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Nursery Treatment Efficacy Study within Worcester County, Massachusetts, to Support the Asian Longhorned Beetle Cooperative Eradication Program

Environmental Assessment
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I. Introduction

Asian longhorned beetle (*Anoplophora glabripennis*) (ALB) is a foreign wood-boring beetle that threatens a wide variety of hardwood trees in North America. The native range of ALB includes China and Korea. Introduction and establishment of ALB into the United States would likely result in significant economic, environmental, and social impacts. While ALB spreads slowly by natural means, it can spread much faster through artificial spread. New infestations are difficult to detect and are often not found for 10 years or longer. If it becomes established in the United States, ALB could destroy as much as 60 percent of the tree population in some areas. Susceptible host species may never significantly recover and regenerate thus resulting in negative impacts to forest-dependent terrestrial and aquatic species. Negative economic impacts could include loss or reduction of exports of host species logs and live trees and shrubs, loss of saw logs and other forest products, and negative impacts to the maple syrup industry. In addition, the cumulative loss of urban trees over a 30- to 50-year period could amount to hundreds of billions of dollars in replacement costs (USDA–APHIS, 2009a).

ALB is believed to have been introduced into the United States from wood pallets and other wood packing material accompanying cargo shipments from Asia. ALB was first discovered in August 1996 in the Greenpoint neighborhood of Brooklyn, New York. Within weeks, another infestation was found on Long Island in Amityville, New York, after officials learned that infested wood had been moved from Greenpoint to Amityville.

In July 1998, due to the U.S. Department of Agriculture's (USDA) national ALB pest alert campaign, a separate infestation was discovered in the Ravenswood area of Chicago. This discovery prompted USDA's Animal and Plant Health Inspection Service (APHIS) to amend its existing quarantine of wood movement in infested areas and place additional restrictions on importing solid wood packing material into the United States from China and Hong Kong.

In October 2002, ALB was discovered in Jersey City, New Jersey, and in August 2004, ALB was discovered in the Borough of Carteret, the Avenel section of Woodbridge Township, and in the nearby cities of Rahway and Linden, New Jersey. It was subsequently found in 2007 in Richmond County, New York (Staten Island), across the Arthur Kill River from the New Jersey infestation sites.

In August 2008, ALB was discovered in Worcester County, Massachusetts. This infestation appeared to be 8 to 10 years old. The infested area is being treated according to the new pest response guidelines (USDA–APHIS, 2008a). The treatment consists of cutting, chipping, and disposing (either by burning or mulching) of infested trees and other host trees in close proximity to those infested. Uninfested host trees beyond the cutting zone are treated with either trunk injections or soil applications at the base of the tree using the insecticide imidacloprid. Imidacloprid is taken up and distributed throughout the tree, and has been found to be effective against adult ALB as it feeds on small twigs, the female when depositing eggs, and young larvae (USDA–APHIS, 2008b).

A. Biology

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. After mating, adult females chew depressions into the bark of various hardwood tree species in which they lay (oviposit) their eggs. There are 12 known genera of host trees: *Acer* (maple and box elder), *Aesculus* (horsechestnut), *Salix* (willow), *Ulmus* (elm), *Betula* (birch), *Albizia* (mimosa), *Celtis* (hackberry), *Cercidiphyllum* (katsura tree), *Fraxinus* (ash), *Plantanus* (sycamore and London planetree), *Sorbus* (mountain ash), and *Populus* (poplar) (USDA–APHIS, 2008b; Commonwealth of Massachusetts, 2009).

Once the eggs hatch, small white larvae bore into the tree, feeding on the vascular layer beneath. The larvae continue to feed deeper into the tree's heartwood forming tunnels, or galleries, in the trunk and branches. This damage cuts off nutrient flow and weakens the integrity of the tree which will eventually die if the infestation is severe enough. Sawdust debris and insect waste and excrement (or frass) is commonly found at the base of afflicted trees, as well. Infested trees are also prone to secondary attack by other diseases and insects.

Over the course of a year, a larva will mature and then pupate. From the pupa, an adult beetle emerges chewing its way out of the tree, forming characteristic round holes approximately three-eighths of an inch in diameter. The emergence of beetles typically takes place from June through October, with adults then flying in search of mates and new egg-laying sites to complete their life cycle.

B. Purpose and Need

APHIS has the responsibility for taking actions to exclude, eradicate, and/or control plant pests under the Plant Protection Act (7 United States Code (U.S.C.) 7701 et seq.). It is important that APHIS implement a quarantine and eradicate ALB from Massachusetts to prevent damage to hardwood trees in North America. To eliminate ALB in Massachusetts, the program utilizes removal of host trees, intensive tree surveys, insecticide injections into trees or soil applications, and herbicide treatments to host plant stumps and sprouts. Activities undertaken in the Massachusetts eradication effort have been the subject of a previous environmental assessment (EA) (USDA–APHIS, 2008b). Links to this EA, as well as other EAs that are pertinent to ALB eradication, are online and available: http://www.aphis.usda.gov/plant_health/ea/alb.shtml.

Currently, APHIS has only one insecticide to use for soil treatment or trunk injection; this is applied in the spring to ensure effectiveness. Additional chemicals and treatment schedules are being evaluated to determine if additional chemicals and/or fall chemical applications can be used, thereby significantly increasing the amount of time available to conduct eradication treatments, which could expedite the eradication effort given the large size of the Worcester infestation. The potential environmental impact of the additional treatment options was the subject of a previous EA (USDA–APHIS, 2009b). Links to this EA are available, as noted above.

Due to the wide scope of the ALB infestation in Worcester, Massachusetts, (currently 74-square miles) commercial nurseries are now being impacted by quarantine restrictions placed on the movement and sale of nursery stock. Treatment options for the nursery industry are needed so that host material can be moved outside of regulated areas. Guidelines, based on an understanding of the dynamics of uptake and resultant residues from material applied around several species and sizes of trees in a nursery situation, need to be developed. The proposed study has been designed to determine if soil applications of various insecticides will result in effective levels of insecticides in tree tissues upon which beetles feed.

This EA has been prepared consistent with the National Environmental Policy Act of 1969 (NEPA) and APHIS' NEPA implementing procedures (7 Code of Federal Regulations (CFR) part 372) for the purpose of evaluating how the proposed action, if implemented, may affect the quality of the human environment.

II. Alternatives

This EA analyzes the potential environmental consequences associated with the proposed nursery treatment study of insecticides being considered in the ALB Cooperative Eradication Program.

Two alternatives are being considered: (1) no action by APHIS, and (2) the preferred alternative, to determine if soil applications of various insecticides will result in effective levels in tree tissues that beetles feed upon.

A. No Action

Under the no action alternative, APHIS would continue to implement the ALB eradication program in Worcester County, Massachusetts, without consideration of adding treatments specifically for susceptible nursery trees. This would prohibit the movement of host nursery stock out of the quarantine.

B. Preferred Alternative

Under the preferred alternative, APHIS would also continue to implement the ALB eradication program in Worcester County. In addition, APHIS would actively seek to determine whether labeled soil applications of systemic insecticides can achieve acceptable residue levels in several ALB host tree species grown in ground within a commercial nursery. This would be useful information in the design of a fully integrated eradication program for ALB wherever it may occur. If acceptable residue levels are detected, host nursery stock would be able to be moved out of the quarantine.

Several sizes and species of ALB host trees commonly sold and planted in the Worcester area by the public and municipalities will be included in the study in order to provide the most relevant information to the program. The proposed study will test four tree species and two size groupings of red maple, sugar maple, elm, and London plane trees. The applications will be drench treatments along a row of planted trees using industry equipment and personnel (row spacing = 11 feet). All tests will be conducted on property owned and operated by Bigelow Nurseries, Inc., within Worcester County, Massachusetts (see figure 1).

Trees will be randomly assigned to an early summer application of one of four chemical treatments; a minimum of 50 trees of each size class and chemical will be treated. Chemical applications are planned for May/June of 2010, 2011, and 2012. All application rates will be the maximum allowed by the U.S. Environmental Protection Agency (EPA)-registered pesticide label. The chemical treatments will include the following amounts of product in a minimum of 2 gallons of water per 1,000 ft^2:

1) drench application with Marathon 60 WP (60% imidacloprid; 1 packet/1,000 ft of row);
2) drench application with Safari 20 SG (20% dinotefuran; 8 oz/1,000 ft^2);
3) drench application with Arena 50 WDG (50% clothianidin; 8.3 g/1,000 ft^2); and
4) drench application with Flagship 25 WG (25% thiamethoxam; 1.47 oz/1,000 ft^2).

Figure 1. The treatment area (outlined in red) is adjacent to Route 62 just east of Moore's Corners in Worcester, Massachusetts.

The list of tree species selected for this study and their approximate size expressed as diameter at breast height (DBH) is as follows—

Tree Species	DBH	DBH	Minimum # of Trees
Sugar maple	1.0–1.5"	2.0–2.5"	400
Red maple	1.0–1.5"	2.0–2.5"	400
Elm	1.0–1.5"	2.0–2.5"	400
London plane	1.0–1.5"	2.0–2.5"	400

Treatments will be identified by attaching colored flagging tape to trees on each end of a production row: Marathon = red stripe; Safari = yellow stripe; Arena = green stripe; Flagship = blue stripe.

Pesticide residue analysis will be conducted on foliage collections made each year in early July (to coincide with the expected first emergence of ALB adults) and early September (near end of flight season) under supervision by the APHIS Otis Methods Laboratory. Sampling will be done by collecting leaves from terminal branches located within the lower one-third to one-half of each tree canopy, selecting a total of 4 to 8 samples from all sides of the tree. The goal is to determine whether applications made using standard nursery practices and rates are sufficient to control ALB. In order to confirm the efficacy of these treatments, they will be compared to parallel studies being conducted in China in 2010 by Otis Methods Laboratory scientists. Potential treatment sites are located in eastern China near/along the Yangtze River, within Anhui Province and Zhejiang Province.

III. Environmental Impacts

A. No Action

Environmental impacts that could result from choosing the no action alternative would likely be related to preventing the eradication program from fully utilizing the information and experience that could be gained from carrying out research to improve the efficacy and efficiency of the program. Information that could be gained from the proposed action could lead to a treatment that would allow certification of nursery stock so it could be moved out of the regulated area, thus reducing the potential impact of the ALB program on the nursery industry.

B. Preferred Alternative

APHIS proposes to evaluate the efficacy of four insecticides to control ALB. All four compounds are systemic neonicotinoid insecticides that are currently registered for use on a wide variety of crops to control a variety of pests. All products proposed for use in this study are currently registered for these types of applications. The potential environmental risks from the proposed use of each of the four proposed insecticides (imidacloprid, dinotefuran, clothianidin, and thiamethoxam) are discussed below.

1. Imidacloprid

a. Toxicity

Technical and formulated imidacloprid has low to moderate acute oral mammalian toxicity with median toxicity values ranging from 400 to greater than 2,000 milligram/kilogram (mg/kg). The technical material, as well as several formulations, are considered practically nontoxic from dermal or inhalation exposure (FS, 2005; USDA–APHIS, 2002a). Acute lethal median toxicity values are typically greater than 2,000 mg/kg and 2.5 milligrams/Liter (mg/L) for dermal and inhalation exposures, respectively. Available data for imidacloprid and associated metabolites suggest a lack of mutagenic, carcinogenic, or genotoxic effects at relevant doses. Developmental, immune, and endocrine related effects have been observed in some mammalian toxicity studies. In all developmental studies, the noted effects were observed at doses above maternal effects and at concentrations and durations not expected in the proposed efficacy study (FS, 2005).

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 (USDA–APHIS, 2002a; EPA, 2008; FS, 2005). Reproduction studies using the mallard and bobwhite quail have shown no effect concentrations of approximately 125 parts per million (ppm) for both species.

Technical and formulated imidacloprid is considered acutely toxic to some terrestrial invertebrates, such as honey bees and other related bee species, by oral and contact exposure. Median lethal toxicity values range from 3.7 to 230 nanograms (ng)/bee (Schmuck et al., 2001; Tasei, 2002; FS, 2005; EPA, 2008). Acute sublethal effects in laboratory studies have shown that the no observable effect concentrations (NOEC) may be less than 1 ng/bee (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 nontoxic (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, and colony health, as well as other endpoints, revealed that there was a lack of effects, or effects were observed at test concentrations not expected to occur 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).

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 (LC_{50}) typically exceeding 100 mg/L (EPA, 2008; FS, 2005). Chronic toxicity to fish is in the low parts per million range, depending on the test species and endpoint. Aquatic invertebrates are more sensitive to imidacloprid when compared to fish with acute median toxicity values in the low parts per billion range to greater than 100 mg/L, depending on the test species (USDA–APHIS, 2002a; EPA, 2008; FS, 2005).

b. Exposure and Risk

Based on the limited use pattern for the proposed use of imidacloprid in this study, potential exposure will be primarily to applicators and workers. Exposure to applicators will be reduced by following label directions, including recommendations for personal protective equipment, resulting in minimal risk to applicators. Exposure to the general public is not expected because none of the treated trees will be used to yield products that would be used for human consumption. Dietary risk from exposure to contaminated drinking water is also not expected based on the limited area of application, the proposed method of application, adherence to label recommendations regarding the protection of ground water, and monitoring data that has been collected in association with ALB eradication efforts in other States. Ground water sampling between 2003 and 2006 in Suffolk County, New York, demonstrated that approximately half of the samples had no detectable levels of imidacloprid and, of those where detections occurred, the average concentration was 3.2 parts per billion (ppb) which is below levels of concern for human health. 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.

Exposure and risk to terrestrial vertebrates is expected to be minimal, based on the proposed method of application, limited area of application, and available effects data. Exposure from drift is not expected, nor is any significant runoff, based on the use pattern for imidacloprid in the ALB eradication program. There is the possibility of some imidacloprid exposure to mammals and birds that may feed on insects or vegetation from treated trees; however, under worst-case exposure scenarios, the risk is considered minimal.

Imidacloprid exposure to terrestrial invertebrates, especially honey bees, is also not expected to result in significant risk to pollinators. Impacts to individual insects that feed on twigs and leaves from treated trees and are sensitive to imidacloprid are expected to occur; however, population level impacts are not expected due to the small area of treatment within the managed nurseries. Pollinator exposure to imidacloprid will be minimized by the fact that only treated trees and their associated flowers and pollen could have residues while other flowering plants in the area of treatment will not contain residues. The potential level of imidacloprid in pollen and nectar from trees that have been treated for ALB is unknown, but is expected to be low based on the available pollen and nectar residue data for other plants. Previous studies have shown that imidacloprid levels in pollen and flowers are low compared to other parts of the plant. Schmuck et al. (2004) found that levels of imidacloprid and associated metabolites were below the level of detection (0.001 mg/kg) in sunflowers. Laurent and Rathahao (2005) found average imidacloprid residues from sunflower pollen of 13 micrograms/kilograms (µg/kg), while Bonmatin et al. (2005) found average imidacloprid levels of 6.6 and 2.1 µg/kg in flowers and pollen from treated maize seed. These reported sunflower and corn pollen residues are within the range of values from other studies, and are similar to imidacloprid residue levels found in the nectar and pollen for rape (Maus et al., 2003). Chauzat et al. (2006) found that approximately 50 percent of the pollen samples collected from pollen traps in apiaries contained measurable levels of imidacloprid with an average concentration of 1.2 µg/kg.

As part of the environmental monitoring program, APHIS analyzed for imidacloprid residues in flowers collected from imidacloprid-treated willow, horse chestnut, and maple trees from New York during and after ALB eradication efforts (USDA–APHIS, 2003, 2002b). With the exception of one maple flower sample (0.13 mg/kg), all residues were below the level of quantification or detection (level of detection = 0.03 mg/kg) over a 2-year sampling period. Residues in flowers were lower than in twig and leaf residues which are similar to observations in other plant species, such as corn and sunflowers. APHIS is working in cooperation with USDA–Agriculture Research Services (ARS) to collect pollen and nectar samples, where applicable, to characterize exposure to honey bees in nursery trees from these types of treatments. The risk to honey bees and other pollinators is expected to be minimal based on the small number of trees proposed for treatment, expected residues from the method of application and the presence of other nontreated flowering plants (both of which minimize exposure), and the available acute and chronic honey bee toxicity data for imidacloprid.

Imidacloprid exposure in aquatic environments is also expected to be minimal and not pose a significant risk to aquatic biota. The method of application eliminates the potential for drift and reduces the probability of off-site transport via runoff. There is a potential for subsurface transport of imidacloprid to aquatic habitats from applications made directly into soil. This type of exposure will be minimized by only making applications where the ground water table is not in proximity to the zone of application, and avoiding soils that have a high leaching potential. Any aquatic residues that could occur would be below effect levels for aquatic biota due to the low probability of off-site transport and environmental fate for imidacloprid. There is the potential for leaf litter from treated trees to be washed into surface water during leaf drop. Kreutzweiser et al. (2009, 2008, 2007) demonstrated imidacloprid-related impacts on decomposition rates in aquatic systems, as well as sublethal impacts to some aquatic invertebrates that feed on leaf litter. The risk to benthic aquatic invertebrates through this exposure pathway in the proposed study is expected to be minimal. A small number of trees are being treated (relative to the amount of leaf litter from nontreated plants) that could be deposited into aquatic systems and applications will not occur in close proximity to natural bodies of water. A small retention pond is present at the nursery and is within the drainage area where applications will occur. APHIS will collect water samples to determine if any residues are present as a result of the proposed treatment.

c. Environmental Quality

Imidacloprid is soluble in water and is considered to have moderate mobility based on soil adsorption characteristics for several soil types.

Based on field dissipation studies, the foliar half-life is less than 10 days while the persistence in soil can range from 27 to 229 days, (CDPR, 2006; FS, 2005). In cases of soil application, exposure of imidacloprid to soil invertebrates is possible; however, the impacts would be localized to the areas of treated soil and would be transient, based on available data (FS, 2005). In water, imidacloprid is stable to hydrolysis at all relevant pH values but breaks down rapidly in the presence of light with aqueous photolysis half-life values typically less than 2 hours. Imidacloprid does exhibit properties consistent with pesticides that can contaminate surface and ground water; however adherence to label recommendations and the lack of close proximity of the study site to ground water and surface water resources mitigates the potential for ground and surface water contamination.

The closest surface water resource to the proposed treatment site is a retention pond at the nursery which will be sampled to insure protection of surface water resources. The low volatility and proposed method of application in this program minimizes the potential for impacts to air quality from the use of imidacloprid.

2. Dinoferuran

a. Toxicity

The available acute mammalian toxicity data suggest that technical and formulated dinotefuran has low oral, dermal, and inhalation toxicity (EPA, 2004). Irritation to the eye and skin is considered minor based on the available material safety data sheet. Dinotefuran is not considered to be mutagenic, carcinogenic, or teratogenic based on the available mammalian toxicity data. Subchronic and chronic no observable effect levels (NOEL) for mammals range from less than 3 mg/kg/day in chronic dosing studies in mice to 5,414 mg/kg /day in a 90-day dosing study in mice (EPA, 2004).

The toxicity of dinotefuran to nontarget terrestrial vertebrates is low. Available mammalian and avian toxicity data show low toxicity to both groups based on available surrogate toxicity data. Acute oral and dietary median lethality studies using the quail and mallard duck show toxicity to be greater than the highest test concentration (EPA, 2009, 2004). Chronic toxicity to birds is also low with reproductive NOEC values of 2,150 and 5,270 ppm for the mallard and bobwhite quail, respectively.

Toxicity to insects, such as the honey bee, is high with oral and contact median lethal dose (LD_{50}) values of 0.023 and 0.047 µg/bee, respectively.

Available acute freshwater and marine fish toxicity data suggest that dinotefuran is practically nontoxic with LC_{50} values greater than the highest test concentration. Acute and chronic toxicity to aquatic invertebrates is low for most test organisms with the exception of the mysid shrimp, which reports a LC_{50} of 0.79 ppm. Acute toxicity to other aquatic invertebrates is low, with acute and chronic toxicity values greater than 95 ppm (EPA, 2009, 2004).

b. Exposure and Risk

Dinotefuran applications proposed in this efficacy study are expected to have minimal impacts to human health, based on the available toxicity data and low potential for exposure. Applications will be made as a soil drench where the active ingredient will then be taken up and distributed throughout the tree. None of the treated trees will be used to yield products that would be used for human consumption; therefore, dietary exposure is not expected. The potential for exposure is greatest for applicators; however, the low mammalian toxicity and adherence to label recommendations regarding personal protective equipment will minimize exposure and risk to applicators (EPA, 2004).

Exposure and risk of dinotefuran applications proposed in this study are expected to be low for most terrestrial nontarget organisms. Applications of dinotefuran to soil will result in exposure to terrestrial vertebrates that may feed on treated twigs, leaves, or seeds as part of their diet. Significant dietary risk to terrestrial vertebrates is not expected, based on the available toxicity data and conservative assumption that feeding would occur only from treated trees. Indirect impacts to birds and mammal populations that rely on insects for food would also not be significant because of the method of application for dinotefuran and the low number of trees being treated. There could be impacts to some terrestrial invertebrates that feed on treated trees and are sensitive to dinotefuran. No negative impacts to invertebrate populations are expected due to the low number of trees being treated and the availability of other nontreated vegetation.

Similar to other neonicotinoid insecticides, there are concerns regarding dinotefuran risk to honey bees. Treatments will occur in the spring and, based on the systemic nature of this class of insecticides, there is the potential for dinotefuran exposure to nectar and pollen. Residue data for this class of insecticides from nectar and pollen have been measured in several crops and, to date, residues have typically been below levels that would suggest impacts (Franklin et al., 2004; USDA–APHIS, 2008). There is some uncertainty in this assessment because the potential residues from dinotefuran applications using this method of application have not been characterized in trees. To address this uncertainty, nectar and pollen samples will be collected from treated trees during this study and analyzed for dinotefuran by USDA–ARS. Residue data on pollen and nectar will allow for a more accurate characterization of exposure to honey bees.

This data can then be compared to the available toxicity data for dinotefuran, and related insecticides, to provide a more accurate representation of risk to honey bees from these types of treatments. The study itself is not anticipated to result in a major impact to honey bees because of the small number of trees that will be treated relative to the number available to bees. In addition, dinotefuran exposure to honey bee populations from these types of treatments will be reduced compared to conventional broadcast applications of insecticides and the presence of other flowering vegetation in the area that has not been treated.

Exposure and risk of dinotefuran to aquatic organisms is not expected. Dinotefuran has low toxicity to most aquatic organisms, and significant exposure from drift and runoff are not expected because the material will be applied as a soil drench. The potential exists for leaf litter from treated trees to be washed into surface water during leaf drop the following fall. Studies using another neonicotinoid insecticide, imidacloprid, have demonstrated some impacts on decomposition rates in aquatic systems, as well as sublethal impacts to some aquatic invertebrates that feed on leaf litter (Kreutzweiser et al., 2009, 2008, 2007). There is uncertainty whether this type of impact could result from dinotefuran applications; however, the potential for contamination through this pathway is expected to be minor in this study because there will be leaf litter contributions from plants that have not been treated in the area, and trees selected for treatment will not be in proximity to surface water. A retention pond present at the nursery and in the same drainage as the proposed efficacy study will be sampled to determine if dinotefuran is present as a result of treatment.

c. Environmental Quality

Dinotefuran degrades slowly in soil with a reported aerobic soil metabolism half-life of 138 days. Degradation in water is rapid in the presence of light with a half-life of 1.8 days, but is stable to hydrolysis. Dinotefuran is highly soluble in water, and does not absorb well to soil; therefore, it could be susceptible to runoff and potential contamination of ground and surface water (EPA, 2004). The closest surface water feature to the treatment area is a retention pond which will be sampled for residues; however, other surface water resources are not expected to be impacted due the distance from the treatment site and the proposed small area of treatment. Adherence to precautionary label language regarding the protection of ground water (such as avoiding applications to permeable soils and avoiding sites with a high water table) will ensure protection of ground water resources. Impacts to air quality are not expected based on the chemical properties of dinotefuran, which show low volatility, as well as the method of application, which will eliminate impacts to air quality from drift.

3. Clothianidin

a. Toxicity

The clothianidin technical ingredient and formulation proposed for use in this program has low acute mammalian oral, dermal, and inhalation toxicity. The median lethal toxicity values for oral exposure range from 3,900 to 4,700 mg/kg, and the dermal and inhalation toxicity values are greater than 5,000 mg/kg and 3.2 mg/L, respectively. The formulation proposed for use in this program is moderately irritating to the eye and is a slight skin irritant (Valent, 2007). Clothianidin is not considered to be teratogenic, mutagenic, or carcinogenic and, based on the range of subchronic and chronic studies that are available, the NOEL ranged from 9.8 mg/kg/day in reproduction studies to 1,000 mg/kg/day in subchronic dermal toxicity studies (EPA, 2003).

Clothianidin also has low toxicity to birds, based on available toxicity values for surrogate test species. Acute oral and dietary LC_{50} are greater than 2,000 mg/kg in oral testing and greater than 5,000 ppm in dietary studies. Chronic studies using birds show low toxicity with a NOEC of 205 ppm (EPA, 2009).

Clothianidin is highly toxic to honey bees with an acute contact median lethal concentration of 0.0439 µg/bee (EPA, 2003). Sublethal impacts (such as colony health and foraging ability) have been evaluated for other pollinators, such as the bumble bee, with no impacts observed at pollen residue values up to 36 ppb (Franklin et al., 2004).

Clothianidin has low acute toxicity to freshwater and marine vertebrates with LC_{50} values greater than 94 ppm. Chronic toxicity studies of fish using the fathead minnow reports a NOEC of 9.7 ppm (EPA, 2009). Toxicity to aquatic invertebrates is variable with median effective concentration (EC_{50}) and LC_{50} values ranging from highly toxic with EC_{50} value of 0.022 ppm for the midge, to practically nontoxic with EC/LC_{50} values greater than 100 ppm for the freshwater crustacean *Daphia magna* and eastern oyster (EPA, 2003; Barbee and Stout, 2009).

b. Exposure and Risk

Human exposure and risk to clothianidin is expected to be minimal based on the method of application, the small area of treatment, and available toxicity data. The pesticide will be applied directly to the soil where the active ingredient will be translocated upward in the tree. None of the treated trees will be used to yield products that would be used for human consumption; therefore, dietary exposure would not be expected. Exposure through contaminated drinking water is also not expected because treated trees will not be in proximity to surface water or ground water drinking sources. The greatest chance for exposure to clothianidin will occur with applicators; however, risk will be minimal based on the low oral, dermal, and inhalation toxicity. In addition, exposure will be low based on the method of application and adherence to label recommendations regarding personal protective equipment.

Exposure and risk to most nontarget organisms is expected to be minimal. Toxicity to terrestrial vertebrates is low, and exposure to clothianidin would only occur through ingestion of soil under treated trees or by consuming leaves, twigs, or seeds from treated trees. Using the available toxicity data and the unrealistically conservative assumption that only items from treated trees are fed upon, residue data for these types of treatments using similar insecticides show that levels in various parts of the tree would not pose a risk to terrestrial vertebrates. Actual exposure and risk would be less based on the different types of food items used by terrestrial vertebrates and the relatively small number of trees that will be treated within the area.

Indirect effects to terrestrial vertebrates through the loss of invertebrate prey is also not expected because only certain insects would be impacted by feeding on treated trees, and terrestrial vertebrates would be able to forage in the area on insects that are present on untreated trees and other vegetation.

Some insects that feed on treated trees could be impacted; however, based on the method of application, no drift would be expected and impacts would be restricted only to those insects that are sensitive to clothianidin and feed on treated trees. Similar to other neonicotinoid insecticides, there are concerns regarding clothianidin risk to honey bees. For this class of insecticides, residue data from nectar and pollen have been measured in several crops and, to date, residues have typically been below levels that would suggest impacts (Franklin et al., 2004; USDA–APHIS, 2008). There is some uncertainty in this assessment because the potential residues from clothianidin applications using this method of application have not been characterized. To address this uncertainty USDA–ARS, in cooperation with APHIS, will be collecting nectar and pollen samples from treated trees during the efficacy study to evaluate clothianidin levels to better characterize exposure and risk to honey bees. The efficacy study, itself, is not anticipated to result in major impacts to honey bees because of the small number of trees that will be treated relative to the available sources for bees to choose from and expected residues based on previously published literature for similar insecticides.

Applications of clothianidin, as proposed in this program, are not expected to impact aquatic organisms. Although toxicity to fish is low, clothianidin is toxic to some aquatic invertebrates. The method of application will eliminate the potential for off-site drift, and runoff is not expected because soil applications will not occur in proximity to natural bodies of water. There is the potential for leaf litter from treated trees to be washed into surface water during leaf drop the following fall. Studies using another neonicotinoid insecticide, imidacloprid, have demonstrated some impacts on decomposition rates in aquatic systems, as well as sublethal impacts to some aquatic invertebrates that feed on leaf litter (Kreutzweiser et al., 2009, 2008, 2007). The relatively small number of trees proposed for treatment, the leaf litter contributions from plants that have not been treated in the area, and not treating trees that are in proximity to surface water will reduce the potential for clothianidin leaf litter residues in surface water.

c. Environmental Quality

Clothianidin is considered stable in soil with metabolic half-lives of 148 to 1,155 days, and dissipation half-lives of 277 to 1,386 days. Impacts to some soil invertebrates could occur; however, those impacts would be restricted to invertebrates sensitive to clothianidin and in areas immediately under treated trees. In aquatic environments, clothianidin breaks down rapidly in the presence of light with a half-life of 1 day, but is considered stable to hydrolysis. Clothianidin is soluble in water and considered mobile to highly mobile in soil (EPA, 2003). Impacts to ground and surface water quality are not anticipated, based on the method of application which reduces the potential for runoff and drift, and adherence to precautionary label language regarding the protection of these types of resources. A retention pond present at the nursery will be sampled for clothianidin to determine if treatments result in residues that could impact water quality. Based on the method of application and low potential to volatilize into the atmosphere, clothianidin applications are not expected to impact air quality (EPA, 2003).

4. Thiamethoxam

a. Toxicity

Thiamethoxam has low acute oral, dermal, and inhalation toxicity, based on available data for the technical active ingredient and the formulated material. The technical active ingredient is not considered an eye or skin irritant; however, the proposed formulation is a slight to mild irritant to the eye and skin (EPA, 1999). The active ingredient or formulated material is not considered a skin sensitizer, based on available data from the material safety data sheet. Thiamethoxam is not considered to be neurotoxic, based on acute and subchronic exposures which reported NOEL of 100 and 95.4 mg/kg/day, respectively (EPA, 1999). Thiamethoxam is not mutagenic, and reproductive and developmental effects occur at levels that are maternally toxic.

Thiamethoxam has moderate to low acute toxicity to avian species, based on surrogate toxicity data required for pesticide registration. Median lethal oral doses range from 576 mg/kg in the mallard to 1,552 mg/kg in the bobwhite quail. Median lethal dietary concentrations show that thiamethoxam is considered practically nontoxic to the mallard and bobwhite quail with values exceeding 5,200 ppm (EPA, 2010). Chronic reproductive effects are low with a NOEC of 300 ppm for the mallard and greater than 900 ppm for the bobwhite quail.

Similar to other neonicotinoids, thiamethoxam is very highly toxic to honey bees in both oral and contact exposures. Contact toxicity is reported as an LD_{50} of 0.024 µg/bee with a NOEC of 0.005 µg/bee while oral toxicity is greater with an LD_{50} of 0.005 µg/bee and a reported NOEC of 0.002 µg/bee (EPA, 2010).

Aquatic toxicity of thiamethoxam is variable depending on the test organism. Acute toxicity ranges from 0.035 mg/L for the midge to greater than 90 mg/L for all other aquatic test organisms including fish, aquatic invertebrates, and plants (EPA, 2010; Stark, 2005). Chronic toxicity data is limited to the cladoceran and the rainbow trout; however, based on those values chronic aquatic toxicity is expected to be low with NOEC values greater than 20 mg/L (EPA, 2010).

b. Exposure and Risk

Exposure and risk to human health is not expected, based on the proposed use pattern for thiamethoxam and the available mammalian toxicity data. Exposure through the dietary route is not expected for applicators or the general public. No products from the treated trees will be used for human consumption and adherence to label requirements for personal protective equipment will reduce exposure to workers via dermal and inhalation routes. Exposure through contaminated drinking water is also not expected because treated trees will not be in proximity to surface water or ground water drinking sources. In addition, adherence to label directions regarding protection of these resources will further insure protection of water quality. The low potential for exposure and the available mammalian effects data for thiamethoxam demonstrate wide margins of safety to human health.

Exposure to wild mammals and birds may occur for those groups that feed on trees which have been treated; however, based on the available effects data, the small area of treatment, varied diets for these types of animals, and foraging that would occur where untreated trees are present would all result in minimal risk. Indirect impacts to wild mammals and birds that depend on invertebrate prey is also not expected to be significant since not all insects will be impacted due to differences in sensitivities, the number of trees being treated is relatively small and wild mammals and birds will forage in areas where treatments have not occurred.

Insects that are sensitive to thiamethoxam, and feed on treated trees, could be impacted; however, due to the small area of treatment in these managed areas the impacts would not be expected to be widespread. Honey bees that may feed from treated trees may be exposed to thiamethoxam levels in nectar and pollen. As previously mentioned with the other insecticides proposed for use in this study, there is uncertainty regarding the exposure levels in nectar and pollen from thiamethoxam soil applications adjacent to host trees. Data for other neonicotinoids in other crops suggest the pollen and nectar insecticide residues may not reach levels where lethal or sublethal impacts have been observed. To address this uncertainty, ARS and APHIS will be collecting nectar and pollen samples to better characterize exposure of thiamethoxam to honey bees. Impacts to honey bee populations from this study are not expected based on the available data regarding impacts to honey bees from this class of insecticides and the relatively small number of trees being treated relative to other flowering plants in the area that have not been treated and would reduce exposure and risk.

Potential aquatic exposure of thiamethoxam is not expected to result in adverse impacts to aquatic nontarget organisms. The method of application which will eliminate drift as a major pathway of off-site transport, as well as low toxicity to most aquatic organisms, will result in minimal risk. There is the potential for surface water contamination from a retention pond on site; however, following precautionary label language to protect surface water will reduce the likelihood of contamination. APHIS will collect water samples during the efficacy study to determine if thiamethoxam residues occur as a result of the proposed treatments and if those residues exceed aquatic organism threshold values.

c. Environmental Quality

Thiamethoxam degrades slowly in soil, based on available aerobic soil metabolism and soil photolysis half-lives which ranged from 54 to 385 days (NYSDEC, 2002). In water, thiamethoxam is stable to hydrolysis at pH values of five and seven; however, under more alkaline conditions breaks down with a half-life of 8.4 days at a pH of nine. Aerobic aquatic metabolism half-lives range from 9.5 to 21.9 days, while under anaerobic conditions reported half-lives range from 9.8 to 353.5 days (NYSDEC, 2002). Field dissipation studies using treated seed demonstrate half-lives of 72 to 111 days. Thiamethoxam is considered mobile based on the high reported water solubility and low soil adsorption coefficients which range from 43 to 77. Concerns over mobility are reflected on the label with surface and ground water advisories and recommendations for mitigating off-site transport. Adherence to label recommendations and the small area of treatment proposed in the study will mitigate potential contamination of surface or ground water. A small retention pond is present on site and will be sampled to determine if any residues may be present as a result of treatment. Impacts to air quality from drift are not expected based on the method of application. Significant volatilization into the atmosphere will not occur based on the chemical properties reported for thiamethoxam (Syngenta, 2009).

C. Cumulative Effects

The proposed nursery study in Worcester, Massachusetts, is unlikely to result in significant cumulative impacts to the environment. While the nursery where the study is to be conducted has not been found to be infested with ALB, it is located in the same county and in close proximity to the quarantine area and, thus, potentially at risk of future infestation.

Over 26,000 trees have been removed from the quarantine area in Worcester, and an additional 1,200 trees are currently known to be infested and will be removed in the near future. In addition, current plans call for the chemical treatments of approximately 40,000 trees over the next year and for each year in the foreseeable future. The addition of at least 1,600 trees for chemical treatment in this study is not expected to significantly add to the total number of trees likely to be treated in Worcester during the eradication program. The trees proposed for treatment are located in a commercial nursery and are subject to pesticide treatments, within the legal constraints of the Federal Insecticide, Fungicide, and Rodenticide Act, at the discretion of the nursery managers. Considering the proximity of the nursery to the Worcester ALB infestation, it is reasonable to anticipate that the trees would be treated whether they were included in this study or not.

D. Threatened and Endangered Species

Section 7 of the Endangered Species Act and its implementing regulations require Federal agencies to ensure that their actions are not likely to jeopardize the continued existence of threatened or endangered species or result in the destruction or adverse modification of critical habitat. There are no federally listed species within the Federal quarantine area or in proximity to the nursery where the proposed action will take place. Therefore, the proposed action will have no effect on federally listed species.

E. Other Considerations

Executive Order (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 manner 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 from the proposed applications are expected to be minimal and are not expected to have disproportionate adverse effects to any minority or low-income family.

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 because of developmental stage, greater metabolic activity levels, and 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. The proposed treatments are to be made directly to trees in a commercial nursery and are not in areas where children would be expected to play and climb trees. Based on the anticipated lack of significant exposure, no disproportionate risks to children are anticipated as a consequence of implementing the preferred alternative.

IV. Agencies Consulted

U.S. Department of Agriculture
Animal Plant Health Inspection Service
Plant Protection and Quarantine
Emergency and Domestic Programs
4700 River Road, Unit 137
Riverdale, MD 20737

U.S. Department of Agriculture
Animal Plant Health Inspection Service
Plant Protection and Quarantine
Environmental Compliance
4700 River Road, Unit 150
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U.S. Department of Agriculture
Animal Plant Health Inspection Service
Policy and Program Development
Environmental and Risk Assessment Services
4700 River Road, Unit 149
Riverdale, MD 20737

U.S. Department of Agriculture
Animal and Plant Health Inspection Service
Plant Protection and Quarantine
Center for Plant Health Science and Technology
Insecticide and Applied Technology Section
Otis Pest Survey, Detection and Exclusion Laboratory
Buzzards Bay, MA 02542

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