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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.

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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–\$60M (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 moth 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

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