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Potential impacts of emerald ash borer invasion on biogeochemical and water cycling in residential landscapes across a metropolitan region

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Abstract Trees provide important ecological services in cities, yet the vulnerability of the urban forest to massive tree losses from pest outbreaks could threaten those services, with unknown environmental consequences. The outbreak of emerald ash borer is an imminent threat to the ash population in North America. In the Minneapolis–Saint Paul, Minnesota, metropolitan area, ash trees are present in 50 % of residential landscapes in Ramsey and Anoka Counties. We used a large survey of household activities, a tree inventory, a Household Flux Calculator accounting tool, and a set of annual evapotranspiration measurements, to quantify the current carbon, nitrogen, and phosphorus storage in ash trees, the cycling of these elements, and the total evapotranspiration from ash trees in residential areas in the metropolitan region. Ash represented 6 % of the trees in residential areas and the removal of the entire ash population would correspondingly reduce net primary production and carbon sequestration by only a few percent and would have negligible effects on losses of nitrogen and phosphorus from residential landscapes. Similarly, the effects of ash loss on

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the hydrologic cycle would be minimal and would depend largely on management choices for the ground currently underneath ash tree canopies. Overall, the percentage change in biogeochemical and hydrological fluxes corresponded closely with the percent of the total urban tree population that was represented by ash, suggesting that areas with higher densities of ash would experience correspondingly larger effects. A hypothetical tree replacement scenario with similar broadleaf species was determined to be likely to re-establish the original biogeochemical and hydrological conditions once the replacement trees reach maturity.

Keywords Urban landscape · Households · Carbon flux · Nitrogen flux · Phosphorus flux · Agrilus planipennis · Evapotranspiration · Water flux

Introduction

Trees modify the cycling of carbon (C), nitrogen (N), phosphorus (P), and water in urban areas, potentially providing ecosystem services such as nutrient retention, storm water runoff mitigation, and other benefits (Westman 1977; Bolund and Hunhammar 1999). Yet trees in urban areas are highly vulnerable to outbreaks of pathogens and pests due to the limited species and genetic diversity typical of many urban forests. For example, the historic demise of the American chestnut (*Castanea dentata* (Marsh.) Borkh.) and, in recent decades, the devastation of American elm (*Ulmus americana* L.) and butternut (*Juglans cinerea* L.) by exotic fungal diseases resulted in large tree populations being lost from urban forests (Schlarbaum et al. 1999). Similarly, the asian longhorned beetle (*Anoplophora glabripennis* Motschulsky), which infests maple (*Acer* spp.) and other hardwoods, could potentially kill 10 % of the tree population in the U.S., with peaks of ≥60 % of the trees in some large cities (e.g., Chicago, Illinois and Boston, Massachusetts) resulting in a loss of >30 % canopy cover (Nowak et al. 2001).

In North America, the recent outbreak of emerald ash borer (*Agrilus planipennis* Fairmaire), a non-native phloem-feeding beetle, has the potential to kill most of the ash (*Fraxinus* spp.) population across a 25-state area over the next decade (Kovacs et al. 2010). Emerald ash borer was discovered in 2002 in the Detroit, Michigan area, in the upper midwestern United States, and neighboring areas of Ontario, Canada, and it has spread rapidly across the region (Poland 2007). Its impact is particularly severe in urban areas because white and green ash (*F. americana*, *F. pennsylvanica*) have been planted widely in boulevard and residential areas. Emerald ash borer was discovered in Saint Paul, Minnesota in the summer of 2009, less than a year later in the bordering city of Minneapolis, and in the summer of 2011 it was reported in the second-ring suburb of Shoreview. The limited success of measures to contain the spread of the beetle (Poland and McCullough 2006) suggests that virtually the entire ash population in the metropolitan region eventually will be lost.

The emerald ash borer outbreak in Minneapolis–Saint Paul (hereafter Twin Cities) provides a case study of how the loss of a widely planted urban tree species could affect biogeochemical and water cycling across a large metropolitan region. Changes in the composition of vegetation on residential lands can have a significant impact on urban areas. In the seven-county Twin Cities metropolitan area, residential land use represents 20 % of total land use as compared to 8 % for parks and preserves, and in some areas residential land use is even greater, with Ramsey County at 43 % (Metropolitan Council 2005). Ash represents >25 % of the total trees on public land in the city of Saint Paul, with approximately 35,000 ash trees in rights of way and 120,000 in parks, natural areas, and other open spaces (Saint Paul Department of Parks and Recreation 2009). However, it is unknown how many ash trees currently populate residential areas of the city. A 2010 urban forest inventory of residential areas in Saint Louis Park, a first-ring suburb of the Twin Cities, found that ash was the third most common genus, accounting for 12 % of the residential tree population (Gulsvig et al. 2010). A tree inventory in Shoreview, a second-ring suburb of the Twin Cities, indicated that ash represented 9 % of trees in residential areas (Sower et al. 2009). The responses of other cities affected by emerald ash borer and the management plans currently in place in the Twin Cities suggest that most of the ash tree population in managed parks and boulevards will be removed through proactive efforts to slow the spread of emerald ash borer using structural removal and immediate response when individual trees die. Conversely, in residential areas, the responses of homeowners are more difficult to predict, including whether and when they will remove ash trees on their property, whether they will replant, and which species they will select to replace the lost ash trees. In a Saint Louis Park survey, 77 % of homeowners expressed the intent to replace their ash trees if they become infested by emerald ash borer and a majority reported that they preferred hardwood shade trees as replacements (Fechtelkotter et al. 2010).

While the loss of ash trees in urban areas has been considered in terms of its economic impact (e.g. Kovacs et al. 2010), the biogeochemical and hydrologic impacts of losing a major urban forest species are largely unknown, especially for residential land. Information on changes in ecosystem processes associated with tree mortality due to insect infestation is scarce for urban ecosystems, as most studies have been conducted in natural forests (Kenis et al. 2009). However, natural forest ecosystem dynamics can differ strongly from those in urban areas. For example, in natural forests, N mineralization increases after tree mortality (e.g. Jenkins et al. 1999), but the reduced vegetative uptake of elements is often transitory because it is balanced by increased growth of understory vegetation (Jenkins et al. 1999). In urban ecosystems this situation is very unlikely, with the death of trees leaving a gap in vegetation cover, resulting in altered biogeochemical cycling of elements until the trees are replaced. Opposite to natural forests, where on-site decomposition of dead trees allows elements to return to the soil and contribute to internal recycling, diseased and dead trees in urban ecosystems are removed with virtually no element return to the soil. In urban areas, evapotranspiration from trees, which is the combination of transpired water loss and evaporation of intercepted rainfall, can provide ecosystem services (McPherson et al. 2005) such as reduced surface temperatures (Leuzinger et al. 2010; Oke 1989) and mitigation of storm water runoff (Mitchell et al. 2008; Wang et al. 2008). The removal of trees represents a potential reduction in urban evaporative water loss and increase in storm water runoff.

In this paper, we used data from the Twin Cities Household Ecosystem Project (TCHEP, Fissore et al. 2011), including a large survey of household activities, a tree inventory, a Household Flux Calculator accounting tool, as well as a unique set of annual evapotranspiration measurements (Peters et al. 2011) to quantify the current C, N, and P storage in ash trees, the cycling of these elements, and the total evapotranspiration from ash trees in residential areas in the metropolitan region. Then, using literature and recent survey results from a suburb within the study area, we developed scenarios about the percentage of the ash tree population that would ultimately be replaced and which tree species would be selected by homeowners for replanting. We used these scenarios as inputs to the Household Flux Calculator to generate estimates of the biogeochemical and water cycling rates that could potentially occur in the residential landscape once replacement had occurred and the new trees reached maturity. Our objectives were (1) to quantify the existing ash tree population's effects on C, N, P, and water cycling, effects that will be lost from these urban ecosystems according to a total tree loss scenario; and (2) to estimate the long-term effects on C, N, P, and water cycling, assuming likely replacement scenarios for the ash tree population.

Methods

Effect on cycling of C, N, and P in residential landscapes

In the spring and summer of 2008 we conducted a mail survey among single-family, owner-occupied, detached households in Ramsey and Anoka Counties, Minnesota (Nelson et al. 2008; Fissore et al. 2011) and asked for permission from homeowners to visit their properties to conduct on-the-ground vegetation assessments. Among the households that gave us permission to visit, we randomly selected 360 households distributed proportionally to housing density and visited them for vegetation measurements between May and August of 2008. The measurements included tree species, height, diameter at breast height (DBH, 1.4 m above ground), canopy width, tree mortality, and distance and direction from buildings for those trees >7 m tall and ≤20 m from the buildings. We did not include boulevard trees because these are managed by municipalities and not by homeowners. In addition, we obtained information concerning landscape management from the mailed survey. These included annual frequency of N fertilizer application (0, 1–2, 2–3, 4–5, >5 or done by lawn care company), irrigation regime (never, occasionally when the grass is dry, or once or more per week), and clippings and leaf litter management (left on site or removed from property).

Tree C, N, and P fluxes represent an integral component to the total household landscape biogeochemical budget, which is part of the Household Flux Calculator (HFC), a computational tool we developed to estimate C, N, and P fluxes thorough households associated with major human activities (Fissore et al. 2011). In estimating household landscape biogeochemical fluxes, the HFC accounts for a number of element inputs (including N and P atmospheric deposition and pet waste), outputs (including leaves and grass clippings removal), internal cycling of litterfall, and accumulation (in wood and soil). Through a series of computations and transformations, the HFC converts the collected data into units of elements that cycle through the household landscape in 1 year (Fissore et al. 2011, 2012).

Tree measurements were entered into the Urban Forest Effects (UFORE) model (Nowak et al. 2008) to obtain estimates of wood and leaf biomass and net primary production (NPP). For each tree we also measured the basal area (m^2, calculated across a sectional area of tree bole at breast height) and projected canopy ground area (m^2). We used the GlopNet database (Wright et al. 2004) to estimate conifer leaf longevity and leaf N and P content for the different species observed on the properties. Adjustments to take into account element resorption in senescent leaves and consequently estimate element concentration in litterfall were obtained from Kobe et al. (2005). We multiplied litter nutrient concentrations by leaf biomass to determine litterfall N and P fluxes. For leaf litter remaining on site, we assumed that litterfall was in equilibrium with litter decomposition such that all litterfall C was respired each year and all litter N and P entered the soil. Leaf litter removal by the household caused all elements (C, N, P) to leave the household system; hence, they did not contribute N and P to soil. Tree gross C sequestration in wood (net wood production) was also provided by the UFORE model, based on tree characteristics and from this we obtained tree wood NPP (kg Cyr^{-1}). Tree wood N and P was calculated based on C:N and C:P stoichiometry (Rodin and Bazilevich 1967).

Net landscape accumulation or loss of elements are estimated as the difference between element inputs to and outputs from the landscape. Landscape C accumulation rates equal the soil C accumulation rate, estimated as the difference between grass NPP and the sum of heterotrophic respiration and grass clipping removal rates (if clippings were removed, as reported in the survey), plus the calculated wood NPP (expressed in units of C). Specifically, grass NPP, soil heterotrophic respiration, and clipping fluxes were estimated using the biome-BGC ecosystem process model for Minneapolis, Minnesota (Milesi et al. 2005; C. Milesi, *personal communication*) that takes into account unmanaged lawn as well as well-watered, moderately to heavily fertilized lawns, with clippings removed or left on site to decompose (see also Fissore et al. 2011 for more details). Additional C, N, and P inputs were taken into account to estimate total landscape biogeochemical fluxes as part of the HFC. Briefly, these included atmospheric deposition of N and P based on local data (P data from Barr Engineering 2004; N data measured at the Cedar Creek National Atmospheric Program site in 2008), pet waste as estimated based on metabolic requirements of dogs and nutrient content in dog food (Fissore et al. 2011, 2012), and fertilizer application, based on reported application events per year by homeowners and by assuming that each application event occurs at 48.9 kg N ha^{-1}, as recommended by fertilizer companies.

Nitrogen and P accumulation or loss from the landscape was driven by C accumulation, assuming fixed C:N and C:P ratios of soil (Elliott 1986; Horgan et al. 2002) and wood (Rodin and Bazilevich 1967). If N and P inputs exceeded nutrient uptake by vegetation and soils plus exports as tree leaves or lawn clippings, we assumed that excess nutrients were lost, likely through runoff, leaching, and/or denitrification (N only); our model did not allow us to specify the loss pathways (see Fissore et al. 2011).

Tree replacement scenario development involved running the landscape component of the Household Flux Calculator using allometric equations specific to selected replacement species (*Acer* spp.*,* *Quercus* spp., and *Tilia* spp.) in place of each ash tree and comparing the resulting C, N, and P fluxes.

Effect on evapotranspiration from residential landscapes

Ash tree evapotranspiration rates were quantified using 2 years of sap flow measurements on trees in Ramsey County as part of a study examining species differences in water use (Peters et al. 2010). A total of four ash trees were measured, including two trees in a small neighborhood park in the first-ring suburb of Lauderdale and two trees on a residential lot in Saint Paul. The trees were healthy with open full canopies and were representative of mature ash trees in the study region, with a mean DBH of 38.6 cm, mean height of 17.7 m, and mean projected canopy area of 75.9 m^2. As described in Peters et al. (2011), evapotranspiration was calculated as the sum of transpiration, measured continuously using Granier-type heat dissipation sap flow (Granier 1987; Lu et al. 2004), and rainfall interception, estimated using a tree-based adaptation of the Rutter canopy interception model (Rutter et al. 1975; Valente et al. 1997). The interception model treated each tree as a closed canopy, and interception loss was calculated on a canopy-area basis using the Priestley–Taylor equation for potential evapotranspiration (Priestley and Taylor 1972). Following Wang et al. (2008), we modeled crown storage capacity as a function of leaf area index (LAI) considering specific leaf storage of 0.2 mm. The seasonal pattern of LAI was modeled using piece-wise logistic equations fit to stand-level LAI measurements (LAI-2000, LI-Cor, Lincoln, Nebraska, USA) that were collected starting prior to leaf-out and continuing until after senescence in 2007 and 2008 in the same study area (Peters and McFadden 2010).

To estimate total annual evapotranspiration from all other non-ash residential trees, we used average evapotranspiration rates for evergreen needleleaf and deciduous broadleaf trees measured within the study area (Peters et al. 2011). In the tree replacement scenario, we quantified 2008 annual transpiration rates for *Tilia* spp. using measurements from two *Tilia americana* trees, for *Acer* spp. using measurements from four diffuse-porous trees (*Tilia* and *Juglans* spp.), and for *Quercus* spp. using measurements from four ring-porous trees (*Fraxinus* and *Ulmus* spp.), as reported in Peters et al. (2010). Diffuse and ring-porous species differ in their wood structure, such that xylem vessels formed early in the growing season in ring-porous species are larger and have a higher capacity to conduct water than vessels formed later in the growing season, while vessels formed in diffuse-porous species are relatively uniform in size. Interception rates of replacement species were modeled the same as for ash trees.

Residential landscape management practices

Our approach for estimating element fluxes through the household included mailed surveys and landscape measurements. We developed a 40-question survey on household characteristics and choices (Nelson et al. 2008). Here, we use survey responses that related to landscape management practices (fertilization, irrigation, grass clipping, and leaf litter management), and landscape decision-making criteria. The survey, which took approximately 40 minutes to complete, was conducted between May and August of 2008. We asked household members to report activities for the previous year. The total response rate was 21 %, corresponding to 3,100 households.

Spatial scaling and statistical analyses

The 360 households that we sampled for field measurements were randomly selected and stratified by housing density, providing a representative sample of the residential landscapes of single-family, owner-occupied households in Ramsey and Anoka Counties (Fissore et al. 2011). We used a geographic information system (GIS) to extrapolate the number of ash trees and the biogeochemical fluxes (e.g. C accumulation) from the sampled households to the entire single-family residential area in Ramsey and Anoka Counties, Minnesota (Fig. 1). Briefly, using the GIS we computed the total area of single-family, detached residential land use in Anoka and Ramsey counties combined. Ash tree inventory data from each of the 360 sampled households were used to generate Thiessen polygons, which were used to delineate the area of influence around each household. Boundaries of these polygons define the area that is closest to each household relative to all other sampled households, therefore each polygon has a different area. To extrapolate tree measurements from the individual household to the single-family, detached residential area we first divided the measurement by the household property size to obtain a measure of density (values expressed in m^{-2}) and then we multiplied it by the Thiessen polygon size for that specific household. The sum of values across Thiessen polygons provided an estimate of the ash tree population in residential landscapes across Anoka and Ramsey Counties. This approach, in contrast with using the average for the 360 households we sampled and multiplying it to the entire residential area of Anoka and Ramsey Counties, allowed us to represent landscape heterogeneity across households. Total ash tree evapotranspiration for the study region was calculated by multiplying evapotranspiration on a projected canopy area basis by the total ash projected canopy area in the region.

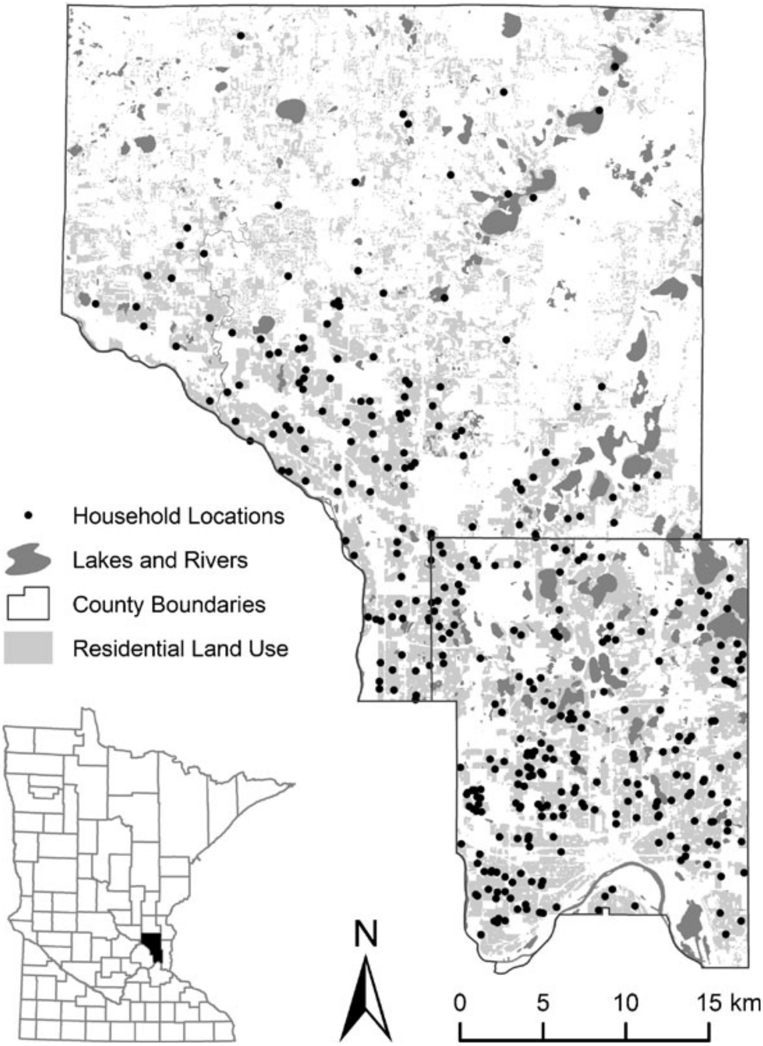


Fig. 1 Total single-family, detached residential area (pale grey area) in Ramsey (to the north) and Anoka (to the south) Counties, MN. Dots represent sampled sites (n=360) and each household location has been randomly shifted to protect the anonymity of survey respondents

Results

Residential ash tree population characteristics and landscape management practices

Among the measured 360 household landscapes, ash species composed about 6 % of the total trees occurring in the residential landscape, accounting for a total of 883 trees (Tables 1 and 2). About one-half of the sampled households had at least one ash tree on the property (172 out of 360 households), corresponding to a mean density of 1.8 trees km^{-2} of residential land (Table 2). Tree density varied along the gradient in housing density (Fig. 2), but not systematically, such that there was no significant relationship between tree

Table 1 List of ten most common genera measured in 360 household landscapes in the summer of 2008

Genus	Total number of trees	Percent of total trees
Pinus	2074	14.3
Acer	2068	14.3
Quercus	1736	12.0
Picea	985	6.8
Populus	928	6.4
Fraxinus	883	6.1
Ulmus	763	5.3
Prunus	746	5.2
Syringa	565	3.9
Rhamnus	517	3.6
Total number of trees§		14470

§All species identified on property, including those not listed in the table

density and housing density or between tree density and the distance from the center of the city of Saint Paul. Among those households with at least one ash tree on the property, ash trees represented between 1 % and 100 % of the total number of trees (mean ± SD=18± 16 %, median =13 %). For 10 % of households, ash trees represented >40 % of the total trees on the property. The size of ash trees varied greatly in our sample with an average diameter (DBH) of 28±21 cm (mean ± SD, Fig. 3). A majority (58 %) had a diameter of <30 cm, whereas only 9 % had diameter >60 cm (maximum =106 cm), resulting in an overall mean basal area of 0.1 m^2 km^{-2} (Table 2). The mean ash tree height was 12±7 m (mean ± SD). Among our sample of 360 households, the mean projected canopy area of ash trees was 85.4 m^2 km^{-2}, accounting for about 5 % of total projected canopy area of all residential trees. Scaled to the entire single-family, detached residential area in Ramsey and Anoka counties, we estimated a population of 53.0×10^4 ash trees, accounting for a total of 44.8×10^3 m^2 basal area and 28.8×10^6 m^2 projected canopy area.

Households that did not remove leaves from their property typically had larger landscape size than those that removed leaves from the entire landscape (2003±2400 m^2 and 1093± 1945 m^2, respectively). Homeowners who removed leaves from only part of their property had on average slightly smaller landscape areas (1023±720 m^2) and had fewer ash trees and total trees than those who either completely removed leaves or allowed leaves to remain on

Table 2 Characteristics of ash trees (*Fraxinus* spp.) inventoried in 360 household landscapes in Ramsey and Anoka counties. Values are means, with maxima and minima given in parentheses

Ash tree population characteristics	
Total single-family residential area size (ha) for 360 households	49.6
Total number of ash trees for 360 households	883
Percent of total trees that were ash trees (%)	6
Mean diameter of ash trees (cm)	28.8 (2.5 106.3)
Mean height of ash trees (m)	12.2 (1.3 72.0)
Ash tree density (number trees km^{-2})	1.8
Total ash tree basal area (m^2)	5.7 (0 5.6)
Total basal area (m^2 km^{-2})	0.1
Projected canopy ash tree area (m^2)	57.4 (0.6 576.8)
Projected canopy ash tree area (m^2 km^{-2})	85.4

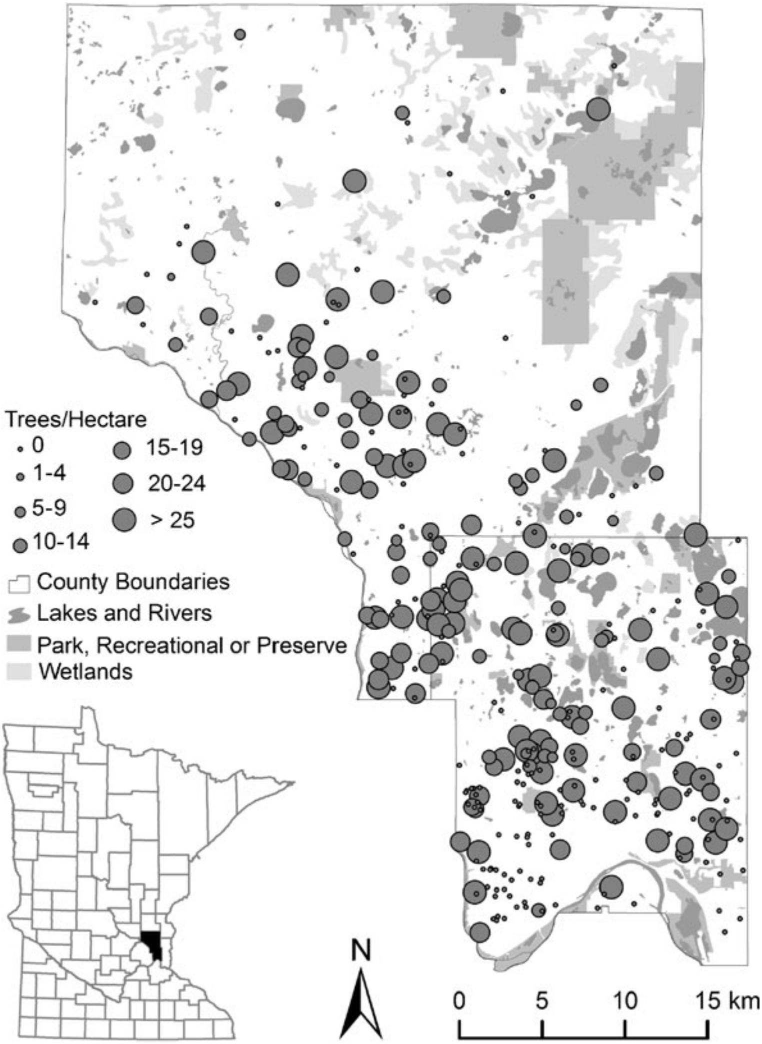
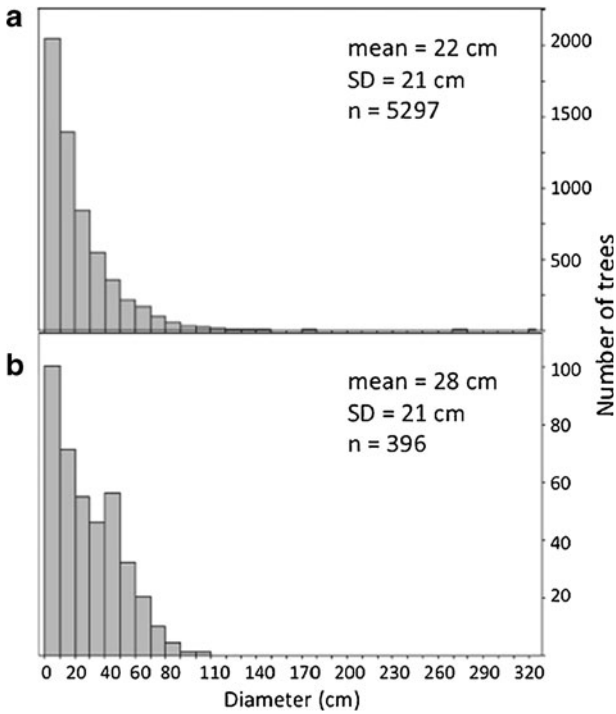


Fig. 2 Distribution of ash tree density classes for the 360 measured household landscapes. Dots represent sites with no ash trees and each household location has been randomly shifted to protect the anonymity of survey respondents

site. In contrast, we did not observe any significant difference in number of trees between households that removed or left all leaves (data not shown). Further, those households that left leaves on site were typically located at greater distance from the Saint Paul city center than those that did remove or that only partially removed the leaves (19±10 km, 14±8 km, and 13±9 km, respectively).

Twenty seven percent of the households having at least one ash tree on the property did not fertilize their lawn, whereas the remainder fertilized their lawn or hired a lawn care company to do so. Among the households with ash trees, 20 % irrigated rarely, 49 % irrigated occasionally (when the grass is dry), and 31 % irrigated regularly (once or more per week). Among homeowners with ash trees, 87 % left grass clippings on site, whereas 13 %

Fig. 3 Distribution of diameters of **a** all residential trees and **b** residential ash trees (measured at breast height, DBH)



removed grass clippings from the property. Overall, 62 % of the ash trees were located <20 m away from the house itself, a proximity that could afford shading effects on home energy use.

Effects of ash trees on biogeochemical cycling and evapotranspiration

For the entire Anoka and Ramsey Counties residential area, we estimated that the ash tree population contributed 3.1 % of total C input to the landscape. Accumulation of C, N, and P in ash tree wood corresponded to 4.5 % of total household landscape C, N, and P wood accumulation among all trees.

Leaves that remained on site rather than being removed from the property were assumed to completely decompose within 1 year and contribute to soil nutrient cycles that eventually could become available for plant uptake. Our survey showed that 41 % of households removed all and 16 % removed part of the leaf litter from the property, whereas 42 % did not remove leaves. Ash trees alone, through their litter, contributed between 6.3 and 8.4 % of total leaf litter element fluxes (Table 3). Assuming that all existing ash trees in residential areas of Ramsey and Anoka Counties were removed due the invasion of emerald ash borer (and assuming no change in N fertilizer application rate), we calculated that the reduced element uptake, reduced accumulation in wood and soil, and reduced recycling of N through litterfall together would result in a 1.9 % increase in the amount of N available to be potentially lost from the household landscape through storm water runoff, denitrification, leaching, etc. We did not do the same analysis for P because household landscapes in our study region generally retain P (i.e., inputs of P are less than ecosystem demand, Fissore et al. 2011) and would be expected to do so even if ash trees were removed.

Table 3 Estimated total C, N, and P in litterfall of ash trees for all single-family residential households in Ramsey and Anoka counties, Minnesota, and the percent of total household landscape element input fluxes (including all trees on property) they represented. Total litterfall C, N, and P rates refer to those elements in the total ash tree litterfall, whereas litter-soil flux represents the fraction of total litterfall left on site to decompose (i.e., the fraction that was not raked) and therefore contribute to soil nutrient cycling

	Carbon (kg C yr^{-1})	Nitrogen (kg N yr^{-1})	Phosphorus (kg P yr^{-1})
Total litterfall	3.4×10^{6}	6.3×10^{4}	5.9×10^{3}
% of total household landscape inputs	7.3	6.3	8.4
Litter-soil flux	2.6×10^{7}	5.8×10^{5}	4.2×10^{4}
% of total household landscape inputs	7.2	6.1	7.8

Residential ash trees in Anoka and Ramsey Counties transpired 3.6×10^{6} m^{3} water yr^{-1} and intercepted 2.7×10^{6} m^{3} water yr^{-1}, for a total annual evapotranspiration rate of 6.3×10^{6} m^{3} water yr^{-1}. Evapotranspiration from ash trees represented 6.4 % of the total evapotranspiration from all residential trees (9.9×10^{7} m^{3} water yr^{-1}). Assuming that all residential trees had a turfgrass understory equal in area to the projected canopy area, and with evapotranspiration rates equal to those reported in Peters et al. (2011) from an open turfgrass lawn in the same study area, then residential tree covered areas had a total annual evapotranspiration rate of 13.8×10^{7} m^{3} water yr^{-1} (Table 4). If all residential ash trees were removed due to the emerald ash borer invasion, and with no change in understory evapotranspiration, this would reduce evaporative water loss from the ecosystem and could result in an increase in runoff during rain events. Removal of the ash canopy, however, would expose understory turfgrass to greater solar radiation and thereby increase turfgrass water use. If all residential areas currently covered by ash canopies were replaced by open-grown turfgrass with evapotranspiration rates the same as those reported in Peters et al. (2011), total evaporative water loss actually would increase by 4.3 % to 14.4×10^{7} m^{3} water yr^{-1}.

Effects of ash tree replacement scenarios

We explored the hypothetical scenario that all ash trees would be removed and that 100 % of them eventually would be replaced with a different tree species based on homeowner preferences and suitability to the Twin Cities regional climate. Using the Household Flux Calculator, we estimated the long-term, ultimate biogeochemical consequences of replacing ash trees with different broadleaf deciduous species that would likely be selected. Replacing all ash trees with an equal combination of maple, oak, and basswood trees (based on a survey

Table 4 Changes in biogeochemical and water cycles from current household landscape conditions to different hypothetical scenarios of ash tree replacement

	Present conditions		Ash replacement scenarios		
	All trees[§]	Maple	Oak	Basswood	Mix[$]
Total C inputs (kg C yr^{-1})	219.27×10^{6}	220.95×10^{6}	222.45×10^{6}	218.72×10^{6}	220.71×10^{6}
Total C accumulation (kg C yr^{-1})	869.82×10^{5}	895.93×10^{5}	910.77×10^{5}	877.68×10^{5}	894.68×10^{6}
Total N accumulation (kg N yr^{-1})	393.50×10^{4}	364.04×10^{4}	364.04×10^{4}	394.39×10^{4}	394.16×10^{4}
Total P accumulation (kg P yr^{-1})	317.28×10^{2}	322.28×10^{2}	322.39×10^{2}	325.52×10^{2}	323.40×10^{2}
Total evapotranspiration[a] (m^{3} H_2O yr^{-1})[b]	1.38×10^{8}				1.39×10^{8}

of homeowner preferences in Saint Louis Park. Fechtelkotter et al. 2010) resulted similar C inputs to the landscape (Table 4). For the same tree replacement scenario, and assuming no change in N fertilizer application regime, our model estimated only minimal reduction in N that is in excess of ecosystem demand (0.3 %) and hence potentially lost from the landscape. Assuming no change in understory turfgrass evapotranspiration rates, the tree replacement scenario resulted in a 0.7 % increase in annual evaporative water loss (1.4×10^{8} m^{3} water yr^{-1}) due to slightly higher annual transpiration rates of diffuse-porous tree species (*Acer* and *Tilia* spp.) as compared to ring-porous species (*Fraxinus* and *Quercus* spp.) (Peters et al. 2010). The estimates above assume a tree replacement rate of 100 %; however, it may be more accurate to assume a 75 % replacement rate (Fechtelkotter et al. 2010), which would reduce the estimates in Table 4 by 25 %.

Discussion

Biogeochemical and water cycling effects due to loss of ash trees

In Ramsey and Anoka Counties, ash trees represented 6 % of the total tree population in residential areas, a much smaller proportion of all trees than the 25 % estimated on public land in Saint Paul, for example. However, the importance of ash trees varied among municipalities and neighborhoods. For example, our survey estimated that ash accounted for 9 % of the residential tree population in Shoreview, a second-ring suburb of Saint Paul, which was consistent with an estimate of 9.5 % from another, independent tree inventory (Sower et al. 2009). Residential ash trees represented 12 % of the tree cover in the suburb of Saint Louis Park (Gulsvig et al. 2010). Overall, approximately 50 % of the households surveyed in Ramsey and Anoka counties had at least one ash tree on the property, consistent with a recent survey conducted in the Saint Anthony Park neighborhood in Saint Paul, where 40 % of 104 surveyed households had at least one ash tree (Jorgensen et al. 2010). This, along with the fact that many were large, mature ash trees, suggests that their importance to many homeowners for shade or aesthetic value could be greater than is apparent from the percentage of trees that ash represents for all residential land in the region.

In residential areas, homeowner willingness to treat ash trees with pesticides appears to be fairly low, based on a recent survey of homeowners in a Saint Paul neighborhood (Jorgensen et al. 2010). Even if insecticide treatments were used to slow the spread of emerald ash borer in some areas, the majority of ash trees in the region likely will be infested by emerald ash borer and eventually will have to be removed. There is evidence indicating high likelihood of near-total loss in the proximity of emerald ash borer outbreak points (MacFarlane and Meyer 2003). However, it is important to consider the number of diverse initiatives that have been proposed or undertaken. Among some of the most relevant 20-years plans in the region that concern residential intervention are planting a new tree before decline—near the original ash tree location—to minimize perturbation (neighborhood initiative) and anticipating gradual removal with ash decline (municipal plan). For the purpose of this work, we focused on two opposite scenarios aimed at investigating biogeochemical and hydrological potential impacts of total loss of ash trees (by assessing current ash tree C, N, P and water fluxes) and long-term ultimate biogeochemical effects of total ash tree replacement. The latter used a range of tree replacement options based on climatic suitability and homeowner preferences. Our methods (use of the Household Flux Calculator to estimate C, N, and P flux rates; Fissore et al. 2011), did not allow us to model trajectories of the population over time, but rather we quantified the biogeochemical and water flux rates of the hypothetical replacement tree species once they had reached the same stage of maturity as the current urban forest.

Our results showed that the current residential ash tree population has relatively small effects on the total landscape biogeochemistry and water cycling the residential area of the metropolitan region. This was because, despite the large variability in tree size of the ash population in the residential areas, there was a close correspondence between the percentage of the urban forest represented by ash trees and the percentage by which total biogeochemical fluxes were changed. The loss of all existing ash trees, without replacement, would result in a small C loss in residential landscapes, corresponding to the C removed with ash tree wood. The effects of ash removal on N and P cycling are predominantly related to household management decisions about disposing of leaf litter and fertilization regime. For example, if ash tree loss is not followed by a reduction in N fertilizer application, more N will be lost through storm water runoff with potential negative consequences for water quality because of reduced N uptake by trees.

Currently, P in the household landscapes appears to be less than ecosystem demand (Fissore et al. 2011, 2012), likely as a consequence of legislation that currently restricts P fertilizer use in Minnesota. Under these circumstances, P necessary to support ecosystem demand is derived from internal recycling of leaf litterfall P, atmospheric deposition, pet waste, and uptake from soil pools (Fissore et al. 2011, 2012). The removal of all ash trees in the region is not estimated to reverse this situation.

The net hydrologic effect of residential ash tree removal due to emerald ash borer, while similarly small and proportional to the total ash tree cover in our study area, depended on ash tree evapotranspiration rates as well as the management of underlying ground cover. For example, if ash canopies overlie impervious surfaces such as asphalt or concrete, then runoff is likely to increase when trees are removed because the trees would no longer return water to the atmosphere by evapotranspiration. In contrast, if ash canopies overlie vegetated surfaces, such as turfgrass with higher per area evapotranspiration rates than trees when openly grown (Peters et al. 2011), then total ecosystem evapotranspiration may increase and result in less runoff, assuming soil water holding capacity is sufficiently high to store water immediately after rain events. Municipal water use could also increase if ash canopies were replaced with open grown turfgrass because homeowners tend to irrigate turfgrass lawns more than trees. Additionally, our per-tree evapotranspiration measurements came from fully mature trees, which tend to have a higher leaf area index than small trees. Therefore, our scaled-up estimates for the metropolitan region are likely an overestimate of the actual total tree evapotranspiration.

In addition to the effects on biogeochemical and hydrological fluxes that we have described here, the loss of a large number of ash trees would also diminish ecosystem services in residential areas. Among these, ash removal would likely result in a decrease in property value, decreased shading and consequent lost energy saving from cooling bills (Parker 1983), reduced air pollution removal, and other effects. Large trees are often preferred for shading and other environmental services as well as for their social benefits (Lohr et al. 2004; Kuo 2001), as in Toledo, Ohio where residents expressed high aesthetic value of a canopied street (Heimlich et al. 2008).

Effects of potential ash tree replacement scenarios

It is likely, given that ash trees were present in more than half of the households in the region and that they represented 18 % of all trees on those properties, that homeowners will undertake some degree of tree replacement in response to emerald ash borer invasion. Considering that more diverse forest communities are less susceptible to insect and pest invasions (Brockerhoff et al. 2006), the incentive to replace dead or diseased ash trees in residential landscapes could represent an opportunity to increase species diversity. For example, in the Hamline-Midway neighborhood of Saint Paul, volunteers inventoried residential properties for the total number of trees, percent of ash trees, and availability of space to plant new trees. The neighborhood council's 2011 goal was to plant 100 trees in this area in anticipation of ash tree loss. Initial species selected were bur oak, Kentucky coffee tree, basswood, and Princeton elm; preference was for big trees that grew quickly. Cities are also responding to emerald ash borer infestation by changing regulations and programs. For example, Savage and Prior Lake, Minnesota removed ash trees from the 'significant status list'; developers no longer have to replace these trees when they prepare a site for construction. Other cities provide discounted replacement trees and management support for residents who want to replace their ash trees (Fairfield 2007).

A recent survey in Saint Louis Park, Minnesota found that about 77 % of households would consider replacing infested ash trees, preferentially with species with similar aesthetic characteristics and shading properties (Fechtelkotter et al. 2010). The same survey indicates that homeowners' choices would likely fall among maple, oak, basswood, and pine, which are already among the most common species in the residential landscape. We ran our model based on those reported preferences to estimate the long-term ultimate biogeochemical consequences of ash tree replacement assuming that the trees had reached similar stages of maturity as the current ash population.

We estimated that if ash trees were replaced by other broadleaf species with similar canopy and shade characteristics, the change in biogeochemical cycles would be minimal across residential landscapes, although these values may be more or less important depending on ash trees presence in different neighborhoods. Although different trees have small differences in rates of wood and leaf NPP, ultimately the number of trees being replanted and the homeowners' management practices (e.g., allowing for internal recycling of N and P by leaving leaf litter on site) have a larger impact on the cycling of C, N, P, and water than does tree species choice within a large range of broadleaf trees common to urban areas.

Conclusions

Our results suggest that, despite the large number of ash trees currently populating the residential landscape, the biogeochemical and hydrological effects of an emerald ash borer infestation and hypothesized total ash tree loss would be modest, with changes in C, N, P, and evapotranspiration flux rates between 4.5 % and 7.8 %. This is because ash, although it was one of the ten most common genera in household landscapes, represented only 6 % of the total tree population in the residential areas of the metropolitan region. The potential biogeochemical consequences of the emerald ash borer invasion are likely to be greater on public land in cities such as Saint Paul, where ash trees represent 25 % of total trees. The close correspondence between the percent change in flux and the percent of the total urban forest removed, independent of tree size, suggests that, as far as residential landscapes are concerned, the ash tree population in the study area was similar to the "average tree" in the urban forest.

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