Maintenance of Forest Ecosystem Health and Vitality

5

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Introduction

FOREST HEALTH WILL likely be threatened by a number of factors—including fragmentation, fire regime alteration, and a variety of diseases, insects, and invasive plants—along with global climate change (Krist et al. 2007, Tkacz et al. 2008). By itself, global climate change could dramatically and rapidly alter forest composition and structure (Allen and Breshears 1998, Allen et al. 2010). In conjunction with other threats, global climate change poses unique challenges to forest management by influencing forest dynamics at virtually all levels: disturbance regimes in forest ecosystems; rates of resource availability and utilization; canopy gap formation and woody debris dynamics; fire regimes; community composition; and forest distribution, structure, biodiversity, and biogeochemistry. Global climate change and other threats, in turn, could favor the establishment of invasive species. The following pages discuss effects of the most pertinent threats to the future health of forests in the Northern United States.

OVERVIEW OF THREATS

The major biotic threats affecting northern forests (USDA FS, n.d.a) include:

- Oak decline
- Asian longhorned beetle (*Anoplophora glabripennis*)
- Gypsy moth (*Lymantria dispar*)
- Emerald ash borer (*Agrilus planipennis*)
- Hemlock woolly adelgid (*Adelges tsugae*)
- Beech bark disease
- Sirex woodwasp (*Sirex noctilio*)
- Spruce budworm (*Choristoneura fumiferana*)
- Winter moth (*Operophtera brumata*)

The threats vary considerably in the amount of forest area they affect and in their rate of spread, but collectively the North surpasses other regions of the country in the number of invasive forest pests per county (Liebhold et al. 2013). Some threats are not yet broadly established but pose serious concern for future forests nonetheless (Tables 5.1, 5.2). Although some have been successfully mitigated, contained, controlled, or even eradicated, others continue to spread unimpeded throughout northern forests (Fig. 5.1).

Key Findings

- Forest threats like emerald ash borer, hemlock woolly adelgid, beech bark disease, and gypsy moth are causing major changes to forests.
- Insects such as the Asian longhorned beetle, spruce budworm, Sirex woodwasp, and winter moth have exhibited the potential to decimate a variety of tree species and could become more formidable to forests in urban and rural areas.
- Emerald ash borer could decimate the entire U.S. population of ash, which is culturally significant to Native American populations, valuable for specialty products, and valued in urban landscapes.
- Decline-disease complexes such as oak decline are causing negative ecological and economic effects.
- Fire regime changes have substantially decreased the abundance of fire-tolerant and fire-adapted vegetation like oaks and have contributed to invasion by fire-intolerant mesic species.
- Invasive plants are contributing to habitat loss, ecosystem degradation, and decreasing species diversity.
- Other threats such as deforestation and fragmentation contribute to worsening forest health conditions, which provide increased opportunities for invasions to spread.
- Although forest threats have always existed, present-day challenges to forest ecosystem health, diversity, and resilience are unprecedented.

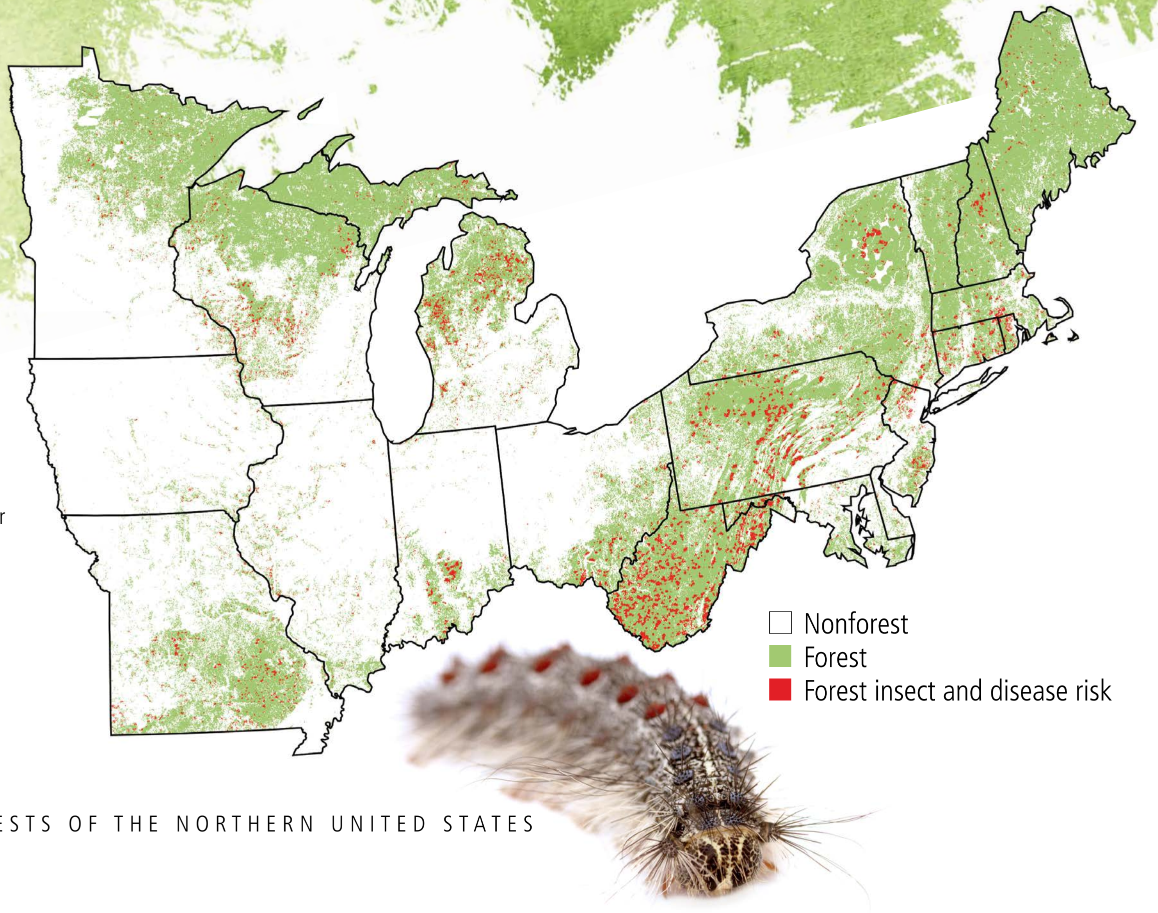


FIGURE 5.1

Areas in the North predicted to have ≥25-percent mortality risk from oak decline, southern pine beetle, gypsy moth, pine engraver beetle, or hardwood decline by 2030 (USDA FS, n.d.e).

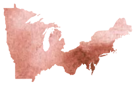


Table 5.1—Occurrence of the major invasive insect and disease threats affecting forests in the North

State	Threat (X indicates presence)								
	Oak decline	Asian longhorned beetle[a]	Gypsy moth[b]	Emerald ash borer[c]	Hemlock woolly adelgid[a]	Beech bark disease[a]	Sirex woodwasp[d]	Spruce budworm[a]	Winter moth[a]
Connecticut	X		X	X	X	X	X		X
Delaware	X		X		X				
Illinois	X		X	X					
Indiana	X		X	X					
Iowa	X		X	X					
Maine	X		X		X	X		X	X
Maryland	X		X	X	X	X			
Massachusetts	X	X	X	X	X	X			X
Michigan	X		X	X		X	X	X	
Minnesota	X		X	X				X	
Missouri	X			X					
New Hampshire	X		X	X	X	X		X	X
New Jersey	X	X	X	X	X	X	X		
New York	X	X	X	X	X	X	X	X	X
Ohio	X	X	X	X		X	X		
Pennsylvania	X		X	X	X	X	X		
Rhode Island	X		X		X	X			X
Vermont	X		X		X	X	X	X	
West Virginia	X		X	X	X	X			
Wisconsin	X		X	X		X		X	

[a] *USDA FS, n.d.a.*

[b] *Slow the Spread Project, n.d.a.*

[c] *USDA 2014*

[d] *Purdue University, n.d.*

Table 5.2—Major invasive insect and disease threats affecting forests of the Northern United States (Purdue University, n.d.; USDA FS, n.d.a.)

Threat	Type	Counties (out of 1037) where severe outbreaks occurred					Characteristics
		2009	2010	2011	2012	2013	
Oak decline	Native decline complex	NA	NA	NA	NA	NA	Combination of tree and site factors, drought, fungi, and insects. Periodic widespread mortality is triggered when all factors occur on sites with a high proportion of mature trees in the red oak group.
Asian longhorned beetle	Nonnative invasive insect	9	10	11	11	11	Currently infests a very small area, but has the potential to cause substantial regionwide damage. Feeds on a variety of species but prefers maples.
Gypsy moth	Nonnative invasive insect	122	123	54	36	89	Currently found in 19 of 20 States, gradually spreading west from eastern Massachusetts. Feeds on a variety of species but prefers oaks.
Emerald ash borer	Nonnative invasive insect	187	228	282	330	403	Currently found in 16 of 20 States, spreading at approximately 12 miles per year from southeastern Michigan. Infests ash (*Fraxinus* spp.) exclusively.
Hemlock woolly adelgid	Nonnative invasive insect	181	202	204	217	228	Currently found in 13 of 20 States, spreading northeast and southwest from Virginia. Infests eastern and Carolina hemlocks.
Beech bark disease	Nonnative decline complex	246	254	254	257	261	Currently found in 14 of 20 States and infects >90 percent of American beech trees >6 inches d.b.h. Affects American beech exclusively.
Sirex woodwasp	Nonnative invasive insect	1	6	2	1	5	Currently infests a very small area, but has the potential to cause substantial regionwide damage. Feeds on a variety of species but prefers pines.
Spruce budworm	Native invasive insect	23	38	29	33	24	Biggest defoliator of spruce and fir in eastern forests of North America. Periodic outbreaks are part of natural cycle in spruce-fir forests, but fire suppression and other anthropogenic disturbances increase the likelihood of outbreaks.
Winter moth	Nonnative invasive insect	8	13	13	14	17	Currently found in 6 of 20 States. Feeds on a variety of species.

DECLINE-DISEASE COMPLEXES

Oak Decline

Oak decline is a disease complex recognized in North America since the middle of the 19th century (Hopkins 1902, Oak et al. 1996). Characterized by progressive branch dieback (Fig. 5.2) and eventual mortality, oak decline has been described as the combination of predisposing, inciting, and contributing factors primarily affecting trees from the red oak group (Manion 1991, Wargo et al. 1983).

Oak decline could be responsible for the loss of more than 2 billion square feet of oak basal area over the next 15 years, causing substantial economic losses throughout North America (Krist et al. 2007). In addition, considering that oak-hickory (*Quercus* spp. – *Carya* spp.) is the largest species group in the United States and accounts for 35 percent of all land in northern forests (Smith et al. 2009), the consequences of oak decline could be widespread (Fralish 2004).

FIGURE 5.2

Oaks showing symptoms of oak decline.
Photo by USDA Forest Service, Forest Health Protection, St. Paul Archive, Bugwood.org.

Although studies have described relatively localized incidences of oak decline and probable primary causes (Fan et al. 2006) and have detected shifts in species composition away from oak (Rogers et al. 2008), broad-scale information on oak decline is lacking. Identifying the controlling factors and extent of oak decline across a broader scale will be critical for the management of oaks throughout northern forests.

A variety of long-term factors related to tree characteristics and growing site conditions predispose forests to oak decline:

- Affected tree species are primarily from the red oak group (Fan et al. 2008, Haavik and Stephen 2010) and include black oak (*Quercus velutina*), red oak (*Q. rubra*) and scarlet oak (*Q. coccinea*).
- The average age of affected trees is >70 years (Oak et al. 1996).
- The size of affected trees is >10 inches d.b.h. (Johnson et al. 2002).
- Growth of affected trees is <0.08 inches per year (Shifley et al. 2006).
- The canopy position of affected trees is classified as intermediate or suppressed crown class (Fan et al. 2006).
- Stand basal area of affected stands is >56 square feet per acre (Fan et al. 2006).
- Affected trees are generally on poor-quality sites characterized by low productivity, shallow and rocky soils, dry ridges or slopes, and southerly aspect (Kabrick et al. 2008).

Short-term factors that increase susceptibility to oak decline (Houston 1987, Staley 1965, Wargo et al. 1983) by increasing tree stress include: drought (Huddle and Pallardy 1996, Jenkins and Pallardy 1995); structural damage from wind and ice (Shirakura et al. 2006); and defoliation by frost, ice, and insects such as gypsy moth and oak leaf roller (*Archips semiferanus*).

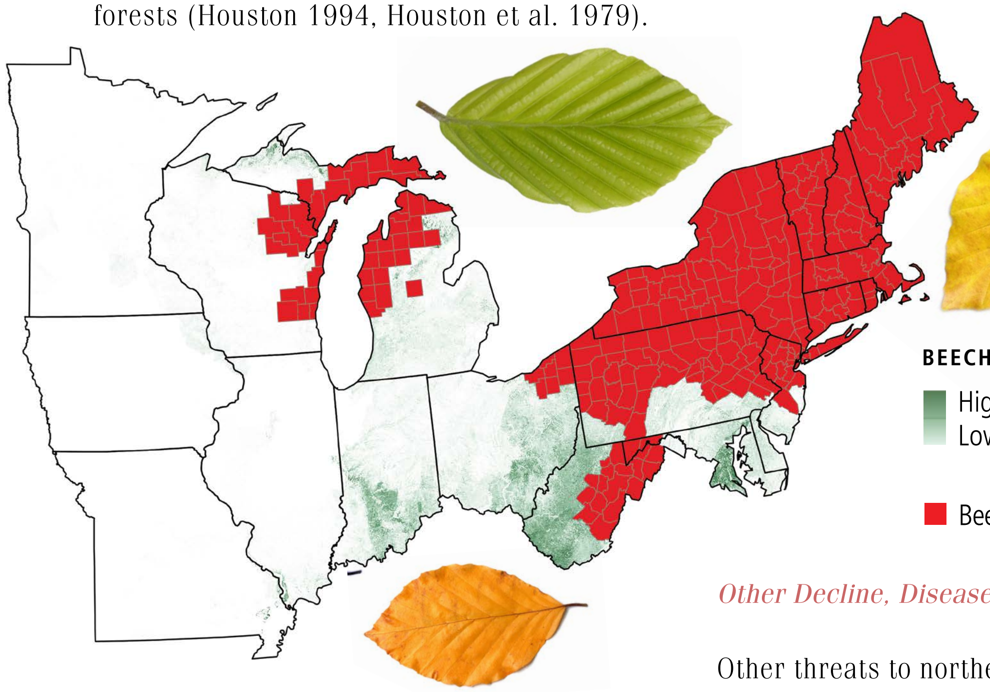
A number of factors contribute to the oak decline complex, including: fungal pathogens, such as *Armillaria mellea* and *Hypoxylon* spp. (Wargo et al. 1983), and insects from the family Buprestidae, such as two-lined chestnut borer (*Agrilus bilineatus*) and red oak borer (*Enaphalodes rufulus*) studied by Starkey et al. (2004). Both are opportunistic organisms that inhabit weakened trees. In combination with predisposing and inciting factors, these organisms increase stress and damage to trees, leading to mortality (Manion 1991, Wargo 1996).

Identifying areas that are particularly susceptible to oak decline can help land managers mitigate negative economic and ecological consequences. Although some factors such as drought are beyond the control of land managers, others can help identify conditions presently conducive to oak decline (for example species, age, and stand basal area). This information can help guide management strategies for reducing incidence and impacts of oak decline.

Beech Bark Disease

Beech bark disease, a complex caused by several species of native and nonnative fungi (*Nectria* spp.) that are vectored by a nonnative scale insect (Shigo 1972), has caused 50- to 85-percent mortality of American beech (*Fagus grandifolia*) in the heaviest infected areas (Fig. 5.3) of northern forests (Houston 1994, Houston et al. 1979).

FIGURE 5.3
Range of American beech and counties with beech bark disease in the North (USDA FS, n.d.a; Wilson et al. 2012).



Research suggests that <1 percent of beech trees exhibit genetic resistance to the disease complex and that beech bark disease infects >90 percent of American beech trees >6 inches d.b.h. (Latty et al. 2003, Sage 1996). Although smaller trees have lower susceptibility, beech bark disease has the potential to cause substantial basal-area loss of this species throughout its range. Because American beech is a major component of northern forests, the ecological effects of its demise would be far-reaching and could have negative effects on associated wildlife (Rosemier and Storer 2010).

Other Decline, Disease, and Pathogen Threats

Other threats to northern forest hardwoods include butternut canker (*Sirococcus clavigignenti-juglandacearum*); thousand cankers disease (*Geosmithia morbida*) of walnut (*Juglans* spp.); oak wilt (*Ceratocystis fagacearum*); sudden oak death (*Phytophthora ramorum*); and decline of ash (*Fraxinus* spp.), maple (*Acer* spp.), and basswood (*Tilia americana*). As with oak decline, tree, site and stand conditions, abiotic factors, insects, and pathogens have been implicated as contributing agents (Manion 1991). In addition to decline, ash is susceptible to pathogens that cause chlorosis, witches' brooms, and abundant epicormic branching (PSU CAS 1987).

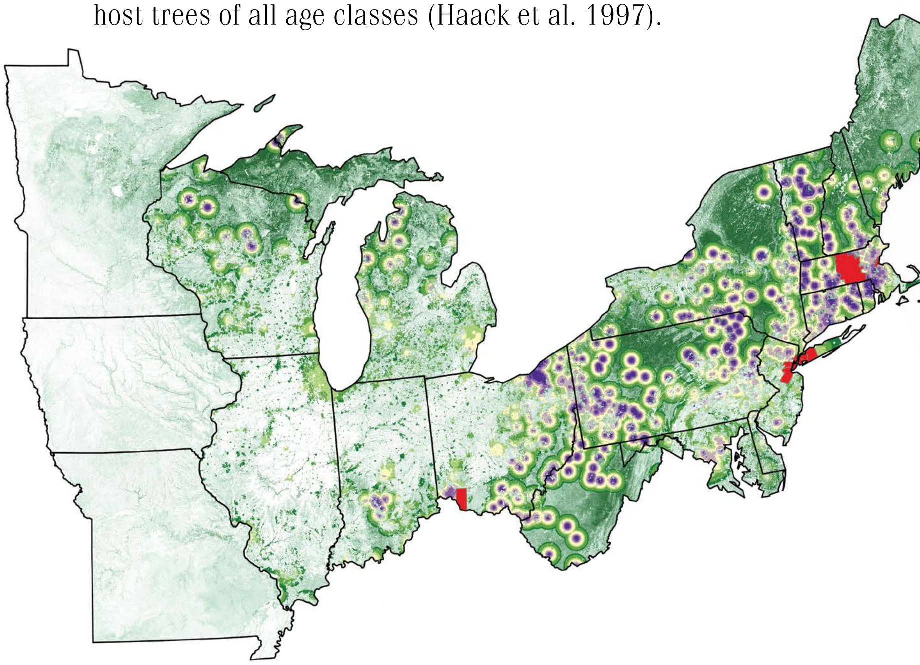
INVASIVE INSECT SPECIES

Asian Longhorned Beetle

Among the most serious of nonnative insect pests in northern forests is the Asian longhorned beetle. This species is capable of causing extensive damage to many hardwood species (Poland et al. 1998, Solomon 1995). As with emerald ash borer (described in a subsequent section of this chapter), Asian longhorned beetle larvae (Fig. 5.4) cause extensive damage to phloem and xylem in both healthy and declining host trees of all age classes (Haack et al. 1997).

Asian longhorned beetles feed on a variety of species, but they prefer maples (*Acer* spp.), a major concern given the importance of maples in northern forests and urban areas (Haack et al. 2010). An estimated 111 million acres, 64 percent of all northern forest land (Fig. 5.4), are at risk from infestation.

Although quarantines and eradication programs have been somewhat successful since the discovery of Asian longhorned beetle in the mid-1990s, they have come at a high economic cost (Hu et al. 2009).



SUSCEPTIBILITY POTENTIAL

🟪 High
🟩 Low

🟥 Asian longhorned beetle detected, 2013

FIGURE 5.4

Spatial distribution of Asian longhorned beetle occurrence and susceptibility in the North; susceptibility is defined as the potential for introduction and establishment of a forest pest within a tree species or group (USDA FS, n.d.a; USDA FS, n.d.b; Wilson et al. 2012).

Adult Asian longhorned beetle on log showing damage.

Photo by E. Richard Hoebeke, Cornell University, Bugwood.org

Asian longhorned beetle larva.

Photo by Kenneth R. Law, USDA APHIS PPQ, Bugwood.org

Consequently, the long-term and broad-scale effectiveness of inspection, interception, and eradication programs is uncertain. As of 2013, Asian longhorned beetle infestations were confined to Massachusetts, New Jersey, New York, and Ohio (Fig. 5.4). Because spread rates and host mortality probabilities have not yet been established, accurately projecting the future range of Asian longhorned beetle is challenging. However, it clearly poses a serious threat to northern forests for two reasons: (1) it can utilize numerous tree species as hosts; and (2) new introductions associated with imported goods are highly likely.

Susceptibility of Host Trees to Nonnative Invasive Insects

Susceptibility is defined as the potential for introduction and establishment, over a 15-year period, of a forest pest within a tree species or species group (USDA FS 2014; USDA FS, n.d.e). Potential for introduction considers the import of commodities (such as bricks, stones, metal, and glass materials) that are shipped with wood packing or pallets from countries where invasive insects are native, as well as the businesses and individuals engaged in importing and handling commodities of concern (represented as polygon ZIP code centroids). Potential for establishment considers the mean basal area for all host trees (Wilson et al. 2012), by ZIP code centroids with 18-mile buffers, to include the potential cumulative spread of the invasive insect over a 15-year period at the rate of 1 mile per year (Bancroft and Smith 2001).

Gypsy Moth

Gypsy moth was brought to North America and accidentally released in 1869. Since then it has continued to spread across northern forests from east to west (Liebhold et al. 1995). Although it prefers white oak (*Q. alba*), host trees can include other oak species as well as aspen (*Populus* spp.), birch (*Betula* spp.), willow (*Salix* spp.), beech, hemlock (*Tsuga* spp.), pine (*Pinus* spp.), and spruce (*Picea* spp.). During population outbreaks, foliage of nearly all forest trees is at least partially consumed (Liebhold et al. 1995). Outbreaks can cause extensive defoliation, reducing tree growth and vigor. Repeated defoliation (Fig. 5.5) can result in mortality, often in conjunction with secondary fungal and insect invaders that attack weakened trees (Liebhold et al. 1995).

A 2005 study suggested that the infested area is only 23 percent of the total area susceptible to the gypsy moth (Morin et al. 2005). Thus far, management consists of slowing the spread of its advancing border (Figs. 5.5, 5.6, 5.7) along a line from central Wisconsin southward and eastward to West Virginia (Tkacz et al. 2008).

Gypsy moth larva feeding on foliage.

Photo by John H. Ghent, USDA Forest Service, Bugwood.org

Foliage consumed by late instar gypsy moth larvae.

Photo by E. Bradford Walker, Vermont Department of Forests, Parks and Recreation, Bugwood.org

Adult male (brown) and female (white) gypsy moths.

Photo by USDA APHIS PPQ Archive, Bugwood.org

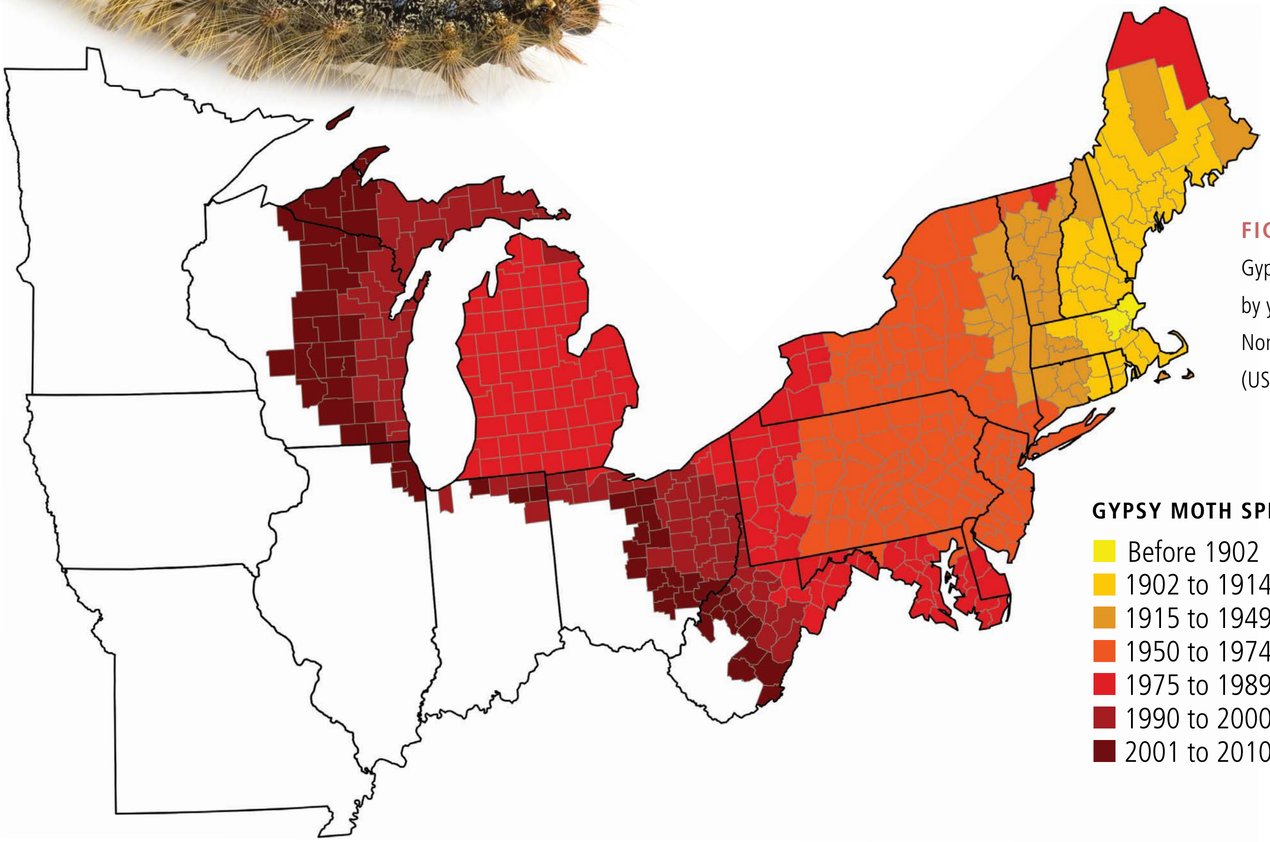


FIGURE 5.5

Gypsy moth spread by year throughout Northern U.S. counties (USDA FS, n.d.a).

Because females (Fig. 5.5) are generally incapable of flight, the rate of gypsy moth spread has been fairly slow (Slow the Spread Project, n.d.b). Given the slow spread rate and the fact that the gypsy moth is a resource generalist, modeling the expansion of the core infested area is important. The Gypsy Moth Slow the Spread Project has monitored the gypsy moth population expansion and has been successful at slowing its spread for more than 20 years; the average spread rate in northern forests since the project's inception is about 3 miles per year (Tobin and Blackburn 2007), a gradual decrease from previous decades. Nevertheless, gypsy moth could occupy most of the northern forest by 2060 based on the current spread rate. This means that the continued success of the Gypsy Moth Slow the Spread Project will likely be crucial for sustaining the health of northern forests over the next 50 years.

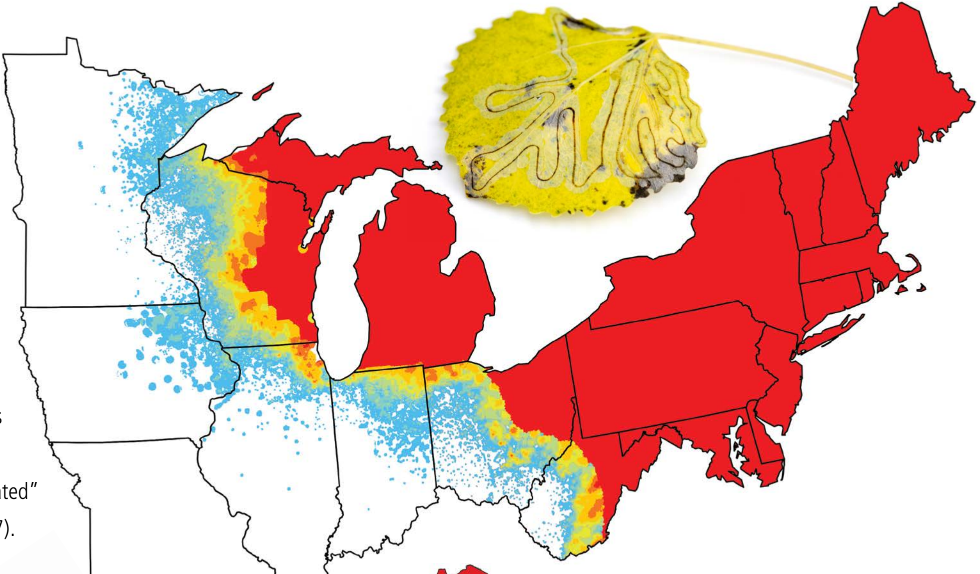


FIGURE 5.6

Occurrence of gypsy moth throughout its range in the North in 2010; occurrence levels for areas in red range from "populated" to "infested" (Tobin and Blackburn 2007).

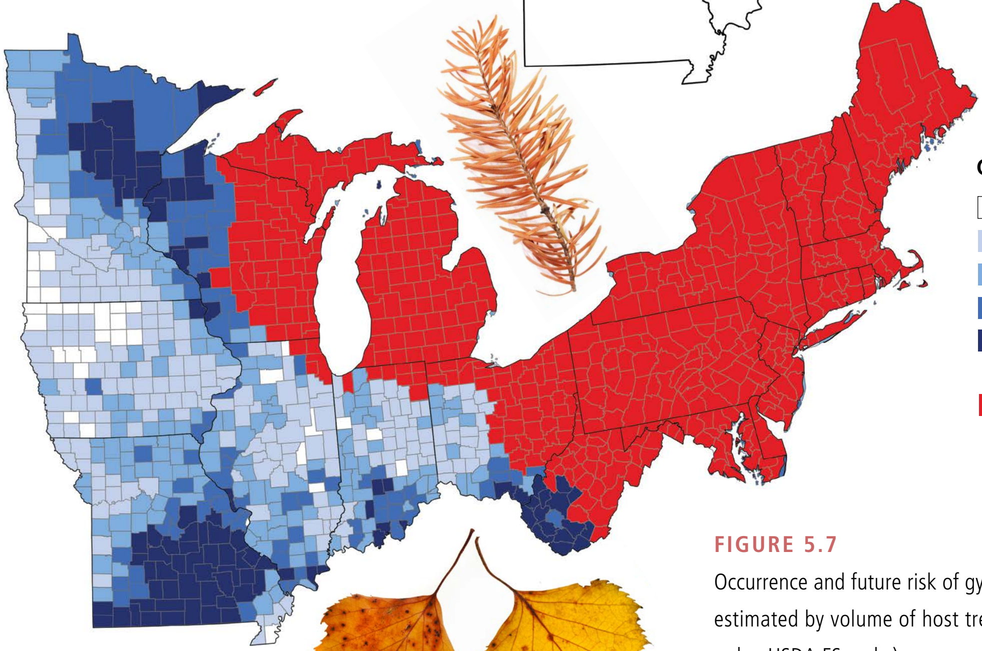


FIGURE 5.7

Occurrence and future risk of gypsy moth infestation in the North, estimated by volume of host tree species (Slow the Spread Project, n.d.a; USDA FS, n.d.c).

Emerald Ash Borer

The nonnative invasive emerald ash borer (EAB) kills trees when damage from larval phloem galleries (Fig. 5.8) and outer sapwood cavities girdles their stems, disrupting water and nutrient transport (Cappaert et al. 2005).

Adult emerald ash borer.

Photo by David Cappaert, Michigan State University, Bugwood.org

Emerald ash borer larva in phloem gallery.

Photo by David Cappaert, Michigan State University, Bugwood.org

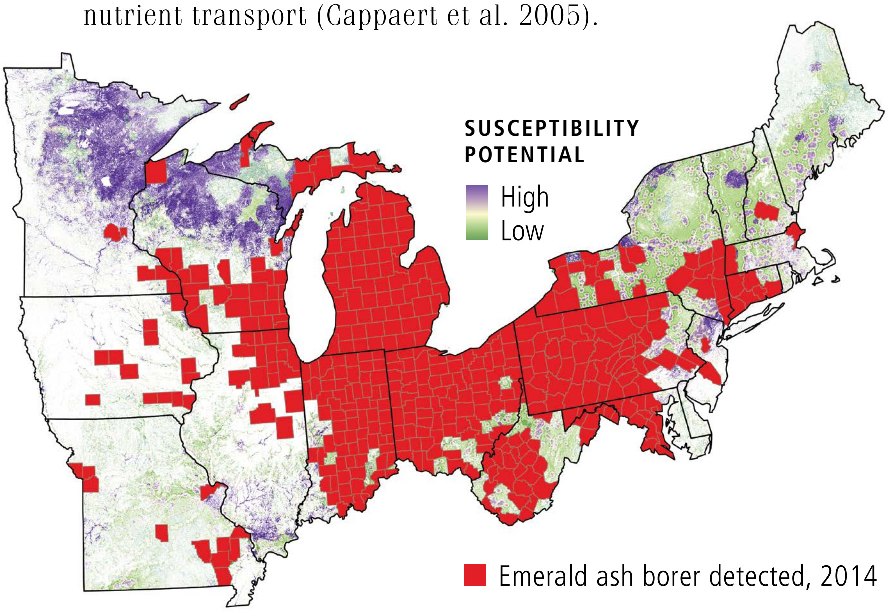


FIGURE 5.8

Spatial distribution of emerald ash borer occurrence and susceptibility as a function of preferred host range, presence of urban ash trees, proximity of urban ash trees to natural forests, and past rates of phloem insect interceptions at U.S. ports of entry; susceptibility is defined as the potential for introduction and establishment of a forest pest within a tree species or group (USDA FS, n.d.b; Wilson et al. 2012).

In Asia, the beetles feed on ash throughout its native range but do not usually cause extensive damage because their hosts have developed defensive mechanisms, including production of volatile compounds (Eyles et al. 2007, Jendek 1994, Rebek et al. 2008). Because North American ash species lack these defenses and are suitable hosts, infestations are usually fatal to affected trees (Poland and McCullough 2006).

Since its introduction to North America in the 1990s, emerald ash borer has spread across 23 percent of the range of ash (Fig. 5.8) and killed millions of ash trees in northern forests (Gandhi and Herms 2010, Haack et al. 2002, Pureswaran and Poland 2009). It is spreading at about 12 miles per year, which suggests that it will occupy the entire range of northern ash (Fig. 5.9) before 2050 (DeSantis et al. 2013b).

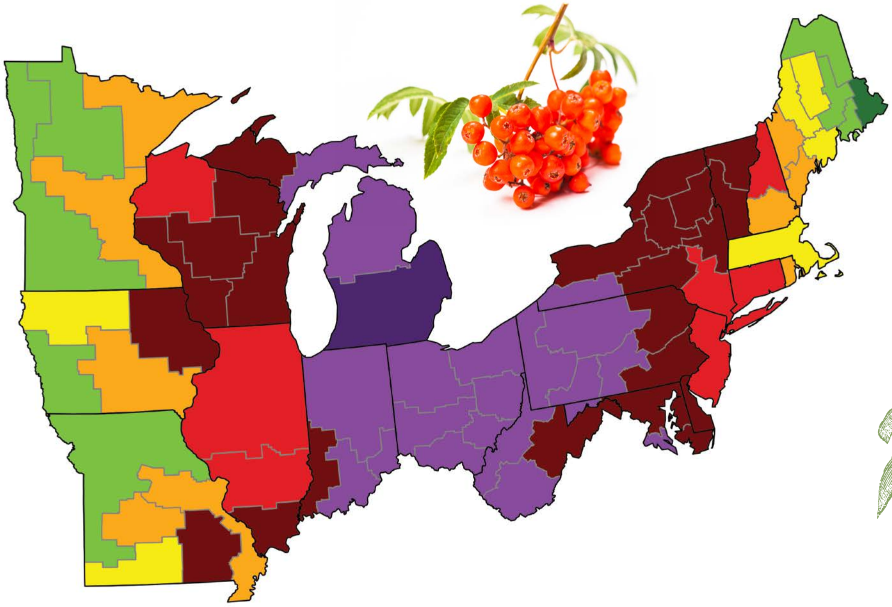


FIGURE 5.9

Projected ash mortality caused by emerald ash borer in forests of the North by survey reporting unit within each State (source: FIA data) and based on the following assumptions: (1) emerald ash borer spread leads to 100-percent ash mortality once the spread subsumes each survey unit, (2) the spread in New York, Vermont, New Hampshire, and Maine will be influenced by present infestations in the Canadian municipalities of Ontario and Québec but not from infestations in other Canadian or U.S. locations, (3) the spread rate is 12 miles per year, with the initial extent of the insect based on detection in U.S. counties and Canadian regional municipalities (CFIA 2014; DeSantis et al. 2013b; Personal communication, 2011, Paul Chaloux, Emerald Ash Borer National Program Manager, U.S. Department of Agriculture, APHIS, Riverdale, MD.

Therefore, given enough time, nearly all ash trees in northern forests will likely be killed (Herms et al. 2010, Prasad et al. 2010).

Modeling results suggest infestations will contribute to a small decrease in the total number of trees and saplings in survey units measured by Forest Inventory and Analysis (FIA) from 2010 to 2060, but also that the effects of ash mortality on the elm-ash-cottonwood (*Ulmus* spp.–*Fraxinus* spp.–*Populus* spp.) group will differ among FIA inventory units. Although the loss of ash trees will alter forest species composition in non-urban forests measured by FIA, modeling results suggest EAB effects will not cause substantial forest-type group changes. In many areas, the impacts of emerald ash borer on ecosystem functioning could be minimal because increases in associated non-ash species may offset the loss of ash trees and their associated volume. Despite the assumption of 100-percent ash mortality, the transition from ash to other species could progress slowly as it is replaced by a variety of associated species. The emerald ash borer infestation could also produce canopy gaps that would facilitate an increase in native and nonnative invasive plant species (Gandhi and Herms 2010).

Because this analysis focused on species groups, the geographic differences in the composition of each group are important including:

- In Maine inventory units, ash is not present in significant amounts in any species groups.
- In Minnesota inventory units, the elm-ash-cottonwood group is mainly composed of green ash (*Fraxinus pennsylvanica*) and black ash (*F. nigra*) whereas white ash (*F. americana*) is a very small component of the oak-hickory forest-type group and is largely absent from the elm-ash-cottonwood forest-type group.
- In most northern inventory units, the ash component of the elm-ash-cottonwood forest-type group is larger than the ash component of the oak-hickory forest-type group.

Furthermore, this modeling was not designed for urban areas not measured by FIA; therefore, urban areas could experience larger impacts because they support an extensive distribution of ash (Chapter 10).

Emerald ash borer infestation will likely have negative economic consequences for the wood products industry, in urban areas where ash has been widely used for landscaping and street trees, and for Native American tribes that value ash as a culturally important forest resource (Cappaert et al. 2005). Infestation could also affect wildlife and ecosystem functioning, especially on wet sites where black ash and pumpkin ash (*F. profunda*) are common (Burns and Honkala 1990b).

Hemlock Woolly Adelgid

Hemlock woolly adelgid was first discovered in North America in the Pacific Northwest in 1924, but the insect was not reported to cause extensive hemlock damage until its 1951 discovery in Virginia (Annand 1924, Gouger 1971, McClure 1987). It has subsequently spread northeast and southwest from the initial Virginia infestation to its present range (Fig. 5.10) and has caused considerable damage to eastern hemlock (*Tsuga canadensis*) and Carolina hemlock (*T. caroliniana*).

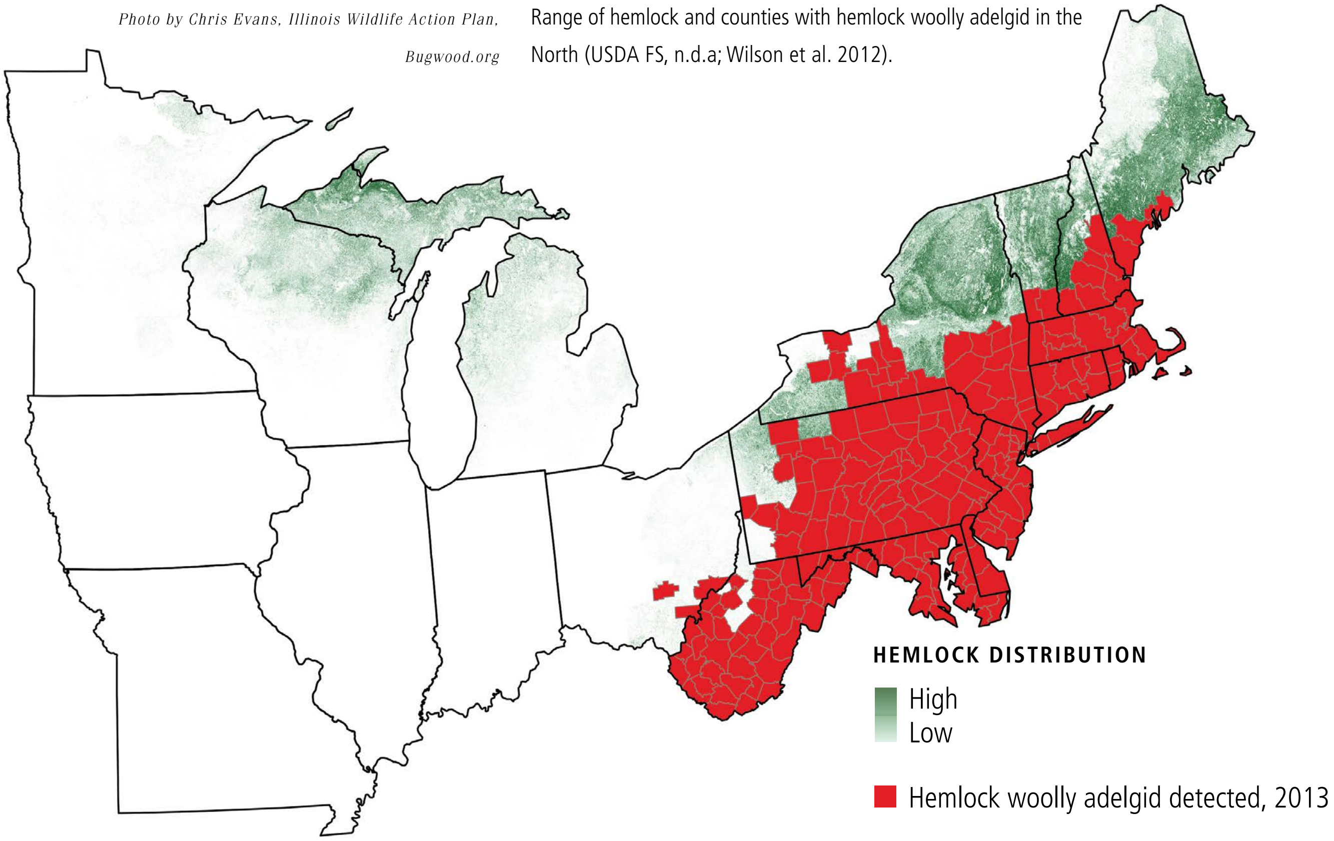
Throughout the native range of hemlock woolly adelgid in Asia, as well as in western areas of North America, hemlocks appear to be resistant (McClure and Cheah 1999). However, mortality in northern forests can be >95 percent of infested hemlock trees, and defoliation of surviving trees can range from 50 to 75 percent. This means that the probability of hemlock mortality in infested stands can eventually reach 100 percent (Orwig and Foster 1998, 2000). Mortality commonly occurs 4 to 10 years after infestation in the northern range of hemlock woolly adelgid, but in its southern range mortality occurs more rapidly (Georgia Forestry Commission, n.d.). Because defoliated hemlock trees lose the ability to produce seed or to sprout, changes in tree species composition are common following infestation (Orwig and Foster 1998, Spaulding and Rieske 2010).

Hemlock woolly adelgid infestation.

Photo by Chris Evans, Illinois Wildlife Action Plan, Bugwood.org

FIGURE 5.10

Range of hemlock and counties with hemlock woolly adelgid in the North (USDA FS, n.d.a; Wilson et al. 2012).



Establishing the spread rate of hemlock woolly adelgid and the probability of hemlock mortality is necessary for the management of hemlock in northern forests. However, this determination is complicated by (1) an asymmetric pattern of adelgid spread that is influenced by host abundance and (2) geographical variation in lethal cold winter temperatures (Parker et al. 2002, Skinner et al. 2003). Although hemlock woolly adelgid can succumb to low winter temperatures, it appears to be more susceptible to extreme low temperature events than to long periods of exposure to typical winter low temperatures. The spread rate ranges from 5 to 18 miles per year, and southwest spread is more rapid than to the northeast (where winter temperatures are lower). Thus, if winter low temperatures rise with climate change, the rate of hemlock woolly adelgid spread could increase across northern landscapes.

Sirex Woodwasp

Sirex woodwasp, which attacks both living and dead trees, was first detected in North America in 2004 and has been increasingly present in solid wood packing materials at U.S. ports of entry (de Groot et al. 2006, Haugen and Hoebeke 2005).

Native to Asia, Europe, and northern Africa, Sirex woodwasp was accidentally introduced to Australia, New Zealand, South Africa, and parts of South America, where it now causes ≤80-percent tree mortality in pine plantations (Haugen and Hoebeke 2005). Although it currently infests a very small area of northern forests, it has the potential to cause substantial regionwide damage (Fig. 5.11).

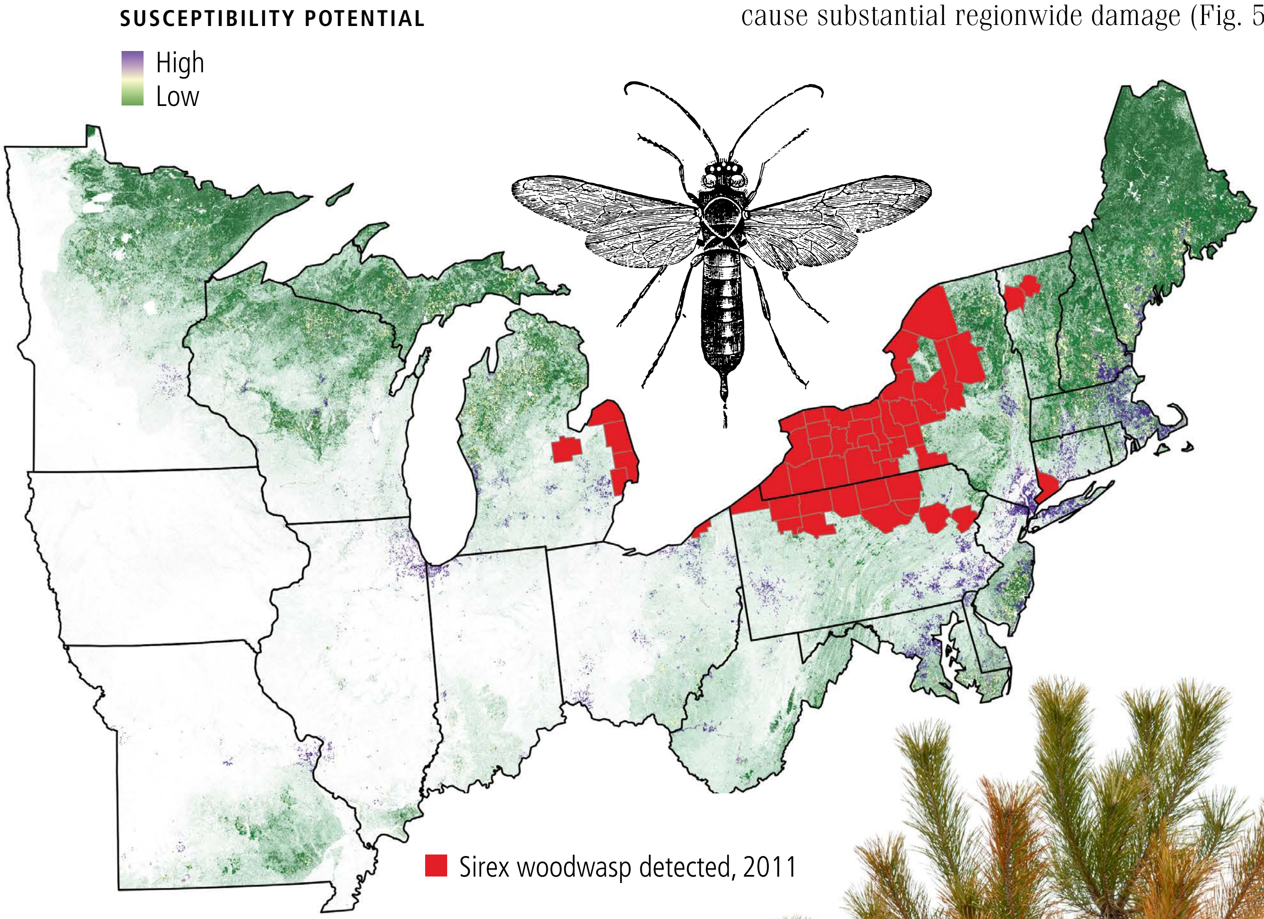


FIGURE 5.11

Spatial distribution of Sirex woodwasp occurrence and susceptibility; susceptibility is defined as the potential for introduction and establishment of a forest pest within a tree species or group (Purdue University, n.d.; USDA FS, n.d.d ; Wilson et al. 2012).

FIGURE 5.12

Counties with spruce budworm and range of four host tree species—spruce, fir, pine, and larch—in the North (USDA FS, n.d.a; Wilson et al. 2012).

Spruce Budworm

Eastern spruce budworm is one of the most destructive defoliators in North America and is the major defoliator of spruce and fir (*Abies* spp.) throughout eastern Canada and the eastern United States (Kucera and Orr, n.d.; PSU CAS 1987; Williams and Liebhold 2000). Over-mature, fire-suppressed stands of balsam fir (*Abies balsamea*), white spruce (*Picea glauca*), red spruce (*Picea rubens*), and black spruce (*Picea mariana*) are particularly susceptible to spruce budworm outbreaks, but other softwoods are also vulnerable (Fig. 5.12).

Spruce budworm feeds on foliage and new shoots, causing extensive root mortality (Burns and Honkala 1990a). Although periodic budworm outbreaks are part of a natural cycle in spruce-fir forests, fire suppression and other human disturbances leading to unhealthy stand conditions can predispose forests to spruce budworm outbreaks (Boulanger and Arsenault 2004).

Winter Moth

Winter moth was first introduced in eastern Canada in the 1930s and is now established throughout coastal New England and New York (Childs et al. 2011). Native to Europe, this insect became a serious pest in eastern Canada, western Canada, and the U.S. Pacific Northwest; it now threatens to infest northern forests. Winter moth larvae defoliate a variety of woody species, including oaks, maples, cherries (*Prunus* spp.), basswood (*Tilia americana*), ash, elms, apples (*Malus* spp.), and spruces (Childs et al. 2011).

Other Invasive Insects

Other major pests of North American conifers include the nonnative balsam woolly adelgid (*Adelges piceae*) and the native jack pine budworm (*Choristoneura pinus*), which could cause considerable ecological and economic damage (McCullough et al. 1994).

Southern pine beetle (*Dendroctonus frontalis*) and pine engraver beetle (*Ips* spp.) cause extensive damage to yellow pines throughout the southeastern and western United States. Although damage is not as widespread in the U.S. North, both beetles are established and remain a threat. Other biotic agents that could pose serious problems to forests in the region include European spruce bark beetle (*Ips typographus*) and Mediterranean pine engraver beetle (*Orthotomicus erosus*).

NONNATIVE INVASIVE PLANTS

Invasive plants can cause considerable damage to ecosystem function, productivity, and species diversity (Lavoie 2010, McCullough et al. 2006). Other than habitat loss, invasive plants could pose the greatest threat to already imperiled North American plants and wildlife, with research showing that invasive plants contributed to the endangerment of 49 percent of all imperiled wildlife species and 57 percent of the imperiled plant species in the United States (Wilcove et al. 1998). In northern forests, invasive plants of major concern include tree, shrub, vine, herbaceous, and grass species (Olson and Cholewa 2009). Among the most widespread are garlic mustard (*Alliaria petiolata*), common buckthorn (*Rhamnus cathartica*), Morrow's honeysuckle (*Lonicera morrowii*), Japanese honeysuckle (*Lonicera japonica*), reed canarygrass (*Phalaris arundinacea*), Canada thistle (*Cirsium arvense*), Amur honeysuckle (*Lonicera maackii*), and autumn olive (*Elaeagnus umbellata*). Multiflora rose (*Rosa multiflora*), the most common invasive plant (Table 5.3) throughout northern forests (Kurtz 2013, Moser et al. 2009, Webster et al. 2006), is described here.

Table 5.3—Known occurrence of the major invasive plant threats in Northern States, 2005-2010 (Kurtz 2013)

State	Threat (marked by an X)								
	Multiflora rose	Garlic mustard	Common buckthorn	Morrow's honeysuckle	Japanese honeysuckle	Reed canarygrass	Canada thistle	Amur honeysuckle	Autumn olive
Connecticut	X	X	X	X	X	X	X		X
Delaware	X	X	X		X			X	X
Illinois	X	X	X	X	X	X	X	X	X
Indiana	X	X	X		X	X	X	X	X
Iowa	X	X	X	X		X	X	X	X
Maine	X		X	X		X	X	X	
Maryland	X	X	X	X	X		X	X	X
Massachusetts	X	X	X	X		X			X
Michigan	X	X	X	X	X	X	X	X	X
Minnesota	X	X	X	X	X	X	X	X	X
Missouri	X	X	X	X	X	X	X	X	X
New Hampshire	X		X			X			X
New Jersey	X	X	X	X	X	X		X	X
New York	X	X	X	X		X	X	X	X
Ohio	X	X	X	X	X	X	X	X	X
Pennsylvania	X	X	X	X	X	X	X	X	X
Rhode Island	X			X					X
Vermont	X		X	X					
West Virginia	X	X	X	X	X	X	X	X	X
Wisconsin	X	X	X	X	X	X	X	X	X
Percent[a]	17	5	4	4	4	3	3	3	3

[a] *The percentage of FIA plots where the monitored species was recorded.*

A native of Japan, Korea, and eastern China, multiflora rose (Fig. 5.21) was introduced to the eastern United States as rootstock for ornamental roses in 1866 (Plant Conservation Alliance 2005). This nonnative plant is extremely invasive in human-disturbed prairies, old fields, savannas, forest edges, and woodlands, forming dense thickets that can exclude native plant species, thereby decreasing species diversity. Multiflora rose has a broad ecological niche, tolerating various light, moisture, and soil conditions ranging from open fields to stream banks and roadsides to dense woods. It also produces abundant fruits that are widely consumed by birds, enhancing seed distribution.

COMPOUNDING EFFECTS OF ABIOTIC AGENTS

Forest Fragmentation

Forest land constitutes nearly 172 million acres, or 42 percent of the total land area in the U.S. North (Smith et al. 2009). In recent decades, U.S. forests have experienced a net increase in area. However, this increase includes conversions of land from agriculture to forest land, and from forest land that is contiguous to forest land that is fragmented by urbanization. As forest land becomes increasingly fragmented, the plants and wildlife that depend on forest ecosystems are subjected to increased competitive interactions among populations (O'Neill et al. 1988, 1992; Riitters et al. 2002). Fragmentation not only decreases the amount of habitat suitable for species adapted to the forest interior but also increases forest edge, thereby raising the chances of invasion from other land cover types (for example, agricultural, urban, barren, grassland, and shrub land) or water (Gardner et al. 1993, Tkacz et al. 2008).

In this manner, fragmentation can alter forest composition and structure and reduce forest health by predisposing the spread of nonnative and invasive species, reducing biodiversity, and dramatically changing ecological processes (Boulinier et al. 2001, Smith et al. 2009).

FIGURE 5.21

Multiflora rose in May. Photos by James H. Miller, U.S. Forest Service, Bugwood.org.

Human-caused fragmentation involves more frequent disturbances that lack the randomness of natural fragmentation, and often is more permanent than natural fragmentation because it is frequently the result of conversion to urban uses (Best 2002). Northern forest area is projected to decrease 5 percent by 2060 (Miles and Wear 2015). However, this includes both losses and gains, with much of the new acreage resulting from conversion of agricultural land to fragmented forest. Forest fragmentation is discussed in more detail in Chapter 3; conversion of urban and rural forest land is discussed in Chapter 10.

Climate Change

Climate is a major force that influences the composition of plant communities. Disturbances such as drought or fire often influence forests by promoting early successional, light-demanding vegetation (Abrams 1996). By changing successional pathways, severe or prolonged droughts can increase tree mortality and contribute to shifts in species composition (Allen et al. 2010). Although decreased available precipitation, increased temperature, and increased extent and severity of drought can have substantial impacts on forest ecosystems, the importance of these factors drastically increases when coupled with other forest threats, such as invasive species.

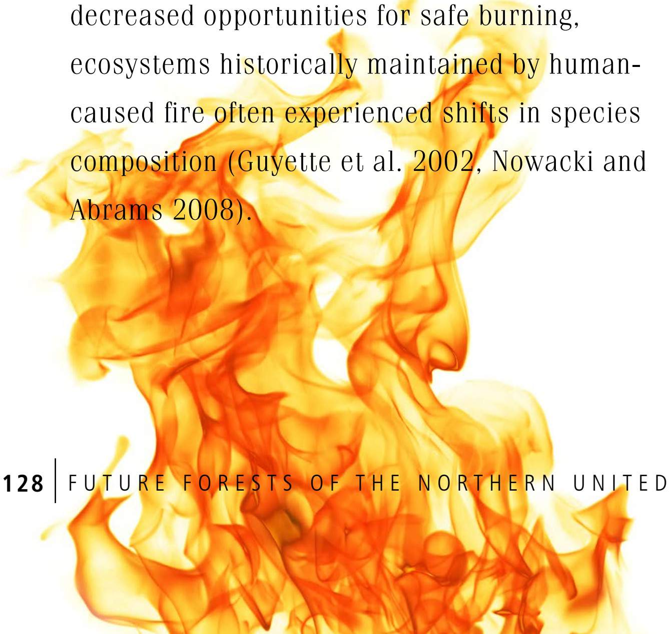
The Intergovernmental Panel on Climate Change generally suggests future global mean temperature will increase (Solomon et al. 2007). As with other invasive insect ranges, climate change may create warmer conditions favorable to nonnative invasive plant and insect habitat in areas that are currently unsuitable. For example, while the northward spread of insects has historically been limited by winter conditions, insects such as the emerald ash borer have rapidly spread to the north recently thanks to increasing winter temperatures (Chown and Terblanche 2006). However, the northern spread of emerald ash borer can be affected by complex weather relationships (DeSantis et al. 2013a).

Biotic and abiotic forces have altered forest ecosystems for millennia. Research suggests that climate has always played an important role in determining forest composition and structure (Allen 2009, Swetnam and Betancourt 1998, van Mantgem et al. 2009). One example is the decline of eastern hemlock in North America during the period that started 4700 years ago (and ended 3800 years ago), which one study (Filion and Quinty 1993) attributes to increasingly wet conditions and another (Allison et al. 1986) attributes to a pathogen. Both studies reported an eventual increase in eastern hemlock following the decline, and taken together they suggest that the interaction of biotic and abiotic forces is important to long-term changes in forest composition and structure. Along with evidence that climate is changing at an increasingly rapid pace (Solomon et al. 2007), there is the growing concern that background rates of forest change could increase and substantially alter the composition and structure of forests globally.

Fire

Wildfires pose several concerns, including risks to human life and property, species loss, adverse effects on forest productivity and health, carbon-cycle alteration, and air pollution. With a few exceptions, the annual amount of acreage burned in the North from 2004 to 2013 did not substantially change (Table 5.4). Although prescribed fire was used, most of the acreage that burned in northern forests was caused by wildfire.

Both the presence and absence of fire can have serious negative implications for forest ecosystems. The long-term effects of fire can be especially important for wildlife because of the potential for substantial changes to habitat and food sources (Bendell 1974). The long-term effects of fire suppression have indirectly altered forest health by influencing stand structure and composition (Lorimer 1985). For example, in northern forests that were once dominated by fire-adapted and fire-tolerant woody species or herbaceous vegetation, the suppression or decreased use of fire has enabled encroachment by other woody plants (Mutch 1970). When human population pressures and forest fragmentation decreased opportunities for safe burning, ecosystems historically maintained by human-caused fire often experienced shifts in species composition (Guyette et al. 2002, Nowacki and Abrams 2008). The result was a decrease in fire-tolerant and fire-adapted vegetation such as oaks, which have substantially decreased in abundance in northern forests (Abrams 1992). Similar trends have occurred with yellow pine species in southern forests (Aldrich et al. 2010, Lafon et al. 2007, South and Buckner 2003). Given the amount of oak and pine in the region (30 percent of volume according to 2010 FIA data), such changes in species composition have broad-scale economic and ecological implications (Beisner et al. 2003, Scheffer et al. 2001). In addition, a decrease in fire is thought to have played an important role in oak decline throughout the range of northern oaks (Brose et al. 2001, Lorimer et al. 1994, Van Lear 2001).



Forests play an essential role in the global carbon cycle by acting as carbon sinks or sources. To address concerns about the effects of increasing atmospheric carbon dioxide on global climate, policy makers need accurate quantification of forest ecosystem carbon stocks, carbon fluxes between forests and the atmosphere, and effects of disturbances on forest carbon stocks and fluxes (Schulze et al. 2000). Depending on the forest type and stage of stand development, forests can act as carbon sinks or sources, but human-caused and natural disturbances can quickly change forest carbon sinks into sources. For example, stand-replacing wildfires that consumed jack pine (*Pinus banksiana*) forests in northern Michigan caused the rapid conversion of a carbon sink to a carbon source due to the consumption of respirable substrate (Rothstein et al. 2004).

Table 5.4—Acres burned by wildfire in North (NIFC, n.d.)

State	Area burned (acres)										
	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	Average
Connecticut	94	263	419	288	893	246	262	244	417	238	336
Delaware	0	231	126	153	36	44	32	0	0	17	64
Illinois	476	733	1,040	611	16	1,155	630	2,019	1,848	55	858
Indiana	3,351	2,568	3,468	5,730	1,064	4,706	132	530	1,846	806	2,420
Iowa	585	10,566	827	684	2,217	8,565	2,722	2,145	349	14,704	4,336
Maine	960	822	1,794	423	535	481	318	116	489	743	668
Maryland	3,147	4,350	6,396	5,790	2,421	4,956	1,577	8,310	838	182	3,797
Massachusetts	1,534	2,530	2,821	2,687	2,918	1,144	2,117	545	1,027	879	1,820
Michigan	2,754	4,684	8,128	23,344	2,675	3,899	11,441	1,865	28,377	940	8,811
Minnesota	42,881	24,944	66,540	163,111	19,712	13,858	33,969	135,650	64,682	22,107	58,745
Missouri	32,542	19,621	24,308	10,537	9,765	42,241	17,620	55,395	26,456	1,660	24,015
New Hampshire	289	172	472	212	195	159	145	48	221	137	205
New Jersey	549	1,532	2,514	20,849	3,322	1,131	10,630	794	3,095	1,430	4,585
New York	390	656	2,508	855	3,629	1,404	2,066	232	2,127	1,073	1,494
Ohio	3,126	2,920	3,866	1,578	1,078	10,962	3,995	1,203	1,250	152	3,013
Pennsylvania	3	1,100	6,822	1,047	7,694	6,132	3,203	636	3,157	1,788	3,158
Rhode Island	81	84	120	61	132	45	23	27	41	27	64
Vermont	244	574	232	230	168	192	86	14	405	281	243
West Virginia	6,007	2,762	13,663	7,149	10,296	13,591	14,319	2,680	6,703	8,577	8,575
Wisconsin	2,385	6,311	2,287	4,713	1,050	3,746	2,185	719	2,880	9,196	3,547
Total	101,398	87,423	148,351	250,052	69,816	118,657	107,472	213,172	146,208	64,992	130,754

FIGURE 5.13
Serotiny of jack pine cones. Photo by Paul Wray, Iowa State University, Bugwood.org.

However, the extraordinary regeneration abilities of jack pine—producing seed from serotinous cones (Fig. 5.13)—enabled rapid establishment and conversion of a carbon source to a carbon sink. In other cases, the depletion of carbon sinks can sometimes be long term or even permanent (Balshi et al. 2009, Metsaranta et al. 2010).

FUTURE THREATS

In 1900, American chestnut (*Castanea dentata*) could be found throughout northern forests. Little did people know that one of the most dominant hardwood species would be devastated within 50 years. Since that time, Dutch elm disease (*Ophiostoma nova-ulmi*) has decimated American elm (*U. americana*), hemlock woolly adelgid has begun to decimate hemlock, and more recently, emerald ash borer has begun to decimate ash in northern forests. In addition, plants such as multiflora rose, intentionally introduced, have rapidly spread, competitively excluding native plant species. Asian longhorned beetle, Sirex woodwasp, and other recently introduced forest pests threaten to devastate maples and pines. Over the coming decades, new threats, including those from climate change and extreme storms, will inevitably materialize and negatively impact northern forests. Unfortunately, as has previously occurred, some destructive threats will emerge for which we lack the capacity to predict. Keeping forests intact and managing for forest health are the best defenses for resilience to these forest threats.

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