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The cost of gypsy moth sex in the city

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ABSTRACT

Since its introduction in the 1860s, gypsy moth, *Lymantria dispar* (L.), has periodically defoliated large swaths of forest in the eastern United States. Prior research has suggested that the greatest costs and losses from these outbreaks accrue in residential areas, but these impacts have not been well quantified. We addressed this lacuna with a case study of Baltimore City. Using two urban tree inventories, we estimated potential costs and losses from a range of gypsy moth outbreak scenarios under different environmental and management conditions. We combined outbreak scenarios with urban forest data to model defoliation and mortality and based the costs and losses on the distribution of tree species in different size classes and land uses throughout Baltimore City. In each outbreak, we estimated the costs of public and private suppression, tree removal and replacement, and human medical treatment, as well as the losses associated with reduced pollution uptake, increased carbon emissions and foregone sequestration. Of the approximately 2.3 M trees in Baltimore City, a majority of the basal area was primary or secondary host for gypsy moth. Under the low outbreak scenario, with federal and state suppression efforts, total costs and losses were $5.540 M, much less than the $63.666 M estimated for the high outbreak scenario, in which the local public and private sectors were responsible for substantially greater tree removal and replacement costs. The framework that we created can be used to estimate the impacts of other non-native pests in urban environments.

Introduction

Increased connectivity between people, places and markets has unintentionally resulted in the introduction of many non-native species (Brockerhoff et al., 2006; Hulme et al., 2008; McCullough et al., 2006; Ruiz et al., 2000; Work et al., 2005). Although only a minority of introduced species are believed to establish upon their arrival (Ludsin and Wolfe, 2001; Simberloff and Gibbons, 2004), and even fewer are thought to cause environmental and economic damage (Aukema et al., 2010; Mack et al., 2000), those that do can impose considerable environmental and socioeconomic costs (Aukema et al., 2011). One commonly cited estimate of the control costs and damages caused by non-native species in the United States is $120 billion in market value annually (Pimentel et al., 2000, 2005). Biological invasions also affect ecosystem services such as carbon sequestration, rainwater interception, microclimate regulation, and esthetic value, many of which are provided by urban forests (Dwyer et al., 1992; Morales, 1980; Nowak et al., 2002; McPherson et al., 1997). Non-native species diminish these services, causing both losses and mounting costs as governments and homeowners attempt to mitigate their impacts (Aukema et al., 2011; Holmes et al., 2009).

Management costs and potential economic losses in urban settings have been documented for a few non-native forest pests. For example, in nine U.S. cities the Asian longhorned beetle, *Anoplophora glabripennis* (Motschulsky), could reduce canopy cover by 13–68%, yielding a maximum cost of $669 billion based on the compensatory value of killed trees (Nowak et al., 2001). Emerald ash borer, *Agrilus planipennis* Fairmaire, could have a total impact of $1.8–$ 7.6 billion in urban areas of Ohio alone due to losses in landscape value and the costs of tree removal and replacement (Sydnor et al., 2007). In Anoka County, Minnesota the spread of oak wilt, *Ceratocystis fagacearum*, over a ten year period is projected to result in the death of 80,000–270,000 trees and tree removal costs of $18M–$60 M (Haight et al., 2011).

Gypsy moth, *Lymantria dispar* (L.), is a well-studied biological invader, and consequently, there is a considerable amount of information on its invasion dynamics (Elkinton and Liebhold, 1990;

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Johnson et al., 2006; Tobin et al., 2007, 2009). Research on gypsy moth has largely focused on its negative ecological impacts on natural forest ecosystems (Campbell and Sloan, 1977; Herrick and Gansner, 1987; Redman and Scriber, 2000; Thurber et al., 1994) and financial impacts on timberlands (Gansner et al., 1978; Gansner and Herrick, 1987; Herrick, 1981; McCay and White, 1973), with relatively little attention paid to impacts in urban areas (but see Moeller et al., 1977; Payne et al., 1973; Onstad et al., 1997). This is somewhat surprising since cost-benefit analyses of gypsy moth management programs have shown that most of the benefits are due to delaying the onset of residential impacts (Leuschner et al., 1996; Mayo et al., 2003). These impacts include the costs of management, tree removal and replacement, as well as the nuisance associated with outbreaks.

As a polyphagous defoliator, gypsy moth larvae feed on over 300 species of deciduous and coniferous host trees including the highly preferred oak, aspen, birch, and larch (Elkinton and Liebhold, 1990; Liebhold et al., 1995a). The density of preferred gypsy moth host trees is an important factor determining the severity of outbreaks in forest settings; outbreaks rarely occur in forests that are not dominated by oaks or other preferred host species (Herrick and Gansner, 1986). During outbreaks, which typically occur over a 3 year time period, gypsy moths can severely defoliate trees and cause mortality in trees that were previously defoliated, are diseased, or are otherwise stressed (Gansner and Herrick, 1984; Herrick and Gansner, 1987). After 1–3 years, outbreak populations generally collapse, largely due to regulation by two gypsy moth-specific entomopathogens: the fungus *Entomophaga maimaiga* (Hajek et al., 1995) and a nuclear polyhedrosis virus (Elkinton and Liebhold, 1990).

There have been significant periodic gypsy mouth outbreaks in North America starting with the initial outbreak in Medford, Massachusetts in the 1890s (Forbush and Fernald, 1896; Johnson et al., 2005). The severity of impacts, as well as costs, depends in part on public and private investments in suppression. Long-term losses from tree mortality may be compounded in the short-run by the costs of removing dead trees and planting new trees. During and after an outbreak there are clean-up costs associated with frass and dead insects, as well as the costs and losses related to allergic reactions experienced by people that come into contact with gypsy moth (Tuthill et al., 1984; Allen et al., 1991). Consequently, there has been significant public investment in suppressing outbreaks in the infested area and slowing the spread of the gypsy moth to delay the onset of costs and losses (Tobin et al., 2012), including those in urban and suburban forests (Webb et al., 1991).

To model urban gypsy moth outbreaks and impacts, we first synthesized information from the literature and expert informants to create a range of outbreak scenarios. We then applied these outbreak scenarios to an urban forest database, resulting in varying levels of simulated tree defoliation and mortality that depend on the forest structure, composition, and distribution across urban land uses. Next, we identified appropriate data sources to estimate the costs and losses that would be incurred under each scenario. Specifically, we used information from public management programs; forestry, arboriculture, and entomology literature; surveys of private households; and the professional tree care industry. We used this information to estimate the costs of public and private suppression, tree removal and stump grinding, tree replacement, and medical treatment as well as losses of pollution removal benefits and carbon sequestration. We illustrate this framework with an application to the forests of Baltimore City, Maryland – highlighting the importance of public suppression efforts and demonstrating how available data can be harnessed to plan a response to non-native forest pest in urban forests.

Methods

Urban forest data, Baltimore City, Maryland

We focused our study on Baltimore City (hereon referred to as "Baltimore") because of the availability of tree composition data from an i-Tree Eco (formerly UFORE) study. In 1999, US Forest Service scientists collected data from a land-use-based stratified random sample of two hundred and two 0.1 acre (0.04 ha) circular plots in Baltimore and input these data into the i-Tree Eco model (Nowak et al., 2004). Among the model outputs were estimates of the total number of trees for each species by land use, the diameter distribution of each species, and the distribution of trees across condition classes (Nowak and Crane, 2000). We used the i-Tree Eco model outputs as our base population of trees affected by gypsy moth.

Because most randomly located circular plots did not fall along linear street right-of-ways, the i-Tree Eco sampling methodology was not an efficient way to sample for street trees. To improve the street tree estimate, we augmented the i-Tree Eco output with data from a U.S. Forest Service Forest Health Monitoring (FHM) Pilot Street Tree inventory of Maryland. Taking the i-Tree Eco output, we extracted the street trees from the total number of trees in each land use, assuming the street trees followed the same diameter distributions as the total population. We then combined this with data from the FHM Pilot Street Tree inventory, which were collected between 2002 and 2006 from plots randomly located along right-of-ways (Cumming et al., 2001, 2006). We extracted the FHM Street Tree data collected in Baltimore, a 0.14% sample of right-of-ways, to estimate the city's total street tree population. We assumed the i-Tree Eco and FHM Street Tree outputs represented independent estimates of the street tree population and averaged the number of trees of each species in each diameter class from the two population estimates to derive our estimate of Baltimore's street tree population.

Gypsy moth outbreak scenarios

We developed four gypsy moth outbreak scenarios using research on outbreak cycles. Gypsy moth outbreaks are periodic and typically exhibit a dominant period of 8–10 years (Johnson et al., 2005). In addition to the dominant period, less severe subdominant outbreaks can occur every 4–5 years (Haynes et al., 2009). The outbreaks result in varying levels of defoliation and mortality depending on forest structure, tree species composition, and outbreak severity (Liebhold et al., 1995a; Herrick and Gansner, 1986). The four scenarios we created reflect responses to a range of environmental conditions, entomopathogen levels, and public suppression efforts that result in outbreaks of varying severity (Table 1).

The outbreak scenarios were characterized by the percent defoliation and mortality of host trees. We classified trees as primary hosts, secondary hosts, or non-hosts (Liebhold et al., 1995a) and assumed defoliation and mortality of only primary and secondary hosts to occur over 2–3 years depending on outbreak severity. The high outbreak scenario, occurring over 3 years, represented conditions that were optimal for tree defoliation and mortality, including no public suppression efforts and environmental conditions that limit regulation by entomopathogens. The low outbreak scenario, occurring over 2 years, posited less defoliation and mortality due to the opposite conditions: successful public suppression efforts and regulation by entomopathogens. The two medium scenarios represented outbreaks that collapse after 2 or 3 years due to successful suppression efforts or regulation by entomopathogens. Rates of defoliation and mortality in each scenario were based on reports from forested stands (Campbell and Sloan, 1977; Gansner

Table 1
Gypsy moth outbreak scenarios. We created four outbreak scenarios, each occurring over a three year period. In each outbreak the percent of primary and secondary host trees defoliated varied along with the number of trees killed.

Scenario	Year	Trees defoliated		Trees killed	
		Primary host trees (%)	Secondary host trees (%)	Primary host trees (%)	Secondary host trees (%)
High outbreak	1	90	25	0	0
	2	90	50	25	5
	3	50	10	25	5
Medium outbreak, 3 yr.	1	50	12.5	0	0
	2	75	25	12.5	2.5
	3	25	5	12.5	2.5
Medium outbreak, 2 yr.	1	50	12.5	0	0
	2	75	25	12.5	2.5
	3	0	0	0	0
Low outbreak	1	25	0	0	0
	2	50	10	5	0
	3	0	0	0	0

and Herrick, 1984; Herrick and Gansner, 1987) and were modified to reflect the varied intensity of outbreaks across scenarios.

Estimating costs and losses from gypsy moth outbreaks

Impacts of gypsy moth outbreaks included direct expenditures and reduced wellbeing, i.e., costs and losses, due to defoliation and public nuisance. Defoliation leads to losses in tree benefits, such as pollution removal and carbon sequestration, and tree mortality. Mortality imposes costs such as tree removal, stump grinding, and tree replacement and losses in carbon storage and reduced sequestration. However, defoliation and mortality can be mitigated by private suppression efforts that require private expenditures but reduce the costs associated with dead trees (Fig. 1).

Our scenarios consider both public and private expenditures for mitigating gypsy moth outbreaks and addressing their aftermath. Specifically, the costs included are for suppression, dead tree removal and replacement, and health care to treat allergic reactions. Losses included reductions in monetary or non-monetary values due to gypsy moth outbreaks. We considered losses of two ecosystem services: pollution removal and carbon sequestration and storage. To estimate these costs and losses, we assumed the number of host trees experiencing complete defoliation – and therefore a cost or a loss – was directly proportional to the percent defoliation from a given outbreak scenario. Most of our economic data were collected between 2006 and 2008, and we therefore converted all dollar values into 2007 dollars using the Consumer Price Index (2013). Losses were calculated over an 8 year period: 3 years during an outbreak and 5 years following the outbreak, at which time subdominant outbreaks may occur resulting in additional losses (Haynes et al., 2009). Costs and losses were calculated for each year of the outbreak and discounted to the first year of the scenario using a 5% discount rate. These annual values were summed to calculate the total economic cost in 2007 dollars for all 8 years of the outbreak scenarios.

Suppression costs

We included public suppression costs in the low, medium low, and medium high outbreak scenarios only. Public suppression costs were estimated by applying the average management costs of the USDA Forest Service-Maryland Cooperative Gypsy Moth Suppression Program during the 2006–2008 outbreak, $7413/km^2 to the annual treatment area (USDA Forest Service, 2013). Most treatments are coordinated through USDA Forest Service cooperative suppression program, which limits aerial application to areas greater than 10 acres (0.04 km^2) and with a minimum of 50% canopy. Private residential areas are unlikely to be covered by public suppression efforts because they do not meet minimum canopy requirements and homeowners may object to spraying. This is exemplified by cooperative suppression in 2006 and 2007 in Pennsylvania, where less than 0.5% of the area sprayed was within medium or high density residential areas (2001 National Land Cover Database, Homer et al., 2004). Therefore, we assumed that

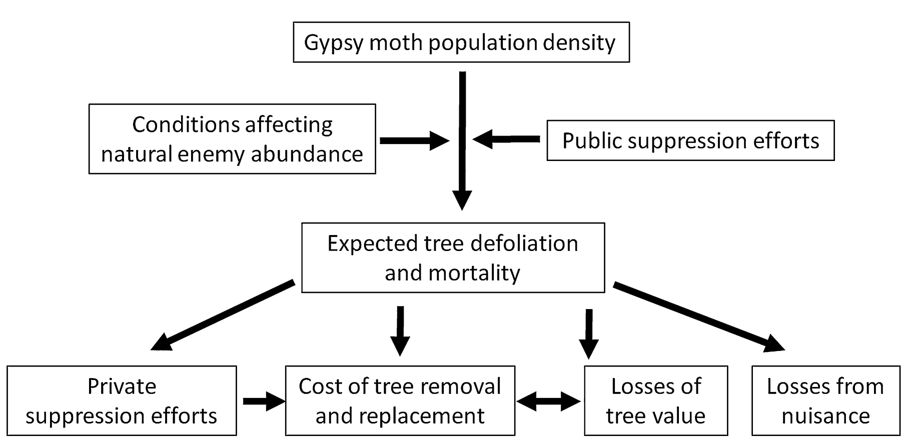

Fig. 1. Conceptual framework for estimating gypsy moth costs and losses in urban areas. Gypsy moth population density is influenced by suppression efforts and conditions affecting natural abundance, both of which contribute to the defoliation and mortality of gypsy moth host trees. Defoliation and mortality result in private suppression efforts, tree removal and replacement, and nuisance impacts. All of these factors have varying levels of costs and losses.

Table 2
Gypsy moth host trees in Baltimore and tree removal and replacement rates.

Land use category	Number of trees			Dead tree removal (%)	Dead tree replacement (%)
	Primary host species	Secondary host species	Non-host species		
Barren	0	1009	0	25	0
Commercial/industrial	4938	64,198	247,721	90	25
Forest	136,624	348,646	188,801	25	0
High-density Residential	20,307	81,255	77,378	90	50
Institutional	18,376	87,313	101,066	90	75
Med/low-density residential	13,803	179,624	94,337	50	25
Street trees	12,322	49,833	54,251	90	75
Transportation	2936	19,083	2936	90	25
Urban open	144,071	181,977	174,596	90	25
Citywide	353,377	1012,939	941,086	–	–

aerial spraying would occur only in areas zoned as parks, excluding parks that are predominantly athletic facilities. For treatment under the public suppression program, we selected only parks in Baltimore greater than 10 acres (0.04 km^2) and with at least 50% canopy cover, according to the 2001 National Land Cover Dataset (Homer et al., 2004). Treatment was applied to each block in each year of the low outbreak, the first and second year of the medium low outbreak, and the second year of the medium high outbreak scenario at a cost of \$7413/km^2 (USDA Forest Service, 2013).

Private suppression costs were derived from a telephone survey in 2008 of approximately 200 residents of counties affected by the gypsy moth. The survey included one county with no public spraying, Center County, Pennsylvania, which was in the first year of an outbreak in 2007 and two counties with substantial public spraying programs, Pike County, Pennsylvania and Burlington County, New Jersey, which were both in their third year of an outbreak in 2007. Starting from a random sample of residents in urban and suburban zip codes in these counties, we screened for respondents who lived in single family homes and were responsible for their yards including the trees. We considered only respondents with yards ≤5 acres (2.02 ha) and ≤60 large deciduous trees per acre (24 per ha) to eliminate the influence of outlier households with very large properties. From each homeowner we elicited information on time spent and out-of-pocket costs for protecting yard trees from gypsy moth.[1] This total homeowner cost was then divided by the number of large deciduous trees in the yard. Respondents who reported that their yard was not covered by a public spraying program had out-of-pocket costs that averaged \$1.90 and spent an average of 0.3 h per large deciduous tree in the yard. We valued the work at the minimum wage of \$6.55/h and applied the total – \$3.90 per large deciduous tree – to our high outbreak scenario. Respondents whose yards were covered by a public spraying program spent an average of \$1.70 out of pocket and 0.2 h on suppression – in total \$3.00 per large deciduous yard tree. These estimates are low in part because they were averaged over all large deciduous trees, potentially including many trees that were not treated because they were not threatened or not considered valuable, and in part because of the low cost of inputs required for treatments such as scraping egg masses or using barrier bands to trap caterpillars. As expected, private expenditures were lower where public suppression had limited the severity of the outbreak. We used the \$3.00 per tree estimate for our medium and low outbreak scenarios.

We determined the cost of private suppression by applying \$3.90 and \$3.00 in the high and medium/low scenarios, respectively, to each large deciduous tree in the private land use classes. To represent the survey concept of "large trees," we used approximately the upper quartile of trees in the Baltimore tree data: trees larger than 30 cm diameter at breast height (DBH). In an urban environment, these trees are likely to be large or valuable enough to warrant treatment. Specifically, we multiplied the estimated cost per tree – \$3.90 and \$3.00 – by the number of large trees in high and medium/low density residential and commercial/industrial land uses.

[1] Cost estimates were obtained by asking the landowner about their total expenditures and time spent on suppression. Specifically, we asked: "Considering all of the control methods that you used last year, how much did your household pay out of pocket for private services and materials?" and "How much time did you personally spend on these methods last year?" as well as how much time other family members spent on these methods.

Tree removal and replacement costs

We modeled tree removal and replacement by considering how rates would vary according to the potential hazard of standing dead trees, the likelihood of natural regeneration, and the benefits of tree replacement in different land uses (Table 2). The scenarios were largely based on discussions with the office of the City Arborist in Baltimore City, Maryland. We assumed the highest removal rate, 90%, in areas where dead trees would pose an imminent threat. This included high density residential, institutional, commercial/industrial, street, transportation, and urban open land uses. We assumed that 90% of dead trees were removed – as opposed to 100% – due to the time lag between identifying and removing a dead tree. The middle rate, 50%, was applied in medium/low density residential land use and reflects an intermediate hazard. We assumed a low removal rate of 25% in forests and barren land uses because dead trees in these areas are a minor hazard due to limited public use. Finally, we assumed that tree removal occurred the year after a tree died because that is when its death is likely to be noted, due to its failure to foliate.

We assumed that 50% of the trees removed in high density residential areas were replaced because trees are important to residents and they are unlikely to regenerate naturally. In urban open, medium/low density residential, commercial/industrial, and transportation land uses, we assumed a 25% replacement rate, reflecting the lower priority given to trees in these areas and increased likelihood of natural regeneration. We assumed no replanting in forests and barren land uses because natural regeneration is the most likely source of replacement trees. We also assumed that 75% of dead trees along streets and in institutional land uses were replaced by planting new trees, reflecting more active landscaping in these areas. Finally, we assumed that tree replacement occurred the year a tree was identified as dead – i.e., the year after mortality occurs.

The cost of tree removal and stump grinding vary with tree size, shape, and proximity to structures, but costs are typically correlated with tree diameter at breast height (DBH) (McPherson et al., 2005). We used cost estimates of tree removal/stump grinding by DBH class provided by Bartlett Tree Experts, Annapolis, Maryland, which were \$120/\$40: 0–15 cm DBH, \$230/\$80: 16–30 cm DBH, \$675/\$120: 31–76 cm, and \$2700/\$250: >76 cm DBH. We applied these removal costs to the percent of trees that were preferred

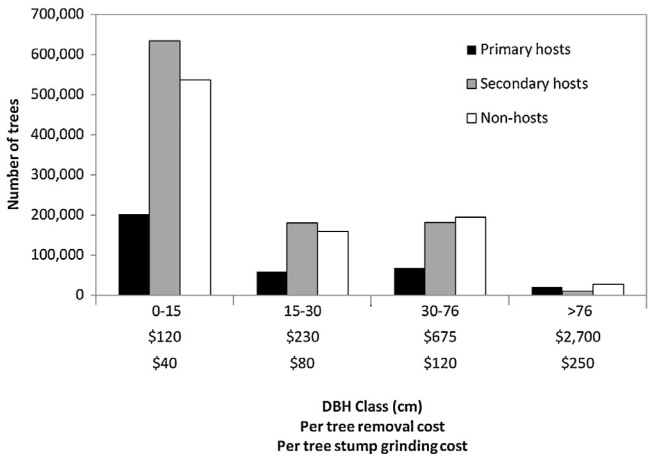

Fig. 2. Distribution of gypsy moth host trees. The number of trees in Baltimore for each gypsy moth host type categorized into the 4 classes that we used to calculate tree removal and stump grinding costs.

or secondary gypsy moth hosts that were assumed to die under different outbreak scenarios (Fig. 2). We also assumed that stump grinding was required only for trees that were replaced with new trees, at the replacement rates described for different land uses.

The cost of tree replacement depends largely on the size, which is likely to vary according to land use. In residential areas, homeowners may plant trees < 1 in. (2.54 cm) diameter above root collar that cost as little as $7. However, many city governments have ordinances that require larger and more expensive trees to be planted on public property. In Baltimore, the minimum size public tree required at planting, 2.5 in. (6.35 cm) diameter above root collar, was estimated to cost $700 by the City Arborist. Therefore, we assumed a tree replacement cost of $700 per tree planted on public property – urban open, street, institutional, and transportation land uses. For trees planted in commercial/industrial and high and medium/low density residential land uses, we assumed $40 as the cost of a replacement tree, reflecting the typical price of trees sold at big box retailers.

Public nuisance from gypsy moth outbreaks

Nuisance from gypsy moth outbreaks occurs when larvae and frass interact with human populations. Forbush and Fernald (1896) provided graphic descriptions of the nuisance during the first gypsy moth outbreak in North America, in which streets and sidewalks were so slick with squashed caterpillars and frass that they were impassable. There are clearly clean-up costs associated with larvae and frass, but these costs are difficult to quantify. Instead, we obtain a lower bound on nuisance costs by estimating human health impacts due to 'gypsy moth rash'. Tuthill et al. (1984) reported that during a severe gypsy moth outbreak, 10.4% of the human population reported allergic reactions to the urticating hairs of larvae. During a less severe outbreak only1.6% of people reported a rash. We used the midpoint of these values and assumed that 6% of the population would experience a rash in a medium outbreak.

Because rates of defoliation vary in our outbreak scenarios, we applied different rash rates according to the defoliation of primary hosts. When defoliation was 90% we used the high rash rate, 10.4%; when defoliation was 50% or 75%, we used the medium rash rate, 6%; and when deforestation was 25%, we used the low rash rate, 1.6%. We applied these rates to the human population that would interact with gypsy moth. Specifically, we assumed the number of people that encountered gypsy moth was proportional to the tree cover (BES LTER, 2009) and the human population in each census block group (US Census Bureau, 2011). We then assumed that people experiencing a rash – dependent on the defoliation rate – would seek an over-the-counter treatment at a cost of $5 per case, the cost of a tube of cortisone. While a small number of people may have more severe reactions, requiring medical treatment by a doctor, we used an estimate of out-of-pocket costs for over-the-counter treatment as a conservative estimate of the human health impacts.

Losses due to reduced tree benefits

Gypsy moth outbreaks also result in losses due to reduced tree benefits. Some of these non-use values, esthetics for instance, can be most directly and completely quantified with stated preference non-market valuation methods (e.g., Jakus, 1994). Other losses, for example, air pollution removal and carbon sequestration and storage, can be quantified with existing models, such as those provided in i-Tree Eco (Nowak and Crane, 2000).

Our framework for calculating impacts on carbon and pollution removal considered both dead and defoliated trees over 8 years, 3 years of the outbreak and the 5 years following. We calculated carbon emissions using the output of the i-Tree Eco model, which provided an estimate of above and below ground carbon storage and sequestration per tree in each size class. For each outbreak scenario we calculated the amount of carbon stored in dead trees in each size class that would be emitted when the tree decomposed, was burned, or was chipped and composted. While these emissions may occur over many years after a tree dies, we assumed an immediate loss of credit for the carbon that was stored in the tree. We also calculated the loss in sequestration by defoliated and dead trees. We assumed that trees have zero net carbon sequestration during the years they were defoliated. In addition, we calculated lost sequestration benefits from trees that died for 5 years after the outbreak, assuming that each tree would have continued to sequester carbon at pre-mortality rates. Of course, many of the dead trees would have continued to sequester carbon for much longer than five years, but the replacement trees would also start to sequester carbon, so we assumed that these two factors balance.

To determine monetary losses for carbon storage and sequestration we summed the amount of carbon emitted from dead trees and lost carbon sequestration for each year. We then converted carbon to the CO_2 equivalent and multiplied this estimate by $10 per Mt of CO_2 ($9.51 in $2007), the floor price established by the California Environmental Protection Agency (2011). While we estimated the market value loss in carbon storage and sequestration, this value is close to the lowest estimate of the social value of carbon used by the U.S. Environmental Protection Agency (2013).

We calculated pollution removal losses using the Pollution Removal Calculator in i-Tree Eco, which assumes that pollution removal and monetary benefits are proportional to the amount of tree canopy. In Baltimore, tree canopy was estimated to be 24% based on high resolution land cover data (BES LTER, 2009). We assumed that tree canopy was distributed proportional to basal area across all trees in our tree database. Thus, we began with a baseline estimate of pollution removal and in each outbreak scenario calculated the reduction based on the basal area of defoliated and killed trees. We monetized pollution removal using the externality value of four pollutants (2007 dollars per Mt): particulate matter 10 ($6307), nitrogen dioxide ($9447), sulfur dioxide ($2313), and carbon monoxide ($1342) (Murray et al., 1994).

We calculated pollution removal losses for each year of the outbreak. For the first outbreak year we only calculated losses due to defoliation, assuming that new leaves produced in response to defoliation did not significantly offset losses because refoliated trees generally have <40% of the original leaf area (Heichel and Turner, 1976). In the second year of the outbreak, we again only considered defoliated trees with no offset of losses from leaf-out

Table 3
Dominant trees in Baltimore. The ten most abundant species and those most dominant in terms of basal area in Baltimore. Species in bold are primary or secondary gypsy moth hosts.

Most abundant		Dominant in basal area	
Tree species	Number	Tree Species	Basal area[a] (m^2)
Slippery elm	172,447	Silver maple	14,936.9
American beech	**158,972**	**Northern red oak**	**11,730.3**
White ash	158,900	**Black oak**	**10,165.7**
Black cherry	**143,867**	White ash	9007.8
Green ash	128,009	Tulip tree	8598.0
Black locust	121,267	Black locust	8593.5
Tree of heaven	**104,390**	**White oak**	6967.1
White oak	**97,944**	Slippery elm	5951.8
Red maple	**81,492**	**American beech**	**5933.7**
Boxelder	75,045	**Red maple**	**5226.8**

[a] The data listed the number of trees per DBH class in 3 in. intervals; thus, we used the midpoint of each class to estimate basal area.

later in the year. By the end of the second year, some of the trees that had been defoliated 2 years in a row die. Thus, in the third year, we considered defoliated trees and the trees that died in the previous year's outbreak. For the next 5 years, we assumed a loss in benefits based on the basal area of trees that died during the outbreak.

Results

We estimated a total of 2.3 M live trees in Baltimore. The most abundant tree was slippery elm and the most dominant in terms of basal area was silver maple, neither of which are suitable gypsy moth hosts (Table 3). However, many other dominant tree species in Baltimore were primary or secondary hosts, including white oak and northern red oak. The total basal area of all trees was 1,583,459 m^2, of which 425,246 m^2 (27%) and 485,055 m^2 (31%) are primary and secondary gypsy moth hosts, respectively. Most trees in Baltimore were <15 cm DBH and a very small number >76 cm DBH (Fig. 2). Tree basal area was greatest in medium/low density residential (25%), forest (22%), and high density residential (16%). Urban open (13%), commercial (12%), and streets (7%) contained smaller fractions of Baltimore's basal area, while institutional (5%), transportation (1%), and barren (<1%) had the smallest fractions.

We calculated12.7 km^2 of Baltimore was suitable for public suppression given the minimum canopy and land use requirements. In this area, costs ranged between $0 in the high outbreak scenario and $0.269 M in the low outbreak scenario (Table 4). We assumed private suppression applied to only large trees (>30 cm) in residential and commercial/industrial land uses. Costs were greater in more substantial outbreaks with no public suppression, ranging from $0.189 M to $0.245 M for the low and high outbreak scenarios, respectively (Table 4).

Tree defoliation and mortality also varied across years (Fig. 3) and outbreak scenarios (Table 4). Annual defoliation of host trees reduced total urban canopy between 36% and 4% in the greatest and least severe cases, respectively. In the low outbreak scenario, with public suppression efforts, fewer trees were defoliated (362,000) than in the high outbreak scenario (1,658,000). Mortality varied with defoliation rates, ranging between 17,000 and 275,000 trees in the low and high outbreak scenarios, respectively.

The removal of dead trees in each outbreak scenario was a significant cost. In the high outbreak scenario 171,000 trees (8%) were removed, while only 11,000 trees (<1%) were removed during the low outbreak. Tree removal and replacement costs were consistently greater than all other costs, ranging from $1.502 M/$1.738 M in the low outbreak scenario to $24.060 M/$26.022 M in the high outbreak scenario (Table 4). Stump grinding costs ranged from $0.138 M to $2.317 M in the low and high outbreak scenarios, respectively.

Out of a total human population of 637,455 in Baltimore, we estimated that 11,275 people (1.8%) would encounter gypsy moth and experience an allergic reaction during the low outbreak scenario. In the high outbreak scenario, allergic reactions affected 40,832 people (6.4%). Using a conservative estimate – the cost of a tube of cortisone – the total cost for treating allergic reactions ranged from $0.054 M to $0.196 M in the low and high outbreak scenarios, respectively (Table 4).

Dead trees in Baltimore released 319,882 Mt of carbon in the high outbreak and 22,851 Mt of carbon in the low outbreak scenarios (Table 5). Lost sequestration from dead trees over the 5 year period after the outbreak ranged between 18,667 Mt and 1083 Mt of carbon in the high and low outbreak scenarios, respectively. Lost carbon sequestration benefits caused by defoliation were 37,517 Mt in the high and 7771 Mt low outbreak scenarios. After converting carbon to CO_2 equivalent, at $9.51 per Mt CO_2, tree mortality and defoliation yielded $3.419 M and $0.285 M in emissions and sequestration losses in the high and low outbreak scenarios, respectively (Table 4).

Trees in Baltimore provided $4.727 M in benefits *annually* by removing local atmospheric pollutants, but these benefits were reduced by gypsy moth outbreaks. The greatest *annual* loss, $1.778 M, occurred during the second year of the high outbreak scenario (38% reduction), while the *annual* loss during the third year of a low outbreak scenario was $0.054 M (1% reduction). Total reduction in pollution removal benefits for 8 years after the beginning of an outbreak ranged between $1.365 M in the low outbreak and $7.407 M in the high outbreak scenario.

The total costs and losses of gypsy moth outbreak in Baltimore were significant. Costs and losses ranged from $5.540 M to $63.666 M in the low and high outbreak scenarios, respectively (Table 4). The medium outbreak scenarios resulted in costs and losses of $32.251 M for the three year outbreak and $17.644 M for the two year outbreak (Table 4).

Discussion

Our framework included a range of gypsy moth outbreak scenarios that provided an upper and lower bound on tree defoliation and mortality, and costs and losses over an 8 year time period. The total *annual* defoliation of all trees in our scenarios – between 4% and 36% – was consistent with a previous model of a gypsy moth outbreak in Chicago IL, which found that an outbreak would result in 14–26% defoliation city-wide (Onstad et al., 1997). Our high outbreak scenario represented the most extreme, catastrophic case for an urban forest, with costs of $52.840 M and losses of $10.826 M per outbreak. In contrast, the low outbreak scenario depicted successful public suppression and minimal defoliation, mortality, costs ($3.890 M) and losses ($1.650 M). The

Table 4
Gypsy moth impacts. The total number of affected trees (×1000) and costs and losses (×1000 $USD) summed over 8 years; 3 years during and 5 years after the outbreak.

	Outbreak Scenario			
	High	Medium, 3 yr.	Medium, 2 yr.	Low
Total number of defoliated trees	1658	951	814	362
Total number of dead trees	275	138	69	17
Carbon emissions and lost sequestration	$3419	$1713	$933	$285
Loss of pollution uptake	$7407	$4128	$3009	$1365
Tree removal costs	$24,060	$11,875	$6005	$1502
Stump grinding costs	$2317	$1146	$580	$138
Tree replacement costs	$26,022	$13,011	$6664	$1738
Public suppression costs	$0	$90	$175	$269
Private suppression costs	$245	$189	$189	$189
Health costs	$196	$99	$89	$54
Total costs and losses	$63,666	$32,251	$17,644	$5540

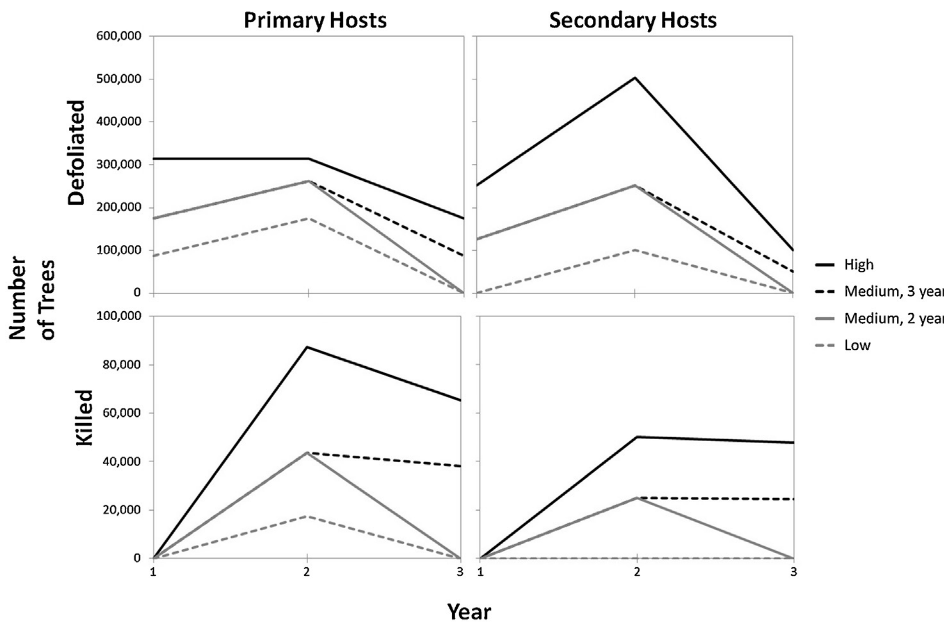

Fig. 3. Tree defoliation and mortality from gypsy moth. The number of primary and secondary host trees defoliated or killed in each outbreak scenario.

two medium scenarios represented outbreaks that were regulated by entomopathogens and public suppression efforts, resulting in costs of $26.410 M/$13.702 M and losses of $5.841 M/$3.942 M per outbreak.

The likelihood of the high outbreak scenario occurring in an intensively developed urban area is small because of limited and fragmented forest cover (e.g., Cadenasso et al., 2007). This is particularly true in a densely developed urban area like Baltimore where the tree cover is highly fragmented. Medium or low outbreaks are more likely in cities where parks, remnant forests, or surrounding forests provide sufficient habitat to support high density gypsy moth populations (Gottschalk, 1993). Suburban or exurban areas with more extensive and contiguous forest cover are more likely to be affected by an outbreak analogous to our high outbreak scenario.

Even our conservative estimates suggest that gypsy moth can impose significant economic costs in urban areas. Because gypsy moth is highly polyphagous, it affects a much greater proportion of trees than wood-boring insects that tend to be more host-specific. In Baltimore, for example, more than half the basal area was a primary or secondary gypsy moth host. Even when a small percentage of urban trees die due to gypsy moth attack, the costs associated with tree removal alone can be considerable (Table 4). These are unavoidable costs for urban forest managers due to the hazards and associated liability of dead and dying trees. Our estimates of removal costs are conservative because they do not include costs associated with impacts on traffic and businesses closures that may be required to remove street trees. Private residents may also be affected by liability concerns when dead trees become hazardous

Table 5
Carbon Losses from gypsy moth. Carbon emissions (Mt CO_2) from tree mortality and lost sequestration benefits (Mt CO_2/yr.) summed over 8 years; 3 years during and 5 years after the outbreak.

	Outbreak Scenario			
	High	Medium, 3 yr.	Medium, 2 yr.	Low
Emissions: dead tree (Mt CO_2)	319,882	159,941	79,971	22,851
Lost sequestration: outbreak (Mt CO_2/yr.)	37,517	22,158	18,837	7771
Lost sequestration: dead tree 5 years post outbreak (Mt CO_2/yr.)	18,667	9334	4667	1083

to adjacent properties, as well as suffering a loss in the functional value of trees (Holmes et al., 2009).

We found the costs of gypsy moth outbreak tended to fall mostly on local government and private residents. Federal and state public expenditure to suppress gypsy moth populations was comparatively low, comprising a small portion of total costs regardless of the outbreak scenario (Table 4). Tree removal and replacement costs borne by private residents and Baltimore city government alone were an order of magnitude greater than the public suppression costs. Aukema et al. (2011) found similar evidence of the unequal distribution of the costs of non-native forest pests. Specifically, Aukema et al. (2011) examined a set of non-native forest "poster-pests" from three guilds and found that the costs borne by local government and private household were consistently greater than those incurred by federal and state government. In our model, the discrepancy between the cost of preventative public suppression – generating the low outbreak scenario – and the cost of tree removal and replacement in the high outbreak scenario highlights the importance of cost-share programs for suppression.

More direct comparisons between our estimates and those of Aukema et al. (2011) were challenging because we modeled costs and losses of an outbreak in one city over 8 years while Aukema et al. (2011) modeled the annual costs and losses of gypsy moth's presence in the U.S. However, this comparison is facilitated by converting our estimates to their *annual* equivalents – costs and losses of $9.850 M and $0.857 M in the high and low outbreak scenarios, respectively. The *annual* equivalents for Baltimore show general agreement with estimates presented by Aukema et al. (2011) for total annual costs and losses for gypsy moth for federal government ($33 M), local government ($50 M), home owner expenditures ($46 M), and residential property value ($120 M) in all areas managed for gypsy moth across the U.S. But additional research on the per capita costs and losses of non-native forest pests in cities and throughout the entire U.S would produce a more robust comparison.

Because the largest fraction of land inside cities is in private residential use (Nowak et al., 1996) and because trees are an important component of the private residential landscape (Troy et al., 2007), many costs and losses associated with gypsy moth impact residential property values. Aukema et al. (2011) considered losses in residential property value to be the largest component of the total costs and losses from gypsy moth. These losses incorporate the effects of nuisance, defoliation, and mortality. We consider these effects individually, rather than estimating residential property value losses. Because the contingent valuation estimate of property value losses used by Aukema et al. (2011) could be inflated by compliance bias in the survey, summing the individual components of costs and losses provides a more conservative estimate.

We found that tree replacement was the single greatest cost even though it occurs at a much lower rate than tree removal. This is primarily driven by urban forestry standards, including those in Baltimore, that set a minimum tree size for planting on public property. However, with constrained urban forestry budgets, local governments might allocate more resources toward tree removal because of the risks posed by dead trees, using only the leftover funds for tree replacement, which would undermine progress toward tree canopy goals that have been set by many cities, including Baltimore (McPherson et al., 2011; Baltimore Tree Trust, 2013).

Our approach was designed to provide conservative cost estimates based on reasonable assumptions and available data and models. We demonstrated how to combine these into consistent scenarios, for example, with higher public suppression costs associated with fewer losses of tree benefits. In our framework, we varied removal and replacement rates across land use classes based on the likelihood that dead trees would pose a hazard, the potential for natural regeneration, and the relative value assigned to trees in various urban settings. Losses due to reduced tree benefits were challenging to quantify because of the wide range of non-market values affected (e.g., esthetic, energy, recreational). Because we could only measure a few of those values with existing data and tools, we obtained a conservative estimate of losses. One exception to our conservative assumptions is that we assumed immediate emissions of all carbon into the atmosphere when a tree died. However, we offset this by using a conservative value of CO^2, $10 per Mt, which is the floor price set by the California Environmental Protection Agency (2011) and less than the social value used by the U.S. Environmental Protection Agency (2013).

There are multiple potential sources of error in our model including the input tree data and the assumptions about gypsy moth impacts on trees and cost and loss estimates. However, our purpose was to demonstrate how to develop reasonable assumptions based on the scientific literature and/or input from subject matter experts and apply them to publicly available data. Clearly, different assumptions could be made in different cities. For example, the $700 cost of a replacement tree on public land in Baltimore might be different in another city, leading to different cost estimates.

Sampling error was also inherent in estimates of the city-wide tree species composition and size class distribution because these estimates were based on a random sample of plots (Nowak and Crane, 2000). With a sample-based inventory, sampling error decreases with increasing number of sample plots. The i-Tree Eco outputs were based on >200 sample plots, but even so, the standard errors for individual tree species were relatively high due to the variability of tree distribution in different land uses. However, combining trees into three host classes reduced the standard errors on the estimated total basal area and total number of trees within each class. We believe our estimates of Baltimore's tree population – characterized to the level of gypsy moth host class – and the potential impacts of a gypsy moth outbreak are robust.

Conceptually, the approach we developed for gypsy moth could be applied to other non-native forest pests and pathogens, such as the emerald ash borer, Asian longhorned beetle, or oak-wilt. While there would be less information available to populate the framework for these pests, it could be simplified in some instances. For example, defoliation rates and outbreak scenarios are unnecessary for emerald ash borer, which is host specific and causes much higher rates of tree mortality than gypsy moth. An even more critical constraint in urban areas could be the lack of knowledge about the structure and composition of forests, although the recent increase in urban forestry programs and tools have helped to address this deficiency (Nowak et al., 2008). If analyses in different cities or with different pests provided enough evidence to support cost-share programs (e.g., Kokotovich and Zeilinger, 2011), this could have spill-over benefits by reducing the movement of pests among cities and between cities and recreational areas, thus mitigating an important pathway for the spread of non-native forest pests (Colunga-Garcia et al., 2010; Tobin et al., 2010; Bigsby et al., 2011; Koch et al., 2012).

Conclusion

As the number of biological invasions continues to increase globally, more socioeconomic systems will be affected. While there is wide recognition of the potential impacts on commercial agriculture systems and on the composition and function of native communities (Liebhold et al., 1995b; Niemelä and Mattson, 1996; Pimentel et al., 2005), urban forests represent a very costly battlefield as well (Lard et al., 2002; Sydnor et al., 2007; Nowak et al., 2001; Aukema et al., 2011). This is partly due to the greater potential of biological invaders to directly impact humans and their potential to be spread through anthropogenic pathways. Our work synthesizes knowledge of a well-studied, non-native forest pest – the gypsy moth – in a framework that quantifies costs and losses in an urban forest setting. This study confirmed the potential for non-native forest pests to impose large economic costs in urban areas, demonstrated that costs and losses accrued primarily to private residents and local governments – supporting a cost-share approach to management – and provided a framework for conducting similar analyses of other non-native forest pests.

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