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Modeled $PM_{2.5}$ removal by trees in ten U.S. cities and associated health effects

David J. Nowak [a,*], Satoshi Hirabayashi [b], Allison Bodine [b], Robert Hoehn [a]

[a] USDA Forest Service, 5 Moon Library, SUNY-ESF, Syracuse, NY 13210, USA
[b] Davey Institute, 5 Moon Library, SUNY-ESF, Syracuse, NY 13210, USA

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ABSTRACT

Urban particulate air pollution is a serious health issue. Trees within cities can remove fine particles from the atmosphere and consequently improve air quality and human health. Tree effects on $PM_{2.5}$ concentrations and human health are modeled for 10 U.S. cities. The total amount of $PM_{2.5}$ removed annually by trees varied from 4.7 tonnes in Syracuse to 64.5 tonnes in Atlanta, with annual values varying from $1.1 million in Syracuse to $60.1 million in New York City. Most of these values were from the effects of reducing human mortality. Mortality reductions were typically around 1 person yr^{-1} per city, but were as high as 7.6 people yr^{-1} in New York City. Average annual percent air quality improvement ranged between 0.05% in San Francisco and 0.24% in Atlanta. Understanding the impact of urban trees on air quality can lead to improved urban forest management strategies to sustain human health in cities.

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1. Introduction

Fine particulate matter less than 2.5 microns ($PM_{2.5}$) is associated with significant health effects that include premature mortality, pulmonary inflammation, accelerated atherosclerosis, and altered cardiac functions (e.g., Pope et al., 2004). A 10 μg m^{-3} increase in fine particulate matter has been associated with an approximately 4%, 6%, and 8% increased risk in all-cause, cardiopulmonary and lung cancer mortality, respectively (Pope et al., 2002). The regulation of these pollutants by the U.S. Environmental Protection Agency (U.S. EPA) has resulted in significant improvements in air quality over the last decade with reductions in monitored $PM_{2.5}$ from 2000 to 2007 associated with 22 000–60 000 net avoided premature mortalities in the United States (Fann and Risley, 2011).

Trees are often a major element of the city landscape with urban tree cover in the United States averaging 35.0% (Nowak and Greenfield, 2012a). Trees directly affect particulate matter in the atmosphere by removing particles (e.g., Beckett et al., 2000a; Freer-Smith et al., 2004) and emitting particles (e.g., pollen) or through resuspension of particles captured on the plant surface. Some captured particles can be absorbed into the tree, though most particles that are intercepted are retained on the plant surface. The intercepted particle often is resuspended to the atmosphere, washed off by rain, or dropped to the ground with leaf and twig fall. Consequently, vegetation is only a temporary retention site for many atmospheric particles. Trees can also affect particulate matter concentration by altering air temperatures, emitting volatile organic compounds and altering energy use (e.g., tree shade on building, altering wind speeds, cooling air temperatures) and consequent emissions from power plants (e.g., Heisler, 1986; Smith, 1990; Beckett et al., 1998). At the local scale, interior parts of forest patches within urban areas can have substantially lower concentrations of particulate matter than forest edges (Cavanagh et al., 2009).

To date, most research related to urban trees and particulate matter has focused on removal of particulate matter less than 10 microns (PM_{10}) by trees. Increasing total tree cover in West Midlands, UK from 3.7% to 16.5% is estimated to reduce average primary PM_{10} concentrations by 10% from 2.3 to 2.1 μg m^{-3} (removing 110 tonnes per year); increasing tree cover from 3.6% to 8% in Glasgow, UK is estimated to reduce PM_{10} concentrations by 2%, (removing 4 tonnes per year) (McDonald et al., 2007). In the Greater London area (UK), urban tree canopies are estimated to remove between 852 and 2121 tonnes of PM_{10} annually, which equates to 0.7%–1.4% PM_{10} air quality improvement (Tallis et al., 2011). A 10 × 10 km grid in London with 25% tree cover was estimated to remove 90.4 tonnes of PM_{10} per year, which equated to the avoidance of 2 deaths and 2

* Corresponding author.
E-mail address: dnowak@fs.fed.us (D.J. Nowak).

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hospital emissions per year (Tiwary et al., 2009). PM_{10} removal by urban trees in the United States has been estimated at 214 900 tonnes per year (Nowak et al., 2006a).

Various studies to date have investigated the removal rate and resuspension of $PM_{2.5}$ by trees (e.g., Beckett et al., 2000b; Freer-Smith et al., 2004, 2005; Pullman, 2009), but none have estimated the overall impact of the trees and forests in a city on $PM_{2.5}$ concentrations. The objective of this paper is to estimate, on an hourly basis over the course of a year, the amount of $PM_{2.5}$ removal and resuspension by trees within 10 U.S. cities and its effect on $PM_{2.5}$ concentrations, including the associated values and impact on human health.

2. Methods

To estimate the effects and associated values of $PM_{2.5}$ removal by urban trees in 10 cities (Table 1), four types of analyses were conducted that estimated: 1) the total leaf area in the city on a daily basis, 2) the hourly flux and resuspension of $PM_{2.5}$ to and from the leaves, 3) the effects of hourly $PM_{2.5}$ removal by trees on $PM_{2.5}$ concentration in the atmosphere, and 4) the health incidence impacts and monetary value of the change in $PM_{2.5}$ concentration using information from the U.S. EPA Environmental Benefits Mapping and Analysis Program (BenMAP) model (U.S. EPA, 2012a).

2.1. City tree population parameters

To determine the leaf surface area within the 10 U.S. cities, field data on trees were measured within randomly selected 0.04 ha plots and analyzed using the i-Tree Eco model (Table 1; Nowak et al., 2008). The model estimated the total leaf area index per unit of tree cover (LAI = one-sided leaf area in crown divided by projected crown area on the ground; i.e., the number of layers of leaves within the crown) and percent of the tree population that is evergreen. Tree cover within each city was estimated by photo interpreting random points throughout each city with recent imagery (Table 1; Nowak and Greenfield, 2012b). Total city leaf area was estimated by multiplying city tree cover (m^2) by city LAI per unit of tree cover ($m^2\ m^{-2}$). Leaf area index values were combined with percent evergreen information and local leaf on and leaf off dates to estimate total daily leaf surface area in each city.

2.2. $PM_{2.5}$ removal by trees

Hourly pollution removal or flux (F in $\mu g\ m^{-2}\ hr^{-1}$) can be estimated as:

$$F = V_d \times C \quad (1)$$

where V_d is the deposition velocity of the pollutant to the leaf surface ($m\ h^{-1}$) and C is pollutant concentration ($\mu g\ m^{-3}$) (e.g., Hicks et al., 1989). Daily (24-h) pollution concentrations of $PM_{2.5}$ in each city were obtained from U.S. EPA monitors for the year of 2010. If more than one monitor existed, the daily values were averaged for each day to produce a city average value. The average daily value was used to represent the hourly concentration values throughout the day.

Deposition velocities of $PM_{2.5}$ to trees were estimated from the literature and varied with wind speed (Beckett et al., 2000b; Freer-Smith et al., 2004; Pullman, 2009). These papers measured deposition velocities to tree leaves from 17 tree species under wind speeds of 1, 3, 6, 8, 9 and 10 $m\ s^{-1}$. For each wind speed, the median deposition velocities from the measured deposition velocities was used to estimate the V_d for that wind speed per unit leaf area (Table 2). The standard error of the estimates among the species was used to estimate a potential range of values of deposition velocities. The 95 percent confidence interval of median deposition velocity per wind speed was used to estimate a maximum deposition for the wind speed. As 95 percent confidence interval for the lower range of deposition velocities produced negative deposition velocities, the minimum average V_d from any species was used to represent the minimum V_d for the wind speed. To estimate the V_d for wind speeds between 1 and 10 $m\ s^{-1}$ that did not have a measured V_d, values were interpolated between the closest measured values. For wind speeds above 10 $m\ s^{-1}$, the V_d for 10 $m\ s^{-1}$ was used; for a wind speed of 0 $m\ s^{-1}$, the V_d was assumed to be 0 $m\ s^{-1}$ (Table 3).

Resuspension of $PM_{2.5}$ from trees was estimated from Pullman (2009) and varied with wind speed. This paper measured percent resuspension of $PM_{2.5}$ from tree leaves of three tree species under wind speeds of 6.5, 10 and 13 $m\ s^{-1}$. The average percent resuspension for the trees species and wind speed was calculated (Table 3). As the percent resuspension for the wind speed of 6.5 $m\ s^{-1}$ was 9.5%, a value of 9% was assumed for a wind speed of 6 $m\ s^{-1}$ and 10% for 7 $m\ s^{-1}$. The percent resuspension for a wind speed of 0 $m\ s^{-1}$ was assumed to be 0%. To estimate the percent resuspension for wind speeds between 0 and 13 $m\ s^{-1}$ that did not have measured resuspension rates, values were interpolated from the closest measured values (or assumed value at wind speed of 0 $m\ s^{-1}$). For wind speeds above 13 $m\ s^{-1}$, the percent resuspension rate for 13 $m\ s^{-1}$ was used (Table 3).

To calculate pollution removal, local city weather data from the National Climatic Data Center were used to obtain hourly wind speed and precipitation data. Hourly flux values to trees in the city (Eq. (1); $\mu g\ m^{-2}\ h^{-1}$) were multiplied by total leaf surface area (m^2) with hourly V_d based on local wind speed (Table 3). Flux values were accumulated hourly with a percent of the total accumulated $PM_{2.5}$ over the current and previous hours resuspended back to the atmosphere hourly based on local wind speed. $PM_{2.5}$ was accumulated upon leaves and resuspended from leaves

Table 1
Number of field plots and tree cover estimates in cities.

City	Plots			Tree cover (%)[b]	Cover year[c]
	#	#km^{-2}	Year[a]		
Atlanta, GA[d]	205	0.6	1997	52.1	2009
Baltimore, MD[d]	195	0.9	2009	28.5	2005
Boston, MA[d]	217	1.5	1996	27.9	2008
Chicago, IL[e]	745	1.2	2007	18.0	2009
Los Angeles, CA[f]	348	0.3	2007–2008	20.6	2009
Minneapolis, MN[g]	110	0.7	2004	34.1	2008
New York, NY[h]	206	0.3	1996	19.7	2009
Philadelphia, PA[i]	210	0.6	1996	20.8	2010
San Francisco, CA[j]	194	1.6	2004	16.0	2011
Syracuse, NY[d]	198	3.0	2009	26.9	2009

#Number of plots.
[a] Year of plot field data collection.
[b] Cover estimates from photo-interpretation (Nowak and Greenfield, 2012b). Philadelphia and San Francisco are unpublished estimates, U.S. Forest Service, Syracuse, NY.
[c] Year of imagery for cover estimates.
[d] Unpublished data from U.S. Forest Service, Syracuse, NY.
[e] Nowak et al., 2010.
[f] Nowak et al., 2011.
[g] Nowak et al., 2006b.
[h] Nowak et al., 2007a.
[i] Nowak et al., 2007b.
[j] Nowak et al., 2007c.

Table 2
Summary of average deposition velocities ($cm\ s^{-1}$) of $PM_{2.5}$ by wind speed from the literature per unit leaf area.

Species	Wind speed ($m\ s^{-1}$)				
	1	3	6	8.5[a]	10
Quercus petraea[b]		0.831	1.757	3.134	
Alnus glutinosa[b]		0.125	0.173	0.798	
Fraxinus excelsior[b]		0.178	0.383	0.725	
Acer pseudoplatanus[b]		0.042	0.197	0.344	
Pseudotsuga menziesii[b]		1.269	1.604	6.04	
Eucalyptus globulus[b]		0.018	0.029	0.082	
Ficus nitida[b]		0.041	0.098	0.234	
Pinus nigra[c]	0.13	1.15		19.24	28.05
Cupressocyparis x leylandii[c]	0.08	0.76		8.24	12.2
Acer campestre[c]	0.03	0.08		0.46	0.57
Sorbus intermedia[c]	0.04	0.39		1.82	2.11
Populus deltoides[c]	0.03	0.12		1.05	1.18
Pinus strobus[d]	0.0108				
Tsuga canadensis[d]	0.0193				
Tsuga japonica[d]	0.0058				
Picea abies[e]	0.0189				
Picea abies[e]	0.038				
Median	0.030	0.152	0.197	0.924	2.110
SE[f]	0.012	0.133	0.281	1.610	5.257
Maximum[g]	0.057	0.442	0.862	5.063	14.542
Minimum[h]	0.006	0.018	0.029	0.082	0.570

[a] Combination of 8 and 9 $m\ s^{-1}$ wind speeds.
[b] From Freer-Smith et al. (2004).
[c] From Beckett et al. (2000b).
[d] From Pullman (2009). Included particles up to 3.0 μm in diameter.
[e] From Pullman (2009). Based on maximum and minimum of reported range. Included particles up to 3.8 μm in diameter.
[f] Standard error.
[g] Based on 95 percent confidence interval above median value.
[h] Based on lowest recorded value for any species.

Table 3
Deposition velocities and percent resuspension by wind speed per unit leaf area.

Wind speed ($m\ s^{-1}$)	Deposition velocity (V_d)			Resuspension (%)
	Average ($cm\ s^{-1}$)	Minimum ($cm\ s^{-1}$)	Maximum ($cm\ s^{-1}$)	
0	0.00	0.000	0.000	0
1	0.03	0.006	0.042	1.5
2	0.09	0.012	0.163	3
3	0.15	0.018	0.285	4.5
4	0.17	0.022	0.349	6
5	0.19	0.025	0.414	7.5
6	0.20	0.029	0.478	9
7	0.56	0.056	1.506	10
8	0.92	0.082	2.534	11
9	0.92	0.082	2.534	12
10	2.11	0.570	7.367	13
11	2.11	0.570	7.367	16
12	2.11	0.570	7.367	20
13	2.11	0.570	7.367	23

during non-precipitation periods. During precipitation events, the accumulated $PM_{2.5}$ was assumed to be washed off to the ground surface depending upon the magnitude of the precipitation event (Pe in mm). As leaves capture about 0.2 mm of precipitation (Wang et al., 2008) before runoff from the leaf, the total precipitation storage capacity (Ps in mm) of the canopy was calculated as 0.2 × LAI. If Pe was greater than Ps, then all particles were assumed to be removed from the leaves and resuspension dropped to zero. When the Pe was less than Ps, no particles were removed from the leaves as there was no runoff from the leaves. After the rain stopped, $PM_{2.5}$ began accumulating on and resuspending from leaves again. Water on the leaves after rain events was reduced hourly based on evaporation rates calculated from meteorological conditions. The annual flux to tree leaves was estimated as the total $PM_{2.5}$ washed off leaves during the year plus the amount remaining on leaves at the end of the year.

2.3. Change in $PM_{2.5}$ concentration

To estimate percent air quality improvement due to dry deposition (Nowak et al., 2000), hourly boundary layer heights were used in conjunction with local hourly fluxes and resuspension rates in each city. Daily morning and afternoon mixing heights were calculated using the EPA's mixing height program (U.S. EPA, 1981) with upper air data from the nearest radiosonde station. These mixing heights were then interpolated to produce hourly boundary layer height values using the EPA's PCRAMMIT program (U.S. EPA, 1995). Minimum boundary-layer heights were set to 150 m during the night and 250 m during the day based on estimated minimum boundary-layer heights in cities. Hourly mixing heights (m) were used in conjunction with pollution concentrations ($\mu g\ m^{-3}$) to calculate the amount of pollution within the mixing layer ($\mu g\ m^{-2}$). This extrapolation from ground-layer concentration to total pollution within the boundary layer assumes a well-mixed boundary layer, which is common in the daytime (unstable conditions) (Colbeck and Harrison, 1985). Hourly change in $PM_{2.5}$ concentration was calculated as:

$$\Delta C = \Delta P_t / (BL \times CA) \tag{2}$$

where ΔC = change in $PM_{2.5}$ concentration ($\mu g\ m^{-3}$), ΔP_t = change in $PM_{2.5}$ mass (μg) due to the net of effect of $PM_{2.5}$ removal (flux) and resuspension from leaves, BL = boundary layer height (m) and CA = city area (m^2). Percent air quality improvement was calculated as:

$$\%\Delta = \Delta P_t / (\Delta P_t + P_a) \tag{3}$$

where P_a = $PM_{2.5}$ mass in the atmosphere (μg), which equals measured concentration ($\mu g\ m^{-3}$) × BL × CA.

2.4. Health incidence effects and monetary value of $PM_{2.5}$ removal

For the 10 U.S. cities, the U.S. EPA's BenMAP program was used to estimate the incidence of adverse health effects (i.e., mortality and morbidity) and associated monetary value that result from changes in $PM_{2.5}$ concentrations. BenMAP is a Windows-based computer program that uses Geographic Information System (GIS) based data to estimate the health impacts and economic value when populations experience changes in air quality (U.S. EPA, 2012a). The model uses air quality grids to determine the change in pollution concentration, concentration-response functions to estimate the change in adverse health effects, and valuation functions to calculate the associated economic value (Table 4). BenMAP was used to obtain incidence and value results for each county within which the 10 cities reside.

The air quality grids used for this analysis were for baseline (2000) and control (2006) years that had the greatest change in pollution concentration based on national pollution trends (http://www.epa.gov/airtrends/index.html). The pollution concentration for the grids was interpolated from existing pollution data sets from EPA pollutant monitors using Voronoi neighborhood averaging.

Several functions were used to estimate incidence and value for the following common health effects of $PM_{2.5}$: acute bronchitis, acute myocardial infarction, acute respiratory symptoms, asthma exacerbation, chronic bronchitis, emergency room visits, hospital admissions – cardiovascular or respiratory, lower respiratory symptoms, mortality, upper respiratory symptoms, and work loss days. The concentration-response functions that were used for the $PM_{2.5}$ analysis (Table 4) have several inputs including air quality metrics (e.g., 24-h mean) and age of the population (e.g., 18–64 years old, 65–99 years old).

The model was run using population statistics from the U.S. Census 2010 county dataset using an economic forecasting model described in the BenMAP user manual (Abt Associates, 2010). BenMAP configures Census block populations into grid cell level data and the calculation is at grid cell level. BenMap data were then aggregated to the county level. The health effects categories potentially had multiple estimates corresponding to different air quality metrics and age groups. Different age groups were represented because the concentration-response functions are age specific and incidence rate can vary across different age groups. Multiple estimates were pooled by either averaging the estimates using the random/fixed effects method or summing the estimates depending on which process was appropriate. In the end, a final estimate was produced to cover all possible metrics and age groups within a health category. For example, equations for 0–17, 18–64, and 65–99 age groups were summed to produce an estimate for 0–99 age group. More details on the BenMAP model are found in the literature (Davidson et al., 2007; Abt Associates, 2010; U.S. EPA, 2012a).

To estimate each individual health category incidence and dollar value effect at the city scale, the county estimates were divided by the county population by age group and change in pollution concentration to produce an estimate of number of incidences or dollar value per person per age group per change in $\mu g\ m^{-3}$, similar to the procedure used in U.S. EPA (2012b). This value was then multiplied by the city population per age group and change in $PM_{2.5}$ concentration due to trees in the city to estimate the tree effects on incidence and value for each health category. The dollar values for all health categories were summed to determine the total value of $PM_{2.5}$ effects from trees in each city.

3. Results

Total amount of $PM_{2.5}$ removal annually by trees varied from 4.7 tonnes in Syracuse to 64.5 tonnes in Atlanta, with values varying from \$1.1 million in Syracuse to \$60.1 million in New York City (Table 5). Most of these values were dominated by the effects of reducing human mortality (Table 6). The average value per mortality incidence was \$7.8 million. Mortality reductions were typically around 1 person yr^{-1} per city, but were as high as 7.6 people yr^{-1} in New York City. The net removal amounts per square meter of canopy cover varied from 0.13 $g\ m^{-2}\ yr^{-1}$ in Los Angeles to 0.36 $g\ m^{-2}\ yr^{-1}$ in Atlanta. The average annual percent air quality improvement ranged between 0.05% in San Francisco and 0.24% in Atlanta (Table 5).

The average health benefits value per hectare of tree cover was about \$1 600, but varied from \$500 in Atlanta and Minneapolis to \$3800 in New York (Table 5). The value per tonne of $PM_{2.5}$ averaged \$682 000, but varied from \$142 000 in Atlanta to \$1 610 000 in New York. The health benefits value per reduction of 1 $\mu g\ m^{-3}$ also varied from \$122 million in Syracuse to \$6.2 billion in New York, with an overall average of \$1.6 billion.

The interactions among variable V_d, resuspension, and precipitation can be seen in an hourly graph of total accumulation by tree canopies, in which removal of $PM_{2.5}$ by trees occurs during precipitation events when particles on leaves are washed off and transferred to the soil. Total accumulation stabilizes around 3500 $\mu g\ m^{-2}$ of tree cover among the cities with variations up (net removal) and down (net resuspension) hourly (Fig. 1). Average hourly cumulative flux in the cities ranged between 2100 $\mu g\ m^{-2}$ of tree cover in Philadelphia to 5700 $\mu g\ m^{-2}$ of tree cover in San Francisco. Average reduction in $PM_{2.5}$ concentrations ranged between 0.006 $\mu g\ m^{-3}$ in Philadelphia and San Francisco to 0.03 $\mu g\ m^{-3}$ in Atlanta (Table 5). Of all the particles intercepted by leaves, on average 34.0 percent were resuspended, with percent resuspension varying from 26.7 percent in Syracuse to 42.6 percent in San Francisco.

Table 4
Concentration-response functions used for $PM_{2.5}$ analyses. Daily 24-h mean concentrations data were aggregated by seasonal metric. Valuation procedure for health effects are also noted.

Health effect	Concentration response function reference	Seasonal metric	Start age	End age
Acute Bronchitis[a]	Dockery et al., 1996	Quarterly	8	12
Acute myocardial infarction[b]				
Acute myocardial infarction, nonfatal	Peters et al., 2001	Annual	18	99
	Pope et al., 2006	Annual	0	99
	Sullivan et al., 2005	Annual	0	99
	Zanobetti and Schwartz, 2006	Annual	0	99
	Zanobetti et al., 2009	Annual	0	99
Acute respiratory symptoms[a]				
Minor restricted activity days	Ostro and Rothschild, 1989	Annual	18	64
Asthma exacerbation[a]				
Asthma exacerbation, cough	Mar et al., 2004	Annual	6	18
Asthma exacerbation, shortness of breath	Mar et al., 2004	Annual	6	18
Asthma exacerbation, wheeze	Ostro et al., 2001	Annual	6	18
Chronic bronchitis[a,b]	Abbey et al., 1995	Quarterly	27	99
Emergency room visits, respiratory[b]				
Emergency room visits, asthma	Mar et al., 2010	Annual	0	99
	Norris et al., 1999	Annual	0	17
	Slaughter et al., 2005	Annual	0	99
Hospital admissions, cardiovascular[b]				
HA, all cardiovascular (less myocardial infarctions)	Bell et al., 2008	Annual	65	99
	Moolgavkar, 2000	Annual	18	64
	Moolgavkar, 2003	Annual	65	99
	Peng et al., 2008	Annual	65	99
	Peng et al., 2009	Annual	65	99
	Zanobetti et al., 2009	Annual	65	99
Hospital admissions, respiratory[b]				
HA, all respiratory	Zanobetti et al., 2009	Annual	65	99
Lower respiratory symptoms[a]	Schwartz and Neas, 2000	Annual	7	14
Mortality[c]				
Mortality, all cause	Laden et al., 2006	Quarterly	25	99
	Woodruff et al., 1997	Quarterly	0	1
	Woodruff et al., 2006	Quarterly	0	1
Upper respiratory symptoms[a]	Pope et al., 1991	Quarterly	9	11
Work loss days[d]	Ostro, 1987	Annual	18	64

Valuation procedure (U.S. EPA, 2012a).
[a] Willingness to pay.
[b] Cost of illness.
[c] Value of statistical life.
[d] Median daily wage.

4. Discussion

The removal of $PM_{2.5}$ by urban trees is substantially lower than for PM_{10} (Nowak et al., 2006a), but the health implications and values are much higher. The value of $PM_{2.5}$ removal in the cities ranged from \$1.1 million yr^{-1} (Syracuse) to 60.1 million yr^{-1} (New York). The annual values per tonne removed ranged between \$142 000 (Atlanta) and \$1.6 million (New York). These values are substantially higher than value estimates for PM_{10} (\$4500 t^{-1}), which are based on median national externality values (Murray et al., 1994). Most of this $PM_{2.5}$ removal value is derived from the reduction in human mortality due to reduced $PM_{2.5}$ concentrations. Reduction in human mortality ranged from 1 person per 365 000 people in Atlanta to 1 person per 1.35 million people in San Francisco (average = 1 person per 990 000 people). Overall, the greatest effect of trees on reducing health impacts of $PM_{2.5}$ occurred in New York due to its relatively large human population and moderately high removal rate and reduction in concentration. The greatest

Table 5
Estimated removal of $PM_{2.5}$ by trees and associated value in several U.S. cities.

City	Total ($t\ yr^{-1}$)	Range ($t\ yr^{-1}$)	Value (\$ yr^{-1})	Effect[a]: $m^{-2}\ yr^{-1}$		ΔC[b] ($\mu g\ m^{-3}$)	AQ[c] (%)
				(g)	(\$)		
Atlanta, GA	64.5	(8.5–140.4)	9 170 000	0.36	0.05	0.030	0.24
Baltimore, MD	14.0	(1.8–29.5)	7 780 000	0.24	0.13	0.010	0.09
Boston, MA	12.7	(2.0–35.6)	9 360 000	0.32	0.23	0.020	0.19
Chicago, IL	27.7	(4.0–68.1)	25 860 000	0.26	0.24	0.011	0.09
Los Angeles, CA	32.2	(4.2–70.3)	23 650 000	0.13	0.09	0.009	0.07
Minneapolis, MN	12.0	(1.6–28.2)	2 610 000	0.23	0.05	0.010	0.08
New York, NY	37.4	(5.1–97.2)	60 130 000	0.24	0.38	0.010	0.09
Philadelphia, PA	12.3	(1.6–28.1)	9 880 000	0.17	0.14	0.006	0.08
San Francisco, CA	5.5	(0.8–14.4)	4 720 000	0.29	0.25	0.006	0.05
Syracuse, NY	4.7	(0.6–10.8)	1 100 000	0.27	0.06	0.009	0.10

[a] Average effects per square meter of tree cover per year: removal in grams and dollar value.
[b] Average annual reduction in hourly concentration.
[c] Average percent air quality improvement.

Table 6
Reduction in number of incidences and associated dollar value for various health effects due to $PM_{2.5}$ reduction from trees.

Health effect[a]	No.	Value	No.	Value	No.	Value	No.	Value	No.	Value
	Atlanta, GA		Baltimore, MD		Boston, MA		Chicago, IL		Los Angeles, CA	
Acute bronchitis	0.6	60	0.4	30	0.5	50	1.8	160	2.1	180
Acute myocardial infarction	0.3	26 300	0.2	14 600	0.3	28 400	0.9	78 800	0.6	49 300
Acute respiratory symptoms	488.7	47 900	240.9	23 600	502.5	49 200	1125.2	110 300	1263.6	123 900
Asthma exacerbation	243.8	19 800	138.3	11 200	243.0	19 800	770.0	62 600	936.4	76 100
Chronic bronchitis	0.4	104 000	0.2	53 000	0.3	96 000	0.9	247 000	1.0	285 000
Emergency room visits	0.4	180	0.9	390	0.4	190	1.2	510	1.1	470
Hospital admissions, cardiovascular	0.2	7700	0.2	6200	0.2	6300	0.5	17 400	0.3	12 700
Hospital admissions, respiratory	0.1	4400	0.1	2300	0.1	4600	0.4	13 800	0.3	9000
Lower respiratory symptoms	7.2	400	4.4	200	6.5	300	22.9	1200	25.5	1300
Mortality	1.2	8 940 000	1.0	7 670 000	1.2	9 140 000	3.2	25 300 000	3.0	23 000 000
Upper respiratory symptoms	6.4	300	3.7	200	5.2	200	18.3	800	21.0	900
Work loss days	84.8	16 300	40.8	6000	87.5	15 300	192.1	35 000	217.4	37 000
Total	na	9 170 000	na	7 780 000	na	9 360 000	na	25 900 000	na	23 600 000
	Minneapolis, MN		New York, NY		Philadelphia, PA		San Francisco, CA		Syracuse, NY	
Acute bronchitis	0.2	20	4.5	400	0.5	50	0.2	20	0.1	10
Acute myocardial infarction	0.1	5800	1.4	129 300	0.2	22 400	0.1	7400	0.0	2400
Acute respiratory symptoms	146.8	14 400	2930.9	287 300	313.8	30 800	207.3	20 300	49.5	4900
Asthma exacerbation	80.9	6600	1919.3	156 000	205.8	16 700	77.2	6300	37.7	3100
Chronic bronchitis	0.1	29 400	2.4	682 000	0.3	71 500	0.2	51 600	0.0	9600
Emergency room visits	0.1	40	8.0	3300	0.4	160	0.1	30	0.0	20
Hospital admissions, cardiovascular	0.0	1200	1.2	46 200	0.1	5400	0.1	2000	0.0	500
Hospital admissions, respiratory	0.0	600	0.7	22 700	0.1	3000	0.0	1500	0.0	300
Lower respiratory symptoms	2.2	100	55.7	2900	6.1	300	2.0	100	1.0	50
Mortality	0.3	2 550 000	7.6	58 700 000	1.2	9 720 000	0.6	4 620 000	0.1	1 080 000
Upper respiratory symptoms	1.9	100	45.0	2000	5.1	200	1.7	100	0.8	40
Work loss days	25.0	4800	504.0	92 100	53.7	8500	36.0	7900	8.3	1400
Total	na	2 610 000	na	60 100 000	na	9 880 000	na	4 720 000	na	1 100 000

[a] Incidence values of 0.0 indicate a value of less than 0.05.

overall removal by trees was in Atlanta due to its relatively high percent tree cover and $PM_{2.5}$ concentrations (12.6 $\mu g\ m^{-3}$).

The net removal rates per square meter of tree cover varied among cities between 0.36 $g\ m^{-2}\ yr^{-1}$ (Atlanta) and 0.13 $g\ m^{-2}\ yr^{-1}$ (Los Angeles), with Los Angeles having the highest $PM_{2.5}$ concentrations (13.8 $\mu g\ m^{-3}$), but the lowest amount (392 mm yr^{-1}) and frequency of rainfall (247 h yr^{-1}). The average amount and frequency of precipitation among the cities were 644 mm yr^{-1} and 394 h yr^{-1} respectively. On average, about 24 $g\ m^{-2}\ yr^{-1}$ of $PM_{2.5}$ removal equated to 1 $\mu g\ m^{-3}$ reduction in $PM_{2.5}$ concentrations, but results varied from 12 $g\ m^{-2}\ yr^{-1}$ in Atlanta to 45 $g\ m^{-2}\ yr^{-1}$ in San Francisco.

Removal rates per unit canopy and effects on local $PM_{2.5}$ concentration varies among cities based on amount of tree cover – increased cover increases removal, pollution concentration – increased concentration increases removal, length of growing season and percent evergreen leaf area – longer growing season increases removal by deciduous species, and meteorological conditions. The meteorological conditions (precipitation, wind speed and boundary layer heights) interact to affect $PM_{2.5}$ removal and concentrations. Increased precipitation tends to increase tree removal via the washing of particles from the leaf surfaces. The low removal rate in Los Angeles is likely due, in part, to limited precipitation. Wind speeds affect resuspension and boundary layer heights. Greater resuspension reduces the overall removal rate by trees; increased boundary layer heights reduce the overall percent impact of trees on pollutant concentrations, but also reduce $PM_{2.5}$ concentrations. Maximum percent air quality improvements tended to occur under windy conditions (increased V_d) with low boundary layer heights (increased impact of removal on pollutant concentration) and relatively clean leaves (low amount of particles to be resuspended).

When resuspension is greater than the removal rate, trees can increase local concentrations due to previously deposited particles reentering the atmosphere (Fig. 1). Although $PM_{2.5}$ removal by trees in the analyzed cities lead to reduced overall particulate concentrations, it is possible that even though trees remove particulate matter, they could increase overall particulate concentrations. This overall increase in concentrations could occur depending upon the meteorological conditions when particles are deposited and resuspended. If particulate removal occurs under high boundary layer conditions, but resuspension occurs mostly under low boundary layer conditions, the amount of removal would cause a lower reduction in concentrations than the increased concentration effect due to resuspension. Thus timing of removal relative to boundary layer heights has a substantial impact on overall concentration changes. Overall impacts and dollar values also varied based on population density and composition, along with the tree effects on concentration.

Though there are various limitations to these estimates, the results indicate a first-order approximation of the magnitude of tree effects on $PM_{2.5}$ concentrations. Limitations of the analysis include: a) assumption that all particles are removed from leaves by precipitation events that cover the entire leaf area as some particles may remain on leaves or some particles may be removed in light rain events ($Pe < Ps$), b) there is no assumed interaction with water on leaves after precipitation events, c) some precipitation events may be in the form of snow, which may limit removal; however these events are relatively infrequent and limited to only evergreen trees removal that accounts for only about 18% of the total leaf area among the cities, d) measured deposition velocities used to calculate the average V_d are based on varying particle sizes with some particles greater than 2.5 μm (up to 3.8 μm) and particle size affects the deposition velocity (e.g., Gallagher et al., 1997) – it is assumed the measured deposition velocities represent the average for the particle distribution in the atmosphere, e) wind speeds and therefore V_d and resuspension can vary locally, though an average wind speed is used to represent the entire city, f) tree volatile

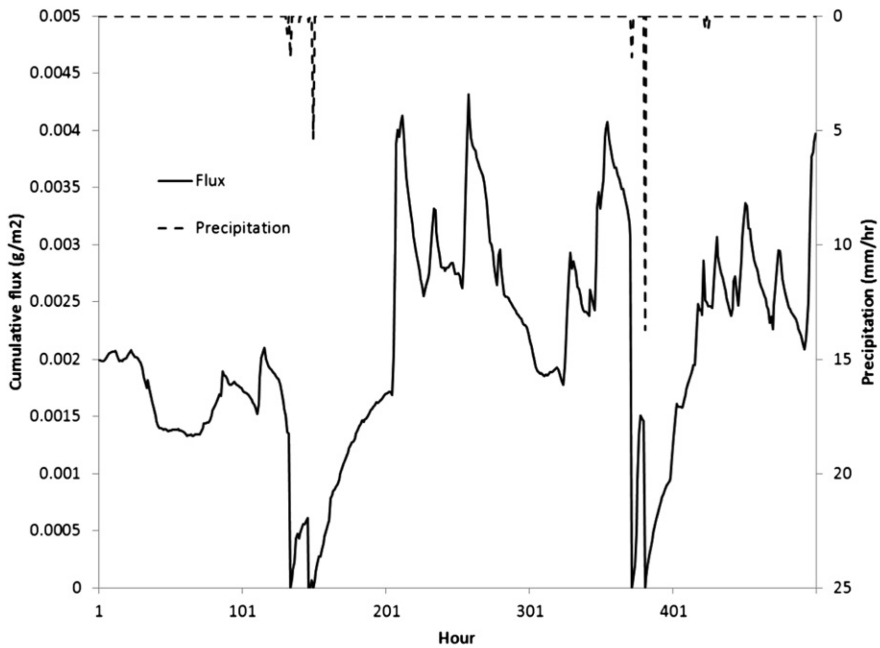

Fig. 1. Cumulative hourly flux of $PM_{2.5}$ per square meter of tree cover in New York City starting at 1 am on July 8, 2010. Increasing flux values indicate hourly removal, decreasing values indicate a net resuspension. Precipitation periods could remove particles from leaves and transport them to the ground. This transported amount was calculated as a net removal from trees.

organic compound emissions and their potential contribution to $PM_{2.5}$ concentrations are not considered (e.g., Hodan and Barnard., 2004), g) V_d is assumed equal for all leaves within a tree canopy; however interior leaves are likely to have lower wind speeds and therefore lower V_d and resuspension rates, but most leaf surface area is not within the interior of the tree canopy, h) rainfall intensity is not considered and may affect washoff rates; i) use of 24-h average concentration data to estimate the hourly concentrations during the day as concentrations will vary locally (e.g., likely higher concentrations near roadways) and temporally, and j) the boundary layer is assumed to be well-mixed (unstable), which will likely lead to conservative estimates of concentration reductions during stable conditions. Future research and more detailed modeling may help overcome these current limitations.

Despite the limitations, there are advantages to these modeling estimates, which include: a) use of locally measured tree, weather and pollution data to assess $PM_{2.5}$ effects, b) use of measured V_d and resuspension rates to estimate removal and resuspension, and c) interaction of V_d and resuspension with local hourly wind speeds. The interactions and variations of $PM_{2.5}$ removal and resuspension with wind speed (Fig. 1) illustrate how the $PM_{2.5}$ flux can vary hourly, yielding positive and negative concentration changes throughout a day. Average wind speed in the cities was 3.7 m s^{-1} with a maximum speed of 20.6 m s^{-1}. The average deposition velocity to tree canopies was 0.65 cm s^{-1}, which is above the typical range listed for particles less than 2 μm (<0.5 cm s^{-1}; Lovett, 1994). However, the average V_d estimate for $PM_{2.5}$ (0.65 cm s^{-1}) does not include resuspension, which considering a 34 percent average resuspension rate, would lower the V_d estimate to about 0.43 cm s^{-1}.

In this simulation, the movement of the particles from the tree leaves to the soil environment occurs via precipitation. The greater the amount of particles on a leaf just prior to a precipitation event, the greater the overall effect of the trees on removal of $PM_{2.5}$ from the atmosphere. Between rainfall events, the amount of particles retained on tree canopies averages 3500 μg m^{-2}, but fluctuates through time based on wind speed. Frequent rainfall would likely maximize tree effectiveness on removing particles from the atmosphere and transferring them to the soil environment. However, not all particles will be resuspended or washed off with precipitation, some particle will adhere to waxy leaf surfaces and be transferred to the soil via leaf drop and leaf decomposition (e.g., Joureava et al., 2002).

This citywide modeling focuses on broad-scale estimates of tree effects on $PM_{2.5}$. Local-scale effects likely differ depending upon vegetation designs. At the local scale, $PM_{2.5}$ concentrations can be increased if trees: a) trap the particles beneath tree canopies near emission sources (e.g., along road ways, Gromke and Ruck (2009)), b) limit dispersion by reducing wind speeds (e.g., Vos et al., 2012) and/or c) lower boundary layer heights by reducing wind speeds (e.g., Nowak et al., 2000). Under stable atmospheric conditions (limited mixing), particle removal by trees could lead to increased reductions in pollution concentrations at the ground level. In addition, if some local sources of $PM_{2.5}$ come from wind-borne soils, tree cover can reduce these particles by reducing wind speeds. Large stands of trees can also reduce pollutant concentrations in the interior of the stand due to increased distance from emission sources and increased dry deposition (e.g., Dasch, 1987; Cavanagh et al., 2009). Thus, local scale design with trees and forests are important for reducing local scale $PM_{2.5}$ concentrations. More research is needed on these local scale issues as local scale designs with trees need to consider vegetation configuration and source–sink relationships to maximize tree effects on reducing $PM_{2.5}$ concentrations and minimizing human exposure to $PM_{2.5}$.

In addition to $PM_{2.5}$ removal, tree also remove other air pollutants (e.g., ozone, sulfur dioxide, nitrogen dioxide; Nowak et al., 2006a) and emit volatile organic compounds that can contribute to ozone formation (e.g., Chameides et al., 1988). Managers need to understand the magnitude of tree effects on air pollution to better manage urban vegetation to improve air quality. To aid in assisting urban forest planners and managers, a free model (i-Tree: www.itreetools.org) has been developed to aid cities in quantifying pollution removal by trees and other environmental services. Improving air quality with vegetation in cities can lead to improved human health and substantial health care savings.

5. Conclusions

Modeling of broad-scale effects of pollution removal by trees on $PM_{2.5}$ concentrations and human health reveal that trees can produce substantial health improvements and values in cities. More research is needed to improve these estimates and on local scale effects of vegetation designs. These local scale effects include potentially increasing local concentrations due to limiting pollution dispersion or reducing concentrations through enhanced deposition and reducing the production of particulate matter. Urban forest designs that consider source–sink relationships of $PM_{2.5}$ and other pollutants can be developed to reduce $PM_{2.5}$ concentrations and minimize human exposure to $PM_{2.5}$ in cities across the globe.

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