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Quantifying air pollution removal by green roofs in Chicago

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ABSTRACT

The level of air pollution removal by green roofs in Chicago was quantified using a dry deposition model. The result showed that a total of 1675 kg of air pollutants was removed by 19.8 ha of green roofs in one year with O_3 accounting for 52% of the total, NO_2 (27%), PM_{10} (14%), and SO_2 (7%). The highest level of air pollution removal occurred in May and the lowest in February. The annual removal per hectare of green roof was 85 kg ha^{-1} yr^{-1}. The amount of pollutants removed would increase to 2046.89 metric tons if all rooftops in Chicago were covered with intensive green roofs. Although costly, the installation of green roofs could be justified in the long run if the environmental benefits were considered. The green roof can be used to supplement the use of urban trees in air pollution control, especially in situations where land and public funds are not readily available.

1. Introduction

City air often contains high levels of pollutants that are harmful to human health (Mayer, 1999). The American Lung Association (ALA, 2007) reported that over 3700 premature deaths annually in the United States could be attributed to a 10-ppb increase in O_3 levels. Worldwide, the World Health Organization (WHO, 2002) estimated that more than 1 million premature deaths annually could be attributed to urban air pollution in developing countries. The United Nations Population Fund (UNFPA, 2007) predicted that the urban population worldwide would increase from 3.3 billion in 2008 to 5 billion by 2030, meaning that there will be an increase in sensitive population groups such as children and the elderly. Therefore, cities with serious air pollution problems need to come up with ways to control the problem and reduce the damages.

Conventional air pollution management programs focus on controlling the source of air pollutants (Schnelle and Brown, 2002). This strategy effectively reduces the emission of new air pollutants but does not address the pollutants already in the air. Innovative approaches can be adopted to remove existing air pollutants thereby reducing air pollution concentrations to an acceptable level. One way to reach that goal is the use of urban vegetation which can reduce air pollutants through a dry deposition process and microclimate effects. The high surface area and roughness provided by the branches, twigs, and foliage make vegetation an effective sink for air pollutants (Beckett et al., 1998; Hill, 1971). Vegetation also has an indirect effect on pollution reduction by modifying microclimates. Plants lower the indoor air temperature through shading, thus reducing the use of electricity for air conditioning (Heisler, 1986). The final result is that the emission of pollutants from power plants decreases due to reduced energy use. Vegetation also lowers the ambient air temperature by changing the albedos of urban surfaces and evapotranspiration cooling. The lowered ambient temperature then slows down photochemical reactions and leads to less secondary air pollutants, such as ozone (Akbari, 2002;

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Rosenfeld et al., 1998). Studies show that trees could contribute significantly to air pollution reduction in cities (Nowak, 1994; Nowak et al., 2006; Rosenfeld et al., 1998; Scott et al., 1998). Nowak et al. (2006) estimated that urban trees remove a total of 711 000 metric tons annually in the U.S. These findings led to the inclusion of tree planting as a state implementation strategy for improving air quality by the United States Environmental Protection Agency (EPA) in 2004 (US EPA, 2004).

While it is desirable to use trees for controlling air pollution, it is not always easy to plant trees in a densely populated city. For example, the percentage of impervious area in New York City is 64%; it can reach as high as 94% in districts like Mid-Manhattan west (Rosenzweig et al., 2006). The green roof can be a solution to this dilemma since it makes use of rooftops, usually 40–50% of the impermeable area in a city (Dunnett and Kingsbury, 2004). Nevertheless, the limited number of studies on the air pollutant removal capacity of green roofs does not provide enough information for people to judge their effectiveness in air pollution control. The methods and main findings of the few reported studies are summarized in the following section.

Currie and Bass (2005) estimated that 109 ha of green roofs in Toronto could remove a total of 7.87 metric tons of air pollutants annually. They pointed out in their paper that the urban forest effects (UFORE) model they used was developed specifically for trees and shrubs. The majority of plants used on green roofs are herbaceous plants which would have an impact on estimates when using this model. Deutsch et al. (2005) conducted a simulation of different planting scenarios of green roofs in Washington, DC, using the UFORE model. They showed that 58 metric tons of air pollutants could be removed if all the roofs in the city were converted to green roofs. Corrie et al. (2005) estimated the annual reduction of NO_2 by green roofs in Chicago and Detroit. Their study showed by covering 20% of the roof surface in Chicago the reduction of NO_2 was between 806.48 and 2769.89 metric tons depending on the type of plants used. These estimates were reached by assuming the NO_2 uptake rates by green roof plants were constant. This could be problematic because NO_2 uptake is influenced by many factors (e.g., meteorological conditions, concentration of NO_2, plant physiology). In one field study, Tan and Sia (2005) measured the concentrations of acidic gaseous pollutants and particulate matters on a 4000 m^2 roof in Singapore before and after the installation of a green roof. They found that the levels of particles and SO_2 in air above the roof were reduced by 6% and 37%, respectively, after installation of the green roof. This field measurement proved that green roofs can reduce certain air pollutants but it is difficult to extrapolate their results to other places or to a larger scale. The measurement was site specific and the volume of air that was influenced by the green roof was not given.

The cases discussed above have shown the potential benefit of using green roofs in air pollution control. However, there are many aspects of this mitigation measure that remain unclear. More studies are needed to help cities decide whether the green roof can be an effective way to improve air quality. We believe the following questions need to be answered: How can we quantify the level of air pollutant removal after installing green roofs in one city? Is there a difference between different types of green roofs in the level of air pollutant removal? How does the green roof compare to other mitigation measures such as planting trees? In this paper, we will address those questions with a case study in Chicago, Illinois.

2. Study site and methods

2.1. Study site

This study took place in Chicago, Illinois, which is located along the southwest shore of Lake Michigan with a center coordinate of 41°53′N and 87°39′W. The total area of the city is 588.3 km^2. Chicago is the third most populous city in the U.S with a population of 2.9 million in 2000. According to ALA (2007), over 2 million people in Chicago were at heightened risk for health problems resulting from acute exposure to O_3 and particulate matters.

Chicago is ranked number one in terms of total area of installed green roofs among North American cities. According to Taylor (2007), green roofs were installed on 300 buildings resulting in a total area of 27.87 ha by June 2007. There are three types of green roofs in Chicago: extensive green roofs, intensive green roofs, and semi-intensive green roofs. Extensive green roofs are planted with low height and slow growing plants. The depth of the growth media is less than 15 cm. Intensive green roofs consist of large perennial herbaceous plants and, occasionally, shrubs and small trees. The depth of growth media on an intensive green roof usually varies between 20 cm and 1.2 m. The semi-intensive green roof is a mixture of extensive and intensive green roof with 25% or less of the area as extensive green roof.

2.2. Survey of green roofs in Chicago

A request for information was submitted to Chicago's Department of Environment for a list of green roofs resulting in a list of 170 green roofs. Two steps were taken to verify the list. First, information including the address of the green roof, type of the green roof, size, and the date it was completed was gathered from various sources. We then searched the address of each green roof through an image database hosted by Pictometry International Crop. Digital aerial photographs covering Chicago were taken by Pictometry International Corp in July 2006. Because the photographs have a ground resolution of 16 cm and were taken from multiple angles, the location, size, type of the green roof, and the type of building could be clearly interpreted. For each green roof, the area of grass, trees, and other surfaces was measured and the percentage to the total area was calculated. Pictometry software allows users to directly measure distances and areas on those georeferenced images. The error margin of the measurement was estimated to be 1% or smaller (Federal Emergency Management Agency, 2005).

2.3. Removal of air pollutants by green roofs

In this study, a big-leaf resistance model was used to quantify the dry deposition of air pollutants. The structure of the model and how the input parameters were fitted are explained below.

The removal of a particular air pollutant at a given place over a certain time period was calculated as (Nowak, 1994):

$$Q = F \times L \times T \tag{1}$$

where Q is the amount of a particular air pollutant removed by certain area of green roofs in a certain time period (g), F is the pollutant flux ($\mathrm{g\,m^{-2}\,s^{-1}}$), L is the total area of green roof ($\mathrm{m^2}$), and T is the time period (s). The pollutant flux F is calculated as in Eq. (2):

$$F = V_d \times C \times 10^{-8} \tag{2}$$

where V_d = dry deposition velocity of a particular air pollutant ($\mathrm{cm\,s^{-1}}$), and C = concentration of that pollutant in the air ($\mathrm{\mu g\,m^{-3}}$). The dry deposition process can be described as the inverse of total resistance (Baldocchi et al., 1987):

$$V_d = \frac{1}{R_a + R_b + R_c} \tag{3}$$

where R_a = aerodynamic resistance, R_b = quasi-laminar boundary layer, and R_c = canopy resistance. The algorithms for calculating R_a and R_b were reported in Yang et al. (2005). In this study, the roughness length z_0 and displacement length d for short grasses were used to represent extensive green roofs. The intensive green roofs were treated as mixtures of short grass, tall herbaceous plants, and small deciduous tree. The z_0 and d values used in the model are listed in Table 1.

The hourly canopy resistances R_c for O_3, SO_2, and NO_2 are calculated as (Walmsley and Wesely, 1996).

$$R_c = \Big[(R_{sx} + R_{mx})^{-1} + R_{lux}^{-1} + (R_{dc} + R_{clx})^{-1} + (R_{ac} + R_{gsx})^{-1}\Big]^{-1} \tag{4}$$

In Eq. (4), R_{sx} is leaf stomata resistance, R_{mx} is leaf mesophyll resistance, R_{lux} is leaf cuticles resistance, R_{dc} is the resistance for gas-phase transfer by buoyant convection in canopies, R_{clx} is resistance by leaves, twigs, bark or other exposed surfaces in the lower canopy, R_{ac} is transfer resistance which depends only on canopy height and density, and R_{gsx} is ground surface resistance. Resistance components can vary with solar intensity, seasons, and vegetation types. Algorithms are available for calculating resistance components for grass and deciduous trees. The tall herbaceous plants were modeled as crops in this study. Details of the algorithms were described in Wesely (1989); Walmsley and Wesely (1996); Zhang et al. (2002).

The deposition velocity of PM over green roofs was calculated as (Zhang et al., 2001).

Table 1
Value of roughness lengths and displacement heights used in the model

Vegetation type	Average height h_0 (m)	$Z_0 = 0.1h_0$ (m)	$d = 0.7h_0$ (m)
Short grass	0.15	0.015	0.105
Tall herbaceous plants	1.0	0.1	0.7
Deciduous trees	5.0	0.5	3.5

$$V_d = V_g + \frac{1}{(R_a + R_s)} \tag{5}$$

Where V_g is the gravitational settling velocity, R_a is the aerodynamic resistance above the canopy, R_s is the surface resistance.

The gravitational settling is calculated as.

$$V_g = \frac{\rho d_p^2 g C}{18\eta} \tag{6}$$

Where ρ is the density of the particle, in this study, a value of 1800 $\mathrm{kg\,m^{-3}}$ was used as suggested by Lim et al. (2006), d_p is the particle diameter, g is the acceleration of gravity, C is the correction factor for small particles and is calculated as (Zhang et al., 2001), η is the viscosity coefficient of air.

The aerodynamic resistance R_a is calculated as before. The surface resistance R_s is based on the size of deposition particles, atmospheric conditions, and surface properties. It was calculated as (Zhang et al., 2001).

$$R_s = \frac{1}{\varepsilon_0 \mu^* (E_B + E_{IM} + E_{IN}) R_1} \tag{7}$$

Where ε_0 is an empirical constant and taken as 3 here, $\mu*$ is the friction velocity. E_B, E_{IM}, and E_{IN} are collection efficiency from Brownian diffusion, impaction and interception, respectively. The re-suspension of particles after hitting a surface was modeled by modifying the total collection efficiency by the factor of R_1, which represents the fraction of particles sticking to the surfaces. The extensive green roofs and intensive green roofs were modeled in the same manner as in calculating R_c. Details on how those parameters were fitted can be found in Zhang et al. (2001).

The final deposition velocity for PM_{10} was the weight-averaged V_d for all particles with a size less than 10 μm. Information on size classes and mass concentration of particles in Chicago were obtained from Offenberg and Baker (2000).

Hourly air pollution data including NO_2, SO_2, O_3, and PM_{10} concentration from an air pollution monitoring station in central Chicago between 8/1/2006 and 7/31/2007 were obtained from the U.S. EPA. Hourly surface meteorology data including sky condition, air temperature, relative humidity, atmospheric pressure, wind speed, precipitation, and snow cover measured by a station located at O'Hara International Airport for the same time period was obtained from the National Climatic Data Center. The hourly solar radiation intensity was simulated by using the meteorological/statistical solar radiation model (METSTAT, Maxwell et al., 1995). During precipitation and when the ground was covered by snow, the value of V_d was set as zero because the dry deposition process could not occur. Hourly fluxes of NO_2, SO_2, O_3, and PM_{10} to green roofs in Chicago were calculated by using weather data, concentration of pollutants, and the modeled deposition velocities.

2.4. Additional removal with different planting scenarios and costs

Three future planting scenarios were assumed and the amount of air pollution removal for each scenario calculated.

The first scenario assumed planting all roofs in Chicago with the same ratio of extensive vs. intensive green roofs used currently. The second scenario assumed the remaining roofs would only be planted with extensive roofs. The third scenario assumed only intensive roofs would be used in future projects. In all these scenarios, the intensive roof was treated as a mixture of tall herbaceous plants and small deciduous trees and shrubs at a ratio of 50:50. The total area of roofs in Chicago was obtained from Gray and Finster (2000) study, which showed that Chicago's roof surface was 27.86% of the urban area. According to information gathered from the green roof companies and the literature, the average installation cost for green roofs are as follows: extensive green roofs between \$107.64 and \$161.46 per m^2 (\$10–\$15 per ft^2); intensive green roofs between \$161.46 and \$269.1 per m^2 (\$10––\$25 per ft^2). The medians of those ranges were used in the calculation. The maintenance cost of green roofs was not included in this calculation.

3. Results

Among the 170 green roofs included in the list, detailed information for 71 green roofs was obtained and verified through aerial photographs. The total area of those 71 green roofs is 19.8 ha, 71% of the total area of green roofs in Chicago reported by Taylor (2007).

The information about those green roofs is shown in Table 2.

The green roofs surveyed were located mainly on commercial building and the size of each individual roof was relatively large. Among the 71 green roofs, half had an area larger than 500 m^2 and 23 green roofs were larger than 1000 m^2. The green roof in the Soldier Field was 22 445 m^2 while the one in Millennium Park was 99 983 m^2.

Based on the analysis of aerial photographs, the 19.8 ha of green roof consisted of 63% short grass and other low growing plants, 14% large herbaceous plants, 11% trees and shrubs, and about 12% various structures and hard surfaces.

The monthly air quality between August 2006 and July 2007 in Chicago is shown below (Fig. 1).

It can be seen from Fig. 1 that O_3 was the main air pollutant in Chicago. PM_{10} ranked second while the SO_2 pollution was low. PM_{10} and O_3 pollution peaked in summer while SO_2 and NO_2 peaked in winter.

The monthly mean deposition velocities for air pollutants calculated for different vegetation types showed a seasonal trend (Table 3). The deposition velocities for all air pollutants were highest in May and lowest in February.

The modeled monthly uptake of air pollutants by green roofs is shown in Fig. 2.

The total air pollution removal by 19.8 ha of green roofs was 1675 kg between August 2006 and July 2007. If the reported 27.87 ha of green roofs were all completed and had the same ratio of extensive vs. intensive green roofs, the air pollutants removed could reach 2388 kg.

Table 2
Percentages of different type of green roofs in Chicago

Type of green roof	On residential buildings (%)	On commercial buildings (%)	On office buildings (%)	Total (%)
Extensive	4.05	20.55	7.98	32.58
Intensive/semi-intensive	0.10	61.42	5.90	67.42
Sub total (%)	4.15	81.97	13.88	100.00

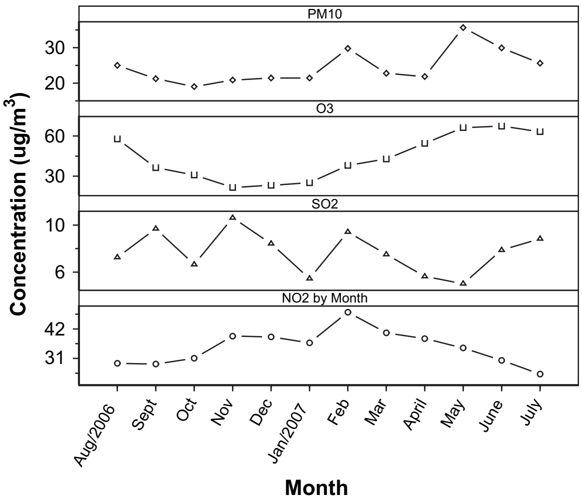

Fig. 1. Concentrations of criteria air pollutants in Chicago between August 2006 and July 2007. The monthly mean values were shown in the figure.

Among the four air pollutants, the uptake of O_3 was the largest, 52% of the total uptake followed by NO_2 (27%), PM_{10} (14%), and SO_2 (7%). Seasonally, the highest uptake occurred in May and the lowest in February. The annual removal rate among different vegetation types is compared in Table 4.

If all remaining roofs in Chicago were planted with intensive green roofs, the direct removal of air pollutants could reach as high as 2046.89 metric tons, assuming the same level of air pollution as 2006–2007. However, the installation cost would be \$35.2 billion.

4. Discussion

4.1. Evaluation of results

The results showed that air pollutant removal by green roofs in Chicago was affected by air pollutant concentrations, weather conditions, and the growth of plants. The highest air pollutant removal occurred in May when leaves of plants were fully expanded and the concentration of pollutants was high. The lowest removal was in February when the vegetation was covered in snow. The reliability of the estimate was evaluated by comparing it to values reported in other studies.

The dry deposition velocities of air pollutants influence the magnitude of air pollutant removal most. We found that the modeled deposition velocities were within a reasonable range compared to the measured values reported in the literature (Tables 4 and 6). It should be noted that the size of PM has a strong influence on the deposition velocity. In Chicago, Offenberg and Baker (2000) found that the bulk mass of PM was at particles with a d_p less than 2 μm. The modeled V_d values for PM_{10} in this

Table 3
Annual removal rate of air pollutants per canopy cover by different vegetation types in Chicago between August 2006 and July 2007

Type of vegetation	SO_2 ($g\,m^{-2}\,yr^{-1}$)	NO_2 ($g\,m^{-2}\,yr^{-1}$)	PM_{10} ($g\,m^{-2}\,yr^{-1}$)	O_3 ($g\,m^{-2}\,yr^{-1}$)	Total ($g\,m^{-2}\,yr^{-1}$)
Short grass	0.65	2.33	1.12	4.49	8.59
Tall herbaceous plants	0.83	2.94	1.52	5.81	11.10
Deciduous trees	1.01	3.57	2.16	7.17	13.91

The non-vegetated surfaces were excluded from the calculation.

study were comparable to the values for fine particles reported in the literature.

The removal rate was compared to the removal rate of air pollutants, including SO_2, NO_2, PM_{10}, and O_3, extracted from similar studies. The results showed that the annual removal per hectare of green roof was 85 kg ha^{-1} yr^{-1} and the annual removal per hectare of canopy cover was 97 kg ha^{-1} yr^{-1}. The annual removal per hectare of canopy cover reported in this study was higher than the removal rate of 69 kg ha^{-1} yr^{-1} estimated for green roofs in Toronto by Currie and Bass (2005). Deutsch et al. (2005) reported a removal rate of 77 kg ha^{-1} yr^{-1} for Washington, DC. As suggested by Nowak et al. (2006), the different pollution removal rates among cities can be caused by factors such as the amount of vegetation cover, pollution concentration, length of growing season, and meteorological conditions. Furthermore, the different methods used in modeling the air pollution removal by grass and large herbaceous plants in those studies also contributed to difference in results. Currie and Bass (2005) did not model grass and large herbaceous plants separately. Instead, they adjusted the estimated V_d value of air pollutants from trees to grasses by using the ratio of leaf area index (LAI) of grasses to trees (3:6). The ratio of 1:2 was supported by Shreffler (1978) study on modeled deposition velocity for SO_2 over grasslands vs. forests. However, based on the V_d values modeled in this study, and also from the observed values reported in the literature (Table 6), we found that the V_d values of air pollutants for trees may not always be two times those of grass and large herbaceous plants. Finally, the UFORE model tends to give conservative estimation of PM_{10} removal because it assumes a fixed deposition velocity of 0.064 m s^{-1} and a 50% re-suspension rate for PM_{10} (Nowak et al., 2006). As pointed out by Ould-Dada and Baghini (2001), the 50% re-suspension was much larger than the re-suspension rate they measured for fine particles. All those differences can lead to the relatively high removal rates reported in this study.

4.2. Uncertainties of the approach

The estimated air pollutant removal for green roofs in Chicago should be treated as an approximation rather than an accurate estimation of actual air pollution removal. Several uncertainties should be noted. The green roofs in Chicago were generalized as continuous surfaces of short grass, tall herbaceous plants, and deciduous trees with uniform heights. This generalization was necessary for running a big-leaf model at a city scale. Nevertheless, small-scale effects such as the differing heights of green roofs, arrangement of vegetation, and relation to the geometry of street canyons could influence turbulence and

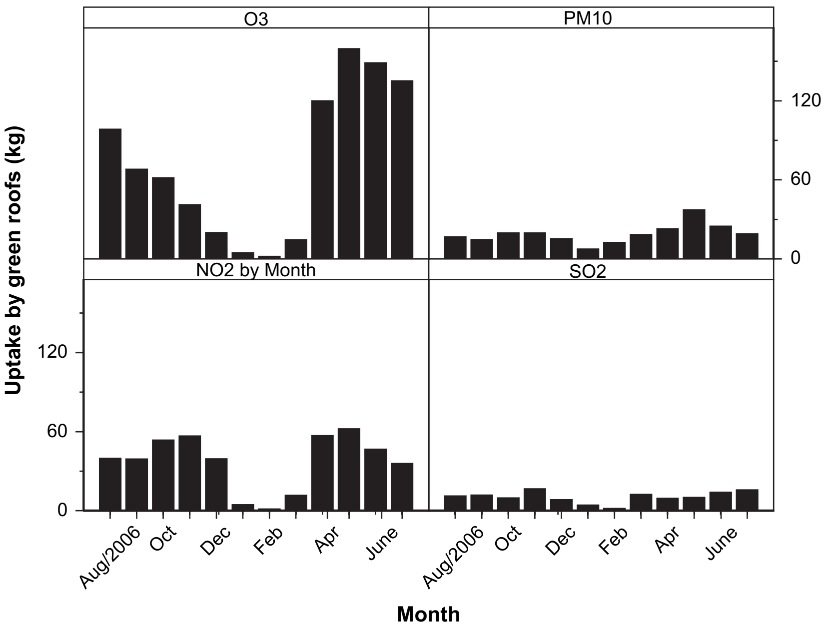

Fig. 2. Monthly uptake of air pollutants by green roofs in Chicago between August 2006 and July 2007.

Table 4
Modeled deposition velocities of pollutants over different vegetation types

Type of vegetation	SO_2 ($cm\ s^{-1}$)	NO_2 ($cm\ s^{-1}$)	PM_{10} ($cm\ s^{-1}$)	O_3 ($cm\ s^{-1}$)
Short grass	0.04 (0.005) 0.39 (0.006)	0.01 (0.001) 0.39 (0.006)	0.10 (0.005) 0.19 (0.003)	0.01 (0.001) 0.42 (0.007)
Tall herbaceous plants	0.04 (0.006) 0.48 (0.007)	0.01 (0.001) 0.49 (0.007)	0.10 (0.006) 0.25 (0.004)	0.01 (0.001) 0.54 (0.008)
Deciduous trees	0.05 (0.006) 0.57 (0.007)	0.01 (0.001) 0.58 (0.008)	0.13 (0.008) 0.36 (0.006)	0.01 (0.001) 0.65 (0.008)

The minimum and maximum monthly average deposition velocities were shown here. The numbers inside the parenthesis were standard errors.

transport in wind canopies (McDonald et al., 2007). The concentrations of air pollutants were considered uniform for the entire study area. This assumption works for situations where a well-mixed boundary layer exists in daytime under unstable conditions (Colbeck and Harrison, 1985). Nevertheless, the influence of buildings and the distances to sources of emission could cause the concentrations of air pollutants to vary spatially. Green roofs close to highly polluted streets could have higher uptake of air pollutants than those located in relatively clean areas.

Another source of uncertainty is the way the V_d was modeled. The resistance components were modeled by simplifying all plants into three prototypes: grass, crops, and deciduous trees. Values adopted from existing literatures were used to represent the vegetation characteristics. However, the differences among plant species (e.g., photosynthetic pathways, stomatal densities, LAI, growth speed) can introduce uncertainties into the estimate of V_d. In the future, more field measurements on the dry deposition velocities of pollutants on urban grass should be conducted to calibrate the dry deposition model and verify the modeling results.

Finally, green roofs can also become a source of pollutants. Pollens produced by plants and erosion of growth media under a strong wind can increase particle pollution (Tan and Sia, 2005). Plants can also emit volatile organic compounds (VOC) that can result in O_3 production (Benjamin and Winer, 1998). Those factors were not considered in this study but they can potentially lower the estimate of air pollutant removal by green roofs.

4.3. Practical considerations

It can be seen from Table 5 that a large amount of air pollutants can be removed if all roofs in Chicago were converted to green roofs. However, it was also obvious that the cost of constructing the specified area of green roofs would be prohibitively high. Compared to the cost of traditional air pollution controls, between $935 per metric ton for CO and $4482 per metric ton for NO_2 (McPherson, 1994), the green roof is not an economically viable measure in air pollution control.

Although the removal rate of 97 $kg\ ha^{-1}\ yr^{-1}$ is comparable to the removal rates for urban forests reported by Nowak et al. (2006) in 55 cities, which range between 59 $kg\ ha^{-1}\ yr^{-1}$ and 168 $kg\ ha^{-1}\ yr^{-1}$, green roofs cost more than planting trees. Based on the results of Nowak (1994), a medium size tree can remove the same amount of air pollutants as a 19 m^2 extensive green roof in one year but the planting costs for them are around $400 and $3059, respectively.

Even with their high cost, there are several reasons why the green roof is a viable alternative to trees in air pollution control. The high initial installation cost of a green roof can be justified by its long-term benefits. Benefits contributed by green roofs include reduction of storm water runoff, saving energy, reducing urban heat islands, and extending the life span of roofs (Carter and Keeler, 2007; Wong et al., 2003). Acks (2005) did a cost-benefit analysis of several planting scenarios of green roofs in New York City and found the medium benefit/cost ratio was 1.02 over a period of 55 years. The cost-benefit ratio of building green roofs can be further improved by increasing the efficiency of air pollutant removal and simultaneously lowering the construction cost. Plant species used in green roofs can be selected to increase the amount of air pollutants removed and reduce the emission of VOC (Benjamin et al., 1996). The construction and maintenance costs of a green roof can be reduced if the industry is standardized and a complete system for green roof production, delivery, and installation is formed. Currently, as estimated by Philippi (2006), the unit installation cost of the extensive green roof in the U.S. was ten times that in Germany. Furthermore, unlike tree planting programs where land has to be set aside for the plantings, green roofs do not occupy land; they are built on rooftops. This is an important factor for high-density urban communities.

Table 5
Additional air pollution removal from planting more green roofs and the estimated installation cost

Scenarios	Total air pollutants removed (metric tons)	Total installation cost ($ million)	Cost of removal ($ million/ metric ton)
Current ratio	1835.23	3086.52	1.68
Extensive only	1405.50	2201.51	1.57
Intensive only	2046.89	3522.42	1.72

5. Conclusion

Air pollution in the urban environment is a major threat to human health. As the global population is becoming more concentrated in urbanized areas, new ideas and approaches are needed to help maintain clean air that is safe for everyone to breathe. This study evaluated one such innovative approach: using green roofs for air pollution control. By using a big-leaf dry deposition model, the air pollutants removed by green roofs in Chicago were quantified. The result showed that the green roofs in Chicago can remove a large amount of pollutants from air. Currently, the green roof cannot be used as a stand-alone measure in air pollution controls because of its high cost. However, a comprehensive look at its environmental benefits shows that it can be an effective option to mitigate air pollution as well as other environmental problems.

Table 6
Observed deposition velocities of SO_2, NO_2, PM_{10}, and O_3 over different vegetation types reported in the literature

Pollutants	Vegetation (h_0 in m)	V_d Value ($cm\,s^{-1}$)	References
SO_2	Short grass (0.1)	0.2 ± 0.1–0.4 ± 0.2	Sorimachi et al. (2003)
	Grass (0.3)	0.6–0.8	Feliciano et al. (2001)
	Heathland	0.8 ± 0.4	Erisman et al. (1993)
	Grassland	1.2 ± 0.3	Erisman et al. (1993)
	Grassland (0.1–0.8)	0.4–0.7	Pio and Feliciano (1996)
	Deciduous forest	0.48 ± 0.45	Zhang et al. (2002)
	Deciduous forest (22)	0.30–1.04	Finkelstein (2001)
NO_2	Heathland	0.10–0.35	Coe and Gallagher (1992)
	Grass (0.15)	0.27 ± 0.017	Watt et al. (2004)
	Wheat	0–0.35	Pilegaard et al. (1998)
	Grassland	0.11–0.24	Hesterberg et al. (1996)
	Orchard (2.1)	0.2–0.6	Walton et al. (1997)
	Coniferous forest	0.4	Rondón et al. (1993)
O_3	Short grass (0.1)	0.2 ± 0.2–0.4 ± 0.3	Sorimachi et al. (2003)
	Grassland (0.22)	0.22–0.36	Stocker et al. (1993)
	Grass (0.1–0.8)	0.1–0.5	Pio et al. (2000)
	Mooreland	0.2–0.7	Fowler et al. (2001)
	Deciduous trees (33)	0.2–1.0	Padro (1996)
	Deciduous forest (22)	0.10–0.75	Finkelstein (2001)
PM_{10}	Grass (0.06)	0.16–0.12 ($d_p = 5$)	Chamberlain (1967)
	Nature grass (0.3–0.5)	0.22 ± 0.06	Wesely et al. (1985)
	Rye grass (0.75–1)	0.16 ± 0.072 (NGMD = 0.52)	Vong et al. (2004)
	Urban grass (0.1–0.25)	0.33–0.38 ($d_p = 0.6$–0.8)	Fowler et al. (2004)
	Urban woods (25)	0.7–1.07 ($d_p = 0.6$–0.8)	
	Deciduous trees (22)	0.1 ($d_p < 2$)	Hicks et al. (1989)
	Beach (24–25)	0.45 (NGMD[a] = 0.02–0.03)	Pryor (2006)
		0.15 (NGMD[a] = 0.06–0.07)	

[a] NGMD is the number geometrical mean diameter (μm).

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References

American Lung Association (ALA), 2007. State of the Air. Available from: http://lungusa.kintera.org/sota07pdf.

Acks, K., 2005. A framework of cost-benefit analysis of green roofs: initial estimates. Available from: http://ccsr.columbia.edu/cig/greenroofs/Green_Roof_Cost_Benefit_Analysis.pdf.

Akbari, H., 2002. Shade trees reduce building energy use and CO_2 emissions from power plants. Environmental Pollution 116, 119–126.

Baldocchi, D.D., Hicks, B.B., Camara, P., 1987. A canopy stomatal resistance model for gaseous deposition to vegetated surfaces. Atmospheric Environment 21, 91–101.

Beckett, K.P., Freer-Smith, P., Taylor, G., 1998. Urban woodlands: their role in reducing the effects of particulate pollution. Environmental Pollution 99, 347–360.

Benjamin, M.T., Winer, A.M., 1998. Estimating the ozone-forming potential of urban trees and shrubs. Atmospheric Environment 32, 53–68.

Benjamin, M.T., Sudol, M., Bloch, L., Winer, A.M., 1996. Low-emitting urban forests: a taxonomic methodology for assigning isoprene and monoterpene emission rates. Atmospheric Environment 30, 1437–1452.

Carter, T., Keeler, A., 2007. Life-cycle cost-benefit analysis of extensive vegetated roof systems. Journal of Environmental Management. doi: 10.1016/j.jenvman.2007.01.024.

Chamberlain, A.C., 1967. Transport of lycopodium spores and other small particles to rough surfaces. Proceedings of the Royal Society London 296, 45–70.

Coe, H., Gallagher, M.W., 1992. Measurements of dry deposition of NO_2 to a Dutch heathland using the eddy-correlation technique. Quarterly Journal of the Royal Meteorological Society 118, 767–786.

Colbeck, I., Harrison, R.M., 1985. Dry deposition of ozone: some measurements of deposition velocity and of vertical profiles to 100 meters. Atmospheric Environment 19, 1807–1818.

Corrie, C., Talbot, B., Bulkley, J., Adriaens, P., 2005. Optimization of green roofs for air pollution mitigation. In: Proceedings of Third Annual Greening Rooftops for Sustainable Communities Conference, Awards and Trade Show, Washington, DC, May 4–6, 2005.

Currie, B.A., Bass, B., 2005. Estimate of air pollution mitigation with green plants and green roofs using the UFORE model. In: Proceedings of Third Annual Greening Rooftops for Sustainable Communities Conference, Awards and Trade show, Washington, DC, May 4–6, 2005.

Deutsch, B., Whitlow, H., Sullivan, M., Savineau, 2005. Re-greening Washington, DC: A Green Roof Vision Based on Quantifying Storm Water and Air Quality Benefits. Available from: http://www.greenroofs.org/resources/greenroofvisionfordc.pdf.

Dunnett, N., Kingsbury, N., 2004. Planting Green Roofs and Living Walls. Timber Press, Portland.

Erisman, J.W., Versluis, A.H., Verplanke, T.A.J.W., Hann, D.D., Anink, D., van Elzakker, B.G., Mennen, M.G., van Aalst, R.M., 1993. Monitoring the dry deposition of SO_2 in the Netherlands: results for grassland and heather vegetation. General Topics. Atmospheric Environment. Part A 27, 1153–1161.

Federal Emergency Management Agency, 2005. Appendix H: Pictometry explanation. In: Evaluation of Alternatives in Obtaining Structural Elevation Data. Available from: http://www.fema.gov/business/nfip/alt_elevations.shtm.

Feliciano, M.S., Pio, C.A., Vermeulen, A.T., 2001. Evaluation of SO_2 dry deposition over short vegetation in Portugal. Atmospheric Environment 35, 3633–3643.

Finkelstein, P.L., 2001. Deposition velocities of SO_2 and O_3 over agricultural and forest ecosystems. Water Air and Soil Pollution: Focus 1, 1573–2940.

Fowler, D., Skiba, U., Nemitz, E., Choubedar, F., Branford, D., Donovan, R., Rowland, P., 2004. Measuring aerosol and heavy metal deposition on urban woodland and grass using inventories of ^{210}Pb and metal concentrations in soil. Water Air and Soil Pollution: Focus 4, 483–499.

Fowler, D., Flechard, C.R., Cape, J.N., Storeton-West, R.L., Coyle, M., 2001. Measurements of ozone deposition to vegetation quantifying the flux, the stomatal and non-stomatal components. Water Air and Soil Pollution 1, 63–74.

Gray, K.A., Finster, M.E., 2000. The Urban Heat Island, Photochemical Smog, and Chicago: Local Features and the Problem and Solution. Northeastern University, Evanston, IL. Available from: http://www.epa.gov/hiri/resources/pdf/post_chicago/chicago_toc_exsum.pdf.

Heisler, G.M., 1986. Effects of individual trees on the solar radiation climate of small buildings. Urban Ecology 9 (3/4), 337–359.

Hesterberg, R., Blatter, A., Fahrni, M., Rosset, M., Neftel, A., Eugster, W., Wanner, H., 1996. Deposition of nitrogen-containing compounds to an extensively managed grassland in central Switzerland. Environmental Pollution 91, 21–34.

Hicks, B.B., Matt, D.R., McMillen, R.R., Womack, J.D., Wesely, M.L., Hart, R. L., Cook, D.R., Lindberg, S.E., de Pena, R.G., Thompson, D.W., 1989. A field investigation of sulfate fluxes to deciduous forest. Journal of Geophysical Research 94, 13003–13011.

Hill, A.C., 1971. Vegetation: a sink for atmospheric pollutants. Journal of the Air Pollution Control Association 21, 341–346.

Lim, J.H., Sabin, L.D., Schiff, K.C., Stozenbach, K.D., 2006. Concentration, size distribution, and dry deposition rate of particle-associated metals in the Los Angeles region. Atmospheric Environment 40, 7810–7823.

Maxwell, E.L., Marion, W., Myers, D., Rymes, M., Wilcox, S., 1995. NREL/TP-463-5784. National Solar Radiation Data Base (1961–1990), Final Technical Report, vol. 2. National Renewable Energy Laboratory, Golden, CO.

Mayer, H., 1999. Air pollution in cities. Atmospheric Environment 33, 4029–4037.

McDonald, A.G., Bealey, W.J., Fowler, D., Dragstis, 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. Atmospheric Environment 41, 8455–8467.

McPherson, E.G., 1994. Benefits and costs of tree planting and care in Chicago. General technical report NE-186. In: McPherson, E.G. (Ed.), Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project. United States Department of Agriculture, Forest Service, Northeastern Forest Experimental Station, Randnor, PA, pp. 115–133.

Nowak, D.J., Crane, D.E., Stevens, J.C., 2006. Air pollution removal by urban trees and shrubs in the United States. Urban Forestry & Urban Greening 4, 115–123.

Nowak, D.J., 1994. Air pollution removal by Chicago's urban forest. General technical report NE-186. In: McPherson, E.G. (Ed.), Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project. United States Department of Agriculture, Forest Service, Northeastern Forest Experimental Station, Randnor, PA, pp. 63–81.

Offenberg, J.H., Baker, J.E., 2000. Aerosol size distributions of elemental and organic carbon in urban and over-water atmospheres. Atmospheric Environment 34, 1509–1517.

Ould-Dada, Z., Baghini, N.M., 2001. Resuspension of small particles from tree surfaces. Atmospheric Environment 35, 3799–3809.

Padro, J., 1996. Summary of ozone dry deposition velocity measurements and model estimates over vineyard, cotton, grass and deciduous forest in summer. Atmospheric Environment 30, 2363–2369.

Philippi, P.M. How to get cost reduction in green roof construction. In: Proceedings of Fourth Annual Greening Rooftops for Sustainable Communities Conference, Awards and Trade Show, Boston, MA, May 11–12, 2006.

Pilegaard, K., Hummelshoj, P., Jensen, N.O., 1998. Fluxes of ozone and nitrogen oxide measured by eddy correlation over a harvested wheat field. Atmospheric Environment 32, 1167–1177.

Pio, C.A., Feliciano, M.S., Vermeulen, A.T., Sousa, E.C., 2000. Seasonal variability of ozone dry deposition under southern European climate conditions, in Portugal. Atmospheric Environment 34, 195–205.

Pio, C.A., Feliciano, M.S., 1996. Dry deposition of ozone and sulphur dioxide over low vegetation in moderate southern European weather conditions. Measurements and modeling. Physics and Chemistry of the Earth 21, 373–377.

Pryor, S., 2006. Size-resolved particle deposition velocities of sub 100 nm diameter particles over a forest. Atmospheric Environment 40, 6192–6200.

Rondón, A., Johansson, C., Granat, L., 1993. Dry deposition of nitrogen dioxide and ozone to coniferous forest. Journal of Geophysical Research 98, 5159–5172.

Rosenfeld, A.H., Akbari, H., Romm, J.J., Pomerantz, M., 1998. Cool communities: strategies for heat island mitigation and smog reduction. Energy and Buildings 28, 51–62.

Rosenzweig, C., Solecki, W., Parshall, L., Gaffin, S., Lynn, B., Goldberg, R., Cox, J., Hodges, S., 2006. Mitigating New York City's heat island with urban forestry, living roofs, and light surfaces. In: Proceedings of Sixth Symposium on the Urban Environment, January 30–Feburary 2, Atlanta, GA. Available from: http://ams.confex.com/ams/pdfpapers/103341.pdf.

Schnelle, K.B.J., Brown, C.A., 2002. Air Pollution Control Technology Handbook. CRC Press, Boca Raton, EL.

Scott, K.I., McPherson, E.G., Simpson, J.R., 1998. Air pollutant uptake by Sacramento's urban forest. Journal of Arboriculture 24, 224–234.

Shreffler, J.H., 1978. Factors affecting dry deposition of SO_2 on forests and grasslands (1967). Atmospheric Environment 12, 1497–1503.

Sorimachi, A., Sakamoto, K., Ishihara, H., Fukuyama, T., Utiyama, M., Liu, H., Wang, W., Tang, D., Dong, X., Quan, H., 2003. Measurements of sulfur dioxide and ozone dry deposition over short vegetation in northern China – a preliminary study. Atmospheric Environment 37, 3157–3166.

Stocker, D.W., Stedman, D.H., Zeller, K.F., Massman, W.J., Fox, D.G., 1993. Fluxes of nitrogen oxides and ozone measured by eddy correlation over a shortgrass prairie. Journal of Geophysical Research 98, 12619–12630.

Tan, P.Y., Sia, A., 2005. A pilot green roof research project in Singapore. In: Proceedings of Third Annual Greening Rooftops for Sustainable Communities Conference, Awards and Trade Show, Washington, DC, May 4–6, 2005.

Taylor, D.A., 2007. Growing green roofs, city by city. Environmental Health Perspectives 115, 307–311.

United Nations Population Fund (UNFPA), 2007. State of world population 2007: unleashing the potential or urban growth. Available from: http://www.unfpa.org/swp/2007/english/introduction.html.

United States Environmental Protection Agency (US EPA), 2004. Incorporating Emerging and Voluntary Measures in a State Implementation Plan (SIP). US Environmental Protection Agency, Research Triangle Park, NC. Available from: http://www.epa.gov/ttn/oarpg/tl/memoranda/evm_ievm_g.pdf/.

Vong, R.J., Vickers, D., Covers, D.S., 2004. Eddy correlation measurements of aerosol deposition to grass. Tellus B 56, 105–117.

Walmsley, J.L., Wesely, M.L., 1996. Modification of coded parameterizations of surface resistances to gaseous dry deposition. Atmospheric Environment 30, 1181–1188.

Walton, S., Gallagher, M.W., Choularton, T.W., Duyzer, J., 1997. Ozone and NO_2 exchange to fruit orchards. Atmospheric Environment 31, 2767–2776.

Watt, S.A., Wagner-Riddle, C., Edwards, G., Vet, R.J., 2004. Evaluating a flux-gradient approach for flux and deposition velocity of nitrogen dioxide over short-grass surfaces. Atmospheric Environment 38, 2619–2626.

Wesely, M.L., 1989. Parameterization of surface resistance to gaseous dry deposition in regional scale, numerical models. Atmospheric Environment 23, 1293–1304.

Wesely, M.L., Cook, D.R., Hart, R.L., Speer, R.E., 1985. Measurements and parameterization of particulate sulfur dry deposition over grass. Journal of Geophysical Research 90, 2131–2143.

Wong, N.H., Tay, S.F., Wong, R., Ong, C.L., Sia, A., 2003. Life cycle cost analysis of rooftop gardens in Singapore. Building and Environment 38, 499–509.

World Health Organization (WHO), 2002. The World Health Report 2002: Reducing Risks, Promoting Healthy Life. WHO, Geneva.

Yang, J., McBride, J., Zhou, J., Sun, Z., 2005. The urban forest in Beijing and its role in air pollution reduction. Urban Forestry & Urban Greening 3, 65–78.

Zhang, L., Moran, M.D., Makar, P.A., Brook, J.R., Gong, S., 2002. Modeling gaseous dry deposition in AURAMS: a unified regional air-quality modeling system. Atmospheric Environment 36, 537–560.

Zhang, L., Gong, S., Padro, J., Barrie, L., 2001. A size-segregated particle dry deposition scheme for an atmospheric aerosol module. Atmospheric Environment 35, 549–560.