Resource and Energy Economics 36 (2014) 270–289

Contents lists available at ScienceDirect

Resource and Energy Economics

journal homepage: www.elsevier.com/locate/ree

A bioeconomic analysis of an emerald ash borer invasion of an urban forest with multiple jurisdictions

Kent F. Kovacs [a,*], Robert G. Haight [b], Rodrigo J. Mercader [c], Deborah G. McCullough [d,e]

[a] Department of Agricultural Economics and Agribusiness, University of Arkansas, Fayetteville, AR 72762, United States
[b] U.S. Forest Service Northern Research Station, 1992 Folwell Avenue, St. Paul, MN 55108, United States
[c] Department of Biology, 1700 SW College Avenue, Washburn University, Topeka, KS 66621, United States
[d] Department of Entomology, Michigan State University, East Lansing, MI 48824, United States
[e] Department of Forestry, Michigan State University, East Lansing, MI 48824, United States

A R T I C L E   I N F O

Article history:
Received 27 April 2012
Received in revised form 19 March 2013
Accepted 18 April 2013
Available online 28 May 2013

JEL classification:
Q23
Q24
Q28

Keywords:
Invasive species
Management
Spatial control
Emerald ash borer
Non-linear programming

A B S T R A C T

Bio-invasions occur in management mosaics where local control affects spread and damage across political boundaries. We address two obstacles to local implementation of optimal regional control of a bio-invasion that damages public and private resources across jurisdictions: lack of local funds to protect the public resource and lack of access to protect the private resource. To evaluate these obstacles, we develop a spatial-dynamic model of the optimal control of emerald ash borer (EAB) in the Twin Cities metropolitan area of Minnesota, USA. We focus on managing valuable host trees with preventative insecticide treatment or pre-emptive removal to slow EAB spread. The model includes spatial variation in the ownership and benefits of host trees, the costs of management, and the budgets of municipal jurisdictions. We develop and evaluate centralized strategies for 17 jurisdictions surrounding the infestation. The central planner determines the quantities of trees in public ownership to treat and remove over time, to maximize benefits of surviving trees net costs of management across public and private

* Corresponding author. Tel.: +1 479 575 2323.
E-mail addresses: kkovacs@uark.edu (K.F. Kovacs), rhaight@fs.fed.us (R.G. Haight), rodrigo.mercader@washburn.edu (R.J. Mercader), mccullo6@msu.edu (D.G. McCullough).

http://dx.doi.org/10.1016/j.reseneeco.2013.04.008

ownerships, subject to constraints on municipal budgets, management activities, and access to private trees. The results suggest that centralizing the budget across jurisdictions rather than increasing any one municipal budget does more to increase total net benefits. Strategies with insecticide treatment are superior to ones with pre-emptive removal because they reduce the quantity of susceptible trees at lower cost and protect the benefits of healthy trees. Increasing the accessibility of private trees to public management substantially slows EAB spread and improves total net benefit.

1. Introduction

The management of bio-invasions is an example of spatial-dynamic processes that are receiving increasing attention by economists (Wilen, 2007). Bio-invasions generate damages as they spread, and optimal control – where, when, and how intensively control activities are undertaken – depends on the spatial configuration of resource benefits and management costs. The optimal control of bio-invasions is addressed with spatially explicit models of invasive species dynamics, in which the landscape is divided into a set of discrete patches, control activities are defined for each patch, and the growth and dispersal of the invasive species is a function of the selected controls (Bhat et al., 1993; Hof, 1998; Albers et al., 2010; Blackwood et al., 2010; Kaiser and Burnett, 2010; Epanchin-Niell and Wilen, 2012). While these models show that optimal control depends on the configuration of the threatened resource and natural barriers to spread, they do not explore management in landscapes where multiple ownerships and political boundaries separate the threatened resource (Epanchin-Niell et al., 2010).

Bio-invasions usually take place at landscape scales that encompass multiple landowners – a management mosaic – where each landowner's control decisions directly impact his/her neighbors' decisions by affecting invasion spread across boundaries (Wilen, 2007; Epanchin-Niell et al., 2010). When landholders make control decisions based only on damages occurring on their own land, an externality occurs because controllers confer uncompensated benefits to those in advance of the invading front (Wilen, 2007). As a result, managers may under-control from a system-wide perspective, leading to increased invasion of the landscape (Wilen, 2007). A central planner who determines the optimal control of a bio-invasion can internalize this diffusion externality and increase total net benefits across ownerships (e.g., Feder and Regev, 1975; Bhat and Huffaker, 2007; Richards et al., 2010; Sims et al., 2010).

While the potential for welfare gain from centralized planning is well known, there are practical obstacles to realizing that gain. For example, the central authority may establish uniform controls across jurisdictions because of imperfect information about local payoffs. This will not represent the first-best outcome if preferences, damages, environmental attributes, or control costs differ across jurisdictions (List and Mason, 2001; Albers et al., 2010). Instead, market-based corrections such as taxes, permits, or subsidies (e.g., Richards et al., 2010; Sims et al., 2010) or transfer payments (e.g., Bhat and Huffaker, 2007) may induce local control decisions that approach the levels of a centralized plan that accounts for asymmetries in local payoffs.

We address two other obstacles that arise where the resource at risk is owned by both public and private entities. First, a local government may not have sufficient funds to implement the control strategy that is optimal at the regional level thereby increasing the likelihood of infesting neighboring jurisdictions and increasing damages. Second, local governments may not have access to private land within their jurisdiction to implement control activities.

To address these obstacles, we develop a spatial-dynamic model of an insect invasion of an urban forest with municipal jurisdictions and with both public and private ownership of trees.

Our spatial-dynamic model is for the optimal control of emerald ash borer (*Agrilus planipennis* Fairmaire) (EAB) in the Twin Cities metropolitan area of Minnesota, USA. EAB is a non-native forest insect that was discovered in the USA and Canada in 2002 and is poised to spread, colonize and kill native ash (*Fraxinus* spp.) trees throughout North America (Anulewicz et al., 2008). EAB is projected to cost homeowners and local governments billions of dollars for treatment or removal and replacement of landscape ash trees in cities (Kovacs et al., 2010). Satellite populations of EAB can become established when humans inadvertently transport infested nursery trees, logs, firewood, or related material. The established satellite population in the Twin Cities was detected in 2009 and is projected to expand and accelerate ash mortality throughout the Midwestern U.S. (Kovacs et al., 2011).

Our spatial-dynamic model focuses on managing host trees, which are a valuable, non-market resource in an urban setting (Sander and Haight, 2012). Managing the host is appropriate for pests that are difficult to detect while the host is readily identifiable and mediates pest population growth and spread. For example, Sims et al. (2010) address the optimal control of a destructive native forest insect by focusing on timber harvesting where trees provide both market and non-market benefits and mediate insect population density and damage. Management tactics for EAB include the application of a highly effective systemic insecticide to ash trees to kill EAB adults or larvae that are present and the pre-emptive removal and destruction of infested trees before larvae can complete development. Each tactic can effectively reduce EAB numbers, but only the insecticide preserves the urban forest benefits of a healthy tree. These tactics are considered more for infestations like the one in the Twin Cities that are localized and recently established than for infestations that have high EAB densities and an abundance of declining trees, which may preclude large-scale management efforts (McCullough et al., 2009c).

Our model accounts for spatial variation in the benefits of host trees and the costs of management actions, which may influence the spatial-dynamic paths of optimal management (e.g., Epanchin-Niell and Wilen, 2012). The model also accounts for the spatial variation in budgets of municipal jurisdictions, which is important when inaction due to budget restrictions in some jurisdictions allows the pest to grow and become a source of new invaders in neighboring jurisdictions. We use the model to develop and evaluate centralized control strategies for 17 municipal jurisdictions in the vicinity of the EAB infestation. Each jurisdiction may manage public trees but not private trees. The central planner determines the quantities of trees in public ownership to treat and remove over time, to maximize benefits of surviving trees net costs of management across public and private ownerships, subject to constraints on municipal budgets, management activities, and access to private trees. By aggregating budgets across jurisdictions, we estimate the gains in total net benefit attainable from budget sharing. By increasing the quantity of host trees accessible to public management, we estimate the gains in net benefit from coordinating management across public and private ownerships.

2. Methods

2.1. Dynamics of the emerald ash borer host

There is increasing interest in developing optimal control strategies for invasive insects and pathogens using models of host population dynamics coupled with economic models of management and damage (e.g., Ndeffo Mbah and Gilligan, 2010; Sims et al., 2010). Here, we model the spatial-dynamics of EAB in an urban forest by focusing on the population of ash trees that are potential hosts for EAB. Our model follows from a map grid representation of the spatially explicit growth and dispersal of an EAB population following initial colonization (Mercader et al., 2011). Our model consists of a grid of m square cells (sites) of equal size, each classified into a single land use. Within each cell, ash trees are subdivided into n ownership classes (e.g., public and private). The model accounts for the amounts (m^2) of susceptible and infested ash tree phloem, the inner bark of the tree consumed by larvae, by ownership class and time period based on assumptions about the growth and dispersal of fertilized EAB females. Phloem amounts can be converted to numbers of trees based on assumptions about tree size (McCullough and Siegert, 2007).

Let $S_{ij}(t)$ be the amount of susceptible phloem (m^2) in site i and ownership class j at the end of period t (December). We assume susceptible trees can be treated with an insecticide containing the active ingredient emamectin benzoate, which provides virtually 100% effective EAB control for 2 years (McCullough et al., 2011). We track the cumulative amount of treated phloem at the end of period t with the variable $R_{ij}(t)$ and define control variable $X_{ij}(t)$ as the amount of susceptible phloem treated with insecticide in the spring during period t. Susceptible trees can also be removed in the fall, as defined by the control variable $Y_{ij}(t)$, which prevents larvae within the tree from completing development (McCullough et al., 2009a). Insecticide treatment and removals both reduce the amount of phloem available for EAB larvae, and the cumulative amount of phloem removed is tracked with the variable $L_{ij}(t)$. The total amount of phloem removed from each site, ownership class, and time period must be less than the susceptible phloem present at the end of the previous period: $X_{ij}(t) + Y_{ij}(t) \leq S_{ij}(t-1)$.

The amount of susceptible phloem at the end of period t, $S_{ij}(t)$, also depends on phloem consumed by EAB larvae. We define $G_{ij}(t)$ as the reduction in susceptible phloem in site i and ownership j during period t from larval progeny of adult beetles that emerge from infested phloem, disperse, and lay eggs in susceptible phloem during the spring and summer of period $t-1$. Using these definitions, we model the dynamics of EAB in each site and ownership as a system of difference equations:

$$S_{ij}(t) = S_{ij}(t-1) - X_{ij}(t) - Y_{ij}(t) - G_{ij}(t) + X_{ij}(t-2) \quad (1)$$

$$C_{ij}(t) = C_{ij}(t-1) + G_{ij}(t) \quad (2)$$

$$R_{ij}(t) = R_{ij}(t-1) + X_{ij}(t) - X_{ij}(t-2) \quad (3)$$

$$L_{ij}(t) = L_{ij}(t-1) + Y_{ij}(t) \quad (4)$$

Each period, the susceptible phloem is reduced by the amount of phloem treated, the amount of phloem removed by managers, and the amount of phloem consumed by larvae during the period (Eq. (1)). The cumulative amount of phloem consumed by the end of period t, $C_{ij}(t)$, is the amount of phloem consumed in earlier periods and the amount of phloem consumed by larvae during period t (Eq. (2)). The cumulative amount of treated phloem or removed phloem is updated at the beginning of the period (Eqs. (3) and (4)). The insecticide is effective for only 2 years, consistent with field studies (Herms et al., 2009; McCullough et al., 2011).

Following Mercader et al. (2011), we assume that EAB adults emerge from infested phloem, disperse, mate, and produce larvae during period $t-1$, and these larvae overwinter and consume phloem during period t. The number of EAB adults that emerge from infested phloem in site k and ownership j is $\mu G_{kj}(t-1)$, where $\mu = 88.9$, the average number of adults that can emerge per m^2 of phloem (McCullough and Siegert, 2007). While larvae in newly infested trees often require 2 years to develop (Cappaert et al., 2005; Tluczek et al., 2011), to simplify our model, we assume all larvae mature and adults emerge after 1 year of development. We define p_{ik} as the expected proportion of EAB adults that emerge from phloem in site k and move to site i where p_{ik} is a negative exponential function of the distance between sites i and k. The number of adults arriving in site i is, $\sum_{k=1}^{m} p_{ik} \sum_{j=1}^{n} \mu G_{kj}(t-1)$ and the number of eggs that can potentially be laid by adults reaching site i is, $r\sum_{k=1}^{m} p_{ik} \sum_{j=1}^{n} \mu G_{kj}(t-1)$ where $r = 10.35$ is the annual population growth rate (Mercader et al., 2011).

We model the reduction in susceptible phloem during period t, $G_{ij}(t)$, as the product of the susceptible phloem in site i and ownership j (after treatment and removal in period t) and the proportion of the aggregate susceptible and treated phloem in site i that is colonized by EAB at the end of the previous period:

$$G_{ij}(t) = (S_{ij}(t-1) - X_{ij}(t) - Y_{ij}(t) + X_{ij}(t-2)) \left(\frac{r\sum_{k=1}^{m} p_{ik} \sum_{j=1}^{n} \mu G_{kj}(t-1)}{\mu \sum_{j=1}^{n} (S_{ij}(t-1) + R_{ij}(t-1))} \right) \quad (5)$$

This equation models the spatial dynamics of the spread of EAB among cells. The proportion of the susceptible and treated phloem colonized is the ratio of two terms. The numerator is the numbers of eggs that can potentially be laid by adults reaching cell i. The denominator is the number of larvae that could be potentially supported by both susceptible and treated phloem.

There are several assumptions associated with this model of phloem consumption. First, we assume that infested phloem on public and private land in period $t-1$, $G_{ij}(t-1)$, is identified and removed during period $t-1$ after one cohort of adult EAB has emerged. Under this assumption, the variable $G_{ij}(t-1)$ represents the mandated removal of infested phloem and differs from the decision variable $y_{ij}(t-1)$, which represents pre-emptive removal of susceptible phloem during period $t-1$. Second, adult EAB disperse to each site and segregate between susceptible and treated phloem in proportion to the amounts of susceptible and treated phloem present. Adults reaching phloem in period $t-1$

that is treated in periods $t-1$ or t either die because they consume toxic foliage or lay eggs and produce larvae that die before damaging the tree and do not produce a new generation of adults in period t (McCullough et al., 2011). Third, larvae of adults that reach phloem in period $t-1$ that is pre-emptively removed in period t also perish before maturing. Adults that reach phloem in period $t-1$ that is neither treated nor pre-emptively removed produce larvae that consume phloem and produce the next generation of adults in period t. Finally, for simplicity, we assume a constant EAB population growth rate rather than logistic growth (e.g., Burnett et al., 2012; Eiswerth and Johnson, 2002). As a result, a non-negativity constraint,

$$1-\left(\frac{r\sum_{k=1}^{m}p_{ik}\sum_{j=1}^{n}\mu G_{kj}(t-1)}{\mu\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))}\right)\geq 0 \tag{6}$$

is necessary to prevent actual larval production from exceeding the capacity of ash phloem to produce larvae. When the potential larval production exceeds the phloem capacity, the proportion in Eq. (5) equals 1, indicating all of the susceptible phloem is consumed.

2.2. Management problem

The central planner determines the quantities of trees in public ownership to treat and remove over time, to maximize benefits of surviving trees net costs of management across public and private ownerships, subject to constraints on municipal budgets, management activities, and access to private trees. Several economic parameters are needed to complete the formulation, and the units of the parameters are in terms of m^2 of phloem. The annual benefit of a healthy tree (\$/m^2) is b_i, which depends on the land use of site i. The discount factor to make value consistent across time is δ_t. The present value of an annuity that delivers an annual payment of one unit of the currency for k years is $a_{k|}$. The net cost (\$/m^2) at site i of removing a m^2 of phloem, and replacing it with the phloem of another species of tree is $c_1^i=d_i-b_ia_{40}\delta_{30}$ where d_i is the cost of removing each m^2 of phloem in site i and replacing it with another species of tree and $b_ia_{40|}\delta_{30}$ is the present value of the stream of benefits from each m^2 of replacement phloem. The stream of benefits from replacement phloem is assumed to begin 30 years after the tree is planted and lasts for 40 years (McPherson et al., 2005). The cost (\$/m^2) of treating each m^2 of phloem with insecticide in site i is c_2^i. The planner has a budget $B(t)$ (\$) each period to spend on treating and removing susceptible trees in public ownership. The planner also knows the management activities that are applied to trees in the private ownership.

The problem is to maximize net benefits of the ash trees:

$$\max_{X_{i1}(t),Y_{i1}(t),S^{ij}_{T+1},C^{ij}_{T+1},R^{ij}_{T+1},L^{ij}_{T+1}}:\sum_{t=1}^{T}\alpha\delta_t\left(\sum_{i=1}^{m}\sum_{j=1}^{n}b_i(S_{ij}(t)+R_{ij}(t))-c_1^i(G_{ij}(t)+Y_{ij}(t))-c_2^iX_{ij}(t)\right)+V(T+1) \tag{7}$$

subject to:

$$B(t)\geq\sum_{i=1}^{m}(c_2^iX_{i1}(t)+d_iY_{i1}(t)) \tag{8}$$

$$S_{ij}(0)=S_0^{ij},\quad C_{ij}(0)=C_0^{ij},\quad R_{ij}(0)=0,\quad L_{ij}(0)=0 \tag{9}$$

$$S_{ij}(T+1)=S_{T+1}^{ij},\quad C_{ij}(T+1)=C_{T+1}^{ij},\quad R_{ij}(T+1)=R_{T+1}^{ij},\quad L_{ij}(T+1)=L_{T+1}^{ij} \tag{10}$$

$$V(T+1)=\alpha\delta_{T+1}\sum_{i=1}^{m}\sum_{j=1}^{n}(h_iS_{T+1}^{ij}+e_iR_{T+1}^{ij}) \tag{11}$$

where

$$h_i=b_ia_3-\delta_3c_1^i,\quad e_i=b_ia_5-\delta_5c_1^i,\quad c_1^i=d_i-b_ia_{80}\delta_{60} \tag{12}$$

$$X_{i1}(t) \geq 0, \quad Y_{i1}(t) \geq 0, \quad X_{i2}(t) = \varphi_i, \quad Y_{i2}(t) = 0 \tag{13}$$

and the spatial dynamics of EAB (Eqs. (1)–(6)). The objective (Eq. (7)) is to determine $X_{i1}(t)$ and $Y_{i1}(t)$, the numbers of trees in public ownership to treat and remove over time, to maximize the present value of the benefits of surviving trees net costs of management across both public and private ownership over a fixed time horizon T. The number of trees in private ownership that are treated, represented by $X_{i2}(t)=\varphi_i$, and removed, as represented by $Y_{i2}(t)=0$, is determined outside the model (see Section 3). Benefits accrue to surviving trees, which can either be susceptible or treated. Management costs include the cost of treating susceptible ash trees, $c_2^i X_{ij}(t)$, the cost of pre-emptively removing susceptible ash trees, $c_1^i Y_{ij}(t)$, and the cost of removing ash trees that are infested and killed by EAB in the current year, $c_l^i G_{ij}(t)$.

Eq. (8) is the planner's budget constraint, which is an upper bound on the total cost of treating and removing susceptible ash trees in the public ownership. The budget constraint does not include the costs of removing infested trees because we focus on the budget available for implementation of management tactics to slow population growth of EAB. Eq. (9) is the initial conditions of the state variables. Note that the objective function includes a term for the value of the ending inventory, $V(T+1)$. Eq. (10) defines the control variables for period $T+1$, and Eqs. (11) and (12) define the value of the ending inventory. We assume susceptible and treated trees present in period $T+1$ produce benefits for 3 and 5 years without management, respectively, before eventually being infested and then removed and replaced. Eq. (13) includes the non-negativity constraints on the management activities for trees on public ownership and the fixed values of the management activities for trees on private ownership. We solve this problem with the Generalized Algebraic Modeling System (GAMS) 23.5.1 using the non-linear programming solver CONOPT from AKRI Consulting and Development.

2.3. Optimality conditions

The optimal treatments and removals at site i for the control problem (7) can be examined from the intuition offered by the necessary conditions (A1) and (A2) derived in the Supplementary Material. The necessary conditions for the optimal level of phloem treated or removed and replaced are as follows:

Condition for optimal treatment:

$$\begin{aligned}
&\overbrace{c_1^i \delta_t \left(\frac{r\sum_{k=1}^{m} p_{ik} \sum_{j=1}^{n} G_{kj}(t-1)}{\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))} \right) + \lambda_R^{ij}(t)}^{\text{Marginal benefit for site}\,i} \\
&+ \overbrace{\left(\sum_{h=1}^{n} (c_1^h \delta_{t+1} + \lambda_S^{hj}(t+1))(S_{hj}(t) - X_{hj}(t+1) - Y_{hj}(t+1) + X_{hj}(t-1)) \left(\frac{r p_{hi}}{\sum_{j=1}^{n}(S_{hj}(t)+R_{hj}(t))} \right) \right)}^{\text{Marginal benefit outside site}\,i} \\
&\quad \times \left(\frac{r\sum_{k=1}^{m} p_{ik} \sum_{j=1}^{n} G_{kj}(t-1)}{\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))} \right) \\
&= c_2^i \delta_t + \lambda_S^{ij}(t) \left(1 - \frac{r\sum_{k=1}^{m} p_{ik} \sum_{j=1}^{n} G_{kj}(t-1)}{\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))} \right)
\end{aligned} \tag{14}$$

Condition for optimal pre-emptive removal and replacement:

$$\overbrace{\left(\sum_{h=1}^{n}(c_1^h\delta_{t+1}+\lambda_S^{hj}(t+1))(S_{hj}(t)-X_{hj}(t+1)-Y_{hj}(t+1)+X_{hj}(t-1))\left(\frac{rp_{hi}}{\sum_{j=1}^{n}(S_{hj}(t)+R_{hj}(t))}\right)\right)}^{\text{Marginal benefit outside site}\,i}$$
$$\times\left(\frac{r\sum_{k=1}^{m}p_{ik}\sum_{j=1}^{n}G_{kj}(t-1)}{\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))}\right)$$
$$=(\delta_t c_1^i+\lambda_S^{ij}(t))\left(1-\frac{r\sum_{k=1}^{m}p_{ik}\sum_{j=1}^{n}G_{kj}(t-1)}{\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))}\right) \tag{15}$$

Eq. (14) indicates the treatment of a tree at site i continues as long as the marginal benefit for site i – present value of the savings from not replacing a healthy tree, $c_1^i\delta_t$, for the proportion of the tree affected by EAB in this period, $\left(r\sum_{k=1}^{m}p_{ik}\sum_{j=1}^{n}G_{kj}(t-1)\right)/\left(\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))\right)$, plus the shadow value of the resistant tree created, $\lambda_R^i(t)$ – plus the marginal benefit of treatment outside site i – the present value of the savings from not replacing healthy trees plus the shadow value of healthy trees not consumed in the next period for all the trees at the sites h surrounding site i, $\sum_{h=1}^{n}(c_1^h\delta_{t+1}+\lambda_S^{hj}(t+1))(S_{hj}(t)-X_{hj}(t+1)-Y_{hj}(t+1)+X_{hj}(t-1))\left(rp_{hi}/\left(\sum_{j=1}^{n}(S_{hj}(t)+R_{hj}(t))\right)\right)$, weighted by the proportion of the tree affected by EAB this period at site i, $\left(\left(r\sum_{k=1}^{m}p_{ik}\sum_{j=1}^{n}G_{kj}(t-1)\right)/\left(\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))\right)\right)$ – exceeds the present value of the treatment cost, $\delta_t c_2^i$, plus the shadow value of the proportion of the treated trees that would have not been consumed by EAB this period, $\lambda_S^i(t)\left(\left(r\sum_{k=1}^{m}p_{ik}\sum_{j=1}^{n}G_{kj}(t-1)\right)/\left(\sum_{j=1}^{n}\left(S_{ij}(t-1)+R_{ij}(t-1)\right)\right)\right)$.

The removal and replacement of trees at site i (Eq. (15)) continues as long as the marginal benefit of removal and replacement outside site i (equivalent to the formulation of marginal benefits outside site i in (Eq. (14))) exceeds the present value of the replacement cost plus the shadow value of the healthy trees, $\delta_t c_1^i+\lambda_S^{ij}(t)$, for the proportion of the removed trees that would have not been consumed by EAB during this period, $\lambda_S^i(t)\left(\left(r\sum_{k=1}^{m}p_{ik}\sum_{j=1}^{n}G_{kj}(t-1)\right)/\left(\sum_{j=1}^{n}(S_{ij}(t-1)+R_{ij}(t-1))\right)\right)$. The optimality of removal and replacement (Eq. (15)) indicates that all the trees should be removed from the site if the trees would instead be killed by EAB.

Comparing the left-hand side of Eqs. (14) and (15) indicates that the insecticide treatment provides a greater marginal benefit than tree removal because while each action provides an identical marginal benefit outside site i, only the insecticide treatment provides a marginal benefit at site i. However, if the cost of treatment is higher than the cost of removal, the preferred management option for that site could be removal. Comparing the right-hand side of Eqs. (14) and (15) indicates that as the proportion of the trees affected by EAB in this period approaches one, the cost of the removal action approaches zero, while the cost of insecticide treatment approaches the present value of treatment cost. This suggests that in sites close to a heavy infestation, removals may be a priority unless there is a large marginal benefit at site i of the insecticide treatment.

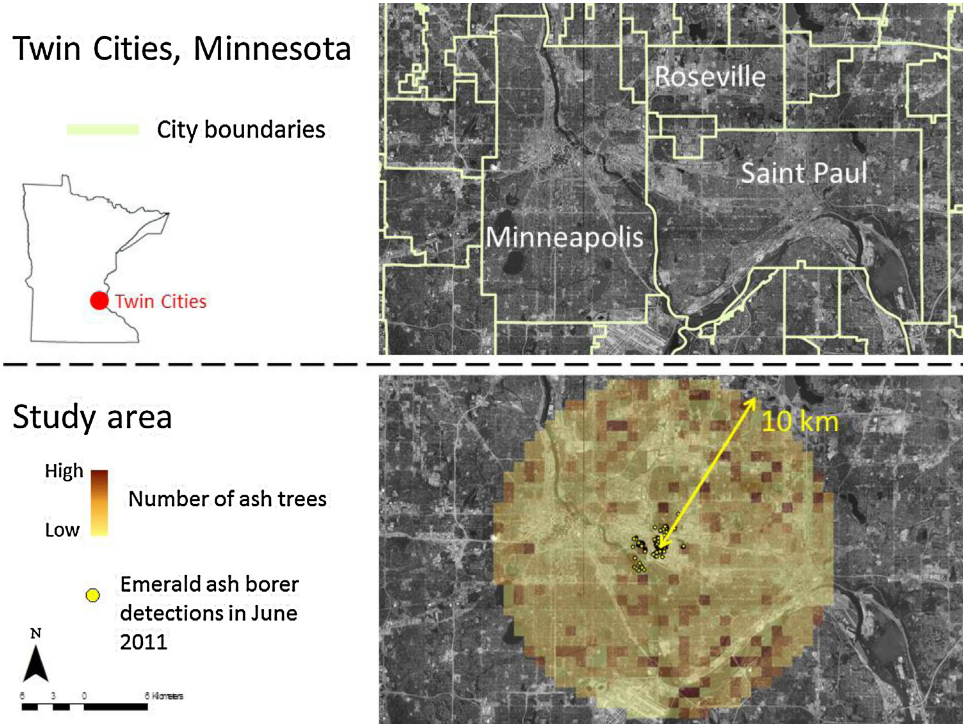

Fig. 1. Municipalities and the 10-km circular study area within the Twin Cities, Minnesota.

3. Data

In our model, 17 cities in the Minneapolis – St. Paul metropolitan area could have EAB infestations by 2016 (Fig. 1). The study area is defined by a 10 km radius around the known epicenter of the infestation, as delineated in the summer of 2009. The 10 km radius is based on an estimated distance that EAB could naturally spread in the next 5 years projected from the dispersal observed in field studies (Siegert et al., 2010). We chose a 5-year period, 2012–2016, for the purpose of examining a short-horizon policy response to the invasion. Projecting the infestation and management program beyond 5 years requires assumptions about potential human transport of EAB to other parts of the metropolitan area that are difficult to justify.

To compute the discounted net benefits of an EAB management strategy, we compile three types of data. First, we estimate the initial amounts of susceptible and infested ash phloem public and private ownership in 600 m^2 square cells. Next, we estimate the benefits of healthy ash trees, costs of treating trees, costs of removing trees, and the budgets available for treating or removing ash trees in each of the cities. Finally, we identify the probabilities that EAB will spread from each cell in the study area to other sites using a modified model developed for predicting the localized spread of EAB (Mercader et al., 2011).

3.1. *Estimating susceptible and infested ash phloem in public and private ownership*

We first classify each 600 m^2 cell into one of three land uses: residential, park, and non-residential. The cell classification is based on a 2005 land use classification system developed from aerial photos and assessor information (Metropolitan Council, 2006). Residential land use includes single and multi-family residential and mixed-use. Park land use includes parks, preserves, golf courses, and undeveloped land. The non-residential land use includes office, retail and other commercial, industrial and utility, and institutional uses.

While each cell is classified into a single land use, any ash phloem in a cell is divided between public and private ownerships. In residential and non-residential cells, we assume that 30% of the ash phloem is in public ownership and accessible to public management. This represents an estimate of the proportion of street trees on residential land (McPherson et al., 2005; Nowak et al., 2006), and we assume the same proportion of street trees on non-residential land. Based on an estimate that 20% of open space in St. Paul is covered by golf courses (City of Saint Paul, 2008), we assume that 20% of the ash phloem in park cells is in private ownership and inaccessible to management. We also assume that a proportion of the ash phloem in private ownership is treated by owners to prevent EAB infestation. On residential land, 25% of private ash phloem is treated. On non-residential and park land, 5% of the private ash phloem is treated. These proportions are based on estimates of the proportions of phloem associated with mature ash trees, which provide higher benefits and are more likely to be treated than younger ash trees (Kovacs et al., 2010).

Next, we estimate the amount of ash phloem in 600 m^2 sites using an urban ash tree inventory for the Minneapolis – St. Paul metropolitan area collected by the Minnesota Department of Agriculture (MDA) in 2009 and 2010 and the National Land Cover Database (NLCD) 2001 tree canopy layer. The MDA conducted three ash tree assessments in summer of 2009 and in winter and summer of 2010 in the neighborhoods surrounding the infested ash trees found in St. Paul in summer of 2009. The three ash tree assessments identify nearly all of the ash, both public and private, in the 1.6 km radius around the known infestations. The MDA urban ash inventory also includes the boulevard and park ash trees from inventories conducted in the cities of Minneapolis, Falcon Heights, and St. Paul.[1]

Many municipalities in the study area have not inventoried ash growing on boulevards or in parks, and none of the cities have inventoried ash trees on private property. For this reason, we predict the number of ash trees in each cell based on tree canopy and land use. For each land use, we use ordinary least squares regression to estimate a relationship between the number of ash trees from the MDA urban ash inventory and the NLCD index of tree canopy at a 30 m^2 cell size resolution. To simplify modeling, we assume that all ash trees are 30 cm in diameter at breast height (DBH), which is the average size of trees in the MDA inventory. Then, we subdivide each 600 m^2 cell into 30 m^2 cells, each with tree canopy index from the NLCD, and use the regression equation to estimate the number of ash trees present. Finally, we aggregate those estimates to obtain the number of ash trees in the 600 m^2 cell. Ash trees are converted to phloem by assuming 14.14 m^2 of ash phloem are present in each ash tree (30 cm DBH) (McCullough and Siegert, 2007).

Finally, we estimate the amount of ash phloem that is consumed by EAB in the beginning of the study period, $G_{ij}(0)$ and $C_{ij}(0)$. Personnel from the MDA conducted surveys to delineate sites with EAB infestations in 2011. We assume eight larvae per m^2 of phloem are present in each infested site in 2011 because this is the minimum larval density that can be detected (Mercader et al., 2012). On average, 1 m^2 of phloem can develop 88.9 larvae (McCullough and Siegert, 2007). Therefore, we assume that 9% of the phloem in infested sites is consumed in period zero.

3.2. Estimating benefits of healthy ash trees and costs of treatment and removal

McPherson et al. (2005) compute the values for the annual benefits of a healthy green ash tree, which include improved local air quality, energy savings from a tree growing near buildings, sequestration of carbon dioxide, reduced storm water runoff, and the shading, recreation, and the esthetic value of the trees. For residential land use, we assume the annual benefits of an ash tree other than the sequestration of carbon dioxide, are capitalized into the value of the residential properties close to the tree. McPherson et al. (2005) estimate the average annual increase in property value from an ash tree on the street to be \$40.06. Based on the land cover categories in McPherson et al. (2005), we estimate this will be higher at \$49 on residential streets because of the greater proximity of the street

[1] Most of the boulevard and park ash are not inventoried in St. Paul, but are inventoried in Minneapolis and Falcon Heights.

tree to valuable property. McPherson et al. (2005) calculate the average annual carbon sequestration benefit of a green ash is $5. The annual benefit of an ash tree in the residential land use is then $54.

We estimate the annual benefits of trees growing in parks or non-residential lands using the direct value of some of their ecosystems services because no studies have looked at how a single tree contributes to property values for these land uses. The annual benefit of a tree growing in a park is $11, based on its value derived from improved local air quality and carbon dioxide sequestration (McPherson et al., 2005). The annual benefit of a tree in non-residential land use is $30, based on its value derived from carbon dioxide sequestration, half of the energy savings generated by a tree growing near buildings (since energy use is lower after business hours), and improved local air quality (McPherson et al., 2005). Recognizing that the values of all ecosystem services may not be included in these estimates, we determine the sensitivity of optimal EAB management strategies to increasing the value of individual tree benefits.

When an ash tree is killed by EAB or pre-emptively removed, there is an immediate removal and replacement cost. We assumed the cost of removal and replacement of an ash tree in the residential and non-residential land use is $800 and in the park land use is $600 (Kovacs et al., 2010). There is a delay of 30 years until the annual benefit of a replacement tree equals that of the ash tree removed because of EAB, and the replacement tree provides a constant annual benefit for 40 years (McPherson et al., 2005). We use an annual real rate of discount of 2% (Howarth, 2009).

The trunk injection of the insecticide with emamectin benzoate was assumed to prevent colonization and injury from EAB for 2 years (McCullough et al., 2011). We assumed trees would be treated by private contractors, rather than by in-house labor. Treatment cost for an ash tree in the residential and non-residential land use is, therefore, assumed to be $120 and in the park land use is $100 (Kovacs et al., 2010). The treatment cost for 10-year commitment of insecticide is $540 for non-park land use and $450 for the park land use, slightly lower than five individual treatments because of a city's ability to buy in bulk over a long time horizon. The purpose is to see if and where communities would adopt a 10-year commitment with a slight cost advantage but the obvious constraint of a commitment over 10 years. McCullough and Mercader (2012) report the cost of treating municipal trees with in-house labor is lower at $48.

3.3. *Calculating EAB dispersal probabilities*

Field studies indicate that most adult EAB females select hosts within 200 m of their point of emergence when ash trees are available (Mercader et al., 2009; Siegert et al., 2010). Little is known about dispersal of EAB beyond 800 m, but a small proportion of beetles probably go further. Laboratory tests showed mated EAB females are physiologically capable of flying 1.8 km/day (Taylor et al., 2006). Dispersal of EAB females is also likely to be affected by host searching behaviors, which are predicted to have large effects on the spread of the EAB (Mercader et al., 2011).

To project EAB spread over time, we used field data collected from the SL.ow A.sh M.ortality (SLAM) pilot project, which represents the largest intensively sampled EAB infestation to date. As part of this project, ash trees and EAB distribution were intensively surveyed from 2008 to 2010 in an area centered on the origin of the infestation (SLAME info, 2012). The EAB infestation was estimated to have originated 4–6 years prior to the detection of the infestation in 2007 (approximately 2002) (Poland and McCullough, 2010). We estimated the detectable spread of the infestation from the estimated origin from 2002 to 2010.

This information was subsequently used to adjust the dispersal function developed by Mercader et al. (2011) to account for the lower resolution of the data at the SLAM pilot project. The model presented in Mercader et al. (2011) uses a negative exponential function to estimate the spread of EAB based on results from field studies of EAB dispersal (Mercader et al., 2009; Siegert et al., 2010). To adjust the dispersal function, we ran simulations varying the parameters of the dispersal function in the SLAM pilot project and contrasted the predicted spread with the observed spread. The mean square difference was used to determine the best fitting parameters. For these simulations, maximum dispersal of individual EAB females was assumed to be 5 km per year.

We used this adjusted model to calculate the expected dispersal probabilities for the St. Paul – Minneapolis study area. We subdivided each 600 m^2 cell in the St. Paul – Minneapolis study area into nine 200 m^2 cells. We then calculated the expected dispersal probability between cells in the subdivided environment, and used these results to calculate the expected dispersal probabilities for the study area containing 600 m^2 cells.

4. Management strategies

We develop and evaluate five centralized strategies for 17 municipal jurisdictions surrounding the infestation. For each strategy, the central planner determines the quantities of trees in public ownership to treat and remove over time, to maximize benefits of surviving trees net costs of management across public and private ownerships, subject to constraints on municipal budgets, management activities, and access to private trees. For constraints on municipal budgets, we use projections of 5-year budgets made by the municipalities. St. Paul anticipates spending \$3.8 million, Roseville \$0.1 million, and the remaining cities plan to spend nothing. Strategy one is no-management where EAB population spread is simulated without any effort to treat or remove ash trees. Strategy two is centralized management using an aggregate budget constraint of \$3.9 million that may be shared among cities. Strategy three is centralized management using local municipal budget constraints equal to their 5-year projections. Strategy four is centralized management with an aggregate budget of \$3.9 million where the planner may use pre-emptive removal as a management action but not insecticide treatment. Strategy five is centralized management with an aggregate budget of \$3.9 million where action is limited to insecticide treatment.

We performed sensitivity analyses for some of the strategies. For strategy 2, we explored how changing the level of the aggregate budget and increasing the percentage of phloem that is accessible to public management influences optimal public management and total net benefit. We also determined how changing the value of individual trees influences optimal management. For this latter analysis, the low-end value of a tree in the residential land use is half the baseline value, or \$27 per tree, and in park or non-residential land the value per tree is derived solely from the carbon dioxide sequestration, \$5. The high-end value of a tree includes the value of reduced storm water runoff for all land uses, which increases the value of a tree in residential land use to \$102 per tree, in the park land use to \$59, and in the non-residential land use to \$78 (McPherson et al., 2005). For strategy 3, we explored the impact of changing the total budget by proportionally changing the budget constraints for St. Paul and Roseville, the two cities committed to spending. We also explored the effects of increasing the percentage of phloem that is accessible to public management. Finally, to further explore the effects of budget sharing, we developed a strategy with an aggregate budget constraint of \$3.8 million for the cities of Minneapolis and Saint Paul while the other cities have budget constraints equal to their 5-year projections.

The City of Saint Paul has the largest budget for EAB management, and in strategy 2 with an aggregate budget constraint, we reallocate this budget to surrounding cities to examine the gain in total net benefits throughout the study area. To investigate if St. Paul has an incentive to permit such a reallocation, we analyze a set of optimal centralized management strategies with increasingly aggregated budget constraints. The strategies include separate budget constraints for each city (strategy 3) and aggregate budget constraints for St. Paul and Minneapolis, for St. Paul and Roseville, for St. Paul, Minneapolis, and Roseville, and for all cities in the study area (strategy 2). For each strategy, we compute the net benefit accruing to the City of Saint Paul. Then, we use the net benefit for St. Paul obtained under strategy 3 as the baseline for comparison with the net benefit for St. Paul obtained under the other strategies in which St. Paul's budget is shared. The difference is the gain or loss in net benefit for St. Paul from reallocating its management budget to surrounding cities. Presumably, reallocating a portion of St. Paul's budget to surrounding cities will increase the net benefits for St. Paul because the reallocation slows EAB spread across city borders.

For all strategies, we assume that trees treated with insecticide are immune to infestation for 2 years, and then return to the susceptible class. We also included a management activity that assumes a 10-year commitment to insecticide treatment.

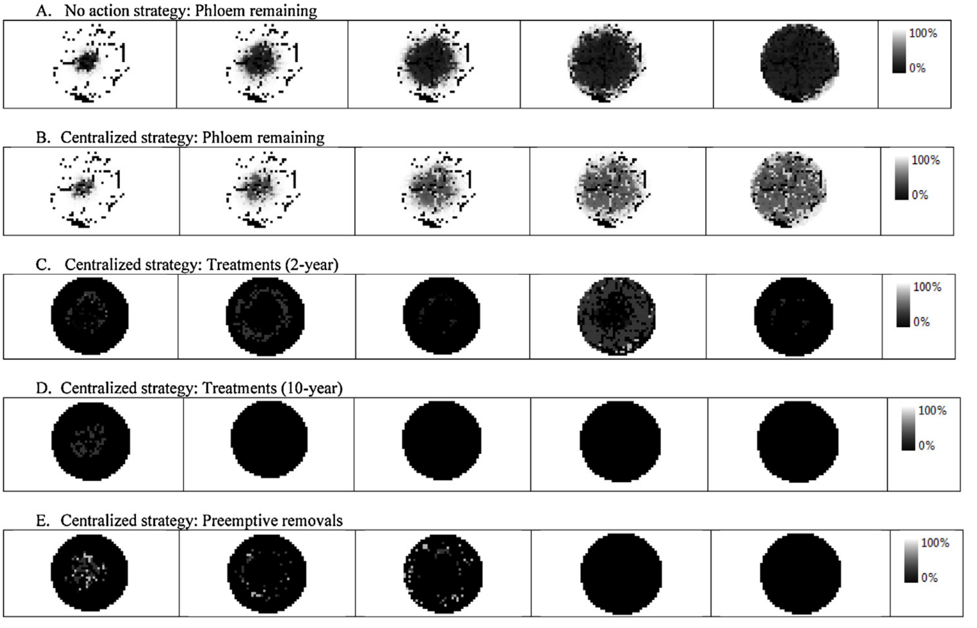

Fig. 2. (A–E) Predicted phloem consumed (No action and centralized strategy) and management action (Centralized strategy) from 2012 to 2016.

5. Results

With no management, the EAB population consumes phloem in the central part of the study area where the infestation originated. Over time, the population expands toward the periphery of the study area (Fig. 2A). After 5 years, one-third of the phloem remains within the area bounded by the 10 km radius. As a result of ash mortality associated with the infestation, the urban forest benefit of the ash trees decreases from \$0.435 million in year 1 to \$0.135 million in year 5, while costs of removing infested ash trees increase from \$0.161 to \$1.981 million (Table 1). Costs of tree removal exceed the benefits of the urban forest, resulting in a negative total net benefit (\$−0.330 million).

With an aggregate budget constraint of \$3.9 million (strategy 2), a combination of insecticide treatments and pre-emptive removal of ash trees in the area surrounding the initial infestation slows EAB spread and reduces phloem consumption (Fig. 2B). The total net benefit (\$0.220 million) is positive and at least two times greater than the net benefits of the other strategies, except strategy 5, which is constrained to treatment only (Table 1). Compared with no management (strategy 1), the management activities applied in strategy 2 protect trees from EAB consumption, thereby increasing the benefits of the urban forest and reducing the costs of removing infested trees. Treatment activities increase net benefits because the cost of treating a healthy tree is less than the cost of removing an infested tree. Further, treatments combined with pre-emptive removals reduce susceptible phloem and thereby slows EAB population growth and spread.

With local municipal budget constraints (strategy 3), insecticide treatment and pre-emptive removal are concentrated in the city of St. Paul (the SE portion of the study area), which has the largest budget (Fig. 3). The EAB infestation expands and larvae consume most of the ash phloem outside of St. Paul. Compared with strategy 2, the benefits of the urban forest decrease at a greater rate over time as infested ash trees are removed and the total net benefit (\$−0.140 million) is lower (Table 1).

With constraints that allow an aggregate budget and only pre-emptive tree removal (strategy 4), total net benefit (\$−0.160 million) is less than the benefits of strategies 2 and 3, which allow both pre-emptive removal and insecticide treatment (Table 1). A strategy that is restricted to pre-emptive removal is less efficient because the costs of removal are much higher than the costs of insecticide

Table 1
Benefits and costs ($thousands) of five centralized EAB management strategies for 17 municipal jurisdictions in the Twin Cities metropolitan area of Minnesota, USA.

Strategy	Year					PV	Total net benefits[a]
Benefits and costs	2012	2013	2014	2015	2016		
(1) No management							
Urban forest benefit	435	408	345	248	135	1523	−330
Removal cost	161	460	1083	1607	1981	4985	
(2)Aggregate budget for all cities							
Urban forest benefit	438	426	393	348	287	1825	220
Removal cost	97	197	544	728	1034	2448	
2yr treat cost	15	30	2	233	1	266	
10yr treat cost	119	0	0	0	0	119	
Preempt removal cost	132	81	126	0	0	333	
(3) Local budgets							
Urban forest benefit	436	417	367	290	187	1641	−140
Removal cost	130	339	845	1274	1777	4107	
2yr treat cost	6	9	0	65	0	77	
10yr treat cost	75	0	0	0	0	75	
Preempt removal cost	76	19	26	0	0	120	
(4) Aggregate budget for all cities – removal only							
Urban forest benefit	438	423	386	332	240	1756	−160
Removal cost	101	250	582	867	1510	3110	
Preempt removal cost	245	189	328	473	0	1189	
(5) Aggregate budget for all cities – treatment only							
Urban forest benefit	438	427	399	351	289	1835	210
Removal cost	96	192	498	768	1081	2479	
2yr treat cost	25	71	8	246	4	337	
10yr treat cost	152	0	0	0	0	152	

[a] Total net benefits, calculated in Eq. (7), indicates the net present value of the tree benefits less the costs of management action across the study horizon plus the value of the ending inventory.

treatment. When pre-emptive removal is the only management action, susceptible phloem available to EAB is reduced, and the slower population growth of EAB retains some urban forest benefits. Pre-emptive removal can only slow the population growth as long as there are trees to remove.

With constraints that allow an aggregate budget and only insecticide treatment (strategy 5), total net benefit ($0.210 million) is comparable to strategy 2, which has an aggregate budget and allows both

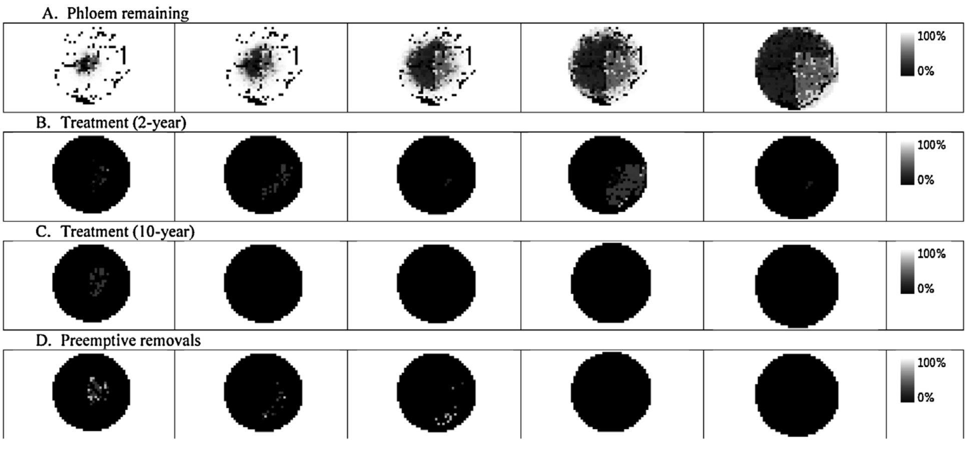

Fig. 3. (A–D) Predicted phloem consumed and remaining, and management action from 2012 to 2016: Localized strategy.

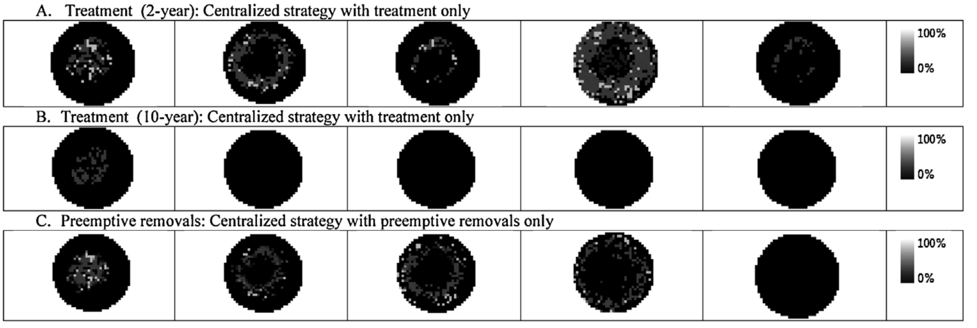

Fig. 4. (A–C) Predicted management action from 2012 to 2016: Centralized strategy with pre-emptive removals only and insecticide treatment only.

insecticide treatment and pre-emptive removal (Table 1). Insecticide treatment reduces the susceptible phloem available to EAB at a lower cost than pre-emptive removal. Further, treated trees continue to provide urban forest benefits. Without pre-emptive removal, no action is taken on low-value ash trees in parks and industrial areas, and relative to strategy 2, this susceptible phloem is partially consumed and increases EAB population growth and removal of infested trees in later periods.

Insecticide treatment protects individual tree benefits and slows EAB population growth in nearby areas. Trees treated five times over 10 years (Figs. 2D and 3D) begin in the first period in the cells surrounding the initial infestation. This 10-year treatment commitment reduces the reproductive capacity of EAB in centrally located cells where the infestation has the potential to grow rapidly and spread to the widest possible area. Ten-year treatment is entirely on residential land where individual tree benefits are greatest.

Short term insecticide treatment, where trees are treated once over 2 years (Figs. 2C and 3C), occurs primarily in periods one, two, and four. Short term treatment is widespread in the first two periods to slow the population growth of the initial infestation and in the fourth period to retain tree value in the terminal period. Treatment mostly occurs on the edge of the advancing front to anticipate the short distance most mated females travel. To protect the highest tree benefits, treatment occurs mostly on residential land, but some treatment occurs on non-residential and park land. When treatment is the only management action allowed, more treatment is undertaken to compensate for the lack of pre-emptive tree removal (Fig. 4A and B).

Preemptive tree removal (Figs. 2E and 3E) occurs in the first three periods. The extensive insecticide treatment performed in period four makes pre-emptive removal of trees after the third period unnecessary. Preemptive removal occurs mostly in a few cells distant from the advancing front of the infestation to reduce phloem available for consumption and thereby reduce population growth. Most of pre-emptive removal occurs on park and non-residential land where individual trees provide the lowest benefits and are less expensive to remove. When the only management action allowed is pre-emptive tree removal, more removal is undertaken to compensate for the loss of EAB control provided by insecticide treatment (Fig. 4C).

We varied the aggregate budget available for treatment and pre-emptive removal under strategy 2 and found that total net benefit increased from \$−0.33 million with no management to \$0.19 million with a budget of \$1.00 million (Table 2). Increasing funding from \$1.00 to \$4.00 million improved total net benefit by \$0.03 million, indicating diminishing returns from management. These results suggest an aggregate budget of \$3.90 million, which is the projected budget for EAB control in the Twin Cities for the next 5 years, is located on the plateau of the benefit curve where a decrease in funding will not greatly influence total net benefit, provided that the aggregate budget is shared among cities.

For strategy 3 with the local budget constraints, we varied the total budget using proportional changes in the budgets for St. Paul and Roseville, the two cities with funding commitments. Total net benefit increased from \$−0.33 with no management to \$−0.16 million with a \$0.20 million budget (Table 2). There are no significant increases in net benefits for budgets higher than \$0.20 million.

Table 2
Percentage of phloem in the final period by condition class (treated, removed, consumed, and healthy) for centralized management strategies with alternative budget constraints (aggregate and local), benefits per tree, and budget levels.

Strategy Budget level (thousands)	Percentage treated	Percentage removed	Percentage consumed	Percentage healthy	Total net benefit[a] ($millions)
Aggregate budget for all cities					
$0	0.00	0.00	78.54	21.46	−0.33
$200	5.85	0.00	66.24	33.76	−0.02
$600	17.87	5.58	51.30	43.12	0.05
$1000	23.52	9.41	41.35	49.24	0.19
$4000	23.52	9.41	41.35	49.24	0.22
Local budgets					
$0	0.00	0.00	78.54	21.46	−0.33
$200	6.47	3.29	66.80	29.91	−0.16
$600	6.71	3.31	66.58	30.11	−0.15
$1000	6.93	3.63	65.64	30.73	−0.15
$4000	7.17	4.09	65.08	30.83	−0.14
Aggregate budget for all cities – low-end benefits per tree					
$0	0.00	0.00	78.54	21.46	−3.90
$200	4.98	3.62	64.95	31.43	−3.71
$600	12.40	12.89	47.35	39.76	−3.62
$1000	12.76	12.73	47.24	40.03	−3.61
$4000	12.94	18.41	41.73	39.85	−3.60
Aggregate budget for all cities – high-end benefits per tree					
$0	0.00	0.00	78.54	21.46	12.60
$200	11.41	0.00	59.53	40.47	13.56
$600	31.67	0.00	41.63	58.37	14.46
$1000	34.72	0.00	39.90	60.10	14.84
$4000	35.28	2.39	37.75	59.86	16.15

Note: The percentage of healthy phloem includes the unconsumed and treated phloem.

[a] Total net benefits, calculated in Eq. (7), indicates the net present value of the tree benefits less the costs of management action across the study horizon plus the value of the ending inventory.

Optimal management is two-thirds devoted to insecticide treatment and one-third to pre-emptive removal of trees throughout the budget range.

The value given to individual tree benefits affects optimal management in strategy 2 with the aggregate budget (Table 2). When the value per tree is low, insecticide treatment and pre-emptive removal are used in equal proportions. When the value per tree is high, the budget is allocated to insecticide treatment except at the highest budget level where a small proportion is used for pre-emptive removal. The value per tree also affects the shape of the total net benefit curve. With a low value per tree, the net benefit curve plateaus at a budget of $0.60 million. With a high value per tree, total net benefit is still increasing with $4.0 million budget.

We investigated how changing the percentage of phloem accessible to public management on residential and non-residential land affects total net benefits and optimal management in strategies 2 and 3 with aggregate and local budget constraints, respectively (Table 3). On private residential land that remains inaccessible to public management, we assume that 25% of the phloem is treated, and on non-residential land 5% is treated. In strategy 2 with an aggregate budget constraint of $3.9 million, total net benefits increase at a slowly diminishing rate, suggesting that increasing access to phloem for management is a good investment. The percentage of phloem treated and removed peaks with 70% access and then decreases as access increases to 100%, suggesting that centralized management with an aggregate budget of $3.9 million that is shared among the cities and spent on treatment and pre-emptive removal of ash in public and private ownership can effectively slow the spread of EAB. In strategy 3 with local municipal budget constraints, the percentage of phloem treated and removed rises steadily for all percentages of phloem available for management. Compared with strategy 2, total net benefit does not increase as quickly because phloem can be managed in only a portion of the study

Table 3
Percentage of phloem in the final period by condition class (treated, removed, consumed, and healthy) for centralized management strategies with alternative budget constraints (aggregate and local) and percentages of phloem accessible to public management on residential and non-residential land.[a]

Strategy Percentage of phloem available for public management	Percentage treated	Percentage removed	Percentage consumed	Percentage healthy	Total net benefits[b]
Aggregate budget for all cities					
10%	9.16	9.62	54.02	36.36	−0.08
30%	22.28	11.12	40.53	48.35	0.22
70%	47.43	11.12	16.28	72.59	0.76
100%	40.09	2.36	5.50	92.14	1.09
Local budgets					
10%	2.83	3.38	68.03	28.59	−0.23
30%	7.17	4.09	65.08	30.83	−0.14
70%	15.15	5.01	59.90	35.09	0.05
100%	20.13	3.64	58.53	37.83	0.16

[a] For phloem on residential land inaccessible to public management, we assume 25% is treated, and on non-residential land 5% is treated. The percentage of healthy phloem includes the unconsumed and treated phloem.
[b] Total net benefits, calculated in Eq. (7), indicates the net present value of the tree benefits less the costs of management action across the study horizon plus the value of the ending inventory.

area, allowing EAB to reproduce and expand in other areas. These results suggest that the value of increasing management activities on private land in St. Paul is a consequence of EAB spread from surrounding municipalities that have no management budgets.

It is useful to look at the effects of aggregating the local municipal budget constraints ($3.9 million total) on optimal management actions and total net benefit (Table 4). Relative to no-management, strategy 3 with local municipal budget constraints increases the percentage of healthy phloem in the final period by only 9.37% because most of the study area does not have funds for management. With an aggregate budget constraint for Minneapolis and St. Paul, an additional 11.29% of the phloem remains healthy in the final period because insecticide treatments and pre-emptive removals can be applied in a much greater portion of the study area. With an aggregate budget constraint for all municipalities (strategy 2), an additional 6.23% of the phloem remains healthy in the final period. The change from local municipal budget constraints (strategy 3) to an aggregate budget constraint (strategy 2) increases

Table 4
Percentage of phloem in the final period by condition class (treated, removed, consumed, and healthy) for centralized management strategies with alternative budget constraints and percentages of phloem accessible to public management.

Management strategy	Percentage treated	Percentage removed	Percentage consumed	Percentage healthy	Total net benefits[a]
No management	0.00	0.00	78.54	21.46	−0.33
Local budgets	7.17	4.09	65.08	30.83	−0.14
Aggregate budget for St. Paul – Minneapolis	16.16	9.48	48.40	42.12	0.08
Aggregate budget for all cities and 30% phloem accessible to public management	22.28	11.12	40.53	48.35	0.22
Aggregate budget for all cities and 100% phloem accessible to public management	40.09	2.36	5.50	92.14	1.09

Note: The percentage of healthy phloem includes the unconsumed and treated phloem.
[a] Total net benefits, calculated in Eq. (7), indicates the net present value of the tree benefits less the costs of management action across the study horizon plus the value of the ending inventory.

Table 5
Net benefit to St. Paul under for management strategies with different aggregate budget constraints across neighboring cities.

Management strategy	Net benefit to St. Paul[a]
Initial infestation at the center of the study area	
Local budgets	1.00
Aggregate budget for St. Paul – Minneapolis	11.10
Aggregate budget for St. Paul – Roseville	3.70
Aggregate budget for St. Paul – Minneapolis – Roseville	13.60
Aggregate budget for all cities	12.30
Initial infestation at the center of St. Paul	
Local budgets	1.00
Aggregate budget for St. Paul – Minneapolis	1.01
Aggregate budget for St. Paul – Roseville	1.02
Aggregate budget for St. Paul – Minneapolis – Roseville	1.00
Aggregate budget for all cities	1.02

[a] Net benefit is computed as the ratio of the net benefit to St. Paul in a given strategy to the net benefit to St. Paul under the strategy with the local budgets.

the percentage of trees treated from 7% to 22% and the percentage of trees pre-emptively removed from 4% to 11%. In addition, this change increases the percentage of healthy trees remaining in the final period by 18%, and more than doubles total net benefit.

The biggest increases in the percentage of healthy phloem and total net benefit occur when 100% of the phloem is available for management under an aggregate budget constraint (Table 4). In this case, 40% of the ash phloem is treated, less than 5% is pre-emptively removed, and EAB spread is stopped almost completely. As a result, over 90% of the ash phloem is healthy at the end of the 5-year study period and total net benefit ($1.09 million) is five times greater than the total net benefit ($0.22 million) of strategy 2, which also has an aggregate budget constraint but only has 30% of the ash phloem accessible.

The net benefit to the City of Saint Paul under strategy 3 in which budget constraints are imposed for each city is $100 thousand. The changes in St. Paul's net benefit from allowing its management budget to be re-allocated to surrounding cities are shown in Table 5. The strategy with an aggregate budget constraint for St. Paul and Minneapolis increases net benefit for St. Paul by 11 times because the infestation is slowed in Minneapolis, which slows the spread back to St. Paul. The strategy with an aggregate budget constraint for St. Paul and Roseville increases St. Paul's net benefits by 3.7 times. The maximum gain (13.6 times) is obtained from the strategy with an aggregate budget for St. Paul, Minneapolis, and Roseville. The management activities in St. Paul are largely unaffected by the budget reallocations suggesting that St. Paul has a surplus budget and benefits by sharing its surplus with neighboring cities to slow EAB spread between cities.

The results in Table 5 depend on the location of the initial EAB infestation. We repeated the analysis with the initial infestation moved from the border between St. Paul and Minneapolis to the center of St. Paul. In this case, there is little EAB spread between St. Paul and surrounding cities and the net benefit for St. Paul barely rises under strategies with aggregate budget constraints across those cities (Table 5).

6. Conclusions

We address two obstacles to local implementation of optimal regional control of a bio-invasion that damages public and private resources across jurisdictions: lack of funds to protect the public resource and lack of access to protect the private resource. To evaluate these obstacles, we develop a spatial-dynamic model of the optimal control of emerald ash borer in the Twin Cities metropolitan area of Minnesota, USA. The model focuses on managing host trees, which are a valuable resource that can be protected with an insecticide or removed to slow the EAB population growth. The model includes spatial variation in the ownership and benefits of host trees, the costs of management, and the budgets of municipal jurisdictions. We develop and evaluate centralized strategies for 17 jurisdictions surrounding the infestation. The central planner determines the quantities of trees in public ownership to treat and remove over time, to maximize benefits of surviving trees net costs of management across public and private ownerships, subject to constraints on municipal budgets, management activities, and access to private trees. We use the model to explore how centralizing the budget across political boundaries, restricting the use of treatments or removals, and increasing the quantity of host trees accessible to public management affect total net benefits.

The results suggest that enabling municipalities to aggregate their budgets greatly improves total net benefits. The change from local to aggregated budget constraints increases the percentage of trees treated from 7% to 22% and the percentage of trees pre-emptively removed from 4% to 11%. In addition, an aggregate budget increases the percentage of healthy trees remaining in the final period by 18%, and the total net benefits more than double. Local government officials in the Twin Cities currently focus on EAB and potential management activities in their own neighborhoods. Although officials are curious about regional approaches to EAB management and whether they could be more effective than jurisdiction-based approaches, there is little active coordination among jurisdictions (Dunens et al., 2013).

The results also suggest that even greater benefits can be attained by increasing the quantity of trees that are accessible to public management. When all trees in both public and private ownerships on residential and non-residential land are available for management with an aggregate budget, 40% of the ash is treated, less than 5% is pre-emptively removed, and EAB spread is stopped almost completely. As a result, over 90% of the ash phloem is healthy at the end of the 5-year study period and total net benefit is five times greater than the net benefit of the strategy with an aggregate budget and with inaccessible ash on private land. These potential gains suggest the value of incentive or regulatory programs for ash treatment and removal aimed at private landowners.

We find that strategies emphasizing insecticide treatment rather than pre-emptive tree removal are superior because treatments reduce susceptible phloem available to EAB at a lower cost while protecting the urban forest benefits of healthy trees. Two other simulation studies of EAB population growth in urban landscapes reach similar conclusions. McCullough and Mercader (2012) find that annual insecticide treatment of 20% of the ash population can protect 99% of trees after 10 years, and the cumulative costs of treatment are substantially lower than costs of removing dead or severely declining ash trees. Vannatta et al. (2012) find that a strategy of applying insecticide treatment to the entire ash population provides higher urban forest value than a strategy of pre-emptive removal and replacement of ash trees.

We find that short and long term commitments to insecticide treatment of trees are optimal in residential areas along the advancing front of the infestation where tree benefits are greatest. These findings corroborate results of previous studies with spatial-dynamic models, which show that pest control in a border around the invasion is most effective (Hof, 1998; Epanchin-Niell and Wilen, 2012). In addition, we find pre-emptive tree removal occurs on park and non-residential land distant from the advancing where tree benefits and removal costs are lower. This result is an example of the "forward-looking" nature of invasion control (Epanchin-Niell and Wilen, 2012) in the sense that pre-emptive removal reduces the phloem available for EAB when it arrives and thereby reduces population growth.

The model for EAB dynamics and control can be extended and further refined. One management strategy to slow the population growth and spread of EAB is to girdle ash trees by removing a band of outer bark and phloem, which stresses the tree, increasing its attraction to adult EAB beetles, including egg-laying females (Rodriguez-Saona et al., 2006; McCullough et al., 2009a,b). The use of a modified dispersal function accounting for attraction to girdled trees could be used to direct more EAB toward the grid cells where ash trees are girdled (Mercader et al., 2011). This would make tree removals more effective sinks for EAB and further increase the net benefits of management.

An important caveat to our model is our assumption of perfect information about the location and spread of the EAB infestation under different management activities. In practice, there is considerable uncertainty about EAB spread, and several years can pass before managers become aware of a tree's infestation. An important extension is to incorporate optimal detection effort as part of management activities. Given these uncertainties, it is important to recognize that a centralized management plan developed under the assumption of perfect information may not be optimal for the study area. Local controls based on improved local information may be superior to a centralized plan based on imperfect information (e.g., List and Mason, 2001).

Inadvertent human transport of invasive species promotes the establishment of satellite populations that substantially increase environmental damage unless management actions slow population growth and expansion. At least a dozen established EAB satellites have been found since 2007, many of them in cities, and slowing their spread is a critical management problem. The model we develop for managing the host of an invasive species provides insight into the optimal spatial-dynamic path of management activities and total net benefits in response to the coordination of management among municipalities and the degree of public access to the host.

Acknowledgements

The authors are grateful to Mark Abrahamson for sharing digital maps of the Twin Cities' ash tree cover and EAB detections. We thank Becky Epanchin-Niell, Frances Homans, Andrew Liebhold, Noel Schneeberger, Robert Venette, and Lynne Westphal for helpful comments on earlier drafts of this manuscript. We thank the U.S. Dept. of Agriculture Forest Service, Northeastern Area State and Private Forestry and the Northern Research Station for their support.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.reseneeco.2013.04.008.

References

Albers, H.J., Fischer, C., Sanchirico, J.N., 2010. Invasive species management in a spatially heterogeneous world: effects of uniform policies. Resource and Energy Economics 32, 483–499.

Anulewicz, A.C., McCullough, D.G., Cappaert, D.L., Poland, T.M., 2008. Host range of the emerald ash borer (*Agrilus planipennis* Fairmaire) (Coleoptera: Buprestidae) in North America: results of multiple-choice field experiments. Environmental Entomology 37, 230–241.

Bhat, M.G., Huffaker, R.G., 2007. Management of a transboundary wildlife population: a self-enforcing cooperative agreement with renegotiation and variable transfer payments. Journal of Environmental Economics and Management 53 (1), 54–67.

Bhat, M.G., Huffaker, R.G., Lenhart, S.M., 1993. Controlling forest damage by dispersive beaver populations: centralized optimal management strategy. Ecological Applications 3 (3), 518–530.

Blackwood, J., Hastings, A., Costello, C., 2010. Cost-effective management of invasive species using linear-quadratic control. Ecological Economics 69, 519–527.

Burnett, K., Pongkijvorasin, S., Roumasset, J., 2012. Species invasion as catastrophe: the case of Brown tree snake. Environmental and Resource Economics 51, 241–254.

Cappaert, D., McCullough, D.G., Poland, T.M., Siegert, N.W., 2005. Emerald ash borer in North America: a research and regulatory challenge. American Entomologist 51, 152–165.

City of Saint Paul, 2008. The Parks and Recreation Plan. Commission of Parks and Commission of Planning for the 2020 Saint Paul Comprehensive Plan, http://www.stpaul.gov/DocumentView.aspx?DID=11884 (accessed January 2012).

Dunens, E., Haase, R., Kuzma, J., Quick, K., 2013. Facing the Emerald Ash Borer in Minnesota: stakeholder Understandings and Their Implications for Communication and Engagement. Humphrey School of Public Affairs, University of Minnesota.

Eiswerth, M.E., Johnson, W.S., 2002. Managing nonindigenous invasive species: insights from dynamic analysis. Environmental and Resource Economics 23, 319–342.

Epanchin-Niell, R.S., Hufford, M.B., Aslan, C.E., Sexton, J.P., Port, J.D., Waring, T.M., 2010. Controlling invasive species in complex social landscapes. Frontiers in Ecology and the Environment 8, 210–216.

Epanchin-Niell, R.S., Wilen, J.E., 2012. Optimal spatial control of biological invasions. Journal of Environmental Economics and Management 63, 260–270.

Feder, G., Regev, U., 1975. Biological interactions and environmental effects in the economics of pest control. Journal of Environmental Economics and Management 2, 75–91.

Herms, D.A., McCullough, D.G., Smitley, D.R., Sadof, C.S., Williamson, R.C., Nixon, P.L., 2009. Insecticide Options for Protecting Ash Trees from Emerald Ash Borer. National IPM Center, Illinois, 12 p. Available for download on the national EAB.info website, www.emeraldashborer.info (accessed March 2012).

Hof, J., 1998. Optimizing spatial and dynamic population-based control strategies for invading forest pests. Natural Resource Modeling 11, 197–216.

Howarth, R.B., 2009. Discounting, uncertainty, and revealed time preference. Land Economics 85, 24–40.

Kaiser, B.A., Burnett, K.M., 2010. Spatial economic analysis of early detection and rapid response strategies for an invasive species. Resource and Energy Economics 32, 566–585.

Kovacs, K.F., Haight, R.G., McCullough, D.G., Mercader, R.J., Siegert, N.W., Liebhold, A.M., 2010. Cost of potential emerald ash borer damage in U.S. communities, 2009–2019. Ecological Economics 69, 569–578.

Kovacs, K.F., Mercader, R.J., Haight, R.G., Siegert, N.W., McCullough, D.G., Liebhold, A.M., 2011. The influence of satellite populations of emerald ash borer on projected economic costs in U.S. communities, 2010–2020. Journal of Environmental Management 92, 2170–2181.

List, J.A., Mason, C.F., 2001. Optimal institutional arrangements for transboundary pollutants in a second-best world: evidence from a differential game with asymmetric players. Journal of Environmental Economics and Management 42, 277–296.

McCullough, D.G., Siegert, N.W., 2007. Estimating potential emerald ash borer (*Agrilus planipennis* Fairmaire) populations using ash inventory data. Journal of Economic Entomololgy 100, 1577–1586.

McCullough, D.G., Poland, T.M., Anulewicz, A.C., Cappaert, D., 2009a. Emerald ash borer (*Agrilus planipennis* Fairmaire) (Coleoptera: Buprestidae) attraction to stressed or baited ash trees. Environmental Entomology 38, 1668–1679.

McCullough, D.G., Poland, T.M., Cappaert, D., Anulewicz, A.C., 2009b. Attraction of the emerald ash borer to ash trees stressed by girdling, herbicide and wounding. Canadian Journal of Forest Research 39, 1331–1345.

McCullough, D.G., Siegert, N.W., Bedford, J., 2009. SL.owing A.sh M.ortality: a potential strategy to SLAM emerald ash borer in outlier sites. Abstract. In: Proceedings of the USDA Interagency Research Forum on Gypsy Moth & Other Invasive Species, January 9–13, 2008, Annapolis, MD. USDA Forest Service, Northern Research Station (Gen. Tech. Rep. NRS-P-36), Newtown Square, PA.

McCullough, D.G., Poland, T.M., Anulewicz, A.C., Lewis, P., Cappaert, D., 2011. Evaluation of *Agrilus planipennis* control provided by emamectin benzoate and two neonicotinoid insecticides, one & two seasons after treatment. Journal of Economic Entomology 104, 1599–1612.

McCullough, D.G., Mercader, R.J., 2012. Evaluation of potential strategies to SLow Ash Mortality (SLAM) caused by emerald ash borer (*Agrilus planipennis*): SLAM in an urban forest. International Journal of Pest Management 58 (1), 9–23.

McPherson, E.G., Simpson, J.R., Peper, P.J., Maco, S., Gardner, S., Cozad, S., et al., 2005. City of Minneapolis, Minnesota Municipal Tree Resource Analysis. Center for Urban Forest Research, USDA Forest Service, Pacific Southwest Research Station.

Mercader, R.J., Siegert, N.W., Liebhold, A.M., McCullough, D.G., 2009. Dispersal of the emerald ash borer, *Agrilus planipennis*, in newly colonized sites. Agricultural and Forest Entomology 11, 421–424.

Mercader, R.J., Siegert, N.W., Liebhold, A.M., McCullough, D.G., 2011. Influence of foraging behavior and host spatial distribution on the localized spread of the emerald ash borer, *Agrilus planipennis*. Population Ecology 53, 271–285.

Mercader, R.J., Siegert, N.W., McCullough, D.G., 2012. Estimating the influence of population density and dispersal behavior on the ability to detect and monitor *Agrilus planipennis* (Coleoptera: Buprestidae) populations. Journal of Economic Entomology 105, 272–281.

Metropolitan Council, 2006. Generalized Land Use 2005 for the Twin Cities Metropolitan Area, http://www.datafinder.org/metadata/GeneralizedLandUse2005.htm (accessed July 2011).

Ndeffo Mbah, M., Gilligan, C.A., 2010. Balancing detection and eradication for control of epidemics: sudden oak death in mixed-species stands. PLoS ONE 5 (9), e12317.

Nowak, D.J., Hoehn III, R.E., Crane, D.E., Stevens, J.C., Walton, J.T., Bond, J., et al., 2006. Assessing urban forest effects and values: Minneapolis' urban forest. In: Resource Bulletin NE-166. U.S. Department of Agriculture, Forest Service, Northeastern Research Station, Newtown Square, PA, pp. 24.

Poland, T., McCullough, D.G., 2010. SLAM: a multi-agency pilot project to SL.ow A.sh M.ortality caused by emerald ash borer in outlier sites. Newsletter of the Michigan Entomological Society 55, 4–8.

Richards, T.J., Ellsworth, P., Tronstad, R., Naranjo, S., 2010. Market-based instruments for the optimal control of invasive insect species: *B. tabaci* in Arizona. Journal of Agricultural and Resource Economics 35, 349–367.

Rodriguez-Saona, C., Poland, T.M., Miller, J.M., Stelinski, L.L., Grant, L.G.G., de Groot, P., et al., 2006. Behavioral and electrophysiological responses of the emerald ash borer, *Agrilus planipennis*, to induced volatiles of Manchurian ash, *Fraxinus mandshurica*. Chemoecology 16, 75–86.

Sander, H.A., Haight, R.G., 2012. Estimating the economic value of cultural ecosystem services in an urbanizing area using hedonic pricing. Journal of Environmental Management 113, 194–205.

Siegert, N.W., McCullough, D.G., Williams, D.W., Fraser, I., Poland, T.M., 2010. Dispersal of *Agrilus planipennis* (Coleoptera: Buprestidae) from discrete epicenters in two outlier sites. Environmental Entomology 39, 253–265.

Sims, C., Aadland, D., Finnoff, D., 2010. A dynamic bioeconomic analysis of mountain pine beetle epidemics. Journal of Economic Dynamics and Control 34 (12), 2407–2419.

SLAMEAB.info, 2012. SLow Ash Mortality (SLAM) Website, www.slameab.info (accessed February 2012).

Taylor, R.A.J., Poland, T.M., Bauer, L.S., Haack, R.A., 2006. Is emerald ash borer an obligate migrant? In: Mastro, V., Reardon, L.R. (Eds.), Proceedings of the Emerald Ash Borer Research and Technology Development Meeting 2005. USDA Forest Service, Forest Health Technology Enterprise Team, Morgantown, WV, pp. 26–27.

Tluczek, A.R., McCullough, D.G., Poland, T.M., 2011. Influence of host stress on emerald ash borer (*Agrilus planipennis* Fairmaire) (Coleoptera: Buprestidae) adult density, development, and distribution in *Fraxinus pennsylvanica* trees. Environmental Entomology 40, 357–366.

Vannatta, A.R., Hauer, R.H., Schuettpelz, N.M., 2012. Economic analysis of emerald ash borer (Coleoptera: Buprestidae) management options. Journal of Economic Entomology 105 (1), 196–206.

Wilen, J., 2007. Economics of spatial-dynamic processes. American Journal of Agricultural Economics 89, 1134–1144.