The Urban Forest Effects (UFORE) Model: Quantifying Urban Forest Structure and Functions

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Abstract.—The Urban Forest Effects (UFORE) computer model was developed to help managers and researchers quantify urban forest structure and functions. The model quantifies species composition and diversity, diameter distribution, tree density and health, leaf area, leaf biomass, and other structural characteristics; hourly volatile organic compound emissions (emissions that contribute to ozone formation) throughout a year; total carbon stored and net carbon sequestered annually; and hourly pollution removal by the urban forest and associated percent improvement in air quality throughout a year.

Few people know how many and what kind of trees are found in urban areas, or how these trees affect a city's environment and the health and well-being of its inhabitants. Measuring the urban forest is one of the first steps toward understanding this resource and developing appropriate management plans. Understanding and quantifying the impact of urban trees is an important prerequisite to managing city vegetation for optimal beneficial effects.

The Urban Forest Effects (UFORE) computer model was developed to help managers and researchers quantify urban forest structure and its functions. Written in SAS, the program incorporates vegetation data and local hourly meteorological and pollution-concentration measurements to quantify city-specific vegetation structure and functions. The model currently has four modules: (1) UFORE-A: Anatomy of the Urban Forest, which quantifies urban forest structure (e.g., species composition, tree density, tree health, leaf area, leaf biomass); (2) UFORE-B: Biogenic Volatile Organic Compound (VOC) Emissions, which quantifies hourly urban forest VOC emissions (emissions that contribute to ozone formation); (3) UFORE-C: Carbon Storage and Sequestration, which calculates total carbon stored and net carbon sequestered annually by urban trees; and (4) UFORE-D: Dry Deposition of Air Pollution, which quantifies hourly pollution removal by the urban forest and associated percent improvement in air quality. Pollution removal is calculated for ozone, sulfur dioxide, nitrogen dioxide, carbon monoxide, and particulate matter less than 10 microns. UFORE methods can be applied to areas of any size and to non-urban areas. Model results have been cross-checked and verified against test data sets and field measurements.

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DATA REQUIREMENTS

Four types of data are needed to run all four UFORE modules: field, tree cover, meteorological, and pollution concentration data. Field data are collected within randomly located 0.04-ha plots throughout the study area, stratified by land-use type. Within each plot, data are collected on land use, ground and tree cover, shrub characteristics, and building features, as well as on individual-tree attributes of species, stem diameter at breast height (1.37 m), tree height, height to base of live crown, crown width, percent crown dieback, and distance and direction from buildings (to be used in a module under development: UFORE-E: Energy Effects on Buildings). Although tree cover measurements are made in the field, tree cover attributes often are also measured by sampling aerial photographs (Nowak *et al.* 1996).

Running the UFORE-B and D modules requires digital hourly meteorological data from the National Climatic Data Center (NCDC) (U.S. Department of Commerce 1970). UFORE-D also requires hourly pollution-concentration data from the U.S. Environmental Protection Agency's (EPA) Aerometric Information Retrieval System (AIRS database).

UFORE-A: ANATOMY OF THE URBAN FOREST

UFORE-A uses field data to quantify forest structure of an entire urban area, individual land uses, and/or of individual tree species. It quantifies total or means as well as standard errors (Cochran 1977) of number of trees, species composition, tree density, dbh and condition-class distribution, leaf area, and leaf biomass. This module also calculates species richness, Shannon-Wiener diversity index values, population distribution by region of origin (e.g., percent native species), and ground-cover type distribution. The proportions of leaf area and live tree population in various susceptibility classes to gypsy moth feeding (Liebhold *et al.* 1995) also are calculated to reveal an urban forest's susceptibility to gypsy moth defoliation.

Leaf area and leaf biomass of individual trees are calculated using regression equations for deciduous urban trees (Nowak 1996). For deciduous trees whose size is beyond the appropriate input range for the regression equation (typically less than 10 percent of the population), leaf area is estimated using extrapolations of calculated leaf-area indices (LAI) for similar trees. For trees that are too large, average LAI is calculated by the regression equation for the maximum allowable tree size based on the appropriate height-width ratio and shading coefficient class of the tree. This calculated LAI is applied to the ground area covered by the crown of the specific tree to calculate its leaf area. For trees with height-to-width ratios that are too large or too small, tree height or width is scaled downward or upward to allow the crown to reach the maximum (2) or minimum (0.5) height-to-width ratio; the appropriate LAI for the maximum or minimum ratio is used to determine total leaf area. Leaf area is then scaled proportionally to obtain the original crown volume.

For all conifer trees, average LAI per height-to-width class for deciduous trees with a shading coefficient of 0.91 is applied to the tree's ground area to calculate leaf area. The 0.91 shading coefficient class is believed to be the best class to represent conifers because conifer forest LAI typically is about 1.5 times greater than deciduous forest stands (Barbour *et al.* 1980). The average shading coefficient for deciduous trees is 0.83 (Nowak 1996); 1.5 times the 0.83 class LAI is equivalent to the 0.91 class LAI.

Tree leaf biomass is calculated by converting leaf-area estimates using species-specific measurements from the literature (see Nowak 1994a) of g leaf dry weight per m^2 of leaf area. Shrub leaf biomass is calculated as the product of the crown volume occupied by leaves (m^3) as measured in the field and measured leaf biomass factors ($g\ m^{-3}$) for individual species (e.g., Nowak 1991, Winer *et al.* 1983). Shrub leaf area is calculated by converting leaf biomass to leaf area based on measured species conversion ratios ($m^2\ g^{-1}$). If there are no leaf biomass-to-area conversion factors or leaf biomass factors for an individual species, genera or hardwood/softwood averages are used. Estimates of leaf area and leaf biomass are adjusted downward based on percent crown dieback (tree condition).

Urban Forest Structure

The UFORE model has been used to quantify the urban forest structure in five cities (approximately 200 plots per city): Atlanta, GA, Baltimore, MD, Boston, MA, New York, NY, and Philadelphia, PA. Number of trees, tree cover, and tree density are highest in Atlanta (table 1). New York will be used to illustrate specific model results. The most common trees in New York are tree of heaven (*Ailanthus altissima*, 468,000 trees, 9 percent of total city tree population), black cherry (*Prunus serotina*, 424,000 trees, 8.1 percent), and sweetgum (*Liquidambar styraciflua*, 411,000 trees, 7.9 percent). Tree species that dominate the city in terms of leaf area are London planetree (*Platanus* x *acerifolia*, 14.3 percent of total city leaf area), sweetgum (10.1 percent), and northern red oak (*Quercus rubra*, 9.5 percent). These and other structural attributes are used to quantify various urban forest functions. An understanding of urban forest structure and how it impacts forest functions can lead to better urban forest management.

UFORE-B: BIOGENIC VOLATILE ORGANIC COMPOUND EMISSIONS

Trees emit volatile organic compounds (VOC's) that can contribute to the formation of ozone and carbon monoxide

Table 1.—*Estimates of number of trees and tree density (trees ha^{-1}) for cities analyzed with the UFORE model. Estimates of percent tree cover are based on satellite images or sampling of aerial photographs. Data from Oakland, CA (Nowak, 1991) and Chicago, IL (Nowak, 1994a) were not analyzed with UFORE (SE = standard error).*

City	Number of trees		Tree density		Tree cover (%)	
	Total	SE	Mean	SE	Mean	SE
Atlanta, GA	9,420,000	749,000	276	22	32.9	na
New York, NY	5,220,000	719,000	65	9	16.6	0.3
Chicago, IL	4,130,000	634,000	68	10	11.0	0.2
Baltimore, MD	2,600,000	406,000	109	17	18.9	na
Philadelphia, PA	2,110,000	211,000	62	6	21.6	0.4
Oakland, CA	1,590,000	51,000	120	4	21.0	0.2
Boston, MA	1,180,000	109,000	83	8	21.2	0.4

na = not analyzed; base data for Atlanta from American Forests; base data for Baltimore from Grove (1996).

(Brasseur and Chatfield 1991). The amount of emissions depends on tree species, leaf biomass, air temperature, and other environmental factors. UFORE-B estimates the hourly emission of isoprene, monoterpenes, and other volatile organic compounds (OVOC's) by species for each land use and for the entire city. Species leaf biomass (from UFORE-A) is multiplied by genera-specific emission factors (e.g., Benjamin *et al.* 1996, Geron *et al.* 1994) to produce emission levels standardized to 30°C and PAR flux of 1000 μmol m^{-2} s^{-1} (full sunlight). If genera- specific information is not available, median emission values for the family or order are used. Standardized emissions are converted to actual emissions based on light and temperature correction factors (Geron *et al.* 1994) and local meteorological data.

Urban Forest Influence on VOC Emissions

The species composition and leaf biomass of urban trees directly affect VOC emissions. Model results of planting of additional low VOC-emitting genera have been shown to decrease ozone concentrations in the Los Angeles Basin (Taha 1996). Shifting from high VOC-emitting genera may be another strategy to help improve air quality. High-emitting genera (emissions >35 μg g^{-1} leaf weight) include *Liquidambar* spp., *Quercus* spp., *Platanus* spp., *Populus* spp., *Rhamnus* spp., and *Salix* spp. The proportion of total tree leaf biomass in high-emitting genera was highest in New York (44.3 percent; 35,500 t leaf biomass in high-emitting genera), followed by Atlanta (29.7 percent; 27,900 t), Chicago (29.4 percent; 11,200 t), Baltimore (25.4 percent; 7,300 t); Boston (22.1 percent; 3,400 t), and Philadelphia (21.0 percent; 6,300 t). The three most dominant trees in New York (London planetree, northern red oak, and sweetgum) are high emitters and make up 35 percent of the city's total leaf biomass.

UFORE-C: CARBON STORAGE AND SEQUESTRATION

Carbon dioxide is a dominant greenhouse gas, and trees, through their growth process, sequester and store carbon (from carbon dioxide) in their tissue. Thus, trees are one potential means to help reduce atmospheric carbon dioxide. To calculate current carbon storage, biomass for each measured tree is calculated using allometric equations from the literature. If there is more than one equation, the mean of the biomass equation results is used. If there is no allometric equation for an individual species with a specific dbh and/or height, the average of results from appropriate equations of the same genera is used. If there are no genera equations, biomass is computed separately for each hardwood and conifer equation, and the average result from the hardwood or conifer group is used. For large trees (>97 cm dbh), volumetric equations from Hahn (1984) and specific gravity of wood are used to estimate biomass.

Biomass equations differ in what portion of tree biomass is calculated and in whether fresh or oven-dry weight is estimated. Equations that predict aboveground biomass are divided by 0.78 to convert to whole-tree biomass, as average belowground biomass of trees is approximately 22 percent of total tree biomass (see Nowak 1994b). Equations that compute fresh-weight biomass are multiplied by species- or genera- specific-conversion factors from the literature to yield dry-weight biomass. Open-grown, maintained trees tend to have less biomass than predicted by forest-derived biomass equations (Nowak 1994b). To adjust for this difference, biomass results for urban trees are multiplied by 0.8. No adjustment is made for trees found in more natural stand conditions (e.g., on vacant lands or in forest preserves). For dead and dying trees, leaf biomass is removed from the estimate of total tree biomass. Total tree and shrub dry-weight biomass is converted to total stored carbon by multiplying by 0.5 (e.g., Chow and Rolfe 1989).

Urban Tree Growth and Carbon Sequestration

To estimate the gross amount carbon sequestered annually, average diameter growth from the appropriate genera and diameter class is added to the existing tree diameter (year x) to estimate tree diameter in year $x+1$. For trees in forest stands, average dbh growth is estimated as 0.38 cm yr^{-1} (Smith and Shifley 1984); for trees in a park-like structure (e.g., parks, cemeteries, golf courses), average dbh growth is 0.61 cm yr^{-1} (deVries 1987); for more open-grown trees, dbh and genera specific growth rates, which average about 0.87 cm yr^{-1}, are used based on Nowak (1994b). Average height growth is calculated with formulas from Fleming (1988) and the specific dbh growth factor used for the tree. Growth rates are adjusted based on tree condition. For trees in excellent to fair condition (0-25 percent crown dieback), growth rates are multiplied by 1 (no adjustment). For trees with greater than 25 percent crown dieback, growth rates are adjusted downward based on percent dieback associated with the condition class. Dead tree growth rates are multiplied by 0. The difference in estimates of carbon storage between year x and year $x+1$ is the gross amount of carbon sequestered annually.

Carbon Storage and Sequestration in New York

Trees in New York City store approximately 1.2 million metric tons of carbon. This carbon, which took years to store, is equivalent to the amount emitted from New York's population in about 10 days based on average per capita carbon emissions (U.S. Dept. of Energy 1997). New York's trees sequester an estimated 39,000 t C yr^{-1}.

However, based on estimated mortality and tree removals (given New York's tree-condition distribution), net sequestration is likely negative due to carbon emissions from decomposition of dead and/or removed trees. New York City land uses that contain the most carbon in trees are open space (44 percent of total carbon stored by trees in the city), one- to two-family residential (28 percent), and vacant land (19 percent). Tree species that currently store the most carbon in New York are London planetree (13.6 percent of the total carbon stored), northern red oak (12.8 percent), and pin oak (*Quercus palustris*; 9.7 percent). Through proper planting strategies, urban trees also can reduce atmospheric carbon by reducing energy use in buildings and consequent emission of carbon dioxide from fossil fuel-based power plants. This avoidance of carbon emissions from power plants could be four times greater than direct carbon storage over the life of a mature tree (Nowak 1993).

UFORE-D: DRY DEPOSITION OF AIR POLLUTION

This module calculates the hourly dry deposition (i.e., pollution removal during nonprecipitation periods) of ozone (O_3), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), and carbon monoxide (CO), and daily deposition of particulate matter less than 10 microns (PM10) to tree canopies throughout the year. Pollutant flux (F) is calculated as the product of the deposition velocity (V_d) and the pollutant concentration (C):

$$F\ (g\ m^{-2}\ s^{-1}) = V_d\ (m\ s^{-1}) \times C\ (g\ m^{-3})$$

Deposition velocity is calculated as the inverse of the sum of the aerodynamic (R_a), quasi-laminar boundary layer (R_b) and canopy (R_c) resistances (Baldocchi *et al.* 1987):

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

Specific procedures used to calculate R_a and R_b are given in Nowak *et al.* (1997). In-leaf, hourly tree-canopy resistances for O_3, SO_2, and NO_2 are calculated based on a hybrid of big-leaf and multi-layer canopy deposition models (Baldocchi *et al.* 1987, Baldocchi 1988). Hourly inputs to calculate canopy resistance are photosynthetic active radiation (PAR; $\mu E\ m^{-2}\ s^{-1}$), air temperature (K°), windspeed ($m\ s^{-1}$), u_* ($m\ s^{-1}$), carbon dioxide concentration (set to 360 ppm), and absolute humidity ($kg\ m^{-3}$). Air temperature, windspeed, u_*, and absolute humidity are measured directly or calculated from measured hourly meteorological data. Total solar radiation is calculated based on the National Renewable Energy Laboratory Meteorological / Statistical Solar Radiation Model (METSTAT) with inputs from the NCDC data set (Maxwell 1994). PAR is calculated as 46 percent of total solar radiation input (Monteith and Unsworth 1990).

Because the removal of carbon monoxide and particulate matter by vegetation is not directly related to transpiration, R_c for CO was set to a constant for in-leaf season (50,000 $s\ m^{-1}$) and leaf-off season (1,000,000 $s\ m^{-1}$) based on data from Bidwell and Fraser (1972). For particles, the deposition velocity (based on average V_d from the literature) was set at 0.0064 $m\ s^{-1}$ for the in-leaf season and 0.0014 $m\ s^{-1}$ for the leaf-off season, both of which incorporate a 50 percent resuspension rate of particles back to the atmosphere (Zinke 1967).

The model uses an urban tree LAI of 6, bark area index of 1.7, and a distribution of 90 percent deciduous and 10 percent coniferous leaf surface area (Nowak 1994a, Whittaker and Woodwell 1967). Local leaf-on and leaf-off dates are input into the model so that deciduous-tree transpiration and related pollution deposition are limited to the in-leaf period, and seasonal variation in removal can be illustrated for each pollutant. To limit deposition estimates to periods of dry deposition, deposition velocities were set to zero during periods of precipitation.

Hourly pollution-concentration data for gaseous pollutants (ppm) and average daily concentrations of PM10 ($\mu g\ m^{-3}$) are obtained from local EPA monitors. Hourly ppm values are converted to $\mu g\ m^{-3}$ based on measured atmospheric temperature and pressure (Seinfeld 1986). Missing hourly meteorological or pollution-concentration data are estimated using the monthly average for the specific hour. In a few locations, an entire month of pollution-concentration data may be missing and are estimated based on interpolations from existing data. For example, ozone concentrations may not be measured during winter months and existing O_3 concentration data are extrapolated to missing months based on the average national O_3 concentration monthly pattern.

Average hourly pollutant flux ($g\ m^{-2}$ of tree-canopy coverage) among the pollutant monitor sites is multiplied by city tree-canopy coverage (m^2) to estimate total hourly pollutant removal by trees across the city. Bounds of total tree removal of O_3, NO_2, SO_2, and PM10 are estimated using the typical range of published in-leaf dry deposition velocities (Lovett 1994). Monetary value of pollution removal by trees is estimated using the median externality values for the United States for each pollutant. The externality values are: NO_2 = \$6,750 t^{-1}, PM10 = \$4,500 t^{-1}, SO_2 = \$1,650 t^{-1}, and CO = \$950 t^{-1} (Murray *et al.* 1994). Externality values for O_3 were set to equal the value for NO_2.

To approximate boundary-layer heights in the study area, local daily morning and afternoon mixing heights are obtained and interpolated to produce hourly values using the EPA's PCRAMMIT program (U.S. EPA 1995). Minimum boundary-layer heights are set to 150 m during the night and 250 m during the day based on estimated minimum boundary-layer heights in cities. Hourly mixing heights (m) are used in conjunction with pollution concentrations ($\mu g\ m^{-3}$) to calculate the amount of pollution within the mixing layer ($\mu g\ m^{-2}$). This extrapolation from ground-layer concentration to total pollution within the boundary layer assumes a well-mixed boundary layer, which is common during the daytime (unstable atmosphere) (e.g., Colbeck and Harrison 1985). The amount of pollution in the air is contrasted with the amount of pollution removed by trees to calculate the relative effect of trees in reducing local pollution concentrations:

$$E = R\,(R+A)^{-1}$$

where E = relative reduction effect (%); R = amount removed by trees (kg); A = amount of pollution in the atmosphere (kg).

Air Pollution Removal by Urban Trees

In 1994, trees in New York City removed an estimated 1,821 metric tons of air pollution at an estimated value to society of \$9.5 million (table 2). Air pollution removal by urban forests in New York was greater than in Atlanta and Baltimore (table 2), but pollution removal per m^2 of canopy cover was fairly similar among these cities (New York: 13.7 g $m^{-2}\,yr^{-1}$; Baltimore: 12.2 g $m^{-2}\,yr^{-1}$; Atlanta: 10.6 g $m^{-2}\,yr^{-1}$). These standardized pollution removal rates differ among cities according to the amount of air pollution, length of in-leaf season, precipitation, and other meteorological variables.

Air quality improvement in New York due to pollution removal by trees during daytime of the in-leaf season averaged 0.47 percent for PM10, 0.45 percent for O_3, 0.43 percent for SO_2, 0.30 percent for NO_2, and 0.002 percent for CO. Air quality improves with increased percent tree cover and decreased boundary-layer heights. In urban areas with 100 percent tree cover (i.e., contiguous forest stands), short-term improvements in air quality (one hour) from pollution removal by trees were as high as 15 percent for O_3, 14 percent for SO_2, 13 percent for PM10, 8 percent for NO_2, and 0.05 percent for CO.

CONCLUSION

The UFORE model and data-collection methodology provide a relatively easy and low-cost means of assessing and quantifying urban forest structure and functions in cities across the United States. The resulting data will aid urban forest managers in understanding their resource and developing management plans and vegetation designs to increase human and environmental health and well-being.

Table 2.—*Total estimated pollution removal (metric tons) by trees during nonprecipitation periods (dry deposition) and associated monetary value (thousand dollars) for New York (800 km²), Atlanta (341 km²), and Baltimore (209 km²) in 1994. Estimates are for ozone (O_3), particulate matter less than 10 microns (PM10), nitrogen dioxide (NO_2), sulfur dioxide (SO_2), and carbon monoxide (CO). Numbers in parentheses represent expected range of values (no range determined for CO). Monetary value of pollution removal by trees was estimated using the median externality values for United States for each pollutant (Murray* et al. *1994). Externality values for O_3 were set to equal the value for NO_2.*

	New York, NY		Atlanta, GA		Baltimore, MD	
Pollutant	Removal	Value	Removal	Value	Removal	Value
O_3	506	3,417	514[1]	3,471[1]	180	1,214
	(124-631)	(839-4,263)	(101-604)	(684-4,081)	(42-221)	(284-1,494)
PM10[2]	470	2,120	406	1,833	137	618
	(182-834)	(819-3,761)	(157-706)	(709-3,184)	(53-239)	(239-1,079)
NO_2	510	3,441	145	979	115	733
	(216-593)	(1,459-4,004)	(72-165)	(483-1,115)	(48-134)	(322-907)
SO_2	238	394	95	158	55	91
	(117-358)	(193-593)	(42-137)	(69-227)	(26-85)	(42-140)
CO	97	93	35	33	13	12
Total	1,7821	9,465	1,196	6,474	499	2,709
	(736-2,514)	(3,404-12,713)	(407-1,648)	(1,979-8,640)	(181-692)	(900-3,632)

[1] Average national O_3 monthly trend data were used to estimate missing data for January, February, and December.
[2] Assumes 50 percent resuspension of particles.

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LITERATURE CITED

Baldocchi, D. 1988. A multi-layer model for estimating sulfur dioxide deposition to a deciduous oak forest canopy. Atmosphere Environment. 22: 869-884.

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

Barbour, M.G.; Burk, J.H.; Pitts, W.D. 1980. Terrestrial plant ecology. Menlo Park, CA: Benjamin/Cummings Publication Co. 604 p.

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. Atmosphere Environment. 30(9): 1437-1452.

Bidwell, R.G.S.; Fraser, D.E. 1972. Carbon monoxide uptake and metabolism by leaves. Canadian Journal of Botany. 50: 1435-1439.

Brasseur, G.P.; Chatfield, R.B. 1991. The fate of biogenic trace gases in the atmosphere. In: Sharkey, T.D.; Holland, E.A.; Mooney, H.A., eds. Trace gas emissions by plants. New York: Academic Press: 1-27.

Chow, P.; Rolfe, G.L. 1989. Carbon and hydrogen contents of short-rotation biomass of five hardwood species. Wood and Fiber Science. 21(1): 30-36.

Cochran, W.G. 1977. Sampling methods. New York: John Wiley and Sons. 428 p.

Colbeck, I.; Harrison, R.M. 1985. Dry deposition of ozone: some measurements of deposition velocity and of vertical profiles to 100 metres. Atmosphere Environment. 19(11): 1807-1818.

deVries, R.E. 1987. A preliminary investigation of the growth and longevity of trees in Central Park. New Brunswick, NJ: Rutgers University. 95 p. M.S. thesis.

Fleming, L.E. 1988. Growth estimation of street trees in central New Jersey. New Brunswick, NJ: Rutgers University. 143 p. M.S. thesis.

Geron, C.D.; Guenther, A.B.; Pierce, T.E. 1994. An improved model for estimating emissions of volatile organic compounds from forests in the eastern United States. Journal of Geophysical Research. 99(D6): 12,773-12,791.

Grove, J.M. 1996. The relationship between patterns and processes of social stratification and vegetation of an urban-rural watershed. New Haven, CT: Yale University. Ph.D. dissertation.

Hahn, J.T. 1984. Tree volume and biomass equations for the Lake States. Res. Pap. NC-250. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 10 p.

Liebhold, A.M.; Gottschalk, K.W.; Muzika, R.; Montgomery, M.E.; Young, R.; O'Day, K.; Kelley, B. 1995. Suitability of North American tree species to the gypsy moth: a summary of field and laboratory tests. Gen. Tech. Rep. NE-211. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 34 p.

Lovett, G.M. 1994. Atmospheric deposition of nutrients and pollutants in North America: an ecological perspective. Ecological Application. 4: 629-650.

Maxwell, E.L. 1994. A meteorological/statistical solar radiation model. In: Proceedings of the 1994 annual conference of the American solar energy society. San Jose, CA: American Solar Energy Society: 421-426.

Monteith, J.L.; Unsworth, M.H. 1990. Principles of environmental physics. New York: Edward Arnold.

Murray, F.J.; Marsh, L.; Bradford, P.A. 1994. New York State energy plan, Vol. II: issue reports. Albany, NY: New York State Energy Office.

Nowak, D.J. 1991. Urban forest development and structure: analysis of Oakland, California. Berkeley, CA: University of California. 232 p. Ph.D. dissertation.

Nowak, D.J. 1993. Atmospheric carbon reduction by urban trees. Journal of Environmental Management. 37: 207-217.

Nowak, D.J. 1994a. Urban forest structure: the state of Chicago's urban forest. In: McPherson, E.G.; Nowak, D.J.; Rowntree, R.A., eds. Chicago's urban forest ecosystem: results of the Chicago Urban Forest Climate Project. Gen. Tech. Rep. NE-186. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 3-18; 140-164.

Nowak, D.J. 1994b. Atmospheric carbon dioxide reduction by Chicago's urban forest. In: McPherson, E.G.; Nowak, D.J.; Rowntree, R.A., eds. Chicago's urban forest ecosystem: results of the Chicago Urban Forest Climate Project. Gen. Tech. Rep. NE-186. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station: 83-94.

Nowak, D.J. 1996. Estimating leaf area and leaf biomass of open-grown deciduous urban trees. Forest Science. 42(4): 504-507.

Nowak, D.J.; McHale, P.J.; Ibarra, M.; Crane, D.; Stevens, J.; Luley, C. 1997. Modeling the effects of urban vegetation on air pollution. In: Proceedings of the 22nd NATO/CCMS international technical meeting on air pollution and its application. Clermont-Ferrand, France: Imprimerie des UFR Sciences: 276-283.

Nowak, D.J.; Rowntree, R.A.; McPherson, E.G.; Sisinni, S.M.; Kerkmann, E.; Stevens, J.C. 1996. Measuring and analyzing urban tree cover. Lands. Urban Planning. 36: 49-57.

Seinfeld, J.H. 1986. Atmospheric chemistry and physics of air pollution. New York: John Wiley and Sons. 738 p.

Smith, W.B.; Shifley, S.R. 1984. Diameter growth, survival, and volume estimates for trees in Indiana and Illinois. Res. Pap. NC-257. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 10 p.

Taha, H. 1996. Modeling impacts of increased urban vegetation on ozone air quality in the South Coast Air Basin. Atmosphereic Environment. 30(20): 3423-3430.

U.S. Department of Commerce. 1970. National Climatic Data Center WBAN Hourly Surface Observations TD-1440 Documentation Manual. Asheville, NC: National Oceanic and Atmospheric Administration. 14 p.

U.S. Department of Energy, Energy Information Administration. 1997. Emissions of greenhouse gases in the United States 1996. Washington, DC: U.S. Department of Energy, Energy Information Administration.

U.S. Environmental Protection Agency. 1995. PCRAMMIT user's guide. Research Triangle Park, NC: U.S. Environmental Protection Agency.

Whittaker, R.H.; Woodwell, G.M. 1967. Surface area relations of woody plants and forest communities. American Journal of Botany. 54: 931-939.

Winer, A.M.; Fitz, D.R.; Miller, P.R.; Atkinson, R.; Brown, D.E.; Carter, W.P.; Dodd, M.C.; Johnson, C.W.; Myers, M.A.; Neisess, K.R.; Poe, M.P.; Stephens, E.R. 1983. Investigation of the role of natural hydrocarbons in photochemical smog formation in California. Riverside, CA: Statewide Air Pollution Research Center. 234 p.

Zinke, P.J. 1967. Forest interception studies in the United States. In: Sopper, W.E.; Lull, H.W., eds. Forest hydrology. Oxford, UK: Pergamon Press: 137-161.