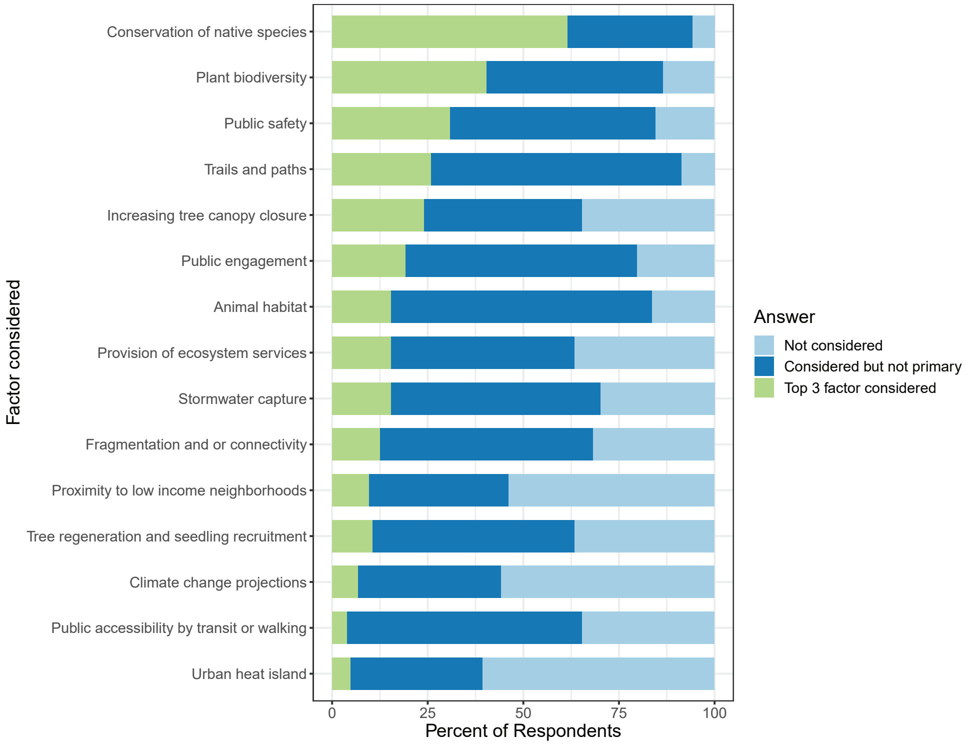

Figure 1. Primary factors survey respondents (n = 108) considered in decisionmaking for urban natural area forest management. Green bars represent the proportion of each factor that was ranked in the top three, by the 108 organizations that responded, of all the listed factors considered for decisionmaking.

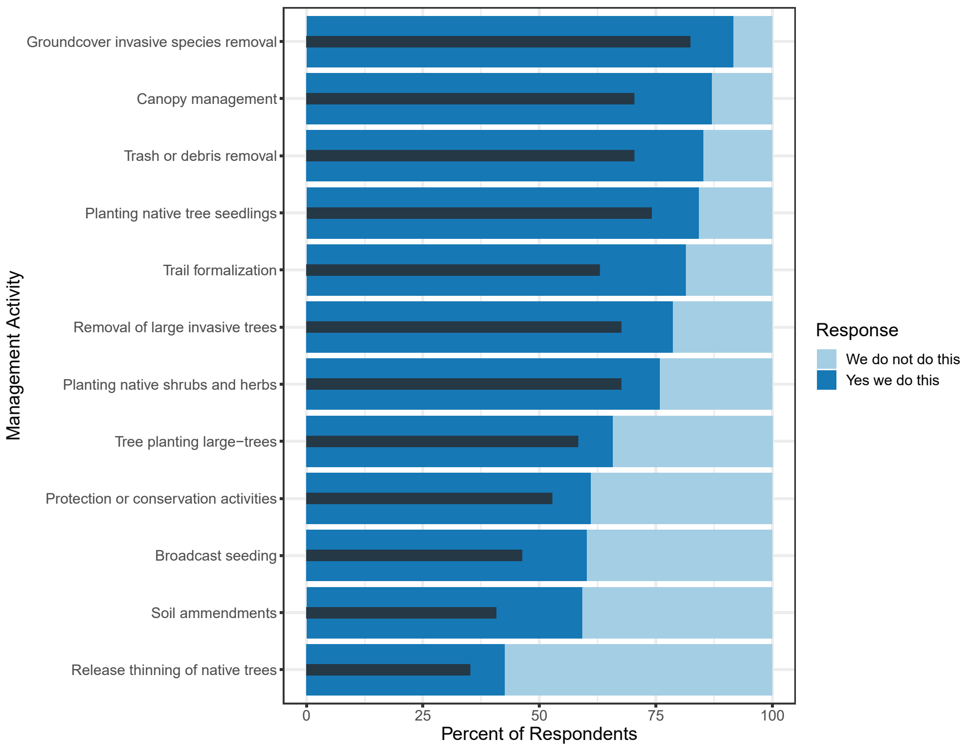

Figure 2. Management activities conducted in urban forested natural areas by organizations (n = 108) in cities across the United States. Responses show the proportion of the responding organizations that do each activity (dark blue) and, if they do that activity, the proportion that does some monitoring of those actions (gray, narrow embedded bars).

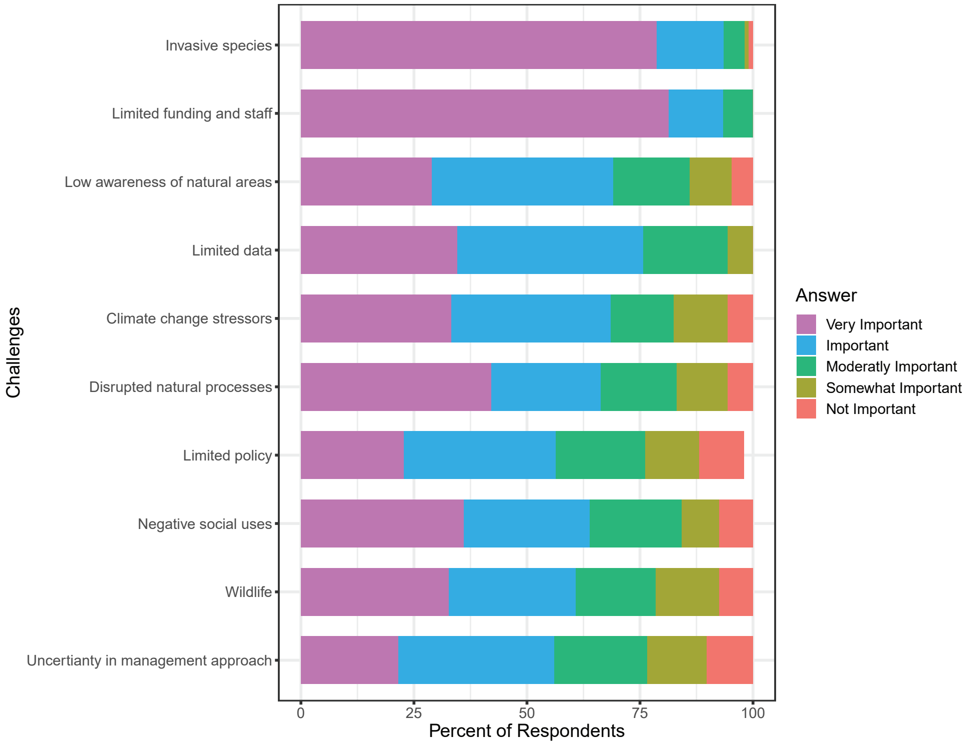

Figure 3. Challenges that organizations (n = 108) face in urban forested natural area management. Responses show the level of importance, as rated by each responding organization, of each factor.

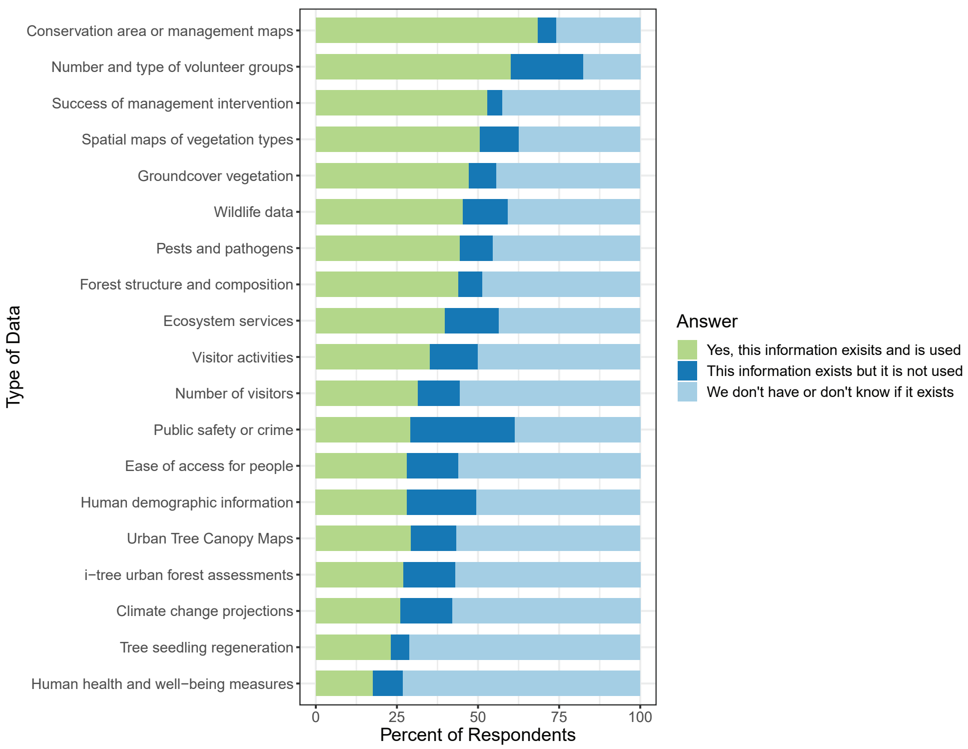

Figure 4. Data available and used for decisionmaking by organizations (n = 108) managing urban natural area forests. Responses show the proportion of responding organizations that have access to and use different social and ecological data to manage forests in their city.

type of volunteer groups were available and used by the majority of respondents (60%). Whereas other measures, such as data on human health and well-being effects of natural areas were not widely available, with only 17% of respondents having such data. In some cases, data that were available were not used for decisionmaking. For example, social data on crime, demographics (e.g., race and income), and the number and types of volunteer groups were sometimes not used (21–32%), and ecological data on ecosystem services, tree canopy cover, and i-Tree reports (www.itreetools.org) were not used by 15–16% of respondents (Figure 4).

Discussion

Net Loss of Urban Natural Area Parkland

Overall, we found a net loss of natural area parkland across US cities. Reported changes were especially dramatic in some cities. For example, the city of Houston (TX) lost 30% of its natural area parkland (7,960 hectares), and Nashville (TN) lost 28% of natural area parkland (3,294 hectares), between 2014 and 2019. As human populations increase, open land is converted to accommodate development, and both Houston and Nashville saw an increase in population during the same five-year period. Because the total hectares of natural areas lost is greater than the hectares of total parkland lost, we expect that some of the natural area parkland remained parkland but was converted to designed parkland (which saw a net gain). The consequences of natural area parkland decline could lead to losses in quality of life for residents and of biodiversity. For example, less access to nature or lower-quality nature can lead to lower levels of physical activity for people (Oyebode et al. 2015) and lower city resilience to increased temperatures (Melaas et al. 2016), and plants and animals can become locally extinct through habitat loss.

At the same time, natural area parkland in many cities increased. The city of Detroit (MI) saw a 353-hectare increase in natural area parkland, and New York City saw an increase of 95 hectares. This could be a result of proactive park acquisition. During that period in New York City, the conversion of a former landfill, Freshkills Park, was completed and added natural area parkland to the city's portfolio. Detroit has experienced significant population decline, losing more than 50% of its population in the last 70 years (change from ~1.8 million to ~700,000 people), because of a decline in industry and economic collapse. The increase in parkland there could be connected to the conversion of vacant houses to open space and changes in zoning and land ownership because of these circumstances. Notably, the city of North Las Vegas saw a 3,797% increase, or a change from 169 hectares in 2014 to 6,595 hectares in 2019, which is due to the inclusion of natural areas under the jurisdiction of the Bureau of Land Management that fall within the municipal boundaries and not likely because of a change in land use type or ownership. This example highlights how self-reporting of land cover, rather than using common quantifiable methodologies could provide some inaccuracies. However, in all cases, these shifts are specific to the circumstances and data available in each city, and given that much of urban parkland is typically owned, regulated, and managed locally by municipal governments, there is opportunity for more coordination within and across cities on land use metrics. Urban natural area parkland is a primary way in which the majority of the population experiences everyday nature and seeks refuge (Sonti et al. 2020). Therefore, it is important to learn more about the factors driving decisions to convert natural area land to other uses and to look at drivers of decisions that add natural area parkland to a city. Factors such as the quality of natural areas or human and neighborhood demographics could be important to evaluate in the decisions to convert, protect, or acquire natural areas.

Management of Urban Forested Natural Areas

We found evidence of well-established urban natural area management programs in cities across the United States. Although many factors are considered in decisionmaking, the evidence that native species conservation is a dominant factor suggests that the functioning of native-dominated forest ecosystems in cities is highly valued. Most organizations are using multiple approaches across forest structural layers to promote native species and healthy forests. Removing invasive species and planting tree seedlings in the groundcover layer are especially common management activities, suggesting the long-term trajectories of city forests are considered. Invasive species groundcover can outcompete native species, and this could lead to a decline in forest succession and health (Martin 1999, Stinson et al. 2006). The management strategies reported to conserve native species suggest that the establishment of native species and shifting the trajectory of areas with invaded groundcover toward native species are applied across most cities.

Invasive species removal and large-scale tree planting efforts are, however, expensive. It is perhaps then not surprising that limited resources and staff are a top challenge. Despite these limited resources, our survey found that most organizations are conducting similar management activities and monitoring the effectiveness of management interventions. Yet, uncertainty in management approaches remains. Uncertainty in management approach was considered the least important challenge, yet 56% still considered it important or very important. This result may suggest that there is an opportunity to synthesize management outcomes from multiple cities to begin to document, and then establish, tested techniques in urban settings across multiple cities. Certainly, documentation of adaptive management in different contexts was the basis for establishing proven silvicultural recommendations that ultimately have turned into management principals in rural forests. The same approach might therefore be profitably adopted for urban forests. However, unlike national parks and state or national forest systems, the ownership and management of forests in cities is siloed and bounded by individual-city municipal governments and local organizations. Therefore, to cross city boundaries and distill information on urban natural area forests and their management, additional coordination and incentive will likely be needed to connect city stakeholders.

The majority of organizations list data availability as a challenge. A lack of data does not mean a lack of appropriate management. Managers often act on experience and personal observations as evidence to justify a specific management intervention (McKinnon et al. 2015) but view that these decisions can be improved when vetted with data. Notably, baseline data (e.g., forest structure and composition and climate projection) are less commonly available/used than data on monitoring and management activities. Keeping track of management outcomes can help to justify the resources needed and being spent to achieve desired conditions and to adapt appropriately. However, it appears there could be an opportunity to come to consensus on some fundamental data sets (e.g., amount of natural area forests, forest structure, and forest composition) that could be collected uniformly across cities. These types of data across multiple cities and metropolitan regions could help to bridge understanding of forests and best-management principles across cities, regions, and the nation.

Part of the reported lack of awareness and policy in urban forested natural areas could be due to lack of common metrics across cities (aside from what we provide in this article). The use of evidence-based conservation targets (sensu Odum 1970) is an approach to connect science and data to policy and decisionmaking. Building policy or management decisions based on anecdotes and personal experience, rather than standardized evidence, can lead to unsuccessful, expensive, and repeated mistakes (Sutherland et al. 2004). In the case of cities, it appears there is growing awareness that a collective resource that documents management interventions and their outcomes might be valuable for informing forest management in urban environments. Such a structured resource could help to contextualize the unique and specific management that occurs in the urban environment and build a foundation of evidence that can expand the field of practice.

Conclusions and Opportunities for the Future

We found that natural areas are a dominant type of parkland in US cities. There appears to be an emerging field of common forest management strategies adapted to dealing with urban situations that could be further developed into best-management practices for urban environments. Investing in knowledge sharing and synthesis from land managers who have expertise and local data could help to connect local cities to a regional network of practitioners facing similar challenges. There also seems to be an opportunity to reposition the relative importance and role for this type of urban greenspace in cities. Bringing together the field of practice across cities might be one mechanism to redress the lack of awareness about natural area forests that our survey reveals. As urban land continues to expand, safeguarding natural areas within cities will have lasting impacts on the quality of life for millions of people, and yet our work reveals that in some cities and overall, there is a net decline in natural area forest cover. Below we offer opportunities to advance science and management of forested natural areas in cities:

- Accurately characterize city natural areas across the country using methodologies that produce high-resolution maps that distinguish between different types of greenspace, including natural areas across many cities (O'Neil-Dunne et al. 2014). This approach would result in more transparent and accurate estimates of natural area parkland and facilitate the incorporation of natural area planning more easily into decisionmaking.
- Establish guidelines and case studies for legal protection of existing natural area parkland and innovative approaches for land acquisition in cities.

- Further understand and synthesize the types of monitoring data that exist across cities. Use these results to build the case for increased investment in the documentation and management of urban natural areas.
- Strengthen partnerships locally, regionally, and nationally between land managers, decisionmakers, and researchers across cities. Such connectivity could lead to multicity designed experiments that allow for comparisons within and across cities of management interventions. The networks could also serve as educational opportunities for the public to increase awareness of natural resource management and governance of urban forested natural areas.
- Further understand the barriers of using existing climate change projection data for urban areas and gaps in data. Learn from city land managers and decisionmakers what data and delivery of information could support action toward mitigating the negative impacts of climate change impacts in cities. If connected with socioeconomic data, which was also rarely used in decisionmaking, it would be feasible to couple natural area management with environmental justice goals, which seems especially important given that climate change is expected to exacerbate such injustices.

Supplementary Materials

Supplementary data are available at *Journal of Forestry* online.

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Endnote

1. Removing native trees in a stand often as an approach to reduce competition and improve forest stand structure and composition toward a more desired trajectory.

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Natural Areas in the Twenty-First Century

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Conservation Issues

Natural Areas in the Twenty-first Century

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INTRODUCTION

The natural areas movement is one of North America's distinct contributions to conservation. In the July 2023 issue of the *Natural Areas Journal*, we provided a brief history of this movement (Noss et al. 2023). The history article was excerpted from a report by the Scientific Advisory Committee (SAC) of the Natural Areas Association (NAA), produced in August 2022. The full report is now available on the NAA web site (https://www.naturalareas.org/docs/NAA_21st_century_6_2.pdf). The SAC report addressed several questions relevant to the NAA and its future. These questions include: Are natural areas still relevant to the public in the twenty-first century? Do they still serve the purposes for which they were established? How might natural areas be better designed, managed, and marketed to meet changing environmental and social conditions over the remainder of this century? In the present article, we summarize the SAC report.

What Qualifies as a Natural Area?

We favor a broad, relativistic definition of natural area: "*A natural area is an area of land or water of any size where relatively natural geomorphological, ecological, and evolutionary processes predominate over anthropogenic processes and where assemblages of native species in natural communities generally prevail over non-native species.*" Given this definition, many kinds of formally designated areas in the United States and Canada may qualify as protected or conserved natural areas. These kinds of conservation areas are listed in the full report. Because "natural" is a relative concept, for all kinds of natural areas there are two continua: a continuum of naturalness (or quality) and a continuum of protection. A worthwhile objective is to use management and restoration to help guide natural areas toward higher-quality states and higher-protected states.

Role and Function of Natural Areas Historically and Today

To what extent are the traditional perceived values of natural areas still accepted and relevant? Below, we summarize some of the long-recognized values of natural areas and offer some suggestions of emerging values that are likely to become more important within the near future. Values in addition to those summarized below are discussed in our full report.

As Places to Protect Biodiversity: Biodiversity (short for biological diversity) can be defined as "the variety of life and its processes. It includes the variety of living organisms, the genetic differences among them, the communities and ecosystems in which they occur, and the ecological and evolutionary processes that keep them functioning yet ever changing and adapting" (Noss and Cooperrider 1994, modified from Keystone Center 1991). The loss of biodiversity, particularly species extinctions, has become one of the most prominent global crises, and it is occurring in North America as well as on other continents. For example, a recent study showed that 51 species and 14 subspecies and varieties of vascular plants have become extinct in the continental United States and Canada since European settlement (Knapp et al. 2021). This is undoubtedly a gross underestimate of the true extinction rate given the dearth of plant surveys in many areas.

Direct destruction as well as fragmentation and degradation of habitat is generally considered the greatest proximate threat to biodiversity, even more so in these times of rapidly changing climate (Noss and Cooperrider 1994; Wilcove et al. 1998; Haddad et al. 2015; Fletcher et al. 2018). Protection, restoration, and management of habitat is therefore the most promising strategy for reducing extinction rates and maintaining the healthy ecosystems and ecosystem services upon which all species, including humans, depend.

Among the kinds of species and habitats most in need of protection, restoration, and enlightened management are (1) imperiled and vulnerable taxa; (2) endemic taxa and disjunct and peripheral populations; (3) ephemeral habitats for migratory species; (4) representative, under-represented, or imperiled ecosystem types; and (5) areas of high ecological integrity.

As Benchmarks or Control Areas for Scientific Comparison with Anthropogenic or More Strongly Manipulated Areas: The value of natural areas as benchmarks—where natural processes dominate—was recognized right from the beginning of the natural areas movement. As Leopold (1949) commented, "A science of land health needs, first of all, a base datum of normality, a picture of how healthy land maintains itself as an organism." Manipulative research in land management benefits from having relatively unmanaged control areas, which represent the same ecosystem types as those being managed, to better gauge the success of management experiments. Natural areas are not ideal controls because no landscape is a perfect replicate of any other, and many human impacts (such as air pollution and climate change) are far-reaching, but they can be the best available and are far superior to an absence of unmanipulated areas.

Historical, Cultural, Scenic, and Recreational Values: Non-biological factors, such as historical, scenic, and recreational values, may be as important as biological values for stakeholders engaged in many conserved natural areas. The key consideration for managers is to ensure that these values are supported in ways that are compatible with the primary natural area values present on site. Scenic and recreational values of natural areas are important because people appear to have a psychological need for nature, whether they realize it or not. A substantial body of research has confirmed the salubrious effect of nature on human physical and emotional health and intellectual development (e.g., Louv 2011; Flies et al. 2017; Oh et al. 2017; Aerts et al. 2018).

Natural Areas as Important Functional Components of Ecosystems and Landscapes

Historically, most attention from natural areas professionals has been given to species populations and to natural communities defined narrowly (e.g., a calcareous fen) and at a fine spatial extent. Beginning in the 1980s, several authors called for more attention to planning on a regional landscape scale (Noss 1983), for an expanded coarse filter that includes functional landscape mosaics (Noss 1987; Aplet and Keeton 1999; Poiani et al. 2000; Groves 2003), and for generally greater attention to ecosystem dynamics and the landscape matrix (Franklin 1993; Lindenmayer and Franklin 2002) in conservation planning and management.

As noted by Franklin (1993), "Designing an appropriate system of habitat reserves is one landscape-level concern. Understanding and appropriately manipulating the landscape matrix is at least equal in importance to reserve issues, however, since the matrix itself is important in maintaining diversity, influences the effectiveness of reserves, and controls landscape connectivity." The landscape context of sites, specifically their connectivity or proximity to other protected areas, is just as important as the content of sites (Noss and Harris 1986). This consideration has grown more urgent with increased recognition of the need for species to shift their distributions in response to climate change (Heller and Zavaleta 2009).

Challenges for Natural Areas in the Twenty-first Century

Natural areas managers now face unprecedented challenges that will continue well into the future. Many of these issues are not new threats to biodiversity and typically can be managed using conventional conservation approaches (e.g., managing for species viability, removing invasive species, and restoration of altered natural disturbance regimes). Visitor usage rates also can be managed or regulated to mitigate risks to natural and cultural resources. However, when these threats are experienced synergistically, or as extreme events, they can cause increased stress on species and ecosystems, especially those that are already degraded or endangered. Below, given space limitations, we address just a few of these challenges; others are discussed in our full report.

The Effects of Climate Change and Frameworks for Response: The twenty-first century has seen increasing calls for the consideration of climate change in conservation planning and action (e.g., Noss 2001; Millar et al. 2007; Heller and Zavaleta 2009; Aplet and Cole 2010; Cross et al. 2012; Stein et al. 2013; Prober et al. 2019; Peterson St-Laurent et al. 2021). Growing recognition of this problem indicates an urgent need for new skills, tools, and improved understanding of ecological responses and transformations to help make informed decisions for conservation action (Abrahms et al. 2017; Belote et al. 2017a, 2017b; Lam et al. 2020; Hylander et al. 2022).

One crucial consideration is that climate change is occurring in landscapes that have been highly fragmented and degraded by human activities. Species that once could have tracked shifting climate zones through natural dispersal no longer can do so. They must now attempt to disperse across landscapes containing fragments of natural or seminatural habitat, and the landscape matrix is occupied by various human land uses that create movement barriers. Also, many invasive nonnative species may fare better than native species under future climate scenarios, though outcomes are uncertain (Hellmann et al. 2008).

Many of Earth's ecosystems are undergoing major transformations with uncertain endpoints. Ecosystem transformations can sometimes be rather abrupt, as when an ecosystem passes some tipping point or is subjected to a major disturbance and flips relatively quickly into an alternative stable state. An example is a fire-excluded pine savanna becoming increasingly less combustible as mesic hardwood trees with nonflammable leaves invade and gain dominance while grasses and other flammable ground cover diminishes. Eventually a point is reached where the community will not burn, except perhaps a small distance in from the edges or during extreme drought (Noss 2018). Alternately, a woodland may convert to a grassland after invasion by flammable nonnative grasses and an increase in fire frequency or intensity.

Various strategies have been proposed for coping with transformations of ecosystems due to climate change. One well accepted framework, called "resist-accept-direct" (RAD), recognizes three basic strategies: resist change, accept change (at some point), or try to direct or guide change in a desirable or tolerable direction (Aplet and McKinley 2017; Jackson 2021; Lynch et al. 2021). Resistance is the most common strategy applied today, as natural areas managers struggle to maintain ecosystems in their historical states, or restore them to those states, even as climate change makes that increasingly difficult. Often resistance eventually becomes futile or at least too expensive to continue over long periods of time, so managers must ultimately switch to another strategy. Thus, identifying the appropriate timeframes for adaptive responses is crucial.

Guidance for Responding to Climate Change in Natural Areas Management: Natural areas managers increasingly recognize that they not only need to consider climate change in the conservation planning process, but they must also actively invest in the implementation of climate adaptation actions. Given the conundrum of options, none of which is entirely satisfying, some **best management practices** (or at least guidance) for addressing climate-driven environmental change include the following:

- Understand that adaptation in a broad sense includes evolutionary, ecological, and social changes that are likely to reduce the vulnerability of ecosystems to climatic disruption (Moore and Schindler 2022).
- Recognize that climate change is not just a long-term, gradual threat; rather, changes in the frequency and magnitude of climatic extremes are an immediate threat (Butt et al. 2016) and major changes in disturbance regimes (e.g., fire severity) linked to climatic change may result in drastic near-term change.
- Identify and protect climate refugia, which range in spatial extent from small, localized habitats such as sinkholes, seepage areas, north-facing slopes, and edaphic communities (hypothetically) to entire landscapes with relatively stable climates due to topographic heterogeneity, proximity to moderating ocean currents, disturbance regimes (such as frequent fire) that produce resilient ecosystems, and other factors (Noss 2001; Dobrowski 2011; Keppel et al. 2012; Bátori et al. 2017; Harrison and Noss 2017).
- Avoid simplistic “solutions” to climate change, such as massive tree-planting for carbon sequestration. Afforestation of natural and seminatural grasslands is a major threat to global biodiversity (Veldman et al. 2015, 2019).
- In geophysically or geoclimatically diverse landscapes, with heterogeneous topographic and edaphic conditions, allow for opportunities for species to adjust to climate change by moving relatively short distances into newly favorable habitats (Ackerly et al. 2010; Anderson and Ferree 2010; Beier and Brost 2010; Anderson et al. 2015).

Invasive Nonnative Species Control: Most natural areas suffer to some degree from invasions by nonnative plant species and sometimes animal species. Managers of natural areas have often assumed that all nonnative species are bad and should be eliminated as soon as possible. However, many studies have found that some nonnative species play useful roles in ecosystems, often substituting for native species that have experienced population losses or have gone extinct and can actually increase native biodiversity (Davis et al. 2011). Moreover, management to eliminate invasives and restore native plants can have unintended negative consequences on rare native species of conservation concern (Buckley and Han 2014; Casazza et al. 2016).

On the other hand, nonnative species often can have devastating impacts on native biodiversity. One of the most problematic impacts stems from the effects of nonnative plants on disturbance regimes, which in turn affect the structure, composition, and function of the ecosystem in multiple ways. Exotic annual grasses not only are highly competitive with native vegetation (Humphrey and Schupp 2004), they also are often highly flammable and increase the amount and continuity of fine fuels as well as the length of time that these fuels are dry enough to burn (Knapp 1995; Davies and Nafus 2013).

Clearly there is a need for more research and monitoring of invasive species to inform adaptive management interventions. Based on existing evidence, the following are some **best management practices** for invasive nonnative species on natural areas:

- Gather evidence through research and monitoring to determine which nonnative species should be eradicated or controlled and which can potentially be left in place. This is a cost-effective strategy, as controlling invasives can be expensive.
- Remember that native species can be invasive as well, for example oaks and other hardwoods invading fire-excluded pine savannas (e.g., Brockway and Outcalt 2000).
- Be careful that restoration treatments to remove exotics and restore native plant cover do not harm native species of conservation concern.
- Monitor the effects of invasive species management to determine if expected responses of native ecosystems to management actually occur.
- Recognize that the optimal strategy for addressing nonnative plant invasions may be to develop and maintain a natural community with high ecological integrity and resistance to invasion (Sheley and Krueger-Mangold 2003).

Viability of Species of Conservation Concern: Many species of conservation concern will require species-specific management and recovery actions, but the following **best management practices** have considerable generality:

- Strive to maintain ecologically effective populations of species of conservation concern, not just minimally viable populations. Species exist in communities and ecosystems and their interactions with other species and processes will vary with their abundance.
- To simplify consideration of conservation needs and actions for large groups of species, consider clustering species according to shared ecosystem types or geophysical habitats, shared threats, or shared functional traits (Clark and Harvey 2002; Kooyman and Rossetto 2008; Noss et al. 2021).

Conclusions: Lessons for Success in the Twenty-first Century

None of the current or foreseeable future challenges to natural areas addressed in this paper are completely new. The magnitude of these challenges is, however, becoming unprecedented. Given these major threats, important lessons emerge from our research and experiences and our understanding of the values of natural areas.

First, we should not rush to discard the values and norms that mobilized the natural areas movement through the twentieth century and remain prominent today. These values all are still relevant and true. Many recent criticisms of natural areas preservation (e.g., Rohwer and Marris 2021) are caricatures of the movement. Few, if any, conservationists seek to prevent ecological change. Most conservationists would agree that evolutionary change, such as improved adaptation to changing climate, is highly desirable. Awareness of the dynamism of nature has grown, however, in concert with improvements in our understanding of disturbance ecology and observations of the impacts of climate change. This new level of awareness of environmental change and the dynamic nature of ecosystems should stimulate questions about some long-cherished assumptions about natural areas conservation, restoration, and management. For example, a long-unquestioned assumption in ecological restoration is that seeds for plantings should be locally sourced. But is this assumption still valid given knowledge of the rapidity of climate change? Or would sourcing from lower latitude populations be more defensible?

Second, as environmental change accelerates, the value of natural areas as benchmarks increases, as does their role in safeguarding biodiversity and ecological integrity. Novel ecosystems are already emerging inside and outside of natural areas, and they are not devoid of conservation value (Hobbs et al. 2009). Recognizing the conservation value of "historic, hybrid, and novel ecosystems" (Hobbs et al. 2014) is consistent with the resist change, accept change, or guide change options for addressing climate change (Aplet and McKinley 2017; Jackson 2021; Lynch et al. 2021).

Third, one major development in ecology and conservation biology in the late twentieth and early twenty-first centuries is increased recognition of landscape ecology. Large natural areas are landscapes in themselves, but they are still influenced by activities and processes in the larger landscape that surrounds them. In many regions, most natural areas are small sites embedded in human-dominated landscapes. The effects of the surrounding landscape are more profound for these small natural areas, due to edge effects, dispersal limitation, and other processes (Laurance and Yensen 1991; Murcia 1995). Natural areas managers, where possible, should work with land-use planners to improve the landscape context surrounding natural areas. Expanding the size of reserves to mitigate deleterious edge effects may be possible in some cases.

Fourth, conflicts between species-level and ecosystem-level management remain problematic today. Most natural areas managers are aware that both species and ecosystems deserve conservation attention. Because the needs of individual species sometimes conflict, managing for ecosystems seems a sensible way to reduce disputes (Noss 1996). Especially in regions with many conservation-reliant species, there are only so many species that we can conserve or manage individually without being overwhelmed. The biological status of species is usually linked directly to the condition of the ecosystems with which they are associated. Protecting and managing ecosystems is therefore a cost-efficient way to protect multiple species with shared biological needs and shared threats (Noss et al. 2021).

On the other hand, among the best indicators of the quality or integrity of ecosystems is the presence and viability of species that are characteristic of that ecosystem. Hence, species-based indices such as the Floristic Quality Index (FQI) are used to assess the quality and conservation importance of natural areas (Wilhelm 1977). Moreover, foundation species, apex predators, ecological engineers, and other strongly interacting species commonly control the structure and diversity of the ecosystem (Soulé et al. 2003, 2005); these species must be maintained in ecologically functional, not just minimally viable, populations. Some species demand individual attention because they are so highly imperiled that they would perish without it. It is inescapable that natural areas managers must attend to at least some individual species as well as to the ecosystems in which they occur.

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State of the Birds Report
United States of America
2022

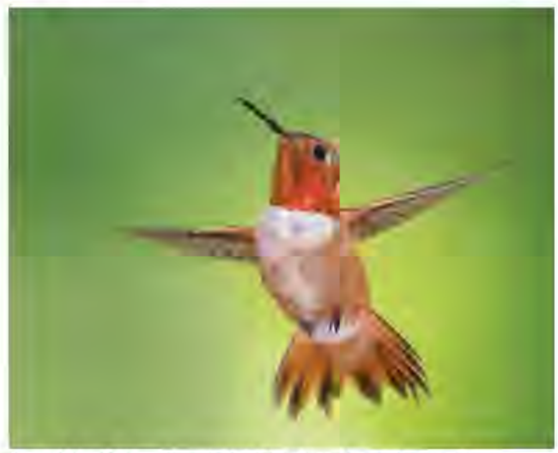
North American Bird Conservation Initiative (NABCI)
https://www.stateofthebirds.org/2022/

Sounding an Alarm About Steep Population Losses

In 2019, a study of 529 bird species with adequate long-term data for analysis (*Science*, Rosenberg et al.) found that 303 species in North America were declining—more than half of the bird species studied.

Now scientists with the Road to Recovery initiative have issued an alert for 90 declining bird species—birds that are not yet federally listed as threatened or endangered, but that have lost half or more of their breeding population since 1970. The scientists further identified a subset of 70 Tipping Point species that could lose another half or more of their populations in the next 50 years, based on recent trajectories and expert assessments.

These Tipping Point species are high priorities for science and conservation because of their high vulnerability to extinction, high urgency, and steep population declines where known. All are included on the Birds of Conservation Concern List of the U.S. Fish and Wildlife Service and/or state lists of Species of Greatest Conservation Need.



Two-thirds of Rufous Hummingbirds have been lost in the past 50 years.

COMMITTEE NOTE: Two birds on the list below (Chimney Swift and Wood Thrush) have been confirmed in White's Woods

On Alert: All of these bird species have lost half of their populations in the past 50 years

Baird's Sparrow
Black-billed Cuckoo
Black Skimmer
Black Swift
Canada Warbler
Cerulean Warbler
Clark's Grebe
Eastern Whip-poor-will
Grace's Warbler
Long-billed Dowitcher
Mourning Warbler
Olive-sided Flycatcher
Red-headed Woodpecker
Rock Sandpiper
Snowy Owl
Surfbird
Thick-billed Longspur
Western Grebe
Wilson's Plover
Wood Thrush

Allen's Hummingbird
American Golden-Plover
Ashy Storm-Petrel*
Audubon's Shearwater*
Bachman's Sparrow
Band-rumped Storm-Petrel*
Bendire's Thrasher
Bicknell's Thrush*
Black-capped Petrel*
Black-chinned Sparrow
Black-footed Albatross*
Black-vented Shearwater*
Black Rail*
Black Rosy-Finch*
Black Scoter
Bobolink
Bristle-thighed Curlew*
Brown-capped Rosy-Finch*

Buff-breasted Sandpiper
Cassia Crossbill*
Chestnut-collared Longspur
Chimney Swift
Craveri's Murrelet*
Elegant Tern*
Evening Grosbeak
Fea's Petrel*
Golden-winged Warbler
Great Black-backed Gull
Greater Sage-Grouse
Guadalupe Murrelet*
Harris's Sparrow
Heermann's Gull*
Henslow's Sparrow
Hudsonian Godwit
Ivory Gull*
King Eider

King Rail
Kittlitz's Murrelet*
Laysan Albatross*
Least Tern
LeConte's Sparrow
LeConte's Thrasher
Lesser Prairie-Chicken*
Lesser Yellowlegs
Mottled Duck
Mountain Plover
Murphy's Petrel*
Parkinson's Petrel*
Pectoral Sandpiper
Pinyon Jay
Prairie Warbler
Red-faced Cormorant
Red-legged Kittiwake*
Ruddy Turnstone

Rufous Hummingbird
Saltmarsh Sparrow*
Scripps's Murrelet*
Seaside Sparrow*
Semipalmated Sandpiper
Short-billed Dowitcher
Sprague's Pipit
Stilt Sandpiper
Townsend's Storm-Petrel*
Tricolored Blackbird*
Wandering Tattler
Whimbrel
Whiskered Auklet*
Yellow-billed Loon
Yellow-billed Magpie
Yellow Rail*

These Tipping Point species are on a trajectory to lose another 50% of their remnant populations in the next 50 years if nothing changes.

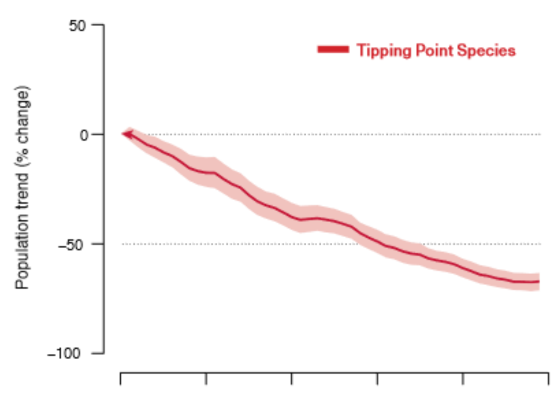
These 90 bird species lost 50% or more of their populations during 1970–2019. The Tipping Point species are on a trajectory to lose another 50% of their populations in the next 50 years (39 species), or already have perilously small populations and continue to face high threats, but lack sufficient monitoring data (31 species, indicated with an asterisk). For the USFWS Birds of Conservation Concern list, visit fws.gov/media/birds-conservation-concern-2021pdf.

The Next Set of Species Plummeting Toward Endangered Status

Of the 1,093 bird species protected under the Migratory Bird Treaty Act, 89 birds have received additional protections as either threatened or endangered under the U.S. Endangered Species Act to prevent their extinction.

The Tipping Point species represent another 70 birds that could be next to face threatened or endangered status. Cumulatively, the Tipping Point species that have sufficient data for monitoring have lost more than two-thirds of their populations in the past 50 years.

Tipping Point species come from varied habitats, but they all have the same urgency—immediate science and conservation actions are needed to turn around declines.



70 Tipping Point Species

Urgent action is needed to help these birds before they become endangered.

GRASSLAND BIRDS
Prairie specialists such as Mountain Plover, Sprague's Pipit, and Chestnut-collared Longspur have lost more than 75% of their populations since 1970.

FOREST BIRDS
The steepest declines among forest birds include species that specialize on tree seeds (e.g., Pinyon Jay), long-distance migrants (e.g., Bicknell's Thrush), and aerial insect-eating specialists (e.g., Chimney Swift).

ARCTIC & ALPINE TUNDRA
Shrinking habitats due to changing climate and resource extraction threaten Arctic-breeding shorebirds (e.g., Hudsonian Godwit and Whimbrel) and alpine birds (e.g., Black Rosy-Finch).

COASTAL & OCEANIC
Nearly a quarter of seabirds

ARIDLAND BIRDS
Aridland birds are experiencing

WETLAND BIRDS
Though many waterbirds benefited

Agricultural Sciences

Pennsylvania Forests Changing From Red Oak To Red Maple Dominated

March 26, 2003

UNIVERSITY PARK, Pa. -- Whether they blame Smoky Bear, acid rain or white-tailed deer, experts in Penn State's College of Agricultural Sciences agree that the species composition of forests in Pennsylvania is changing and warn that economically important species such as red oaks are not regenerating at historic levels.

Scientists may debate the reasons for forest change, but it now appears that they might all be right.

"The decline of oak in our forests is a big story," says Marc Abrams, a professor of forest ecology and physiology who has been honored several times in recent years for outstanding research on systematic change in Eastern forests. "The change actually started in the early 1900s when forest fires first were suppressed."

Abrams tracks a fascinating trend over the last century when red maple -- a tree species that originated in swampy habitats -- started taking over eastern forests. "Originally, because of its sensitivity to fire, red maple was relegated to the swamps," explains Abrams. "In fact, it used to be called swamp maple. But now that we suppress forest fires, red maple has emerged from the swamps and taken over upland sites, and can be found on just about every landscape in the eastern deciduous forest. This change in our forests may have profound economic and ecological consequences.

"Forest regeneration is a huge concern," adds Abrams. "Trees that historically dominated this region -- the pines, oaks, hemlocks and hickories -- no longer regenerate very well. Red maple is replacing trees that have high economic value. Its soft wood, color and grain aren't as highly valued as that of black cherry and oak. Also, many wildlife species depend on the trees that are being replaced."

Forest hydrology professor Bill Sharpe -- who has chronicled the effects of acid rain in Pennsylvania for several decades -- also has watched red oaks decline and red maples become predominant. But his explanation for the trend is a bit different. He maintains that soils in many places have become too acidic to support adequate growth of red oak.

According to Sharpe, Pennsylvania's forest soils for many decades have been absorbing acidic precipitation originating in the Ohio Valley -- the greatest industrial complex in the world. "The acid comes from sulfur dioxide in the emissions from coal-fired generating plants in Ohio, Indiana, Illinois, West Virginia and western Pennsylvania. Our forests long have been the victim of the most acidic precipitation in North America and our data show that forests soils are much more acidic now than they were 40 to 50 years ago.

"The acid deposition leaches aluminum out of the soils, which is toxic to plants, and also lowers the availability of calcium and magnesium, both essential elements for plant growth," Sharpe says. "We have a forest regeneration problem and a forest health problem -- our forests are sick. We know there is very little regeneration of red oak and large, mature red oaks are dying. That cannot be blamed on deer or the lack of fire."

Sharpe has completed several research projects that suggest soil acidification may be responsible for the rising fortunes of red maple. In a simulated deer browsing study, red maple grew better after simulated browsing than red oak, and in plant bioassays red maple was much less sensitive to aluminum and low calcium than red oak. In deer studies done at Penn State in the 1970s, deer actually preferred to browse red maple over red oak, so Sharpe does not subscribe to the deer hypothesis. "We can do something now and that is to demand tighter emissions controls, including controls on tail pipe emissions," insists Sharpe. "We also should lime areas to be harvested where regeneration is problematic."

Abrams believes a shift in wildlife populations is likely to parallel this shift in tree species. Oaks and hickories supply many small mammals and birds with nuts and acorns. And the oak's rough bark -- unlike the maple's smooth bark -- houses bark-dwelling insects for insect-eating birds. Red maple's proliferation also poses a biodiversity concern, he points out.

"Very diverse forests -- with six to 12 different species in the overstory -- may be changing to red maple-dominated stands," he says. "And stands of single species are more susceptible to total devastation by insects and disease."

Abrams believes many forests can be managed with controlled burns on a case-by-case basis. "In many instances," he says, "a controlled understory fire is highly realistic and will go a long way in encouraging oak regeneration and retarding further development of red maple."

Selective browsing by an overpopulation of white-tailed deer also has been widely blamed for a lack of forest regeneration. Deer damage has caused an increase in species such as hay-scented ferns, which compete successfully with tree seedlings. Some experts, such as wildlife resources professor Gary San Julian, say it seems clear that where deer are most numerous, seedlings are devoured before they can grow out of reach of voracious herds of whitetails.

"In many areas of Pennsylvania in the last 40 years or so, deer numbers and densities significantly altered habitat," says San Julian, who has done research in wildlife damage management. "In some areas, very few deer -- because of past heavy browsing -- can greatly affect regeneration."

But, San Julian concedes, it is likely that several factors are contributing to the change in forest composition in Pennsylvania. "All three theories about why young oaks have become scarce in our forests have merit," he says. "It may be that decades of fire suppression, acid rain and deer damage have all combined to create an environment that is not favorable to red oaks and a few other desirable tree species."

The subjects of deer damage to forests and deer management have proven to be extremely controversial in Pennsylvania, to say the least, but San Julian is philosophical about the debate surrounding the state's efforts to decrease deer numbers for the benefit of the forests.

"What if the cause of poor regeneration of oaks and other economically important tree species in our forests is a combination of three factors, as we suspect?" he asks. "What if it is caused by fire suppression, acid rain and too many deer? What can we do?

"We no longer can just let wild fires burn -- there are too many people and too many valuable properties to take that kind of risk. And there is little that we in Pennsylvania can do about acid rain in the short term. The kind of pollution abatement needed will take a national effort, and the results won't be felt for years. But we can bring deer numbers into harmony with the habitat. We can do that much right now for the health of our forests."

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- Agricultural Sciences

The HawkEye
IT'S YOUR MEDIA

Hunting in White's Woods? Bad idea

Posted on March 4, 2024 by David Loomis

Photo: David Loomis

Reported opinion

By Willard Radell

WHITE TOWNSHIP — Dana Milbank's emotional plea to eliminate the deer-browsing menace through ramped-up hunting ignores some complicating facts about hunting in selected zones in suburban areas. Once these facts are faced, hunting in suburban areas for ecological reasons is revealed to be an ineffective deer management side-trip that makes people feel like something is being done, with little or no improvement in the problem that motivated it.

Milbank blames the "ecological bullies" for eating up the forest understory and worries that deer density in his part of Virginia's Rappahannock County with 40-50 deer per square mile is an ecological disaster for the 27 humans per square mile who live there.

It could be asked of Milbank, How's the understory doing on the spacious, herbicide-sotted lawns and savannahs of suburban and exurban northern Virginia? The ecological damage of those 27 humans per square mile probably dwarfs the damage of those 40 deer claimed to be in his square mile by orders of magnitude.

In The Washington Post version of Milbank's column, he invites a "walk into the forest …. past the edge between field and woods where invasive vines now dominate, and you will find a manicured scene: all mature trees and no understory – none of the seedlings, saplings, flowers and shrubs that once covered the forest floor."

The same could be said of the typical, sprawling, suburban human homestead: A walk in the former forest reveals that it is gone, replaced by McMansions, pools, sprawling lawns and savannahs, supported by ample applications of herbicide to prevent incursion by both invasive and native plants that might disturb the green uniformity of the grass.

Milbank seems to believe that when he kills his deer as a novice gun owner, "I'll be donating what I don't eat (for 'locavores,' there's no food more local than a deer consumed on the very land where it lived) to Hunters for the Hungry, a nonprofit that processes and donates venison." I wonder if Milbank has thought this through. The .30-06 caliber he plans to use easily has a range of 1,000 yards. So for his neighbors' sake, I hope he's learned enough in his few hours at a gun range that he's good enough to take only clean shots.

Also, as Milbank is a brand-new hunter, I'm a little concerned that he thinks he's going to field dress the deer, butcher it, eat what he wants, and take the rest to Hunters for the Hungry. Like cooking chitterlings in the South, better learned from Grandma than a cookbook, field dressing and butchering safely is best learned from an experienced hunter. Hunters for the Hungry will not accept deer carcasses unless they are properly field dressed and not at all if they have been partially butchered.

THE HAWKEYE apparently reprinted excerpts from the Milbank article on Bambi, an ecological bully, because deer hunting in White's Woods Nature Center is a hot issue in the local community. Many people see the damage done by deer and assume that anything that culls deer must make matters better. But the evidence is murky on that. There are several practical problems with deer hunting in White's Woods that are worth considering.

White's Woods Recreation Area (aka White's Woods Nature Center). White Township map. Click to enlarge.

White's Woods has 250 acres arranged in a Y-shape. The south and west segments are about 450 and 500 yards wide; the north segment is about 650 yards wide. Assuming the White's Woods hunters are on the high ground near the center of each arm, that gives hunters only 225 to 325 yards to work with. The range of a hunting rifle is over 1,000 yards. Since the hunters will be shooting down-slope, missed shots will easily go 3,000 to 4,000 yards into private property outside the nature center.

BOW HUNTING in White's Woods lessens the range problems of rifle hunting but aggravates other problems. Since White's Woods is on high ground and wounded deer tend to run downhill, you can expect a number of wounded deer to be running into and through private property in White Township and Indiana Borough before they finally die.

The problem of deer struck by arrows taking a long time to die is well enough known in the bow hunting community that you can read articles on the importance of good blood trailing skills to recover the deer you shot "1000 yards back." Maiming with bow hunting is a known problem. Some studies have shown that for every deer killed there is another unrecovered wounded deer.

There is also the problem of unrecovered arrows. Bow hunters take many shots per deer killed and the result is unrecovered arrows in the woods where they can be stepped on by walkers and dogs.

There would be a problem of field dressing in White's Woods. The entrails would be left to rot to be explored by dogs. That would surely compromise the enjoyment of the park for the many and the blood trails and gut piles from field dressing are likely media for the spread of deer wasting disease prions and multiplication of flies and gnats.

There isn't much solid evidence that deer hunting sustainably lowers the deer population in an open bio-system. After the hunt, better fed deer survivors bear more fawns and more of the fawns survive to reproduce, while other deer move in to take advantage of the now more abundant food.

Unless the deer management area is an island or peninsula, the population rebounds and we are dependent on annual hunting to harvest the surplus population without reducing the deer density enough to significantly impact deer browse pressure.

When a wounded deer refugee from White's Woods Nature Center ends up dying at a private residence, who do you call – state police? borough police? the sheriff? White Township supervisors? game commission? What's the plan?

DANA MILBANK and proponents of a hunt in White's Woods Nature Center have good intentions but have not fully explored the practical challenges. Stray bullets and arrows, wounded animals, small hunting space surrounded by "no hunting" residential areas, population rebound, immigration from the many herds whose ranges abut and overlap White's Woods, the narrow, irregular configuration of White's Woods Nature Center, and the topography with high ground at the center are all factors that show hunting in White's Woods to be unable to solve the problem it's proposed to solve.

Sign at North 12th Street entrance to White's Woods Recreational Area, White Township, Pa. Photo: David Loomis.

HawkEye readers should think unemotionally about the issue of deer hunting in White's Woods Nature Center. Yes, deer are eating our stuff. But the key issue is, will hunting on half of a square mile in the middle of a 20-square-mile range with overlapping herds make a dent in the deer density? And will it be worth the costs imposed on WWNC users and nearby property owners?

Intensive hunting and culling can decrease deer browse pressure and other deer-related problems — in a closed bio-system like a peninsula or island. If it were feasible to fence in White's Woods Nature Center and either kill all the deer inside or have an annual deer drive to push any stragglers out, then the deer browse problem would cease in the Nature Center.

But WWNC is not a closed system. It is surrounded by multiple deer herds for many miles. Dana Milbank and White Township decision-makers need to realize that hunting/culling in places like WWNC will not produce the good results wanted. Ineffective vengeance against large, hooved, vegetarian rats in an open bio-system is folly.

White Township supervisors have postponed a decision on hunting in White's Woods. They should extend the postponement indefinitely.

Willard Radell is a resident of White Township.

Editor's note:

Public comments on White Township's plan for White's Woods are invited through March 22, 4:30 p.m. Comments may be submitted to the township by:

— email address: wtinfo@whitetownship.org

—postal mail address: White Township, 950 Indian Springs Road, Indiana, PA 15701

— an online form on the township's website

— David Loomis

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ORIGINAL RESEARCH

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Red oak seedlings as indicators of deer browse pressure: Gauging the outcome of different white-tailed deer management approaches

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Abstract

After decades of high deer populations, North American forests have lost much of their previous biodiversity. Any landscape-level recovery requires substantial reductions in deer herds, but modern societies and wildlife management agencies appear unable to devise appropriate solutions to this chronic ecological and human health crisis. We evaluated the effectiveness of fertility control and hunting in reducing deer impacts at Cornell University. We estimated spring deer populations and planted *Quercus rubra* seedlings to assess deer browse pressure, rodent attack, and other factors compromising seedling performance. Oak seedlings protected in cages grew well, but deer annually browsed ≥60% of unprotected seedlings. Despite female sterilization rates of >90%, the deer population remained stable. Neither sterilization nor recreational hunting reduced deer browse rates and neither appears able to achieve reductions in deer populations or their impacts. We eliminated deer sterilization and recreational hunting in a core management area in favor of allowing volunteer archers to shoot deer over bait, including at night. This resulted in a substantial reduction in the deer population and a linear decline in browse rates as a function of spring deer abundance. Public trust stewardship of North American landscapes will require a fundamental overhaul in deer management to provide for a brighter future, and oak seedlings may be a promising metric to assess success. These changes will require intense public debate and may require new approaches such as regulated commercial hunting, natural dispersal, or intentional release of important deer predators (e.g., wolves and mountain lions). Such drastic changes in deer management will be highly controversial, and at present, likely difficult to implement in North America. However, the future of our forest ecosystems and their associated biodiversity will depend on evidence to guide change in landscape management and stewardship.

KEYWORDS

deer management, forest regeneration, oak browse rate, *Odocoileus virginianus*, *Quercus rubra*, recreational hunting

1 | INTRODUCTION

Temperate forests in eastern North America face a crisis due to accelerated development, climate change, and introduced pests and diseases (Aukema et al., 2010; Liebhold et al., 2013). In addition, high populations of white-tailed deer (*Odocoileus virginianus*, Figure 1) cause dramatic and wholesale changes in habitats across much of North America, that threaten the continent's biodiversity, economies, and human health (Côté, Rooney, Tremblay, Dussault, & Waller, 2004). This once iconic species has turned into an ecological villain and human health threat, yet modern societies struggle to find appropriate responses (Sterba, 2012).

Overexploitation nearly led to extinction of white-tailed deer in the late 1800s. However with changes in hunting regulations and establishment of state wildlife agencies to manage recovery of the species in the early 1900s, deer herds rebounded quickly (Halls, 1984). Population recovery was aided by subsidies from human activities (agriculture) and the regrowth of eastern forests. Early dire warnings about long-term ecological consequences of deer population increases in the absence of traditional predators, such as mountain lions (*Puma concolor*) and timber wolves (*Canis lupus*; Leopold, Sowls, & Spencer, 1947) were ignored by state wildlife agencies. Today, scientific evidence regarding negative impacts of historically high white-tailed deer populations is voluminous, increasing, and largely uncontested.

White-tailed deer are ruminant browsers with a variable diet composed of woody species, herbs, grasses, and mushrooms. Diet composition is influenced by geography, season, habitat features, primary human land uses, deer abundance, legacy effects, and plant community composition (Anthony & Smith, 1974; Arceo, Mandujano, Gallina, & Perez-Jimenez, 2005; Daigle, Crete, Lesage, Ouellet, & Huot, 2004; Johnson et al., 1995; Nixon, Hansen, Brewer, & Chelsvig, 1991; Ramirez, Quintanilla, & Aranda, 1997; Royo, Kramer, Miller, Nibbelink, & Stout, 2017). Deer make daily feeding decisions based on their seasonal nutritional needs, individual preferences, nutritional value and defense chemistry of forage species, and presence/absence of predators (Berteaux, Crete, Huot, Maltais, & Ouellet, 1998; Cherry, Warren, & Conner, 2017; Hanley, 1997; Lavelle et al., 2015; Masse & Cote, 2009). Differences in nutritional value and palatability among plant species lead to distinct feeding preferences. Although deer can adapt as food quality declines due to selective removal of the most desirable species, resulting in smaller deer with reduced body size (Simard, Cote, Weladji, & Huot, 2008). Deer continue to seek out strongly preferred plant species, even if they occur at low densities, further increasing threats of local extinction for particularly vulnerable populations (Erickson et al., 2017).

Long-term consequences of high deer populations have been documented for herbaceous and woody species alike. The impact of deer browse on herbaceous species may result in direct mortality, but tissue removal preventing flowering and reproduction has dramatic demographic consequences that play out on a decadal time scale. For example, high deer populations caused declines of >90% for many orchids in the mid-Atlantic region in Maryland (Knapp & Wiegand, 2014). Deer browsing also threatens understory herbs like Trilliums (*Trillium grandiflorum* and *T. erectum*) and American ginseng (*Panax quinquefolius*; Bialic-Murphy, Brouwer, & Kalisz, 2019; Dávalos, Nuzzo, & Blossey, 2014, 2015a; Knight, Caswell, & Kalisz, 2009; McGraw & Furedi, 2005), however, these are only a few well-researched examples, and threats are widespread (Frerker, Sabo, & Waller, 2014). In contrast to herbaceous species that experience deer browsing without reprieve, most woody plants have the ability of vertical escape once terminal shoots grow out of browse height (1.5–2 m). However, current deer densities across much of eastern North America prevent transition from seedlings (<1 year old; up to 20 cm tall) to saplings (Kelly, 2019; Long, Brose, & Horsley, 2012; Miller & McGill, 2019). Despite abundant seed production by mature overstory trees and successful germination, deer browsing is now so extensive that forest regeneration after harvests or natural mortality is largely prevented, creating a regeneration debt (Miller & McGill, 2019) that plays out over centennial time scales and affects not just the highly palatable species. High deer browse pressure not only creates less diverse forests that will exist long into the future, but it also prevents dispersal of many tree species northward in response to climate change, which in turn has large economic consequences for timber management (Côté et al., 2004), and limits potential for climate change mitigation through reforestation (Bastin et al., 2019).

High deer populations and their impact on primary producer diversity and abundance led to dramatic abundance declines in forest macrolepidoptera specialized on understory plant species in New Jersey (Schweitzer, Garris, McBride, & Smith, 2014). In Pennsylvania, aboveground insect abundance, richness, and diversity were up to 50% higher where deer were excluded for 60 years (Chips et al., 2015). Furthermore, deer facilitate spread of invasive plants and

FIGURE 1 White-tailed deer female (yellow ear tag and VHF collar) and male in velvet (blue ear tags) on the Cornell campus in summer 2009 (photos by B. Blossey)

invasive earthworms (Dávalos, Nuzzo, & Blossey, 2015b; Dávalos, Simpson, Nuzzo, & Blossey, 2015; Eschtruth & Battles, 2009; Kalisz, Spigler, & Horvitz, 2014; Shelton, Henning, Schultz, & Clay, 2014), which individually and collectively have far reaching consequences on soils, erosion, nutrient cycling, and food webs (Maerz, Nuzzo, & Blossey, 2009; Nuzzo, Maerz, & Blossey, 2009). In summary, elevated deer densities create depauperate landscapes, and the resulting successional forest trajectories have long-lasting (>100 years) legacy effects that negatively affect all trophic levels including migratory birds (Bressette, Beck, & Beauchamp, 2012; Martin, Arcese, & Scheerder, 2011; Nuttle, Ristau, & Royo, 2014; Nuttle, Yerger, Stoleson, & Ristau, 2011). High deer populations also represent a human health threat due to deer-vehicle collisions and amplification of tick populations and prevalence of tick-borne diseases including Lyme (Kilpatrick, LaBonte, & Stafford, 2014; Raizman, Holland, & Shukle, 2013).

In the US, legal authority to manage deer and other wildlife as a public trust resource (except for endangered or migratory species) rests with state wildlife agencies, which follow the North American model of wildlife management, with hunting and trapping as core management tools (Geist, Mahoney, & Organ, 2001; Hare & Blossey, 2014; NYSDEC, 2011). However, the assertion that recreational hunting as currently implemented and regulated can achieve deer population regulation has been challenged (Williams, DeNicola, Almendinger, & Maddock, 2013). Further complications arise from strong opposition to hunting and lethal deer management by animal rights groups, particularly in suburbia (Sterba, 2012).

We used simultaneous experimental implementation of different deer management approaches (no management, sterilization, and recreational hunting) to assess competing claims by wildlife agencies (recreational hunting is able to control deer populations and their impacts) and animal rights activists (nonlethal control can reduce deer populations, and deer do not drive ecosystem deterioration). We know of no other study that simultaneously assessed effects of different deer management approaches for their effect on the size of a free-roaming deer population and the impact on ecological resources. We used browse incidence and seedling growth of a bio-indicator, red oak (*Quercus rubra*) to assess outcomes of different deer management approaches. The species is widespread in eastern North America, an important timber species, a major source of food for wildlife, and a species of intermediate preference for deer (Averill, Mortensen, Smithwick, & Post, 2016; McShea et al., 2007; Tallamy & Shropshire, 2009). In addition, *Q. rubra*, like other oak species, shows regional regeneration failures in eastern North America (Abrams & Johnson, 2012), but the species is flourishing when deer numbers are kept low, for example on tribal lands (Reo & Karl, 2010). We chose to focus on browse frequency and growth as the important variables determining the likelihood of seedlings to advance to the sapling stage in woody plant recruitment (Kelly, 2019). We included rodent attack, insect herbivory, and the role of competing vegetation into our assessments (a more complete justification for our approach is detailed in Section 2.3) due to their potential influence on oak recruitment and demography (Crow, 1988; Davis, Tyler, & Mahall, 2011). We evaluated the following hypotheses:

1. Deer browse intensity on red oak seedlings will vary in different management zones. Specifically, we expected browse rates to be highest in the no management zone, be intermediate in the sterilization zone, and be lowest in areas with recreational hunting.
2. The proportion of oak seedlings browsed by deer will be higher than the proportion of oaks affected by other factors (rodents, insects, and winter mortality).
3. Oaks protected from deer herbivory will grow, while height of oaks exposed to deer herbivory under the same forest conditions will regress or remain stable.
4. Browse intensity on red oak seedlings is a function of the deer population size.

2 | MATERIALS AND METHODS

2.1 | Study area and deer population estimation

Our study area was located in central New York State, USA, and incorporated major portions of the Cornell University campus and surrounding areas in the Towns of Ithaca and Dryden (Figure 2). Historically, hunting, as regulated by the New York State Department of Environmental Conservation (NYSDEC), has occurred on Cornell University lands for decades. Lack of success in reducing deer populations and their associated impacts resulted in the establishment of an Integrated Deer Research and Management (IDRM) Program in 2007 (Boulanger, Curtis, & Blossey, 2014). The goal of this program was to reduce deer populations, human health threats, and ecological and economic deer impacts by 75% over a 10-year time frame. Core elements of IDRM were coordination of deer management efforts, surgical sterilization, a recreational hunting program, monitoring of deer abundance on core campus, and assessment of ecological health using bio-indicators.

We initially established three zones with different deer management approaches: (1) no management (approx. 281 ha) where neither sterilization nor hunting was permitted; (2) sterilization (approx. 446 ha); and (3) a hunting zone (approx. 1,600 ha) where recreational hunting (bows, crossbows, and firearms) occurred in accordance with local and state laws (Boulanger et al., 2014). These three zones did not overlap but were adjacent to each other, each representing a mix of suburban, residential and rural agricultural and forested lands (Figure 1).

Obtaining accurate estimates of abundance for free-ranging deer is notoriously difficult and cost prohibitive, particularly over large areas. Traditional survey methods have included track or pellet counts, spotlight surveys, drive counts, aerial or thermal imagery surveys, or population reconstruction based on hunter reports and sex ratios. However, all of these methods produce unreliable results, and some may only be available in open habitats (Fritzen, Labisky, Easton, & Kilgo, 1995; Goode et al., 2014; Keever et al.,

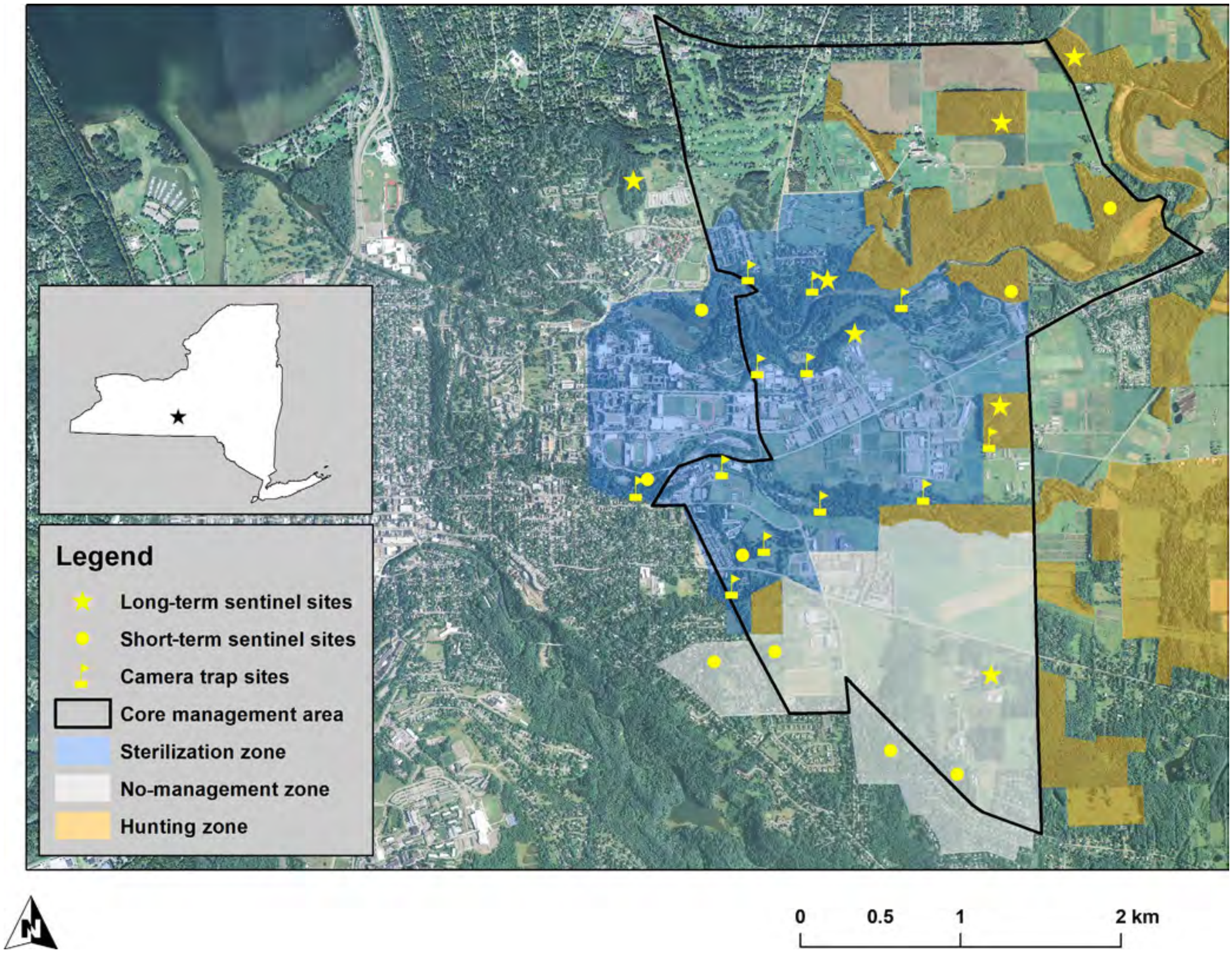

FIGURE 2 Delineation of no management, sterilization, and hunting zones (2008–2013) and core deer management area (after 2013) surrounding the main Cornell University campus in Ithaca, New York, USA. Short-term (2010 and 2011) and long-term (2010–2015) *Q. rubra* planting and camera trap locations are indicated by yellow markers

2017; Marques et al., 2001; Norton, Diefenbach, Wallingford, & Rosenberry, 2012). Lately, use of camera traps has become popular. However, accurate population estimation still requires identification of individuals, and individual deer are impossible to distinguish, except for branch-antlered male deer (hereafter bucks) in the fall. Furthermore, density estimates are influenced by detection probabilities that vary seasonally and with terrain, human development, and hunting pressure (Parsons et al., 2017). The development of genetic tools using DNA extracted from pellet groups to estimate deer density and spatially explicit habitat use shows great promise (Brinkman, Person, Chapin, Smith, & Hundertmark, 2011), but costs associated with sample processing make this still cost prohibitive in most circumstances (Goode et al., 2014).

To obtain accurate deer population estimates to quantify responses to our management activities, we utilized a cohort of 120 individually marked deer. We captured and sedated deer in the sterilization zone (Figure 2), and veterinary surgeons performed tubal ligations and ovariectomies (Boulanger & Curtis, 2016). We captured most of the 120 deer in the first two years of the program, but continued to target immigrating individuals to maintain a high sterilization rate. We fitted captured deer with individually numbered livestock ear tags (Premier1 Supplies) and fitted most sterilized adult females with very high-frequency (VHF) radio collars (Telonics, Inc.; Figure 1). We released all deer at their original capture location and monitored their movements, which varied widely among individuals (Figure 3). We then conducted an annual camera census (mark-recapture study) in the sterilization zone each spring using 12 digital infrared-triggered cameras that took pictures at bait stations continuously for 5–7 days. Our population estimation thus occurred at a time when potential behavioral responses to fall hunting pressure and spatial escape of deer into the sterilization or no-hunting zones would have been minimal. We placed cameras in a grid system comprised of 40-ha blocks (Figure 1) and calibrated them to take a photograph every four minutes, if deer were present at bait. We tallied photographs and then modeled deer abundance using programs MARK and NOREMARK (Curtis, Boldgiv, Mattison, & Boulanger, 2009; White, 1996). An initial test of this approach obtained accurate and precise estimates of deer abundance (Curtis et al., 2009).

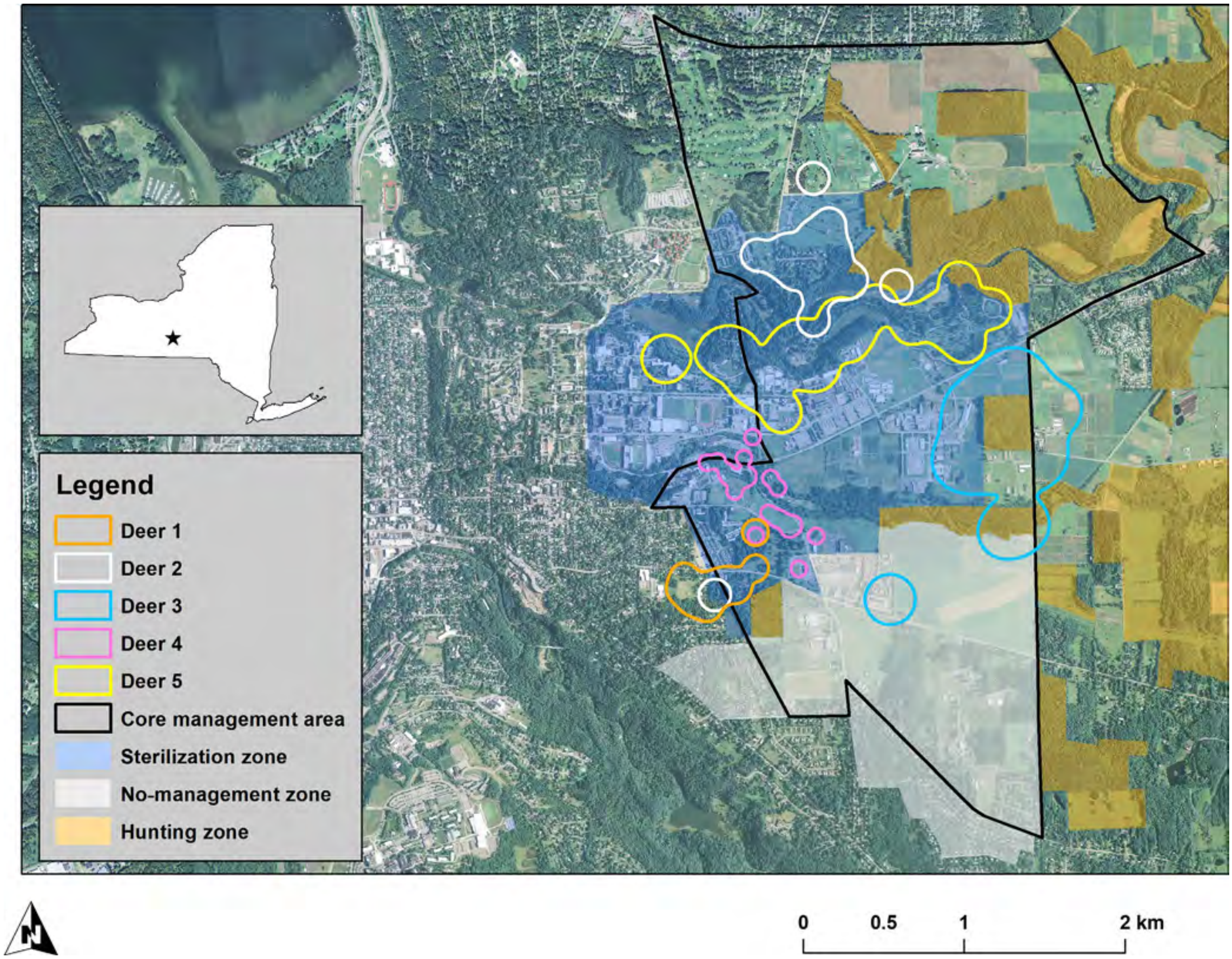

FIGURE 3 A sample of variation in shape and size of 95% adaptive kernel home range estimates for surgically sterilized radio-collared adult female deer on Cornell campus (2008–2013; adapted from Boulanger et al., 2014)

2.2 | Deer management

In addition to continuing sterilization efforts of deer immigrating into our sterilization zone, we established a coordinated recreational hunting program in accordance with New York State hunting seasons each fall from October to December. For safety reasons, we restricted hunting close to campus or suburban neighborhoods to archery, but elsewhere allowed shotguns and/or muzzleloaders. We experimented with various approaches to increase antlerless harvests by the >500 recreational hunters who annually registered for the Cornell University Hunting Program. These included Earn-A-Buck approaches (hunters were required to shoot a female before they can shoot a buck), and use of Deer Management Assistant Permits (additional nonantlered tags) issued by the NYSDEC. Beginning with the 2012 season, the NYSDEC established a special Deer Management Focus Area that allowed harvest of two antlerless deer per hunter per day through the regular hunting season and added a unique 3-week antlerless season in January that included our core management area (Boulanger et al., 2014) to assist in deer management efforts.

Despite hundreds of deer taken by hunters on Cornell lands and doe sterilization rates of >90%, our camera surveys indicated that by 2012, five years into the program, we had not achieved any reduction in the core deer population (Boulanger & Curtis, 2016). In response to our failure to reduce the deer population, we eliminated sterilization efforts and established a larger core management area (CMA, approx. 953 ha) that included most of the sterilization zone plus selected areas previously designated as no management or hunting zones (Figure 1). In 2013 and 2014, we allowed recreational archery hunting in designated areas of the CMA during the hunting seasons and added use of Deer Damage Permits (DDPs) as permitted by NYSDEC. Use of DDPs allowed use of bait (typically maize [*Zea mays*]) and shooting at night using artificial lights, both of which are otherwise illegal in New York State, from the end of the regular season in December to the end of March the following year. We allowed use of bows and crossbows with no tag limits placed on volunteer participants. Each participant was further required to report their efforts (hours in stand), the fate of every arrow shot, distance lethally wounded deer travelled, wounding rates, and other observations. This allowed us to make adjustments in the program as needed and be accountable to hunters, the state management agency, university administration as well as those questioning methods and security of our approach. In 2015, we eliminated all recreational hunting in our CMA and focused exclusively on volunteer archers using DDPs to limit behavioral changes in deer exposed to hunting pressure (Williams, DeNicola, & Ortega, 2008). Our highly structured DDP program restricts shooting at bait locations to no more than once per week (or less) in an attempt to limit deer behavioral changes while increasing our ability to achieve management goals. Recreational hunting has continued outside of the core management area. In addition, two adjacent villages (Cayuga Heights and the Village of Lansing) use their own DDPs to remove deer, while the City of Ithaca has a discharge ordinance that prohibits the ability to take deer within City limits.

2.3 | Indicator selection, *Q. rubra* natural history, seedling performance, and procedures

Ideally, any comprehensive measurement of the status of forest biodiversity should include multiple metrics or indicators at different trophic levels; however, there are currently no agreed upon or sensitive metrics available. While desirable, it is typically impossible to measure many different variables in different trophic levels when assessing outcomes of human activities, including landscape or deer management, effects of pollution, etc. However, applied ecology has a long history of using indicator species (Bachand et al., 2014; Dale & Beyeler, 2001) to better gauge the outcome of management interventions. Using an indicator species, or a restricted portfolio of indicators, would also facilitate adoption of metrics by land managers who do not have the resources nor expertise that typically are required in scientific experiments. For the purpose of assessing differences in outcomes of alternative deer management approaches, an indicator should be sensitive to changes in deer browse pressure, for example due to fencing or culling.

We selected *Q. rubra* as our bio-indicator to assess the impact of different deer management approaches or changes in deer abundance on ecological health. In a previous study (Blossey, Dávalos, & Nuzzo, 2017), we demonstrated the utility and sensitivity of *Q. rubra* to respond to changes in deer browse pressure (fencing) through improved growth. We chose *Q. rubra* for multiple reasons, including its potential to serve as a general indicator of forest health that can be planted with reasonable expertise at low cost. This allows communities or individual landowners to assess whether their selected deer management approaches result in improvements in the ability to regenerate a diverse forest that includes *Q. rubra*. Many different oaks, including *Q. rubra* have shown persistent regeneration failures in the Northeast for decades, and various factors including lack of fire, too much shade, and high deer browse pressure are implicated (Abrams, 2003; Abrams & Johnson, 2012). These regeneration failures, as in many other woody species, occur despite abundant mature oak trees that mast frequently followed by successful acorn germination. However, seedlings are unable to advance to the sapling stage, a pattern that can be reversed through fencing, suggesting that deer play an important role in preventing this transition (Abrams & Johnson, 2012; Leonardsson, Lof, & Gotmark, 2015; Long et al., 2012; Long, Pendergast, & Carson, 2007; Schwartz & Demchik, 2015; Thomas-Van Gundy, Rentch, Adams, & Carson, 2014). These patterns suggested that selecting *Q. rubra* was an appropriate and sensitive indicator for assessing the outcome of our different deer management approaches. Changes in browse frequency for *Q. rubra*, while not expected to be identical for other species, should indicate the direction of overall browsing pressure experienced by other taxa.

Quercus rubra is a widely distributed deciduous tree in eastern North America ranging from Ontario and Quebec south to Georgia and Alabama in the east, and from Minnesota and Iowa south to eastern Oklahoma, with isolated populations in Louisiana (USDA NRCS, 2017). Mature trees are typically 20–30 m tall, start to produce acorns at age 30–40, and may live for up to 500 years. Wood of *Q. rubra* is widely used to make furniture, veneer, cabinets, and flooring. Due to its vibrant fall foliage and qualities as a shade tree, *Q. rubra* was widely planted as an ornamental. Acorns need 2 years to mature, require cold stratification after dropping off the tree, and all surviving acorns germinate in the following spring. There is no seed bank. Mass fruiting occurs every 2–5 years. Acorns may be consumed by insects, many mammals, and birds. Successful seedling recruitment is episodic and often only occurs after mass-fruiting events due to insect attack and acorn predation, particularly by rodents (Crow, 1988). Depending on site conditions, young trees may need to spend many years, or even decades, in the forest understory before gap creation due to natural mortality or harvesting of overstory trees creates opportunities to enter the overstory.

For *Q. rubra*, germination and seedling establishment is possible on many different soils, and in full or partial shade. Seedling and sapling densities of 1,000–2,500 stems/ha are required to ensure sufficient regeneration for future canopy recruitment, and in many places in the Northeast sapling densities are much lower indicating a regeneration debt (Miller & McGill, 2019). Competing herbaceous vegetation, poor soils, or shade intolerance have been proposed as factors limiting the ability of *Q. rubra* to survive more than a few years in the understory (Abrams, 2003; Crow, 1988; Lorimer, Chapman, & Lambert, 1994). However, experimental investigations have shown that oak seedlings are similarly shade tolerant as many other species, (no growth or survival benefits beyond 15% full sun'; Dillaway, Stringer, & Rieske, 2011; Kaelke, Kruger, & Reich, 2001; Long et al., 2012). Liming does not affect oak seedling growth (Long et al., 2012), and fire and herbicide treatments to reduce effects of competing vegetation actually negatively affect oak seedlings compared with untreated controls (Miller, Brose, & Gottschalk, 2016). However, in all these studies, fencing had substantial and sustained beneficial effects on oak seedling growth and survival. SORTIE, a model to predict Northeastern hardwood forest successional dynamics based on field assessments, indicates that a 1-cm-diameter *Q. rubra* sapling has a 30% probability to survive for 5 years in 1% sunlight, and it will take 125 years to reach 3 m in height (compared with 12 years in full sun; Pacala et al., 1996). Unfortunately, SORTIE, as so many other early investigations into forest regeneration failures, ignores the transitions in the very early life history of *Q. rubra*. It also does not incorporate biotic pressures (insect, rodent, or deer browse intensity), which, as recent evidence suggests (Kelly, 2019; Miller & McGill, 2019), appear crucially important, but are also difficult to capture if deer rapidly consume emerging seedlings.

Matrix populations models (Caswell, 2001), while popular with ecologists for many different species, have not been used frequently for long-lived species such as oaks, and none exists for *Q. rubra*. Therefore, we can only speculate about the importance of shade, other abiotic factors, competition, insect, rodent, or deer herbivory on the demography of *Q. rubra* and in prohibiting transition from germinated seedling to sapling. The successful transition from seedling to sapling and vigorous sapling growth in fenced plots suggests that deer browse is of overriding importance. This is supported by elegant experiments to assess the importance of fecundity and biotic factors (cattle, deer, and rodents) on population growth rates of Valley oak (*Quercus lobata*) in California (Davis et al., 2011). While survival rates for *Q. lobata* varied among years, population growth rates were primarily limited by survivorship and growth of established seedlings and saplings, which were strongly affected by ungulate browsing and rodent damage. The terminology and criteria distinguishing seedlings from saplings vary among investigators (typically height or stem diameter). In our assessment, we follow natural history and, in part, the demographic model using *Q. lobata* (Davis et al., 2011). We define seedlings as oaks that recently germinated and are <20 cm tall. We define saplings as individuals >20 cm tall, regardless of age.

We were not interested in building a full demographic model, but we were looking for a quick assessment (every year or in short intervals) that allowed us to evaluate whether differences in deer management approaches and changes in deer abundance would affect the growth and transition from seedling to sapling for *Q. rubra*. We therefore chose to assess deer browse frequency and rodent or insect attack in annual oak cohorts that we followed for a growing season up to a year. We incorporated rodent and insect attack into our assessments due their importance in affecting oak seedling survival and growth in other studies. We did not focus on survival, because browsed oaks, or oaks cut by rodents may produce secondary sprouts with very small leaves, and these individuals may linger for many years (very few return to vigorous growth; B. Blossey personal observation). We also chose to plant propagated oaks to standardize our approach across many different forests. In many of our local forest fragments, naturally germinating oak seedlings are extremely rare, occur only in microsites protected from deer browse, such as in treefalls or on steep slopes, are not produced annually, and their abundance varies with overstory tree composition. This

FIGURE 4 Top row L to R: Oaks seedlings ready to transplant, individual oak, and field cages to protect seedlings. Bottom row L to R: Healthy oak protected by wire-mesh cage, oak in matrix vegetation, healthy surviving oak, and partially browsed oak with a single leaf remaining (all photos by B. Blossey)

variation prevented use of naturally occurring *Q. rubra* seedlings for our assessments.

Each September and October, we collected *Q. rubra* acorns from local sources and stored them over winter in gauze bags buried in moist sand in a dark walk-in environmental room (Nor-lake) at 4°C. We planted acorns each February/March in individual SC7U Ray-Leach Cone-tainers (3.8 cm diameter × 14 cm deep; Stuewe and Sons) using commercial potting soil (Farfard Canadian growing mix No. 1-P) and allowed them to germinate and grow in a greenhouse (20–25°C daytime, 10°C at night) under natural photoperiod. After seedlings developed 2–4 leaves (late April to mid-May), we hardened them outside on elevated metal greenhouse benches with legs standing in buckets filled with soapy water to prevent earthworm colonization. We protected seedlings against deer or rodent herbivory in walk-in field cages (Lumite® screening, shade 15%, porosity 1629CFM; Synthetic Industries).

For each site, we selected 40 well-watered seedlings with 3–8 leaves (Figure 4) usually 8–15 cm tall. We typically selected a 100 m × 100 m area and planted seedlings >3 m apart along multiple meandering transects (Figure 4) from mid-May to mid-June, the same time field germinated oaks would appear in our region. We avoided planting seedlings next to live large trees or in windfalls, on very steep slopes, or among large boulders that could function as refuges by limiting physical access by deer. We used a handheld drill with a 5-cm diameter, 30-cm long masonry drill bit to create tapered planting holes (10–15 cm deep × 5–10 cm wide). We removed rooted seedlings from their Cone-tainers, removed the acorn (to reduce rodent predation), and then planted seedlings firmly covering potting soil with local soil. We placed a numbered metal tag (Racetrack aluminum tags; Forestry Suppliers) staked into the ground next to each seedling. Immediately after planting, we measured seedling height (cm), recorded the number of leaves, and then measured "average" height of vegetation at four locations approximately 50 cm away from the seedling (for seedlings planted in 2010 only). Surrounding vegetation could either function as aboveground competition, or possibly as camouflage, and hence protect oak seedlings (Underwood, Inouye, & Hambäck, 2014). We protected half of the seedlings at each site (randomly alternating caged and uncaged oaks) with individual wire-mesh or plastic hardware net cages (Tenax Corporation; 50 cm diameter × 1 m tall, mesh size 1 × 1 cm, Figure 4), to prevent deer access.

We revisited each planting location after 7–10 days to assess each seedling (we recorded no transplant mortality), and thereafter at monthly intervals to record deer browse, rodent attack (recognized by a 45° cut angle), other herbivory or other causes of mortality (usually winterkill). We terminated monthly visits with leaf senescence in October and recorded attack one last time after leaf out in May or June 2011. We repeated the same procedures in 2011, using a new cohort of seedlings planted into the same locations. However, because most damage occurred before leaf senescence, we followed the 2011 cohort only until October. We lost one location in the no management zone; thus, we planted 600 oak seedlings in 2010 and 560 in 2011.

The assessments of the 2010 and 2011 cohorts allowed us to evaluate the impacts of no management (no deer removal, except through deer-vehicle accidents), sterilization, and recreational hunting (Figure 1) on oak browse rates, rodent attack, and growth for oaks protected in individual cages or exposed to deer. Because our different management approaches did not result in sufficient deer population reductions, we changed our management regime beginning with the fall 2013 season (see Section 2.2 for details). We continued assessment of oak seedling browse and growth at a subset of seven sites located within or at the perimeter of the CMA (Figure 2) to assess whether deer browse rates on oak seedlings were sensitive to changes in the deer population from 2010–2011 to 2014–2015 (omitting 2012 and 2013 due to lack of funding). For the latter cohorts, we did not cage any oaks and therefore were able to reduce the number of planted oaks/site to 20. We continued to use baited camera traps to assess the status of the spring deer population each year and to determine whether our changes in deer management in the CMA resulted in herd reduction. Both camera trapping and oak sentinel assessments occurred at a time when known behavioral responses to fall hunting pressure and spatial escape of deer into areas without hunting pressure did not exist.

2.4 | Data analysis

We evaluated deer browse rate as a function of management regime and fencing (open or caged) with Cox proportional hazard models implemented in the R statistical (R Core Team, 2016) package "coxme" (Therneau, 2015). We included initial oak height at planting and average vegetation height (for 2010 only) as covariates. We included site as a random factor in all models to reflect the hierarchical structure of the data. The test compared time (number of days since planting) to deer browse among experimental groups. Data were right-censored because no information about oak browse rates was available after the study period. Deer browsed 113 oaks protected in cages (94 in 2010 and 19 in 2011) by physically dislocating fencing material to gain access. We excluded these oaks from further analyses after deer damaged fences. We used competing risk analysis package "cmprsk", (Gray, 2014) to evaluate probability of an event (defined as a change in the status of an oak due to deer browse) occurring in the presence of competing factors (rodent attack and unknown mortality; Scrucca, Santucci, & Aversa, 2010). We excluded fenced oaks in Cox proportional models and cumulative risk analyses. We fitted separate models for oaks planted in 2010 and 2011 because we lost one study site in 2011.

We used linear mixed models (LMM, package lme4; (Bates, Maechler, Bolker, & Walker, 2014)) to evaluate effect of year, fencing, deer management regime, and second-order interactions on daily growth rates (cm/day) of *Q. rubra* seedlings. We estimated growth rate as the difference in oak height between the first and last sampling date divided by the number of days between samplings. We included site as a random factor to reflect the hierarchical structure of the data. We used variance inflation factors (VIF) to assess

TABLE 1 Number of oaks browsed by deer, attacked by rodents, or dead due to unknown causes when planted without (open) or with individual mesh cages (fenced) in 2010 (15 sites, *N* = 600) and 2011 (14 sites, *N* = 560) at sites with different deer management regimes

	Deer		Rodent		Unknown mortality	
Management	Open	Fenced[a]	Open	Fenced	Open	Fenced
2010						
No management	79	35	12	8	2	10
Sterilization	58	29	32	29	4	9
Hunting	59	30	12	4	2	2
2011						
No management[b]	53	2	1	2	0	1
Sterilization	77	11	6	14	1	1
Hunting	52	6	3	1	0	0

[a]Deer browsed some oaks after breaching fencing. We excluded these oaks from analyses after fence breaches.
[b]One no management site was excluded in 2011.

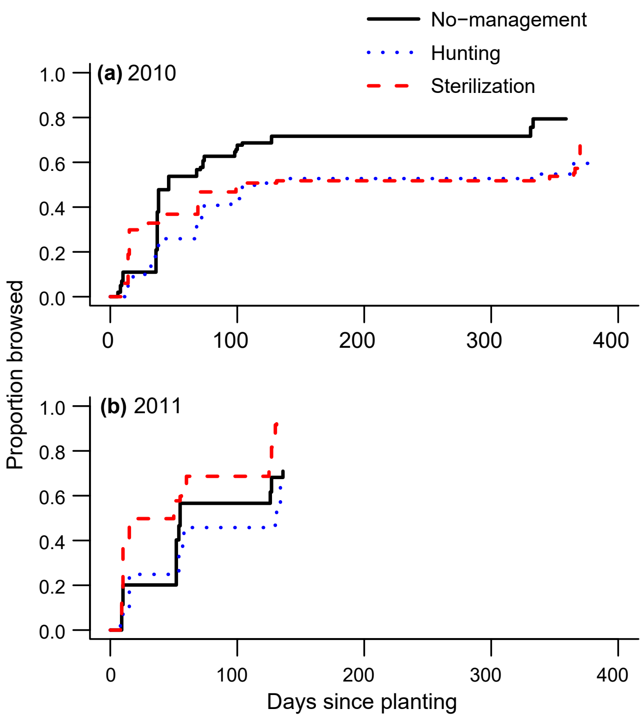

FIGURE 5 Proportion of browsed *Q. rubra* seedling cohorts planted in spring 2010 and 2011 in areas using different deer management (no management, hunting, or sterilization). Only unfenced oaks were included in the analysis (*N* = 20 oaks per site; 5 sites per management regime; one site in the no management area was omitted in 2011). Lines represent expected values according to mixed effects Cox regression (site included as random factor, Table 2). For clarity, we omitted standard errors

collinearity among explanatory variables (Zuur, 2009). Variables were not correlated (VIF < 3).

We used generalized linear mixed models (GLMER) to evaluate the effects of management regime, fencing, and initial oak height on the probability of transitioning into a sapling stage. We used log-likelihood tests between a full model and a model where we deleted the term of interest to assess significance.

We used Akaike Information Criterion (AICc; Burnham & Anderson, 2002) to evaluate explanatory power among competing models (for LMM, GLMER, Cox proportional hazard models, and competing risk analysis). We ranked candidate models according to the difference between model's AICc and min AICc (ΔAICc). We considered all models within two AICc to be similar. For LMM only, we evaluated percent variance explained by the model with conditional (full model) and marginal (fixed effects only) R^2 (Nakagawa & Schielzeth, 2013).

We used linear regression to evaluate changes in the proportion of oaks browsed during the growing season (June–October) as a function of spring deer abundance estimates. We calculated mean oak browse rate during the growing season per year across seven sites located within the core management area (Figure 1). Oak browse by site was estimated as the number of browsed oaks 200 days after planting over the total number of oaks planted at the site (*N* = 20).

3 | RESULTS

We encountered differences in the fate of *Q. rubra* seedlings among locations, management regimes, and in 2010 or 2011 cohorts (Table 1). Across all three management zones, deer browsed 65% of unprotected oaks (*N* = 196 of 300 planted in 2010 and 182 of 280 planted in 2011). In both years, but particularly in 2010, deer compromised and physically dislocated cages to gain access to protected *Q. rubra* seedlings (Table 1). Deer browse resulted in complete or partial removal of leaves, but most often deer removed entire upper stem portions of the seedling (Figure 4). Deer browse did not always result in immediate death, and surviving seedlings produced small replacement leaves. This also sometimes occurred after rodent attack that severed the stem a few cm above ground. Rodent attack and mortality due to unknown causes were similar for unprotected and fenced *Q. rubra* seedlings, but differed among deer management regimes and sites (Table 1). Deer browse and rodent attack occurred

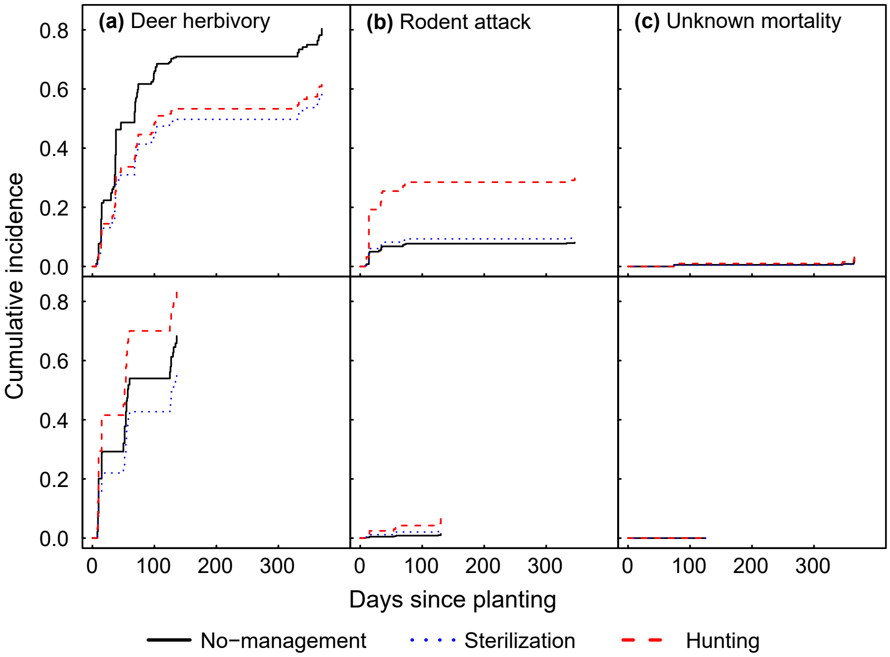

FIGURE 6 Cumulative incidence of deer herbivory (a), rodent attack (b), and unknown mortality (c) for unprotected *Q. rubra* seedling cohorts planted in spring 2010 (top row) and 2011 (bottom row) in areas with different deer management (no management, hunting, or sterilization; *N* = 20 oaks per site; 5 sites per management regime; one site in the no management area was omitted in 2011)

TABLE 2 Results for mixed effects Cox regression evaluating effects of fencing (fenced or open), deer management (no management, sterilization, and hunting,), and average vegetation height on oaks planted in 2010 (15 sites) and 2011 (14 sites)

	Coef (*SE*)	Exp (coef)	*z*-Value	*p*
(A) 2010				
Fixed effects				
Management (hunting)	−0.14 (0.93)	1.15	0.15	.88
Management (sterilization)	2.30 (1.51)	9.89	2.18	.03
Initial height	0.09 (0.04)	1.10	2.34	.02
Initial height: management (hunting)	−0.05 (0.06)	0.85	−0.91	.36
Initial height: management (sterilization)	−0.20 (0.07)	0.82	−2.74	.01
Random effects	Std dev			
Site	0.27			
(B) 2011				
Fixed effects				
Management (hunting)	−0.37 (0.37)	0.70	−1.00	.32
Management (sterilization)	0.55 (0.36)	1.73	1.52	.13
Random effects	Std dev			
Site	0.46			

Note: We present only results for the best model. Estimates and standard errors (*SE*) reported from the model fitted with restricted maximum likelihood.

rapidly after planting, typically within 1–2 months before trailing off (Figures 5 and 6).

In 2010, the risk of browsing by deer was significantly higher for *Q. rubra* seedlings in the no management zone compared with seedlings in hunting and sterilization zones (Figure 5; Tables 2A and S1A). The best model indicated that browse risk significantly increased as a function of initial oak height (Table S2A) and was associated with a significant interaction between management zone and initial oak height, such that taller oaks were more likely to be browsed in the no management zone than in the hunting and sterilization zones. In 2010, initial oak height at planting averaged 14.7 ± 0.13 cm and oaks in the sterilization zone were slightly but significantly shorter at planting (mean ± *SEM*: 13.88 ± 0.19 cm) than oaks planted in no management (14.99 ± 0.22 cm) or hunting (15.17 ± 0.24 cm) zones ($F_{2,594}$ = 10.4, p < .005; a posteriori Tukey test p < .05). However, oak height at planting was similar between caged (14.5 ± 0.18 cm) and unprotected individuals (14.85 ± 0.17 cm; $F_{1,594}$ = 2.01, p = .15) in each management zone. Average height of the surrounding vegetation at planting (measured only in 2010) was significantly lower in the sterilization zone (mean ± *SEM*: 6.9 ± 1.5 cm) than no management

(15.1 ± 1.9 cm) and hunting (11.3 ± 2.7 cm) zones, but did not differ between hunting and no management zones (a posteriori Tukey test; $p < .05$). Average vegetation height at planting was not a significant variable in our analyses and dropped from the best model (Table S1A).

In 2011, we found a marginally significant effect of management zone (log-likelihood test between the model including management zone and the null model: $\chi^2 = 5.9$, $df = 2$, and $p = .05$) and no significant effect of initial oak height at planting (log-likelihood test between the model including height and the null model: $\chi^2 = 0.35$, $df = 1$, and $p = .85$) on the risk of being browsed by deer. However, the best model (lowest AICc) included management zone (Table S1B) and indicated that the risk of deer browsing was highest in the sterilization zone, followed by the no management zone, and the hunting zone (Figure 5b; Table 2B). Initial height of oaks planted in 2011 averaged 12.9 ± 0.11 cm and did not differ among management regimes or fencing treatments ($p > .05$).

Cumulative risk analysis indicated that risk of deer herbivory was significantly higher than risk of attack by rodents or unknown mortality (Figure 6; Table 3). For oaks planted in 2010, the risk of deer herbivory was significantly higher in the no management zone than in sterilization or hunting zones, whereas risk of rodent attack was higher in sterilization than no management or hunting zones (Figure 6; Tables 3 and S2). Unknown mortality (almost exclusively winterkill) was similar across all management zones and significantly lower than the risk of being browsed by deer or attacked by rodents (Figure 6; Table S2). For oaks planted in 2011, risk of deer herbivory was significantly higher in the sterilization zone, but risk did not differ between no management and hunting zones (Figure 6; Table S2). Rodent attack and unknown mortality were similar across management zones and insignificant (Figure 6).

TABLE 3 Results of cumulative risk analyses evaluating effects of deer management (no management, hunting, and sterilization) and average vegetation height (cm) on risk of deer herbivory and rodent attack occurring in presence of competing factors for oaks planted in 2010 (15 sites) and 2011 (14 sites)

	Coef (*SE*)	Exp (coef)	*z*-Value	*p*
(A) 2010				
Deer herbivory				
Hunting	−0.59 (0.16)	0.56	−3.68	<.001
Sterilization	−0.49 (0.18)	0.62	−2.75	.006
Rodent attack				
Hunting	0.20 (0.47)	1.22	0.42	.67
Sterilization	1.43 (0.40)	4.18	3.62	<.001
(B) 2011				
Deer herbivory				
Hunting	−0.33 (0.18)	0.72	−1.81	.07
Sterilization	0.44 (0.17)	1.55	2.56	.01

Note: Initial vegetation height was not significant and dropped from best models. The null model was the best model predicting unknown mortality (for 2010 and 2011) and rodent attack (2011). For procedures of model selection, see Table S2.

Protected *Q. rubra* seedling grew significantly faster than unprotected oaks across all management zones in 2011 but not in 2010 (significant treatment × year interaction; Table 4; Figure 7). We also found a significant interaction between management regime and year (Table 4) such that growth rate was lower in the sterilization zone in 2011 compared with 2010 (Table 4). The proportion of variance explained by the fixed factors marginal $R^2 = 0.40$, whereas the conditional $R^2 = 0.43$, indicating the proportion of variance explained by the full model. Over the study period, 67 oaks transitioned into saplings (>20 cm; 64 and 3 of the 2010 and 2011 cohorts, respectively). Of the 67 oaks that transitioned into saplings, 54 were not browsed by deer, and 13 were browsed at least once. Probability of transitioning into saplings was significantly higher for unbrowsed oaks ($\chi^2 = 6.4$, $df = 1$, $p = .01$) and positively correlated with initial planting height (log-likelihood ratio; $\chi^2 = 234.36$, $df = 1$, $p < .001$). Deer management zone had no significant effect on probability of transitioning into a sapling stage.

Our spring deer population estimates indicated a stable population in our CMA from 2009–2012 (Figure 7). With our switch to using DDPs in 2013, our 2014 spring population estimate for the first time indicated a reduced deer population and this trend continued in 2015, although immigration offset these gains in 2016 (Figure 8).

Annually, our hunters (and vehicle collisions) removed 40%–100% of the estimated spring deer population (a total of >440 deer from 2009 to 2017) from the CMA. Immigration, rutting activity, and foraging deer from areas adjacent to the CMA are included in this tally and indicate the importance of dispersal in open populations. Mean oak browse rate was significantly and positively correlated with mean deer spring abundance estimates ($F_{1,2} = 71.5$, $p = .01$; $R^2 = 0.96$; Figure 9); that is, as the deer population in the CMA was reduced, oak browse rates declined linearly. The proportion of *Q. rubra* browsed by deer varied annually and among the seven sites located within the CMA (Table S3).

4 | DISCUSSION

Despite differences among locations and years, our study demonstrated that deer browse was the overwhelming threat to growth of unprotected *Q. rubra* seedlings, with rodents and other factors relatively unimportant (Figure 6), confirming our second hypothesis. These results align well with results of regional studies (Kelly, 2019; Miller & McGill, 2019) and the demographic model for *Q. lobata* in California (Davis et al., 2011), all indicating that after successful germination, seedlings are unable grow and transition to larger saplings under high deer browse pressure. This browse (and rodent attack) occurred rapidly in spring and early summer, and we would expect the same to occur for naturally germinating oaks. This will not allow seedlings to accumulate sufficient resources for successful regrowth should they be browsed, ultimately resulting in recruitment failure. In addition, because it occurs so rapidly after germination, and browsed seedlings are almost impossible to detect,

TABLE 4 Results of linear mixed model to evaluate effects of fencing, deer management regime (MR) and year planted on growth rate (cm/day) of fenced and deer accessible oak seedlings at 15 sites in 2010 and 14 sites in 2011

	Est	*SE*	*df*	*t*-Value	*p*
Factor					
Intercept	0.002	0.004	40.23	0.36	.72
Year planted	−0.005	0.004	1,153.05	−1.17	.24
Treatment (open)	−0.006	0.003	1,165.00	−1.96	.05
MR (hunting)	0.004	0.006	25.57	0.72	.48
MR (sterilization)	0.001	0.006	37.64	0.18	.86
Year planted:Treatment (open)	−0.041	0.004	1,163.04	−10.19	.00
Year planted:MR (hunting)	0.003	0.005	1,163.75	0.55	.59
Year planted:MR (sterilization)	−0.019	0.005	1,164.31	−3.50	.00
Random effects	Std dev				
Site	0.007				

Note: Only results for the best model are presented. Estimates and standard errors are reported from the model fitted with restricted maximum likelihood. *p*-Values are estimated using Satterthwaite's or Kenward–Roger's methods for degrees of freedom and *t*-statistics (Kuznetsova, Brockhoff, & Christensen, 2017).

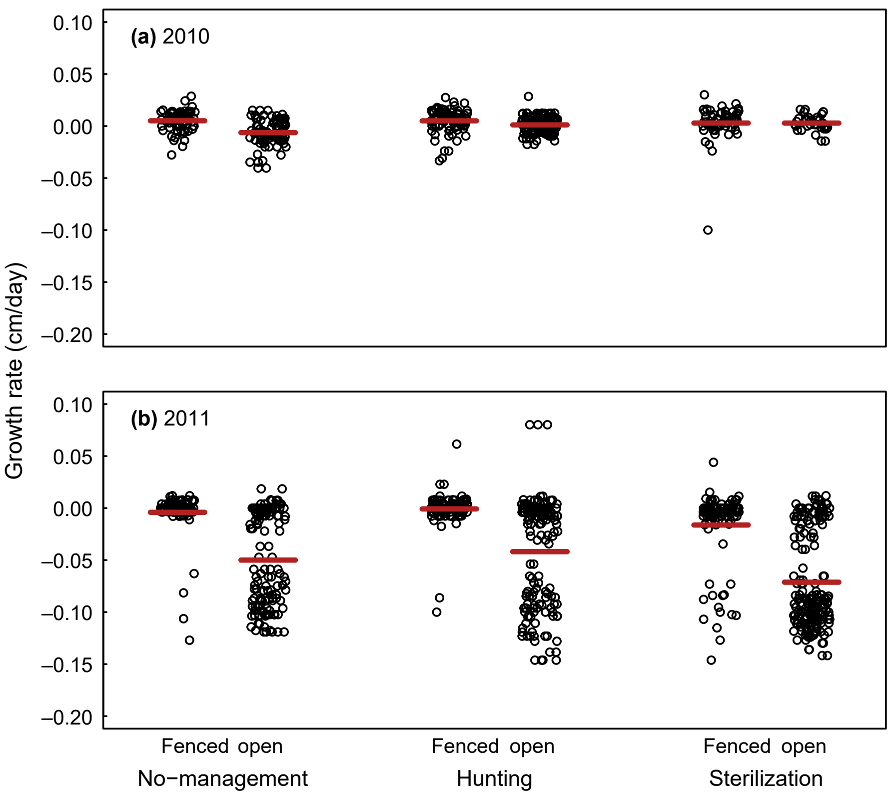

FIGURE 7 Growth (cm/day) of *Q. rubra* seedling cohorts planted in spring (a) 2010 and (b) 2011 at sites with different deer management (no management, hunting, or sterilization; *N* = 5 sites/management regime, one site omitted in the no management area in 2011). Oaks were either protected from deer in individual cages (fenced, Figure 4) or accessible by deer (open). Points (slightly jittered to reduce overlap) represent growth rates of individual seedlings and red horizontal lines indicate mean growth rate of caged and unprotected oaks in each management regime. For model results, see Table 4

even experienced observers will likely miss the deer browse effect on small seedlings.

We need to reject our first hypothesis. Differences in management regimes (no management, sterilization, or recreational hunting) did not result in meaningful differences in *Q. rubra* browse rates (Figure 5) despite some inconsistencies across years. This may not be surprising, given that we were initially unable to reduce the deer population in the CMA (Figure 8). There was a small but noticeably higher level of deer browse in the no management zone in 2010, but no differences in browse intensity among management regimes during 2011 (Figures 5 and 6).

Specifically, recreational hunting was unable to decrease deer densities sufficiently to protect growth of the majority of *Q. rubra* seedlings, as reported elsewhere (Bengsen & Sparkes, 2016; Blossey et al., 2017; Simard, Dussault, Huot, & Cote, 2013; Williams et al., 2013). This inability of woody species to transition from seedlings to saplings over much of the eastern US, and not just of palatable species (Kelly, 2019; Miller & McGill, 2019), occurs in a region where

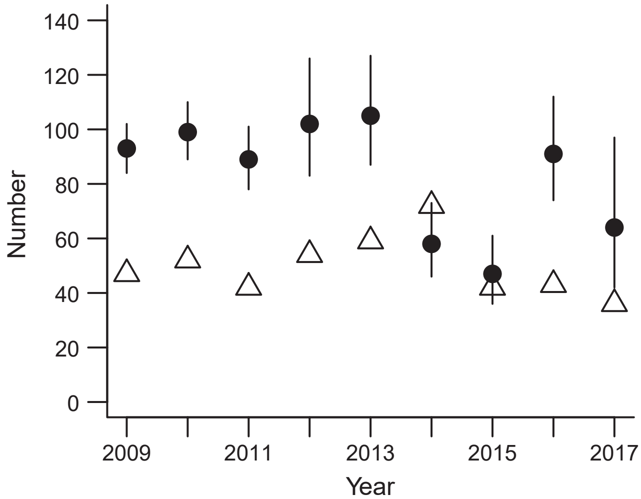

FIGURE 8 Annual spring deer population estimate (and 95%CI; circles; estimated using 12 infrared-triggered cameras set over bait for 5–7 days) and number of deer removed the following fall/winter by volunteer hunters and deer-vehicle accidents (open triangles) in the core management area (Figure 2). In some years, deer removals exceed spring population estimates due to immigration, rutting, or foraging activity typical in open ungulate populations

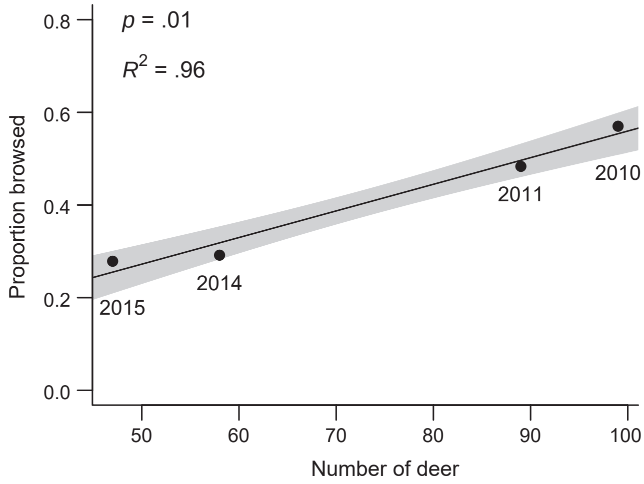

FIGURE 9 Proportion of *Q. rubra* seedlings browsed during the growing season (June–October) as a function of annual spring deer abundance (estimated using 12 baited infrared-triggered cameras) in the core management area (Figure 2). Line and shaded areas depict linear model predictions and 95% CI

recreational hunting is widespread, ubiquitous, and accepted by the vast majority of citizens (Brown, Decker, & Kelley, 1984; Decker, Stedman, Larson, & Siemer, 2015). Some authors claim that hunting can reduce deer browse pressure on herbaceous and woody species, but browse reductions were either small (Hothorn & Müller, 2010), or we lack information about differences in hunting pressure in reference areas that also saw improvements in woody and herbaceous plant performance (Jenkins, Jenkins, Webster, Zollner, & Shields, 2014; Jenkins, Murray, Jenkins, & Webster, 2015). We therefore need to reject claims by wildlife management agencies that recreational hunting is sufficient to allow forest regeneration and can protect biodiversity (NYSDEC, 2011; Rogerson, 2010).

Animal rights and animal welfare organizations have long claimed that deer are not responsible for lack of forest regeneration and that there are more humane methods for managing populations (HSUS, 2018a, 2018b; PETA, 2018). However, there is no evidence to date that can support claims that fertility control alone can sufficiently reduce deer abundance in free-ranging populations (Hobbs & Hinds, 2018; Raiho, Hooten, Bates, & Hobbs, 2015; Ransom, Powers, Hobbs, & Baker, 2014), including our own (Boulanger & Curtis, 2016). Examples cited as success stories show reduced fertility on islands or in fenced populations (Naugle, Rutberg, Underwood, Turner, & Liu, 2002; Rutberg, Naugle, Thiele, & Liu, 2004). To the best of our knowledge, no study has linked fertility control efforts to changes in other ecological parameters, such as changes in plant growth or plant communities, a long overlooked aspect of fertility control research (Ransom et al., 2014). Our study is the first attempt to associate performance of an indicator plant species to deer fertility control. We saw no evidence that fertility control is a viable tool for reducing herbivore populations or browse rates on *Q. rubra* seedlings in a fragmented suburban landscape. Despite a >90% doe sterilization rate and near elimination of deer fawns in our sterilization zone, the deer population remained stable due to immigration, particularly of bucks (Boulanger & Curtis, 2016). There was no reduction in the browse intensity on oak seedlings (Figures 5 and 6). Our results, including that oak seedlings protected from deer browse performed well at all sites, and results of other studies showing recruitment success in fenced areas, indicate that deer are indeed the major stressors in preventing forest regeneration. Our data offer no support for the promise of fertility control as a means to reduce deer browsing pressure.

We found support for our third hypothesis, that growing conditions at all our field sites enabled oak seedling growth (if protected by cages; unless compromised by deer; Figure 7), regardless of site-specific growing conditions, differences in land-use history, or potential presence of other associated stressors (invasive earthworms and invasive plants). Thus, at least in our area and probably across much of the eastern US, *Q. rubra* should be able to transition from seedlings to saplings successfully once white-tailed deer populations are sufficiently reduced. We can also confirm our fourth hypothesis that the browse intensity on *Q. rubra* seedlings is a function of the deer population size (Figure 9), indicating that our sentinel approach is a sensitive and useful way to measure deer browse pressure and the success, or lack thereof, of different deer management approaches. We eventually achieved a deer population reduction (Figure 8) using methods typically not available to the recreational hunter, such as shooting over bait, and at night over extended periods. However, these intensive efforts will need to continue due to immigration pressure from the areas surrounding our CMA.

We are working with communities surrounding the Cornell campus to develop a regional approach. We are hopeful, although not certain, that collectively we may reduce deer populations to levels where *Q. rubra* seedlings will grow and ultimately transition to the sapling stage. Hunting, despite allowing access to every possible safe location on and near campus, removed about 50% (together with car accidents) of our annually estimated spring deer population in the CMA, and this temporary population reduction was not sufficient to affect oak browse rates or the deer population. Only after implementation of our DDP approach did we see an appreciable drop in the CMA deer population. Combined, over nine years, our efforts removed nearly 750 deer from our core management area of <1,000 ha demonstrating the effort required to locally manage open deer populations. In some years, we lethally removed as many deer as we estimated existed in our core management area (Figure 8) highlighting the importance of deer dispersal and deer foraging. Populations quickly rebounded (our population estimation occurred before fawning season), although the long-term trajectory is showing declines despite persistent immigration.

Since their establishment in the early 1900s, state wildlife agencies have been able protect and recover deer populations in North America to historically high levels. However, they are financially and philosophically poorly equipped to effectively address current conservation challenges associated with negative impacts of high deer populations (Jacobson, Organ, Decker, Batcheller, & Carpenter, 2010). Ecological or human health concerns have minimal impact on decisions about desirable deer population goals, in part, because management agencies do not implement routine assessments of ecological health indicators to guide deer management decisions, and thus such (unrecognized) impacts cannot inform public attitudes or management decisions (Riley et al., 2002). Further complicating the issue is that deer impacts are not necessarily a function of deer abundance or density, the metric often used to define landscape-level population management goals (Putman, Watson, & Langbein, 2011). Despite repeated calls to adopt accountability and good governance principles in more holistic stewardship and wildlife management (Decker et al., 2016; Hare & Blossey, 2014; Leopold et al., 1947), agencies continue to focus largely on interests of stakeholders who buy hunting and fishing licenses. Our own experience and the overwhelming scientific evidence for the primary role of deer in the deterioration of ecological, economic, and health of our landscapes in the presence of recreational hunting (Côté et al., 2004; Kelly, 2019; Kilpatrick et al., 2014; Miller & McGill, 2019; Nuttle et al., 2011; Raizman et al., 2013) does not bode well for the future, unless major changes are implemented.

Restoring and maintaining diverse and healthy landscapes into the future will require, first and foremost, changes in deer management. We have no evidence that this can be accomplished using recreational hunting. In the past, strong winters caused major deer mortality in traditional winter yards, however, with climate change and milder winters with less snow cover, this deer mortality is no longer a major mortality factor. Use of regulated market hunting may be an important tool in the immediate future (Vercauteren et al., 2011). We further believe that healthy landscapes require top predators (Estes et al., 2011) and argue that species such as mountain lions and wolves should be afforded federal protection and be allowed to return and recolonize their traditional ranges across the continent. Through their consumptive effects and the creation of a landscape of fear, we anticipate cascading effects that will benefit not just primary producers but a beneficial restructuring of entire food webs (Clinchy, Sheriff, & Zanette, 2013; Manning, Gordon, & Ripple, 2009; Suraci, Clinchy, Dill, Roberts, & Zanette, 2016). We recognize that this is currently highly controversial in North America, but Europe is leading the way in trying to restore large terrestrial predator communities (Chapron et al., 2014). Regardless what options are implemented, the development of indicators or metrics to gauge deer impacts and to determine how changes in deer management affect the health of ecosystems and people is paramount. Society will need to decide how to fund regular assessments, and whether the responsibility for implementation of assessments will rest solely with wildlife management agencies. But managing wildlife as a public trust resource demands that all citizens will have the ability to obtain regularly updated information about the status of land health, and hold management agencies accountable if performance is lacking (Hare & Blossey, 2014).

Our oak sentinel approach showed great promise as an assessment tool. A large number of methods and metrics have been proposed to assess deer impacts, including plant community composition (Habeck & Schultz, 2015), woody browse indices (Morellet, Champely, Gaillard, Ballon, & Boscardin, 2001; Pierson & DeCalesta, 2015; Waller, Johnson, & Witt, 2017), and performance (height and flowering) of herbaceous species (Balgooyen & Waller, 1995; Fletcher, McShea, Shipley, & Shumway, 2001; Williams, Mosbacher, & Moriarity, 2000). Woody browse indices fail to measure impacts on herbaceous species, and other methods require presence of existing specimens. In areas with long-existing large deer populations and depauperate landscapes, these species may no longer be present. By not relying on existing seedlings, saplings, or herbaceous plants that may differ in composition, age, or abundance among sites, we were able to standardize assessment protocols across sites and years. As such, our methodology is applicable at the local and regional scale and allows rapid assessment (within 100 days) of local deer browsing pressure helping managers rapidly evaluate outcomes following potential changes in deer management regulations or approaches. Under low deer browsing pressure, *Q. rubra* seedling mortality is low (20% over a 6-year period in Wisconsin) and 3% per year in the southern Appalachian Mountains, although annual mortality for slow growing individuals may increase to 10%–15% (Kaelke et al., 2001; Wyckoff & Clark, 2002). Annual *Q. rubra* seedling browse rates exceeding 10%–15% are unlikely to enable regeneration in a species needing a decade or longer to grow sufficiently tall to place the top leader out of danger of being browsed by deer. However, we likely need to reduce acceptable rates of oak seedling browse even further if we want to protect more sensitive plant species. Herbaceous species, such as *Trillium grandiflorum* or *T. erectum*, continue to suffer browse rates that will lead to local extinction (Knight et al., 2009), even in areas where browse rates of oak seedlings fall below 15% (B. Blossey, unpublished data).

Due to its ease of implementation and the demonstrated sensitivity to changes in the size of the deer population, we believe oak sentinels are an important tool in assessing landscape health. We recognize that oak sentinels alone will not suffice and that additional more browse-sensitive indicator species will need to be developed to allow assessments once deer populations have declined. Holistic management will also require that additional ecological, social, human health, and economic metrics will be required to create a portfolio of indicators that can guide decision making in holistic deer and landscape management. The future of our forests, the biodiversity contained in them, climate change mitigation, and human health are closely linked to our ability to embrace the required changes in deer management.

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CONFLICT OF INTEREST

No conflict of interest exists with the submission of this manuscript.

AUTHOR CONTRIBUTIONS

BB developed the oak study; PC and JB developed the deer project; AD analyzed the data; and BB led the MS writing to which all authors contributed.

DATA AVAILABILITY STATEMENT

Data associated with this work are available in the Dryad Digital Repository: https://doi.org/10.5061/dryad.6q573n5v5.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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Pennsylvania Woodlands

Number 7

Dead Wood for Wildlife

Most of us would have little difficulty responding if asked what value trees have for people. Living trees provide shade. Trees filter air and produce oxygen with their leaves. Trees can soften the impact of rain, prevent soil erosion, produce food, and are pleasing to the eye. Harvested trees provide many valuable products for people. When a tree is cut, it can be used to frame, insulate, or heat a house. This publication was written and reproduced on paper made from trees.

But most of us would have much difficulty relating the value that trees have for wildlife, especially dead trees. Trees do have special value for wildlife. Dead parts of live trees and dead trees, whether standing (snags) or fallen (logs), are particularly important resources.

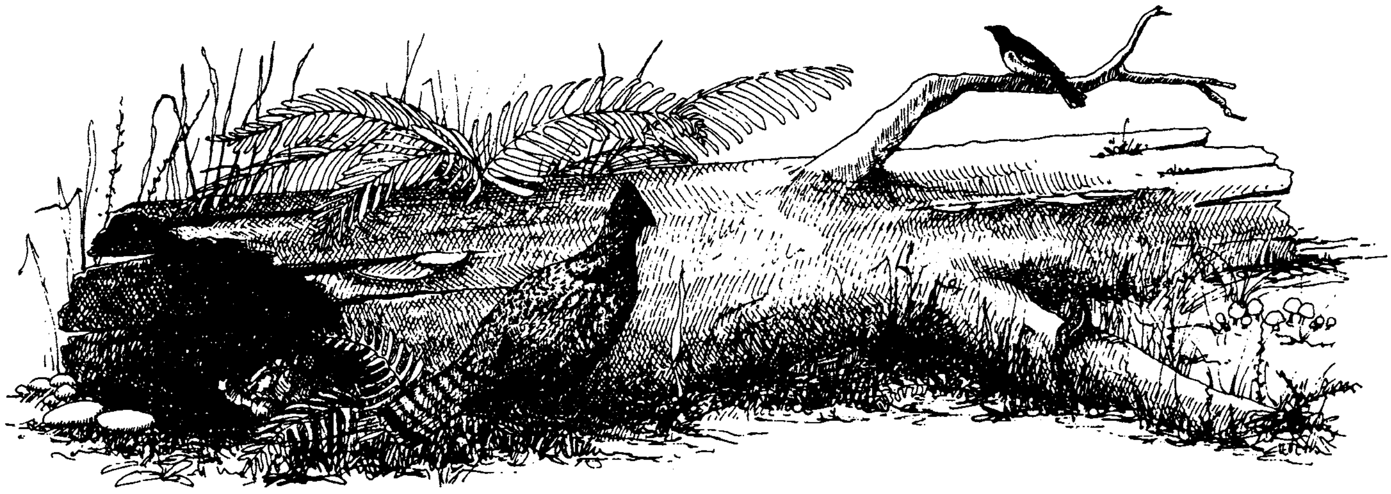
Felling a tree for whatever reason alters wildlife habitat. The effects can be beneficial or detrimental, planned or haphazard. Some people believe leaving dead trees in the forest to rot is a waste of resources. However, dead trees offer both shelter and food to many wildlife species. Dead limbs and trees are a natural and desirable part of wildlife habitat. The existence of numerous species depends on the presence of dead trees. A fallen tree becomes infested with fungi and insects. As the tree decomposes, nutrients are recycled into the soil and a microhabitat favorable for the growth of new tree seedlings is often created.

Insects, salamanders, snakes, mice, and shrews seek refuge in rotting logs. Skunks, bears, and woodpeckers repeatedly return to these cafeterias for easy pickings. Depending on a log's location relative to good cover, a grouse may use it as a drumming site. Some rot-resistant logs have been used by generations of ruffed grouse.

The accumulation of organic material, including damp, rotting wood and leaves, favorably affects mushroom populations. Mushrooms are food for insects, turtles, birds, mice, squirrels, and deer. During critical winter periods, highly nutritious mushrooms can compensate for nutrient deficiencies in deer's native forage.

Ruffed grouse and eastern towhees, among other species, nest under partially elevated logs. Depending on their size, hollow logs can shelter a variety of forest mammals such as shrews, chipmunks, and bears. Foxes and coyotes also may use logs for dens. For some mammals, including deer mice, chipmunks, and squirrels, log tops are highways over the forest floor. Rattlesnakes often coil next to a log and wait for food to arrive.

Logs and stumps meet the special habitat requirements



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of the redback and slimy salamanders. The four-toed and longtailed salamanders hide in moist, decaying wood. The eggs of the northern spring salamander are laid in running water under logs. Greater and lesser gray tree frogs may be found in hollow trees, under loose bark, or in rotted logs during the summer. Seven species of turtles bask on logs that are in or near water. The eastern box turtle may burrow under a log during hot weather. The northern fence lizard is found in log piles and around stumps and hibernates in the rotting wood. Special habitat requirements of the five-lined skink include open woods with logs and slash piles.

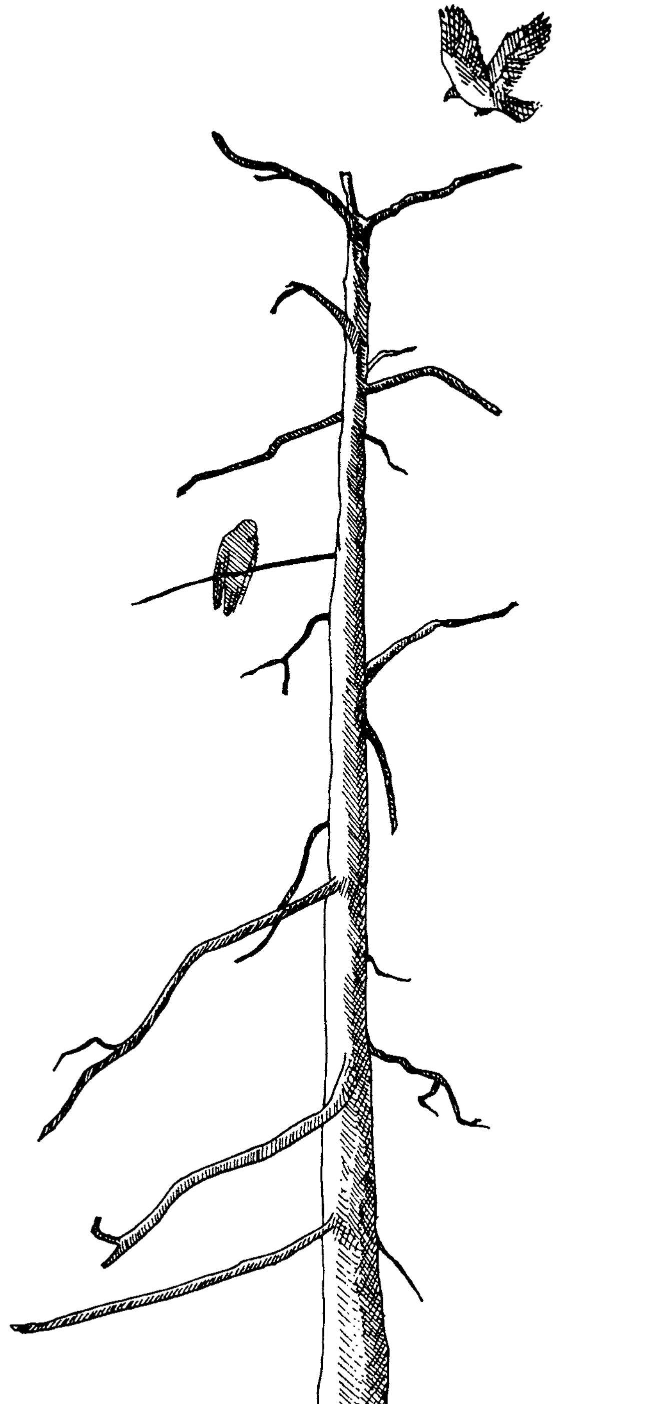
Snakes use logs for shelter and food-seeking activity. Some species, such as the eastern garter snake and the eastern worm snake, hibernate in rotting wood. At least 19 kinds of salamanders and 26 species of reptiles make some use of logs, stumps, bark, and slash piles in Pennsylvania's forests. Ecologists believe dead wood is one of the greatest resources for animals species in the forest.

Wildlife use of dead snags and cavity trees

Standing dead trees (snags) and dead parts of live trees offer both room and board for many kinds of wildlife. Tree cavities in live or dead trees are used by 35 species of birds and 20 species of mammals in Pennsylvania (Tables 1 and 2).

Wood ducks look for tree cavities near water. Barn owls look for nest sites that are near large fields. Bluebirds can nest in wooden fence posts bordering farm fields, or they can occupy holes in snags that are left in recently clearcut areas. Unlike the barn owl and bluebird, pileated woodpeckers are birds of the big woods and next in tree holes far from fields. Table 1 lists the habitat of 35 bird species that nest in free cavities.

In addition to location, the nature of the cavity tree is important to wildlife. Some species choose a cavity in either a live or a dead tree; this is not true of all species. The yellow-bellied sapsucker, for example, constructs a new cavity each year in a live tree. The northern flicker, on the other hand, uses or excavates cavities in dead trees. Whether a snag is hard (sound) or soft (plunky) also determines which birds use it. The pileated and hairy woodpeckers choose to nest in hard snags. The brown creeper nests under exfoliating bark of hard snags. The black-capped and Carolina chickadees prefer to excavate nesting cavities in soft snags.



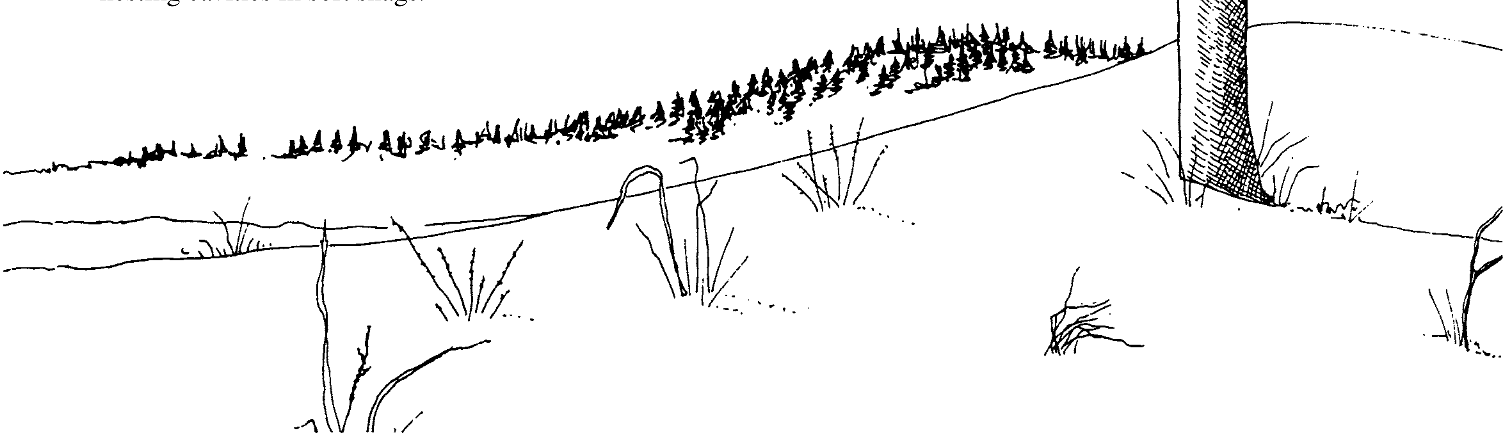


TABLE 1. Birds that use tree cavities in Pennsylvania.

CAVITY-USING BIRDS	CUTTING SITE AND TYPE OF CUTTING ACTIVITY*					FREQUENTLY CHOSEN CAVITY TREE TYPE**		
	FOREST INTERIOR		FOREST-FIELD EDGE	NEAR WATER, WETLAND	SCATTERED TREES, LARGE FIELDS		DEAD	
	A	**B**	**C**	**D**	**E**	**LT**	**HS**	**SS**
Wood duck				x		x	x	x
Hooded merganser				x		x	x	x
American kestrel			x		x	x	x	x
Barn owl			x		x	x	x	?
Screech owl		x	x		x	x	x	x
Barred owl	x			x		x	x	x
Sawwhet owl	x			x			x	x
Great horned owl	x	x	x	x	x	x	x	x
Chimney swift			x	x	x	x	x	x
Northern flicker		x	x	x	x		x	x
Pileated woodpecker	x			x		x	x	
Yellow-bellied sapsucker	x	x	x	x		x		
Hairy woodpecker	x			x		x		
Downy woodpecker	x	x	x	x			x	x
Red-headed woodpecker		x	x	x	x	x	x	x
Red-bellied woodpecker	x	x		x		x	x	?
Great crested flycatcher	x	x	x	x		x	x	x
Tree swallow			x	x		x	x	x
Black-capped chickadee	x	x	x	x				x
Carolina chickadee	x	x	x	x				x
Tufted titmouse		x	x	x		x	x	x
White-breasted nuthatch	x	x	x	x		x		
Red-breasted nuthatch	x			?		x	x	x
Brown creeper	x			x			x	
Winter wren	x	?		x		x	x	x
Carolina wren	x	x	x	x		x	x	x
House wren		x	x	x	x	x	x	x
Bewick's wren		x	x	x	x	x	x	x
Prothonotary warbler	x			x		x	x	x
Eastern bluebird		x	x		x	x	x	x
Purple martin		x	x		x		x	?
European starling			x		x	x	x	x
House sparrow			x		x	x	x	x
Turkey vulture	x	x	x	x				x
Black vulture	x	x	x	x				x
TOTAL:	19	19	24	27	14	26	29	27
PERCENT:	54	54	69	77	40	74	83	77

* **Type of tree cutting activity. A:** partial cutting within a woodlot, often a diameter limit cut or thinning; **B:** cutting heavy enough to create clearings within a woodlot, often a clearcut; **C:** cutting within 100 feet of a field, often fuelwood removal; **D:** any cutting near a stream, pond, or within other wetland sites; **E:** removal of trees competing with crops or for purposes of site development, often the elimination of a fencerow.

** **Cavity tree type. LT:** a live tree with a cavity large enough to shelter the indicated species; **HS:** a hard or firm, dead snag with or without bark and with a cavity large enough to shelter the indicated species; **SS:** a soft, punky, dead snag with a suitable cavity.

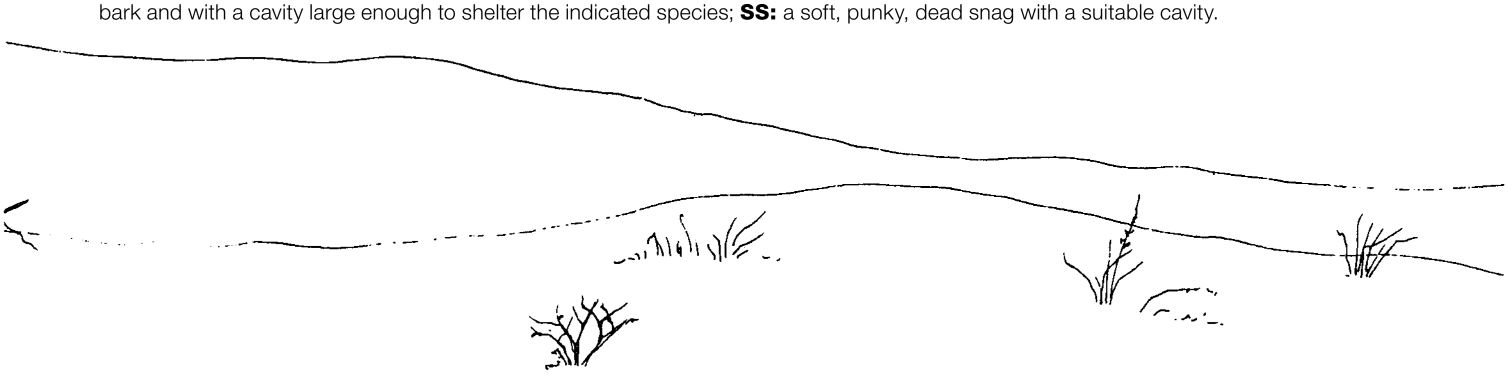


TABLE 2. Mammals that use tree cavities in Pennsylvania.

Opossum	Red squirrel
Pipistrel bat	Eastern flying squirrel
Little brown bat	Northern flying squirrel
Keen bat	Chipmunk
Indiana bat	Deer mouse
Silver-haired bat	White-footed mouse
Big brown bat	Porcupine
Evening bat	Raccoon
Gray squirrel	Black bear
Fox squirrel	Long-tailed weasel

Only the squirrels and perhaps one or two kinds of bats are obligate cavity nesters. Other species may use cavities if they are available.

In addition to the soundness and location of a cavity tree, the following other factors may affect its use by wildlife:

- The size of the cavity. Will the entrance accommodate a bluebird, a barn owl, a squirrel, a raccoon, or a bear?
- The diameter and height of the cavity tree. The house wren and bluebird rarely nest in holes more than 12 feet above the ground, while pileated woodpecker cavities are found higher than 15 feet. Generally speaking, the larger the cavity nester, the larger diameter of the tree selected for nesting.
- The direction faced by the cavity entrance. Screech owls, for example, often choose cavities with north-facing entrances and, consequently, low internal light levels.
- The relationship to other cavity trees. Cavity trees chosen by gray and fox squirrels are often located near other cavity trees.
- The nature of the woodlot. Although most species choose stands of deciduous trees or mixed stands including some evergreens, the sawwhet owl prefers stands of evergreens. Whether a cavity tree is located in a woodlot with a dense or open understory also affects its use by some species. Hairy and downy woodpeckers prefer open and dense understories, respectively. Similarly, dense understories favor gray squirrels, whereas more open understories attract fox squirrels.
- The time of the year. Cavity trees are used for nesting, roosting, winter shelter, escape, food storage, and foraging. One researcher found that amphibian and reptilian use was highest in the summer and early fall, followed by high mammalian use in late fall and winter. Bird use is greatest in spring and early summer. People cleaning bird boxes in early March frequently evict deer mice from the winter apartment.

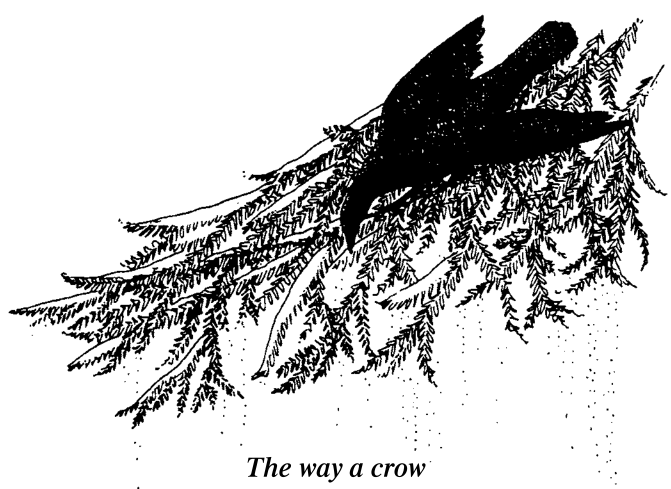
The presence of cavities or the possibility of excavating cavities in wood with heart rot or other decay is not the only attraction of a dying or dead tree for wildlife. Snags are a common source of insects and other invertebrates. This food source may be exceptionally important for overwintering birds.

If snags are houses and cafeterias, they are also airports. Flycatchers use snags for launch sites as they sally forth time and again after flying insects. A snag that borders a field or orchard may be used constantly by hawks and owls while they wait for an errant field mouse. Similarly, kingfishers, ospreys, and bald eagles perch on or fish from dead trees standing in or near water. At least 30 kinds of birds commonly use snags for foraging perches. In addition, the indigo bunting, northern mockingbird, and crow are among species that regularly use snags for singing perches.

Using dead wood for wildlife rather than fuelwood requires some choices. The fuelwood value of a hollow tree must be weighed against the possible value of the wildlife it attracts.

Aside from food or dollar values, the recreational value of such species are, for many of us, worth leaving a few hollow trees and logs on every acre. You may be hunting squirrels, wood ducks, or grouse, or trying to take that special photograph of a bluebird. The entertaining chickadee on your bird feeder may have been born in the hollow aspen tree behind your house. These values are not measured by dollars but by feeling.

The poet Robert Frost put one such intangible value in perspective:



The way a crow
Shook down on me
The dust of snow
From a hemlock tree

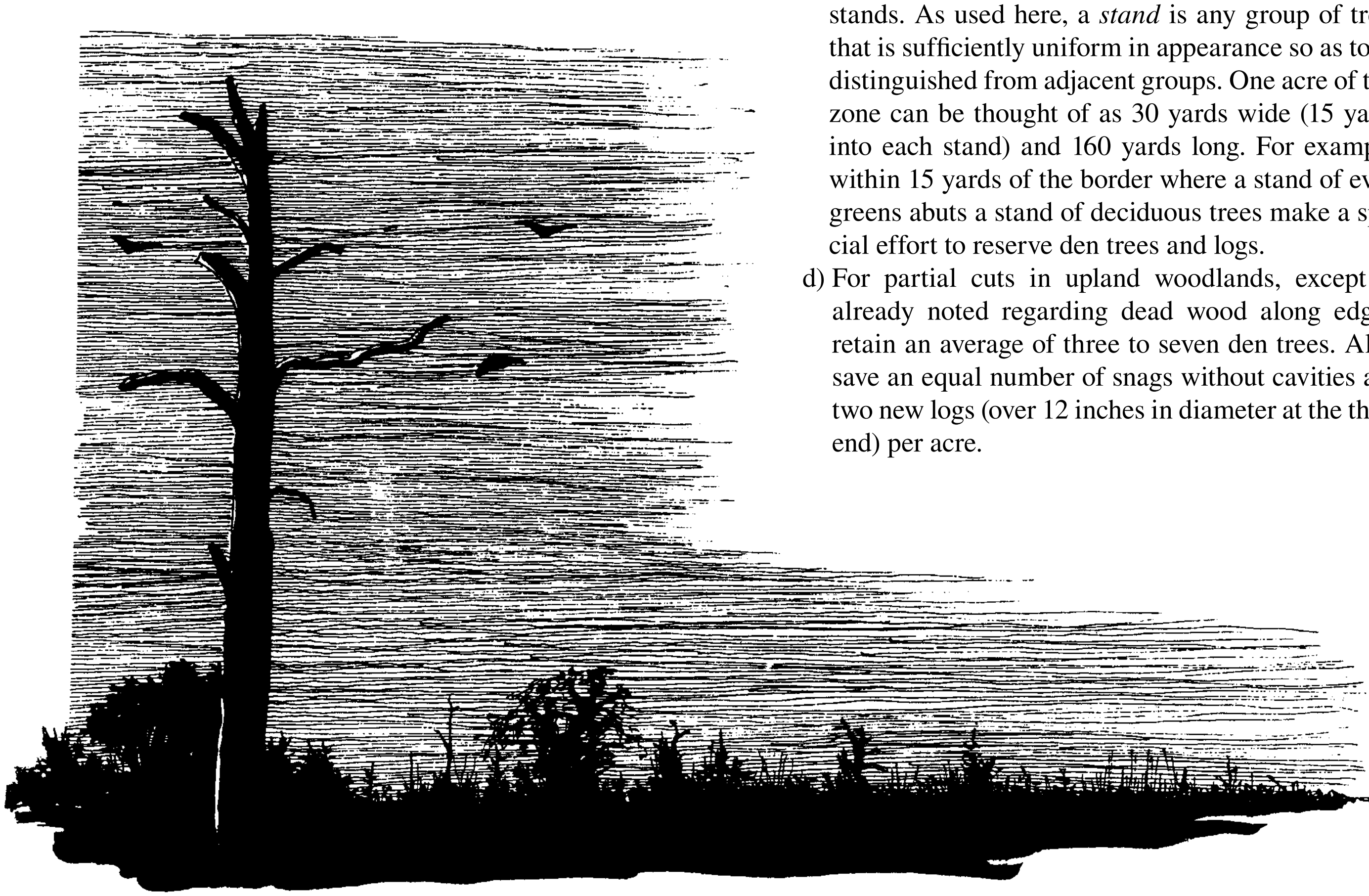
Has given my heart
A change of mood
And saved some part
Of a day I had rued.

Insect populations

The regulation of insect populations is a complex issue. Insects form a major part of the diet of 80 percent of the cavity-using birds in Table 1. Nine of 20 mammals using tree cavities depend on insects for food. Shrews, salamanders, and reptiles that make use of logs, stumps, bark, and slash piles constitute an additional 50 species that forage for insects.

Insect damage to trees is a significant cause of loss. Insectivorous cavity-nesting birds, in many cases, play an important role in the regulation of forest insect populations. Scientists believe that the most important role of birds is the prevention, rather than the suppression, of insect infestations. The protection of cavity-nesting bird populations by promoting forest diversity and leaving snags and den trees is advocated as an economical means to help prevent insect outbreaks in the managed forest.

Many people are familiar with the purple martin's ability to consume large quantities of flying insects. A single purple martin may consume hundreds of mosquitoes in one evening, but bats are the champion. They are the only major predator of night-flying insects. A single big brown bat can consume thousands of mosquitoes before dawn. It is evident that woodlots are best protected from insects by a full complement of species including birds, mammals, reptiles, and amphibians. The alternative could be loss of annual tree growth or expensive spraying of insecticides.



MANAGEMENT CONSIDERATIONS

1. Selective cutting, when only a portion of the trees in a stand is removed in activities such as fuelwood cutting or timber stand improvement, is most likely to be concentrated in areas of vehicle accessibility. This procedure results in removal of snags and logs from *woodland border zones* and fencerows and from *wooded bottoms* traversed by both stream and road. These are precisely the sites where use by wildlife and competition for available nesting and cover sites are greatest (Table 1). Observing the following guidelines can lessen the detrimental aspects of tree harvests in these "edge zones."
   a) Avoid cutting or removing hollow trees and limbs on the ground or standing trees (live or dead) within 15 yards of a field. In woodland areas that are immediately adjacent to this zone, reserve an average of five to ten den trees per acre. Retain all existing logs with varying degrees of composition and at least four new logs (e.g., hollow butt sections of felled trees) per acre.
   b) Avoice cutting or removing hollow trees and limbs within 30 yards of water zones (e.g., streamside riparian zones). In woodland acres that are immediately adjacent to this waterside zone, reserve up to 25 den trees (average 15) per acre. Again, retain logs as described in "a" above.
   c) Retain an average of five to ten cavity trees and two new logs per acre in boundary zones of adjacent stands. As used here, a *stand* is any group of trees that is sufficiently uniform in appearance so as to be distinguished from adjacent groups. One acre of this zone can be thought of as 30 yards wide (15 yards into each stand) and 160 yards long. For example, within 15 yards of the border where a stand of evergreens abuts a stand of deciduous trees make a special effort to reserve den trees and logs.
   d) For partial cuts in upland woodlands, except as already noted regarding dead wood along edges, retain an average of three to seven den trees. Also, save an equal number of snags without cavities and two new logs (over 12 inches in diameter at the thick end) per acre.

2. Clearcutting, when most of the trees in an area are removed, creates a temporary opening and edge in a woodland, and extra bird species are attracted to the forest (Table 1). Under these circumstances, larger woodlands can be attractive to 27 cavity-nesting birds, and most cavity-using mammals and other species *if* the following guidelines are applied.

   Clearcuts, in which all trees, dead or alive, are removed, have a long-term detrimental impact on wildlife dependent on dead wood. The young trees that spring up following clearcutting are not large enough to provide the configuration of dead wood accumulated in the mature stands before clearcutting, and dead-wood deficit develops about 15 years after cutting. This deficit occurs earlier if slash is removed by fuelwood cutters. This deficit may span 40 or more years. For example, depending on its location in Pennsylvania, a clean 20-acre clearcut site is relatively unattractive to 100 or more wildlife vertebrates (birds, mammals, amphibians, and reptiles) for 40 or more years. Conversely, observing the following guidelines can help provide long-term benefits for these same species.

   a) Do not clearcut within 30 yards of water. Partial cutting in this waterside buffer strip should be confined to the solid, live hardwood trees. Note: Slopes next to streams should have wider buffer strips; the steeper the slope, the wider the buffer.
   b) Within clearcuts, reserve at least a ⅕- to ⅓-acre clump of trees for every 5 acres clearcut. Each clump should contain one or more live trees with a squirrel-sized (2½ inches) or larger den entrance. Partial cutting within these tree clusters should be avoided.
   c) Beyond clumps, an average of six to thirteen individual den trees and other snags can be reserved per acre. Den trees should be maintained along clearcut borders, in finger draws, and at the low end of slopes that will help minimize blowdowns. Blowdowns are not, however, wholly objectionable because they contribute logs to the forest floor over time. This process benefits a different set of wildlife species.
   d) Logs are important as wildlife habitat because they last longer than slash. For best distribution of logs on clearcut sites, noncommercial sections of butt logs should not be piled at the log-loading site. Rather, they should be severed from the saleable portion of the log and left at the felling site. Logs oriented along the contour will slow erosion and trap debris. In addition to all older logs with varying degrees of decomposition, at least two new logs (over 12 inches in diameter at the large end) should be retained for every acre cut.
   e) Woody debris (slash) should be reserved on at least 10 percent of the area being clearcut.

3. Additional management tips:
   a) No one can have everything on an acre. In effect, all of the above guidelines should be prefixed with: "If the choice exists… ."
   b) A uniform distribution of cavity trees may be both impractical and, from the standpoint of wildlife, undesirable. The figures used in #1 and #2 above are averages that should be used as guidelines. A few acres may have an excess of cavity trees. This excess can compensate for the many acres that have few or no cavity trees.
   c) If the choice exists, large (over 19 inches dbh), medium (10 to 19 inches dbh), and smaller (less than 10 inches dbh) den trees should be reserved on the same acre, especially in edge zones. A mixture of both live and dead cavity trees is also desirable.
   d) If cavity trees do not exist where you want them, reserve trees with potential for developing a cavity. Candidates include dead or partially dead trees, e.g., a live tree with a broken top.

SUMMARY

Dead wood, both standing and down, serves as important wildlife habitat. Wildlife evolved in forests where dead wood was never removed in the name of woodland management. The increasing demand for forest products has, in many instances, resulted in a lack of dead-wood habitat for wildlife. Application of the management guidelines listed in this publication can help provide some of this important habitat in your woodland.

Can you afford to provide some wood for wildlife habitat, to leave some dead and dying trees, as well as a few hollow logs, on every area? Considering the many rewarding values of wildlife that depend on this resource, the question might better be phrased "Can you afford not to do this?"

Prepared by Jerry Hassinger, wildlife biologist, the Pennsylvania Game Commission; and Jack Payne, extension wildlife specialist, The Pennsylvania State University.

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In Spring, Nature's Cycle Brings a Dead Tree to Life

By Jane E. Brody

- March 24, 1992

IF you don't believe there's life after death, look closer some spring day at a dead tree lying on the forest floor. Chances are, if it has been there for a while, it is teeming with more life now, after death, than when it was standing erect lifting its leafy arms to pray.

Though it lacks the spring finery that inspires poets and lovers, a leafless tree is often more valuable to its forest dead than alive, say ecologists working in the old-growth forests of the Pacific Northwest. This fact, they say, has been largely ignored by wood-hungry forest managers in most of the United States and Europe, where overzealous harvesting of "dead wood" has depleted forests and rendered them highly susceptible to environmental stresses like acid rain.

"Rotten wood was once considered just a fire hazard, a waste, an impediment to travel," remarked Dr. Michael Amaranthus, a soil scientist with the United States Forest Service in Grants Pass, Ore. "More and more we are seeing it as an essential part of the forest system, crucial to its long-term productivity. It provides a reservoir of moisture and nutrients and a variety of habitats and food resources for a wide diversity of organisms. Our understanding of the importance of dead wood has increased a lot in the last 10 years."

When nature cries "timber," countless unseen denizens of the forest rush to take up lodging in the fallen tree. Dead trees serve as warehouses and even factories for essential nutrients that enrich the soil and foster new growth. They store carbon, thus curbing atmospheric carbon dioxide and the pace of global warming. They hold volumes of water that sustain growing trees in droughts. And they serve as nurseries for new plant life, providing cozy niches where seeds can gain a firm roothold and outgrow other seedlings struggling to capture the light that penetrates where the tree once stood.

The trunk of a dead tree is consumed by a varied succession of microbes, plants and animals, which help to replenish the soil as they break down the wood. A result, say the two forest ecologists, Chris Maser and James M. Trappe, is "an accumulation of life and nutrients that is greater than the sum of its original parts."

"In a forest where the trees are repeatedly cut and removed, the soil becomes depleted, the structures deteriorate and the forest loses its resilience for coping with stress," said Dr. Trappe, a forest mycologist at Oregon State University in Corvallis. This has already happened in Germany, where the forests are being severely damaged by air pollution and acid rain, Dr. Trappe said in an interview last week. "And Germany is the country whose concept of intensive forest management served as a model for our own," he noted.

Fallen trees help to preserve the forest by stemming the erosion of soil from wooded slopes and diverting streams that in straight courses might gouge out soil. In fresh waterways, fallen trees trap nutrient-rich sediments and create pools where fish can spawn and fry develop.

Beyond the forest, dead trees help stabilize beaches and create habitats for wildlife in estuaries and salt marshes. Logs that reach the open sea serve as a major source of carbon and other foodstuffs for marine life.

"Unfortunately, very little of this is now happening because the oceans are being deprived of this resource," said Mr. Maser, an author and consultant living in Las Vegas, Nev. "We are beginning to starve the oceans as well as the soil because we are not reinvesting the biological capital nature provides into the forest, ocean, air or land."

"Te function of dead trees in the ecosystem has rarely received the consideration that it deserves," says Dr. Jerry F. Franklin, an ecosystem analyst at the University of Washington's College of Forest Resources in Seattle. "At the time a tree dies, it has only partially fulfilled its potential ecological function. In its dead form, a tree continues to play numerous roles as it influences surrounding organisms. The woody structure may remain for centuries and influence habitat conditions for millennia."

So, these forest scientists urge, woodsman, woodsman, spare thine ax for fallen as well as standing trees. Think twice before hacking up and carting off those logs dead in name only and dooming them to a brief and limited life as firewood. A Long, Rich Afterlife

As scientists with the United States Forest Service in Corvallis in the 1980's, Mr. Maser and Dr. Trappe produced a technical review, "The Seen and Unseen World of the Fallen Tree," that could easily become Exhibit A in the ongoing case to preserve forests. Their publication, number PNW-164, is available for $5.50 from the Superintendent of Documents, U.S. Government Printing Office, Washington, D.C. 20402.

Using the unmanaged 450-year-old forests of Douglas fir in the Pacific Northwest, Mr. Maser and Dr. Trappe demonstrated that dead wood was far more than mere waste or a fire hazard to be removed as quickly as possible. Rather, they showed that dead trees were very much a part of the living forest.

"A dead or fallen tree is simply an altered state of a live tree and has hundreds of years of contribution it can make to the earth," Mr. Maser said. "The big question now is how much wood needs to be left in the landscape as a biological reinvestment in the land that supports us all."

Once a tree falls, it passes through five distinct phases of decay, they wrote. At each stage, the tree supports new life for which it is the sole or principal habitat.

At stage 1 are newly fallen trees with intact bark, a condition soon to be changed as bark and wood-boring beetles tunnel through. These brazen beetles blithely disregard the chemical and mechanical defenses of the conifer's bark that discourage most insect predators. The first beetles create channels for their successors. The beetles also carry in fungi and bacteria that provide food and essential nitrogen for future invaders.

At stage 2, trees still retain bark but as the beetles feast away, the nutritious growing layer of inner bark and the nearby phloem, which transported sugars, become spongy. These tissues are likely to be eaten in a few years. Next in line is the sapwood, which in the living tree housed the water-carrying structures called xylem.

By stage 3 the bark sloughs off. Roots from sprouting seeds now invade the sapwood, and the trunk begins to break into large, solid pieces. In a fallen Douglas fir, the sapwood succumbs to insects and fungi in 10 to 20 years, Dr. Trappe said, although the bark of this tree "probably hangs around for centuries."

At stage 4 the heartwood, composed of the dead xylem that forms the bulk of the tree trunk, is all that remains. It now breaks apart into soft blocks as roots invade this dense, highly resistant and not very nutritious wood. This is the stage, the longest in the decay process, that hosts the most diverse array of wildlife, including mites, centipedes and snails, as well as salamanders, shrews and voles.

Finally, in stage 5, the tree is no more than a soft, powdery mass. Ashes to ashes, dust to dust, soil to soil. A Succession of Life

Stocked with nutrients, a fallen tree supports more life than when it was alive. Invading fungi ooze out enzymes that liberate the tree's nitrogen for use by other organisms. More nitrogen is provided by bacteria that extract it from the air. The tiny organisms that inhabit the log fertilize it with their excrement. Leaf litter and rainwater laden with nutrients and lichens from the forest canopy fall on the dead tree, adding further enrichment.

Carpenter ants are most active in stage 2. Their catholic diet includes butterflies and the honeydew of aphids. Nesting in fallen logs, they carry nutrients into the tree from the outside. Termites take over late in stage 2, importing in their wood-chomping bodies both protozoa that digest cellulose and bacteria that capture atmospheric nitrogen. By the time a termite colony is ready to move on, it has created a labyrinth of passageways in the tree that can be used by other animals and by the roots of invading plants.

As logs reach stage 3, their bark and sapwood is sloughed off and plants have taken root. The logs become ready for occupation by a wide range of animals. As Mr. Maser and Dr. Trappe wrote about the trees when they reach stage 4: "Various mites, insects, slugs and snails feed on the higher plants that become established on the rotten wood. These plants also provide cover for the animals, as do the lichens, mosses and liverworts that colonize fallen trees."

In this microenvironment, mites thrive on the dead plant and animal matter that accumulates on fallen trees. The skeletons of dead mites, in turn, serve as incubators for fungal spores, and the fungi provide sustenance for other invading plants and animals.

The folding-door spider is among the many arthropods that thrive in these conditions. It constructs a silky tube in one of the many cracks in the outer layer of a fallen tree that has reached stage 3 or 4 of its decay. The outer edges of the tube are pulled inward to form a slitted cover and the spider waits on the inside for the arrival of suitable prey, which are abundant in the decaying wood. Diversity or Monoculture?

Among the ecologically important denizens of fallen Douglas fir is the California red-backed vole. The rodent eats mostly fungi and lichens but has a particular passion for truffles, Mr. Maser has shown. The vole then disperses the spores of the truffle, inoculating decaying trees with this valued foodstuff. This benefits other truffle-eaters, including the squirrels and mice that are the principal foodstuffs of the spotted owl and other carnivores.

"The spotted owl debate is not a case of owls versus people," Dr. Trappe said. "It's a question of whether we want the diversity of organisms that the natural forest provides, or in its place a monoculture in which many organisms will disappear, not just the spotted owl."

If Dr. Mark E. Harmon, a forest ecologist at Oregon State University, has his way, dead trees as well as living forests will become valued as critical elements in containing global warming. When a tree is cut and processed into paper or a fallen tree turned into firewood, carbon dioxide is ultimately released into the atmosphere. "But a dead tree left on the forest floor holds onto its carbon for decades, even centuries," he explained.

Dr. Harmon is directing a project whose lofty time horizon rivals that of the earthmade plaque sent aboard the spacecraft Pioneer 10 to Jupiter and beyond. More than 500 logs of four different species have been placed throughout the H. J. Andrews Experimental Forest outside Eugene, and their patterns of decomposition are to be studied over the 200 years they will take to decay. Biologists will monitor the insects and microorganisms that colonize the logs, the small plants and large trees that become established on them and the birds, reptiles and mammals that use them as dwellings and food sources.

In a parallel experiment on two sides of the Cascades, 800 large trees were felled in 1987 and 1988 and placed in streams. Dr. James Sedell, an aquatic biologist at Oregon State University, said the project had already restored habitats for juvenile coho salmon and steelhead trout. Fallen Logs and Streams

"When a large log falls in a stream, the current scours out a pool around it and other wood gets trapped to form a debris jam," Dr. Sedell said. Fish then go into the pool, which serves as a safe harbor during winter floods and a secure habitat in summer droughts, he explained. The next step is to see if more fish leave the stream and grow up in the sea.

"I'm optimistic," the biologist remarked. "Worldwide there's been much more interest in the role of wood in rivers and streams. The Forest Service and several states have begun to recognize that on forested land they need to allow big fallen logs to remain in streams to protect the fish resources."

Now, he and other scientists say, the question on land and water is: How much dead wood must be kept to bring back the many habitats needed to sustain the diversity of life on earth?

A version of this article appears in print on March 24, 1992, Section C, Page 1 of the National edition with the headline: In Spring, Nature's Cycle Brings a Dead Tree to Life.

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Dead Trees: Unveiling Their Hidden Role in Sustaining Biodiversity

When one encounters a forest populated with seemingly lifeless trees, their initial reaction might be one of sadness or despair, interpreting the sight as an emblem of deterioration or ruin. Such a perspective, however, is markedly incorrect. Deceased trees, whether they remain upright or have collapsed to the forest floor, are integral components in the maintenance and propagation of biodiversity within forests and other ecosystems. They are, surprisingly, a hub of bustling activity and vitality.

As a tree makes the natural progression from living to dead, it experiences a metamorphosis that confers benefits to a broad spectrum of life forms. This transformative journey encompasses stages such as tree decay and decomposition, which inherently cultivate environments suitable for the habitation of myriad species. This cycle significantly enriches the biodiversity of the ecosystem, subtly supporting a wealth of life often unnoticed or unappreciated by the casual observer. In this discourse, we seek to illuminate this underappreciated, yet fundamental role that dead trees fulfill within our ecosystems.

The prevalent perception of dead trees as mere 'waste' or 'debris' is a misguided view that requires rectification. Through a deeper examination of **the ecology surrounding dead trees**, an understanding of their ecological functions, and an exploration of **the importance of deadwood**, we hope to demystify the essential role these seemingly lifeless entities play in underpinning and nourishing life.

Indeed, these lifeless sentinels that punctuate our landscapes are far from being inert or redundant. Rather, they form a unique ecological niche that supports and nurtures a rich tapestry of life. They create habitats, act as food sources, and even influence the forest's microclimate. To truly appreciate and protect our natural ecosystems, we need to recognize and appreciate the indispensable role that dead hardwoods like oak, birch, and ash trees play within them.

Why Dead Trees Matter: The Ecological Function of Dead Trees

The ecological contributions of dead trees are multifaceted, broad, and consequential, extending far beyond what may initially meet the eye. A crucial role that these seemingly lifeless giants perform is acting as sanctuaries and food reserves for a diverse array of organisms. The spectrum of life forms that find haven and nourishment in dead trees encompasses birds, mammals, insects, and fungi, among other species.

Standing dead trees, often referred to as snags, offer valuable nesting sites for a variety of bird species, including woodpeckers, owls, and nuthatches. Apart from providing a secure abode, these trees also become a food source that supports a complex food web. The **decaying wood attracts a host of insects**, which subsequently entice insectivorous birds, contributing to a dynamic and self-sustaining ecosystem. Moreover, dead trees, irrespective of whether they are upright or have tumbled over, are crucial for fungi. Fungi, often overlooked, play a pivotal role in the decomposition process. They break down the deadwood, converting it into essential nutrients that serve to enrich and replenish the forest soil.

In addition to their role in the sustenance of biodiversity and nutrient cycling, **dead trees are also key players in the global carbon cycle**, thereby contributing to climate regulation. As trees mature, they absorb substantial amounts of carbon dioxide from the atmosphere, thus mitigating greenhouse gas emissions. **When a tree dies**, it does not immediately release this stored carbon back into the atmosphere. Instead, the carbon remains sequestered within the structure of the tree, thereby continuing to act as a carbon sink and helping to alleviate the impacts of climate change.

These unique functions of dead trees, from providing habitats and food resources to playing a significant role in carbon storage, highlight their indispensability within our ecosystems. It underscores the need to recalibrate our perception of dead trees from mere landscape 'waste' to invaluable **contributors to biodiversity**, nutrient cycling, and climate regulation. Dead trees, far from being symbols of decay and death, are, in fact, symbols of life and continuity, silently playing their part in the grand scheme of nature's intricate web.

Dead Tree Ecology: Tree Mortality Factors and Tree Decay

The factors contributing to tree mortality are diverse, encompassing both natural processes and human activities. These include natural aging of the tree, susceptibility to diseases, infestation by pests, as well as anthropogenic influences such as logging and urban development. Irrespective of the trigger, the demise of a tree sets off a cascade of ecological processes, with tree decay being the initial phase.

The process of tree decay is instigated by the activity of fungi and bacteria, initiating the breakdown of the wood structure. This biological activity results in the formation of cavities and hollows within the tree's body. These natural structures present in dead trees offer essential shelter to a wide array of species, acting as safe havens for everything from nesting birds to small mammals, insects, and even other plant species. Over time, the decaying wood becomes a complex labyrinth of cavities and tunnels, providing diverse niches for a variety of species, thus enhancing the biodiversity of the forest.

Simultaneously, as the decomposition continues, **the once upright and robust tree gradually morphs into a source of deadwood.** This transformation not only enriches the forest floor with organic matter but also plays a significant role in the nutrient cycle of the forest ecosystem. Deadwood acts as a nutrient reservoir, gradually releasing the stored nutrients back into the soil as it decays, thereby nourishing the surrounding vegetation and promoting the growth of new life. Therefore, a dead tree's journey from standing sentinel to nurturing the forest's soil reveals the enduring ecological importance of these silent, lifeless giants.

Standing Dead Trees and Snags: Unseen Wildlife Habitat

Standing dead trees, also known as snags, are undeniably integral to the well-being and continuity of wildlife. They act as multifunctional **high-rises in the forest**, offering nesting, roosting, and hibernation sites for a wide array of species including birds, bats, and myriad insects. These vertical ecosystems provide numerous cavities and crevices that serve as unique wildlife habitat trees, offering shelter and refuge to a multitude of species. Snags become homes, nurseries, and hiding spots for a diverse set of creatures, echoing with life despite their apparent lifelessness.

In addition to offering refuge to wildlife, snags perform an important role in promoting the resilience and continuity of forest ecosystems. As snags decay, they become catalysts for forest regeneration. The decomposition process slowly releases stored nutrients back into the soil, which in turn fuels the growth of tree seedlings and other vegetation. This nutrient recycling system **helps to nourish the forest floor**, creating an ideal setting for the emergence of new life.

By offering a wildlife haven and supporting the regeneration of forest ecosystems, snags contribute immensely to the dynamism and diversity of the forest. Their decay cycle is a perfect illustration of the phrase **"life springs from death."** Indeed, the story of snags is one of the most compelling narratives in nature. It paints a vivid picture of how the cycle of life and death in the forest intertwines, each phase seamlessly leading into the next, ensuring the resilience and richness of the **forest's biodiversity.** It reminds us that even in death, trees continue to give life, underscoring the interconnectedness of all living things within the intricate tapestry of the forest ecosystem.

The Importance of Deadwood and Fallen Trees in Biodiversity

The transition of a tree from a towering entity to **fallen timber gives birth to deadwood**, an asset that is crucial to the maintenance and promotion of biodiversity. Deadwood is not as lifeless as it might appear; it is a dynamic habitat teeming with a multitude of organisms, including **insects, fungi, mosses, and lichens.** These inhabitants play a vital role in decomposing the wood, thus triggering a series of ecological processes that culminate in nutrient cycling within the ecosystem. As they break down the wood, they unlock its stored nutrients, making them available once again for uptake by plants and other organisms, thereby completing the nutrient cycle.

In addition to being hotspots of biological activity, fallen trees significantly contribute to the physical structure of ecosystems. They morph the terrain, creating microhabitats and altering the landscape in ways that can benefit a host of organisms. The presence of a fallen tree can provide cover and **habitat for ground-dwelling animals**, modify water flow, and influence soil properties. For instance, a fallen log can create a dam in a stream, influencing water flow and creating new habitats for aquatic species.

Deadwood and fallen trees, far from being mere detritus, are foundational elements within the ecosystem, playing a crucial role in both its biological and physical structure. They foster biodiversity, enhance nutrient cycling, and contribute to landscape diversity, thereby bolstering the overall health and functionality of the ecosystem. Recognizing their value is integral to fostering an understanding and appreciation of the complexity and interconnectivity of forest ecosystems and underscores the importance of including these elements in conservation and management efforts.

Tree Decomposition and its Impact on the Ecosystem

The decomposition of trees is a gradual yet pivotal process that profoundly influences the functioning of ecosystems. Initiated by the actions of fungi, bacteria, and various invertebrates that feast on the wood, this process results in the breakdown of the tree into smaller constituents. Over time, these remnants of once-majestic **trees get assimilated into the soil**, transforming them from sturdy physical structures into foundational elements of the earth itself.

The release of these nutrients from decomposing trees significantly enhances the fertility of the soil, providing a nutrient-dense base that supports and fosters the growth of a myriad of plant species. The enriched soil becomes a nurturing ground, allowing seeds to sprout and plants to flourish. The **process of decomposition**, therefore, is central to sustaining the vitality and dynamism of forest ecosystems.

Moreover, tree decomposition is not solely about nutrient cycling; it also plays a substantial role in carbon cycling. As the trees decompose, they gradually release the carbon they had stored during their lifetimes, a process that has implications for climate regulation. By storing carbon and slowly releasing it over time, decomposing trees contribute to the mitigation of climate change by serving as carbon sinks. Consequently, **tree decomposition**, though often overlooked, is a critical ecological process that supports the functionality of ecosystems at multiple levels. It underpins nutrient cycling, contributes to carbon cycling, and fundamentally shapes the structure and health of the ecosystems it occurs in.

Dead Trees as Habitats: The Role of Tree Rot

Tree rot, a phenomenon stemming from tree decay, is instrumental in converting lifeless trees into vibrant habitats teeming with diverse wildlife. The rot leads to the creation of cavities and hollows within the tree structure, carving out spaces that become homes for an array of species, such as birds, bats, and insects. These natural formations within the **rotting trees** transform them into veritable high-rises in the forest, providing shelter for various species that depend on these structures for survival.

These converted abodes, known as wildlife habitat trees, act as multifunctional sanctuaries, serving as nesting, breeding, and hibernation sites. They contribute significantly to the sustenance of wildlife populations, playing an essential role in the preservation and promotion of biodiversity within forest ecosystems. Birds lay eggs and raise their young in these sheltered spaces, bats find secluded spots to hibernate, and numerous insects make their homes within the rotting wood, thus ensuring the continuation of their respective species.

Furthermore, the presence of rotting wood accelerates the decomposition process, which is crucial for **nutrient cycling in the forest.** As the wood decays, it gradually releases nutrients back into the soil, providing essential nourishment for the growth of surrounding vegetation. This process is integral to maintaining the health and resilience of the ecosystem, ensuring that the cycle of life continues unabated. Thus, the phenomenon of tree rot, though it may seem destructive, is in fact a vital and beneficial process that fosters biodiversity, nutrient cycling, and the overall resilience of forest ecosystems. Through these processes, even in death, trees continue to play an active and indispensable role in the vibrant tapestry of life in the forest.

Dead Tree Conservation: The Need for Dead Tree Retention and Restoration

Considering the profound **ecological significance of dead trees**, it becomes evident that dead tree conservation is a necessity, not an option. This conservation includes two main strategies: dead tree retention and restoration. Both these strategies are paramount for the maintenance of biodiversity and ensuring the smooth functioning of the ecosystem.

Dead tree retention is the practice of leaving dead trees in their natural habitat, allowing them to undergo a natural process of decay and decomposition. This strategy is akin to adopting a hands-off approach, allowing nature to run its course without human intervention. By doing so, we ensure the preservation of the myriad ecological processes associated with dead trees. These processes range from providing a diverse array of habitats for various wildlife to facilitating nutrient cycling by decomposing into the soil, enriching it and paving the way for new growth.

On the flip side, dead tree restoration involves the deliberate reintroduction of dead trees into areas where they have previously been removed, typically due to human activities like logging or urban development. By restoring dead trees to these areas, we help replenish the ecological functions that these trees embody, aiding in the restoration of a balanced and diverse ecosystem.

Each of these strategies - retention and restoration - forms a crucial aspect of **dead tree conservation**, playing a unique role in preserving biodiversity, promoting ecosystem health, and building resilience against environmental stressors. These conservation practices also serve as an important reminder of the intricate interdependencies within nature, and our responsibility to respect and protect these complex relationships for the health of our planet and future generations.

Management and Monitoring of Dead Trees

Effective stewardship and surveillance of dead trees are integral for preserving their invaluable ecological functions. Management practices should be geared towards retaining dead trees within their natural environment whenever feasible and ensuring their undisturbed and natural decay. This approach respects the natural process of decomposition and fosters the wide array of ecological benefits that derive from it.

Monitoring the state of dead trees in an ecosystem is equally vital. This process offers insights into the overall health of the ecosystem and elucidates the role that dead trees play within it. Regular monitoring of the condition and impact of dead trees can help detect shifts and changes in the ecosystem that may otherwise go unnoticed. This continual observation aids in identifying patterns, understanding ecological dynamics, and determining if and when intervention is necessary.

By implementing regular monitoring, we can ensure the informed management of dead trees, allowing for evidence-based decision-making that contributes to the sustainable management of forests. This practice can help to safeguard the crucial contribution of dead trees to biodiversity and the overall health of ecosystems. Regular monitoring and management not only preserve the biological integrity of our forests but also help us better understand the intricate web of life these forests support. In this way, we ensure that dead trees continue to play their essential role in our ecosystems, thereby supporting the intricate balance of nature.

The Controversial Practice of Dead Tree Removal

The practice of dead tree removal is frequently undertaken due to aesthetic preferences or safety considerations. However, this practice is met with controversy due to the pivotal ecological role that dead trees fulfill. Eliminating these arboreal life stages disrupts a suite of interconnected processes that hinge on their existence, including the provision of habitats for a plethora of species, the cycling of nutrients, and the storage of carbon, a key factor in mitigating climate change.

Safety concerns, especially in areas frequented by people, such as parks, residential neighborhoods, or near infrastructure, are indeed legitimate. Dead trees, particularly those that are standing or partially standing, can pose a risk if they were to fall unexpectedly. However, it's imperative to harmonize these concerns with **the ecological significance of dead trees.** It's a delicate balance, requiring careful assessment and thoughtful decision-making to meet both human safety needs and ecological considerations.

Where circumstances allow, and the risk to human safety is negligible, dead trees should ideally be left undisturbed to decay naturally. This allows them to provide their multitude of ecological benefits to the surrounding environment. By adopting such a balanced approach, we can ensure the safety and aesthetic appeal of our surroundings while also preserving the ecological integrity of our natural environments. In doing so, we honor the critical role that all **stages of a tree's life** - including its death - play in supporting the intricate web of life that constitutes our precious ecosystems.

The Future of Dead Tree Management and Conservation

The future of dead tree management and conservation hinges on our recognition and appreciation of the critical roles these seemingly lifeless structures play in the health and functionality of our ecosystems. A profound shift in our perception is needed: rather than viewing dead trees as mere 'waste' or unsightly remnants, we need to understand them as vital pillars supporting biodiversity and ecosystem resilience.

The integration of strategies such as dead tree retention and restoration into our conservation efforts is paramount. Coupling these strategies with efficient management and rigorous monitoring, we can ensure that dead trees continue to fulfill their myriad ecological roles, from serving as wildlife havens to enriching the soil through nutrient cycling.

The time has come to grant dead trees the acknowledgement they warrant. They should not be seen as symbols of death or decline but rather as sustainers of life, silently playing their part in the grand cycle of nature. These **silent sentinels of the forest**, whether standing tall as snags or lying as fallen logs, harbor a rich tapestry of life within them and foster the continuous interplay of life and death that is at the heart of all ecosystems. Recognizing and honoring their role is an essential step towards a more holistic understanding and stewardship of our planet's precious biodiversity.

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Protect, manage and then restore lands for climate mitigation

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Limited time and resources remain to constrain the climate crisis. Natural climate solutions represent promising options to protect, manage and restore natural lands for additional climate mitigation, but they differ in (1) the magnitude and (2) immediacy of mitigation potential, as well as (3) cost-effectiveness and (4) the co-benefits they offer. Counter to an emerging preference for restoration, we use these four criteria to propose a general rule of thumb to protect, manage and then restore lands, but also show how these criteria explain alternative prioritization and portfolio schemes. This hierarchy offers a decision-making framework for public and private sector actors to optimize the effectiveness of natural climate solutions in an environment in which resources are constrained, and time is short.

We need drastic reductions in emissions to—and increased removals from—the atmosphere to avoid the worst effects of climate change[1]. Reducing fossil fuel emissions is the most critical action[1–4], but natural climate solutions (NCS) are also required to meet this goal[5]. The latter are 'additional' land-stewardship actions that capture or reduce greenhouse gas (GHG) emissions by protecting existing ecosystems, improving the management of working lands or restoring natural ecosystems[6–8]. Unlike emergent technologies, such as the direct air capture of carbon dioxide (CO_2), NCS are often lower cost, more readily available and can improve air, soil and water quality[9].

Here we propose the 'NCS hierarchy' as a framework for public and private sector decision-makers that suggests considering NCS related to protection, improved management and then restoration when prioritizing among different NCS (Fig. 1). Despite the need for—and recent indications of—an increased investment in NCS to respond to the urgency of climate change, resource constraints remain and decision-makers need to select among options. We describe a general hierarchy based on four principal criteria: (1) the magnitude and (2) immediacy of mitigation potential, (3) cost-effectiveness and (4) co-benefits. However, we note that protection, improved management and restoration NCS are not mutually exclusive; in planning and practice, these actions can be highly complementary[10]. As the priorities at the national and local scales depend on context (for example, biophysical, political, institutional, economic and socio-cultural factors), we also show how this framework provides a process to improve the overall impact of climate mitigation efforts, rather than a rigid set of prescriptions.

NCS hierarchy

Natural resource management has utilized mitigation hierarchies for over a century, stretching back to the conservation and preservation theories of Pinchot and Muir, respectively[11]. In 2012, a mitigation hierarchy (hereafter 'biodiversity hierarchy') was formalized to mitigate the negative effects of economic development projects on biodiversity and ecosystem services[12,13] and to support global biodiversity conservation[14]. The first three steps of the biodiversity hierarchy are (1) avoid negative impacts to biodiversity, (2) minimize unavoidable impacts and (3) remediate negative impacts by restoring the affected sites or species. Recently, the Science Based Targets Network, a collaboration of non-governmental organizations, business associations and consultancies, developed a hierarchy to help private and public sector entities advance general sustainability goals[15]. Their version, the AR^{3}T framework (avoid, reduce, regenerate, restore and transform), effectively shares the first two steps of the biodiversity hierarchy but differentiates actions that 'remediate' (in the case of the biodiversity hierarchy) into those that improve the ecosystem functions within the existing land uses ('regenerate') from actions that fully re-establish natural cover in places previously converted ('restore'). The Science Based Targets Network also included a transform category, which we acknowledge is essential but do not further expand on here. Transform actions include system- and jurisdictional-wide changes needed to tackle large-scale environmental problems (for example, granting and enforcing tenure rights) that are additive rather than sequential to the AR^3 steps[15].

Our NCS hierarchy focuses specifically on reducing GHG emissions or increasing carbon sequestration with constraints to ensure no negative impacts on biodiversity or human well-being[8,16]. NCS readily align with the biodiversity hierarchy and the AR^{3}T framework (Supplementary Fig. 1). Protection NCS avoid emissions from the conversion of forests, grasslands or wetlands, or from changing wetland hydrology (for example, when salt marshes are diked)[8]. Improved management NCS minimize and/or reduce emissions in working agricultural and forest lands. Improved manure management reduces methane emissions[6] and improved timber felling techniques reduce damage to the residual forest stand[17]. Improved management NCS can also regenerate carbon pools when, for example, cover crops increase soil carbon sequestration[7]. Finally, restoration NCS remediate and/or restore forest, wetland and grassland cover where those ecosystems historically occurred. We note that 'restoration' can also describe a diverse suite of actions to recover

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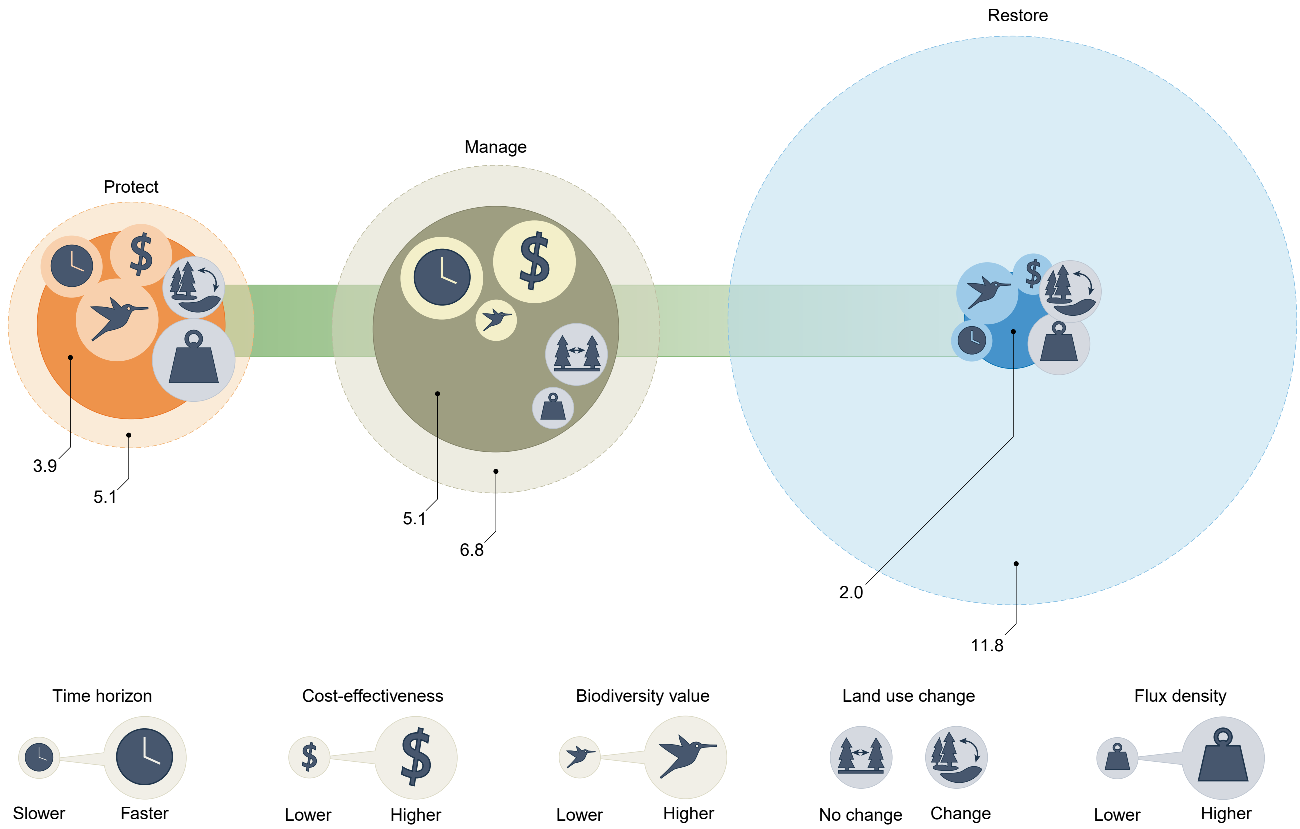

Fig. 1 | The NCS hierarchy starts with the protection of ecosystems and flows to improved management and restoration. Although the NCS hierarchy describes an order, in practice, protection (orange), improved management (green) and restoration (blue) can be complementary and part of a portfolio of NCS to optimize climate mitigation outcomes. The dashed circles indicate the global maximum mitigation potential and the solid circles indicate the mitigation potential at ≤US$100 tCO_2e^{-1} with numbers to indicate estimated $GtCO_2e\,yr^{-1}$ in 2030. Icons are described in the legend at the bottom and a larger size indicates more positive outcomes (for example, faster time horizon or higher cost-effectiveness). Note that the biodiversity benefits reflect on-site per hectare benefits and are somewhat hypothetical in the absence of a systematic review of biodiversity outcomes across NCS. We include icons to show whether land use change is required and relative flux density per hectare, which are additional to the four criteria we describe but influence the order of the NCS hierarchy. Credit: Vin Reed.

degraded, damaged or destroyed ecosystems[18]. This broader definition could encompass some improved management NCS, so here we use 'restoration' narrowly to describe recovering an ecosystem that has been lost. Restoration NCS do not include replacing native with non-native ecosystems (for example, the afforestation of natural grasslands), which have negative biodiversity consequences and ultimately limited mitigation potential[19].

The need for a NCS hierarchy

As conservation practitioners and scientists at three international non-governmental organizations, we often observe land-based climate mitigation strategies that prioritize restoration over improved management or protection. In the public sector, for example, the Canadian government announced a notable Can$3.8 billion investment in NCS over the next 10 years, allocating 81% to restoration (that is, planting 2 billion trees), but only 3% to improved land management and 16% to protection[20]. This relative allocation contrasts with recent research, which suggests that protection and improved management NCS offer the most cost-effective options for nature-based climate mitigation in Canada[7]. More broadly, countries that include the land sector in their nationally determined contributions to the Paris Agreement tend to include protection, afforestation and forest restoration, rather than the improved management of ecosystems[21]. This restoration tilt is also evident in forest sector commitments, as 42% of countries include afforestation and reforestation, 38% include forest management and 32% focus on avoided deforestation[22].

The private sector shows similar patterns. The Carbon Removal Corporate Action Tracker includes 93 corporate pledges and shows that—among those that provide detail on the NCS actions—78% mention restoration, 41% mention protection and 43% mention improved land management[23], although this tool is biased towards corporations that pledge removals. In contrast, land sector emissions from corporate supply chains stem principally from land conversion and management, and thus reducing these activities is critical to decrease supply-chain climate impacts[21,24]. Over 400 companies have pledged to remove deforestation from their supply chains, but with little progress to date[25], and in the meantime there has been a surge of corporate tree-planting commitments[4]. Further, notable corporate commitments have prioritized removals rather than reduced emissions (for example, ref. [26]). However, as we describe below, failure to consider the full range of NCS systematically and comprehensively will unnecessarily constrain efforts to address global warming.

Four criteria of the NCS hierarchy

Four interrelated criteria influence the general order of the NCS hierarchy and can explain variations to the hierarchy in practice and by location (Fig. 1). These criteria are (1) the size of mitigation potential, (2) cost-effectiveness, (3) time horizon and (4) co-benefits. Additional factors may drive NCS feasibility in a given geography, such as technical constraints, availability of ecosystems to conserve or manage and/or policies or regulations that incentivize or disincentivize NCS adoption[3,27]. Further, the preferences and development needs of local communities are critical and will influence the durability of a NCS intervention[4,28]. However, here we focus on criteria that are quantifiable at the global level, recognizing that other factors are best assessed at local levels with impacted communities ultimately determining priorities for climate action.

The size of the mitigation potential (typically estimated in million metric tons of CO_2-equivalents per year ($MtCO_2e\,yr^{-1}$)) refers to removed carbon or reduced GHG emissions that, importantly, are additional to a business-as-usual baseline. Additionality means, for example, that protection NCS should focus on ecosystems at a high risk of losing carbon stores[29]. Mitigation potential is a product of the extent of opportunity (for example, number of hectares restored or heads of cattle affected) and the change in GHGs (CO_2e) per unit extent[8]. The latter we term 'flux density', in which a positive flux density is a reduction in emissions or an increase in removals relative to business as usual. Thus, a high potential can be a function of a large extent, a high positive flux density or both. We note that mitigation potential is most useful for selecting NCS or geographies with a high opportunity to achieve scale. For example, reforestation offers more than six times the mitigation potential of avoided forest conversion in the United States[6], which suggests that the former may be better suited for initiatives with large-scale ambitions. In contrast, flux density is better suited for project-level decisions to identify the NCS or geographies with the greatest mitigation returns per unit extent.

An additional criterion is cost-effectiveness, which measures the resource investment required per tCO_2e reduced. Relevant costs include the sum of (1) net cost to land managers (the sum of the initial NCS implementation costs, opportunity costs—foregone profits associated with switching land uses—and transaction costs) and (2) implementation and transaction costs to others whose actions are also required (for example, government programmes to conduct outreach or enrol land managers). Here we estimated cost-effectiveness as the mean marginal abatement cost (MAC) per NCS by reanalysing data from prior publications[6–8] (see Supplementary Methods). NCS can range from highly cost-effective (low, zero or even negative net cost per tCO_2e, for example, when changes in agricultural practices increase farm profitability) to marginally cost-effective (high net cost per unit GHG mitigation, for example, urban tree planting)[6,8,30] (Fig. 2 and Supplementary Figs. 2 and 3). With limited resources, it makes sense to target the most cost-effective NCS first. However, the pool of NCS with a low cost per unit mitigation can be limited, which requires consideration of options with higher unit costs. In Canada, for example, only one-third of the total mitigation potential is estimated to be available at ≤Can\$50 tCO_2^{-1} (ref. [7]). High costs may also be due to labour-intensive NCS projects, which represent opportunities to create jobs as part of green recovery plans after the COVID-19 pandemic, and governments may prioritize NCS that can stimulate green economic activity. Further, the above costs do not capture non-monetizable values, such as urban shade and mental health benefits.

The time horizon needed to realize a positive change in flux after implementation is another important criterion[3]. For some NCS, changes in flux occur upon implementation or shortly thereafter. For example, manure acidification can rapidly reduce methane emissions[31]. Other NCS take longer to achieve a net positive flux. For example, peatland rewetting releases methane and increases emissions in the short term, but eventually reduces emissions by halting soil carbon loss[32]. Thus, both mitigation and cost-effectiveness depend on the accounting horizon. For example, restoration of forest cover in Canada provides limited mitigation potential within the first decade of planting, but offers the highest potential of all the examined NCS 20 years after tree planting[7]. As this example demonstrates, a long time horizon does not mean a NCS should be avoided as most NCS require acting now to yield meaningful GHG reductions in time to constrain the climate crisis[33].

Here we focus on time horizon, given the need for near-term actions to constrain the climate crisis, but another important temporal component is 'permanence', or the likelihood of reversals due to anthropogenic or natural disturbance[34]. Approaches to deal with project-level permanence risk include buffer pools or discounting to account for the potential project failure. There are also institutional conditions, such as tenure security and benefit-sharing mechanisms that can improve permanence[35,36]. Regardless, mitigation actions with a higher permanence should be prioritized over actions with higher reversal probabilities.

A fourth criterion relates to the ability of each NCS to deliver benefits beyond climate mitigation. NCS can improve human health and livelihoods, support Indigenous cultures, protect biodiversity and increase resilience to future climate impacts[37–40]. Co-benefits can mitigate some feasibility constraints. For example, on-site co-benefits of agroforestry, such as heat mitigation for people and livestock[41], may help to sway potential adopters towards NCS. The financial value of these on-site co-benefits can be captured in the cost-effectiveness criterion because they reduce the opportunity costs for landowners. For example, reduced heat stress from agroforestry can increase livestock productivity to offset the tree-establishment costs[42]. Landowners may also incorporate non-financial co-benefits (for example, reduced human heat stress from agroforestry) into their cost-effectiveness decisions, although these are highly context-dependent and thus difficult to quantify. Compensating landowners for off-site co-benefits may further improve financial feasibility (for example, through payments for environmental services, such as improved downstream water quality[43]). However, trade-offs between climate mitigation and co-benefits are possible. For example, restoring tree cover via plantation forestry may offer a lower-cost climate mitigation than the restoration of native forest, but a lower or negative biodiversity value[44]. Similarly, protecting forests with the highest carbon stores may not protect forests with the highest biodiversity value[45].

These criteria can also help prioritize actions within a single NCS. For example, there are multiple improved forest-management practices (for example, extended rotations, reduced impact logging and partial set asides in planned harvest blocks[8,17]) and multiple ways to restore forest cover (such as timber plantations, agroforestry systems, tree planting to restore native forest and natural forest regrowth[46]). Each of these will vary in mitigation potential, cost-effectiveness, time horizon and co-benefits[47].

Why protection is first

Protection NCS are first in the hierarchy because they can offer a high per-hectare mitigation that can be realized quickly and at a comparatively low cost per tCO_2e, typically with many co-benefits. Protection NCS also align with global commitments to stop deforestation, limit forest degradation and halt biodiversity loss. We describe each of these in more detail below.

Natural ecosystems can store large amounts of carbon, sequester additional carbon and represent more-stable and long-term carbon stores compared with working and restored lands[48,49]. Avoiding the conversion of mature and young secondary ecosystems prevents carbon from being released into the atmosphere and maintains their ability to keep sequestering carbon[50]. Protection NCS can also offer a higher flux density than improved management or restoration NCS (Supplementary Table 1). For example, avoiding mangrove drainage could prevent 29 $tCO_2\,ha^{-1}\,yr^{-1}$ on average[51], which far outweighs the flux density from improved forest management (0.2 $tCO_2\,ha^{-1}\,yr^{-1}$ on average[8]) or natural forest regrowth (13.1 $tCO_2\,ha^{-1}\,yr^{-1}$ on average[46]).

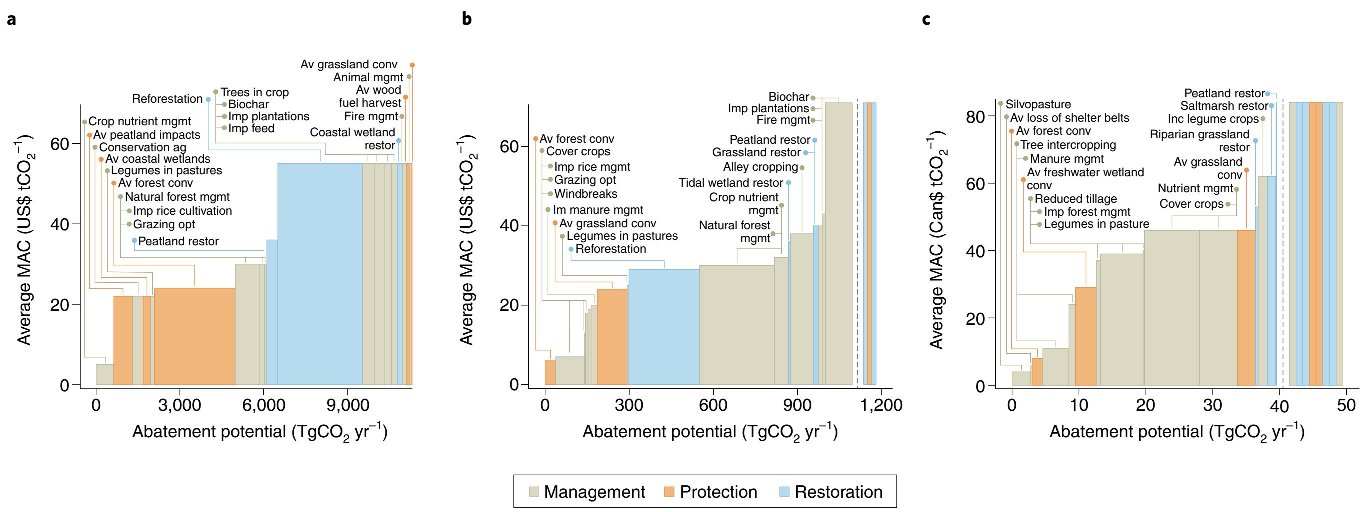

Fig. 2 | Average MAC for NCS available for ≤$100 tCO_2e^{-1}. a–c, Global[8] (**a**) and US[6] (**b**) estimates are in US dollars, and Canadian[7] (**c**) estimates are in Canadian dollars. Comparisons should not occur across geographies given the differences in the underlying methods (Supplementary Methods). The bar height indicates the average MAC and the width indicates the mitigation potential available for ≤$100 tCO_2e^{-1} in 2030 (2025 for the United States). The bars are ordered from left to right by lowest to highest unit cost ($ tCO_2e^{-1}) and then largest-to-smallest mitigation potential. We aggregated labels for visual clarity; when labels share a branch, the left-to-right position of the circular node indicates the left-to-right position in the chart. As most NCS have some mitigation potential at low and high costs, we show how these patterns change if there is no cost threshold (Supplementary Fig. 2) or if the cost threshold is ≤$50 tCO_2e^{-1} (Supplementary Fig. 3). Note that in the United States and Canada, there are NCS with no opportunity at ≤$100 tCO_2e^{-1}, which we indicate with uniformly sized bars to the right of the vertical dashed line, ranked by order when unconstrained by cost (Supplementary Fig. 2). In the United States, these are urban reforestation, avoided seagrass loss and seagrass restoration (left to right). In Canada, these are biochar, urban canopy cover, seagrass restoration, avoided seagrass disturbance, avoided peatland disturbance, freshwater mineral wetland restoration, restoration of forest cover and riparian tree planting (left to right). Ag, agriculture; av, avoided; conv, conversion; crop, cropland; imp, improved; inc, increased; mgmt, management; opt, optimization; restor, restoration.

Protection NCS can also often offer large near-term climate mitigation[49]. Ecosystems can rapidly lose carbon when disturbed, such as when forests are harvested or grasslands are tilled for crops. In many cases, it can take decades to centuries for the carbon to recover. Loss of this 'irrecoverable' carbon is an effectively permanent debit from the remaining global carbon budget for keeping global warming below catastrophic levels[52]. Prioritizing the protection of the irrecoverable carbon stores at risk of disturbance is critical as improved management and restoration NCS will be unable to compensate for this loss on meaningful timescales.

Protection NCS may also offer more cost-effective mitigation than improved management or restoration, although not always (Fig. 2). The latest global estimates, which stem from multiple recent publications[8,46,53,54], suggest that at ≤US$100 tCO_2e^{-1}, protection NCS offer up to 4,245 $MtCO_2e\,yr^{-1}$ in 2030, compared with 2,884 and 3,153 $MtCO_2e\,yr^{-1}$ for improved management and restoration NCS, respectively. This pattern is strongest for the global estimates (Fig. 2), but protection NCS also offer substantial low-cost per tCO_2e potential in the United States and Canada (Supplementary Figs. 2 and 3).

There are many potential co-benefits linked to the protection of ecosystems, such as the protection of Indigenous peoples and local communities' livelihoods and cultures[40], avoidance of extreme heat conditions[55] and reduced negative impacts to coastal communities from rising seas and other coastal hazards[56]. Intact forest ecosystems are noted for their exceptional value with respect to habitat for biodiversity, water provisioning and maintaining human health[57]. Additionally, given the many goals to conserve biodiversity and end deforestation, private and public sector actors can address both biodiversity and climate mitigation goals with the single action of ecosystem protection[25,58]. Protection of forests may also be one of the most cost-effective ways to prevent zoonotic virus spillover to humans, as financing efforts to stop deforestation amounts to just 2% of the cost of the COVID-19 pandemic[59]. However, although protection NCS support in situ biodiversity, the leakage or the displacement of activities from one area to another must be minimized to reap the biodiversity benefit, as well as the climate mitigation benefit[38]. Solutions to minimize leakage include, for example, improved agricultural practices to reverse land degradation and preventing the clearing of forests for new agricultural lands[10], or jurisdictional approaches such as REDD+[60].

Finally, failure to protect native ecosystems can undermine the potential effectiveness of other NCS in the same area. For example, relying on natural forest regrowth to restore forest cover can be cost-effective[61], but it depends on having nearby seed sources[62]. Failure to protect adjacent forests can thus preclude using natural forest regrowth as a climate solution.

Why improved management is next

Improved management NCS are second in the hierarchy primarily because they often offer lower-cost mitigation potential than restoration NCS (Fig. 2). They can also deliver mitigation alongside commodity production and are thus less prone to leakage issues than protection or restoration NCS[38]. However, as noted above, they usually have a lower flux density than protection NCS. Unlike protection NCS, improved management NCS also include carbon removals from the atmosphere—in addition to avoided emissions—and removals may offer lower mitigation when deployed at scale[63,64].

At the global level, improved management NCS account for two-thirds of the mitigation potential available at low cost in 2030 (≤US$30 tCO_2e^{-1}). This pattern is further accentuated in national-level analyses (Supplementary Fig. 3). In Canada, improved management NCS account for 85% of the mitigation potential in 2030 with mean MAC of ≤Can$10 tCO_2e^{-1} and 75% of the potential with mean MAC of ≤Can$50 tCO_2e^{-1}. Similarly, in the United States, improved management NCS account for 75% of the total mitigation potential in 2025 with mean MAC of ≤US\$10 tCO_2e^{-1} and 55% of the total mitigation potential with mean MAC of ≤US\$50 tCO_2e^{-1}.

Improved management NCS also require little-to-no changes in land use (for example, from cropland to forest). Displacing alternative land uses may result in a higher average MAC than estimated here for protection and restoration NCS (Fig. 2), as land cost data used to estimate opportunity costs may be systematically biased low[65,66]. This bias would make improved management NCS even more cost-effective relative to other NCS.

The mitigation potential of many improved management NCS can be realized almost immediately, especially those that reduce emissions ('minimize') rather than increase ('regenerate') stocks. For example, on implementation or shortly thereafter, an improved fertilizer application reduces NO_x emissions[67], and reduced-impact logging techniques can halve logging emissions[17]. However, improved management NCS usually have a lower flux density than either protection or restoration NCS (Supplementary Table 1).

Improved management NCS produce multiple co-benefits. Cover crops can improve soil health and boost yields[68]. Trees along riparian corridors in agricultural lands can help to protect water quality[69] and provide habitat for biodiversity[70]. We hypothesize that the in situ biodiversity benefits of improved management are smaller than those linked to protection NCS or the restoration of native ecosystems (Fig. 1), given the high biodiversity value of intact landscapes[57] and the common intention to restore native biodiversity via restoration projects[71]. However, improved management NCS may have substantial off-site benefits when, for example, improved forest management practices protect the integrity of downstream freshwater and marine ecosystems[72].

Why restoration is third

Restoration NCS have the potential to offer substantial climate mitigation. Indeed, the restoration of forest cover represents the single largest NCS based on the global biophysical potential[8]. However, restoration NCS are third in the hierarchy because failure to protect intact lands where conversion or disturbance pressures are high will release large amounts of carbon that cannot be balanced in a timely manner by the gradual carbon accrual from restoration. Moreover, the mitigation from restoration NCS primarily stems from removals, which may be less effective than avoided emissions at lowering atmospheric GHG concentrations[63,64]. Additionally, restoration NCS can have high costs and feasibility constraints (Fig. 1).

Restoration NCS can be less cost-effective than protection or improved management NCS due to high opportunity and implementation costs. For example, in Canada only three of the eight restoration NCS have any mitigation potential in 2030 at ≤Can\$100 tCO_2e^{-1} (Fig. 2c) and those three restoration NCS have some of the highest average MACs compared with other NCS with mitigation potentials at ≤Can\$100 tCO_2e^{-1}. Similarly, in the United States, restoration NCS account for about 4% of the total mitigation potential in 2025 among NCS with mean MAC ≤US\$10 tCO_2e^{-1}. In the tropics, a spatially explicit analysis of the MAC of avoided deforestation versus reforestation found that the former offered seven to nine times more mitigation potential at ≤US\$20 tCO_2e^{-1} (ref. [54]). Globally, no restoration NCS has an estimated mean MAC of ≤US\$30 tCO_2e^{-1} in 2030 (Fig. 2), even though peatland restoration offers some potential at ≤US\$10 tCO_2e^{-1}.

However, net costs depend on location as well as on the approach used to restore native cover. Lower-cost options can be readily available. For example, relying on natural regrowth where possible rather than active planting can reduce costs by 77% (ref. [73]). Similarly, some lands may have limited value for human uses and thus lower opportunity costs[74], although if these lands are highly degraded they may face higher implementation costs to enable ecosystem recovery. Finally, high material and labour costs associated with active restoration efforts can represent employment and economic opportunities.

From a feasibility perspective, restoration NCS may require shifting land use, which can face a host of cultural, social and economic barriers. For example, some higher-end estimates of mitigation potential from reforestation assume that pasture in historically forested areas can be returned to forest, due to improved efficiencies in livestock production or a shift towards plant-based diets[8,30]. Yet, alternative scenarios are possible where the future extent of the agricultural lands remains constant or increases[75]. Land use will also influence the likelihood of leakage, although restoring degraded lands could have lower leakage risks than lands with a high human land use value. Moreover, landscape-level planning that simultaneously balances multiple criteria, such as climate mitigation, cost and biodiversity conservation, can improve the overall outcomes compared with those of non-systematic planning[76]. For example, coordinated restoration across the Brazilian Atlantic Forest can increase biodiversity benefits by 257%, double climate mitigation potential and reduce costs by 57% compared with those of uncoordinated action by individual land managers[76].

In addition to cost and feasibility constraints, the time horizon for mitigation from restoration is generally longer than those for improved management or protection NCS. Although there are some examples of rapid removals, such as secondary tropical forests recovering in sites conducive to growth, carbon accumulation will take longer in slower-growing forest types or in places with degraded conditions[46]. Similarly, although restoration of inland wetlands can reduce CO_2 emissions relative to those of disturbed sites, it can take decades or centuries to achieve a net cooling effect given the initial releases of methane after restoration[77]. Moreover, sequestration rates of restored wetlands seldom achieve the same level as those of similar natural wetlands[78].

When done properly, restoration has the potential to offer high co-benefits, particularly in regions that have experienced severe loss and degradation of the native vegetation[79]. Restoring tree cover in urban landscapes can capture carbon[6,7], improve air and water quality, and reduce urban heat effects[80–82]. Restoration of forest cover can also provide habitat for biodiversity, as well as an improved flow regulation of water[83]. Restoration of coastal wetlands can protect coastal communities from storm surge and erosion[56]. However, restoration NCS may not achieve the same level of co-benefits as those observed in protection NCS. Restoration often does not bring back the full function of undisturbed ecosystems[79], and longer time horizons and spatial trade-offs can limit the co-benefits[84].

Applying the hierarchy in practice

The NCS hierarchy is a general framework for considering how to prioritize NCS—not a pre-determined outcome of that prioritization. From a climate mitigation perspective, the generic NCS hierarchy is better suited to locations with high land-conversion pressures. For example, Borneo has high deforestation rates due to the expansion of industrial plantations and a need for regulatory and enforcement reforms[85]. Although improved forest management can alleviate some deforestation pressures[86], insufficient protections put all forests—intact, managed and restored—at risk of conversion.

In other locations, it may be possible to deprioritize protection or improved management NCS. For example, in Gabon, deforestation pressures are low and the four criteria that inform location within the hierarchy point towards improved management NCS. Specifically, improved forest management has the largest mitigation potential[87] and is immediately available via reduced-impact logging practices[88], which are low cost and deliver many co-benefits[17].

There are also some countries in which restoration opportunities are much larger than either protection or improved management opportunities[53]. For example, restoring forest cover in Ethiopia could provide up to 22.0 $MtCO_2 yr^{-1}$ in 2030, compared with 8.4 $MtCO_2 yr^{-1}$ for avoided deforestation and 7.5 $MtCO_2 yr^{-1}$ for improved forest management at ≤US\$100 tCO_2e^{-1} (ref. [89]). Further, at US$20 tCO_2e^{-1}, reforestation in Ethiopia offers 25% more mitigation potential than avoided deforestation[54].

Beyond climate mitigation, we also acknowledge that context can result in a prioritization scheme that differs from the NCS hierarchy. For example, it can take years for communities to build up the infrastructure and social capital needed to restore forests at scale[90]. Given the historical momentum, it may make sense to continue to focus on restoration in such a location, while also considering opportunities to protect ecosystems or improve the management of working lands. Ultimately, the ability to act sooner rather than later by adopting the most feasible actions is critical, as the window to constrain warming below catastrophic levels is narrowing[33]. Land-based mitigation actions will be most effective if undertaken in the next decade[8].

Finally, the NCS hierarchy is intended to guide a decision-making process that flows from protection to improved management to restoration, but also considers whether and how to use all three categories. The best approach is probably to advance a portfolio of NCS[91]. Although we partition protection, improved management and restoration, a mix of NCS may be required at the project and landscape levels to achieve mitigation goals. For example, where forests are cleared primarily for agricultural land or wood fuel, avoiding forest conversion typically requires improved management of degraded agricultural lands or restoration of tree cover to meet the needs of local communities[10]. There is no universal panacea to climate change, so a balanced approach across NCS may best optimize carbon and non-carbon benefits, as well as local needs. Further, NCS with a longer time horizon (such as reforestation) must be started now to reap the benefits in a timely manner[33]. Coupling more immediate NCS with those that have a longer time horizon, but higher mitigation potential, may lead to the best long-term outcomes.

Priorities within the NCS hierarchy also probably differ between public and private sector actors. For example, restoration actions, such as tree planting, are a common way to improve corporate image[92,93] and some corporate actors view restoration NCS as an underdeveloped sector of the carbon market that needs investment now to tap its future potential (for example, ref. [94]). Private sector actors may also prefer improved management NCS because they may be subject to fewer social and cultural constraints, such as changes in land use, and involve non-carbon financial returns over time (for example, ref. [95]). However, public actors may be best positioned for protection NCS, given that these often require policy mechanisms capable of addressing landscape-scale issues linked to leakage and social–institutional or policy–regulatory feasibility constraints[96]. However, a coordinated effort across all actors will lead to better outcomes. The Science-Based Targets Initiative has methods for corporates to develop 1.5-°C-relevant mitigation targets that align with global commitments, and will issue specific guidance in 2022 for the agriculture and forestry sectors to address supply-chain emissions from land use.

Finally, carbon credit certifications may also prioritize actions differently than the NCS hierarchy. Additionality is essential. To reduce net emissions and reach neutrality by 2050, the global community must take steps beyond business as usual. However, the additionality criterion risks creating perverse incentives, and may, for example, lead to a preference for restoration over protection NCS if 'doing' restoration is perceived as more additional than 'stopping' conversion or disturbance. Similarly, the additionality criterion may lead to a preference for labour- or resource-intensive restoration if, for example, tree planting is viewed as more additional than natural forest regrowth. Finally, actors in degraded or at-risk systems have the opportunity to demonstrate how additional action would restore and protect natural lands[49]. In contrast, Indigenous or traditional rural communities may be excluded from incentive mechanisms, such as carbon offsets, if their successful and long-standing traditions of protection and land management are not considered additional to the status quo. The solution is to continue to develop accounting and verification methodologies that minimize these perverse incentives, such as those that better-quantify stable carbon stocks in the climate ledger[49].

Comparison of biodiversity and NCS hierarchies

The original biodiversity hierarchy highlighted several key challenges beyond additionality. These included selecting an appropriate biodiversity metric, demonstrating equivalency between biodiversity losses and gains, and identifying the appropriate multiplication factor for biodiversity gains required by offset projects[14]. These challenges are reduced for the NCS hierarchy, in which CO_2e serves as a more universal currency to measure climate impacts within the NCS hierarchy. However, we note emerging research that flags the weaker climate mitigation effect of carbon removals compared with that of avoided emissions when removals are deployed at large scales[64]. Moreover, other factors beyond CO_2e, such as biodiversity and other co-benefits, can and do influence decisions about the merits of different NCS projects.

The biodiversity hierarchy also includes a final 'offset' step in which the remaining negative impacts on biodiversity are supposedly countered by conservation efforts elsewhere[14,97]. We do not include an 'offset' stage within the NCS hierarchy. The NCS hierarchy can be used to prioritize action within a geography or company's supply chain, as well as to prioritize climate offsets elsewhere to compensate for unavoidable emissions. Thus, including an offset step within the NCS hierarchy itself leads to circularity. However, we flag that GHG offsets should (1) only be used after an entity has implemented all the possible emission reductions from their footprint, (2) ensure that environmental issues are not exported from one community or sector to another, (3) equitably and fairly benefit local communities, and (4) be planned with the NCS hierarchy in mind[3].

Conclusion

Preventing catastrophic climate warming will require radical transformation across all sectors—energy, industry, transportation and land[1]. NCS do not replace or delay the deep decarbonization needed to achieve the 2015 Paris Agreement goal to keep global warming well below 2 °C (ref. [98]). However, they do represent a promising set of options to constrain the climate crisis and also help to conserve biodiversity. Although there are many reasons why individual decision-makers may choose to adopt different prioritization schemes, considering the NCS hierarchy will help these land-based options reach their highest potential.

Data availability

Data underlying Fig. 2 and Supplementary Figs. 2 and 3 are available as Supplementary Data.

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Author contributions

P.W.E. proposed the initial idea. S.C.C.-P. further developed the idea and wrote the manuscript. T.K. conducted the economic analyses. S.Y. created Fig. 2 and all the Supplementary figures. All the co-authors contributed ideas and revised the manuscript.

Competing interests

The authors declare no competing interests.

Additional information

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PLOS CLIMATE

REVIEW

Climate-Smart Forestry: Promise and risks for forests, society, and climate

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Abstract

Climate change is presenting a global challenge to society and ecosystems. This is changing long-standing methods to determine the values of forests to include their role in climate mitigation and adaptation, alongside traditional forest products and services. Forests have become increasingly important in climate change dialogues, beyond international climate negotiations, because of their framing as a Natural Climate Solution (NCS) or Nature-Based Solution (NBS). In turn, the term "Climate-Smart Forestry" (CSF) has recently entered the vernacular in myriad disciplines and decision-making circles espousing the linkage between forests and climate. This new emphasis on climate change in forestry has a wide range of interpretations and applications. This review finds that CSF remains loosely defined and inconsistently applied. Adding further confusion, it remains unclear how existing guidance on sustainable forest management (SFM) is relevant or might be enhanced to include CSF principles, including those that strive for demonstrable carbon benefits in terms of sequestration and storage. To contribute to a useful and shared understanding of CSF, this paper (1) assesses current definitions and framing of CSF, (2) explores CSF gaps and potential risks, (3) presents a new definition of CSF to expand and clarify CSF, and (4) explores sources of CSF evidence.

Introduction

For many millennia, forests have provided sustenance, materials, ecosystem services, and cultural values to human societies, who in turn have advanced various interventions to support these values. The long-standing roles of forests as providers are well documented [1–3]. Also documented is wide variation of culturally acceptable tradeoffs in protection, management, and material use of forests [4–6].

Climate change is presenting a global challenge to society and the forested ecosystems society relies on. This climate crisis, arising because of land use change and emissions from production and fossil fuels burning of, changing long-standing forest valuation to include climate mitigation and adaptation, alongside traditional forest products and services. In turn, forests have received scholarly recognition as a so-called "Natural Climate Solution" (NCS) or "Nature-Based Solution" (NBS), meaning society can address climate change through forestry by reducing emissions from forest loss or by removing atmospheric greenhouse gases via photosynthesis and sequestration [7]. Previous NCS assessments have considered the potential of various land cover and management approaches in terms of opportunity scale (e.g., sequestration of million metric tons of CO_2e) and the costs of implementation. Assessments at international [7] and national levels [8, 9] point to currently- or potentially-forested lands as the dominant opportunity for nature-based climate change mitigation strategies.

Policy makers at various scales, from nation states to local governments, are considering and advancing forest and climate policies with broad implications for society and the environment. Forests have become increasingly important in international climate change dialogue, as seen in the Warsaw Agreement [10, 11], Paris Agreement [10, 12] and the recent COP26 Glasgow Leaders' Declaration on Forests and Land Use, which pledges to end and reverse deforestation by 2030 [13], and continued dialogue in COP27 on the role of market mechanisms to link emitters with forest nations via Article 6 [14]. These efforts are 'next steps' to the Kyoto Protocol and Clean Development Mechanism (CDM) [15]. These examples include Reducing Emissions from Deforestation and Degradation (REDD) investments and substantial international investments to measure, monitor, and promote change in global forest trends [16, 17].

In another context, various regulatory and voluntary markets using forest-based carbon credits have been initiated (e.g., European Union Emissions Trading System [18]), sputtered (e.g., Chicago Climate Exchange [19]), or gained traction (e.g., voluntary markets [20]) in the last two decades. Critiques of market-based activities are that they permit continued pollution [21], outsource mitigation activities, and are capitalistic measurement-intensive interventions. Some scholars [22] have further asserted that such attributes benefit only certain program participants, and that they are often the same actors responsible for global emissions. Regardless, forest carbon projects and innovative incentive programming continue to operate and grow, pointing to an increasing acceptance of mechanisms to finance GHG benefits of trees and forests.

The new emphasis on forest-based NCS has a wide range of interpretations and applications. Such divergent interpretations of forest connections with climate change adaptation and/or mitigation can conflict with one another and have already done so. For example, scholarly work has captured tensions between conservation versus utilization [23], issues with carbon commodification [24], and assertions that carbon credits are a form of 'greenwashing' [21]. Further, it brings an increasingly large assembly of policymakers, program designers, natural resource professionals, land managers, and private sector actors interested in developing, selling, buying, and assessing forest carbon credits. Attention to NCS in political, scholarly, public, and private sectors has dramatically altered forest management and sustainability framing in recent years, reshaping a long-standing dialogue about our relationship with trees and forests.

With a diversity of considerations, myriad actors are embracing a phrase that intends to capture a connection between forests, society, and climate: Climate-Smart Forestry (CSF). However, specific definitions for CSF vary widely, with some emphasizing sustainability [25] or economics [26], and others highlighting landscape carbon reserves [27] (see S1 Table for specific examples). As such, CSF is seemingly being applied to a wide swath of activities and interpreted uniquely by each audience, landowner type, and practice.

Considering the complexity of climate change and human relationships with forests, this paper questions whether the term CSF is adequately defined and if some CSF interpretations present new risks to the environment, society, and climate. This paper also seeks to enhance the emerging scholarly discussion on whether forest management can be sustainable without being 'climate smart' and if other forestry activities, including avoided conversion and restoration, are adequately recognized under the umbrella of CSF. To assess, we explore how different actors are included or excluded in current CSF definitions and consider how other values for forests (e.g., biodiversity) relate to so-called 'climate-smart' outcomes.

To contribute to a useful and shared understanding of CSF, the authors have undertaken a literature review, qualitative assessment of documents, and statistical analysis of datasets from related studies. The results are presented in this paper, which (1) assesses current definitions and framing of CSF, (2) explores CSF gaps and potential risks, (3) presents a new definition of CSF to broaden intervention types and engage multiple scales of decision-makers, and (4) explores sources of evidence of CSF.

Current definitions and ideas in CSF, and their linkage to SFM

Use of the term CSF is rapidly increasing in usage in recent years and other examples can be seen across wide-ranging disciplines, from academia [28] to applied practice by policymakers [29–31], planners and builders [26], conservation NGOs [27], and certification body standards ([32]; see S1 Table for these examples and others). This section assesses current definitions of CSF (e.g., interpretations, applications, and principles) as found in current scientific reporting and literature, policymaking, and mainstream media.

Within academic literature, CSF has a range of definitions (see S2 Table). Consider that Web of Science searches for "carbon + forests" returned 132,532 results, "carbon + climate + forests" had 50,697 results, and "carbon + climate + forests + mitigation" returned 17,595 results. In contrast, as of January 2022, "Climate-Smart Forestry" returned just 18 results via Web of Science and Science Direct. Thus, despite a great body of scholarly work on topics intersecting carbon, forests, and climate, the term CSF is relatively new and has been minimally adopted and explored in scientific literature. Further demonstrating the limited scope, no CSF results are earlier than 2017, nearly all are European-focused (15 focused on Europe, 1 in sub-Saharan Africa, and 1 in the Pacific Northwest of the United States), and most pertain to industrial forest management (see, for example, [33])

One recent definition, [25, p 2] defines CSF with the following principles:

1. Increasing carbon storage in forests and wood products, in conjunction with the provisioning of other ecosystem services,
2. Enhancing human health and community resilience through adaptive forest management, and
3. Using wood resources sustainably to substitute for non-renewable, carbon-intensive materials.

With the word 'sustainable' explicit or implicit in most CSF applications, it is relevant to consider previous ideas about sustainable forestry, particularly Sustainable Forest Management (SFM). SFM is an approach, closely linked with the notion of 'sustainable development', that has been a central focus of forestry research since the 1980s and is well documented in scientific literature [34]. SFM has an emphasis on productive forest landscapes, or 'working' forests, thus denoting 'sustainable' in terms of sustained production and the ability to meet the needs of society now and into the future (see S3 Table for definitions relevant to this paper).

In recent years, while additional forest values (e.g., habitat provisioning) have received new emphasis in SFM [35], SFM still largely reflects industrialized, development-oriented framing. Scholars have critiqued SFM for not adequately encompassing socio-cultural values [35] and political ecologists have noted that industrial forestry generally includes utilitarian tactics that favor economic production above other values [36, 37]. Moreover, forestry, as a science, is dominated by ideas developed for and practiced in temperate forest ecosystems, with a focus

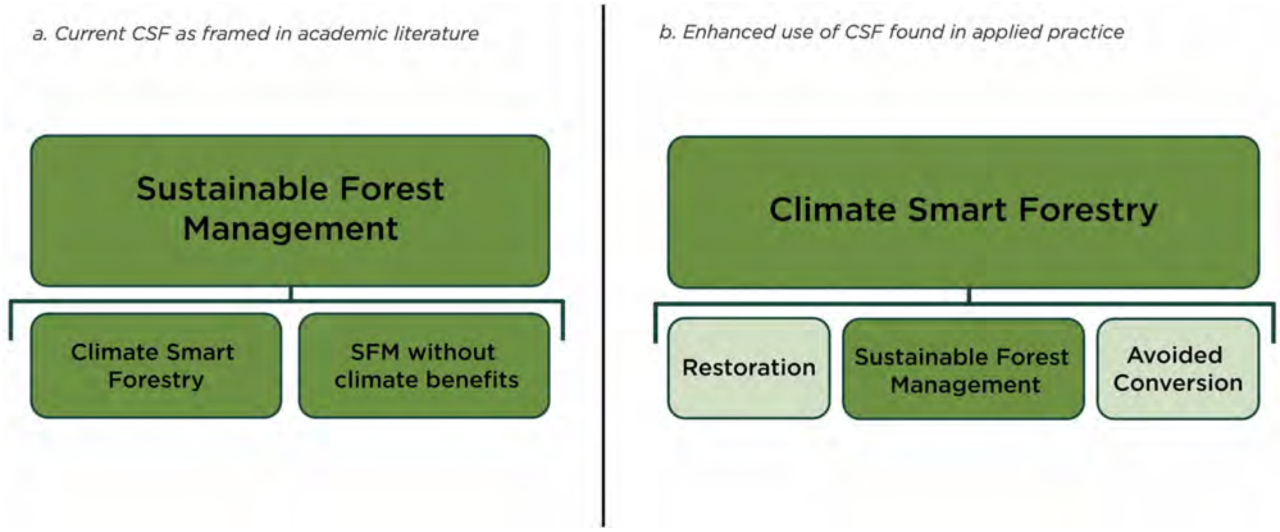

Fig 1. CSF theory examples in literature (a) and in applied practice (b).

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on timber over non-timber products [37]. Still, compared with conventional forest management, SFM is considered more interdisciplinary, inclusive, "less hierarchical", and more "socially accountable" [34, p 205].

Linking CSF and SFM, [28] suggest that CSF is a subset of SFM (Fig 1A), asserting that SFM can be advanced with climate considerations and that the resulting CSF is appropriate on myriad forested landscapes and use types. In their definition, CSF explicitly includes ecosystem services and acknowledges that climate change threatens production which would have previously been assumed under SFM practices alone, acknowledging that previous assurances may no longer be sufficient to ensure long-term outputs (e.g., due to drought or major disturbance). However, this definition implies that SFM can still be accomplished without climate benefits (Fig 1A) and overlooks forests or potentially forested lands that are not managed for productivity. Under an SFM framing (Fig 1A), CSF might be considered as an optional component of SFM. In contrast to *Current* CSF framing described here, this article introduces the *Enhanced* CSF framework (Fig 1B), where SFM is considered just one element of the forest-climate decision portfolio and is explored in more detail later in this paper.

Gaps and potential risks in current CSF

This section explores potential gaps and risks under a current framing of CSF that focuses only on forests managed for production by exploring 'science-practice gaps' and various risks to *Current* CSF framing. It addresses considerations for actors or actions represented in CSF manifestations and considers how bringing in underrepresented values for forests, like biodiversity preservation, or engaging rural communities could improve so-called 'climate-smart' outcomes.

Science–Practice gaps

Use of CSF and related terms, such as *Climate-Smart Forest Economy* [38] or *Climate-Smart Forest Products* [39], are emerging and seemingly rely on an assumption that CSF has been adequately defined and is well understood. This leads to the term being adopted and used colloquially, without critical examination and robust scientific rationale, constituting a 'science-practice gap'. Science, Technology, and Society (STS) scholars have undertaken work in

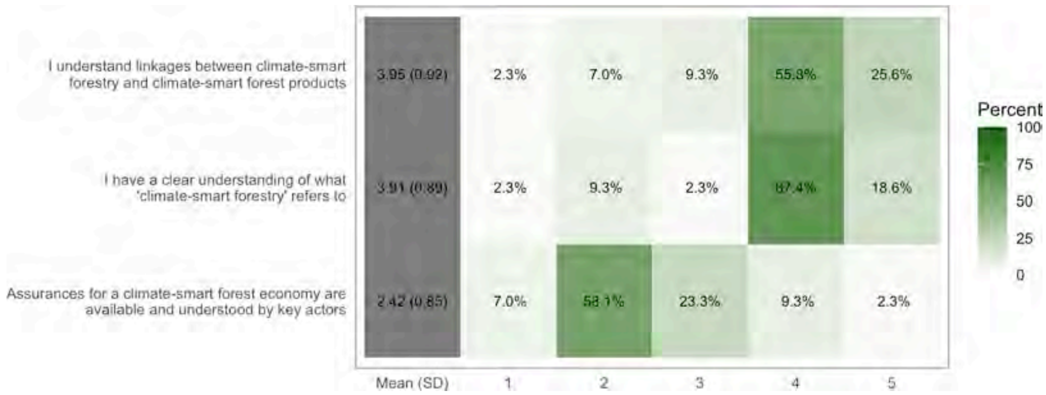

Fig 2. Survey responses from forest, forest product, conservation, and economic development organization professionals. Level of agreement: 1 = Strongly Disagree, 2 = Disagree, 3 = Neutral or I don't know, 4 = Agree, and 5 = Strongly Agree.

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myriad disciplines on such applied and data-driven science research-implementation gaps [40] or knowledge-action gaps [41]. With CSF, these manifest as challenges in interpreting and applying forestry (e.g., growth, carbon, biodiversity, health) and climate (e.g., forest-climate interactions) *sciences* to the *practice* and on-the-ground decision-making of CSF.

To explore perceptions of CSF, consider results from a recent survey distributed to a network of diverse professionals (based largely in North America and Europe) that are affiliated with or in the network of the Climate-Smart Forest Economy Program [42]. These professionals represent organizations that cross forestry, conservation, economic development, sustainability, building and construction. The survey assessed their understanding of CSF definitions and potential assurances for positive outcomes (Fig 2). When asked level of agreement with the statement "*I have a clear understanding of what CSF refers to*", 84% of respondents (n = 44) responded Agree or Strongly Agree. They demonstrated a similar level of agreement (81% responded Agree or Strongly Agree) with "*I understand linkages between CSF and climate-smart forest products*". However, only 26% of those participants agreed that *"Assurances for a climate-smart forest economy are available and understood by actors"*. These results show that the sampled professionals perceived an individual understanding but acknowledged a limited ability to provide adequate assurances to achieve CSF outcomes.

A different dataset derived from pre-course survey responses (n = 178) from domestic and international professionals participating in a United States university-level forest carbon training short course from 2019 to 2021 [43] presents further evidence (Fig 3). In this survey, 94% Agree or Strongly Agree that *"Forest carbon is becoming increasingly important in my profession*" (70% responding Strongly Agree). Interestingly, only 28% Agree or Strongly Agree with *"There is adequate knowledge of forest carbon amongst my colleagues"*. Note that 80% responded Strongly Agree that "*A better understanding of forest carbon will improve policy development and implementation"*. Despite this level of Agreement, decision-maker needs may not be clearly reflected in research and resulting material due to inadequate translation.

These data (Figs 2 and 3) can be interpreted as evidence of a 'science-practice gap' in CSF, by demonstrating gaps in forest carbon knowledge and CSF definitions, linkages, and assurances. This is not unique to CSF; STS scholars have found that "two-way knowledge flow between science and practice through joint knowledge-production/integration processes" is rare [40, p 93]. Considering the ongoing climate crisis coupled with the scale of investments in forest-based NCS, there is a pressing need for two-way knowledge flow to enhance science-

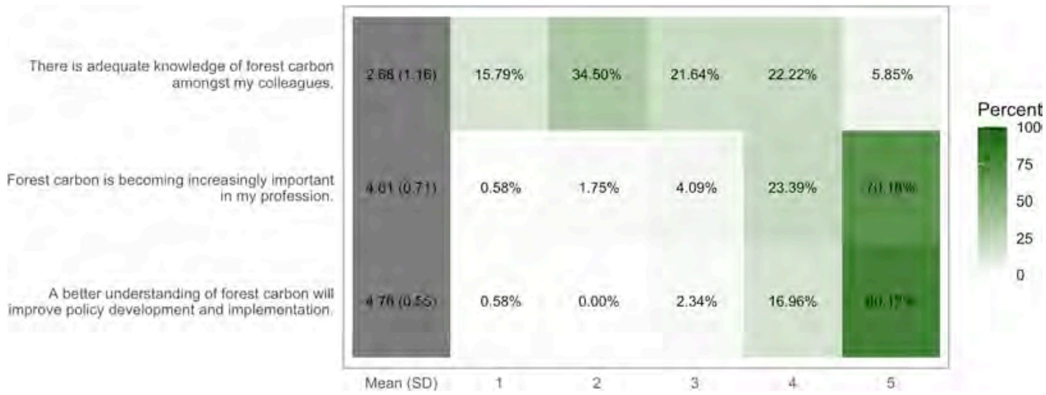

Fig 3. Data 2018–2021 pre-course participant questionnaire for understanding forest carbon management, MSU forest carbon and climate program. Level of agreement: 1 = Strongly Disagree, 2 = Disagree, 3 = Neutral or I don't know, 4 = Agree, and 5 = Strongly Agree.

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based CSF framing, metrics, and assessment. To overcome this and to ensure research is not overlooked by practitioners, [41, p 671] recommend making CSF research language:

1. Salient (relevant to decision-makers and readily accessible)
2. Credible (trustworthy, reliable, and sufficiently authoritative) and
3. Legitimate to both scientists and decision makers (developed via inclusive processes)

Risks of current CSF framing

This section explores a range of risks in *Current* CSF framing including (i) overly simplified relationships between carbon sequestration and forest management, (ii) emphasis on above-ground tree volume as forest carbon stocks, (iii) 'carbonization' of forest values, (iv) unintended social effects and unequal benefit distribution, (v) misinterpreting climate effects, and (vi) overlooking efficiency gains and economic misalignments.

Overly simplified relationships between carbon sequestration and forest management. Forest carbon storage is the outcome of complex, ecological processes. Oversimplifying carbon dynamics risks inadequately valuing late succession, primary, and/or old growth forests because of the misconception that sequestration diminishes rapidly or ceases altogether past the optimal harvest year, despite evidence that older trees and forests continue to sequester carbon at high rates well past this point [44, 45]. In fact, [44] uses the term 'financially mature' as a notable distinction from a presumably different biophysical threshold for reaching maturity (e.g., a substantial slowdown in annual growth), which may be over a hundred years later.

Forest management and public policy can strongly influence the sequestration process and some forestry systems may even emit carbon for a variety of reasons [46, 47]. While sequestration rates are generally higher in younger forests with more vigorous trees, old forests (of the same type), with larger trees, can have vastly more stored carbon. This has long generated debate within the scientific community in terms of resulted in a perceived tradeoff between maintaining carbon in older forests, with potentially declining sequestration, and conversion to younger forests, with potentially higher carbon sequestration rates [48–50].

As such, *Current* CSF literature framing may be appropriate for commercial forests but can leave out other important strategies linking forests and climate, such as avoiding deforestation or preventing reductions in forest complexity (e.g., biodiversity loss associated with conversions to monoculture plantations) [51]. Furthermore, many forests do not require management to help them remain healthy and in a state of net carbon sequestration or long-term storage. In fact, many forest health problems today have been caused or made worse by human interference. Examples include fire suppression [52], disease in monoculture [53], and invasive species expansion post-harvest [54].

Management activities that disturb the forest result in biogenic carbon emissions [55]; nearby trees damaged during management may die, the disturbed litter pool increases decomposition, trees are felled for skid roads, equipment can cause deep soil ruts and loss of stored soil carbon, and there is a reduction of woody material being transferred to litter and dead material pools. Studies [56, 57] have estimated management-related losses of 30 ± 6% in forest floor carbon of temperate forests depending on species composition (among other factors). While wood products provide important biomaterials, scaling up production management in forests with currently low-impact or no management would result in immediate and near-term, and possibly long-term, losses to stored ecosystem carbon [56, 58]. This lower carbon persists for at least decades [59] and can contribute to a shifting-baseline syndrome, wherein the 'baseline' forest carbon levels used for reference are already much lower than historical conditions [60].

Losing large trees can have especially negative climate implications. Researchers have found that large-diameter trees "store disproportionally massive amounts of carbon and are a major driver of carbon cycle dynamics in forests worldwide" [61, p 1]. This study, using forest inventory data from over 3,300 plots to assess the role of large diameter trees (greater than 53 cm, diameter at breast height), found that such trees stored 42% of total aboveground carbon despite accounting for only 3% of trees in the inventory [61].

Emphasis on aboveground tree volume as forest carbon stocks. There are also multiple, distinct challenges with forest carbon measurement, particularly as they relate to *Current* CSF initiatives (e.g., forest carbon projects). One issue is that most forest carbon inventories focus on aboveground biomass as the principal data for measurement, reporting, and verification (MRV) of forest carbon stocks. It is convenient that inventories of forest merchantable stem volume can be correlated with forest carbon stocks, because this greatly increases data availability for estimating carbon sequestration through biomass expansion factor approaches (such systems have been employed in both the U.S. and Canadian national forest inventories; [62, 63]). However, such forest carbon estimates can be biased towards carbon in tree boles [64] and can minimize or leave out carbon pool estimates in other parts of the trees and forests, leading to bias in model predictions (e.g., tree crowns, see [65]).

One major issue is the use of forest volume inventories or timber yield curves as proxies for forest carbon accumulation; this has implications for how forests are measured in terms of understanding and pursuing carbon benefits. Forest carbon storage dynamics are more complex than timber growth and yield curves imply [61, 66]. Timber yield is often maximized in monospecific plantations, but studies have found that multi-species forests store more carbon overall, maintain high sequestration rates over time (avoiding boom/bust cycle), and store more carbon across other pools [67, 68]. Framing forest carbon dynamics in terms of timber growth and yield curves may give the false impression that plantation-style forests are ideal for carbon sequestration rates, but with underperforming results for net climate benefit. Consider that trees spend the first portion of their lives in lower productivity [67], so harvesting can return a stand to a very low period of productivity. On the other end of the life spectrum, forest trees accumulate high rates of carbon at later lifetime stages as, for example, an estimated 70–80% accumulated after tropical trees reach 70 years [69].

Constraints to measuring carbon in other pools (e.g., soil, litter, downed wood, belowground) is a significant barrier [70]. For example, only up to 50% of total tropical forest carbon is found in aboveground, living pools [56]. Forest soil carbon and root biomass are much more difficult to quantify and remain a challenge in forest carbon measurements [71]. Some forest ecosystems have most of their carbon stored belowground; notably mangrove forests, which have been shown to have some of the highest carbon stores of any ecosystem worldwide [72, 73]. Aboveground carbon sequestration rates may be greater in highly productive plantation systems, but they may have lower total carbon storage than alternative forestry systems with lower above-ground sequestration rates (e.g., analog forestry systems versus teak plantations, see [68]). Another major challenge in forest carbon inventory comes from assumptions related to, or simply lack of data on, dead tree carbon stocks [74] and decay rates of dead material [71, 75].

Emerging science aims to better link other methods (e.g., remote sensing and tracking of fluxes through eddy covariance towers) that can enhance measurements and may be more appropriate in some cases and in other forest types [76]. Until such rates are quantified well, it will be difficult to determine more precisely the balance between carbon capture and emissions under different forest management scenarios.

'Carbonization' of forest values over resilience and biodiversity. Overemphasizing atmospheric carbon sequestration at the expense of other forest values has been called 'carbonization' of forest governance [77]. If CSF places a majority emphasis on carbon sequestration to mitigate climate change, there may be inadequate emphasis on adaptive mitigation strategies to ensure forests can respond to future climate trajectories. This may result in oversight of the peak ecological function necessary for forest resilience [78, 79]. While the CSF definition presented earlier [25]) includes 'adaptation', emphasis is largely placed on maintaining productivity and carbon storage levels.

Given the global biodiversity crisis co-occurring with the climate crisis, prioritizing carbon over biodiversity may have severe near and long-term consequences [80, 81]. Without a heavy emphasis on biodiversity in resilience, CSF can leave out the growing consensus around a need for diverse forests and to protect species richness. Notions of CSF currently do not appear to account for climate change effects on both floral and faunal diversity or intra-species genetic diversity [78, 82, 83]. Tree species diversity itself is centrally important in adaptation to maintain carbon stocks into the future [70].

Scholarly work has considered different management strategies that link carbon and biodiversity outcomes (see [84]). While some studies show clear linkages between increased carbon and biodiversity co-benefits, for example in tropical forest restoration [85], other studies point to key species benefits when aboveground carbon is lower. As an example, the Kirtland warbler in Michigan, USA jack pine (*Pinus banksiana*) forests, requires early succession habitat to thrive [86]. Other species require dense undisturbed or late succession forests, including many which we have more limited understanding (e.g., fungi, insects) of their unique roles in contributing to overall ecosystem function and resilience.

Researchers are finding carbon-focused conservation has limitations, as pursuing high carbon storage and habitat for specific species (e.g., birds or primates, see [87]) can still overlook overall biodiverse complexity [88]. This carries a risk for forests globally, as studies point to extended recovery times of species composition following a disturbance. For example, [87] showed that species requiring mature forests can still be absent from secondary forests 100 years later. The latter study goes on to assert that our understanding of secondary forest recovery is so limited it invalidates "any reliance upon the value of secondary forest for future conservation of tropical forest biodiversity" [87, p 28]. These results highlight risks of contributing to extinction for mature-forest dependent species by promoting management intensification in pursuit of higher sequestration rates, particularly with confounding stresses from a changing climate and the limited extent that mature forests have today.

Considering there is not a clear (e.g., linear) relation between carbon and biodiversity across different forest types [89], there is a pressing need to frame and assess biodiversity within CSF.

Unintended societal effects and unequal benefit distribution. CSF at scale implies substantial shifts in material use, investments, economics, and policies affecting land management, which will have wide-ranging societal impacts and creates the potential for unintended negative effects. *Current* CSF tends to emphasize large-scale opportunities, potentially overlooking smaller scale interventions (e.g., tree planting efforts, see [90]). Under a narrow framing of *Current* CSF, well-positioned beneficiaries (e.g., global investors and private companies) may stand to gain, while small landowners and Indigenous communities remain overlooked or even negatively affected by increasing pressure (e.g., timber production) or shifting values (e.g., monoculture) [21, 91]. Focusing mainly on commercially productive forests leaves out many potential actors or could create perverse incentives to develop more productive forests.

Moreover, *Current* CSF has a focus on complex, technical carbon stock measurements, which creates barriers for actors without training or sufficient resources to engage in complicated schemes [92–94]. Traditional ecological knowledge and cultural values may not be adequately embraced [95], and communities and rural actors likely perceive tradeoffs between livelihoods and biodiversity very differently than in industrialized outlooks [22, 96]. Because of this, not all such peoples are interested in advancing production-oriented forest management as defined in *Current* CSF framing. Despite optimistic views of far-reaching benefits for all [97], industry, investors, and governments are more likely to benefit as sophisticated participants and proponents of complex carbon schemes and global markets, an example of elite capture [21, 98].

The case of Indigenous people highlights how narrowly framed CSF risks overlooking best practices in justice and inclusion, by supporting entrenched systems of extraction, exploitation, and inequity. Roughly 1.5 billion Indigenous and rural peoples depend on forests for food and livelihoods, occupying approximately 28% of global land [81] and nearly 20% of global forests, with either formal or informal tenure rights [99]. Forests cover more than 80 percent of indigenous land area, totaling over 330 million hectares [100] of some of the most ecologically important, carbon rich biodiversity hotspots on Earth [81, 101]. Of these, 173 million hectares are considered "intact forests" meaning they have had little to no human modifications in the last 60–80 years [81]. Recent decades have seen an increased focus on indigenous and rural rights in relation to conservation and climate mitigation [91], which can be leveraged to reduce risk of CSF oversight.

The challenge of engaging rural landowners in CSF can be exemplified in data from a 2019 survey issued to farmland owners with over 100 years of consecutive ownership in the Kalamazoo River watershed in Michigan, USA (n = 116). Farm owners were asked questions about their land and if it could be leveraged to contribute to climate solutions (e.g., afforestation or forest restocking). The results were notable; 54% reported having marginal land (654 acres total) and 73% reported having fallow land (410 acres total) (Fig 4). Put together, this points to at least 1000 acres in one watershed, across 110 properties, as having potential for restoration with climate benefits. However, when participants shared their level of agreement with the statement "*Climate change influences my land management decisions*", only 16% responded Agree or Strongly Agree. Contrarily, 57% responded Agree or Strongly Agree with *"Environmental stewardship and beliefs influence my land management decisions"*. Finally, when asked level of agreement with the statement "*I would like to learn how my land can provide carbon sequestration*" the majority (41%) responded Neither Agree nor Disagree, indicating either disinterest or uncertainty. These data suggest that many rural landowners may not be ready to engage in or prioritize forest carbon but would be amenable to CSF activities framed as "environmental stewardship" or another culturally relevant term or phrase.

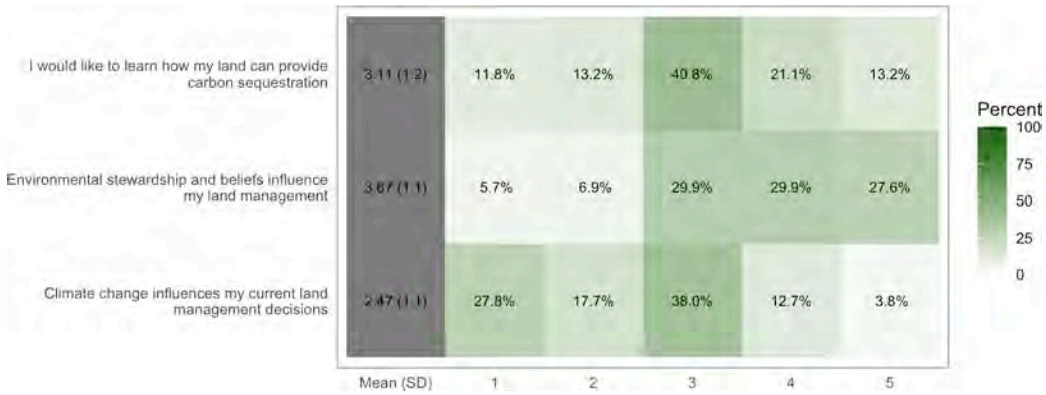

Fig 4. Survey response data by Michigan Kalamazoo watershed farmland owners (2019). Level of agreement: 1 = Strongly Disagree, 2 = Disagree, 3 = Neutral or I don't know, 4 = Agree, and 5 = Strongly Agree.

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Misinterpreting forest ecosystem management and climate effects. The emphasis on carbon sequestration as the dominant indicator of *Current* CSF brings in many potential misinterpretations in (1) non-carbon forest interactions with climate, (2) fossil fuel use in forestry operations and transportation, and (3) long-term and end-of-life carbon storage.

Non-carbon forest interactions with climate

Forest-climate effects are an area of active research and cannot be calculated from only biogenic carbon sequestration and emissions estimates. While the correlation between atmospheric CO_2 and global climate change is well documented [102], this does not fully explain forests' role in global temperature regulation. Forests generally have lower surface albedo (energy reflectance) and higher evapotranspiration (ET) compared to open land and non-tree vegetation due to the dark shade of the foliage. This reduced albedo can increase local warming, but this warming can be offset by increased ET. The precise connection is correlated with latitude; magnitudes and even effect direction on climate varies among tropical, temperate, and boreal forests [103]. These interactions complicate the carbon estimations, as recent research has found increased absorption of solar radiation creates localized warming that can outstrip the benefit of calculable increased carbon storage, for example by warming soil and increasing decomposition and release of soil carbon [104]. Because of these dynamics, climate benefits (e.g., local temperature, increased carbon storage) of interventions may have different effects based on forest type and geographic location. One study in Norway found net climate *warming* from increasing extent and quantity of high latitude mountain birch forests, even when considering the climate benefit of the additional stored carbon [105]. Comparatively, [106] explored temperature regulation and drought feedbacks ('savannization') in the Amazon basin linked to degradation, which points to prioritizing tropical forest protection over other forest-climate strategies.

Fossil fuel use in forestry operations and transportation

Fossil fuel use in forestry also undermines climate benefits calculated as part of harvested wood products (HWPs) in CSF. An established science, estimating carbon stored in HWPs requires robust life cycle analyses, including GHG emissions from production and shipping. A 2010 study estimated total yearly global emissions, considering both management and transport, to be 88.1 million metric tons CO_2e. Management-related emissions (36.9 million metric tons CO_2e) includes productivity interventions such as thinning as well as harvesting activities. Total transport-related forestry emissions were estimated to be over 50 million metric tons of CO_2e annually, with nearly 60 percent of these emissions associated with international trade [107]. Beyond this, fossil fuels are used in nearly all in-forest management activities including thinning, harvest, and hauling to mills, suggesting a need for CSF strategies that reduce carbon emissions associated with forest management.

Long-term and end-of-life carbon storage

Materials from sustainably managed forests, or with low-intensity management, provide society with essential goods and have an important role in CSF. Carbon in wood products is estimated as stored carbon (based on wood density and carbon ratio estimates), embodied carbon (overall emissions using life-cycle analysis), and substitution effect (net benefit from replacing a more emissions-intensive material). Carbon in HWPs is stored if the material remains in its physical form. Buildings, furnishings, and infrastructure across the built environment hold carbon for the longest time when compared to other wood products.

Wood products are second only to concrete in US annual waste material production, producing 40.8 million tons of waste in 2018 [108]. At end of life, wood products are typically incinerated or landfilled. Once in landfills, HWPs release gas (approximately half methane, CH_4, and half CO_2 by volume) from decomposition of degradable organic carbon unless in strictly anaerobic conditions [107, 109]. To estimate stored carbon eventually reentering the atmosphere, IPCC provides a default value of 0.5 (50%) [110]; note that site-specific studies have reported much lower estimates (e.g., 0–3% in US landfills, [111]). However, conditions in anaerobic landfills vary globally, particularly as open dumps and incineration are still common waste management strategies in developing countries. Even in developed nations, CO_2 and CH_4 emissions from landfills are substantial (e.g., roughly 2% of annual GHG emissions in Europe) and undermine the carbon storage of HWPs in landfills [112]. Because of this, the HWP duration of use and end-of-life of need to be considered in CSF.

Overlooking efficiency gains and economic misalignments

Inefficiencies in harvesting, processing, and material transport can undercut climate benefits of CSF and reduce landscape carbon storage. The modern globalized economy creates new opportunities, as well as concerns, for timber and agroforestry commodities about large scale and rapid impacts on forested landscapes [113]. Global trade is rife with inefficiency and international trade has been linked to higher increased GHG emissions [114]; transporting materials globally that could be produced and used locally is a major source of emissions [81].

To pursue climate benefits under CSF, there are ample opportunities to alter traditional economic flows of goods and materials to better value and emphasize waste reduction and material re-use, extend residence time of forest-based material in circulation, and incentivize innovative carbon storage and HWP production in landscapes outside of traditional forest management (e.g., abandoned urban and peri-urban areas). Recycling and waste product utilization can also help meet demands of scaled applications (e.g., bioenergy, mass timber).

Data from the US EPA ([108]; see Table 1) estimated that only 17% of US wood (3.1 of 18.09 million tons) was re-used and recycled and 67% (12.15 million tons) ended up in a landfill in 2018. Note that such recycling is almost entirely from chipping wood used in transport and packaging (e.g., pallets), but it is unclear if the chips are re-used in new materials or go towards emissions (e.g., via decomposition or burning). Recycling of durable wood products (6.51 million tons of waste generated in 2018) remains “negligible” [108].

Table 1. 2018 US wood material data with estimated equivalent forest acreage (EPA 2020).

US estimate (2018)	Amount
Wood material produced (million tons)	18.09
Wood material landfilled (million tons)	12.15
Wood material recycled (million tons)	3.1
Approximate equivalent forest extent (estimating 100 tons per acre)	121,500 acres (49,169 hectares)

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Table 2. Estimate of avoided emissions with MSU campus wood material diversion program (in metric tons CO_2).

	Without MSU SWRI	With MSU SWRI
Biomass removed from campus	146.09	146.09
Biomass–Diverted to SWRI	0.00	- 68.66
Net biomass removed from campus	146.09	77.43
Emissions—Processing mulch	0.52	0.28
Emissions—IPF tree removal vehicles	27.17	27.17
Emissions—SWRI step truck	0.00	6.28
Emissions—SWRI sawmill	0.00	0.47
Emissions—SWRI wood lab	0.00	12.00
Total emissions	173.78	123.63
Total avoided emissions	0.00	**50.15**

https://doi.org/10.1371/journal.pclm.0000212.t002

Consider a hypothetical scenario to grasp the scale of this material (Table 1). Assuming an average US southeast softwood clear-cut produces 100 tons of wood material per acre (40.5 tons per hectare), one could estimate annual wood landfilled as equivalent to harvest of 121,500 acres (49,169 hectares) of such a forest. These figures demonstrate how wood in use now could support CSF by providing substantial source material that can minimize forest pressure in the case of increased demand for 'climate-smart' wood products.

A 2020 study from the Michigan State University campus assessed the carbon benefits of a wood material diversion (sustainable wood recovery initiative, or SWRI) program that uses trees felled on campus to create artisanal wood products (e.g., furniture and housewares) [115]. The analysis found that between 2015 and 2017, MSU SWRI reduced net CO_2 emissions from the MSU campus urban wood system by approximately 28.9%, diverting 68.66 metric tons of CO_2e in logs and securing 28.42 metric tons of CO_2e in final wood products, with 40.24 metric tons of CO_2e remained in storage. Without this intervention, removal of campus trees would have resulted in emissions of 173.78 metric tons of CO_2, whereas with the MSU SWRI, the system emitted 123.63 metric tons of CO_2, equating to total avoided emissions of approximately 50.15 metric tons CO_2 (see Table 2). While fossil fuel emissions from chipping diminished, overall energy used for processing increased, including a net increase in fossil fuel use. Despite some campus solar, energy is largely from natural gas, diesel, and gasoline, which reflect net additive emissions distinct from the biogenic carbon cycle stored in the wood.

While this case demonstrates the added value of storing biogenic carbon longer, the use of fossil fuels contributes a net increase in atmospheric carbon from pre-industrial times. Soon, it will be necessary to eliminate fossil fuels from a 'climate-smart' forest product system to improve the comparative scenarios of wood use.

Expanding and clarifying CSF

Enhanced CSF framework. This analysis finds there is ample opportunity to broaden the concept of CSF to a spectrum of activities currently underrepresented that will increase climate benefits as well as social and environmental co-benefits. Adding to [25] three pillars (see *Current definitions and ideas in CSF* above), a broader definition could explicitly include additional landscapes, forest types, and interventions with climate benefits. Here, we propose the following additions as two new pillars to create an '*Enhanced*' CSF definition:

4) Protecting natural places by avoiding loss of forests, intact forests, forest complexity, biodiversity, or connectivity, or conversion to higher management intensity;

5) Promote restoration of degraded landscapes, improved ecosystem function, and connectivity (e.g., through corridors)

To better understand the distinction between *Current* and *Enhanced* CSF framing, Fig 5 distinguishes activities that dominate *Current* CSF (dark green) from those on either end of the forest condition and type spectrum that are not adequately represented (light green). These *Enhanced* columns capture the new pillars presented above. Further, the left column, *Phases*, reflects assessment and implementation phases that have not yet been clearly defined for CSF. Phases 1–4 reflect those could be considered generally present in *Current* CSF framing. The addition of a new Phase 5 captures broader assessment and impacts of *Enhanced* CSF

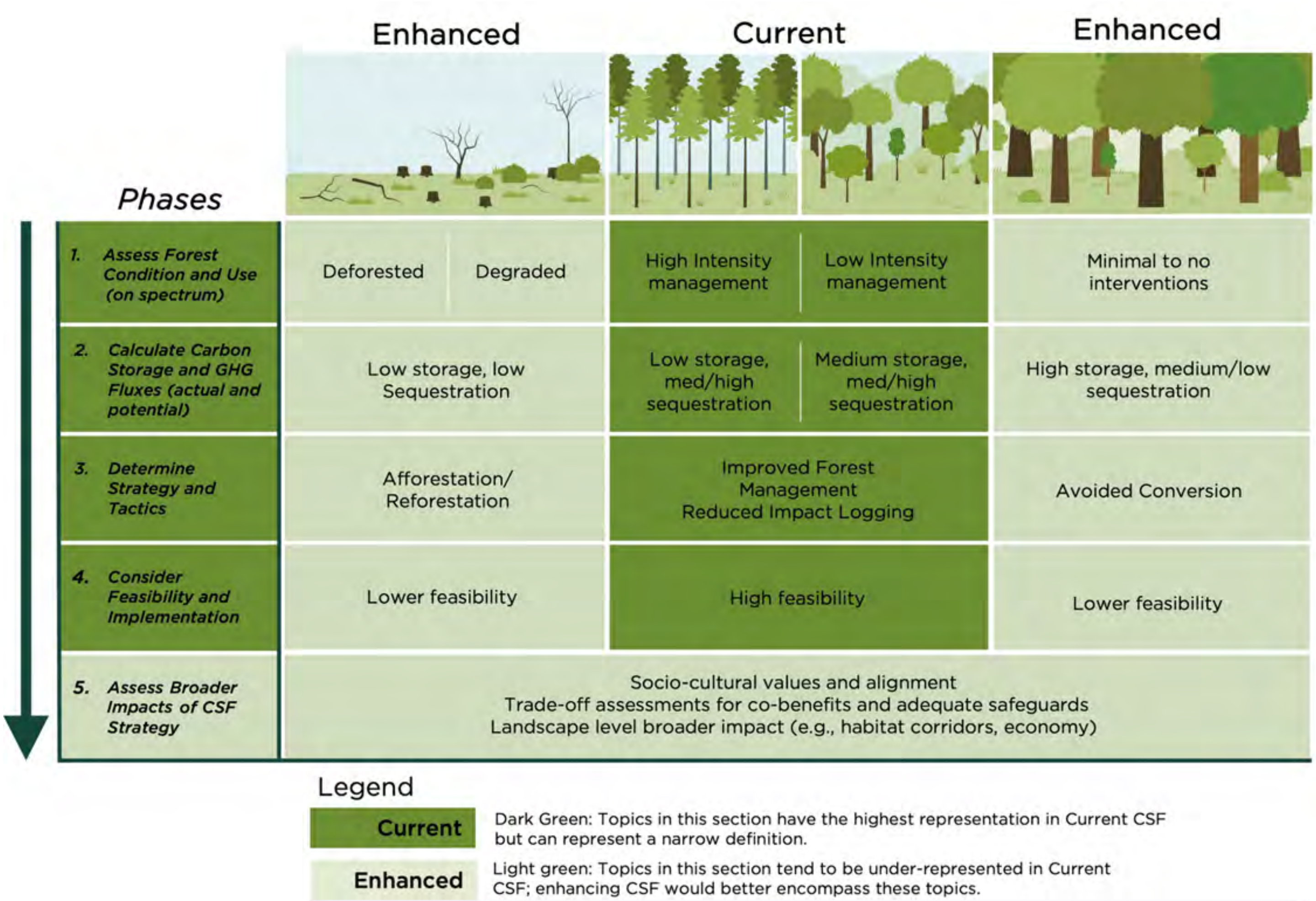

Fig 5. Planning and implementation phases of both *Current* CSF and proposed *Enhanced* CSF frameworks. This figure shows conceptual planning and implementation Phases (numbered 1–5) of both *Current* CSF and the *Enhanced* CSF proposed in this paper. The dark green center column indicates common features of *Current* CSF, particularly reflecting the emphasis on productive and managed forests in Improved Forest Management carbon projects. In Phase 1, *Enhanced* CSF, the light green columns on the right and left, encompass a broader spectrum of potential CSF landscapes from deforested or degraded (right, light green column) to minimal intervention, remote areas (left, light green column) than is seen in *Current* CSF alone (center, dark green column). After the landscape is assessed, GHG benefit (e.g., carbon storage and sequestration) is analyzed in Phase 2. Phase 3 includes a strategy assessment to achieve climate benefit, with tactics including reforestation and restoration (left, light green column), improved forest management (center, dark green column), and protection (right, light green column). Phase 4 captures feasibility challenges (e.g., finance, social license, additionality) that may be associated with each tactic; reflecting the high feasibility of *Current* CSF and the feasibility challenges facing *Enhanced* CSF. Phase 5, with the entire row in light green indicating it is a part of *Enhanced* CSF, reflects the increasingly dominant themes of landscape and biodiversity planning, inclusion, safeguards, and forest products that explicitly to link multiple scales and disciplines of actors that can be absent from *Current* CSF.

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currently absent from many strategies. Note that Phase 5 is increasingly discussed in CSF-related dialogues (e.g., climate-smart forest economies or mass timber) and we propose should have a role in a broader CSF definition.

Enhanced CSF components. The following sections explore key components of the *Enhanced* CSF framework presented above by highlighting details of the proposed *Phases* (see Fig 5, first column, for phase names).

1. Assess current forest condition and use (on a spectrum). CSF science would benefit from additional linkages across the spectrum of forest conditions and climate benefits to include these land and forest classifications as additional starting points to assess potential for CSF solutions. The *Current* CSF framework focuses on carbon in productive forests, including in most domestic US forest carbon projects and major international investments from development banks [116, 117]. However, as this review shows, there is a range of landscapes that could be included and promoted in CSF, including degraded areas, savannas, trees outside of forests, and intact areas with limited or no human interventions currently (e.g., remote tropical or boreal forests). These cover types are underrepresented in *Current* CSF literature but are relevant under *Enhanced* CSF framing. This aligns with several examples of colloquial usage (see Table 2, [117]) and makes for direct connections to REDD+ and restoration activities that are proven to provide highly impactful climate and carbon storage benefits.

2. Calculate carbon storage and GHG fluxes (Actual and Potential). CSF interventions must consider actual and potential greenhouse gas fluxes when considering benefits of wood use and stored carbon. There is an emerging emphasis on sequestration rates over carbon storage, which, as this paper explores, presents a narrow understanding of climate benefits compared to, for example, considering long-term resilience of forests and other treed landscapes. These oversights could undermine any carbon storage or sequestration by way of large-scale disturbance or die-off. On the other hand, *Enhanced* CSF principles can augment traditional forestry metrics by identifying and promoting additional indicators (e.g., tree longevity and biomass residency time) as part of CSF analysis to appraise multiple forest types more appropriately. These additional data will make it more likely that actors can adequately assess higher storage, lower productivity forests [56, 118, 119], as well as bring attention to maintaining large and secure carbon pools in place now [58].

Moreover, some forest carbon projects leave out carbon pools and GHGs considered 'not significant' or too difficult to assess, though some of them are potentially immensely important (e.g., such as forested peat soils, see [120]). While it may not be possible to adequately measure them now, their inclusion, event with default values, can provide important insights to support decision-making. Further, if CSF intends to make claims about carbon in the HWP pool, these calculations must be data-driven to avoid overestimating substitution benefit [121, 122] or underestimating emissions in forestry practices.

3. Determine strategy and tactics. While SFM focuses on forests managed for productivity, *Enhanced* CSF encompasses additional decisions for forested and potentially forested landscapes. An emphasis solely on 'productive', 'managed', or 'working' forests overlooks other opportunities for optimal climate benefits, particularly when planning at a landscape scale. Based on GHG information from Phase 2, an optimal mix of tactics can be determined that may include afforestation or reforestation, improved forest management (a type of SFM common in temperate carbon projects that pursues adjustments to practices to increase carbon storage on the landscape and in products) or Reduced Impact Logging (RIL), Avoided conversion of forested lands (including changes that result in loss of biodiversity or key species), or a combination of these.

Considering momentum on forest carbon projects and jurisdictional approaches, these methods and strategies could be explicitly linked in *Enhanced* CSF framing. Such interventions

Table 3. Examples sources of low feasibility scenarios in *Current* CSF framing.

Type	Example
Opportunity costs	High land value for commodity production overrides finances available for forest protection
Additionality	An inability to prove additionality (e.g., in the case of communally held remote tropical forests)
Carbon	Lower estimated sequestration rates despite immense carbon pools, (e.g., mature tropical Amazon)
	Low carbon return and high initial implementation costs for a period (e.g., for afforestation or reforestation)
Scale	Small scale intervention on a 20-acre parcel

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are well-documented in methodologies (e.g., Verra [123]), and access to a wider range of solutions can avoid potential pitfalls such as overlooking unique value of old or late succession forests, inappropriately prioritizing trees over prairies, or promoting more intense or even commercial forest management in communal forests with no history or interest in that activity. Further, if HWPs are part of the CSF strategy mix, it is essential to pursue efficiency for optimal climate benefits. CSF strategies could include identifying cascading value for wood materials to increase emphasis on long-lived products, reuse, and recycling.

4. Consideration of feasibility and implementation. Forest carbon projects typically have a feasibility stage that includes assessment of carbon stocks and fluxes, carbon market access, technical capacity, governance and management, and financial considerations (e.g., opportunity, inventory, and monitoring costs). *Current* CSF, particularly efforts that alter production management regimes (Improved Forest Management, or IFM), are highly feasible and have become the dominant source of carbon credits. For example, in the United States around 50% of projects on the Verra registry [123] and 87% on California ARB [117] are IFM projects. On the other hand, activities can be considered lower feasibility for a range of reasons, including high value of alternative land uses (known as opportunity costs), an inability to prove additionality, lower estimated sequestration rates, and scale of intervention (e.g., smaller parcels) (see Table 3).

CSF should include considerations beyond implementation costs and additionality to create more inclusive incentive structures. Carbon schemes that require evidence of deforestation risk to claim additionality or offer low payments to rural actors to protect forests, can undervalue stored carbon. For example, a major Peruvian conservation program, National Program for the Conservation of Forests (PNCB in Spanish), pays Indigenous communities 10 soles—approximately 3USD—per hectare, even in areas of demonstrably high risk [124] and of well-documented high-biodiversity ecological value [125].

Inclusive CSF interventions could benefit more actors by being easy to understand and with low barriers to entry (e.g., cost and knowledge). Increasingly, programs that provide *Enhanced* CSF benefits are reaching additional actors with programs that are comprehensible and with reasonable requirements (e.g., short time commitments). The Peruvian PNCB, discussed previously, performs well in this aspect, requiring commitments of only 5 years and presents the program in a simpler framing (avoiding forest conversion) and avoids technical carbon knowledge. Similarly, the US-based Family Forest Carbon Program offers shorter timeframes when compared to traditional carbon projects and compensates landowners for undertaking and committing to specific practices, like removing invasive species or allowing their forests to increase in maturity [126].

Key aspects of feasibility are social dimensions, like governance, participation, and inclusion of grievance mechanisms. [127] pointed out that policies should focus on how to ensure meaningful participation of local users in developing forest management and protection plans

[128]. Considering the example programs above, these tactics are helping overcome social barriers and increase feasibility that will be essential to scale and incentivize robust and diverse CSF interventions.

5. Assess broader impacts of CSF strategy

Enhanced CSF does not occur only at the parcel level. Instead, parcel-level initiatives are considered as component tactics of strategies to produce optimum outcomes at a landscape scale. This requires considerations like balancing production with protection and connecting natural areas as a restoration strategy.

Considering how and where to distribute benefits efficiently and equitably will be the central challenge going forward if CSF-oriented climate finance continues to arrive in forests globally [129]. Tactics should include horizontal (lateral) and vertical (top down or bottom up) benefit distribution across actors from forest decision-makers to wood users in built environments [130].

Multiple levels of governance and incentives require integrated approaches to sustainable land use, which will underpin CSF implementation. Further, multiple scales of government reporting (e.g., national level commitments in UNFCCC Nationally Determined Contributions (NDCs), jurisdictional approaches by sub-national actors) indicate different levels of uncertainty and possibilities for interventions. There remain opportunities to improve linkages between carbon stocks with landscape scale planning and management, to ensure carbon pool levels are maintained. Because of the wide range of possible actors in *Enhanced* CSF, it is becoming increasingly imperative and yet difficult to merge and layer this information in ways that neither inflate nor overlook benefits; or push increased sequestration at the detriment of stored carbon, communities, or other ecosystem services or forest inhabitants. Finally, efforts should strengthen incorporation of social and environmental safeguards (limiting negative consequences) in CSF, including unique approaches to eliminating harm (e.g., biodiversity loss) and increasing co-benefits across scales like local or regional economies and watersheds.

Sources of evidence of CSF

As shown in this analysis, *Enhanced* CSF reflects a complex interdisciplinary realm, crossing guidance and metrics for carbon storage and sequestration, biodiversity, sustainability, governance, and development. Dialogue on CSF can include wide ranging expertise, from architects to foresters to development organizations. Currently, there are substantial limitations in assuring sustainability in global forest management and product use, and it is unclear if or how available assurances can adequately assess and communicate CSF principles in an efficient and robust manner [93]. Further, the range of actors and decision-makers engaging in CSF makes it challenging to work across existing frameworks to safeguard against negative consequences.

The determination of whether forestry is 'climate smart' is a multistage process; the phases described here (Fig 5) represent a conceptualization of that process. Stacking and layering CSF methods and assurances will be necessary to assess these impacts; requiring the ability to translate data and methodologies for parcel level certifications, forest carbon projects, jurisdictional areas, and along the chain of custody for wood products. As many of these actors have good practice guidance or requirements in place, this section explores potential CSF additional metrics and assurances as well as additional sources of guidance useful for clarifying CSF and points of initiation for additional growth going forward.

Established implementation science

Scientific information can shape behavior through various processes of '*implementation science*'. In forestry, these can include regulations, voluntary guidelines, extension and knowledge transfer, evaluation frameworks, and professional organizations. Such examples of implementation science act as a translator between research and practitioner communities, i.e., overcoming the science-practice gap. As climate change becomes an increasing and persistent threat to society and forests, there are efforts to rapidly expand previous evaluation sustainable forestry frameworks (e.g., sustainable management Certification, Criteria and Indicators, Laws and Policies, Nationally Determined Contributions), Trade agreements, Best Management Practices) with new initiatives (e.g., Climate Smart Forest Economy Program, jurisdictional approaches). There have been relevant scholarly efforts assessing SFM criteria and indicators to identify which indicators are applicable for CSF, in a largely managed forest context [28, 131]. *Monitoring*, Reporting, and Verification (MRV*)* is the science of metrics and indicators for forest carbon and other GHG measurements. As a well-established approach in line with national commitments, direct linkages to the emerging theories around CSF have not yet been made clear, though they presumably match with a variety of measurement approaches related to carbon and forests. Considering the immense MRV efforts by nation-states and increasingly sub-state actors to establish MRV systems, there is increasingly ample data on landscape carbon stocks in above and below ground pools, and increasingly in soils. However, as this paper explores, carbon stocks alone are limited in their ability to frame climate benefits more broadly (e.g., climate "smartness") and MRV protocols might need to be *Enhanced* to include broader CSF principles.

Sustainable forest management certification, particularly for landowners, is a central interface to close the science-knowledge gap (see S3 Table for the language in the standard as well as other examples). However, in areas with weak governance and high levels of illegal activity, chain of custody can be nearly impossible to determine, limiting the power of existing assurances like certification. This means that promoting wood used from unknown origins can have major social and environmental impacts. Generally, certification and their implementing organizations provide not only guidance, but a two-way communication platform to engage and train rural decision-makers and utilize inclusive stakeholder engagement process to develop guidance. Straka and Khanal [132] describe how forest certification is a tool for knowledge transfer, and, as an example, the latest Sustainable Forestry Initiative (SFI) standard includes a new Objective titled "Climate Smart Forestry" [32].

Emerging implementation science

Implementation science can also strive to include participatory and mutually beneficial data collection and sharing efforts (e.g., community-based monitoring). Increasing value for communities to participate, value their role as protectors of resources. These established sources of implementation offer opportunities to add specific guidance and planning related to CSF to reach practitioners [133].

Unique methodologies can address risk and improve outcomes. Project level requirements (forest carbon projects) are well established and emerging guidance on climate-smart forest economies is forthcoming. At the nation-state level, National Safeguard Information Systems (SIS) include indicators like 'No Net Loss' of biodiversity (NNL), or even strive for a 'Net Gain' (NG) [134]. Additional methodologies to provide guidance for identifying areas that are ideal for restoration (e.g., where previous high carbon storage areas, boost habitat connectivity) remain underdeveloped and underemphasized.

Overall, more research and engagement work at assessing CSF, particularly a broader definition is needed to implement the principles across additional landscapes and scales.

Conclusion

Largescale application of the *Current* CSF framework could result in paltry or even undesirable outcomes for climate, biodiversity, and society. This review finds that, despite its increasing use in professional and applied contexts, definitions and analysis of CSF are limited in the literature, reflecting a 'science-practice gap'. Our analysis reveals additional planning and implementation components are necessary to assess and ensure the degree to which forest interventions are in fact 'climate-smart', including broadening forest cover types, conditions, and climate interactions. To do so, this paper presents a framework with an *Enhanced* CSF definition to better link scholarly work in carbon, climate, communities, and forests which evolving interpretations of forest-based climate action. This expanded framework aims to support the translation of the theoretical CSF, defined by the researcher, into practice, with interventions that to engage rural and marginal actors, build local and regional economies, minimize waste, limit and eventually eliminate fossil fuels, and value diverse forest values (e.g., carbon storage alongside cultural values, habitat).

CSF diverges from SFM in the depth of existing research, eligible land and forest categories, and indicators needed for assessment. Still under-studied, scholarly work appears to generally indicate that CSF could be understood as a niche component of SFM [29], implying that there exists forest management that is sustainable but not considered 'climate-smart' (as in, there are no calculable carbon or GHG benefits) as well as leaving out potentially forested landscapes and intact areas that can be targeted for protection. This analysis offers a different conclusion, contending CSF is a broad umbrella under which to assess additional forested or potentially forested landscapes, particularly those that may not be managed primarily for timber. In this definition, SFM is only one element *under* the broad umbrella of CSF, which encompass a diversity of land management and conservation practices beyond carbon, but that are essential to adapting to climate, actively consider other species, and supporting resilient landscapes for multiple values in society.

CSF is being used broadly outside of academia, demonstrating a need to incorporate climate-oriented decision-making across many landscapes *including*–not excluding–protected lands, urban areas, and in restoration. This framing reaches multiple professional disciplines crossing forestry, development and planning, timber production, conservation, natural resource management, social sciences, and governing bodies. For CSF to reach its potential, it should include wood use as efficiently and for as long as possible, eliminate risk of perverse incentives to replace more natural landscapes with plantation forests, maintain or increase landscape-level carbon, improve inefficient wood use practices, and restore degraded lands. An *Enhanced* framing of CSF would reduce risks associated with applying production-oriented CSF too broadly–undesirable outcomes for environment and society–by drawing in the robust body of science on carbon, climate, forests, habitats, and social science (including participation, economics, justice, and diverse values of nature).

Future work will need to pursue improved methods to estimate and model forest carbon across pools, as well incorporate climate sensitivity, uncertainty assessments, and quantification of other ecosystem services (e.g., biodiversity, hydrologic processes). Building on best practices from across sustainable development and forestry disciplines, CSF requires inclusive dialogue to navigate this profound opportunity for radical revisioning of forestland decision-making, forest product use, conservation, transparency, economic indicators, inclusion, and benefit sharing.

Supporting information

S1 Table. Examples of Climate-Smart Forestry (CSF) in use by diverse actors [27, 29, 30, 32, 100, 137–142].
(DOCX)

S2 Table. Definitions of CSF in academic literature (as of early 2022) [25, 29, 131, 143, 144].
(DOCX)

S3 Table. Approaches and phrases related to CSF, defined [38, 39, 42, 135, 136, 145–147].
(DOCX)

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FEBRUARY 1, 2024

Edge habitats along roads and power lines may be key to conserving rare plants

by Chuck Gill, Pennsylvania State University


Wild lupine, a plant of conservation concern across most of its natural range in eastern North America, grows along a Pennsylvania roadside. Credit: Isabella Petitta

Managing forest edge habitats to maintain a gradient of canopy cover and plant density could be key to conserving some threatened native plant species such as wild lupine, according to Penn State researchers.

Edge habitats created by natural or human-caused disturbances, including corridors along roadways and utility rights-of-way, provide prime opportunities for encouraging the establishment and reproduction of rare native plants, the researchers reported in a new study published in *Plant Ecology*.

The authors reviewed and synthesized the findings of 33 published studies examining the biology and management of wild lupine and associated plants and insects. Their case study suggests that land management—including prescribed burning, mowing and mechanical thinning—can promote the conservation of wild lupine and other forest edge plants.

"Most Eastern ecosystems are managed to maintain dense, forested habitats," said lead author Isabella Petitta, master's degree candidate in Penn State's intercollege ecology graduate program. "The lack of disturbance in these woodlands generates homogenous, closed canopy forests that result in losses of habitat for early successional plants such as wild lupine."

Across almost 60% of its original range in eastern North America, wild lupine is a species of conservation concern that requires management strategies for its protection, the researchers said.

Petitta, a U.S. National Science Foundation Graduate Research Fellow, explained that one of wild lupine's primary habitats is oak savanna, an early successional habitat with a canopy cover of less than 50%. Considered transition areas between prairie and forest, oak savannas provide a mix of canopy cover that allows for diverse plant communities and microhabitats.

Wild lupine prefers open or partially shaded conditions with a canopy cover of 50% or less. It grows to about 8 to 24 inches tall, and each mature stem produces between 30 and 50 white, light pink, purple or blue flowers. The pollinator-dependent perennial has been deemed an indicator species of quality oak savanna habitat.

A patch of wild lupine grows along a forest edge in a road and utility right-of-way. With the decline of the plant's original habitat, 80% of wild lupine populations in Pennsylvania are located along rights-of-way for human infrastructure, researchers said. Credit: Nash Turley

"But fire suppression, development, demand for timber, conversion to agricultural land and other factors have reduced oak savannas to be among the most endangered habitats in North America," Petitta said, noting that oak savannas cover only about 0.02% of their original land area.

Study co-author Autumn Sabo, assistant professor of biology at Penn State Beaver and Petitta's co-adviser, noted that the loss of oak savanna habitat means that forest edges that are maintained for infrastructure have become an important habitat for rare plant species.

"In 2017, for example, there were more than 700,000 miles of high-voltage transmission lines and 6.5 million miles of local power distribution lines in the United States," she said.

"And in Pennsylvania, about 80% of wild lupine populations are located along road, trail, rail, gas or power line rights-of-way."

Wild lupine habitat generally supports other early successional, prairie and forest edge plants, some of which may benefit from wild lupine's ability to fix atmospheric nitrogen in the soil, the researchers said.

"Wild lupine habitats also support a diverse insect community, and the flowers produce pollen and nectar that attract insect pollinators and visitors," said study co-author Margarita López-Uribe, associate professor of entomology and Lorenzo L. Langstroth Early Career Professor in the College of Agricultural Sciences.

López-Uribe, who also co-advises Petitta, pointed out that roadsides and power line rights-of-way adjacent to a forest edge are considered pollinator-friendly habitats because they provide diverse floral resources and movement corridors.

"Also, the vegetative parts of wild lupine serve as a host for the larvae of three butterflies of conservation concern—the endangered Karner blue butterfly, the persius duskywing and the frosted elfin—and one moth, the lupine leafroller," she said. "Declines in wild lupine habitat are directly related to the decline of these species."

The researchers said several environmental conditions are needed to increase wild lupine cover and density, including light intensity levels around 65% of full sunlight, canopy cover that provides intermediate or partial shade, and the minimal presence of leaf litter. Management practices that can achieve these conditions, they suggested, include prescribed fire, herbicide application, mowing and mechanical tree removal.

In addition to habitat management, wild lupine populations can be enhanced through seeding and transplanting, the researchers added. But they recommended that land managers should focus on increasing existing populations before attempting to establish new ones.

The researchers cautioned that although various edge habitat management practices have been shown to have benefits for early successional plants, these methods need to be planned and timed correctly to be effective, and more study is needed to fine-tune recommendations.

"Wild lupine offers an opportunity to study and optimize management of rare plants in early successional edge habitats, including human-made habitats such as power line rights-of-way and roadsides," Petitta said. "Restoring and managing these edge habitats is critical for the conservation of wild lupine and its associated plant and insect communities."

More information: Isabella R. Petitta et al, Biology and management of wild lupine (Lupinus perennis L.): a case study for conserving rare plants in edge habitat, *Plant Ecology* (2023). DOI: 10.1007/s11258-023-01371-9
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WILD CARBON

A synthesis of recent findings

MARK G. ANDERSON, PhD

We find ourselves not at the edge of a precipice, but beyond it. Climate change is altering the world as we know it, no matter how quickly we act to reduce our collective carbon footprint. But the worst impacts are still avoidable with natural climate solutions. Permanently protecting forests and allowing them to grow in landscapes free from direct human manipulation is proving to be one of the most effective and cost efficient methods available to address the climate crisis. While wild nature has a right to exist simply for its intrinsic value, recent science is shedding peer-reviewed light on the exceptional carbon storage capacity of unmanaged land, and its equally important benefits for safeguarding biodiversity. In this short synthesis, ecologist Mark Anderson summarizes recent studies which demonstrate that in our fragmented, fast-developing world, wilderness offers the earth and its community of life the precious gift of time.

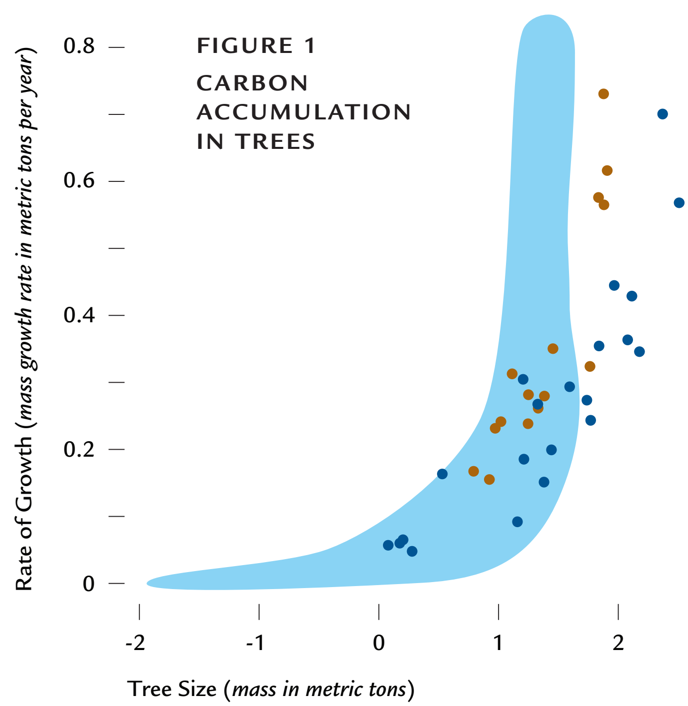
—Jon Leibowitz, Executive Director, Northeast Wilderness Trust

A long-standing debate over the value of old forests in capturing and storing carbon has prompted a surge of synthesis studies published in top science journals during the last decade. Here are five emerging points that are supported by solid evidence.

1) Trees accumulate carbon over their entire lifespan. Plants absorb carbon dioxide from air and transform it into carbon-rich sugars. These are then converted to cellulose to create biomass (trunk, bark, leaf) or transferred below-ground to feed the root-fungal networks. Over the long lifespan of the tree, large amounts of carbon are removed from the air and stored as biomass. Growth efficiency declines as the tree grows but corresponding increases in the tree's total leaf area are enough to overcome this decline and thus the **whole-tree carbon accumulation rate increases with age and size** (Figure 1). A study of 673,046 trees across six countries and 403 species found that at the extreme, a large old tree may sequester as much carbon in one year as growing an entire medium size tree (Stephenson et al. 2014). At one site, large trees comprised 6 percent of the trees but 33 percent of the annual forest growth. Young trees grow fast, but old trees store a disproportional amount of carbon.

2) Old forests accumulate carbon and contain vast quantities of it. Old-growth forests have traditionally been considered negligible as carbon sinks. Although individual trees experience an increasing rate of carbon sequestration, forest stands experience an "S-curve" of net sequestration rates (e.g. slow, rapid, slow). The expected decline in older stands is due to tree growth being balanced by mortality and decomposition. To test the universality of carbon neutrality in old forests, an international team of scientists reviewed 519 published forest carbon-flux estimates from stands 15 to 800 years old and found that, in fact, net carbon storage was positive for 75 percent of the stands over 180 years old and the chance of finding an old-growth forest that was carbon neutral was less than one in ten (Luyssaert et al. 2014). They concluded that **old-growth forests are usually carbon sinks, steadily accumulating carbon and containing vast quantities of it**. They argued that carbon-accounting rules for forests should give credit for leaving old-growth forest intact. This is important globally, as old forests in the tropics have acted as long-term net biomass/carbon sinks but are now vulnerable to edge effects, logging and thinning, or increased mortality from disturbances (Brienen et al. 2015, Lan Qui et al. 2018).

3) Old forests accumulate carbon in soils. The soil carbon balance of old-growth forests has received little attention, although it was generally accepted that soil organic carbon levels in old forests are in a steady state. In 2017, Guoyi Zhou and colleagues measured the 24-year dynamics of the soil carbon in an old-growth forest at China's Dinghushan Biosphere Reserve. They found that **soils in the top 20-cm soil layer accumulated atmospheric carbon at an unexpectedly high rate**, with soil organic carbon concentration increasing from about 1.4 percent to 2.4 percent



Aboveground mass growth rates for 58 species (shaded area) juxtaposed with two of the most massive tree species on earth: Swamp Gum (*Eucalyptus regnans*—brown dots) and Coast Redwood (*Sequoia sempervirens*—blue dots). Mass growth rate equals the total mass accumulated each year after accounting for respiration. The mass of a tree is primarily carbon, so the figure shows that annual carbon accumulation increases with the size of the tree. (Adapted from Stephenson et al. 2014.)

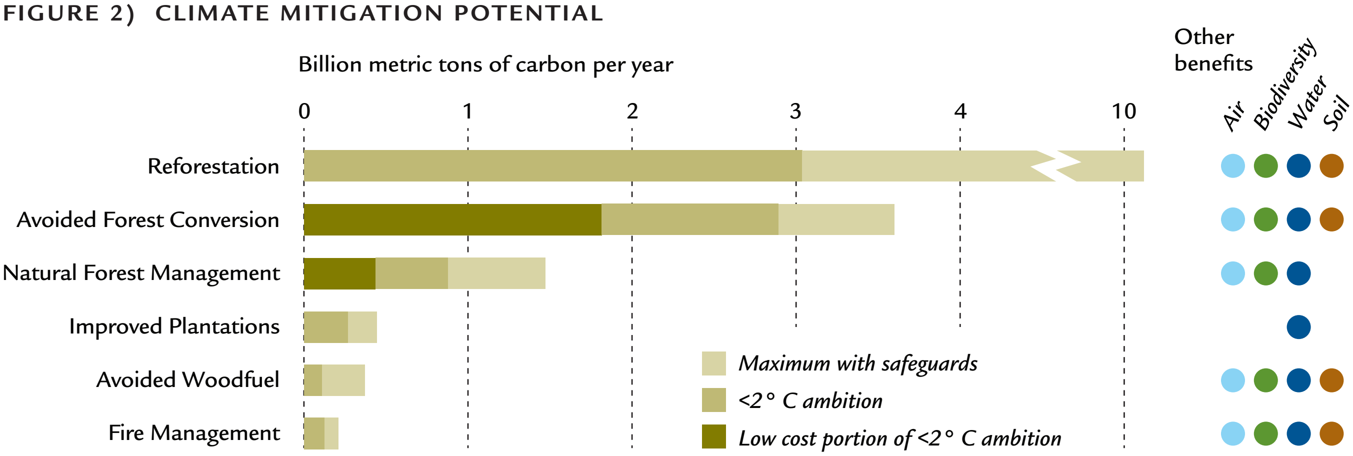
and soil carbon stock increasing significantly at an average rate of 0.61 metric tons of carbon per hectare per year (Zhou, G. et al. 2006). Their result directly challenges the prevailing belief in ecosystem ecology regarding carbon budget in old-growth forests and calls for further study.

4) Forests share carbon among and between tree species. Forest trees compete for light and soil resources, and competition for resources is commonly considered the dominant tree-to-tree interaction in forests. However, recent research made possible by stable carbon isotope labeling indicates that trees interact in more complex ways, including substantial exchange and sharing of carbon. In 2016, Tamir Klein and colleagues applied carbon isotope labeling at the canopy scale, and found that carbon assimilated by a tall spruce was traded with neighboring beech, larch, and pine trees via overlapping root spheres. Aided by mycorrhiza networks, **interspecific transfer accounted for 40 percent of the fine root carbon** totaling roughly 280 kilograms per hectare per year tree-to-tree transfer (Klein et al. 2016). In a subsequent study, Morrie et al. (2017), found that mycorrhiza soil networks become more connected and take up more carbon as forest succession progresses even without major changes in dominant species composition.

A large old tree accumulates impressive amounts of carbon every year while also releasing oxygen, filtering pollution, and creating food and habitat for wildlife.

5) Forest carbon can help slow climate change. There has been debate about the role of forests in sequestering carbon and the role of land stewardship in achieving the Paris Climate Agreement goal. In 2017, Bronson Griscom and colleagues systematically evaluated twenty conservation, restoration, and improved land management actions that increase carbon storage and/or avoid greenhouse gas emissions. They found the maximum potential of these natural climate solutions was almost 24 billion metric tons of carbon equivalent per-year while safeguarding

FIGURE 2) CLIMATE MITIGATION POTENTIAL



Climate mitigation potential of six forest pathways estimated for reference year 2030. Bars represent maximum possible with safeguards (i.e. constraints applied to safeguard the production of food and fiber and habitat for biological diversity). Darker portions represent cost-effective mitigation levels assuming a global ambition to hold warming to <2° C. Darkest portions indicate low cost portions. Ecosystem service benefits linked with each pathway are indicated by colored dots for biodiversity, water (filtration and flood control), soil (enrichment), and air (filtration). (Adapted from Griscom et al. 2017.)

food security and biodiversity. About half of this could be delivered as cost-effective contributions to the Paris Agreement, equivalent to about **30 percent of needed mitigation as of 2030**, with 63 percent coming from forest-related actions (Figure 2). **Avoided forest conversion** had the highest carbon potential among the low-cost solution (Griscom et al. 2017). New research suggests this strategy is the most cost-feasible option by a large margin (Busch et al. 2019) and it should receive high priority as a policy consideration in the U.S. (McKinley et al. 2011). An analysis of 18,507 forest plots in the Northeast found that old forests (greater than 170 years) supported the largest carbon pools and the highest simultaneous levels of carbon storage, timber growth, and species richness (Thom et al. 2019). In addition to carbon, old forests also build soil, cycle nutrients, mitigate pollution, purify water, release oxygen, and provide habitat for wildlife.

CONCLUSION

Recently published, peer-reviewed science has established that unmanaged forests can be highly effective at capturing and storing carbon. It is now clear that trees accumulate carbon over their entire lifespan and that old, wild forests accumulate far more carbon than they lose through decomposition and respiration, thus acting as carbon sinks. This is especially true when taking into account the role of undisturbed soils only found in unmanaged forests. In many instances, the carbon storage potential of old and wild forests far exceeds that of managed forests. We now know that the concept of overmature forest stands, used by the timber industry in reference to forest products, does not apply to carbon.

In the Northeast, a vigorous embrace of natural climate solutions to mitigate global overheating does not require an either/or choice between managed and unmanaged forests. **Conserving unmanaged wild forests is a useful, scalable, and cost-effective complementary strategy to the continued conservation of well-managed woodlands.**

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THE AMERICAN FORESTS.

The forests of America, however slighted by man, must have been a great delight to God; for they were the best he ever planted. The whole continent was a garden, and from the beginning it seemed to be favored above all the other wild parks and gardens of the globe. To prepare the ground, it was rolled and sifted in seas with infinite loving deliberation and forethought, lifted into the light, submerged and warmed over and over again, pressed and crumpled into folds and ridges, mountains and hills, subsoiled with heaving volcanic fires, ploughed and ground and sculptured into scenery and soil with glaciers and rivers, — every feature growing and changing from beauty to beauty, higher and higher. And in the fullness of time it was planted in groves, and belts, and broad, exuberant, mantling forests, with the largest, most varied, most fruitful, and most beautiful trees in the world. Bright seas made its border with wave embroidery and icebergs; gray deserts were outspread in the middle of it, mossy tundras on the north, savannas on the south, and blooming prairies and plains; while lakes and rivers shone through all the vast forests and openings, and happy birds and beasts gave delightful animation. Everywhere, everywhere over all the blessed continent, there were beauty, and melody, and kindly, wholesome, foodful abundance.

These forests were composed of about five hundred species of trees, all of them in some way useful to man, ranging in size from twenty-five feet in height and less than one foot in diameter at the ground to four hundred feet in height and more than twenty feet in diameter, — lordly monarchs proclaiming the gospel of beauty like apostles. For many a century after the ice-ploughs were melted, nature fed them and dressed them every day; working like a man, a loving, devoted, painstaking gardener; fingering every leaf and flower and mossy furrowed bole; bending, trimming, modeling, balancing, painting them with the loveliest colors; bringing over them now clouds with cooling shadows and showers, now sunshine; fanning them with gentle winds and rustling their leaves; exercising them in every fibre with storms, and pruning them; loading them with flowers and fruit, loading them with snow, and ever making them more beautiful as the years rolled by. Wide-branching oak and elm in endless variety, walnut and maple, chestnut and beech, ilex and locust, touching limb to limb, spread a leafy translucent canopy along the coast of the Atlantic over the wrinkled folds and ridges of the Alleghanies, — a green billowy sea in summer, golden and purple in autumn, pearly gray like a steadfast frozen mist of interlacing branches and sprays in leafless, restful winter.

To the southward stretched dark, level-topped cypresses in knobby, tangled swamps, grassy savannas in the midst of them like lakes of light, groves of gay sparkling spice-trees, magnolias and palms, glossy-leaved and blooming and shining continually. To the northward, over Maine and the Ottawa, rose hosts of spiry, rosiny evergreens, — white pine and spruce, hemlock and cedar, shoulder to shoulder, laden with purple cones, their myriad needles sparkling and shimmering, covering hills and swamps, rocky headlands and domes, ever bravely aspiring and seeking the sky; the ground in their shade now snow-clad and frozen, now mossy and flowery; beaver meadows here and there, full of lilies and grass; lakes gleaming like eyes, and a silvery embroidery of rivers and creeks watering and brightening all the vast glad wilderness.

Thence westward were oak and elm, hickory and tupelo, gum and liriodendron, sassafras and ash, linden and laurel, spreading on ever wider in glorious exuberance over the great fertile basin of the Mississippi, over damp level bottoms, low dimpling hollows, and round dotting hills, embosoming sunny prairies and cheery park openings, half sunshine, half shade; while a dark wilderness of pines covered the region around the Great Lakes. Thence still westward swept the forests to right and left around grassy plains and deserts a thousand miles wide: irrepressible hosts of spruce and pine, aspen and willow, nut-pine and juniper, cactus and yucca, caring nothing for drought, extending undaunted from mountain to mountain, over mesa and desert, to join the darkening multitudes of pines that covered the high Rocky ranges and the glorious forests along the coast of the moist and balmy Pacific, where new species of pine, giant cedars and spruces, silver firs and sequoias, kings of their race, growing close together like grass in a meadow, poised their brave domes and spires in the sky three hundred feet above the ferns and the lilies that enameled the ground; towering serene through the long centuries, preaching God's forestry fresh from heaven.

Here the forests reached their highest development. Hence they went wavering northward over icy Alaska, brave spruce and fir, poplar and birch, by the coasts and the rivers, to within sight of the Arctic Ocean. American forests! the glory of the world! Surveyed thus from the east to the west, from the north to the south, they are rich beyond thought, immortal, immeasurable, enough and to spare for every feeding, sheltering beast and bird, insect and son of Adam; and nobody need have cared had there been no pines in Norway, no cedars and deodars on Lebanon and the Himalayas, no vine-clad selvas in the basin of the Amazon. With such variety, harmony, and triumphant exuberance, even nature, it would seem, might have rested content with the forests of North America, and planted no more.

So they appeared a few centuries ago when they were rejoicing in wildness. The Indians with stone axes could do them no more harm than could gnawing beavers and browsing moose. Even the fires of the Indians and the fierce shattering lightning seemed to work together only for good in clearing spots here and there for smooth garden prairies, and openings for sunflowers seeking the light. But when the steel axe of the white man rang out in the startled air their doom was sealed. Every tree heard the bodeful sound, and pillars of smoke gave the sign in the sky.

I suppose we need not go mourning the buffaloes. In the nature of things they had to give place to better cattle, though the change might have been made without barbarous wickedness. Likewise many of nature's five hundred kinds of wild trees had to make way for orchards and cornfields. In the settlement and civilization of the country, bread more than timber or beauty was wanted; and in the blindness of hunger, the early settlers, claiming Heaven as their guide, regarded God's trees as only a larger kind of pernicious weeds, extremely hard to get rid of. Accordingly, with no eye to the future, these pious destroyers waged interminable forest wars; chips flew thick and fast; trees in their beauty fell crashing by millions, smashed to confusion, and the smoke of their burning has been rising to heaven more than two hundred years. After the Atlantic coast from Maine to Georgia had been mostly cleared and scorched into melancholy ruins, the overflowing multitude of bread and money seekers poured over the Alleghanies into the fertile middle West, spreading ruthless devastation ever wider and farther over the rich valley of the Mississippi and the vast shadowy pine region about the Great Lakes. Thence still westward the invading horde of destroyers called settlers made its fiery way over the broad Rocky Mountains, felling and burning more fiercely than ever, until at last it has reached the wild side of the continent, and entered the last of the great aboriginal forests on the shores of the Pacific.

Surely, then, it should not be wondered at that lovers of their country, bewailing its baldness, are now crying aloud, "Save what is left of the forests!" Clearing has surely now gone far enough; soon timber will be scarce, and not a grove will be left to rest in or pray in. The remnant protected will yield plenty of timber, a perennial harvest for every right use, without further diminution of its area, and will continue to cover the springs of the rivers that rise in the mountains and give irrigating waters to the dry valleys at their feet, prevent wasting floods and be a blessing to everybody forever.

Every other civilized nation in the world has been compelled to care for its forests, and so must we if waste and destruction are not to go on to the bitter end, leaving America as barren as Palestine or Spain. In its calmer moments in the midst of bewildering hunger and war and restless over-industry, Prussia has learned that the forest plays an important part in human progress, and that the advance in civilization only makes it more indispensable. It has, therefore, as shown by Mr. Pinchot, refused to deliver its forests to more or less speedy destruction by permitting them to pass into private ownership. But the state woodlands are not allowed to lie idle. On the contrary, they are made to produce as much timber as is possible without spoiling them. In the administration of its forests, the state righteously considers itself bound to treat them as a trust for the nation as a whole, and to keep in view the common good of the people for all time.

In France no government forests have been sold since 1870. On the other hand, about one half of the fifty million francs spent on forestry has been given to engineering works, to make the replanting of denuded areas possible. The disappearance of the forests in the first place, it is claimed, may be traced in most cases directly to mountain pasturage. The provisions of the code concerning private woodlands are substantially these: No private owner may clear his woodlands without giving notice to the government at least four months in advance, and the forest service may forbid the clearing on the following grounds: to maintain the soil on mountains, to defend the soil against erosion and flooding by rivers or torrents, to insure the existence of springs and watercourses, to protect the dunes and seashore, etc. A proprietor who has cleared his forest without permission is subject to heavy fine, and in addition may be made to replant the cleared area.

In Switzerland, after many laws like our own had been found wanting, the Swiss forest school was established in 1865, and soon after the Federal Forest Law was enacted, which is binding over nearly two thirds of the country. Under its provisions, the cantons must appoint and pay the number of suitably educated foresters required for the fulfillment of the forest law; and in the organization of a normally stocked forest, the object of first importance must be the cutting each year of an amount of timber equal to the total annual increase, and no more.

The Russian government passed a law in 1888, declaring that clearing is forbidden in protection forests, and is allowed in others "only when its effects will not be to disturb the suitable relations which should exist between forest and agricultural lands."

Even Japan is ahead of us in the management of her forests. They cover an area of about 29,000,000 acres. The feudal lords valued the woodlands, and enacted vigorous protective laws; and when, in the latest civil war, the Mikado government destroyed the feudal system, it declared the forests that had belonged to the feudal lords to be the property of the state, promulgated a forest law binding on the whole kingdom, and founded a school of forestry in Tokio. The forest service does not rest satisfied with the present proportion of woodland, but looks to planting the best forest trees it can find in any country, if likely to be useful and to thrive in Japan.

In India systematic forest management was begun about forty years ago, under difficulties — presented by the character of the country, the prevalence of running fires, opposition from lumbermen, settlers, etc. — not unlike those which confront us now. Of the total area of government forests, perhaps 70,000,000 acres, 55,000,000 acres have been brought under the control of the forestry department, — a larger area than that of all our national parks and reservations. The chief aims of the administration are effective protection of the forests from fire, an efficient system of regeneration, and cheap transportation of the forest products; the results so far have been most beneficial and encouraging.

It seems, therefore, that almost every civilized nation can give us a lesson on the management and care of forests. So far our government has done nothing effective with its forests, though the best in the world, but is like a rich and foolish spendthrift who has inherited a magnificent estate in perfect order, and then has left his rich fields and meadows, forests and parks, to be sold and plundered and wasted at will, depending on their inexhaustible abundance. Now it is plain that the forests are not inexhaustible, and that quick measures must be taken if ruin is to be avoided. Year by year the remnant is growing smaller before the axe and fire, while the laws in existence provide neither for the protection of the timber from destruction nor for its use where it is most needed.

As is shown by Mr. E. A. Bowers, formerly Inspector of the Public Land Service, the foundation of our protective policy, which has never protected, is an act passed March 1, 1817, which authorized the Secretary of the Navy to reserve lands producing live-oak and cedar, for the sole purpose of supplying timber for the navy of the United States. An extension of this law by the passage of the act of March 2, 1831, provided that if any person should cut live-oak or red cedar trees or *other timber* from the lands of the United States for any other purpose than the construction of the navy, such person should pay a fine not less than triple the value of the timber cut, and be imprisoned for a period not exceeding twelve months. Upon this old law, as Mr. Bowers points out, having the construction of a wooden navy in view, the United States government has to-day chiefly to rely in protecting its timber throughout the arid regions of the West, where none of the naval timber which the law had in mind is to be found.

By the act of June 3, 1878, timber can be taken from public lands not subject to entry under any existing laws except for minerals, by *bona fide* residents of the Rocky Mountain States and Territories and the Dakotas. Under the timber and stone act, of the same date, land in the Pacific States and Nevada, valuable mainly for timber, and unfit for cultivation if the timber is removed, can be purchased for two dollars and a half an acre, under certain restrictions. By the act of March 3, 1875, all land-grant and right-of-way railroads are authorized to take timber from the public lands adjacent to their lines for construction purposes; and they have taken it with a vengeance, destroying a hundred times more than they have used, mostly by allowing fires to run into the woods. The settlement laws, under which a settler may enter lands valuable for timber as well as for agriculture, furnish another means of obtaining title to public timber.

With the exception of the timber culture act, under which, in consideration of planting a few acres of seedlings, settlers on the treeless plains got 160 acres each, the above is the only legislation aiming to protect and promote the planting of forests. In no other way than under some one of these laws can a citizen of the United States make any use of the public forests. To show the results of the timber-planting act, it need only be stated that of the 38,000,000 acres entered under it, less than 1,000,000 acres have been patented. This means that less than 50,000 acres have been planted with stunted, woebegone, almost hopeless sprouts of trees, while at the same time the government has allowed millions of acres of the grandest forest trees to be stolen, or destroyed, or sold for nothing. Under the act of June 3, 1878, settlers in Colorado and the Territories were allowed to cut timber for mining and agricultural purposes from mineral land, which in the practical West means both cutting and burning anywhere and everywhere, for any purpose, on any sort of public land. Thus, the prospector, the miner, and mining and railroad companies are allowed by law to take all the timber they like for their mines and roads, and the forbidden settler, if there are no mineral lands near his farm or stock-ranch, or none that he knows of, can hardly be expected to forbear taking what he needs wherever he can find it. Timber is as necessary as bread, and no scheme of management failing to recognize and properly provide for this want can possibly be maintained. In any case, it will be hard to teach the pioneers that it is wrong to steal government timber. Taking from the government is with them the same as taking from nature, and their consciences flinch no more in cutting timber from the wild forests than in drawing water from a lake or river. As for reservation and protection of forests, it seems as silly and needless to them as protection and reservation of the ocean would be; both appearing to be boundless and inexhaustible.

The special land agents employed by the General Land Office to protect the public domain from timber depredations are supposed to collect testimony to sustain prosecution, and to superintend such prosecution on behalf of the government, which is represented by the district attorneys. But timber-thieves of the Western class are seldom convicted, for the good reason that most of the jurors who try such cases are themselves as guilty as those on trial. The effect of the present confused, discriminating, and unjust system has been to place almost the whole population in opposition to the government; and as conclusive of its futility, as shown by Mr. Bowers, we need only state that during the seven years from 1881 to 1887 inclusive the value of the timber reported stolen from the government lands was $36,719,935, and the amount recovered was $478,073, while the cost of the services of special agents alone was $455,000, to which must be added the expense of the trials. Thus for nearly thirty-seven million dollars' worth of timber the government got less than nothing; and the value of that consumed by running fires during the same period, without benefit even to thieves, was probably over two hundred millions of dollars. Land commissioners and Secretaries of the Interior have repeatedly called attention to this ruinous state of affairs, and asked Congress to enact the requisite legislation for reasonable reform. But, busied with tariffs, etc., Congress has given no heed to these or other appeals, and our forests, the most valuable and the most destructible of all the natural resources of the country, are being robbed and burned more rapidly than ever. The annual appropriation for so-called "protection service" is hardly sufficient to keep twenty-five timber agents in the field, and as far as any efficient protection of timber is concerned these agents themselves might as well be timber.

That a change from robbery and ruin to a permanent rational policy is urgently needed nobody with the slightest knowledge of American forests will deny. In the East and along the northern Pacific coast, where the rainfall is abundant, comparatively few care keenly what becomes of the trees as long as fuel and lumber are not noticeably dear. But in the Rocky Mountains and California and Arizona, where the forests are inflammable, and where the fertility of the lowlands depends upon irrigation, public opinion is growing stronger every year in favor of permanent protection by the federal government of all the forests that cover the sources of the streams. Even lumbermen in these regions, long accustomed to steal, are now willing and anxious to buy lumber for their mills under cover of law: some possibly from a late second growth of honesty, but most, especially the small mill-owners, simply because it no longer pays to steal where all may not only steal, but also destroy, and in particular because it costs about as much to steal timber for one mill as for ten, and therefore the ordinary lumberman can no longer compete with the large corporations. Many of the miners find that timber is already becoming scarce and dear on the denuded hills around their mills, and they too are asking for protection of forests, at least against fire. The slow-going, unthrifty farmers, also, are beginning to realize that when the timber is stripped from the mountains the irrigating streams dry up in summer, and are destructive in winter; that soil, scenery, and everything slips off with the trees: so of course they are coming into the ranks of tree-friends.

Of all the magnificent coniferous forests around the Great Lakes, once the property of the United States, scarcely any belong to it now. They have disappeared in lumber and smoke, mostly smoke, and the government got not one cent for them; only the land they were growing on was considered valuable, and two and a half dollars an acre was charged for it. Here and there in the Southern States there are still considerable areas of timbered government land, but these are comparatively unimportant. Only the forests of the West are significant in size and value, and these, although still great, are rapidly vanishing. Last summer, of the unrivaled redwood forests of the Pacific Coast Range the United States Forestry Commission could not find a single quarter-section that remained in the hands of the government.

Under the timber and stone act of 1878, which might well have been called the "dust and ashes act," any citizen of the United States could take up one hundred and sixty acres of timber land, and by paying two dollars and a half an acre for it obtain title. There was some virtuous effort made with a view to limit the operations of the act by requiring that the purchaser should make affidavit that he was entering the land exclusively for his own use, and by not allowing any association to enter more than one hundred and sixty acres. Nevertheless, under this act wealthy corporations have fraudulently obtained title to from ten thousand to twenty thousand acres or more. The plan was usually as follows: A mill company desirous of getting title to a large body of redwood or sugar-pine land first blurred the eyes and ears of the land agents, and then hired men to enter the land they wanted, and immediately deed it to the company after a nominal compliance with the law; false swearing in the wilderness against the government being held of no account. In one case which came under the observation of Mr. Bowers, it was the practice of a lumber company to hire the entire crew of every vessel which might happen to touch at any port in the redwood belt, to enter one hundred and sixty acres each and immediately deed the land to the company, in consideration of the company's paying all expenses and giving the jolly sailors fifty dollars apiece for their trouble.

By such methods have our magnificent redwoods and much of the sugar-pine forests of the Sierra Nevada been absorbed by foreign and resident capitalists. Uncle Sam is not often called a fool in business matters, yet he has sold millions of acres of timber land at two dollars and a half an acre on which a single tree was worth more than a hundred dollars. But this priceless land has been patented, and nothing can be done now about the crazy bargain. According to the everlasting laws of righteousness, even the fraudful buyers at less than one per cent of its value are making little or nothing, on account of fierce competition. The trees are felled, and about half of each giant is left on the ground to be converted into smoke and ashes; the better half is sawed into choice lumber and sold to citizens of the United States or to foreigners: thus robbing the country of its glory and impoverishing it without right benefit to anybody, — a bad, black business from beginning to end.

The redwood is one of the few conifers that sprout from the stump and roots, and it declares itself willing to begin immediately to repair the damage of the lumberman and also that of the forest-burner. As soon as a redwood is cut down or burned it sends up a crowd of eager, hopeful shoots, which, if allowed to grow, would in a few decades attain a height of a hundred feet, and the strongest of them would finally become giants as great as the original tree. Gigantic second and third growth trees are found in the redwoods, forming magnificent temple-like circles around charred ruins more than a thousand years old. But not one denuded acre in a hundred is allowed to raise a new forest growth. On the contrary, all the brains, religion, and superstition of the neighborhood are brought into play to prevent a new growth. The sprouts from the roots and stumps are cut off again and again, with zealous concern as to the best time and method of making death sure. In the clearings of one of the largest mills on the coast we found thirty men at work, last summer, cutting off redwood shoots "in the dark of the moon," claiming that all the stumps and roots cleared at this auspicious time would send up no more shoots. Anyhow, these vigorous, almost immortal trees are killed at last, and black stumps are now their only monuments over most of the chopped and burned areas.

The redwood is the glory of the Coast Range. It extends along the western slope, in a nearly continuous belt about ten miles wide, from beyond the Oregon boundary to the south of Santa Cruz, a distance of nearly four hundred miles, and in massive, sustained grandeur and closeness of growth surpasses all the other timber woods of the world. Trees from ten to fifteen feet in diameter and three hundred feet high are not uncommon, and a few attain a height of three hundred and fifty feet, or even four hundred, with a diameter at the base of fifteen to twenty feet or more, while the ground beneath them is a garden of fresh, exuberant ferns, lilies, gaultheria, and rhododendron. This grand tree, *Sequoia sempervirens*, is surpassed in size only by its near relative, *Sequoia gigantea*, or big tree, of the Sierra Nevada, if indeed it is surpassed. The sempervirens is certainly the taller of the two. The gigantea attains a greater girth, and is heavier, more noble in port, and more sublimely beautiful. These two sequoias are all that are known to exist in the world, though in former geological times the genus was common and had many species. The redwood is restricted to the Coast Range, and the big tree to the Sierra.

As timber the redwood is too good to live. The largest sawmills ever built are busy along its seaward border, "with all the modern improvements," but so immense is the yield per acre it will be long ere the supply is exhausted. The big tree is also to some extent being made into lumber. Though far less abundant than the redwood, it is, fortunately, less accessible, extending along the western flank of the Sierra in a partially interrupted belt about two hundred and fifty miles long, at a height of from four to eight thousand feet above the sea. The enormous logs, too heavy to handle, are blasted into manageable dimensions with gunpowder. A large portion of the best timber is thus shattered and destroyed, and, with the huge knotty tops, is left in ruins for tremendous fires that kill every tree within their range, great and small. Still, the species is not in danger of extinction. It has been planted and is flourishing over a great part of Europe, and magnificent sections of the aboriginal forests have been reserved as national and state parks, — the Mariposa Sequoia Grove, near Yosemite, managed by the State of California, and the General Grant and Sequoia national parks on the King's, Kaweah, and Tule rivers, efficiently guarded by a small troop of United States cavalry under the direction of the Secretary of the Interior. But there is not a single specimen of the redwood in any national park. Only by gift or purchase, so far as I know, can the government get back into its possession a single acre of this wonderful forest.

The legitimate demands on the forests that have passed into private ownership, as well as those in the hands of the government, are increasing every year with the rapid settlement and upbuilding of the country, but the methods of lumbering are as yet grossly wasteful. In most mills only the best portions of the best trees are used, while the ruins are left on the ground to feed great fires which kill much of what is left of the less desirable timber, together with the seedlings on which the permanence of the forest depends. Thus every mill is a centre of destruction far more severe from waste and fire than from use. The same thing is true of the mines, which consume and destroy indirectly immense quantities of timber with their innumerable fires, accidental or set to make open ways, and often without regard to how far they run. The prospector deliberately sets fires to clear off the woods just where they are densest, to lay the rocks bare and make the discovery of mines easier. Sheep-owners and their shepherds also set fires everywhere through the woods in the fall to facilitate the march of their countless flocks the next summer, and perhaps in some places to improve the pasturage. The axe is not yet at the root of every tree, but the sheep is, or was before the national parks were established and guarded by the military, the only effective and reliable arm of the government free from the blight of politics. Not only do the shepherds, at the driest time of the year, set fire to everything that will burn, but the sheep consume every green leaf, not sparing even the young conifers when they are in a starving condition from crowding, and they rake and dibble the loose soil of the mountain sides for the spring floods to wash away, and thus at last leave the ground barren.

Of all the destroyers that infest the woods the shake-maker seems the happiest. Twenty or thirty years ago, shakes, a kind of long boardlike shingles split with a mallet and a frow, were in great demand for covering barns and sheds, and many are used still in preference to common shingles, especially those made from the sugar-pine, which do not warp or crack in the hottest sunshine. Drifting adventurers in California, after harvest and threshing are over, oftentimes meet to discuss their plans for the winter, and their talk is interesting. Once, in a company of this kind, I heard a man say, as he peacefully smoked his pipe: "Boys, as soon as this job 's done I 'm goin' into the duck business. There 's big money in it, and your grub costs nothing. Tule Joe made five hundred dollars last winter on mallard and teal. Shot 'em on the Joaquin, tied 'em in dozens by the neck, and shipped 'em to San Francisco. And when he was tired wading in the sloughs and touched with rheumatiz, he just knocked off on ducks, and went to the Contra Costa hills for dove and quail. It 's a mighty good business, and you 're your own boss, and the whole thing 's fun."

Another of the company, a bushy-bearded fellow, with a trace of brag in his voice, drawled out: "Bird business is well enough for some, but bear is my game, with a deer and a California lion thrown in now and then for change. There 's always a market for bear grease, and sometimes you can sell the hams. They 're good as hog hams any day. And you are your own boss in my business, too, if the bears ain't too big and too many for you. Old grizzlies I despise, — they want cannon to kill 'em; but the blacks and browns are beauties for grease, and when once I get 'em just right, and draw a bead on 'em, I fetch 'em every time." Another said he was going to catch up a lot of mustangs as soon as the rains set in, hitch them to a gang-plough, and go to farming on the San Joaquin plains for wheat. But most preferred the shake business, until something more profitable and as sure could be found, with equal comfort and independence.

With a cheap mustang or mule to carry a pair of blankets, a sack of flour, a few pounds of coffee, and an axe, a frow, and a cross-cut saw, the shake-maker ascends the mountains to the pine belt where it is most accessible, usually by some mine or mill road. Then he strikes off into the virgin woods, where the sugar-pine, king of all the hundred species of pines in the world in size and beauty, towers on the open sunny slopes of the Sierra in the fullness of its glory. Selecting a favorable spot for a cabin near a meadow with a stream, he unpacks his animal and stakes it out on the meadow. Then he chops into one after another of the pines, until he finds one that he feels sure will split freely, cuts this down, saws off a section four feet long, splits it, and from this first cut, perhaps seven feet in diameter, he gets shakes enough for a cabin and its furniture, — walls, roof, door, bedstead, table, and stool. Besides his labor, only a few pounds of nails are required. Sapling poles form the frame of the airy building, usually about six feet by eight in size, on which the shakes are nailed, with the edges overlapping. A few bolts from the same section that the shakes were made from are split into square sticks and built up to form a chimney, the inside and interspaces being plastered and filled in with mud. Thus, with abundance of fuel, shelter and comfort by his own fireside are secured. Then he goes to work sawing and splitting for the market, tying the shakes in bundles of fifty or a hundred. They are four feet long, four inches wide, and about one fourth of an inch thick. The first few thousands he sells or trades at the nearest mill or store, getting provisions in exchange. Then he advertises, in whatever way he can, that he has excellent sugar-pine shakes for sale, easy of access and cheap.

Only the lower, perfectly clear, free-splitting portions of the giant pines are used, — perhaps ten to twenty feet from a tree two hundred and fifty in height; all the rest is left a mass of ruins, to rot or to feed the forest fires, while thousands are hacked deeply and rejected in proving the grain. Over nearly all of the more accessible slopes of the Sierra and Cascade mountains in southern Oregon, at a height of from three to six thousand feet above the sea, and for a distance of about six hundred miles, this waste and confusion extends. Happy robbers! dwelling in the most beautiful woods, in the most salubrious climate, breathing delightful doors both day and night, drinking cool living water, — roses and lilies at their feet in the spring, shedding fragrance and ringing bells as if cheering them on in their desolating work. There is none to say them nay. They buy no land, pay no taxes, dwell in a paradise with no forbidding angel either from Washington or from heaven. Every one of the frail shake shanties is a centre of destruction, and the extent of the ravages wrought in this quiet way is in the aggregate enormous.

It is not generally known that, notwithstanding the immense quantities of timber cut every year for foreign and home markets and mines, from five to ten times as much is destroyed as is used, chiefly by running forest fires that only the federal government can stop. Travelers through the West in summer are not likely to forget the fire-work displayed along the various railway tracks. Thoreau, when contemplating the destruction of the forests on the east side of the continent, said that soon the country would be so bald that every man would have to grow whiskers to hide its nakedness, but he thanked God that at least the sky was safe. Had he gone West he would have found out that the sky was not safe; for all through the summer months, over most of the mountain regions, the smoke of mill and forest fires is so thick and black that no sunbeam can pierce it. The whole sky, with clouds, sun, moon, and stars, is simply blotted out. There is no real sky and no scenery. Not a mountain is left in the landscape. At least none is in sight from the lowlands, and they all might as well be on the moon, as far as scenery is concerned.

The half dozen transcontinental railroad companies advertise the beauties of their lines in gorgeous many-colored folders, each claiming its as the "scenic route." "The route of superior desolation" — the smoke, dust, and ashes route — would be a more truthful description. Every train rolls on through dismal smoke and barbarous melancholy ruins; and the companies might well cry in their advertisements: "Come! travel our way. Ours is the blackest. It is the only genuine Erebus route. The sky is black and the ground is black, and on either side there is a continuous border of black stumps and logs and blasted trees appealing to heaven for help as if still half alive, and their mute eloquence is most interestingly touching. The blackness is perfect. On account of the superior skill of our workmen, advantages of climate, and the kind of trees, the charring is generally deeper along our line, and the ashes are deeper, and the confusion and desolation displayed can never be rivaled. No other route on this continent so fully illustrates the abomination of desolation." Such a claim would be reasonable, as each seems the worst, whatever route you chance to take.

Of course a way had to be cleared through the woods. But the felled timber is not worked up into firewood for the engines and into lumber for the company's use: it is left lying in vulgar confusion, and is fired from time to time by sparks from locomotives or by the workmen camping along the line. The fires, whether accidental or set, are allowed to run into the woods as far as they may, thus assuring comprehensive destruction. The directors of a line that guarded against fires, and cleared a clean gap edged with living trees, and fringed and mantled with the grass and flowers and beautiful seedlings that are ever ready and willing to spring up, might justly boast of the beauty of their road; for nature is always ready to heal every scar. But there is no such road on the western side of the continent. Last summer, in the Rocky Mountains, I saw six fires started by sparks from a locomotive within a distance of three miles, and nobody was in sight to prevent them from spreading. They might run into the adjacent forests and burn the timber from hundreds of square miles; not a man in the State would care to spend an hour in fighting them, as long as his own fences and buildings were not threatened.

Notwithstanding all the waste and use which have been going on unchecked like a storm for more than two centuries, it is not yet too late, though it is high time, for the government to begin a rational administration of its forests. About seventy million acres it still owns, — enough for all the country, if wisely used. These residual forests are generally on mountain slopes, just where they are doing the most good, and where their removal would be followed by the greatest number of evils; the lands they cover are too rocky and high for agriculture, and can never be made as valuable for any other crop as for the present crop of trees. It has been shown over and over again that if these mountains were to be stripped of their trees and underbrush, and kept bare and sodless by hordes of sheep and the innumerable fires the shepherds set, besides those of the millmen, prospectors, shakemakers, and all sorts of adventurers, both lowlands and mountains would speedily become little better than deserts, compared with their present beneficent fertility. During heavy rainfalls and while the winter accumulations of snow were melting, the larger streams would swell into destructive torrents; cutting deep, rugged-edged gullies, carrying away the fertile humus and soil as well as sand and rocks, filling up and overflowing their lower channels, and covering the lowland fields with raw detritus. Drought and barrenness would follow.

In their natural condition, or under wise management, keeping out destructive sheep, preventing fires, selecting the trees that should be cut for lumber, and preserving the young ones and the shrubs and sod of herbaceous vegetation, these forests would be a never failing fountain of wealth and beauty. The cool shades of the forest give rise to moist beds and currents of air, and the sod of grasses and the various flowering plants and shrubs thus fostered, together with the network and sponge of tree roots, absorb and hold back the rain and the waters from melting snow, compelling them to ooze and percolate and flow gently through the soil in streams that never dry. All the pine needles and rootlets and blades of grass, and the fallen decaying trunks of trees, are dams, storing the bounty of the clouds and dispensing it in perennial life-giving streams, instead of allowing it to gather suddenly and rush headlong in short-lived devastating floods. Everybody on the dry side of the continent is beginning to find this out, and, in view of the waste going on, is growing more and more anxious for government protection. The outcries we hear against forest reservations come mostly from thieves who are wealthy and steal timber by wholesale. They have so long been allowed to steal and destroy in peace that any impediment to forest robbery is denounced as a cruel and irreligious interference with "vested rights," likely to endanger the repose of all ungodly welfare.

Gold, gold, gold! How strong a voice that metal has!

"O wae for the siller, it is sae preva'lin'."

Even in Congress, a sizable chunk of gold, carefully concealed, will outtalk and outfight all the nation on a subject like forestry, well smothered in ignorance, and in which the money interests of only a few are conspicuously involved. Under these circumstances, the bawling, blethering oratorical stuff drowns the voice of God himself. Yet the dawn of a new day in forestry is breaking. Honest citizens see that only the rights of the government are being trampled, not those of the settlers. Merely what belongs to all alike is reserved, and every acre that is left should be held together under the federal government as a basis for a general policy of administration for the public good. The people will not always be deceived by selfish opposition, whether from lumber and mining corporations or from sheepmen and prospectors, however cunningly brought forward underneath fables and gold.

Emerson says that things refuse to be mismanaged long. An exception would seem to be found in the case of our forests, which have been mismanaged rather long, and now come desperately near being like smashed eggs and spilt milk. Still, in the long run the world does not move backward. The wonderful advance made in the last few years, in creating four national parks in the West, and thirty forest reservations, embracing nearly forty million acres; and in the planting of the borders of streets and highways and spacious parks in all the great cities, to satisfy the natural taste and hunger for landscape beauty and righteousness that God has put, in some measure, into every human being and animal, shows the trend of awakening public opinion. The making of the far-famed New York Central Park was opposed by even good men, with misguided pluck, perseverance, and ingenuity; but straight right won its way, and now that park is appreciated. So we confidently believe it will be with our great national parks and forest reservations. There will be a period of indifference on the part of the rich, sleepy with wealth, and of the toiling millions, sleepy with poverty, most of whom never saw a forest; a period of screaming protest and objection from the plunderers, who are as unconscionable and enterprising as Satan. But light is surely coming, and the friends of destruction will preach and bewail in vain.

The United States government has always been proud of the welcome it has extended to good men of every nation, seeking freedom and homes and bread. Let them be welcomed still as nature welcomes them, to the woods as well as to the prairies and plains. No place is too good for good men, and still there is room. They are invited to heaven, and may well be allowed in America. Every place is made better by them. Let them be as free to pick gold and gems from the hills, to cut and hew, dig and plant, for homes and bread, as the birds are to pick berries from the wild bushes, and moss and leaves for nests. The ground will be glad to feed them, and the pines will come down from the mountains for their homes as willingly as the cedars came from Lebanon for Solomon's temple. Nor will the woods be the worse for this use, or their benign influences be diminished any more than the sun is diminished by shining. Mere destroyers, however, tree-killers, spreading death and confusion in the fairest groves and gardens ever planted, let the government hasten to cast them out and make an end of them. For it must be told again and again, and be burningly borne in mind, that just now, while protective measures are being deliberated languidly, destruction and use are speeding on faster and farther every day. The axe and saw are insanely busy, chips are flying thick as snowflakes, and every summer thousands of acres of priceless forests, with their underbrush, soil, springs, climate, scenery, and religion, are vanishing away in clouds of smoke, while, except in the national parks, not one forest guard is employed.

All sorts of local laws and regulations have been tried and found wanting, and the costly lessons of our own experience, as well as that of every civilized nation, show conclusively that the fate of the remnant of our forests is in the hands of the federal government, and that if the remnant is to be saved at all, it must be saved quickly.

Any fool can destroy trees. They cannot run away; and if they could, they would still be destroyed, — chased and hunted down as long as fun or a dollar could be got out of their bark hides, branching horns, or magnificent bole backbones. Few that fell trees plant them; nor would planting avail much towards getting back anything like the noble primeval forests. During a man's life only saplings can be grown, in the place of the old trees — tens of centuries old — that have been destroyed. It took more than three thousand years to make some of the trees in these Western woods, — trees that are still standing in perfect strength and beauty, waving and singing in the mighty forests of the Sierra. Through all the wonderful, eventful centuries since Christ's time — and long before that — God has cared for these trees, saved them from drought, disease, avalanches, and a thousand straining, leveling tempests and floods; but he cannot save them from fools, — only Uncle Sam can do that.

John Muir.

SOME UNPUBLISHED LETTERS OF DEAN SWIFT.

I.

John Forster, who lived to complete but one of the three volumes in which he had planned to write the Life of Jonathan Swift, speaks in the preface of his hero's correspondence "with his friend Knightley Chetwode, of Woodbrooke, during the seventeen years (1714–1731) which followed his appointment to the deanery of St. Patrick's. Of these letters," Forster goes on to say, "the richest addition to the correspondence of this most masterly of English letter-writers since it was first collected, more does not need to be said here; but of the late representative of the Chetwode family I crave permission to add a word. His rare talents and taste suffered from his delicate health and fastidious temperament, but in my life I have seen few things more delightful than his pride in the connection of his race and name with the companionship of Swift. Such was the jealous care with which he preserved the letters, treasuring them as an heirloom of honour, that he would never allow them to be moved from his family seat; and when, with his own hand, he had made careful transcript of them for me, I had to visit him at Woodbrooke to collate his copy with the originals. There I walked with him through avenues of trees which Swift was said to have planted."

As Forster did not bring down the Life later than 1711, — three years and more before the first of these letters was written, — he made scarcely any use of the correspondence. He refers to it twice, and twice only. On his death, the copy of the originals, with the corrections he

Managing for Forest Resilience in a Changing Climate

A Factsheet for Forest Landowners in the Western Allegheny Plateau Region

W. Allegheny Plateau

Your Forest and Climate Change

Climate change is impacting our forests. The Bureau of Forestry recommends taking steps to establish forests resilient to a changing climate.

Current and Projected Climate Shifts

Pennsylvania's climate has already warmed by 1.8^{o}F since the early 1900s. Scientists know the rate of warming is accelerating and expect as much as a 5.9^{o}F increase by 2050.

Average winter temperatures are increasing more than any other season, by 1.3^{o}F per decade since 1970.

Our climate has also become wetter. Average annual rainfall has increased 10% over the last century and heavy downpours have increased by 71% in the northeastern US.

The following information identifies potential forest vulnerabilities to climate change and management strategies to encourage forest resilience.

Forest Vulnerabilities

- New forest pests and greater impact of existing forest pests
- More invasive plants
- More fungal outbreaks
- More windthrow
- More soil erosion

- Streambank destabilization
- Changing forest community
- Unpredictable seasonal temperatures and extremes

Management Strategies

- Increase vigilance for forest pests, invasive species, and pathogens (with aggressive follow-up)
- Protect existing forests
- Reforest deforested lands
- Plant a diversity of native trees, including known climate resilient trees in your region
- Work with your county service forester and consulting forester to create a forest management plan

pennsylvania
DEPARTMENT OF CONSERVATION AND NATURAL RESOURCES
www.dcnr.pa.gov

Resilient Trees

Oaks: scrub, white, black, chinkapin, scarlet, pin, shingle

Hickories: mockernut, pignut, shagbark, bitternut

Other: black walnut, sycamore, slippery elm, black gum, eastern redbud, osage orange, cottonwood, hackberry, hophornbeam, hornbeam, sassafras, Virginia pine, pitch pine, boxelder, black locust, honey locust, yellow buckeye, persimmon, pawpaw, flowering dogwood, black willow

Trees at Risk

Maples: mountain, striped, black
Aspens: big tooth, quaking

Birches: paper, yellow, gray

Evergreens: hemlock, red pine, jack pine, red spruce, white spruce

Other: American mountain ash, fire cherry, balsam poplar, American beech, chokecherry, pin cherry

**Resilient and At Risk lists are based on modeling by the USDA Forest Service. With all models, there is some uncertainty. Some species may fare better (or worse) in different settings depending on prevailing ecological factors at the site. Landowners should enhance diversity to improve climate resilience and not necessarily limit their management decisions based solely on these models.*

More Information

DCNR Bureau of Forestry
PaForester@pa.gov
717-787-2703

DCNR's Climate Change Page
https://www.dcnr.pa.gov/Conservation/ClimateChange/Pages/default.aspx

Service Forester Directory
https://maps.dcnr.pa.gov/landownerassist/

Pennsylvania

States

Climate Change Atlas Tree Species

Current and Potential Future Habitat, Capability, and Migration

USDA Forest Service
Northern Research Station
Landscape Change Research Group
Iverson, Peters, Prasad, Matthews

	sq. km	sq. mi	FIA Plots
Area of Region	117,294	45,287	2,980

Species Information

The columns below provide breif summaries of the species associated with the region and described in the table on the next pages. Definitions are provided in the Excel file for this region.

Genus	Species
Ash	3
Hickory	4
Maple	8
Oak	15
Pine	8
Other	53
	91

Abundance	
Abundant	5
Common	21
Rare	65
Absent	36
	127

	Model Reliability	Adaptability
High	27	30
Medium	42	67
Low	46	24
FIA	13	
	128	**121**

Potential Change in Habitat Suitability	Scenario RCP45	Scenario RCP85
Increase	26	32
No Change	17	15
Decrease	35	31
New	29	30
Unknown	21	20
	128	**128**

Capability to Cope or Persist	Scenario RCP45	Scenario RCP85
Very Good	9	10
Good	19	21
Fair	8	11
Poor	20	15
Very Poor	22	19
FIA Only	7	7
Unknown	8	7
	93	**90**

Migration Potential	SHIFT RCP45	SHIFT RCP85
Likely	1	1
Infill	14	17
Migrate	13	12
	28	**30**

Potential Changes in Climate Variables

Temperature (°F)

	Scenario	**2009**	**2039**	**2069**	**2099**
Annual Average	CCSM45	48.7	50.6	53.0	53.2
	CCSM85	48.7	51.0	53.6	56.8
	GFDL45	48.7	52.1	54.7	55.8
	GFDL85	48.7	52.2	55.9	60.2
	HAD45	48.7	51.6	55.0	56.6
	HAD85	48.7	51.7	56.0	61.1
Growing Season May—Sep	CCSM45	65.0	66.9	69.0	69.5
	CCSM85	65.0	67.3	69.9	73.8
	GFDL45	65.0	68.7	72.1	73.7
	GFDL85	65.0	69.4	73.6	78.3
	HAD45	65.0	68.3	71.3	73.5
	HAD85	65.0	67.9	73.1	78.7
Coldest Month Average	CCSM45	23.8	25.7	27.5	27.9
	CCSM85	23.8	26.4	27.7	29.5
	GFDL45	23.8	26.9	28.0	28.7
	GFDL85	23.8	27.1	28.4	30.2
	HAD45	23.8	25.3	28.0	28.2
	HAD85	23.8	26.3	28.4	31.0
Warmest Month Average	CCSM45	70.8	73.0	74.2	74.5
	CCSM85	70.8	73.4	75.1	77.3
	GFDL45	70.8	73.9	76.2	77.5
	GFDL85	70.8	75.4	77.9	80.9
	HAD45	70.8	74.5	76.4	77.9
	HAD85	70.8	74.8	78.1	82.0

Precipitation (in)

	Scenario	**2009**	**2039**	**2069**	**2099**
Annual Total	CCSM45	43.3	45.2	45.3	48.1
	CCSM85	43.3	45.4	47.2	49.8
	GFDL45	43.3	48.0	50.1	49.8
	GFDL85	43.3	45.1	50.0	51.5
	HAD45	43.3	45.1	45.8	45.4
	HAD85	43.3	46.9	44.0	47.2
Growing Season May—Sep	CCSM45	20.4	21.8	21.7	22.8
	CCSM85	20.4	21.2	22.1	22.4
	GFDL45	20.4	21.6	21.6	21.7
	GFDL85	20.4	20.0	21.3	21.3
	HAD45	20.4	21.2	19.4	19.7
	HAD85	20.4	21.1	18.5	19.3

NOTE: For the six climate variables, four 30-year periods are used to indicate six potential future trajectories. The period ending in 2009 is based on modeled observations from the PRISM Climate Group and the three future periods were obtained from the NASA NEX-DCP30 dataset. Future climate projections from three models under two emission scenarios show estimates of each climate variable within the region. The three models are CCSM4, GFDL CM3, and HadGEM2-ES and the emission scenarios are the 4.5 and 8.5 RCP. The average value for the region is reported, even though locations within the region may vary substantially based on latitude, elevation, land-use, or other factors.

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Pennsylvania

States

Climate Change Atlas Tree Species

Current and Potential Future Habitat, Capability, and Migration

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Iverson, Peters, Prasad, Matthews

Common Name	Scientific Name	Range	MR	%Cell	FIAsum	FIAiv	ChngCl45	ChngCl85	Adap	Abund	Capabil45	Capabil85	SHIFT45	SHIFT85	SSO	N
red maple	Acer rubrum	WDH	High	86.6	1664.6	19.3	Sm. dec.	Lg. dec.	High	Abundant	Good	Good			1	1
black cherry	Prunus serotina	WDL	Medium	70.7	914.0	13.4	Sm. dec.	Sm. dec.	Low	Abundant	Fair	Fair			0	2
sugar maple	Acer saccharum	WDH	High	46	624.7	12.4	No change	No change	High	Abundant	Very Good	Very Good			1	3
northern red oak	Quercus rubra	WDH	Medium	62.9	531.9	8.8	No change	No change	High	Abundant	Very Good	Very Good			1	4
chestnut oak	Quercus prinus	NDH	High	42.8	504.1	12.8	Sm. inc.	No change	High	Abundant	Very Good	Very Good			1	5
sweet birch	Betula lenta	NDH	High	57.5	433.1	7.9	Sm. dec.	Lg. dec.	Low	Common	Poor	Very Poor			0	6
white ash	Fraxinus americana	WDL	Medium	57.4	403.0	7.4	No change	No change	Low	Common	Poor	Poor			0	7
eastern hemlock	Tsuga canadensis	NSH	High	35.4	370.1	10.3	Lg. dec.	Lg. dec.	Low	Common	Very Poor	Very Poor			0	8
white oak	Quercus alba	WDH	Medium	41.1	319.8	8.1	Lg. inc.	Lg. inc.	High	Common	Very Good	Very Good			1	9
American beech	Fagus grandifolia	WDH	High	37.7	285.5	7.7	No change	No change	Medium	Common	Fair	Fair			1	10
eastern white pine	Pinus strobus	WDH	High	28.3	216.0	7.5	No change	Sm. dec.	Low	Common	Poor	Poor			0	11
yellow-poplar	Liriodendron tulipifera	WDH	High	29.9	186.0	8.2	Lg. inc.	Lg. inc.	High	Common	Very Good	Very Good			1	12
black locust	Robinia pseudoacacia	NDH	Low	22.4	164.0	7.8	Sm. inc.	Sm. inc.	Medium	Common	Good	Good			1	13
black oak	Quercus velutina	WDH	High	34.6	157.0	5.3	Lg. inc.	Lg. inc.	Medium	Common	Very Good	Very Good			1	14
blackgum	Nyssa sylvatica	WDL	Medium	33.6	124.2	4.1	Sm. inc.	Lg. inc.	High	Common	Very Good	Very Good			1	15
sassafras	Sassafras albidum	WSL	Low	34.1	115.8	4.0	Sm. inc.	Sm. inc.	Medium	Common	Good	Good			1	16
scarlet oak	Quercus coccinea	WDL	Medium	21.5	101.2	4.7	Lg. inc.	Lg. inc.	Medium	Common	Very Good	Very Good			1	17
black walnut	Juglans nigra	WDH	Low	17.4	98.6	7.5	Sm. inc.	Sm. inc.	Medium	Common	Good	Good			1	18
quaking aspen	Populus tremuloides	WDH	High	11.9	95.6	7.2	Lg. dec.	Lg. dec.	Medium	Common	Poor	Poor			0	19
American elm	Ulmus americana	WDH	Medium	19.4	95.2	4.9	No change	Sm. inc.	Medium	Common	Fair	Good			1	20
bigtooth aspen	Populus grandidentata	NSL	Medium	17.5	94.4	5.4	Lg. dec.	Lg. dec.	Medium	Common	Poor	Poor			0	21
yellow birch	Betula alleghaniensis	NDL	High	21.3	91.5	4.1	Lg. dec.	Lg. dec.	Medium	Common	Poor	Poor			0	22
pignut hickory	Carya glabra	WDL	Medium	23.5	76.5	3.5	Lg. inc.	Lg. inc.	Medium	Common	Very Good	Very Good			1	23
American basswood	Tilia americana	WSL	Medium	18.5	69.0	3.8	Sm. inc.	Sm. inc.	Medium	Common	Good	Good			1	24
shagbark hickory	Carya ovata	WSL	Medium	18.7	64.2	3.6	Sm. inc.	Sm. inc.	Medium	Common	Good	Good			1	25
eastern hophornbeam; ironw	Ostrya virginiana	WSL	Low	22.7	53.4	2.3	No change	Sm. inc.	High	Common	Good	Very Good			1	26
Scots pine	Pinus sylvestris	NSH	FIA	5.3	47.7	8.9	Unknown	Unknown	NA	Rare	NNIS	NNIS			0	27
Virginia pine	Pinus virginiana	NDH	High	5.4	46.4	9.7	No change	Sm. inc.	Medium	Rare	Poor	Fair	Infill +	Infill +	1	28
serviceberry	Amelanchier spp.	NSL	Low	24.6	44.0	1.8	Sm. dec.	Sm. dec.	Medium	Rare	Very Poor	Very Poor			0	29
bitternut hickory	Carya cordiformis	WSL	Low	11.9	41.4	3.9	Sm. inc.	Lg. inc.	High	Rare	Good	Good	Infill ++	Infill ++	1	30
mockernut hickory	Carya alba	WDL	Medium	12.8	37.9	3.9	Lg. inc.	Lg. inc.	High	Rare	Good	Good			1	31
Norway spruce	Picea abies	NSH	FIA	3.8	35.0	8.5	Unknown	Unknown	NA	Rare	NNIS	NNIS			0	32
pin cherry	Prunus pensylvanica	NSL	Low	8.5	34.4	4.2	Lg. dec.	Lg. dec.	Medium	Rare	Very Poor	Very Poor			0	33
red pine	Pinus resinosa	NSH	Medium	4.5	32.8	8.0	Lg. dec.	Very Lg. dec.	Low	Rare	Very Poor	Lost			0	34
pitch pine	Pinus rigida	NSH	High	6.6	32.5	4.8	Sm. dec.	Sm. dec.	Medium	Rare	Very Poor	Very Poor			0	35
slippery elm	Ulmus rubra	WSL	Low	8.7	32.1	3.9	No change	No change	Medium	Rare	Poor	Poor	Infill +	Infill +	1	36
American hornbeam; muscle	Carpinus caroliniana	WSL	Low	11.6	31.0	2.5	Sm. dec.	Sm. inc.	Medium	Rare	Very Poor	Fair			1	37
cucumbertree	Magnolia acuminata	NSL	Low	10.8	27.5	2.4	Sm. dec.	Sm. dec.	Medium	Rare	Very Poor	Very Poor			0	38
black willow	Salix nigra	NSH	Low	1.3	25.1	13.4	Sm. dec.	No change	Low	Rare	Very Poor	Very Poor			2	39
ailanthus	Ailanthus altissima	NSL	FIA	7.7	25.1	5.2	Unknown	Unknown	NA	Rare	NNIS	NNIS			0	40
boxelder	Acer negundo	WSH	Low	7.8	23.8	5.6	No change	No change	High	Rare	Fair	Fair	Infill +	Infill +	1	41
sycamore	Platanus occidentalis	NSL	Low	3.3	22.0	5.8	Sm. inc.	Lg. inc.	Medium	Rare	Fair	Good	Infill +	Infill ++	2	42
green ash	Fraxinus pennsylvanica	WSH	Low	3.1	20.1	7.7	No change	Sm. inc.	Medium	Rare	Poor	Fair	Infill +	Infill +	2	43
silver maple	Acer saccharinum	NSH	Low	2.6	18.4	8.3	No change	No change	High	Rare	Fair	Fair	Infill +	Infill +	2	44
eastern redcedar	Juniperus virginiana	WDH	Medium	3.5	16.4	5.9	Lg. inc.	Lg. inc.	Medium	Rare	Good	Good	Infill ++	Infill ++	2	45
pin oak	Quercus palustris	NSH	Low	3.9	16.1	6.1	No change	No change	Low	Rare	Very Poor	Very Poor			2	46
hackberry	Celtis occidentalis	WDH	Medium	4.3	15.9	5.2	No change	No change	High	Rare	Fair	Fair	Infill +	Infill +	2	47

Pennsylvania

States

Climate Change Atlas Tree Species

Current and Potential Future Habitat, Capability, and Migration

USDA Forest Service
Northern Research Station
Landscape Change Research Group
Iverson, Peters, Prasad, Matthews

Common Name	Scientific Name	Range	MR	%Cell	FIAsum	FIAiv	ChngCl45	ChngCl85	Adap	Abund	Capabil45	Capabil85	SHIFT45	SHIFT85	SSO	N
paper birch	Betula papyrifera	WDH	High	3.1	15.6	4.4	Lg. dec.	Lg. dec.	Medium	Rare	Very Poor	Very Poor			0	48
Norway maple	Acer platanoides	NSL	FIA	4.9	15.1	6.9	Unknown	Unknown	NA	Rare	NNIS	NNIS			0	49
white spruce	Picea glauca	NSL	Medium	1.4	12.9	10.8	Lg. dec.	Lg. dec.	Medium	Rare	Very Poor	Very Poor			0	50
shingle oak	Quercus imbricaria	NDH	Medium	1.5	11.8	6.4	Sm. dec.	Sm. dec.	Medium	Rare	Very Poor	Very Poor			2	51
swamp white oak	Quercus bicolor	NSL	Low	1.7	10.6	7.9	Sm. dec.	Lg. dec.	Medium	Rare	Very Poor	Very Poor			2	52
gray birch	Betula populifolia	NSL	Low	3.2	9.7	3.1	Lg. dec.	Lg. dec.	Medium	Rare	Very Poor	Very Poor			0	53
sweetgum	Liquidambar styraciflua	WDH	High	0.4	9.0	8.8	Lg. inc.	Lg. inc.	Medium	Rare	Good	Good			2	54
butternut	Juglans cinerea	NSLX	FIA	1.7	7.8	5.5	Unknown	Unknown	Low	Rare	FIA Only	FIA Only			0	55
black ash	Fraxinus nigra	WSH	Medium	0.4	7.6	9.7	Lg. dec.	Lg. dec.	Low	Rare	Very Poor	Very Poor			0	56
eastern cottonwood	Populus deltoides	NSH	Low	0.7	7.4	7.9	No change	No change	Medium	Rare	Poor	Poor	Infill +	Infill +	2	57
bear oak; scrub oak	Quercus ilicifolia	NSLX	FIA	1.6	7.3	4.8	Unknown	Unknown	Medium	Rare	FIA Only	FIA Only			0	58
flowering dogwood	Cornus florida	WDL	Medium	7.2	6.3	1.1	Sm. inc.	Sm. inc.	Medium	Rare	Fair	Fair	Infill +	Infill +	1	59
Osage-orange	Maclura pomifera	NDH	Medium	1.8	5.8	9.1	Sm. dec.	No change	High	Rare	Poor	Fair	Infill +	Infill +	2	60
red spruce	Picea rubens	NDH	High	1.2	5.6	4.7	Lg. dec.	Lg. dec.	Low	Rare	Very Poor	Very Poor			0	61
chokecherry	Prunus virginiana	NSLX	FIA	1.3	5.2	3.8	Unknown	Unknown	Medium	Rare	FIA Only	FIA Only			0	62
black maple	Acer nigrum	NSH	Low	0.6	4.9	8.2	Lg. dec.	Lg. dec.	High	Rare	Poor	Poor			0	63
loblolly pine	Pinus taeda	WDH	High	0.2	3.3	5.7	Lg. inc.	Lg. inc.	Medium	Rare	Good	Good			2	64
eastern redbud	Cercis canadensis	NSL	Low	1.6	2.1	1.9	Lg. inc.	Lg. inc.	Medium	Rare	Good	Good			0	65
jack pine	Pinus banksiana	NSH	Medium	0.2	1.9	11.1	Lg. dec.	Lg. dec.	High	Rare	Poor	Poor			0	66
tamarack (native)	Larix laricina	NSH	High	0.6	1.8	5.7	Lg. dec.	Lg. dec.	Low	Rare	Very Poor	Very Poor			0	67
honeylocust	Gleditsia triacanthos	NSH	Low	0.3	1.5	5.9	Sm. dec.	Lg. inc.	High	Rare	Poor	Good			2	68
red mulberry	Morus rubra	NSL	Low	1.2	1.5	8.0	No change	No change	Medium	Rare	Poor	Poor	Infill +	Infill +	2	69
paulownia	Paulownia tomentosa	NSL	FIA	0.8	1.4	14.5	Unknown	Unknown	NA	Rare	NNIS	NNIS			0	70
Table Mountain pine	Pinus pungens	NSL	Low	0.5	1.3	2.6	Sm. dec.	Sm. dec.	High	Rare	Poor	Poor		Infill +	2	71
common persimmon	Diospyros virginiana	NSL	Low	0	1.2	4.0	Lg. inc.	Lg. inc.	High	Rare	Good	Good			2	72
river birch	Betula nigra	NSL	Low	0.2	0.9	1.1	Sm. dec.	Sm. inc.	Medium	Rare	Very Poor	Fair		Infill +	2	73
American chestnut	Castanea dentata	NSLX	FIA	1.9	0.8	0.5	Unknown	Unknown	Medium	Rare	FIA Only	FIA Only			0	74
balsam poplar	Populus balsamifera	NSH	Medium	0.2	0.6	3.3	Lg. dec.	Lg. dec.	Medium	Rare	Very Poor	Very Poor			0	75
bur oak	Quercus macrocarpa	NDH	Medium	0.1	0.4	4.7	Sm. dec.	Sm. dec.	High	Rare	Poor	Poor		Infill +	2	76
yellow buckeye	Aesculus flava	NSL	Low	0.1	0.4	3.8	Lg. dec.	Lg. dec.	Low	Rare	Very Poor	Very Poor			0	77
white mulberry	Morus alba	NSL	FIA	0.5	0.3	1.5	Unknown	Unknown	NA	Rare	NNIS	NNIS			0	78
swamp chestnut oak	Quercus michauxii	NSL	Low	0.5	0.3	0.8	No change	No change	Medium	Rare	Poor	Poor	Infill +	Infill +	2	79
chinkapin oak	Quercus muehlenbergii	NSL	Medium	0	0.2	0.2	Lg. inc.	Lg. inc.	Medium	Rare	Good	Good			2	80
Kentucky coffeetree	Gymnocladus dioicus	NSLX	FIA	0.1	0.2	2.1	Unknown	Unknown	Medium	Rare	FIA Only	FIA Only			0	81
northern catalpa	Catalpa speciosa	NSHX	FIA	0.3	0.1	5.9	Unknown	Unknown	Medium	Rare	FIA Only	FIA Only			0	82
pawpaw	Asimina triloba	NSL	Low	0.4	0.1	1.0	Lg. dec.	Very Lg. dec.	Medium	Rare	Very Poor	Lost			0	83
American holly	Ilex opaca	NSL	Medium	0.1	0.1	0.5	Lg. inc.	Lg. inc.	Medium	Rare	Good	Good			2	84
striped maple	Acer pensylvanicum	NSL	Medium	2.7	0.1	4.1	Sm. dec.	Sm. dec.	Medium	Rare	Very Poor	Very Poor			0	85
rock elm	Ulmus thomasii	NSLX	FIA	0.3	0.1	4.0	Unknown	Unknown	Low	Rare	FIA Only	FIA Only			0	86
sourwood	Oxydendrum arboreum	NDL	High	0	0.1	0.3	Lg. inc.	Lg. inc.	High	Rare	Good	Good			2	87
southern red oak	Quercus falcata	WDL	Medium	0	0.1	0.3	Lg. inc.	Lg. inc.	High	Rare	Good	Good			2	88
northern pin oak	Quercus ellipsoidalis	NSH	Medium	0.3	0.0	2.0	Lg. dec.	Lg. dec.	High	Rare	Poor	Poor			0	89
laurel oak	Quercus laurifolia	NDH	Medium	0.1	0.0	0.2	Lg. inc.	Lg. inc.	Medium	Rare	Good	Good			2	90
mountain maple	Acer spicatum	NSL	Low	0.1	0.0	0.2	Lg. dec.	Sm. dec.	High	Rare	Poor	Poor			0	91
balsam fir	Abies balsamea	NDH	High	0	0	0	Unknown	Unknown	Low	Modeled	Unknown	Unknown			0	92
Atlantic white-cedar	Chamaecyparis thyoides	NSH	Low	0	0	0	Unknown	Unknown	Low	Absent	Unknown	Unknown			0	93
shortleaf pine	Pinus echinata	WDH	High	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat	Migrate +	Migrate ++	3	94

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Common Name	Scientific Name	Range	MR	%Cell	FIAsum	FIAiv	ChngCl45	ChngCl85	Adap	Abund	Capabil45	Capabil85	SHIFT45	SHIFT85	SSO	N
slash pine	Pinus elliottii	NDH	High	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			3	95
longleaf pine	Pinus palustris	NSH	Medium	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			3	96
pond pine	Pinus serotina	NSH	Medium	0	0	0	New Habitat	New Habitat	Low	Absent	New Habitat	New Habitat	Migrate +		3	97
bald cypress	Taxodium distichum	NSH	Medium	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat	Migrate +	Migrate +	3	98
northern white-cedar	Thuja occidentalis	WSH	High	0	0	0	Unknown	Unknown	Medium	Absent	Unknown	Unknown			0	99
florida maple	Acer barbatum	NSL	Low	0	0	0	New Habitat	New Habitat	High	Absent	New Habitat	New Habitat	Migrate +	Migrate +	3	100
Ohio buckeye	Aesculus glabra	NSL	Low	0	0	0	Unknown	Unknown	Medium	Absent	Unknown	Unknown			0	101
cittamwood/gum bumelia	Sideroxylon lanuginosum ssp.	NSL	Low	0	0	0	New Habitat	New Habitat	High	Absent	New Habitat	New Habitat			0	102
water hickory	Carya aquatica	NSL	Medium	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			3	103
pecan	Carya illinoinensis	NSH	Low	0	0	0	New Habitat	New Habitat	Low	Absent	New Habitat	New Habitat			3	104
shellbark hickory	Carya laciniosa	NSL	Low	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			3	105
black hickory	Carya texana	NDL	High	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			0	106
sugarberry	Celtis laevigata	NDH	Medium	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			3	107
blue ash	Fraxinus quadrangulata	NSL	Low	0	0	0	Unknown	Unknown	Low	Absent	Unknown	Unknown			0	108
silverbell	Halesia spp.	NSL	Low	0	0	0	Unknown	Unknown	Medium	Absent	Unknown	Unknown			0	109
southern magnolia	Magnolia grandiflora	NSL	Low	0	0	0	Unknown	New Habitat	Medium	Absent	Unknown	New Habitat			3	110
sweetbay	Magnolia virginiana	NSL	Medium	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat	Migrate ++	Migrate ++	3	111
bigleaf magnolia	Magnolia macrophylla	NSL	Low	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			3	112
mountain or Fraser magnolia	Magnolia fraseri	NSL	Low	0	0	0	New Habitat	New Habitat	NA	Absent	New Habitat	New Habitat			0	113
water tupelo	Nyssa aquatica	NSH	Medium	0	0	0	New Habitat	New Habitat	Low	Absent	New Habitat	New Habitat	Migrate +	Migrate +	3	114
swamp tupelo	Nyssa biflora	NDH	Medium	0	0	0	New Habitat	New Habitat	Low	Absent	New Habitat	New Habitat	Migrate +	Migrate +	3	115
redbay	Persea borbonia	NSL	Low	0	0	0	New Habitat	New Habitat	High	Absent	New Habitat	New Habitat	Migrate +		3	116
cherrybark oak; swamp red o	Quercus pagoda	NSL	Medium	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat	Migrate +	Migrate +	3	117
overcup oak	Quercus lyrata	NSL	Medium	0	0	0	New Habitat	New Habitat	Low	Absent	New Habitat	New Habitat			3	118
blackjack oak	Quercus marilandica	NSL	Medium	0	0	0	New Habitat	New Habitat	High	Absent	New Habitat	New Habitat	Migrate +	Migrate ++	3	119
water oak	Quercus nigra	WDH	High	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat	Migrate +	Migrate ++	3	120
willow oak	Quercus phellos	NSL	Low	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat	Likely +	Likely +	3	121
Shumard oak	Quercus shumardii	NSL	Low	0	0	0	New Habitat	New Habitat	High	Absent	New Habitat	New Habitat		Migrate +	3	122
post oak	Quercus stellata	WDH	High	0	0	0	New Habitat	New Habitat	High	Absent	New Habitat	New Habitat	Migrate ++	Migrate ++	3	123
live oak	Quercus virginiana	NDH	High	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			0	124
bluejack oak	Quercus incana	NSL	Low	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat			0	125
American mountain-ash	Sorbus americana	NSL	Low	0	0	0	Unknown	Unknown	Low	Absent	Unknown	Unknown			0	126
winged elm	Ulmus alata	WDL	Medium	0	0	0	New Habitat	New Habitat	Medium	Absent	New Habitat	New Habitat	Migrate +	Migrate ++	3	127
cedar elm	Ulmus crassifolia	NDH	Medium	0	0	0	New Habitat	New Habitat	Low	Absent	New Habitat	New Habitat			0	128