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Modeling the Effects of Emerald Ash Borer on Forest Composition in the Midwest and Northeast United States

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Abstract

The nonnative invasive emerald ash borer (*Agrilus planipennis* Fairmaire; EAB) has caused considerable damage to the ash (*Fraxinus* spp.) resource in North America. While there are methods to mitigate, contain, control, or even eradicate some nonnative invasive insects, EAB continues to spread across North America. Considering strong evidence suggesting >99 percent probability of host tree mortality, the loss of the North American ash resource is possible. To examine anticipated effects of EAB on tree species composition, we modeled future spatial and temporal changes in forest composition over the next 50 years with and without ash mortality anticipated from EAB spread. We used U.S. Forest Service Forest Inventory and Analysis (FIA) data, the current extent of EAB in the United States and Canada, estimated spread rate and host mortality data, and a suite of human population, energy, consumption, land use, and economic models to project the future condition of forests in the Midwest and Northeast United States. Our results suggest that in most cases EAB will not have a substantial effect on ecosystem function of future forests measured by FIA because of the replacement of ash by other species. The transition from ash to other species may take many decades, but forests can eventually recover when a variety of associated species replace ash.

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SUMMARY

Since being introduced to a novel environment in North America in the 1990s, the emerald ash borer has done extensive damage to green, white, black, blue and pumpkin ash throughout the Midwest and Northeast United States. It is possible that given enough time, EAB will kill nearly 100 percent of all ash throughout their ranges in eastern North America. EAB infestation will likely have a variety of negative economic consequences and the ecological impacts could affect associated wildlife and ecosystem functioning, especially in hydric systems where black ash and pumpkin ash are common. To determine the potential effects of EAB on forest composition, we used a series of models to project the future composition of forests. We used U.S. Forest Service Forest Inventory and Analysis (FIA) data and incorporated EAB current range, estimated spread rate, and host mortality, as well as human population distribution, global economic conditions, energy and technology use, population and economic growth, climate change models, timber harvesting, land use change, and natural succession. Our modeling assumed EAB will cause 100 percent ash mortality. In this report, we describe our modeling framework and provide an explanation for our results. Our results suggest EAB will contribute to a small decrease in the total number of trees and saplings from 2010 to 2060 in Maine, Michigan, Minnesota, New York, Pennsylvania, and Wisconsin, but EAB-caused changes in the elm-ash-cottonwood forest-type group will differ among states. Ultimately, our results indicate ash will be replaced by a variety of species as forests slowly recover from EAB infestation. Although forest composition will change, in many cases the impacts of EAB on ecosystem function may be minimal because non-ash species have the potential to offset the loss of ash. However, these results only apply to forests measured by FIA. There may be different outcomes in urban forests not measured by FIA due to the increased importance of ash, the preemptive removal of EAB-infested ash trees, and the chemical treatment of individual trees. While these factors are probably not relevant on FIA plots, they play an important role in the survival of urban ash trees outside areas measured by FIA.

INTRODUCTION

Due to the volume of international commerce, the likelihood of nonnative insect and disease introductions to novel environments in North America is at an all-time high (Aukema et al. 2011, Gandhi and Herms 2010a, Work et al. 2005). In many cases, introductions had or are having drastic consequences for native flora and fauna (e.g., beech bark disease, hemlock woolly adelgid [*Adelges tsugae* Annand], and gypsy moth [*Lymantria dispar* L.] [Latty et al. 2003, Liebhold et al. 1995, Orwig and Foster 1998, Shigo 1972]), and have caused substantial economic losses (e.g., chestnut blight) (Wallner 1996). Future projections of ecosystem changes due to nonnative insect pests often suggest negative impacts on ecosystem function (e.g., emerald ash borer [*Agrilus planipennis* Fairmaire; EAB] infestation of black ash [*Fraxinus nigra* Marsh.] in hydric systems [Poland and McCullough 2006]). Under epidemic conditions, native invasive insect pests such as pine engraver beetle (*Ips pini* Say), eastern larch beetle (*Dendroctonus simplex* LeConte), mountain pine beetle (*Dendroctonus ponderosae* Hopkins), and forest tent caterpillar (*Malacosoma disstria* Hubner) caused substantial increases in North American tree mortality. Prior knowledge of insect spread rates and risk of host tree infestation by location are important for efforts to help mitigate deleterious economic and ecological effects (Tobin et al. 2004).

Different approaches can be taken to model the susceptibility of forest stands to specific insect pests and determine pest spread rates. Risk maps can be created by integrating models of anthropogenic impacts, pest biology, and ecological attributes over a given geographical extent (Morin et al. 2005). In some cases, monitoring insect pest populations can help decrease future forest resource losses. Insect trapping programs coordinated by the U.S. Forest Service in cooperation with several state programs have allowed for the long-term monitoring of gypsy moth populations since 1988 (gypsy moth Slow the Spread project [STS]; Tobin et al. 2004). STS has provided information that enabled more effective forest land management, despite impending defoliation by gypsy moth. Likewise, modeling EAB spread can help decisionmaking for the purpose of detecting, monitoring, and perhaps slowing the spread (Prasad et al. 2010).

EAB was first discovered in North America in southeastern Michigan and nearby Windsor, Ontario, in 2002, but may have been established there since the early- to mid-1990s (Haack et al. 2002, Siegert et al. 2007). For nearly 20 years, this nonnative invasive insect has caused considerable damage to the North American ash (*Fraxinus* spp.) resource. EAB feeds on Chinese ash (*Fraxinus chinensis* Roxb.), Manchurian ash (*Fraxinus mandshurica* Rupr.), and other ash species throughout its native range in Asia but does not usually cause extensive damage because host trees are somewhat resistant due to their defensive mechanisms (Eyles et al. 2007, Jendek 1994, Rebek et al. 2008). Ash species in North America are suitable hosts for EAB and highly susceptible to EAB-caused decline and mortality (Poland and McCullough 2006). During infestations, EAB's larval galleries in phloem and outer sapwood girdle trees, disrupting water and nutrient transport and eventually killing trees (Cappaert et al. 2005). Currently there are no known methods of broad-scale EAB eradication, control, containment, or mitigation, and it is estimated that given enough time, nearly 100 percent of the ash resource in eastern North America could be killed by EAB (Herms et al. 2010). Likewise, green ash in riparian systems of western North America could also be decimated by EAB.

EAB infestation of ash is already having extremely negative economic consequences for forest landowners, in urban areas where ash has been widely used for landscape and street trees, for tree nurseries, and for Native American tribes using ash as a cultural resource (Poland and McCullough 2006). Literature suggests the combined economic value of the loss of ash to EAB infestation on residential property and to forest landowners will be $4.4 billion in the United States over the next decade (Aukema et al. 2011; Kovacs et al. 2010, 2011). The ecological impacts of widespread EAB infestation will include altered forest composition and structure and negative effects on associated wildlife and ecosystem function, especially in hydric and mesic systems where ash is common (Gandhi and Herms 2010a, 2010b). Considering the economic and ecological consequences of EAB, projections of future ash forest composition would be beneficial for the management of North American ash forest resources.

Literature suggests the effects of EAB on North American ash forests may already be visible in U.S. Forest Service Forest Inventory and Analysis data (FIA). For instance, while ash increased throughout parts of the Midwest United States in the 1980s and 1990s, it has since substantially decreased, especially within 50 km of the invasion epicenter in southeastern Michigan where average ash volume decreased from 12.7 to 3.2 m^2 ha^{-1} between 2004 and 2009 (Pugh et al. 2011). Therefore, we used FIA data along with EAB current range, estimated spread rate, and host mortality data to provide both an assessment of the current ash resource and a projection of future EAB spread and subsequent ash mortality. These data were used in a series of submodels (Wear and Greis, in press, Wear et al. 2013) to project changes in the species composition, volume, and size class distribution of Midwest and Northeast ash forests between 2010 and 2060. Projections of land use change and climate were integrated into the modeling framework to simulate stand dynamics. In this paper, we briefly describe our modeling structure and provide some insight into the intensity and trajectory of the impact of EAB and its consequences for future stand development. Although ash can be found throughout the Midwest and Northeast, the highest concentrations of ash by number of trees and saplings are located in Maine, Michigan, Minnesota, New York, Pennsylvania, and Wisconsin. Therefore, this paper focuses on future projections for FIA inventory units in these six states (Figure 1, 2).

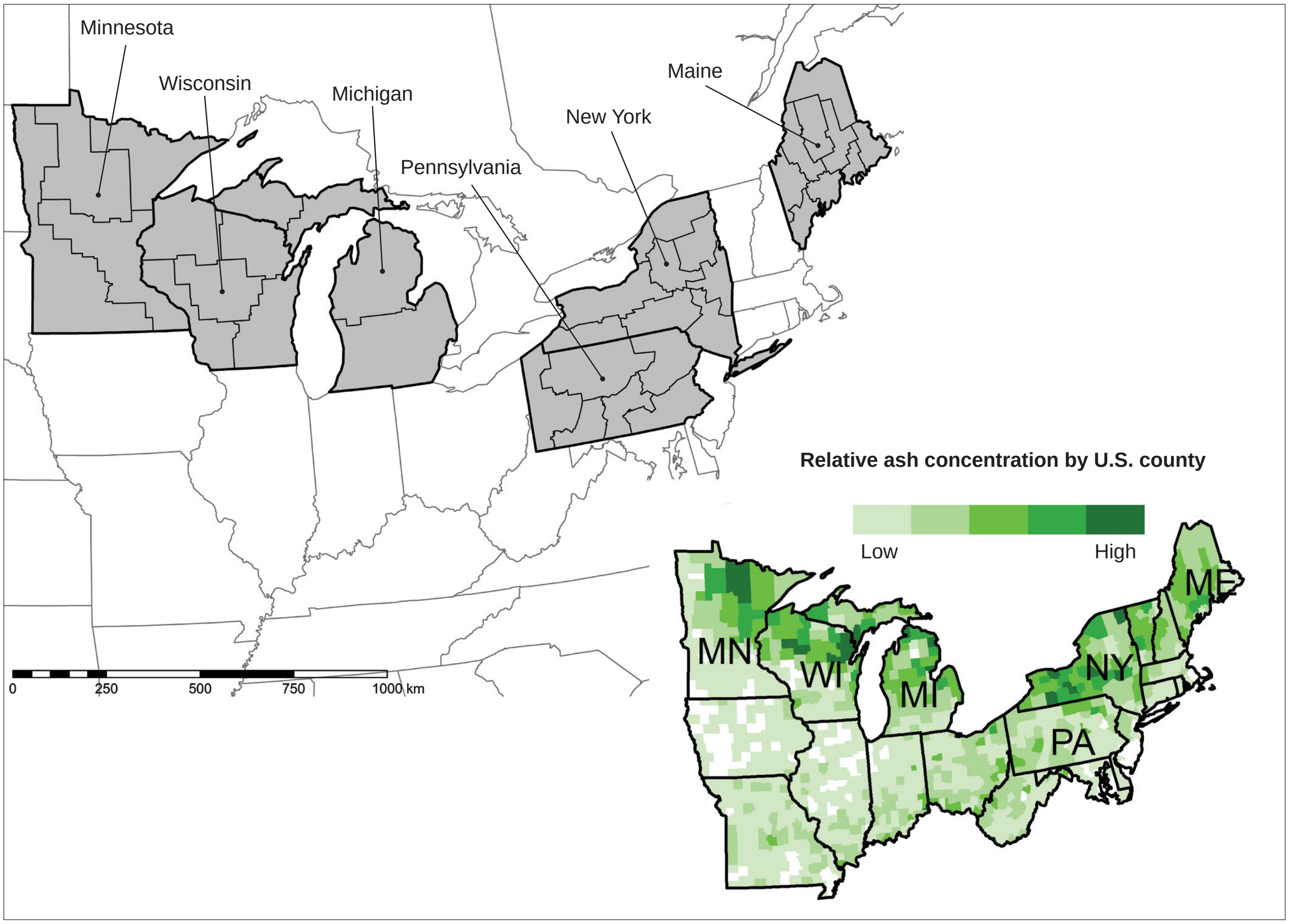

Figure 1.—Location of FIA inventory units in Maine, Michigan, Minnesota, New York, Pennsylvania, and Wisconsin in relation to the eastern United States and Canada. Map inset shows the concentration of ash (number of saplings and trees) by county in the Midwest and Northeast United States.

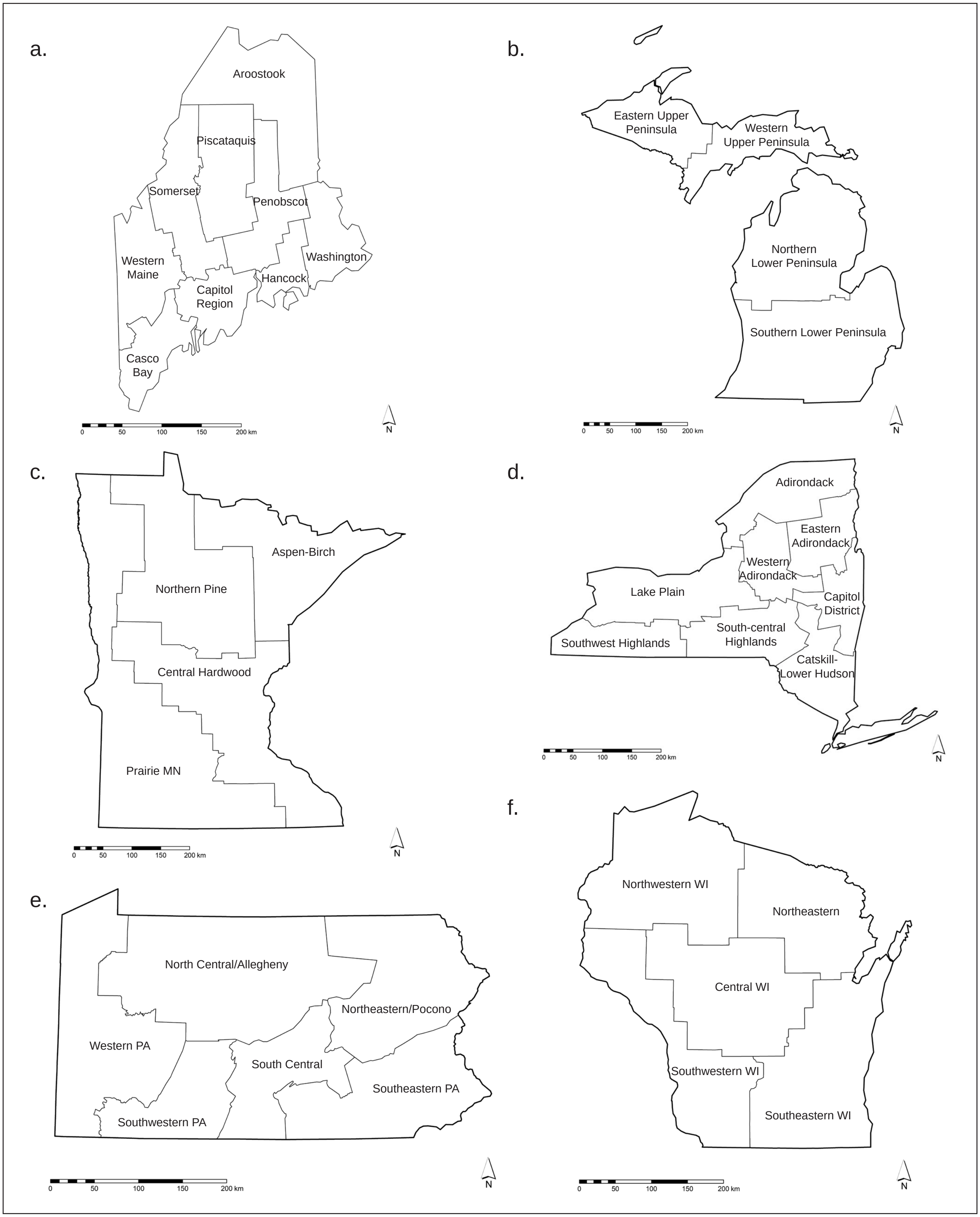

Figure 2.—Location, number, and name of FIA inventory units by state: a. Maine; b. Michigan; c. Minnesota; d. New York; e. Pennsylvania; f. Wisconsin.

METHODS

Forest Inventory and Analysis Program

FIA estimates forest area, volume, species composition, and other attributes using a nationwide sampling system with a tessellated design. The base sampling intensity is one plot per 2,400 ha, but it is sometimes augmented by intensification within selected states and on certain ownerships, such as National Forest land. The Northern Research Station FIA program covers 24 states in the upper Midwest and Northeast United States (the 20 states shown in inset map, Figure 1, plus North Dakota, South Dakota, Nebraska, and Kansas). FIA plots in this region are inventoried on a 5-year interval, with 20 percent of the plots in a state inventoried each year. The FIA program defines forest land as land that is a minimum of 0.4 ha in size, at least 36.6 m in width at the smallest dimension, and at least 10 percent stocking by live trees of any size, unless the land has been recently harvested and is anticipated to remain forested. Each FIA plot is approximately 0.067 ha and made up of four 7.32-m-radius subplots where all trees 12.7 cm diameter at breast height (d.b.h.) and greater are measured. Each subplot also contains a 13.5 m^2-microplot where trees 2.4 cm d.b.h. through 12.7 cm d.b.h. are measured. The plot design is one central subplot and three other subplots arranged in a spoke-like fashion at azimuths of 0, 120, and 240 degrees and 36.6 m from the center of the central subplot. A detailed explanation of design, techniques, and estimation procedures can be found in Bechtold and Patterson (2005).

U.S. Forest Service Northern Forest Futures Project

The purpose of the Northern Forest Futures Project is to forecast how current and future societal and natural resource trends might change the structure and composition of future forests and how those changes alter forest ecosystem services (Shifley et al. 2012). The knowledge gained through predicting future forest conditions can be used for decisionmaking, strengthening relationships between agencies, and influencing policy at multiple levels. The Northern Forest Futures Project focuses on a number of issues and trends, including improving environmental literacy, determining forest area, wood supply, fragmentation, parcelization, recreation pressures, forest management, water quality and supply, and wildlife habitat, and predicting future effects of invasive insects and disease. Using a baseline assessment of current forest conditions, the Northern Forest Futures Project creates projections of future forest conditions for 20 of the Northern Research Station states. Future projections are based on FIA forest-type groups and forecasts are created in 5-year increments for the period 2010-2060 (USDA Forest Service 2012; Wear[1]). The forest-type group category is used by FIA to group forest types which were developed from multiple sources including lists from FIA, the Society of American Foresters, and FIA analysts (Woudenberg et al. 2010). Forecasting is conducted using a scenario approach where there are ranges of plausible futures which are responsive to human population distributions, global economic conditions, energy and technology use, climate (three Intergovernmental Panel on Climate Change [IPCC] scenarios and four General Circulation Models [GCMs]), timber harvesting, land use change, other disturbance factors and natural succession (Wear[1]). In this report we compare results from a non-EAB (standard) future forest model and an EAB future forest model that incorporates the additional effects of EAB into the standard model, by analyzing percentage trend changes over time.

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[1] Wear, D., n.d. USFAS - the United States Forest Assessment System: Analysis to support forest assessment and strategic analysis. Proposal and project plan (version 3). Study plan on file at U.S. Department of Agriculture, Forest Service, Southern Research Station, Forestry Sciences Laboratory, Research Triangle Park, NC. 12 p.

Modeling Structure

The U.S. Forest Assessment System (USF^A^S) was used in Northern Forest Futures Project modeling to create projections of future forest composition by forecasting the potential role of human, physical, and biological factors in altering future forest inventories (Wear[1]; Wear et al. 2013). Projections were created for each of three IPCC scenarios and each scenario was linked to one of three GCMs, for a total of nine different storylines (combinations of IPCC scenarios and GCMs; Table 1). Timber harvest models inferred from historical harvest relationships were applied to each storyline, and projections utilized FIA annual inventory data (Ince et al. 2011). These projections did not include urban tree inventories. To project future forests, the USF^A^S incorporated models of forest succession along with the effects of changing climate, timber harvesting, and land use changes (Wear and Greis, in press). Empirical trends and relationships in FIA data between the two latest inventory periods for each state (2003 and 2008 for most of the states we analyzed here) were used to develop a set of transition and clustering models to simulate future forest inventories. The transition models predicted the age and movement of plots between forest types as well as any harvest activity across the 5-year time step. The clustering models produced a set of rules that predicted the plot productivity according to a set of plot characteristics such as age, ownership, and climate. These models, reflecting the 2003-2008 inventory dynamics, were applied to the 2008 FIA inventory to simulate the 2013 inventory. Subsequent applications of the models to the simulated inventories resulted in a set of projected forest inventories at 5-year increments from 2013 to 2058. We employed the convention of reporting the results at decadal and semi-decadal increments according to the closest projection year (i.e., the 2008 FIA inventory is referred to as 2010, the 2013 projection is referred to as 2015, … , the 2058 projection is referred to as 2060; e.g., Wear and Greis, in press).

Table 1.—Overview of IPCC scenarios used by the USF^A^S system. The initial drivers were population growth and GDP growth. Adapted from Nakicenovic et al. 2000, Coulson et al. 2010, Environment Canada, n.d.

IPCC scenario characteristics	IPCC scenario A1B	IPCC scenario A2	IPCC scenario B2
Global GDP growth	Very high	Medium	Medium
Global energy use	Very high	High	Medium
Oil and gas availability	High	Low	Medium
Technological pace and direction	Rapid: Gas, biomass, and other renewables	Slow: Coal and gas	Medium: Gas, oil, and biomass
Global population growth	Low	High	Medium
General description	Globalization, Economic convergence	Heterogenic regionalism, Less trade	Localized solutions, Slow change
General development themes	Introduction of new and more efficient technologies; Capacity building	Self-reliance, Preservation of local identities	Sustainable development, Diversified technology
Associated GCMs	MIROC, CGCM, CSIRO-b	MIROC, CGCM, CSIRO-b	HADLEY, CGCM, CSIRO-a

To contribute to knowledge benefitting the management of North America's ash resource, a tenth storyline incorporating EAB into the A2 CGCM storyline was used to project the effects of EAB on future forests. We used the A2 CGCM storyline for the standard model and as a baseline for the EAB model because we think this storyline generally represented intermediate levels of forest change compared to the other storylines (Table 1).

Inclusion of the EAB model necessitated establishment of the current range, spread rate, and host tree mortality for EAB. To determine the current range we used data from the U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Plant Protection and Quarantine program and the Canadian Food Inspection Agency to identify counties in the United States and regional municipalities in Canada where EAB was detected as of December 31, 2010 (Figure 3). Given its recent discovery in a novel

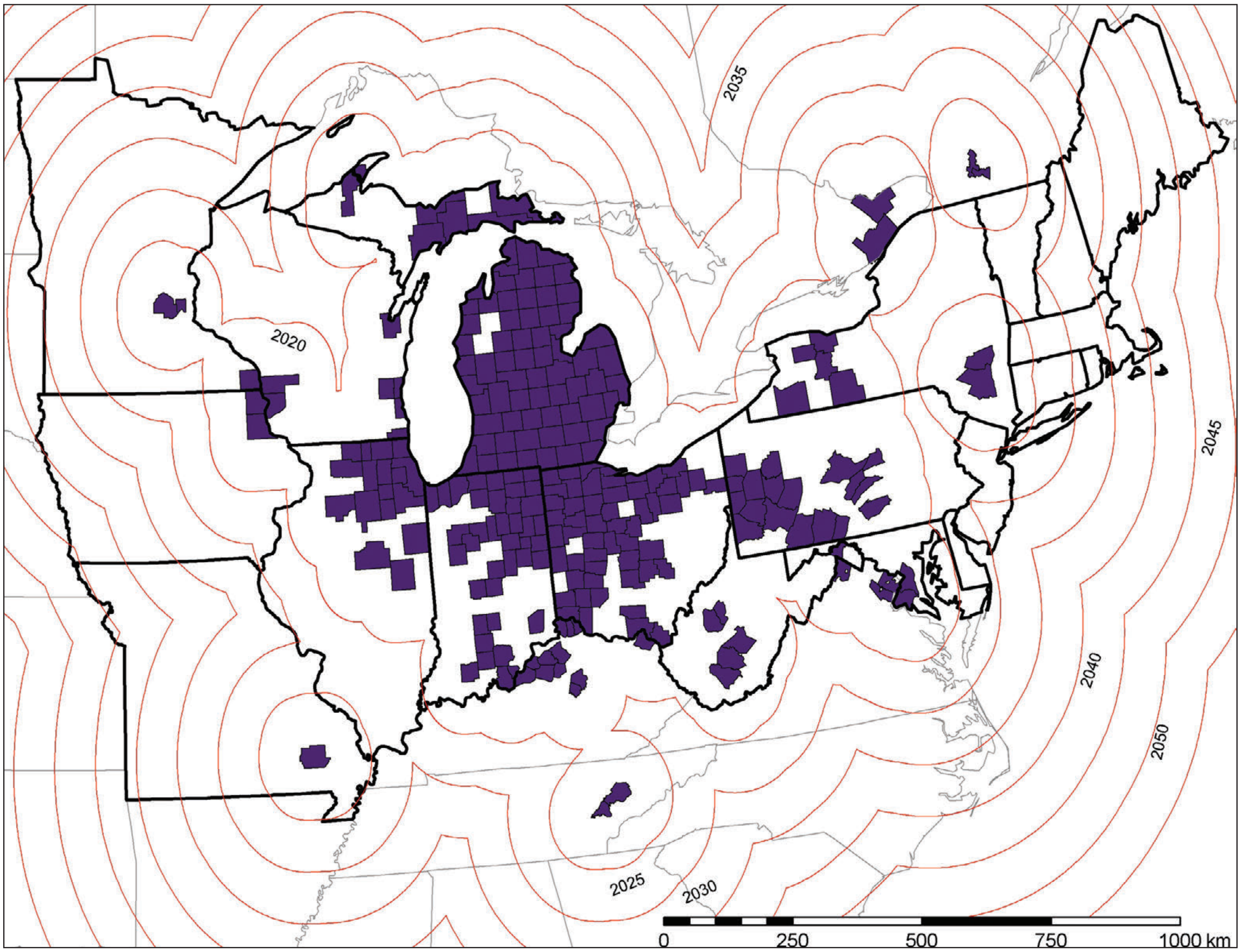

Figure 3.—Counties in the United States and regional municipalities in Canada where EAB was detected as of December 31, 2010, and projected 20 km yr $^{-1}$ EAB spread rate in 5-year intervals. Innermost red spread line corresponds with 2020 and outermost 2050. EAB presence is indicated by purple-shaded counties where EAB was detected and is based on data from U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Plant Protection and Quarantine program (Chaloux personal communication[2]); and Canadian Food Inspection Agency.

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[2] Paul Chaloux. 2011. Personal communication. National program manager, emerald ash borer program, U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Plant Protection and Quarantine, 4700 River Road, Riverdale, MD 20737.

environment, determining the EAB spread rate and host mortality probability was difficult (Poland and McCullough 2006). For the purpose of modeling future North American forests following EAB infestation, we assigned spread rate and host mortality probability estimates. Our analysis was confined to the Midwest and Northeast United States.

Spread Rate

Determining the spread rate of EAB for future projections of the North American ash resource was problematic for a number of reasons:

1. EAB-caused ash mortality occurs after extensive damage from larval galleries to phloem and outer sapwood girdles trees, disrupting water and nutrient transport (Cappaert et al. 2005). However, EAB may feed on individual trees at low population densities and damage can be difficult to detect due to the low probability of finding external signs such as characteristic adult-stage D-shaped exit holes (McCullough and Roberts 2002, Siegert[3]). For this reason, it is estimated that it can take up to 10 years from EAB site establishment until it is detected (Poland and McCullough 2006). This discrepancy between establishment and detection is common with other invasive insect pests, which remain at low densities until some other predisposing factor leads to tree stress, an exponential increase in insect density, or both (Shigesada and Kawasaki 1997). In addition, detecting EAB by assessing ash tree health status is difficult because many North American ash species are susceptible to numerous diseases which cause chlorosis, witches' broom, and abundant epicormic branching (PSU 1987).
2. EAB population dynamics in North America are still not entirely understood because of its fairly recent identification (Haack et al. 2002, Poland and McCullough 2006). This complicates modeling of its rate of spread.
3. Literature suggests two components to the spread rate of EAB: the initial spread from the core infested area in southeastern Michigan and human-assisted spread (Prasad et al. 2010, Siegert[3]). In addition, there are usually two or more phases of spread, whereby the initial rate is lower due to lower EAB density. Later, at high densities, EAB may exhibit quicker life cycles and satellite colonies coalesce, resulting in a much faster spread rate (Siegert et al. 2007, Siegert[3]). This makes determination of a single spread rate of EAB difficult.

The strongest line of evidence suggests the spread rate from the core infested area is influenced by short-range insect dispersal and short-range human-facilitated dispersal. As infestation satellites of human-assisted site establishment coalesce with the core infested area, EAB spread from the core infested area is estimated to be 20 km yr^{-1} (Iverson et al. 2010). Although new, long-range satellites of human-assisted establishment are possible, most current satellite infestations are new discoveries that became established before any regulations were in place (Siegert[3]). Current regulations prohibiting the transportation of firewood may help decrease the incidence of long-range EAB spread (BenDor et al. 2006, Poland and McCullough 2006). In the future it seems likely that fewer distant satellites will emerge and EAB spread will be driven mostly by the occurrence of satellites located near the periphery of the infestation.

Mortality Probability

Throughout its native range in northeastern China, Korea, Japan, Mongolia, Taiwan, and eastern Russia, EAB feeds on Chinese ash, Manchurian ash, and other ash species (Anulewicz et al. 2008, Cappaert

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[3] N.W. Siegert. 2010. Personal communication. Forest entomologist, U.S. Department of Agriculture, Forest Service, Northeastern Area State and Private Forestry, 271 Mast Road, Durham, NH 03824.

et al. 2005, Jendek 1994) but usually does not cause extensive damage because EAB remains at low population densities and host trees have developed some level of host resistance (Chen and Poland 2009, McCullough et al. 2009, Pureswaran and Poland 2009). When introduced to a novel environment like North America, related species of ash are suitable hosts for EAB but do not contain the same level of resistance as do Asian ash species (Rebek et al. 2008). Asian ash species contain much higher levels of host volatiles and other defensive mechanisms unfavorable to EAB (Eyles et al. 2007). EAB uses both olfactory and visual cues to determine host suitability and has demonstrated host preference for the major North American ash species (green [*Fraxinus pennsylvanica* Marsh.], white [*Fraxinus americana* L.] and black ash), as well as blue (*Fraxinus quadrangulata* Michx.) and pumpkin ash (*Fraxinus profunda* (Bush) Bush). However, no evidence exists for EAB attacking North American non-ash species (Anulewicz et al. 2008, Pureswaran and Poland 2009). As with other insects from the family Buprestidae, EAB is generally attracted to trees stressed by other factors (e.g., girdling), but in North America EAB will also attack healthy ash trees (Anulewicz et al. 2008).

For modeling purposes, we assumed EAB-caused mortality of green, white, and black ash (>2.5 cm d.b.h.) in the Midwest and Northeast United States will be approximately 100 percent upon full EAB exposure (Herms[4], Herms et al. 2010).

EAB Simulation Protocol

We simulated the effects of EAB on forests over 50 years in 5-year time steps beginning with FIA inventory year 2008 and ending with 2058. However, as previously discussed, we employed the convention of reporting the results at decadal and semi-decadal increments according to the closest projection year (e.g., 2010, 2015, … , 2060). EAB spread subsumes the entirety of the Midwest and Northeast United States by 2050 but our projections of future forests were carried out through 2060. To determine the core infested area and satellite infestations, we identified counties in the United States and regional municipalities in Canada where EAB was detected as of December 31, 2010, and projected a 20 km yr^{-1} spread rate from the core infested area in 5-year intervals (Figure 4) as related to Midwest and Northeast FIA inventory units (FIA inventory units are essentially groups of counties; Figure 4). EAB detection in each inventory unit corresponded with a 5-year increment in the following way: Since trends between 2003 and 2008 (i.e., 2010) data were used to project future forests, the first projection period was 2008 to 2013 (i.e., 2015). Because we identified EAB detection as of 2010 and because 2010 occurred during the 2015 time step (i.e., 2008 to 2013), the first EAB projection was for 2015. We analyzed different scenarios of EAB spread and subsequent ash mortality by county and by inventory unit, including: 1) assuming EAB spread leads to ash mortality immediately upon spread arrival in each analysis unit, and 2) assuming EAB spread leads to ash mortality once the spread subsumes the centroid of each analysis unit, but for the purposes of this exercise, we chose to 3) assume EAB spread leads to ash mortality once the spread subsumes each inventory unit (Figure 4).

Considering strong evidence suggesting: 1) EAB causes >99 percent host tree mortality probability including sprouts >2.5 cm d.b.h., 2) EAB site establishment can occur >10 years before detection, and 3) EAB-infested trees do not typically live long, we created a spread model that assumed complete ash mortality in a given inventory unit once EAB spread subsumed that inventory unit in its entirety (Herms et al. 2010, Poland and McCullough 2006). We selected 5-year intervals in which each inventory unit would be subsumed by EAB infestation and used these intervals as temporal indicators to simulate total ash mortality in each inventory unit (Figure 4). In each inventory unit, once ash mortality due to EAB was simulated, forests were projected following USF^{A}S protocols.

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[4] D. Herms. 2010. Personal communication. Professor and associate chairperson, Department of Entomology, Ohio Agricultural Research and Development Center, Ohio State University, 1680 Madison Avenue, Wooster, OH 44691.

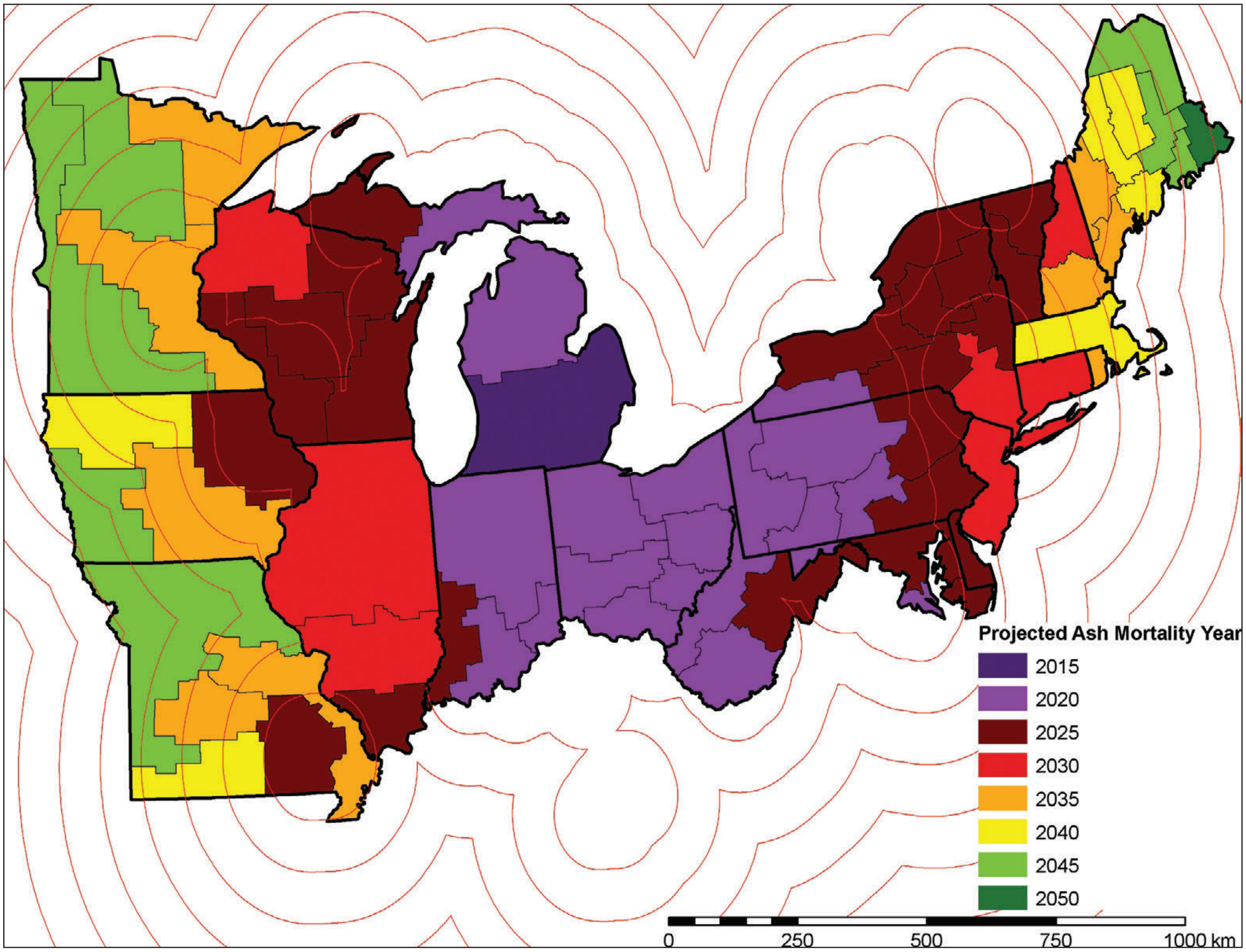

Figure 4.— Projected total mortality of ash due to EAB in each FIA inventory unit of Midwest and Northeast United States, using dates when EAB spread subsumes each inventory unit. EAB spread in New York, Vermont, New Hampshire, and Maine is influenced by present EAB infestations in regional municipalities of Ontario and Québec, Canada. Projections assume a) EAB spread is not influenced by EAB infestations in other Canadian locations or southeastern United States locations in Tennessee, Kentucky, or Virginia; and b) EAB spread leads to ash mortality once the spread subsumes each inventory unit.

Forest-type Groups Potentially Affected by EAB

We analyzed projections by forest-type groups instead of species because: 1) our projections were modeled by forest-type groups and not by species, and 2) ash is primarily a component of the elm-ash-cottonwood (E-A-C) forest-type group (for the states we analyzed here, ash constituted 34 percent of E-A-C by number of trees and saplings) and to a lesser extent the oak-hickory (O-H) forest-type group (6 percent; Table 2). However, there are differences within forest-type groups based on geographic area. For example, the ash component of E-A-C may be predominately white ash in Maine inventory units, while in northern Minnesota inventory units, the ash component of E-A-C may be entirely composed of green and black ash. Since white ash is typically found in more upland and less mesic areas than are black or green ash, there is no white ash forest type in the fairly mesic E-A-C; white ash forest types are included in, but do not

Table 2.—Estimated total number of saplings and trees in billions, on forest land by state, year, and model.

				Standard model		EAB model	
State	2010 ash percent of E-A-C	2010 ash percent of O-H	2010 number of saplings and trees	2060 number of saplings and trees	Percent change (2010 to 2060)	2060 number of saplings and trees	Percent change (2010 to 2060)
ME	19	3	23.32	20.87	-11	20.54	-12
MI	32	6	14.03	12.25	-13	11.25	-20
MN	47	12	13.06	11.37	-13	9.97	-24
NY	26	10	12.19	11.54	-5	10.93	-10
PA	14	3	8.35	7.27	-13	6.94	-17
WI	37	6	10.92	9.56	-12	8.67	-21
Total	34	6	81.87	72.86	-11	68.30	-17

make up a particularly large part, of O-H. Therefore, in areas where most ash is white ash and there is little green ash or black ash found, EAB effects on E-A-C could be minimal. If ash composes a substantial component of O-H in those areas, EAB effects on O-H could be more apparent than those on E-A-C. On the other hand, effects of EAB on E-A-C should be large in areas abundant with black and green ash, considering these species are a substantial component of E-A-C. This might be the case in Minnesota's "Aspen-Birch" and "Northern Pine" inventory units, where white ash is uncommon and green ash and black ash are common (Figure 2c). Generally, ash is a defining component of riparian systems in the northern Midwest and Northeast, but often does not constitute a large component of any forest-type group on FIA plots (Table 2).

RESULTS

Current Ash Resource

According to 2008 FIA data, the five species of ash native to the Midwest and Northeast total over 1.3 billion trees (≥12.7 cm d.b.h.) comprising an estimated volume of 427 million m^3 and more than 225 million metric tons of above and belowground carbon. Approximately 78 percent of these trees are located on private land, 17 percent on State and local government land, and 5 percent on Federal land. There are an additional 4.3 billion ash saplings (2.5 to 12.6 cm d.b.h.); 3.3 billion are <7.6 cm d.b.h. Ash is present on 32 percent of all forest land in the Midwest and Northeast, but usually makes up ≤25 percent of total stand basal area where present (Miles 2011). When measured by the total number of stems >2.5 cm d.b.h., ash is most abundant in Minnesota, New York, Michigan, Wisconsin, Maine, and lastly Pennsylvania (Figure 1 inset). Black ash is the predominant species in these states, followed by green ash and then white ash; collectively, these three species constitute more than 99 percent of the ash in the Midwest and Northeast (Miles 2011).

Future Forest Land

Trends in future forests generally differ among states and FIA inventory units. Our projections suggest a decrease in forest land in all states, with the greatest percentage decrease in states with the highest population densities and least amount of forest land, such as New Jersey and Rhode Island (data not shown). In states with the most forest land, the projections generally indicate a small decrease in forest land area, and larger decreases in the most populated FIA inventory units of each state, such as Michigan's "Southern Lower Peninsula" inventory unit (including the Detroit, Grand Rapids, Lansing, Ann Arbor, and Flint metro areas), Minnesota’s “Central Hardwoods” inventory unit (Minneapolis-St. Paul, Rochester, and St. Cloud metro areas), and Wisconsin’s “Southeastern” inventory unit (Milwaukee, Madison, and Green Bay metro areas; Figure 2).

Future Forests: Number of Trees

Our projections suggest a decrease in the number of all trees and saplings with the standard model, and a larger decrease with the EAB model by 2060 (Table 2). The EAB model projects the loss of all ash by 2050 for all states (Figure 4). The standard model projects E-A-C and O-H forests to increase in some FIA inventory units, except in FIA inventory units where ash decreased prior to the projections timeframe due to EAB or other factors (e.g., the Southern Lower Peninsula inventory unit of Michigan; Table 3 and Figure 2). The EAB model projects E-A-C forests to decrease in most FIA inventory units. However, in heavily forested, mostly undeveloped FIA inventory units, such as both inventory units of Michigan’s Upper Peninsula and Minnesota’s Aspen-Birch inventory unit, the EAB model projects a substantial increase in E-A-C and O-H forests. The EAB model projects a substantial increase in E-A-C forests in Pennsylvania’s Northeastern/Pocono inventory unit and Maine’s Aroostook County inventory unit. In the Northern Pine inventory unit of Minnesota, both the standard and EAB models project substantial decreases in O-H forests. In the Central Hardwood inventory unit of Minnesota and both Lower Peninsula inventory units of Michigan, both the standard and EAB models project substantial decreases in O-H forests.

Future Forests: Tree Volume

Volume projections vary substantially across states (Table 4). E-A-C and O-H volume varies among FIA inventory units and does not appear to follow any discernible trend (Table 5). However, both the standard and EAB models project substantial increases in O-H forests in Minnesota’s Aspen-Birch inventory unit, E-A-C forests in Pennsylvania’s Western inventory unit and Minnesota’s Central Hardwood inventory unit, O-H and E-A-C forests in New York’s South-Central Highlands inventory unit, O-H forests in New York’s Western Adirondack inventory unit, and O-H and E-A-C forests in Michigan’s Upper Peninsula inventory units, Wisconsin’s northern inventory units, and Maine’s Aroostook County inventory unit (Table 5, Figure 2).

Future Forests in General

The EAB model projects greater volume decreases in most forest-type groups than does the standard model (Table 6). This is especially true for forest-type groups where ash is a major component such as E-A-C, while this is not true for forest-type groups where ash is a minor component, such as spruce-fir. However, the standard model projects greater volume decreases in E-A-C than does the EAB model in Maine, the standard model projects greater decreases in O-H than does the EAB model in Minnesota, and the standard model projects greater decreases in O-H than does the EAB model in Pennsylvania.

DISCUSSION

Regardless of the model type used, all invasive insect pest modeling systems have their drawbacks (Neubert and Caswell 2000, Prasad et al. 2010). Many modeling systems incorporate projections of new EAB satellite infestations (e.g., BenDor et al. 2006, Crocker and Meneguzzo 2009, MacFarlane and Meyer 2005, McCullough and Siegert 2007, Mercader et al. 2009, Muirhead et al. 2006, Siegert et al. 2010). Our approach used a series of sub-models which partitioned plots from current forest inventories by identifying important attributes, forecasted those attributes, created future forest inventories from those forecasts, and linked the future inventories to land use changes. It is important to note that our projections are for land measured by FIA and do not include urban tree inventories. The inclusion of FIA time-series data and human population, energy, consumption,

Table 3.—Estimated number of saplings and trees in millions, in the elm-ash-cottonwood (E-A-C) and oak-hickory (O-H) forest-type groups on forest land, by state, FIA inventory unit, model, and year.

		Number of E-A-C saplings and trees				Number of O-H saplings and trees			
			Standard model	EAB model			Standard model	EAB model	
State	FIA inventory unit	2010	2060	Initial post-EAB[1]	2060	2010	2060	Initial post-EAB[1]	2060
ME	Washington	35.17	26.30	3.13	10.98	0	20.31	23.18	19.36
	Aroostook	18.27	65.37	56.49	25.74	0	50.73	22.77	25.66
	Penobscot	31.65	11.42	28.23	25.21	11.43	24.63	10.33	15.44
	Hancock	10.38	0.86	13.88	8.26	13.78	3.48	5.20	9.03
	Piscataquis	36.52	45.79	21.29	42.66	6.34	43.29	0	28.01
	Capitol Region	62.26	0	8.04	5.42	44.54	37.40	30.85	17.90
	Somerset	45.39	64.20	77.08	46.89	0	12.19	22.19	21.09
	Casco Bay	74.19	20.32	25.50	10.28	154.14	19.57	80.29	43.09
	Western Maine	19.71	13.77	48.61	29.77	23.46	34.18	17.26	16.98
	ME Total	333.55	248.03	282.25	205.20	253.68	245.78	212.08	196.55
MI	Eastern Upper Peninsula	223.92	271.02	222.62	264.05	34.88	344.66	108.07	193.61
	Western Upper Peninsula	170.45	355.70	266.26	214.47	43.01	400.72	161.29	329.08
	Northern Lower Peninsula	453.57	486.30	444.10	350.40	771.39	520.15	722.26	525.08
	Southern Lower Peninsula	439.26	191.05	237.77	112.99	875.99	319.05	686.40	365.62
	MI Total	1287.20	1304.06	1170.75	941.90	1725.27	1584.58	1678.03	1413.39
MN	Aspen-Birch	358.47	739.71	547.25	433.46	22.10	308.24	183.88	298.63
	Northern Pine	322.81	588.83	558.96	237.88	403.98	404.77	440.95	371.90
	Central Hardwood	181.00	217.73	237.23	131.55	576.44	171.88	263.53	232.64
	Prairie MN	71.15	52.45	61.88	33.33	94.07	26.51	46.84	60.52
	MN Total	933.42	1598.71	1405.33	836.23	1096.59	911.39	935.21	963.70
NY	Adirondack	163.59	0	64.46	20.57	133.91	243.73	214.90	154.54
	Lake Plain	239.51	0	67.07	8.45	375.60	261.89	426.61	252.21
	Western Adirondack	83.85	0	39.33	1.48	47.90	218.66	145.63	216.38
	Eastern Adirondack	0.31	0.31	0.00	0	53.92	211.16	109.78	176.42
	Southwest Highlands	41.79	7.55	31.00	13.33	301.27	541.61	339.51	402.30
	South-Central Highlands	12.89	17.90	16.29	6.60	224.32	520.94	393.14	452.15
	Capitol District	36.85	0	2.62	0.03	190.22	119.51	198.10	122.19
	Catskill-Lower Hudson	67.28	67.39	89.59	41.44	428.14	481.93	514.11	483.55
	NY Total	646.07	93.15	310.35	91.90	1755.28	2599.43	2341.78	2259.73
PA	South Central	8.11	12.88	8.06	0	692.52	449.51	666.74	470.48
	Western PA	41.12	5.43	31.31	7.16	722.05	616.87	665.58	728.64
	North Central/Allegheny	4.28	2.55	4.10	1.48	1281.24	1452.55	1266.66	1453.69
	Southwestern PA	3.88	6.56	1.42	5.62	586.50	469.87	508.39	433.08
	Northeastern/Pocono	26.51	39.50	21.98	31.48	909.31	758.04	840.22	773.39
	Southeastern PA	23.47	26.28	27.41	9.02	398.45	215.30	314.38	185.35
	PA Total	107.36	93.19	94.29	54.76	4590.06	3962.14	4261.97	4044.63
WI	Northeastern	226.76	232.10	194.30	194.51	187.61	515.35	297.06	408.23
	Northwestern WI	297.31	266.92	305.35	205.47	528.70	677.85	641.40	757.01
	Central WI	167.58	134.76	130.75	69.69	604.91	320.55	535.43	414.96
	Southwestern WI	84.75	60.63	71.94	65.54	686.69	363.90	512.25	338.96
	Southeastern WI	160.99	51.00	153.73	65.45	232.11	151.26	206.41	129.09
	WI Total	937.39	745.41	856.07	600.66	2240.01	2028.91	2192.55	2048.26
Grand Total		4244.98	4082.56	N/A (different years)	2730.65	11660.89	11332.23	11621.61	10926.25

[1] Refers to the first 5-year interval following EAB-caused total ash mortality, for each FIA inventory unit individually (Figure 4).

Table 4.—Total tree volume on forest land, in million cubic meters, by state, year, and model.

			Standard model		EAB model	
State	2010 percent of total land in forest land	2010 volume	2060 volume	Percent volume change	2060 volume	Percent volume change
ME	89	721.53	765.77	6	735.67	2
MI	55	893.47	970.97	9	927.14	4
MN	33	512.51	634.39	24	591.84	15
NY	63	1121.59	1197.25	7	1152.14	3
PA	58	1002.02	986.75	-2	984.24	-2
WI	48	658.13	817.09	24	766.66	16
Total	53	4909.25	5372.22	9	5157.69	5

Table 5.—Volume of trees in the elm-ash-cottonwood (E-A-C) and oak-hickory (O-H) forest-type groups on forest land, in million cubic meters, by state, FIA inventory unit, model, and year.

		E-A-C volume			O-H volume		
State	FIA inventory unit	2010	Standard model 2060	EAB model 2060	2010	Standard model 2060	EAB model 2060
ME	Washington	0.25	0.24	0.04	0	2.22	2.45
	Aroostook	0.77	2.30	1.55	0	3.91	3.84
	Penobscot	1.50	0.52	1.46	0.66	2.30	1.26
	Hancock	0.49	0.06	0.61	1.30	0.34	0.68
	Piscataquis	1.31	1.45	2.18	0.29	2.67	1.49
	Capitol Region	2.37	0	0.51	2.84	3.13	2.49
	Somerset	1.20	0.97	0.89	0	0.98	1.16
	Casco Bay	3.14	0.46	0.36	11.07	1.78	3.83
	Western Maine	0.51	0.40	1.25	2.14	3.40	2.04
	ME Total	11.54	6.40	8.85	18.30	20.72	19.23
MI	Eastern Upper Peninsula	9.46	13.04	10.42	1.67	19.67	22.21
	Western Upper Peninsula	8.29	19.08	14.04	3.05	34.58	31.81
	Northern Lower Peninsula	23.26	25.20	22.26	58.35	53.67	55.37
	Southern Lower Peninsula	43.97	22.03	14.86	87.84	43.81	41.54
	MI Total	84.98	79.35	61.58	150.92	151.73	150.93
MN	Aspen-Birch	12.93	25.47	12.27	1.42	22.65	26.46
	Northern Pine	16.01	27.49	12.60	26.51	27.81	30.31
	Central Hardwood	15.58	40.81	38.09	53.54	16.49	21.93
	Prairie	7.47	17.85	9.43	9.59	3.21	4.95
	MN Total	52.00	111.63	72.38	91.06	70.16	83.64
NY	Adirondack	6.58	0	1.09	5.92	14.93	9.97
	Lake Plain	28.11	0	0.46	28.44	25.12	25.04
	Western Adirondack	4.76	0	0.05	1.81	11.20	12.90
	Eastern Adirondack	0.02	0.05	0	3.46	12.42	9.42
	Southwest Highlands	3.53	0.01	0.19	19.85	55.81	39.49
	South-Central Highlands	0.74	3.23	3.29	23.84	70.81	67.74
	Capitol District	4.99	0	0	23.37	17.80	15.77
	Catskill-Lower Hudson	8.99	9.06	4.86	60.80	78.45	64.63
	NY Total	57.72	12.35	9.95	167.49	286.54	244.94

(Table 5 continued on next page)

Table 5 (continued).—Volume of trees in the elm-ash-cottonwood (E-A-C) and oak-hickory (O-H) forest-type groups on forest land, in million cubic meters, by state, FIA inventory unit, model, and year.

		E-A-C volume			O-H volume		
State	FIA inventory unit	2010	Standard model 2060	EAB model 2060	2010	Standard model 2060	EAB model 2060
PA	South Central	0.72	0.79	0	78.39	54.84	55.41
	Western PA	4.64	0.86	2.18	90.23	107.23	105.16
	North Central/Allegheny	0.21	0.25	0.03	160.98	192.47	183.48
	Southwestern PA	0.29	0.14	0.08	49.74	51.07	49.85
	Northeastern/Pocono	4.29	11.26	8.73	91.93	97.46	89.90
	Southeastern PA	4.47	9.44	1.37	66.97	39.93	39.89
	PA Total	14.62	22.73	12.39	538.25	543.00	523.69
WI	Northeastern	8.86	18.90	17.41	14.45	42.78	44.43
	Northwestern WI	13.24	22.15	18.01	35.63	63.00	60.42
	Central WI	11.46	14.75	7.76	50.21	34.74	42.75
	Southwestern WI	10.88	8.93	11.03	59.98	33.15	31.89
	Southeastern WI	13.82	6.91	5.16	22.58	16.70	15.36
	WI Total	58.27	71.64	59.37	182.85	190.37	194.85
Grand Total		279.14	304.10	224.53	1148.87	1262.52	1217.28

Table 6.—Total volume of trees on forest land, in million cubic meters, and percent change in total volume of trees on forest land, by state, forest-type group, and model.

		Percent change 2010-2060	
Maine Forest-type group	2010 total volume	Standard model	EAB model
White-red-jack-pine	77.38	0	1
Spruce-fir	212.64	11	4
Oak-hickory	18.30	13	5
Elm-ash-cottonwood	11.54	-45	-23
Maple-beech-birch	308.64	10	5
Aspen-birch	67.19	-13	-9

		Percent change 2010-2060	
Michigan Forest-type group	2010 total volume	Standard model	EAB model
White-red-jack-pine	100.58	52	49
Spruce-fir	109.21	-6	-9
Oak-hickory	150.92	1	0
Elm-ash-cottonwood	84.98	-7	-28
Maple-beech-birch	313.00	8	6
Aspen-birch	105.35	2	-9

		Percent change 2010-2060	
Minnesota Forest-type group	2010 total volume	Standard model	EAB model
White-red-jack-pine	49.70	21	23
Spruce-fir	79.76	5	-3
Oak-hickory	91.12	-23	-8
Elm-ash-cottonwood	52.00	115	39
Maple-beech-birch	53.77	-18	-26
Aspen-birch	170.84	48	44

		Percent change 2010-2060	
New York Forest-type group	2010 total volume	Standard model	EAB model
White-red-jack-pine	90.04	22	3
Spruce-fir	35.84	17	33
Oak-hickory	167.49	71	46
Elm-ash-cottonwood	57.72	-79	-83
Maple-beech-birch	663.28	1	3
Aspen-birch	29.89	-57	-61

		Percent change 2010-2060	
Pennsylvania Forest-type group	2010 total volume	Standard model	EAB model
White-red-jack-pine	26.46	16	15
Spruce-fir	1.32	-25	-25
Oak-hickory	538.25	1	-3
Elm-ash-cottonwood	14.62	55	-15
Maple-beech-birch	364.00	-8	-1
Aspen-birch	10.37	61	53

		Percent change 2010-2060	
Wisconsin Forest-type group	2010 total volume	Standard model	EAB model
White-red-jack-pine	80.51	126	114
Spruce-fir	40.27	10	4
Oak-hickory	182.85	4	7
Elm-ash-cottonwood	58.27	23	2
Maple-beech-birch	180.86	-4	-7
Aspen-birch	88.20	11	-1

land use, and economic models likely improved our modeling system's ability to project future forest composition. Due to the complicated nature of EAB spread dynamics, our projections of the timing of EAB establishment in specific FIA inventory units could be a model weakness. However, considering strong evidence for the EAB spread rate and host mortality probability we utilized, there is a high likelihood of EAB affecting our entire study area by 2050 as our projections suggest (Herms et al. 2010, Iverson et al. 2010) (Figure 4). Therefore, due to the likely substantial decrease in ash, we focus our conclusions on projected forest composition changes.

In states or FIA inventory units where ash is not an important genus (e.g., Maine; 2 percent of total growing-stock volume), there is little or no difference between the standard and EAB models (Table 7). In states or FIA inventory units where ash is a more prominent genus (e.g., Minnesota; 8 percent of total growing-stock volume), detecting a difference between the standard and EAB models is more likely. Minnesota and Maine are on opposite sides of the spectrum regarding the effect of ash mortality on differences between the EAB and standard models. Ash constitutes a much greater portion of the total growing stock volume in Minnesota than it does in Maine, which contributes to a greater difference between the standard and EAB models in Minnesota than in Maine. In Minnesota, because ash represents a larger component of forest and is predominately found in E-A-C forests, the majority of changes in forest types involve forest types in the E-A-C forest-type group. In addition, there are differences between the standard and EAB model results for both number of trees and volume in Minnesota (Tables 2, 3, 4, 5, and 6). In Maine, because ash represents a very small component of the forest and since ash is more prevalent in O-H than E-A-C forests, most changes in forest types do not involve forest types in E-A-C forests and the standard and EAB model results are similar for both number of trees and volume. Therefore, the EAB model does not appear to substantially alter Maine's E-A-C projections trajectory. In addition, O-H number of trees and volume trends between the standard and EAB model results are similar. The similarity between the standard and EAB model results for E-A-C and O-H forests is likely an effect of ash representing a very small proportion of total growing stock in Maine.

Since we summarize results at the scale of states and inventory units and EAB is only known to kill ash, the coarse scale of our analysis units and the relative importance of ash in each analysis unit play important roles in our results. There could be more and greater differences between the standard and EAB models if our analysis were to be conducted on a finer scale (e.g., sub-inventory unit) and in locations where ash is a more prominent genus. Assuming ash is not a prominent genus in most Midwest and Northeast states, removing it from the landscape altogether may not substantially affect forest composition in terms of analysis by forest-type group. On the other hand, this

Table 7.—Net volume of growing-stock trees at least 12.70 cm d.b.h. on forest land, in million cubic meters, by state.

State	2008 net growing-stock volume	2008 ash growing-stock volume	2008 ash percentage of growing-stock volume
ME	673.66	14.87	2
MI	827.05	40.08	5
MN	443.82	35.11	8
NY	1032.29	76.11	7
PA	939.29	46.54	5
WI	601.61	37.15	6

may only be true for FIA plots, and not necessarily for urban areas not measured by FIA, where EAB could have a greater impact due to the abundance of ash; data from Atlanta, Baltimore, Boston, Chicago, New York, Syracuse, Oakland, and Philadelphia suggest ash trees contribute up to 14 percent of the total urban leaf area, and perhaps even more in north-central and western states (Federal Register 2003, Poland and McCullough 2006). In contrast, according to FIA data, ash represents only 5 percent of the total basal area in Midwest and Northeast FIA plots.

Generally, the less ash there is in each state, the more stochastic the model results are, thereby increasing the chances of the standard and EAB models producing similar results (Tables 3 and 5). In addition, the probability of forest compositional changes including transitioning into or out of forest-type groups with ash such as E-A-C or O-H is highly influenced by trends found between the 2003 and 2008 inventories. In other words, the standard model in Minnesota projects an increase in E-A-C because E-A-C forests increased from 2003 to 2008, whereas the standard model in Maine projects a decrease in E-A-C because E-A-C forests decreased from 2003 to 2008. The standard model in Michigan also projects a decrease in E-A-C forests because E-A-C decreased from 2003 to 2008. However, the decrease may have been partly due to EAB effects on ash, especially in the Southern Lower Peninsula inventory unit, in which EAB has likely been established since the early- to mid-1990s (Haack et al. 2002, Siegert et al. 2007).

For most FIA inventory units, the EAB model projects a small decrease in the number of E-A-C saplings and trees immediately following EAB-caused ash mortality (Table 3). This is followed by a small increase in the number of E-A-C saplings and trees, after which the EAB model appears to mimic the standard model trends in the number of E-A-C saplings and trees. These results suggest other mesic species in the E-A-C forest-type group could increase and fill gaps left by ash tree mortality. However, EAB-caused ash mortality and subsequent canopy gaps could enable invasion by exotic invasive plant species such as Canada thistle (*Cirsium arvense* (L.) Scop.) and Japanese honeysuckle (*Lonicera japonica* Thunb.) (Hausman et al. 2010, Ruzicka et al. 2010). Oak-hickory trends vary among states, with some net increases and some net decreases after 50 years. FIA inventory units in northern Maine, Michigan, Minnesota, and Pennsylvania contain some of the highest ash concentrations by inventory unit, yet they are sparsely populated and contain abundant riparian area protected by Federal, State, or local government (Miles 2011). Therefore, while EAB infestation may lead to the removal of ash, the lack of land development in these FIA inventory units could allow other species to increase enough to compensate for the loss of ash.

Wildlife is generally not dependent on ash, but benefits from a variety of species in E-A-C and O-H (unpublished report[5], Myers and Buchman 1984, Poland and McCullough 2006). However, Gandhi and Herms (2010b) documented a large number of arthropod species that utilize ash, including at least 44 species that utilize ash exclusively and thus are at risk of coextirpation.

Dutch elm disease (*Ophiostoma ulmi* (Buism.) Nannf. and *Ophiostoma novo-ulmi* Brasier) and EAB will likely cause a substantial decrease in elm (*Ulmus* spp.) and ash, especially considering the rapid spread of EAB and the establishment of Dutch elm disease in all midwestern and northeastern states (Schlarbaum et al. 2002). However, considering E-A-C contains more than just elm and ash species, increases in other E-A-C species have the potential to mitigate the loss of elm and ash. Likewise, given the importance of oak and hickory in the O-H forest-type group, the loss of ash may be mitigated by genera more prominent than ash

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[5] Draft report by R. Heyd. 2005. Ash (*Fraxinus* spp.) management guidelines. Emerald ash borer response strategy. On file at Michigan Department of Natural Resources, Lansing, MI. 24 p.

in O-H. There are a number of potential replacements for ash in mesic and hydric E-A-C forests, including red maple (*Acer rubrum* L.), river birch (*Betula nigra* L.), American sycamore (*Planatus occidentalis* L.), eastern cottonwood (*Populus deltoides* Bartram ex Marsh.), willow (*Salix* spp.), pecan (*Carya illinoensis* (Wangenh.) K. Koch), sugarberry (*Celtis laevigata* Willd.), hackberry (*Celtis* spp.), and silver maple (*Acer saccharinum* L.) (Burns and Honkala 1990, Woudenberg et al. 2010). In addition, before succumbing to Dutch elm disease, American elm (*Ulmus americana* L.) in smaller size classes could serve as another replacement for ash in mesic and hydric areas. Potential replacements for ash in drier upland O-H forests include eastern white pine (*Pinus strobus* L.), northern red oak (*Quercus rubra* L.), cherry (*Prunus* spp.), yellow-poplar (*Liriodendron tulipfera* L.), elm (*Ulmus* spp.), black locust (*Robinia pseudoacacia* L.), eastern redcedar (*Juniperus virginiana* L.), post oak (*Quercus stellata* Wangenh.), blackjack oak (*Quercus marilandica* Münchh.), chestnut oak (*Quercus prinus* L.), white oak (*Quercus alba* L.), sassafras (*Sassafras albidum* (Nutt.) Nees), persimmon (*Diospyros virginiana* L.), sweetgum (*Liquidambar styraciflua* L.), bur oak (*Quercus macrocarpa* Michx.), scarlet oak (*Quercus coccinea* Münchh), and black walnut (*Juglans nigra* L.) (Burns and Honkala 1990, Woudenberg et al. 2010).

CONCLUSIONS

Our modeling summarizes the effects of EAB at the coarse scale of FIA inventory units and the broad category of forest-type groups. Since we analyzed changes by forest-type group, it is important to consider geographic differences in the composition of each forest-type group. For instance, E-A-C in northern Minnesota inventory units is composed mainly of green and black ash, whereas ash might make up a larger portion of O-H than E-A-C in other states and does not make up a substantial amount of any forest-type group in Maine inventory units. Despite our assumption that EAB will cause 100 percent ash mortality, our results suggest the transition to other species may not be rapid. Ash in previously ash-dominated forests may be replaced by a variety of species and future forests may contain less saplings and trees but more volume on less land. Due to the slow transition, time still exists for the forest products industry reliant on ash to shift to other species. Although there does not appear to be any effective broad-scale treatment to mitigate the effects of EAB, on a smaller scale, private landowners can protect individual trees with chemical treatments (McCullough et al. 2012, Rebek et al. 2008). Our results suggest the impact of EAB-caused ash mortality in non-urban forests measured by FIA may only cause minor forest-type group changes because associated species not prone to EAB infestation have the potential to offset the loss of ash. However, EAB-killed ash could contribute to canopy gaps which facilitate an increase in native and nonnative invasive plant species (Gandhi and Herms 2010a). In addition, our results may not hold true for urban areas not measured by FIA, where there could be much more of an impact due to the extensive distribution of urban ash.

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ENGLISH EQUIVALENTS

1 centimeter (cm) = 0.394 inches

1 kilometer (km) = 0.621 miles

1 hectare (ha) = 2.471 acres

1 cubic meter (m^3) = 35.315 cubic feet

1 metric ton = 1.102 short tons

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The nonnative invasive emerald ash borer (*Agrilus planipennis* Fairmaire; EAB) has caused considerable damage to the ash (*Fraxinus* spp.) resource in North America. While there are methods to mitigate, contain, control, or even eradicate some nonnative invasive insects, EAB continues to spread across North America. Considering strong evidence suggesting >99 percent probability of host tree mortality, the loss of the North American ash resource is possible. To examine anticipated effects of EAB on tree species composition, we modeled future spatial and temporal changes in forest composition over the next 50 years with and without ash mortality anticipated from EAB spread. We used U.S. Forest Service Forest Inventory and Analysis (FIA) data, the current extent of EAB in the United States and Canada, estimated spread rate and host mortality data, and a suite of human population, energy, consumption, land use, and economic models to project the future condition of forests in the Midwest and Northeast United States. Our results suggest that in most cases EAB will not have a substantial effect on ecosystem function of future forests measured by FIA because of the replacement of ash by other species. The transition from ash to other species may take many decades, but forests can eventually recover when a variety of associated species replace ash.

KEY WORDS: vegetation change, emerald ash borer, ash, modeling,
Forest Inventory and Analysis, Northern Forest Futures Project

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