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Characteristics and distribution of potential ash tree hosts for emerald ash borer

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Abstract

The emerald ash borer (EAB) (*Agrilus planipennis*) is a recently discovered (July 2002) exotic insect pest, which has caused the death of millions of ash trees (*Fraxinus* spp.) in Detroit, MI, USA and has also spread into other areas of Michigan, isolated locations in Indiana, Ohio, Maryland and Virginia, and nearby Windsor, Ont., in Canada. Ash trees occur in many different forest ecosystems in North America, are one of the more widely planted trees in urban areas, and are a valuable commercial timber species. If emerald ash borer populations are not contained and eventually eradicated, the ash resource in North America could be devastated. The destruction caused by EAB and its rate of spread are likely to be strongly influenced by the spatial distribution and status of the ash tree host, but general information regarding the abundance, health and distribution of ash trees is diffused throughout the literature. Here, we summarize what is currently known regarding the characteristics and potential spatial distribution of various species of *Fraxinus* in natural and planted ecosystems in North America and evaluate this information with specific regard to assessing the relative risk of ash populations to EAB.

Keywords: Emerald ash borer; Ash; Exotic pest; Host range

1. Introduction

The emerald ash borer (EAB) (*Agrilus planipennis*, Fairmaire; Coleoptera; Buprestidae) is a recently discovered (July 2002) exotic insect pest, which has become established across large areas of MI, USA and isolated areas in Ohio, Indiana, Maryland and Virginia and Windsor, Ont., Canada. The beetle has caused the death of millions of ash trees (*Fraxinus* spp.) and evidence suggests that the beetle has been established in Michigan for at least 6–10 years (McCullough and Katovich, 2004). Forty-eight U.S. counties have populations of EAB (Fig. 1) and a number of counties in southern lower Michigan, Ohio and Indiana have been quarantined to regulate the movement of live ash trees and ash tree products. *Fraxinus* species are widely distributed across the eastern U.S. (Fig. 1) and portions of southeastern Canada, occurring in many different forest ecosystems (Harlow et al., 1991). According to the USDA Forest Inventory and Analysis

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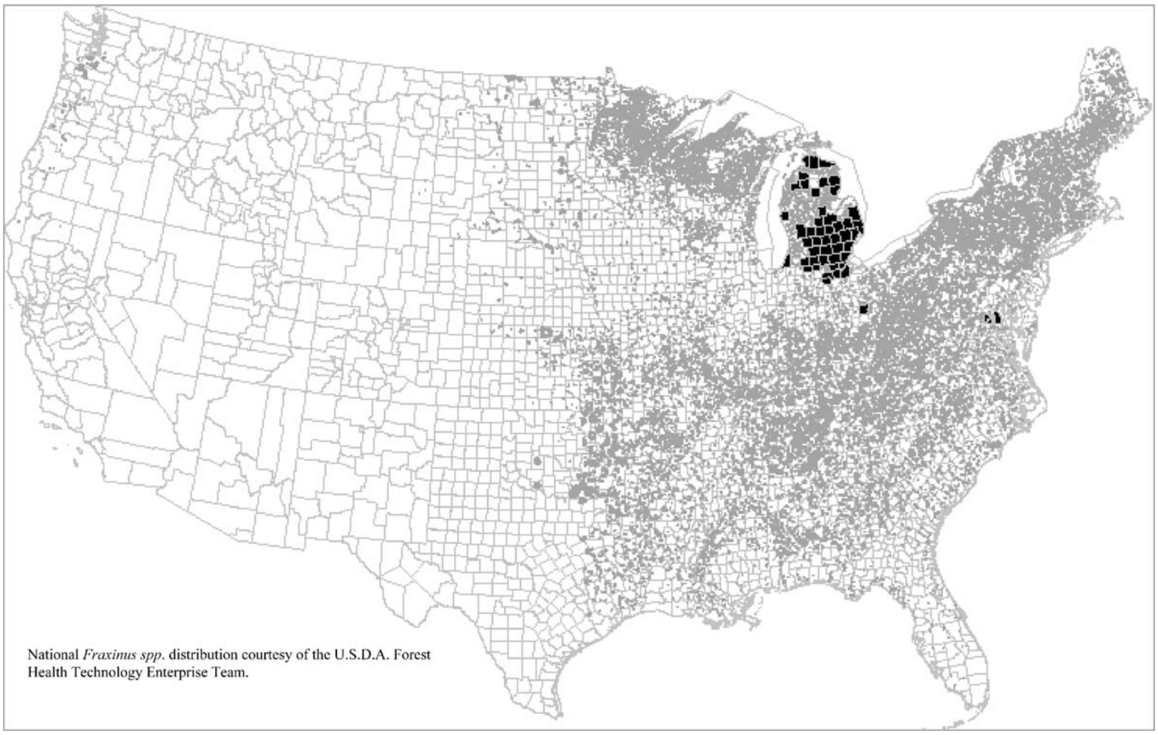

Fig. 1. Distribution of *Fraxinus* species in the United States (gray shaded areas). Populations of emerald ash borer (*Agrilus planipennis*) have been discovered in 48 U.S. counties (black shaded), as well as Windsor, Ont., Canada (not shown).

(FIA) database (http://fia.fs.fed.us), there are over 802.5 million ash trees on timberlands in Michigan alone. Ash trees are also one of the more widely planted trees in urban areas of the U.S. (Ottman and Kielbaso, 1976; Giedriaitis and Kielbaso, 1982). If emerald ash borer populations are not contained and eventually eradicated, the North American ash resource could be devastated. In natural forests, EAB may dramatically change forest biodiversity and forest stand dynamics. In urban areas, dead and dying trees will pose hazards to people and property and will require removal, often a costly process. Substantial economic losses will be sustained by the wood products and horticultural industries through direct destruction of the resource and quarantines affecting the movement of ash trees and products. Unfortunately, little is known regarding the rate of spread of EAB (Haack et al., 2002), but rate and direction of spread is likely to be correlated with the spatial distribution and status of the ash tree host.

An understanding of the distribution and condition of potential ash hosts in the U.S. is essential to assessing the potential risk of forest ecosystems to devastation by EAB. General information regarding the abundance, health and distribution of ash trees is available through the USDA Forest Service's FIA and Forest Health Monitoring (FHM) programs, but the resolution of this data is coarse, particularly in urban ecosystems. The purpose of this paper is to summarize what is currently known regarding the characteristics and potential spatial distribution of various species of *Fraxinus* in natural and planted ecosystems in North America and to evaluate this information with specific regard to assessing the relative risk of ash populations to EAB.

2. Ash in natural forest systems

It is important to first consider the characteristics and distribution of potential ash tree hosts for EAB in North America in light of the native host range. Preliminary investigation suggests that EAB is found on *F. chinensis*, *F. rhynchophylla* (a.k.a. *F. chinensis* var. *rhynchophylla*), *F. mandshurica* and some North American *Fraxinus* species in its native range, which includes most of China, Korea and Japan and possibly parts of Russia and Mongolia (McCullough and Katovich, 2004). Ash species are found in a variety of climates across most of China, in all provinces except Xinjiang and Tibet (i.e., western China) but are much less abundant and geographically isolated relative to North American ash species (Dr. Xie Yingping, College of Life Science and Technology of Shanxi University, pers. com.). In North America, by contrast, ash species are widely distributed across many interconnected forested ecosystems.

There are 16 or 17 arborescent species of *Fraxinus* (Harlow et al., 1991), depending on phenotypic and genetic relationships amongst the ash species (Wright, 1944, 1959a,b). Harlow et al. (1991) list white ash and green ash as being taxonomically ''important'' species of ash in the U.S., but also list a number of species of lesser importance, including black ash (*F. nigra*), ranging from southeastern Canada to northeastern U.S.; Carolina ash (*F. caroliniana*), found on the coastal plain from northeastern Virginia, south to Florida and west to southeastern Texas; pumpkin ash (*F. profunda*), found on the coastal plain from southern Maryland to northern Florida, west to Louisiana and north to southern Illinois; blue ash (*F. quadrangulata*), found on dry limestone uplands in Ohio and the Upper Mississippi valleys; Oregon ash (*F. latifolia*), ranging along the west coast from southern British Columbia to southern California; single-leaf ash (*F. anomala*) and velvet ash (*F. velutina*), in the arid southwestern U.S. Stewart and Krajicek (1973) list six of these species: white, pumpkin, blue, black, green, and Oregon, as commercially important.

Preliminary testing of host preferences suggests that that all eastern North American ash species are susceptible to EAB, but certain ash cultivars (i.e., *Fraxinus americana*, 'Autumn Purple') may be more resistant to EAB than others (Herms et al., 2004). If species-specific susceptibility or population status is to be considered, it must be recognized that there has long been confusion regarding the identification of ash species in the field (Wright, 1944, 1959a,b). Green ash and red ash are common names often used interchangeably to describe morphs of *F. pennsylvanica*, although taxonomists have been unable to agree over the years whether red ash is a subspecies or separate species of green ash (Wright, 1944, 1959b; Harlow et al., 1991). Similarly, Oregon ash, velvet ash, and Carolina ash are close relatives of the more widespread green ash and have all been classified as both separate species and sub-species of green ash (Wright, 1944; Wright, 1959b). Green and white ash, the two most common species, have been known to hybridize, complicating identification (Wright, 1959a). Many forest inventories separate white and green ash based upon landscape position (i.e., upland and lowland sites, respectively) or do not attempt to differentiate ash species at all. If ash species becomes important for EAB risk assessment, one might be tempted to refine existing databases describing ash species by landscape position. However, this would likely be either redundant or logically circular.

We found that information regarding North American *Fraxinus* species other than green, black and white ash in the literature is scarce. Here, we assume that species-specific characteristics referred to in this report reflect a greater than usual effort to identify ash to the species level by investigators. Below, we outline the various factors which might be used to differentiate natural host tree distribution, abundance and vigor as potential spatial indicators of risk for EAB.

2.1. Physical and genetic characteristics of ash

The physical–genetic properties of ash trees should be meaningful for EAB risk assessment if genetically based resistance to the borer becomes evident. Stewart and Krajicek (1973) noted that white ash has superior strength qualities, which differentiate it commercially from other ash species, and this may relate to the physical–chemical properties of the wood. Genetically, white ash is more distinct than many of the other ashes listed (Wright, 1959a), although it has one close relative in the rare Texas ash (*F. texensis*). Since white ash is the most commercially important of the ash species, used for products such as tool handles, baseball bats, flooring, and furniture, susceptibility of white ash would not bode well for North American timber resources. Black ash, sometimes marketed as ''brown ash'', and pumpkin ash seem to be differentiable from other ash trees (Stewart and Krajicek, 1973). Green, Carolina, velvet and Oregon ashes are believed to be genetically related (Wright, 1944, 1959b). It may be useful to consider these latter species together from a standpoint of genetic resistance, at least initially, before more detailed studies are undertaken.

2.2. Site preferences for different ash species

White ash is generally a species of mesophytic uplands in the northeastern and north central U.S., but is also found on lowlands as well (Fowells, 1965; Buchholz, 1981; Harlow et al., 1991). Along a gradient of soil types, white ash typically occupies wet-mesic to dry-mesic site conditions, with a general preference for brown and grey-brown podzolic soils (Fowells, 1965). White ash is drought sensitive, and as such, is only found on higher slope positions when there are seeps and ephemeral streams to enhance moisture content of the soil (Woodcock et al., 1993). Elliott (1953, cited from Wright, 1959a) noted that white ash in Lower Michigan is found on podzolic soils underlain by heavy clay and high water tables, but is nearly absent on lighter soils underlain by well-drained glacial drift. Forests growing on lighter soils are more prone to wildfire and, since white ash is severely affected by fire (Henning and Dickmann, 1996), this may help limit its presence on such sites. Another study showed white ash to be limited to areas underlain by a compacted glacial till at a depth of less than 20 in., which supports a perched water table in rainy periods (Stout, 1952). Van Breemen et al. (1997) found that white ash canopy dominance was highest where the silt fraction of the soil was highest. Van Breemen et al. (1997) and Finzi et al. (1998) found white ash to be most abundant in areas of Connecticut forests where total soil calcium [Ca] and magnesium [Mg] concentrations measured in the study were highest. These areas also had the lowest quantities of exchangeable iron [Fe] and aluminum [Al] measured, indicating that ash has a preference for richer soils with higher pH (Van Breemen et al., 1997). Under deep shade, white ash seedling mortality was noted to be much higher in non-calcareous soils versus calcareous, which also suggests that [Ca], in particular, is important for the development of this species (Kobe, 1996). Thus, in general, white ash seems fairly sensitive to both soil fertility and available water and should be less abundant and more stressed outside of moist, fertile environments.

White ash also occurs on lowlands, generally in areas where gley soils are present in the subsoil and where flood duration is limited (Buchholz, 1981). Fowells (1965) noted that on the lowlands of the coastal plain, white ash is usually limited to slightly elevated ridges in major stream bottoms. In the central region it is most common on slopes along major streams, although it also found in many upland situations.

Green ash is generally a bottomland species (Wright, 1959b) which prefers alluvial soils in floodplains along rivers and streams. In an old-growth Indiana eastern hardwood forest, green ash was found where ephemeral spring ponds remained the longest (Badger et al., 1998). Geographically, green ash has a wide ecological distribution (Stewart and Krajicek, 1973), conferred by its resistance to salt, flooding, drought (Mueli and Shirley, 1937) and high alkalinity (McComb, 1949). Because of this stress tolerance, green ash is commonly planted in strip mine reclamations (Fowells, 1965). Green ash is also an important component of the Great Plains prairie-woodland ecosystems (Hansen and Hoffman, 1988) and is one of most commonly planted species in the Great Plains shelterbelts, because it can persist on dry soils once established (Harlow et al., 1991). Hence, while green ash is principally a wetland species, it is capable of surviving under a wide range of conditions.

Black ash is typically a hydric species occurring in bogs, swamps, along small streams, in poorly drained depressions and other poorly drained sites with high water tables (Fowells, 1965). Black ash can grow in both low pH (less than 4.1) and high pH (greater than 5.5) soil conditions (Fowells, 1965). Black ash prefers wet muck and shallow organic peat, although it is also found on fine sands and loams underlain by clay that impede surface drainage. Black ash may occasionally occur on uplands (Ronald, 1972; Erdmann et al., 1987; Arevalo et al., 2000).

Pumpkin ash and Carolina ash are trees of flooded bottomlands. Oregon ash is found on both moist, well-drained uplands (Stewart and Krajicek, 1973) and rich alluvial bottomlands (Harlow et al., 1991). Blue ash is typically found on dry, limestone uplands and is tolerant of high pH and drought (Harlow et al., 1991). Velvet ash is found in washes, canyons and stream banks in the arid west, as well as desert woodlands between 760 and 2130 m above sea level (Harlow et al., 1991). Single-leaf ash is found in dry canyons and foothills in desert woodlands to 1980 m (Harlow et al., 1991).

2.3. Associations of ash with other species

The potential susceptibility of different forested areas to EAB can be considered as a function of ash abundance and vigor in different forest community types. Such knowledge should provide not only a baseline perspective on the health of the ash resource beyond the current EAB infestation but also allow for an assessment of potential structural changes that might occur in different forest communities, given various levels of decline in ash species within them.

White ash trees are rarely found in great abundance in the forest, rather they are more typically a minor component of many forest community types. Wright (1959a), for example, noted that white ash was listed as a component species of 26 cover types recognized by the Society of American Foresters for the eastern U.S. He also noted that white ash usually comprises only about 3–4% of stand volume but is rarely completely absent on any appropriate site. Some of the major associate species of white ash are eastern white pine (*Pinus strobus*), northern red oak (*Quercus rubra*), white oak (*Q. alba*), sugar maple (*Acer saccharum*), red maple (*A. rubrum*), yellow birch (*Betula alleghaniensis*), American beech (*Fagus grandifolia*), black cherry (*Prunus serotina*), American basswood (*Tilia americana*), eastern hemlock (*Tsuga canadensis*), American elm (*Ulmus americana*), and tulip polar (*Liriodendron tulipifera*) (Fowells, 1965; Stewart and Krajicek, 1973). In the south, white ash occurs on loamy ridges and bottoms with hickories (*Carya* spp.), willow (*Salix* spp.), sweetgum (*Liquidambar styraciflua*), swamp chestnut oak (*Quercus michauxii*) and other oaks (*Quercus* spp.). At the northern limits of its range it occurs with white pine and the beech–birch–maple–hemlock mixture, as a scattered tree (Harlow et al., 1991). White ash is also an important component of pioneer forests regenerating on fallow agricultural lands (Wright, 1959a; Meiners and Gorchov, 1998).

Black ash occurs in a number of associations with other trees across its range. Primack (2000) found that that associations of black ash, ironwood (*Ostrya virginiana*), musclewood (*Carpinus caroliniana*) and basswood that were located primarily along river bends that were inundated from 1 to 27% of the time. A study by Erdmann et al. (1987) revealed that black ash is most frequently associated with American elm-red maple-ash forest cover type, a long-lived sub-climax on somewhat poorly drained mineral soils. Seedlings and sprouts of black ash are usually the only hardwood regeneration occurring in gaps in the elm-maple-ash type growing on wet organic soils. The elm-red maple-black ash type grades into an almost pure black ash type on poorly drained sites with organic peat and muck soils, where it has been considered as the climax species (Erdmann et al., 1987). Tardif and Bergerson (1999) noted pure stands of black ash in the northern portion of its range, along lakes and rivers. Black ash communities on boreal forest floodplains were identified by Tardif and Bergeron (1992), who used a cluster analysis to produce four vegetation types: (1) black ash/speckled alder (*Alnus rugosa*)/bog willow (*Salix pedicellaris*), (2) black ash/balsam poplar (*Populus balsamifera*)/ostrich fern (*Matteuccia struthiopteris*), (3) black ash/pussy willow (*Salix discolor*)/sensitive fern (*Onoclea sensibilis*), and (4) black ash/speckled alder/sensitive fern. In the northern Lower Peninsula of Michigan, water-covered northern white cedar (*Thuja occidentalis*) lowlands were also found to have high percentages of black ash (Gates and Erlanson, 1925). Gates (1942) found that black ash-red maple stands reach only about 11 m in height before replacement by northern white cedar. Black ash is also occasionally found as a scattered tree in stands of balsam fir (*Abies balsamea*), black spruce (*Picea mariana*), eastern hemlock-yellow birch, white spruce (*Picea glauca*)-balsam fir-paper birch (*Betula papyrifera*), and tamarack (*Larix laricina*) (Fowells, 1965).

Species most commonly associated with green ash are box-elder (*Acer negundo*), red maple, pecan (*Carya illinoinensis*), sugarberry (*Celtis laevigata*) sweetgum, American sycamore (*Platanus occidentalis*), cottonwood (*Populus deltoides*), quaking aspen (*Populus tremuloides*), black willow (*Salix nigra*), willow oak (*Quercus phellos*), and American elm (Fowells, 1965; Stewart and Krajicek, 1973). Oregon ash is most commonly associated with red alder (*Alnus rubra*), black cottonwood (*Populus trichocarpa*), willow, big leaf maple (*Acer macrophyllum*), Oregon white oak (*Quercus garryana*) and Douglas-fir (*Psuedotsuga menziesii*) (Stewart and Krajicek, 1973; Harlow et al., 1991). Pumpkin ash is found with baldcypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*). Blue ash is found with northern red oak, mockernut hickory (*Carya tomentosa*), sweetgum, white oak, white ash, slippery elm (*Ulmus rubra*), American elm and sugar maple (Stewart and Krajicek, 1973). Velvet ash is found in association with black cottonwood, Fremont cottonwood (*Populus fremontii*) and salt cedar (*Tamarisk pentandra*) on low land riparian areas (source: Nevada Gap Analysis, Utah State University) and in association with desert oaks, such as canyon live oak (*Quercus chrysolepsis*) and desert scrub oak (*Q. turbinella*), and ponderosa pine (*Pinus ponderosa*). Single-leaf ash is found in shrubby woodlands and in ponderosa pine forests (Harlow et al., 1991).

2.4. *Ash abundance, vigor and landscape population dynamics*

A number of studies suggest that ash abundance and vigor are related to disturbance and successional patterns in the landscape (Fowells, 1965; Taylor, 1971; Ronald, 1972; Hansen and McComb, 1958; Langlais and Begin, 1993; Ward, 1997; Arevalo et al., 2000; Lesica, 2001; Battaglia et al., 2002). Taylor (1971), for example, noted that large scale disturbance of the landscape increased both the distribution and the relative importance of both white and green ash species in Michigan, relative to pre-settlement vegetation. In a Minnesota oak forest, black ash was originally a minor component, but in years following a catastrophic wind event its basal area increased 900% (Arevalo et al., 2000). In an Ohio Nature Preserve, a look at gap dynamics revealed that white ash is important in larger and more recent canopy gaps versus older, smaller gaps (Spies, 1987). Fowells (1965) noted that the proportion of ash trees usually decreases with increasing stand age and crown closure in mixed stands, due to slower growth and decreasing tolerance as the trees age. In combination, these studies suggest that one can expect that ash will be more abundant, with younger and more vigorous populations, in more open and more recently disturbed forests. Conversely, ash trees should be less abundant and declining in older, less disturbed forest communities. Thus, given different forest ages and community types, EAB might be faced with either a more vigorous and more abundant host versus a weaker, but more difficult to locate host.

Trends in ash population dynamics in areas unaffected by EAB are critical to consider when evaluating potential susceptibility of ash populations and the relative damage caused by EAB. For example, a declining population may have less vigorous, and thus more susceptible, members, although a relationship between tree vigor and EAB colonization has not yet been established. On the other hand, it might be possible that more vigorous hosts are preferred by EAB, as was found in the case of lilac borer (*Podosesia syringae*) (Santamour and Stenier, 1986), which attacks ash trees, lilacs and other plants in the olive family. Herms et al. (2004) recently reported that EAB preferred ash hosts which were fertilized with nitrogen in experimental trials. While the physiological condition of potential hosts is critical to understand, it is also critical to understand trends in the ash population, i.e., what is the background level of decline *sans* EAB? For example, one might consider the relative damage caused by EAB to be high in a declining population, if it is unlikely to rebound, or low if the resource is already considered to be in poor shape anyway. Conversely, the potential loss associated with a well-established resource may be relatively low if it is believed that it will rebound easily after infestation, or high, if a developing resource may be lost before its full potential is realized.

Since the 1920s, there has been an overall concern regarding ‘‘ash decline’’ (Woodcock et al., 1993; Ward, 1997), with many causal elements identified, but unproven, including ozone air pollution and the vascular disease ‘‘ash yellows’’ (Luley et al., 1992; Feeley et al., 2001). For example, ash yellows was detected in only 11 of 145 trees exhibiting external signs of the disease (Feeley et al., 2001). Woodcock et al. (1993) suggested that white ash in the northeastern U.S. was declining on dry-mesic sites, located on steep slopes, because these sites were more vulnerable to drought. Ward (1997), on the other hand, suggested that white ash decline was caused by a decline in ash reproduction and lower canopy tree survival as forests age, in the absence of substantial disturbance. While ash may be declining in some parts of the landscape, we found that ash populations have been on the rise over the last 2 decades in Michigan, the epicenter of the outbreak of EAB, with the total number of ash trees on timberlands estimated to be

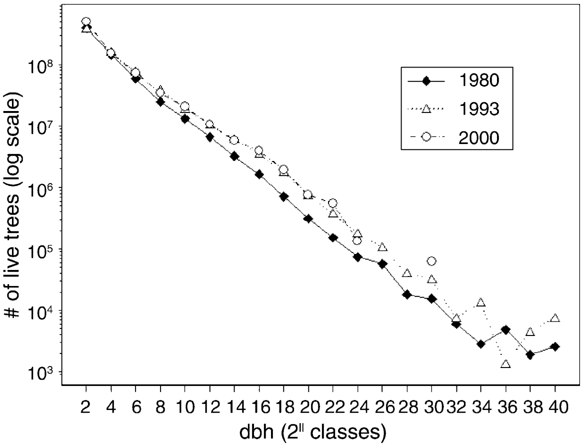

Fig. 2. Twenty year demographic change for ash trees in Michigan, based on the number of ash trees in different stem diameter classes at three points in time (data from USDA Forest Inventory and Analysis Program).

649.4 million trees in 1980, 692.9 million in 1993 and 802.5 million in 2000 (see Fig. 2, note the last estimate was computed from a smaller percentage of the total permanent inventory plots in Michigan and is missing some tree size classes; data from http://fia.fs.fed.us); this trend includes an increase in almost every size class of tree (diameter at breast height, dbh, see Fig. 2). Hence, at least in the vicinity of the current outbreak, the value of the ash resource was increasing at the point of introduction of EAB. Thus, it may be prudent to consider the total risk of EAB to ash populations in the context of a currently increasing resource base, rather than against a general background of decline in the health of ash populations.

3. The urban ash resource

Much of the damage caused by EAB to date has occurred in urban–suburban areas of the Detroit, MI–Windsor, ON metropolitan area, so it is important to address the ash resource in urban ecosystems. Urban ecosystems are often stressful to trees and are also common launch points for many exotic pests and pathogens of trees. Ash trees have been a popular street tree for decades and were widely planted to replace American elms (*Ulmus americana* L.) killed by Dutch elm disease. Green ash (*Fraxinus pennsylvanica*), for example, is tolerant of salt, drought stress, compacted (anoxic) soils and a variety of soil pH conditions ranging from acid to alkaline (McComb, 1949). This wide ecologic amplitude allows the species to occupy a large natural geographic range, ranging from riparian flats in Montana and Saskatchewan, south to Texas, and across the entire eastern U.S. This natural stress tolerance has pre-adapted green ash to grow in urban environments. Soil root pits on city streets are often in compacted, poorly drained soils with a generally high pH. Tolerance to alkalinity is particularly important in urban settings where calcium leacheate from concrete accumulates in the tree root pit, causing often extremely high pH. Similarly salt tolerance has pre-adapted green ash to urban conditions in areas where snowfall accumulates, because the rock salt, that is used to melt ice and snow, accumulates in snow piles around urban trees. Other ash species have proven hardy in urban environments as well (Street Tree Fact Sheets, 1989 (Gerhold et al., 1989)).

The popularity of ash as an urban tree in the United States began in the 1940s with the introduction of the ‘‘Marshall Seedless’’ cultivar of green ash, which was not only tolerant of urban ecosystems, but, as a male clone, eliminated the messy cleanup associated with ash seed mast. Other cultivated varieties of green ash and white ash subsequently became popular street trees (Table 1). European ash (*Fraxinus excelsior*) has been used in place of green or white ash in some cities. Eurasian flowering ash (*Fraxinus ornus*) is sometimes used as an ornamental in the Pacific Northwest (Harlow et al., 1991) and ‘‘Modesto ash’’ (*F. velutina* var. ‘Modesto’) is a common street tree in California and other western states. The popularity of ash cultivars, particularly green ash, continued to rise nationally through the 1980s (Giedraitis and Kielbaso, 1982), except in the southern U.S. where ash has not been commonly used as a street tree (Ottman and Kielbaso, 1976; Giedraitis and Kielbaso, 1982). The greatest popularity has been in the north central region of the U.S. (Giedraitis and Kielbaso, 1982), where unfortunately EAB was introduced. Ash trees are still abundant in many cities in the eastern central and western parts of the country, although green ash may be declining in popularity relative to white ash (*F. americana*) cultivars (as seen in Table 1), possibly due to both the knowledge that green ash is over-planted and the attractive purple fall foliage color in white ash cultivars. In a recent study (Boris and Kielbaso, 1999),

Table 1
Ash tree cultivars recommended for Michigan streets in 1999

Species	var.	Year	Origin	1999 Rank (top 100)
F. pennsylvanica		na	na	66
	"Marshall Seedless"	1946	UT, USA	36
	"Summit"	1957	MN, USA	[a]
	"Patmore"	1975	Manitoba, CA	15
	"Urbanite"	1987	IL, USA	98
	"Cimmaron"	1992	OH, USA	39
F. americana		na	na	23
	"Autumn Purple"	1956	WI, USA	4
	"Rosehill"	1966	MO, USA	[a]
	"Autumn Applause"	1975	IL, USA	2
	"Champaign County"	1975	IL, USA	[a]

Data from Boris and Kielbaso (1999) and Street Tree Fact Sheets (Gerhold et al., 1989).
[a] Indicates that the cultivar is not currently in the top 100 trees recommended.

white ash ranked 2nd and 4th amongst trees recommended for Michigan streets (Table 1).

In assessing potential risk of urban forests to EAB, both the abundance of ash and biological diversity of ash trees must be considered. Data abstracted from a 1994 study of the demography and health of trees in Michigan cities (unpublished data from J. Kielbaso, related to Michigan Forest Health Report, 1994 (Randall, 1994)) revealed that cities that are currently within the core zone of EAB infestation have substantial components of white and green ash trees on their streets (ranging between 5 and 29% of all street trees). The genetic diversity of planted ash in cities in the U.S. is quite low. Consider, for example, that abundant green ash cultivars, in the eastern U.S., and predominant velvet ash cultivars (i.e., 'Modesto') in the western U.S., were selected from already closely related species. In Michigan, only five green and four white ash cultivars (Table 1) were determined to be common in urban ecosystems (Jim Kielbaso, pers. com.). If Michigan cities are representative of the urban forest resource in other parts of the country, then the combined abundance and low genetic diversity of ash should enhance the risk of damage by EAB in urban ecosystems nationwide.

4. Conclusion

Fraxinus species are an important component of many forest ecosystems throughout North America, usually occurring as a minor component of many different forest types. White, blue and Oregon ash are found on fertile uplands and river terraces; green, black, Carolina and pumpkin along river bottoms and in wetlands (black is most abundant in bogs); and velvet and single-leaf ash in dry semi-deserts and canyons. North American ash populations have been put at substantial risk from the introduction of EAB. Large scale losses of ash trees expected as a result of EAB infestations would likely result in dramatic changes in the composition and successional dynamics of many natural forests, cause widespread damage to urban forests and have a severe negative impact on hardwood timber industry in the central and eastern U.S.

Below is a summary of our major findings regarding ash host characteristics and their relationship to the potential risk of EAB infestation:

1. Urban areas with a significant component of ash trees are at a high level of risk.
2. Green and velvet ash cultivars are major components of urban forests, in all areas but the southeastern U.S. The ash resource in urban forests has a low genetic diversity which enhances risk.
3. Certain (white) ash cultivars may have some resistance to EAB.
4. Maturing second growth forests on uplands contain fewer, less vigorous ash trees, but contain much of the economic value in ash wood products nationally. White ash is by far the most important species economically.
5. Younger more open forests tend to have a greater number of more vigorous ash trees.

6. The general phenomenon referred to as "ash decline" may be a natural successional process, which is not occurring equally across differing landscapes.
7. The potential devastation of EAB should not necessarily be considered in light of a "declining" ash resource. Ash populations were not declining in Michigan before the period where EAB was introduced.

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