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Asian Longhorned Beetle Eradication Program

Draft Programmatic Environmental Impact Statement—March 2015

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Executive Summary

The U.S. Department of Agriculture, Animal and Plant Health Inspection Service (APHIS) has prepared a draft programmatic environmental impact statement (EIS) for the Asian Longhorned Beetle (ALB) Eradication Program. ALB is a serious insect pest of certain hardwood tree species, with the potential to cause significant economic and environmental impacts if allowed to establish and spread throughout the United States. The Program is a cooperative effort between Federal and State agencies to identify and eradicate ALB infestations in the United States. To date, there have been ALB outbreaks in five States including Illinois, Massachusetts, New Jersey, New York, and Ohio. This draft ALB EIS considers potential environmental impacts from each of the alternatives proposed for the APHIS ALB Eradication Program, should ALB be discovered elsewhere in the continental United States. APHIS can tier subsequent site-specific environmental assessments (EAs) to this EIS, incorporating, by reference, analyses included in this document, thus reducing response time for APHIS to act on new detections. In addition, this EIS will provide the interested public with a programmatic analysis of the potential for environmental impacts from the alternatives available to APHIS to eradicate ALB from the United States.

On August 16, 2013, APHIS published a notice in the Federal Register (FR) describing its intent to prepare a programmatic EIS for the ALB Eradication Program (78 FR 50022–23, August 16, 2013). The public was invited to comment on the proposed EIS; APHIS received 27 comment letters during the 45-day scoping period. Comments were received from the public, non-governmental organizations, and State agencies regarding different aspects of the ALB Eradication Program. APHIS has addressed the comments to the extent possible in this draft EIS.

Five alternatives were evaluated in this draft EIS. The analysis is a general assessment of the alternatives, and their potential impacts to human health and the environment. The five alternatives include:

(1) No Action—APHIS would implement quarantine restrictions but no eradication program;

(2) Removal of Infested Trees—APHIS would implement quarantine restrictions and remove only ALB-infested trees;

(3) Full Host Removal—APHIS would implement quarantine restrictions and remove only ALB-infested trees, and all high-risk host trees up to a ½-mile radius of infested trees;

(4) Insecticide Treatment—APHIS would implement quarantine restrictions, remove all infested trees, and chemically treat all high-risk host trees with an insecticide up to a ½-mile radius of infested trees; and

(5) An Integrated Approach—APHIS would implement quarantine restrictions, remove infested trees, and use a combination of removal and insecticide treatments of high-risk host trees (preferred alternative).

The potential impacts from the implementation of the five alternatives suggest that there could be some effects to the human environment. The largest impacts, both economic and environmental, are expected to occur under the no action alternative, which would effectively allow ALB to become established and spread throughout the United States. Economic impacts related to the establishment of ALB in the United States in forested areas include a decrease in revenues related to various timber markets, maple syrup production, and tourism, where ALB-host trees are a primary component of impacted forests. Economic losses related to ALB establishment in urban areas include tree replacement costs and impacts to property value. Environmental impacts are anticipated for terrestrial and aquatic wildlife that are dependent upon ALB-host trees. The extent of the impacts will vary based on the prevalence of ALB-host trees in forests, and whether affected species depend on ALB-host trees to meet critical periods in their life history. The implementation of quarantine, in association with an eradication program, will reduce these impacts to varying degrees, depending on which eradication approach is used.

The potential impacts from the proposed alternatives, and applicable environmental laws and statutes, are discussed on a programmatic basis in this draft EIS. No site-specific eradication projects will be implemented as a direct result of the decision that will follow this EIS. The decision to implement any treatment project will be made after site-specific EAs are conducted and documented, in accordance with the implementing procedures of the National Environmental Policy Act of 1969 (NEPA). Site-specific EAs will evaluate similar topics as there may be changes over time in the available data regarding this analysis, as well as applicable laws and statutes.

Selection of the preferred alternative allows the Program to implement a proven eradication program that has been successful in other ALB eradication efforts in the United States. The preferred alternative integrates survey and quarantine with the removal of infested trees, and site-specific management of high-risk host trees, thus allowing the Program the greatest flexibility in responding to ALB outbreaks and achieving eradication.

I. Purpose of and Need for Action

The Asian longhorned beetle (*Anoplophora glabripennis* (Motchulsky)) (ALB) is a foreign wood-boring beetle that threatens a wide variety of hardwood trees in North America. The introduction of ALB into the United States was likely from infested wood pallets, and other wood packaging material (WPM), accompanying cargo shipments from Asia. The purpose of the proposed action is to protect the forests and trees of the United States from the adverse effects of ALB.

Why is there a need to eradicate this pest?

There is a need to eradicate ALB wherever it occurs because it is potentially one of the most destructive and costly invasive species to enter the United States. The beetle bores through the tissues that carry water and nutrients throughout the tree, causing the tree to weaken and eventually die. Symptoms occur approximately 3 to 4 years after infestation, and tree death can occur in 10 to 15 years, depending on site conditions. Infested host trees do not recover and regenerate (APHIS, 2009). Tree mortality caused by ALB has been noted in countries where the beetle is endemic, and where it has been introduced (Haack et al., 2010). In the United States, foresters have observed ALB-related tree mortality in New York, New Jersey, Massachusetts, Ohio, and Illinois.

The insect threatens urban and suburban shade trees, and recreational and forest resources valued at hundreds of billions of dollars (Nowak et al., 2001). In addition, ALB is likely to have negative impacts on forest-dependent terrestrial and aquatic species, including threatened and endangered (T&E) species; soil and water quality could also be significantly impacted in forested areas where ALB-host trees are dominant (APHIS, 2009).

Who has authority to act?

The U.S. Department of Agriculture (USDA), Animal and Plant Health Inspection Service (APHIS) has a broad mission area that includes protecting and promoting U.S. agricultural health, and protecting and promoting food, agriculture, natural resources, and related issues. Specifically, the Plant Protection Act of 2000 (7 United States Code (U.S.C.) 7701 et seq.) provides the authority for APHIS to take actions to exclude, eradicate, and control plant pests, including ALB. Under this authority, APHIS works to prevent new infestations of ALB from entering the United States by regulating WPM, by restricting movement of items that may be infested with ALB (known as regulated articles) from areas under quarantine for ALB, and by conducting programs to eradicate ALB where it is found in the United States.

This programmatic environmental impact statement (EIS) discloses the different methods and alternatives that APHIS can use to eradicate ALB from areas it occurs in the contiguous United States.

Why do this environmental impact statement?

As a Federal Government agency subject to compliance with the National Environmental Policy Act of 1969 (NEPA) (42 U.S.C. 4321–4347), APHIS prepared this EIS in accordance with the applicable implementing and administrative regulations (40 Code of Federal Regulations (CFR) §§ 1500–1508; 7 CFR §§1b, 2.22(a)(8), 2.80(a)(30), 372). This programmatic EIS presents program alternatives APHIS could adopt as part of the ALB Eradication Program, and examines the potential consequences of implementing them.

APHIS has prepared 14 site-specific environmental assessments (EAs), since 1996, for ALB eradication programs and research studies in Illinois, New York, New Jersey, Massachusetts, and Ohio (appendix B). This EIS will consider potential environmental impacts from the APHIS ALB Eradication Program should ALB be discovered elsewhere in the contiguous United States. APHIS can tier subsequent site-specific EAs to this EIS, incorporating by reference analyses included in this document, thus reducing response time for APHIS to act on new detections, should these occur. In addition, this EIS will provide the interested public with a programmatic analysis of the potential for environmental impacts from the alternatives available to APHIS to eradicate ALB from the United States.

A. Background

1. Description of ALB

a. Life Cycle

ALB is in the wood-boring beetle family Cerambycidae. Adults are 1 to 1½ inches in length with long antennae, and are shiny black with small white markings on the body and antennae (figure 1–1).

Adult females chew depressions into the bark of various hard-wood tree species in which they lay (oviposit) their eggs. Eggs hatch within 2 weeks of oviposition (Haack et al., 1997; EPPO, 1999). After hatching, small white larvae bore into the tree, feeding on the vascular layer which transports nutrients and water throughout the tree. Larvae molt and can go through as many as 13 instar stages (Keena, 2008). The larvae continue to feed deeper into the tree's heartwood, forming tunnels (or galleries) in the trunk and branches.

Figure 1–1. Adult Asian longhorned beetle.

Over the course of a year, a larva will mature and then pupate inside the tunnels formed by the larva within the tree. The pupal stage lasts 13 to 24 days (Lingafelter and Hoebeke, 2002; Haack et al., 2006). From the pupa, an adult beetle emerges, chewing its way out of the tree, forming characteristic round exit holes approximately ⅜ inch in diameter. After emerging from the tree, adults will feed on leaves and bark for 10 to 14 days before mating and laying eggs (Keena, 2002; Smith et al., 2002). Adults tend to remain on the tree from which they emerge, resulting in infestation by future generations (EPPO, 1999). Adult activity is usually observed from May to July, although adults have been observed from April to December (Haack et al., 2010).

The damage from larvae burrowing into the tree and adults burrowing out of the tree cuts off nutrient flow and weaken the tree. The tree will eventually die if the infestation is severe enough. Sawdust-like debris and insect waste (called frass) from the insect's burrowing activity may be found at the base of infested trees. Infested trees are also prone to secondary attack by disease and other insects.

In most locations, ALB produces one generation per year; however, in other countries where the pest is established, the number of annual generations varies with climate and latitude. For example, in northern China, one generation takes 2 years to develop (EPPO, 1999; Lingafelter and Hoebeke, 2002). Because ALB can overwinter in multiple life stages (egg, larval, and possibly pupal stages), adult emergence is staggered over time, resulting in adults emerging, feeding, mating, and laying eggs (ovipositing) throughout the summer and fall (Haack et al., 2006).

b. Hosts

There are 13 genera of host trees that APHIS regulates for ALB and are considered high-risk hosts: *Acer* (maple and box elder), *Aesculus* (horse chestnut and buckeye), *Salix* (willow), *Ulmus* (elm), *Betula* (birch), *Albizia* (mimosa), *Celtis* (hackberry), *Cercidiphyllum* (katsura tree), *Fraxinus* (ash), *Koelreuteria* (golden raintree), *Platanus* (sycamore and London planetree), *Sorbus* (mountain ash), and *Populus* (poplar) (APHIS, 2009), 7 CFR 301.51–2). These trees are hosts because the ALB can derive its food supply and complete its life cycle. A host tree is still a host even if it is not infested. *Acer* is the most commonly infested tree genus in the United States, followed by *Ulmus* and *Salix* (Haack et al., 2010). *Acer*, *Aesculus*, *Salix*, *Ulmus*, and *Betula* are considered preferred ALB-hosts (APHIS, 2008). Other regulated genera are considered rare or occasional hosts, including *Albizia*, *Celtis*, *Cercidiphyllum*, *Fraxinus*, *Koelreuteria*, *Plantanus*, *Sorbus*, and *Populus* (APHIS, 2008) (see appendix C). APHIS expects to remove *Celtis* from the list of regulated host trees based on observation that this is not a host tree. A recent study indicates that ALB egg laying (oviposition) and reproductive success are higher on red maple (*Acer rubrum*), compared to Norway maple (*A. platanoides*) or sugar maple (*A. saccharum*) (Dodds et al., 2014). (See appendix C for an annotated list of ALB hosts.)

c. Dispersal and Spread

(1) Human-Mediated Spread

ALB entered the United States, most likely, in WPM from Asia. USDA–APHIS regulations in 7 CFR §319.40–1 define WPM as “wood or wood products (excluding paper products) used in supporting, protecting, or carrying a commodity (includes dunnage).” WPM (e.g., pallets, crates, etc.) is made from low-grade lumber with higher moisture content than seasoned wood or heat-treated wood. Low-grade lumber may be of low quality due to pest damage (Cavey, 1998). (Bush et al., 2002) reported that hardwood species accounted for about two-thirds of the total wood used for pallets during the 1990s. The National Plant Protection Organizations (NPPO) recognize WPM as a pathway for pest movement between countries.

In response to the threat posed by untreated WPM, the Commission on Phytosanitary Measures of the International Plant Protection Convention (IPPC) of 2002 adopted International Standards for Phytosanitary Measures No. 15 (ISPM 15) titled “Guidelines for Regulating Wood Packaging Material in International Trade” (FAO, 2002). ISPM 15 prescribes either heat treatment or fumigation for all WPM to mitigate risk of pests. WPM subjected to these approved measures is required to display a mark verifying compliance. Since September 16, 2005, the United States began full enforcement of ISPM 15, requiring either fumigation or heat treatment for all WPM entering the United States from any country; however, compliance with ISPM 15 is not required for WPM used domestically. WPM made from infested wood and used in domestic shipments would continue to be a potential source of spread.

Once ALB is present in the United States, it can move through domestic pathways. For example, movement of recently felled trees could carry ALB, and potentially spread it a long distance from the infested site. The level of risk depends on the end use of felled trees, that is, chipping for mulch, disposal in landfills, incineration, and so on. Although ALB prefers to infest branches in the tree crown rather than the trunk, lumber and saw logs are, nonetheless, a potentially important pathway to spread ALB.

The production and distribution of fuel wood (this category includes firewood) also could transport infested material long distances. Moreover, many people use fallen limbs or felled trees from their private property as firewood, sometimes transporting and using these on vacations or in other homes. While educational programs and quarantine regulations in affected areas seek to prevent this kind of spread, this transport may still occur. Transport of infested stumps and limbs for disposal are another potentially important ALB pathway, although the level of risk depends on how municipal solid waste is treated in the infested area. ALB can also be transported through the movement of infested nursery stock.

Establishment of quarantines and quarantine regulations, which restrict the movement of woody material, reduce the likelihood of human-mediated spread of ALB. In quarantine areas in the United States, businesses or individuals working with trees or related products (e.g., firewood) must be under a compliance agreement with APHIS. These include landscapers, tree pruning and tree removal companies, firewood dealers, pallet distributors, nurseries, and sanitation workers, as well as other municipal or community services and associated contractors. The ALB Eradication Program provides a debris disposal site for use by companies under compliance agreement, and by municipalities within the quarantine area.

Private residents have to secure the services of a company under compliance to remove host material from their property. All host material for disposal, regardless of infestation, must be chipped prior to leaving the quarantine area. Chipping infested trees is effective in destroying ALB (Wang et al., 2000). Nursery stock in the quarantine area is subject to inspection. Uninfested host material in the nursery trade may leave the quarantine area if accompanied by a certificate of inspection and approved permits. Any infested host material found in the nursery trade must be chipped.

(2) Natural Spread

Adult beetles are prompted to move when the density of beetles in a given area reaches high levels (Williams et al., 2004), or when they are in search of a mate (Bancroft and Smith, 2005). ALB is a weak flier therefore spread through natural means (self-motility) is a minor contributor to spread potential (Bancroft and Smith, 2005).

Various studies have examined the distance ALB travels per day. Smith et al. (2001) found, on average, that adult females and males move 23 meters/day (m/day) and 17 m/day, respectively. Dispersal rates may be affected by the age of the beetle, population density, temperature, proximity of host trees, and other factors. In a mark-recapture study, Smith et al. (2004) found the median daily dispersal rate for both sexes was 30 m/day (range of 11 to 56 m). Williams et al. (2004) tracked adult beetles using harmonic radar, and found movement rates were 1.9 m/day for females and 3.7 m/day for males. Most beetles tended to move in one direction and rarely backtracked. In a mark-recapture study, Smith et al. (2004) found the median daily dispersal rate for both sexes was 30 m/day (range of 11 to 56 m). Williams et al. (2004) tracked adult beetles using harmonic radar, and found movement rates were 1.9 m/day for females and 3.7 m/day for males. Most beetles tended to move in one direction and rarely backtracked. Bancroft and Smith (2005) estimated a probability of 62 percent that beetles move from the tree on which they were last recorded to a tree nearby. They characterized daily movement at 20 m/day.

Smith et al. (2001) measured a maximum ALB dispersal distance (including that of female beetles carrying mature eggs) of 1,442 m. In another mark-recapture study, Smith et al. predicted that "98% of beetles were recaptured at distances <920 m from the release point." Beetles were recaptured at the most outlying sampling points (1,000 to 1,080 m from release point), and even at distances up to 2,600 m from the release point (Smith et al., 2004). Of the gravid (carrying eggs) females, 86 percent were recaptured within 1,080 m, and 77 percent were recaptured within 600 m. Eight gravid females were recaptured at distances greater than 2,000 m (2 kilometers (km) (Smith et al., 2004).

d. Geographic Distribution: Current and Projected

ALB is native to China and Korea (Cavey, 1998; Lingafelter and Hoebeke, 2002), but it was not considered a pest in natural forests there (Hajek, 2010). The distribution and abundance of ALB increased dramatically throughout China because in the 1960s and 1970s the Chinese government supported reforestation programs using fast-growing poplar trees, many of which were susceptible to ALB (Cao et al., 2010; Hajek, 2010). Millions of ALB-infested trees have been cut in China over the past decades, and it is likely that this wood was used as WPM (Haack et al., 2010). Because of the increase in trade with Asia that involved WPM, ALB was introduced to Austria, Belgium, Canada, France, Germany, Italy, Japan, the Netherlands, Switzerland, United Kingdom, and the United States (Carter et al., 2010; Haack et al., 2010; EPPO, 2014). Belgium and Japan have eradicated the insect (EPPO, 2014).

Peterson et al. (2004) predicted the potential distribution of ALB in North America based on the climatic factors in its native distribution. Their models suggest, ". . . the species has the potential to invade much of eastern North America, but only limited areas in western North America, and that a focus of initiation of invasion is likely to lie in the area south of the Great Lakes."

Haack et al. (1997) overlaid the distribution of ALB in China (21 °N to 43 °N) onto North America, and determined that the range coincides with the span from southern Mexico to the Great Lakes. Keena (2006) noted that summer temperatures, throughout most of the lower 48 States, should support beetle survival and reproduction. Based on preferred host availability, Nowak et al. (2001) estimated the percentage of trees and number of trees at risk in nine U.S. cities, including one city in the Southeast and one city in the West. They estimated that 20 percent (or 1.8 million) trees in Atlanta, Georgia were at risk of infestation, and in Oakland, California, 12 percent (or 0.19 million) trees were at risk. They also estimated that cities in the Northeast, such as Boston, Massachusetts and Syracuse, New York may lose as much as 60 percent of their trees should ALB be introduced and allowed to spread. All available studies indicate that ALB is able to survive and reproduce in any location within the lower 48 States that has host trees (APHIS, 2009). (See figure 2–1 for the ALB susceptibility potential for *Acer* (species of maple) in the United States.)

2. Economic Impacts of ALB in the United States

The spread of ALB beyond the current isolated outbreaks could have severe economic effects. Although it is difficult to estimate the extent of expected impacts over time, it is clear that several industries in the Northeastern United States would be affected if ALB were to spread beyond the current areas of infestation to other urban and forested areas. These impacts would include not only the loss of urban trees at an estimated cumulative cost of at least $948 million in cities in the Northeastern United States, but also losses in several other industries, as well as non-market economic losses (Nowak et al., 2001).

The timber industry produces over $178 million of products from host species annually in the Northeastern United States alone. Maple syrup production was worth nearly $67 million in 2010. The tourism industries in nine of the Northeastern States reported revenues around $143 billion in 2012. In addition, sales of deciduous shade trees were estimated at an additional $56 million for eight of the Northeastern States (including Ohio). The annual production values at risk in these three industries alone are significantly larger than the projected annual budget of the ALB Eradication Program.

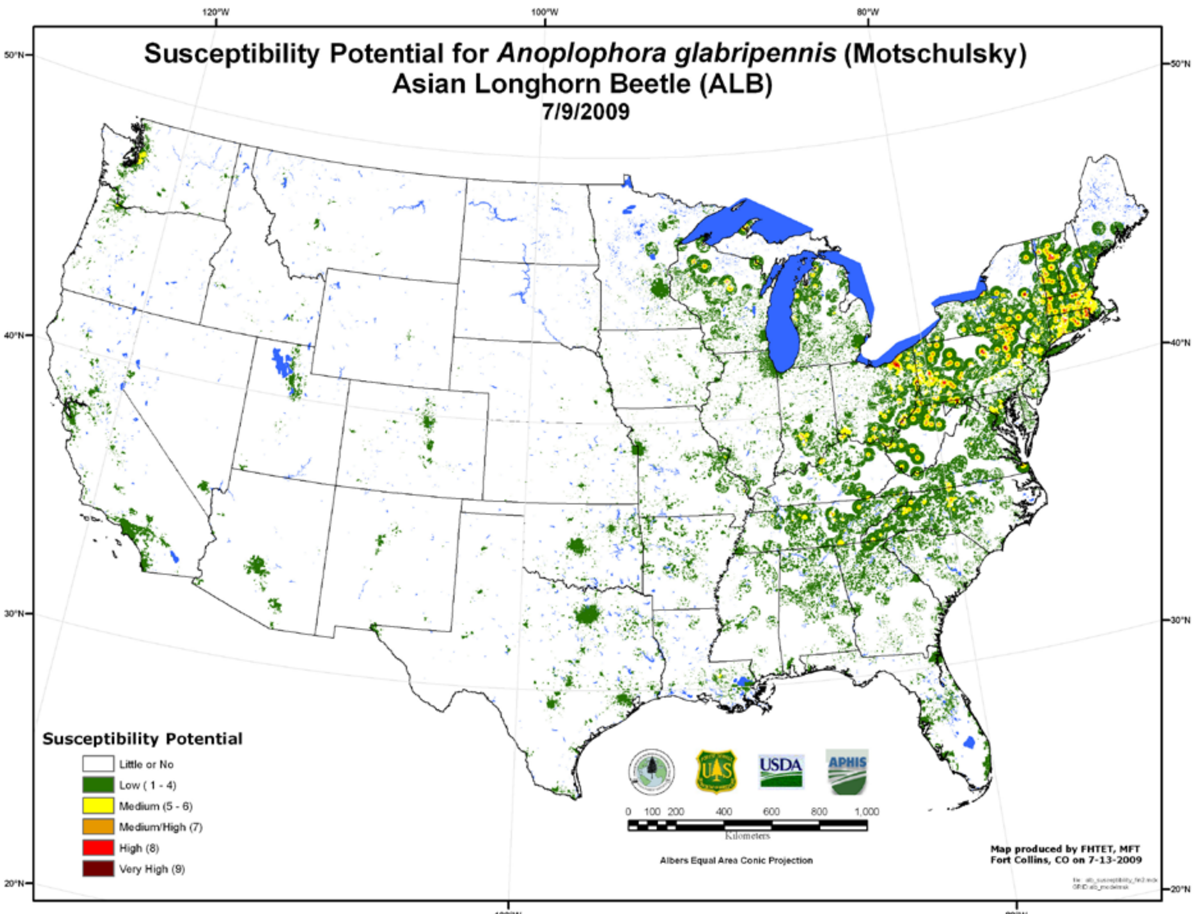

Figure 1–2. Susceptibility potential for the Asian longhorned beetle in the United States based on the basal area for *Acer* (maple). Source: (FS, 2009a).

While the expected economic losses in forest-related products in the Northeastern United States and Ohio are large, previous research indicates non-market economic losses due to forest-related pests are expected to be larger than any market losses. This loss of host trees would have economic impacts in urban, suburban, and forested environments due to the loss of aesthetic and ecosystem benefits. Previous studies have shown that the presence of healthy trees increases property values by 2 to 6 percent (Morales et al., 1976; Anderson and Cordell, 1988; Dombrow and Sirmans, 2000). In addition, the loss of host species would likely decrease fall foliage tourism in the Northeastern United States, which relies on a number of host species for the vibrant fall color displays. Many communities in the Northeastern States and the State of Ohio generate significant economic activity from fall foliage tourism, and would be unlikely to recover from the loss of so many of the species that provide fall colors. Other non-market impacts would be likely due to the loss of recreational benefits and ecosystem service provisions. The widespread establishment of ALB would also likely affect U.S. trade in green lumber, and live trees and shrubs if importing countries establish quarantines or other restrictions for ALB.

Although a complete cost-benefit analysis of the current Program is outside the scope of this EIS, even a relatively modest annual decrease in only two forest-related industries would result in economic losses more than sufficient to justify the anticipated costs of the ALB Eradication Program. The market losses in the timber and maple syrup industries do not include any impacts related to nursery stock or non-market losses related to urban trees and other aesthetic and ecosystem impacts.

Considering that non-market losses are likely to be greater than any market losses, the benefits of an eradication program would outweigh program costs with a relatively small annual loss in the markets for forest products (APHIS, 2009). Further research to quantify these non-market benefits of eradication would help to strengthen this conclusion.

3. Existing and Projected Ecological Impact in the United States and Elsewhere

Widespread establishment of ALB in the United States will cause significant ecological damage, affecting terrestrial and aquatic fauna and flora. Multiple factors dictate the extent of damage and its impact on ecological community function. In cases where the host trees are not a dominant component of the forest stand, the impacts may be less; however, in locations where host trees occur in high density and are keystone species within that particular forest type, the impacts are almost certain to be significant. The loss of keystone tree species and the associated cascading impacts to ecosystem function have been documented in other forest types where invasive forest insect pests were introduced (Ellison et al., 2005). The spatial and temporal factors related to an ALB infestation also will dictate the magnitude of impacts to ecosystems, particularly for species that are dependent on these habitats at critical periods in their life history.

History of APHIS Eradication Actions and Other Regulatory Actions Against ALB

The first detection of ALB in the United States was in Brooklyn, New York, in 1996, although the actual date of introduction is much earlier. Shortly after, another infestation was detected in Amityville on Long Island. Since then infestations have been found in Islip on Long Island, in Queens, in Manhattan and on Staten Island, including Pralls Island. Most recently, an infestation was detected in portions of Babylon Township on Long Island.

Additional outbreaks occurred in Chicago, Illinois in 1998, and in Middlesex and Union Counties in New Jersey in 2002 (APHIS, 2007a).

In August 2008, ALB was found in Worcester County, Massachusetts and in July 2010, six ALB-infested trees were discovered in Suffolk County in Massachusetts. ALB was discovered in Clermont County, Ohio in June 2011.

In response to the introduction of ALB in these areas of the United States, APHIS implemented an adaptive management eradication program. Currently, the ALB eradication program in the United States uses—

- visual surveys from the ground or aerial surveys by tree climbers or bucket trucks,
- quarantine areas (1½ mile radius from infested trees) to restrict movement of regulated materials,
- application of a preventative/protective systemic insecticide to un-infested trees,
- removal of infested and high-risk host trees within the quarantine area, and
- public involvement, outreach, and education about ALB.

APHIS and its partners declared ALB eradicated from the following areas:

- Chicago, Illinois and Hudson County, New Jersey, in 2008
- Islip (Suffolk County, New York) in 2011
- Manhattan (New York County, New York) and Staten Island (Richmond County, New York) in 2013

- Middlesex and Union Counties, New Jersey in 2013

- Boston (Suffolk County, Massachusetts) in 2014

Active eradication programs are continuing in Queens, Brooklyn and Suffolk and Nassau Counties, New York as well as in Worcester County, Massachusetts, and Clermont County, Ohio.

B. Public Involvement

1. Prior EAs Published by the Agency and Results from Public Comments

APHIS has prepared 14 site-specific EAs regarding ALB eradication programs or research in New York, Illinois, New Jersey, Massachusetts, and Ohio, available for public comment since 1996 (appendix B).

2. Past Agency Outreach Efforts on ALB Issues

APHIS provides many opportunities for public involvement and outreach regarding program activities in ALB-quarantined areas. As such, APHIS—

- provided media interviews for newspapers, and radio and television outlets;
- issued press releases;
- conducted an annual advertising awareness campaign;
- provided public service announcements on radio and television stations;
- had a presence at industry shows, expos, and various outreach venues;
- secured multiple airings on public television cable access stations across the United States of the "Lurking in the Trees," ALB documentary, APHIS produced in conjunction with The Nature Conservancy, and made the documentary available on YouTube and iTunes;
- posted information on social media including Facebook, Twitter, Pinterest and Flickr;
- held public meetings as well as meetings with Federal and State officials, town administrators, and other impacted groups and persons;
- provided informational materials and Web sites to the public, including an online reporting function and the arrangement of a national-use ALB hotline telephone number.

3. Locations of Addressed Issues Raised by Stakeholders

Scoping is an open and early process to determine the issues to be addressed in an EIS, and to identify significant issues related to the proposed action covered in the EIS. As part of this process, APHIS sent out letters to all federally recognized tribal nations in the contiguous United States, inviting them to participate in a conference call on July 10, 2013. During this conference call, APHIS asked tribal nations to provide input on how ALB and its eradication could affect them, and also provided opportunity for them to ask questions about the ALB Eradication Program. Approximately 20 tribes participated in the conference call.

On August 16, 2013, APHIS published a notice of intent (NOI) in the FR describing its intent to prepare a programmatic EIS for the ALB Eradication Program (78 FR 50022–23, August 16, 2013). The public was invited to comment on the proposed EIS.

In the NOI, APHIS identified the following resources requiring further examination, in this EIS, of the potential environmental effects:

- wildlife, including consideration of migratory bird species and changes in native wildlife habitat and populations;
- federally listed T&E species;
- soil, air, and water quality;
- forests and trees in residential areas, and impacts on property values;
- wood products industry and other economic impacts, including impacts on the firewood industry;
- human health and safety; and
- cultural and historic resources.

APHIS made available a press release regarding the NOI to media contacts in New York, Massachusetts, and Ohio, and through the APHIS Stakeholder Registry that contains almost 12,000 contacts, and in the ALB e-newsletter on August 29, 2013. In addition, APHIS conducted the following notification activities:

- ALB project managers in New York, Massachusetts, and Ohio shared the NOI with their staff members, and any key contacts that may have an interest in an ALB EIS;
- Facebook post(s) on the ALB Facebook account located at https://www.facebook.com/asianlonghornbeetle;

- Twitter post(s) on the ALB Twitter account located at https://twitter.com/StopALB;

- posted on the AsianLonghornedBeetle.com homepage and each State page;

- posted on the APHIS ALB News and Information page;

- Massachusetts State Department of Agricultural Resources and Department of Conservation and Recreation partners were notified and asked to share the NOI;

- Ohio Department of Agriculture, Ohio Department of Natural Resources, Ohio State University, and Clermont County communication partners were notified and asked to share the NOI;

- notification to tribal contacts;

- notification to USDA Forest Service (FS) contacts;

- notification to U.S. Fish and Wildlife Service (FWS) contacts;

- notification to various partners and organizations, such as the Tree Care Industry Association, Nature Conservancy, Arbor Day Foundation, American Forest Foundation, various arboretums, Sierra Club, Society of American Foresters, and so on;

- APHIS–PPQ State Plant Health Directors in New York, Ohio, and Massachusetts shared the press release with FS, FWS, USDA–Agricultural Research Service (ARS), USDA–Natural Resource Conservation Service (NRCS), State agriculture/forestry/natural resource/heritage, and industry contacts.

Although the comment period was to end on September 16, 2013, APHIS extended it to September 30, 2013. APHIS received 27 comment letters during the 45-day scoping period. All comments were fully considered by APHIS in the planning of this EIS. Issues and concerns identified by the public and tribal contacts included:

- impacts from heavy equipment use and other property damage from program activities;

- impacts to soil and water quality;

- impacts to maple syrup producers;

- impacts to federally owned lands, such as national forests and national parks, State-owned nature preserves, natural heritage areas, and other high-value forest landscapes;
- use of adaptive management as new studies or best management practices (BMPs) are identified;
- insecticide pollution;
- impacts to T&E species, migratory birds, and their habitats;
- potential for ALB infestations through imported wood products;
- biological control of ALB;
- consideration of the Coastal Zone Management Act (CZMA) of 1972;
- potential for yard waste and brush to spread ALB in the United States;
- risk of ALB to all of North America if ALB is allowed to spread; and
- replacement of removed trees.

Many comments did not raise specific issues for analysis in this EIS; however, opinions were provided for or against the selection of certain program control methods.

APHIS and its cooperators recognize the public's concern about the potential impacts of ALB and program activities on human health, biological resources, and the physical environment. These concerns will be addressed as part of this EIS and will be made available for further public input.

C. Decision Framework

In the earliest EAs for ALB eradications in Illinois and New York, alternatives available to APHIS included quarantine only, suppression, and eradication. Although eradication was the preferred alternative, no chemical methods of control were available at the time. Only host tree removal followed by burning or chipping of removed wood material was available. After 2000, APHIS added a prophylactic treatment using the insecticide imidacloprid as an option to the ALB Eradication Program after tests proved it effective in protecting trees from ALB under certain conditions. However, imidacloprid treatments do not ensure complete control of ALB within a tree due to variability in treatments, weather conditions, and tree health, all of which can result in uneven distribution of imidacloprid within a tree.

In the most recent EA prepared for Clermont County, Ohio, in May 2013 (APHIS, 2013b), APHIS analyzed four alternatives: (1) no action by APHIS; (2) removal of infested trees and high-risk host trees up to ½ mile from infested trees (full host removal); (3) removal of infested trees and imidacloprid treatment of high-risk host trees up to ½ mile from infested trees; and (4) infested host removal and combination of removal or imidacloprid treatment of high-risk hosts (preferred alternative). Alternatives 2 through 4 were eradication program options.

Listed below are five alternatives for further examination in this EIS. (Chapter 2 describes the alternatives in greater detail.)

1. No Action

Under the no action alternative, APHIS would not undertake eradication efforts. However, APHIS would continue to implement quarantine restrictions in the event of a confirmed ALB detection. This alternative represents the baseline against which a proposed action may be compared.

2. Removal of Infested Trees

Under this alternative, APHIS would implement quarantine restrictions, and would only remove trees infested with ALB. High-risk host trees would not be removed or treated.

3. Full Host Removal

Under this alternative, APHIS would implement quarantine restrictions, remove infested trees, and would remove high-risk host trees up to ½ mile from infested trees.

4. Insecticide Treatment

Under this alternative, APHIS would implement quarantine restrictions, remove infested trees, and treat high-risk host trees with an insecticide up to ½ mile from infested trees.

5. Integrated Approach (Preferred Alternative)

Under this alternative, APHIS would implement quarantine restrictions, would remove infested trees, and would use a combination of removal and insecticide treatments of high-risk host trees (preferred alternative).

APHIS will not implement site-specific eradication projects as a direct result of the decision that will follow this EIS. Rather, APHIS will conduct site-specific EAs before the agency decides to implement any treatment project. EAs will address unique local issues, beyond the scope of this document, for site-specific management projects for ALB. Site-specific EAs are more detailed and precise as to geographical locations and strategies appropriate for the type of outbreak.

The decision on this EIS will serve as the primary guide for management of ALB in the contiguous United States. Treatments and strategies allowed by prior EA decisions will continue to be available for use. The decision whether to plan and implement an ALB project in the United States will occur on a case-by-case basis by APHIS.

D. Scope of this Document and NEPA Requirements

This EIS concerns only the APHIS ALB Eradication Program carried out by APHIS, directly or in conjunction with others (States, other Federal agencies, and tribal governments). The information and analysis contained in this EIS can be incorporated by reference into EAs and any other environmental documents prepared for ALB eradication projects, in accordance with NEPA. Some ALB-related activities, such as WPM importation regulations, inspection of WPM, and other ALB-regulated articles at the point of entry in the United States, and research and methods development activities are outside the scope of this document and were not examined. (For more information regarding environmental impacts of importation of WPM, see the APHIS EIS for Importation of WPM (APHIS, 2003), and the 2007 supplemental EIS (APHIS, 2007b)).

E. Consultations

Section 7 of the Endangered Species Act (ESA) and its implementing regulations require Federal agencies to ensure their actions are not likely to jeopardize the continued existence of T&E species, or result in the destruction or adverse modification of critical habitat. If necessary, APHIS conducts Section 7 consultation with the FWS and National Marine Fisheries Service (NMFS) on a site-specific basis for ALB eradication activities.

APHIS considers whether critical habitat or listed species are present in the program area. If none are present, no Section 7 consultation is required. For the ALB Eradication Program in Worcester County, Massachusetts, APHIS consulted informally with FWS on a threatened plant, the small whorled pogonia (*Isotria medeoloides*), in 2008. In 2011, APHIS consulted informally with FWS on the impact of the ALB Eradication Program on the small whorled pogonia and piping plover (*Charadrius melodus*) in Norfolk and Suffolk Counties in Massachusetts.

In June, 2011, APHIS first contacted FWS in Columbus, Ohio for technical assistance regarding impacts to federally listed species in Clermont County, Ohio. Seven endangered species (Indiana bat, *Myotis sodalis*; running buffalo clover, *Trifolium stoloniferum*; fanshell, *Cyprogenia stegaria*; rayed bean, *Villosa fabalis*; snuffbox, *Epioblasma triquetra*; pink mucket pearlymussel, *Lampsilis abrupta*; and sheepnose, *Plethobasus cyphyus*) occur in Clermont County.

Since that time, APHIS submitted biological assessments and consulted with FWS on those species, receiving concurrence with "may affect, not likely to adversely affect" determinations, with the implementation of protection measures. In addition, FWS personnel have made site visits to the infested area, and have provided Indiana bat training to APHIS and the Ohio Department of Agriculture personnel. In addition, APHIS conducted surveys for Indiana bats in the Clermont County eradication work zones and reported those findings to FWS. Most recently, APHIS entered into a formal consultation with FWS on the Indiana bat (receiving a biological opinion dated June 4, 2014) and a conference on the northern long-eared bat (*Myotis septentrionalis*), a species proposed for listing as endangered (receiving a conference opinion dated July 3, 2014). In the near future, APHIS has initiated a conference for the northern long-eared bat in New York and Massachusetts.

APHIS will conduct ESA Section 7 consultations with the appropriate agency, as necessary, for any eradication programs if ALB is detected in new locations in the United States. Consultation with FWS, and NMFS, if necessary, at the local level will ensure that ALB eradication actions will not jeopardize the continued existence of a listed species or adversely modify critical habitat in the program area. APHIS will ensure the implementation of any protection measures for T&E species or critical habitat that result from such consultations. In addition, APHIS will ensure that site-specific consultations will be done, as necessary, under the National Historic Preservation Act, Migratory Bird Treaty Act, Bald and Golden Eagle Protection Act, and any other laws, regulations, Executive orders, and agency policies that apply to site-specific projects.

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II. Alternatives

This EIS analyzes the potential environmental consequences associated with the alternative options to eradicate ALB from areas in the contiguous United States where the insect establishes. The purpose of the alternatives is to describe the reasonable strategies the agency could take to achieve its goal.

APHIS conducts survey activities and imposes Federal quarantines, according to the agency's authority under the Plant Protection Act. Survey and quarantine are part of each alternative proposed, and are not unique to any one alternative. Therefore, descriptions for these two activities are independent from the descriptions for each alternative.

A. Quarantine

Federal quarantine authority for ALB includes 7 CFR § 301.51 for eradication programs, 7 CFR § 319.40 for solid wood packing material, and 7 CFR § 330 for plant pests. Under these regulations, APHIS establishes quarantines and regulates international and interstate movement of regulated plant host material, also referred to as regulated articles. APHIS cannot regulate intrastate movement without the State Plant Regulatory Agency first establishing an intrastate quarantine. Intrastate quarantine facilitates regulatory activities within a geographical area less than an entire State.

APHIS and State plant regulatory agencies establish quarantine boundaries 1.5 miles from a tree with ALB-exit holes, and 0.5 miles from a tree with egg sites only.

Under quarantine, APHIS restricts the movement of articles, known as regulated articles, which present a risk of spreading ALB interstate from the quarantine area. The regulated articles listed under the quarantine 7 CFR § 301.51, as published in 2013, include the beetle and all its life stages; firewood (all hardwood species, not restricted to ALB-host trees); green lumber, and other living, dead, cut, or fallen material, including nursery stock, logs, stumps, roots, branches, and debris from ALB-host trees of ½ inch or more in diameter.

As of December 2014, 13 genera of trees in the United States are regulated hosts for ALB, and are considered high-risk hosts because of ALB's preference for feeding and completing its life cycle on these genera: *Acer* spp. (maple and box elder), *Aesculus* spp. (horse chestnut and buckeye), *Albizia* spp. (mimosa), *Betula* spp. (birch), *Celtis* spp. (hackberry), *Cercidiphyllum* spp. (katsura tree), *Fraxinus* spp. (ash), *Koelreuteria* spp. (golden rain tree), *Platanus* spp. (sycamore and London planetree), *Populus* spp. (poplar), *Salix* spp. (willow), *Sorbus* spp. (mountain ash), and *Ulmus* spp. (elm). ALB can complete its life cycle in these materials, and transport of these materials could spread ALB to non-infested areas. APHIS plans to remove the genus *Celtis* from the list of regulated hosts based on evidence that ALB is unable to complete its life cycle on species within this genus. (Appendix C provides additional information and references on the list of regulated host trees of ALB.)

Regulated articles originating from within the quarantine area may move outside the quarantine area if accompanied by a certificate or limited permit, unless the USDA moves the articles for scientific or research purposes. Issuance of a certificate or permit verifies that the host material underwent inspection by an APHIS-approved inspector, and is free of ALB. At this time APHIS has no approved regulatory treatments, (treatments applied to ALB-regulated articles), that would allow for the interstate movement of regulated articles, except for heat treatment for firewood. The quarantine does not limit the natural spread of ALB. (A full description of the quarantine is found in 7 CFR 301.51.)

Removal of an area under quarantine occurs upon declaration that the area is free of ALB. APHIS makes this declaration when the final survey finds no ALB infestations after control activities, secondary surveys, and a minimum of 4 years has passed since the last evidence of ALB in an area. The basis of the minimum 4-year survey interval is the beetle's life cycle length (1 to 2 years) and the minimum length of time for the beetle's population to grow to detectable levels should the beetle still be present.

To declare eradication, a final round of negative survey is needed at some time after surveys conducted as part of delimitation and control measures have failed to find ALB in the area. The final time interval selected would depend on the size/area of the original infestation, the prevalence of 2-year ALB life cycles (if any) in the program area, and other programmatic and logistic factors. (Maps of ALB quarantine areas in the United States are available from the Plant Health link at http://www.aphis.usda.gov/wps/portal/aphis/home)

B. Survey

ALB Program inspectors conduct surveys to (1) determine the scope of the infestation; (2) establish the quarantine area; (3) determine that ALB has not spread outside of the established quarantine area and, if it has, to expand the quarantine area; and (4) determine when to release an area from quarantine. The types of surveys conducted in an area depend on the scope of the infestation, and the circumstances surrounding the infestation. Below is a general description of surveys; however, there is flexibility in the survey protocols. For example, the number of times ALB Program inspectors survey an area depends on forest and host composition, the degree of infestation, and the time it takes between survey cycles. The control strategy employed in an area (e.g., removal and/or treatment of all host trees in an area) will affect the number of survey cycles. As new biological information becomes available on ALB and survey strategies evolve, APHIS expects the survey protocols for the ALB eradication program to change. Site-specific EAs for the ALB eradication program will indicate changes to the following survey description.

1. Level 1 Survey (Core Survey)

The discovery of ALB triggers a Level 1 survey, also known as core survey, to determine the scope of the infestation. Before the survey begins, the APHIS National Identification Service confirms the suspect specimen as ALB through structural and/or genetic identification. Once confirmed, Program inspectors survey neighboring host trees for oviposition (egg) sites and exit holes that may indicate an ALB infestation.

During the Level 1 survey, ALB Program inspectors visually look for signs of infestation on every regulated host tree in a circular radius around the infestation, until they fail to find ALB within approximately a ½-mile radius of infested trees. Signs of infestation on host trees include exit holes, oviposition sites, frass, tunneling (formed as larvae and bore into the tree), and sap flow from damaged sites. Inspectors search for signs using binoculars from the ground, and may conduct aerial tree inspections through trained professionals using bucket trucks to peer into trees from above, and by trained tree climbers to search for signs of an infestation within tree canopies. Interest groups and organizations voluntarily assist inspectors by searching trees from the ground using binoculars. Use of tree climbers is the most effective method of detecting signs of ALB; however, this is also a slower and more costly method (Hu et al., 2009). Currently, no method of visual survey, including visual surveys for ALB, is completely effective in pest detection (Bulman et al., 1999).

If inspectors find additional infestations, APHIS extends the ½-mile radius from the outermost find. APHIS bases the ½-mile radius on studies published in the scientific literature, as well as unpublished data on ALB's natural spread potential (Smith et al., 2001; Smith et al., 2004; Williams et al., 2004; Bancroft and Smith, 2005).

Survey areas may include residential, commercial, or public land; access to these properties is necessary for the Program. The Program removes trees infested with ALB.

2. Level 2 Survey (Buffer)

The Level 2 survey is a safeguard to ensure that ALB is not spreading beyond the ½-mile radius around infested host trees established during the Level 1 survey. During the Level 2 survey, also known as a buffer survey, inspectors survey host trees within a minimum of 1 mile beyond the survey boundary set during the Level 1 survey. This results in approximately a 1½-mile radius from the point of infestation for trees with less than 100 exit holes. Level 2 Survey areas should be expanded to a minimum of 2.5 miles from areas that are or were centers of high populations, as denoted by the presence of a cluster of trees with many exit holes or one or more trees with >100 exit holes. This boundary expansion only applies to survey and not regulatory boundaries (the two can differ). APHIS bases the additional 2.5 mile boundary expansion on the high infestation or population level which indicates ALB has likely been in the area for some time as well as the beetle's behavior to move away from its hatch tree when ALB populations are high (Williams et al., 2004).

During the Level 2 survey, inspectors focus on maple trees (*Acer* spp.) where they are present. Additional preferred hosts (*Aesculus*, *Betula*, *Populus, Salix*, and *Ulmus*) are surveyed when maples are not present or are within 100 yards of the last inspected preferred host tree. Sweet birch (*Betula lenta*) and Japanese maple (*Acer palmatum*) are not considered preferred hosts. The Program may adjust the survey radius upon discovery of additional infested trees.

Similar to the Level 1 survey, inspectors complete the survey using binoculars from the ground, and may use bucket trucks and/or tree climbers to perform aerial surveys.

Depending upon terrain or geographic layout (e.g., urban/suburban streets versus a forested area), inspectors use differing techniques to move quickly through an area to determine the extent of the infestation and determine the quarantine boundary. For example, in an urban setting, inspectors may skip to the next street or block; in wooded locations, inspectors may use a grid pattern. The intention of this survey is to quickly understand the general area of infestation, and determine the type of response needed in terms of resources and quarantine enactment.

3. Level 3 Survey (High-Risk Survey)

The ALB Program may conduct a Level 3 survey, also called a high-risk site survey. This survey extends beyond established survey boundaries to inspect high-risk sites.

The Program conducts high-risk site surveys (Level 3) to identify and inspect high-risk locations where potentially infested host material may have been transported, stored, processed, or sold. Site inspections primarily focus on maples when they are present. High-risk sites include, but are not limited to:

- Landscape and nursery businesses
- Tree and lawn care companies

- Firewood producers and transporters
- State and local parks/forestry departments
- Local utility and sanitation services
- Parks and campsites
- Landfills and disposal sites
- Import facilities that receive or have received high-risk cargo from known source countries
- Stone dealers or stone cutting facilities such as monument/headstone companies near infestations

4. Secondary Survey

After APHIS completes the first survey cycle and removes or treats infested trees and trees with a high risk of becoming infested, inspectors conduct secondary surveys, which is a repeat of survey Levels 1 and 2. Secondary surveys may take place more than once in an area, depending on the amount of time that has passed since the last survey cycle.

The amount of time between surveys can influence the survey frequency due to the understanding that ALB infestations develop over time and signs of infestation become more readily detectable.

Secondary surveys continue until a final cycle confirms there are no infested trees. Four years is the minimum amount of time between an initial detection and a final survey cycle is completed. The Program may conduct an interim survey should more than 4 years lapse since the first survey cycle occurred and the Program has yet to conduct a final round of negative survey. This would provide an extra level of assurance that a population no longer remains and could limit the scope and expense of the ensuing program should detection of ALB occur.

C. Alternatives Evaluated in this EIS

APHIS is considering five alternative options, four of which are eradication program options. The five alternative options derive from scientific research published in peer-reviewed scientific journals, and experience in the eradication programs in Illinois, New York, New Jersey, Ohio, and Massachusetts. These alternatives are not unique to this EIS; other EAs prepared by APHIS include one or more of these alternatives (appendix B). Briefly, the five alternative options considered in this EIS include No Action, Removal of Infested Trees, Full Host Removal, Insecticide Treatment, and An Integrated Approach (see table 2–1).

Adaptive Management

This EIS proposes the use of specific chemical treatments as part of the ALB Eradication Program under the various alternatives with the exception of the no action alternative. Herbicides are proposed for use in

Table 2–1. Overview of Alternatives in this EIS.

Methods	Alternatives*				
	1 – No Action	**2** – Removal of Infested Trees	**3** – Full Host Removal	**4** – Insecticide Treatment	**5** – Integrated Approach
Remove and chip or incinerate infested trees		X	X	X	X
Remove and chip/incinerate high-risk host trees within a ½-mile radius of infested trees			X		X
Grind stumps of cut trees		X	X	X	X
Treat stumps and sprouts of cut trees with herbicide		X	X	X	X
Allow stumps of host trees to regenerate			X		X
Site restoration		X	X	X	X
Imidicloprid treatment of high-risk host trees within a ½-mile radius of infested trees (only with landowner's permission)				X	X

* Imposing Federal quarantines and conducting surveys to determine the extent of ALB infestation are part of all the alternatives.

Adaptive Management

This EIS proposes the use of specific chemical treatments as part of the ALB Eradication Program under the various alternatives with the exception of the no action alternative. Herbicides are proposed for use in all the alternatives with the exception of the no action alternative, while insecticide use is proposed under alternatives 4 and 5. The Program could add other treatment(s) that may become available in the future to currently approved treatments for managing ALB, referred to as adaptive management. A new treatment would be available for use upon the agencies' finding that the treatment is registered by EPA for use on ALB, and poses no greater risks to human health and nontarget organisms than are disclosed in this EIS for the currently approved treatments. The protocol for making the necessary finding that a treatment is authorized by adaptive management is as follows:

1. Conduct a human health and ecological risk assessment (HHERA). In this risk assessment, review scientific studies for toxicological and environmental fate information relevant to effects on human health and nontarget organisms. Use this information to estimate the risk to human health and nontarget organisms. Include these four elements in the HHERA: (a) hazard evaluation, (b) exposure assessment, (c) dose response assessment, and (d) risk characterization. The HHERA will do the following:

    - Identify potential use patterns, including formulation, application methods, application rate, and anticipated frequency of application.
    - Review hazards relevant to the human health risk assessment, including systemic and reproductive effects, skin and eye irritation, dermal absorption, allergic hypersensitivity, carcinogenicity, neurotoxicity, immunotoxicity, and endocrine disruption.
    - Estimate exposure of workers applying the chemical.
    - Estimate exposure to members of the public.
    - Characterize environmental fate and transport, including drift, leaching to ground water, and runoff to surface streams and ponds.
    - Review available ecotoxicity data, including hazards to mammals, birds, reptiles, amphibians, fish, and aquatic invertebrates.
    - Estimate exposure of terrestrial and aquatic wildlife species.
    - Characterize risk to human health and wildlife.

2. Conduct a risk comparison of the human health and ecological risks of a new treatment with the risks identified for the currently authorized treatments. This risk comparison will evaluate quantitative expressions of risk (such as hazard quotients), and qualitative expressions of risk that put the overall risk characterizations into perspective. Qualitative factors include scope, severity, and intensity of potential effects, as well as temporal relationships, such as reversibility and recovery.

3. If the risks posed by a new treatment fall within the range of risks posed by the currently approved treatments, publish a notice in the FR of the agencies' preliminary findings that the treatment meets the requirements of this alternative. The notice must provide a 30-day public review and comment period, and must advise the public that the HHERA and the risk comparison are available upon request.

4. If consideration of public comment leads to the conclusion that the preliminary finding is correct, publish a notice in the FR that the treatment meets the requirements of this alternative and, therefore, is authorized by this alternative for use in the APHIS ALB management program. APHIS will make available to anyone, upon request, a copy of the comments received and the agencies' responses.

The decision to be made as a result of this EIS will be programmatic. Decisions to use specific treatments in projects (including new treatments authorized under the protocol in this alternative) will be made after site-specific environmental analyses are conducted and documented, in accordance with agency NEPA implementing procedures.

1. No Action

Under this alternative, activities for ALB would not involve APHIS. Other Federal or non-Federal entities may apply control measures; however, APHIS would not manage or fund these measures. Under this alternative, APHIS could conduct surveys to determine the extent of an infestation, and implement quarantine restrictions where ALB infestations occur.

This alternative represents the baseline against which to compare a proposed action. This alternative is not an eradication program.

2. Removal of Infested Trees

This alternative would consist of the following:

- Implementing or maintaining the quarantine and expanding or contracting the quarantine area based on ALB's distribution, as determined through survey

All of the alternatives in this EIS include survey and quarantine

- Removing all infested trees
- Chipping or incineration of cut trees
- Grinding or herbicide treatment of stumps
- Restoring the tree-removal site

APHIS and its cooperators, which includes State or forestry officials, contractors, and other entities collaborating with APHIS (referred to collectively as the Program), would remove trees infested with ALB. APHIS declares a tree infested if program inspectors find at least one oviposition site or exit hole characteristic of the shape and size as those formed by ALB, regardless of being able to determine the viability of ALB inside the tree. APHIS uses this criterion after official confirmation through structural or genetic identification that ALB is in the area. APHIS will notify in writing by direct mailing, or in person, any landowner who will have infested trees removed from their property.

After tree removal, the Program destroys cut trees through incineration or by passing the trees through a chipper to destroy the ALB life stages (larvae, pupae, and adults) that may be within those trees, thus eliminating potential adult beetle emergence, dispersal, and mating. Eggs may be able to survive the chipping process due to their small size; however, after hatching, larvae develop by feeding on the thin layer of generative tissue lying between the bark and the wood of a stem (cambium)—chipping makes this area unsuitable for the development of larvae (Wang et al., 2000). Chipping of trees to a size of less than 1 inch, in at least two dimensions, removes the risk of ALB (Wang et al., 2000). Chips of this size are no longer subject to Federal or State regulations. The Program chips trees in place or takes the tree to an approved establishment for chipping.

ALB can reinfest stumps, above-ground roots, and any shoots that may grow from these tree parts, so it can be beneficial to remove and grind stumps from infested trees. APHIS may use herbicides when there are limitations to physically removing stumps. For example, the area is inaccessible to equipment used for stump removal and grinding, or the area is sensitive to erosion or compaction. The Program uses the herbicide triclopyr by spraying or painting the root collar area, the sides of the stump, and the outer portion of the cut surface, including the cambium, until thoroughly wet but not to runoff. Foliar applications of triclopyr mixed with two other herbicides, imazapyr and metsulfuron-methyl, are applied to sprouting foliage from stumps that remain after tree removal to prevent regrowth. APHIS would apply adaptive management as described earlier in this section of the EIS. In some locations, APHIS may leave stumps and allow regrowth, particularly in areas prone to soil erosion or sensitive wildlife habitats.

After tree removal from yards and landscaped settings, the Program restores the area through grading and planting groundcover consistent with the area where the removals took place. This reduces the opportunity for invasive weeds to establish, and provides a groundcover that will help hold the soil in place. However, in woodlot settings a seed bank already exists in the soil that would result in rapid vegetation growth, therefore, planting groundcover may be unnecessary.

APHIS will remove an area from quarantine as described previously.

3. Full Host Removal

The Program, under this alternative, would consist of the following:

- Implementing or maintaining the quarantine and expanding or contracting the quarantine area based on ALB's distribution as determined through survey

All of the alternatives in this EIS include survey and quarantine

- Removing infested trees
- Chipping or incineration of cut trees
- Grinding or herbicide treatment of stumps
- Restoring the tree-removal site

As described in Alternative 2—Removal of Infested Trees

- Removing high-risk host trees within a ½-mile radius of an infested tree(s)

Under this alternative, the Program would conduct pest surveys, implement quarantine restrictions, and remove infested host trees, as described in alternative 2. Signs of low infestation levels may not be readily apparent on high-risk host trees, and can remain unnoticed by visual survey. Consequently, due to their proximity to known infested trees, there are chances that nearby high-risk host trees have undetected infestations or are at risk of infestation. Therefore, the Program would also remove high-risk host trees within a ½-mile radius of infested host trees.

The basis for removing trees within the ½-mile radius is on the dispersal behavior of ALB, and the level of effectiveness of visual survey on lightly infested trees. In a study in Illinois, 99 percent of trees with ALB egg-laying sites were within ¼ mile of a tree from which adult ALB exited (Sawyer, 2006). APHIS may add an additional ¼ mile to the radius for host tree removal around infested trees to capture beetles that may have spread further (Sawyer, 2006). APHIS will notify the landowner prior to the removal of infested and high-risk host trees. If the landowner refuses to allow removal of high-risk host trees, the Program would not remove them, but would continue to survey and, if ALB infests those trees, APHIS would remove the trees after notifying the landowner.

The cutting and removal of trees is the same as described in alterative 2. The Program would focus tree removal efforts first by removing infested trees to eliminate the population of ALB, and then remove high-risk host trees located within the ½-mile radius of infested trees. However, tree removal may not always occur in this order. For example, it is more time and cost effective to finish removing all the trees in an area before relocating equipment to a new staging area.

APHIS removes and grinds or treats stumps and roots of felled trees with herbicides, as described in alternative 2. However, in some cases, such as woodlots, stumps may not be ground or treated with herbicides to allow for regrowth of host trees. APHIS would apply adaptive management, as described earlier in this section of the EIS.

APHIS will remove an area from quarantine as described previously.

4. Insecticide Treatment

The Program, under this alternative, would consist of the following:

- Implementing or maintaining the quarantine and expanding or contracting the quarantine area based on ALB's distribution, as determined through survey

All of the alternatives in this EIS include survey and quarantine

- Removing infested trees
- Chipping or incineration of cut trees
- Grinding or herbicide treatment of stumps
- Restoring the tree-removal site

As described in Alternative 2—Removal of Infested Trees

- Treating high-risk host trees within a ½-mile radius of infested trees with an approved insecticide, following product label requirements.

Under this alternative, APHIS would remove only infested trees, as described in Alternative 2, Removal of Infested Trees. Due to their proximity to known infested trees, there is a chance nearby host trees are at risk of infestation. APHIS would treat high-risk host trees, located up to ½ mile of an infested tree, with the insecticide imidacloprid, but only with permission from the landowner. APHIS uses the insecticide to protect trees from ALB infestation but does not use the insecticide to treat infested trees. If the landowner does not allow imidacloprid treatment of trees on their property, the Program will not treat them; however, survey efforts would continue. Once trees become infested with ALB, the Program would remove them following notification of the landowner.

Application of imidacloprid is through trunk or soil injection, according to product label requirements. The Program makes insecticide treatments in the spring, early summer or fall, prior to and during the adult emergence period, in order to allow the insecticide to distribute throughout the tree and, therefore, be most effective.

The rate of imidacloprid used depends on the application method, as well as the size of the tree to be treated. Pesticide applicators adhere to the requirements on the product label. For trunk injections, applicators drill holes around the trunk, 2 to 6 inches above the soil-wood line, and inject imidacloprid. It takes 1 to 3 weeks for the insecticide to distribute throughout the tree, depending on the size and condition of the tree and weather conditions. Trunk injections are the most common application used by the Program. For soil injection, applicators inject imidacloprid at a minimum of four injection sites, spaced evenly around the base of the tree. Application occurs under the soil around the base of the tree, normally no more than 12 inches from the base. No material may puddle or run offsite. It may take up to 3 months before plant tissues absorb sufficient quantities of imidacloprid, depending on the size and condition of the tree and weather conditions.

For maximum efficacy, APHIS repeats insecticide applications once a year over a consecutive 3-year period to ensure that the concentration of the insecticide within the treated tree is at an adequate level to protect the tree from ALB infestation. Imidacloprid treatments do not ensure complete control of ALB within a tree due to variability in treatments, weather conditions, and tree health, all of which can result in uneven distribution of imidacloprid within a tree. APHIS would apply adaptive management as described earlier in the EIS.

APHIS will remove an area from quarantine as described previously.

5. An Integrated Approach (Preferred Alternative)

An integrated approach would consist of the following:

• Implementing or maintaining the quarantine restrictions, and expanding or contracting the quarantine area based on ALB's distribution as determined through pest surveys	All of the alternatives in this EIS include survey and quarantine
• Removing infested trees • Chipping or incineration of cut trees • Grinding or herbicide treatment of stumps • Restoring the tree-removal site	As described in Alternative 2—Removal of Infested Trees
• Removing high-risk host trees within a ½-mile radius of an infested tree(s)	As described in Alternative 3—Full Host Removal
• Removing or treating with an approved insecticide, according to label requirements, high-risk host trees up to ½ mile-radius from infested tree	As described in Alternative 4—Insecticide Treatment

For this alternative, the Program will remove infested trees and treat with imidacloprid, or remove high-risk host trees within a ½-mile radius of infested trees. APHIS will base the determination to treat or remove high-risk host trees on levels of infestation in the area, host tree density and distribution, potential environmental impacts, and logistical resources. This alternative provides the most flexibility in selecting an appropriate control method for a location. It is also the most cost effective method because this alternative does not prescribe that all high-risk host trees must be treated or removed; rather, it allows flexibility in focusing treatments on the host trees most preferred by ALB (i.e., *Acer*, etc.), or certain locations that would be higher risk than others.

As with the other action alternatives, APHIS would remove or treat with imidacloprid the high-risk host trees located up to ½ mile of an infested tree only with permission from the landowner. If the landowner does not allow removal or imidacloprid treatment of trees on their property, the Program will not remove or treat them but will continue survey efforts. If trees become infested with ALB, the Program would remove them following notification of the landowner.

APHIS would apply adaptive management as described earlier in this section of the EIS.

APHIS will remove an area from quarantine as described previously.

D. Alternatives Considered but Not Included in this EIS

1. Use of Other Chemical Control Agents

APHIS evaluated other insecticide chemistries for their efficacy towards ALB (appendix B). Imidacloprid is currently the only effective, registered product available for use against ALB.

2. Other Integrated Pest Management/ Suppression Strategies

a. Biological Control

Biological control (biocontrol) is the means of reducing or mitigating pests and pest effects with natural enemies. ALB causes damage within its native range despite the presence of associated natural enemies. Research on biocontrol methods for ALB are ongoing, however, none are available for large-scale use.

b. Sterile Insect Technique

Sterile insect technique (SIT) involves breeding and releasing large populations of sterile male insects with the goal of outcompeting fertile males during mating with females. This leads to a reduction in the size of the population, and potential eradication of an insect population. Current SIT technology is not feasible based on the ALB life cycle.

c. Insect Pheromones

Insect pheromones are naturally produced chemicals that can be used by insects to attract each other, and have been used in pest management for survey and suppression of target pests. The identification of insect pheromones specific to ALB is currently an ongoing area of research; however, additional work is needed before it can be successfully implemented in the ALB Eradication Program.

d. Regulatory Treatments

APHIS has determined that it is necessary to add a treatment schedule for ALB in the Plant Protection and Quarantine Treatment Manual. A treatment evaluation document has been prepared that discusses the existing treatment schedule and explains why this change is necessary. The document was made available to the public for review and comment until April 11, 2014. The public can access the "Notice of Availability of a Treatment Evaluation Document for Heat Treatment for Asian Longhorned Beetle" (Docket No. APHIS-2013-0094) online at http://www.regulations.gov/#!documentDetail;D=APHIS-2013-0094-0002).

e. Additional Methods Development and Research

Additional methods development projects include looking at chip size and grinding techniques to deregulate host material, and assessing the use of pesticide treatments in the fall. Research continues with analysis to determine how fast the insect spreads on its own, evaluating the host trees the beetle attacks for preference and range, DNA analysis and behavioral experiments.

III. Affected Environment

This section presents a baseline description of the environmental resources that would be affected by implementation of the alternatives presented in chapter 2. Environmental resources include physical and biological resources, as well as the economic and social factors affecting people.

Because this is a programmatic EIS, the description of the affected environment contained in this chapter is, by necessity, general. The potentially affected environment in the United States is anywhere host trees susceptible for ALB infestation are found. Given the known distribution of ALB, the insect is probably capable of surviving anywhere in the United States where suitable host plants and climatic conditions are available.

A. Affected Forest Communities

Forest trees grow either in pure stands comprised of a single species or in mixed stands. Tree density and type influence the sub-canopy plant species by altering the shade levels, soil composition, moisture levels, and other attributes. Forests fall on public or private lands, and occur on uplands, riparian areas (the interface between terrestrial and aquatic ecosystems), wetlands, and coastal environments.

Indicators of forest conditions include tree mortality and growth rates, degree of insect and pathogen damage, and species composition in the understory and canopy. Tree mortality from ALB can affect the composition of forest communities. As ALB and other introduced insects and pathogens spread, they add stress to forest communities. Other forest stressors, including reduced water and air quality, and geological and soil disturbance often result from human activity. As the proximity and size of human populations increase, the forest canopy coverage and biological diversity decreases, while fragmentation and ecological disturbance increases.

In the United States, 13 tree genera are regulated hosts for ALB: *Acer* (maple and box elder), *Aesculus* (horse chestnut and buckeye), *Salix* (willow), *Ulmus* (elm), *Betula* (birch), *Albizia* (mimosa), *Celtis* (hackberry), *Cercidiphyllum* (katsura tree), *Fraxinus* (ash), *Koelreuteria* (golden raintree), *Platanus* (sycamore and London planetree), *Sorbus* (mountain ash), and *Populus* (poplar) (APHIS, 2009), 7 CFR 301.51–2). APHIS expects to remove *Celtis* from the list of regulated host trees based on observation that this is not a host tree. The degree of susceptibility of a forest to ALB is dependent upon the composition of tree species, including the total density (basal area per acre) and proportion of an area covered by susceptible stands. Within the United States, the distribution of host species is largely concentrated in the mid-Atlantic, New England, and Great Lakes regions (referred to as the Northeast Region by the FS (figure 3–1), occupying a range of ecosystems including riparian and wetland areas. In the Northeast Region, forests (mostly privately owned) cover 42 percent of the land base (Shifley et al., 2012). The composition of trees in the Northeast Region includes the maple-beech-birch forest-type group that accounts for 29 percent of the forest area; elm-ash-cottonwood for 6 percent; and aspen-birch for 10 percent (Shifley et al., 2012). Combined, these forest-type groups account for 45 percent of the forest area. In other regions of the United States, ALB-host species are found but make up a smaller component of the tree species growing in these forest areas (figure 3–1). However, many of these species are planted as ornamentals and urban shade trees elsewhere in the United States (Peterson et al., 2004).

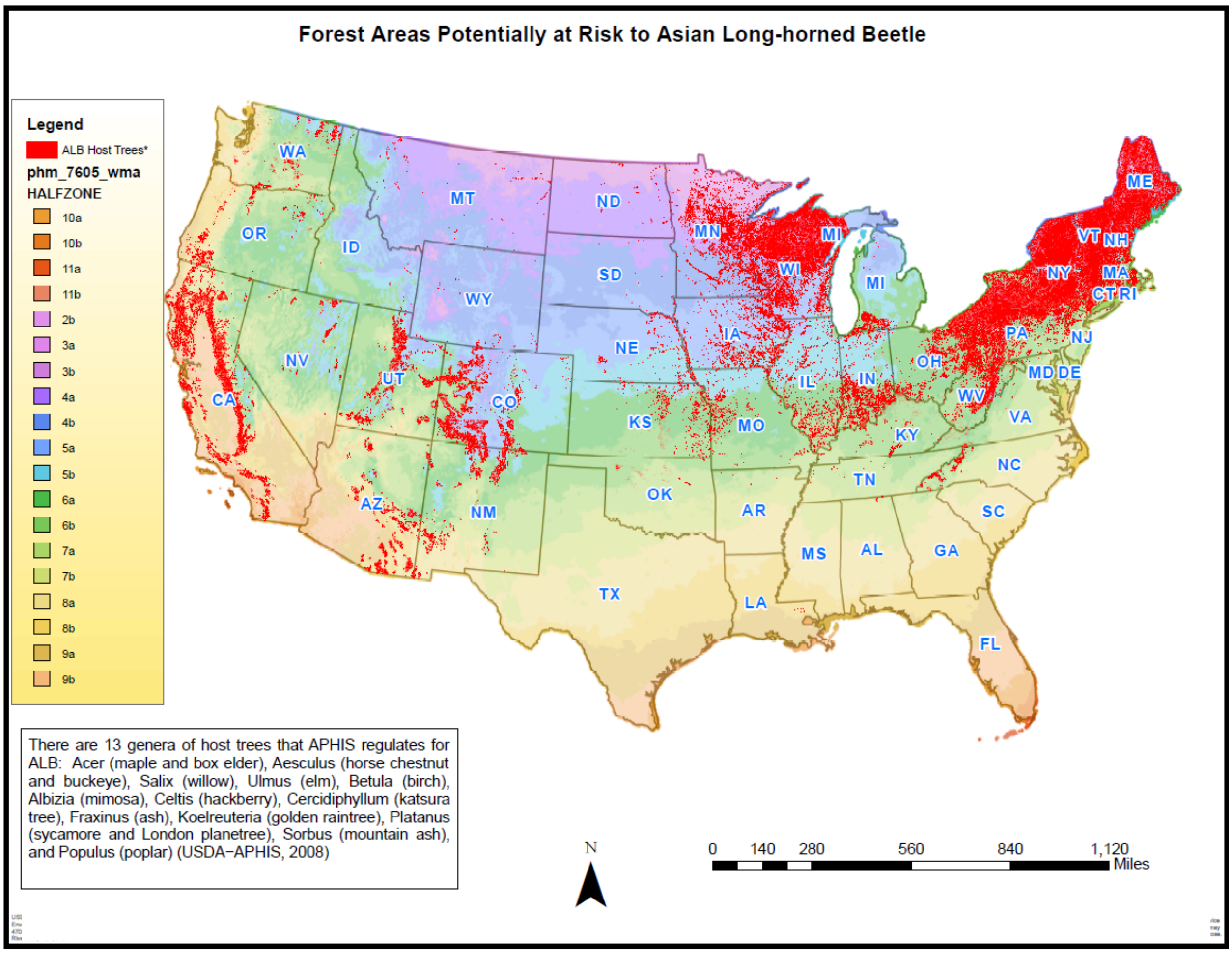

Figure 3–1. Forest areas potentially at risk to Asian longhorned beetle.
**This map does not include host genera planted in urban and suburban areas.
(Data source: (USGS, 2014))

It is possible that other tree species could become confirmed hosts for ALB, which may increase the geographic scope of the environmental impacts. (See appendix C for an annotated list of ALB hosts.)

B. Affected Water Resources

Surface and ground waters within or near affected forest communities may be part of the affected environment. Surface waters include streams, lakes, wetlands, and reservoirs. Surface water seeps underground forming aquifers, also known as ground water. The combination of surface water and underlying ground water within an area that drains to a common outlet, (e.g., a waterway, wetland, reservoir, aquifer, or ocean) is a watershed. In the contiguous United States, there are 2,110 watersheds that may contain ALB-host trees. Nationally, around 53 percent of the water supply originates on forestlands (Shifley et al., 2012). In the Northeast Region, 48 percent of the water supply originates on forestlands; in the South Region, 51 percent; in the Pacific Coast Region, 72 percent; and the Interior West Region, 43 percent (Shifley et al., 2012).

American Heritage Rivers are rivers that represent natural, cultural, and historical resources and flow through forest ecosystems, including suburban and urban forests. Created by Executive Order 13061 (September, 11, 1997), with selection criteria developed by the Council on Environmental Quality (CEQ), the EPA has designated 14 rivers in the contiguous United States as American Heritage Rivers. Examples include the Connecticut River, Cuyahoga River, Hudson River, Mississippi River, and the Potomac River.

The National Estuary Program is a network of voluntary community-based programs which safeguards the health of important coastal ecosystems across the country. The National Estuary Program, established under Section 320 of the 1987 Clean Water Act (CWA) amendments, calls for each national estuary program to develop and implement a comprehensive conservation and management plan that contains specific targeted actions designed to address water quality and habitat challenges in its estuarine watershed (EPA, 2013c). Twenty-seven national estuary programs are in place throughout the contiguous United States.

The Coastal Zone Management Act of 1972 (as amended), administered by the U.S. Department of Commerce, National Oceanic and Atmospheric Administration (NOAA), Office of Ocean and Coastal Resource Management (OCRM), provides for management of the nation's coastal resources, including the Great Lakes. Federal agencies are to cooperate with NOAA if their activities affect any land or water use, or natural resource of the coastal zone (16 U.S.C. § 1456, Section 307). Host trees of ALB occur in some coastal zones; ALB infestation and pursuant management activities can affect natural resources in these areas.

1. Water Quality

Physical, biological, and chemical alterations to ecosystems through natural means or invasive species and human activity can affect water quality and quantity. Indicators of water quality include the flow rate, as well as water and sediment chemistry.

Forested landscapes directly and indirectly affect water resources through a variety of mechanisms, including the stabilization of soil and prevention of sediment runoff; influence on the water cycle by controlling rainfall runoff, flooding, and uptake and evapotranspiration of water; and effects on water temperature and understory plant growth by casting shade.

Canopy gaps (reduced shade cover) can contribute to a decrease in soil moisture due to the increase in levels of light reaching the ground, as seen with the loss of hemlock trees to the hemlock wooly adelgid, an insect pest introduced in the United States (Orwig and Foster, 1998; Orwig et al., 2008). Canopy gaps can also contribute to an increase in water temperature thereby affecting aquatic life, including plants, animals, and invertebrates (Kaushal et al., 2010). Reduced tree density and canopy gaps can affect forest undergrowth and soil stability, and can increase overland water flow, water yield, and runoff; however, the magnitude, timing, and duration of the response varies considerably among forest communities (Hornbeck et al., 1993; Wear and Greis, 2013). Increases in water yield coincide with the mobilization and leaching of nutrients in some forest communities (Hornbeck et al., 1993). Urban development and conversion of forested lands to agriculture are major contributors to reduced forest canopies and increases in water runoff, affecting the temperature and water chemistry of streams and water bodies (Wear and Greis, 2013).

The vegetation in riparian areas and wetlands contribute to water quality and quantity. A wetland is a land area saturated with water at a frequency and duration to support vegetation adapted for life in saturated soil conditions. Many of the tree species that are regulated hosts to ALB grow in riparian and wetland areas, including maple (*Acer*), elm (*Ulmus*), ash (*Fraxinus*), cottonwood (*Populus*), sycamore (*Platanus*), willow (*Salix*), and birch (*Betula*) (USGS, 1998; NHDES, 2008; NCFS, n.d.). Changes in tree cover and density in these aquatic habitats can have negative impacts on streambank stabilization, water temperature, sediment loading, hydrology, nutrient cycling, and contaminant removal (Wenger, 1999; Lee et al., 2004; Jones et al., 2006).

For example, maple trees are a critical component in soil nitrification in the Northeast United States, and their loss could affect nitrogen retention and cycling in forested watersheds (Lovett and Rueth, 1999; Lovett and Mitchell, 2004). Nitrification is the oxidation of nitrogen-containing compounds into nitrate, which is an important component of the nitrogen cycle. A decrease in maple stands would lead to higher nitrogen retention in soils and reductions in nitrogen transport into aquatic systems. The alteration of nitrogen cycling, due to the loss of maples, would alter plant succession and diversity in terrestrial environments, as well as affect aquatic ecosystems that are dependent on higher nitrogen inputs.

Forested landscapes, including floodplains, wetlands, and riparian areas moderate flooding and filter sediments and pollutants (Dosskey et al., 2010), protecting water quality. Alterations to the quality of these surface waters can impair the values they impart. Human communities rely on surface and ground water for drinking water, irrigation, and recreational activities. Waters in urban and agricultural areas tend to have an increased nutrient concentration and load. Excess nutrients mostly arise from fertilizers, wastewater effluent, and industrial waste in urban areas; and animal waste, fertilizers, and chemicals in agricultural areas (Wear and Greis, 2013).

The Clean Water Act (CWA) provides a structure for regulating the discharge of pollutants into waters, and regulates quality standards for surface waters. Section 303(d) of the CWA requires States to develop lists of its impaired and threatened waters (stream/river segments, lakes) (EPA, 2014c). Causes for impairment are numerous and include categories such as pathogens, metals, salinity, sediments, pesticides, trash, and other organic and inorganic compounds. Most States have one or more impaired waters, meaning that the water is not meeting one or more of its designated uses (EPA, 2014c). Assessments are not complete for all watersheds.

2. Interactions of Soils with Water Quality

Soil quality impacts water quality and availability. A well-managed soil will have good porosity (space between soil particles), allowing it to be an efficient receiver of rainwater. Water that infiltrates the soil, in the absence of excessive nutrient or contaminant loads, is generally purified before entering ground water sources or returning to surface water bodies (Karlen et al., 1997). However, improperly managed or disturbed soil typically results in poor porosity, leading to surface water runoff carrying potential pollutants and soil particles with it. Soil enters surface waters as sediment, and can negatively affect water quality. Smaller particles (e.g., clay) stay in suspension contributing to water turbidity (Cook, 1990). Riparian and floodplain habitats are especially sensitive to changes in water quality (Doupé et al., 2010). By volume, sediment is the largest cause of impairment of rivers and streams across the United States (Cunningham et al., 2001; EPA, 2013b).

C. Affected Soil

Soil types capable of supporting ALB-host trees are part of the affected environment. The physical, chemical, and biological characteristics of soil affect the health of the vegetation it supports by changing the availability of water and nutrients.

Soil is composed of a diversity of mineral and organic components. Soil stores, cycles, and moderates the release of nutrients and other elements. Soil sequesters carbon, reducing the prevalence of atmospheric carbon dioxide, a gas linked to climate change. In the Northeast, roughly as much carbon is stored in the soil, mostly in the form of organic matter, as in live biomass (e.g., trees) (Shifley et al., 2012).

In most ecosystems, soil biota help to regulate a number of key ecosystem services, including plant production, nutrient and carbon cycling, maintenance of soil structure, and water regulation (Wall et al., 2012). Soil biota can have direct and indirect impacts on land productivity and other ecosystem functions, such as fresh water; food and pollination services; timber, fiber, and fuel; nutrient and waste management; and climate regulation (Barrios, 2007).

Vegetation alters soil nutrient cycling, especially in situations where certain vegetation types serve a unique ecological function. For example, maple trees are a critical component in soil nitrification in the Northeast United States, and their loss could affect nitrogen retention and cycling in forested watersheds (Lovett and Rueth, 1999).

1. Soil Types of the United States

Soils are categorized by type, which describe the physical properties of the soil (including permeability, water-holding capacity, soil texture, and soil structure), and its chemical properties (including pH, salinity, cation exchange capacity, organic matter, and carbon-to-nitrogen ratio). The Natural Resources Conservation Service categorizes soil type into 12 soil orders and 64 suborders (McDaniel, 2006). Based on their various characteristics, different soil orders and suborders have varying capacities to support ecosystem services (i.e., retain water, filter water impurities, cycle nutrients, anchor plant roots, and absorb air pollutants). As a result, disturbance affects different soil types to varying degrees and in a variety of ways.

Changes in physical soil characteristics occur when ground-based equipment makes repeated passes over the soil. These activities compact soils and, if soils are wet enough, can result in rutting and leaching of soil nutrients. Different soil types compact more readily than others; clay and loam soils generally compact more than sandy soils. These changes to the physical soil characteristics reduce the pore space and the ability of the soil to retain water. In turn, this reduces infiltration rates, slows soil drainage, impedes root growth, and reduces plant-available water and nutrients. Physical soil disturbances also decrease gas exchange, affecting both plants and soil biota.

Organic matter affects the chemical soil characteristics. In its various forms, organic matter provides nutrients and retains moisture for soil organisms and plants. Because organic matter derives from decomposing plant material, the type of vegetation growing in an area influences the chemical composition of soil.

2. ALB-Host Species and Soil Quality

Soil quality refers to "the capacity of a soil to sustain biological productivity, maintain environmental quality, and promote plant and animal health" (Doran and Parkin, 1994). Erosion, compaction, loss of soil structure, loss of nutrient content, and changes in soil salinity degrade soil quality (Cook, 1990). Atmospheric acid deposition has led to a concentration of highly acidic soils in the Northeastern States and southward along the Appalachian Mountains (Shifley et al., 2012). A continued increase in acidity is likely to make sensitive trees more prone to other stressors, for example, insect attacks. Soil types vary in response to different impacts and the preparation of EAs at the local or regional level for specific Program actions will address the site-specific issues.

The types of trees that grow in a forest ecosystem affect the quality of the forest soil. Predominant or "foundation" species often define the structure of a forest community by creating locally stable conditions for other species, and by modulating and stabilizing soil quality, productivity, and water balance (Dayton, 1972; Ellison et al., 2005). For example, red maple and sugar maple, high-risk host tree species for ALB, are two of the most common tree species in many Northeastern forests in the United States (Lovett et al., 2006), and play a critical role in shaping the physical and chemical aspects of the soil (Mroz et al., 1985). Removal of foundation tree species can have dramatic effects on ecosystem function and stability. The physical characteristics of trees within forest ecosystems define forest structure and alter microclimates, while the leaf litter contributes substantially to ecosystem processes, such as nutrient cycling (Ellison et al., 2005).

Host species that grow in riparian forest ecosystems offer a variety of ecosystem services specific to soil, including stability and nutrient cycling. The riparian zone is the interface between terrestrial and aquatic ecosystems through which water and materials move, and are often impacted by a change in soil conditions or vegetation (Knoepp and Clinton, 2009). The stability and nutrient cycling these ecosystems provide help to protect aquatic environments against excessive sedimentation, polluted surface runoff, and erosion. The overall health of riparian forest ecosystems is critical to maintaining good water quality and the health of stream ecosystems (Knoepp and Clinton, 2009).

3. Soil Erosion

Soil erosion is the movement of soil particles by water, wind, or ice. Erosion is a natural process within ecosystems that removes and redistributes soil. However, anthropogenic activities (i.e., construction, agriculture) can accelerate erosion. A soil system is in equilibrium when soil erosion is in balance with the formation of new soil (Wall et al., 2012).

In forested sites on steep slopes, water is generally the most common driver of erosion. Erosion is usually infrequent in undisturbed forest soils because organic matter provides a protective layer on the soil surface, limiting the impact of raindrops, and allows water to infiltrate. The surface soil below the organic layer is generally porous, allowing water to infiltrate into and through the soil profile. Soil erosion can occur when the surface soil is compacted, or when the loose surface soil and its protective layer of organic material are changed or removed, such as by disturbances associated with management activities.

Forest soils may be compacted by grazing animals and by the roots of the trees themselves, but more noticeably by vehicles used for a range of mechanized forest operations. Soil compaction drastically reduces the number and size of pores that naturally occur throughout the soil. This reduces the exchange of gases and infiltration of water through the soil. Although compaction may allow surface soils to hold more water, it will tend to pool without soaking through to deeper layers of soil. In turn, runoff of surface water may increase, and tree growth may be reduced because of a reduced water supply, restricted root space, and poor aeration. In contrast, soil compaction may increase traction and, therefore, increase efficiency of vehicles moving on roads and tracks in the forest.

D. Air

This section provides a general overview of air quality as it pertains to the proposed action and alternatives. All site-specific assessments will include an analysis of local air quality, and may tier to this section of the programmatic EIS.

Air quality is affected by two types of pollutants (primary and secondary), and greenhouse gases (GHG) that can pollute the air for human health, forest health, visibility, acid deposition, and climate change. Primary pollutants, such as particulate matter (PM), volatile organic compounds (VOCs), carbon monoxide (CO), nitrogen oxides (NOx), and sulfur oxides (SOx) that can affect human health, are directly generated from sources such as industrial facilities, cars and other mobile sources, and forest processes and activities, including fire (Stern, 1977). Secondary pollutants, for example ozone (O_3), are chemically transformed from primary pollutants such as VOCs and NOx (Stern, 1977). Forest fire emissions, when added to primary pollutants that affect human and forest health, contain carcinogenic air toxins such as acrolein, benzene, mercury, and formaldehyde (Langmann et al., 2009). Black carbon (BC) and GHGs, including carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) affect climate. Pollutants such as sulfate (SO_4), nitrate (NO_3), and organic and elemental carbon (soot) reduce visibility (Stern, 1977).

Atmospheric GHGs, such as CO_2 and CH_4, can trap solar energy and affect climatic conditions (EPA, 2012b). Elevated temperatures can lead to higher ozone levels (EPA, 2012b). GHG emissions in the United States increased 7.3 percent from 1990 to 2009, primarily due to emissions from electricity generation and transportation (EPA, 2012b) (table 3–1).

Table 3–1. GHG Emissions in the United States Allocated by Source*

Source	1990	2005	2008	2009	2010	2011	2012
Energy	5,260.1	6,243.5	6,071.1	5674.6	5,860.6	5,712.9	5498.9
Industrial Processes	316.1	334.9	335.9	287.8	324.6	342.9	334.4
Solvent and other Product Use	4.4	4.4	4.4	4.4	4.4	4.4	4.4
Agriculture	473.9	512.2	543.4	538.9	534.2	528.3	526.3
Land-use change	13.7	25.5	27.3	20.5	20.0	36.0	37.8
Waste	165.0	133.2	136.0	136.5	131.1	128.5	124.0
Total Emissions	**6,233.2**	**7,253.8**	**7,118.1**	**6,662.9**	**6,874.7**	**6,753.0**	**6,525.6**
Land Use and Forestry (sinks)	(831.1)	(1,030.7)	(981.0)	(961.6)	(968.0)	(980.3)	(979.3)
Net Emissions (sources and sinks)	**5,402.1**	**6,223.1**	**6,137.1**	**5,701.2**	**5,906.7**	**5,772.7**	**5,546.3**

* Teragram (Tg), or million metric tons CO_2 Eq. (Data source: (EPA, 2014b))

Different Federal, State, and local air regulatory agencies have created laws, rules, and regulations for control and reduction of air pollutants. Under the Clean Air Act (CAA), last amended in 1990, EPA set National Ambient Air Quality Standards (NAAQS) for pollutants considered harmful to public health and the environment (EPA, 2012a). NAAQS cover six criteria pollutants—sulfur dioxide (SO_2), nitrogen dioxide (NO_2), CO, ozone (O_3), particulate matter (PM_{10} and $PM_{2.5}$), and lead.

EPA requires all States to develop attainment plans (State implementation plans (SIPs)) to improve air quality in nonattainment areas.

Air quality monitoring data is collected and reviewed by EPA and State and local regulatory agencies, and is available to the public. This data is often published with respect to a local air quality index (AQI). AQI is a measurement of the level of pollutants in the atmosphere. An AQI above 100 indicates that air quality conditions exceed human health standards, while values below 100 indicate pollutant levels are below air quality standards. An AQI that exceeds 100 suggests that air quality may be unhealthy for certain sensitive groups of people.

In general, air quality is improving on a national scale, particularly through regulations and voluntary measures taken by industry to reduce emissions (EPA, 2012b). Cleaner cars, industries, and consumer products have contributed to cleaner air for much of the United States (EPA, 2012b). Control programs for mobile sources and facilities (e.g., chemical plants, dry cleaners, coke ovens, and incinerators) are primarily responsible for these reductions (EPA, 2012b). Despite the downward trend in pollutant levels observed across the United States, numerous counties have reported nonattainment for one or more of the six criteria pollutants (EPA, 2013a).

Trees have a beneficial effect on air quality by removing pollutants from the air, thus reducing human exposure to these substances and associated risk. Trees absorb CO_2 from the atmosphere and release oxygen through photosynthesis (Shifley et al., 2012). Trees also absorb or intercept pollutant particles (PM_{10}, e.g., dust, ash, dirt, pollen, and smoke) and gaseous pollutants (e.g., O_3, NO_2, and SO_2) in the air, reducing human exposure to these substances and associated risks (Beckett et al., 1998; Nowak et al., 2000; Nowak et al., 2006; Tiwary et al., 2009). One estimate implies that a mature urban tree can intercept up to 50 pounds of particulates per year (Dwyer et al., 1992). In urban areas, ozone, SO_2 and nitrogen oxide are some of the most common pollutants, all of which trees may absorb (Bell and Treshow, 2002). Trees protect human health by reducing pollutant exposure which, in turn, can diminish respiratory illnesses related to pollutant exposure (Kim and Bernstein, 2009).

Forests can serve as a sink for GHG emissions, sequestering the gases that contribute to GHG levels (table 3–1). In 2009, estimates of the total carbon on forest land in the Northern States, including soil organic carbon, was 14,413 million dry tons, which represents about 32 percent stored on forest lands and soil in the United States (Shifley et al., 2012). In 2010, an estimated 12.4 billion tons of carbon was stored in Southern forests, within tree biomass, soil organic carbon, and understory plants above and below ground (Wear and Greis, 2013). Urban forests also contribute to carbon sequestration. In an early estimate, urban forests stored approximately 800 million tons of carbon, nearly 5 percent of live tree carbon storage in all U.S. forests (Dwyer et al., 1992). Fifty-year projections indicate the forest carbon pool in the South is 5 percent smaller than the pool in 2010 (a net emission of about 600 million tons) (Wear and Greis, 2013). Trees release stored carbon when burned or through the decay process. Conversion of trees to lumber does not immediately release carbon stores within the tree.

Trees can have some negative effects on air quality as well. Trees release natural VOCs, which play a part in the formation of ozone and carbon monoxide (Beckett et al., 1998). These VOCs also can congregate with other particles in the atmosphere to create a haze over some stands of trees (Beckett et al., 1998). In a 2005 study, national VOC emissions from biogenic (natural) sources were larger than the VOC emissions from anthropogenic (human caused) sources, accounting for approximately 74 percent of VOC emissions (EPA, n.d.). Anthropogenic sources of VOCs are from industrial processes and manmade products, such as power plants, chemical production, solvents, vehicles, and other machinery (EPA, 2012b, n.d.). On a national level, anthropogenic VOC emissions have been declining (EPA, n.d.).

In the Northeast region of the United States (comprised of 20 States), 42 percent of the land cover is forested (Shifley et al., 2012). Projected urban development and other land use will shift the location of forested land cover, but overall coverage in the region is expected to remain stable in the near term (Shifley et al., 2012). The loss of trees in developed areas will affect local air quality; however, the greatest contribution to air quality improvements over the last decade is due to the reduction of mobile and industry emissions. “EPA expects air quality to continue to improve as recently adopted regulations are fully implemented and States work to meet current and recently revised national air quality standards.” (EPA, 2012b). Stricter air-quality regulations anticipated in coming years may add to the regulatory constraints on use of prescribed burning (Wear and Greis, 2013).

E. Affected Biological Organisms

All aquatic and terrestrial plant, vertebrate, and invertebrate species living in the environments that could support ALB are part of the affected environment. Changes in the composition of tree species can alter ecosystems, and can affect the species dependent upon them. Biological diversity, a term given to the variety of life and the natural patterns it forms, provides a large number of goods and services that sustain our lives, including food security, fresh air and water, energy, and biological-based products, such as wood products. In forests, high biological diversity enables these ecosystems to respond to external influences and recover from disturbances, while maintaining their ecosystem services, such as nutrient cycling, support to wildlife, and the purification of air and water (Shifley et al., 2012). Due to the importance of biological diversity in the health and sustainability of human populations, national and international programs and organizations work to preserve biological diversity, including the 150 Government-Member Convention on Biological Diversity, of which the United States is a member. Biological diversity is generally lower in suburban and urban forest communities and, in these areas, native species tend to be fragmented and small. Wildlife is usually limited to those species that are adapted to living near people, including raccoons, squirrels, deer, opossums, and a variety of birds.

Domestic animals and pets also comprise a sector of animal life that cohabitates with people. Disturbances to wildlife through habitat destruction from development, traffic, and noise are common in developed communities. In contrast, less developed forest communities (e.g., State and national parks) sustain higher levels of biological diversity and harbor larger mammal species that tend to have a larger home range. In the Northeast region of the United States, forests support 780 known animal species (Shifley et al., 2012). Common mammals in the Northern forest include deer, black bear, porcupine, raccoon, and squirrel (Smith et al., 2007).

Migratory birds contribute to the biological diversity in the United States and bring enjoyment to millions of Americans. Neotropical migratory birds depend on forest stands for nesting and foraging (Donovan et al., 1995; Suarez et al., 1997). Neotropical bird species vary in their habitat preference; some species depend on interior forest habitats, while others prefer edge habitats (Suarez et al., 1997; Thompson, 2005). In response to the importance of migratory birds, the United States established the Migratory Bird Treaty Act for the conservation of migratory birds and their habitats. The contiguous United States has four migratory game bird flyways: Atlantic Flyway, Central Flyway, Mississippi Flyway, and the Pacific Flyway. Managed by FWS and its partners, the goal for administering these flyways is to conserve migratory game bird species and allocate bird resources. Flyways are routes taken by a concentration of migratory birds between breeding and wintering areas.

In the United States (not limited to the contiguous United States), there are over 1,400 (T&E) species federally listed through the Endangered Species Act. Approximately 60 federally listed species and their critical habitats co-occur with ALB-host trees and forest-dependent listed species in the Northeastern and mid-Atlantic areas (figure 4–2). Threatened species are plants and animals that are likely to become endangered throughout all or a significant portion of its range.

Endangered species are those plants and animals that have become so rare that they are in danger of becoming extinct. The forest tree composition and its interaction with the air, water, and soil resources (discussed previously) affect the habitats in which T&E species live. T&E species are generally more sensitive to changes in their habitats. The threatened Virginia round-leaf birch, *Betula uber* (species of *Betula* are hosts to ALB), is found in only one location in Smythe County, Virginia.

According to FWS, Virginia round-leaf birch is associated with second growth deciduous and mixed conifer/deciduous forests. Vegetation associated with the known extant population of the species includes oak-pine and maple-beech-birch associations, with some tendencies to elm-ash-cottonwood associations perhaps because of the riparian setting (Garrison et al., 1977). In prior EAs, USDA–APHIS consulted with FWS on several T&E species that occur in areas where the Program is operating (see table 3–2 for a list of these T&E species). Several species are forest-dependent, including the Indiana bat (*Myotis sodalis*) (FWS, 2007), and the small whorled pogonia (*Isotria medeoloides*) (Mehrhoff, 1989). State species of concern, as well as federally listed plant species that are preferred hosts for ALB, occur in areas where ALB could become established (NHNHB, 2008).

Table 3–2. Threatened or Endangered Species that Occur in Areas the Program is Operating*

Genus Species	Common Name	Type	State Occurrence
Charadrius melodus	Piping plover	Bird	Massachusetts
Cyprogenia stegaria	Fanshell	Aquatic mollusk	Ohio
Epioblasma triquetra	Snuffbox	Aquatic mollusk	Ohio
Isotria medeoloides	Small whorled pogonia	Terrestrial plant	Massachusetts
Lampsilis abrupta	Pink mucket pearly mussel	Aquatic mollusk	Ohio
Myotis sodalist	Indiana bat	Mammal	Ohio
Myotis septentrionalis+	Northern long-eared bat	Mammal	Ohio, NY, MA
Plethobasus cyphyus	Sheepnose	Aquatic mollusk	Ohio
Setophaga kirtlandii	Kirtland's warbler	Bird	Ohio
Trifolium stoloniferum	Running buffalo clover	Terrestrial plant	Ohio
Villosa fabalis	Rayed bean	Aquatic mollusk	Ohio

* T&E species previously analyzed by APHIS for the ALB Program; APHIS consulted with FWS.
+ Proposed for listing.

Pollinators

Pollination is a process of fertilization that occurs through the transference of pollen granules from the anthers (male parts) of a flower to the stigma (female parts) of the same or different flower. This ensures that a plant will produce a full-bodied fruit, and a full set of viable seeds. Pollen moves from flower to flower by wind, rain, and gravity, as well as by pollinating animals, such as birds, bees, bats, butterflies, moths, beetles, ants, and other animals. Pollinators use both pollen and nectar as food sources; some pollinators (e.g., honey bee) survive exclusively on pollen and nectar collected from flowers.

In most terrestrial and aquatic environments, pollination and pollinators render vital ecological services. These services often have economic consequences; many agricultural crops rely on pollination to turn out the food on which humans and other animals depend for survival (Allen-Wardell et al., 1998). In fact, the ecological services that pollinators provide are necessary for the reproduction of over two-thirds of the world's crop species, and 60 to 90 percent of the world's flowering plants (Klein et al., 2007; Kremen et al., 2007).

Reproduction of many flowering plants in the forest ecosystem is dependent upon insect pollinators (Coulson et al., 2005). Forest fragmentation caused by tree removal affects plant-animal interactions. Fragmentation introduces "edge" into a landscape—the changes in population or community structures that occur at the boundary of two habitats. The restricted size, discontinuity, and increased edge of fragments may impose many ecological and genetic effects on plants, both directly and indirectly, through pollination (Aizen and Feinsinger, 1994).

Red maple, a regulated host of ALB in the United States, is an important early spring food resource for European honey bees (*Apis mellifera*) and other pollinators (Batra, 1985). When few other flowers are available, the red maple undergoes massive bloom between March and May, depending on elevation and latitude (Walters and Yawney, 1990). Boxelder and willow, both regulated hosts of ALB, are important food resources for mason bees (*Osmia lignaria lignaria* Say) during nest construction in the Northeastern United States and mid-Atlantic States (Kraemer and Favi, 2005).

Global pollinator decline has become an issue of concern for agricultural crop science. In North America and many parts of the world, the viability of multiple agricultural crops and broader ecosystems is threatened by unsustainable declines in the populations of honey bees, bumblebees, and other insect pollinators (NRC, 2007; Pettis and Delaplane, 2010). Colony collapse disorder (CCD) is a recent, pervasive syndrome affecting honey bee (*Apis mellifera* L.) colonies in the Northern Hemisphere, and is characterized by a sudden disappearance of adult honey bees from the hive. Multiple causes of CCD and general pollinator decline have been proposed, such as poor nutrition, pesticides, pathogens, parasites, and natural habitat degradation (Thompson, 2003; Desneux et al., 2007; Gill et al., 2012).

F. Affected Economic, Social, and Cultural Factors

1. Market Factors

Trees provide a range of products and support several industries within the United States. Tree species, such as those that are host trees for ALB, provide timber, maple syrup, and nursery trees, and generate income related to recreational activities. These industries operate predominantly in less developed forest communities, including uninhabited forest and forest recreational areas; these are important to the economies of many communities in the United States. The discussion below focuses on market and non-market values in the Northeastern United States as the number of ALB-host trees is higher in those areas when compared to other parts of the United States. In 2006, the Northern States wood products industry and the pulp and paper industry was estimated at $112 billion; primary wood products added $52 billion to the economy (Shifley et al., 2012). Hardwood trees compose the majority of the industry in the North (Shifley et al., 2012). Economic returns on the market factors discussed below would be much greater when considering all States where ALB-host trees are present.

2. Timber

The volume of sawtimber varies from State to State when making comparisons between Northeastern States where a significant amount of ALB-host trees are present (table 3–3). Sawtimber refers to a growing stock tree containing at least a 12-foot saw log or 2 noncontiguous sawlogs 8 feet or longer, free from defects. The percentage of hardwoods that are ALB-host trees ranges from 24 percent for Rhode Island to 65 percent for Vermont.

Timber from ALB-host trees has a variety of uses in roundwood products within these nine States. Roundwood products are logs, bolts, or chips cut from trees for industrial and nonindustrial uses (sawlogs, veneer logs, pulpwood, fuelwood, etc.) (FS, 2014). A majority of the volume from these tree species in these Northeastern States is used as sawlogs or as fuelwood (table 3–4) (APHIS, 2009).

This production of timber products translates into hundreds of millions of dollars in value in the Northeastern States. The total value of host species sawlogs for the eight States, as shown in table 3–5 (New Jersey is not included because stumpage prices were not available), was $171 million in 2006. Production of veneer logs was worth an additional $720,000 in

Table 3–3. Volume of Sawtimber (ALB-Host Species) in Northeastern States and the State of Ohio—2012.

State	Net Volume* (billion board feet)	% of Hardwood Volume at Risk	% of Total Volume at Risk
Massachusetts	5.55	40%	22%
Connecticut	4.53	35%	30%
Maine	12.08	60%	21%
New Hampshire	7.49	49%	24%
Vermont	14.19	65%	38%
Rhode Island	0.43	24%	16%
New York	32.94	47%	35%
New Jersey	1.85	30%	15%
Pennsylvania	40.67	36%	34%
Ohio	17.33	36%	34%

* Net volume eguals gross volume less deductions for other defects that affect use for lumber.
(Source: USDA–FS (2014), Northern Research Station (NRS-171:188)).

Table 3–4. Volume of Roundwood Products (ALB-Host Species) in the Northeastern States—2006.

State	Sawlogs	Veneer Logs	Pulpwood	Composite Products	Fuel-wood	Post-poles-pilings	Other Products	All Products
	MCF *							
Connecticut	1,264	0	97	0	2,094	0	0	3,457
Maine	19,211	0	0	0	2,506	0	0	21,718
Massachusetts	950	0	0	0	7,095	0	0	8,044
New Hampshire	12,320	572	0	0	2,880	0	48	15,822
New Jersey	323	0	32	0	5,641	0	0	5,998
New York	28,902	588	21,879	1,440	41,805	0	2	94,612
Ohio**	60,272	478	23,538	695	No data	0	0	91,204
Pennsylvania	33,828	2,656	2,368	0	1307	298	554	41,014
Rhode Island	81	0	0	0	49	0	0	130
Vermont	12,360	0	0	0	7,598	0	0	19,956
TOTAL	109,239	3,816	24,376	1,440	70,975	298	604	210,751

* = Thousand cubic feet
** Data for Ohio is for 2010. (Source: (FS, 2009b))

New York. Fuelwood production of host species was worth an additional $6.6 million in 2006. Although there is substantial fuelwood production in many States, the overall value of this product is less because the prices for fuelwood are significantly lower than many other products. Host species are also used for pulpwood, composite products, and other products in some Northeastern States, but are not valued here due to lack of price data (APHIS, 2009).

Table 3–5. Value of Selected Timber Products for Species and State—2006.

	CT	ME	MA	NH	NY	PA	RI	VT
	(million dollars)							
Sawlogs (Total)	0.833	22.313	0.586	10.033	67.245	49.014	0.087	20.463
Ash	0.131	0.000	0.107	0.552	8.438	2.189	0.002	1.423
Sugar Maple	0.451	13.503	0.312	3.947	45.238	23.474	0.073	17.047
Red Maple	0.117	2.266	0.070	1.833	13.410	15.182	0.010	0.000
Yellow Birch	0.049	4.119	0.025	2.421	0.075	0	0.001	1.381
Other Birch	0.025	2.425	0.045	1.281	0.026	0	0.001	0.611
Elm	0	0	0	0	0.005	0	0	0
Poplar	0.060	0	0.027	0	0.053	8.169	0.001	0
Veneer Logs* (Total)	0	0	0	NA	0.7196	NA	0	0
Ash	0	0	0	NA	0.1323	NA	0	0
Sugar Maple	0	0	0	NA	0.3437	NA	0	0
Red Maple	0	0	0	NA	0.2391	NA	0	0
Yellow Birch	0	0	0	NA	0.0036	NA	0	0
Other Birch	0	0	0	NA	0.0007	NA	0	0
Elm	0	0	0	NA	0.0002	NA	0	0
Poplar	0	0	0	NA	0.0001	NA	0	0
Fuelwood	0.137	0.501	0.466	NA	4.703	NA	0.003	0.807
TOTAL	0.970	22.815	1.052	10.033	72.668	49.014	0.090	21.270

Price data was not available for veneer logs and fuelwood in NH and PA. (Source: (APHIS, 2009))

The United States exports sawlogs to various countries. Figure 3–2 shows historical trends for the U.S. hardwood log exports for four ALB-host species. In 2008, log exports for birch, ash, maple, and yellow poplar were valued at $153 million in total (FAS, 2009). Historically, exports of maple logs have been the highest in value of the four species; however, the total value of maple log exports has been decreasing since 2005. These four ALB-host species comprise 21 percent of the value of all U.S. hardwood log exports in 2008 (APHIS, 2009).

In 2012, log exports for birch, ash, maple, yellow poplar, and cherry (not a host of ALB) were valued at $188.7 million in total (GTIS, 2013). China

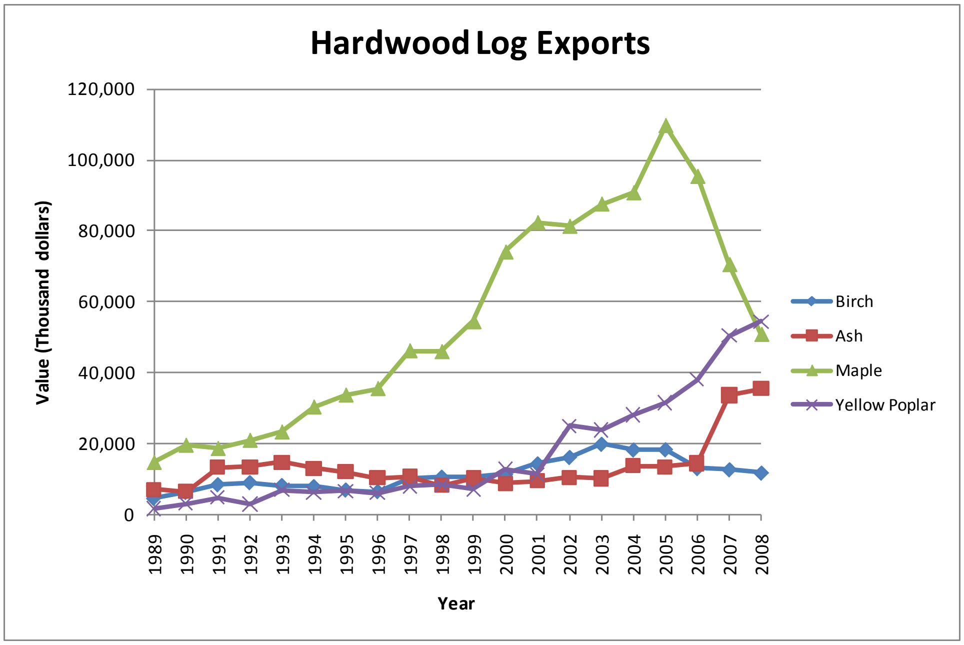

Figure 3–2. U.S. hardwood log exports for select ALB-host species, 1989–2008. (Source: (FAS, 2009)

accepts the largest percentage of these exports at 33 percent, followed by Canada at 25 percent, European Union countries at 12percent, Vietnam at 8 percent, Japan at 4 percent, and Thailand, Turkey, and Hong Kong at 2 percent each (GTIS, 2013).

3. Maple Syrup

In addition to timber production, ALB-host trees (e.g., maple) provide economic returns through maple syrup production. In 2013, total maple syrup production in the Northern region exceeded $131 million, with 3.5 million gallons collected (tables 3–6 and 3–7). Production in 2013 was greatest in Vermont with 1.4 million gallons ($49.4 million), followed by New York with 574,000 gallons ($25 million), and Maine with 560,000 gallons ($17.9 million) (tables 3–6 and 3–7). Other Northeastern States producing maple syrup include New Hampshire, Pennsylvania, Massachusetts, Connecticut, and Ohio, each of which produces 200,000 gallons or less annually. Demand in the United States has been increasing; currently only 0.4 percent of tappable maples are being utilized, suggesting that economic returns and production could increase if demand continues (Farrell and Chabot, 2012).

Table 3–6. Maple Syrup Production 2010–13.

	Production (*1,000 Gallons*)			
	2010	**2011**	**2012**	**2013**
Vermont	890	1,140	750	1,480
New York	312	564	360	574
Maine	315	360	360	560
New Hampshire	87	120	76	124
Pennsylvania	54	128	96	134
Massachusetts	29	62	40	63
Connecticut	9	17	11	20
Ohio	65	125	100	155
TOTAL	1,960	2,794	1,908	3,523

Source: U.S. Crop Production, June 2014 USDA–NASS and USDA–NASS, New England Field Office, June 2014.

4. Tree Nurseries

The tree nursery industry grows ALB-host trees. Between 2005 and 2009, producers in Pennsylvania sold 479,950 deciduous shade trees (not all were hosts of ALB) for $12.4 million; producers in New Jersey sold 300,119 trees for $22.2 million; producers in New York sold 363,008 trees for $12 million; and, producers in Ohio sold 218,341 trees for $16.6 million (USDA, 2009). Within four Northeast States (Connecticut, New Jersey, New York, and Pennsylvania) there were 216 producers with over $56 million dollars in gross sales in 2006 (USDA, 2009).

5. Non-Market Factors

Non-market factors relate to the benefits of ALB-host trees in residential and developed forest communities. Aesthetic values for residents and tourists; use values from recreation activities such as hiking, hunting, bird watching, and fishing; and ecosystem values from watershed services and carbon sequestration are examples of non-market factors related to urban and forested areas where ALB-host trees occur. Recreational activities, including fall foliage and wildlife viewing, hiking, and hunting generate revenue for many States. For example, visitors to Vermont spend $1.719 billion in the State annually, of which approximately 27 percent ($460 million) occurs during the fall season, from September through November (University of Vermont Tourism Data Center, 2012). Direct expenditures of visitors to Maine totaled $4.9 billion in 2012. Approximately 31 percent of trips to Maine occur during the fall season (October through November), which would account to around $1.5 billion in direct spending. Fifty-five percent of Maine visitors reported “beautiful scenery” as a motivation for their visit (DPA, 2013).

Many factors affect tree value and benefits, including species composition, age distribution, condition, amount of canopy cover, and location. Urban trees provide valuable benefits to residents, including air temperature regulation, carbon sequestration, pollution reduction, stormwater runoff reduction, and lowering heating and cooling costs by serving as windbreaks, and casting shade resulting indirectly in the reduction of emissions from energy generation (Huang et al., 1987; Dwyer et al., 1992; McPherson and Simpson, 1995; Akbari et al., 1997; Simpson, 1998; McPherson, 2005). However, trees can also have the opposite effect and increase energy costs by casting shade on buildings in the winter, or blocking cooling winds in the summer (Nowak, 2002).

In a literature review conducted by EPA, studies have found general increases of about 3 to 10 percent in residential property values associated with the presence of trees and vegetation on a property (EPA, 2008). Homeowners generally place a high value on their trees for shade, aesthetics, privacy, investment, and wildlife habitat, and are, consequently, concerned when this resource is threatened.

6. Social Factors

The forestry industry supplies jobs to thousands of people, and is a dominant employer in some communities. In the Northern States, 441,000 workers are employed in forest management, logging, forest products, and pulp and paper industries (Shifley et al., 2012). Other benefits of urban trees to society, that are more difficult to quantify, include increased job satisfaction, sentimental attachment, and improved child development (Kane and Kirwan, 2009).

A person's health is affected by the quality of the environment where they live. Access to greenness varies among demographic groups (Donovan et al., 2013). Trees are an important part of the natural environment, particularly in urban areas where they provide various health benefits to humans (Sarajevs, 2011), in addition to scenic views and environmental benefits.

Trees can reduce stress and have positive physiological and psychological effects on human health (HCN and DAC, 2004; Guite et al., 2006). For example, post-surgical patients had shortened hospital stays and a reduced need for pain-relieving drugs when they viewed trees through their windows (Ulrich, 1984). People recovered faster from stress, mental fatigue, illness, and experienced a long-term overall health and well-being improvement after viewing natural landscapes (Velarde et al., 2007; Lee et al., 2009). Trees reduce noise levels resulting in reduced stress and improved mental health (Sarajevs, 2011).

Trees also decrease human exposure to ultraviolet radiation through shading, which can reduce eye cataracts, and morbidity and mortality from skin cancer (Heisler and Grant, 2000; Heisler et al., 2003; Heisler, 2010; Grant et al., 2002). Greater tree-canopy cover has also been associated with improved birth outcomes, suggesting that trees may affect the health of a pregnant woman and reduce the risk of babies being born underweight (Donovan et al., 2011).

Trees are part of urban green space that provides an environment conducive to physical activity. When there are green areas in the neighborhood, people tend to spend a greater amount of time outdoors and are more physically active (Humpel et al., 2002; Tzoulas et al., 2007). Children are less prone to become overweight when green space is available (Bell et al., 2008). Seniors in urban areas live longer when there are walkable green spaces along streets (Takano et al., 2002). Trees positively affect behavior and reduce crime (Kuo and Sullivan, 2001a; Kuo and Sullivan, 2001b; Taylor et al., 2002).

7. Cultural Factors

Trees affect the air, water, and soil in and around cultural and archaeological resources, as described in the respective sections above. Placement of trees around cultural, historic, and archaeological resources affect the aesthetic quality of the historic resource, and possibly the physical quality by providing wind buffer, shading, particulate adsorption, hydrological functions, and other protective attributes. Trees also can negatively affect cultural resources by damaging resources through falling limbs or trees, root growth, pollen production, and other natural phenomena.

Trees themselves may also be part of the cultural traditions and heritage of various human groups. For example, wood fiber from ash trees is an important material used in baskets made by several Native American communities. Loss of ash trees in the United States from infestations of the emerald ash borer has affected the availability of this core resource to tribal communities (http://www.emeraldashborer.info/files/EABImpactsOnAmericanIndianCommunities.pdf).

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IV. Environmental Consequences

The purpose of this chapter is to discuss the potential impacts related to each of the proposed alternatives. Information presented in Chapter 3, Affected Environment, serves as the baseline for the evaluation of impacts to human health and the environment from the proposed alternatives. The potential impacts reflect those identified in the scoping notice and applicable comments received during the scoping process for this EIS, as well as the impacts evaluated in previous NEPA documentation for the ALB Eradication Program. Due to the programmatic nature of this EIS, the discussion in this chapter is a qualitative evaluation of the impacts of the proposed alternatives. A quantitative approach is used where possible and, in some cases, analysis for a specific geographic area is used as an example to relate the potential impacts of a proposed alternative. The information in this section, and the relevant appendices, are also applicable to a site-specific environmental analysis for an ALB outbreak.

A. Program Alternatives

1. Alternative 1: No Action

a. Forest Resources

Under the no action alternative, a worst-case scenario model where ALB appears all at once in the Northeastern United States without mitigation in place, estimates a loss of 71 billion board feet of wood within 25 years, and death of all host trees within 60 years (Jacobson et al., 2012). This would be a loss of about 45 percent of the forest (Jacobson et al., 2012), followed by succession with non-host plant and tree species that would result in a change in the current forest composition. This vegetation could replace ecological and economic voids left by the loss. For example, the growth of non-host trees may open other wood product markets. ALB has been in the United States for approximately 20 years. Under no eradication or mitigation measures, ALB populations would expand through natural spread. The spread of ALB through human-mediated pathways would also likely occur without quarantine restrictions.

b. Environmental Resources

(1) Water

Tree loss and defoliation of ALB-infested trees have the potential to impact water quality. Many of the preferred hosts for ALB are tree species that occupy riparian and wetland areas. Riparian zones ensure high quality stream habitat for aquatic fauna. Loss of tree cover and density from ALB infestations can have negative impacts on streambank stabilization, water temperature, sediment loading, hydrology (increasing runoff), nutrient cycling, and contaminate removal in aquatic habitats (Wenger, 1999; Lee et al., 2004; Jones et al., 2006). The degradation in water quality from the loss of riparian function can also impact drinking water supplies, which has implications for human health.

(2) Air

Trees intercept pollutants from the air, reducing human exposure and associated risks, such as respiratory illnesses (Beckett et al., 1998; Nowak et al., 2000; Bell and Treshow, 2002; Nowak et al., 2006; Lovasi et al., 2008; Kim and Bernstein, 2009; Tiwary et al., 2009; Donovan et al., 2013). The estimated total annual air pollution removal for ozone, nitrogen oxides, sulfur oxides, carbon monoxide and particulate matter by urban trees in the United States was at 711,000 metric tons (Nowak et al., 2006). One estimate implies that a mature urban tree can intercept up to 50 pounds of particulates per year (Dwyer et al., 1992). Trees infested with ALB reach mortality within 10 to 15 years. During the declining stage, trees continue to intercept air pollutants and sequester carbon dioxide. However, stress to these trees decreases their ability to sequester carbon dioxide (Bréda et al., 2006). The loss of trees from ALB would cause a reduction in the interception of air pollutants and other air quality improvements; however, through natural succession or replanting with non-host trees, the air quality contributions would recover over time.

Trees sequester the gases, including carbon dioxide, that contribute to GHG levels. During photosynthesis, plants and trees absorb carbon dioxide, store carbon above and below ground, and release oxygen as a byproduct. Trees release carbon back to the atmosphere through respiration, decomposition, and burning. Conversion of trees to lumber does not immediately release carbon stores within the tree. In 2009, estimates of the total carbon on forest land in the northern States, including soil organic carbon, was 14,413 million dry tons, which represents about 32 percent of the stored carbon on forest lands and soil in the United States (Shifley et al., 2012). In 2010, an estimated 12.4 billion tons of carbon was stored in southern forests within tree biomass, soil organic carbon, and understory plants above and below ground (Wear and Greis, 2013). Urban forests also contribute to carbon sequestration. In an early estimate, urban forests stored approximately 800 million tons of carbon, nearly 5 percent of live tree carbon storage in all U.S. forests (Dwyer et al., 1992).

Climate change is the global shift in climate and weather from increased temperatures, mostly because of human activity. Carbon dioxide (CO_2) and GHGs trap solar energy in the atmosphere, leading to increases in temperature. Trees, including ALB-host trees, store carbon and play a role in the reduction of CO_2 in the atmosphere. The loss of trees to ALB reduces carbon sequestration, and the decomposition of dead trees release CO_2 into the atmosphere.

Climate change may affect the distribution of host trees and ALB. In a study of Northeastern U.S. forests, 36 out of 80 species assessed show the potential to shift their growing range approximately 62 miles (100 km) to the north, including 7 that could shift greater than 155 miles (250 km) (Iverson and Prasad, 1998). Sugar maple (*Acer saccharum*, 14.5%) and black cherry (*Prunus serotina*, 10.0%) would decline sharply, while oak and southern pines would expand northward. Temperature increases caused by climate change could affect ALB's life cycle. In cold climates, ALB produces one generation every 2 years. In the areas of the United States where ALB has established, the insect typically produces one generation per year. The observance of one generation every 2 years may be possible if ALB establishes in colder climates in the United States. However, future increases in temperature may shift this to one generation per year.

In the extreme scenario where ALB infests all host-tree species in the contiguous United States, (13.9 million metric tons (mt)) of CO_2 would be released over time (depending on the rate of tree decomposition) to the atmosphere in the absence of any eradication (appendix D). If released in a single 1-year pulse, this would represent 37.8 percent of total global CO_2 annual emissions, based on the Year 2010 estimate of 9.1 Gt C/year (33.5 gigatons (Gt) CO_2) global emissions from industrial sources and an estimated total of 10.0 Gt C/year, including land-use change and deforestation (http:// co2now.org/Current-CO2/CO2-Now/global-carbon-emissions.html). It is likely that this would have a significant but transient impact on overall climate change.

Forests are capable of undergoing carbon recovery. Through sequestration in new growth, tree regeneration would recapture the equivalent of the CO_2 released within 76 years (appendix D). Under improved management of tree health and tree cover, the level of CO_2 sequestration in new tree growth could exceed initial losses due to tree mortality, and recapture fully all CO_2 within as little as 43 years (appendix D).

(3) Soil

The loss of host trees in forests where ALB-host trees are dominant would alter soil nutrient cycling, especially in situations where a host tree serves a unique ecological function. For example, maple trees are considered a critical component in soil nitrification in the Northeastern United States; this loss could impact nitrogen retention and cycling in forested watersheds (Lovett and Rueth, 1999; Lovett and Mitchell, 2004).

A decrease in maple stands from ALB infestation, would lead to higher nitrogen retention in soils and reduction in nitrogen transport into aquatic systems. The alteration of nitrogen cycling from the loss of maple would alter plant succession and diversity in terrestrial environments, as well as impact aquatic ecosystems, that are dependent on higher nitrogen inputs. Impacts to nitrogen cycling in forested ecosystems have been reported with other defoliating invasive forest insects (Kizlinski et al., 2002; Lovett et al., 2002). Additional soil impacts may include increased soil erosion and increased soil temperature from the loss of trees, or foliage dieback resulting from repeated ALB infestations.

c. Ecological Resources

The dieback over successive seasons and loss of trees because of ALB infestation could result in a reduction of forested stands where ALB-host trees are dominant, and form canopy gaps in mixed forest habitats where ALB-host trees are present. The creation of canopy gaps and edge forest habitat can adversely impact those species that depend on contiguous blocks of forest for nesting and reproduction. In particular, neotropical migratory birds may be impacted by the loss or alteration of forest stands from ALB infestations. The species of birds impacted vary depending on whether it is dependent on interior forest habitats or edge habitats. Those species dependent on interior forest habitats may be negatively impacted due to the loss of habitat, while edge-dependent species may benefit. The loss of habitat for nesting and foraging impacts reproduction. For example, some neotropical migratory bird species could be negatively impacted by increased nest parasitization and predation from birds (e.g., brown-headed cowbird) in forested areas fragmented by the loss of ALB-host trees (Thurber et al., 1994; Donovan et al., 1995; Suarez et al., 1997; Thompson, 2005).

Impacts to migratory birds vary by species and its habitat requirements, as well as the size of the ALB infestation (Chalfoun et al., 2002). Federal and State natural resource agencies, as well as non-governmental organizations, recognize the importance of habitats to migratory birds; these habitats are protected through bird conservation plans and bird conservation regions (BCRs). Two of those regions where ALB-preferred host plants are prevalent are the Atlantic Northern Forest and the Appalachian Mountain BCRs (FWS, 2006). The conservation plans for these areas identify threats to a range of bird species by invasive species, and forest habitat loss is a concern for multiple species. The added impacts from an ALB infestation on these regions increase the difficulty in implementing effective restoration and habitat protection plans.

As with birds, other terrestrial fauna that depend on ALB-host trees and contiguous forested areas may be negatively impacted. Based on the spatial scale of forest loss, these impacts vary and are dependent upon the specific habitat requirements of the species. Canopy gaps or open areas resulting from an infestation may benefit some species while selecting against others that depend on ALB-host trees or depend on contiguous forest stands. Certain species may benefit from ALB-infested trees, such as secondary wood-boring pests and associated vertebrates that depend on snags for habitat and prey.

A reduction in canopy cover or complete loss of ALB-host trees would favor understory vegetation. This would benefit species that depend on fragmented stands, and would create more open riparian areas for foraging and nesting (Bell and Whitmore, 2000). However, potential benefits to certain species from the loss of ALB-host trees are minor compared to widespread terrestrial and aquatic benefits related to the conservation of forested habitats.

In addition to upland areas that could be impacted by ALB infestation, many of the preferred host trees for ALB commonly occur in riparian areas; impacts to these habitats can have negative impacts to bird species, as well as amphibians and other terrestrial, semi-aquatic and aquatic fauna (Naiman and Decamps, 1997; Petranka and Smith, 2005; Perkins and Hunter, 2006). Riparian areas are dynamic and complex habitats that serve as an interface between terrestrial and aquatic ecosystems, providing unique functions that, if significantly impacted, have negative consequences to a wide range of ecological processes within terrestrial and aquatic ecosystems (Naiman and Decamps, 1997). These areas provide biodiversity and habitat for numerous species, and regulate nutrient cycling and microclimate conditions in terrestrial and aquatic ecosystems.

The loss of ALB-preferred host trees that are typical in these areas would impact the function of riparian zone habitats. In cases where ALB-host trees decline or are lost, other plant species grow that may contribute different or lower nutritive qualities, and alter decomposition rates. This can impact aquatic vertebrate and invertebrate species that exist in the particular habitat (Smock and MacGregor, 1988). The magnitude of impacts would vary, depending on the density and abundance of host trees and the plants and animals that rely on them for habitat.

Figure 4–1 shows the types of direct and indirect impacts that could occur to representative nontarget aquatic and terrestrial species that are dependent on ALB-host trees.

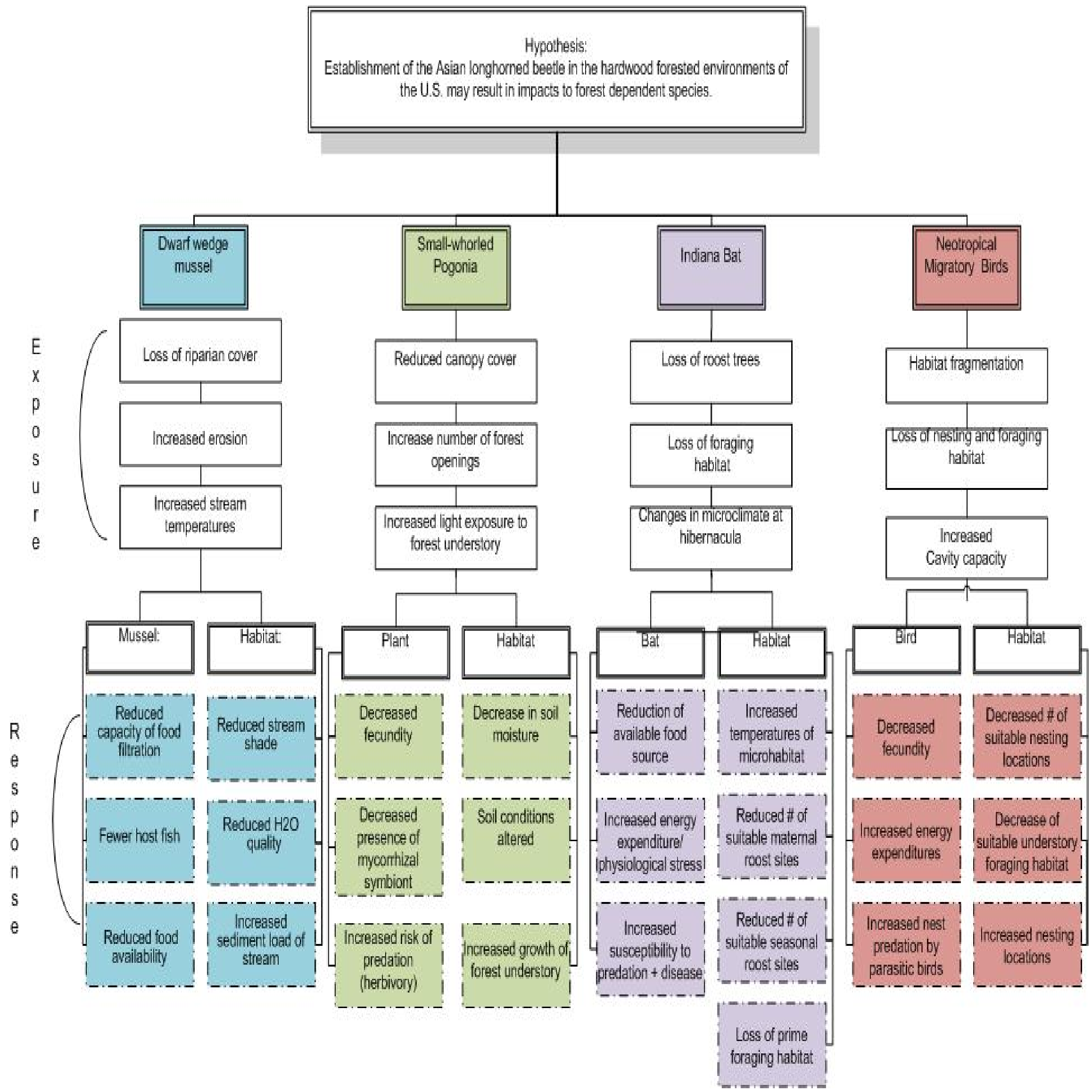

Figure 4–1. Direct and indirect effects of ALB establishment to representative aquatic and terrestrial organisms.

d. Economic, Social, and Cultural Resources

Both market and non-market impacts will occur as ALB-host trees are lost to the insect. Impacts will vary depending on the density of host trees in an area, the rate of spread of the insect, and the rate of decline of the trees. Hardwood trees compose the majority of the tree product industry in the north (Shifley et al., 2012), although not all of these are ALB hosts. Under a worst-case scenario model where ALB establishes in all Northeastern States, the loss of ALB-host trees in the Northeast region of the United States would result in a $20 billion loss of harvestable timber over 100 years, using a 3 percent discount rate to reflect the time value of money (Jacobson et al., 2012). This estimate takes into account the replacement of ALB-killed trees with other harvestable trees.

The Northeastern region accounts for most of the maple syrup production in the United States and, based on the worst-case scenario where all maples would be lost, the maple syrup industry valued at $131 million in 2013 would be lost (NASS, 2014). The tree nursery industry would lose part of their inventory; however, for those nurseries that carry non-host trees or are able to convert their inventory to non-host trees, an economic recovery may occur. Tree industries located within quarantine boundaries could incur additional costs due to restrictions (including treatments) imposed on the movement of ALB-host material.

Countries listing ALB as a quarantine pest would restrict trade of commodities derived from host trees located within quarantine areas. Trees differ in the quality and utility of their wood. If replacement tree species do not meet export market needs, revenue from exports would decline. The United States may increase imports of timber and wood products derived from ALB-host trees should the country lose its domestic source.

Other forest product industries could benefit from the loss of host trees. For example, host trees would supply material for firewood, biofuels, and pulpwood. However, a surplus in supply may cause a decrease in price for the commodity because the supply is greater than the demand.

An immediate disappearance of ALB-host trees is unlikely to occur; rather, the natural spread of ALB is expected to occur slowly, as would the death of host trees, based on observations in the United States and elsewhere. As ALB-host trees disappear, other tree species may take their place, potentially sustaining the forestry industry, perhaps with the exception of the maple syrup industry. ALB is not the only problem affecting the health of Northeastern forests. Other invasive pests and diseases, as well as environmental stress (including that from climate change) and development pressures are affecting forest health throughout the United States. These stressors would be expected to affect forest recovery from ALB.

Research measuring the non-market value of trees is limited. The loss of ALB-host trees, particularly in areas dominated by these species, would change the composition and age of tree stands. Outdoor recreationists prefer areas with older trees (Scarpa et al., 2000; Englin et al., 2006), along with a preference for sites with more deciduous trees versus evergreen trees (Scarpa et al., 2000).

Changes in forest composition, particularly with the loss of maples and other hardwood trees, would negatively affect the fall foliage tourism in the Northeastern United States. Maples and other hardwood trees provide much of the brilliance in fall foliage. For example, in New Hampshire, fall foliage viewing contributes $292 million annually to the State's economy (NH Dept. of Environ. Svcs.). Visitors to Vermont spend $1.719 billion in the State annually, of which, approximately 27.0 percent ($460 million) occurs during the fall season from September through November (University of Vermont Tourism Data Center, 2012).

The loss of mature host trees would diminish the value and aesthetics of residential properties. Several studies show a correlation between both the presence and health of trees on property values (Holmes et al., 2006; Price et al., 2010). Holmes et al. (2006) studied the impact of hemlock wooly adelgid infestations on residential housing prices in northern New Jersey; a positive correlation was found between healthy hemlock trees and housing prices, and a negative correlation between infested hemlock trees and housing prices. In this study, unhealthy hemlock trees, up to 0.62 miles (1 km) away from the property, had a negative effect on housing prices.

A study in Portland, Oregon showed that street trees added an average $8,870 to sales prices, and reduced the time of housing on the market by 1.7 days (Donovan and Butry, 2010). Other positive attributes trees impart to infrastructure include protection from stormwater runoff, temperature regulation, and electricity usage reduction, which would be affected by the loss of host trees. These effects would diminish over time as replacement trees mature.

Human Health

Under the no action alternative, ALB would continue to spread resulting in potential impacts to human health due to loss of trees. Tree loss in urban and rural residential areas was associated with declining cardiovascular and lower-respiratory-tract illness in an ecological study evaluating the relationship between the presence of emerald ash borer and human health (Donovan et al., 2013). Tree loss from the spread of the emerald ash borer among 15 States was associated with 6.8 additional deaths per year per 100,000 adults over the 18-year period for the study (1990–2007) (Donovan et al., 2013). Tree loss contributes to risk factors for cardiovascular and respiratory diseases including stress, lack of physical activity, and poor air quality (Pope III et al., 2003; Everson-Rose and Lewis, 2005; Lucas and Platts-Mills, 2005; Donovan et al., 2013). Uncontrolled tree loss reduces the health benefits of trees discussed in chapter 3, and may cause negative effects on human health.

Tree loss can cause stress and have psychological effects (Velarde et al., 2007; Whitelaw et al., 2008). A comparison study on psychophysiological stress recovery and directed attention restoration of young adults, in natural and urban field settings, showed that natural settings with tree views have a positive impact on improved attention functioning, and lowered blood pressure levels (Hartig et al., 2003). This study found an increased positive affect and decreased anger for individuals in a natural reserve (Hartig et al., 2003). Another study analyzed survey results from 953 individuals in 9 Swedish cities correlated increased participation in outdoor urban, open-green spaces with reduced stress levels (Grahn and Stigsdotter, 2003). (An open-green space is any open piece of undeveloped land with public access that is partly or completely covered with grass, trees, shrubs, or other vegetation.) Stress may also come from other sources, including decreased property values, and increased heating and cooling costs associated with tree loss (APHIS, 2013b). Tree loss can increase human exposure to ultraviolet (UV) radiation by removing the shading effects from trees that provide UV-radiation protection. UV radiation can negatively affect human health, primarily causing skin cancer and eye cataracts. UV radiation can also positively affect human health because it is essential for the synthesis of vitamin D. Vitamin D is required for bone health, and can reduce non-skin cancers.

2. Alternative 2: Removal of Infested Trees

a. Forest Resources

In one study, researchers modeled a worst-case scenario where ALB infests all host trees in the Northeastern United States all at once, with no eradication or mitigation programs in place. In this scenario, the Program would need to remove host trees from approximately 45 percent of the forested area (not including urban forests) (Jacobson et al., 2012). However, based on the historical introduction and spread of ALB, as well as APHIS' response to the pest, the percentage of trees removed would be less, and would not occur throughout the United States or the Northeastern States all at once. APHIS has removed woodlot areas in Massachusetts and Ohio, resulting in the removal of hundreds of trees from one location. Urban forest cover would diminish if ALB-host trees dominate streets, parks, or residential plantings, as observed in cities and townships in Illinois, New York, New Jersey, Massachusetts, and Ohio. Regrowth of trees on woodlots through natural succession would take many years. Similarly, recovery of urban tree canopy cover will also take some time as replacement trees will likely be with immature non-host trees.

Canopy cover influences understory plant composition, structure, and diversity (Oliver and Larson, 1996). Understory vegetation serves an important role in ecosystem health, biodiversity, and nutrient cycling, as well as contributions to soil and water quality in forests and wooded urban areas (Yarie, 1980; Oliver and Larson, 1996). Many vertebrate and invertebrate species depend on understory vegetation (Oliver and Larson, 1996; Koide and Wu, 2003; Pineda et al., 2005). The removal of trees infested with ALB would reduce the canopy cover, creating an environment favorable to shade-intolerant vegetation.

The process of removing trees can damage surrounding vegetation. Felled trees, vehicles, and other tree removal equipment can compress vegetation and soil. The introduction of weeds and invasive plant species on equipment could alter the vegetative understory. In some instances, the Program applies an herbicide to stumps to prevent sprouting, although the preference is to grind the stumps in place. During recent ALB eradication efforts, greater than 75 percent of the stumps were removed rather than being treated with an herbicide. While herbicide application is directly on the stump surface, and according to label instructions, damage to nearby vegetation could occur from drift or runoff.

b. Environmental Resources

(1) Water

The removal of infested trees near aquatic resources can impact water quality. In particular, the movement of soil into aquatic resources (rivers, lakes, and other bodies of water) can result in sedimentation, excessive nutrients (eutrophication), increased turbidity or cloudiness, and alteration of stream flow. In addition, tree removal adjacent to aquatic resources can reduce shading, which is important in maintaining water temperature.

Degradation of water quality due to sedimentation can result in negative effects to aquatic organisms through direct or indirect impacts to fish, aquatic insects, and crustaceans, such as freshwater mussels and crayfish (Richter et al., 1997; Henley et al., 2000). The risk to soil quality and aquatic resources from erosion, due to tree removal, can be reduced by the implementation of timber BMPs (Aust and Blinn, 2004).

(2) Air

As described in the environmental consequences for the no action alternative, the loss of trees to ALB infestation could affect air quality and contribute to climate change. The degree to which air quality and climate change are affected depends on the human-facilitated contribution of air pollutants, as well as the number of trees infested with ALB.

Trees infested with ALB reach mortality within 10 to 15 years. During the declining stage, trees continue to intercept air pollutants and sequester CO_2; however, stress to these trees decreases their ability to sequester carbon dioxide (Bréda et al., 2006). Removal of trees prior to mortality reduces the interception of air pollutants and sequestration of carbon dioxide. Trees release stored carbon during decomposition or burning.

Under this alternative, the Program removes and chips infested trees. Wood chips decompose at a faster rate than intact woody material, resulting in a faster release of carbon dioxide (McPherson and Simpson, 1999). Replacement of trees through succession or planting would restore air quality attributes to the area; however, this would occur slowly over time and would vary depending on the types of species that may regrow in areas.

The loss of trees through the removal of ALB-infested trees by the Program would temporarily affect the local carbon sequestration. For example, the potential total CO_2 release estimate from trees and soil at five ALB eradication sites active in 2012 is 20,187 mt (appendix D). These levels are below the CEQ reference level of 25,000 mt for all GHGs; other GHGs (methane, nitrous oxide, hydro-fluorocarbons, perfluorocarbons, and sulfur hexafluoride) were not included in this study. The restoration of vegetation and trees reduce the contribution of CO_2 to the atmosphere from the removal of trees by the Program.

In urban areas, where trees now shade buildings, tree removal under ALB eradication would increase energy requirements and emissions of GHGs from power plants to compensate for increased heating in winter and air conditioning in summer. For example, in New York City, trees are estimated to reduce energy costs from residential buildings by $11.2 million annually based on 2002 energy costs. Trees also provide an additional $167,000 in value per year by reducing the amount of carbon released by the city's fossil-fuel based power plants (a net reduction of 9,100 tons of carbon emissions) (Nowak et al., 2007).

Several U.S. cities have greening programs (e.g., Boston (http://www.growbostongreener.org/gbg/) and Baltimore (http://www.baltimoretreetrust.org/) aimed at increasing tree cover to achieve benefits trees provide, including improved air quality, sequestration of carbon, reduced energy consumption, and flooding control (Nowak et al., 2007; Nowak et al., 2010). The removal of infested trees prior to mortality in urban areas and adjacent forest would negatively affect these benefits. However, infested trees weaken over time and eventually fail to benefit the urban environment; rather, they may become a fall hazard. In addition, leaving infested trees leads to additional tree infestations due to ALB spreading resulting in more tree loss.

The combustion of gasoline and diesel fuel in machinery used to remove and chip trees release air pollutants and GHGs (McPherson and Simpson, 1999). Estimates of release for these activities are scarce, including data from ALB Eradication Program activities. Emissions from tree removal and chipping activities would have the greatest impact to air quality in urban areas where air quality may already be impacted.

(3) Soil

Soil quality impacts under this alternative (Alternative 2: Removal of Infested Trees) would be similar or, in some cases, more significant than those described under the no action alternative. Under this alternative infested trees would be removed, which could result in physical and chemical impacts to soils, especially in areas where soils are vulnerable to erosion. The removal of only infested trees without addressing host trees nearby at risk of infestation could also allow ALB to continue to spread, resulting in additional tree removals in areas where soils may be susceptible to erosion.

c. Ecological Resources

The removal of infested trees would have impacts to ecological resources similar to those described under the no action alternative, although at a potentially slower rate in the long term. Initially the rate of tree loss under this alternative would be greater in the infested areas compared to the no action alternative since trees would be removed more quickly than through natural loss from ALB. In cases where infested trees are removed, some of the impacts previously noted could occur. However, the expansion of ALB resulting from not removing high-risk host trees within the known dispersal range of ALB would leave those trees vulnerable, and infestations would continue to occur, resulting in additional removal of infested trees.

Herbicide Use

APHIS uses herbicides when there are limitations to physical removal of stumps. The limitations include those areas that are inaccessible to equipment used for stump grinding, and those areas that are sensitive to erosion or compaction.

The Program uses the herbicide triclopyr by spraying or painting the root collar area (the sides of the stump) and the outer portion of the cut surface, including the cambium (thin layer of generative tissue lying between the bark and the wood of a stem), until thoroughly wet, but not to runoff. Foliar applications of triclopyr mixed with two other herbicides, imazapyr and metsulfuron-methyl, would be applied to sprouting foliage from stumps that remain after tree removal to prevent regrowth.

Triclopyr triethylamine salt (TEA) toxicity to terrestrial wildlife is considered low. Toxicity to avian species is low for triclopyr TEA with oral and dietary median lethal toxicity values greater than the highest test concentrations tested (EPA, 1998; Durkin, 2003). Chronic toxicity to birds is also expected to be low with reproductive toxicity. The no observable effect levels (NOEL) are 100 and 500 parts per million (ppm) for the mallard and bobwhite quail, respectively, when exposed to triclopyr acid (EPA, 1998). Available avian toxicity data for triclopyr butoxyethyl ester (BEE), another triclopyr product available for use by the Program, demonstrates slight toxicity, with median lethal dose values ranging from 735 to 849 mg/kg for the bobwhite quail (EPA, 1998). These values are well above any residues that would occur due to Program applications. Triclopyr TEA is not toxic to honey bees based on acute contact studies (EPA, 1998). Triclopyr TEA does exhibit toxicity to some terrestrial plants based on results from seedling emergence, germination, and vegetative vigor studies. The primary degradation product of triclopyr TEA, triclopyr acid, is similar in toxicity to terrestrial nontarget organisms based on the available toxicity data.

TEA toxicity to aquatic organisms is low for fish and aquatic invertebrates. Available acute fish toxicity data demonstrates median lethal concentrations greater than 100 mg/L for Garlon® 3A and technical triclopyr TEA (Wan et al., 1987; EPA, 2014a). Triclopyr TEA is not considered toxic to aquatic invertebrates in freshwater and marine environments, with toxicity values exceeding 300 mg/L. Chronic toxicity to fish and aquatic invertebrates is also low with chronic toxicity no observable effects concentration (NOEC) ranging from approximately 80 mg/L to greater than 100 mg/L, depending on the test organism and endpoint. Although Triclopyr BEE may be toxic to aquatic invertebrates and fish, they will not be exposed to levels that could result in adverse effects from applications made by the Program. The primary metabolite of triclopyr TEA and BEE, triclopyr acid, is not considered toxic to aquatic organisms, based on available toxicity data (EPA, 1998, 2014a).

For foliar treatments, Garlon® 3A is proposed for use as a mixture with the active ingredients imazapyr and metsulfuron-methyl. Imazapyr is an imidazolinone herbicide while metsulfuron-methyl is a sulfonylurea herbicide, with both products used as a mixture with triclopyr in the control of woody vegetation.

The toxicity of imazapyr and metsulfuron-methyl is considered low for mammals. The formulation containing metsulfuron-methyl, Escort® XP, is not considered toxic to mammals via inhalation, dermal, and oral exposures. All toxicity values were reported as greater than the highest test concentration. In addition, metsulfuron-methyl is not considered to be carcinogenic, nor has it been shown to be a reproductive, teratogenic, or developmental hazard (Klotzback and Durkin, 2004). Escort® XP is considered a slight eye irritant, but is not considered a skin irritant or sensitizer. The other herbicide in the mixture, Arsenal®, containing the active ingredient imazapyr, has a similar mammalian toxicity profile to metsulfuron-methyl, and is considered practically nontoxic in acute inhalation, dermal, and oral exposures. Imazapyr is not considered a carcinogen or mutagen, and is not known to be a reproductive, teratogenic, or developmental hazard (Durkin and Follansbee, 2004).

The toxicity of imazapyr and metsulfuron-methyl is low to all nontarget organisms, with the exception of some aquatic and terrestrial plants. Neither product is considered toxic to mammals, birds, or terrestrial invertebrates (Durkin and Follansbee, 2004; Klotzback and Durkin, 2004; EPA, 2014a). Toxicity to fish and aquatic invertebrates is very low, with median lethal acute concentrations typically exceeding 100 mg/L for both chemicals (Durkin and Follansbee, 2004; Klotzback and Durkin, 2004; EPA, 2014a). Chronic toxicity to fish and aquatic invertebrates is also considered low, based on the available NOECs that were reported from standardized toxicity studies.

Exposure to terrestrial and aquatic nontarget organisms is expected to be minimal from each proposed formulation and mix. Significant drift or runoff is not expected as applications are not broadcast applied, but are made using either a backpack sprayer to deliver a coarse droplet size, or by brushing the material on individual stumps and associated sprouting vegetation. The low probability of offsite transport for any of the products results in very low exposure to most nontarget organisms. The low probability of exposure and the favorable available effects data demonstrate that all products have a very low risk of causing adverse ecological risk (see appendix E). Risk to nontarget organisms is greatest for plants as they are the most sensitive group to each application; however, the application methods and label directions minimize impacts to terrestrial plants, restricting potential harm to those plants that are immediately adjacent to treated stumps or sprouts. Exposure in aquatic systems is not expected to occur at levels that could result in any direct impacts to aquatic plants, or at levels that would suggest indirect impacts to aquatic organisms that depend on aquatic plants as a food source or as habitat. (Appendix E provides the risk assessment for herbicides the Program proposes to use.)

d. Economic, Social, and Cultural Resources

As described in the environmental consequences section for the no action alternative, ALB-host trees are important to the forestry products industry and their loss, particularly on forested lands, would result in negative economic impacts. The Program removes and chips infested trees, making them unavailable to the timber and forestry products industry. ALB larvae create tunnels or galleries inside the tree, damaging the structural integrity of the wood. It is possible that lightly infested trees could have salvageable wood for timber and other end-use products; however, the Program does not allow the diversion of infested trees from chipping to saw mills because of the risk of spreading ALB.

One maple tree can produce 10 to 60 gallons of sap for maple syrup in one season, depending on the tree (including size), weather conditions, length of sap season, and the method of collecting sap (http://maple.dnr.cornell.edu/index.html). Maple syrup producers can absorb a loss of a percentage of maple trees but, depending on the size of the producer, there is a threshold where economic loss would shut down the business. APHIS does not recommend replacing maple trees with ALB-host trees in an area infested with ALB; therefore, replanting would not be an option for maple producers.

The impact to landowners is similar to those described under the environmental consequences for the no action alternative. However, the Program will remove infested trees rather than leaving them to die in place. In the early stages of infestation, trees can appear healthy and continue to provide the aesthetic qualities and other benefits. Trees decline and die at different rates, depending on type of tree, its size, the population of ALB, exposure to other stressors, and other factors. Symptoms occur in approximately 3 to 4 years after infestation, and tree death can occur in 10 to 15 years, depending on site conditions. The Program would remove trees at all stages of infestation. This alternative would slow the spread of ALB; however, due to the difficulties in identifying infested trees, it is likely that some infested trees would be missed and, therefore, ALB would spread.

(1) Human Health

Under this alternative, the overall rate of tree loss from ALB infestation is expected to be reduced from the no action alternative because the Program removes infested trees, which reduces the ALB population and spread to other host trees. However, tree removal activities would initially cause an increase in tree loss, compared to the no action alternative where infested trees remain in place to die from ALB-infestation. In the short term, tree removal activities may result in increased noise levels (from use of mechanical equipment and increased traffic), increased stress (from decreased property values), increased cooling and heating costs, and other localized negative human health consequences from a lack of trees, (as discussed in the no action alternative) to the general public living in the infested areas. In the long term, the negative human health consequences could be less than those compared to the no action alternative because the overall tree loss is expected to decrease to some degree.

(2) Herbicide Use

APHIS evaluated the risk to workers and the general public from the Program use of the three herbicides (appendix E). Two triclopyr formulations, Garlon® 3A (active ingredient is TEA) and Pathfinder® II (active ingredient is BEE), for the treatment of stumps were analyzed. Pathfinder® II is used to treat the bark instead of direct application to cut areas of the stem. Minor foliar applications of Garlon® 3A mixed with two other herbicides, imazapyr and metsulfuron-methyl, are used to treat sprouting foliage from stumps that have been removed as part of the eradication program. The applications are made by hand either by brushing undiluted material on the stump or directly spraying stumps and/or sprouting foliage using a backpack sprayer. The TEA formulation can cause significant eye irritation, but has low acute inhalation and dermal toxicity. Acute oral median lethal concentrations range from approximately 600 to 1,000 mg/kg, suggesting low to moderate toxicity (Durkin, 2003).

Long-term toxicity studies have shown that triclopyr TEA is not a carcinogen or mutagen, and that toxicity in developmental and reproductive studies primarily occurs at high doses, and at levels that are also maternally toxic (EPA, 1998). The concentrations at which these effects have been reported would not occur under normal program uses.

The other proposed BEE formulation, Pathfinder® II, can cause slight temporary eye irritation during application, and some skin irritation under prolonged exposure. Acute oral median lethal concentrations are 1,000 mg/kg, with acute inhalation and dermal toxicity median lethality values greater than the highest test concentration, suggesting low acute mammalian toxicity under various exposure pathways. Triclopyr BEE is not considered carcinogenic or mutagenic and, in cases where developmental and reproductive studies demonstrate effects, doses were at levels considered maternally toxic. The concentrations at which these effects have been reported would not occur under normal program uses.

TEA breaks down in soil (~12 days) to triclopyr acid, and to a lesser extent, triethanolamine. Triclopyr BEE has low water solubility, and adsorbs more strongly to soil when compared to the amine. Triclopyr BEE also breaks down quickly to triclopyr acid in soil and water, with hydrolysis half-lives of less than 1 day. Imazapyr degradation and dissipation half-lives are variable, ranging from approximately 25 days to greater than 300 days. Metsulfuron-methyl half-lives in soil range from 17 to 180 days. The use of these herbicides may negatively affect sensitive individuals or those who inadvertently become exposed; however, these herbicides are relatively short lived in the environment so any health effects from herbicide use are likely to be temporary.

The human health risk assessment results show that using triclopyr, or using triclopyr and mixing it with imazapyr and metsulfuron-methyl to control ALB populations should pose minimal risks to human health for workers and the general public (appendix E).

In addition, notification to landowners also occurs in the case of any chemical treatments that may be used to kill stumps (herbicides), as well as other label risk reduction requirements for herbicide use that are designed to protect workers, the general public, and the environment.

3. Alternative 3: Full Host Removal

a. Forest Resources

Host trees within a ½-mile radius of an infested tree are at risk of ALB infestation because they are within the dispersal range of the beetle. Under the full host removal alternative, removal by the Program of both infested trees and high-risk host trees would occur. Depending on the density of host trees in the quarantine area, this will likely translate to the removal of a larger population of trees compared to the no action alternative and two of the eradication alternatives. The intent is to protect urban and rural forests from ALB through the removal of host trees potentially infested with ALB but not at detectable levels.

The impacts to forestry resources, as described in alternative 2, could also occur under this alternative. Trees infested with ALB typically die within 10 to 15 years. Full host removal by the Program prematurely eliminates the ecological services the trees provide. Host trees provide habitats and food for wildlife, and contribute to nutrient and water cycles. The long-term benefits of full host removal in a quarantine area include the reduction in the beetle's spread rate and eradication of the beetle.

Removal of host trees classified as invasive in the United States would be beneficial. For example, several Northeastern States designate the Norway maple as an invasive species.

b. Environmental Resources

(1) Water

Impacts to water quality from the selection of alternative 3 would be increased when compared to alternative 2. The removal of infested trees and high-risk host trees would result in soil sediment runoff to aquatic areas. Current water quality data for rivers and streams in the

United States lists sediments as the second leading cause of impairment under Section 303(d) of the CWA (EPA, 2014c). Sediments are the ninth leading cause of impairment in lakes, reservoirs, and ponds. The source of sediments causing impairment in waterways varies; however, silviculture-related activities, such as harvesting and forest road construction, are contributing factors. These impacts would be more likely in forested areas where large numbers of trees may be removed as a result of an ALB infestation. Replanting vegetation and following silviculture BMPs will protect soils vulnerable to erosion, reducing potential for impacts to water quality.

(2) Air

Under this alternative, the potential removal of a greater number of trees could cause greater impact to air resources compared to those described in Alternative 2—Removal of Infested Trees. Loss of all host-tree species across the contiguous United States is estimated to release 13.9 million mt of CO_2 to the atmosphere (appendix D). This large pulse of CO_2 to the atmosphere is unlikely as ALB spreads slowly, trees become infested at different times, and trees die at different rates. Large urban and forested areas under quarantine with a high density of ALB-host trees would see impacts on the local air quality.

Forest areas are typically a mixture of tree species. For example, in the Northeastern United States, forests are typically a mixture of hardwood and hardwood-conifer (Shifley et al., 2012); full host removal would not result in complete deforestation of an area. Emissions from tree removal and chipping activities could result in some localized impacts to air quality, but these would be reduced over time as eradication efforts are implemented.

(3) Soil

This alternative would result in potentially greater impacts to soil quality, compared to the other eradication alternatives, because both infested trees and high-risk host trees would be removed. Impacts to soil would be greatest where large numbers of trees are removed from a concentrated area.

Changes in soil temperature and moisture, as well as soil erosion and loss of nutrients in areas, can impact the ability of a forest to regenerate (Ballard, 2000). These impacts are more prevalent in cases where clear-cutting is being used for forest harvesting; however, that type of removal is not likely to occur in the Program because other non-ALB-host trees would be left standing.

Compaction from the use of heavy equipment may result in increased soil bulk density values that may limit regrowth of vegetation in areas where trees are removed. These physical impacts to soil may result in increased erosion of soil from wind and rain, both during and after tree removal. Seeding areas with grass or other vegetation reduces these types of soil quality impacts.

c. Ecological Resources

Overall impacts to ecological resources, under alternative 3, are expected to be less than those described under the no action alternative and alternative 2 because of the prevention of ALB dispersal to non-infested areas within the United States. Impacts on a local level may be more significant to ecological resources because removal of infested and high-risk host trees may result in some fragmentation, as well as a reduction in tree density within riparian zones. While these impacts would be more localized, compared to the loss of trees resulting from the no action alternative, the impacts would be more immediate as trees would be removed at a more rapid rate than if they were lost to ALB. The extent of ecological impacts would be dependent upon the size of the infestation and the predominance of ALB-host trees in natural and urban areas. The density of host trees will likely be greater in forest areas than urban settings, potentially causing greater impacts to ecological resources in forest areas.

d. Economic, Social, and Cultural Resources

Full host tree removal would result in the removal of more trees within the quarantine area compared to the other three eradication alternatives. The extent of the impact depends on the density of host trees within the area and the intended use of the host trees. Significant impacts could occur if a quarantine area overlaps with a woodlot, forest, tree plantation, maple syrup production area, or other commercial forestry area dependent upon hardwood trees that are hosts to ALB.

Urban areas predominantly planted with ALB-host trees would see a reduction in tree cover. This may result in an initial reduction in aesthetic qualities of the landscape, as well as the ecosystem services trees provide; this includes reduction in water and sediment runoff, interception of air pollutants, and buffering from solar radiation and wind.

Restoration of areas through the planting or natural regrowth of non-host trees and other vegetation may eventually lead toward recovery of resources. Suppression in the tourism industry may occur if a high proportion of the ecological resource (e.g., park, community, etc.) falls within the quarantine area. Full host removal is expected to have less of an economic impact compared to the no action alternative because this alternative slows the spread of ALB and leads to eradication. Compared to Alternative 2—Removal of Infested Trees, Alternative 3—Full Host Removal is expected to produce greater short-term economic impacts because of the potential removal of more host trees. However, in the long-term, it is expected that impacts would be reduced because of the protection of forest resources.

Human Health

Alternative 3 would result in increased efficacy to eradicate ALB, and prevent its spread to new areas. In the short term, tree loss would occur from removal of infested and surrounding host trees. These removals could result in localized negative human impacts related to tree loss, such as increased stress, reduced air quality, and so on, as discussed in the no action alternative. However, in the long term, the rate and spread of tree loss due to ALB would be reduced, minimizing human health related impacts.

The potential for human exposure and risk to herbicide use is the greatest under this alternative because more trees are being removed and herbicide use would be expected to increase. Human health risks are still expected to be low based on the herbicide risk assessment included in appendix E of this EIS. In some cases, stumps may be left to allow for regrowth in areas where ALB reinfestation would not occur. When stump treatment is needed, grinding is the preferred method over herbicide treatments; therefore, while there would be an increase in herbicide use, it would not necessarily be proportional to the increase in the number of trees removed.

4. Alternative 4: Infested Tree Removals and Insecticide Treatment of High-Risk Host Trees

a. Forest Resources

Under this alternative, the Program removes host trees infested with ALB, and treats the high-risk host trees located within a ½-mile radius of infested trees with the insecticide imidacloprid. APHIS uses the insecticide imidacloprid through trunk or soil injections to protect trees from ALB infestation. The Program applies insecticide treatments in the fall, spring, and early summer prior to and during the adult emergence period. The impact to forestry resources from this alternative include the impacts described under forestry resources for Alternative 2: Removal of Infested Trees.

In addition to those impacts, the treatment of high-risk host trees with imidacloprid within a ½-mile radius of infested trees may affect beneficial insects (e.g., honey bees) associated with host trees. Research is ongoing to understand these interactions and their impacts on tree health and reproduction. Research indicates imidacloprid and the application methods do not affect tree growth. Treatment of high-risk host trees with imidacloprid may protect these trees from ALB infestation, and prevent their removal by the Program. Treatment helps preserve high-risk host trees in the quarantine area and the ecosystem services they impart.

b. Environmental Resources

(1) Water

The removal of infested trees under this alternative has similar water quality impacts as Alternative 2—Removal of Infested Trees. Under this alternative, insecticide treatment would slow the spread of ALB, however, it will not stop the spread as treatment does not provide 100 percent protection to all trees. In comparison to Alternative 3—Full Host Removal, we expect this alternative to have less of a short-term impact to water quality in regards to host tree removal because the Program would remove fewer trees, at least initially, under this alternative.

Alternative 4 would have the greatest potential for imidacloprid to move into surface or ground water) because chemical treatment is the only option available for high-risk host trees. Imidacloprid exhibits physical and chemical properties that suggest it could contaminate surface and ground water. Detections in ground water have occurred in various parts of the United States, including States where ALB is present. Solubility and a lack of affinity for binding to soil or sediment suggest that imidacloprid could move offsite through runoff or leaching (appendix F). The ability to leach into ground water would depend on site-specific conditions, such as soil type and depth to the water table. However, label restrictions regarding applications near surface water and other information regarding ground water reduces the potential for water contamination. In addition, the preferred use of tree injections of imidacloprid by the Program further reduces the possibility of impacts to water quality when compared to soil injection.

APHIS collected water samples as part of its monitoring effort to determine the potential for imidacloprid to move to surface and ground water from Program applications. (The data is available in appendix F.) The imidacloprid level in a majority of the water samples is below detection in surface and ground water, or below levels that would impact human health or the environment. However, detection of imidacloprid in some water samples is above the level the Program expects from their applications. Due to the widespread use of imidacloprid for other uses, especially home and garden, it is difficult to attribute detections to program applications in urban areas where a majority of program treatments have taken place. The State of New York, after finding imidacloprid in water samples from all uses, restricts the application of imidacloprid to trunk injection.

(2) Air

The impacts on air quality from the removal of infested trees (as described in Alternative 2—Removal of Infested Trees) are the same for this alternative. Treatment of ALB-host trees with imidacloprid usually occurs through trunk injection, however, sometimes through soil injection. The trunk injection method involves injecting imidacloprid directly into the tree trunk; this is unlikely to affect air quality due to minimal exposure of the insecticide to the ambient air. A minor amount of the insecticide volatilizes during the soil injection until full adsorption into the soil and uptake by the plant roots occurs. Drift is not expected to impact air quality because both application methods would not have any associated drift. The chemical and physical properties for imidacloprid suggest that there is a low probability of imidacloprid volatilizing into the atmosphere and impacting air quality (see appendix F).

(3) Soil

Physical impacts to soil from alternative 4 would be expected to be less than those described in alternative 3. Tree removal would not occur for high-risk host trees reducing the potential for soil disturbance that could occur during removal. There is the potential for impacts to soil as a result from imidacloprid treatments of high-risk host trees. Impacts to soil quality would be greatest for soil injections compared to tree injections.

Sensitive soil terrestrial invertebrates would be impacted in the immediate area of treatment in the case of soil injection applications; however, soil injections rarely occur, and tree injections would minimize these types of impacts. Imidacloprid negatively affects earthworms in soil at concentrations that have been observed in previous ALB eradication efforts using soil injections; however, reported levels do not appear to cause long-term impacts to soil microbes (appendix F; (Tu, 1995; Kreutzweiser et al., 2008a)).

c. Ecological Resources

The impacts to ecological resources, under alternative 4, is expected to be less than those described under Alternative 1—No Action Alternative and Alternative 2—Removal of Infested Trees because of the prevention of ALB dispersal to non-infested trees. Impacts to ecological resources from fragmentation and loss of trees in riparian areas would be less than alternative 3 because ALB high-risk host trees would receive an insecticide treatment rather than being removed. Under this alternative, ecological impacts from tree removal would be reduced; however, there is the potential for impacts from the use of the insecticide, imidacloprid, to these same resources that would not occur under alternatives 1–3. These impacts are summarized in the following section with a more detailed analysis in appendix F.

There may be circumstances where imidacloprid use is not practical and, under this alternative, the inability to remove high-risk host trees would allow for the spread of ALB to other trees. In addition, imidacloprid efficacy is variable as a prophylactic treatment for ALB (Poland et al., 2006). Site-specific conditions, regarding tree health and other factors, can impact the uptake and distribution of imidacloprid in trees, allowing ALB to survive even within imidacloprid-treated trees. In these situations, ALB would be able to spread, with the possibility of additional infested tree removal and insecticide treatments, increasing the risk to terrestrial and aquatic ecological resources.

Insecticide/Herbicide Use

Imidacloprid has low to moderate acute toxicity to wild mammals, based on the available toxicity data. Imidacloprid is considered toxic to birds with acute oral median toxicity values ranging from 25 to 283 mg/kg (Anatra-Cordone and Durkin, 2005; EPA, 2014c). Reproduction studies using test species to support pesticide registration (e.g., the mallard and bobwhite quail) have shown NOECs of 125 ppm for both species. The application method and the available effects data indicate low exposure and risk to terrestrial vertebrates (e.g., birds and mammals) (appendix F). Applications using trunk and soil injection remove the risk of exposure from drift or runoff. There is the possibility of imidacloprid exposure to mammals and birds that may feed on insects or vegetation from treated trees. APHIS measured imidacloprid leaf and twig residue values, and demonstrated that most birds and mammals would have to consume several times their daily intake to reach an adverse effect. The risk to terrestrial vertebrates consuming insect prey with residues of imidacloprid is unknown; however, as these prey do not forage exclusively on treated trees and residue levels are not enough to kill them, the risk to predators is likely low. Imidacloprid is also specific to certain groups of insects, and would not affect all insects that are present on treated trees.

Technical and formulated imidacloprid is acutely toxic to honey bees, and other related bee species, at exposures of 3.7 to 230 nanograms (ng)/bee by oral and contact exposure (Schmuck et al., 2001; Tasei, 2002; Anatra-Cordone and Durkin, 2005; EPA, 2014a). Acute sublethal effects in laboratory studies show NOECs at less than 1 ng/bee (Anatra-Cordone and Durkin, 2005). Imidacloprid metabolite toxicity to honey bees is variable with some of the metabolites having equal toxicity to imidacloprid, while other metabolites are practically nontoxic (Anatra-Cordone and Durkin, 2005). Several studies have been conducted to determine potential sublethal effects in laboratory and field situations. Studies to assess the effects of imidacloprid on homing behavior, colony development, foraging activity, reproduction, wax/comb production, and colony health, as well as other endpoints, generally reveal effects at levels above those measured in nectar and pollen in the field from agricultural crops under various application methods (appendix F).

Impacts to honey bees from sublethal exposure to imidacloprid in the presence of other stressors have also been evaluated in laboratory studies. These studies suggest that pesticides, such as imidacloprid, in combination with pathogens may impact colony health and immune function in honey bees (Alaux et al., 2010; Vidau et al., 2011; Pettis et al., 2012). Due to the uncertainty of the risk to honey bees from the proposed treatments in this program, APHIS provided funding for a multiyear study to determine the potential for exposure and effects to honey bees from the proposed treatments (Johnson, 2012). Results from that work suggest that these types of applications do not adversely impact honey bees and their hives, and that imidacloprid residue pollen levels collected from maple trees is low, with an average concentration of 5.3 micrograms (µg)/kg from tree injection applications, and 0.28 µg/kg from soil injections of imidacloprid. Residues of imidacloprid and six associated metabolites were reported as below detection in nectar samples.

Imidacloprid exposure to terrestrial invertebrates from the ALB Eradication Program, especially honey bees, is not expected to result in significant risk to pollinators. Pollinator exposure to imidacloprid is reduced because only treated trees and their associated flowers and pollen could have residues, while other flowering plants that have not been treated would not contain residues. Exposure and risk would increase in cases where large numbers of trees are treated over large areas prior to flowering, and in cases where only flowers from treated trees are the primary nectar source. This may occur in the Northeastern United States where maple trees bloom prior to many other flowering plants.

Applications of imidacloprid, particularly via soil injection, could expose soil-dwelling invertebrates sensitive to the insecticide; however, environmental fate data indicate the effects are transient (Anatra-Cordone and Durkin, 2005). In cases where imidacloprid is tree-injected, there would be reduced exposure and risk to soil-dwelling terrestrial invertebrates; exposure would occur primarily from leaves that drop from treated trees. These risks would be proportional to the number of treated trees in a given area.

Imidacloprid has low toxicity to aquatic organisms including fish, amphibians, and some aquatic invertebrates. Acute toxicity to fish and amphibians is low, with acute median lethal concentrations typically exceeding 100 mg/L (Anatra-Cordone and Durkin, 2005; EPA, 2014a). Chronic toxicity to fish is in the low ppm depending on the test species and endpoint. Aquatic invertebrates are more sensitive than fish to imidacloprid (Anatra-Cordone and Durkin, 2005; EPA, 2014a) (appendix F).

Imidacloprid exposure in aquatic environments and risk to aquatic biota is low (appendix F). The method of application eliminates the potential for drift and, in the case of tree injections, eliminates the probability of offsite runoff. Another potential pathway of exposure to aquatic organisms is imidacloprid residues in leaf litter from treated trees. Aquatic invertebrates feeding on leaf litter containing imidicloprid residues have measurable sublethal impacts, as well as impacts on decomposition rates (Kreutzweiser et al., 2007; Kreutzweiser et al., 2008a, b; Kreutzweiser et al., 2009). Mortality to some leaf-shredding insects occurred when they were intentionally overdosed at levels higher than they would encounter at typical field applications (Kreutzweiser et al., 2007; Kreutzweiser et al., 2009). Not all plant material available to aquatic decomposers will contain imidacloprid as ALB-host trees are part of a greater ecosystem that will contain other plant and organic material. Exposure and risk to aquatic organisms will increase in situations where large numbers of trees may be treated within a watershed. The risk to aquatic organisms from this exposure can be reduced by not treating trees, or treating a small number of trees, and avoiding treatments close to surface water.

There is a potential for subsurface transport of imidacloprid to aquatic habitats for applications made directly into soil. This exposure will be minimized by only making applications where the ground water table is not in proximity to the zone of injection, and avoiding soils that have a high-leaching potential. Conservative estimates of potential aquatic residues in static, shallow bodies of water from soil injections, based on maximum label rates, demonstrate values that are not expected to have indirect or direct impacts to aquatic biota (appendix F). Actual aquatic residues will be below levels that could harm aquatic biota due to the low chance of moving offsite, which is a result of the method of application and environmental fate of imidacloprid.

The potential impacts to ecological resources from herbicide use would be low and less than those described under alternatives 2 and 3 (appendix E) because fewer trees would be cut down. Expansion of infested areas or new infestations would occur at a slower rate under alternative 4, when compared to alternative 2. This would result in less herbicide use in an infested area, and in new introductions reducing the impacts to aquatic and terrestrial ecological resources. Herbicide use would be greater under alternative 3, compared to alternative 4, because more trees would be removed under alternative 3 (both infested trees and host trees within a ½-mile radius).

d. Economic, Social, and Cultural Resources

The economic impacts, under alternative 4, would be less than those described under alternatives 1 and 2. In relation to preventing the spread of ALB, the economic impacts may be similar to those described under alternative 3 when imidacloprid treatments are successful in serving as a prophylactic treatment of high-risk host trees. However, due to the variability in imidacloprid uptake by trees and other site-specific conditions (e.g., general tree health), insecticide treatments may not provide the level of protection that removal of high-risk host trees would provide. In cases where imidacloprid treatments cannot be successfully used for high-risk host trees, the economic impacts would be greater than those under alternative 3 because ALB would be able to expand its range and infest new areas. In addition, the cost of multiple insecticide treatments to a tree is greater than the costs of removal.

Potential economic losses specific to the use of imidacloprid could occur for the maple syrup industry because imidacloprid label restrictions would render treated trees unusable for syrup production. Imidacloprid is a systemic insecticide, and concentrations of the insecticide will be present in the tree sap. The economic impact to maple syrup producers will be dependent upon the proportion of their maple trees removed from production through either tree removal or treatment with imidacloprid. Similarly, nursery tree producers would incur costs by treating host trees with imidacloprid to protect their inventory, and to meet regulatory requirements for shipping trees outside the quarantine boundary. Sales of nursery trees that are hosts to ALB, regardless of treatment with imidacloprid, may drop as buyers select trees that are not hosts to ALB.

As discussed in alternative 2, the removal of infested trees from public and private lands may result in economic, social, and cultural impacts. The option to be able to treat trees with imidacloprid will be based on an evaluation, by the Program, of site-specific conditions for a given infestation.

The forestry and tourism industries, as well as public and private landowners located near an ALB quarantine area, will gain protection from ALB through the confinement of pest populations, and eradication through removal of infested host trees and treatment of host trees in proximity to infested trees. These benefits will be realized in situations where imidacloprid can be used to successfully provide protection to high-risk host trees.

(1) Human Health

Under alternative 4, in the short term, tree loss would occur from removal of infested host trees. Initially, the number of trees lost from removal activities is similar to alternative 2 as only infested host trees would be removed in this alternative. Prophylactic treatments with imidicloprid would protect some high-risk host trees and may result in an overall decrease in the removal of infested trees compared to alternative 2. APHIS expects the removal of fewer trees under alternative 4 compared to alternative 3—Full Host Removal. Some localized negative human health consequences from tree loss are expected as discussed in alternative 2. This alternative is less effective than alternative 3 because insecticide treatment is unlikely to stop the spread of ALB resulting in additional tree loss in the future.

(2) Insecticide/Herbicide Use

APHIS uses the insecticide imidacloprid through trunk or soil injections to protect trees from ALB infestation. For trunk injections, applicators drill holes around the trunk, 2 to 6 inches above the soil-wood line. For soil injection, applicators inject imidacloprid, at a minimum of four injection sites spaced evenly around the base of the tree. Application occurs under the soil around the base of the tree, normally no more than 12 inches from the base. No material may puddle or run offsite. In addition, APHIS uses herbicides to treat stumps and roots of felled trees, as described in alternative 2.

APHIS prepared a risk assessment to evaluate the risks for workers and the general public associated with the use of imidacloprid applications using trunk and soil injections (appendix F). Several formulations are available for this particular application. Technical and formulated imidacloprid (Merit® 2F) has low to moderate acute oral mammalian toxicity, with median toxicity values ranging from 400 to greater than 4,000 mg/kg. Acute lethal median toxicity values are typically greater than 2,000 mg/kg and 5.3 mg/L for dermal and inhalation exposures, respectively. Chronic oral exposure toxicity studies in rat, dog, and mouse showed that the rat was the most sensitive test species with a NOEL of 5.7 mg/kg/day. The EPA acute and chronic reference doses (RfD) are 0.14 mg/kg/day and 0.057 mg/kd/day, respectively.

Imidacloprid is a neurotoxic insecticide, based on its mode of action. Available literature for imidacloprid and associated metabolites suggest a lack of carcinogenic, mutagenic, or genotoxic effects at relevant doses.

Developmental, immune, and endocrine-related effects were observed in some mammal studies; however, these effects occurred at doses that would not be expected to occur under program use.

Exposure to imidacloprid is greatest for applicators, but is reduced by following label directions regarding personal protective equipment (PPE). The human health risk assessment quantified risks for occupational worker exposure and general public exposure using a child soil ingestion (pica) scenario. Results from that assessment show that imidacloprid used to control ALB poses minimal risk to human health under both exposure scenarios. The human health risk assessment (included in appendix F) provides details on toxicity, exposure, and risk associated with imidacloprid.

Herbicide risks to human health would be similar to those described under alternative 2. Notification to landowners also occurs in the case of any chemical treatments that may be used to kill stumps (herbicides) or treat high-risk host trees for ALB (insecticide). In addition to the notification process, there are other risk reduction requirements for pesticide use that are designed to protect workers, as well as the general public and the environment.

Another potential exposure pathway for the public is the use of imidacloprid-treated trees for firewood. The levels of imidacloprid in treated trees that could be used as firewood is expected to be low because the insecticide moves to the leaves, and smaller, actively growing branches in the tree where insect feeding is greatest; these parts of the tree would not typically be used as firewood. In cases where trees are treated, their removal would not be expected to occur in the same growing season as treatment, allowing degradation of imidacloprid. In addition, trees harvested for firewood are usually allowed to dry before they are used as fuel, which would allow for additional degradation of imidacloprid. The rapid combustion of wood at high temperatures, as can occur in a fireplace, results in rapid degradation of other types of pesticides; residues are more likely under slow combustion and temperatures less than 1112 ^{o}F (600 ^{o}C) (McMahon et al., 1985; Bush et al., 1987a; Bush et al., 1987b). Imidacloprid would be expected to degrade at temperatures similar to those that would occur from burning firewood, based on its measured thermal decomposition temperature, which is below 932 ^{o}F (500 ^{o}C).

Potential thermal degradation products from the use of the imidacloprid formulations that could be used in the Program include hydrogen cyanide, carbon monoxide, and oxides of nitrogen and carbon. Concentrations of these degradation products would be very low due to the expected concentrations of imidacloprid in firewood, and potential temperatures that could occur in burning firewood.

5. Alternative 5: Integrated Approach (Preferred Alternative)

a. Forest Resources

The impacts of the preferred alternative on forestry resources will be similar to those impacts described in alternatives 3 and 4. Under the preferred alternative, the Program may either remove high-risk host trees or treat high-risk host trees with imidacloprid. An integrated approach alternative allows for the adaptation of eradication methods to suit site-specific needs and resources. In the long term, adverse impacts to forest resources would be greatest under Alternative 1—No action Alternative and Alternative 2—Removal of Infested Trees when compared to the preferred alternative, the integrated approach. Alternative 1 is not an eradication program and ALB would continue to spread under alternative 2.

b. Environmental Resources

(1) Water

Impacts to water quality (described for the previous alternatives, other than the no action alternative), would apply to the preferred alternative as they require some level of tree removal that can result in impacts to water quality. These impacts will be greatest in watersheds that have soils that are vulnerable to erosion and large numbers of host trees in proximity to water. However, the short-term impacts to water quality would be reduced under the preferred alternative when compared to the implementation of the exclusive use of tree removal or insecticide treatment of high-risk host trees.

Flexibility in addressing high-risk host trees would allow the Program to implement measures with the greatest chance of success based upon site-specific conditions and resources. Potential long-term impacts to water quality would be reduced as ALB eradication efforts would have a greater probability of success, and prevent the spread and introduction of ALB to other watersheds in the United States where host trees are present.

(2) Air

The impacts to air quality (described for the previous alternatives, other than the no action alternative) would apply to the preferred alternative, as it requires some level of tree removal, which can affect air quality. However, the impacts for the preferred alternative would be reduced compared to those alternatives because neither imidacloprid treatment nor removal of high-risk hosts would be used exclusively. Impacts to air quality from tree removal activities would be more localized when compared to full host tree removal. Air quality impacts from the use of the herbicides, or the insecticide imidacloprid, are not anticipated based on their use pattern and the low probability of any chemical volatilizing into the atmosphere.

(3) Soil

The impacts to soil would be similar to those described for alternatives 3 and 4, Full Host Removal and Insecticide Treatment of High-Risk Host Trees. Both Alternative 1—No Action and 2—Removal of Infested Trees would allow ALB to continue to expand its distribution within the United States, with impacts to soil quality dependent upon the importance of ALB-host species in soil nutrient cycling. The potential for soil quality impacts would increase with the removal of infested trees under alternatives 2 through 4; however, impacts under the preferred alternative would be less as there would be an option for tree removal of ALB high-risk host trees or insecticide treatment. Insecticide treatment can impact soil quality for those soil fauna that are sensitive to imidacloprid; nonetheless, these impacts would be more localized than tree removal activities that can result in the physical disruption of soil through the use of heavy equipment.

c. Ecological Resources

Environmental impacts from the selection of the preferred alternative may result in long-term impacts that are less than those described under the no action alternative and the other eradication alternatives. The implementation of alternative 5 has been shown to be a successful eradication program strategy in site-specific infestations. Risks to ecological resources would be localized to the areas of infestation similar to alternatives 3 and 4; however, in the case of high-risk host trees, the Program would have the flexibility to select the treatment option that best fits site-specific conditions. There would be potential risks related to removal of trees, as described under alternative 3, and insecticide treatment, described under alternative 4 however, flexibility to select either method would reduce the risk when compared to the exclusive use of either eradication method.

d. Economic, Social, and Cultural Resources

Selection of the preferred alternative may have economic, social, and cultural impacts similar to those described for alternatives 3 and 4. These impacts would be reduced under the preferred alternative because there is flexibility in how high-risk host trees would be treated. Alternative 1 and alternative 2 would have the greatest potential impacts because neither approach eradicates ALB.

Human Health

Under alternative 5, the rate of tree loss is expected to be reduced the most compared to the other alternatives. In the short term, tree loss may occur from removal of infested and some surrounding host trees. The amount of tree loss from the removal activities would be more than alternative 2 and 4 but less than alternative 3. Localized negative human health consequences from tree loss (e.g., increased stress and reduced air quality as discussed in the no action alternative) are expected in the affected areas where ALB infested and high-risk host trees are removed.

This alternative is expected to eradicate ALB resulting in less tree loss in the long term. Eradication of ALB would allow all the benefits of trees to continue in broader areas, and the negative consequences from a lack of trees can be avoided. Specifically, shading and improved air quality would continue to be available for people living in and travelling through affected areas. Trees would continue to provide opportunities for stress reduction and beautify the landscape.

e. Adaptive Management Approach

The adaptive management approach evaluates future pesticide use for similar or less impacts than those described for the pesticides in each of the alternatives discussed in this EIS. This includes any changes in herbicide or insecticide use that could occur in the future as the Program evaluates new chemical treatment options. The use of the adaptive management approach is only in cases where a human health and ecological risk assessment demonstrates that proposed pesticides would have equal or less risk to human health and the environment, in comparison to the pesticides currently used by the program. The criteria for this approach ensures that the potential pesticide-related impacts to the various resources discussed in this EIS are similar, or less, than those associated with any new pesticide use proposed in the future.

6. Cumulative Impacts

Cumulative impacts are those impacts on the environment that result from the incremental impact of an action when added to other past, present, and reasonably foreseeable future actions, regardless of what agency (Federal or non-Federal) or person undertakes such actions. Cumulative impacts can result from individually minor, but collectively significant actions taking place over time. The potential for cumulative impacts in the ALB Eradication Program will vary across the United States due to the various site-specific conditions where an infestation may occur.

The cumulative impacts discussion in this EIS is meant to be general because the baseline environmental conditions vary between urban and forested areas, and other site-specific conditions that may occur where

ALB-host trees are present. Site-specific EAs may incorporate this information to better characterize the potential cumulative impacts related to an ALB Eradication Program.

For the Program, the cumulative impacts are primarily associated with the loss of trees, and the proposed use of an insecticide and herbicides. The severity and intensity of tree loss varies under the different alternatives, as well as the dominance of ALB-host trees in a given area.

Cumulative impacts related to forest resources would be expected under those alternatives where ALB is likely to spread and infest new areas. Cumulative impacts to forest resources would also be expected during successful eradication efforts; however, those impacts would be more localized and short term compared to instances where ALB is allowed to spread. The effects of natural and manmade stressors to forests (e.g., timber harvests, acid rain, climate change, and other pests and diseases) can be additive or synergistic, that is, the effects of all of the stressors together become greater than the individual stressors alone (Cox, 1999; Logan et al., 2003). The effect of these other stressors added to the impact of an ALB infestation could increase the severity of an ALB outbreak. An example of this type of cumulative impact is evident with the decline of oaks in the Ozark forests in the Central United States. A native cerambycid beetle, the red oak borer (*Enaphaledes rufulus*) was not a major pest of oak, but is now considered a major contributor to oak decline in the area due to other factors that predispose the trees to infestations at higher levels than previously recorded (Coulson and Stephen, 2006).

The loss of ALB-host trees may have cumulative impacts by increasing the ability of invasive plants to establish in areas where canopy gaps occur, or other areas where host trees are lost. These types of introductions have been noted in disturbed forested areas, and may impact natural plant succession, as well as nutrient cycling (Woods, 1997; Meiners et al., 2002).

Currently, FS is addressing approximately 70 invasive species (plants and insects) in areas where ALB-host trees are present. The impact from the loss of ALB-host trees could result in cumulative impacts that would benefit the establishment of many of these species. These impacts would be more widespread under the no action alternative (1) and alternative 2 because ALB would be able to establish and spread to other areas of the United States. These types of cumulative impacts may also occur under alternatives 3 through 5 because tree removal would occur, however, for the alternatives that result in eradication, the potential cumulative impacts would be reduced.

Under the no action alternative, it is possible other State and local agencies or tribes will implement an eradication program without involvement from APHIS. Given the limited jurisdiction of these entities, the eradication programs would lack regional or national coordination, which could result in the spread of ALB to other host trees over a larger geographic area. The expansion of ALB beyond the current areas could result in additional stressors to host trees, causing both economic and environmental impacts. Abiotic and biotic stressors (e.g., climate change, other invasive pests, and air pollution) all pose threats to ALB-host trees; the addition of ALB to urban and natural forest ecosystems would be expected to result in cumulative impacts beyond those already identified as potential stressors (Horsley et al., 2002; Poland and McCullough, 2006; Iverson et al., 2008). Economic losses to the timber and maple syrup industries would be anticipated, as well as increased costs to homeowners that choose to treat trees or have them removed once they are infested. Economic data for the loss of ash trees in Ohio from an invasive pest, which is just one of the hosts for ALB, show that landscape loss, tree removal, and replacement costs could range between $1.8 and $7.6 billion due to the emerald ash borer (*Agrilus planipennis*) (Sydnor et al., 2007). The loss of one tree species is significant; ALB has many more host trees than emerald ash borer.

Cumulative impacts to the environment would also be expected as ALB-host trees are lost from urban and natural forests. The potential for cumulative impacts would be dependent upon whether trees are removed from urban or forested areas, and their dominance in those areas. Tree removal, under the various alternatives, could result in impacts to soil and water quality. These impacts are reduced by implementation of BMPs. In urban areas, residential and business development result in soil disturbance and potential water quality impacts due to land disturbance, and increase impervious surfaces which have the ability to transport a variety of pollutants to surface and ground water.

ALB eradication efforts in urban areas are expected to have incrementally minor impacts to environmental quality when put in the context of other activities that may impact air, soil, and water. In forested areas, the cumulative impacts from tree removal may result in cumulative impacts to soil and water quality, resulting in increased erosion and transport of sediments and nutrients to water and, in particular, in cases where large areas of timber are removed in proximity to water, or from watersheds vulnerable to soil erosion. The magnitude of these contributions to natural and other man-made sources of sediment and nutrients will vary, depending on site-specific conditions and other activities that may be occurring in a given watershed. The potential for cumulative impacts in these scenarios would be reduced due to the implementation of BMPs by the Program, where applicable. The selection of an alternative that will ensure eradication will reduce the likelihood of cumulative impacts occurring over a larger geographic area. There would be some loss of wildlife habitat in areas where host trees are removed; however, those losses would not be considered permanent because in unmanaged habitats (e.g., woodlots), stumps of high-risk host trees would be allowed to resprout, and replanting activities may occur in managed areas.

Cumulative impacts to human health and the environment are anticipated to be incrementally minor for the proposed use of herbicides. All three herbicides have agricultural and non-agricultural (e.g., right-of-way and forestry) uses. FS uses triclopyr and, to a lesser extent, imazapyr in many of its invasive weed control programs where ALB-host trees are present. The proposed use of herbicides in the ALB Eradication Program is not expected to contribute significantly to the overall use of herbicides by other entities. The Program applies herbicides to stumps or sprouting vegetation from cut stumps, using hand painting or backpack spray applications that minimize offsite transport of the proposed formulations. Recent ALB eradication efforts suggest that stump removal is the preferred treatment method, with greater than 75 percent of the stumps being removed rather than being treated with an herbicide.

Imidacloprid is widely used in urban and agricultural settings; however, the increase in loading beyond current use, in addition to that which could be added due to ALB treatments, is difficult to quantify because the number of treated trees is unknown relative to current use patterns for areas where ALB-host trees may occur. In addition, the use of imidacloprid under the preferred alternative would vary because of the choice of tree removal and insecticide treatment.

Cumulative impacts from the proposed use of imidacloprid would not be expected to result in significant impacts to human health. The toxicity profile, method of application, and notification to the public regarding its use would minimize exposure and risk to the public. Adherence to label requirements would also minimize exposure and protect human health for workers involved with ALB eradication efforts.

The amount of imidacloprid added to the environment would be greatest under alternative 4 because high-risk host trees within a ½-mile radius of infested trees would receive treatment, compared to the preferred alternative where only select trees would receive imidacloprid treatments.

The cumulative risk to aquatic resources would be greatest when considering large-scale imidacloprid treatments of deciduous trees, such as ALB-host trees. Imidacloprid residues in leaf litter from treated trees can be transported to aquatic environments and result in sublethal impacts to some aquatic invertebrates (Kreutzweiser et al., 2007; Kreutzweiser et al., 2008b; Kreutzweiser et al., 2009). In those studies, the more significant impacts occurred in cases where exposure to imidacloprid in leaf litter was at concentrations greater than anticipated under the current proposed use pattern. These impacts are selective to certain types of aquatic invertebrates due to their feeding preference, and would not be anticipated for other aquatic invertebrates. Cumulative impacts from the addition of treated leaves would be expected to only occur for some aquatic invertebrates in cases where host trees are a dominant species and large numbers are treated. The cumulative impacts to aquatic invertebrates would also be reduced by the presence of non-treated leaf litter and other organic matter present in aquatic habitats.

Streams that may already be impacted by other factors could have cumulative impacts related to imidacloprid use in cases of large-scale treatments. Available water monitoring data for imidacloprid and other neonicotinoid insecticides in the United States shows that detections are common in urban and agricultural areas (Phillips and Bode, 2004; Starner and Goh, 2012; Hladik et al., 2014). These detections occur along with multiple other pesticides, and other organic and inorganic contaminants.

Water quality data show pesticide mixtures to be a common occurrence. Water quality data collected by the U.S. Geological Survey, as part of a national monitoring effort, shows that 50 percent of the surface water samples contain four or more pesticides in urban, agricultural, and mixed-use watersheds (Gilliom et al., 2007). The impact of these mixtures on human health and aquatic ecosystems is poorly understood. Chemical mixtures may have additive, synergistic, antagonistic, or potentiation effects on biological systems. Studies testing imidacloprid and other chemical mixtures have reported results showing no interactions between chemicals or additive, synergistic, and antagonistic effects to test organisms (Chen et al., 2010; Loureiro et al., 2010; Zhang et al., 2010; Pavlaki et al., 2011). The chemical mixture, timing and level of dosing, and endpoints measured were all variables that can impact how these chemicals may act in combination. Interactions between pesticides, such as imidacloprid and other stressors (e.g., water quality and predators) have also been reported for various aquatic and terrestrial species (Holmstrup et al., 2010; Laskowski et al., 2010).

Uncertainties regarding the prediction of effects to nontarget organisms from imidacloprid and all possible mixtures, as well as the large area of the United States that is being considered in this EIS, make it difficult to determine cumulative impacts from program insecticide treatments. However, the incremental increase in risk due to imidacloprid use is expected to be minor for aquatic communities because the potential for imidacloprid risk to aquatic habitats is low; this is based on the proposed method of application, environmental fate, and available information regarding imidacloprid effects to aquatic organisms.

Cumulative impacts to terrestrial wildlife associated with imidacloprid use are also expected to be incrementally minor for most animals. Imidacloprid residues on food items for mammals and birds are not expected to result in significant risk to wildlife. The low risk to mammals and birds suggests that any cumulative impacts of imidacloprid would be minor when put in context with other stressors, such as loss of habitat. Imidacloprid will affect sensitive nontarget invertebrates that consume plant material from treated trees; however, the mode of action of imidacloprid, and the method of application, which is targeted to certain trees, will reduce impacts to most nontarget invertebrates. The risk to sensitive terrestrial invertebrates from proposed imidacloprid applications is expected to be minor in relation to other stressors, such as other pesticide applications and loss of habitat. The lack of significant cumulative impacts to most terrestrial invertebrates will also ensure that mammals and birds that prey on insects are not impacted.

Large-scale treatment of trees using imidacloprid could also increase pesticide exposure to pollinators above current levels. A variety of factors stress native pollinators, as well as honey bees. Stressors include environmental pollution, habitat loss, poor nutrition, pests and diseases (e.g., Varroa mites on honey bees), and some pesticides, including imidacloprid (Potts et al., 2010; USDA, 2012; Sanchez-Bayo and Goka, 2014). Recent studies have shown that honey bees exposed to sublethal concentrations of insecticides and pathogens indicate interactive negative effects from exposure (Alaux et al., 2010; Pettis et al., 2012).

Interactions between imidacloprid, as well as other neonicotinoids and pathogens (e.g., *Nosema*), have resulted in colony and immune function impacts to honey bees (Alaux et al., 2010; Vidau et al., 2011; Pettis et al., 2012). Colony collapse disorder (CCD) is a problem impacting domestic honey bee populations, with potentially significant impacts regarding pollination of agricultural and native plants. The causal agents for CCD are a complex variety of stressors, of which imidacloprid and other insecticides may contribute (USDA, 2012; Lu et al., 2014). The potential for exposure and cumulative impacts to honey bees, and other pollinators, from imidacloprid use is reduced by the availability of other species of flowering plants, and selectively treating trees in the ALB quarantine area.

B. Special Programmatic Considerations

1. Applicable Environmental Statutes

a. Endangered Species Act

Section 7 of the ESA and its implementing regulations require Federal agencies to ensure their actions are not likely to jeopardize the continued existence of listed threatened or endangered species, or result in the destruction or adverse modification of critical habitat.

(1) Potential Effects of ALB Establishment on Listed Species

ALB has the potential to affect listed species and their habitats throughout the United States. ALB-host species, including maple, poplar, and birch, among others, play a critical role in the life histories of numerous listed species; impacts to these host species could affect their survival and recovery. Approximately 60 federally listed species and their critical habitats could be impacted by the introduction and spread of ALB in the United States based on the co-occurrence of preferred ALB-host trees and forest-dependent listed species in the Northeastern and mid-Atlantic areas (figure 4–2). Approximately half of the species in these areas are listed freshwater mussels, while approximately 18 percent are listed terrestrial plants. The remaining listed species are primarily mammals and fish.

Widespread establishment of ALB in the United States will cause significant ecological damage. Impacts to terrestrial and aquatic fauna and flora are expected, especially in areas where ALB-host plants are prevalent. T&E species that depend on ALB-host trees would be most affected. The extent of damage and its impact on ecological community function would be dictated by multiple factors. In cases where the host trees are not a dominant component of the forest stand, impacts may be less; however, in situations where host trees occur in high density and are keystone species within that particular forest type, the impacts are almost certain to be significant.

(2) Potential Effects of the ALB Program on Listed Species

Effects to listed species from the proposed alternatives for eradicating ALB may also pose a risk to protected species and their designated critical habitat without proper mitigation. A recent example of these impacts is with listed bat species that may occur where ALB eradication efforts are currently implemented. Bat species in the program area (e.g., Indiana bat and northern long-eared bat) use trees for roosting, as travel corridors, and as foraging habitat. A portion of these trees could be ALB-host trees that may require removal.

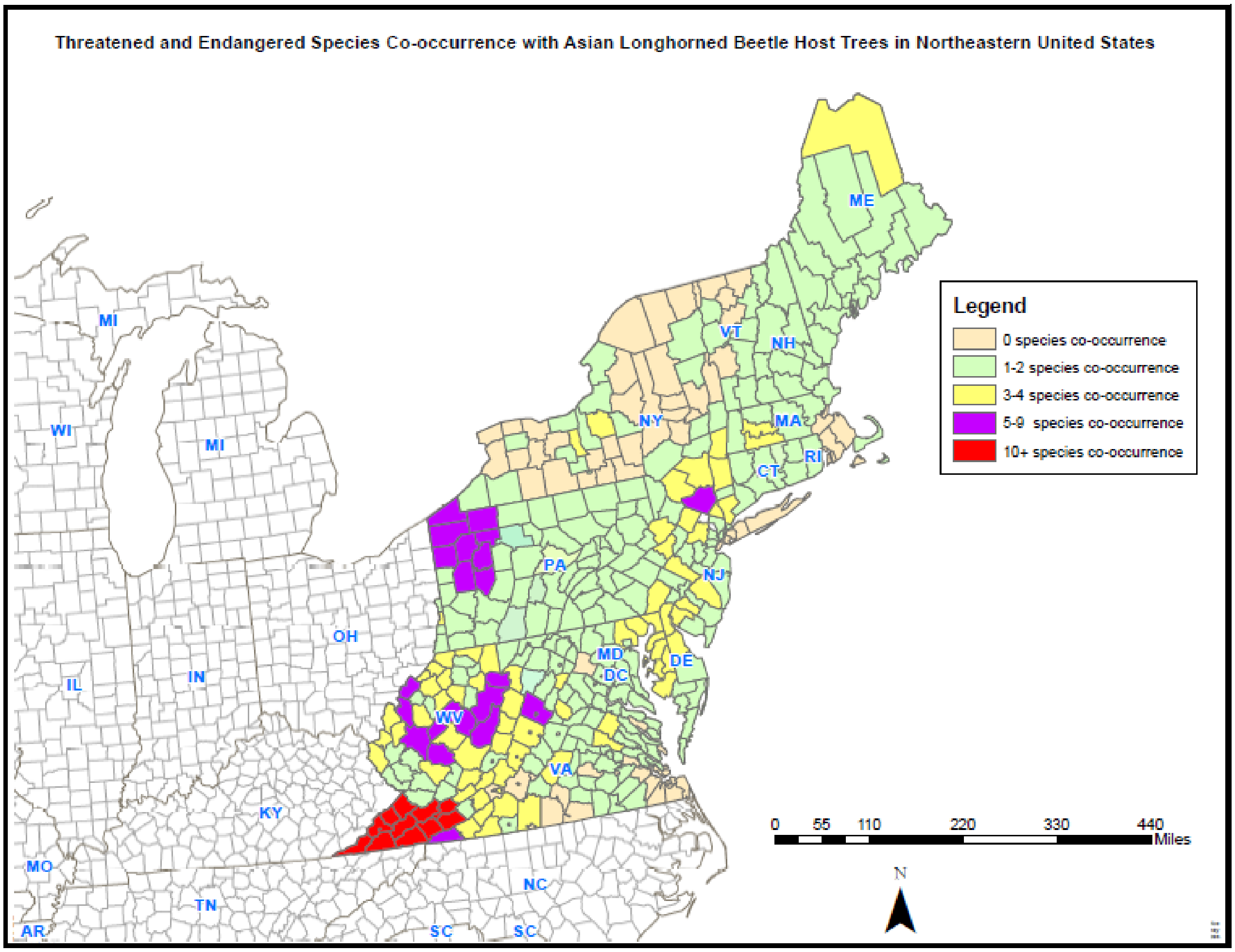

Figure 4–2. Federally listed threatened and endangered species by county that co-occur with ALB-host trees in the Northeastern United States.

Direct effects are those effects that are the result of the direct or immediate effects of the project on the species or its habitat. For bats, these effects can include increases in disturbance (i.e., in the form of noise, human activity, and vibrations from equipment) from tree clearing and would vary in intensity, depending upon the source (chainsaws, chippers, stump grinders, trucks). Tree removal can result in loss of maternity and non-maternity roost trees. Removal of roost trees could impact bats by requiring them to expend time and energy to identify an alternate roost site. Removal of an unidentified roost tree during the summer occupancy season could result in crushing or injury of bats.

Indirect effects to bat species are those effects that are caused by or would result from the proposed action and are later in time, but are still reasonably certain to occur. These effects could result from reduction of bat insect prey from exposure to insecticides or herbicides. Tree removal may also indirectly affect bats by decreasing roost and habitat availability, and requiring additional time to forage and search for suitable roost trees. Elimination or significant reduction of fence rows and tree lines (used as travel corridors) may require bats to expend additional energy to find alternative routes.

The removal of ALB-host trees can also impart beneficial effects to bats by creating canopy openings and more open understory, providing a higher quality habitat and an increase in maternity trees. Beneficial effects to bats from removal of ALB-host trees could create canopy openings in the interior of forests; this would increase solar exposure that may allow these larger trees to become suitable as maternity habitat. In addition, many of the trees removed are smaller trees; this would create a more open understory, providing higher quality habitat.

Other listed terrestrial species may also be impacted by ALB eradication activities. Physical disturbance of areas where ground-nesting listed species occur can also have direct and indirect impacts. For example, disturbance of nesting areas of listed birds during the breeding season could occur from program activities near breeding areas. Disturbance can result in temporary abandonment of nests, exposing adults to aerial predation, and eggs and chicks to predation and inclement conditions. Tree removal activities may also result in take of listed species that may not be able to disperse during tree removal activities, and could be crushed by equipment. Forest-inhabiting plant species (e.g., running buffalo clover and small whorled pogonia) may be affected by clearing infested trees that would decrease forest floor shade, thus reducing habitat suitability. Tree removal could also result in trampling or physical destruction of plant populations. Insecticides applied to host trees could reduce pollinators of listed plant species. Program application of herbicides to stumps could directly harm listed plants if they are near treatments.

ALB-host trees serve to protect water quality; their removal can destabilize stream banks resulting in excess sedimentation which would be detrimental to listed freshwater mussels (e.g.,fanshell and pink mucket pearlymussel), fish, or other aquatic species in the vicinity of an ALB-infested area. As previously mentioned, over half of the listed species in the Northeastern United States that could be impacted by the loss of ALB-host trees are aquatic. Sedimentation and changes to water quality from tree loss (temperature, shading, and pesticide residues) may result in impacts to aquatic species due to program activities.

(3) Consultation History

APHIS considers whether listed species, species proposed for listing, or critical habitat are present in the proposed program area. If none are present, no Section 7 consultation is required. If species or critical habitat is present in the proposed program area, APHIS conducts Section 7 consultation with the FWS and NMFS, on a site-specific basis for ALB eradication activities.

For the ALB Eradication Program in Worcester County, Massachusetts, APHIS consulted informally with FWS on a threatened plant, the small whorled pogonia, *Isotria medeoloides*, in 2008. In 2011, APHIS consulted informally with FWS on the impact of the eradication program on the small whorled pogonia and piping plover (*Charadrius melodus*) in Norfolk and Suffolk Counties in Massachusetts.

In June, 2011, APHIS first contacted the FWS in Columbus, Ohio for technical assistance regarding impacts to federally listed species in Clermont County, Ohio. Seven endangered species (Indiana bat, *Myotis sodalis*; running buffalo clover, *Trifolium stoloniferum*; fanshell, *Cyprogenia stegaria*; rayed bean, *Villosa fabalis*; snuffbox, *Epioblasma triquetra*; pink mucket pearlymussel, *Lampsilis abrupta*; and sheepnose, *Plethobasus cyphyus*) occur in Clermont County. Since that time, APHIS submitted biological assessments and consulted with FWS on those species, receiving concurrence with their "may affect, not likely to adversely affect" determinations, with the implementation of protection measures. In addition, FWS personnel made site visits to the infested area and have provided Indiana bat training to APHIS and the Ohio Department of Agriculture personnel. Also, APHIS conducted surveys for the Indiana bat, and other bat species, in the Clermont County eradication work zones and reported those findings to FWS. Most recently, APHIS entered into a formal consultation with FWS on the Indiana bat (receiving a biological opinion, dated June 4, 2014) and a conference on the northern long-eared bat (*Myotis septentrionalis*), a species proposed for listing as endangered. APHIS received a northern long-eared bat conference opinion on July 3, 2014.

APHIS would continue to consult with FWS or NMFS, as necessary, when a known ALB infestation has been confirmed. In addition, APHIS would implement measures prior to the initiation of program activities to protect federally listed species and critical habitat.

b. Migratory Bird Treaty Act

The Migratory Bird Treaty Act of 1918 (16 U.S.C. 703–712) established a Federal prohibition, unless permitted by regulations, to pursue, hunt, take, capture, kill, attempt to take, capture or kill, possess, offer for sale, sell, offer to purchase, purchase, deliver for shipment, ship, cause to be shipped, deliver for transportation, transport, cause to be transported, carry, or cause to be carried by any means whatever, receive for shipment, transportation or carriage, or export, at any time, or in any manner, any migratory bird or any part, nest, or egg of any such bird.

Prior ALB eradication efforts and consultations with FWS have resulted in several management recommendations to minimize impacts to migratory birds. These include:

- minimize tree removals during nesting season
- minimize disturbance as much as possible (avoid impacts to areas of nonhost shrub/brush areas)
- replant areas that have been significantly deforested
- use existing trails for equipment to avoid disturbance to pastures/open fields that could be used as breeding sites for ground-nesting birds
- have the names and contact information for local wildlife rehabilitators so that if there is an issue (e.g., as a raptor nest or fledging in the area), guidance can be provided regarding how to handle the situation

c. Bald and Golden Eagle Protection Act

The Bald and Golden Eagle Protection Act (16 U.S.C. 668–668c) prohibits anyone, without a permit issued by the Secretary of the Interior, from “taking” bald eagles, including their parts, nests, or eggs. The Act provides criminal penalties for persons who “take, possess, sell, purchase, barter, offer to sell, purchase or barter, transport, export or import, at any time or any manner, any bald eagle…[or any golden eagle], alive or dead, or any part, nest, or egg thereof.” The Act defines “take” as “pursue, shoot, shoot at, poison, wound, kill, capture, trap, collect, molest or disturb.”

Without the implementation of the protective measures outlined below, tree cutting could disturb nesting eagles. FWS has recommended buffer zones from active nests which require different levels of protection (FWS, 2007). They are as follows:

- Avoid clear-cutting or removal of over-story trees within 330 feet of a nest at any time. (The Program will not use clear-cutting under any alternative discussed in this document.)

- Avoid timber harvest operations (including road construction, and chain saw and yarding operations) during the breeding season, within 660 feet of the nest. The distance may be decreased to 330 feet around alternate nests within a particular territory—

  - including nests that were attended during the current breeding season but not used to raise young, and

  - after eggs are laid in another nest within the territory have hatched.

To ensure that program activities do not disturb eagles, APHIS would contact State fish and wildlife agencies, as well as the FWS, to determine the location of eagle nests in the program area. APHIS would also work with the FWS prior to any tree removal during the breeding season within 660 feet of a nest to confirm that all eagles have left the nest. Outside of the breeding season, cutting may occur within the buffer zone around nests.

d. National Historic Preservation Act

Section 106 of the National Historic Preservation Act requires Federal agencies to evaluate the potential impacts of their actions on historic properties. Historic properties are those that are on the National Register of Historic Places or meet the criteria for the National Register (NHPA, 2014). To date, the ALB Eradication Program has met its Section 106 responsibilities by contacting the State Historic Preservation Office (SHPO) in each State where eradication activities may occur. Due to the programmatic approach used in this EIS, and the uncertainty regarding where an ALB infestation may occur, APHIS has not contacted SHPOs in States where infestations have not occurred. APHIS will continue its current policy of evaluating program impacts to historical properties, and working with the SHPO in States with known ALB infestations.

e. Executive Order 13045

Executive Order (EO) 13045, "Protection of Children from Environmental Health Risks and Safety Risks," acknowledges that children, as compared to adults, may suffer disproportionately from environmental health and safety risks due to their developmental stage, greater metabolic activity levels, and their behavior patterns. This EO (to the extent permitted by law and consistent with the agency's mission) requires each Federal agency to identify, assess, and address environmental health risks and safety risks that may disproportionately affect children. An analysis of the potential impacts to children from the proposed alternatives, including the available human health risk assessments for proposed program pesticide use, suggest that no disproportionate risks to children are anticipated.

f. Executive Order 12898

EO 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations," focuses Federal attention on the environmental and human health conditions of minority and low-income communities, and promotes community access to public information and public participation in matters relating to human health and the environment. This EO requires Federal agencies to conduct their programs, policies, and activities that substantially affect human health or the environment in a way so as not to exclude persons and populations from participation in or benefiting from such programs. It also enforces existing statutes to prevent minority and low-income communities from being subjected to disproportionately high or adverse human health or environmental effects.

The human health and environmental effects resulting from the preferred alternative are expected to be minimal, and are not expected to have disproportionate adverse effects to any minority or low-income family. Low-income families may depend on woodlots for firewood to heat their homes; however, the most valuable species used for firewood (including oak, hickory, beech, and locust) are not ALB-host species and would not be removed. Although some maple species may be less valued for firewood, they are commonly used for that purpose and are a preferred ALB-host. Nevertheless, if no action is taken, allowing ALB to spread could result in permanent loss of maples and all other ALB-hosts from the area. For full host removal, stumps from high-risk host trees in woodlots may be allowed to resprout, which would then allow more rapid regrowth. Wood treated with imidacloprid and used as firewood is not expected to cause adverse health effects. Therefore, the human health and environmental effects from the action alternatives are not expected to have disproportionate adverse effects to any minority or low-income family.

g. Executive Order 13175

EO 13175, "Consultation and Coordination with Indian Tribal Governments," was issued to ensure that there would be "meaningful consultation and collaboration with tribal officials in the development of Federal policies that have tribal implications…." APHIS prepared and sent a letter to each of the tribes in the contiguous United States in June, 2013, prior to publishing the NOI to prepare the EIS. The letter provided background on ALB in the United States, and provided the tribes an opportunity to dialogue with APHIS regarding potential impacts to tribal resources. APHIS also hosted a teleconference with all interested tribes in July, 2013, to discuss the Program and answer any questions from tribal representatives. APHIS will continue to actively engage with the tribes on eradication program activities, and address questions and concerns. This would occur through tribal consultation prior to the development of site-specific EAs.

h. Executive Order 13186

EO 13186, "Responsibilities of Federal Agencies to Protect Migratory Birds," directs Federal agencies taking actions with a measurable negative effect on migratory bird populations to develop and implement a memorandum of understanding (MOU) with FWS that promotes the conservation of migratory bird populations. On August 2, 2012, an MOU between APHIS and FWS was signed to facilitate the implementation of this EO. The MOU provides APHIS with guidance to avoid and minimize, to the extent practicable, detrimental migratory bird habitat alteration or unintentional take during management activities.

i. Other Federal and State Regulations and Statutes

Other Federal and State regulations may also apply to the proposed alternatives identified in this EIS. APHIS complies with all applicable Federal regulations and Executive orders discussed in this EIS, such as the:

- Clean Air Act,
- Clean Water Act,
- Coastal Zone Management Act, and
- Federal Insecticide, Fungicide and Rodenticide Act.

States may also have applicable regulations regarding various proposed activities related to the ALB Eradication Program. APHIS works cooperatively with State agencies in the implementation of any proposed ALB activities, and to identify applicable State regulations to ensure compliance.

2. Mitigation and Risk Reduction

APHIS recognizes that the various alternatives proposed in this EIS may pose some risk to human health and the environment. This includes the no action alternative, as well as the proposed alternatives that were evaluated in this EIS. Many of the potential risks associated with implementation of an eradication program can be mitigated based on program requirements and, in the case of pesticide use, Federal regulations; this includes pesticide label language to reduce risk to human health and the environment. In addition, coordination with State agencies and other stakeholders on site-specific eradication efforts ensures that other State and local requirements are considered and implemented, as appropriate.

Hazards to the public from tree removal in urban and natural areas are mitigated through proper notification to ensure that the public would not be in areas when tree removal operations are taking place. All tree removals are coordinated with the respective landowners prior to any removal. Notification to landowners also occurs when chemical treatments may be used to kill stumps (herbicides) or treat high-risk host trees for ALB (insecticide). In addition to the notification process, there are other risk reduction requirements for pesticide use that are designed to protect workers, as well as the general public, and the environment. The mitigations are included on the pesticide label for a given formulation, and must be followed to ensure compliance with applicable Federal statutes.

The greatest potential for human exposure to program pesticides is for workers who handle the concentrated form of each product and make applications. In the case of workers, there are several requirements for PPE that will reduce exposure and risk including the use of:

- long sleeved shirt and long pants,
- protective eyewear,
- chemical resistant gloves, and
- shoes and socks.

These PPE requirements are based on the known toxicity profile for each of the proposed pesticides and the various formulations. Other labeling restrictions (e.g., applicable reentry intervals) are also designed to protect the public, including children as well as domestic pets.

Risk reduction to the environment from the proposed alternatives may occur through program requirements, as well as pesticide labeling to restrict use. As an example of program measures to reduce risk, the current eradication activities in Ohio are implementing Ohio Department of Forestry BMPs that have been developed to protect soil and water quality during tree removal operations (Ohio Department of Natural Resources, nd). These BMP's identify stream management zones (SMZs) that are protected areas that have been established adjacent to bodies of water that increase with increasing slope (Ohio Department of Natural Resources, nd). These types of buffers have been shown to provide protection to receiving waters from sedimentation and nutrients that may be of concern for a given watershed (Wenger, 1999).

Excess sediments and nutrients are primary causes for listing waterways in the United States as impaired, under Section 303(d) of the CWA. The SMZs are areas where no, or very limited, cutting would be allowed. Infested ALB-host trees that could occur within a SMZ would still need to be selectively removed, but without the use of heavy equipment. This could also occur with some high-risk host trees; however, in those cases, the removals would only be conducted with landowner approval and consultation with other Federal and State agencies to ensure the risk to water quality is minimized.

The risk to human health and the environment from the offsite transport of pesticides that are proposed for use in the ALB Eradication Program may also be reduced by pesticide labeling requirements designed to reduce the potential for contamination. Labeling restrictions and recommendations regarding application rate, timing of application, and application proximity to sensitive areas are all designed to reduce the risk of effects to human health and the environment. In addition, pesticide storage and disposal requirements are designed to reduce the risk of accidental spills and contamination of areas which could result in effects to human health and the environment.

3. Program Monitoring

APHIS conducts various monitoring for several of its programs. Monitoring efforts are typically directed towards measuring potential exposure to program pesticide treatments. Monitoring may consist of grab samples collected from different matrices (e.g., soil, water, air, biota) based on potential concerns regarding exposure which may occur to the environment and human health. These monitoring efforts are conducted to support analysis and compliance with applicable Federal regulations, such as the NEPA, ESA, and Federal Insecticide Fungicide and Rodenticide Act. Monitoring efforts in previous and current ALB eradication programs have been directed towards analysis of imidacloprid in various matrices (APHIS, 2013a). The objectives of the ALB environmental monitoring program are to:

- demonstrate the effectiveness of ALB-operational procedures in excluding or minimizing exposure of the public and the environment to Program-applied imidacloprid;

- collect data which can be used to evaluate whether the assumptions used in the environmental assessments are valid estimates of potential exposure of the public and the environment to Program-applied imidacloprid; and

- investigate any Program-related complaints or reports of adverse effects on public health, worker safety, environmental quality, or nontarget species.

The APHIS environmental monitoring staff coordinates the collection, packaging, and shipment of samples to accredited laboratories for processing and analysis according to standard operating procedures (APHIS, 2013a). The results of the laboratory's residue analyses are then correlated with environmental conditions data recorded at the time of treatment and sampling. The APHIS' environmental monitoring staff further analyzes the results to determine whether there are any human or environmental risks related to the use of the pesticide. The monitoring staff reports the data and analyses to Program managers at the end of the program, or intermittently during the program, as required.

ALB-related environmental monitoring for imidacloprid has occurred in Massachusetts and New York. To date, monitoring efforts for imidacloprid have focused on soil, water, plant, and bee-related samples to evaluate potential residues. Monitoring efforts will continue to be evaluated on a site-specific basis for any current and potential new infestations in the future.

4. Irreversible and Irretrievable Commitment of Resources

APHIS has been working on ALB eradication efforts since the beetle was first discovered in Brooklyn, New York, in 1996. Subsequently, resources have been committed to address infestations, as well as conduct research to understand the impacts of ALB introduction into the United States.

Research to develop successful eradication methods have occurred, and will continue, so that APHIS implements the most effective methodologies in current and future infestations. Since the initial infestation, other infestations have been documented requiring additional APHIS resources. APHIS works with multiple stakeholders regarding the implementation of ALB-eradication activities. This includes cost-sharing to implement various aspects of the ALB Eradication Program. Federal share of the total costs to operate ALB eradication activities has varied from 45 to 95 percent since the first Program was initiated (table 4–1). To date, Federal spending for the ALB Eradication Program has been over $500 million.

Table 4–1. ALB Eradication Program Historical Cost-Share Contributions (from 1997 to 2009 (thousands of dollars)).

Fiscal Year	Federal Funding (Appropriated + Emergency Funding)	Cooperator Funding	Total	Federal Share
1997	849	149	998	85%
1998	1,327	1,634	2,961	45%
1999	5,510	2,573	8,083	68%
2000	16,180	1,555	17,735	91%
2001	49,098	2,654	51,752	95%
2002	31,656	4,000	35,656	89%
2003	33,181	4,000	37,181	89%
2004	42,851	4,000	46,851	91%
2005	28,933	11,071	40,004	72%
2006	19,859	11,218	31,077	64%
2007	19,904	13,731	33,635	59%
2008	19,867	11,602	31,469	63%
2009	44,618	16,052	60,670	74%
2010	74,472	12,856	87,328	85%
2011	32,456	9,843	42,299	77%
2012	56,732	4,191	60,923	93%
2013	39,731	2,240	42,151	94%
Total	**$517,057**	**$111,599**	**$628,656**	**79%**

Appendix A. Preparers

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Background: Risk scientist with expertise on pest risk assessment, pest outbreak dynamics, and the role of climate drives and climate change on pest incidence. Eight years of service in APHIS. Experience in risk analysis especially as it relates to risk assessment of exotic pests entering on commodities under import permit application; climate as a drive on the outbreak of high-threat pests; and compliance to climate change and invasive pest legislation including Executive Orders 13122 and 13514 and Departmental Regulation 1070–001.

EIS Responsibility: EIS analyst – evaluated the effects of removing ALB-infested trees on greenhouse gas (GHG) emissions, and the effects of climate change impacts on/by tree removal actions. Identified management alternatives and "preferred option" to minimize impacts.

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B.S. Mathematical Economics
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Background: Applied economist analyzing the economic impact(s) from a proposed change in an agricultural policy/rule (domestic or foreign) and presenting the decision makers with the outcome of the potential economic benefits and costs for all the participants. Sixteen years of service with APHIS' Policy Analysis and Development Programs Division.

Worked as an analyst with the International Markets Division in Economic Research Service of USDA, and in the International Food and Policy Research Institute (IFPRI) for 7years.

EIS Responsibility: Economic analyst

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Appendix B. List of ALB NEPA Links Inventory of USDA Analyses to Date with Links to the Resources

Asian Longhorned Beetle Cooperative Eradication Program in Clermont County, Ohio, Revised Environmental Assessment, May 2013 (PDF; 2.65 Mb)

- FONSI signed May 1, 2013 (PDF: 71 Kb)

Asian Longhorned Beetle Cooperative Eradication Program in Clermont County, Ohio, Environmental Assessment, May 2012 (PDF; 1.27 Mb)

Asian Longhorned Beetle Eradication Efforts in Clermont and Brown Counties, Ohio, Environmental Assessment, September 2011 (PDF; 3.03 Mb)

- FONSI signed September 6, 2011 (PDF; 103 Kb)

Asian Longhorned Beetle Cooperative Eradication Program in Essex, Norfolk and Suffolk Counties, Massachusetts, Environmental Assessment, May 2011 (PDF; 83 Kb)

- FONSI signed May 20, 2011 (PDF; 4.28 Mb)

Chemical Treatment Study in New York City, New York, and Central New Jersey for the Asian Longhorned Beetle Eradication Program Environmental Assessment, September 2010 (PDF; 5.10 Mb)

- FONSI signed September 14, 2010 (PDF; 93 Kb)

Nursery Treatment Efficacy Study within Worcester County, Massachusetts, to Support the Asian Longhorned Beetle Cooperative Eradication Program, Environmental Assessment, April 2010 (PDF; 192 Kb)

- FONSI signed May 6, 2010 (PDF; 1.07 Mb)

New Chemical Treatment Study within the Worcester, Massachusetts, Quarantine Zone for the Asian Longhorned Beetle Eradication Program, Environmental Assessment, September 2009 (PDF; 903 Kb)

- FONSI signed October 30, 2009 (PDF; 162 Kb)

Asian Longhorned Beetle Cooperative Eradication Program in Worcester and Middlesex Counties, Massachusetts Environmental Assessment September 2008 (PDF; 1.28 Mb) [FONSI signed 11/21/08]

- Amended FONSI signed September 11, 2009 (PDF; 140 Kb)
- Addendum to the FONSI signed March 29, 2010 (PDF; 469 Kb)

Asian Longhorned Beetle Cooperative Eradication Program in the New York Metropolitan Area, Environmental Assessment May 2007 (PDF; 43 Kb)

Asian Longhorned Beetle Cooperative Eradication Program - Hudson County, New Jersey, Environmental Assessment, March 2003 (PDF; 36 Kb)

Asian Longhorned Beetle Program, Environmental Assessment, February 2000 (PDF; 892 Kb)

Asian Longhorned Beetle Field Trial, Environmental Assessment, January 2000 (PDF; 698 Kb)

Asian Longhorned Beetle Control Program - Illinois, Environmental Assessment, August 1998 (PDF; 23 Kb)

Asian Longhorned Beetle Control Program, Environmental Assessment, December 1996 (PDF; 23 Kb)

Appendix C. ALB: Annotated Host List

Asian Longhorned Beetle: Annotated Host List

Updated By Baode Wang on February 1, 2012

Additional comments by Baode Wang on June 25, 2014

USDA-APHIS-PPQ, Center for Plant Health Science and Technology, Otis Laboratory

Genus[1]	**Common Name**	**Host Abundance and Other Notes**[2]	**Treated, Surveyed**[3]
		Preferred Host in US[4]	
Acer	Maple, boxelder	Very common trees. Many US records, all species: Norway, red, silver, sugar, sycamore maple and boxelder especially favored; Amur maple less favored; Japanese maple seldom attacked.	**Yes**
Aesculus	Horsechestnut, buckeye	Fairly common trees. Several US records, some heavily infested.	**Yes**
Betula	Birch	Fairly common trees. Several US records: gray, European white and river birches. Some gray birches with many exits. Birches are apparently less preferred than maple. No exit holes found in laboratory studies with black & yellow birches yet although some larva developments inside trees of these two species have been observed.	**Yes**
Salix	Willow	Fairly common trees. Several US records: weeping, pussy and white willows highly favored; black willow (oviposition only) less favored.	**Yes**
Ulmus	Elm	Very common trees. Many US records: American, Siberian and Chinese elms. Elms are apparently less preferred than maple.	**Yes**
		Occasional to Rare Host in US[4]	
Albizia	Mimosa, silk tree, *A. julibrissin*	Occasional ornamental. Exit holes: 2 records from field in NY with additional emergence in laboratory. No Chinese record.	**Yes**
Cercidiphyllum	Katsura tree, *C. japonicum*	Occasional ornamental. Four records from Worcester, MA, including 2 trees with exit holes.	**Yes**
Fraxinus	Ash (especially green ash, *F. pennsylvanica*)	Very common tree, but injury infrequent relative to host abundance. Several US records, all from IL, most of these unverified (but at least two exit holes confirmed). Chinese ash, *F. chinensis* and white ash, *F. americana* were confirmed to be host in China	**Yes**
Platanus	London plane tree, *P. acerifolia*	Very common urban trees. 12 US records (including 4 with exit holes, NY); no record for *P. occidentalis*, American sycamore. Host in Chinese literature. Exit holes observed in China.	**Yes**
Populus	Poplar	Very common trees. Diverse and variable group, hybrids occur. Suitability apparently varies; some species and hybrids are prime hosts in China. Nine US records (NY, NJ, MA). Complete life cycle on eastern cottonwood, *P. deltoides* and quaking aspen, *P. tremuloides.* Oviposition on balsam popular, *P. balsamifera*, Balm-of-Gilead (a hybrid cultivar), unidentified *Populus sp*. Generally, *Populus* section Aigeiros (black poplars) are more preferred than other sections.	**Yes**
Sorbus	European mountain-ash, *S. aucuparia*	Occasional ornamental. Exit hole: 1 record from field in IL with additional emergence in laboratory. No Chinese record. Note: this is not a true ash; *Sorbus* is a member of the rose family.	**Yes**

Genus[1]	Common Name	Host Abundance and Other Notes[2]	Treated, surveyed[3]
Questionable US Records[4]			
***Celtis*[5]**	Hackberry, *C. occidentalis*	Fairly common tree. Oviposition: 1 unverified record from IL, with small/medium-sized larva identified as ALB. No Chinese record. No egg sites were found in lab studies with caged trees and beetles and active egg sites or exit holes were found in surveys in China.	**survey only**
Hibiscus	Rose-of-Sharon, *H. syriacus*	Common ornamental shrub. Exit: 1 unverified report, NY; Oviposition: several records, NY, but no larval development, possibly incidental to heavy damage on nearby hosts. No Chinese record.	**No**
Malus	Apple, crab apple	Common ornamental. Oviposition: 1 questionable record, IL. Host in Chinese literature. Oviposition observed in China.	**No**
Morus	Mulberry	Very common tree. Oviposition: 1 record, NY. No Chinese record.	**No**
Prunus	Cherry, plum	Very common ornamental. Oviposition: 2 records, NY & IL, but no survival. Host in Chinese literature.	**No**
Pyrus	Pear	Common ornamental. Exit: 1 questionable record, IL. Host in Chinese literature. Few exit holes were observed on *Pyrus bretschneideri* trees in China.	**No**
Quercus	Oak, (pin oak, *Q. palustris*)	Very common tree. Oviposition: 1 record, NY (incidental to heavy damage on nearby hosts). No Chinese record.	**No**
Robinia	Black locust, *R. pseudoacacia*	Common tree. Exit: 2 doubtful records, IL. Host in Chinese literature. Quite a few egg sites were observed in China, no exit holes.	**No**
Tilia	Linden (little-leaf linden, *T. cordata*)	Common tree. Oviposition: 2 records (IL & NY) but no survival. Oviposition but no survival in Canada. Host in Chinese literature.	**No**
No US Record[4]			
Alnus	Alder	Locally common tree or shrub. No US record. Host in Chinese literature. Exit hole observed in gray alder, *A. incana*, in cage study in China.	**No**
Elaeagnus	Russian olive (Oleaster), *E. angustifolia*	Widely-distributed ornamental shrub and escaped weed; quite variable, easily confused with other *Elaeagnus* species. No US record. Host in Chinese literature; Heavy feeding damage and few exit holes observed in China.	**No**
Koelreuteria	Goldenraintree, *K. paniculata*	Occasional ornamental. No US record. Heavy feeding, oviposition sites and 2 exit holes observed in field studies in China. Other exit holes were also found on trees along roadside.	**Yes**
Melia	Chinaberry, *M. azedarach*	Uncommon shrub. No US record; reported *not* to be a host in Chinese literature but damage observed.	**No**
Non-Host[4]			
Ailanthus	Tree of heaven, *A. altissima*	Common tree. No US record; reported *not* to be a host in Chinese literature.	**No**

Appendix D. Assessment of Greenhouse Gas Fluxes and Climate Change Impacts Related to ALB

Table of Contents

Executive Summary

The five key findings in this assessment of ALB (Asian Longhorned Beetle) eradication impacts are:

1. Total carbon dioxide (CO_2) emission from trees and soil at the Clermont County Ohio ALB Eradication Project site is estimated to be 7,872 metric tons (mt); total at all five ALB eradication sites is estimated to be 20,187 mt. These levels are below the CEQ (Council on Environmental Quality) reference level of 25,000 mt for all greenhouse gases (GHG); other GHGs (methane, nitrous oxide, hydro-fluorocarbons, perfluorocarbons, and sulfur hexafluoride) were not assessed in this study. Alternative scenarios of 'no action' indicate 2,254 million mt of CO_2 would be released from host trees in urban areas and adjacent forests under the current tree removal protocols across 48 States of the contiguous US. In the extreme scenario where all host tree species are infested by ALB across the 48 States, 13,906 million mt would be released in the absence of any eradication.
2. Cumulative CO_2 emission is zero by 76 years post-eradication where trees removed are artificially re-planted and/or natural forest regeneration is allowed. Most cities in the US this decade show diminished tree cover due to increased infrastructure (i.e., road, residential, and commercial building). Compared to American Forests standards of 40% urban tree cover, the US-wide average is 23%, indicating considerable leeway to increase CO_2 sequestration and C storage through expansion of urban tree cover. Alternative scenarios indicate that under methods that enhance tree growth by 10%, provide added protection, and expand tree cover from 25% to 40%, complete CO_2 recovery could be achieved in 43 years and total C storage could double over 'business-as-usual' approaches.
3. Tree removal in urban areas and adjacent forest impacts negatively on building energy consumption, local air pollution levels, and flooding control. Promoting tree cover to non-host, rapidly growing tree species increases adaptation to climate change. Examples are given of US cites with 'greening' programs aimed at increased tree cover to achieve benefits of adaptation.
4. Both host trees and ALB are strongly impacted by present and continuing climate change. In a study of Northeast US forests, 36 out of 80 species assessed show potential for their ecological optima to shift at least 100 km to the north, including seven that could move >250 km. Sugar maple (*Acer saccharum, -14.5%)* and black cherry (*Prunus serotina, -10.0%)* would decline sharply, while oak and southern pines would expand northward – hence an expected shift away from prime ALB host trees in the near future. ALB under present climate has the potential to occupy all 48 contiguous States. Temperature increases would enhance ALB survival and growth over large areas. Data on a diverse array of 612 crop pests and pathogens worldwide demonstrate an average pole-ward shift of 2.7 +/-0.8 km / year since 1960, supporting the hypothesis that global warming-driven pest movement is already underway.
5. Long-term shifts in average temperature and total precipitation at Cincinnati, Ohio (1895-2013) are modest, highly variable, and cyclical; temperature increased 0.42C/100 years compared to a global temperature change 0.73C/100 years. A simple ALB 'Index of Climate Favorability', 1950 to 2013 at Cincinnati is dominated by a cyclic pattern. Linear regression shows a decrease rate of 1.6 %/100 years. The present outbreak in Clermont County Ohio is preceded by a six year interval of favorable climate for ALB; since first detection in1996,

Years 1996, 2000, 2001, 2003, 2004, 2009 stand out as particularly favorable for ALB growth; this contrasts to any earlier 18-year period which had only two to three such events. Index values suggest an overall static trend but one punctuated by especially favorable or adverse yearly conditions. Outbreak sites imply that ALB thrives within a ‘humidity corridor’ and that commonly applied degree-day models require revision to fully incorporate moisture parameters.

1.0 Introduction

USDA APHIS evaluated the potential for greenhouse gas (GHG) emissions related to taking no action against the Asian Longhorned Beetle (ALB) and the preferred alternative which includes removal of infested host trees and a combination of removal and imidacloprid treatments of high risk host trees.

Estimating these types of emissions can be difficult to do on a national scale especially in this case where ALB has not been detected in areas where it could become established but could be in the future due to the presence of host trees and favorable environmental conditions. Estimates regarding the release of GHG in this assessment were quantified based on the most recent ALB outbreak in Ohio with information also provided on the other ALB outbreaks that have occurred in the United States. The emission of GHGs associated with tree removal, and GHG sequestration where tree replanting and regeneration are allowed (albeit into different tree species) was evaluated against the tree and soil carbon balances typical of similar but non ALB-infested areas.

2.0 GHG Emissions and Project Impact on Climate Change

2.1 GHG Emissions and Sequestration

2.1.1 Direct and indirect GHG emissions

Direct Emissions. The most conspicuous impact of the removal of large numbers of ALB-infested trees is the loss of CO_2 sequestration and release of carbon storage in trees and soil. As of mid-2013, about 9,400 trees had been removed in the Ohio ALB eradication program (Table 1).

Levels of tree removal at five ALB eradication sites were used as input to a CO_2 Emissions and Sequestration Model. This model developed by us applied the number of trees removed in tree species biomass equations of the USDA Forest Service, assumptions of tree size distribution among urban and forest tree populations, tree carbon and soil carbon content to estimate total CO_2 emitted to the atmosphere through decomposition of wood, and that sequestered from the atmosphere in regrowth over tree life expectancy of 100 years (Attachment 1).

Release of CO_2 to the atmosphere through decomposition follows removal, cutting, and chipping. Rapid release initially occurs from fine, nutrient-rich branches; this is followed by breakdown of residual heartwood and larger branches, resulting in a transient spike of release

8-12 years later. Decomposition in most temperate hardwoods is more or less complete by Year 22 (Fig. 1).

Assuming artificial replanting and natural regeneration follows immediately after tree removal, sequestration of CO_2 in new growth will match that released annually in decomposition by Year 16 and will completely re-capture all the CO_2 released by Year 76. Hence, the net positive flux (to atmosphere) in early decades reverts to a net negative flux (from atmosphere) in later decades. This changing dynamic is an important element in the overall, complete assessment of CO_2 impacts on GHGs (Fig. 1). Assertive management of tree health and expanded tree cover (as discussed below under Alternative 2) can actually benefit in shortening the time to full carbon recovery, and may substantially exceed the initial losses through better tree growth and through the selection of replacement species both resistant to ALB and better suited to warming conditions and associated weather variability.

The Ohio site to date is estimated to have released a total of 1762 mt of CO_2, about a quarter of that from soil disturbance; eventually that level could rise to 7872 mt. The total eventual release of CO_2 from all five ALB eradication sites in the northeast and north central States is estimated to be 20,200 mt through the near future, based on removal estimates of about 114,000 trees (Table 1).

Figure 1. Estimates for Ohio ALB eradication program of the emission of CO_2 from trees removed (blue) and sequestration of CO_2 in new tree growth (green). Net CO_2 flux (red) is emissions *minus* sequestration; dashed red line is zero net flux. Full recovery of all initial CO_2 emission is captured in new tree growth (dashed blue line, Year 76).

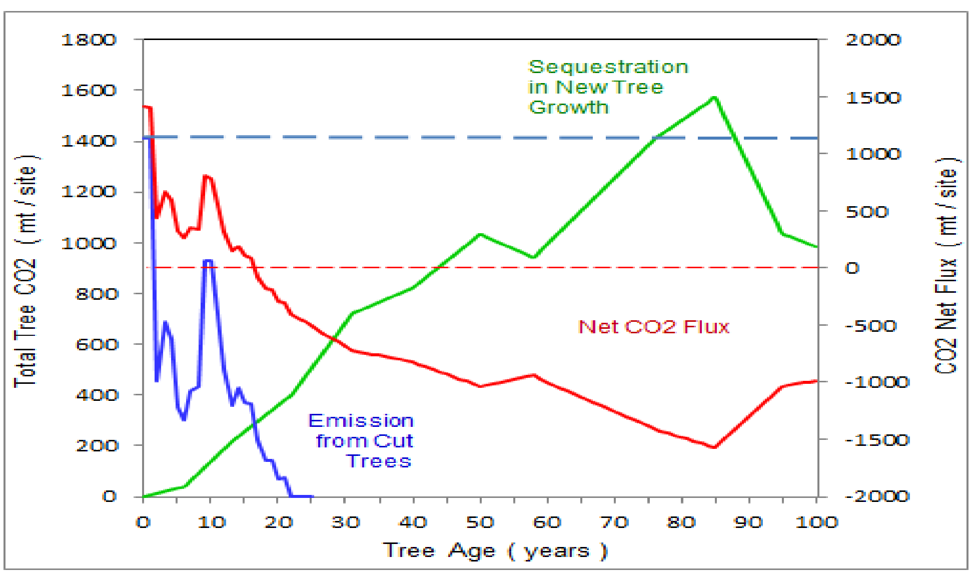


Table 1. Summary of tree removal levels to date (7/2013) and estimated final number at five locations in Northeastern United States, part of USDA APHIS Asian Longhorn Beetle Eradication Program. Estimated percent urban/forest, CO_2 released following tree removal and chipping, years required for total CO_2 release to atmosphere, and years required for total CO_2 recovery through sequestration in new tree growth.

YEAR	LOCATION	Actual No. Trees Removed	Estimated Urban %	Estimated Forest %	Estimated Biomass C Release CO2 (Mt)	Estimated Soil C Release CO2 (Mt)	Estimated Biomass+Soil C Release CO2 (Mt)	Estimated C Release (Years)	Estimated C Recovery (Years)
1996	First detection at US Port	0							
2000	Illinois	1771	100	0	223	56	279	22	76
2002	New York	8142	52	48	1150	288	1438	22	76
2005	New Jersey	32306	100	0	4076	1019	5095	22	76
2010	Massacheusetts (to 2013)	23463	35	65	3443	861	4304	22	76
2013	Massacheusetts (final est.)	30000	35	65	4402	1101	5503	22	76
2013	Ohio (to 2013)	9402	25	75	1410	352	1762	22	76
2015	Ohio (final est.)	42000	25	75	6298	1574	7872	22	76
TOTAL	All Locations (to 2013)	75084	62	38	10302	2576	12878	22	76
	All Locations (final est.)	114219	62	38	16149	4037	20187	22	76

<u>*Indirect Emissions*</u>. Other GHGs tied to the carbon cycle, notably NO_2 and CH_4 can have important atmospheric warming effects but were not quantified in this study. Methane production following tree removal would occur where anaerobic conditions exist, such as in very wet or submerged soils. Nitrogen compounds that eventually may end up in the atmosphere as GHG are released as tree wood decomposes. The production of nitrates ($^-NO_3$), studied in-depth by researchers more than other forms of organic nitrogen, is closely tied to soil processes and stream water discharge. Christopher et al (2004) observed that $^-NO_3$ discharge from forested watersheds in New York was largely from surface soils following snowmelt and major storm events (i.e., the 'flushing effect'). Even though the geology and hydrology of the two adjacent deciduous watersheds they studied were similar, the levels of $^-NO_3$ in discharge showed significant differences -- for reasons likely attributed to dissimilar nitrogen-fixing and decomposition properties. The lack of ALB Program data on soils and soil disturbances during cutting operations makes meaningful estimates of methane and nitrate production/release to the atmosphere difficult. In the eastern US, atmospheric nitrogen deposition is high, complicating an overall assessment of nitrogen release due to tree removal.

The USDA OCE (2013) recently released a draft report on *Science-Based Methods for Entity-Scale Quantification of Greenhouse Gas Sources and Sinks from Agriculture and Forestry Practices*. Nitrogen and methane estimates form part of the agricultural methods documented, but do not appear in any of the forest or tree-based management models (e.g., forestry, afforestation, agro-forestry, wood-carb biomass).

2.1.2 GHG emissions in carbon equivalents

The only GHG quantified in detail is CO_2. Figure 1 and Table 1 are representative estimates of CO_2 release and sequestration for the Ohio and four other ALB eradication projects to date. Throughout, we used the commonly reported conversion factor of 3.67 grams CO_2 per gram of carbon.

2.1.3 Cumulative impacts project would contribute to/have on global climate change

The estimated total CO_2 release from five ALB eradication sites in the northeast US is 20 thousand mt (Table 1). This is approximately 0.024 % of total global CO_2 annual emissions based on the Year 2010 estimate of 9.1 Gt C / year (33.5 Gt CO_2) global emissions from industrial sources and an estimated total of 10.0 Gt C / year, including land-use change and deforestation (http:// co2now.org/Current-CO2/CO2-Now/global-carbon-emissions.html accessed 10/30/2013).

Unlike industrial emissions from fossil fuel sources, forests have the advantage of carbon recovery. Through sequestration in new growth, natural and artificial regeneration would re-capture the equivalent of the CO_2 released within 76 years. Under improved management of urban tree health and tree cover, the level of CO_2 sequestration in new tree growth could exceed initial losses due to tree removal and recapture fully all CO_2 within as little as 43 years (see Alternative 2 below).

2.2 Impacts of Project on/by Climate Change

2.2.1 Direct and indirect impact of climate change on the ALB eradication project

Long-term change in climate will vividly impact on *both* the (a) host trees (in urban settings as well as in natural forests) and on (b) ALB survival and population vigor (and that of most other invasive forest pests).

(a) Direct impact of climate change on host trees and on vegetation.
Anticipated changes in the New York City, Chicago, and Toronto urban areas (Nowak et al 2007, Nowak et al 2010, Toronto 2007) include:

- Warmer winter temperatures and longer growing seasons;
- Changes in the seasonality of precipitation and extreme events like droughts and heavy rainfalls;
- Expanded ranges of insects and increased over-winter survival rates;
- Increased frequency and severity of storm events.

Model projections for the northeastern US show a range of impacts on forest productivity due to anticipated climate change. The overall effect on net ecosystem productivity is positive (+75 gC/m^2/year but with notably large variations: from -150 to 350 gC/m^2/year for the region, and -85 to +275 gC/m^2/year at the level of four specific sites) (Aber et al

1995). Tree growth predictions vary because of the intermixing of positive and negative climate effects. Increases in carbon dioxide and temperature may have a positive effect by increasing the rate of tree growth, but only up to a point. Increased temperatures will also increase evapotranspiration, soil drying, and the frequency of short-term droughts, which would limit water availability for tree growth (Wilmont 2011). Overall, drops in tree productivity and associated changes in micro-climate imply stress and vulnerability of trees to pathogen and insect attack.

The expected shift is also toward more damaging events: increases and/or altered frequencies are expected for fire, drought, insect and pathogen outbreaks, prevalence of introduced species, hurricanes, windstorms and ice-storms (Dale and Beyeler 2001).

Important to the issue in ALB management are large differences among major forest species in how and to the extent they will be affected. For example, for northeast US forests, Iverson and Prasad (1998) show that roughly 30 species could expand their range and/or weighted importance at least 10%, while an additional 30 species could decrease by at least 10%. Depending on the global change scenario used, 4–9 species would potentially move out of the United States to the north. Nearly half of the species assessed (36 out of 80) showed the potential for the ecological optima to shift at least 100 km to the north, including seven that could move >250 km. They chose Vinton County in southern Ohio to model climate projections -- among 15 species projected to decline, sugar maple (*Acer saccharum, -14.5%)* and black cherry (*Prunus serotina, -10.0%)* would decline sharply, while sourwood (*Oxydendrum arboretum, +7.5%)* and white oak (*Quercus alba, +4.0%)* would increase in importance. An additional 14 species are projected to change very little under the changed climate. Hence, a major ALB host is projected to be one of the tree species in the northeast US most affected under expected near-future climate shifts.

Overpeck et al (1991) predict the above general patterns for northern pines, as well as large increases in oak abundance in the northern Great Lakes and New England -- namely for black, northern red, and white oak (*Quercus velutina, Q. rubra, and Q. alba,*respectively). They also predicted a severe northern shift for white birch (*Betula papyrifera*), and a large northward expansion for southern pines, as exemplified by loblolly pine (*Pinus taeda*). The significance here is that there is an expected shift away from prime ALB host trees in the future.

A recent case study of impacts on Vermont forests is particularly instructive of the degree of on-going and *already* visible changes within the current geographic reach of ALB. Wilmont (2011) notes that climate changes are already clearly evident in in the State. Temperatures have increased in the larger Northeast region by 1.8F (1.0C) since 1970, with winter temperatures rising faster than summer temperatures. Precipitation has increased by 15-20% over the past 50 years with 67% of this now falling in heavy precipitation events. These and other medium-term trends in climate are anticipated to affect Vermont's forests, including:

- More frequent hot (over 90^{o}F), humid days;

- Longer growing seasons;
- Worsening of air quality in areas where air quality problems already exist;
- Increased heavy downpours;
- More frequent winter thaws and earlier springs;
- Less winter precipitation falling as snow and more as rain;
- Earlier spring snowmelt resulting in earlier peak river flows; and,
- More frequent short-term droughts in late summer and fall.

Species distribution has shifted at high elevations in Vermont in the past 40 years. Northern hardwood trees are now able to survive at increasing elevations due to moderating temperatures, outcompeting spruce and fir trees which themselves are increasingly vulnerable to warming. Only slightly less vulnerable are northern hardwood forests whose dominant species are sugar maple, yellow birch and American beech. These forests are expected to be *nearly eliminated* in Vermont, replaced by species that prefer the warmer drier conditions, such as oak and pine species (Karl et al, 2009). Trees stressed from low water availability tend to reduce their defense mechanisms and are more susceptible to insect or disease invasion. Currently, three non-native pests are expanding across Vermont forests, with the potential to severely impact hemlock, ash, and maple species (Wilmont, 2011).

Indirect impact of climate change on host trees and on vegetation. Elsewhere, such as in Britain, surveys of forest tree condition illustrate correlations between drought years and tree crown defoliation as drought, combined with high temperature induces stress in trees and thereby predisposes them to attacks by pests and pathogens. Such stresses are likely to be most intensely felt by street trees because of the 'heat island effect' and also because urban trees are often planted in suboptimal conditions with little area for root expansion and frequent root disturbance from utilities operations. This stress is often found to increase susceptibility to insect pests although evidence is more limited or at least variable, when it comes to pathogens. Changes in the plants' environment have the capacity to alter their palatability to pests, as they can change the way in which plants allocate available resources to growth or defense. For example, increased CO_2 levels can make plants more palatable to some pest species through increases in soluble carbohydrates within phloem tissue (Tubby and Webber 2010).

(b) Direct impact of climate change on Asian longhorned beetle. We anticipate that ALB and other insects will be strongly impacted by projected levels of global (and local) warming. Writ-large, predicted climate changes are likely to increase developmental rates and reduce winter mortality for many insects, leading to multi-voltinism in some species. The increased number of generations per year will also enable pests to evolve and adapt much more effectively to climatic change than their tree hosts, through enhanced dispersal as well as phenotypic and genotypic plasticity.

Prior to Bebber et al (2013), the extent to which crop pests and pathogens alter their latitudinal ranges in response to global warming was *largely unknown*. Using CABI (Centre for Agriculture and Biosciences International) data on a diverse array of 612 crop pests and pathogens worldwide, they demonstrate an average pole-ward shift of 2.7 +/- 0.8 km / year since 1960, but with significant variation in trends among taxonomic groups. Insect pests are strongly influenced: warming generally stimulates insect herbivory at higher latitudes, primarily through increased winter survival; rainfall has also an obvious effect, with defoliating insects responding negatively to drought and borers positively. The observed pole-ward shift in many taxa support the hypothesis that global warming-driven pest movement is already underway.

Because of their generally short life-cycles, great reproductive potential, sensitivity to changes in temperature and, in many cases, great capacity for dispersal, even moderate changes in climate have already had significant rapid impacts on the distribution and abundance of many pests and pathogens. Climate change will also affect pest vectors, parasites and natural enemies. Native pests, some not currently perceived as problematic, may become more damaging. In comparison to insects, trees are very long-lived and will only be able to adapt much more slowly to changes in their local environment. Potentially, this makes them more vulnerable to the rapidly moving and changing organisms around them.

There are few studies on the effect climate change will have on ALB in North America. On the other hand, there are several comprehensive laboratory studies of temperature and humidity impacts on the species' reproduction by Keena and colleagues (Keena, 2006; 2009; Keena and Moore, 2010; Sanchez and Keena, 2013); these provide much insight into the insect's likely response to warming conditions.

At an early point in her ALB research, Keena (2006) noted there was a critical need for information on the basic biology of the ALB. Accurate data and models are seen as essential underpinnings for scientifically predicting the insect's development – with the management goals of optimizing exclusion and eradication treatments and predicting spread rates under different environmental conditions. Detailed studies followed on individuals from Bayside, NY and Ravenswood, IL to assess temperature effects on developmental rates and survival over a wide temperature range -- from 10°C (larvae only), 15°C, 20°C, 25°C, 30°C (egg and larvae only) to 35°C (larvae only).

Keena and colleagues (Keena, 2006; 2009; Keena and Moore, 2010; Sanchez and Keena, 2013) found:

- a nearly *linear* relationship exists between developmental rate of ALB eggs and temperatures between 15°C and 30°C. Using this relationship, they predict that eggs would not hatch at temperatures of 10°C or less. Based on the lower percentage hatch of viable eggs at 30°C compared to 25°C, the upper temperature at which egg development ceases and eggs die is at or above 35°C.

- defining lower and upper temperature limits are essential to accurate modeling. The minimum developmental threshold for instars 1 to 8 is close to 10°C. There is some development at 10°C, at least for early instars; about 20 percent of the larvae will molt to the second instar after about 5 months. The upper threshold at which development stops and death occurs, is probably between 35 and 40°C. Larval survival was higher at 25°C than at lower temperatures and was zero for 35°C by the beginning of the fifth instar. Larvae held at 15°C, 30°C, and 35°C had narrower head capsules and weighed less than those held at 20°C or 25°C from the third instar.

- temperature response depended on ALB provenance. Bayside NY females laid fewer eggs at both 20°C and 25°C than Chicago IL females. The percentage of eggs that were viable did not vary between temperatures or strains. The Chicago IL larvae gained weight faster than those from Bayside NY in later instars. Temperature and its influence on larval weight had profound impacts on whether a larva proceeded to pupation.

- there is no significant difference in female longevity between populations of the two provenances or temperatures. Males tended to lived longer than females at all temperatures and lived longer at 20°C than 15°C or 25°C. The time the females began laying eggs and the order of average number of eggs laid was significant: 25°C > 20°C > 15°C.

- the estimated lower threshold temperature for development of instars 1-5 and the pupal stage was near 10C and was near 12C for the higher instars. Developmental rate was less temperature sensitive for instars 5-9 compared with instars 1-4. Development for all but the first instar was inhibited at constant temperatures at about 30C, and all instars failed to develop at 40C.

- differences in humidity shorten adult longevity, especially at low and high temperatures. Hence, caution should be used in applying predictions based only on temperature relationships developed in the laboratory to field conditions. Moreover, the beetle's ability to seek out locations with optimum temperatures (e.g., sunny perches when it is too cool and shady locations when it is hot) may lessen the adverse effects in both the summer and fall; at least 2^{o}C should be added to air temperatures to adjust for the mediation of temperature by the wood.

In summary, temperature has significant impact on all life history parameters (esp. female fecundity, longevity) assessed and comparable with that Zhou et al (1984) found for *A. glabripennis* (form *nobilis*) adults in China. Straightforward mathematical relationships greatly facilitate development of predictions of potential geographic range that are essential for effective control and eradication efforts. Current summer temperatures throughout most of the lower 48 states should support ALB survival and reproduction. Given ALB survival is brief where summer temperatures for a full day exceed 30^{o}C, continued future warming may adversely impact on the beetle's vigor and establishment in specific regions of the country.

Indirect impact of climate change on Asian long-horn beetle. The level of uncertainty regarding specific climate change impacts makes planning more challenging. The potential implications of this for ALB management cannot be understated. Forest-specific strategies for climate change adaptation are now urgently needed to inform management plans. Most cities and many natural forests across the northeast US and southeast Canada face threats from multiple insect pests; many of the most difficult to manage are exotic invasives that have no or few indigenous biological enemies. In addition to ALB, emerald ash borer (EAB), gypsy moth (GM) and Dutch elm disease (DED) pose significant threats and are being monitored closely as part of control programs to slow widespread damage (Nowak et al, 2010; Toronto, 2007). Climate change is already having an impact by increasing the severity of outbreaks by these pests. This increases greatly the costs of their control, as well as raises the need for and expenditures on tree replacement and for programs designed to maintain tree health. A high premium is placed on strategic management of urban forests and pests to maintain and expand tree cover. Under warming conditions and enhanced needs for pest control efforts, there is increasingly less leeway for delay and discretionary management action.

It is now clear that ALB and other exotic pests are having sizable impact on urban trees. For example, in Chicago, ALB could potentially cause losses to the urban forest of $1.3 billion in structural value (53.6 percent of live tree population), GM $595 million of losses (19% of all trees), EAB $295 million (12% of all trees) and DED, an additional $31 million (5.5 % of all trees). In Toronto, ALB discovered in 2003 can affect 43% of the City's tree population and potentially cause structural losses estimated at $4 billion. Over the 1930-1950 period, DED killed as many as one-third to half of all City trees and could kill an additional 1.6% ($279 million); current outbreaks of GM and EAB threaten an additional 24% of the tree population ($2.1 billion structural value). One result (compounded by urban conversions) is that adequate tree cover in ALB-infested cities is difficult to maintain. Toronto's tree cover diminished 7% over the 1985-2005 period -- in spite of accelerated tree re-planting programs. Tree diversity is now less and vulnerability greater as individual species are selectively killed in successive pest outbreaks (Toronto 2007). In New Your City, ALB represents a potential loss to urban forest of $2.25 billion in structural value (43.1% of the tree population); GM could potentially kill 23% of all trees ($2.21 billion loss); EAB 0.5% of all trees ($9.8 million loss), and DED an additional 0.7% of all trees ($111 million loss).

A recommended tree cover target is an average of 40% canopy cover -- to ensure the sustainability of the urban forest and maximize the ecological, social and economic benefits derived from urban trees (American Forests 2007). Using this standard, Baltimore, Chicago, New York City and Toronto require increases of 16%, 23%, 20% and 20%, respectively, to attain the tree cover target (Nowak et al 2007, 2010, Toronto 2007). A tabulation of percent tree cover in 17 North American cities shows a range of 7% to 37%, with an average of about 25% (Nowak et al 2010) suggesting considerable room to expand urban tree cover to capture the benefits of CO_2 sequestration, carbon storage and environmental benefits as climate changes.

In summary, the nexus of climate change and the presence of ALB and other exotic pests is greatly increasing the demand for concerted management for tree health and sustainability of urban and adjacent natural forests.

2.2.2 Direct and indirect impact of ALB eradication project on climate change

Direct Impacts of eradication project on climate change. The most conspicuous impact of the removal of large numbers of ALB-infested trees is the loss of CO_2 sequestration and release of carbon storage in trees and soil. Our estimates indicate that across the northeast US about

120,000 trees have been removed and chipped, releasing a total of 20,000 mt of CO_2 to the atmosphere since Year 2000. At risk over the five affected States is an estimated 2.4 million mt of CO_2 release from host trees (Table 1).

If left to regenerate or if artificially replanted, the amount of CO_2 released upon removal would be sequestered in new growth over an interval of 76 years on average (or less, if warming generally accelerates tree growth). A potential benefit is the removal of older infested trees. This assumes old trees more vulnerable than young age classes to disease and to damaging weather events (e.g., wind, snow, icing, drought), and are slower growing.

<u>Indirect Impacts of eradication project on climate change</u>. Tree removal under current ALB eradication programs serves to expand and exacerbate the impacts of climate change as we understand them. Below, we identify three categories of impacts where tree removal feeds back to exaggerate/exacerbate the on-going GHGs linked to global climate warming and attendant increases in extreme weather variations.

(1) In urban areas, where trees now shade buildings, tree removal under ALB eradication will increase energy requirements and emissions of GHG from power plants -- to compensate for increased heating in winter and air conditioning in summer.

Based on average state energy costs in February 2009, trees in Chicago are estimated to reduce energy costs from residential buildings by $360,000 annually. Trees are estimated to slightly increase the amount of carbon released by fossil-fuel based power plants. However, this estimated increase in emissions (1,200 tons) is more than offset by annual carbon sequestration by trees (25,200 tons) (Nowak et al 2010). Trees in Toronto are estimated to reduce energy costs from residential buildings by $9.7 million annually. Trees also provide an additional $483,000 CND in value per year by reducing the amount of carbon released by fossil-fuel based power plants, representing a reduction of 17,000 mt of carbon emissions. These values could be increased through more strategic tree planting to maximize the potential energy effects of trees (Toronto, 2007). In New Your City, the energy and cost savings are even greater. Trees there are estimated to reduce energy costs from residential buildings by $11.2 million annually based on 2002 energy costs. Trees also provide an additional $167,000 in value per year by reducing the amount of carbon released by the City's fossil-fuel based power plants (a net reduction of 9,100 tons of carbon emissions) (Nowak et al., 2007).

(2) Reduced tree cover typically results in loss of storm-water regulation and flood moderation. A single, average- size tree (21 m height) evapo-transpires 10-200 liters water per day or about 2000-40000 liters per year per tree, pending tree species and climate conditions (Wullschleger et al., 1997). Trees also lessen soil erosion and nutrient loss to streams and ground water. For example, of total annual precipitation, the combination of canopy interception, surface evaporation and transpiration loss is estimated to be 40-64% in deciduous broadleaf forests and 55-80% in confers forests in the UK (Nisbet, 2005). Hence, trees exert a powerful modulating influence on water dynamics and this function is put at risk as tree cover is diminished.

(3) By removing significant levels of air pollutants, urban trees and forests improve local air quality in five main ways (Nowak et al., 2007; 2010; Toronto, 2007), namely by:

• absorbing gaseous pollutants through leaf surfaces, including ozone (O3) and nitrogen dioxide (NO2)
• intercepting particulate matter (e.g., dust, ash, dirt, pollen, smoke)
• reducing emissions from power generation for heating and cooling of buildings
• releasing oxygen through photosynthesis
• transpiring water and shading surfaces, lowing local air temperatures, and thereby reducing O3 levels.

Not all gases above are GHGs but in total and across the composite of urban areas, these do affect local and regional climates (e.g., heat island effect) and ultimately feed-back variously to the global warming phenomenon. Although trees do emit volatile organic compounds (VOCs) that can contribute to ozone formation, integrative studies have revealed that an increase in tree cover actually leads to reduced ozone formation (Toronto, 2007).

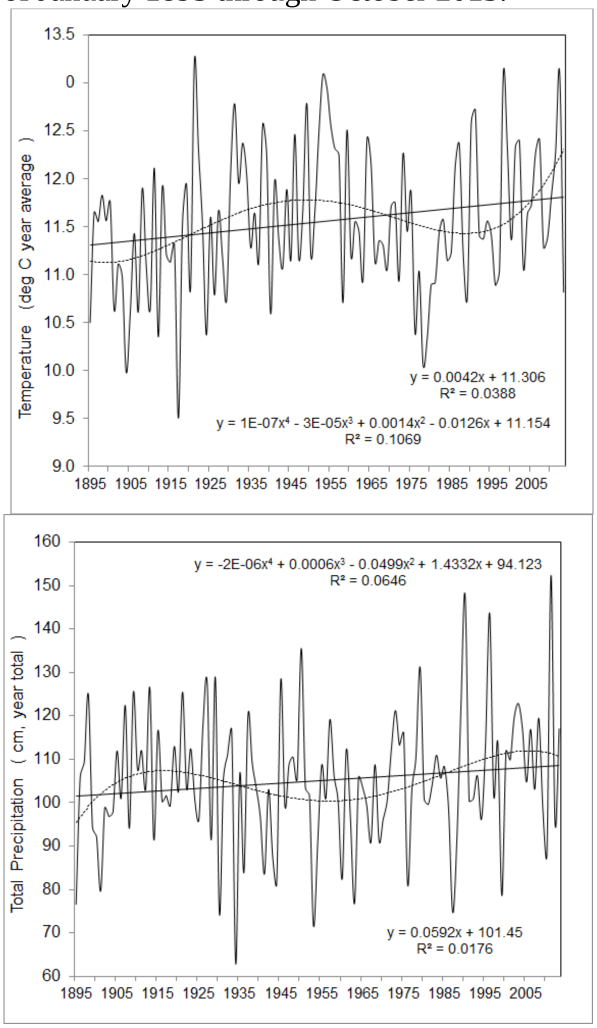
In Chicago trees remove an estimated 888 tons of air pollution (CO, NO_2,O_3, PM10, SO_2) per year with an associated value of $6.4 million (Nowak et al., 2010). In Toronto trees and shrubs remove an estimated 1,430 mt of the same air pollutants per year, with an associated value of $16.1 million (Toronto 2007). In New York City, removal is estimated at 2,202 tons/year with an associated value of $10.6 million/year (Nowak et al., 2007).

2.2.3 Manifestation of evident/known impacts of global climate change in geographic area

Keena and colleagues (Keena, 2006; 2009; Keena and Moore, 2010; Sanchez and Keena, 2013) demonstrate that climate has a strong impact on ALB growth and survival. Hence, the question -- what actual evidence is available to demonstrate impacts on ALB populations have occurred already and continue to occur as a result of global climate change?

Long-term shifts recorded for average temperature and total precipitation at Cincinnati, Ohio are both modest, highly variable, and may be cyclical in nature (Fig. 2a, 2b) (NCDC, 2013). Long-term air temperature records for southwestern Ohio (NOAA Zone 8) indicate an increase of 0.42C / 100 years over the 1895 through 2013 period. This magnitude is modest relative to global temperature change, estimated to be 0.73C / 100 years (Attachment 3).

Figure 2. Yearly (a) average air temperature and (b) average total precipitation in NOAA's meteorological station at Cincinnati's Lunken Field airport. Linear and 4th order polynomial regression trends and equations shown. Data obtained from http://www.esrl.noaa.gov/psd/data/timeseries/ (accessed 11/6/2013) includes observations of January 1895 through October 2013.

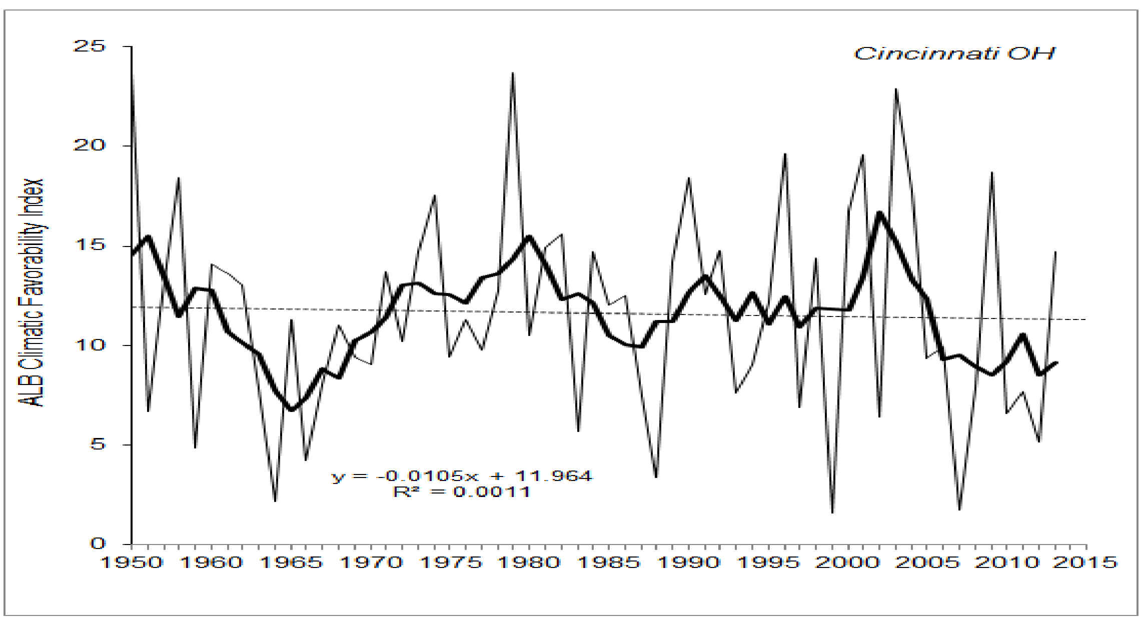


A simple model of the 'favorability' of daily weather conditions, May through October over the 1950 – 2013 period was developed using Keena (2009) observations on ALB response to temperature and humidity conditions. The ALB Climatic Favorability Index is the sum of daily average temperatures 10 C to 33 C, minus the sum of temperature <10 C and > 32C, multiplied by the sum of total precipitation > 0 mm, an calibrated on a scale of 0-100 by dividing by 10,000,000.

The Index is dominated by a cyclic pattern. Based on the linear regression, there is a decrease of 0.95 index points (or 1.6 % per 100 years) relative to 1950. Since first detection on wood packaging material arriving from China in 1996, six years, (1996, 2000, 2001, 2003, 2004, 2009) stand out as particularly favorable for ALB growth (i.e., >17.5 Index); this contrasts to any earlier 18-year period which had two to three such events over 18 years. Six years were relatively unfavorable (<7.5) over the 1996-2013 interval (Fig. 3). These Index values suggest an overall static trend but one punctuated by especially favorable or adverse yearly conditions. It is noteworthy that the present outbreak in Clermont County Ohio was preceded by a five year interval (2000-2004) of favorable climate for ALB, with the one exception of 2002.

In brief, our analysis shows moderate temperature and precipitation shifts long-term in the southwest Ohio zone, consistent with but more moderate than global trends (Attachment 3). Our Index of climatic suitability for ALB shows both favorable and adverse years occur on a frequency of about 1.4 and 2.0 per decade, respectively, but with virtually no overall trend since 1950. Cyclical patterns are conspicuous in all trends, implying a need for caution in stating definitively there is clear evidence of impact of global climate change on geographic area. Outbreak sites on the Great Lakes (Chicago, Toronto), the Atlantic Ocean (Worcester, Boston, New York City, Jersey City) and the Ohio River Valley (Cincinnati) imply that ALB thrives within a 'humidity corridor' and that commonly applied degree-day models require revision to fully incorporate moisture parameters.

Figure 3. Daily estimate of favorability of climate for ALB survival and growth in Cincinnati, OH. ALB Climatic Favorability Index based on Keena et al (Keena, 2006; 2009; Keena and Moore, 2010; Sanchez and Keena, 2013) temperature and precipitation thresholds. Daily maximum and minimum air temperature and daily total precipitation January 1950 through October 2013 observed at Cincinnati Lunken FAA AP, OH, Station 1916 (http://www.esrl.noaa .gov/psd/cgi-bin/data/timeseries/ accessed 11/6/2013). Dashed line is linear regression of the Index trend.



3.0 Alternatives

3.1 No Action Alternative

We posit the question of what would be the impact on CO_2 release under hypothetical conditions in which there was no ALB eradication effort. Keena and Moore (2010) concluded that climatic conditions are suitable for ALB establishment and survival in all contiguous 48 States. To estimate release US-wide, we use tree carbon estimates for urban areas in 48 States published by Nowak and Crane (2000) and convert their weights of C to metric tons (mt) of CO_2 using the standard 3.67 conversion factor.

We then assume a percentage of all trees in any State are ALB host tree species. The percent of all trees that were ALB host species was the percent basal area of trees in each State that were known host trees estimated by Nowak et al. (2003). Second, we estimated the forest area affected adjacent to each urban area. The fraction of trees in forest areas at the last two ALB eradication sites is 2.33-fold greater than that removed from those two urban areas. We use this conversion (vs. that for all five sites) on the consideration that these are more representative of on-going and near-future conditions (compared to the ALB locations early in 2000). Third, the CO_2 released from soil (as in the main CO_2 model) was estimated as 25% of that released from the urban and adjacent forest trees removed. Total potential CO_2 release is the sum of that released from urban trees, forest trees, and soils.

This total was estimated for each of the 5 States with APHIS ALB eradication efforts to date. For comparison, we also estimate the total release for each of the 20 States in the Northeast *plus* North central regions, and for all 48 States (Attachment 4).

Alternative 1a. In this alternative, only urban host trees and those in adjacent forests are involved. Estimates indicate that in the absence of ALB eradication there is a potential release of CO_2 of 478 million mt in the five States, 2,111 million mt from the twenty Northeast *plus* North central States, and 2,255 million mt for all forty-eight States. These estimates represent potential releases under tree removal methods standardly used to date to eradication ALB found in and adjacent to city areas.

Alternative 1b. Urban forest area is only 3.5% of US total area on average; even with the addition of adjacent forest areas (2.33 x that of the urban trees), the area involved is only 11.7% of total. Using the same hypothetical approach and assumptions as above but involving all host trees in the landscape (not just those in urban and adjacent areas), the estimated potential releases are two to six-fold larger (Attachment 5). Potential release of CO_2 is estimated to be 1,030 million mt in the five States, 6,959 million mt from the twenty Northeast *plus* North central States, and 13,906 million mt for all forty-eight States.

3.2 Preferred Alternative

We posit several alternative CO_2 management scenarios to achieve enhanced carbon storage and sequestration rates in trees. Two of these, increased tree health and expanded tree cover were identified as obvious opportunities in urban tree studies, notably those of Nowak et al (2007, 2010) and the City of Toronto (Toronto 2007). The fact that soils typically contain significant quantities of carbon, their treatment during and after eradication efforts also merits attention.

(1) Tree Health. Under programs of active urban tree and forest management, there is an opportunity for replacing trees removed and some of the current trees in poor or declining health with tree species resistant to ALB, and at the same time, better adapted to enhanced warming and climate variability. Beyond this are opportunities to increase tree protection from pests and improve growth of urban trees, thereby increasing CO_2 sequestration and storage.

The fraction of urban trees in unhealthy condition is notable. For example, an urban forest health survey across 20 Northeast and North central States found only 23% of all trees were in good (22%) to excellent (1%) condition. The remaining 77% ranged from fair (27%), to poor (16%), or declining (12%) and otherwise improving and variable condition (18%, and 4% 'other') (Pokorny, 1998) (http://www.na.fs.fed.us/spfo/pubs/uf/survey/execsum.htm accessed 11 08 2013).

(2) Tree Cover. Recent surveys over urban areas in the US (and Canada) show the percent in tree cover is a fraction of what the American Forests (2007) recommends as adequate. Compared to their standard of 40% tree cover, the average over 20 cities was 23.0%. The large five cities with active tree planting programs (Baltimore, Boston, Chicago, New York City, Toronto) had an average of 20.4% and could benefit by doubling their tree cover (Nowak et al., 2010). Re-planting can be difficult to achieve; many US cites over the past decade in fact show increased conversion to building, roads and other infrastructure, hence a relative decrease in urban tree cover (Nowak and Greenfield, 2012).

(3) Soil Disturbance. Impacts of tree removal operations on soil surface disturbance are unreported in the ALB data and remain a large uncertainty in our GHG estimates.

We posit two tree management scenarios designed to address the issues of sub-optimal tree health and sub-par total percent urban area in tree cover. Details of our procedure are documented in Attachment 8.

In the first case, we run the CO_2 Emissions and Sequestration Model to estimate the effect that a continuous increased tree growth rate of 10% would have on the number of years to compensation point (i.e., total sequestration in new growth = total emission from removed trees), and on the magnitude of additional CO_2 sequestration. We consider ten percent is conservative considering that three-quarters of all trees are in fair, poor or worst condition. We assumed that better silviculture and protection against pests will be part of the effort to ensure the enhanced growth is sustained.

The current Ohio eradication site was used to demonstrate what could be achieved in an actual operation. This alternative (Scenario 2) to a 'business-as-usual' approach (Scenario 1) shortened the compensation time by only two years but resulted in an additional 330 mt of CO_2 capture (Fig. 4, Table 2).

The Ohio eradication site was also used to estimate the additional CO_2 sequestered in a hypothetical expansion of tree cover. We assumed an urban tree cover of 25% at the Ohio site and increased this 1.6-fold to attain the American Forests (2007) standard of 40% (Scenario 3), while still ensuring the better silviculture that is part of Scenario 2. The compensation point was reached in 43 years -- or in about half the time of the 'business as usual' scenario (Scenario 1). The additional CO_2 captured by year 85 at the peak of tree growth was significant. Fully1471 mt more CO_2 was sequestered and stored, or twice that in the 'business as usual' case (Fig. 4, Table 2).

Clearly, there is considerable discretionary leeway to improve the sequestration and capture of CO_2 through active management of tree populations in the post-eradication period. Further gains can be made by reducing soil disturbance and by ensuring adequate or improved soil drainage properties.

Figure 4. Estimates of CO_2 sequestration in new tree growth and dynamic of CO_2 net flux under alternative management options of the Ohio eradication site. **Scenario 1** is business as usual. **Scenario 2** is management for 10% increased tree growth rate with improved silviculture and pest protection. **Scenario 3** is a 1.6-fold expansion of tree cover while maintaining enhanced tree growth and protection. Dashed blue line aids in visualizing timing of compensation points (total sequestration = total emissions) of each scenario.

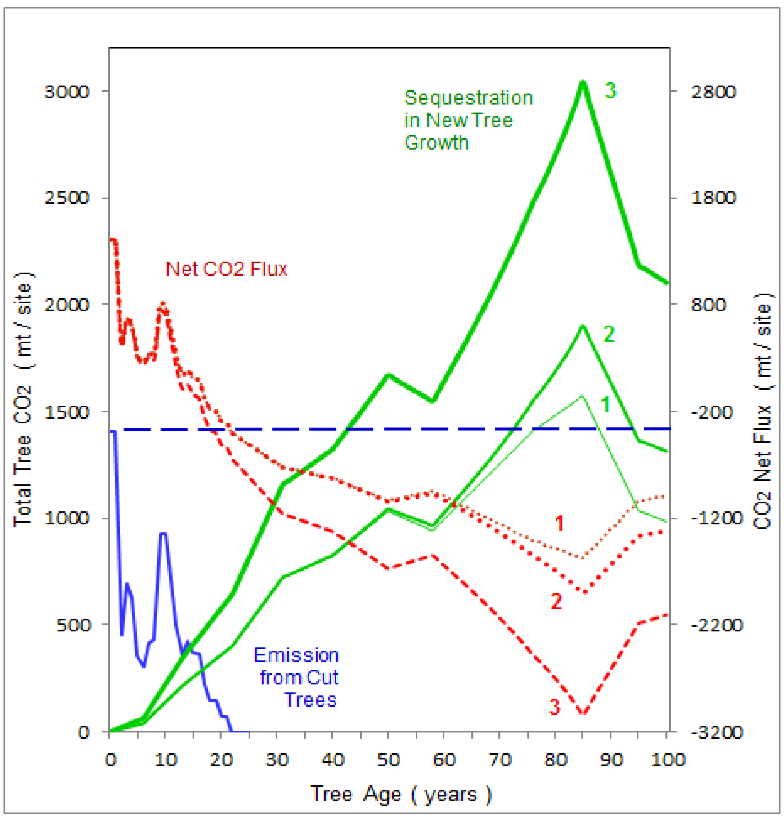


Table 2. Three alternative ALB post-eradication alternatives for management of tree CO_2 sequestration and other benefits. Scenario 1 is 'business-as-usual'. Scenario 2 is management for a continuous 10% acceleration of tree growth, while maintaining enhanced tree protection. Scenario 3 is urban tree cover expanded from 25% to 40% together with enhanced tree growth and protection. Years to achieve compensation, peak CO_2 sequestration level, and gain in total CO_2 over 'business as usual' are given.

CO_2 Management Parameter		**Scenario 1** business as usual	**Scenario 2** accelerated growth	**Scenario 3** expanded tree cover
Compensation point (yrs)		76	74	43
Peak sequestration (mt)		1572	1902	3043
Management gain (mt)		0	330	1471

4.0 Mitigation Measures

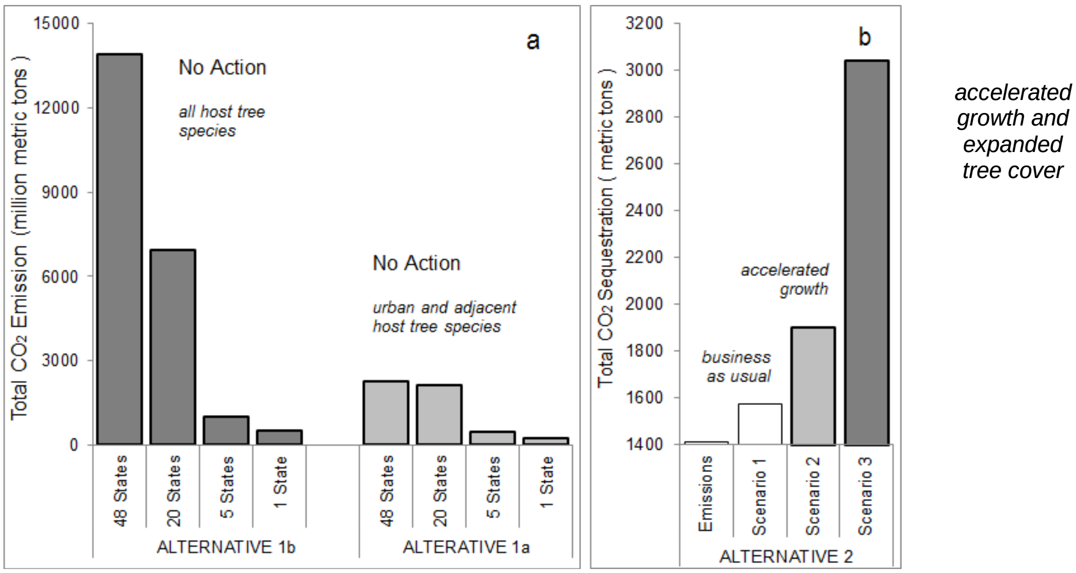
4.1 Reduction of GHG Emissions and Environmental Impacts

The President's Council on Environmental Quality (CEQ) directs agencies to consider and evaluate reasonable alternatives to proposals (Title 40, Code of Federal Regulation, Part 1502.14 (40 CFR 1502.14)). Alternatives proposed to address climate change issues need to be relevant to the proposed action's purpose and need as well as technically and scientifically feasible. Alternatives may include mitigation measures to reduce GHG emissions, affect carbon cycling, or enhance adaptive capacity. Alternatives developed to respond to climate change issues should clearly relate to the cause-effect relationship between the proposal and climate change and have meaningfully different climate change-related effects when compared to the proposal and other alternatives.

4.1.1 Project design to minimize GHG emissions and environmental impacts

ALB eradication approaches will continue to require removal of trees (and pesticide injections) as applied in current methods. The newly introduced regulations on climate change require attention to details of how to minimize release of carbon stored in trees, how to maximize CO_2 sequestration in the post-removal period, and enhance climate adaptation. A comparison of scenarios assessed in this study indicate large emissions in the unlikely event of no action to achieve eradication nation-wide. Alternatively, there is considerable management leeway to increase the rates of tree regeneration post-removal and expand tree cover to levels exceeding pre-treatment (Fig. 5).

Figure 5. Summary of CO_2 emissions under 'No Action' Alternative 1 and 'Enhanced Action' Alternative 2. (a) Alternative 1b is extreme case where all host trees in US are killed; Alternative 1a is case where only urban and adjacent forest host trees are killed by ALB. (b) Alternative 2 is case where CO_2 emissions and sequestration of 'business-as-usual' (Scenario 1) is compared with 10% accelerated growth (Scenario 2), and accelerated growth plus expanded tree cover (Scenario 3) (see Table 2).



4.1.2 Operational changes

Two significant changes in ALB eradication efforts could significantly improve the efficacy of the ALB Eradication Project nationwide. The first is to enhance the surveillance efforts to find ALB early through improved surveillance technology. This could include models of especially high-risk years and locales through the development of GIS-based maps (e.g., Nowak et al., 2003; Kalaris and Crane, 2013) coupled with routinely updated models of ALB climate suitability. Identified "hotspots" based on the most recent climate data could be the focus of intensive field detection surveys. Second, data collection during tree removal in the future could include the records of tree species, tree diameter, soil type and degree of soil disturbance following tree removal. These data would then be used in refining the current approach and model precision.

4.1.3 Compensatory measures

The removal of infected and high-risk trees will result in emissions of CO_2 (and possibly other GHG such as NOx) that feed into the global levels of GHG in the atmosphere. Our analyses show there are two main "offsets" that compensate for these initial emissions.

First, there is recapture of emitted carbon through natural tree regeneration; beyond this, enhanced tree management options that can capture all and even more than the total CO_2

released. Practical and economically feasible silviculture programs can be designed to both accelerate the rate of CO_2 capture by regenerating trees and exceed initial loss levels.

Second, tree cover expansion and continuing tree protection provide numerous advantages of climatic adaptation for urban areas, with measureable human benefits. Payoffs include reduced energy consumption, moderation and/or capture of air pollutants, improved storm-water management, and the presence of shade trees. Adding a strong diversity of tree species is important given successive waves of mortality by exotic pests in the past, and their continuing introduction and establishment (Aukema et al., 2010)

In addition to the cities frequently referred to in this report (Chicago, New York City, Toronto), other cities in the Northeast have active tree expansion programs in progress, such as Boston (http://www. growbostongreener.org/gbg/ accessed 11/05/2013) and Baltimore (http://www.baltimoretreetrust.org/ accessed 11/05/2013). These urban programs provide insight into the practicality and cost advantages of on-going operations and the many added human benefits from increased tree cover in urban areas.

5.0 References

CEQ. 2010. Draft NEPA Guidance on Consideration of the Effects of Climate Change and Greenhouse Gas Emissions. February 18, 2010 Release, Council on Environmental Quality. Washington DC 22p.

USDA APHIS 2013. Asian Longhorned Beetle Cooperative Eradication Program in Clermont County, Ohio Eradication Program in Clermont County, Ohio. United States Department of Agriculture, Marketing and Regulatory Programs, Animal and Plant Health Inspection Service. Revised Environmental Assessment, Draft January 2013. (Agency Contact: Robyn Rose, National Program Manager Asian Longhorned Beetle Eradication Program, USDA APHIS PPQ, 4700 River Road, Riverdale, MD 20737 107 p.

Aber, J.D., S.V. Ollinger, C.A. Federer, P.B. Reich, M.L. Goulden, D.W. Kicklighter, J.M. Melillo and R.G. Lathrop. 1995. Predicting the effects of climate change on water yield and forest production in the northeastern United States. Clim. Res. 5: 205-222. http://cedarcreek.umn.edu/biblio/fulltext/t1002.pdf

American Forests 2007. Urban tree cover forty percent standard. http://www.americanforests.org/

Arthur, M.A., L.M. Tritton, and T.J. Fahey. 1992. Dead bole mass and nutrients remaining 23 years after clear-felling of a northern hardwood forest. Can J For Res 23:1298–1305.

Aukema, J.E., D.G. McCullough, B. Von Holle, A.M. Liebhold, K. Britton and S.J. Frankel. 2010. Historical accumulation of nonindigenous forest pests in the continental United States. BioScience 60: 886-897.

Bebber, D.P., M.A.T. Ramotowski, and S.J. Gurr. 2013. Crop pests and pathogens move polewards in a warming world. Nature Climate Change, Advance Online Publication www.nature.com/natureclimate change doi: 10.1038/nclimate1990.

Christopher, S.F., M.J. Mitchell, M.R. McHale, E.W. Boyer, and D.A. Burns. 2008. Factors controlling nitrogen release from two forested catchments with contrasting hydrochemical responses. Hydrol. Process. 22: 46–62.

Dale, V.H. and S.C. Beyeler. 2001. Challenges in the development and use of ecological indicators. Ecological Indicators 1:3-10..

Iverson, L.R. and A.M. Prasad 1998. Predicting abundance of 80 tree species following climate change in the eastern United States. Ecological Monographs 68:465-485.

Kalaris, T. and M. Crane 2013. Asian Longhorned Beetle degree day modeling and usage. Geospatial Information System (GIS) Presentation, USDA APHIS, Riverdale MD November 19, 2013.

Karl, T.R., J.M. Melillo, and T.C. Peterson. 2009. Global Climate Change Impacts in the United States (eds.). Cambridge Univ. Press. http://downloads.globalchange.gov/usimpacts/pdfs/climate-impacts-report.pdf.

Keena, M.A. 2006. Effects of temperature on *Anoplophora glabripennis* (Coleoptera: Cerambycidae) adult survival, reproduction, and egg hatch. Environ. Entomol. 35(4): 912-921.

Keena, M.A. 2009. Phenology of the Asian longhorned beetle under simulated annual environmental fluctuations. In: K. McManus and K.W. Gottschalk eds. Proceedings, 19th US Department of Agriculture interagency research forum on invasive species 2008: January 8-11 2008; Annapolis, MD. Gen. Tech. Rep. NRS-P-36. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: 37. http://www.nrs.fs.fed.us/pubs/gtr/gtr-nrs-p-36papers/28keena-p-36.pdf

Keena, M.A. and P.M. Moore 2010. Effects of temperature on *Anoplophora glabripennis* (Coleoptera: Cerambycidae) larvae and pupae. Environ. Entomol. 39(4): 1323-1335.

NCDC. 2013. National Climate Data Center. Cincinnati Municipal Airport Lunken Field, OH. US. http://www1.ncdc.noaa.gov/.

Nisbet, T. 2005. Water use by trees. Information Note (April 2005) Forest Research, Forestry Commission, Surrey UK. 8 p. http://www.forestry.gov.uk/pdf/fcin065.pdf/$FILE/fcin065.pdf

Nowak D.J. and D.E. Crane. 2002. Carbon storage and sequestration by urban trees in the USA. Environ. Pollution 116: 381–389.

Nowak, D.J. and E.J. Greenfield. 2012. Tree and impervious cover change in U.S. cities. Urban Forestry and Urban Greening 11: 21– 30.

Nowak, D.J., D. Crane, J. Stevens, and J. Walton 2003. Potential Damage from Asian Longhorned Beetle. Northern Research Station, USDA Forest Service. Syracuse, New York. (http://www.nrs.fs.fed.us /disturbance/invasive_species/alb/local-resources/docs/ALB_potential.pdf accessed 11/13/2013)

Nowak, D.J., R.E. Hoehn, D.E. Crane, J.C. Stevens, and C. Leblanc Fisher. 2010. Assessing urban forest effects and values, Chicago's urban forest. Resour. Bull. NRS-37. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 27 p. http://www.itreetools.org/resources/ reports/ Chicago's%20Urban%20Forest.pdf

Nowak, D.J., R.E. Hoehn, D.E. Crane, J.C. Stevens, and J.T. Walton 2007. Assessing urban forest effects and values, New York City's urban forest. Resour. Bull. NRS-9. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 22 p. http://www.milliontreesnyc.org/ download /pdf/ufore_study.pdf

Overpeck, J.T., P.J. Bartlein, and T. Webb III. 1991. Potential magnitude of future vegetation change in eastern North America: comparisons with the past. Science 254: 692-695.

Pokorny, J.D. 1998. Urban forest health: Needs assessment survey, Results and recommendations. Report NA-TP-01-98 Northeastern State and Private Forestry, USDA Forest Service, St. Paul MN.

Pouyat, R.V., M.J. McDonnell, and S. T. Pickett. 1997. Litter decomposition and nitrogen mineralization in oak stands along an urban-rural land use gradient. Urban Ecosystems 1:117–131.

Sanchez, S.A. and M.A. Keena 2013. Development of the teneral Adult *Anoplophora glabripennis* (Coleoptera: Cerambycidae): time to initiate and completely bore out of maple wood. Environmental Entomology 42: 1-6.

Tubby, K.V. and J.F. Webber. 2010. Pests and diseases threatening urban trees under a changing climate. Forestry 83: 451-459.

Toronto. 2007. Every tree counts: a portrait of Toronto's urban forest. (Forward by R. Ubbens, director), Parks, Forestry and Recreation: Urban Forestry. City of Toronto, Toronto, Ontario, Canada. 63 p. http://www.sfu.ca /grow/science/resources/1304545703.pdf

USDA OCE. 2013. Science-based methods for entity-scale quantification of greenhouse gas sources and sinks from agriculture and forestry practices. Draft report Climate Change Program Office, Office of the Chief Economist, USDA. Washington DC http://usda.gov/oce/climate _change/estimation.htm

Wilmont, S. 2011. Climate change and Vermont's forests. Climate Change Adaptation White Paper Series (May 2011), Vermont Agency of Natural Resources, Secretary's Office, Montpelier, Vermont. http://www.anr.state.vt.us/anr/ climatechange/Pubs/VTCCAdaptForestry.pdf

Wullschleger, S.D., F.C. Meinzer, and R.A. Vertessy. 1997. A review of whole-plant water use studies in trees. Tree Physiology 18: 499-512.

Zhou, J., K. Zhang, and Y. Lu. 1984. Study on adult activity and behavioral mechanism of *Anoplophora nobilis* Ganglbauer. Sci. Silv. Sin. 20: 379-379F.

Attachments

Attachment 1

Major elements, sources of data and assumptions used in development of the CO_2 Emissions and Sequestration Model.

Trees Removed number of trees removed at each of five ALB Eradication Program sites

Percent Urban percent of all trees in urban area and in forest area for each of five sites

Tree Volumes total tree volume, including roots, trunks, branches, leaves estimated using tree species equations of USDA Forest Service COLE Program (http://www. ncasi2.org/COLE/), using 'Individual Tree Spreadsheets' option. Algorithm requires number of trees of selected tree species by diameter (dbh) class, tree age of dbh class, tree height of dbh class, and tree specific equation. Tree age and height estimates are the same as those of Nowak et al (2010). The example below is for sugar maple in urban areas:

DBH Class (inches)	DBH Mid-Point (inches)	(Nowak) Age Estimate (years)	(Nowak) Height Estimate (feet)	Crown Leaf Area (ft2/tree)	Tree Species Volume Coefficient b (scale)	% Number Trees Removed	Number Trees Removed	Tree Volume (cubic feet)
0	0.0	0.0	0.0	0.0	0	0.0	0.0	0
1 to 3	2.0	6.0	17.5	349.8	0.0019290	30.9	2161.0	2631
3.1-6	4.5	13.5	27.7	600.1	0.0019290	22.9	1601.3	15568
6.1-9	6.5	19.5	35.0	828.8	0.0019290	12.9	903.7	23233
9.1-12	10.5	31.5	48.0	1262.1	0.0019290	10.1	708.0	64992
12.1-15	13.5	40.5	56.0	1684.6	0.0019290	6.3	443.3	78551
15.1-18	16.5	49.5	62.6	2085.5	0.0019290	4.9	344.4	101937
18.1-21	19.5	58.5	67.9	2481.1	0.0019290	3.0	210.9	94472
21.1-24	22.5	67.5	71.7	2978.9	0.0019290	3.2	223.9	141015
24.1-27	25.5	76.5	74.1	3468.7	0.0019290	2.6	185.2	154811
27.1-30	28.5	85.5	75.1	4760.4	0.0019290	1.3	93.6	99058
>30	31.5	94.5	74.6	6070.9	0.0019290	1.8	122.7	157756

Tree Species Three tree species are selected to represent a range of coefficient b for tree volume representative of northern hardwoods, namely sugar or "hard" maple, red maple, and silver or “soft” maple. Tree species are not recorded for tree removals on eradication sites (Attachment 1).

Urban and Forest Tree diameter (dbh) and tree heights differ between urban and forest-grown trees such that separate equations are used.

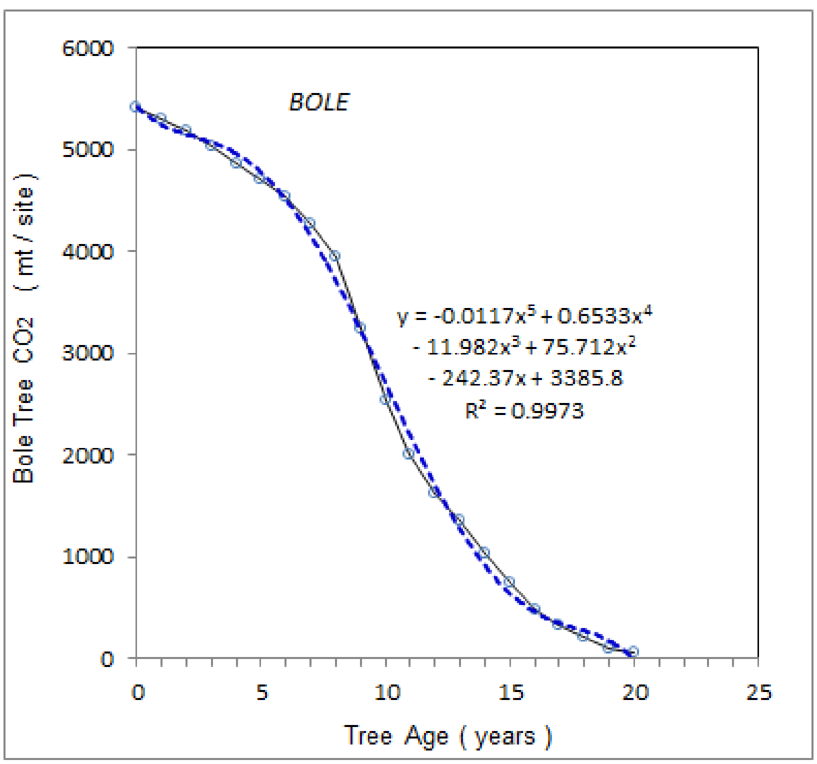
Tree Dry Weights tree volume converted to tree dry weight using coefficient b (above) of each tree species. Total dry weight of each tree component (e.g., foliage) for each dbh size class is summed to calculate total tree dry weight. For example, calculations for sugar maple in urban areas:

Total Foliage Dry Weight (Biomass) (lbs)	Total Top-Only Dry Weight (Biomass) (lbs)	Total Foliage+Top Dry Weight (Biomass) (lbs)	Total Bole-Only Dry Weight (Biomass) (lbs)	Total Stump-Only Dry Weight (Biomass) (lbs)	Total Root-Only Dry Weight (Biomass) (lbs)	Total (AG)Tree Dry Weight (Biomass) (lbs)	Total (AG+BG)Tree Dry Weight (Biomass) (lbs)
0	0	0	0	0	0	0	0
642	7197	7838	17318	1462	5155	26619	31774
3424	38391	41814	92383	7801	27499	141999	169497
4729	53028	57757	127607	10775	37984	196140	234123
11907	133505	145411	321266	27128	95628	493805	589433
13746	154126	167872	370889	31319	110399	570080	680479
17402	195122	212523	469540	39649	139763	721713	861476
16006	179475	195481	431888	36470	128556	663839	792395
24072	269911	293983	649513	54846	193334	998342	1191676
26998	302719	329717	728462	61513	216834	1119692	1336526
17890	200596	218486	482714	40762	143685	741962	885647
29923	335515	365437	807381	68177	240325	1240995	1481321

Tree and Soil CO_2 conversion factors are used to estimate CO_2 levels from estimates of dry weights. Nowak et al (2010) use 0.8 to calibrate for urban trees that typically have less biomass than predicted by forest-derived biomass equations. The species carbon factor (0.5) is the carbon content of wood, converted to CO_2 content using the standard 3.67coefficient. The total carbon at soil surface in urban areas is obtained from Pouyat et al (1997). Soil carbon exposed is a fraction of the crown leaf area (above); this is multiplied by an assumed exposure factor (5%) likely once the tree canopy is removed. The total soil CO_2 release is product of the three prior terms. Total CO2 is the sum of the tree and soil carbon. All US units are then standardized in metric tons (mt).

Nowak Adjust Factor	Species Carbon Factor	C to CO2 Convert Factor	Total Estimated Tree CO2 (US tons)	Total Soil Soil Carbon at Surface (lbs / sq ft)	Total Estimated Soil Carbon Exposed (US tons)	Total Estimated Release Factor %	Estimated Total Soil CO2 Release to Atmosphere (US tons)	Total Estimated Tree C + Soil C Released as CO2 (US tons)
0.8	0.5	3.67	0	1.70	0	5	0	0
0.8	0.5	3.67	24	1.70	643	5	59	83
0.8	0.5	3.67	156	1.70	817	5	75	230
0.8	0.5	3.67	215	1.70	637	5	58	273
0.8	0.5	3.67	541	1.70	759	5	70	610
0.8	0.5	3.67	624	1.70	635	5	58	683
0.8	0.5	3.67	790	1.70	610	5	56	846
0.8	0.5	3.67	727	1.70	445	5	41	768
0.8	0.5	3.67	1093	1.70	567	5	52	1145
0.8	0.5	3.67	1226	1.70	546	5	50	1276
0.8	0.5	3.67	813	1.70	379	5	35	847
0.8	0.5	3.67	1359	1.70	633	5	58	1417

CO_2 Emission The rate of decomposition and release of CO_2 to atmosphere from chipped wood following tree removal are based on those for deciduous hardwood tree species studied in the Northeast US (e.g., Arthur et al 1992 at Hubbard Brook NH). Assumption is made that chipping wood of large boles does not significantly alter natural decomposition rate. Example of CO_2 emission from decomposition of bole wood is shown below.



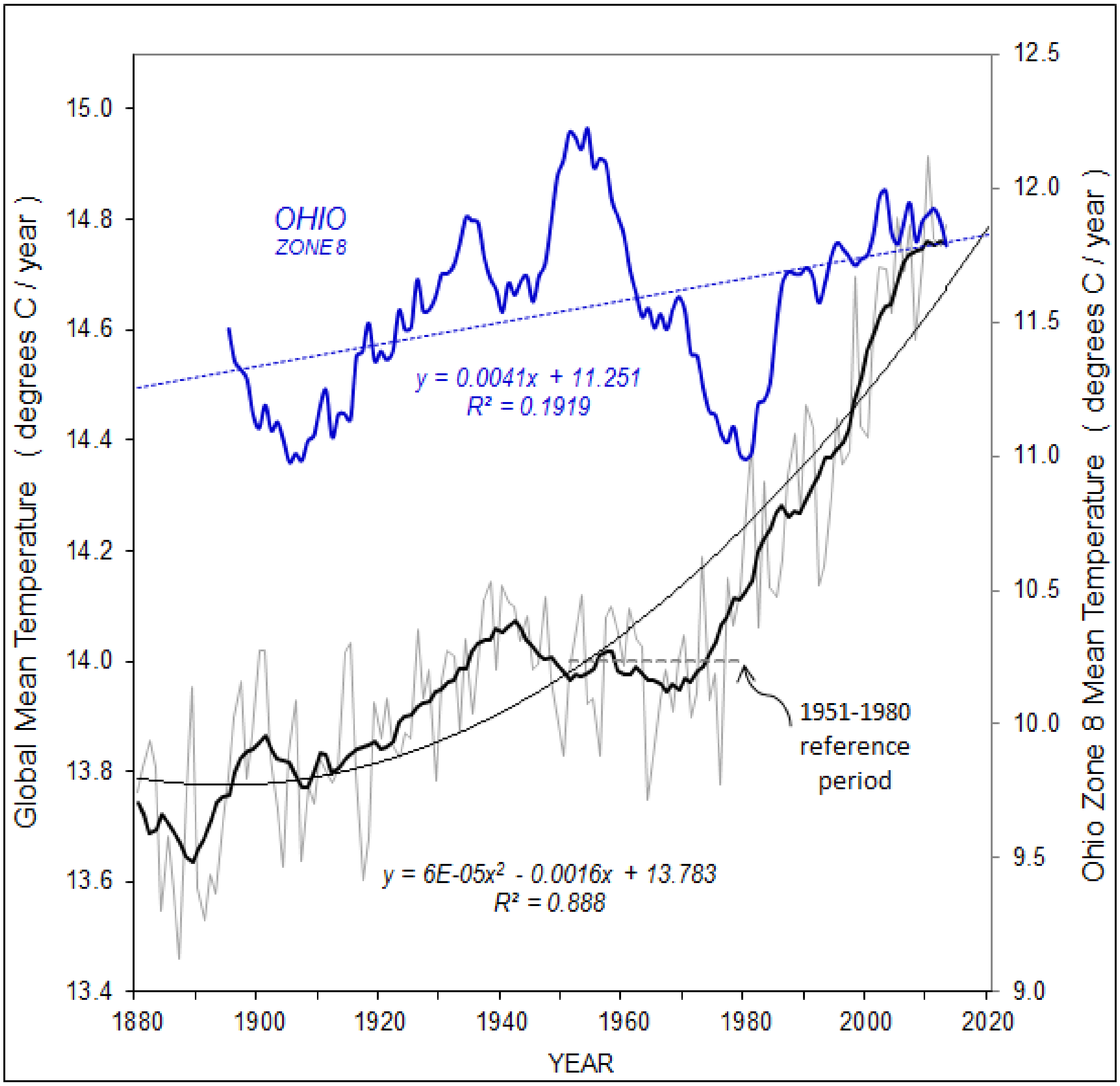
Attachment 2

Coordinates of Cincinnati, OH US Meteorological Station, website and types of data accessed used in models of ALB Climatic Favorability Index for Alternative 2 evaluations.

Latitude: 39° 14' 27" N (deg min sec), 39.2408° (decimal), 3914.45N (LORAN)
Longitude: 84° 29' 58" W (deg min sec), -84.4995° (decimal), 08429.97W (LORAN)
Elevation: 245 meters (804 feet) validated against 244 meters (799 feet) from NED
Contiguous US 1/3E arc second elevation data
Location: Cincinnati, OH US
County: Hamilton, OH
GHCND: USW00093812 Cincinnati Municipal Airport Lunken Field, Oh US
Website: http://www1.ncdc.noaa.gov/
Data Types: TMAX - Maximum temperature (tenths of degrees C)
TMIN - Minimum temperature (tenths of degrees C)
PRCP - Precipitation (tenths of mm)

Attachment 3

Comparison of trend in global mean temperature, 1880-2013, (http://data.giss.nasa.gov/gistemp/tabledata/GLB.Ts.txt accessed 11/13/2013) with that of southwest Ohio, Zone 8, 1895-2013 (http://data.giss.nasa.gov/gistemp/tabledata/GLB.Ts.txt accessed 11/6/2013).
Dark trend lines are 11-point moving averages.



Attachment 4

Estimate of CO_2 release across US 48 contiguous States under Alternative 1 (no action). Assumes full mortality of host tree species in urban and adjacent forests. Metric tons (mt) of carbon of urban areas in each State from Nowak and Crane (2002). Conversion to tons CO_2 based on 3.67 carbon: carbon dioxide ratio. CO_2 released in adjacent forest areas based on a multiplier of 2.33 (see text for details). CO_2 released from soils is 25% of total CO_2 release from urban plus forest trees. Estimates are total tree plus soil CO_2 release in each 48 State, in 20 Northeast and Northcentral States (USDA Forest Service Region 9) and in the 5 States to date with ALB Eradication Programs.

State	Carbon in urban trees	CO2 in urban trees	Percent Host Trees	CO2 release from urban host trees	CO2 release from forest host trees	CO2 release from soils	Total CO2 release host trees + soils	Total CO2 release USDA FS Region 9	Total CO2 release APHIS ALB Programs
	million mt	million mt	%	million mt	million mt	million mt	million mt	million mt	million mt
Alabama	37.8	139	7.8	10.8	25.3	9.0	45.1		
Arizona	9.7	35.7	10.0	3.6	8.3	3.0	14.9		
Arkansas	7.9	29.2	3.8	1.1	2.6	0.9	4.6		
California	27.6	101.2	1.4	1.4	3.3	1.2	5.9		
Colorado	5.2	19.2	22.6	4.3	10.1	3.6	18.1		
Connecticut	8.2	30.2	36.0	10.9	25.4	9.1	45.3	56.7	
Delaware	2.4	8.9	27.4	2.4	5.7	2.0	10.2	16.7	
Florida	31.3	115.0	7.2	8.3	19.3	6.9	34.5		
Georgia	42.7	156.5	8.0	12.5	29.2	10.4	52.2		
Idaho	2.3	8.4	39.9	3.3	7.8	2.8	14.0		
Illinois	28.6	104.9	2.9	3.0	7.1	2.5	12.7	196.6	12.7
Indiana	14.4	53.0	34.4	18.2	42.5	15.2	75.9	99.3	
Iowa	9.6	35.4	35.1	12.4	29.0	10.3	51.7	66.3	
Kansas	4.9	17.9	37.2	6.7	15.6	5.6	27.8		
Kentucky	10.4	38.3	20.0	7.7	17.9	6.4	31.9		
Louisiana	12.6	46.2	12.7	5.9	13.7	4.9	24.4		
Maine	12.7	46.7	33.4	15.6	36.4	13.0	65.1	87.7	
Maryland	16.8	61.6	23.2	14.3	33.3	11.9	59.5	115.5	
Massachusetts	16.1	59.2	28.8	17.0	39.8	14.2	71.0	111.0	71.0
Michigan	20.6	75.6	48.2	36.4	85.0	30.3	151.7	141.7	
Minnesota	23.4	86.0	44.4	38.2	89.1	31.8	159.1	161.3	
Mississippi	12.0	44.1	11.7	5.2	12.0	4.3	21.5		
Missouri	16.0	58.7	9.8	5.8	13.4	4.8	24.0	110.1	
Montana	19.9	73.2	1.9	1.4	3.2	1.2	5.8		
Nebraska	2.1	7.6	13.3	1.0	2.4	0.8	4.2		
Nevada	2.9	10.7	79.0	8.5	19.8	7.1	35.3		
New Hampshire	7.6	28.0	45.7	12.8	29.8	10.7	53.3	52.4	
New Jersey	26.5	97.2	31.7	30.8	71.9	25.7	128.4	182.2	128.4
New Mexico	1.0	3.8	24.7	0.9	2.2	0.8	3.9		
New York	24.6	90.4	8.3	7.5	17.5	6.3	31.3	169.5	31.3
North Carolina	25.5	93.5	19.5	18.2	42.5	15.2	76.0		
North Dakota	0.3	1.2	50.6	0.6	1.4	0.5	2.6		
Ohio	35.2	129.0	43.7	56.4	131.6	47.0	234.9	241.9	234.9
Oklahoma	10.7	39.1	13.5	5.3	12.3	4.4	22.0		
Oregon	6.4	23.5	6.8	1.6	3.7	1.3	6.7		
Pennsylvania	26.6	97.7	44.5	43.5	101.4	36.2	181.1	183.1	
Rhode Island	0.8	2.8	30.2	0.8	2.0	0.7	3.5	5.2	
South Carolina	16.1	59.2	9.8	5.8	13.5	4.8	24.2		
South Dakota	1.1	4.0	10.5	0.4	1.0	0.4	1.8		
Tennessee	30.0	110.0	18.1	19.9	46.5	16.6	83.0		
Texas	25.8	94.7	8.2	7.8	18.1	6.5	32.4		
Utah	3.3	12.2	28.5	3.5	8.1	2.9	14.5		
Vermont	1.4	5.1	12.8	0.7	1.5	0.5	2.7	9.5	
Virginia	29.0	106.3	42.8	45.5	106.1	37.9	189.5		
Washington	17.7	64.8	10.2	6.6	15.4	5.5	27.5		
West Virginia	4.2	15.6	46.6	7.2	16.9	6.0	30.2	29.2	
Wisconsin	10.9	40.0	23.4	9.4	21.8	7.8	39.0	75.0	
Wyoming	0.3	1.0	7.8	0.1	0.2	0.1	0.3		
Total or Average	703.0	2,580.1	24.1	541.1	1,262.6	450.9	2,254.7	2,111.0	478.3

Attachment 5

Extreme estimate of CO_2 release from mortality of all host tree species across US 48 contiguous States under Alternative 1 (no action). Same as in Attachment 6 except that <u>all</u> host trees assumed to be killed (not just those in urban and adjacent forests).

State	Carbon in all trees	Percent host trees	CO2 release from all host trees	CO2 release from soils	Total CO2 release host trees + soils	Total CO2 release USDA FS Region 9	Total CO2 release APHIS ALB Programs
	million mt	%	million mt	million mt	million mt	million mt	million mt
Alabama	530.2	7.8	151.8	37.9	189.7		
Arizona	175.0	10.0	64.2	16.1	80.3		
Arkansas	460.3	3.8	64.2	16.0	80.2		
California	1,157.2	1.4	59.5	14.9	74.3		
Colorado	383.0	22.6	317.7	79.4	397.1		
Connecticut	68.6	36.0	90.7	22.7	113.4	113.4	
Delaware	13.1	27.4	13.2	3.3	16.5	16.5	
Florida	329.7	7.2	87.1	21.8	108.9		
Georgia	613.4	8.0	180.1	45.0	225.1		
Idaho	489.9	39.9	717.4	179.4	896.8		
Illinois	137.4	2.9	14.6	3.7	18.3	18.3	18.3
Indiana	148.6	34.4	187.7	46.9	234.6	234.6	
Iowa	68.3	35.1	87.9	22.0	109.9	109.9	
Kansas	47.6	37.2	64.9	16.2	81.2		
Kentucky	372.5	20.0	273.4	68.4	341.8		
Louisiana	338.3	12.7	157.7	39.4	197.1		
Maine	389.0	33.4	476.8	119.2	596.0	596.0	
Maryland	99.2	23.2	84.4	21.1	105.5	105.5	
Massachusetts	116.4	28.8	123.0	30.8	153.8	153.8	153.8
Michigan	480.9	48.2	850.6	212.7	1,063.3	1,063.3	
Minnesota	275.8	44.4	449.4	112.3	561.7	561.7	
Mississippi	459.9	11.7	197.5	49.4	246.9		
Missouri	359.5	9.8	129.3	32.3	161.6	161.6	
Montana	477.0	1.9	33.3	8.3	41.6		
Nebraska	26.3	13.3	12.8	3.2	16.0		
Nevada	76.1	79.0	220.5	55.1	275.6		
New Hampshire	159.4	45.7	267.3	66.8	334.1	334.1	
New Jersey	63.1	31.7	73.4	18.4	91.8	91.8	91.8
New Mexico	207.0	24.7	187.6	46.9	234.6		
New York	618.7	8.3	188.5	47.1	235.6	235.6	235.6
North Carolina	554.2	19.5	396.6	99.2	495.8		
North Dakota	10.9	50.6	20.3	5.1	25.4		
Ohio	264.7	43.7	424.5	106.1	530.6	530.6	530.6
Oklahoma	171.1	13.5	84.8	21.2	106.0		
Oregon	1,155.8	6.8	288.5	72.1	360.6		
Pennsylvania	579.4	44.5	946.2	236.6	1,182.8	1,182.8	
Rhode Island	13.4	30.2	14.8	3.7	18.6	18.6	
South Carolina	348.6	9.8	125.4	31.3	156.7		
South Dakota	26.0	10.5	10.0	2.5	12.5		
Tennessee	437.0	18.1	290.3	72.6	362.8		
Texas	570.2	8.2	171.6	42.9	214.5		
Utah	189.7	28.5	198.4	49.6	248.0		
Vermont	157.1	12.8	73.8	18.5	92.3	92.3	
Virginia	508.4	42.8	798.5	199.6	998.2		
Washington	1,024.3	10.2	383.4	95.9	479.3		
West Virginia	446.5	46.6	763.7	190.9	954.6	954.6	
Wisconsin	357.9	23.4	307.4	76.8	384.2	384.2	
Wyoming	162.0	7.8	46.4	11.6	58.0		
Total or Average	15,956.8	23.7	11,124.8	2,781.2	13,906.0	6,959.1	1,030.0

Attachment 6

Assumptions used in runs of the CO_2 Emissions and Sequestration Model to estimate levels of sequestration under the three scenarios of Alternative 2.

Scenario 1 all data and model algorithms as documented in Attachment 3 for 'standard' run of CO_2 Emissions and Sequestration Model.

Scenario 2 initial total CO_2 sequestration of Year 1 is compounded annually at rate of 0.10. This estimate of CO_2 sequestration in accelerated growth is then added to total CO_2 estimates of Scenario 1 standard run over full 100 year sequence post-tree removal by ALB Eradication Program.

Scenario 3 yearly estimates of total CO_2 sequestration of Scenario 2 are multiplied by 15% increase over the extant 25% urban tree cover, or by 1.6. This provides an estimate of CO_2 sequestration where total urban tree cover is 40% as recommended by standards for US cities (American Forests 2007).

Appendix E. Herbicide Human Health and Ecological Risk Assessment: Asian Longhorned Beetle Eradication Program

Table of Contents

EXECUTIVE SUMMARY

The United States Department of Agriculture (USDA) Animal and Plant Health Inspection Service (APHIS) proposes to use the herbicides triclopyr or as a mixture using triclopyr, imazapyr, and metsulfuron-methyl to control sprouting vegetation from stumps that are host species for the Asian Longhorned Beetle (ALB). These herbicide treatments are needed as a way to prevent reinfestation of ALB-host trees that have been removed as part of the ALB Eradication Program. The preferred method of control for stumps is physical removal; however, in some cases, the use of herbicides is required.

USDA APHIS evaluated the potential human health and ecological risks from the proposed use of triclopyr, imazapyr, and metsulfuron-methyl for the ALB Eradication Program. The risks to human health are expected to be negligible based on limited exposure from the proposed use pattern of these herbicides (hand painting and backpack spraying). Exposure is greatest for workers who will apply the product. The potential exposure for workers is low with the proper use of required protective equipment. The potential exposure for the general public is also minimal. Risks are quantified for workers and the general public to represent extreme exposure scenarios including accidental conditions. The conservative risk evaluation results show that the hazard index for workers and the general public do not exceed one (the USEPA level of concern), indicating that the exposure is unlikely to cause adverse health effects. Therefore, triclopyr and triclopyr mixed with imazapyr and metsulfuron-methyl used in the ALB Eradication Program should pose minimal risk to human health.

The risk of herbicide use to non-target fish and wildlife is also minimal. The proposed use pattern reduces potential exposure to most non-target fish and wildlife. Wild mammals and birds are at very low risk from herbicide applications due to the low toxicity of all three herbicides and the lack of anticipated effects to food sources that they use. Aquatic organisms are also at low risk based on the favorable toxicity profile for all three herbicides and expected residues that could occur in aquatic environments from the proposed applications. Non-target terrestrial plants are at the greatest risk from herbicide treatment; however, the method of application and selective use of herbicides as a treatment for stumps will reduce the risk to terrestrial plants.

1.0 INTRODUCTION

This human health risk assessment (HHRA) and ecological risk assessment (ERA) provides a quantitative and qualitative evaluation of the potential risks and hazards to human health, nontarget fish, and wildlife from exposure to the herbicides triclopyr, imazapyr, and metsulfuron-methyl when used to control the regrowth of stumps of host species of the Asian Longhorned Beetle (ALB).

The methods used in this HHRA to assess potential human health effects follow standard regulatory guidance and methodologies (NRC, 1983; USEPA, 2014a), and generally conform to other Federal agencies such as U.S. Environmental Protection Agency (USEPA), Office of Pesticide Programs (USEPA/OPP). The methods used in this ERA to assess potential ecological risk to nontarget fish and wildlife follow USEPA methodologies and other published methodologies with an emphasis on those used by USEPA/OPP in the pesticide registration process.

The risk assessment is divided into four sections. The first is the problem formulation (identifying hazard), followed by the the effects analysis or dose-response assessment, and then the exposure analysis (identifying potentially exposed populations and determining potential exposure pathways for these populations). The fourth section of the risk assessment integrates the information from the exposure and effects analysis to characterize the risks to human health and the environment. In addition, the uncertainties associated with the risk assessment and potential cumulative impacts are discussed.

2.0 PROBLEM FORMULATION

APHIS is proposing the use of the herbicides triclopyr, or triclopyr mixed with imazapyr and metsulfuron-methyl to treat stumps and associated sprouts from host trees that were removed to prevent the further spread of ALB. When possible, APHIS will physically remove host trees along with the stumps to prevent reinfestation. However, physical removal of the stumps may not be possible in some situations such as when it is impractical to move stump removal equipment into an area, or there may be restrictions for habitat protection. In situations where stump removal is not feasible, APHIS will apply herbicides to treat the remaining stumps and associated sprouts.

Triclopyr is an herbicide that was first registered in 1979 (USEPA, 1998; USDA FS, 2011a). Triclopyr imitates a plant hormone (indoleacetic acid) classified as an auxin, and is used to control woody plants and broadleaf weeds (Cox, 2000). Imazapyr is an herbicide first registered in 1985 (USEPA, 2006). Imazapyr is an imidazolinone herbicide that inhibits acetohydroxyacid synthase, an enzyme involved in the biosynthesis of amino acids such as leucine, isoleucine and valine (HSDB, 2014b). Imazapyr is a systemic, non-selective, pre- and post-emergent herbicide that acts by inhibiting the synthesis of branched-chain amino acids in plants (USEPA, 2006). It is used for the control of a broad range of terrestrial and aquatic weeds. Metsulfuron-methyl is a sulfonylurea herbicide that inhibits the enzyme that catalyzes the biosynthesis of branched-chain amino acids (valine, leucine, and isoleucine) which are essential for plant growth (USDA FS, 2011c).

APHIS proposes the use of two triclopyr formulations for the treatment of stumps, Garlon® 3A and Pathfinder® II. Garlon® 3A contains the active ingredient triclopyr triethylamine salt (TEA). Pathfinder® II contains the active ingredient triclopyr butoxyethyl ester (BEE). Pathfinder® II allows more flexibility in being able to treat the bark instead of direct application to cut areas of the stem. In addition, APHIS is proposing some foliar applications of Garlon® 3A that will be mixed with two other herbicides. The active ingredients in these two other herbicides are imazapyr and metsulfuron-methyl in Arsenal® and Escort® XP, respectively. They will be used to treat sprouting foliage from stumps that are removed as part of the eradication efforts. All applications will be made either by hand painting undiluted material on the stump or directly spraying stumps and/or sprouting foliage using a backpack sprayer.

The following sections discuss the chemical description and product use; physical and chemical properties; environmental fate; and hazard identification for these herbicides.

2.1 Chemical Description and Product Use

Triclopyr or triclopyr acid (C7-H 4-Cl3-NO3) (CAS No. 55335-06-3) is the common name for [(3,5,6-trichloro-2-pyridinly)oxy]acetic acid. There are no active commercial products for triclopyr acid. Triethylamine (TEA) salt and butoxyethyl ester (BEE) are two commercial forms of triclopyr. Garlon® 3A (EPA Reg. No. 62719-37) contains the active ingredient TEA (44.4%) and Pathfinder® II (EPA Reg. No. 62719-176) contains the active ingredient BEE (13.6%). Garlon® 3A also includes the chelating agent ethylene diamine tetraacetic acid and ethanol (USDA FS, 2011a). The other ingredients of Pathfinder® II were not specified by the manufacturing company.

The proposed application methods for triclopyr are painting the undiluted triclopyr formulation on the surface of stumps or directly spraying stumps and /or sprouting foliage using a backpack sprayer with triclopyr mixed with imazapyr and metsulfuron-methyl. In spray applications, the herbicide sprayer or container is carried by backpack. The nozzle on the wand or gun jet of the backpack sprayer should not be positioned higher than the handlers' waist, reducing the likelihood that the chemical will come into direct contact with the arms, hands, or face of the worker. In addition, a large coarse droplet size applied as close to the target area as possible is used to minimize the potential for drift.

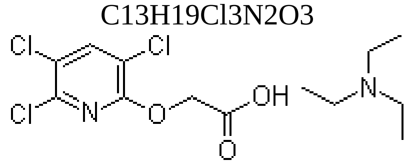
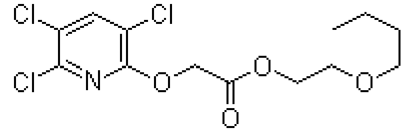
Garlon® 3A is used at rates of 3/4 to 9 pound (lb) acid equivalent (a.e.) of triclopyr (1/4 to 3 gallons of Garlon® 3A) per acre to control broadleaf weeds and woody plants (Dow AgroSciences, 2011a). Pathfinder® II is used at no more than 8 lb a.e. per acre per year on non-crop areas (Dow AgroSciences, 2011b). The Garlon® 3A and Pathfinder® II label requirements for the restricted-entry interval are 48 hours and 12 hours, respectively (Dow AgroSciences 2011a; 2011b). During the restricted entry interval, entry is not allowed except for workers with the proper personal protective equipment (PPE).

Imazapyr (CAS No. 81334-34-1) is an imidazolinone herbicide while metsulfuron-methyl (CAS No. 74223-64-6) is a sulfonylurea herbicide. Both products are a common mix partners with triclopyr in the control of woody vegetation. Arsenal® (EPA Reg. No. 241-346) contains 27.8% or 28.7% isopropylamine salt of imazapyr and 72.2% or 71.3% other ingredients. The Escort® XP formulation (EPA Reg. No. 352-439) contains 60% metsulfuron-methyl and 40% other ingredients. For low-volume foliar brush control, the Arsenal®application rate is 0.5 to 1% by volume and 2 oz per acre when in a tank mix with Escort® XP. Garlon® 3A may be added to the mix at 1 to 2 pints per acre (BASF, 2012a). The Arsenal® and Escort® XP label requirements for the restricted-entry interval are 48 hours and 4 hours, respectively (BASF, 2012a; DuPont, 2012a).

2.2 Physical and Chemical Properties

Triclopyr is a colorless solid with a melting point of 148-150^{o}C. Triclopyr TEA is a grayish white granular solid with a melting point of 111-117^{o}C (USEPA, 1998). Triclopyr acid, triclopyr TEA, and triclopyr BEE have vapor pressures of 1.26 x 10^{-6}, <1 x 10^{-8}, and 3.6 x 10^{-6} mm Hg, respectively, indicating that these compounds can volatilize into vapor and be transported as a vapor or in aparticulate phase into the ambient air. Triclopyr acid and triclopyr TEA are soluble (water solubility of 430 mg/L and 4.12 x 10^5 mg/L, respectively) (HSDB, 2014a). Triclopyr BEE has relatively low solubility (6.8 mg/L) (USEPA, 1998). The basic physical and chemical properties of the commercial products, triclopyr TEA and triclopyr BEE are summarized in table 2-1.

Table 2-1. Physical and chemical properties for triclopyr TEA and BEE.

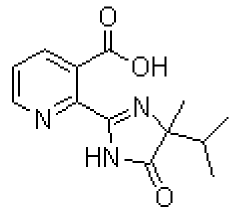
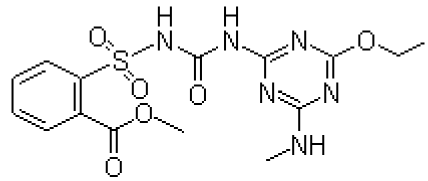
Parameters	**Triclopyr TEA**	**Triclopyr BEE**
CAS No.	57213-69-1	64700-56-7
Molecular Formula	C13H19Cl3N2O3	C13H16Cl3NO4
Molecular Structure		
Molecular Weight	358.67	356.63
Melting Point (^{o}C)	111-117^{o}C	NA
Henry Law Constant (atm m3 mol-1)	1.15 x 10^{-14}	2.47 x 10^{-7}
Vapor Pressure (mm Hg) (25^{o}C)	<1 x 10^{-8}	3.6 x 10^{-6}
Water Solubility, mg/L	412,000	7.4
Log Kow	1.5011	4.0133

Sources: USEPA, 1998; HSDB, 2014a

Imazapyr is a clear, slightly viscous, pale yellow to dark green aqueous liquid, or white to tan powder with a slight ammonia odor. Its melting point is 171^{o}C. Imazapyr has a low vapor pressure (1.79x10^{-11} mm Hg), and low Henry's Law constant (7.08x10^{-17} atm-cu m/mol), suggesting low volatility from soil and water. It is considered highly soluble (water solubility of 1.13x10^4 mg/L) (HSDB, 2014b). The basic physical and chemical properties of imazapyr are summarized in table 2-2.

Metsulfuron-methyl is a white or colorless crystal, or white to pale yellow solid with a faint, sweet, ester-like odor with a melting point of 163^{o}C. Metsulfuron-methyl has low vapor pressure (2.5x10^{-12} mm Hg), and low Henry's Law constant (1.32x10^{-16} atm-cu m/mol). It also has a high solublility in water (water solubility of 9.50x10^3 mg/L) (HSDB, 2014c). The basic physical and chemical properties of metsulfuron-methyl are summarized in table 2-2.

Table 2-2. Physical and chemical properties for imazapyr and metsulfuron-methyl.

Parameters	Imazapyr	Metsulfuron-methyl
CAS No.	81334-34-1	74223-64-6
Molecular Formula	C13H15N3O3	C14H15N5O6S
Molecular Structure		
Molecular Weight	261.28	381.37
Melting Point (^{o}C)	171	163
Henry Law Constant (atm m3 mol-1)	7.08 x 10^{-17}	1.32 x 10^{-16}
Vapor Pressure (mm Hg) (25^{o}C)	9.0 x 10^{-11}	2.5 x 10^{-12}
Water Solubility, mg/L	1.13E+04	9.5E+03
Log Kow	0.22	2.2

Source: HSDB, 2014b, c

2.3 Environmental Fate

The environmental fate describes the processes by which triclopyr, imazapyr, and metsulfuron-methyl move and degrade in the environment. Environmental fate processes include: 1) mobility, persistence, and degradation in soil, 2) movement to air, 3) migration potential to groundwater and surface water, and 4) plant uptake.

In soil, triclopyr is expected to have high mobility (Koc's ranging from 1.5 to 134 mL/g organic carbon (oc)). Under anaerobic conditions, triclopyr is persistent with a half-life of approximately 1,300 days. Under aerobic conditions, triclopyr biodegrades in silty clay loam and silty loam soils with half-lives of 8 and 18 days, respectively. The half-life of the major metabolite, 3,5,6-trichloro-2-pyridinol (TCP), is 30 to 90 days (HSDB 2014a). In soil, both forms of triclopyr degrade into several intermediates before ultimately degrading to carbon dioxide (CO_2) (NPIC, 2002). In air (ambient atmosphere), triclopyr is expected to exist in both the vapor and particulate phase based on its vapor pressure (1.26 x 10^{-6} mm Hg at 25 ^{o}C). Triclopyr in the vapor phase will degrade to hydroxyl radicals through a photochemical reaction with an estimated half-life of 3.3 days. Triclopyr undergoes photodecomposition with a half-life of <12 hours. In water, triclopyr is mainly broken down by exposure to sunlight. The half-life of triclopyr in water ranges from 1 to 10 days depending on water conditions such as turbidity (NPIC, 2002). Triclopyr degrades slowly in a soil:water system incubated aerobically, with a reported half-life of 142 days (HSDB 2014a). The potential for bioconcentration in aquatic organisms is low with an estimated bioconcentration factor (BCF) of 3. Triclopyr's half-life in plants ranges from 3 to 10 days, and the primary metabolite is 3,5,6-trichloro-2-methoxypyridine (NPIC, 2002).

In soil, imazapyr is expected to have high mobility (Koc's ranging from 8.2 to 110 mL/g oc for soil) (USEPA, 2007). Imazapyr biodegrades with aerobic soil half-lives ranging from 17.7 to 63.1 days. In ambient atmosphere, imazapyr will likely exist in the particulate phase based on its vapor pressure (an estimated vapor pressure of 9.0×10^{-11} mm Hg at 25^{o} C). In water, the potential for bioconcentration in aquatic organisms is low due to an estimated BCF of 3. Exposure to ultraviolet (UV) light in aqueous solutions results in complete degradation of imazapyr in 48 hrs, with a half-life of 7 hrs (HSDB, 2014b).

In soil, metsulfuron-methyl is expected to have moderate to high mobility (Koc values ranging from 4 to 345 mL/g oc). Metsulfuron-methyl is more mobile in alkaline soils than in acidic soils. A typical half-life for metsulfuron-methyl in soil is 30 days (ranging from 14 to 180 days). Soil temperature, moisture content, and pH influence degradation of metsulfuron-methyl. Metsulfuron-methyl degrades faster under high moisture content and high temperature and acidic conditions (Trevathan, 2002). In air, metsulfuron-methyl will exist solely in the particulate phase (vapor pressure of 2.50×10^{-12} Hg at 25^{o}C). Metsulfuron-methyl may undergo direct photolysis based on 50 and 76% degradation in an aqueous solution after a 15 and 36 hour exposure to UV irradiation, respectively. In water, metsulfuron-methyl may undergo direct photolysis. It is not expected to adsorb to suspended solids and sediment based upon its Koc values. Metsulfuron-methyl has BCF values ranging from 1-17 suggesting the potential for bioconcentration in aquatic organisms is low (HSDB, 2014b).

2.4 Hazard Identification

Technical triclopyr acid, TEA, and BEE are slightly toxic for the oral and dermal exposure routes (Toxicity Category III). They are practically non-toxic for the inhalation exposure route (Toxicity Category IV) and are not dermal irritants. However, triclopyr TEA was corrosive and BEE was minimally irritating in the primary eye irritation study. Triclopyr TEA and BEE are dermal sensitizers. The primary target organs of triclopyr are the liver and kidney (USEPA, 1998; USEPA, 2002a).

Imazapyr is practically non-toxic for oral (Toxicity Category IV) and dermal (Toxicity Category III) routes of exposure. For the acute inhalation route of exposure, imazapyr is classified in Toxicity Category II. It is not irritating to the skin, and is negative for dermal sensitization. However, imazapyr causes acute irreversible eye damage (Toxicity Category I) (USEPA, 2006).

Metsulfuron-methyl is non-toxic for oral, dermal, and inhalation exposure routes (Toxicity Category IV). It is not a skin irritant or sensitizer, but is a slight eye irritant (USEPA, 2002b).

2.4.1 Mechanism of Action and Pharmacokinetics

Triclopyr, a weak acid, is excreted primarily from the kidney through an active transport process (Timchalk and Nolan, 1997; Timchalk et al., 1990; 1997). At very high doses, it may interfere with the excretion of other weak acids. However, concentrations of weak acids in the body under normal environmental exposures will be far below levels that would interfere in the active transport process. Therefore, this mechanism of active transport is not expected to play a substantial role in potential health effects. Imazapyr is a plant amino acid synthesis inhibitor, which inhibits acetolactate synthase (ALS), an enzyme found only in plants and microorganisms. Plants require ALS for the synthesis of essential branched chain amino acids for their growth. Animals lack ALS and do not synthesize these amino acids by themselves (USDA FS, 2011b). A mechanism of action for imazapyr in mammals is currently unknown (USEPA, 2005a; HSDB, 2014b). Metsulfuron-methyl is also a plant amino acid synthesis inhibitor with the same mechanism of action as imazapyr. The mechanism of action of metsulfuron-methyl in mammals is not clear (USDA FS, 2005).

Pharmacokinetic studies show that triclopyr is absorbed and excreted almost exclusively in the urine through acid hydrolysis (Shackelford et al., 1999). Following oral exposure, triclopyr is absorbed and excreted relatively rapidly, with half-lives for oral absorption and urinary excretion of 3.61 and 1.1 hours, respectively. A majority of ingested triclopyr is excreted unchanged in the urine, although minor metabolites can be formed (USFS, 2011a). Triclopyr has very low potential to be absorbed through the skin or to accumulate in humans at acutely toxic levels because it is absorbed slowly through the skin and is rapidly eliminated (HSDB, 2014a). Imazapyr and metsulfuron-methyl are absorbed and excreted through urine and feces mostly unchanged (HSDB 2014b,c). In a rat study, 87% of an orally administered imazapyr dose was excreted in the urine and feces within 24 hours (HSDB, 2014b). Metsulfuron-methyl is eliminated from rats quickly (9-16 hours at low doses, and 23-29 hours for high doses) (HSDB, 2014c).

2.4.2 Acute Toxicity

The acute oral rat LD_{50} values for TEA, Garlon® 3A, BEE and Pathfinder® II formulations show low toxicity (Category III) (table 2-3). Acute toxicity values for BEE are lower compared to those for TEA and Garlon® 3A, but are still within the same toxicity category. The TEA, BEE and Pathfinder® II formulations have low dermal toxicity (Category III) and Garlon® 3A has very low (Category IV) acute dermal toxicity. The acute inhalation toxicities for TEA, Garlon® 3A, BEE and Pathfinder® II formulations are all very low (Category IV). None of the formulations are dermal irritants. Triclopyr TEA, Garlon® 3A, and BEE are dermal sensitizers. However, Pathfinder® II is not a dermal sensitizer. The primary eye irritation study results between the technical active ingredients and the proposed formulations vary from corrosive (triclopyr TEA - Category I), to may cause severe irritation with corneal injury (Garlon® 3A), and from minimally irritating (BEE) to slight temporary eye irritation (Pathfinder® II).

Table 2-3. Comparative acute mammal toxicity between triclopyr technical active ingredients and proposed formulations.

Toxicity Study	TEA (44.4% a.i.)	Garlon® 3A	BEE (97.1% a.i.)	Pathfinder® II
Acute Oral LD_{50} (rat)	1,847 mg/kg (M&F) (III)	1,847 mg/kg (III)	803 mg/kg (M&F) (III)	1,000 mg/kg (F) (III)
Acute Dermal LD_{50} (rat for technical and rabbit for Garlon® 3A)	>2,000 mg/kg (III)	>5,000 mg/kg (IV)	>2,000 mg/kg (III)	>2,000 mg/kg (III)
Acute Inhalation LC_{50} (rat)	>2.6 mg/L (IV)	>2.6 mg/L (IV)	>4.8 mg/L (IV)	>5.0 mg/L (IV)
Primary Eye Irritation (rabbit)	Corrosive (I)	May cause severe irritation with corneal injury	Minimally irritating (III)	Slightly temporary irritation
Primary Skin Irritation	Not irritating (IV)	Brief contact- not irritating; prolonged contact may cause slight skin irritation with local redness; repeated contact may cause skin burns	Not irritating (IV)	Prolonged skin contact may cause moderate skin irritation with local redness
Dermal Sensitization (Guinea pig)	Sensitizer	Sensitizer	Sensitizer	Non-sensitizer

Sources: USEPA, 1998; Dow AgroSciences, 2010; 2011c

Acute mammal toxicity data for imazapyr and Arsenal® show very low toxicity for oral, dermal, and inhalation routes (table 2-4). The formulation is much less toxic causing no eye irritation while the technical imazapyr can cause irreversible eye damage.

Table 2-4. Comparative acute mammal toxicity between imazapyr technical active ingredient and the proposed formulation.

Toxicity Study	**Imazapyr Technical**	**Arsenal®**
Acute Oral LD_{50} (rat)	>5,000 mg/kg (IV)	>5,000 mg/kg (IV)
Acute Dermal LD_{50} (rat)	>2,000 mg/kg (III)	>2,000 mg/kg (III)
Acute Inhalation LC_{50} (rat)	1.3 mg/L(gravimetric) 5.1 mg/L (nominal) (III)	>5.3 mg/L (IV)
Primary Eye Irritation (rabbit)	Irreversible Eye Damage (I)	Non-irritating (IV)
Primary Skin Irritation	Non-irritating to slight erythema and edema (IV)	Mildly-irritating (IV)
Dermal Sensitization (Guinea pig)	Negative	Negative

Sources: USEPA, 2005a; BASF, 2012b

Acute mammal toxicity for technical metsulfuron-methyl and Escort® XP show low to very low toxicity for oral, dermal, and inhalation routes (table 2-5). The formulation is less toxic for the dermal route and causes less eye and skin irritation.

Table 2-5. Comparative acute mammal toxicity between metsulfuron-methyl technical active ingredient and proposed formulation.

Toxicity Study	Metsulfuron-methyl Technical	Escort® XP
Acute Oral LD_{50} (rat)	>5,000 mg/kg (IV)	>5,000 mg/kg
Acute Dermal LD_{50} (rat)	3,000 mg/kg (III)	>5,000 mg/kg
Acute Inhalation LC_{50} (rat)	>2 mg/L (IV)	>5 mg/L
Primary Eye Irritation (rabbit)	Irritation reversible in 7 days	Slight irritation
Primary Skin Irritation	Moderate irritation in 72 hours	No irritation
Dermal Sensitization (Guinea pig)	Negative	Negative

Sources: US FS, 2005; DuPont, 2012b

2.4.3 Sub-Chronic/Chronic Toxicity

Based on chronic and subchronic toxicity studies, the kidney appears to be the most sensitive target organ for triclopyr. Decreased phenolsulfonphthalein urinary excretion and reduced absolute and relative kidney weights in dogs were observed after exposure to a dose of 2.5 mg/kg/day for 183/184 (male/female) days (USDA FS, 2011a). Kidney effects on hematological and histopathological changes and increased kidney weight in rodents were observed after subchronic exposure to triclopyr doses at 70 mg/kg/day for 90 days. Damage was characterized as degeneration of the proximal tubules of the kidneys (≥20 mg/kg/day for 90 days) and increases in kidney weight (USDA FS, 2011a). The No Observed Adverse Effect Level (NOAEL) for kidney toxicity in rats is 5 mg/kg bw/day from a two generation dietary reproduction study in rats. This is the basis of the chronic reference dose (RfD) for triclopyr (USEPA, 2002a). Other general systemic effects of triclopyr include signs of liver damage and a decrease in food consumption, growth rate, and gross body weight occurring at high doses (USDA FS, 2011a).

Subchronic and chronic studies using imazapyr have been conducted in rats, rabbits, and dogs. The 90-day oral toxicity study in rats shows that the dermal and systemic NOAEL was 1,695 mg/kg/day for males and 1,784 mg/kg/day for females at the highest dose tested (HDT). The 21/28-day dermal toxicity study in rabbits showed that the dermal and systemic NOAEL was 400 mg/kg/day (HDT). The chronic toxicity study in dogs (1 year) showed that the NOAEL was 250 mg/kg/day (HDT). The NOAEL of 250 mg/kg/day was selected as the basis of the chronic RfD for imazapyr because it was the lowest NOAEL. The dose of 250 mg/kg/day was the highest dose tested in the dog study and no adverse effects were observed. USEPA used this dose for calculating the chronic RfD based on a structural analog of imazapic to choose a toxic endpoint. Imazapic causes skeletal muscle effects in dogs at 137 mg/kg/day (male) and 180 mg/kg/day (female) (USEPA, 2006).

Subchronic and chronic studies with metsulfuron-methyl have been conducted using rats, rabbits, and dogs. The 90-day oral toxicity study in rats reported a NOAEL of 68 mg/kg/day (male) and 64 mg/kg/day (female) with a Lowest Observable Adverse Effect Level (LOAEL) of 521 mg/kg/day (male) and 659 mg/kg/day (female) based on a transient decrease in body weight. The 21-day dermal toxicity study in rabbits showed that the dermal NOAEL was 125 mg/kg/day with a LOAEL of 500 mg/kg/day based on skin lesions (diffuse/multifocal dermatitis). The systemic NOAEL was 125 mg/kg/day with a LOAEL of 500 mg/kg/day based on an increased incidence of diarrhea. The chronic toxicity study in male and female rats showed that the NOAEL was 25 mg/kg/day with a LOAEL of 250 mg/kg/day based on body weight loss. The NOAEL of 25 mg/kg/day was selected as the basis for the chronic RfD for metsulfuron-methyl because it was the lowest NOAEL observed in the toxicity studies (USEPA, 2002b).

2.4.4 Carcinogenicity/Mutagenicity

Based on the USEPA/OPP chemical evaluation of carcinogen potential (USEPA, 2013), triclopyr is "Not Classifiable as to Human Carcinogenicity (Group D)". Triclopyr was classified as a Group D chemical because the evidence of the increase in mammary tumors in the female rat and mouse, and adrenal pheochromocytomas in the male rat are marginal. There was no additional support from structural analogs or genotoxicity studies (USEPA, 1998). Triclopyr is not considered mutagenic or genotoxic based on a lack of evidence from several in vitro and in vivo studies (USEPA, 2002a). However, two unpublished studies on triclopyr ingestion by rats and mice have suggested increased frequency of mammary gland cancer at high doses. Mutagenicity studies using triclopyr (triclopyr technical acid, triclopyr BEE, and non specified triclopyr) in mice and rat did not show mutagenicity (USDA FS, 2011a).

USEPA (2013a) classifies imazapyr as "Evidence of Noncarcinogenicity for Humans (Group E)" based on a study in mice that showed no evidence of carcinogenicity. In the carcinogencity study, the NOAEL was 10,000 ppm (1,301 mg/kg/day in males and 1,639 mg/kg/day in females). The level was the HDT and there was no LOAEL (USEPA, 2005a). Imazapyr is not considered mutagenic or genotoxic based on a lack of evidence from several in vitro and in vivo studies (USEPA, 2005a). The results for the Ames assay was negative up to 5,000 μg/plate. The results of in vitro mammalian cell gene mutation and chromosome aberration studies were negative up to subchronic toxic doses (5,000 μg/ml) with and without activation.

USEPA (2013a) classifies metsulfuron-methyl as "Not Likely to Be Carcinogenic to Humans" based on studies in rats that show no evidence of carcinogenicity. In one carcinogenicity study, the NOAEL was 25 mg/kg/day for both male and female rats. The LOAEL was 250 mg/kg/day for both male and female rats based on reduced body weight. In another carcinogenicity study, the NOAEL was 666 mg/kg/day (male) and 836 mg/kg/day (female) (USEPA, 2002b). Metsulfuron-methyl is not considered mutagenic or genotoxic based on a lack of evidence from several in vitro and in vivo studies (USEPA, 2002b).

2.4.5 Development and Reproductive Effects

The current chronic RfD for triclopyr was based on a two-generation reproduction study (USEPA, 1998). In this study, male and female rats were exposed to dietary concentrations of triclopyr resulting in doses of 0, 5, 25, or 250 mg/kg/day, except that the parent males in the high dose group were exposed to a concentration resulting in a daily dose of 100 mg/kg bw/day. The 5 mg/kg/day dose groups showed no evidence of adverse effects in the parents or offspring. At 25 mg/kg/day, degeneration of renal proximal tubules was observed only in adult animals. At 250 mg/kg/day, parental effects included decreased food consumption and body weights as well as histopathological changes in the liver and kidney. Fetotoxic effects, including decreased pup survival and litter sizes, were noted at 250 mg/kg/day. The NOAEL of 25 mg/kg/day for reproductive effects from this study is supported by a three-generation reproduction study using the same strain of rats with no adverse effects observed to offspring at doses of 3, 10, or 30 mg/kg/day (Hanley et al., 1984); and by an earlier study summarized by the Forest Service in which no adverse reproductive effects were observed in rats exposed to doses up to 30 mg/kg bw/day (USDA FS, 2011a).

The developmental studies on triclopyr acid, triclopyr TEA, and triclopyr BEE show that triclopyr can cause adverse developmental effects including birth defects at sufficiently high doses. The developmental studies for the triclopyr salt and ester in rats show that the maternal NOAEL was 100 mg/kg/day, with a LOAEL of 300 mg/kg/day based on mortality (triclopyr salt and ester), clinical signs, necropsy findings, decreased body weight gains, decreased food consumption, increased water consumption, and increased relative kidney and liver weight (triclopyr ester). The developmental NOAEL was 100 mg/kg/day with a LOAEL of 300 mg/kg/day based on decreased fetal weight, increased fetal and litter incidence of skeletal anomalies, increased fetal incidence of unossified sternebrae (triclopyr salt), increased incidence of hydrocephalus, cleft palate, microphthalmia/anophthalmia, retinal folds, thin diaphragm/protrusion of the liver, decreased fetal weight, and visceral and skeletal amomalies and variants (triclopyr ester). The maternal and developmental NOAELs and LOAELs in rabbits were 30 mg/kg/day and 100 mg/kg/day, respectively (USEPA, 2002a). A consistent pattern with triclopyr, however, is that adverse developmental effects occur only at doses that are maternally toxic. The developmental studies conducted in rats and rabbits do not suggest substantial or consistent differences in the developmental effects of the various forms of triclopyr (USDA FS, 2011a).

The developmental studies in rats on imazapyr show that the maternal NOAEL was 300 mg/kg bw/day with a LOAEL based on salivation at 1,000 mg/kg bw/day. The developmental NOAEL was 1,000 mg/kg/day (HDT) and no LOAEL was reported. The developmental studies in rabbit showed that the maternal NOAEL was 400 mg/kg bw/day (HDT) and the developmental NOAEL was 400 mg/kg bw/day (HDT). The reproduction studies in rats showed that the NOAEL for parental systemic, reproductive andoffspring was 10,000 ppm (738 mg/kg bw/day in males and 933.3 mg/kg bw/day in females) (HDT) (USEPA, 2005a).

The developmental studies in rodents on metsulfuron-methyl show that the maternal NOAEL and LOAEL were 250 mg/kg/day, and 1,000 mg/kg/day, respectively, with effects on salivation and decreased body weight gain-compensatory increase after dosing stopped. The developmental NOAEL and LOAEL were <1,000 mg/kg/day (HDT) and >1,000 mg/kg/day (HDT), respectively. The developmental studies in nonrodents show that the maternal NOAEL and LOAEL were 25 mg/kg/day, and 1,000 mg/kg/day, respectively, with effects on increased mortality, decreased body weight gains, and clinical signs of anorexia, red/orange urine and/or exudate. The developmental NOAELwas 700 mg/kg/day (HDT). Reproduction studies in rats showed that the NOAEL for parental systemic was 34 mg/kg/day and 43 mg/kg/day (male and female respectively). The LOAEL was 342 and 475 mg/kg/day (male and female respectively) based on decreased premating body weight gains. The reproductive NOAEL was 342 and 475 mg/kg/day (male and female respectively) (HDT). The offspring NOAEL was 342 and 475 mg/kg/day (male and female respectively) (HDT) (USEPA, 2002b).

2.4.6 Endocrine Effects

A literature search did not identify any study indicating the potential for triclopyr, imazapyr, and metsulfuron-methyl to disrupt endocrine function. None of these herbercides are among the group of pesticide active ingredients to be screened under the USEPA Endocrine Disruptor Screening Program. However, the identity of chemicals for screening is based on exposure potential, not on whether the pesticide is a known or likely potential endocrine disruptor (USEPA, 2014b). Fetal toxicity and abnormalities have been observed at higher doses using triclopyr, however, there is no indication that the effects occurred through a mechanism involving endocrine disruption (USDA FS, 2011a).

2.4.7 Potential Additive, Antagonistic, or Synergistic Effects:

A literature search was performed to identify any studies indicating additive (toxicity of the mixture is equivalent to the sum of the toxicities of the individual compounds), antagonistic (toxicity of the mixture is less than additive), or synergistic (toxicity of the mixture is greater than additive) effects from the mixing of the three herbicides. The search results indicate that the most common type of effect is additive toxicity and there may be mildly synergistic effects associated with the proposed herbicide mixture (Tatum, 2004). The formulations of the three herbicides proposed to be used in the ALB program have very low toxicity and any additive toxicity is anticipated to be low.

3.0. DOSE-RESPONSE ASSESSMENT

3.1 Human Health Dose-Response Assessment

A dose-response assessment evaluates the dose levels (toxicity criteria) for potential human health effects including acute and chronic toxicity. The toxicity criteria sources include documents and on-line sources from the USEPA/OPP, USEPA Integrated Risk Information System, and the Agency for Toxic Substances and Disease Registry. If a criterion was not available from these sources, information in other regulatory documents or the primary literature was used. When toxicity criteria were developed, uncertainty factors (UFs) were incorporated to address data gaps, effects to sensitive groups, and variability in the study and/or human populations.

As discussed in Section 2.4, triclopyr, imazapyr, and metsulfuron-methyl are not classified as human carcinogens. The noncancer toxicity criterion is developed by identifying a NOAEL, or LOAEL if an appropriate NOAEL is not available, and applying one or more uncertainty factors.

These values are used to calculate a RfD which is a dose that will not result in any adverse effects to an individual.

3.1.1 Triclopyr

The acute and chronic RfDs for triclopyr (1 and 0.05 mg/kg bw/day, respectively) are derived from NOAEL values from the rat studies with an uncertainty factor of 100 added for inter- and intraspecies differences (USEPA, 2002a; USDA FS, 2011a). The acute RfD is based on a developmental study with triclopyr BEE in which no effects were noted at 100 mg/kg bw/day but maternal toxicity was noted at 300 mg/kg bw/day. The chronic RfD is based on a two- generation reproduction study in rats with triclopyr acid where no adverse effects were noted at 5 mg/kg bw/day but effects on the kidney (degeneration of renal proximal tubules) were noted at 25 mg/kg bw/day. Because of concerns for the reproductive and developmental toxicity of triclopyr, the chronic RfD is used to assess the risks to women of childbearing age associated with both acute and long-term exposure. USEPA (2002a) indicates that the acute RfD is not applicable to females between the ages of 13-50 years—i.e., of child bearing age. The basis for this recommendation appears to be signs of maternal toxicity observed at 30 mg/kg bw/day with a reported NOAEL of 5 mg/kg bw/day. As discussed below, the chronic RfD for triclopyr is 0.05 mg/kg bw/day, based on a NOAEL of 5 mg/kg bw/day. Thus, for women of childbearing age, the EPA recommends an acute RfD of 0.05 mg/kg/day, which is equivalent to the chronic RfD.

3.1.2 TCP

TCP is a metabolite of triclopyr. The acute and chronic RfDs for TCP (0.025 and 0.012 mg/kg bw/day, respectively) are based on NOAEL values from the rabbit and dog studies (USEPA, 2002a, USDA FS, 2011a). The acute RfD is based on a developmental study in rabbits in which birth defects (an increased incidence of hydrocephaly and dilated ventricles) were noted at a dose of 100 mg/kg bw/day but no adverse effects were observed at 25 mg/kg bw/day. The chronic RfD is based on a chronic study in dogs in which changes in clinical chemistry were observed at the 48 mg/kg/day (LOAEL) and no effects were observed at the 12 mg/kg/day dose (NOAEL).

For both acute and chronic exposures the uncertainty factor for TCP is set at 1000: 10 to account for uncertainties in species-to-species extrapolation and another factor of 10 to encompass sensitive individuals in the population, as well as an additional factor of 10 for the potential for increased sensitivity in children.

3.1.3 Imazapyr

The USEPA developed a chronic RfD of 2.5 mg/kg/day for imazapyr based on a 2-year dog study and a reported NOEL of 25 mg/kg/day that was adjusted using an uncertainty factor of 100 for inter- and intraspecies differences (USEPA, 2006). USEPA did not develop an acute/single dose RfD for imazapyr. A developmental study in rats with imazapyr reported a maternal NOAEL of 300 mg/kg bw/day with a LOAEL based on salivation at 1,000 mg/kg bw/day (see Development and Reproductive Effects under Section 2.4). Based on the maternal NOAEL of 300 mg/kg bw/day applying by an uncertainty factor of 100, an acute RfD will be 3 mg/kg/day, which is similar to the chronic RfD for imazapyr.

3.1.4 Metsulfuron-Methyl

The USEPA developed a chronic RfD of 0.25 mg/kg/day for metsulfuron-methyl based on a 2- year rat study and a reported NOEL of 25 mg/kg/day adjusted using an uncertainty factor of 100 for inter- and intraspecies differences (USEPA, 2002b). USEPA did not develop an acute/single dose RfD for metsulfuron-methyl. A reproduction study in rats reported a parental systemic NOAEL of 34 mg/kg/day, with a LOAEL of 342 mg/kg/day based on decreased body weight (see Development and Reproductive Effects under Section 2.4). Based on the NOAEL of 34 mg/kg bw/day and an uncertainty factor of 100, an acute RfD will be 0.34 mg/kg/day, which is similar to the chronic RfD for metsulfuron-methyl.

3.1.5 Drinking Water

Triclopyr, imazapyr, and metsulfuron-methyl are not currently regulated under the Safe Drinking Water Act. Maximum Contaminant Levels protective of human health are not established for these compunds.

3.2 Ecological Effects Analysis

This section of the risk assessment discusses available ecological effects data for terrestrial and aquatic biota. Available acute and chronic toxicity data are summarized for all major taxa and will be integrated with the exposure analysis section to characterize the risk of triclopyr, imazapyr, and metsulfuron-methyl to nontarget wildlife and domestic animals. Information in this section was gathered from on-line databases and searches for relevant peer reviewed and non-peer reviewed literature. Recent reviews of triclopyr, imazapyr, and metsulfuron-methyl by the Forest Service were used to summarize available ecotoxicological effects data.

3.2.1 Aquatic Effects Analysis

3.2.1.1 Fish and Amphibians

<u>Triclopyr and TCP</u>

The acute fish toxicity studies show that triclopyr and TCP acute toxicities to fish range from practically non-toxic to highly toxic (USEPA Ecotoxicity Categories, USEPA, 2014c) (table 3-1). The acute median effective concentrations (96-hour EC_{50}) for triclopyr TEA, triclopyr acid, TCP, and triclopyr BEE from the fish toxicity studies are 131 mg a.e./L, 15.3 mg a.e./L, 3.19 mg a.e./L, and 0.539 mg a.e./L, respectively (USDA FS, 2011a). Based on the acute toxicity data, triclopyr TEA is much less toxic to fish than triclopyr acid, triclopyr BEE, or TCP. Compared to triclopyr BEE, triclopyr TEA is less acutely toxic by a factor of approximately 240, and TCP is less acutely toxic by a factor of approximately 6.

Table 3-1. Acute fish toxicity data for triclopyr acid, TEA, BEE, and TCP.

Test Organism	**Endpoint/Length**	**Toxicity Value (mg/L)**	**NOEC (mg/L)**
Triclopyr Acid			
Bluegill sunfish	96-hour LC_{50}	155.4	NR
Rainbow trout	96-hour LC_{50}	79.2	NR
Chinook salmon	96-hour LC_{50}	9.7	NR
Coho salmon	96-hour LC_{50}	9.6	NR
Chum salmon	96-hour LC_{50}	7.5	NR
Sockeye salmon	96-hour LC_{50}	7.5	NR
Rainbow trout	96-hour LC_{50}	7.5	NR
Pink salmon	96-hour LC_{50}	6.3	NR
Median		**15.3**	
Triclopyr TEA			
Tidewater silverside	96-hour LC_{50}	40.1	NR
Bluegill sunfish	96-hour LC_{50}	65.1	NR
Catfish, juv	96-hour LC_{50}	78.3	NR
Fathead minnow	96-hour LC_{50}	85.8	NR
Fathead minnow	96-hour LC_{50}	86.4	NR
Chum salmon	96-hour LC_{50}	96.1	NR
Chinook salmon	96-hour LC_{50}	99	NR
Sockeye salmon	96-hour LC_{50}	112	NR
Coho salmon, juv	96-hour LC_{50}	127.2	NR
Catfish, adult	96-hour LC_{50}	141	103
Rainbow trout	96-hour LC_{50}	151	NR
Coho salmon	96-hour LC_{50}	167	NR
Fathead minnow	96-hour LC_{50}	168.5	NR
Bluegill sunfish	96-hour LC_{50}	233.1	NR
Rainbow trout	96-hour LC_{50}	273.7	NR
Rainbow trout	96-hour LC_{50}	286	NR
Fathead minnow	96-hour LC_{50}	422.8	NR
Median		**130.7**	
Triclopyr BEE			
Bluegill sunfish	96-hour LC_{50}	0.25	0.091
Coho salmon	96-hour LC_{50}	0.26	NR
Coho salmon	96-hour LC_{50}	0.47	NR
Fathead minnow	96-hour LC_{50}	0.5	NR
Bluegill sunfish	96-hour LC_{50}	0.54	NR
Bluegill sunfish	96-hour LC_{50}	0.58	NR
Coho salmon	96-hour LC_{50}	1	NR
Fathead minnow	96-hour LC_{50}	1.5	0.97
Median		**0.54**	
TCP			
Rainbow trout	96-hour LC_{50}	1.5	NR
Coho salmon	96-hour LC_{50}	1.8	NR
Chum salmon	96-hour LC_{50}	1.8	NR
Chinook salmon	96-hour LC_{50}	2.1	NR
Sockeye salmon	96-hour LC_{50}	2.5	NR

Test Organism	Endpoint/Length	Toxicity Value (mg/L)	NOEC (mg/L)
Pink salmon	96-hour LC_{50}	2.7	NR
Bluegill sunfish	96-hour LC_{50}	12.5	NR
Rainbow trout	96-hour LC_{50}	12.6	NR
Median		**3.19**	

NR = not reported   Source: USDA FS, 2011a, Table 33

The sublethal effects of Garlon® 4 were investigated on rainbow trout using flow-through systems (Johansen and Green, 1990). Fish were lethargic at concentrations of 0.32-0.43 mg/L. Another study reported behavioral changes in rainbow trout for Garlon® 4 at 0.6 mg/L and Garlon® 3A at 200 mg/L (Morgan et al., 1991).

Chronic fish toxicity studies were conducted using triclopyr TEA, triclopyr BEE, and TCP and are summarized in USDA FS (2011a). The 28-day toxicity study for triclopyr TEA in fathead minnow reported a NOEC of 32.4 mg a.e./L and a LOEC of 50.2 mg a.e./L based on effects to length. The chronic toxicity study for triclopyr BEE in rainbow trout reports a NOEC of 0.019 mg a.e./L and a LOEC of 0.034 mg a.e./L based on effects to larval weight and length (USEPA, 2009). The chronic toxicity study for TCP in rainbow trout reports a NOEC of 0.178 mg/L and a LOEC of 0.278 mg/L based on effects to length and weight (USDA FS, 2011a). The chronic fish effects data show that triclopyr BEE is the most toxic form of triclopyr and that TCP is more toxic than triclopyr TEA.

Acute toxicity data for aquatic phase amphibians is limited. An acute toxicity study using the African clawed frog, *Xenopus laevis*, exposed to Garlon® 3A shows that triclopyr TEA is slightly toxic with a 96-hour LC_{50} of 84 mg a.e./L (USDA FS, 2011a). This LC_{50} is well within the range of LC_{50} values (≈40 to 420 mg a.e./L) for triclopyr TEA in fish. The acute toxicity studies in embryos and tadpoles exposed to triclopyr BEE formulations show triclopyr BEE is slightly toxic to amphibian embryos with a median 96-hour LC_{50} of 17.78 mg a.e./L, and moderately toxic to tadpoles with a median 96-hour LC_{50} of 2.34 mg a.e./L. The acute toxicity of triclopyr BEE to tadpoles varies from species to species (*Rana pipiens* is the most sensitive species and *R. clamitans* is the least sensitive species). Tadpoles are more sensitive than embryos by approximately an order of magnitude. This difference in sensitivity may reflect the rapid uptake of triclopyr BEE through the gills of tadpoles, relative to passive uptake by amphibian embryos (USDA FS, 2011a). The acute toxicity data indicate that amphibians are less sensitive to triclopyr BEE than fish. The acute toxicity data in amphibians for triclopyr BEE all involve triclopyr formulations. No data appears to be available regarding the toxicity of unformulated triclopyr BEE or TCP to amphibians. The acute amphibian toxicity data for triclopyr TEA and BEE formulations are summarized in table 3-2.

Table 3-2. Acute amphibian toxicity data for triclopyr TEA and BEE formulations

Test Organism	Endpoint/Length	Toxicity Value (mg a.e./L)	NOEC (mg a.e./L)
Triclopyr TEA			
African clawed frog, *Xenopus laevis*	96-hour LC_{50}	84	NR
Triclopyr BEE Formulations (Embryos)			
Xenopus laevis	96-hour LC_{50}	13.7	NR
Xenopus laevis	96-hour LC_{50}	15.0	NR
Bufo americanus	96-hour LC_{50}	15.1	NR
Rana pipiens	96-hour LC_{50}	23.3	NR
Rana clamitans	96-hour LC_{50}	24.6	NR
Median		**17.78**	
Triclopyr BEE Formulations (Tadpoles)			
Rana pipiens	96-hour LC_{50}	0.79	NR
Bufo americanus	96-hour LC_{50}	0.88	NR
Xenopus laevis	96-hour LC_{50}	1.70	NR
Rana pipiens	96-hour LC_{50}	2.79	NR
Rana clamitans	96-hour LC_{50}	3.01	NR
Rana pipiens	96-hour LC_{50}	3.39	NR
Rana clamitans	96-hour LC_{50}	11.50	NR
Median		**2.34**	

NR = Not reported Source: USDA FS, 2011a

In addition to standardized acute toxicity studies there have been frog embryo teratogenesis studies conducted using Garlon® 3A and Garlon® 4 (Perkins et al., 2000). In the assay, *X. laevis* embryos were exposed to the test solution in Petri dishes for 96 hours and observed for malformations. No hind limb abnormalities were reported. The abnormalities observed in the study include uncoiling of the gut, edema, blistering, abnormal pigmentation, and axial twisting in control embryos. However, there were no statistically significant increases in abnormalities in any groups exposed to Garlon® 3A or Garlon® 4 at sublethal levels. The study results indicate that triclopyr at sublethal concentrations is not likely to cause reproductive or teratogenic effects in amphibians.

Imazapyr

The acute toxicity of imazapyr is classified as practically non-toxic to fish based on LC_{50} values of >100 mg a.e./L for imazapyr acid (USDA FS, 2011b). The acute toxicity for the isopropylamine salt of imazapyr is also practically non-toxic to fish based on acute bioassays in bluegill and trout. The acute toxicity for Arsenal® herbicide (27.8% a.i isopropylamine salt of imazapyr and 72.2% inert) is slightly toxic with a 96-hour LC_{50} of 41 mg a.e./L in bluegill and 21 mg a.e./L in trout. The acute fish toxicity data for imazapyr are summarized in table 3-3.

Table 3-3. Acute and chronic fish toxicity data for imazapyr.

Test Organism	**Endpoint/Length**	**Toxicity Value (mg a.e./L)**	**NOEC (mg a.e./L)**
Imazapyr Acid			
Bluegill sunfish	96-hour LC_{50}	>100	NR
Atlantic silversides	96-hour LC_{50}	NR	>184
Bluegill sunfish	96-hour LC_{50}	>100	NR
Rainbow trout	96-hour LC_{50}	>100	NR
Channel catfish	96-hour LC_{50}	>184	NR
Fathead minnow	Early life-stage (egg-to-fry)	NR	120
Fathead minnow	Full Life Cycle	NR	118
Rainbow trout	Early life-stage (egg-to-fry)	NR	43.1 (LOEC: 92.4)
Imazapyr Isopropylamine (IPA) Salt			
Bluegill sunfish	96-hour LC_{50}	>815.5	NR
Rainbow trout	96-hour LC_{50}	>110	110
Arsenal® Herbicide			
Bluegill sunfish	96-hour LC_{50}	40.68	NR
Rainbow trout	96-hour LC_{50}	20.8	10.4

NR = Not reported Source: USDA FS, 2011b

A micronucleus assay using *Tilapia rendalli* (an herbivorous fish native to Africa) indicated positive mutagenic activity (USDA FS, 2011b; Grisolia, 2002). In this screening test, fish were exposed to imazapyr at 20, 40, or 80 mg/kg through intra-abdominal injections. A statistically significant increase in erythrocyte micronuclei was observed in the 80 mg/kg dose groups. However, the finding of this study does not have a substantial impact on the hazard identification for fish because the exposure route was atypical, and a positive response was only seen at the maximum dose of 80 mg/kg. In addition, imazapyr does not appear to be mutagenic or carcinogenic in mammals as discussed in the human health section of this risk assessment.

The USEPA/OPP risk assessment for the California red-legged frog summarized two fish kill incidents associated with imazapyr (USEPA, 2007). The first incident reported a 63 fish and algae kill in a pond 60 feet away from a mixed herbicidal spray. A mixture of the isopropylamine salt of imazapyr, diuron and metsulfuron-methyl was sprayed onto a fence row and may have entered the pond from drift and/or runoff. However, it cannot be definitively determined that the fish kill was due to exposure to imazapyr. The second incident involved a goldfish kill. The cause of the kill could not be determined, but there was suspected runoff and drift into the pond after an aerial application of an imazapyr formulation to a nearby 145 acres.

The long-term toxicity of imazapyr acid to fathead minnows has been tested in an early life-stage (egg to fry) study and a full life cycle study (USDA FS, 2011b). Neither study detected adverse effects at concentrations of up to about 120 mg a.e./L. The early life-stage chronic toxicity study in rainbow trout for imazapyr acid reported a NOEC of 43.1 mg a.e./L and a LOEC of 92.4 mg a.e./L due to reduced hatch and fry survival. Both acute and chronic toxicity studies in fish indicate that trout appear to be the most sensitive species.

No long-term toxicity studies on imazapyr formulations have been conducted. The acute NOAEC of 110 mg a.e./L for the isopropylamine salt of imazapyr in rainbow trout is above the longer-term NOAEC of 43.1 mg a.e./L for the imazapyr acid. However, the acute NOAEC of 10.4 mg a.e./L for the Arsenal® herbicide formulation in rainbow trout is below the longer-term NOAEC for the imazapyr acid.

No information is available regarding the toxicity of imazapyr to aquatic-phase amphibians. Following a standard USEPA approach, fish toxicity data is used for aquatic phase amphibians assuming that fish are approximately as sensitive as aquatic phase amphibians.

Metsulfuron-methyl

Acute toxicity studies using rainbow trout and bluegill (USDA FS, 2005) show that metsulfuron-methyl is practically nontoxic to fish with 96-hour LC_{50} values >150 mg/L. Sublethal effects including erratic swimming behavior, laying on the bottom, lethargy and color changes that were observed in rainbow trout with a reported NOEC of 10 mg/L.

The 90-day chronic exposure study in rainbow trout including fish, egg, and fry (USDA FS, 2005) reported no effects on rainbow trout hatching, larval survival, or larval growth at a concentration of up to 4.7 mg/L. The study reported a LOEC of 8 mg/L with decreases in hatching and survival of fry. The acute and chronic fish toxicity data for metsulfuron-methyl is summarized in table 3-4.

No information is available regarding the toxicity of metsulfuron-methyl to amphibian species.

Table 3-4. Acute and chronic fish toxicity data for metsulfuron-methyl.

Test Organism	**Endpoint/Length**	**Toxicity Value (mg/L)**	**NOEC (mg/L)**
Acute			
Rainbow trout	96-hour LC_{50}	>150	100
Bluegill sunfish	96-hour LC_{50}	>150	150
Bluegill sunfish	96-hour LC_{50}	>1000	1000
Rainbow trout	96-hour LC_{50}	>1000	10
Chronic			
Rainbow trout	90-days	NR	4.7 (LOEC: 8.0)

NR = Not reported Source: USDA FS, 2005

3.2.1.2 Aquatic Invertebrates

Triclopyr and TCP

Triclopyr acid and TEA are less toxic to aquatic invertebrates compared to triclopyr BEE (USDA FS, 2011a). TCP appears to be less toxic than triclopyr BEE, but more toxic than triclopyr acid and TEA to aquatic invertebrates. The aquatic invertebrate studies show that the triclopyr acid and TEA are practically nontoxic to non-bivalve aquatic invertebrate species with acute LC_{50} values >100 mg/L. Triclopyr TEA is slightly toxic to bivalve test species with an acute median LC_{50} of 19.7 mg/L. One study using TCP shows that TCP is slightly toxic to *Daphnia magna* with an acute LC_{50} of 10.9 mg/L. Triclopyr BEE and Garlon® 4 are moderately toxic to aquatic invertebrates with an acute median LC_{50} value of 2.9 mg/L. Triclopyr BEE and Garlon® 4 are highly toxic to bivalves with acute EC_{50} values between 0.1 and 1 mg/L based on shell deposition. The acute triclopyr and TCP toxicity data for aquatic invertebrates are summarized in table 3-5.

Table 3-5. Acute aquatic invertebrate toxicity data for triclopyr and TCP

Test Organism	**Endpoint/Length**	**Toxicity Value (mg a.e./L)**	**NOEC (mg a.e./L)**
Triclopyr Acid and TEA (Non-bivalve)			
Grass shrimp	48-hour LC_{50}	103.7	NR
Daphnia magna (Acid)	48-hour LC_{50}	132.9	NR
Pink shrimp	48-hour LC_{50}	270.5	NR
Physella gyrina (Acid)	48-hour LC_{50}	293	NR
Daphnia magna	48-hour LC_{50}	346	NR
Daphnia magna	48-hour LC_{50}	357	NR
Daphnia magna	48-hour LC_{50}	376	<108
Daphnia magna	48-hour LC_{50}	837	NR
Red swamp crayfish	48-hour LC_{50}	6397.5	NR
Median:		**401.6**	
Triclopyr TEA (Bivalves)			
Eastern oyster	96-hour EC_{50} (shell dep)	18.4	NR
Eastern oyster	48-hour EC_{50} (abnormal development)	21.1	NR
Median:		**19.7**	
TCP			
Daphnia magna	48-hour LC_{50}	10.9	NR
Triclopyr BEE and Garlon® 4 (Arthropods)			
Daphnia magna	48-hour LC_{50}	0.25	NR
Daphnia pulex	48-hour LC_{50}	0.54	NR
Grass shrimp	96-hour LC_{50}	0.77	NR
Daphnia magna	48-hour LC_{50}	1.2	NR
Grass shrimp	96-hour EC_{50}	1.8	NR
Red swamp crayfish	48-hour LC_{50}	3.1	1.2
Stonefly (*Calineuria californica*)	48-hour LC_{50}	3.6	NR

Test Organism	Endpoint/Length	Toxicity Value (mg a.e./L)	NOEC (mg a.e./L)
Mayfly (*Ameletus* sp.)	48-hour LC_{50}	3.8	NR
Caddisfly (*Brachycentrus americanus*)	48-hour LC_{50}	5	NR
Daphnia magna	48-hour LC_{50}	8.3	NR
Mayfly (*Cinygma* sp.)	48-hour LC_{50}	8.95	NR
Caddisfly (*Psychoglypha* sp.)	48-hour LC_{50}	12.5	NR
Caddisfly (*Lepidostoma unicolor*)	48-hour LC_{50}	20	NR
Median:		**2.9**	
Triclopyr BEE and Garlon 4 (Bivalves)			
Eastern oyster*	96-hour EC_{50} (shell dep.)	0.14	0.05
Eastern oyster	96-hour EC_{50} (shell dep.)	0.33	NR
Median:		**0.21**	

*Garlon® 4 NR = not reported Source: USDA FS, 2011a

Chronic toxicity to aquatic invertebrates has been assessed for triclopyr TEA and TCP. The 21-day *D. magna* study for triclopyr TEA reported a NOEC of 25 mg a.e./L and a LOEC of 46.2 mg a.e./L based on total number of young and mean brood size (USEPA, 2009). The chronic study using *D. magna* and TCP reported a NOEC of 0.058 mg TCP/L and a LOEC of 0.13 mg TCP/L based on a significant decrease in the mean number of young (USDA FS, 2011a).

A series of 1-hour field exposure studies and triclopyr BEE in several species of stream invertebrates show that the LC_{50} values for these aquatic invertebrates were greater than 290 mg/L (≈200 mg a.e./L) (Kreutzweiser et al., 1992). The LC_{50} values from the 1-hour field studies are two orders of magnitude higher than the standard 48-hour LC_{50} values for triclolpyr BEE suggesting those species are less sensitive.

Imazapyr

The acute toxicity data for aquatic invertebrates show that imazapyr acid and isopropylamine salt of imazapyr are practically non-toxic to *D. magna* (USEPA, 2005b; USEPA, 2007) and saltwater invertebrates—i.e., oysters and pink shrimp (USEPA, 2005b). The bioassays using *D. magna* indicate that the Arsenal® herbicide formulation is slighty toxic to aquatic invertebrates (more toxic than either imazapyr acid or the isopropylamine salt of imazapyr). The EC_{50} of 79 mg a.e./L for Arsenal® to *D. magna* is less than the EC_{50} of >100 mg a.e./L for imazapyr acid to *D. magna*, and the EC_{50} of 614 mg a.e./L for isopropylamine salt of imazapyr (USDA FS, 2011b). The acute aquatic invertebrate toxicity data for imazapyr is summarized in table 3-6.

Table 3-6. Acute and chronic aquatic invertebrates toxicity data for imazapyr

Test Organism	Endpoint/Length	Toxicity Value (mg a.e./L)	NOEC (mg a.e./L)
Imazapyr Acid			
Daphnia magna	24-hour EC_{50}	>100	NR
	48-hour EC_{50}	>100	NR
Eastern oyster	96-hour EC_{50}	>100	132
Eastern oyster	96-hour EC_{50}	NR	109 (LOEC: 173)
Pink shrimp	96-hour LC_{50}	>189	189
Daphnia magna	21-day LC_{50}	>97.1	97.1
Imazapyr IPA Salt			
Daphnia magna	48-hour EC_{50}	614	NR
Arsenal® Herbicide			
Daphnia magna	48-hour LC_{50}	79.1	40.68 (LOEC: 81.36)

NR = Not reported Source: USDA FS, 2011b.

A long-term toxicity study using *D. magna* and imazapyr reported no effects at concentrations up to 97.1 mg a.e./L. Similar to fish, the chronic NOEC in daphnids is above the acute NOEC of 40.68 mg a.e./L for Arsenal® herbicide. A mesocosm study was conducted to assess the long-term impacts of formulated imazapyr to aquatic invertebrates (Fowlkes et al. 2003). In this study, mixed macroinvertebrate species were exposed to mesocosms treated with Arsenal Applicators Concentrate® at concentrations of 0.184, 1.84, or 18.4 mg a.e./L. After a 2-week exposure period (comparable to the exposure period in the chronic daphnid studies), no impacts were noted on species richness or abundance. The NOEC of 18.4 mg a.e./L from the mesocosm study is consistent with the acute NOEC of 40.68 mg a.e./L for Arsenal® herbicide and the chronic NOAEC of 97.1 mg a.e./L in daphnids. However, USEPA considered the results of the mesocosm study are of limited value because potential effects at the species level were not examined (USEPA, 2007; USDA FS, 2011b).

<u>Metsulfuron-methyl</u>

Acute toxicity studies using *D. magna* show that metsulfuron-methyl is practically non-toxic to aquatic invertebrates with 48-hour EC_{50} values for immobility ranging from >150 mg/L to 720 mg/L (USDA FS, 2005). The acute NOECs ranged from 150 to 420 mg/L. The 21-day chronic toxicity studies using *D. magna* reported NOECs ranging from 100 to 150 mg/L for survival, reproduction, and immobility, and a lower NOEC of 17 mg/L for growth. The acute and chronic aquatic invertebrate toxicity data for metsulfuron-methyl are summarized in table 3-7.

Table 3-7. Acute and chronic aquatic invertebrate toxicity data for metsulfuron-methyl

Test Organism	Endpoint/Length	Toxicity Value (mg/L)	NOEC (mg/L)
Acute			
Daphnia magna	48-hour EC_{50}	>150	150
Daphnia magna	48-hour EC_{50}	720	420
Chronic			
Daphnia magna	21-days	NR	>150 (reproduction) (LOEC: >150) 17 (growth)
Daphnia magna	21-days	NR	100

NR = Not reported Source: USDA FS, 2005

3.2.1.3 Aquatic Plants

Triclopyr and TCP

Aquatic plant studies using blue green algae, freshwater diatoms, and green algae testing technical and formulated material have demonstrated that triclopyr TEA and BEE, and TCP, are moderately toxic (EC_{50} >1 to 10 mg/L) to slightly toxic (EC_{50} >10 to 100 mg/L) except for three test species (USDA FS, 2011a; USEPA, 2009). Triclopyr TEA is highly toxic to green algae *Ankistrodesmus* spp. and triclopyr BEE is very highly toxic to diatoms *Navicula pelliculosa* and *Skeletonema costatum.* These studies also show that triclopyr acid is the least toxic, and triclopyr BEE is the more toxic form of triclopyr to algae and diatoms compared to triclopyr TEA (table 3-8).

Table 3-8. Acute aquatic plant (algae) toxicity data for triclopyr and TCP.

Test Organism	Endpoint/Length	Toxicity Value (mg a.e./L)	NOEC (mg a.e./L)
Triclopyr Acid			
Chlorella vulgaris		11	NR
Chlorella pyrenoidosa		80	NR
Kirchneria subcapitata	120-hour EC_{50}	32.8	7
Selanastrum capricornutum	120-hour EC_{50}	50	22
Triclopyr TEA			
Chlorella vulgaris		8	NR
Chlorella pyrenoidosa		54	NR
Ankistrodesmus spp.	96-hour EC_{50}	0.49	0.23
Kirchneria subcapitata	96-hour EC_{50}	5.4	8.1
Skeletonema costatum	96-hour EC_{50}	4.6	0.39
Navicula pelliculosa	96-hour EC_{50}	10.6	5.54
Kirchneria subcapitata	120-hour EC_{50}	12.1	NR
Anabaena flos-aquae	168-hour EC_{50}	4.1	1.39
Triclopyr BEE			
Anabaena flos-aquae	96-hour EC_{50}	1.42	0.37
Navicula pelliculosa	96-hour EC_{50}	0.073	0.0014
Skeletonema costatum	96-hour EC_{50}	0.84	0.15

Skeletonema costatum	120-hour EC_{50}	5.9	1.0
Kirchneria subcapitata	120-hour EC_{50}	2.5	NR
TCP			
Kirchneria subcapitata	120-hour EC_{50}	1.8	0.65
Anabaena flos-aquae	120-hour EC_{50}	1.8	0.36

NR = Not reported. Source: USDA FS, 2011a

Aquatic plant studies using macrophytes have demonstrated that triclopyr acid, triclopyr TEA (to monocots), and triclopyr BEE (to both monocots and dicots) are moderately toxic with 7- or 14-day median EC_{50} values between 1 and 10 mg/L. Triclopyr acid and triclopyr TEA are very highly toxic to dicots with median EC_{50} values <0.1 mg/L (see table 3-9).

Table 3-9. Aquatic plant (macrophytes) toxicity data for triclopyr

Test Organism	Endpoint/Length	Toxicity Value (mg a.e./L)	NOEC (mg a.e./L)
Triclopyr Acid and Triclopyr TEA Monocots			
Lemna gibba	14-day EC_{50}	6.06	2.5
Lemna gibba	14-day EC_{50}	7.6	2.5
Lemna gibba	14-day EC_{50}	7.8	2.5
Lemna gibba	14-day EC_{50}	13.58	NR
Lemna minor	14-day EC_{50}	15.8	NR
Median:		**9.47**	
Dicots			
Watermilfoil (acid)	14-day EC_{50}	0.04	NR
Watermilfoil (TEA)	5-week EC_{50}	0.04	0.01
Milfoil Hybrid	5-week EC_{50}	0.08	0.03
Watermilfoil (acid)	14-day EC_{50}	0.56	NR
Median:		**0.09**	
Triclopyr BEE			
Monocots			
Lemna gibba	14-day EC_{50}	0.86	<0.111
Lemna gibba	14-day EC_{50}	6.25	NR
Median:		**2.32**	
Dicots			
Watermilfoil	14-day EC_{50}	1.49	NR
Watermilfoil	14-day EC_{50}	4.62	NR
Median:		**2.62**	

NR = Not reported Source: USDA FS, 2011a

Imazapyr

Imazapyr is more toxic to aquatic macrophytes than algae based on available EC_{50} values. Toxicity data for blue-green algae, green algae, and diatoms show that imazapyr is slightly toxic to algae (7-day EC_{50} ranging from 11.5 to 92 mg a.e./L) (USDA FS, 2011b). The toxicity data for duckweed and watermilfoil show that imazapyr is very highly toxic to macrophytes (7- or 14-days EC_{50} ranging from 0.018 to 0.029 mg a.e./L) (USDA FS, 2011b) (table 3-10).

Table 3-10. Aquatic plants (algae and macrophytes) toxicity data for imazapyr

Test Organism	**Endpoint/Length**	**Toxicity Value (mg a.e./L)**	**NOEC (mg a.e./L)**
Algae			
Imazapyr Acid			
Green algae (*Selenastrum capricornutum*)	7-day EC_{50}	71	50.9
Blue-green algae (*Anabaena flosaquae*)	7-day EC_{50}	12.2	9.6
Freshwater diatom (*Navicula pelliculosa*)	7-day EC_{50}	>41	41
Marine diatom (*Skeletonema costatum*)	7-day EC_{50}	92	15.6
Imazapyr IPA Salt			
Green algae (*Selenastrum capricornutum*)	7-day EC_{50}	11.5	7.16
Macrophytes			
Imazapyr Acid			
Duckweed (*Lemna gibba*)	14-day EC_{50}	0.024	0.01
Imazapyr IPA Salt			
Duckweed (*Lemna gibba*)	7-day EC_{50}	0.018	0.011
Arsenal®			
Water milfoil (*Myriophyllum sibiricum*)	14-day EC_{50}	0.029	NR

NR = Not reported Source: USDA FS, 2011b

Metsulfuron-methyl

Two 14-day toxicity studies using duckweed and northern watermilfoil demonstrate high toxicity from exposure to metsulfuron-methyl. The duckweed study reported a 14-day EC_{50} value of 0.36 µg/L based on chlorosis of fronds and a NOEC value of 0.16 µg/L. The northern watermilfoil study reported a 14-day EC_{50} value of 0.22 µg/L based on a decrease in dry root mass (USDA FS, 2005) (table 3-11).

The toxicity studies in algae show that metsulfuron-methyl is less toxic to algae compared to aquatic macrophytes. The 120-hour or 72-hour EC_{50} values for algae range from >95.4 to 1,560 µg/L with effects on cell density, growth rate, and growth inhibition. The NOEC values from the algae studies range from 10 to 125 µg/L.

A toxicity study using aquatic cyanobacteria reported significant growth inhibition at a metsulfuron-methyl concentration of 3 µg/L (Peterson et al., 1994).

Table 3-11. Aquatic plants (algae and macrophytes) toxicity data for metsulfuron-methyl.

Test Organism	Endpoint/Length	Toxicity Value (µg/L)	NOEC (µg/L)
Algae			
Green algae* (*Selenastrum capricornutum*)	Cell inhibition/120-hour EC_{50}	NR	10
Blue-green algae* (*Anabaena flosaquae*)	120-hour EC_{50}	>95.4	<95.4
Freshwater diatom* (*Navicula pelliculosa*)	Growth rate/120-hour EC_{50}	>95.4	95.6
Green algae (*Selenastrum capricornutum*)	Growth inhibition/72-hour EC_{50}	1,560	NR
Green algae (*Selenastrum capricornutum*)	Cell density/72-hour EC_{50} Area under the growth curve/72-hour EC_{50} Growth Rate/72-hour EC_{50}	372 359 1307	125 125 125
Green algae (*Chlorella pyrenoidosa*)	Growth inhibition/96-hour EC_{50}	620	NR
Macrophytes			
Duckweed (*Lemna gibba*)	14-day EC_{50}	0.36	0.16
Watermilfoil (*Myriophyllum sibiricum*)	14-day IC_{50}	0.22	NR

* Ally® herbicide. NR = Not reported Source: USDA FS, 2005

3.2.2 Terrestrial Effects Analysis

3.2.2.1 Mammals

Available mammalian data discussed in the human health section of this risk assessment can be used as a surrogate to evaluate the potential effects to wild mammals. In general mammalian toxicity is low for triclopyr, imazapyr, and metsulfuron-methyl. Specific information regarding the acute and chronic effects of each herbicide can be found in section 2.4 of this risk assessment.

3.2.2.2 Birds

Triclopyr

Triclopyr is considered practically non-toxic to slightly toxic to birds based on studies using the mallard and quail (USEPA, 1998; USDA FS, 2011a). The acute oral toxicity of triclopyr varies from 529 to 1,698 mg a.e./kg in mallard duck, and northern bobwhite quail (table 3-12). Reproduction studies in mallard and bobwhite quail report a NOEC of approximately 100 ppm and a LOEC of 200 ppm for both species. The acute dietary study reported a NOEC of 1,000 ppm in Japanese quail. A long-term NOEC of 50 ppm was determined in Zebra finches for triclopyr BEE.

Table 3-12 Acute toxicity of various forms of triclopyr to birds.

Test Species	LD_{50}/LC_{50}	NOEC	LOEC
Acute oral/gavage			
Mallard duck			
triclopyr acid	1,698 mg/kg/day	464 mg/kg bw (mortality)	NA
triclopyr TEA	1,418 mg/kg/day	NA	NA
Northern bobwhite quail			
triclopyr BEE	529 mg a.e./kg bw	≈210 mg a.e./kg bw (mortality)/ ≈126 mg a.e./kg bw (toxicity)	≈350 mg a.e./kg bw
TCP (99.9%)	>2,000 mg a.i./kg bw	125 mg/kg bw	250 mg/kg bw (reduced body weight)
Acute dietary			
Japanese quail			
triclopyr acid	3,272 ppm (LC_{50})	1,000 ppm (≈550 mg a.e./kg bw)	2,000 ppm (≈1100 mg a.e./kg bw)
Northern bobwhite quail			
triclopyr acid	≈2,553 mg a.e./kg bw (LC_{50})	≈1000 mg a.e./kg bw (mortality)	NA
triclopyr TEA	5,189 ppm a.e. (≈3000 mg a.e./kg bw) (LC_{50})	2,150 ppm formulation (995 mg a.e./kg bw)	NA
triclopyr BEE (96.1% a.i.)	5,401 ppm a.i. (3,885 ppm a.e.) (LC_{50})	961 ppm a.i. (≈691 ppm a.e.) (toxicity)	1,711 ppm a.i. (reduced weight gain)
Mallard duck			
triclopyr acid	5,620 ppm (LC_{50})	NA	NA
triclopyr TEA	4,464.8 ppm a.e. (LC_{50})	NA	NA
triclopyr BEE (93% a.i.)	>6,689 ppm a.e. (LC_{50})	2,150 ppm (≈350 mg a.e./kg bw)	4,640 ppm, reduced food consumption and bw gain
triclopyr BEE (96.1% a.i.)	>3,885 ppm a.e. (1087.8 mg a.e./kg bw) (LC_{50})	961 ppm a.i. (246 mg a.e./kg bw/day)	1,711 ppm a.i. (≈313 mg a.e./kg bw/day) (reduced body weight gain)
TCP	>5,620 ppm (LC_{50})	NA	562 ppm (reduced body weight gain)
Reproductive Studies			
Mallard duck			
triclopyr acid (98.9% a.i.)		100 ppm (10 mg/kg bw/day)	200 ppm (20 mg/kg bw/day)
Northern bobwhite quail			
triclopyr acid (98.9% a.i.)		100 ppm (≈7.5 mg a.e/kg bw)	200 ppm (≈15 mg a.e/kg bw)

NA – Not available. Source: USDA FS, 2011a

Field studies using triclopyr applications in Forest Service programs did not cause adverse effects in birds (Boren et al., 1993; Schulz et al., 1992a,b). Benefits were observed in some cases where certain bird species benefited from changes in vegetation.

Based on the acute gavage LD_{50} (>2,000 mg/kg bw) for TCP in bobwhite quail, TCP is less toxic than triclopyr acid, triclopyr TEA, or triclopyr BEE. The NOAEL of 125 mg/kg bw for TCP is similar to the NOAEL of ≅126 mg a.e./kg bw for triclopyr BEE. The dietary LC_{50} for TCP in mallards is >5,620 ppm suggesting it is practically non-toxic with a reduced body weight gain noted at 562 ppm.

Imazapyr

Imazapyr has low acute toxicity to birds based on available avian studies. The acute gavage studies with single oral doses of imazapyr acid resulted in no signs of toxicity at a dose of 2,510 mg a.e./kg bw in either quail or ducks (USEPA, 2007). The study using the Arsenal® herbicide formulation also caused no signs of toxicity at doses up to 2,150 mg formulation/kg bw, equivalent to about 486 mg a.e./kg bw. The acute NOAEL was the highest dose tested based on a lack of adverse effects. Results of these studies suggest imazapyr is considered practically non-toxic to birds.

The longer-term (≈18 week) reproduction studies using the imazapyr acid indicate no adverse effects following exposures to dietary concentrations of up to 2,000 ppm a.e. (USDA FS 2011a).

A field study reported no changes in bird populations after imazapyr was applied at about 3.7 lb a.e./acre for site preparation (Brooks et al., 1995). The visual surveys did not note any impacts on bird diversity, relative to sites treated with picloram, triclopyr, or hexazinone. Another field study indicated that imazapyr can improve bobwhite quail habitat by controlling hardwood establishment in pine stands (Welch et al., 2004).

There was a reported bird kill incident in Aiken County, South Carolina from spraying a mixture of herbicides (imazapyr, diuron, and metsulfuron-methyl) on fence rows that may have drifted onto adjacent bird nest boxes. Imazapyr, as well as diuron and metsulfuron-methyl, were used in the incident and no definitive link to the herbicide treatments could be made (USDA FS, 2011b).

Metsulfuron-methyl

Metsulfuron-methyl has low avian acute toxicity based on acute and subchronic toxicity studies conducted in bobwhite quail and mallards (USDA FS, 2005). The 14-day oral LD_{50} of technical grade metsulfuron-methyl administered by gavage in adult bobwhite quail was >2,250 mg/kg. The 5-day dietary exposure of juvenile ducks and quail to technical grade metsulfuron-methyl at concentrations ranging from 292 to 5,620 ppm did not result in any. observable toxicity. The only sign of toxicity following a 5-day dietary metsulfuron-methyl exposure of 3,160 ppm to 10-day-old bobwhite quail and 5,620 ppm to 8-day old mallard was weight loss. The NOAELs for weight loss ranged from 1,780 to 3,160 ppm in both studies.

Two 23-week feeding studies on reproductive effects for metsulfuron-methyl in bobwhite quail and mallards reported a NOAEL of 1,000 ppm (the highest dose tested) for chronic dietary exposure (USDA FS, 2005).

3.2.2.3 Reptiles and Amphibians (terrestrial phase)

No information regarding the toxicity of imazapyr to reptiles or terrestrial-phase amphibians was identified in the open literature or in studies submitted to the USEPA (USEPA, 2005b; 2006; 2007). The USEPA ecological risk assessment for imazapyr includes a risk evaluation to terrestrial phase amphibians using birds as a surrogates for terrestrial phase amphibians and reptiles (USEPA, 2005b). The same approach is used in the USEPA risk assessment for the California red legged frog (USEPA, 2007). The permeability of amphibian skin to pesticides and other chemicals is a concern for this approach. No data is available on the permeability of amphibian skin to imazapyr. Quaranta et al. (2009) noted that the skin of the frog *R. esculenta* is much more permeable to three pesticides (atrazine, paraquat, and glyphosate) than the skin of pig ear due to the differences in the structure and function of amphibian skin relative to mammalian skin.

An open literature search did not identify any reptile toxicity studies using metsulfuron-methyl but identified one study on the toxicity of metsufluron-methyl to terrestrial-phase amphibians (Lajmanovich et al., 2013). The study determined the individual toxicity of herbicide formulations contining glyphosate, metsulfuron-methyl, bispyribac-sodium, and picloram, and compared the mixture toxicity of three binary combinations (glyphosate and metsulfuron-methyl, glyphosate and bispyribac-sodium, glyphosate and picloram) on *Rhinella arenarum* tadpoles. The 48 hour LC_{50} for metsulfuron-methyl is 105.56 mg a.i./L. The NOEC and LOEC for metsulfuron-methyl were 80 mg a.i./L and 160 mg a.i./L, respectively.

3.2.2.4 Terrestrial Invertebrates and Microorganisms

<u>Terrestrial Invertebrates</u>
USEPA/OPP (1998) classifies triclopyr as practically non-toxic to bees based on the results of acute contact toxicity studies in honey bees using triclopyr acid and triclopyr TEA (contact LD_{50} values >100 μg/bee). The contact LD_{50} for triclopyr BEE to the honey bee is reported as >72 μg/bee suggesting low toxicity (USEPA, 2009).

The acute (14 day) toxicity studies in earthworms using triclopyr, triclopyr TEA, triclopyr BEE, and Garlon® 4 indicate that triclopyr acid is the least toxic to earthworms with an NOAEL of approximately 790 ppm a.e. (i.e., 790 mg a.e./kg soil dry weight), and an LC_{50} of 1,110 ppm a.e. (USDA FS, 2011a). Triclopyr TEA is the most toxic form of triclopyr to earthworms with an LC_{50} of about 146 ppm a.e., and an LOAEC of 134 ppm a.e. based on a significant increase in mortality (35% relative to 0% in the control groups) and a significant decrease in body weight (17%) relative to the control group. This study suggests that triclopyr TEA may be moderately toxic to earthworms relative to triclopyr acid.

A chronic 56-day earthworm bioassay testing Garlon® 4 showed no adverse effects on reproduction or growth at a concentration of 6.9 ppm a.e. (USDA FS, 2011a). The study on the impacts of Garlon® 3A to earthworms and other invertebrates at an application rate of 0.56 kg a.i./ha (≈0.36 lb a.e./acre) to turf plots showed consistent results with no significant reduction in mixed earthworm populations, mites, springtails, or ants in turf and soil core samples (Potter et al. 1990).

A series of field studies using triclopyr in broadcast applications suggested some effects to terrestrial invertebrates including beetles, butterflies, and spiders (USDA FS, 2011a). These effects are considered secondary because they were attributable to changes in vegetation cover, similar to changes in invertebrate populations observed with other vegetation management methods, instead of the potential toxic effect of triclopyr. Imazapyr has low acute toxicity to to the honey bee based on the acute contact LD_{50} of greater than 100 µg/bee, which is equivalent to about 860 mg/kg bw (USEPA, 2005b; 2007). There is no information available on other potential subchronic or non-lethal effects in bees or other invertebrate species.

Honey bee toxicity studies using metsulfuron-methyl indicate that the acute LD_{50} is greater than 25 µg/bee and possibly greater than 100 µg/bee, suggesting very low toxicity (USDA FS, 2005). These values correspond to doses ranging from about 270 to 1,075 mg/kg (0.025 mg/0.000093 kg to 0.1 mg/0.000093 kg) using a body weight of 0.093 g for the honey bee (USDA APHIS, 1993).

A toxicity study on three insect species (large white butterfly, beetle, and grain aphid) placed on plants sprayed with metsufuron-methyl at an application rate of 0.00004 to 0.003 lbs a.i./acre showed no adverse effects on survival or larval growth rate (Kjaer and Heimbach, 2001).

Terrestrial Microorganisms

Triclopyr is unlikely to have an impact on soil microorganisms (i.e., soil microbial function or community structure) based on studies on growth inhibition of fungi species (Chakravarty and Sidhu, 1987; USDA FS, 2011a). A study of Garlon® 4 (Estok et al., 1989) indicated a significant reduction of radial growth in fungi species at concentrations ≥1,000 ppm with total growth inhibition at ≥5,000 ppm. The slowest growing fungus was the least sensitive to triclopyr. Triclopyr TEA did not show any impacts on soil microbial function or community structure at an application rate of 1.9 kg a.i./ha (Houston et al., 1998).

Information on effects on terrestrial microorganisms is not available for imazapyr (USEPA, 2005b; 2007). Metsulfuron-methyl studies showed that at a concentration of 5 ppm in culture inhibited the growth of several strains of *Pseudomonas* (USDA FS, 2005). The inhibition of terrestrial microorganisms was attributed to ALS inhibition because terrestrial microorganisms have an enzyme involved in synthesis of branched chain amino acids. This enzyme is functionally equivalent to the target enzyme in terrestrial macrophytes.

3.2.2.5 Terrestrial Plants

Triclopyr

Triclopyr causes uncontrolled abnormal growth in plants by mimicking indole auxin plant growth hormones. It is a selective herbicide that is most toxic to broadleaf plants (Lewer and Owen, 1990). Triclopyr is effective in the control of dicots and relatively ineffective in controlling monocots through foliar application (USDA FS, 2011a).

The vegetative vigor studies in non-target plants from direct foliar application to young plants indicate that triclopyr TEA and triclopyr BEE have similar toxicities to a range of species (sunflower, sugar beet, tomato, oilseed rape, radish, soybean, wheat, corn, and onion for triclopyr TEA, and alfalfa, carrots, corn, oats, onions, radishes, soybeans, sunflowers, tomatoes, and wheat for triclopyr BEE). The sunflower (a dicot) is the most sensitive species for both TEA (EC_{25} = 0.005 lb a.e./acre) and triclopyr BEE (EC_{25} ≈0.0064 lb a.e./acre). Wheat and oats (monocots) are much more tolerant with EC_{25} values in excess of 0.3 lb a.e./acre for triclopyr TEA and triclopyr BEE (USDA FS, 2011a). The range of NOECs for triclopyr TEA ranges from 0.0028 lb a.e./acre testing sunflowers and soybeans to 0.23 lb a.e./acre for barley based on effects to shoot length (USEPA, 2009). The NOECs in alfafa, carrots, corn, oats, onions, radishes, soybeans, sunflowers, tomatoes, and wheat from triclopyr BEE exposure, based on shoot length and shoot weight effects, range from 4.4 g a.i./ha (sunflowers for shoot length, and carrots and sunflowers for shoot weight) to >2,242 g a.i./ha (oats). The NOECs from another study using triclopyr BEE were <0.063 lb a.e./acre (onions) and 0.028 lb a.e./acre (sunflowers) (USEPA, 2009).

The seedling emergence studies indicate that triclopyr BEE is about equally effective against dicots and at least some monocots. Triclopyr BEE is much more toxic than triclopyr TEA in some species. For example, the EC_{25} in alfafa for triclopyr BEE is 40 g a.i./ha (≈0.02 lb a.e./acre), which is much lower than the EC_{25} values for triclopyr TEA for all tested species (>0.23 lb a.e./acre). The higher phytotoxicity of triclopyr BEE may relate to faster absorption compared to triclopyr TEA that has been reported in chickweed, wheat, and barley (Lewer and Owen 1990). Variation in species sensitivity to triclopyr BEE may directly relate to the plant’s rate of metabolic ester hydrolysis (Lewer and Owen, 1990).

Field studies using triclopyr have shown some adverse impacts at high application rates and using broadcast applications. The field studies in various conifer species show that these species tend to be tolerant to triclopyr exposure in spring (April and May) and after fall dormancy (September) (USDA FS, 2011a).

Exposure to triclopyr drift may cause long-term impacts on some bryophyte and lichen communities (Newmaster et al., 1999). Some Forest Service events suggested that volatilization of triclopyr BEE may damage nontarget plants if it is applied under a poorly ventilated canopy and high temperatures. However, none of the field studies involving triclopyr BEE documented damage to nontarget plant species through volatilization (USDA FS, 2011a).

Imazapyr

Imazapyr inhibits acetolactate synthase, an enzyme that catalyzes the biosynthesis of branched-chain amino acids essential for protein synthesis and plant growth (USDA FS, 2011b). After foliage application, imazapyr is rapidly transported by the phloem from the treated leaves to the roots . Imazapyr has the potential to induce allelopathic effects; however, the potential for allelopathic effects may not substantially increase the risk to non-target plants because its movement in soil is relatively rapid (USDA FS, 2011b). The bioassays for vegetative vigor (i.e., post-emergence applications) and seedling emergence (i.e., pre-emergence applications) in response to foliar applications of imazapyr indicate that dicots are more sensitive to imazapyr than monocots. The differences between dicots and monocots are greater in the vegetative vigor assays compared to the seedling emergence studies. In the vegetative vigor assays, the EC_{25} of the most sensitive dicot, cucumber (0.0009 lb a.e./acre) is about a factor of 13 below the EC_{25} of the most sensitive monocot, wheat (0.012 lb a.e./acre). The NOAECs for the acid (22.6% a.e.) for dicots range from 0.000064 lb a.e./acre in cucumber to 0.0039 lb a.e./acre in sunflower, and for monocots range from 0.0039 lb a.e./acre in wheat and oat, to 0.0078 lb a.e./acre in corn. The NOAECs for the isopropylamine salt for dicots are 0.001 lb a.e./acre in sugar beet and 0.008 lb a.e./acre in soybean, and for monocots is 0.005 lb a.e./acre in onion. In seedling emergence assays, the EC_{25} of the most sensitive dicot, sugar beet (0.0024 lb a.e./acre) is a factor of approximately 2 below the most sensitive monocot species, wheat (0.0046 lb a.e./acre). The NOAECs for imazapyr acid (22.6% a.e.) in dicots are 0.00017 lb a.e./acre in sugar beet and 0.0003 lb a.e./acre in tomato, and in monocots range from 0.00099 lb a.e./acre in wheat to 0.0156 lb a.e./acre in oat. For the most sensitive species of dicot, foliar applications (vegetative vigor) are more toxic than soil exposures (seedling emergence) by about a factor of about 2.6, and vice versa for the most sensitive species of monocot (USDA FS, 2011b; USEPA, 2005b).

Metsulfuron-methyl

Metsulfuron-methyl inhibits acetolactate synthase, an enzyme that catalyzes the biosynthesis of three branched-chain amino acids (valine, leucine, and isoleucine) essential for plant growth.

Laboratory toxicity studies (pre- and post-emergence bioassays) using metsulfuron-methyl to evaluate the effects to terrestrial plants by a direct application has been evaluated for 10 species of plants including both dicots (soybean, cocklebur, cotton, morningglory, wild buckwheat, and sugar beet) and monocots (corn, barnyardgrass, rice, and nutsedge) (USDA FS, 2005). The study results showed that the most sensitive species was the morningglory with 70% growth inhibition at pre-emergence applications of about 0.00022 lbs a.i./acre. At the same application rate, the cocklebur and sugar beet showed 20% and 40% growth inhibition, and rice was the only monocot to respond with 20% inhibition. All of the plants showed 60 to 100% growth inhibition at an application of 0.014 lbs a.i./acre which is approximately a factor of two below the typical application rate used by the Forest Service (0.03 lb/acre).

Other laboratory toxicity studies on pre-emergence and post-emergence were performed in corn, cucumber, onion, pea, rape, sugar beet, sorghum, soybean, tomato, and wheat

(USDA FS, 2005). The pre-emergence assay showed that the most sensitive species were cucumber and onion with a NOEC of 0.000037 lb/acre, and the most tolerant species was wheat with a NOEC of 0.0056 lb/acre. The post-emergence assay showed that the cucumber was also the most sensitive species, with a NOEC of 0.000037 lb/acre, and the most tolerant species was wheat with a NOEC of 0.0039 lb/acre.

Field studies have shown that the lowest application rate associated with adverse effects (a decreased yield of both tomatoes and onions) was 0.1 g/ha (USDA FS, 2005; Obrigawitch et al., 1998), which is similar to the LOEC of 0.25 g/ha reported for other dicots. The most tolerant species in the field studies (Obrigawitch et al., 1998) consisted of various grasses for which NOEC values based on crop yield ranged up to 6 g/ha (NOEC values for wheatgrass and bromegrass). The NOEC value of 6 g/ha is equivalent to an application rate of about 0.0054 lb/acre, which is identical to the NOEC value for wheat (0.0056 lb/acre).

4.0 EXPOSURE ASSESSMENT

4.1 Human Health Exposure Assessment

Exposure assessments estimate the potential exposure of humans to triclopyr, imazapyr, and metsulfuron-methyl. The exposure assessment begins with the use and application methods of triclopyr, or triclopyr mixed with imazapyr and metsulfuron-methyl products. An identified exposure pathway for each herbicide includes (1) a release from a source, (2) an exposure point where contact can occur, and (3) an exposure route such as ingestion, inhalation, or dermal contact by which contact can occur (USEPA, 1989). Exposures for the identified human populations are qualitatively evaluated for each identified exposure pathway.

4.1.1 Identification of Potentially Exposed Human Populations and Complete Exposure Pathways

Under the expected use pattern and the proposed application methods (manual painting or direct spray treatment using a backpack sprayer), workers in the program who mix and apply the herbicides in the field are the most likely human population with the potential to be exposed to these compounds. Dermal exposure may occur from both the manual painting application and backpack spray application as well as and spill/splashing-during mixing and loading herbicide into the containers used during application. Exposure during transportation is not anticipated because the container of the concentrated material is sealed. Following label directions including the use of proper PPE will minimize exposure to workers. Accidental exposure may occur during mixing and applying the formulations. Under an accidental spill scenario, workers may be exposed to triclopyr, imazapyr, and metsulfuron-methyl through dermal contact. However, the potential dermal contact exposure would be limited because these accidental events would be of low frequency and short duration.

In addition to worker exposure, there is the possibility of exposure to residents in areas after treatment. The general public is not expected to be exposed to the herbicides during the application process. Exposure of residents whose property has been treated can be minimized through proper notification prior to treatment. However, there is potential for the general public to be exposed to herbicides shortly after application by walking through a treated area and sitting on treated stumps. Exposure to the general public to triclopyr residues was evaluated. Sensitive members of the population (i.e., adult females and pre-teen and teenaged children (age 11 to 16)) were used for the general public exposure assessment. Dermal contact with treated tree surfaces is the primary exposure route for the general public. Incidental ingestion and inhalation exposure routes are highly unlikely and were not evaluated. A dietary exposure route to the public is not expected because all of the herbicides will be used to only to treat tree stumps or sprouting vegetation.

A significant exposure pathway is not idenified for groundwater or surface water in proximity of herbicide-treated stumps. Although triclopyr, imazapyr, and metsulfuron-methyl are all soluble, mobile, and can leach from soil to groundwater, the leaching potential is limited because the herbicides are applied directly to the tree stump or associated sprouting vegetation, which minimizes contact with the soil. Although surface runoff from treated stumps into aquatic resources is possible, the use pattern of spot treatment, current label restrictions, and low application rates limit the amount of these herbicides runoff to surface water.

4.1.2 Exposure Evaluation

This section quantitatively evaluates worker exposure from direct contact while applying triclopyr using direct stump applications and backpack spraying, as well as to the general public from exposure resulting from direct contact with contaminated vegetation. The exposure scenarios evaluated for workers and the general public are related to short-term exposures because these herbicides are not persistent in the environment and probability of long-term exposure is low.

Workers: The general worker exposure evaluated in this assessment is based on the two application methods, manual painting and backpack direct spray. For the manual painting application, the direct contact exposure scenario to a triclopyr solution is characterized by wearing triclopyr-contaminated gloves for 1 minute and 10 minutes. It is assumed that wearing gloves grossly contaminated with a chemical solution is equivalent to immersing the hands in the pesticide solution. Therefore, the chemical concentration in contact with the skin and the dermal absorption rate are essentially constant. Because the concentration of triclopyr in contact with the skin is nearly constant and the rate of absorption is constant, zero-order absorption (i.e., the dermal absorption rate is constant over time) kinetics is used. The rate of absorption under this exposure scenario is estimated based on central estimate (range) dermal permeability coefficients (K_p) of 1.3×10^{-4} ($7.8 \times 10^{-5} – 2.3 \times 10^{-4}$) cm/hour for triclopyr TEA and 8.3×10^{-3} ($4.4 \times 10^{-3} – 1.6 \times 10^{-2}$) cm/hour for triclopyr BEE (USDA FS, 2011c,d).

For the backpack spray application, the direct contact exposure scenario estimate for a backpack worker to a triclopyr solution are characterized based on the amount of material handled per day. The worker exposure rate is 0.01 mg/kg bw per lb and ranges between 0.001 mg/kg bw per lb and 0.08 mg/kg bw per lb. The worker exposure rates are the geometric mean and 95% confidence intervals, based on a log-transformation of data from a Forest Service assessment of worker exposure rates for triclopyr BEE (USDA FS, 2012).

For the spill/splash related dermal contact exposure scenario, exposure to a triclopyr solution is characterized by a spill on the lower legs as well as the hands, with a certain amount of the chemical adhering to the skin. The absorbed dose is calculated as the product of the amount of chemical on the surface of the skin (i.e., the amount of liquid per unit surface area multiplied by the surface area of the skin over which the spill occurs and the chemical concentration in the liquid), the first-order dermal absorption rate coefficient (Ka) (expressed as a proportion of the deposited dose absorbed per unit time—e.g., hour-1), and the duration of exposure. The rate of absorption under this exposure scenario is estimated based on the first-order dermal absorption rate coefficient (Ka) of 6.7×10^{-4} (2.4×10^{-4} – 1.8×10^{-3}) hour^{-1} for triclopyr TEA and 2.6×10^{-3} (1.0×10^{-3} – 6.7×10^{-3}) hour^{-1} for triclopyr BEE (USDA FS, 2011c,d). Details regarding exposure assumptions are presented in Attachment A.

General Public: The exposure scenario developed for the general public assumes that an individual is exposed to the compound shortly after its application. The exposure scenario from contact with contaminated tree stumps assumes that the herbicide is sprayed at a given application rate, and that an adult female or a pre-teen and teenage child comes in contact with the sprayed tree stump surfaces on the same day. This exposure scenario depends on estimates of dislodgeable residue (the estimated amount of the chemical which could be released from vegetation, expressed in units of pesticide mass/surface area of vegetation), and dermal transfer rates (the rate at which the chemical is transferred from the contaminated vegetation to the surface of the skin). Dislodgeable residues are based on the pesticide, the formulation, and the site-specific conditions. Dermal transfer rates are reasonably consistent for a number of pesticides (Durkin et al.1995). This exposure assessment assumes a contact period of 10 minutes and that the chemical is not effectively removed by washing within 1 or 2 hours of exposure. The assumptions regarding body weight, skin surface area, and first-order dermal absorption rates are summarized in Attachment A.

4.2 Ecological Exposure Assessment

4.2.1 Aquatic Exposure Assessment

The three herbicides proposed for use in the ALB Eradication Program exhibit properties that suggest they could contaminate surface water. High solubilty and low soil adsorption coefficients increase the likelihood of off-site transport from runoff. Drift is not anticipated to be a significant pathway to aquatic environments from the proposed treatments. Herbicides are either hand painted onto stumps or backpack applications are made directly to the the stump and associated vegetation. The use of backpack sprayers, large coarse droplets during application, and other label requirements will result in negligible drift. Estimates of potential aquatic residues from runoff were evaluated for triclopyr BEE since it has a greater toxicity to aquatic vertebrates and invertebrates compared to triclopyr TEA, imazapyr, and metsulfuron-methyl. Aquatic residue values for this assessment were generated using the GLEAMS environmental fate model (USDA FS, 2003). The Forest Service determined the residues of triclopyr BEE in a small stream for three soil types and average annual rainfall amounts ranging from 5 to 250 inches per year. The application rate was1 lb a.e./acre with resulting residues ranging from zero to 0.149 mg/L. The highest concentrations were observed at the maximum average annual rainfall levels and in sandy soils which would be highly permeable and susceptible to runoff. The residues from these estimates were then compared to the available aquatic effects data for triclopyr BEE which is further discussed in the risk characterization section of this risk assessment.

4.2.2 Terrestrial Exposure Assessment

The potential for herbicide exposure to non-target terrestrial vertebrate wildlife is very low under the proposed uses. Treatments to stumps would not result in any appreciable exposure to non-target terrestrial vertebrates through either the oral, dermal, or inhalation pathways. In addition, the available toxicity data for birds and mammals through these exposure pathways demonstrates very low toxicity. There is the possibility that some mammals may graze on treated sprouting vegetation; however, there is no plausible exposure scenario resulting in adverse effects based on daily food consumption rates. There is the potential for exposure from drinking water. However, estimates for triclopyr BEE, discussed above, are very low and any non-target vertebrate wildlife would have to consume several times their daily water intake to receive a dose that would approach an adverse effect.

5.0 RISK CHARACTERIZATION

5.1 Human Health

Risks associated with adverse human health effects are characterized quantitatively in this section. The potential adverse health effects to subgroups within the population from exposure to triclopyr were evaluated using a hazard quotient (HQ) approach. A HQ is the ratio of the estimated level of total exposure to the relevant noncancer toxicity criterion (i.e., appropriate RfD) for a specific exposure pathway (or for a single substance). If there is more than one chemical, the HQs are summed over all exposure pathways and all chemicals to develop a total hazard index (HI). If the HI is less than 1, it is considered unlikely that the exposure will cause adverse health effects; conversely, if the HI is greater than 1, then health effects may result from exposure. Unlike cancer risk estimates that are expressed as probabilities (e.g., 1 in a million), HIs are deterministic methods of estimating risk and do not represent the probability of health effects, but are presented to provide context to the relationship between the known toxicity of a substance and the estimated magnitude of exposure. The risk estimations associated with the triclopyr exposure scenario are expected to be representative of the mixed herbicide scenario. This is because triclopyr has at least an order of magnitude higher toxicity compared to the other two herbicides (RfD of 0.05 mg/kg/day for triclopyr, RfDs of 2.5 mg/kg/day for imazapyr, and 0.25 mg/kg/day for metsulfuron-methyl). Adding HQs from the other two compounds is not expected to raise the total HI to exceed the USEPA level of concern (HI of 1).

Workers: The HQs for general and accidental exposures of workers are calculated and summarized in table 5-1. The HQs are based on the total exposure that a worker might receive during a directed foliar application at a label allowable maximum application rate of 9 lb/acre

(TEA) and 8 lb/acre (BEE) with the acute RfD of 1 mg/kg/day. The use of the acute RfD for risk characterization for the general worker exposure (male workers) is intended only to illustrate the consequences of applying triclopyr sporadically as part of other activities. The HI values for the three worker-exposure scenarios (i.e., manual painting application direct contact, backpack spray application direct contact, and spill/splashing-related dermal contact) are less than or do not exceed the USEPA level of concern of 1 under the central and lower estimations for both triclopyr TEA and triclopyr BEE. The upper estimate HQ of 7 for triclopyr BEE under the contaminated gloves with 10 minutes exposure scenario and the upper estimate HQs of 6 for triclopyr TEA and 5 for triclopry BEE under the backpack application exposure scenario exceeded the level of concern. However, these risk estimations are conservative values based on the maximum labelled application rates (9 lb a.e./acre TEA and 8 lb a.e./acre BEE) instead of a typical application rate (1 lb a.e./acre). The hazard quotients calculated for a typical application rate are less than one (table 5-2). The backpack application exposure scenario also conservatively assumes no protective clothes resulting in overestimation of risk, which will be minimized by the proper use of PPE. The central risk estimates are a more typical exposure scenario. Therefore, it is unlikely that the exposure to workers will cause adverse health effects.

Table 5-1. Hazard quotient values estimated for workers making triclopyr applications at maximum rates.

		Hazard Quotient		
Exposure Scenario	Form	Central Estimate	Lower Estimate	Upper Estimate
Accidental/Incidental Exposure				
Contaminated Gloves, 1 min.	TEA	2×10^{-3}	5×10^{-4}	1×10^{-2}
	BEE	7×10^{-2}	2×10^{-2}	7×10^{-1}
Contaminated Gloves, 10 mins	TEA	1×10^{-2}	5×10^{-3}	1×10^{-1}
	BEE	7×10^{-1}	2×10^{-1}	**7**
Spill on Hands, 1 hour	TEA	3×10^{-3}	7×10^{-4}	4×10^{-2}
	BEE	1×10^{-2}	3×10^{-3}	1×10^{-1}
Spill on lower legs, 1 hour	TEA	7×10^{-3}	1×10^{-3}	9×10^{-2}
	BEE	2×10^{-2}	6×10^{-3}	3×10^{-1}
General Exposures				
Backpack Applications	TEA	4×10^{-1}	1×10^{-3}	**6**
	BEE	4×10^{-1}	1×10^{-3}	**5**
HI	TEA	0.4	0.008	**6**
	BEE	1	0.2	**12**

Bold and shaded - Hazard quotient/index exceeding the target hazard quotient/index of 1.
Note: HI is the sum of HQs of the contaminated gloves (10 minutes), spill on hands, spill on lower legs, and backpack applications exposure scenarios.

Table 5-2. Hazard quotient values estimated for workers making triclopyr applications at typical application rate.

Exposure Scenario	Form	Hazard Quotient		
		Central Estimate	Lower Estimate	Upper Estimate
Accidental/Incidental Exposure				
Contaminated Gloves, 1 min.	TEA	2×10^{-3}	5×10^{-5}	1×10^{-3}
	BEE	9×10^{-3}	3×10^{-3}	8×10^{-2}
Contaminated Gloves, 10 mins	TEA	1×10^{-3}	5×10^{-4}	1×10^{-2}
	BEE	9×10^{-2}	3×10^{-2}	8×10^{-1}
Spill on Hands, 1 hour	TEA	3×10^{-4}	8×10^{-5}	5×10^{-3}
	BEE	1×10^{-3}	3×10^{-4}	2×10^{-2}
Spill on lower legs, 1 hour	TEA	7×10^{-4}	2×10^{-4}	1×10^{-2}
	BEE	3×10^{-3}	7×10^{-4}	4×10^{-2}
General Exposures				
Backpack Applications	TEA	4×10^{-2}	2×10^{-4}	6×10^{-1}
	BEE	4×10^{-2}	2×10^{-4}	6×10^{-1}

General Public: The HQs for potential herbicide exposure to the general public are calculated and summarized in table 5-3. The HQs are based on the exposure scenario where an adult female or a pre-teen and teenaged child (age 11-16) could be exposed to dislodgeable residues shortly after a foliar application at the label allowable maximum application rates of 9 lb/acre (TEA and TCP) or 8 lb/acre (BEE), for a direct contact time of 10 minutes and exposure duration for 1 hour (adult female) and 2 hour (pre-teen and teenaged child). The chronic RfDs of 0.05 mg/kg/day (TEA and BEE) and 0.012 mg/kg/day (TCP) were used for the adult female as a conservative approach. The HIs for the adult female and pre-teen and teenaged child are all less than USEPA level of concern of one suggesting that it is unlikely that herbicide exposure to the general public will cause adverse health effects.

Table 5-3. Hazard quotient values estimated for general public after triclopyr applications.

Exposure Scenario	Form	Hazard Quotient		
		Central Estimate	Lower Estimate	Upper Estimate
Non-accidental Acute Exposure				
Vegetation contact, shorts and T-shirt – an adult female				
	TEA	3×10^{-3}	1×10^{-3}	8×10^{-3}
	BEE	1×10^{-2}	4×10^{-3}	2×10^{-2}
	TCP	2×10^{-1}	7×10^{-2}	6×10^{-1}
HI		0.2	0.08	0.6
Vegetation contact, shorts and T-shirt – a pre-teen and teenaged child (11-16)				
	TEA	3×10^{-4}	1×10^{-4}	8×10^{-3}

	BEE	1×10^{-3}	4×10^{-4}	3×10^{-3}
	TCP	2×10^{-1}	8×10^{-2}	6×10^{-1}
HI		0.2	0.08	0.6

Under the proposed use patterns, triclopyr, and triclopyr mixed with imazapyr and metsulfuron-methyl should pose minimal risks to human health.

5.2 Aquatic and Terrestrial Ecological Risk Characterization

This section of the risk assessment will integrate the effects analysis regarding toxicity studies summarized for terrestrial and aquatic nontarget organisms with the potential for exposure to quantify whether direct or indirect risks would be anticipated for nontarget organisms and domestic animals from program use of triclopyr, imazapyr, and metsulfuron-methyl. Direct risk refers to those risks that could occur from direct exposure to the herbicides while indirect risks refer to impacts that could occur to prey or habitat that nontarget organisms rely on for food and shelter. In cases where the range of response data to each herbicide does not fall within the range of potential exposure values, impacts to individuals and populations are not anticipated. Further evaluation of the assumptions used in the risk characterization is required to refine risk where residues exceed the response data.

5.2.1 Aquatic

The available aquatic toxicity data for vertebrates and invertebrates for all three herbicides suggests that these taxa would be at very low risk from herbicide applications. Triclopyr BEE poses the greatest risk to aquatic vertebrates and invertebrates based on its toxicity profile while imazapyr and metsulfuron-methyl pose the least amount of risk because both products would be considered practically non-toxic to aquatic invertebrates and vertebrates. Based on the low frequency of use in the ALB program and the method of application, there is a low likelihood that any associated drift and runoff from these applications would result in significant risk to aquatic vertebrates or invertebrates. The risk to aquatic plants is greater based on the known toxicity for each herbicide. Effects to aquatic plants can result in indirect effects to aquatic vertebrates and invertebrates that rely on these species for food and shelter. The proposed use pattern would minimize the potential for significant residues from drift or runoff to aquatic habitats. In addition, current label requirements restricting applications in aquatic areas would further reduce the potential for residues.

The acute aquatic risk from the use of triclopyr BEE was determined for aquatic vertebrates, invertebrates and plants based on a typical application rate (1 lb a.e./acre) and the maximum labelled used rate (8 lb a.e./acre) (figure 5-1).

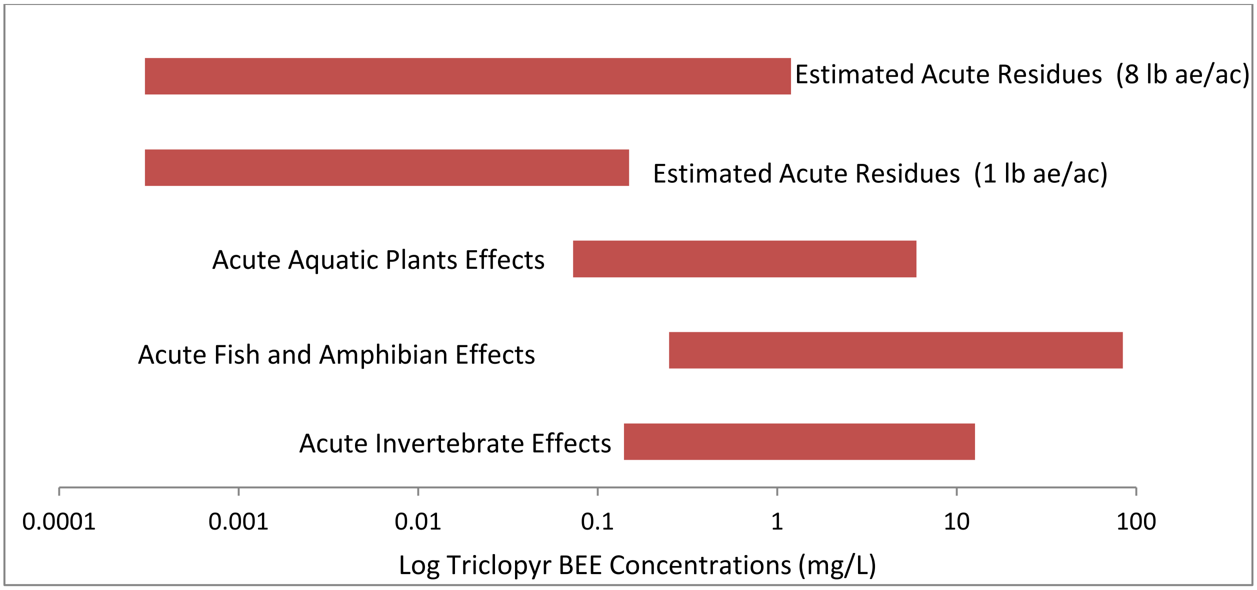


Figure 5-1. Aquatic risk characterization for triclopyr BEE.

Residue values were compared to the range of acute aquatic toxicity data. A presumption of risk occurs when there is an overlap between residues and the range of acute toxicity data. This was done for triclopyr BEE due to its ten fold or greater toxicity when compared to triclopyr TEA, imazapyr, and metsulfuron-methyl. The waterbody that was modeled in this exercise was a small stream (USDA FS, 2003). Residues were estimated using the GLEAMS environmental fate model for three soil types (clay, loam, and sand) and annual yearly rainfall amounts ranging from 5 to 250 inches. Chronic risk comparisons were not made because triclopyr BEE degrades rapidly in water to triclopyr acid, and chronic exposure to the ester would not occur. The residues estimated from this excerise are conservative estimates because they assume applications occur to soil, which would not be the case for proposed stump applications. There is the possibility of some triclopyr BEE moving to soil due to a rainfall event, but applications will be made to minimize the potential for runoff. Residues at 1 and 8 lb a.e./acre exceed the distribution of toxicity data for aquatic plants and exceed the effects distribution for aquatic vertebrates at the maximum labelled rate. Residues were below all toxicity values in all soil types when considering areas that had 100 inches or less of annual rainfall. Areas in the United States with annual average rainfall amounts greater than 100 inches are restricted to coastal areas in the Pacific Northwest and a small portion of northwest California. These are not considered high risk areas for the introduction of ALB, thus, the aquatic risk from triclopyr BEE for stump treatments is very low for a majority of the country based on conservative estimates of exposure. Due to the significant reduction in toxicity for triclopyr TEA, imazapyr, and metsulfuron-methyl the potential for aquatic risk would be much less for these three herbicides when compared to triclopyr BEE.

5.2.2 Terrestrial Wildlife and Domestic Animals

The risk to non-target terrestrial vertebrate wildlife and domestic animals will be negligible based on the proposed method of application for each herbicide and available toxicity data for mammals and birds. Direct application of herbicides to stumps and sprouting vegetation using hand painting or hand held sprayers reduces the likelihood of off-site drift and runoff. Any plausible scenarios where mammals and birds would be exposed to significant residues on stumps or associated sprouting vegetation is unlikely. Some mammals, such as white-tailed deer, may graze on sprouts from stumps that have been selectively treated with herbicides. However, the toxicity profiles for all three herbicides suggest mammals would have to consume many times their daily food consumption rates over an extended period of time to reach a dose that could result in any acute or chronic effects. Treated vegetation would become unpalatable as the sprout dies back due to herbicide treatment, reducing exposure time. Incidental soil ingestion by foraging mammals and birds would also be low because applications are directed to the stump and sprouting vegetation using hand held sprayers. Risk from herbicide exposure through drinking water to nontarget vertebrates would also be very low based on aquatic residues estimated in a small stream using applications of triclopyr BEE. Non-target vertebrate wildlife and domestic animals would have to consume many times their daily water ingestion rate to consume a dose that could result in an adverse effect. There is no plausible exposure scenario where that situation could occur. Risks from ingestion of drinking water containing triclopyr TEA, TCP, imazapyr, and metsulfuron-methyl would be even less due to their comparative lower toxicity, and in the case of imazapyr and metsulfuron-methyl, much lower use rates compared to triclopyr.

5.2.3 Terrestrial Invertebrates and Plants

Herbicide applications are not expected to result in adverse impacts to terrestrial invertebrates. Available toxicity data for terrestrial invertebrates shows that toxicity is low for pollinators, such as honey bees, as well as soil borne invertebrate species for all three herbicides. The greatest potential for exposure would be for soil borne invertebrates however, the method of application suggests that any significant residues would be immediately adjacent to any treated stumps.

All three herbicides are toxic to terrestrial plants, and non-target plants immediately adjacent to the treatment site would be at risk from drift and runoff. The method of application will reduce the risk to terrestrial plants because applications are made by hand directly to stumps and any associated sprouting vegetation which would minimize and drift and runoff.

6.0 UNCERTAINTIES AND CUMULATIVE IMPACTS

The uncertainties associated with this risk evaluation arise primarily from a lack of information about the effects of each herbicide, its formulations, metabolites, and potential mixtures to non-target organisms that can occur in the environment. These uncertainties are not unique to this assessment but are consistent with uncertainties in human health and ecological risk assessments with any environmental stressor. In addition, there is uncertainty in where an ALB infestation may occur in the United States and the extent of herbicide use in a given area because application is based on site-specific factors. As a way to account for some of the uncertainty with the available data APHIS made conservative assumptions regarding exposure and relied on the more sensitive effect endpoints to assess risk to human health and the environment.

Another area of uncertainty is the potential for cumulative impacts to human health and the environment from the proposed use of the three herbicides together and how they may interact with other biotic and abiotic stressors. Areas where cumulative impacts could occur are: 1) repeated worker and environmental exposures to each herbicide in relation to other uses; 2) co-exposure to other chemicals with a similar mode of action; and 3) exposures to other chemicals in mixtures and how that may affect the toxicity of each herbicide.

From a human health perspective, the ALB program use of triclopyr and the tank mix of the three herbicides are expected to result in negligible cumulative impacts when considering the proposed use patterns. The herbicides are not expected to enter the food chain through food and drinking water and significantly add to triclopyr, imazapyr, and melsulfuron-methyl exposures and other stressors in the human population.

Cumulative impacts may occur from the use of each herbicide as well as the three herbicides together in relation to other chemicals that have a similar mode of action, as well as others that have a different mode of action, but could result in synergistic, additive, or antagonistic effects. The cumulative impacts from the individual use of triclopyr or the three herbicides together are expected to be low because they are not persistent in the environment or bioaccumulative (Tatum 2004). In addition, their selective use in the ALB Eradication Program and the methods of application reduce exposure. The low probability of exposure and favorable toxicity profile result in very low risk to human health and to most non-target wildlife so that any cumulative impacts from the use of herbicides relative to other stressors would be incrementally negligible.

7.0 REFERENCES

BASF Corporation (BASF). 2012a. Pesticide label: Arsenal® herbicide, EPA Reg. No. 241-346. 21 pp.

BASF. 2012b. Safety data sheet Arsenal. Avalaible at http://www.env.gov.yk.ca/environment-you/documents/ArsenalSafetyDataSheet.pdf, last accessed 6/25/2014.

Boren, J.C., Lochmiller, R.L., Leslie, D.M. and D.M. Engle. 1993. Long-term effects of woody vegetation management on seasonal body condition of northern bobwhites. J Range Manag. 46: 520-523.

Brooks, J.J., Rodrigue, J.L., Cone, M.A., Miller, K.V., Chapman, B.R. and A.S. Johnson. 1995. Small mammal and avian communities on chemically-prepared sites in the Georgia sandhills. General Technical Report - Southern Research Station, USDA Forest. (No. S):21-23.

Budavari S. (Ed). 1989. The Merck Index: An Encyclopedia of Chemicals, Drugs, and Biologicals, 11th ed. Merck & Co., Inc., Rahway, New Jersey.

Chakravarty P. and Sidhu, S.S. 1987. Effect of glyphosate hexazinone and triclopyr on in vitro growth of five species of ectomycorrhizal fungi. Eur J for Pathol. 17 (4-5): 204-210.

Cox C. 2000. Triclopyr herbicide factsheet. Journal of Pesticide Reform. 20(4): 12-19. Available at http://www.pesticide.org/get-the-facts/pesticide-factsheets/factsheets/triclopyr, last accessed 12/22/2014.

Dow AgroSciences. 2010. Material safety data sheet Pathfinder® II herbicide. Available at http://www.dot.ca.gov/dist05/maint/ivm/msdsfiles/pathfinder2_msds.pdf, last accessed 6/25/2014

Dow AgroSciences. 2011a. Pesticide label: Garlon® 3A, EPA Reg. No. 62719-37, Alternate Brand Name: Element 3A. Dated October 24, 2011. 38 pp.

Dow AgroSciences. 2011b. Pesticide label: Pathfinder® II, specialty herbicide, EPA Reg. No. 62719-176. Dated May 6, 2011. 12 pp.

Dow AgroSciences. 2011c. Material safety data sheet Garlon® 3A herbicide. Available at http://www.aqumix.com/pdf/Garlon3AMSDS.pdf, last accessed 6/25/2014

DuPont. 2012a. Pesticide label: DuPont™ Escort® XP herbicide. EPA Reg. No. 352-439. Dated February 29, 2012. 21 pp.

DuPont. 2012b. Material safety data sheet DuPont™ Escort® XP herbicide. Ref. 130000036195. Available at http://www.plunketts.net/docs/msds/escort_xp-msds.pdf, last accessed 6/25/2014.

Estok, D., Freedman, B., and Boyle, D. 1989. Effects of the herbicides 2 4-D, glyphosate, hexazinone, and triclopyr on the growth of three species of ectomycorrhizal fungi. Bull Environ Contam Toxicol. 42(6): 835-839.

Fowlkes, M.D., Michael, J.L., Crisman, T.L. and J.P. Prenger. 2003. Effects of the Herbicide Imazapyr on Benthic Macroinvertebrates in a Logged Pond Cypress Dome. Environ Toxicol Chem. 22(4): 900-907.

Grisolia, C.K. 2002. A comparison between mouse and fish micronucleus test using cyclophosphamide, mitomycin C and various pesticides. Mutat Res. 518 :145–50.

Hanley, T., Thompson, D.J., Palmer, A.K., Beliles, R.P. and B.A. Schwetz. 1984. Teratology and reproduction studies with triclopyr in the rat and rabbit. Fundam Appl Toxicol. 4:872-882.

Harris, S.A. and K.R. Solomon. 1992. Human exposure to 2,4-D following controlled activities on recently sprayed turf. J Environ Sci Health. B27(1): 9-22.

Houston, A.P.C., Visser, S. and R.A. Lautenschlager. 1998. Response of microbial processes and fungal community structure to vegetation management in mixed wood forest soils. Can J Bot. 76: 2002-2010.

Hazardous Substances Data Bank (HSDB). 2014a. HSDB: Triclopyr, available via Toxnet at http://toxnet.nlm.nih.gov/cgi-bin/sis/search2/f?./temp/~TeYhGy:1, last accessed 6/27/2014.

HSDB. 2014b. HSDB: Imazapyr, available via Toxnet at http://toxnet.nlm.nih.gov/cgi-bin/sis/search2/f?./temp/~JU0Z9j:1, last accessed 6/27/2014

HSDB. 2014c. HSDB: Metsulfuron-methyl, available via Toxnet at http://toxnet.nlm.nih.gov/cgi-bin/sis/search2/f?./temp/~9S9p9F:1, last accessed 6/27/2014.

Lewer, P. and W.J. Owen. 1990. Selective action of the herbicide triclopyr. Pestic Biochem Physiol. 36(2): 187-200.

Johansen, J.A. and G.H. Geen. 1990. Sublethal and acute toxicity of the ethylene glycol butyl ether ester formulation of triclopyr to juvenile Coho salmon (Oncorhynchus kisutch). Arch Environ Contam Toxicol. 19(4): 610-616.

Kjaer, C. and U. Heimbach. 2001. Relationships between sulfonylurea herbicide treatment of host plants and the performance of herbivorous insects. Pest Manag Sci. 57 (12):1161-1166.

Kreutzweiser, D.P. and S.S. Capell. 1992. A simple stream-side test system for determining acute lethal and behavioral effects of pesticides on aquatic insects. Environ Toxicol Chem. 11: 993-999.

Lajmanovich, R.C., Junges, C.M., Attademo, A.M., Peltzer, P.M., Cabagna-Zenklusen, M.C. and A. Basso. 2013. Individual and Mixture Toxicity of Commercial Formulations Containing Glyphosate, Metsulfuron-Methyl, Bispyribac-Sodium, and Picloram on Rhinella arenarum Tadpoles, Water Air Soil Pollut 224:1404.

Mason, R.W. and B.L. Johnson. 1987. Ergonomic factors in chemical hazard control. In: Handbook of Human Factors. Salveny, G; ed. John Wiley and Sons, New York, NY. pp. 772-741.

Morgan, J.D., Vigers, G.A., Farrell, A.P., Janz, D.M. and J.F. Manville. 1991. Acute avoidance reactions and behavioral responses of juvenile rainbow trout (Oncorhynchus mykiss) to Garlon 4, Garlon 3A and Vision herbicides. Environ Toxicol Chem. 10(1): 73-80.

National Pesticide Information Center (NPIC). 2002. Triclopyr general fact sheet http://npic.orst.edu/factsheets/triclogen.pdf last accessed 3/28/2014.

National Research Council (NRC). 1983. Risk assessment in the Federal government: managing the process. National Academy Press, Washington, DC.

Newmaster, S.G., Bell, F.W. and D.H. Vitt. 1999. The effects of glyphosate and triclopyr on common bryophytes and lichens in northwestern Ontario. Canadian Journal of Forest Research. 29 (7): 1101-1111.

Obrigawitch, T.T., Cook, G. and J. Wetherington. 1998. Assessment of effects on non-target plants from sulfonylurea herbicides using field approaches. Pestic Sci. 52 (3):199-217.

Perkins, P.J., Boermans, H.J. and G.R. Stephenson. 2000. Toxicity of glyphosate and triclopyr using the frog embryo teratogenesis assay–Xenopus. Environ. Toxicol. Chem. 19(4): 940-945.

Peterson, H.G., Boutin, C., Martin, P.A., Freemark, K.E., Ruecker, N.J. and M.J. Moody. 1994. Aquatic phytotoxicity of 23 pesticides applied at expected environmental concentrations. Aquatic Toxicology (Amsterdam). 28 (3-4):275-292.

Potter, D.A., Buxton, M.C., Redmond, C.T., Patterson, C.G. and A.J. Powell. 1990. Toxicity of pesticides to earthworms (Oligochaeta: Lumbricidae) and effect on thatch degradation in Kentucky bluegrass turf. J Econ Entomol. 83: 2362-2369.

Quaranta, A., Bellantuono, V., Cassano, G. and C. Lippe. 2009. Why amphibians are more sensitive than mammals to xenobiotics. PLoS ONE. 4: (11): 1-4.

Schulz, C.A., Leslie, D.M. JR. Lochmiller, R.L. and D.M. Engle. 1992a. Autumn and winter bird populations in herbicide-treated cross timbers in Oklahoma. Am Midl Nat. 127 (2): 215-223.
Schulz, C.A., Leslie, D.M., Lochmiller, R.L. and D.M. Engle. 1992b. Herbicide effects on cross timbers breeding birds. J Range Manag. 45: 407-411.

Shackelford, D.D., Young, D.L., Mihaliak, C.A., Shurdut, B.A. and J.A. Itak. 1999. Practical immunochemical method for determination of 3,5, 6-trichloro-2-pyridinol in human urine: applications and considerations for exposure assessment. J Agric Food Chem. 47(1):177-82.

Tatum, V.L. 2004. Toxicity, transport, and fate of forest herbicides. Wildlife Society Bulletin, 32(4): 1042-1048. http://www.bioone.org/doi/pdf/10.2193/0091-7648%282004%29032%5B1042%3ATTAFOF%5D2.0.CO%3B2, last accessed 9/12/2014

Timchalk, C., Dryzga, M.D. and P.E. Kastl. 1990. Pharmacokinetics and metabolism of triclopyr 3,5,6-trichloro-2-pyridinyloxyacetic acid) in Fischer 344 rats. Toxicology. 62(1): 71-87.

Timchalk, C., Finco, D.R. and J.F. Quast. 1997. Evaluation of renal function in rhesus monkeys and comparison to beagle dogs following oral administration of the organic acid triclopyr (3,5,6-trichloro-2-pyridinyloxyacetic acid). Fundam Appl Toxicol. 36(1): 47-53.

Timchalk, C. and R.J. Nolan. 1997. Pharmacokinetics of triclopyr (3,5,6-trichloro-2-pyridinyloxyacetic acid) in the beagle dog and rhesus monkey: perspective on the reduced capacity of dogs to excrete this organic acid relative to the rat, monkey, and human. Toxicology and Applied Pharmacology. 144 (2): 268-278.

Trevathan W. (ed.). 2002. Metsulfuron-methyl pesticide fact sheet: forest use, Oregon State University Extension Service, Enviormental Health Sciences Center. 2 pp. available at http://www.oregon.gov/odf/privateforests/docs/metsulfuronmethyl.pdf, last accessed 6/27/2014.

United States Department of Agriculture Animal and Plant Health Inspection Service (USDA APHIS). 1993. Nontarget Risk Assessment for the MEDFLY Cooperative Eradication Program. USDA APHIS. February 1993.

United States Department of Agricuture Forest Service (USDA FS). 1989. Final environmental impact statement: vegetation management in the Appalachian Mountains, Management Bulletin R8-MB-38, dated July, 1989. 1104 pp.

USDA FS. 2003. Triclopyr - Human health and ecological risk assessment, Final Report. SERA TR 02-43-13-03b.

USDA FS. 2005. Metsulfuron Methyl - Human health and ecological risk assessment, Final Report. SERA TR 04-43-17-01c, Report dated December 9, 2004, Revised on February 28, 2005. http://www.fs.fed.us/foresthealth/pesticide/risk.shtml

USDA FS. 2011a. Triclopyr - Human health and ecological risk assessments, Final Report. SERA TR-052-25-03a, Report dated May 24, 2011 with revisions as of October 20, 2011. http://www.fs.fed.us/foresthealth/pesticide/risk.shtml

USDA FS. 2011b. Imazapyr – Human health and ecological risk assessment, Final Report. SERA TR-052-29-03a, Report dated December 16, 2011, http://www.fs.fed.us/foresthealth/pesticide/risk.shtml

USDA FS. 2011c. Forest Service Program Worksheets Triclopyr TEA Formulations: Terrestrial Applications, Worksheet Version: 5.00.64

USDA FS. 2011d. Forest Service Program Worksheets Triclopyr Triclopyr BEE Formulations: Terrestrial Applications, Worksheet Version: 5.00.64

USDA FS. 2011e. Forest Service Program Worksheets 3,5,6-Trichloro-2-pyridinol (TCP) from Terrestrial Applications of Triclopyr, Worksheet Version: 6.00.01

USDA FS. 2012. Reassessment of worker exposure rates, Final Report. SERA TR-052-30-02a, Report dated June 24, 2012. http://www.fs.fed.us/foresthealth/pesticide/pdfs/Worker_Exposure_2012.pdf

United States Environmental Protection Agency (USEPA). 1989. Risk assessment guidance for Superfund volume I, human health evaluation manual (Part A). interim final. Office of Emergency and Remedial Response, Washington, DC. EPA/540/1-89/002 December 1989.

USEPA. 1998. Registration eligibility document (RED). Triclopyr. EPA 738-R-98-011. Available at http://www.epa.gov/oppsrrd1/REDs/2710red.pdf, last accessed 6/26/2014.

USEPA. 2002a. Triclopyr; Pesticide Tolerance. Federal Register 40 CFR 180 67: 58712 - 58725. Available at: https://www.federalregister.gov/articles/2002/09/18/02-23746/triclopyr-pesticide-tolerance, Last accessed 6/27/2014

USEPA. 2002b. Metsulfuron Methyl; Pesticide Tolerance. Federal Register 40 CFR 180. 67(152): 51088-51097. Available at: https://www.federalregister.gov/articles/2002/08/07/02-19807/metsulfuron-methyl-pesticide-tolerance, Last accessed 2/13/2015

USEPA. 2005a. Imazapyr: Revised HED chapter of the reregistration eligibility document (RED). PC Code: 128821, CAS Reg 81334-34-1, Case #3078, DP Barcode: D324106. Reported dated: December 8, 2005. U.S. EPA E-Docket EPA-HQ-OPP-2005-0495. EPA File Name: EPA-HQ-OPP-2005-0495-0004.pdf. Available at: www.regulations.gov. Last accessed 2/14/2015
USEPA. 2005b. Level I screening ecological risk assessment for the reregistration of imazapyr. U.S. EPA E-Docket EPA-HQ-OPP-2005-0495. EPA File Name: EPA-HQ-OPP-2005-0495-0009.pdf. Available at: www.regulations.gov.

USEPA. 2006. Reregistration eligibility decision for imazapyr. EPA 738-R-06-007/OPP-2005-0495. Available at http://www.epa.gov/oppsrrd1/REDs/imazapyr_red.pdf, last accessed 6/27/2014.

USEPA, 2007. Risks of imazapyr sse to the Federally listed California red-legged frog (Rana aurora draytonii), Effects Determination for Imazapyr and the California Red-legged Frog. Includes Appendices A-K and Attachments 1 and 2. Documents dated July 20, 2007. Available at: http://www.epa.gov/espp/litstatus/effects/redleg-frog/index.html?PHPSESSID=9e2e2b07da820363425c65a13695be5a#imazapyr. Last accessed 2/14/2015.

USEPA. 2009. Risks of triclopyr use to Federally threatened California red-legged frog (*Rana aurora draytonii*). Including Appendices A through Q. Available at: http://www.epa.gov/espp/litstatus/effects/redleg-frog/. Last accessed 6/27/2014

USPEA. 2011. Exposure factors handbook: 2011 Edition, National Center for Environmental Assessment, Office of Research and Development, U.S. EPA, Washington, DC. EPA/600/R-090/052F, September 2011. Available at: http://www.epa.gov/ncea/efh/pdfs/efh-complete.pdf. Last accessed 2/14/2015.

USEPA. 2013. Chemicals evaluated for carcinogenic potential. Office of Pesticide Programs. http://npic.orst.edu/chemicals_evaluated.pdf, last accessed 6/30/2014.

USEPA. 2014a. Overview of Risk Assessment in the Pesticide Program – Human Health Risk Assessment, Office of Pesticide Programs (http://www.epa.gov/pesticides/about/overview_risk_assess.htm) last accessed 3/21/2014.

USEPA. 2014b. Endocrine Disruptor Screening Program, Office of Chemical Safety and Pollution Prevention, Office of Science Coordination and Policy (http://www.epa.gov/endo/), last accessed 3/27/14.

USEPA. 2014c. Ecotoxicity Categories for Terrestial and Aquatic Organisms. Technical Overview of Ecological Risk Assessment, Analysis Phase: Ecological Effects Characterization, Pesticides: Environmental Effects, http://www.epa.gov/oppefed1/ecorisk_ders/toera_analysis_eco.htm, last accessed 1/27/15.

Welch, J.R., Miller, K.V., Palmer, W.E. and T.B. Harrington. 2004. Response of understory vegetation important to the northern bobwhite following imazapyr and mechanical treatments. Wildlife Soc Bull. 32: 1071-1076.

Attachment A: Risk Estimations for Workers and General Public

USDA Forest Service Program Worksheets for triclopyr TEA and triclopyr BEE formulations for terrestrial applications (worksheet version: 5.00.64) and TCP from terrestrial applications of triclopyr (version 6.00.01) (USDA FS, 2011c,d,e) were used for the risk estimations. Attachment A summarizes the exposure concentrations, exposure assumptions and toxicity values used in the hazard quotient estimates to workers and the general public.

Attachment A-1 Concentrations in field solutions and exposure rates.

Parameter/Assumption	Range	Value	Units	Reference
Application rate		9 / 8	Lb/acre	The maximum rate per label (9 lb/acre for TEA and TCP per Garlon 3A label & 8lb/acre per Pathfinder II label)
Application Volume	Central	25	Gal/acre	USDA FS, 2011a
	Upper	40		
	Lower	5		
Concentration in field solution	Central	*0.36/0.32*	Lb/gal	Calculated
	Upper	*0.225/0.2*		
	Lower	*1.8/1.6*		
Conversion factor for lbs/gal to mg/ml		119.8	mg/mL per lb/gal	
Concentration in field solution	Central	*4.30E+01/3.80E+01*	mg/mL	Calculated
	Lower	*2.70E+01/2.40E+01*		
	Upper	*2.20E+02/1.90E+02*		
Number of Applications		1	Unitless	USDA FS 2011a
Application Interval		1	Days	USDA FS 2011a
Worker Exposure Rate	Central	0.01	mg/kg bw per lb	USDA FS 2012
	Lower	0.0001		
	Upper	0.08		
Number of acres that the worker will treat per hour	Central	0.625	Acres/hr	
	Lower	0.25		USDA FS 2012
	Upper	1		
Number of hours of worker exposure per day.	Central	7		
	Lower	6		USDA FS 2012
	Upper	8		

Attachment A-2 Exposure assumptions for herbicide risk estimations

Receptors	Factor	Value	Units	Reference
Backpack Worker	Absorbed dose rate			USDA FS 2012
	Central	0.01	(mg/kg bw)/(lbs	
	Upper	0.0001	handled per day)	
	Lower	0.08		
	Hours of application per day			
	Central	7	Hours per day	USDA FS 1989
	Upper	6		
	Lower	8		
	Acres treated per hour			
	Central	0.625	acres/hour	USDA FS 1989
	Lower	0.25		
	Upper	1		
Adult Male Worker	Body weight	80	kg	USEPA 2011
	Surface Area, Hands	1070	cm^2	USEPA 2011
	Surface Area, Lower legs	2273	cm^2	USEPA 2011
	Exposure duration	1/10/60	minutes	Contaminated gloves and spill, professional judgment
Adult Female	Body weight	71	kg	USEPA 2011
	Surface Area, Wearing shorts and T-shirt	6473	cm^2	USEPA 2011
	Contact time/Exposure Duration	10 /1	minutes/hour	Professional Judgment
Pre-teen and teenaged Child (11-16)	Body weight	56.8	kg	USEPA 2011
	Surface Area, Wearing shorts and T-shirt	5650	cm^2	USEPA 2011
	Contact time/Exposure Duration	10 / 2	minutes/hour	Professional Judgment

Attachment A-3 Conversion factors for herbicide risk estimations.

Receptors	Factor	Value	Units	Reference
Conversion Factors	acrefoot^{2}gal	325900	gal/acre-foot	Budavari 1989
	liters per gal	3.785	L/gal	Budavari 1989
	pound per milligram	453600	mg/lb	Budavari 1989
	lbac^{2}mgcm^2	0.01121	mg/cm^2 per lb/acre	Budavari 1989
	lbac^{2}ugcm^2	11.21	ug/cm2 per lb/acre	Budavari 1989
	conversion for milligrams to pounds	2.204E-06	lb/mg	Budavari 1989
	conversion for milliliters to gallons	3785	ml/gal	Budavari 1989
Misc	Liquid adhering to skin	0.008	mL/cm^2	Mason and Johnson 1987
	Dislodgeable residue as proportion	0.1	none	Harris and Solomon 1992

Attachment A-4 Summary of toxicity values

Compound	Duration	Toxicity Value (RfD – mg/kg/day)	Reference
Triclopyr TEA	Acute	1	USDA FS, 2011a, USEPA 2002a
	Chronic	0.05	
Triclopyr BEE	Acute	1	
	Chronic	0.05	
TCP	Acute	0.025	
	Chronic	0.012	

Appendix F. Imidacloprid Human Health and Ecological Risk Assessment: Asian Longhorned Beetle Eradication Program

Table of Contents

EXECUTIVE SUMMARY

The United States Department of Agriculture (USDA), Animal and Plant Health Inspection Service (APHIS) proposes to use the insecticide imidacloprid as part of an eradication program for the Asian Longhorned Beetle (ALB).

USDA APHIS evaluated the potential human health and ecological risks from the proposed use of imidacloprid for the ALB eradication program. Risks to human health are expected to be negligible based on limited exposure from the proposed use pattern of imidacloprid (trunk and soil injection). Exposure is greatest for workers who will apply the product, but the formulation and the required protective equipment result in a low potential for exposure and risk.

Ecological risks for terrestrial and aquatic non-target organisms is also expected to be low based on the toxicity and environmental fate of imidacloprid, and method of application. Risks to terrestrial invertebrates, including pollinators, is expected to be negligible based on available laboratory data and field collected data for pollinators that has been collected for ALB specific applications of imidacloprid. There is some risk to sensitive terrestrial invertebrates that consume vegetation from treated trees or inhabit the soil where soil injection is used.

1.0 INTRODUCTION

This human health risk assessment (HHRA) and ecological risk assessment (ERA) provide a quantitative and qualitative evaluation of the potential risks and hazards to human health, nontarget fish, and wildlife as a result of exposure to the insecticide, imidacloprid, when used for controlling the ALB.

The methods used in this HHRA to assess potential human health effects follow standard regulatory guidance and methodologies (NRC, 1983; USEPA, 2014a), and generally conform to other Federal agencies such as U.S. Environmental Protection Agency (USEPA), Office of Pesticide Programs (USEPA/OPP). The methods used in this ERA to assess potential ecological risk to nontarget fish and wildlife follow USEPA methodologies regarding eco-risk assessment, with an emphasis on those used by USEPA/OPP in the pesticide registration process.

The risk assessment starts with problem formulation (identifying hazard), then the toxicity assessment (the dose-response assessment), and then the exposure assessment (identifying potentially exposed populations and determining potential exposure pathways for these populations). Lastly, the information from the exposure and toxicity assessments is combined to characterize risk (determining whether there is adverse health and eco-risk).

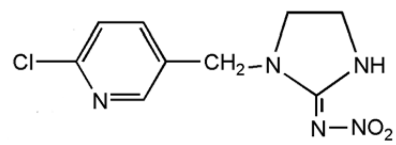
2.0 PROBLEM FORMULATION

APHIS is proposing the use of the insecticide imidacloprid to treat high risk host trees that are not cut to prevent further spread of ALB as part of the ALB eradication. The following sections discuss the chemical description and product use; physical and chemical properties; environmental fate; and hazard identification for imidacloprid.

2.1 Chemical Description and Product Use

Imidacloprid (C_9-H_{10}-Cl-N_5-O_2) (CAS No. 138261-41-3) belongs to a class of insecticides, neonicotinoids, that act by binding directly to the acetylcholine binding receptor. The molecular structure is shown in Figure 1. Imidacloprid is manufactured in several different formulations that can be used for soil, seed and foliar applications. It is registered for use on a wide variety of agricultural commodities as well as in horticultural and turf applications and for animal health. Imidacloprid controls a variety of insects including sucking insects such as psyllids, aphids, thrips, whiteflies, rice hoppers, turf and soil insects, and some beetles. In the ALB eradication program, an imidacloprid formulation such as Merit® 2F will be applied through trunk or soil injection at the base of the tree. After application, imidacloprid is taken up and distributed throughout the tree (USDA APHIS, 2013). Merit® 2F (21.4% imidacloprid as an active ingredient and 78.6% others as inerts) contains 2 pounds of imidacloprid per gallon (Bayer 2004).

Figure 1 - Molecular Structure (a chloronicotinyl nitroguanidine insecticide)



2.2 Physical and Chemical Properties

Imidacloprid is a colorless crystal with a slight odor. Its melting point is 144°C. It is not considered volatile due to a low vapor pressure (3×10^{-12} mmHg at 20°C). The partition coefficient (log Kow) is 0.57 at 21 °C suggesting it would not bioaccumulate in lipids. The solubility in water is 0.61 grams per liter (g/L) at 20°C. The estimated Henry's constant value is 2×10^{-15} atm-m^3/mole at 20°C.

2.3 Environmental Fate

The environmental fate describes the processes by which imidacloprid moves and transforms in the environment. The environmental fate processes include: 1) mobility, persistence, and degradation in soil, 2) movement to air, 3) migration potential to groundwater and surface water, and 4) plant uptake.

In soil, imidacloprid soil organic carbon-water partitioning coefficient (Koc)[1] values range from 157 to 810 in various soil types and organic carbon levels suggesting that in most cases imidacloprid does not adsorb strongly to soil particles (USDA FS, 2005; HSDB 2014). The adsorption capacity of imidacloprid to soil is correlated with the amount of soil organic matter (Liu et al., 2006). Imidacloprid is less mobile in organic carbon and clay soils and more mobile in sand and gravel soils (HSDB, 2014). The leaching potential also depends on the soil type and organic content level in soil. Several studies have shown that imidacloprid does not leach to

[1] KOC values are useful in predicting the mobility of organic soil contaminants. Higher KOC values correlate to less mobile organic chemicals while lower KOC values correlate to more mobile organic chemicals.

ground water in silty loam and silt soils with high organic matter content.(Miles, Inc., 1992; Rouchaud et al., 1994). A leaching study in fine sandy loam soil demonstrated the potential for imidacloprid residues to move downward through the soil with percolating water (Felsot et al., 1998).

Soil photolysis half-lives range from 19 to 39 days (CA DPR, 2006a; Graebing and Chib, 2004). Aerobic soil metabolism half-lives show imidacloprid to be persistent with half-lives ranging from 83 days to greater than one year (CCME, 2007). Imidacloprid degrades more rapidly in soil under aerobic conditions with vegetation (half-lives of 48 days) compared to soil without vegetation (half-lives of 190 days) (HSDB, 2014). Imidacloprid degrades under anaerobic aquatic conditions more rapidly than under aerobic soil conditions (HSDB, 2014). At the soil surface, imidacloprid has a photolysis half-life of 39 days. When incorporated into soil, the photolysis dissipation half-life of imidacloprid varies from 27 to 229 days (CA DPR, 2006a). The half-lives tend to increase with increasing pH values but decrease in the presence of light (Sarkar et al., 2001; CA DPR, 2006a).. Soil half-life values in the absence of light are as high as 229 days in field studies and 997 days under laboratory conditions (CA DPR, 2006a). Several field dissipation studies have been conducted with imidacloprid under various crop, soil, and weather conditions with a resulting half-life range of 7 to 107 days (USDA FS, 2005). Imidacloprid degrades under anaerobic aquatic conditions more rapidly than under aerobic soil conditions (HSDB, 2014).

In air, imidacloprid is expected to exist solely in the particulate phase and not expected to volatilize into the ambient atmosphere based on the low reported volatility (3×10^{-12} mm Hg) and Henry's Law Constant[2] (2×10^{-15} atm m^3/mole) (HSDB, 2014).

Imidacloprid is considered soluble in water with values ranging from 510 to 610 mg/L (USDA FS, 2005). In aquatic environments imidacloprid is resistant to hydrolysis at all relevant pH values but is susceptible to aquaeous photolysis with half-lives of less than 5 hours (USDA FS, 2005).

In plants, imidacloprid applied through trunk or soil injection to host trees will translocate throughout the tree. These applicaton methods were used previously in the ALB eradication programs in New York and Illinois. During the environmental monitoring, paired leaf and twig samples from eight different species of host trees in New York were collected periodically for 15 months to test for the presence of imidacloprid residues. Tree species included Norway, sycamore, sugar and silver maple, poplar, elm, hackberry and mountain ash. Plant tissue sampling of trees treated with imidacloprid using the same application methods indicated that both leaves and twigs have detectable residues of imidacloprid concentrations (USDA APHIS, 2002). The detected concentrations in leaves were higher than the detected concentrations in twigs. Imidacloprid appears to be distributed predominately in areas that are rapidly growing (i.e., leaves) based on the sampling data. The sampling data also shows that uptake and distribution varied by tree species. Blossom sampling results (USDA APHIS, 2001; 2002) show that imidacloprid residues were either not detected or detected at low levels not quantifiable

[2] Henry's law constant is the ratio of a chemical's concentration in the air to its concentration in water at equilibrium.

(0.03 to 0.099 ppm) except for one blossom sample that contained 0.13 ppm (USDA APHIS, 2002).

The half-life of imidacloprid on vegetation from field dissipation studies has been shown to range from 1.17 days in potatoes to 9.8 days on turf. Because applications in this program are not foliar, this route of degradation is not expected to be significant since imidacloprid will be translocated by plant roots into the tree or distributed throughout the tree from trunk injection sites by the tree's vascular system. The main breakdown products of imidacloprid in plants includes a monohydroxy metabolite, imidacloprid guanidine, imidacloprid olefin, and a monoglucoside of 6-chloropicolyl alcohol (CA DPR, 2006a).

2.4 Hazard Identification

Imidacloprid, a neonicotinoid insecticide, results in toxicity through binding or partial binding to specific areas of the nicotinic acetylcholine receptor (nAChR). Acetylcholine is an important neurotransmitter in most animals. It is released at the nerve synapse in response to a membrane depolarization that is the initiator of nerve transmission. Acting as an inhibitor at nicotinic acetylcholine receptors, imidacloprid disrupts the nervous system. Imidacloprid is more toxic to insects than to mammals because it has higher binding strength to the nicotinic receptors of insect nerve cells than to mammalian receptors (Tomizawa and Casida, 2003; Gervais et al., 2010).

Imidacloprid has moderate acute oral toxicity and low acute dermal and inhalation toxicity, but is not an eye or dermal irritant, or dermal sensitizer. The primary target of imidacloprid is the nervous system. Imidacloprid has been shown to have thyroid and/or liver effects in chronic dietary studies in rats and dogs. These reported impacts occurred at levels above those anticipated from imidacloprid use in the ALB eradication program. Reports of imidacloprid deliberate self-poisoning or accidental exposure in humans have shown low lethality (Mohamed et al., 2009; Gervais et al., 2010; Lin, et al., 2013; Kumar et al., 2013). Signs of toxicity include drowsiness, dizziness, vomiting, disorientation, and fever (Gervais et al., 2010). Other more serious health effects from exposure to imidacloprid include compromised respiratory, cardiovascular, and neurological function such as dyspnoea/apnoea, coma, tachycardia, hypotension, mydriasis, and bradycardia (Lin et al., 2013). Kumar et al. (2013) reported signs of acute inhalation intoxication from an accidental imidacloprid exposure that included severe gastrointestinal symptoms, respiratory distress, and neuropsychiatric features. Signs of toxicity from accidental inhalation of a pesticide containing 17.8% imidacloprid included becoming disoriented, agitated, incoherent, with sweating and breathlessness (Gervais et al., 2010). Two fatal incidents of deliberate ingestion were associated with suicides (Gervais et al., 2010; Iyyadurai et al., 2010).

2.4.1 Acute Toxicity

Multiple acute toxicity studies have been conducted using imidacloprid on several mammalian species. These studies along with other available data have been summarized in several reports (CA DPR, 2006b; USDA FS, 2005; USEPA, 2005; USEPA, 2010). Acute oral median lethal toxicity values range from 131 mg/kg for the mouse to 475 mg/kg for the rat, suggesting moderate acute toxicity to mammals. Inhalation and dermal toxicity is considered low for imidacloprid with LC_{50} and LD_{50} values greater than 5.33 mg/L and 2,000 mg/kg, respectively. Acute oral and inhalation sublethal effects have been measured in the rat and mouse with oral no observable effect levels (NOEL) ranging from 10 to 50 mg/kg/day and inhalation NOELs ranging from 3.4 to 192 mg/kg/day (CA DPR, 2006b). Sublethal impacts noted in these studies were apathy, labored breathing, trembling and staggering gait. An acute neurotoxicity study in rats reported a lowest observable adverse effect level (LOAEL) of 42 mg/kg (USEPA, 2010). The California Department of Pesticide Regulations (CA DPR) has used this value to set the acute benchmark dose ($BMD_{0.5}$) of 9 mg/kg/day while USEPA determined that the reported NOEL was a LOEL and calculated an acute Population-Adjusted Dose (aPAD)/Reference Dose (RfD) of 0.14 mg/kg/day based on the application of uncertainty factors (CA DPR, 2006b; USEPA, 2010). An uncertainty factor of 300 (10X for interspecies extrapolation, 10X for intraspecies variation, and 3X for the use of a LOAEL due to the lack of a NOAEL) was applied to the aPAD (USEPA, 2010).

The acute oral LD_{50} values obtained from rat studies for one of the available imidacloprid formulations, Merit® 2F, demonstrates lower acute toxicity when compared to the technical active ingredient (table 2-1). The acute inhalation toxicities are very low (LC_{50} > 2.0 mg/L) for both the technical imidacloprid and the Merit® 2F formulation.

Table 2-1. Comparative acute mammalian toxicity between the technical active ingredient and the formulated imidacloprid .

Toxicity Study	Technical Imidacloprid	Merit® 2F*
Acute Oral LD_{50} (rat)	424 mg/kg (M)/450 mg/kg (F)	>4,870 mg/kg (M)/4,143 mg/kg (F)
Acute Dermal LD_{50}	>5,000 mg/kg (rat)	> 2,000 mg/kg (rabbit)
Acute Inhalation LC_{50} (rat)	>5.3 mg/L	>5.33 mg/L
Dermal Irritation (rabbit)	No irritation	No irritation
Ocular (rabbit)	No irritation	Mild Irritation (Minimally irritating)
Sensitization (Guinea pig)	Non-sensitizing	Non-sensitizing

M= male; F=Female

* Source: USDA FS, 2005; Bayer, 2008

2.4.2 Sub-Chronic/Chronic Toxicity

Chronic toxicity is a condition caused by repeated or long-term exposure to low doses of a toxic substance. Several sub-chronic studies have been submitted to support the registration of imidacloprid. Studies range in duration from 21 to 107 days testing under oral, dermal, and inhalation exposures (CA DPR, 2006b; USDA FS, 2005) (table 2-2).

Table 2-2. Subchronic mammalian effects studies using imidacloprid.

Test Animal	Exposure/Duration	NOEL (mg/kg/day)	LOEL (mg/kg/day)
Rat	Oral/98 days	14.0	61.0
Rat	Inhalation/28 days	0.9	5.2
Rat	Oral Neurtoxicity/91 days	9.3*	NR
Mouse	Oral/107 days	86.0	427.0
Dog	Oral/28 days	7.3	31.0
Dog	Oral/91 days	8.0	24.0
Rabbit	Dermal/21 days	>1,000	>1,000

*Value represents the lowest reported NOEL from the study; NR = Not Reported

Chronic studies have been conducted using the rat, dog and mouse in oral exposures ranging from one to two years. The rat was the most sensitive test species with a NOEL and LOEL of 5.7 and 17 mg/kg/day, respectively. Comparative values for the dog and mouse were higher with NOELs of 15 and 47, respectively, and LOELs of 41 and 143 mg/kg/day (CA DPR, 2006b). The NOEL reported in the two year rat study was used by USEPA/OPP to set the chronic reference dose (RfD) of 0.057 mg/kd/day (USEPA, 2010). RfD is "an estimate of daily oral exposure to the human population that is likely to be without an appreciable risk of deleterious effects during a lifetime" (USEPA Risk Assessment Glossary, http://www.epa.gov/risk_assessment/glossary.htm).

2.4.3 Carcinogenicity/Mutagenicity

There is no evidence of carcinogenic potential in the rat and mouse carcinogenicity studies, and there is no concern for mutagenicity. Based on studies with rats and mice, the USEPA has classified imidacloprid into Group E, no evidence of carcinogenicity (USEPA, 2005; 2013). A study of human lymphocytes exposed to greater than 5,200 μg/mL of imidacloprid demonstrated a slight increase in chromosome abnormalities *in vitro*, but this result was not found with *in vivo* tests (Gervais et al., 2010). Imidacloprid is not considered mutagenic or genotoxic based on the weight of evidence from several in vitro and in vivo studies. The results of an in vitro study on human peripheral blood lymphocytes indicated that imidacloprid at concentrations less than 20 microMolar is not genotoxic to human lymphocytes (Costa et al., 2009).

2.4.4 Development and Reproductive Effects

A two generation reproductive study using the rat resulted in a NOEL of 13 mg/kg/day for adult rats based on a decrease in premating body weights. The offspring NOEL was also 13 mg/kg/day based on decreased pup body weight in both litters (CA DPR, 2006b). Developmental studies using the rat reported a maternal and developmental NOEL of 13 mg/kg/day, and in a rabbit study, the maternal and developmental NOEL was reported as 24 mg/kg/day. A chronic fertility study showed that exposure to imidacloprid at doses of 225 and 112 mg/kg for 60 days can cause histological damage to testicular tissue, sperm mortality, and decreased testosterone levels in mature male rats (Najafi et al., 2010).

2.4.5 Endocrine Effects

A literature search did not identify any study indicating the potential of imidacloprid to affect the endocrine system. Imidacloprid is among the group of 58 pesticide active ingredients on the initial list to be screened under the USEPA Endocrine Disruptor Screening Program. However, the list of chemicals was generated based on exposure potential, not based on whether the pesticide is a known or likely potential endocrine disruptor (USEPA, 2014b). Imidacloprid is not among the European Union (EU) list of chemicals with the potential to impact the endocrine system. The EU list includes three categories: Category 1 – endocrinal effect recorded at least on one type of animal; Category 2 – a record of biological activity in vitro leading to disruption; and Category 3 – not enough evidence or no evidence data to confirm/disconfirm endocrinal effect of tested chemicals (Hrouzková and Matisova, 2012).

2.4.6 Immune System Effects

A literature review of earlier studies indicates that imidacloprid does not have a direct effect on the immune system in mammals (USDA FS, 2005). Recent studies suggest that exposure to imidacloprid may induce immunotoxicity (Mohany et al., 2011; Gawade et al., 2013; Badgujar et al., 2013). A study in male albino rats indicated that exposure to 1/100 of the LD_{50} for imidacloprid can induce immunotoxicity, oxidative stress, lipid peroxidation, and hepatotoxicity (Mohany et al., 2011). A developmental immunotoxicity study in Wistar rats (Gawade et al., 2013) indicated that continuous exposure to imidacloprid during development adversely affected the immune system. An oral exposure study in BALB/c mice over 28 days showed that imidacloprid has immunosuppressive effects at doses >5 mg/kg (Badgujar et al., 2013).

2.4.7 Metabolism

The imidacloprid molecule (composed of a pyridinyl moiety and an imidazolidine ring) is metabolized through two main routes of metabosim. The first route is oxidative cleavage that frees the pyridinyl moiety to yield 6-chloronicotinic acid. Then, 6-chloronicotinic acid is mostly conjugated with glycine to form a hippuric acid-type conjugate. Minor quantities of 6-chloronicotinic acid are dechlorinated to form methylmercaptonicotinic acid and derivatives. The second route is hydroxylation of the imidazolidine ring to form 4- or 5-hydroxyl imidacloprid (HSDB, 2014). In mammalian studies imidacloprid has been shown to be rapidly excreted in feces and urine. In an oral dosing rat study using radiolabelled imidacloprid approximately 90 percent of the radioactivity was excreted through the urine and feces within 24 hours, with 96 percent excreted after 48 hours (USDA FS, 2005).

3.0 DOSE-RESPONSE ASSESSMENT

3.1 Human Health Dose-Response Assessment

A dose-response assessment evaluates the dose levels (toxicity criteria) for potential human health effects including acute and chronic toxicities.

USEPA derived an acute reference dose (RfD) of 0.14 mg/kg for imidacloprid based on the LOAEL of 42 mg/kg body weight divided by an uncertainty factor of 300 (10 for interspecies extrapolation, 10 for intraspecies sensitivity, 3 for using a LOAEL to approximate a NOAEL) (USEPA, 2010).

USEPA also derived a chronic RfD of 0.057 mg/kg/day based on chronic toxicity and carcinogenicity studies using the rat. The NOAEL was estimated to be 5.7 mg/kg/day and the LOAEL was set at 16.9 mg/kg/day based on increased occurrence of mineralized particles in the thyroid gland of male rats (USEPA, 2010).

USEPA has not established a maximum contaminant level for imidacloprid in water (USEPA, 2014c). USEPA developed a chronic drinking water level of comparison (DWLOC) (1,755 ppb) for imidacloprid for an aggregate dietary exposure analysis during registration. The DWLOC determines the theoretical upper limits for a pesticide's concentration in drinking water (USEPA, 2005)

For the ALB eradication environmental monitoring program in Suffolk County, New York, the New York State "Imidacloprid Groundwater Monitoring Project Plan" established an "action threshold" of 25 ppb (half of the New York State drinking water standard) for imidacloprid (USDA APHIS, 2007).

3.2 Ecological Effects Analysis

This section of the risk assessment discusses available ecological effects data for terrestrial and aquatic biota. Available acute and chronic toxicity data are summarized for all major taxa and will be integrated with the exposure analysis section to characterize the risk of imidacloprid to nontarget wildlife and domestic animals. Information in this section was gathered from on-line databases and searches for relevant peer reviewed and non-peer reviewed literature. In cases where multiple toxicity values were located for the same test species, the lowest value was generally used in the effects analysis. This was particularly the case for aquatic invertebrate toxicity data where multiple acute lethality values were available for the same test species. Other values that were not selected were within the range of effects data that was reviewed during the literature search.

3.2.1 Aquatic Effects Analysis

3.2.1.1 Fish and Amphibians

Imidacloprid acute toxicity to fish and amphibians is low based on the available acute median lethal concentrations (USEPA, 2008; USDA FS, 2005; Feng et al., 2004; Jemec et al., 2007) (table 3-1). Values typically exceed 100 mg/L suggesting that imidacloprid is practically non-toxic to fish and amphibians. Sublethal toxicity based on available no observable effect concentrations (NOEC) data ranges from 25 to 58 mg/L for fish with effects such as erratic swimming behavior, discoloration, quiescence and labored respiration noted at higher concentrations (USDA FS, 2005).

Table 3-1. Acute aquatic vertebrate toxicity data for imidacloprid.

Test Organism	Endpoint/Length	Toxicity Value (mg/L)	NOEC (mg/L)
Bluegill	96-hour LC_{50}	>105	25
Rainbow Trout	96-hour LC_{50}	229.1	52.1
Zebrafish	96-hour LC_{50}	241.0	NR
Sheepshead Minnow	96-hour LC_{50}	163	58.2
Rana limnocharis	48-hour LC_{50}	165	30
Rana hallowell	48-hour LC_{50}	219	101.2
Hypsiboas pulchellus	96-hour LC_{50}	84.91	NR

NR = not reported

Limited chronic fish toxicity data is available; however, in a 98-day flow-through exposure the rainbow trout LOEC and NOEC were 2.3 and 1.2 mg/L, respectively, based on a statistically significant reduction in length (USDA FS, 2005). Sanchez-Bayou and Goka (2005) dosed rice paddies at a rate 1.5 times the proposed rate and measured several sublethal responses in Japanese medaka over an approximate three month period. No statistical analysis was completed; although, the authors noted a lack of malformations in imidacloprid treated fields, and did note an increase in parasitism rates by the ectoparasite, *Trichodenia*, when compared to controls. Measured concentrations of imidacloprid ranged from 239.2 µg/L immediately after dosing to 1.1 µg/L 118 days post treatment. Ruiz de Arcautea et al. (2014) reported various genotoxic effects to tadpoles of the Montevideo tree frog, *Hypsiboas pulchellus*, at concentrations ranging from 15 to 45 mg/L in 48 and 96 hour exposures.

3.1.1.2 Aquatic Invertebrates

Aquatic invertebrates are more sensitive to imidacloprid when compared to fish or amphibans with acute median toxicity values in the high part per trillion range to greater than 100 mg/L depending on the test species (Song et al., 1997; Song and Brown, 1998; USDA APHIS, 2002; USEPA, 2008; Overmyer et al., 2005; USDA FS, 2005; Paul et al., 2006; Key et al., 2007; Alexander et al., 2007; Stoughton et al., 2008; Roessink et al. 2013) (table 3-2).

Table 3-2. Representative acute aquatic invertebrate toxicity data for imidacloprid.

Test Organism	Endpoint/Length	Toxicity Value (µg/L)	NOEC(µg/L)
Epeorus longimanus	96-hour LC_{50}	0.65	NR
Caenis horaria	96-hour EC_{50}	1.77	NR
Cloeon dipterum	96-hour EC_{50}	1.02	NR
Daphnia magna	48-hour EC_{50}	85200	42000
Daphnia magna	48-hour EC_{50}	10440	NR
Chironomus tentans	96-hour LC_{50}	5.75	1.03
C. tentans	10-day LC_{50}	3.17	0.67
Hyallela azteca	96-hour LC_{50}	65.43	54.24

H. azteca	96-hour LC_{50}	526.0	0.35
Simulium vittatum	48-hour LC_{50}	6.75 – 9.54	NR
Lumbriculus variegatus	96-hour EC_{50}	6.2	NR
Aedes aegypti (4th instar)	72-hour LC_{50}	84.0	NR
Aedes aegypti (adult)	72-hour LC_{50}	>6300	NR
A. taeniorhynchus	48-hour LC_{50}	13.0	NR
Mysidopsis bahia	96-hour LC_{50}	38.0	32.0
Palaemonetes pugio	96-hour LC_{50}	308.8	100
Artemia sp.	48-hour LC_{50}	361230	NR
Crassostrea virginica	96-hour EC_{50}	>100000	145000

NR = not reported

Sublethal effects have also been observed for various aquatic invertebrates under acute and chronic exposures. Agatz et al. (2014) reported feeding inhibition for *Gammarus pulex* in four day exposures to imidacloprid residues as low as 30 µg/L. Agatz et al. (2013) reported similar effects to the cladoceran, *D. magna*, where 50% reductions in feeding occurred at an exposure concentration of 1.83 mg/L. Reduced feeding resulted in impacts to survival, growth, and reproduction; however, the presence of surplus food after exposure allowed for recovery. In cases of limited food availability, recovery in growth was not observed. Alexander et al. (2007) reported feeding inhibition for the mayfly, *Epeorus longimanus*, and the aquatic oligachaete, *Lumbriculus variegatus*, in pulsed 24-hour doses using imidacloprid at concentrations above 0.5 µg/L. Survivorship effects were noted at concentrations between 1 and 10 µg/L.

Chronic toxicity to aquatic invertebrates has been assessed in both marine and freshwater invertebrate species. In two flow-through studies using *M. bahia*, the maximum allowable toxicant concentration (MATC) values for reproductive success ranged from greater than 643 ng/L to 849 ng/L while the MATC for growth ranged from 230 to 3806 ng/L (USDA FS, 2005). Stoughton et al. (2008) conducted 28-day continuous and pulse dose studies using the midge, *C. tentans*, and the freshwater amphipod, *H. azteca*. The *C. tentans* NOECs for survival and weight were both 1.14 µg/L under continuous exposure and 3.47 µg/L under a pulse dose. *H. azteca* was comparatively less sensitive with a NOEC for survival and weight of 3.44 and 11.46 µg/L, respectively, under continuous exposure, and a NOEC of 3.53 and 11.93 µg/L for the same endpoints under a pulse dose exposure. Chronic toxicity testing using the less sensitive *D. magna* resulted in NOEC and LOEC values of 1.8 and 3.6 mg/L, respectively (USEPA, 2008). Pestana et al. (2009) reported reduced leaf litter decomposition related to decreased feeding activity by stoneflies after pulsed dose exposures in mesocosms receiving three doses of imidacloprid at 17.60 µg/L. No effects were observed at the next lowest concentration, 1.63 µg/L. Nyman et al. (2013) reported reduced feeding activity and lipid content in 14- and 21-day exposures to the amphipod *Gammarus pulex* exposed to imidacloprid residues as low as 15 µg/L.

Sublethal impacts to some aquatic invertebrates that feed on leaf litter containing imidacloprid have been observed, as well as effects on decomposition rates

(Kreutzweiser et al., 2007; Kreutzweiser et al., 2008; Kreutzweiser et al., 2009). Mortality to leaf-shredding insects occurred at higher rates that were intentionally overdosed; however, significant mortality did not occur to shredding insects such as *Pternarcys dorsata* and *Tipula* sp. at typical field applications. Feeding inhibition was observed at imidacloprid leaf concentrations of 18-30 µg/g which have been observed in the field under normal applications (Kreutzweiser et al., 2009).

In a review of a microcosm study that was submitted to support registration of imidacloprid negative impacts to some blue green algae, copepods, and several insect groups were observed at measured average doses ranging from 19 to 180 µg/L (USDA FS, 2005). The NOEC based on total number of invertebrates and species richness was determined to be 2 µg/L from the study where tanks were dosed biweekly four times.

3.1.1.3 Aquatic Plants

Aquatic plant studies testing technical and formulated material on blue green algae, freshwater diatoms, and green algae have demonstrated low toxicity (USDA FS, 2005; USEPA, 2008) (table 3-3).

Table 3-3. Acute aquatic plant toxicity data for imidacloprid.

Test Organism	Endpoint/Length	Toxicity Value (mg/L)	NOEC (mg/L)
*Anabaena flosaquae**	120-hour EC_{50}	32.8	24.9
*Navicula pelliculosa**	120-hour EC_{50}	NR	6.69
Scenedesmus subspicatus	96-hour EC_{50}	>10	10
Selanastrum capricornutum	120-hour EC_{50}	>119	>119

*Formulated material; NR = Not reported

3.2.2 Terrestrial Effects Analysis

3.2.2.1 Mammals and Birds

Imidacloprid has moderate acute oral toxicity to wild mammals based on the available toxicity data used to evaluate human health effects. Imidacloprid is considered toxic to birds with acute oral median toxicity values ranging from 41 to 152 mg/kg (USDA APHIS, 2002a; USEPA, 2008; USDA FS, 2005) (table 3-4). Reproduction studies using the mallard and bobwhite quail have shown NOECs of approximately 125 ppm for both species.

Table 3-4. Acute avian oral and dietary toxicity data for imidacloprid.

Test Species	Study	LD_{50}/LC_{50}	NOEL/LOEL
House Sparrow	Acute Oral	41.0	3.0/NR
Pigeon	Acute Oral	NR	NR/12.5
Canary	Acute Oral	NR	10.0/NR
Ringed Turtle Dove	Subacute Dietary	NR	NR/228.0
Bobwhite Quail	Acute Oral	152.3	25.0

	Subacute Dietary	1535.87	69.0
	Chronic Reproductive	NA	126.0
Mallard	Subacute Dietary	4797.0	69.0
	Chronic Reproductive	NA	125.0

NR = not reported; NA = not available

3.2.2.2 Reptiles and Amphibians (terrestrial phase)

No acute or chronic toxicity data appears to be available for reptiles or terrestrial phase amphibians based on a review of the literature and databases containing toxicity data for imidacloprid. Available data for the aquatic phase of amphibians demonstrates low toxicity and is comparable to the data for surrogate fish species. In cases where effects data are lacking for reptiles USEPA assumes that the sensitivity is comparable to birds which would suggest that imidacloprid is toxic to reptiles. There is uncertainty in that assumption due to physiological and life history differences between birds and reptiles.

3.2.2.3 Terrestrial Invertebrates and Soil Microorganisms

A substantial amount of imidacloprid toxicity data has been collected for a variety of insects and other arthropods with results varying based on the terrestrial invertebrate species tested, the method of application, and other environmental factors (USDA FS, 2005; Pisa et al., 2015). Most of this data has been collected to assess potential impacts to parasitoids and predators to determine compatability of imidacloprid in integrated pest management. In general, imidacloprid offers reduced effects to these types of invertebrates compared to broad spectrum insecticides. Impacts to susceptible insects that feed on treated trees are expected, but due to the method of application and the treatment of specific ALB host trees, the effects are expected to be localized and not widespread.

Due to concerns with pollinators such as honey bees several toxicity studies have been conducted assessing acute and chronic effects of imidacloprid to these species. Technical and formulated imidacloprid is considered acutely toxic to honey bees and other related bee species by oral and contact exposure. Median lethal toxicity values range from 3.7 to 230 ng/bee (Schmuck et al., 2001; Tasei, 2002; USDA FS, 2005; USEPA, 2008). Acute sublethal effects in laboratory studies have shown that the NOEC may be less than 1 ng/bee (USDA FS, 2005). Imidacloprid metabolite toxicity to honey bees is variable with some of the metabolites having equal toxicity to imidacloprid while other metabolites are considered practically non-toxic (USDA FS, 2005).

Due to concerns regarding the potential sublethal impact of imidacloprid to honey bees, several studies have been conducted to determine potential effects in laboratory and field situations. Studies to assess the effects of imidacloprid on homing behavior, colony development, foraging activity, reproduction, wax/comb production, colony health, as well as other endpoints, reveal conflicting results with similar effects not typically observed at test concentrations under realistic exposure scenarios (Tasei et al., 2000; Tasei et al. 2001; Tasei, 2002; Bortolloti et al., 2003; Maus et al., 2003; Morandin and

Winston, 2003; Stadler et al, 2003; Schmuck, 2004; Johnson, 2012; Fischer et al., 2014; Scholer and Krishik, 2014; Feltham et al., 2014). Blacquière et al. (2012) and Pisa et al. (2015) summarize many of these studies with effect concentrations ranging from the low µg/bee or µg/kg to greater, depending on the study design and endpoint. Many of the effects reported in these laboratory studies have not been reported in the field under more realistic exposure conditions (Cresswell, 2010; Blacquière et al., 2012; Fairbrother et al., 2014). The lack of reported effects in field studies may be due to the limitations in the extrapolation of laboratory effects to the field, health of the bees, and the experimental design of field studies. The lack of statistical power in the design of field studies may lead to incorrect conclusions regarding the lack of effects (Cresswell, 2010; van der Sluijs et al., 2013). Recent studies have reported mixed results regarding impacts to pollinators at field relevant doses. Feltham et al. (2014) reported a reduction in pollen collection in the field for bumblebees exposed in the laboratory to sugar water and pollen imidacloprid concentrations of 0.7 and 6.0 µg/L, respectively. No differences were reported for nectar collection efficiency between controls and imidacloprid exposed bees.

Imidacloprid lethality and sublethal studies have been conducted with various earthworm species. Lethality varies with LC_{50} values ranging from 1.5 to 25.5 ppm (Pisa et al., 2015). Sublethal effects have been noted in various earthworm species as well. Effects on reproduction and behavior, such as burrowing, have been noted in various species and under various exposure conditions with effects seen at concentration ranging from 0.1 ppm and above (Bhattacharya and Sahyu, 2013; Pisa et al., 2015). Earthworm studies testing the effects of imidacloprid applied for the ALB have reported effects in both soil and trunk injections. Kreutzweiser et al. (2009) reported no mortality to earthworms that were fed scenescent leaves from trunk injected maple trees containing 3 to 11 mg/kg of imidacloprid. However they did report sublethal impacts such as antifeeding behavior and earthworm weight loss in addition to reduced leaf litter degradation. In soil treatment studies using imidacloprid applied for ALB control a LC_{10} and LC_{50} of 2 and 5.7 mg/kg, respectively, were reported for, the earthworm *Dendrobaena octaedra*. Sublethal effects such as weight loss were reported at 3 mg/kg. The other earthworm species tested, *Eisenia fetida*, was less sensitive with a reported LC_{50} of 25 mg/kg and sublethal effects noted at 14 mg/kg (Kreutzweiser et al., 2008).

3.2.2.4 Terrestrial Plants

Toxicity testing using terrestrial plants is not typically required in the registration of insecticides and little data exists regarding potential effects. Of the available data, no impacts to treated terrestrial plants have been noted in forestry or agricultural settings (Westwood et al., 1998; USDA FS, 2005).

3.2.3 Toxicity of formulations and metabolites to nontarget wildlife

The aquatic toxicity of the primary metabolites of imidacloprid in the environment which are urea-based metabolites, designated as NTN 33823 and NTN 33519, and 6-chloronicotinic acid have been evaluated for *H. azteca* and *C. tentans* which represent the more sensitive aquatic species. Based on the range of toxicity values for each of the metabolites compared to imidacloprid, the toxicity for each metabolite is several orders of magnitude less than the parent (USDA FS, 2005) (table 3-5).

Table 3-5. Comparative aquatic toxicity data for imidacloprid and associated metabolites.

Test Species	Chemical	Endpoint/Length	Toxicity Value (µg/L)	NOEC (µg/L)
Hyallela azteca	imidacloprid	96-hour LC_{50}	526.0	0.35
	NTN 33823	96-hour LC_{50}	51800	22100
	NTN 33519	96-hour LC_{50}	>94830	94830
C. tentans	imidacloprid	96-hour LC_{50}	5.75	1.03
	NTN 33823	96-hour LC_{50}	>82800	8190
	NTN 33519	96-hour LC_{50}	>99800	99800
	6-chloronicotinic acid	96-hour LC_{50}	>1000	1000

Formulation aquatic toxicity data does not appear to be available for Merit®, or some of the other available formulations for use against ALB; however, aquatic toxicity data with other formulations containing imidacloprid have demonstrated comparable, or increased aquatic toxicity in studies conducted with the technical material (Stoughton et al., 2008). The complete list of inerts is considered confidential business information and therefore is currently not available for review for most formulations. The material safety data sheet for Merit® states that glycerin is present although the exact quantity is unknown.

4.0 EXPOSURE ASSESSMENT

4.1 Human Health Exposure Assessment

Exposure assessments estimate the potential exposure of humans to imidacloprid. The exposure assessment begins with the use and application method for imidacloprid. An identified exposure pathway for imidacloprid includes (1) a release from an imidacloprid source, (2) an exposure point where contact can occur, and (3) an exposure route such as ingestion, inhalation, or dermal contact (USEPA 1989). Exposures for the various subgroups of the population are qualitatively evaluated for each identified exposure pathway.

4.1.1 Identification of Potentially Exposed Human Populations and Complete Exposure Pathways

Based on the expected use pattern for both types of imidacloprid applications (trunk or soil injection), applicators and workers in the program who are mixing and applying the insecticide in the field are the most likely subgroup of the human population to be exposed to imidacloprid. Exposure during transportation is not anticipated because the container of the concentrated material is sealed. Following label directions including the use of proper personal protective equipment (PPE) will minimize exposure to workers. Accidental exposure may occur during mixing and applying the formulation. Under an accidental spill scenario, workers may be exposed to imidacloprid through dermal contact. However, the potential dermal contact exposure is anticipated to be limited because these accidental events would be of low frequency and short duration.

In addition to worker exposure there is the possibility of exposure to residents in areas during and after treatment. Generally, exposure to residents during treatment is not expected based on the method of application (trunk and soil injection). Exposure to residents whose property may be treated can be minimized through proper notification prior to treatment. Therefore, a significant exposure pathway is not identified for direct contact to imidacloprid for the general public. A significant exposure pathway is not identified for dietary plant consumption because treated trees will not have products harvested for human consumption. There is the potential for dietary exposure to the public in cases where treated that are used in the production of maple syrup; however, this is not a registered use for imidacloprid and this type of exposure would not be expected to occur. The soil injection application method may result in potential exposure to imidacloprid for children who exhibit pica, the recurrent ingestion of unusually high amounts of soil, on the order of 1,000-5,000 mg/day or more (USEPA, 2011)). In addition, there is the potential for children and adults to be exposed to residual imidacloprid in leaf litter. The environmental monitoring data indicates that imidacloprid can persist in treated trees for at least 12 months after treatment. Chemical uptake occurs throughout the tree with higher residues detected in leaf samples than twig samples for most species of trees sampled (USDA APHIS, 2002).

A significant exposure pathway is not idenified for groundwater or surface water media under the trunk injection use pattern. Imidacloprid is soluble and has the potential to leach from soil to groundwater under the soil injection use pattern. Groundwater sampling between 2003 and 2006 in Suffolk County, New York, demonstrated that approximately half of the water samples had no detectable levels of imidacloprid. Of those where detections occurred, the average concentration was 3.2 parts per billion (ppb) which is below the level of concern for human health (USDA APHIS, 2007). Samples with detectable levels of imidacloprid do not suggest a contribution from the ALB eradication program because other uses of imidacloprid occurred in these areas, and there did not appear to be a significant correlation between ALB-related treatment activities and increased residues (USDA APHIS, 2013). In addition, significant surface runoff into aquatic resources is not expected based on the trunk and soil injection use pattern (USDA FS, 2005).

4.1.2 Exposure Evaluation

This section quantitatively evaluates the worker exposures from direct contact pathways while applying imidacloprid using trunk and soil injection.

The proposed use pattern for imidacloprid in the ALB program involves soil injection applications using the Merit® 2F, or similar, formulation. The rate of application for Merit® 2F is 3-6 ml (0.1-0.2 fl oz) per inch of trunk diameter at breast height with a maximum use rate of 25.6 fl oz product/acre/year. The formulation is applied in water in sufficient quantities to adequately moisten the soil near the base of the tree but not saturate the soil which could increase the likelihood of off-site transport. Due to the method of application, exposure from drift is not expected. Exposure to applicators will be minimal based on the method of application, following the label regarding PPE, and the environmental fate of imidacloprid which reduces exposure via inhalation, dermal, and oral routes. To quantify the potential exposure to workers making ground applications using imidacloprid, estimated doses were derived based on methods developed by the U.S. Forest Service, and in particular, those that were developed to assess human health risks to imidacloprid (USDA FS, 2005; SERA, 2005). A lower and upper bound estimate of exposure was estimated based on assumptions regarding the amount of time and product that is applied during a typical work day.

To quantify the potential exposure to a child from soil ingestion and dermal contact to leaf litter, the soil (lower and average concentrations of 0.13 ppm and 1.32 ppm, respectively) and leaf litter (average and maximum concentrations of 1.966 ppm and 97.5 ppm, respectively) data from the APHIS ALB environmental monitoring program were used.

4.2 Ecological Exposure Assessment

4.2.1 Aquatic Exposure Assessment

Exposure to aquatic organisms will occur primarily from either runoff related to soil applications or from leaf litter that may be deposited into waterbodies. Drift is not expected to be an exposure pathway based on the method of application. Estimating residues that could occur from leaf deposition into water bodies is difficult to quantify using standardized environmental fate models. Aquatic residues from soil injection are based on estimates from the U.S. Forest Service that were modeled for various soil types, precipitation levels and water bodies (USDA FS 2005). Average estimated aquatic residues in ponds ranged from 1.49×10^{-7} ppb under low annual rainfall (25 in) to 0.308 ppb under high annual rainfall (250 in). Concentrations were less in small streams with average imidacloprid levels of 1.09×10^{-8} ppb under low rainfall levels to 0.0829 ppb under high annual rainfall levels. Yearly rainfall levels of 25 inches or less resulted in no detectable levels of imidacloprid. This range of values is based on the assumption of a one pound per acre application applied to one acre of land adjacent to an aquatic habitat. Residues could increase as the amount of acreage treated relative to the size of the water body increases; however, as the treatment is moved further away from the water body more rainfall would be needed to transport imidacloprid. This range of estimated residues was compared to the available effects data for imidacloprid in the aquatic risk characterization section of this assessment.

4.2.2 Terrestrial Exposure Assessment

Exposure to mammals and birds will be primarily through dietary exposure from consuming treated vegetation or invertebrates that consume treated vegetation, or are present in the soil where applications may occur. Dermal and inhalation exposure for mammals and birds is not expected based on the method of application and environmental fate of imidacloprid. Dietary exposure would be greatest for those mammals and birds that feed exclusively on vegetation from treated trees. Average and maximum residues for imidacloprid are 1.966 ppm and 97.5 ppm, respectively, based on environmental monitoring that has been done by the program.

Honeybees and other pollinators may be exposed to imidacloprid from nectar and pollen. Exposure honeybees to imidacloprid from water is expected to be minor based on the methods of application. Field monitoring data from trees treated with imidacloprid during the ALB eradication program suggest exposure levels will be low (Johnson, 2012). Pollen samples collected from trees treated with imidacloprid through soil injection revealed residues below the level of detection (1.0 ppb) to approximately 30.6 ppb with an average concentration of 5.31 ppb. Greater than 50% of the samples collected from treated trees contained levels below detection. Pollen samples collected from trees treated with imidacloprid through trunk injected trees were comparably lower, ranging from below detection to 1.5 ppb with an average concentration of 0.28 ppb. Greater than two-thirds of the pollen samples had levels below detection. The residues collected from this study fall within the low range of imidacloprid residues that have been reported in pollen samples from agricultural crops where imidacloprid is applied as a seed treatment or a broadcast application (Bonmatin et al., 2015). Nectar samples collected from honeybee hives in the ALB-treated tree study revealed no detectable levels of imidacloprid or its metabolites.

5.0 RISK CHARACTERIZATION

5.1 Human Health

Risks associated with human health are characterized quantitatively for complete exposure pathways in this section. Under the PPQ proposed use, imidacloprid use to control ALB populations poses minimal risk to human health when applications are made according to label directions.

Estimated exposure doses were based on methods developed for imidacloprid by the U.S. Forest Service methods to quantify the potential risk to workers making ground applications of imidacloprid (USDA FS, 2005; SERA, 2005). A lower and upper bound estimate of exposure was based on assumptions regarding the amount of time and product that is applied during a typical work day. The exposure estimate was then compared to the acute (0.14 mg/kg/day) and chronic (0.057 mg/kg/day) reference dose to calculate hazard quotient (HQ) values.

Both acute and chronic HQ values were below one under all scenarios suggesting minimal risk to workers making applications of imidacloprid (table 5-1).

Table 5-1. Hazard quotient values estimated for workers making imidacloprid applications.

	Lower Estimate	Central Estimate	Upper Estimate
Hours of Application/Day	6	7	8
Acres Treated/Day	48	77	144
Adsorbed Dose Rate (mg/kg per lb a.i. /day)	0.00001	0.0002	0.0009
Adsorbed Dose (mg/kg bw/day)	1.92×10^{-4}	6.16×10^{-3}	5.18×10^{-2}
Acute Hazard Quotient	0.001	0.04	0.4
Chronic Hazard Quotient	0.003	0.1	0.9

To quantify the HQ from the unusual soil ingestion behavior (pica) and leaf litter ingestion by children (age 1-6), soil concentrations were estimated using the application dosage for a soil injection scenario. The estimated soil concentrations were compared to the soil concentrations collected from the environmental monitoring reports. The imidacloprid concentations from the soil monitoring data are higher and were used for the risk estimation. The leaf litter data collected from the monitoring reports was also used for estimating risk from those types of exposures. HQ values were calculated using the following USEPA soil ingestion risk estimation equation for non-carcinogens:

> Acute Hazard Quotient = (Soil/Leaf Litter Concentration x Soil/Leaf Ingestion Rate)/ Body Weight or
> Chronic Hazard Quotient = (Soil/Leaf Litter Concentration x Soil/Leaf Ingestion Rate x Exposure Duration x Exposure Frequency x Conversion Factor)/(Averaging Time x Body Weight x Reference Dose) (USEPA 2002).

The calculated acute and chronic HQ values were below one under all scenarios suggesting minimal risk to imidacloprid from soil or leaf litter ingestion by children. The upper estimate represents the maximum amount of fluid per acre in a single application (table 5-2).

Table 5-2. Hazard quotient values estimated for children exposures to soil and leaf litter.

	Lower Estimate	Upper Estimate
Application Dosage for Soil Injection	less than 6.0 fl oz per acre in a single application and 2 pounds of imidacloprid per gallon	
Estimated Soil Concentration	0.0445 mg/kg	0.089 mg/kg
Soil Monitoring Data	0.13 mg/kg	1.32 mg/kg
Leaf Litter Concentration	1.966 mg/kg	97.5 mg/kg
Acute Hazard Quotient		
Soil		
Leaf Litter	0.1	1
	0.02	1
Chronic Hazard Quotient		
Soil	0.0008	0.008
Leaf Litter	0.01	0.7

Note: A default soil bulk density of 1.55 g/cm^3 for silt loams, and silty clay loams soil (USDA NRCS, 2014) was used to estimate soil concentration.

The risks for the general public from potential exposure to imidacloprid during application are not quantified because of minimal potential for exposure and low exposure frequency and duration if potential exposure occurs. Dietary consumption of maple syrup from trees treated with imidacloprid is not expected because that is not a labelled use and those trees would not receive treatment. Consequently, the potential adverse health risks for the general public are expected to be minimal.

The risks to the general public from potential exposure to imidacloprid released to soil and transported to surface water or groundwater are not quantified because of the low surface runoff potential based on the proposed use pattern.

5.2 Aquatic and Terrestrial Ecological Risk Characterization

This section of the risk assessment will integrate the effects analysis, regarding toxicological studies summarized for terrestrial and aquatic nontarget organisms, with the potential for exposure, to quantify the risk to nontarget organisms and domestic animals from program use of imidacloprid. Direct risk refers to those risks that could occur from direct exposure to imidacloprid while indirect risk refers to impacts that could occur to prey or habitat that nontarget organisms rely on for food and shelter. Impacts to individuals and populations are not anticipated in cases where the range of imidacloprid toxicity data to imidacloprid does not overlap the range of potential exposure values. Further evaluation of the assumptions used in the risk characterization is required to refine risk where residues exceed the response data.

5.2.1 Aquatic

The potential risk of imidacloprid applications to aquatic environments is low for tree and soil injections. The low risk to these types of habitats is related to the methods of application proposed for ALB where trees are injected directly with imidacloprid or the chemical is injected into the soil under the dripline of the tree. Drift and runoff are not anticipated to be major pathways for off-site transport of imidacloprid to aquatic environments. There is a greater chance for runoff from soil injections compared to tree injections; however, the label restrictions and estimates of residues when compared to the available toxicity data show the risk to be low (Figure 2). The range of effects data discussed in the response section was compared to the residue values estimated by the U.S. Forest Service for soil injections in various soil types, and water bodies, under various annual rainfall amounts. The comparison of the two data sets suggests that acute and chronic risk to aquatic organisms from soil application of imidacloprid would be low based on the assumptions used in the exposure modeling.

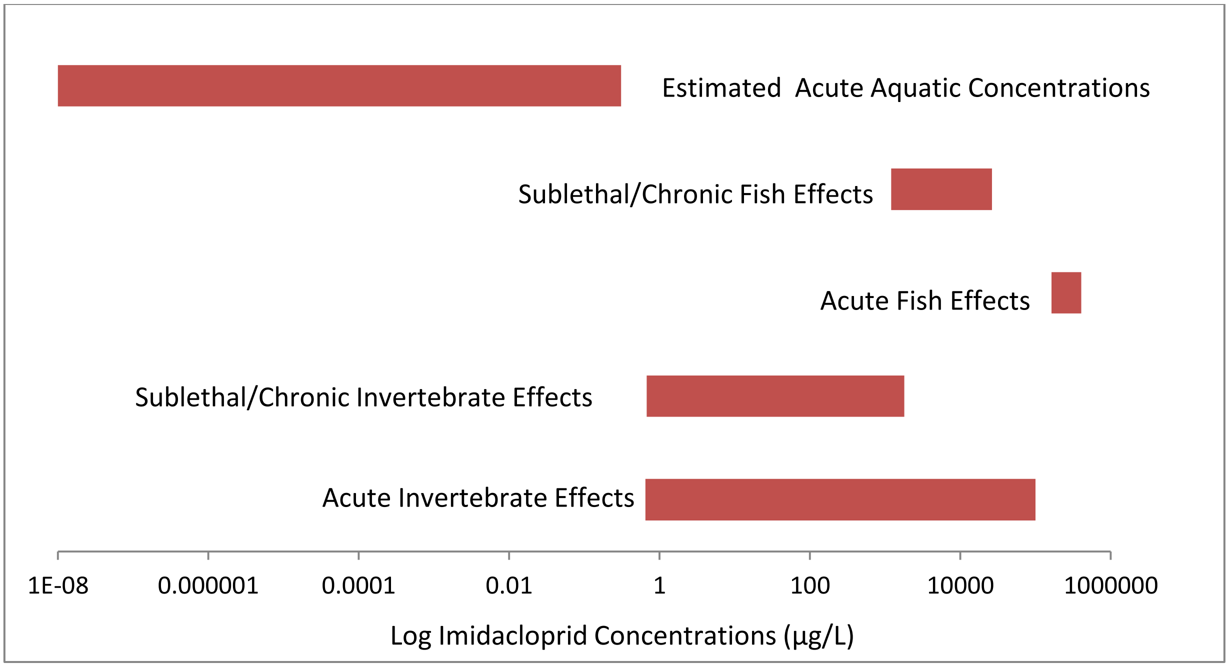


Figure 2. Aquatic risk characterization for imidacloprid soil applications.

The greatest potential for residues to reach aquatic habitats is from the leaves of treated trees. Acute risk of direct effects to fish and amphibians is very low with an increase in risk to aquatic invertebrates from this pathway. Impacts to aquatic invertebrates can result in indirect effects to fish and amphibians that rely on invertebrates as a food source. Previous research has indicated that lethal effects from this pathway are not expected based on field observed levels of imidacloprid in leaf litter deposited in water bodies (Kreutzweiser et al., 2007; Kreutzweiser et al., 2008a; Kreutzweiser et al., 2009). However some sublethal impacts, such as cessation of feeding, have been noted (Kreutzweiser et al., 2009).

5.2.2 Terrestrial Wildlife and Domestic Animals

The direct risk of imidacloprid to mammals and birds is expected to be low based on the available toxicity data and the method of application and label requirements that reduce the potential for dietary exposure. This would include domestic animals such as dogs and cats which would not be expected to forage on treated plants or soil under trees that have been treated. Vegetation collected from previous program treatments show average and maximum concentrations of 1.966 ppm and 97.5 ppm, respectively. Plausible exposure scenarios for wild mammals and birds from either tree injections or soil treatments is difficult to quantify; however, the direct risk to nontarget wildlife is expected to be low based on the available toxicity data and methods of application (USDA FS, 2005). Not all trees and other plant food sources for wild mammals and birds will be treated; therefore, exposure to imidacloprid treated vegetation will be reduced since other non-treated plants will be within the home range of most mammals and birds.

A recent review of published studies regarding direct and indirect risk to vertebrates from imidacloprid use suggests that environmental exposure levels are below concentrations that would result in mortality and most reported sublethal effects (Gibbons et al., 2014). There is some uncertainty regarding the lack of risk due to the extrapolation of available toxicity data and sublethal impacts from one animal group to another or from one use pattern to another. Mineau and Palmer (2013) discuss similar direct and indirect risks regarding impacts to birds; however, the emphasis of that work was related to seed treatments using imidacloprid as well as other neonicotinoid insecticides. Direct risk to birds and mammals from imidacloprid treatments in the ALB program would be expected to be comparatively lower than seed treatments. Treated seed that is exposed on the ground during planting operations in agricultural settings would be more attractive to seed eating birds and mammals resulting in greater exposure. In addition, imidacloprid use would occur within a field with little to no alternative imidacloprid residue-free seed sources available for foraging birds. This would not be the case with imidacloprid treatments from tree or soil injections as part of the ALB program. Select trees are treated within a given area based on site-specific needs, and other non-treated trees and plants producing seed would be available for foraging by birds and mammals. However, there is the possibility that imidacloprid treated trees could result in measurable residues in tree seeds. This risk is expected to be low, especially for maples, that are not preferred food sources for most seed eating birds and mammals compared to other plant species (Rodewald and Abrams, 2002). In addition, maple seeds degrade more quickly than hard mast seeds, such as acorns, and would not be available for foraging. Terrestrial vertebrates that eat insects and other terrestrial invertebrate species may also be exposed to imidacloprid from foraging on species that have consumed plant material from treated trees. The risks to terrestrial vertebrates from this type of exposure is expected to be low due to several factors. Exposure to most insectivorous vertebrates would be low because there would be unexposed invertebrate because the ALB program does not treat all plant material in an area. In cases of soil injections, there would also be the potential for soil inhabiting invertebrates that occur under the tree dripline to be exposed; however, these would be localized to areas within the dripline of an individual tree and would not be area wide. In addition, most vertebrates that feed on invertebrates would preferentially consume live prey compared to those that had received lethal doses of imidacloprid.

Indirect risk to terrestrial vertebrates through the loss of prey items is also not anticipated to be a significant risk because only invertebrates that consume treated material would be potentially impacted by imidacloprid. Thus, many invertebrates with no imidacloprid residues would occur in an area to provide a food source to insectivorous terrestrial vertebrates. Also, invertebrates that occupy treated trees but do not feed on the leaves and twigs would not be impacted by imidacloprid. This would also be the case for soil inhabiting invertebrates that may occur under the dripline of trees that are treated with imidacloprid using soil injection. Falcone and DeWald (2010) evaluated the impacts of imidacloprid broadcast treatments on invertebrate prey items for several neotropical bird species in areas where the invasive balsam wooly adelgid occurs. Hemipteran and lepidopteran abundance declined in treated areas versus control areas, but species richness and composition was similar between treated and nontreated blocks. Total bird abundance for the species evaluated did not differ between treated and untreated areas, suggesting that these species were able to forage on other arthropods in treated blocks. These types of impacts would not be anticipated in the ALB program because imidacloprid is not being applied using broadcast applications.

5.2.3 Terrestrial Invertebrates and Plants

Risks to terrestrial invertebrates will vary based on the method of application. Soil inhabiting invertebrates such as earthworms will be at greater risk from imidacloprid soil applications compared to tree injections. Imidacloprid residue data that has been collected as part of the ALB eradication program from various locations report an average soil concentration of 1.32 ppm with a maximum value of 43 ppm. Soil injection sites will have elevated imidaclprid levels compared to the area under the tree canopy drip line. The reported average soil imidacloprid values are within the range of lethal (LC_{50} > 1.5 ppm) and sublethal (> 0.1 ppm) effects data threshold concentrations, suggesting risk to earthworms. The risk to earthworms in the field will be affected by site-specific conditions that could impact exposure concentration and duration. Any impacts are expected to be restricted to areas where imidacloprid has been injected into the soil under the dripline of trees where applications would occur.

Terrestrial invertebrates that are sensitive to imidacloprid and consume plant material from treated trees are expected to be impacted; however, based on the method of application and the mode of action for imidacloprid, these risks will be reduced when compared to making broadcast applications. Risks to pollinators are expected to be low based on the available residue data that has been collected for pollen from trees treated with imidacloprid either through trunk or soil injection and the lack of detectable residues in nectar (Johnson, 2012). Mean imidacloprid pollen levels were 1.5 and 5.31 ppb in trunk and soil injected trees, respectively. These residues are below a majority of the sublethal effects data for similar types of exposure. In addition, more than half of the pollen samples in both treatment methods contained no detectable levels of imidacloprid and none of the nectar samples contained detectable levels of imidacloprid. Pollen data collected from bees in the same study suggest that other sources of pollen are being utilized, further reducing exposure to foraging bees, as well as other bee life stages. Pollen traps used to collect pollen from foraging bees at the entrance to the hives demonstrated that the percentage of pollen from red maples, a preferred ALB host tree, ranged from 16 to approximately 63 percent. Pollen sources from other plants that are not treated with imidacloprid will further reduce exposure to foraging bees as well as individuals within the hive. Exposure and risk to honeybees that forage on imidacloprid treated trees would be low due to the low residues that have been reported and the frequency of imidacloprid levels below detection observed in nectar and pollen. In addition there would also be pollen sources from plants that have not been treated with imidacloprid that honey bees would use.

Direct risk to plants is expected to be low based on the method of application to selective trees, and the lack of direct toxicity to terrestrial plants. Indirect risks to terrestrial plants from potential impacts to pollinators is also expected to be low based on the available data regarding imidacloprid pollen and nectar levels in imidacloprid treated trees. Invertebrates that may feed on treated trees, and are considered pollinators, may be impacted if they are sensitive to the effects of imidacloprid. However, impacts to the pollination of terrestrial plants would be minor because most plants have various pollinators, many of which would not be exposed to any imidacloprid residues because they would not consume leaves or twigs from treated trees.

6.0 UNCERTAINTIES AND CUMULATIVE IMPACTS

The uncertainties associated with this risk evaluation arise primarily from lack of information about the effects of imidacloprid, its formulations, metabolites, and potential mixtures to non-target organisms that can occur in the environment. These uncertainties are not unique to this assessment but are consistent with uncertainties in human health and ecological risk assessments with any environmental stressor. In addition, there is uncertainty in where an ALB infestation may occur in the United States and the extent of imidacloprid use in a given infestation because its use is based on site-specific factors.

Another area of uncertainty is the potential for cumulative impacts to human health and the environment from the proposed use of imidacloprid in the ALB program. Areas where cumulative impacts could occur are: 1) repeated worker and environmental exposures to imidacloprid from program activities in conjunction with other crop use sources; 2) co-exposure to other chemicals with a similar mode of action; and 3) exposures to other chemicals in mixtures and how that may affect the toxicity of imidacloprid.

Cumulative impacts may occur from imidacloprid use from other APHIS programs and in relation to other chemicals that have a similar mode of action, as well as others that have a different mode of action, that can result in synergism, potentiation, additivity or antagonistic effects. Other neonicotinoid insecticides include thiamethoxam, clothianidin, dinotefuran, nitenpyram, acetamiprid, and thiacloprid (Goulson, 2013). Temporal variability in the occurrence of multiple stressors, as well as their effects, are not well understood. As an example, available water quality monitoring data in the United States indicate the presence of multiple natural and anthropogenic contaminants. Sources for these chemicals can occur from point and non-point sources, and the relative contribution from each is dependent on land use in a given watershed. Based on the most recent United States Geological Survey National Water Quality Assessment (USGS–NAWQA) data for pesticides, frequency of occurrence for two or more pesticides in surface water exceeds 80% nationally (Gilliom et al., 2006). When considering other organics and trace metals, the combination of mixtures can become extremely large, especially when spatial and temporal variability in mixtures that can occur in a given watershed are considered. The seasonal variability in mixtures of pesticides and other contaminants has been well documented nationally in urban and agricultural areas (Ryberg et al., 2010; Gilliom et al., 2006; Stone et al., 2014). An analysis of all detections from agricultural streams indicated more than 6,000 unique mixtures of 5 pesticides (Gilliom et al., 2006). As would be expected, based on the large variability in mixtures, the ecological and human health response data for these types of exposure scenarios is very limited for all organic and inorganic chemicals including those proposed in the program.

From a human health perspective, the ALB program use of imidacloprid is expected to have negligible cumulative impacts. The proposed trunk and soil injection use of imidacloprid in the ALB progam is unlikely to enter the food chain or drinking water and be available to significantly add to imidacloprid exposure in the human population, or occur with other naturally occurring or synthetic compounds that could result in acute or long-term impacts. Use of imidacloprid and other neonicotinoids on animals is another exposure source. Repeated chronic exposure to imidacloprid from pet medicine may pose possible health risks to veterinarians, veterinary technologists, dog caretakers, and owners (Craig et al., 2005). However, due to the low probability of imidacloprid exposure to the human population from applications as part of the ALB program, the potential for significant cumulative impacts would be minor.

7.0 REFERENCES

Agatz, A., Ashauer, R. and C.D. Brown. 2014. Imidacloprid perturbs feeding of *Gammarus pulex* at environmentally relevant concentrations. Environmental Toxicology and Chemistry 33(3):648–653.

Agatz, A., Cole, T.A., Preuss, T.G., Zimmer, E. and C.D. Brown. 2013. Feeding inhibition explains effects of imidacloprid on the growth, maturation, reproduction, and survival of *Daphnia magna*. Environ. Sci. Technol. 47:2909−2917.

Alexander, A.C., Culp, J.M., Liber, K. and A.J. Cessna. 2007. Effects of insecticide exposure on feeding inhibition in mayflies and oligochaetes. Env. Tox Chem. 26(8): 1726-1732.

Badgujar, P.C., Jain, S.K., Singh, A., Punia J.S., Gupta, R.P., and G.A.Chandratre. 2013. Immunotoxic effects of imidacloprid following 28 days of oral exposure in BALB/c mice. Environ. Toxicol. Pharm. 35:408-418.

Bayer (Bayer Environmental Science). 2004. Merit® 2F insecticide specimen label http://www.backedbybayer.com/search?utf8=%E2%9C%93&search=Merit%C2%AE+2F+ last accessed 3/28/2014.

Bayer. 2008. Material safety data sheet (MSDS Number: 102000015064 Version 2.1 Revision Date: 05/19/2008) http://www.backedbybayer.com/system/product/msds_en/52/Merit_2F_Insecticide_19May2008.pdf, last accessed 3/28/2014

Bhattacharya, A. and S.K. Sahu. 2013. Eco-toxicity of neonicotinoid: a case study on the impact of imidacloprid using *Drawida willsi* earthworm as bioindicator. Int. J. Biosci. 3(5): 78-86.

Blacquière, T., Smagghe, G., vanGestel, C.A.M. and V. Mommaerts. 2012. Neonicotinoids in bees: a review on concentrations, side-effects and risk assessment. Ecotoxicology 21:973–992.

Bortolloti, L., Montanari, R., Marcelino, J., Medrzycki, P., Maini, S., and C. Porrini, 2003. Effect of sub-lethal imidacloprid doses on the homing rate and foraging activity of honey bees. Bull. Insectol. 56: 63–67.

Bonmatin, J.M., Giorio, C., Girolami, V., Goulson, D., Kreutzweiser, D.P., Krupke, C., Liess, M., Long, E., Marzaro, M., Mithcell, E.A.D., Noome, D.A., Simon-Delso, N. and A. Tapparo.

2015. Environmental fate and exposure; neonicotinoids and fipronil. Environ. Sci. Pollut. Res. 22: 35-67

CA DPR (California Department of Pesticide Regulation). 2006a. Environmental fate of imidacloprid. revised by Fossen, M., Environmental Monitoring, Department of Pesticide Regulation. 16 pp. April 2006. Available at: http://www.cdpr.ca.gov/docs/emon/pubs/fatememo/Imidclprdfate2.pdf, last accessed 3/25/2014.

CA DPR. 2006b. Imidacloprid risk characterization document dietary and drinking water exposure, Health Assessment Section, Medical Toxicology Branch, Department of Pesticide Regulation. 195 pp.

Canadian Council of Ministers of the Environment (CCME). 2007 Canadian water quality guidelines: imidacloprid scientific supporting document, available at: http://www.ccme.ca/files/Resources/supporting_scientific_documents/imidacloprid_ssd_1388.pdf, last accessed 1/29/2015.

Costa, C., Silvari, V., Melchini, A., Catania, S., Heffron, J.J., Trovato, A. and R.D.Pasquale. 2009. Genotoxicity of imidacloprid in relation to metabolic activation and composition of the commercial product. Mutation Research 672: 40-44.

Craig, M.S., Gupta, R.C., Candery, T.D. and D.A. Britton. 2005. Human exposure to imidacloprid from dogs treated with Advantage. Toxicol. Mech. Methods 15: 287-291.

Cresswell J.E. 2010. A meta-analysis of experiments testing the effects of a neonicotinoid insecticide (imidacloprid) on honey bees. Ecotoxicology 20:149-157.

Fairbrother, A., Purdy, J., Anderson, T. and R. Fell. 2014. Risks of neonicotinoid insecticides to honeybees. Env. Toxicol. Chem. 33(4): 719-731.

Falcone, J.F. and L.E. DeWald. 2010. Comparisons of arthropod and avian assemblages in insecticide-treated and untreated eastern hemlock (*Tsuga canadensis* [L.] Carr) stands in Great Smoky Mountains National Park, USA. Forest Ecol. Mgt. 260:856–863.

Feng, S., Kong, Z., Wang, X., Zhao, L. and P. Peng. 2004. Acute toxicity and genotoxicity of two novel pesticides on amphibian, *Rana N. hallowell*. Chemosphere 56:457–463.

Feltham, H., Park, K. and D. Goulson. 2014. Field realistic doses of pesticide imidacloprid reduce bumblebee pollen foraging efficiency. Ecotoxicology 23:317–323.

Fischer, J., Muller, T., Spatz, A.K., Greggers U. and B. Grunewald. 2014. Neonicotinoids interfere with specific components of avigation in honeybees. PLoS ONE 9(3): e91364. doi:10.1371/journal.pone.0091364

Gawade, L., Dadarkar, S.S., Husain, R., and M. Gatne. 2013. A detailed study of developmental immunotoxicity of imidacloprid in wistar rats. Food Chem. Toxicol. 51:61-70.

Gervais, J. A., Luukinen, B., Buhl, K. and D. Stone. 2010. Imidacloprid technical fact sheet; National Pesticide Information Center, Oregon State University Extension Services. http://npic.orst.edu/factsheets/imidacloprid.html last accessed 3/25/2014

Gibbons, D., Morrissey, C. and P. Mineau. 2014. A review of the direct and indirect effects of neonicotinoids and fipronil on vertebrate wildlife. Environ. Sci. Pollut. Res.

Gilliom, R.J. and P.A. Hamilton. 2006. Pesticides in the Nation's streams and ground water, 1992–2001, The Quality of Our Nation's Waters. U.S. Geological Survey Circular 1291,172 p.

Graebing P. and J.S. Chib 2004. Soil photolysis in a moistureand temperature-controlled environment. 2. Insecticides. Journal of Agricultural and Food Chemistry 52(9): 2606–2614.

Goulson, D. 2013. Review: An overview of the environmental risks posed by neonicotinoid insecticides http://onlinelibrary.wiley.com/doi/10.1111/1365-2664.12111/full last accessed 3/28/2014.

HSDB (Hazardous Substances Databank). 2014. "Imidacloprid"; *IN:* Toxnet. U.S. Department of Health and Human Services, National Institutes of Health, National Library of Medicine. http://toxnet.nlm.nih.gov/cgi-bin/sis/search2/f?./temp/~JQa1Yi:1, last accessed 6/20/2014

Iyyadurai, R., George, I.A, and J.V. Peter, 2010. Imidacloprid poisoning newer insecticide and fatal toxicity, J. Med. Toxicol. 6:77–78

Jemec, A., Tisler, T., Drobne, D., Sepcic, K., Fournier, D. and P. Trebs. 2007. Comparative toxicity of imidacloprid, of its commercial liquid formulation and of diazinon to a non-target arthropod., the microcrustacean *D. magna*. Chemosphere 68:1408–1418.

Johnson, J.D. 2012. The role of pesticides on honey bee health and hive maintenance with an emphasis on the neonicotinoid, imidacloprid. Dissertation submitted to University of Maryland. 170 pp.

Key, P., Chung, K., Siewicki, T. and M. Fulton. 2007. Toxicity of three pesticides individually and in mixture to larval grass shrimp (*Palaemonetes pugio*). Ecotox. Environ. Safety 68(2):272-277.

Kreutzweiser, D., Thompson, D. and T.A. Scarr. 2009. Imidacloprid in leaves from systemically treated trees may inhibit litter breakdown by non-target invertebrates. Ecotox. and Environ. Safety. 72:1053–1057.

Kreutzweiser, D., Good, K., Chartrand, D., Scarr, T. and D. Thompson. 2008. Are leaves that fall from imidacloprid-treated maple trees to control Asian longhorned beetles toxic to non-target decomposer organisms? J. Environ. Qual. 37: 639–646.

Kreutzweiser, D., Good, K., Chartrand, D., Scarr, T. and D. Thompson 2007. Non-target effects on aquatic decomposer organisms of imidacloprid as a systemic insecticide to control emerald ash borer in riparian trees. Ecotox. and Environ. Safety. 68: 315–325.

Kumar, A., Verma, A.B. and A. Kumar. 2013. Accidental human poisoning with a neonicotinoid insecticide, imidacloprid: A rare case report from rural India with a brief review of literature, Egyptian Journal of Forensic Sciences 3:123-126. http://ac.els-cdn.com/S2090536X13000427/1-s2.0-S2090536X13000427-main.pdf?_tid=d32e9344-fd61-11e3-9c00-00000aacb360&acdnat=1403808415_f61bd64cdb42a8f2fd88bd50b8749ca8, last accessed 6/26/2014.

Liu, W., Zheng, W., Ma, Y. and K.K. Liu. 2006. Sorption and degradation of imidacloprid in soil and water, Journal of Environmental Science and Health Part B, 41:623–634.

Lin P.C., Lin,H.J., Liao, Y.Y., Guo, H.R. and K.T. Chen. 2013. Acute poisoning with neonicotinoid insecticides: a case report and literature review. Basic & Clinical Pharmacology & Toxicology, 112, 282–286. http://onlinelibrary.wiley.com/doi/10.1111/bcpt.12027/pdf, last accessed 6/26/2014.

Maus, C., Cure, G. and R. Schmuck. 2003. Safety of imidacloprid seed dressings to honey bees: a comprehensive overview and compilation of the current state of knowledge. Bull. Insectol. 56(1): 51–57.

Hrouzková, S. and E. Matisova. 2012. Chapter 5 Endocrine disrupting pesticides, Pesticides - Advances in Chemical and Botanical Pesticides. http://www.intechopen.com/books/pesticides-advances-in-chemical-and-botanical-pesticides/endocrine-disrupting-pesticides last accessed 3/27/2014.

Mineau, P. and C. Palmer. 2013. The impact of the nation’s mostwidely used insecticides on birds. American Bird Conservancy, USA. 97 pp.

Morandin, L.A., and M.L. Winston. 2003. Effects of novel pesticides on bumble bee (Hymenoptera: Apidae) colony health and foraging ability. Environ. Entomol. 32(3): 555–563.

Mohamed, F., Gawarammana, I., Robertson, T.A., Roberts, M.S., Palangasinghe, C., Zawahir, S., Jayamanne, S., Kandasamy, J., Eddleston, M., Nick A. Buckley, N.A., Dawson, A.H., and

D.M. Roberts. 2009. Acute human self-poisoning with imidacloprid compound: a neonicotinoid insecticide. PLoS ONE 4(4): e5127. doi:10.1371/journal.pone.0005127, http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0005127, last accessed 6/20/2014

Mohany, M., Badr, G., Refaat, I. and El-Feki, M. 2011. Immunological and histological effects of exposure to imidacloprid insecticide in male albino rats. African J Pharm Pharmacol 5(18): 2016-2114.

Najafi, G., Razi, M., Hoshyar, A., Shahmohamadioo, S. and S. Feyzi. 2010. The effect of chronic exposure with imidacloprid insecticide on fertility in mature male ats. International J.Fertility Sterility 4(1):9-16.

NRC (National Research Council). 1983. Risk assessment in the Federal government: managing the process. National Academy Press, Washington, DC.

Nyman, A.M., Hintermeister, A., Schirmeri, K. and R. Ashauer. 2013. The insecticide imidacloprid causes mortality of the freshwater amphipod, *Gammarus pulex* by interfering with feeding behavior. PLoS ONE 8(5): e62472. doi:10.1371/journal.pone.0062472.

Overmyer, J.P., Mason, B.N. and K.L. Armbrust. 2005. Acute toxicity of imidacloprid and fipronil to a nontarget aquatic insect, *Simulium vittatum* Zetterstedt cytospecies IS-7. Bull. Environ. Cont. Tox. 74(5):872-879.

Paul, A., Harrington, L.C. and J. G. Scott. 2006. Evaluation of novel insecticides for control of dengue vector *Aedes aegypti* (Diptera: Culicidae). J. Med. Ent. 43(1):55-60.

Pestana, J.L.T., Alexander, A.C., Culp, J.M., Baird, D.J., Cessna, A.J. and A.M.V.M. Soares. 2009. Structural and functional responses of benthic invertebrates to imidacloprid in outdoor stream mesocosms. Environ. Poll. 157:2328–2334.

Pisa, L.W., Amaral-Rogers, V., Belzunces, L.P., Bonmatin, J.M., Downs, C.A., Goulson, D., Kreutzweiser, D.P., Krupke, C., Liess, M., McField, M., Morrissey, C.A., Noome, D.A., Settele, J., Simon-Delso, N., Stark, J.D., van der Sluijs, J.P., VanDyck, H. and M. Wiemers. 2015. Effects of neonicotinoids and fipronil on non-target invertebrates. Environ. Sci. Pollut. Res. 22:68–102.

Rodewald, A.D. and M.D. Abrams. 2002. Floristics and avian community structure: implications for regional changes in eastern forest composition. For. Sci. 48(2): 267-272.

Roessink, I., Merga, L.B., Zweers, H.J. and P.J. Vanden Brink. 2013. The neonicotinoid imidacloprid shows high chronic toxicity to mayfly nymphs. Env. Toxicol.Chem. 32(5): 1096–1100.

Ruiz de Arcaute, C., Perez-Iglesias, J.M., Nikoloff, N., Natale, G.S., Soloneski, S. and M.L. Larramendy. 2014. Genotoxicity evaluation of the insecticide imidacloprid on circulating blood cells of Montevideo tree frog, *Hypsiboas pulchellus* tadpoles (Anura, Hylidae) by comet and micronucleus bioassays. Ecol. Indicators 45:632–639.

Ryberg, K.R., Vecchia, A.V., Martin, J.D. and R.J. Gilliom. 2010. Trends in pesticide concentrations in urban streams in the United States, 1992–2008: U.S. Geological Survey Scientific Investigations Report 2010–5139, 101 p.

Sánchez-Bayo, F. and K. Goka. 2005. Unexpected effects of zinc pyrithione and imidacloprid on Japanese medaka fish (Oryzias latipes) Aquatic Tox. 74.4: 285-293.

Sarkar, M., S. Roy, Kole, R. and A. Chowdhury. 2001. Persistence and metabolism of imidacloprid in different soils of West Bengal. Pest Manag. Sci. 57: 598—602.
Schmuck, R. 2004. Effects of a chronic dietary exposure of the honeybee, *Apis mellifera* (Hymenoptera: Apidae) to imidacloprid. Arch. Environ. Contam. Toxicol. 47: 471–478.

Schmuck, R., Schoning, R., Stork, A. and O. Schramel. 2001. Risk posed to honeybees (Apis mellifera L., Hymenoptera) by an imidacloprid seed dressing of sunflowers. Pest Manag. Sci. 57: 225–238.

Scholer J. and V. Krischik. 2014. Chronic exposure of imidacloprid and clothianidin reduce queen survival, foraging, and nectar storing in colonies of Bombus impatiens. PLoS ONE 9(3): e91573. doi:10.1371/journal.pone.0091573.

SERA (Syracuse Environmental Research Associates, Inc.). 2005. Documentation for Worksheets Version 4.01 - Human Health and Ecological Risk Assessments. Report dated May 4, 2005. Available at: www.sera-inc.com.

Song, M.Y., Stark, J.D. and J.J. Brown. 1997. Comparative toxicity of four insecticides, including imidacloprid and tebufenozide, to four aquatic arthropods. Env. Toxicol. Chemistry 16:(12):2494-2500.

Song, M.Y. and J.J. Brown. 1998. Osmotic effects as a factor modifying insecticide toxicity on Aedes and Artemia. Ecotox. Environ. Safety 41(2):195-202.

Stadler, T., Gines, D.M. and M. Buteler. 2003. Long-term toxicity assessment of imidacloprid to evaluate side effects on honey bees exposed to treated sunflower in Argentina. Bull. Insectol. 56(1): 77–81.

Stone, W.W., Gilliom, R.J. and J.D. Martin. 2014, An overview comparing results from two decades of monitoring for pesticides in the Nation’s streams and rivers, 1992–2001 and 2002–2011: U.S. Geological Survey Scientific Investigations Report 2014–5154, 23 p., http://dx.doi.org/10.3133/sir20145154.

Stoughton, S.J., Liber, K., Culp, J. and A. Cessna. 2008. Acute and chronic toxicity of imidacloprid to the aquatic invertebrates Chironomus tentans and Hyalella azteca under constant- and pulse-exposure conditions, Arch Environ Contam Toxicol 54: 662–673

Tasei, J.N., Lerin, J. and G. Ripault. 2000. Sublethal effects of imidacloprid on bumblebees, *Bombus terrestris* (Hymenoptera: Apidae), during a laboratory feeding test. Pest Manag. Sci. 56: 784–788.

Tasei, J.N., Ripault, G. and E. Rivault. 2001. Hazards of imidacloprid seed coating to *Bombus terrestris* (Hymenoptera: Apidae) when applied to sunflower. J. Econ. Entomol. 94(3): 623–627.

Tasei, J.N. 2002. Impact of agrochemicals on non-*Apis* bees. *In*: Honey bees: estimating the environmental impacts of chemicals. Eds. J. Devillers and M.H. Pham-Delegue. Taylor and

Francis Publishing. pp. 101–131.

Tomizawa, M. and J.E. Casida. 2003. Selective toxicity of neonicotinoids attributable to specificity of insect and mammalian nicotinic receptors. Annu. Rev. Entomol. 48:339-364.

USDA APHIS (United States Department of Agriculture, Animal and Plant Health Inspection Service). 2001. Enviornmental monitoring report, 2000 - 2001 Asian Longhorned Beetle Cooperative Eradication Program prepared by Enviornmental Monitoirng Team.

USDA APHIS. 2002. Asian Longhorned Beetle Cooperative Eradication Program In New York and Illinois environmental monitoring report 2002, prepared by Environmental Monitoring Team.

USDA APHIS. 2007. Environmental monitoring report: 2006 Asian Longhorned Beetle Cooperative Eradication Program for the active eradication region in Suffolk County, New York. 36 pp.

USDA APHIS. 2013. Asian Longhorned Beetle Cooperative Eradication Program in Clermont County, Ohio Revised Environmental Assessment, May 2013

USDA FS (United States Department of Agriculture, Forest Service). 2005. Imidacloprid: human health and ecological risk assessment (final report). SERA TR 05-43-24-03a. 283 pp.

USDA NRCS (United States Department of Agriculture. National Resource Conservation Service). 2014. Soil Bulk Density/Moisture/Aeration. http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_053260.pdf, last accessed 6/26/2014.

USEPA (United States Environmental Protection Agency). 1989. Risk assessment guidance for Superfund volume I, human health evaluation manual (Part A). interim final. Office of Emergency and Remedial Response, Washington, DC. EPA/540/1-89/002 December 1989.

USEPA. 2002. Supplemental guidance for developing soil screening levels for Superfund sites, Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Washington DC, OSWER 9355.4-24

USEPA. 2005. Imidacloprid; pesticide tolerances for emergency exemptions. Fed Regist. 70(16): 3634-3642. January 26, 2005. Available At: http://www.epa.gov/fedrgstr/EPA-PEST/2005/January/Day-26/p1438.htm

USEPA. 2008. OPP Ecotoxicity Database. Available at: http://www.ipmcenters.org/Ecotox/index.cfm.

USEPA. 2010. Imidacloprid: revised human-health risk assessment for proposed Section 3 seed treatment uses on bulb vegetables (Crop Group 3); cereal grains (Crop Group 15); root and tuber vegetables, except sugar beet (Crop Subgroup IB); tuberous and corm vegetables (Crop

Subgroup I C); leafy vegetables, except *Brassica* (Crop Subgroup 4A); *Brassica* vegetables (Crop Group 5); fruiting vegetables (Crop Group 8); cucurbit vegetables (Crop Group 9), and residential crack and crevice and bed-bug uses. Petition Nos: 8F7414, 8F7415, Section 3 Registration, Memorandum from Kramer et al. of Risk Assessment Branch, Health Effects Division to Kable Davis/Venus Eagle of Registration Division, dated March 16, 2010, available at: http://www.epa.gov/pesticides/chem_search/hhbp/R181434.pdf, last accessed 2/10/2015.

USEPA. 2011. Exposure factor handbook: 2011 Edition. Office of Research and Development, Washington, DC, National Center for Environmental Assessment, EPA/600/R-090/052F

USEPA. 2013. Chemicals evaluated for carcinogenic potential, Office of Pesticide Programs, http://npic.orst.edu/chemicals_evaluated.pdf, last accessed June 20, 2014

USEPA. 2014a. Overview of Risk Assessment in the Pesticide Program – Human Health Risk Assessment, Office of Pesticide Programs (http://www.epa.gov/pesticides/about/overview_risk_assess.htm) last accessed 3/21/2014.

USEPA. 2014b. Endocrine Disruptor Screening Program, Office of Chemical Safety and Pollution Prevention, Office of Science Coordination and Policy (http://www.epa.gov/endo/), last accessed 3/27/14.

USEPA. 2014c. List of contaminants and their maximum contaminant levels, Water: Drinking Water Contaminants. http://water.epa.gov/drink/contaminants/, last accessed 6/24/2014.

Van der Sluijs, J.P., Simon-Delso, N., Goulson, D., Maxim, L., Bonmatin, J.M and L.P Belzunces. 2013. Neonicotinoids, bee disorders and the sustainability of pollinator services. Current Opinion in Environmental Sustainability. 5:293–305.

Westwood, F., Bean, K.M., Dewar, A.M., Bromilow, R.H. and K. Chamberlain. 1998. Movement and persistence of (^{14}C) imidacloprid in sugar-beet plants following application to pelleted sugar-beet seed. Pesticide Science. 52(2):97-103.

Appendix G. Acronyms and Glossary

A

Acute	Having a sudden onset, sharp rise, and short course
Aesthetic	Pleasing in appearance, relating to or dealing with the beautiful.
ALB	Asian Longhorned Beetle
American Heritage River	Rivers designated by the U.S. Environmental Protection Agency to receive special attention to further three objectives: natural resource and environmental protection, economic revitalization, and historic and cultural preservation.
APHIS	Animal and Plant Health Inspection Service; an agency within the U.S. Department of Agriculture.
Application Rate	The amount of pesticide product applied per unit area.
AQI	Air Quality Index
Aquatic	Of, in, or relating to water.
Asian Longhorned Beetle High-Risk Host Tree	There are 13 genera of host trees that APHIS regulates for ALB and are considered high-risk hosts: *Acer* (maple and box elder), *Aesculus* (horse chestnut and buckeye), *Salix* (willow), *Ulmus* (elm), *Betula* (birch), *Albizia* (mimosa), *Celtis* (hackberry), *Cercidiphyllum* (katsura tree), *Fraxinus* (ash), *Koelreuteria* (golden raintree), *Platanus* (sycamore and London planetree), *Sorbus* (mountain ash), and *Populus* (poplar). These trees are hosts because ALB can derive its food supply from them and complete its life cycle.

B

Basal Area	The area of a given section of land that is occupied by the cross-section of tree trunks and stems at their base.
BC	Black carbon is a particulate matter (PM) air pollutant formed by the incomplete combustion of fossil fuels, biofuels, and biomass.
BCR	Bird Conservation Region

BEE Butoxyethyl ester

Best Management Practices A set of preventative measures used to protect soil and water quality from human disturbance.

Biodiversity The number and variety of different organisms in the ecological complexes of which those organisms occur naturally; the relative abundance and frequency of biological organisms within ecosystems.

Biological Control The reduction of pest populations by means of living organisms introduced or supplemented by humans; utilizes competitors, parasites, predators, or sterile insects to reduce pest populations (also called biocontrol).

Biological Opinion Document stating the opinions of the U.S. Fish and Wildlife Service and the National Marine Fisheries Service as to whether a Federal action is likely to jeopardize the continued existence of a threatened or endangered species or result in the destruction or adverse modification of critical habitat.

C

ºC Centigrade or Celsius

CAA Clean Air Act

Cambium A cylindrical layer of tissue in the stems and roots of many seed-bearing plants, consisting of cells that divide rapidly to form new layers of tissue.

Canopy The uppermost layer in a forest, formed by the crowns of the trees.

Carcinogen/ Carcinogenic A substance capable of causing cancer in living tissue.

CCD Colony Collapse Disorder

CEQ Council on Environmental Quality

CFR Code of Federal Regulations (U.S.)

CH_4 Methane

Chemical Sensitivity An adverse reaction(s) of a person or organism to ambient levels of toxic substance(s) contained in environmental media such as air, food, soil, and water.

Chronic Marked by long duration, by frequent recurrence over a long time, and often by slowly progressing seriousness.

CO Carbon Monoxide

CO_2 Carbon Dioxide

Combustion The process of burning something.

Community An assemblage of populations of plants, animals, bacteria, and fungi that live in an environment and interact with one another, forming a distinctive living system with its own composition, structure, environmental relations, development, and function; an association of interacting populations, usually defined by the nature of their interaction or the place in which they live.

Concentration The ratio of the mass or volume of a solute to the mass or volume of the solution or solvent; the amount of active ingredient or herbicide equivalent in a quantity of diluent (e.g., expressed as lb/gal, ml/liter, etc.), or an amount of a substance in a specified amount of medium (e.g., air and water).

Conifer A tree that bears cones and evergreen needlelike or scalelike leaves.

Cooperator State agriculture or forestry officials, contractors, and other entities collaborating with APHIS.

CWA Clean Water Act

CZMA Coastal Zone Management Act

D

Deciduous Something that sheds after a period of time, such as a tree or a shrub that sheds its leaves annually.

Dermal Absorption A route by which substances can enter the body through the skin**.**

E

EA Environmental Assessment; a concise public document which provides sufficient evidence and analysis for determining whether to prepare an EIS or Finding of No Significant Impact. It aids in compliance with the National Environmental Policy Act (NEPA) when no EIS is needed.

Ecosystem	A biological community of interacting organisms and their physical environment.
Ecotoxicity	The potential impact on an ecosystem from the release of toxic substances.
EIS	Environmental Impact Statement; a document prepared by a Federal agency in which anticipated environmental effects of alternative planned courses of action are evaluated; a detailed written statement as required by section 102(2)(C) of the National Environmental Policy Act (NEPA).
Eradication	The complete destruction or elimination of something.
EO	Executive Order
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act

F

ºF	Fahrenheit
Floodplain	An area of low-lying ground adjacent to a river, formed mainly of river sediments and subject to flooding.
Foliage	Plant leaves, collectively.
FONSI	Finding of no significant impact
FR	Federal Register
Frass	Debris or excrement produced by insects.
FS	Forest Service; an agency of the U.S. Department of Agriculture.
FWHA	U.S. Federal Highway Administration
FWS	Fish and Wildlife Service; an agency of the U.S. Department of Interior.

G

g	Gram
GHG	Greenhouse gas

Greenhouse Gas (GHG) — A gas that contributes to the greenhouse effect by absorbing infrared radiation.

Gt — Gigaton

H

Hardwood Tree — A flowering tree that produces seed within an enclosure such as a fruit. It produces hard, compact wood or timber. Examples include oak, cherry, maple, or mahogany.

Heartwood — The older, harder, nonliving, central wood of trees that is usually darker, denser, less permeable, and more durable than the surrounding sapwood.

Herbicide — A substance used to kill plants.

HHERA — Human Health and Ecological Risk Assessment

Hydrology — The science that studies the occurrence, circulation, distribution, and properties of the waters of the earth and its atmosphere.

Hypersensitivity — A state of altered reactivity in which the body reacts with an exaggerated immune response to what is perceived as a foreign substance.

I

Incineration — To burn or reduce to ashes.

Infestation — The state of being invaded or overrun by pests or parasites. It can also refer to the actual organisms living on or within a host.

Inhalation — The act of inhaling or breathing in.

Insecticide — A substance used to kill insects.

Invertebrate — An animal lacking a backbone, such as an insect, snail, mussel, worm, etc.

IPPC — International Plant Protection Convention

ISPM — International standards for phytosanitary measures

K

Keystone Species — A species whose presence and role within an ecosystem has a disproportionate effect on other organisms within the ecosystem.

Kg — Kilogram

L

Larva — The active immature form of an insect, especially one that differs greatly from the adult and forms the stage between egg and pupa.

L — Liter

M

Metabolite — Breakdown products of a chemical.

m — Meter

µ — Microgram

mg — Milligram

MOU — Memorandum of Understanding

Mutagen/ Mutagenic — An agent, such as radiation or a chemical substance, that causes genetic mutation.

N

N_2O — Nitrous oxide

NAAQS — National Ambient Air Quality Standards

Neonicotinoid — A class of neuro-active insecticides chemically similar to nicotine. The insecticide imidacloprid is a neonicotinoid.

Neotropical Bird — A bird that breeds in Canada and the United States in the summer, and spends winter in Central America, South America, or the Caribbean Islands.

NEPA — The National Environmental Policy Act of 1969 and subsequent amendments.

Neurotoxicity The ability of a drug or other agent to destroy or damage nervous tissue.

ng Nanogram

NMFS National Marine Fisheries Service

NO_3 Nitrate

NOAA National Oceanic and Atmospheric Administration

NOx Nitrogen oxide

NOEC No observable effect concentration; the highest tested concentration of a toxicant at which no adverse effects are observed on the test organisms at a specific time of observation.

NOEL No observable effect level; the highest dose levels at which there are no observable differences between the test and control populations.

NOI Notice of Intent

O

O_2 Nitrogen dioxide

O_3 Ozone

OCRM Office of Ocean and Coastal Resource Management

Outbreak A sudden increase in numbers of a harmful organism.

Oviposit/ Oviposition To lay eggs.

P

Pesticide A substance, including insecticides and herbicides, used for destroying insects or other organisms.

Permeability The ability of a substance to allow another substance to pass through it, especially the ability of a porous rock, sediment, or soil to transmit fluid through pores and cracks.

PM Particulate matter

$PM_{2.5}$ Particulate matter that is less than 2.5 micrometers

PM_{10} Particulate matter that is between 2.5 and 10 micrometers

Pollinator An insect or other organism that carries pollen from one flower to another.

PPE Personal protective equipment

ppm Parts per million

PPQ Plant Protection and Quarantine

Prophylactic A preventative measure.

Pupa An insect in its inactive immature form between larva and adult.

Pupate A larva becoming a pupa.

Q

Quarantined Area/ Quarantine A geographic area from which movement of regulated articles is restricted in order to prevent or reduce the human-assisted spread of a pest. APHIS establishes regulations that define the boundaries of quarantined areas. (Regulations for the ALB quarantined areas are located at 7 CFR § 301.51-3.)

R

Regulated Articles Articles that can harbor life stages of ALB. The regulated articles listed under the ALB quarantine (7 CFR § 301.51) include the beetle and all its life stages; firewood (all hardwood species, not restricted to ALB-host trees); green lumber, and other living, dead, cut, or fallen material, including nursery stock, logs, stumps, roots, branches, and debris from ALB-host trees of ½ inch or more in diameter.

RfD Chronic reference dose; the term preferred by EPA to express acceptable daily intake for humans; an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population, including sensitive subgroups, that is likely to be without an appreciable risk of deleterious effects during a lifetime.

Riparian Of or relating to wetlands adjacent to rivers and streams.

S

Salinity	The saltiness or dissolved salt content.
Scoping	An early and open process for determining the scope of issues to be addressed, and for identifying the significant issues related to a proposed action.
Sediment	Matter that settles to the bottom of a liquid.
SHPO	State Historic Preservation Office
Silviculture	The growing and cultivation of trees.
SIP	State implementation plan
SIT	Sterile insect technique
SMZ	Stream management zone
SO_2	Sulfur dioxide
SO_4	Sulfate
SO_X	Sulfur oxides
Soil Biota	All the organisms that spend a significant portion of their life cycle within a soil profile, or at the soil-litter interface.
Soil Compaction	The process in which a stress applied to a soil causes densification as air is displaced from the pores between the soil grains.
Soil Erosion	Removal of topsoil faster than the soil forming processes can replace it, due to natural, animal, and human activity.
Sublethal	Not sufficient to cause death.
Suburb/ Suburban	An outlying part of a city or town; a smaller community adjacent to or within commuting distance of a city.
Systemic Insecticide	An insecticide that is absorbed by the plant and moved throughout its tissues.

T

T&E Species	Threatened and endangered species
TEA	Triethylamine salt
Teratogen/ Teratogenic	An agent or factor that causes malformation of an embryo.
Terrestrial	Of, on, or relating to the earth.

U

Understory	A layer of vegetation beneath the main canopy of a forest.
Urban	Of, pertaining to, or designating a city or town; or living in a city.
U.S.C.	United States Code
USDA	United State Department of Agriculture
UV	Ultraviolet

V

Vertebrate	An animal with a backbone or spinal column, including mammals, birds, reptiles, amphibians, and fishes.
VOC	Volatile organic compound
Volatilize	Cause a substance to evaporate or disperse in vapor.

W

WPM	Wood packaging material

Appendix H. References

Aizen, M. A. and P. Feinsinger. 1994. Forest fragmentation, pollination, and plant reproduction in a Chaco dry forest, Argentina. Ecology 75:330-351.

Akbari, H., D. M. Kurn, S. E. Bretz, and J. W. Hanford. 1997. Peak power and cooling energy savings of shade trees. Energy and Buildings 25.

Alaux, C., J.-L. Brunet, C. Dussaubat, F. Mondet, S. Tchamitchan, M. Cousin, J. Brillard, A. Baldy, L. P. Belzunces, and Y. Le Conte. 2010. Interactions between *Nosema* microspores and a neonicotinoid weaken honeybees (*Apis mellifera*). Environmental Microbiology 12:774-782.

Allen-Wardell, G., P. Bernhardt, R. Bitner, A. Burquez, S. Buchmann, J. Cane, P. A. Cox, V. Dalton, P. Feinsinger, M. Ingram, D. Inouye, C. E. Jones, K. Kennedy, P. Kevan, H. Koopowitz, R. Medellin, S. Medellin-Morales, P. Nabhan, B. Pavlik, V. Tepedino, P. Torchio, and S. Walker. 1998. The potential consequences of pollinator declines on the conservation of biodiversity and stability of food crop yields. Conservation Biology 12:8-17.

Anatra-Cordone, M. and P. Durkin. 2005. Imidacloprid--Human Health and Ecological Risk Assessment--Final Report. United States Department of Agriculture, Forest Service.

Anderson, L. M. and H. K. Cordell. 1988. Influence of trees on residential property values in Athens, Georgia (U. S. A.): A survey based on actual sales prices. Landscape and Urban Planning 15:153-164.

APHIS. 2003. Importation of Solid Wood Packing Material, Final Environmental Impact Statement, August 2003. United States Department of Agriculture, Animal and Plant Health Inspection Service.

APHIS. 2007a. Asian Longhorned Beetle: Questions and Answers (available at: http://www.aphis.usda.gov/publications/plant_health/content/printable_version/faq_alb_07.pdf). United States Department of Agriculture, Animal and Plant Health Inspection Service, Plant Protection and Quarantine.

APHIS. 2007b. Importation of Solid Wood Packing Material, Supplemental to the Final Environmental Impact Statement, October 2007. United States Department of Agriculture, Animal and Plant Health Inspection Service.

APHIS. 2008. Asian longhorned beetle: Annotated host list. Page 2. United States Department of Agriculture, Animal and Plant Health Inspection Service, Plant Protection and Quarantine, Otis Pest Survey Detection and Exclusion Laboratory.

APHIS. 2009. USDA–APHIS Asian Longhorned Beetle Eradication Program— Decision Support for the APHIS Management Team. Page 97 pp. United States Department of Agriculture, Animal and Plant Health Inspection Service.

APHIS. 2013a. 2010-2012 Environmental Monitoring Report, Asian Longhorned Beetle Cooperative Eradication Program: For the Eradication Effort in Worcester and Boston, Massachusetts. United States Department of Agriculture, Animal and Plant Health Inspection Service, Plant Protection and Quarantine.

APHIS. 2013b. Asian Longhorned Beetle Cooperative Eradication Program in Clermont County, Ohio. May 2013 United States Department of Agriculture, Animal and Plant Health Inspection Service, Riverdale, MD.

Aust, W. M. and C. R. Blinn. 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982-2002). Water, Air, and Soil Pollution: Focus 4:5-35.

Ballard, T. M. 2000. Impacts of forest management on northern forest soils. Forest Ecology and Management 133:37-42.

Bancroft, J. and M. T. Smith. 2005. Dispersal and influences on movement for *Anoplophora glabripennis* calculated from individual mark-recapture. Entomologia Experimentalis et Applicata 116:83-92.

Barrios, E. 2007. Soil biota, ecosystem services and land productivity. Ecological Economics 64:269-285.

Batra, S. W. T. 1985. Red Maple (Acer rubrum L.), an Important Early Spring Food Resource for Honey Bees and Other Insects. Journal of the Kansas Entomological Society 58:169-172.

Beckett, K. P., P. H. Freer-Smith, and G. Taylor. 1998. Urban woodlands: their role in reducing the effects of particulate pollution. Environmental Pollution 99:347-360.

Bell, J. F., J. S. Wilson, and G. C. Liu. 2008. Neighborhood greenness and 2-year changes in body mass index of children and youth. Am J Prev Med 35:547-553.

Bell, J. L. and R. C. Whitmore. 2000. Bird nesting ecology in a forest defoliated by gypsy moths. Wilson Bulletin 112:524-531.

Bell, J. N. B. and M. Treshow. 2002. Air pollution and plant life. 2nd edition. John Wiley and Sons, New York.

Bréda, N., R. Huc, A. Granier, and E. Dreyer. 2006. Temperate forest trees and stands under severe drought: a review of ecophysiological responses, adaptation processes and long-term consequences. Ann. For. Sci. 63:625-644.

Bulman, L. S., M. O. Kimberley, and P. D. Gadgil. 1999. Estimation of the efficiency of pest detection surveys. New Zealand Journal of Forestry Science 29:102-115.

Bush, E. B., D. G. Neary, C. K. McMahon, and J. W. Taylor Jr. 1987a. Suitability of hardwoods treated with phenoxy and pyridine herbicides for use as firewood. Archives of Environmental Contamination and Toxicology 16:333-341.

Bush, P. B., J. W. Taylor, C. K. McMahon, and D. G. Neary. 1987b. Residues of lindane and chlorpyrifos in firewood and woodsmoke. J. Entomol. Sci. 22:131-139.

Bush, R. J., J. J. Bejune, B. G. Hansen, and P. A. Araman. 2002. Trends in the use of materials for pallets and other factors affecting the demand for hardwood products.*in* 30th Hardwood Symposium.

Cao, S., T. Tian, L. Chen, X. Dong, X. Yu, and G. Wang. 2010. Damage Caused to the Environment by Reforestation Policies in Arid and Semi-Arid Areas of China. AMBIO: A Journal of the Human Environment 39:279-283.

Cavey, J. F. 1998. Solid wood packing material from China. Initial pest risk assessment on certain wood boring beetles known to be associated with cargo shipments: Asian longhorned beetle (*Anoplophora glabripennis*), *Ceresium, Monochamus* and *Hesperophanes*. National Identification Services - SS - PPQ - APHIS - USDA, Riverdale, MD.

Chalfoun, A. D., F. R. I. Thompson, and M. J. Ratnaswamy. 2002. Nest predators and fragmentation: A review and meta-analysis. Conservation Biology 16:306-318.

Chen, X. D., E. Culbert, V. Hebert, and J. D. Stark. 2010. Mixture effects of the nonylphenyl polyethoxylate, R-11 and the insecticide, imidacloprid on population growth rate and other parameters of the crustacean, *Ceriodaphnia dubia*. Ecotoxicology and Environmental Safety 73:132-137.

Cook, M. 1990. Soil and water quality. AG-North Carolina Agricultural Extension Service, North Carolina State University.

Coulson, R. N., M. A. Pinto, M. D. Tchakerian, K. A. Baum, W. L. Rubink, and J. S. Johnston. 2005. Feral honey bees in pine forest landscapes of east Texas. Forest Ecology and Management 215:91-102.

Coulson, R. N. and F. M. Stephen. 2006. Impacts of insects in forest landscapes: Implications for forest health management. Pages 101-125 Invasive Forest Insects, Introduced Forest Trees, and Altered Ecosystems. Springer.

Cox, G. W. 1999. Eastern forests: The dark side of forest biodiversity. Pages 96-109 Alien species in North America and Hawaii: Impacts on natural ecosystems. Island Press.

Cunningham, W. P., B. W. Saigo, and M. A. Cunningham. 2001. Environmental science: A global concern. McGraw-Hill Boston, MA.

Dayton, P. K. 1972. Toward an understanding of community resilience and the potential effects of enrichments to the benthos at McMurdo Sound, Antarctica. Pages 81-96 *in* Proceedings of the colloquium on conservation problems in Antarctica. Allen Press Lawrence, Kansas, USA.

Desneux, N., A. Decourtye, and J.-M. Delpuech. 2007. The sublethal effects of pesticides on beneficial arthropods. Annual Review of Entomology 52:81-106.

Dodds, K. J., H. M. Hull-Sanders, N. W. Siegert, and M. J. Bohne. 2014. Colonization of three maple species by Asian longhorned beetle, *Anoplophora glabripennis*, in two mixed-hardwood forest stands. Insects 5:105-119.

Dombrow, J. M. R. and C. F. Sirmans. 2000. The market value of mature trees in single-family housing markets. The Appraisal Journal 68:39-43.

Donovan, G. H. and D. T. Butry. 2010. Trees in the city: Valuing street trees in Portland, Oregon. Landscape and Urban Planning 94:77-83.

Donovan, G. H., D. T. Butry, Y. L. Michael, J. P. Prestemon, A. M. Liebhold, D. Gatziolis, and M. Y. Mao. 2013. The relationship between trees and human health. American Journal of Preventive Medicine 44:139-145.

Donovan, G. H., Y. L. Michael, D. T. Butry, A. D. Sullivan, and J. M. Chase. 2011. Urban trees and the risk of poor birth outcomes. Health & Place 17:390-393.

Donovan, T. M., F. R. I. Thompson, J. Faaborg, and J. R. Probst. 1995. Reproductive success of migratory birds in habitat sources and sinks. Conservation Biology 9:1380-1395.

Doran, J. W. and T. B. Parkin. 1994. Defining and assessing soil quality. Special Publication 35.*in* J. W. Doran, D. C. Coleman, D. F. Bezdicek, and B. A. Stewart, editors. Defining soil quality for a sustainable environment. Soil Science Society of America, Madison, WI:.

Dosskey, M. G., P. Vidon, N. P. Gurwick, C. J. Allan, T. P. Duval, and R. Lowrance. 2010. The role of riparian vegetation in protection and improving chemical water quality in streams. Journal of the American Water Resources Association 46:261-277.

Doupé, R. G., J. Mitchell, M. J. Knott, A. M. Davis, and A. J. Lymbery. 2010. Efficacy of exclusion fencing to protect ephemeral floodplain lagoon habitats from feral pigs (Sus scrofa). Wetlands Ecology and Management 18:69-78.

DPA. 2013. Maine Office of Tourism, Visitor Tracker Research, 2012 Calendar Year Annual Report. DPA, Kennebunk, ME.

Durkin, P. 2003. Triclopyr: Revised Human Health and Ecological Risk Assessments - Final Report. Page 264. United States Department of Agriculture, Forest Service.

Durkin, P. and M. Follansbee. 2004. Imazapyr - Human Health and Ecological Risk Assessment - Final Report. United States Department of Agriculture, Forest Service.

Dwyer, J., E. G. McPherson, H. W. Schroeder, and R. A. Rowntree. 1992. Assessing the benefits and costs of the urban forest. Journal of Arboriculture 18:227-234.

Ellison, A. M., M. S. Bank, B. D. Clinton, E. A. Colburn, K. Elliot, C. R. Ford, D. R. Foster, B. D. Kloeppel, J. D. Knoepp, G. M. Lovett, J. Mohan, D. A. Orwig, N. L. Rodenhouse, W. V. Sobczak, K. A. Stinson, J. K. Stone, C. M. Swan, J. Thompson, B. Von Holle, and J. R. Webster. 2005. Loss of foundation species: Consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment 3:479-486.

Englin, J. E., J. M. McDonald, and K. Moeltner. 2006. Valuing ancient forest ecosystems: An analysis of backcountry hiking in Jasper National Park. Ecological Economics 57:665-678.

EPA. 1998. Reregistration eligibility decision (RED): Triclopyr (EPA 738-R-98-011). Page 285. United States Environmental Protection Agency.

EPA. 2008. Reducing Urban Heat Islands: Compendium of Strategies. Available at: http://www.epa.gov/heatisland/resources/compendium.htm. United States Environmental Protection Agency (EPA).

EPA. 2012a. National Ambient Air Quality Standards (NAAQS).

EPA. 2012b. Our Nation's Air: Status and Trends through 2010. EPA-454/R-12-001. Last accessed May 21, 2014 at: http://epa.gov/airtrends/2011/report/fullreport.pdf. U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards.

EPA. 2013a. The Green Book Nonattainment Areas for Criteria Pollutants. Last accessed April 24, 2014 at: http://www.epa.gov/airquality/greenbook/index.html.

EPA. 2013b. Water Quality Assessment and TMDL Information: National Summary of State Information. United States Environmental Protection Agency.

EPA. 2013c. Water: Estuaries and Coastal Watersheds, National Estuary Program Overview. June 10, 2013. Available at: http://water.epa.gov/type/oceb/nep/, last accessed January 10, 2014. United States Environmental Protection Agency (EPA).

EPA. 2014a. Ecological Fate and Effects Division (EFED) Pesticide Ecotoxicity Database. United States Environmental Protection Agency, Office of Pesticide Programs.

EPA. 2014b. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2012 (EPA 430-R-14-003). United States Environmental Protection Agency, Washington, D.C.

EPA. 2014c. Watershed Assessment, Tracking and Environmental Results (Available at: http://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T#imp_water_by_state). United States Environmental Protection Agency.

EPA. n.d. Volatile organic compounds emissions. United States Environmental Protection Agency.

EPPO. 1999. EPPO data sheets on quarantine pests: *Anoplophora glabripennis*. European Plant Protection Organization.

Everson-Rose, S. A. and T. T. Lewis. 2005. Psychosocial factors and cardiovascular diseases. Annual Review of Public Health 26:469-500.

FAO. 2002 International Standards for Phytosanitary Measures (ISPM) No. 15. Guidelines for Regulating Wood Packaging Material in International Trade. .

Farrell, M. L. and B. F. Chabot. 2012. Assessing the growth potential and economic impact of the U.S. maple syrup

industry. Journal of Agriculture, Food Systems, and Community Development 2:11-27.

FAS. 2009. U. S. trade internet system. United States Department of Agriculture, Foreign Agriculture Service.

FS. 2009a. Susceptibility potential for *Anoplophora glabripennis* (Asian longhorned beetle). United States Department of Agriculture, Forest Service.

FS. 2009b. Timber Product Output (TPO) reports. United States Department of Agriculture, Forest Service.

FS. 2014. Forest Inventory and Analysis Glossary. Last accessed July 1, 2014 at http://www.nrs.fs.fed.us/fia/data-tools/state-reports/glossary/default.asp. United States Department of Agriculture, Forest Service.

FWS. 2006. A blueprint for the design and delivery of bird conservation in the Atlantic northern forest. Page 346. United States Department of the Interior, Fish and Wildlife Service.

FWS. 2007. Indiana Bat (*Myotis sodalis*) Draft Recovery Plan: First Revision. United States Department of the Interior, U.S. Fish and Wildlife Service, Great Lakes-Big Rivers Region - Region 3, Fort Snelling, Minnesota.

Garrison, G. A., A. J. Bjugstad, D. A. Duncan, M. E. Lewis, and D. R. Smith. 1977. Vegetation and environmental features of forest and range ecosystems. Page 475 *in* U.-F. A. Handbook, editor. U. S. Government Printing Office.

Gill, R. J., O. Ramos-Rodriguez, and N. E. Raine. 2012. Combined pesticide exposure severely affects individual- and colony-level traits in bees. Nature 491:105-108.

Gilliom, R. J., J. E. Barbash, C. G. Crawford, P. A. Hamilton, J. D. Martin, N. Nakagaki, L. H. Nowell, J. C. Scott, P. E. Stackelberg, G. P. Thelin, and D. M. Wolock. 2007. The Quality of our Nation's Waters: Pesticides in the Nation's Streams and Ground Water, 1992-2001. Circular 1291. United States Department of the Interior, U.S. Geological Survey.

Grahn, P. and U. A. Stigsdotter. 2003. Landscape planning and stress. Urban Forestry & Urban Greening 2:1-18.

Grant, R. H., G. M. Heisler, and W. Gao. 2002. Estimation of pedestrian level UV exposure under trees. Photochemistry and Photobiology 75:369-376.

GTIS. 2013. Global Trade Atlas. Global Trade Information Services (GTIS), Inc.

Guite, H. F., C. Clark, and G. Ackrill. 2006. The impact of the physical and urban environment on mental wellbeing. Public Health 120:1117-1126.

Haack, R. A., L. S. Bauer, R. Gao, J. J. McCarthy, D. L. Miller, T. R. Petrice, and T. M. Poland. 2006. *Anoplophora glabripennis* within-tree distribution, seasonal development, and host suitability in China and Chicago. The Great Lakes Entomologist 39:169-183.

Haack, R. A., F. Hérard, J. Sun, and J. J. Turgeon. 2010. Managing invasive populations of Asian longhorned beetle and citrus longhorned beetle: A worldwide perspective. Annual Review of Entomology 55:521-546.

Haack, R. A., K. R. Law, V. C. Mastro, S. H. Ossenburgen, and B. J. Raimo. 1997. New York's battle with the Asian long-horned beetle. Journal of Forestry 95:11-15.

Hajek, A. E. 2010. Asian Longhorned Beetle: Ecology and Control. Pages 21-24 *in* D. Pimental, editor. Encyclopedia of Pest Management. CRC Press.

Hartig, T., G. W. Evans, L. D. Jamner, D. S. Davis, and T. Gärling. 2003. Tracking restoration in natural and urban field settings. Journal of Environmental Psychology 23:109-123.

HCN and DAC. 2004. Nature and Health: The influence of nature on social, psychological and physical well-being. The Hague: Health Council of the Netherlands and RMNO; publication no. 2004/09E; RMNO publication nr A02ae. Health Council of the Netherlands (HCN) and Dutch Advisory Council (DAC) for Research on Spatial Planning Nature and the Environment.

Heisler, G. and R. H. Grant. 2000. Ultraviolet Radiation, Human Health, and the Urban Forest.

Heisler, G. M. 2010. UV Radiation in Global Climate Change: Measurements, Modeling and Effects on Ecosystems. Page 544 pp. *in* W. Gao, D. L. Schmoldt, and J. R. Slusser, editors. Urban Forest Influences on Exposure to UV Radiation and Potential Consequences for Human Health (Chapter 12). Tsinghua University Press, Beijing.

Heisler, G. M., R. H. Grant, and W. Gao. 2003. Individual- and scattered-tree influences on ultraviolet irradiance. Agricultural and Forest Meteorology 120:113-126.

Henley, W. F., M. A. Patterson, R. J. Neves, and A. D. Lemly. 2000. Effects of sedimentation and turbidity on lotic food webs: A concise review for natural resource managers. Reviews in Fisheries Science 8:125-139.

Hladik, M. L., D. W. Kolpin, and K. M. Kuivila. 2014. Widespread occurrence of neonicotinoid insecticides in streams in a high corn and soybean producing region, USA. Environmental Pollution 193:189-196.

Holmes, T. P., E. A. Murphy, and K. P. Bell. 2006. Exotic forest insects and residential property values. Agricultural and Resource Economics Review 35:155-166.

Holmstrup, M., A.-M. Bindesbøl, G. J. Oostingh, A. Duschl, V. Scheil, H.-R. Köhler, S. Loureiro, A. M. V. M. Soares, A. L. G. Ferreira, C. Kienle, A. Gerhardt, R. Laskowski, P. E. Kramarz, M. Bayley, C. Svendsen, and D. J. Spurgeon. 2010. Interactions between effects of environmental chemicals and natural stressors: A review. Science of The Total Environment 408.

Hornbeck, J. W., M. B. Adams, E. S. Corbett, E. S. Verry, and J. A. Lynch. 1993. Long-term impacts of forest treatments on water yield: a summary for northeastern USA. Journal of Hydrology 150:323-344.

Horsley, S. B., R. P. Long, S. W. Bailey, R. A. Hallett, and P. M. Wargo. 2002. Health of eastern North American sugar maple forests and factors affecting decline. Northern Journal of Applied Forestry 19:34-44.

Hu, J., S. Angeli, S. Schuetz, Y. Luo, and A. E. Hajek. 2009. Ecology and management of exotic and endemic Asian longhorned beetle *Anoplophora glabripennis*. Agricultural and Forest Entomology 11:359-375.

Huang, Y. J., H. Akbari, H. Taha, and A. H. Rosenfeld. 1987. The potential of vegetation in reducing summer cooling loads in residential buildings. Journal of Climate and Applied Meteorology 26:1103–1116.

Humpel, N., N. Owen, and E. Leslie. 2002. Environmental factors associated with adults' participation in physical activity. Am. J. Prev. Med. 22:188- 199.

Iverson, L. R. and A. M. Prasad. 1998. Predicting abundance of 80 tree species following climate change in the eastern United States. Ecological Monographs 68:465-485.

Iverson, L. R., A. M. Prasad, S. N. Matthews, and M. Peters. 2008. Estimating potential habitat for 134 eastern US tree species under six climate scenarios. Forest Ecology and Management 254:390-406.

Jacobson, M. G., R. C. Ready, and C. D. Canham. 2012. Potential ecological and economic impacts of a regional outbreak of the Asian longhorned beetle. Final report for USDA-APHIS Award 10-8100-1442-CA. The Pennsylvania State University, State College, PA.

Johnson, J. 2012. The role of pesticides on honey bee health and hive maintenance with an emphasis on the neonicotinoid, imidacloprid. University of Maryland, Baltimore, Maryland.

Jones, K. L., G. C. Poole, J. L. Meyer, W. Bumback, and E. A. Kramer. 2006. Quantifying expected ecological response to natural resource legislation: a case study of riparian buffers, aquatic habitat, and trout populations. [Online.] Available: http://www.ecologyandsociety.org/vol11/iss2/art15/. Ecology and Society 11.

Kane, B. and J. Kirwan. 2009. Value, Benefits, and Costs of Urban Trees. Virginia Cooperative Extension Publication 420–181. Last accessed March 31, 2014: http://www.pubs.ext.vt.edu/420/420-181/420-181_pdf.pdf

Karlen, D., M. Mausbach, J. Doran, R. Cline, R. Harris, and G. Schuman. 1997. Soil quality: A concept, definition, and framework for evaluation (a guest editorial). Soil Science Society of America Journal 61:4-10.

Kaushal, S. S., G. E. Likens, N. A. Jaworski, M. L. Pace, A. M. Sides, D. Seekell, K. T. Belt, D. H. Secor, and R. L. Wingate. 2010. Rising stream and river temperatures in the United States. Frontiers in Ecology and the Environment 8:461-466.

Keena, M. 2002. *Anoplophora glabripennis* (Coleoptera: Cerambycidae) fecundity and longevity under laboratory conditions: comparison of populations from New York and Illinois on *Acer saccharum*. Environmental Entomology 3:490-498.

Keena, M. 2006. Effects of temperature on *Anoplophora glabripennis* (Coleoptera: Cerambycidae) adult survival, reproduction, and egg hatch. Environmental Entomology 35:912-921.

Keena, M. 2008. Phenology of the Asian longhorned beetle under simulated annual environmental fluctuations. United States Department of Agriculture.

Kim, H. and J. Bernstein. 2009. Air pollution and allergic disease. Current Allergy and Asthma Reports 9:128-133.

Kizlinski, M. L., D. A. Orwig, R. C. Cobb, and D. R. Foster. 2002. Direct and indirect ecosystem consequences of an invasive pest on forest dominated by eastern hemlock Jounal of Biogeography 29:1489-1503.

Klein, A. M., B. E. Vaissière, J. H. Cane, I. Ingolf Steffan-Dewenter, S. A. Cunningham, C. Kremen, and T. Tscharntke. 2007. Importance of pollinators in changing landscapes for world crops. Proceedings of the Royal Society B-Biological Sciences 274:303-313.

Klotzback, J. and P. Durkin. 2004. Metsulfuron Methyl - Human Health and Ecological Risk Assessment - Final Report (SERA TR 04-43-17-01c). United States Department of Agriculture, Forest Service.

Knoepp, J. D. and B. D. Clinton. 2009. Riparian zones in southern Appalachian headwater catchments: Carbon and nitrogen responses to forest cutting. Forest Ecology and Management 258:2282-2293.

Koide, R. T. and T. Wu. 2003. Ectomycorrhizas and retarded decomposition in a *Pinus resinosa* plantation. New Phytologist 158:401-407.

Kraemer, M. E. and F. D. Favi. 2005. Flower phenology and pollen choice of Osmia lignaria (Hymenoptera: Megachilidae) in Central Virginia. Environmental Entomology 34:1593-1605.

Kremen, C., N. M. Williams, M. A. Aizen, B. Gemmill-Herren, G. LeBuhn, R. Minckley, L. Packer, S. G. Potts, T. a. Roulston, I. Steffan-Dewenter, D. P. Vázquez, R. Winfree, L. Adams, E. E. Crone, S. S. Greenleaf, T. H. Keitt, A.-M. Klein, J. Regetz, and T. H. Ricketts. 2007. Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change. Ecology Letters 10:299-314.

Kreutzweiser, D., K. Gooda, D. Chartranda, T. Scarrb, and D. Thompson. 2007. Non-target effects on aquatic decomposer organisms of imidacloprid as a systemic insecticide to control emerald ash borer in riparian trees. Ecotoxicology and Environmental Safety 68:315-325.

Kreutzweiser, D. P., K. P. Good, D. T. Chartrand, T. A. Scarr, and D. G. Thompson. 2008a. Are leaves that fall from imidacloprid-treated maple trees to control Asian longhorned beetles toxic to non-target decomposer organisms? Journal of Environmental Quality 37:639-646.

Kreutzweiser, D. P., K. P. Good, D. T. Chartrand, T. A. Scarr, and D. G. Thompson. 2008b. Toxicity of the systemic insecticide, Imidacloprid, to forest stream insects and microbial communities. Bull. Environ. Contam. Toxicol. 80:211-214.

Kreutzweiser, D. P., D. G. Thompson, and T. A. Scarr. 2009. Imidacloprid in leaves from systemically treated trees may inhibit litter breakdown by non-target invertebrates. Ecotoxicology and Environmental Safety 72:1053-1057.

Kuo, F. and W. Sullivan. 2001a. Aggression and violence in the inner city : Effects of environment via mental fatigue. Environment and Behavior 33:543-571.

Kuo, F. E. and W. C. Sullivan. 2001b. Environment and crime in the inner city: does vegetation reduce crime? Environ Behav 33:343-367.

Langmann, B., B. Duncan, C. Textor, J. Trentmann, and G. R. van der Werf. 2009. Vegetation fire emissions and their impact on air pollution and climate. Atmospheric Environment 43:107-116.

Laskowski, R., A. J. Bednarska, P. E. Kramarz, S. Loureiro, V. Scheil, J. Kudłek, and M. Holmstrup. 2010. Interactions between toxic chemicals and natural environmental factors — A meta-analysis and case studies. Science of The Total Environment 408:3763-3774.

Lee, J., Park, B. J., Y. Tsunetsugu, T. Kagawa, and Y. Miyazaki. 2009. Restorative effects of viewing real forest landscapes, based on a comparison with urban landscapes. Scandinavian Journal of Forest Research 24:227-234.

Lee, P., C. Smyth, and S. Boutin. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. Journal of Environmental Management 70:165-180.

Lingafelter, S. W. and E. R. Hoebeke. 2002. Revision of *Anoplophora* (Coleoptera: Cerambycidae). The Entomological Society of Washington, Washington, D.C.

Logan, J. A., J. Regniere, and J. A. Powell. 2003. Assessing the impacts of global warming on forest pest dynamics. Frontiers in Ecology and the Environment 1:130-137.

Loureiro, S., C. Svendsen, A. Ferreira, L. G., C. Pinheiro, F. Ribeiro, and A. M. V. M. Soares. 2010. Toxicity of three binary mixtures to *Daphnia manga*: Comparing chemical modes of action and deviations from conceptual models. Environmental Toxicology and Chemistry 29:1716-1726.

Lovasi, G. S., J. W. Quinn, K. M. Neckerman, M. S. Perzanowski, and A. Rundle. 2008. Children living in areas with more street trees have lower prevalence of asthma. J Epidemiol Community Health 62:647-649.

Lovett, G. M., C. D. Canham, M. A. Arthur, K. C. Weathers, and R. D. Fitzhugh. 2006. Forest ecosystem responses to exotic pests and pathogens in eastern North America. Bioscience 56:395-405.

Lovett, G. M., L. M. Christenson, P. M. Groffman, C. G. Jones, J. E. Hart, and M. J. Mitchell. 2002. Insect defoliation and nitrogen cycling in forests. Bioscience 52:335-341.

Lovett, G. M. and M. J. Mitchell. 2004. Sugar maple and nitrogen cycling in the forests of eastern North America. Frontiers in Ecology and the Environment 2:81-88.

Lovett, G. M. and H. Rueth. 1999. Soil nitrogen transformation in beech and maple stands along a nitrogen deposition gradient. Ecological Applications 9:1330-1344.

Lu, C., K. M. Warchol, and R. A. Callahan. 2014. Sub-lethal exposure to neonicotinoids impaired honey bees winterization before proceeding to colony collapse disorder. Bulletin of Insectology 67:125-130.

Lucas, S. R. and T. A. E. Platts-Mills. 2005. Physical activity and exercise in asthma: Relevance to etiology and treatment. J. Allergy Clin. Immunol. 115:928-934.

McDaniel, P. 2006. The twelve soil orders: soil taxonomy (Last accessed June 30, 2014 at http://www.cals.uidaho.edu/soilorders/). University of Idaho, College of Agricultural and Life Sciences.

McMahon, C. K., H. B. Clements, P. B. Bush, D. G. Neary, and J. W. Taylor. 1985. Pesticides released from burning treated wood. Pages 145-152 *in* Eighth Conference on Fire and Forest Meteorology, Detroit, Michigan.

McPherson, E. G. 2005. Tress with benefits. American Nurseryman 201:34-40.

McPherson, E. G. and J. R. Simpson. 1999. Carbon dioxide reduction through urban forestry: Guidelines for professional and volunteer tree planters. Gen. Tech. Rep. PSWGTR-171. Albany, CA: . United States Department of Agriculture, Forest Service, Pacific Southwest Research Station.

McPherson, G. and J. R. Simpson. 1995. Shade trees as a demand-side resource. Last accessed July 2, 2014 at http://treebenefits.terrasummit.com/Documents/Air_Quality/Shade%20Trees%20as%20a%20Demand-Side%20Resource.pdf. Home Energy Magazine Online.

Mehrhoff, L. A. 1989. The dynamics of declining populations of an endangered orchid, *Isoria medoloides*. Ecology 70:783-786.

Meiners, S. J., S. T. A. Pickett, and M. L. Cadenasso. 2002. Exotic plant invasions over 40 years of old field successions: Community patterns and associations. Ecography 25:215-223.

Morales, D., B. N. Boyce, and R. J. Favretti. 1976. The contribution of trees to residential property value: Manchester, Connecticut. Valuation October/November:26-43.

Mroz, G., M. Jurgensen, and D. Frederick. 1985. Soil nutrient changes following whole tree harvesting on three northern hardwood sites. Soil Science Society of America Journal 49:1552-1557.

Naiman, R. J. and H. Decamps. 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics 28:621-658.

NASS. 2014. Maple syrup production. United States Department of Agriculture, National Agricultural Statistics Service, Harrisburg, PA.

NCFS. n.d. Riparian and Wetland Tree Planting Pocket Guide for North Carolina. Second Edition. Last accessed May 6, 2014: http://ncforestservice.gov/publications/WQ0206.pdf. North Carolina Forest Service.

NHDES. 2008. A field guide to common aquatic and riparian plants of New Hampshire. Second edition. Last accessed May 6, 2014: http://des.nh.gov/organization/commissioner/pip/publications/wd/documents/wd-06-34.pdf. New Hampshire Department of Environmental Services.

NHNHB. 2008. New Hampshire natural heritage bureau. Rare plants, rare animals, and exemplary natural communities in New Hampshire towns. Page 183 *in* D.-D. o. F. Lands, editor.

Nowak, D. J. 2002. The effects of urban trees on air quality. United States Department of Agriculture, Forest Service.

Nowak, D. J., K. L. Civerolo, S. Trivikrama Rao, S. Gopal, C. J. Luley, and D. E. Crane. 2000. A modeling study of the impact of urban trees on ozone. Atmospheric Environment 34:1601-1613.

Nowak, D. J., D. E. Crane, and J. C. Stevens. 2006. Air pollution removal by urban trees and shrubs in the United States. Urban Forestry and Urban Greening 4:115-123.

Nowak, D. J., R. E. Hoehn, D. E. Crane, J. C. Stevens, and C. Leblanc Fisher. 2010. Assessing urban forest effects and values, Chicago's urban forest. Resour. Bull. NRS-37. Available at: http://www.itreetools.org/resources/reports/Chicago's%20Urban%20Forest.pdf. U.S. Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, PA.

Nowak, D. J., R. E. Hoehn, D. E. Crane, J. C. Stevens, and J. T. Walton. 2007. Assessing urban forest effects and values, New York City's urban forest. Resour. Bull. NRS-9. Available at: http://www.milliontreesnyc.org/download/pdf/ufore_study.pdf. U.S. Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, PA.

Nowak, D. J., J. E. Pasek, R. A. Sequeira, D. E. Crane, and V. C. Mastro. 2001. Potential effect of *Anoplophora glabripennis* (Coleoptera: Cerambycidae) on urban trees in the United States. Journal of Economic Entomology 94:116-122.

NRC. 2007. Status of pollinators in North America. National Academy Press, Washington, DC.

Ohio Department of Natural Resources. nd. Best Management Practices for Logging Operations: Factsheet. Ohio Department of Natural Resources, Division of Forestry, Columbus, Ohio.

Oliver, C. D. and B. C. Larson. 1996. Forest Stand Dynamics. John Wiley and Sons, Inc., New York.

Orwig, D. A., R. C. Cobb, A. W. D'Amato, M. L. Kizlinski, and D. R. Foster. 2008. Multi-year ecosystem response to hemlock woolly adelgid infestation in southern New England forests. Canadian Journal of Forestry Research 38:834-843.

Orwig, D. A. and D. R. Foster. 1998. Forest response to the introduced hemlock woolly adelgid in southern New England, USA. Torrey Botanical Society 125:60-73.

Pavlaki, M. D., R. Pereira, S. Loureiro, and A. M. V. M. Soares. 2011. Effects of binary mixtures on the life traits of *Daphnia magna*. Ecotoxicology and Environmental Safety 74:99-110.

Perkins, D. W. and M. L. J. Hunter. 2006. Effects of riparian timber management on amphibians in Maine. Journal of Wildlife Management 70:657-670.

Peterson, A. T., R. Scachetti-Pereira, and W. W. Hargrove. 2004. Potential geographic distribution of *Anoplophora glabripennis* (Coleoptera: Cerambycidae) in North America. American Midland Naturalist 151:170-178.

Petranka, J. W. and C. K. Smith. 2005. A functional analysis of streamside habitat use by southern Appalachian salamanders: Implications for riparian forest management. Forest Ecology and Management 210:443-454.

Pettis, J. S. and K. S. Delaplane. 2010. Coordinated responses to honey bee decline in the USA. Apidologie 41:256-263.

Pettis, J. S., D. vanEngelsdorp, J. Johnson, and G. Dively. 2012. Pesticide exposure in honey bees results in increased levels of the gut pathogen *Nosema*. Naturwissenschaften 99:153-158.

Phillips, P. J. and R. W. Bode. 2004. Pesticides in surface water runoff in south-eastern New York State, USA: Seasonal and stormflow effects on concentrations. Pest Management Science 60:531-543.

Pineda, E., C. Moreno, F. Escobar, and G. Halffter. 2005. Frog, bat, and dung beetle diversity in the cloud forest and coffee agroecosystems of Veracruz, Mexico. Conservation Biology 19:400-410.

Poland, T. M., R. A. Haack, T. R. Petrice, D. L. Miller, L. S. Bauer, and R. Gao. 2006. Field evaluations of systemic insecticides for control of *Anoplophora glabripennis* (Coleoptera: Cerambycidae) in China. Journal of Economic Entomology 99:383-392.

Poland, T. M. and D. G. McCullough. 2006. Emerald ash borer: Invasion of the urban forest and the threat to North America's ash resource. Journal of Forestry:118-124.

Pope III, C. A., R. T. Burnett, G. D. Thurston, M. J. Thun, E. E. Calle, D. Krewski, and J. J. Godleski. 2003. Cardiovascular mortality and long-term exposure to particulate air pollution: Epidemiological evidence of general pathophysiological pathways of disease. Circulation 109:71-77.

Potts, S. G., J. C. Biesmeijer, C. Kremen, P. Neumann, O. Schweiger, and W. E. Kunin. 2010. Global pollinator declines: Trends, impacts and drivers. Trends in Ecology and Evolution 26:345-353.

Price, J. I., D. W. McCollum, and R. P. Berrens. 2010. Insect infestation and residential property values: A hedonic analysis of the mountain pine beetle epidemic Forest Policy and Economics 12:415-422.

Richter, B. D., D. P. Braun, M. A. Mendelson, and L. L. Master. 1997. Threats to imperiled freshwater fauna. Conservation Biology 11:1081-1093.

Sanchez-Bayo, F. and K. Goka. 2014. Pesticide residues and bees – A risk assessment. PLoS One 9:1-16.

Sarajevs, V. 2011. Health Benefits of Street Trees. Forest Research.

Sawyer, A. J. 2006. Infestation dynamics of the Asian long-horned beetle in the United States.*in* Proceedings 17th U. S. Department of Agriculture Interagency Research Forum on Gypsy Moth and other invasive species 2005.

Scarpa, R., W. G. Hutchinson, S. M. Chilton, and J. Buongiorno. 2000. Importance of forest attributes in the willingness to pay for recreation: a contingent valuation study of Irish forests. Forest Policy and Economics 1:315-329.

Schmuck, R., R. Schöning, A. Stork, and O. Schramel. 2001. Risk posed to honeybees (*Apis mellifera*, Hymenoptera) by an imidacloprid seed dressing of sunflowers. Pest Management Science 57:225-238.

Shifley, S. R., F. X. Aguilar, N. Song, S. I. Stewart, D. J. Nowak, D. D. Gormanson, W. K. Moser, S. Wormstead, and E. J. Greenfield. 2012. Forests of the Northern United States. Gen. Tech. Rep. NRS-90. Last accessed May 29, 2014 http://www.fs.fed.us/nrs/pubs//gtr/gtr_nrs90.pdf. United States Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, PA.

Simpson, J. R. 1998. Urban forest impacts on regional cooling and heating energy use: Sacramento County case study. Journal of Arboriculture 24.

Smith, M. T., J. Bancroft, G. Li, R. Gao, and S. Teale. 2001. Dispersal of *Anoplophora glabripennis* (Cerambycidae). Environmental Entomology 30:1036-1040.

Smith, M. T., J. Bancroft, and J. Tropp. 2002. Age-specific fecundity of *Anoplophora glabripennis* (Coleoptera: Cerambycidae) on three tree species infested in the United States. Environmental Entomology 31:76-83.

Smith, M. T., P. C. Tobin, J. Bancroft, G. Li, and R. Gao. 2004. Dispersal and spatiotemporal dynamics of Asian longhorned beetle (Coleoptera: Cerambycidae) in China. Environmental Entomology 33:435-442.

Smith, W. B., P. D. Miles, C. H. Perry, and S. A. Pugh. 2007. Forest Resources of the United States, 2007 (last accessed June 30, 2014 at http://www.fs.fed.us/nrs/pubs/gtr/gtr_wo78.pdf). United States Department of Agriculture, Forest Service.

Smock, L. A. and C. M. MacGregor. 1988. Impact of the American chestnut blight on aquatic shredding macroinvertebrates. Journal of the North American Benthological Society 7:212-221.

Starner, K. and K. S. Goh. 2012. Detections of the neonicotinoid insecticide imidacloprid in surface waters of three agricultural regions of California, USA, 2010–2011. Bull. Environ. Contam. Toxicol. 88:316-321.

Stern, A. C. 1977. Air Pollution: The effects of air pollution. Elsevier.

Suarez, A. V., K. S. Pfening, and S. K. Robinson. 1997. Nesting success of a disturbance-dependent songbird on different kinds of edges. Conservation Biology 11:928–935.

Sydnor, T. D., M. Bumgardner, and A. Todd. 2007. The potential economic impacts of emerald ash borer (*Agrilus planipennis*) on Ohio, U.S., communities. Arboriculture and Urban Forestry 33:48-54.

Takano, T., K. Nakamura, and M. Watanabe. 2002. Urban residential environments and senior citizens' longevity in megacity areas: the importance of walkable green spaces. J Epidemiol Community Health 56:913-918.

Tasei, J. N. 2002. Impact of agrochemicals on non-*Apis* bees. Pages 101-131 *in* J. Devillers and M.-H. Pham-Delegue, editors. Honey Bees: Estimating the Environmental Impacts of Chemicals. CRC Press.

Taylor, A. F., F. E. Kuo, and W. C. Sullivan. 2002. Views of nature and self-discipline: evidence from inner city children. J Environ Psychol 22:49-63.

Thompson, F. R. I. 2005. Landscape level effects on forest bird populations in eastern broadleaf forests: Principles of conservation. Pages 290-295 *in* USDA-FS, editor. General Technical Report.

Thompson, H. M. 2003. Behavioural effects of pesticides in bees–Their potential for use in risk assessment. Ecotoxicology 12:317-330.

Thurber, D. K., W. R. McClain, and R. C. Whitmore. 1994. Indirect effects of gypsy moth defoliation on nest predation. Journal of Wildlife Management 58:493-500.

Tiwary, A., D. Sinnett, C. Peachey, Z. Chalabi, S. Vardoulakis, T. Fletcher, G. Leonardi, C. Grundy, A. Azapagic, and T. R. Hutchings. 2009. An integrated tool to assess the role of new planting in PM10 capture and the human health benefits: a case study in London. Environmental Pollution 157:2645-2653.

Tu, C. M. 1995. Effect of five insecticides on microbial and enzymatic activities in sandy soil. Journal of Environmental Science and Health, Part B: Pesticides, Food Contaminants, and Agricultural Wastes 30:289-306.

Tzoulas, K., K. Korpela, S. Venn, V. Yli-Pelkonen, A. Ka´zmierczak, J. Niemela, and P. James. 2007. Promoting ecosystem and human health in urban areas using Green Infrastructure: A literature review. Landscape and Urban Planning 81:167-178.

Ulrich, R. S. 1984. View through a window may influence recovery from surgery. Science 224:420-421.

University of Vermont Tourism Data Center. 2012. The Vermont travel and tourism industry - 2012 (last accessed June 30, 2014 at http://www.uvm.edu/~snrvtdc/publications/Vermont_Tourism_2011.pdf).

USDA. 2009. Census of Agriculture: Census of horticultural crops, 2009. Last accessed July 2, 2014 at http://www.agcensus.usda.gov/Publications/2007/Online_Highlights/Census_of_Horticulture_Specialties/. United States Department of Agriculture.

USDA. 2012. Report on the National Stakeholders Conference on Honey Bee Health. United States Department of Agriculture, Alexandria, Virginia.

USGS. 1998. Northeast wetland flora. Last accessed May 6, 2014: http://www.npwrc.usgs.gov/resource/plants/florane/list.htm. United States Geological Service (USGS).

USGS. 2014. Digital representations of tree species range maps from "Atlas of United States Trees" by Elbert L. Little Jr. (and other publications). Last accessed July 1, 2014 at http://esp.cr.usgs.gov/data/little/. United States Department of the Interior, U.S. Geological Survey.

Velarde, M. D., G. Fry, and M. Tveit. 2007. Health effects of viewing landscapes -Landscape types in environmental psychology. Urban Forestry & Urban Greening 6:199-212.

Vidau, C., M. Diogon, J. Aufauvre, R. Fontbonne, B. Viguès, J.-L. Brunet, C. Texier, D. G. Biron, N. Blot, H. El Alaoui, L. P. Belzunces, and F. Delbac. 2011. Exposure to sublethal doses of fipronil and thiacloprid highly increases mortality of honeybees previously infected by *Nosema ceranae*. PLoS One 6:1-8.

Wall, D. H., R. D. Bardgett, V. Behan-Pelletier, J. E. Herrick, T. H. Jones, K. Ritz, and J. Six. 2012. Soil ecology and ecosystem services. Oxford University Press.

Walters, R. S. and H. W. Yawney. 1990. Acer rubrum L. Red maple. Silvics of North America 2:60.

Wan, M. T., D. J. Mout, and R. G. Watts. 1987. Acute toxicity to juvenile pacific salmonids of Garlon 3A™, Garlon 4™, triclopyr, triclopyr ester, and their transformation products: 3,5,6-trichloro-2-pyridinol and 2-methoxy-3,5,6-trichloropyridine. Bull. Environ. Contam. Toxicol. 39:721-728.

Wang, B., V. C. Mastro, and W. H. McLane. 2000. Impacts of chipping on surrogates for the longhorned beetle *Anoplophora glabripennis* (Coleoptera: Cerambycidae) in logs. Journal of Economic Entomology 93:1832-1836.

Wear, D. N. and J. G. Greis. 2013. The Southern Forest Futures Project: Technical Report. Gen. Tech. Rep. SRS-178. Last accessed May 16, 2014, http://www.srs.fs.fed.us/pubs/gtr/gtr_srs178.pdf. U.S. Department of Agriculture, Forest Service, Southern Research Station, Asheville, NC.

Wenger, S. 1999. A review of the scientific literature on riparian buffer width, extent, and vegetation. University of Georgia, Institute of Ecology, Office of Public Service and Outreach.

Whitelaw, S., J. Swift, A. Goodwin, and D. Clark. 2008. Physical activity and mental health: The role of physical activity in promoting mental wellbeing and preventing mental health problems: An evidence briefing. University of Glasgow.

Williams, D. W., G. Li, and R. Gao. 2004. Tracking movements of individual *Anoplophora glabripennis* (Coleoptera: Cerambycidae) adults: application of harmonic radar. Environmental Entomology 33:644-649.

Woods, K. D. 1997. Community Response to Plant Invasion. Pages 56-68 Assessment and Management of Plant Invasions. Springer-Verlag, Inc., New York.

Yarie, J. 1980. The role of understory vegetation in the nutrient cycle of forested ecosystems in the mountain hemlock biogeoclimatic zone. Ecology 61:1498-1514.

Zhang, Y.-H., S.-S. Liu, H.-L. Liua, and Z.-Z. Liuc. 2010. Evaluation of the combined toxicity of 15 pesticides by uniform design. Pest Management Science 99:879-887.

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