Urban Forestry & Urban Greening 29 (2018) 40–48

Contents lists available at ScienceDirect

Urban Forestry & Urban Greening

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

Original article

Air pollution removal by urban forests in Canada and its effect on air quality and human health

David J. Nowak[a,*], Satoshi Hirabayashi[b], Marlene Doyle[c], Mark McGovern[c], Jon Pasher[c]

[a] USDA Forest Service, 5 Moon Library, SUNY-ESF, Syracuse, NY 13210 USA
[b] The Davey Institute, 5 Moon Library, SUNY-ESF, Syracuse, NY 13210 USA
[c] Environment and Climate Change Canada, National Wildlife Research Centre, Ottawa, ON K1S5B6 Canada

ARTICLE INFO

Keywords:
Air pollution removal
Air quality
Ecosystem services
Human mortality
Urban forests

ABSTRACT

Urban trees perform a number of ecosystem services including air pollution removal, carbon sequestration, cooling air temperatures and providing aesthetic beauty to the urban landscape. Trees remove air pollution by intercepting particulate matter on plant surfaces and absorbing gaseous pollutants through the leaf stomata. Computer simulations with local environmental data reveal that trees in 86 Canadian cities removed 16,500 tonnes (t) of air pollution in 2010 (range: 7500–21,100 t), with human health effects valued at 227.2 million Canadian dollars (range: $52.5–402.6 million). Annual pollution removal varied among cities and ranged up to 1740 t in Vancouver, British Columbia. Overall health impacts included the avoidance of 30 incidences of human mortality (range: 7–54) and 22,000 incidences of acute respiratory symptoms (range: 7900–31,100) across these cities.

1. Introduction

Air pollution is a significant problem globally that affects human health and well-being, ecosystem health, crops, climate, visibility and man-made materials. Common air pollutants include carbon monoxide (CO), nitrogen dioxide (NO_2), ozone (O_3), sulfur dioxide (SO_2), and particulate matter less than 2.5 µm ($PM_{2.5}$) and 10 µm (PM_{10}) in aerodynamic diameter. In Canada, air quality standards have been developed for $PM_{2.5}$ and O_3, and work has begun to develop standards for NO_2 and SO_2 (Canadian Council of Ministers of the Environment (CCME), 2016). Health effects related to air pollution include impacts on pulmonary, cardiac, vascular, and neurological systems (e.g., Pope et al., 2002). Outdoor air pollution, mostly $PM_{2.5}$, is estimated to lead to 3.3 million premature deaths per year worldwide, mainly in Asia (Lelieveld et al., 2015). In Canada it is estimated that there are 21,000 premature deaths attributable to air pollution each year (Canadian Medical Association, 2008).

Trees and forests affect air quality through the direct removal of air pollutants, altering local microclimates and building energy use, and through the emission of pollen, which affects allergies (e.g., Ogren, 2000) and volatile organic compounds (VOCs), which can contribute to O_3 and $PM_{2.5}$ formation (e.g., Chameides et al., 1988). However, integrative studies have revealed that trees, particularly low VOC emitting species, can be a viable strategy to help reduce urban O_3 levels (e.g., Taha, 1996; Nowak et al., 2000).

Trees remove gaseous air pollution primarily by uptake via leaf stomata, though some gases are removed by the plant surface. Once inside the leaf, gases diffuse into intercellular spaces and may be absorbed by water films to form acids or react with inner-leaf surfaces. Trees directly affect particulate matter in the atmosphere by intercepting particles, emitting particles (e.g., pollen) and resuspension of particles captured on the plant surface. Some particles can be absorbed into the tree, though most intercepted particles are retained on the plant surface. The intercepted particles often are resuspended to the atmosphere, washed off by rain, or dropped to the ground with leaf and twig fall (Smith, 1990). During dry periods, particles are constantly intercepted and resuspended, in part, dependent upon wind speed. The accumulation of particles on the leaves can affect photosynthesis (e.g., Darley, 1971) and therefore potentially affect pollution removal by trees. During precipitation, particles can be washed off and either dissolved or transferred to the soil. Consequently, vegetation is only a temporary retention site for many atmospheric particles, where particles are eventually moved back to the atmosphere or moved to the soil. Pollution removal by urban trees in the United States has been estimated at 651,000 tonnes (t) per year (Nowak et al., 2014).

While various studies have estimated pollution removal by trees (e.g., Nowak et al., 2006a, 2014, McDonald et al., 2007, Tallis et al., 2011), most studies on pollution removal do not directly link the removal with improved human health effects and associated health

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

https://doi.org/10.1016/j.ufug.2017.10.019
Received 12 April 2017; Received in revised form 29 September 2017; Accepted 26 October 2017
Available online 04 November 2017
1618-8667/ Published by Elsevier GmbH.

values. A few studies that have linked removal and health effects include a study in London, England where a 10 × 10 km grid with 25% tree cover was estimated to remove 90.4 t of PM_{10} annually, which equated to the avoidance of 2 deaths and 2 hospital admissions per year (Tiwary et al., 2009). In addition, Nowak et al. (2013a) reported that the total amount of $PM_{2.5}$ removed annually by trees in 10 U.S. cities in 2010 varied from 4.7 t in Syracuse to 64.5 t in Atlanta. Estimates of the annual monetary value of human health effects associated with $PM_{2.5}$ removal in these same cities (e.g., changes in mortality, hospital admissions, respiratory symptoms) ranged from $1.1 million in Syracuse to $60.1 million in New York City. Mortality avoided was typically around 1 person per year per city, but was as high as 7.6 people per year in New York City. Trees and forests in the conterminous United States removed 17.4 million tonnes (t) of air pollution in 2010 with human health effects valued at 6.8 billion U.S. dollars (Nowak et al., 2014). Most of the pollution removal occurred in rural areas, while most of the health impacts and values were within urban areas. Health impacts included the avoidance of more than 850 incidences of human mortality and 670,000 incidences of acute respiratory symptoms.

As people and trees exist throughout a landscape in varying densities, not only will pollution removal and its effects on local pollution concentrations vary, but so will the associated human health impacts and values derived from this ecosystem service. While studies have been conducted on individual Canadian cities (e.g., McNeil and Vava, 2006, TRCA, 2011, City of Edmonton, 2012, Nowak et al., 2013b), a consistent assessment across all Canadians cities of removal of key air pollutants by urban trees has not yet been completed. Such an analysis will allow for a greater understanding of the services provided by green urban infrastructure and set a baseline for investigating changes in service provision over time. The objectives of this paper are to estimate the amount of air pollution (CO, NO_2, O_3, $PM_{2.5}$, SO_2) removed by trees within 86 Canadian cities in 2010 and its associated monetary value and impact on human health.

2. Methods

To estimate avoided health impacts and associated dollar benefits of air pollution removal by trees in 86 Canadian cities (Suppl. 1) in 2010, four types of analyses were conducted to estimate: 1) the total tree cover and leaf area index on a daily basis to account for seasonal variability, 2) the hourly flux of pollutants to and from the leaves, 3) the effects of hourly pollution removal on pollutant concentration in the atmosphere, and 4) the health impacts and monetary value of the change in NO_2, O_3, $PM_{2.5}$ SO_2, and CO concentration. City areas were delimited using shape files provided by Environment and Climate Change Canada and were based on the Statistics Canada populated places boundary file (Statistics Canada, 2011a). It is important to note that the boundaries used are based on a combination of population densities, roads and other geographic data sets and are often not the same as the administrative municipal boundaries. As a result, population counts, area extents and tree coverage may differ from those reported by municipal or other agencies. To simplify presentation, only data from the 15 most populated cities are presented (Table 1), but results for all 86 cities can be found in the supplemental materials.

2.1. Tree cover and leaf area index

Percent and hectares of tree cover within each city were derived from photo-interpretation of aerial images (c. 2011) as detailed in Pasher et al. (2014) (Table 1). Maximum (mid-summer) leaf area index (LAI: m^2 leaf area per m^2 projected ground area of canopy) values were derived from the level-4 MODIS/Terra global Leaf Area Index product for the 2011 growing season (U.S. Geological Survey (USGS), 2013) based on an average of all maximum pixel values within the city. In areas where LAI values per unit of tree cover were missing or abnormally low, a midsummer LAI value of 4.9 was used based on the average LAI in urban areas (Nowak et al., 2008).

Table 1
City area (km^2), human population in 2011 (Statistics Canada, 2011b), percent tree cover (%TC) and percent evergreen cover (%EG) for the 15 most populated cities. These cities comprise over 75% of the urban population and 60% of the total urban area in Canada.

City	Province	Area	Population	%TC	%EG
Calgary	Alberta	722.8	1,095,404	9.3	3.8
Edmonton	Alberta	872.6	960,015	13.0	3.8
Gatineau	Quebec	172.3	302,728	30.6	6.0
Halifax	Nova Scotia	291.4	297,943	51.8	16.8
Hamilton	Ontario	394.8	670,580	21.6	7.3
Kitchener	Ontario	319.4	444,681	20.5	7.3
London	Ontario	225.7	366,191	20.3	7.3
Montréal	Quebec	1557.6	3,407,963	22.7	6.0
Ottawa	Ontario	389.4	933,596	26.5	6.0
Québec	Quebec	682.9	696,946	47.0	6.0
St. Catharines – Niagara	Ontario	394.4	309,319	23.9	7.3
Toronto	Ontario	1763.4	5,132,794	18.2	7.3
Vancouver	British Columbia	1206.6	2,135,201	40.0	2.1
Victoria	British Columbia	281.6	316,327	45.5	2.1
Winnipeg	Manitoba	460.1	671,551	16.5	12.3

Percent tree cover classified as evergreen was estimated based on the average percent evergreen species for the regional forest type (Table 1, Suppl. 1). LAI values were combined with percent evergreen information and local leaf-on and leaf-off (frost) dates (National Climatic Data Center (NCDC), 2005) to estimate total daily leaf surface area in each city assuming a four-week transition period centered on leaf-on and leaf-off dates for spring and autumn, respectively.

2.2. Pollution removal by trees

Hourly pollution removal or flux (F in µg m^{-2} h^{-1}) was estimated as:

$$F = V_d \times C$$

Where V_d is the deposition velocity of the pollutant to the leaf surface (m hr^{-1}) and C is pollutant concentration (µg m^{-3}) (e.g., Hicks et al., 1989). Hourly concentrations for each pollutant by city were obtained from Environment and Climate Change Canada for the year 2010 (Environment Canada, 2013). Missing pollutant data were filled in based on procedures described in Hirabayashi and Kroll (2017). The average percent missing pollution data were 9.6 percent for NO_2, 8.3 percent for SO_2, 6.9 percent for $PM_{2.5}$, 6.6 percent for CO and 5.3 percent for O_3. For PM data, if hourly data did not exist, then daily and 6-day measurements were used to represent the hourly concentration values throughout the day (i.e., the average daily value was applied to each hour of the day). If multiple monitors existed within a city for the same pollutant, the average hourly value was used. If no pollutant monitors existed within the city, the closest data monitor was assigned to represent that area. The median distance away from city center was 35 km for CO, 21 km for SO_2, 11 km for NO_2 and 7 km for O_3 and $PM_{2.5}$.

To calculate the hourly deposition velocity, local hourly weather data for 2010 from the National Climatic Data Center (National Climatic Data Center (NCDC), 2013) were used. If no weather data existed within the city, the closest monitor data was assigned to represent that area. If more than one monitor existed, the weather data closest to the geographic center of the area was used. The median distance from city center was 24 km, with 12 of the 85 cities having weather stations over 100 km away (maximum distance was 271 km from Moose Jaw).

Deposition velocities for all pollutants and resuspension rates for particulate matter were calculated using the i-Tree model (www.itreetools.org) based on methods detailed in Nowak et al. (2006a, 2013a) and Hirabayashi et al. (2011, 2012). Total removal of a pollutant in a city was calculated as the annual flux value (µg m^{-2} yr^{-1}) times total tree cover (m^2). Minimum and maximum estimates of removal were based on the typical range of published in-leaf dry deposition velocities (Lovett, 1994).

2.3. Change in pollutant concentration

To estimate percent air quality improvement due to dry deposition, hourly mixing heights from the nearest radiosonde station (National Oceanic and Atmospheric Administration (NOAA), 2013) were used in conjunction with local hourly fluxes using the i-Tree model based on methods detailed in Nowak et al. (2013a). As pollution removal by trees affects local measured pollution concentrations, this removal effect is accounted for in the calculation of percent air quality improvement (Nowak et al., 2006a).

2.4. Health incidence effects and monetary value of NO_2, O_3, $PM_{2.5}$ and SO_2 removal

The U.S. EPA's Environmental Benefits Mapping and Analysis Program (BenMAP) was used to estimate the incidence of adverse health effects (i.e., mortality and morbidity) and associated monetary value that result from changes in NO_2, O_3, $PM_{2.5}$ and SO_2 concentrations due to pollution removal by trees in the United States (Nowak et al., 2014). BenMAP is a Windows-based computer program that uses geospatial data to estimate the health impacts and monetary value when populations experience changes in air quality (Davidson et al., 2007; Abt Associates, 2010; U.S. Environmental Protection Agency (US EPA), 2012a). Extrapolation from the U.S. derived health effects were made as follows:

1) Each Canadian city was matched to the U.S. County with the closest required air quality metric. Each U.S. county has a health incidence multiplier (number of incidences per concentration change per person) based on seven annual air quality concentration metrics in BenMAP: a) daily one- hour maximum, b) daily mean for 8–10 am, c) daily mean for 6–9 am, d) daily maximum for 8 h moving average, e) daily mean for 9 am–4 pm, f) daily mean and g) quarterly mean of daily mean. The specific air quality metric for each pollutant in each city was compared against U.S. county values. The health incidence multipliers from the county with the closest concentration value to the city was used for that city. For example, the daily one- hour maximum NO_2 concentration in Halifax in 2010 was 12.67 ppb. As Perry County, Pennsylvania had the same daily one-hour maximum NO_2 concentration, the health incidence multipliers for NO_2 from Perry County were applied to Halifax. This matching process assures similar pollutant concentrations in selecting the incidence multipliers derived from BenMAP.
2) Adjust incidence multiplier to local population and tree effects. Each health incidence multiplier is specific to a population age class and health incidence (e.g., acute bronchitis, acute myocardial infarction, etc.) (U.S. Environmental Protection Agency (US EPA), 2012b). The number of people in each city age class was derived from Statistics Canada (2015). The number of incidences per concentration change per person for the city age class was multiplied by the number of people in the age class and the concentration change due to trees to produce the total number of incidences for each health effect and age class. The class values were summed to produce the total incidences and values for each health effect per pollutant.
3) Convert number of incidences to Canadian health values. Economic values due to avoided adverse health incidences are calculated based on the Air Quality Benefits Assessment Tool (AQBAT) Release 2.01 values (Judek et al., 2006). The AQBAT is a computer simulation program developed by Health Canada that is similar to BenMAP in estimating human health costs and/or benefits associated with changes in ambient air quality. AQBAT values (dollars per incidence) were derived for all BenMAP health incidences except for "Acute Myocardial Infarction", "School Loss Days" and "Work Loss Days". For "Acute Myocardial Infarction", the average ratio of Canadian values per incidence from AQBAT compared to U.S values per incidence from BenMAP (1.72) was used to convert the BenMAP derived dollar value to the Canadian dollar value. For "School Loss Days" and "Work Loss Days" the BenMAP U.S. dollar value was converted to Canadian dollars (CAD) based on the ratio of median per-capita income between the US and Canada (Phelps and Crabtree, 2013) and a currency conversion rate of $1.38 CAD per USD. The value multipliers (dollars per incidence) were applied to each incidence total to estimate the total value of the tree effects on reducing the number of health incidences through reducing pollution concentration.

2.5. Monetary value of CO removal

Pollution removal value for CO was estimated using national median externality values (Murray et al., 1994, Ottinger et al., 1990). These values in dollars per tonne were updated to 2010 values using the producer price index (U.S. Department of Labor Bureau of Labor Statistics, 2012) and a currency conversion rate of $1.38 CAD per USD (CO = $2012 CAD per tonne). Externality values can be considered the estimated cost of pollution to society that is not accounted for in the market price of the goods or services that produced the pollution. All dollar values presented in this paper will be in CAD, unless specifically noted elsewise.

3. Results

The total amount of pollution removal in the 86 cities in 2010 was 16,500 t (range: 7500 t to 21,100 t), with a human health value of $227.2 million (range: $52.5 million to $402.6 million) (Table 2). The range in values is based on the typical range of deposition velocities, but other uncertainties based on input data (e.g., tree cover, pollution concentration) and modeling of health benefits would increase the range. However, the value of these uncertainties is unknown. Removal in 2010 was greatest in Vancouver (1740 t), Toronto (1470 t) and Montréal (1400 t), while pollution removal monetary value was greatest in Montréal ($31.4 million), Toronto ($25.4 million) and Vancouver ($16.2 million) (Table 2, Figs. 1–2). The greatest amount of pollution removal was for O_3 and NO_2 (Table 1), while the greatest value associated with removal was for $PM_{2.5}$ and O_3 (Fig. 2). Most of these benefits were dominated by the effects of reducing human mortality, with a reduction of more than 30 incidences of human mortality (Tables 3–4). Other substantial health benefits include the reduction of more than 21,900 incidences of acute respiratory symptoms, 16,500 incidences of asthma exacerbation and 4500 school loss days.

Average removal per square meter of canopy cover for all pollutants in the 15 cities varied from 5.38 g/m^2/year in Hamilton to 2.14 g/m^2/year in Québec City, while values varied from $1564 ha of tree cover/year in Kitchener to $275 ha of tree cover/year in Québec City (Table 5). Overall pollution removal among all 86 cities averaged 3.72 g/m^2 year with an average value per hectare of tree cover of $511 (Table 6). The average annual percent air quality improvement due to trees varied among pollutants and ranged from a low of 0.001% for CO to a high of 0.273% SO_2 (Table 6).

4. Discussion

Pollution removal and dollar values for each pollutant will vary among cities based on local environmental and human population attributes. Overall pollution removal is related to: a) the amount of tree cover (increased tree cover leading to greater total removal), b) pollution concentration (increased concentration leading to greater downward flux and total removal), c) length of in-leaf season (increased growing season length leading to greater total removal), d) amount of precipitation (increased precipitation leading to reduced total removal via dry deposition), e) percent evergreen leaf area (increased evergreen leaf area increases pollution removal during leaf-off seasons) and f) other meteorological variables that affect tree transpiration and

Table 2
Estimated removal of pollution (tonnes) and associated value ($,CAD) due to trees in the 15 most populated Canadian cities (Results for all 86 cities are given in Supplemental Table 2). Values in parentheses indicate minimum and maximum range of estimate (no range given for carbon monoxide).

City	Unit	CO	NO_2	O_3	$PM_{2.5}$	SO_2
Calgary	$	860	27,150 (22,950–29,700)	911,540 (479,420–1,065,910)	6,297,130 (860,920–11,773,330)	1120 (730–1550)
	t	0.6	48.1 (35.8–53.9)	134.9 (55.5–171.6)	11.7 (1.5–26.3)	8.0 (4.8–12)
Edmonton	$	1970	25,050 (22,540–26,770)	795,960 (512,140–918,270)	12,836,540 (1,770,820–26,364,320)	740 (570–950)
	t	1.3	56.0 (42.4–62.7)	218.4 (95.3–278.4)	18.6 (2.5–39.5)	10.3 (6.3–15.8)
Gatineau	$	2960	7650 (5450–8720)	599,320 (289,720–688,460)	3,079,270 (399,130–5,626,150)	110 (70–140)
	t	2.0	19.6 (12.5–22.3)	150.6 (65.3–181.7)	10.0 (1.3–21.1)	1.3 (0.8–1.8)
Halifax	$	4480	16,610 (11,610–17,900)	2,453,610 (1,258,830–2,668,610)	5,765,560 (952,340–15,579,330)	2150 (1120–2900)
	t	3.0	48.1 (31.6–52.6)	433.8 (217–493.8)	29.1 (4.9–84)	33.0 (16–47.3)
Hamilton	$	1780	32,840 (22,290–38,980)	2,807,390 (1,324,610–3,333,520)	5,818,520 (814,850–11,728,570)	6570 (3630–9030)
	t	1.2	60.8 (36.5–70.3)	332.1 (126.5–399.2)	17.2 (2.2–40.6)	47.4 (24.7–67)
Kitchener	$	1420	13,350 (9540–15,160)	1,536,120 (766,510–1,817,220)	8,696,050 (1,268,140–15,987,680)	610 (330–830)
	t	1.0	29.6 (18.7–33.4)	251.4 (101.6–301.2)	11.5 (1.5–27.2)	7.2 (3.6–10.5)
London	$	120	10,830 (7920–12,790)	1,169,180 (520,130–1,375,870)	2,502,520 (355,690–4,860,290)	660 (400–850)
	t	0.1	22.5 (14.7–26.7)	163.7 (63.8–202.2)	6.6 (0.9–15.7)	6.1 (3.4–8.5)
Montréal	$	30,190	80,530 (55,430–91,300)	5,057,750 (2,247,880–5,714,590)	26,195,850 (3,653,470–50,795,570)	7080 (4690–9500)
	t	20.3	236.8 (141.3–270)	986.9 (397.1–1193.6)	85.4 (10.9–187.1)	70.9 (42.7–99.8)
Ottawa	$	11,630	19,460 (14,160–21,850)	2,340,990 (1,160,520–2,671,820)	4,301,680 (570,310–8,081,040)	240 (180–270)
	t	7.8	37.6 (24.5–42.2)	305.0 (135.5–367.2)	14.0 (1.8–29.4)	1.7 (1.3–2.1)
Québec	$	15,030	27,890 (22,850–30,570)	2,556,250 (1,616,920–2,855,690)	6,235,940 (891,270–12,696,840)	1180 (920–1450)
	t	10.1	93.7 (70–105.2)	518.3 (284.2–628.3)	47.4 (6.7–110.6)	16.6 (11.7–21.7)
St. Catharines − Niagara	$	570	11,660 (8440–13,800)	1,070,900 (505,710–1,212,010)	3,353,650 (460,900–6,627,880)	770 (500–990)
	t	0.4	55.7 (34.8–64)	394.3 (156–469.6)	25.8 (3.4–63.7)	15.7 (8.8–21.7)
Toronto	$	7290	239,840 (160,820–284,490)	12,162,690 (5,246,120–14,181,110)	12,955,730 (1,880,140–23,655,470)	20,900 (14,500–27,470)
	t	4.9	304.1 (185.1–358.7)	1,005.0 (413–1254.8)	54.2 (7.3–106.8)	104.0 (68.8–146.2)
Vancouver	$	2960	167,190 (92,360–193,540)	7,293,320 (2,242,860–8,449,470)	8,744,100 (1,198,910–18,066,540)	17,010 (8450–25,200)
	t	2.0	405.1 (198.1–476.6)	1,178.9 (362.7–1468.9)	22.1 (3–50.1)	136.5 (64.8–212.1)
Victoria	$	770	24,520 (13,610–29,310)	1,187,090 (487,200–1,374,730)	3,066,150 (424,160–5,641,520)	2180 (1210–3680)
	t	0.5	98.3 (45.2–119.3)	309.4 (110.1–392.5)	12.5 (1.6–25.9)	29.4 (14.7–50.4)
Winnipeg	$	3700	17,520 (15,690–18,390)	1,105,230 (773,350–1,222,250)	5,994,950 (762,130–11,895,510)	410 (380–440)
	t	2.5	30.9 (25.5–33)	198.2 (108.3–230.8)	10.8 (1.4–24.6)	2.2 (2–2.4)
All Cities	$	166,137	915,237 (618,661–1,057,413)	61,179,792 (28,439,984–70,611,265)	164,903,803 (23,261,859–330,616,408)	83,601 (50,840–115,242)
	t	112	2434 (1462–2832)	12,370 (5317–15,234)	665 (89–1519)	939 (528–1374)

deposition velocities (factors leading to increased deposition velocities would lead to greater downward flux and total removal). All of these factors combine to affect total pollution removal and the standard pollution removal rate per unit tree cover.

Health effects and dollar values are affected by the amount of pollution removed, but also: a) local boundary layer (atmospheric mixing) heights and pollution concentrations, which affect how much pollution concentrations are altered by trees and b) local population totals (lower human populations mean fewer people receive the associated health benefits). Thus cities can have high pollution removal but low health values if few people receive the health benefits of reduced pollution concentrations.

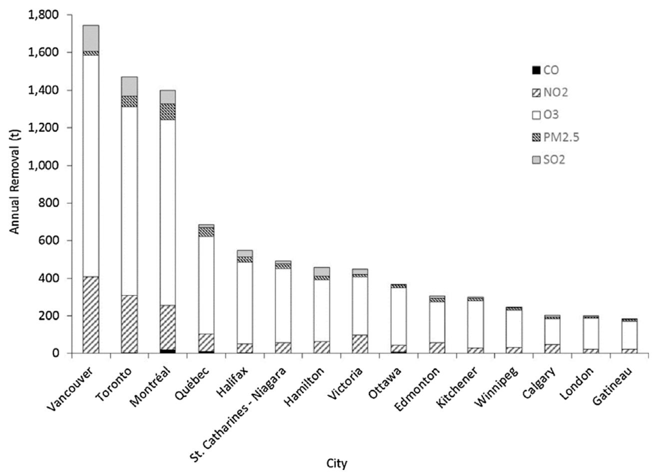

Fig. 1. Urban forest air pollution removal (tonnes/year) by pollutant in the 15 most populated cities.

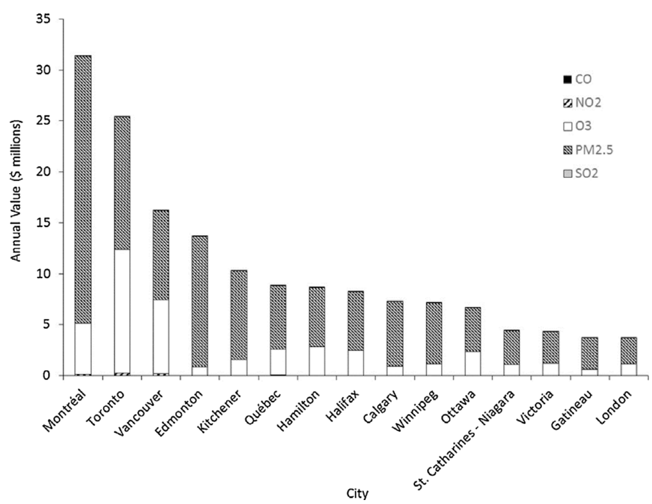

Fig. 2. Urban forest air pollution value (CAD/year) by pollutant in the 15 most populated cities.

In the United States, average pollution removal per square meter of tree cover in urban areas in 2010 was 6.73 g/m^2/year, but this estimate did not include CO removal (Nowak et al., 2014). In the 86 Canadian cities, the average removal rate was 3.72 g/m^2/year. The Canadian removal rate is lower due to reasons detailed above, but most of this difference is likely due to lower pollution concentrations and shorter in-leaf seasons than in the United States. Pollution removal values in the more heavily-populated Canadian cities (Montréal: $31.4 million [human population = 3.4 million], Toronto: $25.4 million [pop. = 5.1 million], Vancouver: $16.2 million [pop. = 2.1 million] are comparable to values found in larger U.S. cities (e.g., Philadelphia, PA: $19 million USD or $26.2 million CAD; [pop. = 1.5 million], Nowak et al., 2016).

Though city trees remove tonnes of air pollutants annually, average annual percent air quality improvement in cities is less than one percent (Table 5), which is comparable to values in Nowak et al. (2006a). Maximum annual air quality improvement among the cities averaged around 0.01 percent for CO, 2 percent for NO_2, 3 percent for SO_2, 4 percent for O_3 and 15 percent for $PM_{2.5}$. These maximum effects are short-lived and tend to occur when the boundary layer height is relatively low. For $PM_{2.5}$, concentrations can increase for certain hours due to trees when previously intercepted particles are resuspended or emitted back to the atmosphere. Percent air quality improvement among pollutants is based on the amount of tree cover and local meteorological and pollution concentration conditions. Percent air quality improvement was typically greatest for sulfur dioxide, ozone, and nitrogen dioxide.

The greatest effect of urban trees on ozone, sulfur dioxide, and nitrogen dioxide is during the daytime of the in-leaf season when trees are transpiring water. Particulate matter removal occurs both day and night and throughout the year as particles are intercepted by leaf and bark surfaces. Carbon monoxide removal also occurs both day and night of the in-leaf season, but at much lower rates than for the other pollutants.

Air pollution removal is only one aspect of how urban trees affect air quality. Trees reduce air temperatures, which can lead to reduced emissions from various anthropogenic sources (e.g., Cardelino and Chameides, 1990). Trees around buildings alter building energy use (e.g., Heisler, 1986) and consequent emissions from power plants. Trees reduce wind speeds, lowering mixing heights and can therefore increase pollution concentrations (e.g., Nowak et al., 2006a). Trees also emit pollen, which affects allergies (e.g., Ogren, 2000), and volatile organic compounds (VOCs) that are precursor chemicals to O_3 and $PM_{2.5}$ formation (e.g., Chameides et al., 1988; Hodan and Barnard, 2004). Ozone studies that integrate temperature, deposition and emission effects of trees are revealing that urban trees can reduce ozone concentrations (Cardelino and Chameides, 1990; Taha, 1996; Nowak et al., 2000). Under stable atmospheric conditions (limited mixing), pollution removal by trees could lead to a greater reduction in pollution concentrations at the ground level. 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).

Estimates of air quality improvement due to pollution removal can underestimate the total effect of the forest on reducing ground-level pollutants because they do not account for the effect of the forest

Table 3
Cumulative number of avoided incidences and value (CAD) for health effects among the 86 Canadian cities.

Health Effects	Incidence	Range	Value	Range
Acute Bronchitis	10	(1.4–20)	4300	(600–8700)
Acute Myocardial Infarction	5.8	(0.8–12)	894,400	(127,700–1,816,100)
Acute Respiratory Symptoms	21,961	(7937–31,082)	395,300	(142,900–559,500)
Asthma Exacerbation	16,539	(8583–23,225)	1,174,300	(609,400–1,649,000)
Chronic Bronchitis	6.1	(0.9–12)	2,135,100	(303,400–4,288,800)
Emergency Room Visits	25	(12–36)	71,200	(34,800–99,900)
Hospital Admissions	62	(36–74)	172,600	(101,300–206,300)
Hospital Admissions, Cardiovascular	2.7	(0.4–5.4)	17,900	(2600–36,400)
Hospital Admissions, Respiratory	2.1	(0.3–4.4)	6000	(800–12,200)
Lower Respiratory Symptoms	124	(18–248)	2200	(300–4500)
Mortality	30	(7.0–54)	221,347,200	(50,756,400–392,560,000)
School Loss Days	4586	(1870–5076)	634,600	(258,900–702,400)
Upper Respiratory Symptoms	98	(14–195)	1800	(200–3500)
Work Loss Days	1168	(166–2348)	225,500	(32,000–453,200)

Table 4
Reduction in number of incidences and associated monetary value (CAD) for various health effects due to pollutant reduction from trees in the 15 most populated Canadian cities (results for all 86 cities are given in Supplemental Table 3).

Pollutant	Adverse Health Effect	Incidence[a]	Value	Incidence[a]	Value	Incidence[a]	Value	Incidence[a]	Value
		Calgary		Edmonton		Gatineau		Halifax	
NO_2	Acute Respiratory Symptoms	24.64	444	20.49	369	5.76	104	12.18	219
	Asthma Exacerbation	343.71	24,403	293.91	20,868	87.91	6242	192.47	13,665
	Emergency Room Visits	0.25	709	0.29	817	0.09	246	0.23	643
	Hospital Admissions	0.57	1594	1.07	2997	0.38	1059	0.74	2081
O_3	Acute Respiratory Symptoms	242.93	4373	203.16	3657	162.89	2932	605.49	10,899
	Emergency Room Visits	0.12	330	0.08	238	0.06	171	0.27	743
	Hospital Admissions	0.23	653	0.28	774	0.28	797	0.81	2269
	Mortality	0.12	896,631	0.11	783,137	0.08	588,301	0.33	2,415,580
	School Loss Days	69.03	9553	58.89	8150	51.44	7119	174.28	24,119
$PM_{2.5}$	Acute Bronchitis	0.29	125	0.54	233	0.16	68	0.51	221
	Acute Myocardial Infarction	0.18	27,408	0.31	48,607	0.07	11,225	0.52	80,208
	Acute Respiratory Symptoms	202.09	3638	382.26	6881	105.82	1905	398.13	7166
	Asthma Exacerbation	138.10	9805	259.73	18,441	72.58	5153	241.13	17,120
	Chronic Bronchitis	0.16	56,024	0.31	107,554	0.09	31,905	0.34	117,847
	Emergency Room Visits	0.20	558	0.37	1047	0.10	292	0.33	924
	Hospital Admissions, Cardiovascular	0.04	264	0.16	1052	0.03	232	0.20	1324
	Hospital Admissions, Respiratory	0.03	92	0.15	418	0.01	39	0.19	519
	Lower Respiratory Symptoms	3.86	69	6.89	124	1.95	35	6.40	115
	Mortality	0.85	6,192,696	1.73	12,639,655	0.41	3,024,930	0.76	5,527,440
	Upper Respiratory Symptoms	2.76	50	5.31	96	1.49	27	4.92	89
	Work Loss Days	33.19	6405	64.44	12,436	17.93	3460	65.20	12,582
SO_2	Acute Respiratory Symptoms	1.21	22	0.87	16	0.12	2	2.56	46
	Asthma Exacerbation	10.53	748	7.24	514	1.03	73	20.77	1475
	Emergency Room Visits	0.03	98	0.04	102	0.004	12	0.09	261
	Hospital Admissions	0.09	250	0.04	106	0.01	23	0.13	366
		Hamilton		Kitchener		London		Montréal	
NO_2	Acute Respiratory Symptoms	25.79	464	10.33	186	8.53	154	66.79	1202
	Asthma Exacerbation	392.69	27,881	161.50	11,467	132.03	9374	1,001.41	71,100
	Emergency Room Visits	0.31	870	0.15	413	0.13	374	0.44	1220
	Hospital Admissions	1.29	3625	0.46	1281	0.33	924	2.50	7012
O_3	Acute Respiratory Symptoms	535.51	9639	335.10	6032	258.18	4647	1,206.52	21,717
	Emergency Room Visits	0.21	574	0.15	421	0.12	332	0.34	958
	Hospital Admissions	0.58	1627	0.50	1388	0.39	1086	1.36	3806
	Mortality	0.38	2,769,559	0.21	1,511,785	0.16	1,151,041	0.68	4,977,442
	School Loss Days	187.78	25,987	119.16	16,491	87.20	12,068	388.93	53,824
$PM_{2.5}$	Acute Bronchitis	0.44	190	0.25	109	0.15	64	1.24	535
	Acute Myocardial Infarction	0.40	62,548	0.21	31,923	0.05	7175	0.91	141,140
	Acute Respiratory Symptoms	312.52	5625	159.13	2864	102.02	1836	898.30	16,169
	Asthma Exacerbation	243.62	17,297	117.46	8340	73.76	5237	591.85	42,021
	Chronic Bronchitis	0.26	88,880	0.13	46,440	0.09	30,387	0.80	279,525
	Emergency Room Visits	0.31	863	0.15	418	0.10	267	0.83	2337
	Hospital Admissions, Cardiovascular	0.18	1188	0.08	565	0.03	179	0.44	2930
	Hospital Admissions, Respiratory	0.10	279	0.04	119	0.03	84	0.37	1028
	Lower Respiratory Symptoms	6.67	120	3.14	57	1.93	35	15.18	273
	Mortality	0.77	5,631,773	1.18	8,600,160	0.34	2,453,930	3.52	25,680,497
	Upper Respiratory Symptoms	4.47	80	2.46	44	1.47	26	12.27	221
	Work Loss Days	50.14	9677	25.95	5008	17.10	3301	151.16	29,172
SO_2	Acute Respiratory Symptoms	7.29	131	0.74	13	0.79	14	7.42	134
	Asthma Exacerbation	60.47	4293	6.44	457	6.48	460	64.62	4588
	Emergency Room Visits	0.31	869	0.02	56	0.02	69	0.29	799
	Hospital Admissions	0.45	1274	0.03	84	0.04	122	0.56	1557
		Ottawa		Québec		St. Catharines – Niagara		Toronto	
NO_2	Acute Respiratory Symptoms	15.68	282	20.89	376	8.96	161	199.27	3587
	Asthma Exacerbation	232.21	16,487	317.34	22,531	137.22	9743	2,958.21	210,033
	Emergency Room Visits	0.21	591	0.38	1072	0.14	405	2.59	7258
	Hospital Admissions	0.75	2096	1.40	3913	0.48	1351	6.77	18,959
O_3	Acute Respiratory Symptoms	452.16	8139	598.85	10,779	228.46	4112	2,819.81	50,757
	Emergency Room Visits	0.15	412	0.22	622	0.07	196	0.88	2469
	Hospital Admissions	0.60	1687	0.74	2081	0.35	971	8.26	23,128
	Mortality	0.32	2,310,006	0.35	2,521,618	0.14	1,054,534	1.64	11,947,817
	School Loss Days	149.89	20,744	152.83	21,151	80.11	11,086	1,000.93	138,520
$PM_{2.5}$	Acute Bronchitis	0.32	138	0.47	200	0.16	69	1.96	843
	Acute Myocardial Infarction	0.08	12,035	0.25	37,999	0.10	15,161	0.83	129,127
	Acute Respiratory Symptoms	203.36	3661	392.25	7061	113.62	2045	1,242.34	22,362
	Asthma Exacerbation	133.68	9491	227.87	16,179	81.40	5779	905.51	64,291
	Chronic Bronchitis	0.18	64,357	0.36	124,470	0.10	35,774	1.08	376,458
	Emergency Room Visits	0.17	479	0.23	630	0.10	277	1.26	3537
	Hospital Admissions, Cardiovascular	0.05	347	0.13	875	0.06	430	0.32	2092
	Hospital Admissions, Respiratory	0.05	144	0.12	326	0.04	101	0.16	459
	Lower Respiratory Symptoms	3.24	58	5.82	105	2.15	39	24.30	437
	Mortality	0.58	4,204,229	0.83	6,035,234	0.45	3,290,200	1.69	12,315,725

(continued on next page)

Table 4 (*continued*)

Pollutant	Adverse Health Effect	Incidence[a]	Value	Incidence[a]	Value	Incidence[a]	Value	Incidence[a]	Value
	Upper Respiratory Symptoms	3.07	55	4.62	83	1.62	29	18.48	333
	Work Loss Days	34.66	6689	66.21	12,777	19.43	3750	207.60	40,063
SO_2	Acute Respiratory Symptoms	0.27	5	1.20	22	0.81	15	24.93	449
	Asthma Exacerbation	2.27	161	10.64	755	6.57	466	193.63	13,748
	Emergency Room Visits	0.01	28	0.03	97	0.04	107	1.02	2866
	Hospital Admissions	0.02	45	0.11	309	0.06	179	1.37	3835
		Vancouver		Victoria		Winnipeg			
NO_2	Acute Respiratory Symptoms	141.70	2551	17.36	313	15.23	274		
	Asthma Exacerbation	2,031.62	144,245	266.06	18,890	216.22	15,352		
	Emergency Room Visits	1.89	5285	0.32	892	0.18	516		
	Hospital Admissions	5.40	15,110	1.58	4429	0.49	1374		
O_3	Acute Respiratory Symptoms	1,924.46	34,640	298.70	5377	280.66	5052		
	Emergency Room Visits	0.86	2402	0.14	382	0.13	365		
	Hospital Admissions	2.55	7130	0.48	1352	0.20	563		
	Mortality	0.98	7,166,381	0.16	1,168,753	0.15	1,086,706		
	School Loss Days	598.07	82,767	81.15	11,230	90.61	12,539		
$PM_{2.5}$	Acute Bronchitis	0.59	254	0.15	66	0.33	143		
	Acute Myocardial Infarction	0.16	24,158	0.14	22,260	0.28	43,943		
	Acute Respiratory Symptoms	406.54	7318	124.48	2241	204.11	3674		
	Asthma Exacerbation	246.44	17,497	69.70	4949	150.51	10,686		
	Chronic Bronchitis	0.38	130,689	0.12	42,187	0.18	63,297		
	Emergency Room Visits	0.32	884	0.09	260	0.20	557		
	Hospital Admissions, Cardiovascular	0.11	711	0.08	531	0.11	756		
	Hospital Admissions, Respiratory	0.11	301	0.05	126	0.12	341		
	Lower Respiratory Symptoms	5.94	107	1.77	32	3.99	72		
	Mortality	1.17	8,548,710	0.41	2,989,480	0.80	5,864,961		
	Upper Respiratory Symptoms	5.68	102	1.45	26	3.18	57		
	Work Loss Days	69.29	13,373	20.71	3997	33.51	6466		
SO_2	Acute Respiratory Symptoms	18.17	327	1.97	35	0.48	9		
	Asthma Exacerbation	155.80	11,061	16.00	1136	3.63	257		
	Emergency Room Visits	0.64	1790	0.11	313	0.02	59		
	Hospital Admissions	1.37	3828	0.25	693	0.03	88		

Table 5
Average pollution removal and value per unit tree cover in the 15 most populated Canadian cities. Results for all 86 cities are given in Supplemental Table 4.

City	g/m^2	CAD/ha
Calgary	3.02	1074
Edmonton	2.69	1206
Gatineau	3.49	701
Halifax	3.63	547
Hamilton	5.38	1016
Kitchener	4.59	1564
London	4.34	804
Montréal	3.95	886
Ottawa	3.55	648
Québec	2.14	275
St. Catharines – Niagara	5.22	471
Toronto	4.60	792
Vancouver	3.62	336
Victoria	3.51	334
Winnipeg	3.22	938

Table 6
Average annual values per tonne ($/t, CAD) of removal and per hectare of tree cover ($/ha), average grams of removal per square meter of tree cover (g/m^2) and average absolute and percent reduction in pollutant concentration in 86 Canadian cities (2010).

Pollutant	$/t	$/ha	g/m^2	ΔC[a]	% ΔC[b]	% ΔCin[c]
NO_2	376	2.06	0.55	0.016	0.181	0.280
O_3	4946	137.62	2.78	0.062	0.246	0.418
$PM_{2.5}$	247,846	370.94	0.15	0.009	0.145	0.249
SO_2	89	0.19	0.21	0.004	0.273	0.485
CO	1486[d]	0.37	0.03	0.001	0.001	0.003
Total		511.18	3.72			

[a] average annual reduction in hourly concentration in ppb, except for $PM_{2.5}$ ($\mu g\ m^{-3}$).
[b] average percent annual reduction in hourly concentration.
[c] average percent reduction in hourly concentration during in-leaf season.
[d] based on externality value, not human health values. Externality estimates tend to be higher than health estimates.

canopy in preventing concentrations of upper air pollution from reaching ground-level air space. Measured differences in O_3 concentration between above- and below-forest canopies in California's San Bernardino Mountains have exceeded 50 ppb (40-percent improvement) (Bytnerowicz et al., 1999). Under normal daytime conditions, atmospheric turbulence mixes the atmosphere such that pollutant concentrations are relatively consistent with height (Colbeck and Harrison, 1985). Forest canopies can limit the mixing of upper air with ground-level air, leading to significant below-canopy air quality improvements. However, where there are numerous pollutant sources below the canopy (e.g., automobiles), the forest canopy could have the inverse effect by minimizing the dispersion of the pollutants away at ground level.

At the local scale, pollution concentrations can be increased if trees: a) trap the pollutants beneath tree canopies near emission sources (e.g., along road ways, Gromke and Ruck, 2009, Wania et al., 2012, Salmond et al., 2013, Vos et al., 2013), b) limit dispersion by reducing wind speeds, and/or c) lower mixing heights by reducing wind speeds (Nowak et al., 2006a). These local scale interactions are important for determining the net effect of trees on air quality and human health. While pollution removal is a positive effect as it removes pollutants from the atmosphere, the effects of trees altering pollution dispersion must be considered. This altering of dispersion can either increase local pollutant concentrations (reduced dispersion that increases concentrations) or decrease local pollutant concentrations (limit pollutants from reaching the area). Trapping pollutants in one area limits the amount of pollution transferred to another area. Thus, while pollution is removed by trees and concentration reduced on average, effects on local concentrations are variable due to dispersion effects. Tree impacts on patterns of dispersion are important to consider in relation to where humans interact with the outdoor atmosphere.

If most people spend their outdoor time near roadways where trees are limiting dispersion, then the local forest could have a negative effect on human health, even though the forest is removing air pollution. On the other hand, if people are spending more time in areas buffered from pollutant sources by trees (e.g., in forested parks with limited traffic), then the forest would likely be producing greater health benefits than estimated by pollution removal alone. This distribution of trees relative to human outdoor activity and local pollutants emissions, particularly from automobiles, is important to consider when designing urban forest landscapes to reduce pollutant concentrations. An issue to consider in urban forest design is that automobiles pollute and people breathe within the same near-ground air space. Creating distance between people and automobiles, or barriers that limit pollution dispersion into areas where people reside, recreate, walk, etc. could reduce human exposure to air pollution. If designed correctly, trees can be used to reduce pollutant concentrations in areas with high population densities, while still removing pollution from the atmosphere. The direct connection of trees to various other health benefits received by humans (e.g., air temperature reduction, aesthetics, connections with nature, recreational activities) also need to be considered when designing urban forests to optimize net benefits to society.

The combination of these numerous local scale interactions (e.g., emissions, wind, trees, people) are important for understanding the ultimate effect of urban forests on pollutant concentrations and human health. More research is needed on how these factors combine to affect air pollution concentrations, particularly along roadways and in heavily populated areas.

Values of air pollution removal, except for CO, are only based on human health impacts and are thus likely conservative. Values of air quality improvement would likely increase if other air pollution impacts such as protection against decreased visibility and damage to animals, crops, vegetation, materials and buildings are included.

There are certain actions managers can take to maximize positive air quality effects from trees. These actions include: a) increase or sustain tree cover to increase or maximize pollution removal, b) use low VOC emitting species if ozone is a local problem, c) use long-lived, low maintenance species to reduce pollutant emission associated tree planting, maintenance and removals, d) plant trees in energy conserving locations to reduce energy use and power plant emissions, e) plant species adapted to the site and maintain these trees to ensure healthy trees that maximize potential effects, f) supply ample water to enhance pollution removal of gaseous pollutants and reduce air temperatures via transpiration (with extra consideration given in areas where water is a limiting factor, e.g., deserts), g) plant trees in polluted or heavily populated areas to maximize pollution removal and health impacts, but ensure that designs do not trap pollutants (increase concentrations) in areas with large outdoor human populations, h) avoid pollution sensitive species to enhance tree health and pollution removal, i) utilize species with large total leaf area, relatively small or complex leaves, textured leaves and/or high water use (transpiration) to enhance pollution removal, and j) utilize evergreen species to enhance particulate removal during leaf off seasons (Nowak et al., 2006b).

This study does not address the issue of advection, where pollution removal in rural areas surrounding urban areas could lower the pollution concentrations arriving into urban areas (or vice versa). As many pollutants are generated locally, this may not be a major factor, but for some pollutants, particularly secondary pollutants such as O_3 that are formed from chemical reactions, the reduction of pollutants in rural areas could have an impact on urban pollutant concentrations. The magnitude of this potential impact is unknown.

Though there are various limitations to these estimates, the results give a first-order approximation of the magnitude of pollution removal by city trees and their effect on human health. Limitations of the analysis include issues associated with modeling particulate matter removal and resuspension (see Nowak et al., 2013a), limited weather and pollution data, tree cover data and estimating human health effects and values (see Nowak et al., 2014). Results are only for pollution removal and do not include other generally positive (i.e., air temperature reduction, building energy use conservation) and negative (VOC and pollen emissions, reduced wind speeds and dispersion) effects of trees on air quality.

As pollution removal is largely driven by tree leaf area, other limitations of this study relate to estimating leaf area in the cities. Three main variables are used to estimate daily leaf area: tree cover (m^2), LAI within tree canopies, and percent evergreen. Tree cover was derived from photo-interpretation of aerial images and had a standard error typically less than one percent (Pasher et al., 2014). LAI was derived from either MODIS or averages from urban field data. Many urban areas had missing LAI estimates due to the coarseness of the MODIS data and relatively low amounts of forest cover in urban areas, thus the average LAI from urban field data was often applied (4.9, standard error = 0.2). Percent evergreen was derived from regional forest type data, but as cities plant a variety of trees, the proportion of evergreen trees between a city and the surrounding region are likely to differ. For example, field data reveal that Toronto's urban forest is 15.4% evergreen (Nowak et al., 2013b), while the regional forest estimate is 7.3% (Table 1). The impact of differences in percent evergreen will likely be minimal as the difference will only affect leaf area during the leaf off season, with most of the impact affecting $PM_{2.5}$ removal estimates as gas exchange is limited during the winter season.

Despite the limitations, there are several advantages to the modeling estimates, which include the use of best available tree, weather, population and pollution data, modeling of tree effects on hourly pollution concentrations and modeling of pollution effects on human health (Nowak et al., 2014). More fine scale modeling across a city (e.g., neighbor-scale analyses) could help illustrate variations in pollution removal and effects on populations throughout cities. Though future research and modeling are needed to help overcome current limitations, these estimates provide the best available and most comprehensive estimates of pollution removal effects by city trees on human health in Canada.

5. Conclusion

Through pollution removal and other ecosystem services (e.g., air temperature reductions), urban trees can help improve air quality in cities and consequently can help improve human health. While the existing percent air quality improvements due to pollution removal are less than one percent, these marginal changes can affect human health by varying degrees across a city. While removal is a positive effect, the health effects from trees could be negative if pollutants are trapped near people, or could be greater than estimated if pollutants are deflected away from people. There are several environmental and social variables that affect air quality and human health. By understanding these variables, urban forests can be better designed and managed to improve air quality and human health for current and future generations.

Acknowledgment

Funding for this project was provided in part by Environment and Climate Change Canada.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ufug.2017.10.019.

References

Abt Associates, 2010. BenMAP Environmental Benefits Mapping and Analysis Program. Office of Air Quality Planning and Standards U.S. Environmental Protection Agency, Research Triangle Park, NC, pp. 283 (Retrieved May, 2012 from http://www.epa.gov/airquality/benmap/models/BenMAPManualAugust2010. pdf).

Bytnerowicz, A., Fenn, M.E., Miller, P.R., Arbaugh, M.J., 1999. Wet and dry pollutant deposition to the mixed conifer forest. In: Miller, P.R., McBride, J.R. (Eds.), Oxidant Air Pollution Impacts in the Montane Forests of Southern California: A Case Study of the San Bernardino Mountains. Springer-Verlag, New York, pp. 235–269.
Canadian Council of Ministers of the Environment (CCME), 2016. Canadian Ambient Air Quality Standards (CAAQS). (Retrieved February, 2016 from http://www.ccme.ca/en/current_priorities/air/caaqs.html).
Canadian Medical Association, 2008. No Breathing Room: National Illness Cost of Air Pollution. CMA, Ottawa.
Cardelino, C.A., Chameides, W.L., 1990. Natural hydrocarbons, urbanization, and urban ozone. J. Geophys. Res. 95 (D9), 13971–13979.
Cavanagh, J.E., Zawar-Reza, P., Wilson, J.G., 2009. Spatial attenuation of ambient particulate matter air pollution within an urbanised native forest patch. Urban Forest. Urban Green. 8, 21–30.
Chameides, W.L., Lindsay, R.W., Richardson, J., Kiang, C.S., 1988. The role of biogenic hydrocarbons in urban photochemical smog: atlanta as a case study. Science 241, 1473–1475.
City of Edmonton, 2012. Urban Forest Management Plan: Edmonton's Urban Forest –Taking Root Today for a Sustainable Tomorrow. (City of Edmonton, 38 p. Retrieved March, 2016 from http://www.edmonton.ca/residential_neighbourhoods/PDF/Urban_Forest_Management_Plan.pdf).
Colbeck, I., Harrison, R.M., 1985. Dry deposition of ozone: some measurements of deposition velocity and of vertical profiles to 100 metres. Atmos. Environ. 19 (11), 1807–1818.
Darley, E.F., 1971. Vegetation damage from air pollution. In: Starkman, E.S. (Ed.), Combustion-Generated Air Pollution. Plenum Press, New York, pp. 245–255.
Dasch, J.M., 1987. Measurement of dry deposition to surfaces in deciduous and pine canopies. Environ. Pollut. 44, 261–277.
Davidson, K., Hallberg, A., McCubbin, D., Hubbell, B., 2007. Analysis of $PM_{2.5}$ using the environmental benefits mapping and analysis program (BenMAP). J. Toxicol. Environ. Health, Part A 70, 332–346.
Environment Canada, 2013. NAPS Data Products. (Retrieved December, 2013 from http://maps-cartes.ec.gc.ca/rnspa-naps/data.aspx?lang=en).
Gromke, C., Ruck, B., 2009. On the impact of trees on dispersion processes of traffic emissions in street canyons. Boundary Layer Meteorol. 131 (1), 19–34.
Heisler, G.M., 1986. Energy savings with trees. J. Arboric. 12 (5), 113–125.
Hicks, B.B., Matt, D.R., McMillen, R.T., 1989. A micrometeorological investigation of the surface exchange of O_3, SO_2, and NO_2: A case study. Boundary Layer Meteorol. 47, 321–336.
Hirabayashi, S., Kroll, C.N., 2017. Single Imputation Method of Missing Air Quality Data for i-Tree Eco Analyses in the Conterminous United States. (Retrieved January 2017 from https://itreetools.org/eco/resources/Single_imputation_method_of_missing_air_quality_data_for_i-Tree_Eco_analyses_in_the_conterminous_United_States.pdf).
Hirabayashi, S., Kroll, C.N., Nowak, D.J., 2011. Component-based development and sensitivity analyses of an air pollutant dry deposition model. Environ. Modell. Software 26, 804–816.
Hirabayashi, S., Kroll, C.N., Nowak, D.J., 2012. i-Tree Eco Dry Deposition Model Descriptions. (Retrieved October, 2013 from http://www.itreetools.org/eco/resources/iTree_Eco_Dry_Deposition_Model_Descriptions.pdf).
Hodan, W.B., Barnard, W.R., 2004. Evaluating the Contribution of PM2.5 Precursor Gases and Re-entrained Road Emissions to Mobile Source PM2.5 Particulate Matter Emissions. (MACTEC under Contract to the Federal Highway Administration, p. 53. Retrieved June, 2012 from http://www.epa.gov/ttnchie1/conference/ei13/mobile/hodan.pdf).
Judek, S., Stieb, D., Jovic, B., 2006. Air Quality Benefits Assessment Tool (AQBAT) Release 1.0. Health Canada, Ottawa.
Lelieveld, J., Evans, J.S., Fnais, M., Giannadaki, D., Pozzer, A., 2015. The contribution of outdoor air pollution sources to premature mortality on a global scale. Nature 525, 367–371.
Lovett, G.M., 1994. Atmospheric deposition of nutrients and pollutants in North America: an ecological perspective. Ecol. Appl. 4, 629–650.
McDonald, A.G., Bealey, W.J., Fowler, D., Dragosits, U., Skiba, U., Smith, R.I., Donovan, R.G., Brett, H.E., Hewitt, C.N., Nemitz, E., 2007. Quantifying the effect of urban tree planting on concentrations and depositions of PM_{10} in two UK conurbations. Atmos. Environ. 41, 8455–8467.
McNeil, J., Vava, C., 2006. Oakville's Urban Forest: Our Solution to Our Pollution. (Town of Oakville, 59 p. Retrieved March, 2016 from https://www.itreetools.org/resources/reports/Oakville's%20Urban%20Forest.pdf).
Murray, F.J., Marsh, L., Bradford, P.A., 1994. New York State Energy Plan Vol. II New York State Energy Office, Albany, NY (Issue Reports).
National Climatic Data Center (NCDC), 2005. NOAA Frost Maps. (Retrieved June, 2005 from http://www.ncdc.noaa.gov/oa/documentlibrary/freezefrost/frost-freemaps.html).
National Climatic Data Center (NCDC), 2013. Climate Data Online: Text & Map Search. (Retrieved February 2013 from http://www.ncdc.noaa.gov/cdo-web/).
National Oceanic, Atmospheric Administration (NOAA), 2013. NOAA/ESRL Radiosonde Database. (Retrieved February 2013 from http://www.esrl.noaa.gov/raobs/).
Nowak, D.J., Civerolo, K.L., Rao, S.T., Sistla, G., Luley, C.J., Crane, D.E., 2000. A modeling study of the impact of urban trees on ozone. Atmos. Environ. 34, 1610–1613.
Nowak, D.J., Crane, D.E., Stevens, J.C., 2006a. Air pollution removal by urban trees and shrubs in the United States. Urban Forest. Urban Green. 4, 115–123.
Nowak, D.J., Hoehn, R., Crane, D.E., Stevens, J.C., Walton, J.T., 2006b. Assessing Urban Forest Effects and Values: Washington D.C.'s Urban Forest. USDA Forest Service, Northern Research Station Resource Bulletin NRS-1. (Newtown Square, PA. 24 p.).
Nowak, D.J., Hoehn, R.E., Crane, D.E., Stevens, J.C., Walton, J.T., Bond, J., 2008. A ground-based method of assessing urban forest structure and ecosystem services. Arboricult. Urban Forest. 34 (6), 347–358.
Nowak, D.J., Hirabayashi, S., Bodine, A., Hoehn, R., 2013a. Modeled $PM_{2.5}$ removal by trees in ten U.S. cities and associated health effects. Environ. Pollut. 178, 395–402.
Nowak, D.J., Hoehn, R.E., Bodine, A.R., Greenfield, E.J., Ellis, A., Endreny, T.E., Yang, Y., Zhou, T., Henry, R., 2013b. Assessing Forest Effects and Values: Toronto's Urban Forest USDA Forest Service, Northern Research Station Resource Bulletin NRS-79. (Newtown Square, PA. 59 p.).
Nowak, D.J., Hirabayashi, S., Ellis, E., Greenfield, E.J., 2014. Tree and forest effects on air quality and human health in the United States. Environ. Pollut. 193, 119–129.
Nowak, D.J., Bodine, A.R., Hoehn, R.E., Low, S.C., Roman, L.A., Henning, J.G., Stephan, E., Taggart, T., Endreny, T., 2016. The Urban Forest of Philadelphia. Resource Bulletin NRS-106. U.S. Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, PA (80 p.).
Ogren, T.L., 2000. Allergy-Free Gardening. Ten Speed Press, Berkeley (267 p.).
Ottinger, R.L., Wooley, D.R., Robinson, N.A., Hodas, D.R., Babb, S.E., Buchanan, S.C., Chernick, P.L., Caverhill, E., Krupnick, A., Harrington, W., Radin, S., Fritsche, U., 1990. Environmental Costs of Electricity. Oceana Publications, White Plains, NY (769 p.).
Pasher, J., McGovern, M., Khoury, M., Duffe, J., 2014. Assessing carbon storage and sequestration by Canada's urban forests using high resolution earth observation data. Urban Forest. Urban Green. 13, 484–494.
Phelps, G., Crabtree, S., 2013. Worldwide, Median Household Income About $10,000. Gallup. December 16, 2013. (Retrieved September 2015 from http://www.gallup.com/poll/166211/worldwide-median-household-income-000. aspx).
Pope, C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., Thurston, G.D., 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. J. Am. Med. Assoc. 287 (9), 1132–1141.
Salmond, J.A., Williams, D.E., Laing, G., Kingham, S., Dirks, K., Longley, I., Henshaw, G.S., 2013. The influence of vegetation on the horizontal and vertical distribution of pollutants in a street canyon. Sci. Total Environ. 443, 287–298.
Smith, W.H., 1990. Air Pollution and Forests. Springer-Verlag, New York, NY (618 p.).
Statistics Canada, 2011a. Population Centre Boundary File, 2011 Census Year. (Catalogue No. 92-166-XWE. Retrieved October, 2015 from http://www12. statcan.gc.ca/census-recensement/2011/geo/bound-limit/bound-limit-2011-eng.cfm).
Statistics Canada, 2011b. 2011 Census Profiles. (Retrieved October 2015 from http://www12. statcan.gc.ca/census-recensement/2011/dp-d/prof/index.cfm?Lang=E&MMUnited).
Statistics Canada, 2015. Estimates of Population by Census Metropolitan Area, Sex and Age Group for July 1, Based on the Standard Geographical Classification (SGC) 2011. (Retrieved December 2015 from http://www5. statcan.gc.ca/cansim/a26?lang=eng&id=510056).
TRCA, 2011. Peel Region Urban Forest Strategy. Toronto and Region Conservation Authority. (83 p. Retrieved March, 2016 from https://www.itreetools.org/resources/reports/Peel_Urban_Forest_Strategy.pdf).
Taha, H., 1996. Modeling impacts of increased urban vegetation on ozone air quality in the South Coast Air Basin. Atmos. Environ. 30 (20), 3423–3430.
Tallis, M., Taylor, G., Sinnett, D., Freer-Smith, P., 2011. Estimating the removal of atmospheric particulate pollution by the urban tree canopy of London, under current and future environments. Landscape Urban Plann. 103, 129–138.
Tiwary, A., Sinnett, D., Peachey, C., Chalabi, Z., Vardoulakis, S., Fletcher, T., Leonardi, G., Grundy, C., Azapagic, A., Hutchings, T.R., 2009. An integrated tool to assess the role of new planting in PM_{10} capture and the human health benefits: a case study in London. Environ. Pollut. 157, 2645–2653.
U.S. Department of Labor Bureau of Labor Statistics, 2012, Retrieved September 2012 from www.bls.gov/ppi/.
U.S. Environmental Protection Agency (US EPA), 2012a. Environmental Benefits Mapping and Analysis Program (BenMAP). (Retrieved May, 2012 from http://www.epa.gov/air/benmap/).
U.S. Environmental Protection Agency (US EPA), 2012b. $PM_{2.5}$ and Ozone Impacts Per Unit Change in Air Quality. (Retrieved May 2012 from http://www.epa.gov/air-quality/benmap/reduced.html).
U.S. Geological Survey (USGS), 2013. Leaf Area Index K Fraction of Photosynthetically Active Radiation 8-day L4 Global 1 km. (Retrieved February 2013 from https://lpdaac.usgs.gov/products/modis_products_table/mod15a2).
Vos, P.E.J., Maiheu, B., Vankerkom, J., Janssen, S., 2013. Improving local air quality in cities: to tree or not to tree? Environ. Pollut. 183, 113–122.
Wania, A., Bruse, M., Blond, N., Weber, C., 2012. Analysing the influence of different street vegetation on traffic-induced particle dispersion using microscale simulations. J. Environ. Manage. 94, 91–101.