



Development of a distributed air pollutant dry deposition modeling framework

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ABSTRACT

A distributed air pollutant dry deposition modeling system was developed with a geographic information system (GIS) to enhance the functionality of i-Tree Eco (i-Tree, 2011). With the developed system, temperature, leaf area index (LAI) and air pollutant concentration in a spatially distributed form can be estimated, and based on these and other input variables, dry deposition of carbon monoxide (CO), nitrogen dioxide (NO₂), sulfur dioxide (SO₂), and particulate matter less than 10 microns (PM₁₀) to trees can be spatially quantified. Employing nationally available road network, traffic volume, air pollutant emission/measurement and meteorological data, the developed system provides a framework for the U.S. city managers to identify spatial patterns of urban forest and locate potential areas for future urban forest planting and protection to improve air quality. To exhibit the usability of the framework, a case study was performed for July and August of 2005 in Baltimore, MD.

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1. Introduction

Air in cities may contain high levels of pollutants that cause human health problems (Mayer, 1999). In the United States, more than 3700 deaths annually are attributable to an increase in ozone levels (Bell et al., 2004). Worldwide, the World Health Organization estimated that 800 000 deaths annually could be attributed to urban air pollutants (WHO, 2002). The United Nations Population Fund predicted that the urban population worldwide would increase from 3.3 billion in 2008 to 5 billion by 2030 (UNFPA, 2007), leading to increased mortality for urban residents. Developing solutions to control air pollutants and reduce exposure risks is a goal for cities worldwide.

Air pollutant management practices often focus on controlling emission sources of air pollutants (Schnelle and Brown, 2002). These practices effectively reduce the local emission of new air pollutants, but do not address pollutants already in the air. To remove existing air pollutants, different approaches need to be employed. One such approach is the use of urban forest that can reduce air pollutants through a dry deposition process. Due to their large leaf surface areas compared to the ground on which they stand, trees can act as biological filters, removing air pollutants and hence improve air quality (Beckett et al., 1998).

For urban forest management, it is crucial to understand the effects of the existing urban forest, and plan future planting and protection to achieve air quality and other environmental goals (Dwyer et al., 2002, 2003; Luley, 2002; Nilsson et al., 2008). The United States Department of Agriculture (USDA) Forest Service's urban forest effects model (formerly called UFORE and now integrated into i-Tree Eco (i-Tree, 2011)) provides a tool to quantify urban forest structure and forest-related effects (Nowak and Crane, 2000; Nowak et al., 2008). UFORE-D is the i-Tree Eco's program that calculates hourly dry depositions of air pollutants to tree canopies based on tree cover and hourly meteorological and air pollutant concentration data. While UFORE-D is widely used to quantify dry deposition in urban areas in North America (Currie and Bass, 2008; Deutsch et al., 2005; Nowak et al., 1998, 2000, 2006; Nowak and Crane, 2000), one limitation of UFORE-D is that the spatial distribution of urban forests is not considered. As a result, pollutant removal is estimated based on average characteristics of an area; it is not possible to assess local effects of urban forests based on their spatial distribution across an area. This limitation stems from UFORE-D's lumped parameter approach. This method assumes input parameters such as meteorology and pollutant concentrations are homogeneous over an area, and quantifies dry deposition across the area as a single value. To enhance UFORE-D's spatial ability, it is desirable to employ a distributed parameter approach in which input parameters with spatial variations are employed. This approach will allow managers to better assess and visualize the local effects of urban forests and create more detailed urban tree management plans.

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Ideally in distributed models all input parameters are available in a distributed form; however, data limitations often exist due to lack or incompleteness of measurements (Mulligan and Wainwright, 2004). As a result, most distributed models use some of their input parameters in a lumped form. This limitation exists for UFORE-D when implemented in a distributed approach. Hirabayashi et al. (2011) performed Monte Carlo with Latin hypercube sampling and Morris one-at-a-time sensitivity analyses to determine the input parameters that had the greatest impact on UFORE-D outputs. They identified temperature and leaf area index (LAI) as the most sensitive model input parameters. In addition, the amount of pollutant removed is directly dependent upon ambient pollutant concentrations. In this study, these three input parameters are distributed and employed with other lumped input parameters over the study area.

Implementing UFORE-D with a distributed approach requires dividing a study region into grid cells, applying UFORE-D within each cell, and composing a distributed result. This analysis can be streamlined by coupling UFORE-D with a geographical information system (GIS). In these circumstances, a strategy called tight coupling is often employed (Fedra, 1996). With tight coupling of a model and GIS, model functionalities are typically built within a GIS framework. Thus two originally independent systems are integrated into one system that provides a common user interface and a transparent data sharing and transfer between the model and GIS. Moreover, with functionalities offered by GIS it is possible to visualize urban forest effects on a municipal map and identify high risk areas that are potential locations for future urban forest planting and protection.

The objective of this study is to develop a distributed air pollutant dry deposition modeling framework by integrating UFORE-D into a GIS. Employing nationally available data, it provides urban forest managers in U.S. cities a framework to quantify and visualize urban forest effects for appropriate management and design plan developments. Three important input parameters for UFORE-D (i.e. temperature, LAI, and air pollutant concentration) are employed in a distributed form. Models to estimate these parameters are also integrated into the system. The model is capable of estimating concentrations and dry depositions of four criteria air pollutants (CAPs): carbon monoxide (CO), nitrogen dioxide (NO₂), sulfur dioxide (SO₂), and particulate matter less than 10 microns (PM₁₀). Using this framework, a case study in Baltimore, MD is performed, in which dry deposition of NO₂ for July and August in 2005 are spatially quantified, and future potential urban forest planting and protecting locations are visually identified.

2. Material and methods

2.1. Temperature calculation

Heisler et al. (2006, 2007) developed empirical models of air temperature differences between multiple weather stations in the city of Baltimore, MD and surrounding neighborhoods. On an hourly basis, Turner atmospheric stability classes are derived from the wind speed and cloud cover (Panofsky and Dutton, 1984), by which hourly meteorological data are stratified. With these explanatory variables as well as raster datasets representing elevation and upwind cover types (i.e. forest, impervious and water) from the National Land Cover Dataset (NLCD) 2001 (Homer et al., 2004), temperature differences between a reference site and grid cells in an area are estimated by regression analysis. Output variables are hourly air temperature (°C) for each cell.

2.2. LAI calculation

LAI is defined as one-sided leaf area of canopy divided by ground projected area of canopy. From field sampled data gathered in Baltimore in 2004, the maximum mid-season LAI can be estimated with UFORE-A, a sibling computer program of UFORE-D integrated in i-Tree Eco. With UFORE-A, leaf area of individual trees is estimated using regression equations for urban trees (Nowak, 1996), and the leaf area and tree cover percentage within six NLCD 2001 landcover types are estimated.

Landcover types employed are developed open space, developed low intensity, developed medium intensity, developed high intensity, barren/agricultural land, and forest/wetland. LAI per unit tree cover for landcover *i* can be calculated as:

$$LAI_i = \frac{LA_i}{A_i \times TC_i} \quad (1)$$

where LA_{*i*}, A_{*i*}, and TC_{*i*} are leaf area (km²), ground area (km²), and tree coverage (%) for landcover *i*, respectively.

2.3. Air pollutant concentration calculation

Air pollutant concentration is calculated based on the methods described in Morani et al. (2011). Air pollutant concentrations are modeled for two emission sources: facility stacks (point sources) and traffic on roads (line sources), and merged into one map and adjusted with monitored data in the area. This method is not designed to estimate actual air pollutant concentrations, rather the potential variabilities in concentration due to these emission sources.

Four national databases are employed to calculate air pollutant maps. The Topologically Integrated Geographic Encoding and Referencing (TIGER) road network data (TIGER, 2008), the U.S. Department of Transportation's highway statistics data (U.S. DOT, 2008), hourly meteorological measurements in 2005 obtained from National Climate Data Center (NCDC) (NCDC, 2008) and the US EPA's National Emission Inventory (NEI) for 2002 (NEI, 2008).

Air pollutant dispersions from roads are estimated in two steps. First air pollutant emissions from automobiles are estimated based on traffic volume and emission factors (Table 1), and then air pollutant dispersion is estimated with a modified General Finite Line Source Model (GFLSM) (Luhar and Patil, 1989; McHugh and Thomson, 2003):

$$C_i = \frac{Q_i}{2\sqrt{2\pi}\sigma_y\sigma_z u} \exp\left[-\frac{1}{2}\left(\frac{z_r - z_s}{\sigma_z}\right)^2\right] \left[\operatorname{erf}\left(\frac{y_r + L_i/2}{\sqrt{2}\sigma_y}\right) - \operatorname{erf}\left(\frac{y_r - L_i/2}{\sqrt{2}\sigma_y}\right) \right] \quad (2)$$

where C_{*i*} (g m⁻³) is the air pollutant concentration for road type *i*, Q_{*i*} (g s⁻¹ m⁻¹) is the pollutant emission rate per unit length for road type *i*, *u* (m s⁻¹) is wind speed, σ_{*y*} (m) and σ_{*z*} (m) are the standard deviations of lateral and vertical concentration distributions, respectively, *y_r* (m) is the crosswind distance between receptor and source, L_{*i*} (m) represents in-cell road length for road type *i*, and z_{*s*} (=0.5 m) and z_{*r*} (=1.5 m) are height of the source and receptor, respectively. *u* must be larger than 0 m s⁻¹ to estimate concentrations with Equation (2).

Emission of nitrogen in both highway statistics and NEI data are reported as oxides of nitrogen (NO_{*x*}). Air quality standards are expressed in terms of nitrogen dioxide (NO₂) because it is closely related to health effects. The concentration of NO_{*x*} estimated with the aforementioned models is converted to the concentration of NO₂ based on the empirical function for the ratio of NO₂ and NO_{*x*} (Derwent and Middleton, 1996).

Pollutants emitted from a point source can be approximated with the Gaussian dispersion equation expressed as (Zannetti, 1990):

$$C_i = \frac{Q}{2\pi\sigma_y\sigma_z u} \exp\left[-\frac{1}{2}\left(\frac{y_r}{\sigma_y}\right)^2\right] \exp\left[-\frac{1}{2}\left(\frac{h_s + \Delta h - z_r}{\sigma_z}\right)^2\right] \quad (3)$$

where C (g m⁻³) is air pollutant concentration at a receptor, Q (g s⁻¹) is pollutant emission rate from a source facility, Δ*h* (m) is emission plume rise, and *h_s* (m) is height of the source (stack height).

Several assumptions are made to employ Equations (2) and (3) for estimating air pollutant concentrations (Turner, 1994). Highway and facility emission data are provided on an annual basis. These data are converted to per-second values to be incorporated in Equations (2) and (3) and assumed to be continuous over time. The mass of emitted pollutants is assumed to remain the same in the atmosphere during transport, and no pollutants are removed through chemical reactions, gravitational settling, or turbulent impaction. The meteorological conditions are assumed to remain unchanged over the time period that the emitted pollutant travels from the source to receptors. It is assumed that the time averaged concentration profiles at any distance in both the crosswind and vertical directions are well represented by

Table 1

Emission factors obtained from U.S. Environmental Protection Agency (US EPA, 1998) for CO and NO_{*x*}, and from EPA's highway vehicle particulate emission modeling software, PART5 (US EPA, 2009a) for PM₁₀ and SO₂.

Road type	Emission factor (g miles ⁻¹)			
	CO	NO _{<i>x</i>}	PM ₁₀	SO ₂
Interstate highway (A1)	7.40	2.58	0.096	0.113
Other freeway and expressway (A2)	10.58	2.02	0.096	0.113
Other principal arterial (A3)	10.58	2.02	0.096	0.113
Local road (A4)	20.52	2.02	0.095	0.113

a Gaussian distribution. In addition, ambient background concentration from other areas and previous hours are not considered. Other potential sources such as homes and vegetation were also not considered. Because of these assumptions, this method aims to estimate air pollutant concentrations in a relative sense (e.g. to identify hotspots).

Since UFORE-D is ultimately intended to assess effects of air quality on human's health, a receptor height (z_r) of 1.5 m, which is the reported human air aspiration height (Vaitiekūnas and Banaitytė, 2007), is used. It can be assumed that the vertical profile of the air pollutant concentration is relatively uniform in the daytime when the atmosphere is unstable and thus is well mixed within the atmospheric boundary layer (Colbeck and Harrison, 1985; Dop et al., 1977; Hov, 1983). Therefore, our model limits its analysis to periods with an unstable atmosphere to ensure that the estimated concentration at 1.5 m may be comparable to that at the canopy height of urban forests. σ_y and σ_z in Equations (2) and (3) are empirically calculated based on atmospheric stability (Green et al., 1980). Emission plume rise, Δh , can be calculated from the buoyancy flux of emitted gas (Briggs, 1969, 1971, 1974).

For a given hour, hourly air pollutant concentration maps separately created for facility stacks and the four road types are merged into one map by taking the summation of values in each cell. The estimated concentrations are averaged for multiple hours and a concentration adjustment is performed with measured data averaged for the same time period. Air pollution concentration data employed were measured in 2005, and obtained from the U.S. Environmental Protection Agency (EPA)'s Air Quality System (AQS) (US EPA, 2009b). If multiple monitoring sites exist in an urban area of interest, the area is divided into Thiessen (Voronoi) polygons that define individual areas of influence around each monitoring site. Thiessen polygons are mathematically defined by the perpendicular bisectors of lines between all points (DeMers, 2000). Within each Thiessen polygon, the average difference between the measured concentration and the estimated concentration of the cell on which the monitoring site resides is considered the background concentration:

$$C_b = C_m - C_{ij} \quad (4)$$

where C_b (g m^{-3}), C_m (g m^{-3}), and C_{ij} (g m^{-3}) represents average air pollutant background concentration, average measured air pollutant concentration, and average estimated air pollutant concentration at cell (i, j) where the monitoring site resides, respectively. C_b is attributable to unidentified or natural emission sources, resuspension of past emissions, and long-range transport (US EPA, 2012). C_b may be negative in cases where the air pollutant emitted within a city is transported by winds across the city boundary. Values of all Thiessen polygon cells are then adjusted with this background concentration:

$$\text{Adj}C_{ij} = C_b + C_{ij} \quad (5)$$

where $\text{Adj}C_{ij}$ (g m^{-3}) represents the adjusted air pollutant concentration at cell (i, j).

2.4. Air pollutant dry deposition calculation

Air pollutant dry deposition is calculated based on the methods described in Hirabayashi et al. (2011). The distributed version of UFORE-D estimates dry deposition of air pollutants to trees on an hourly basis using distributed temperature, pollutant concentration, LAI, and other lumped meteorological parameters. UFORE-D estimates pollutant flux, F ($\text{g m}^{-2} \text{s}^{-1}$), as a product of the dry deposition velocity, V_d (m s^{-1}), and the air pollutant concentration, C (g m^{-3}):

$$F = V_d \cdot C \quad (6)$$

Dry deposition velocity can be estimated as the inverse of the sum of resistances to pollutant transport (Baldocchi et al., 1987):

$$V_d = (R_a + R_b + R_c)^{-1} \quad (7)$$

where R_a represents air movement resistance in the crown space (aerodynamic resistance), R_b represents transfer resistance through the boundary layer immediately adjacent to canopy surfaces (quasi-laminar boundary layer resistance), and R_c represents the chemical and biological absorption capacity of the canopy surfaces (canopy resistance). Estimates of R_a and R_b are calculated using standard resistance formulas (Dyer and Bradley, 1982; Killus et al., 1984; Pederson et al., 1995; US EPA, 1995; van Ulden and Holtslag, 1985; Venkatram, 1980) with meteorological data. Canopy resistance values for NO_2 and SO_2 are calculated based on a modified hybrid of the big-leaf and multilayer canopy deposition models (Baldocchi, 1988; Baldocchi et al., 1987). Canopy resistance has four components: stomatal resistance (r_s), mesophyll resistance (r_m), cuticular resistance (r_t), and soil resistance (r_{soil}):

$$\frac{1}{R_c} = \frac{1}{r_s + r_m} + \frac{1}{r_t} + \frac{1}{r_{\text{soil}}} \quad (8)$$

r_m was set to 0 s m^{-1} for SO_2 (Wesely, 1989), and 100 s m^{-1} for NO_2 (Hosker and Lindberg, 1982) to account for the difference between water vapor and NO_2 transport within mesophyll air spaces, and to ensure the V_d calculated was in the typical range reported by Lovett (1994). r_t was set to 20 000 s m^{-1} for NO_2 based upon Wesely (1989) assuming mixed forest in midsummer, and calculated as

8000 s m^{-1} for SO_2 to account for the typical variation in r_t exhibited among the pollutants (Lovett, 1994; Taylor et al., 1988). r_{soil} was set to 2000 s m^{-1} (Meyers and Baldocchi, 1993). Using a canopy radiative transfer model and a radiation interception model in which the canopy is divided into N layers, r_s is calculated based upon photosynthetic active radiation (PAR) on sunlit and shaded leaves and LAI for each layer (Baldocchi, 1994; Baldocchi et al., 1987; Farquhar et al., 1980; Harley et al., 1992; Hirabayashi et al., 2011; Norman, 1980; Norman, 1982; Weiss and Norman, 1985). All of these values are consistent with those employed by Hirabayashi et al. (2011).

As removals of CO and PM10 by vegetation are not directly related to transpiration, R_c for CO was set to a constant for the in-leaf season (50 000 s m^{-1}) and the out-of-leaf season (1 000 000 s m^{-1}) based on data from Bidwell and Fraser (1972). For PM10, the median deposition velocity (Lovett, 1994) was set to 0.0064 m s^{-1} based on a 50-percent resuspension rate of particles back to the atmosphere (Zinke, 1967). The base V_d was adjusted according to actual LAI and a surface-area index for bark of 1.7 (m^2 of bark per m^2 of ground surface covered by the tree crown) (Whittaker and Woodwell, 1967).

3. Results and discussion

To demonstrate the functionality of the developed modeling framework, temperature, LAI, concentration and dry deposition of NO_2 were estimated for Baltimore, MD. The period from July 1st to August 31st in 2005 was chosen for the analyses because dry depositions to trees are higher in the in-leaf season (i.e. March 21st–October 20th for Baltimore in 2005) due to active photosynthesis. Our model limits its analytical conditions to hours with very unstable atmospheric conditions (Turner Class 1) and no-precipitation to satisfy model assumptions of a fully mixed boundary layer and to limit to condition of dry deposition. Records not meeting these conditions were eliminated from the further analyses, resulting in 160 hourly records.

3.1. Temperature calculation

Temperature raster grids were created for the 160 h (Fig. 1 (a)) Temperature generally exhibited an urban heat island effect, in which the temperature was highest in the downtown Baltimore area and gradually decreased towards suburban areas.

3.2. LAI calculation

The maximum mid-season LAI for the six landcover types were estimated based on UFORE-A outputs (Table 2). Smaller LAI values were observed in the downtown Baltimore area, where the developed high intensity and barren/agricultural landcover types are dominant (Fig. 1(b)). Square shaped patches with high LAI found in downtown corresponded to a developed open space landcover type. Suburban areas tended to have larger LAIs due to forest/wetland and developed open space landcover types.

3.3. Air pollutant concentration calculation

Radiated highways (A1, A2, and A3) run from downtown to suburban areas, while a complex system of local roads (A4) exists in both downtown and suburban areas (Fig. 1(c)). NO_2 dispersions from each of the four road types for the 160 h were estimated. As buildings in the urban area may alter the dispersion of pollutants from motor vehicles, the dispersion buffer was limited to 30 m.

NO_2 dispersions from facilities were also estimated for the 160 h. The study area contains 368 facility stacks such as chemical, construction materials, food, wood, heavy industry manufacturing and utility and medical services. A dispersion buffer of 4000 m was chosen since the concentrations became almost 0 within this distance for all sources. Raster grids representing the hourly distribution of NO_2 concentrations from road traffic and facility emissions were merged.

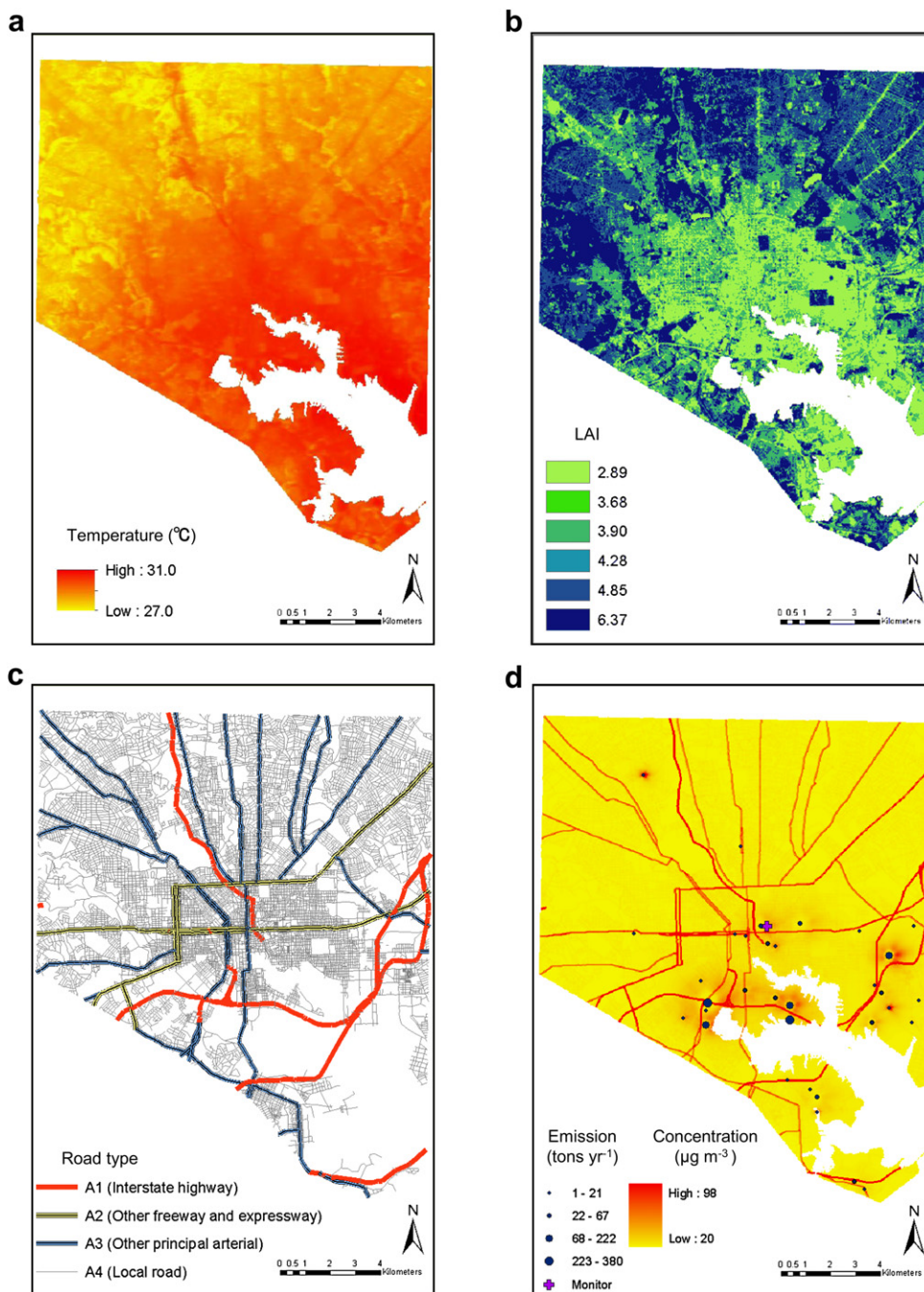


Fig. 1. (a) Temperature estimated for July 1st, 2005 at 9:00 AM, (b) LAI, (c) road networks, (d) facility locations with annual NO_x emission rates and NO_2 concentrations averaged for 160 h analysis period.

A raster representing the average NO_2 concentration over the 160 rasters was created, resulting in concentrations ranged from 0.04 to $73.4 \mu\text{g m}^{-3}$ across the study area. There is only one NO_2 monitoring site in Baltimore, and the average estimated concentration at the cell where the monitoring site exists was $2.0 \mu\text{g m}^{-3}$, while the average measured value over the same period was $26.3 \mu\text{g m}^{-3}$. Based on this difference, all cells in Baltimore were adjusted with Equations (4) and (5), resulting in the NO_2 concentrations ranging from 24.3 to $97.7 \mu\text{g m}^{-3}$ (Fig. 1(d)).

Since a major source of NO_2 is automobiles, high NO_2 concentrations were found along highways, while local traffic contributed less to the concentrations. High NO_2 concentrations radiating from

Table 2

LAI, mean V_d and range of F for NO_2 (calculated across 160 rasters) for six landcover types.

Landcover type	LAI	Mean of V_d (cm s^{-1})	Range of F ($\text{mg m}^{-2} \text{h}^{-1}$)
Developed high intensity	2.89	0.41	0.35–0.97
Developed medium intensity	3.68	0.49	0.42–1.57
Barren/agricultural land	3.90	0.51	0.44–1.60
Developed low intensity	4.28	0.55	0.47–1.62
Forest/wetland	4.85	0.60	0.51–1.42
Developed open space	6.37	0.72	0.61–2.26

some facility stacks were also observed. These facilities emit more than 100 tons of NO_x annually. This NO_2 dispersion pattern is consistent with wind direction and speed in the analysis period (Fig. 2). Winds were mainly from the West and Northwest, and these winds drove NO_2 dispersion to the east of facilities.

In our model, air pollutant concentration is estimated based on air pollutant emission and dispersion. Major factors affecting emission and dispersion are anthropogenic factors (transportation volume and industrial activities) and meteorological factors (wind and atmospheric stability), respectively. Interactions between these and other factors such as plant activities create different seasonal and diurnal variations in CO, NO_2 , PM10 and SO_2 concentrations (Atkins and Lee, 1995; Chen et al., 2001; Hargreaves et al., 2000; Nowak et al., 2006; US EPA, 2010). In addition, depending on study area's geographical factors such as terrain (surface roughness) and wind directions relative to emission sources and measuring points, trends in air pollutant concentrations may greatly vary (Zoras et al., 2006).

Our model is only capable of estimating air pollutant dispersion when the atmosphere is very unstable in daytime. Under such conditions, CO and NO_2 concentrations may generally be lower due to well mixed and diluted air (Ashrafi and Hoshyaripour, 2010; Garnett, 1979; Katsoulis, 1996). Measured CO and NO_2 concentrations exhibited this general trend (Table 3). Because of the model limitation, NO_2 dispersion estimates in Fig. 2 did not capture the worst air quality conditions. The same restriction applies to CO dispersion estimates. For SO_2 , though, the concentrations were higher under unstable conditions, and PM10 concentrations showed no distinct difference due to atmospheric conditions (Table 3). The developed model therefore may capture periods with the worst air quality for SO_2 and PM10.

The US EPA has set national ambient air quality standards (NAAQS) for CAPs considered harmful to public health and the environment (US EPA, 2009c). The NAAQS level of NO_2 is defined as $100 \mu\text{g m}^{-3} \text{ yr}^{-1}$. Areas where air pollution levels persistently exceed the NAAQS are designated as nonattainment areas by the US EPA. Since Baltimore is not designated as nonattainment, the estimated concentrations shouldn't exceed the standard levels. Estimated hourly NO_2 concentration ranged from 24.3 to $97.7 \mu\text{g m}^{-3}$

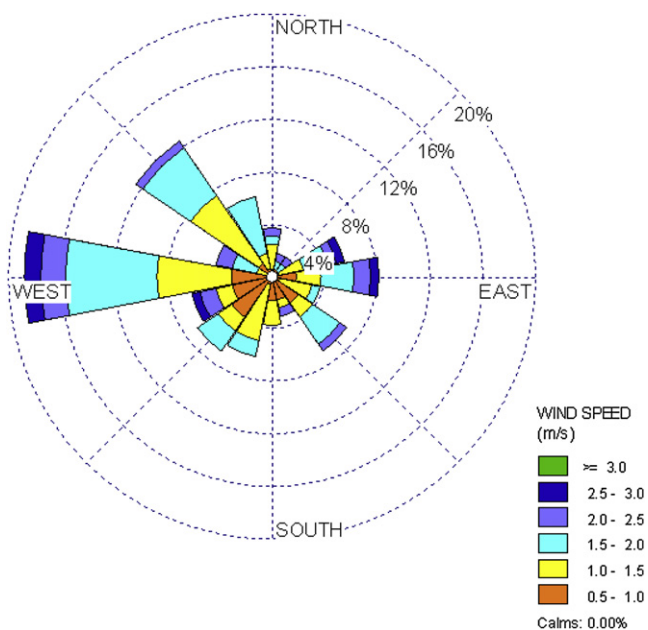


Fig. 2. Frequency of occurrence of wind speed and direction over the 160 h in July and August, 2005 in Baltimore.

Table 3

Average concentrations measured in July and August, 2005 in Baltimore, MD. Stability classes were determined based on cloud cover, ceiling height, solar elevation and wind speed (US EPA, 1995). Stability classes from 1 to 3 occur in daytime, while 5 to 7 occur in nighttime. Stability class 4 occurs both in daytime and nighttime.

Stability class	Average concentration ($\mu\text{g m}^{-3}$)			
	CO	NO_2	PM10	SO_2
1 (very unstable)	297.1	27.0	32.6	23.6
2 (unstable)	307.4	30.4	33.1	20.5
3 (slightly unstable)	337.8	33.1	33.8	17.0
4 (neutral)	383.0	35.1	33.4	13.5
5 (slightly stable)	380.1	40.2	33.6	8.7
6 (stable)	397.2	42.2	33.4	8.2
7 (very stable)	381.2	47.1	33.3	8.3

for the period of July and August 2005, and thus maximum estimated concentrations for this period were estimated to be below the yearly NO_2 standard.

3.4. Air pollutant dry deposition calculation

Both R_a and R_b exhibited very slight variations caused by temperature, in which larger resistances generally correspond to lower temperature (Fig. 3 (a)). R_c , V_d , and F generally reflected the same spatial pattern as LAI (Fig. 3(b), (c) and (d)). Since air pollutant transportation could be enhanced with larger leaf areas, cells with larger LAI resulted in smaller R_c and thus larger V_d and F . For the averaged V_d and F rasters, mean of V_d and range of F for cells with a given LAI value were calculated (Table 2). As shown, both V_d and F are dependent upon LAI values.

To access future urban forest planting and protection needs, areas where V_d was small despite high concentrations were visually identified. V_d and concentration raster grids averaged over 160 h were employed in this analysis. For each raster, percentiles of cell values were calculated and areas with a small percentile of V_d and a large percentile of concentration were determined (Fig. 4). Hotspots are found along highways and areas surrounding NO_2 emitting facilities. It should be noted that these hotspots are only representative for relatively low NO_2 concentrations estimated during very unstable hours by our dispersion model. Although they represent the potential future planting and protecting areas, additional areas may be identified when our dispersion model is upgraded to address atmospheric stabilities other than very unstable conditions. It is expected the air quality in Baltimore can be improved by planting more trees in these areas, though future planting possibilities may be limited because these areas correspond to developed medium and high intensity landcover types. It may not be easy to plant trees in these impervious areas; however, as an alternative, the feasibility of green roofs has been studied in urban areas (Currie and Bass, 2008; Deutsch et al., 2005; Yang et al., 2008). This approach may be adopted in Baltimore.

3.5. Uncertainties and limitations of the modeling system

While the developed modeling framework provides urban forest managers a very useful tool to quantify and visualize urban forest effects, the analysis in Baltimore should be treated as an approximation rather than an accurate estimation of actual processes. Several uncertainties should be noted, which are a combination of uncertainties in input variables, choice of model, and model parameterization.

The temperature regression model employed was developed using more than 3000 h of meteorological data from seven weather stations and more than 130 potential explanatory variables in

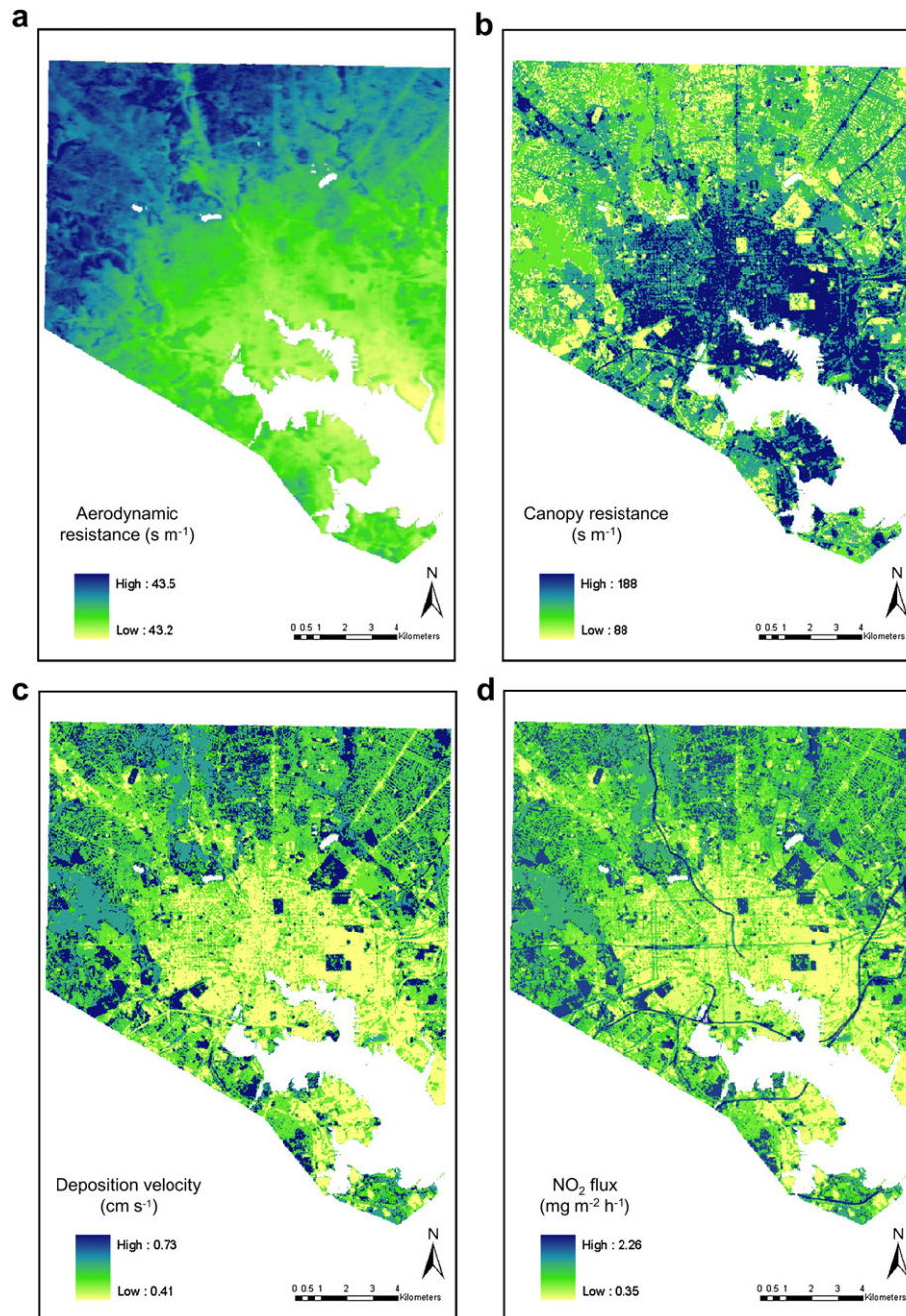


Fig. 3. (a) R_a and (b) R_c estimated for July 1st, 2005 at 9:00 AM, (c) V_d and (d) F estimated for NO_2 averaged over 160 h analysis period.

Baltimore. Nonetheless, the coefficient of determination (R^2) was fairly low, ranging from 0.27 to 0.49 (Heisler et al., 2007). The model could be improved with data from a greater number of weather stations, higher resolution land cover data, and incorporation of a vertical-dimension analysis (Heisler et al., 2007).

Estimating pollutant concentrations at the local scale is technically difficult because it requires knowledge of spatial and temporal variability of pollutant concentrations at a small scale (Jerrett et al., 2007). Methods for estimating spatial patterns of pollution concentration include interpolation of concentration taken from existing monitoring networks (Wong et al., 2004), statistical regressions of observed concentrations with surrounding land use, traffic characteristics (Brauer et al., 2003; Briggs et al.,

1997; Ollinger et al., 1993; Ross et al., 2006) and meteorological processes (Ainslie et al., 2008; Jerrett et al., 2007). However, due to the insufficient density of the monitoring network, small scale concentration variability cannot be resolved with these methods. In this study, therefore, dispersion modeling techniques were employed. One major drawback of this technique is its reliance on detailed spatial and temporal emissions inventories that are known to have large uncertainties (Hanna et al., 2001). In this study, the temporal resolutions of the road and facility emission data employed were originally annual and downscaled to per-second values to be incorporated in the dispersion equations (Equations (2) and (3)). Since weekly or diurnal variations of emissions due to driving and facility operational patterns were not taken into

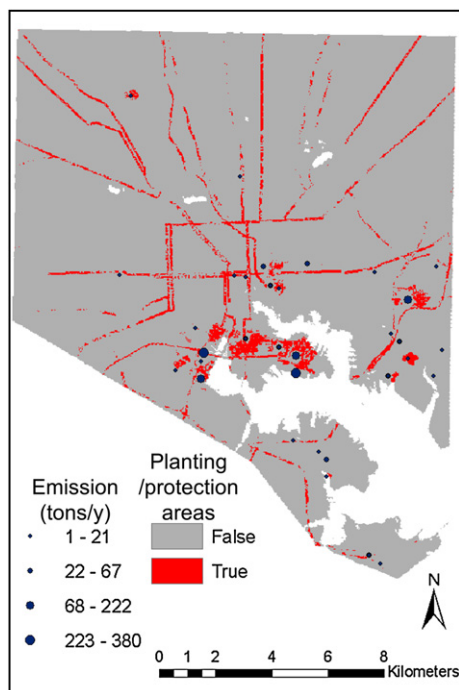


Fig. 4. NO_x emission rate and potential planting and protection areas. Cells smaller than the 30th percentile of V_d (0.49 cm s^{-1}) and larger than the 90th percentile of concentration ($25.6 \mu\text{g m}^{-3}$) are highlighted.

consideration in the downscaling process, the air pollutant concentrations may have large uncertainties, with times of larger emissions overlooked. In addition, street canyon effects (Wang et al., 2008) were ignored and background concentrations (Jensen et al., 2001) were not fully addressed in the model, thus air pollutant concentrations were generally underestimated.

The developed modeling framework has several limiting analytical conditions. Wind and atmospheric stability are the primary factors impacting the dispersion model. Wind speed must be larger than 0 to utilize dispersion equations. To assess the concentrations when there is no wind, a windless model (Jin and Fu, 2005) needs to be added to the framework. To assume the concentrations estimated with the dispersion models are vertically representative in the air, only hours with a very unstable atmosphere are processed. As unstable conditions commonly develop on sunny days with low wind speed (US EPA, 1995), the influences of variations in wind speed may be minimal. On the other hand, wind direction may vary within the temporal resolution of the system (1 h) when the atmosphere is very unstable. This variation in wind direction may affect the dispersion estimates. In our analysis, we took average of 160 h of dispersion estimates to balance out these random effects of wind direction. To handle other atmospheric conditions, vertical profiles of the concentrations need to be accounted for in the system. Due to difficulties in estimating all UFORE-D input parameters in a distributed form, the spatial distributions of only three parameters were estimated in this study. More parameters in a distributed form may lead to model improvements. Such parameters include relative humidity that was found to have a linear influence on dry depositions, and PAR and wind speed that were influential up to its threshold (Hirabayashi et al., 2011). Jerrett et al. (2007) estimated wind fields with an interpolation technique.

Measured regional dry deposition estimates are not available in Baltimore to validate the modeled results and confirm their uncertainties. Dry deposition measurement networks currently

operational in the United States, such as the Clean Air Status and Trends Network (CASTNet) (Clarke et al., 1997) and the Atmospheric Integrated Research Monitoring Network (AIRMoN) (Hicks et al., 2001) are estimating dry depositions based on an inferential method similar to those implemented in UFORE-D with local meteorological data. In addition, all of these sites are located in rural areas to avoid inputs from local pollutant sources. The Air Resource Laboratory (ARL) team at National Oceanic and Atmospheric Administration (NOAA) has developed a movable system for direct measurement of dry deposition fluxes (ARL, 2008). When these systems become more common in the future, calibration and validation exercises of modeled dry deposition processes will be improved.

4. Conclusions

In this study a distributed air pollutant dry deposition modeling framework coupled with a GIS was developed. With the developed system, distributed temperature, LAI, NO₂ concentration and dry deposition were estimated for Baltimore, MD. Based on the model development and case study, the following conclusions were reached:

1. Estimation of the dry deposition processes and its input parameters in a distributed form can be performed within one integrated system that can aid in urban forest management and planning.
2. As the developed framework is based on nationally available input data in the United States, the method is transferable to any U.S. city.
3. Future planting and protection spots can be visually identified at cells with a combination of small V_d and large concentration.

While a number of simplifying assumptions were made to develop the system, the modeling framework presented provides a prototype to aid forest managers to determine the most appropriate area to plant and protect trees to help improve air quality. Future analyses will explore the tradeoffs between model complexity and ease of use, and many of the assumptions made in this paper will be evaluated. The ultimate goal is to develop a model which can be employed by urban planners and managers with limited computational requirements. A system based on this development will evolve into a new i-Tree tool called i-Tree Landscape.

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