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Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project

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Results of the 3-year Chicago Urban Forest Climate Project indicate that there are an estimated 50.8 million trees in the Chicago area of Cook and DuPage Counties; 66 percent of these trees rated in good or excellent condition. During 1991, trees in the Chicago area removed an estimated 6,145 tons of air pollutants, providing air cleansing valued at $9.2 million dollars. These trees also sequester approximately 155,000 tons of carbon per year, and provide residential heating and cooling energy savings that, in turn, reduce carbon emissions from power plants by about 12,600 tons annually. Shade, lower summer air temperatures, and a reduction in windspeed associated with increasing tree cover by 10 percent can lower total heating and cooling energy use by 5 to 10 percent annually ($50 to $90 per dwelling unit). The projected net present value of investment in planting and care of 95,000 trees in Chicago is $38 million ($402 per planted tree), indicating that the long-term benefits of trees are more than twice their costs. Policy and program opportunities to strengthen the connection between city residents and city trees are presented.

Retrevial Terms: urban climate, air pollution, urban forestry, energy conservation, carbon dioxide, urban ecosystem

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Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project

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The Chicago Urban Forest Climate Project (CUFCP) was a 3-year study to quantify the effects of urban vegetation on the local environment and help city planning and management organizations increase the net environmental benefits derived from Chicago's urban forest. The CUFCP study area consists of three sectors: Chicago, Cook County (exclusive of Chicago), and DuPage County (Figure 1). This report presents study results as well as information on continuing urban-forest research in the Chicago area. Numerous interrelated studies in the Chicago region were completed as part of the CUFCP, ranging from region-wide analyses of urban-forest ecosystems to investigations of individual trees and leaves. Research results can be summarized in the following five research topics.

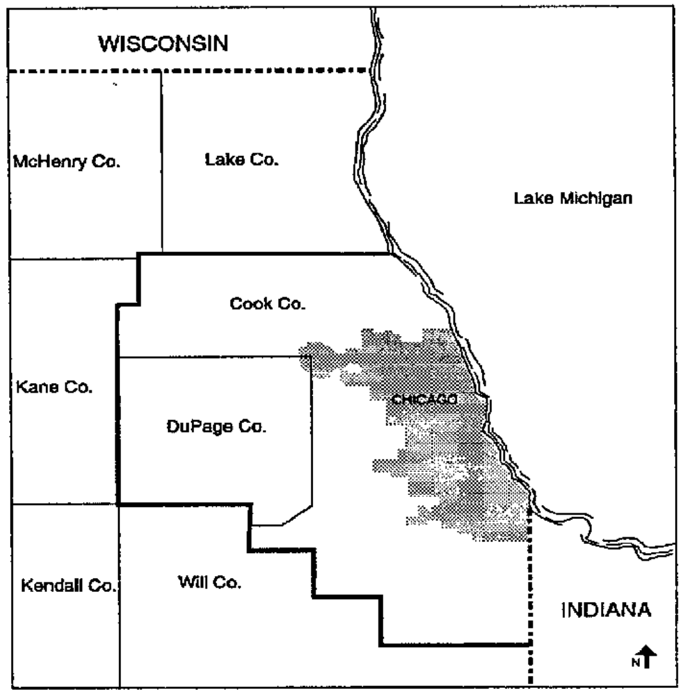

Figure 1.—The Chicago Urban Forest Climate Project study area includes the City of Chicago, and Cook and DuPage Counties.

I. Chicago's Urban Forest Ecosystem and its Effect on Air Quality and Atmospheric Carbon Dioxide

Information on the structure of Chicago's urban forest (e.g., species composition, tree leaf-surface area) provides the basis for understanding the functions of the urban forest that affect the city and its inhabitants. There are currently 4.1 million trees in the City of Chicago, with an estimated 50.8 million trees across the Chicago area of Cook and DuPage Counties. Most of these trees are small and on institutional, residential, and vacant lands. Relatively short-lived pioneer species contribute significantly to the Chicago area's urban forest, are most prevalent on land uses with minimal or naturalistic management (e.g., forest stand conditions), and may constitute an even more important component of the Chicago area's urban forest structure in the future. The most common trees in the Chicago area are buckthorn, green/white ash, *Prunus* spp., boxelder, and American elm.

Field sampling of leaves of urban trees was used to develop equations to estimate leaf-surface area, the plant surface where atmospheric gases are actively exchanged. The most dominant species in leaf area in the Chicago area are silver maple, green/white ash, white oak, American elm, and boxelder. These species likely have the greatest effect on the environment in the Chicago area.

Street trees are a significant part of Chicago's landscape, accounting for 10 percent of the city's trees and 24 percent of the total leaf-surface area. Street trees are less significant in more suburban or rural areas. The most common ground surfaces in the study area are maintained grass, tar, herbaceous cover (e.g., crops), and buildings. Information on the structure of the Chicago urban forest ecosystem was used to help quantify the ecosystem functions of air pollution removal and carbon dioxide sequestration by urban trees.

Removal of Air Pollution

Air pollution is a multibillion dollar problem nationally that affects most major U.S. cities. Air pollution affects human health, damages vegetation and various anthropogenic materials, and reduces visibility. Trees can remove air pollution by intercepting particulates and absorbing gaseous pollutants (Figure 2). In 1991, trees in Chicago removed an estimated 15 metric tons (t) (17 tons) of carbon monoxide (CO), 84 t (93 tons) of sulfur dioxide (SO_2), 89 t (98 tons) of nitrogen dioxide (NO_2), 191 t (210 tons) of ozone (O_3), and 212 t (234 tons) of particulate matter less than 10 microns (PM10). Across the Chicago area, trees (in-leaf season) removed an average of 1.2 t/day (1.3 tons/day) of CO, 3.7 t/day (4.0 tons/day) of SO_2, 4.2 t/day (4.6 tons/day) of NO_2, 8.9 t/day

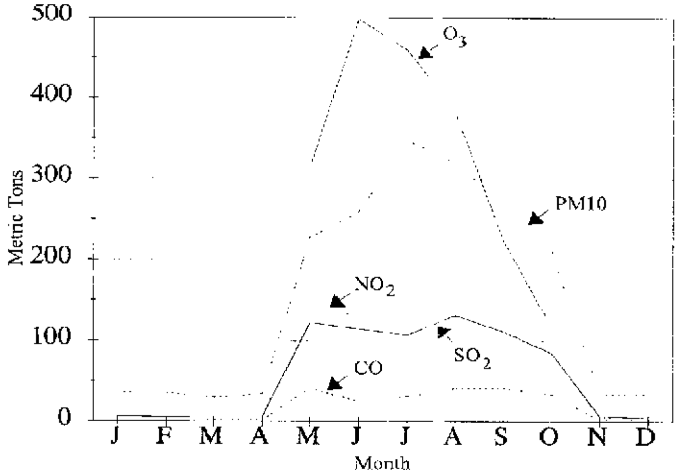

Figure 2. —Monthly estimates of pollution removal by trees in study area in 1991. Ozone removal estimates are for May-October only. PM10 estimates assume 50 percent resuspension of particles.

(9.8 tons/day) of PM10 and 10.8 t/day (11.9 tons/day) of O_3. The estimated value of pollution removal in 1991 was $1 million for trees in Chicago and $9.2 million for trees across the Chicago area. Average hourly improvement (in-leaf season) in air quality due to all trees in the Chicago area ranged from 0.002 percent for CO to 0.4 percent for PM10. Maximum hourly improvement was estimated at 1.3 percent for SO_2, though localized improvements in air quality can reach 5 to 10 percent or greater in areas with relatively high tree cover, particularly under stable atmospheric conditions during the daytime of the in-leaf season. Large, healthy trees remove an estimated 60 to 70 times more pollution than small trees.

Sequestration of Carbon Dioxide

Increasing levels of atmospheric carbon dioxide (CO_2) and other "greenhouse" gases are thought by many to be leading to increased atmospheric temperatures through the trapping of certain wavelengths of heat in the atmosphere. In terms of reducing atmospheric CO_2, trees in urban areas offer the double benefit of direct carbon storage and the avoidance of CO_2 production by fossil-fuel power plants through energy conservation from properly located trees. Trees in Chicago store an estimated 855,000 t of carbon (942,000 tons), and trees throughout the Chicago area store approximately 5.6 million t (6.1 million tons). Carbon storage by shrubs is approximately 4 percent of the amount stored by trees. Total carbon storage and annual sequestration are greatest on 1-3 family residential lands, institutional lands dominated by vegetation (e.g., parks, forest preserves) and vacant lands. The estimated net sequestration of carbon in the Chicago area is 140,600 t (155,000 tons). Carbon storage by urban forests nationally likely is between 400 and 900 million t (440 and 990 millions tons).

Carbon storage by individual trees is up to 1,000 times greater in large than in small trees, with sequestration rates up to 90 times greater for healthy large than healthy small trees. Estimated carbon emissions avoided annually due to energy conservation from existing trees throughout the Chicago area is 11,400 t (12,600 tons). Total carbon stored by trees in the Chicago area, which took years to store, is equivalent to the amount of carbon emitted from the residential sector in the Chicago area during a 5-month period. Net annual sequestration equals the amount of carbon emitted from transportation use in the Chicago area in 1 week. The amount of carbon sequestered annually by one tree less than 8 cm (3 inches) in trunk diameter (d.b.h.) equals the amount emitted by one car driven 16 km (10 miles). Reasonable additional tree planting in conjunction with efforts to sustain existing tree cover could increase carbon storage in the Chicago area by another 1.2 million t (1.3 million tons), or the amount of carbon emitted by transportation use in the Chicago area in less than 2 months.

II. Effect of Urban Trees on Wind and Air Temperature

By transpiring water, blocking winds, shading surfaces, and modifying the storage and exchanges of heat among urban surfaces, trees affect local climate and consequently energy use in buildings, human thermal comfort, and air quality. Models that accurately estimate the effect of urban trees on local windspeed and air temperature at the height of people and residential buildings are lacking, partly because of the complexity of the multiple surfaces in urban areas.

To develop models for estimating the effect of trees on urban microclimates, measurements of windspeed, air temperature, and humidity were taken at 39 sites in and near residential neighborhoods in Chicago over an 11-month period (July 1992 to June 1993). Equations to predict the influence of trees on local climate are being developed by analyzing the interrelationships among climatic variables and local urban morphology (e.g., tree and building attributes).

Preliminary analyses for a 1-week summer period indicate that residential morphology (buildings and trees combined) reduced windspeeds by an average of 46 to 85 percent (relative to an open field site at O'Hare International Airport) depending on the specific neighborhood morphology. The reductions in wind speed were significantly related to indicators of urban morphology. Residential air temperatures generally were warmer than the open-field site due to the predominance of building surfaces which tend to warm the local environment. Continuing work is quantifying the specific effect of urban trees on local windspeed, air temperature, and humidity.

III. Local-Scale Energy and Water Exchanges

The complex mix of anthropogenic surfaces (e.g., buildings, roads) and natural surfaces (e.g., trees, grass) in urban areas affects how energy and water are partitioned and cycled through the urban system (Figure 3). The replacement of natural surfaces with anthropogenic surfaces alters the thermal and moisture properties of the area, thereby modifying the local atmosphere and generating an "urban climate" that is commonly characterized by increased air

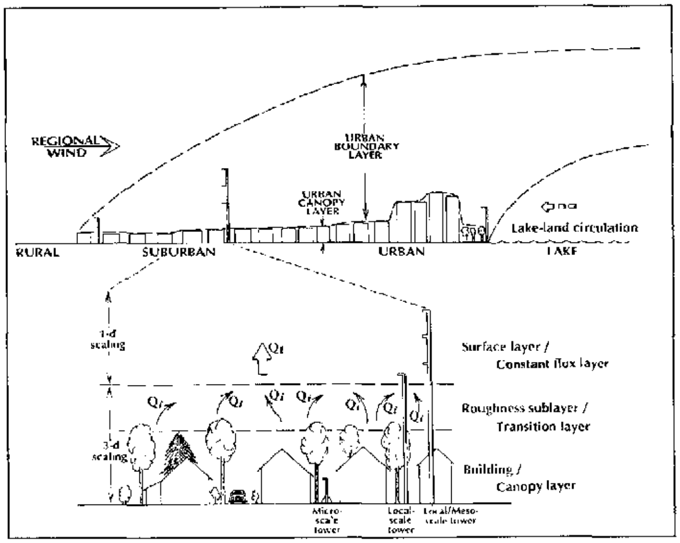

Figure 3.—Schematic representation of spatial scales and atmospheric processes in urban areas (adapted from Oke 1984; Oke et al. 1989).

temperatures and poorer air quality. Extensive climatic measurements across the north-side of Chicago and intensive measurements of a predominantly residential area in and around Chicago were conducted to quantify how urban morphologies affect local energy and water exchanges. Intensive observations consisted of direct measurements of sensible and latent heat flux, and net all-wave radiation. Convective fluxes were quantified using eddy-correlation techniques which seek to measure the flux directly by sensing properties of eddies as they pass through a measurement level on an instantaneous basis.

Calculation of the Bowen ratio for a period during July 1992 indicates that more energy (available from the sun and earth) was going to drying surfaces (latent heat flux) than to warming the air (sensible heat flux). This result is different from that observed in the summer in Tucson, Arizona, and in Sacramento and Los Angeles, California. However, the results for Chicago are realistic considering the meteorological conditions of July 1992 (i.e., relatively high frequency of rainfall). Of the net available energy from solar and earth radiation during the daytime, 32 percent went to heating the air, 38 percent to evaporating water, and 30 percent to heating urban surfaces. Work is in progress to correlate the latent and sensible heat fluxes with tree cover. This correlation will reveal the effect of trees on flux partitioning and help determine to what degree trees cool the local environment. Numerical models are being developed to predict the effect of different tree-planting scenarios on local-scale energy and water exchanges.

IV. Potential Building Energy Savings from Urban Trees

Trees can reduce building energy use by lowering summertime temperatures, shading buildings during the summer, and blocking winter winds. However, trees also can increase building energy use by having their branches shade buildings during the winter, and can increase or decrease building energy use by blocking summertime breezes. Computer simulations of microclimates and building energy performance were used to investigate the potential of shade trees to reduce the use of residential heating and cooling energy in Chicago. Increasing tree cover by 10 percent (or about three trees located in optimal energy-conserving locations per building) could reduce total heating and cooling energy use by 5 to 10 percent ($50 to $90). On a per-tree basis of this mass planting, annual heating energy use can be reduced by about 1.3 percent ($10, 2 MBtu), cooling energy use by about 7 percent ($15, 125 kWh), and peak cooling demand by about 6 percent (0.3 kW). Benefit-cost ratios of 1.40 for trees planted around typical two-story buildings and 1.96 for trees near energy-efficient wood frame buildings indicate that a utility-sponsored shade tree program could be cost-effective for both existing and new construction in Chicago.

Street trees are a major source of building shade in Chicago. Shade from a large street tree located to the west of a typical brick residence can reduce the annual use of air-conditioning energy by 2 to 7 percent ($17 to $25, 138 to 205 kWh) and peak cooling demand by 2 to 6 percent (0.16 to 0.6 kW). Street trees that shade the east side of buildings can produce similar cooling savings, have a negligible effect on peak cooling demand, and can slightly increase heating costs. Shade from large street trees to the south increase heating costs more than they decrease cooling costs. Planting "solar friendly" trees to the south and east can minimize the energy penalty associated with blocking irradiance during the heating season. Design guidelines and recommended tree species for energy-efficient landscapes are presented.

V. Benefits and Costs of Urban Tree Planting and Care

Benefit-cost analysis was used to estimate the net present value, benefit-cost ratio, and discounted payback periods of proposed tree plantings in Chicago. A "typical" tree species, green ash, was located in "typical" park, residential yard, street, highway, and public housing sites. The 30-year stream of annual costs and benefits associated with the planting of 95,000 trees was estimated. Assuming a 7-percent discount rate, a net present value of $38 million, or $402 per planted tree, was projected. Projected benefit-cost ratios were largest for trees planted in residential yards and public housing sites (3.5), and least for parks (2.1) and highways (2.3). Discounted payback periods ranged from 9 to 15 years (Figure 4). Expenditures for planting alone accounted for over 80 percent of projected costs except at public housing sites, while the largest benefits were attributed to "other" benefits (e.g., scenic, social, economic values) and energy savings. Findings indicate that despite the expense of planting and caring for trees in Chicago, with time the benefits that healthy trees produce can exceed their costs.

Several policies and programs could expand the current role of residents, businesses, utilities, and governments in the planning and management of Chicago's future urban forest.

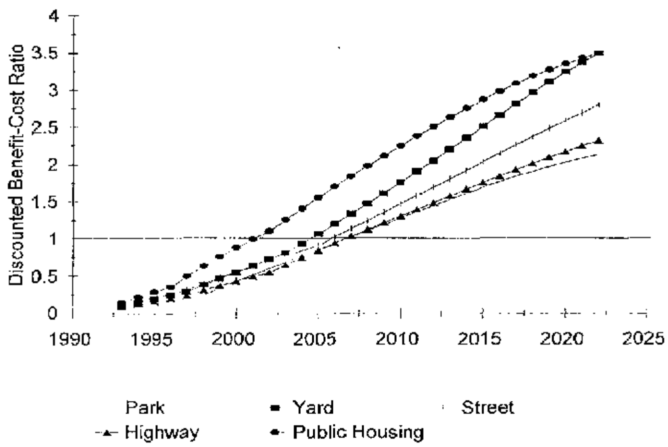

Figure 4. —Discounted payback periods depict the number of years before the benefit-cost ratio exceeds 1.0. This analysis assumes a 30-year planning period and 7-percent discount rate.

Potential new policies and programs include developing a comprehensive set of urban-forest planning principles which address such issues as job training opportunities, conservation education, neighborhood revitalization, mitigation of heat islands, and energy conservation; partnerships to enhance tree planting and care in public and low-income housing areas; an urban-forest stewardship program to provide financial assistance for professional care of existing trees; a yard-tree planting program to reduce building energy use that is sponsored by local utility companies; and a public education program that informs residents about the benefits of healthy and productive urban forests in ways to strengthen the connection between city residents and city trees.

Acknowledgments

There are many individuals and organizations without whose help this project may never have been completed. We sincerely thank The City of Chicago, particularly Mayor Richard M. Daley; Chicago Department of Environment, particularly Henry Henderson, Suzanne Malec Hoerr, Edith Makra, and Dave Inman; Chicago Bureau of Forestry, particularly Steve Bylina Jr., Jerry Dalton, Bill Brown, Timothy Vaughan, and Ray Toren Jr.; Illinois Department of Mental Health, particularly Marva Arnold, Jess McDonald, and the Chicago Read Mental Health Center; Chicago Park District, particularly Geri Weinstein, Robert Megquier, Ron Nemchausky and Al Neiman; USDA Forest Service North Central Forest Experiment Station, particularly John Dwyer; Illinois Environmental Protection Agency, particularly Robert Swinford; Village of Oak Park, particularly Mike Stankovich; Village of Glen Ellyn, particularly Peggy Young; Village of Bloomingdale, particularly Larry Slavicek; Hendricksen The Care of Trees, particularly Larry Hall; Chicago Gateway Green Committee, particularly Vince Pagone; Illinois Department of Transportation, particularly Duane Carlson and Rick Wanner; Commonwealth Edison, particularly Claire Saddler and Tom Hemminger; Peoples Gas, particularly Gene Waas and Bob Pendlebury; Center for Neighborhood Technology, particularly Ray Lau and John Katrakis; Illinois State Police, particularly Ed Bean and Mike Cima; Chicago O'Hare International Airport, particularly Edward McCall; the many residents of the Chicago area who permitted us to survey their vegetation and make climatic measurements; Indiana University undergraduate and graduate student helpers; USDA Forest Service, Northeastern Forest Experiment Station, particularly Robert Lewis, Rich Guldin, Gerald Walton, Lee Philbin, Paul Sacamano, Scott Prichard, Jack Stevens, Una Arnold, Wayne Zipperer, Sue Sisinni, Mary Buchanan, and Marty Jones; USDA Forest Service, Pacific Southwest Research Station, particularly Barbara Weber, Enoch Bell, Esther Kerkmann, Maureen Davis, and Kathy Stewart.

Chapter 1
The Role of Vegetation in Urban Ecosystems

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Abstract

The Chicago Urban Forest Climate Project (CUFCP) evaluates the role of trees and other vegetation in the regional urban forest ecosystem. Ecosystem analysis provides an effective approach to planning and controlling the distribution of benefits and costs associated with ecological effects. The flow of energy, water, carbon, and pollutants through the ecosystem can be changed by changing the amount and spatial distribution of trees. Continuing research in Chicago and collaborating cities will refine the information needs for urban ecosystem management.

Purpose of this Study

The goal of this research is to add to our knowledge of how vegetation in and near cities affects the human environment. This report summarizes the 3-year Chicago Urban Forest Climate Project which examined how trees and plants of the Chicago area affect selected components of the regional urban ecosystem.

Vegetation is part of the region's infrastructure, woven into a complex network of power lines, roads, aqueducts, and sewers that together help to sustain human health and quality of life. Yet, little is known about how this green infrastructure creates benefits and costs for people. In fact, most of the world's cities have scant information about the composition and geography of their urban forest.

Urban forest is now a common term that means all of the vegetation and soils of an urban region. For this study, we occasionally substitute the term "urban forest ecosystem" to emphasize the ecological approach the scientific team has taken in conducting the research. This approach proceeds from the assumption that the Chicago region operates as a result of multiple interactions among vegetation, soils, water, insects, wildlife, climate, anthropogenic surfaces, and people. The goal is to manage that operation so that benefits far exceed costs.

The initial report of this research project, "Chicago's Evolving Urban Forest," describes the history of vegetation and changes in the urban forest in the Chicago region since the beginning of urbanization (McPherson et al. 1993) Because research is continuing into 1995, a book will be published in the next several years updating our knowledge about Chicago's urban forest ecosystem.

Manipulating Vegetation to Guide Ecosystem Operation

Some elements of the urban ecosystem can be readily manipulated and others cannot. Vegetation is one element of the ecosystem that can be manipulated in a planned and cost-effective way. Vegetation is renewable and has the potential to yield a wide range of important benefits. The body of knowledge about the role of vegetation in the urban ecosystem and for enhancing human well being is inadequate for managers to make informed decisions about how much to invest, when and where, and for what outcomes. This weak technical foundation has plagued decisionmakers over the last decades in the face of increasing public interest in urban afforestation and urban forestry.

Planners and managers must know what vegetation does, because it affects nearly every other component of the regional urban ecosystem. Herbs, shrubs, and trees change the temperature and humidity of the air. They intercept rainfall and capture air pollutants. Vegetation mediates chemical exchanges between the soil and the atmosphere. The urban forest provides habitat for local and migratory birds. Therefore, to effectively manage the ecological processes in an urban region, we must manage the vegetation. To do that, we must understand its structure and function.

The ecosystem concept has been used for many years to understand how portions of natural landscapes function. The standard approach is first to describe the main components of the system. The second task is to understand how energy, water, and matter (e.g., nutrients) move through the ecosystem. In this study of the Chicago region, we follow this same sequence. First we quantified the structure of the vegetation. Then the research team examined how vegetation affected the flux, or flow, of energy, water, and air pollution through the ecosystem in ways that produce benefits or costs.

Managing an Urban Region Using the Ecosystem Approach

Today, federal and state land-management agencies are using ecosystem management to bring a science-based approach to caring for complex landscapes. This study is one of the first to approach the analysis of an *urban* landscape with an eye toward employing ecosystem management in the future. The research takes the first steps towards building a model that can support ecosystem management of an urban region by stewarding vegetation.

Given the complexity of ecological and socioeconomic processes in an urban region, ecosystem management is the most effective approach for the following reasons:

(1) *Ecosystem management requires documentation of all components and potential relationships.* No factor is left off the list. The level of documentation and understanding will vary among the components. For example, as a result of this research we know much more about Chicago's urban forest, but our understanding of how the forest cools summer air masses is relatively weak. A survey of how much we know about each component and each potential relationship provides managers with a map of their technical strengths and weaknesses. They can make decisions accordingly and request more technical information where it is needed.

(2) *Ecosystem management views processes that generate benefits and costs at different but related scales of time and space.* Management decisions can be assessed in the context of long-term processes such as changes in tree cover over time. For example, in this report we offer a method for spreading the distribution of benefits and costs of tree planting over future years. This method allows the decisionmaker to see what has been invested and what benefits have been generated at any point in time. Small-scale (in both time and space) processes, such as neighborhood tree planting events, can be assessed in the framework of long-term afforestation programs that will have a spectrum of associated benefits and costs. Thus, a resident planting a tree is seen not as an isolated event but as influencing larger-scale (in both time and space) meteorological, energy, and air-pollution processes. Simply, ecosystem management gives the planner, policymaker, and manager an accounting system and map that aggregates small events into larger processes, and disaggregates large, complex processes into simpler elements.

(3) *Ecosystem management is responsible for inter-regional and inter-generational effects.* Because of the expanded time and space scale cited, this approach makes the management of each ecosystem responsible for how it affects adjacent and distant but related ecosystems. And, ecosystem management is responsible for how future generations of people will be affected. While this may seem to place a greater burden on those who manage an ecosystem, this approach—if applied uniformly across all ecosystems—will result in lower costs and greater benefits for all of society.

(4) *Ecosystem management brings private and public land owners and managers together for a common purpose.* Once it is understood how the ecosystem operates, landowners can see how their actions influence processes that generate benefits and costs. Most ecosystems are made up of private and public land managed for a range of purposes, from parks to supermarkets. When individual land owners and agency officials understand the systemwide effects of their actions, they will be able to better manage their land.

In summary, the information requirements for managing urban ecosystems are high, but the short-, medium-, and long-term benefits far exceed the investment. This is recognized in many cities and urban areas, and citizens and organizations are seeking ways of taking the next step toward ecosystem management in their area.

Transferring the Chicago Ecosystem Model to Other Cities

The Chicago study was conducted with federal funds by a team of USDA Forest Service researchers, in cooperation with several university colleagues, to provide knowledge for future stewardship of the Chicago region, but also to act as a model for other cities in the United States and around the world. Already, several cities are making preparations to conduct similar studies of their ecosystems to determine precisely the role of vegetation. It is the research team's hope that the concepts, methods, and procedures developed in Chicago will be tested and streamlined in the next few years so that cities can do this work themselves with scientists serving only as technical advisors.

Acknowledgments

This research was requested in 1990 by Mayor Richard M. Daley and funded by a special appropriation from the U.S. Congress through the USDA Forest Service, Northeastern Forest Experiment Station. Many people and organizations in the Chicago area assisted the research team in their work. We are grateful for this cooperation. We especially thank Dr. John Dwyer, Project Leader with the USDA Forest Service's North Central Forest Experiment Station in Chicago, for providing the research team with scientific input, office space, and a collegial scientific environment.

Literature Cited

McPherson, E. G.; Nowak, D. J.; Sacamano, P. L.; Prichard, S. E.; Makra, E. M. 1993. **Chicago's evolving urban forest: initial report of the Chicago Urban Forest Climate Project.** Gen. Tech. Rep. NE-169. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 55 p.

Chapter 2

Urban Forest Structure: The State of Chicago's Urban Forest

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Abstract

Information on urban forest structure (species composition, tree size and location, etc.) provides the basis for understanding the urban forest functions that affect urban inhabitants and for improving management to maximize the environmental and social benefits of urban forests. There are an estimated 4.1 million trees in the City of Chicago, with an estimated 50.8 million trees across the study area of Cook and DuPage Counties. Most of these trees are small and on institutional, residential, and vacant lands.

Relatively short-lived pioneer species contribute significantly to the Chicago area urban forest. The invasive buckthorn is the most common tree, accounting for 12.7 percent of the total tree population but only 2.9 percent of total leaf-surface area. Other common trees are green/white ash, *Prunus* spp., boxelder, and American elm. The most dominant species in leaf area are silver maple, green/white ash, white oak, American elm, and boxelder. Native pioneer tree species (e.g., boxelder, green ash, willow, cottonwood) and buckthorn are most prevalent on land uses with minimal or naturalistic management (e.g., forest stand conditions) and may constitute an even more important component of the Chicago area's urban forest structure in the future.

Streets trees are a significant part of Chicago's landscape, accounting for 10 percent of the city's trees and 24 percent of the total leaf-surface area. Street trees are less significant in more suburban or rural areas. Common ground surfaces in the study area are maintained grass, tar, herbaceous cover (e.g., crops) and buildings. This paper presents formulas for estimating the leaf-surface area of urban trees and discusses the importance of urban forest structure, particularly leaf-surface area, and how managers and planners can direct urban forest structure to a desired outcome.

Introduction

Urban forest structure is the three-dimensional spatial arrangement of vegetation in urban areas (species composition, tree size and health, number and location of trees, etc.). Information on this structure provides the basis for understanding the urban forest functions that affect urban inhabitants (air temperature modifications, human stress reduction, air pollution mitigation, improved sense of community, etc.) and for improving management to maximize the environmental and social benefits of urban forests.

Urban forest structure is determined by three broad factors: urban morphology, which creates the spaces available for vegetation; natural factors, which influence the amounts and types of biomass likely to be found within cities; and human management systems, which account for intraurban variations in biomass configurations according to land use distributions (Sanders 1984). There are significant variations in urban forest structure both within and among cities. Aerial photographic analyses of urban tree canopy cover reveal that tree cover varies between 5 and 60 percent among land-use types within four eastern U.S. cities, while overall urban tree cover ranged from 24 to 37 percent among the cities (Rowntree 1984).

There has been little ground-based research evaluating the urban forest structure of an entire city. Many researchers have evaluated the street-tree component of the urban forest (Impens and Delcarte 1979; Richards and Stevens 1979; Dawson and Khawaja 1985; Talarchek 1985; Jim 1986; Stevens and Richards 1986; McPherson and Rowntree 1989) or limited portions of non-street tree urban forests (e.g., Derrenbacher 1969; Schmid 1975; Whitney and Adams 1980; Airola and Buchholz 1982; Boyd 1983; Buhyoff et al. 1984; Dorney et al. 1984; McBride and Froehlich 1984; Miller and Winer 1984; Richards et al. 1984; Schroeder and Green 1985; Schroeder and Cannon 1987; Profous et al. 1988, Profous and Rowntree 1993), but ground-based urban forest structural analyses of an entire urban area have been conducted only for the Los Angeles Basin (Horie et al. 1991) and Oakland, California (Nowak 1991). The Los Angeles study focused on leaf biomass and volatile organic emissions from vegetation. The Oakland study focused on variations in urban forest structure and its overall effect on forest compensatory value, atmospheric carbon storage and volatile organic emissions from vegetation (Nowak 1993a,b).

Since many environmental functions are related to leaf-surface area (e.g., reductions in air temperature, air pollution removal, volatile organic emissions, carbon dioxide sequestration), understanding the leaf-area contribution of various tree species is important to urban-forest researchers, managers and planners. The measure of tree-species dominance reflects the relative contribution of a species to the overall leaf-surface area of the forest. Species with the greatest proportion of leaf-surface area are the most dominant and likely have the greatest influence on the local environment. Many social benefits of trees also may be related to leaf-surface area. For example, large trees contribute more scenic beauty than smaller ones (Buhyoff et al. 1984; Schroeder and Cannon 1987).

Leaf-area indices (LAI) are another common means of comparing the relative contribution of leaf area among different areas or tree species on an equal-area basis. LAI is the total leaf area (one surface only) divided by the ground area occupied by the plant. A LAI of 4 means that for every square meter of ground below the tree canopy, 4 m^2 of leaves lie above it. Net primary productivity (individual plant growth) of forests is greatest at a LAI of approximately 4. However, the yield (growth) per unit of ground area is low in such open stands (LAI < 4). Maximum gross productivity usually occurs at LAI values of 8 to 10 (Kramer and Kozlowski 1979); LAI varies with plant size, age, spacing, species, and site characteristics.

Typical LAI's are 10 to 11 for tropical rain forests, 5 to 8 for deciduous forests, and 9 to 11 for boreal coniferous forests (Barbour et al. 1980). The LAI of some Piedmont hardwood forests range from 4.5 to 7.4 (Hedman and Binkley 1988), and LAI's of a subalpine Sierra Nevada forest range from 3.6 to 11.7 (Peterson et al. 1988). Little research has been conducted on the LAI of urban trees. Data from individual urban trees and shrubs in Warsaw, Poland, show LAI's for individual trees ranging from 1 to 15 with an average LAI of individual trees for various areas in Warsaw of 3.5 to 4.8 (Gacka-Grzesikiewicz 1980).

Because information is scarce on the variation in forest structure within urban areas, on how urban forest structure combines to create an urban forest ecosystem, and on leaf-surface area of urban trees, the objectives of this study were to: 1) quantify urban forest structure and its variation by land-use type in the Chicago area; and 2) measure the leaf-surface area of individual open-grown urban trees and develop predictive equations of leaf-surface area to estimate tree species dominance in the Chicago area. This information will be used to reveal key urban forest characteristics and aid in quantifying various environmental functions (see Nowak 1994a,b: Chapters 5 and 6, this report).

Methods

Study Area

The study area encompasses Cook and DuPage Counties (3,350 km^2; 1,292 mi^2) and contains nearly six million people. To reveal regional variation within the Chicago area, the study area was subdivided into the City of Chicago, Cook County exclusive of Chicago (hereafter referred to as suburban Cook County), and DuPage County (Figure 1). Chicago is the most densely populated sector, accounting for 18 percent of the entire study area and 47 percent of the total population. Suburban Cook County contains 56 percent of the study area and 40 percent of the total population, and many of the older suburban communities in the Chicago region. DuPage County is the least densely populated, most agricultural, and most rapidly urbanizing sector within the study area. It contains 13 percent of the population and occupies 26 percent of the study area. Tree crowns cover an average of 11 percent of the land area in Chicago, 23 percent in suburban Cook County, and 19 percent in DuPage County (McPherson et al. 1993). Crown cover also varies by individual land-use types within each sector (Table 1).

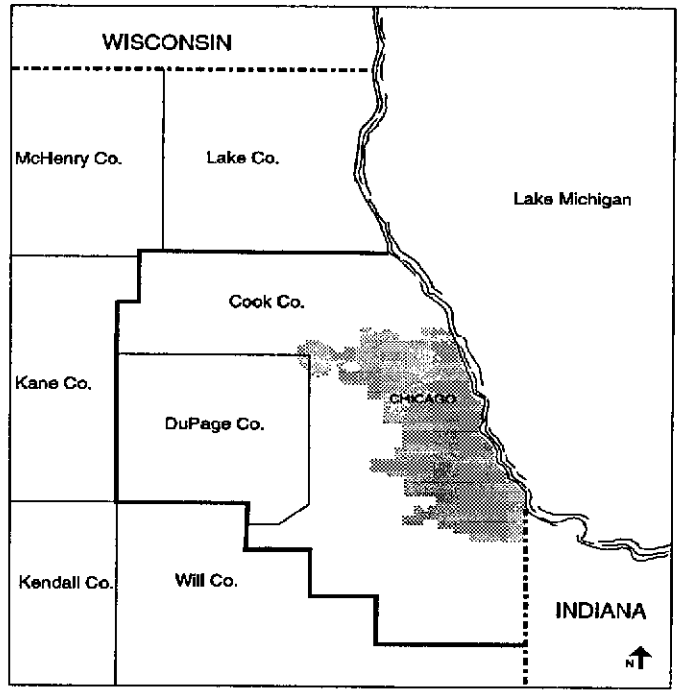

Figure 1. — Study area includes City of Chicago, suburban Cook County, and DuPage County.

Ground Sampling of Vegetation

Urban vegetation and other surface data were collected on 652 randomly located plots established as a sample of grid points (213 plots in Chicago, 222 in suburban Cook County and 217 in DuPage County). Because the focus of this study is on urban trees, the number of sample plots allocated to each land-use type was proportional to the estimated tree cover in the land use.[1]

Plot structure varied by land-use type.[2] Residential plots were subdivided into smaller ground units, whose area was measured to aid in estimating ground-surface cover (to the nearest 5 percent). Building size on each residential plot was measured and building-surface characteristics were noted. The amount of ground area occupied by various materials (tar, cement, buildings, small structures, other impervious material, maintained or unmaintained grass, shrubs, soil, herbaceous, rock, duff, water, wood) was measured or estimated on each plot.

[1] Overall, 249 plots were located on 1-3 family residential lands, 26 plots on multifamily residential lands (apartments with four or more units), 194 plots on institutional lands dominated by vegetation (e.g., parks, cemeteries, golf courses, forest preserves), 22 plots on institutional lands dominated by buildings (e.g., schools, churches), 52 plots on commercial/industrial lands, 45 plots on vacant lands, 39 plots on transportational lands (e.g., airports, freeways), and 25 plots on agricultural lands.

[2] On 1-3 family residential lands, the entire residential lot (mid-road to mid-alley) was measured. For other land use types, 0.04-hectare (ha) (0.1-acre) plots were measured.

Table 1. —Mean percent tree cover and standard error by land-use type in Chicago, suburban Cook County, DuPage County, and entire study area

Land use	Chicago Mean	Chicago SE	Cook Co. Mean	Cook Co. SE	DuPage Co. Mean	DuPage Co. SE	Study area Mean	Study area SE
Transportation (freeway)	3.8	0.8	0.5	0.5	0.0	0.0	1.1	0.3
Transportation (other)	1.8	0.3	2.1	1.0	2.4	2.0	2.1	0.7
Large commercial/industrial	2.9	0.3	2.4	0.5	1.4	0.6	2.3	0.3
Small commercial/industrial[a]	1.8	0.3	3.5	1.2	1.4	1.3	2.6	0.6
Agriculture	0.0	0.0	4.1	0.6	2.4	0.5	2.9	0.4
Institutional (building)[b]	7.1	0.7	6.4	1.2	9.9	1.9	7.3	0.8
Multiresidential[c]	6.6	0.5	8.9	1.7	10.2	2.7	8.1	0.8
Commercial (landscaped)[d]	12.1	7.7	15.6	6.8	6.3	6.1	11.5	4.5
Institutional (vegetation)[e]	26.4	1.0	16.7	1.6	20.4	2.2	19.7	1.1
Residential[f]	15.0	0.4	24.4	0.7	25.3	1.0	22.8	0.5
Vacant	19.6	1.5	39.2	1.9	31.7	2.3	33.7	1.2
Forest preserve	53.8	3.2	66.6	1.4	75.2	2.7	70.0	1.2
Total	11.0	0.2	22.5	0.4	18.6	0.5	19.4	0.3

[a] Small street-front commercial stores, etc.
[b] Dominated by buildings (e.g., schools, churches).
[c] Apartments with four or more units.
[d] Hereafter incorporated in the commercial/industrial land-use class in subsequent tables and analyses.
[e] Dominated by vegetation (e.g., parks, cemeteries, golf courses). This land-use class includes forest preserves in subsequent tables and analyses.
[f] 1-3 family residential units.

SE - denotes the standard error of the corresponding estimate.

The size and species of individual shrub masses were recorded (length, width, height). On every 10th plot measured, stem diameters of individual shrubs at 15 cm (6 inches) above groundline were measured. Data were collected on 8,996 trees and shrubs that were growing in tree form (i.e., relatively large open-grown individuals). The data included species, trunk diameter at breast height (d.b.h. - diameter at 1.37 m or 4.5 ft), total tree height, height to base of crown, crown width, crown shape, percent of crown occupied by leaves, tree location (street-tree locations between sidewalk and road, or on median, were noted), and condition. Estimates of tree condition were based on foliage characteristics. Trees were rated as excellent if less than 5 percent of the crown showed dieback or leaf discoloration. Other ratings were good (5 to 25 percent dieback or discoloration), moderate (26 to 50 percent), poor (51 to 75 percent), dying (76 to 99 percent), and dead (no leaves).

Plot information was combined to produce aggregate estimates on vegetation and other urban-forest attributes by land-use type in each sector of the study area (Gerald Walton, USDA Forest Service, 1992, pers. commun.).

Leaf Area of Urban Trees

To estimate leaf-surface area of urban trees, data were collected from 54 healthy, open-grown park trees in Chicago that were selected specifically for their excellent condition (10 American elm, 10 green ash, 10 hackberry, 10 honeylocust, and 14 Norway maple). The crown height (base of crown to crown top) of sampled trees ranged from 3.4 to 9.1 m (11.2 to 29.9 ft); crown width ranged from 4.1 to 12.0 m (13.5 to 39.4 ft) and individual LAI's ranged from 0.7 to 12.5. The volume of each tree crown was mapped (including areas devoid of leaves) using a telescoping pole.[3] Crown height and distance from the tree base were measured at crown boundary points every 1.5 m (5 ft) vertically and at every 45° angle radially (i.e., eight points around the tree at every 1.5 m vertically). Ten 0.4 m^3 (14.1 ft^3) samples of foliage were collected from random points within the tree crown using a high-lift truck.[4] The number of leaves per sample were counted and approximately 30 leaves were randomly subsampled for analysis of leaf area. For samples with 50 leaves or less, all leaves were analyzed for leaf area. Individual leaf areas were measured with a leaf-area meter (CID Inc., Conveyor Area Meter CI251). Average sample leaf area (one-surface only) per unit crown volume (m^2/m^3) was extrapolated using the total crown volume (m^3) to estimate total leaf area for each tree. Following leaf-area analyses, all leaves were dried at 65°C (149°F) for 24 hours and then weighed.

Total leaf-surface area for smaller urban trees was obtained from Gacka-Grzesikiewicz (1980). Data from 34 trees (12 species) that ranged in crown height (H) from 0.7 to 12.8 m (2.3 to 42.0 ft) and in crown width (D) from 0.5 to 4.6 m (1.6 to 15.1 ft) were combined with field data on leaf-surface area

[3] A sliding pole that displays the height at the top of the pole.

[4] A computer program was written to map the measured tree-crown dimensions and calculate crown volume. Random distances along x, y, and z coordinates from the tree base were selected to determine sampling locations within each tree crown. Sample locations in the tree crown were approached with the high-lift truck bucket so as not to disturb the sample prior to leaf collection.

of individual trees to produce equations for estimating total leaf-surface area of individual urban trees based on crown parameters. Other variables included in the predictive equations were a factor for leaf-surface area based on the outer surface of the tree crown ($S= \pi D(H + D)/2$) (Gacka-Grzesikiewicz 1980) and average shading coefficients for individual species (percent sunlight intercepted by foliated tree crowns) (McPherson 1984).

Least-squares linear regression was used to produce two regression equations for estimating total leaf area of individual urban trees. One equation included shading coefficients, the other excluded shading coefficients to aid in estimating leaf area of species for which shading coefficients are unknown (40 percent of the total population). Because logarithmic equations slightly underestimate leaf area (Crow 1988) a correction factor of one-half of the estimated variance of the estimate was added to the untransformed value ($y = e^x + var(x)/2$) for each equation (G. Walton, 1993, pers. commun.).

The regression formula estimated for log-leaf area of trees with measured shading coefficients was:

$$\ln Y = -4.3309 + 0.2942H + 0.7312D + 5.7217Sh - 0.0148S \quad (r^2 = 0.91),$$

where Y = total leaf area (m^2), H = crown height (m), D = crown diameter (m), Sh = shading coefficient (Appendix A, Table 1), and $S=\pi D(H + D)/2$. The correction factor (0.1159), added to the untransformed estimate, resulted in the following estimate for leaf area:

$$Y = e^{-4.3309 + 0.2942H + 0.7312D + 5.7217Sh - 0.0148S} + 0.1159$$

For trees for which shading coefficients are unknown, the estimated log-leaf area relationship was:

$$\ln Y = 0.6031 + 0.2375H + 0.6906D - 0.0123S \quad (r^2 = 0.86)$$

The correction factor added to the untransformed estimated value was 0.1824.

Total leaf area, derived from trees in excellent condition, was adjusted according to the condition class of the tree. Estimates of total leaf area were multiplied by 1 for trees in excellent condition, by 0.85 for trees in good condition, by 0.625 for moderate trees, by 0.375 for poor trees, by 0.125 for dying trees, and by 0 for dead trees.

For trees with characteristics outside the range of conditions under which the regression equations were derived ($H > 12$ m, $D > 12$ m, $H/D > 3$, $S > 500$ or $S < 1$; $n = 759$, 8.4 percent of the sample), leaf area was estimated using a volumetric approach. The volume of individual crowns occupied by leaves (foliated-crown volume) was estimated based on measured crown height, width, shape, and percent of crown occupied by leaves. Average leaf dry weight (g/m^3) was calculated based on measured data and information from the literature on individual tree species (Winer et al. 1983; Nowak 1991). Factors for average leaf dry weight were applied to the foliated-crown volume to estimate total leaf dry weight of the tree. This estimate was converted to leaf area using conversion factors (m^2/g) calculated from measured data and from the literature (McLaughlin and Madgwick 1968; Monk et al. 1970; Gacka-Grzesikiewicz 1980; Box 1981; Shelton and Switzer 1984; Bacon and Zedaker 1986; Vose and Allen 1988; Reich et al. 1991; Cregg 1992). If no conversion data were found for an individual species, the genera average was substituted; if no genera data were found, the average conversion value for the hardwood or conifer group was used.

Relative dominance of a tree species was calculated as the total leaf-surface area of all trees of one species as a percentage of the total leaf-surface area of trees of all species. Reliable estimates of error of leaf area estimates could not be made because it was not possible to determine the amount of error regarding factors associated with estimates of leaf area, for example, regression formula transformations, conversions used in the volumetric approach, and adjustments for crown condition. Thus, standard errors are not reported for estimates of species dominance.

Average LAI's for individual trees were calculated by dividing the sum of leaf-surface areas by the sum of crown projections (individual ground area = $\pi D^2/4$). The total LAI for the study area was calculated by dividing the estimate of the total leaf-surface area in the study area by the total area occupied by trees (from aerial photograph interpretation) (McPherson et al. 1993). Ground projections based on aerial photographs account for the multiple layering effect of trees (combined effect of overstory and understory trees).

Results

There are approximately 50.8 million trees in the study area, with 4.1 million trees in Chicago, 31.8 million in suburban Cook County, and 14.9 million in DuPage County (Table 2). The largest proportion of trees (49 percent) is on institutional lands dominated by vegetation (e.g., parks, forest preserves, cemeteries, golf courses), followed by 1-3 family residential land (25 percent), and vacant land (21 percent) (Table 2). These land uses also have the highest tree densities with institutional lands dominated by vegetation having 563 trees/ha (228 trees/acre). Vacant lands have 488 trees/ha (197 trees/acre) and 1-3 family residential lands have 93 trees/ha (38 trees/acre) (Table 3). Overall tree density is highest in DuPage County at 173 trees/ha (70 trees/acre), followed by suburban Cook County with 169 trees/ha (68 trees/acre) and Chicago with 68 trees/ha (28 trees/acre) (Table 3). Most of the estimated leaf-surface area (87.5 percent) is on 1-3 family residential lands and institutional lands dominated by vegetation (Table 4).

Cottonwood and green/white ash are the most common species in Chicago. Buckthorn and green/white ash are most common in suburban Cook County, and willow and boxelder are the most common species in DuPage County (Table 5; Appendix A, Tables 2-6). Species that dominate in leaf area are cottonwood and green/white ash in Chicago, silver maple and American elm in suburban Cook County, and white oak and silver maple in DuPage County (Table 5; Appendix A, Tables 2-6). Composition and leaf-area dominance of tree species by land-use type for each sector of the study area are given in Appendix A, Tables 7-14.

Table 2. —Estimated number of trees (in thousands) by land-use type in Chicago, suburban Cook County, DuPage County, and entire study area

Land use	Chicago		Cook County		DuPage County		Study area	
	Total	SE	Total	SE	Total	SE	Total	SE
Institutional (bldg.)	73	55	0	0	57	27	130	61
Transportation	225	175	0	0	28	28	253	178
Agriculture	0	0	0	0	442	342	442	342
Multiresidential	199	134	232	89	153	31	584	164
Commercial/indust.	33	33	1,021	873	81	30	1,136	874
Vacant	494	248	3,863	1,455	6,443	2,406	10,799	2,822
Residential	1,258	180	6,712	586	4,529	647	12,500	892
Institutional (veg.)	1,845	505	19,978	3,300	3,163	706	24,985	3,412
Total	4,128	634	31,806	3,758	14,897	2,612	50,830	4,620

Table 3.—Tree density (no. trees/ha) by land-use type in Chicago, suburban Cook County, DuPage County, and entire study area (divide by 2.471 to convert stems/ha to stems/acre)

Land use	Chicago		Cook County		DuPage County		Study area	
	Total	SE	Total	SE	Total	SE	Total	SE
Institutional (bldg.)	25	19	0	0	20	9	9	4
Agriculture	0	0	0	0	26	20	12	10
Transportation	40	31	0	0	13	13	15	10
Commercial/indust.	2	2	32	27	10	3	21	16
Multiresidential	34	23	56	21	70	14	48	13
Residential	52	7	91	8	124	18	93	7
Vacant	256	128	315	119	810	303	488	127
Institutional (veg.)	332	91	674	111	345	77	563	77
Overall	68	10	169	20	173	30	152	14

Table 4. —Percentage of land area, total number of trees (tree population), and total leaf area within the study area, by land-use type

Land use	Land area	Tree population	Leaf area
Institutional (bldg.)	4.1	0.3	0.6
Transportation	5.2	0.5	1.0
Agriculture	10.6	0.9	0.4
Multiresidential	3.7	1.1	1.3
Commercial/indust.	16.3	2.2	0.8
Vacant	6.6	21.2	8.4
Residential	40.2	24.6	49.7
Institutional (veg.)	13.3	49.2	37.8
Total	100.0	100.0	100.0

Table 5. —Tree-species composition in Chicago, suburban Cook County, DuPage County, and entire study area; includes top 20 species in number and percentage of trees and species dominance based on percentage of total leaf-surface area in each sector

Species	Tree population				Species dominance	
	Number	SE	Percent	Rank	Percent	Rank
CHICAGO						
Cottonwood	535,900	303,100	13.0	1	15.8	1
Green/white ash	495,500	132,100	12.0	2	12.9	2
American elm	297,100	167,200	7.2	3	4.3	6
Prunus spp.	268,200	103,100	6.5	4	2.4	11
Hawthorn	259,500	105,500	6.3	5	1.9	17
Buckthorn	232,100	101,100	5.6	6	0.9	27
Honeylocust	189,000	43,800	4.6	7	3.4	8
Boxelder	178,900	86,700	4.3	8	2.0	15
Mulberry	166,600	49,600	4.0	9	2.3	13
Silver maple	124,700	26,800	3.0	10	7.2	3
Norway maple	122,600	30,900	3.0	11	6.7	5
Yew	112,000	87,700	2.7	12	1.6	20
Ash (other)	107,500	58,100	2.6	13	1.5	21
Ailanthus	89,200	29,900	2.2	14	4.2	7
Crabapple	77,700	28,500	1.9	15	1.9	18
Elm (other)	64,900	49,000	1.6	16	1.0	23
Hackberry	62,100	33,200	1.5	17	2.3	12
Chinese elm	60,000	30,000	1.5	18	0.9	26
Blue spruce	58,900	25,200	1.4	19	1.6	19
White oak	49,600	29,700	1.2	20	7.0	4
Swamp white oak	47,500	34,100	1.2	21	2.3	14
Red/black oak	29,000	26,000	0.7	27	2.5	9
Basswood	26,800	13,600	0.6	28	1.9	16
Linden	18,600	8,900	0.5	31	2.5	10
SUBURBAN COOK COUNTY						
Buckthorn	4,601,600	1,430,800	14.5	1	2.9	12
Green/white ash	3,181,900	745,300	10.0	2	9.6	3
Prunus spp.	2,619,300	660,100	8.2	3	4.0	9
American elm	2,126,400	741,700	6.7	4	9.8	2
Boxelder	1,757,800	447,200	5.5	5	4.6	6
Hawthorn	1,715,600	440,100	5.4	6	3.6	10
Alder	1,337,200	1,130,400	4.2	7	0.5	33
Silver maple	1,220,200	287,900	3.8	8	10.9	1
Red/black oak	1,044,100	328,200	3.3	9	5.2	4
Poplar (other)	841,400	527,800	2.6	10	1.3	21
Black locust	831,000	618,200	2.6	11	0.4	38
Slippery elm	732,900	582,800	2.3	12	1.2	23
Cottonwood	715,700	352,600	2.3	13	3.0	11
Sugar maple	590,400	507,600	1.9	14	1.4	20
White oak	540,100	236,200	1.7	15	4.5	7
Crabapple	490,800	100,300	1.5	16	1.8	15
Honeylocust	430,400	81,200	1.4	17	1.7	16
Mulberry	414,500	132,200	1.3	18	1.2	22
Bur oak	408,000	211,400	1.3	19	1.6	18
Norway maple	407,900	110,700	1.3	20	4.3	8
Willow	317,400	99,800	1.0	26	5.0	5
Swamp white oak	123,100	55,100	0.4	38	2.5	14

Table 5. —continued

Species	Tree population				Species dominance	
	Number	SE	Percent	Rank	Percent	Rank
DUPAGE COUNTY						
Willow	1,819,400	1,754,000	12.2	1	2.3	15
Boxelder	1,630,900	454,500	10.9	2	6.2	3
Buckthorn	1,619,400	572,600	10.9	3	3.7	8
Prunus spp.	1,253,100	333,100	8.4	4	4.3	7
Green/white ash	950,200	381,400	6.4	5	5.2	5
Cottonwood	658,600	442,500	4.4	6	3.4	10
Hawthorn	650,900	175,000	4.4	7	1.2	22
Shagbark hickory	520,700	295,800	3.5	8	2.6	13
American elm	458,200	168,300	3.1	9	4.5	6
Mulberry	299,300	88,300	2.0	10	2.5	14
Red/black oak	299,100	131,100	2.0	11	1.9	16
Blue spruce	295,700	92,900	2.0	12	1.9	17
Silver maple	286,800	47,900	1.9	13	9.4	2
Bur oak	275,700	109,700	1.9	14	5.7	4
Basswood	243,500	144,400	1.6	15	1.3	20
Black locust	236,900	157,300	1.6	16	0.9	25
Jack pine	234,300	169,800	1.6	17	0.2	39
White oak	218,200	66,900	1.5	18	17.3	1
Crabapple	211,200	28,900	1.4	19	1.6	19
Walnut	190,100	121,100	1.3	20	3.4	9
Norway maple	161,700	31,100	1.1	22	3.1	11
Pin oak	112,200	41,600	0.8	25	2.8	12
Honeysuckle	98,800	54,500	0.7	30	1.7	18
STUDY AREA						
Buckthorn	6,453,100	1,544,400	12.7	1	2.9	11
Green/white ash	4,627,500	847,600	9.1	2	8.7	2
Prunus spp.	4,140,600	746,500	8.1	3	3.9	9
Boxelder	3,567,600	643,500	7.0	4	4.8	5
American elm	2,881,700	778,700	5.7	5	7.6	4
Hawthorn	2,626,000	485,300	5.2	6	2.7	13
Willow	2,144,600	1,756,800	4.2	7	3.6	10
Cottonwood	1,910,200	641,900	3.8	8	4.6	6
Silver maple	1,631,600	293,100	3.2	9	10.0	1
Red/black oak	1,372,200	354,400	2.7	10	3.9	8
Alder	1,340,700	1,130,400	2.6	11	0.3	41
Black locust	1,073,000	637,900	2.1	12	0.5	35
Poplar (other)	885,600	528,200	1.7	13	1.0	25
Mulberry	880,300	166,500	1.7	14	1.7	17
Shagbark hickory	864,600	384,800	1.7	15	1.2	22
Slippery elm	841,100	588,200	1.7	16	0.9	28
White oak	807,800	247,300	1.6	17	8.5	3
Crabapple	779,700	108,200	1.5	18	1.8	15
Honeylocust	753,100	96,700	1.5	19	1.7	18
Norway maple	692,300	119,000	1.4	20	4.2	7
Bur oak	690,200	238,300	1.4	21	2.7	12
Siberian elm	332,800	86,100	0.7	31	1.4	20
Norway spruce	265,400	56,300	0.5	32	1.9	14
Walnut	264,100	127,100	0.5	33	1.4	19
Swamp white oak	171,700	64,800	0.3	41	1.8	16

Common and/or dominant species that contribute the most leaf area on a per-tree basis are white oak, swamp white oak, Norway spruce, silver maple, and Norway maple (Table 6). Species that contribute the most large-diameter trees to the study area are silver maple, white oak, American elm, bur oak, and cottonwood (Table 7). Common small-diameter tree species are buckthorn, *Prunus* spp., green/white ash, boxelder, and willow (Table 8).

Fifty-six percent of the trees in the study area are less than 7 cm (3 inches) in diameter and 76.9 percent are less than 15 cm (6 inches) d.b.h. (Table 9). Chicago has the highest proportion of large trees greater than 46 cm (18 inches) d.b.h. (7.5 percent). Land uses with the highest proportion of large trees are institutional land dominated by buildings (29 percent) and 1-3 family residential land (10 percent) (Appendix A, Table 15).

About 55 percent of the trees in the study area were rated in good condition and 10.5 percent were rated as dead or dying (Table 10). Land uses with the highest proportion of dead and dying trees are institutional land dominated by vegetation (16 percent), followed by institutional lands dominated by buildings (11 percent), and vacant land (9.5 percent) (Appendix A, Table 16).

Table 6. —Average leaf-surface area (m^2) per tree for top 20 species (in number and species dominance) in entire study area (index value is average species leaf area per tree divided by average leaf area per tree for entire population (81 m^2))

Species	Leaf area per tree	Index value
White oak	436	5.4
Swamp white oak	422	5.2
Norway spruce	292	3.6
Silver maple	253	3.1
Norway maple	253	3.1
Walnut	219	2.7
Siberian elm	171	2.1
Bur oak	162	2.0
Red oak	117	1.4
American elm	109	1.3
Cottonwood	100	1.2
Crabapple	94	1.2
Honeylocust	91	1.1
Mulberry	79	1.0
Green/white ash	77	1.0
Willow	70	0.9
Shagbark hickory	60	0.7
Boxelder	55	0.7
Poplar (other)	48	0.6
Slippery elm	43	0.5
Hawthorn	42	0.5
Prunus spp.	38	0.5
Black locust	20	0.2
Buckthorn	19	0.2
Alder	10	0.1

The average LAI of individual trees is 4.3 in Chicago, 4.2 in suburban Cook County, 4.5 in DuPage County and 4.3 in the study area. The maximum LAI calculated using the regression equations for an individual tree was 18.1 with only 0.05 percent of the estimated LAI's for individual trees greater than 15. The estimated LAI for the entire study area, which accounts for the multiple layering of trees, is 6.3. The overall LAI may be slightly overestimated because of a likely conservative estimate of tree cover in Chicago. The large amount and size of buildings in Chicago tend to obscure small trees. This obstruction likely results in an underestimation of tree

Table 7. —Most common large trees given as percentage of total number of trees larger than 46 cm (18 inches) d.b.h.

Species	Percent
Silver maple	14.2
White oak	12.3
American elm	8.0
Bur oak	6.8
Cottonwood	6.7
Willow	5.5
Siberian elm	4.6
Green/white ash	4.6
Red oak	4.6
Honeylocust	4.6
Norway maple	2.5
Mulberry	2.2
Prunus spp.	1.5
Boxelder	1.5
Hawthorn	1.5

Table 8. —Most common small trees given as percentage of total number of trees less than 7 cm (3 inches) d.b.h.

Species	Percent
Buckthorn	18.7
Prunus spp.	8.9
Green/white ash	7.5
Boxelder	6.8
Willow	6.7
American elm	5.1
Hawthorn	4.6
Alder	4.4
Cottonwood	3.7
Black locust	2.5
Shagbark hickory	2.3
Red oak	2.2
Slippery elm	2.2
Sugar maple	1.8
Silver maple	1.5
Mulberry	1.4

Table 9. —Distribution of tree diameters in Chicago, suburban Cook County, DuPage County, and entire study area

D.b.h. class (cm)	Chicago		Cook County		DuPage County		Study area	
	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE
0-7	41.3	4.6	58.5	2.2	54.5	5.2	56.0	2.1
8-15	22.2	1.8	20.2	1.2	22.2	3.0	20.9	1.2
16-30	19.9	2.1	12.7	1.2	15.0	2.3	13.9	1.0
31-46	9.1	1.1	5.1	0.6	4.3	0.5	5.2	0.4
47-61	3.5	0.7	2.2	0.3	2.4	0.4	2.3	0.2
62-76	1.9	0.4	0.7	0.2	1.3	0.2	1.0	0.1
77+	2.1	0.8	0.6	0.2	0.4	0.1	0.7	0.1
All classes	100.0		100.0		100.0		100.0	

[a] Percentage of population

Table 10. —Distribution of trees by condition in Chicago, suburban Cook County, DuPage County, and the entire study area

Condition class	Chicago		Cook County		DuPage County		Study area	
	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE
Excellent	9.4	1.2	9.4	1.1	14.6	1.8	10.9	0.9
Good	50.5	3.5	56.0	2.4	53.1	4.4	54.7	2.0
Moderate	25.9	2.4	17.8	1.3	15.3	2.4	17.7	1.1
Poor	7.9	1.3	5.2	0.7	8.0	1.7	6.2	0.7
Dying	1.4	0.2	2.2	0.5	2.4	0.6	2.2	0.3
Dead	5.0	1.0	9.4	1.2	6.6	1.3	8.3	0.8
All classes	100.0		100.0		100.0		100.0	

[a] Percentage of population

cover and consequently a slight overestimation of the overall LAI. Thus, an overall LAI of 6.0 is probably more likely for the Chicago area. Conifers account for 6 percent of the leaf-surface area in the study area.

Populations of Street Trees

There are an estimated 1,463,700 street trees in the study area (SE = 151,900), with 416,000 in Chicago (SE = 48,500), 854,300 in suburban Cook County (SE = 139,400), and 193,400 in DuPage County (SE = 35,700). Norway maple and honeylocust are the most common street trees in Chicago, silver maple and green/white ash in suburban Cook County, and green/white ash and Norway maple in DuPage County (Table 11). Street trees in the study area tend to be larger than trees in general— 51.5 percent of all street trees are 16 to 46 cm (6 to 18 inches) d.b.h. (Table 12). Chicago has the highest proportion of large street trees with 28.7 percent larger than 46 cm d.b.h. (Table 12).

Most street trees in the study area were rated as good (46 percent) or excellent (34 percent) (Table 13). Only 0.5 percent were rated as dead or dying. No dead or dying street trees were found in Chicago or suburban Cook County. Street trees account for only 2.9 percent of the total tree population but 9.5 percent of the total leaf-surface area (Table 14). Street trees are most significant in Chicago where they account for 10.1 percent of the total population and 24 percent of total leaf-surface area. Dominance of street trees varies by land-use type with the greatest proportion occurring on residential lands in Chicago where street trees account for 27.9 percent of the trees and 43.7 percent of leaf-surface area (Table 14).

Urban Ground Cover

The most common ground surfaces in the study area are maintained grass, tar, and herbaceous plants; common surfaces in Chicago are tar, maintained grass, and buildings (Table 15). Ground cover varied by land-use type with maintained grass the most common ground cover type on institutional and 1-3 family residential lands, tar most common on commercial/industrial and transportational lands, herbaceous cover most abundant on agricultural and vacant lands, and building cover most common on multifamily residential lands (Appendix A, Table 17).

Table 11. —Top 25 street tree species in study area by sector

Species	Chicago			Cook County			DuPage County			Study area		
	Percent[a]	SE	Rank	Percent[a]	SE	Rank	Percent[a]	SE	Rank	Percent[a]	SE	Rank
Silver maple	13.1	4.0	3	26.5	9.1	1	17.0	6.8	3	21.5	5.5	1
Green/white ash	12.1	4.0	4	22.1	11.3	2	23.1	6.9	1	19.4	6.8	2
Norway maple	22.2	5.4	1	14.7	5.0	3	22.5	10.5	2	17.9	3.6	3
Honeylocust	22.0	6.5	2	3.2	3.0	8	7.0	4.2	4	9.0	2.6	4
Prunus spp.	0.0	0.0	--	8.9	7.1	4	1.3	1.3	14	5.4	4.1	5
Sugar maple	2.1	1.5	10	5.1	2.9	6	4.1	4.1	7	4.1	1.8	6
Linden	3.2	2.0	6	4.2	2.4	7	5.1	3.6	6	4.0	1.6	7
American elm	1.0	1.0	16	5.7	3.2	5	1.5	1.5	11	3.8	1.9	8
Chinese elm	6.3	6.3	5	0.0	0.0	--	0.0	0.0	--	1.8	1.8	9
Red/black oak	0.0	0.0	--	2.4	1.7	9	0.0	0.0	--	1.4	1.0	10
Siberian elm	0.6	0.6	19	1.8	1.8	10	0.0	0.0	--	1.2	1.0	11
Hackberry	0.6	0.6	18	1.6	1.6	11	0.0	0.0	--	1.1	1.0	12
Pear	0.0	0.0	--	0.3	0.3	15	6.9	5.0	5	1.1	0.7	13
Maple (other)	0.0	0.0	--	1.6	1.6	12	0.0	0.0	--	0.9	0.9	14
Catalpa	2.8	2.0	7	0.0	0.0	--	0.0	0.0	--	0.8	0.6	15
Ailanthus	2.8	2.0	8	0.0	0.0	--	0.0	0.0	--	0.8	0.6	16
Norway spruce	2.7	2.7	9	0.0	0.0	--	0.0	0.0	--	0.8	0.8	17
Golden-rain tree	2.1	2.1	12	0.0	0.0	--	0.0	0.0	--	0.6	0.6	19
Basswood	2.1	2.1	11	0.0	0.0	--	0.0	0.0	--	0.6	0.6	18
Hawthorn	0.0	0.0	--	1.0	1.0	13	0.0	0.0	--	0.6	0.6	20
Pin oak	0.0	0.0	--	0.8	0.8	14	0.0	0.0	--	0.5	0.5	21
Red maple	0.0	0.0	--	0.0	0.0	--	3.4	2.4	8	0.4	0.3	22
Horsechestnut	1.3	1.3	13	0.0	0.0	--	0.0	0.0	--	0.4	0.4	23
White birch	1.2	1.2	14	0.0	0.0	--	0.0	0.0	--	0.3	0.3	24
Oak (other)	0.0	0.0	--	0.0	0.0	--	2.4	2.4	9	0.3	0.3	25
All species	100.0			100.0			100.0			100.0		

[a] Percentage of population

Table 12. —Diameter distribution of street trees in Chicago, suburban Cook County, DuPage County, and entire study area

D.b.h. class (cm)	Chicago		Cook County		DuPage County		Study area	
	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE
0-7	15.7	5.6	7.1	3.4	29.5	11.6	12.5	3.0
8-15	4.0	2.2	24.0	12.5	13.9	5.8	17.0	7.4
16-30	30.3	6.6	26.8	6.8	20.5	6.5	27.0	4.5
31-46	21.4	4.7	27.2	6.5	19.0	9.7	24.5	4.2
47-61	12.8	3.8	10.6	3.5	7.0	4.9	10.7	2.4
62-76	7.6	3.0	4.3	2.6	5.2	3.1	5.4	1.8
77+	8.3	5.3	0.0	0.0	4.9	2.9	3.0	1.5
All classes	100.0		100.0		100.0		100.0	

[a] Percentage of population

Table 13. —Distribution of street trees by condition in Chicago, suburban Cook County, DuPage County, and entire study area

Condition class	Chicago		Cook County		DuPage County		Study area	
	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE
Excellent	18.8	4.8	41.7	13.8	30.3	12.2	33.7	8.3
Good	52.5	9.3	41.2	9.3	55.0	11.0	46.2	6.2
Moderate	26.0	6.9	14.7	4.9	8.2	3.7	17.0	3.5
Poor	2.7	1.6	2.4	1.7	3.0	2.1	2.6	1.1
Dying	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dead	0.0	0.0	0.0	0.0	3.6	2.7	0.5	0.4
All classes	100.0		100.0		100.0		100.0	

[a] Percentage of population

Table 14. —Street trees as a percentage of total tree population (%POP) and percentage of total leaf-surface area (%LSA) in Chicago, suburban Cook County, DuPage County, and entire study area

Land use	Chicago		Cook County		DuPage County		Study area	
	%POP	%LSA	%POP	%LSA	%POP	%LSA	%POP	%LSA
Agriculture	NA	NA	NA	NA	0.0	0.0	0.0	0.0
Institutional (bldg.)	0.0	0.0	NA	NA	0.0	0.0	0.0	0.0
Vacant	1.0	0.9	0.0	0.0	0.0	0.0	0.0	0.1
Institutional (veg.)	0.7	6.3	0.0	0.0	0.0	0.0	0.1	0.6
Multiresidential	10.3	8.5	11.1	10.1	3.2	1.1	8.8	7.6
Residential	27.9	43.7	10.2	19.7	3.8	5.9	9.7	18.0
Transportation	11.5	5.5	NA	NA	0.0	0.0	10.3	3.8
Commercial/indust.	0.0	0.0	14.2	18.5	20.0	41.0	14.2	25.8
Total	10.1	24.0	2.7	9.5	1.3	3.6	2.9	9.5

Table 15. —Distribution of ground-surface materials in Chicago, suburban Cook County, DuPage County, and entire study area

	Chicago		Cook County		DuPage County		Study area	
Surface type	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE	Percent[a]	SE
Grass (maintained)	20.4	1.4	30.7	2.0	32.6	1.8	29.3	1.2
Tar	21.3	2.6	13.3	1.8	11.5	1.2	14.3	1.1
Herbaceous	3.4	0.7	12.6	1.5	20.1	2.0	12.9	1.0
Building	16.5	2.1	9.1	1.3	8.0	1.2	10.1	0.9
Cement	12.2	1.2	5.8	0.7	3.7	0.5	6.4	0.5
Soil	4.5	0.6	7.5	1.4	4.1	1.2	6.1	0.8
Shrub	2.4	0.5	6.2	0.7	6.4	0.7	5.5	0.5
Grass (unmaintained)	2.5	0.8	3.4	0.7	7.7	1.8	4.3	0.6
Other structure	4.2	0.4	3.5	0.9	1.9	0.2	3.2	0.5
Rock	4.9	1.4	2.8	0.7	1.3	0.2	2.8	0.5
Other impervious	5.8	2.0	1.4	1.0	0.3	3.0	1.9	1.0
Duff	1.2	0.4	1.9	0.4	1.4	0.3	1.6	0.3
Water	0.3	0.2	1.8	1.0	1.0	0.3	1.3	0.6
Wood	0.3	0.2	0.1	0.0	0.1	0.0	0.1	0.0
All surfaces	100.0		100.0		100.0		100.0	

[a] Percentage of population

Discussion

Urban Forest Structure in the Chicago Area

The Chicago area's urban forest is composed mostly of small trees less than 15 cm d.b.h. (76.9 percent). Small trees also account for the majority of trees in other cities. In Shorewood, Wisconsin, and Oakland, California, 67 percent and 60.9 percent of the trees are less than 15 cm d.b.h., respectively (Dorney et al. 1984; Nowak 1993a). However, the distribution of tree sizes varies among and within land-use types depending on the duration and intensity of vegetation management. Less-managed (e.g., vacant) or naturalistically managed lands (e.g., forest preserves) had the highest proportion of small trees. Highly managed areas, particularly those managed for a relatively long period (e.g., street trees, residential areas), tend to have a higher proportion of large trees. However, there are some large old remnant trees throughout the Chicago area, particularly in forest preserves.

Most of the trees in the study area were classified as being in good condition. Ratings on tree condition are affected by urban-environmental stresses (e.g., salt, soil compaction, vandalism, injury), plant competition (related to tree density) and natural aging processes (tree size), all of which tend to increase crown discoloration and dieback (e.g., Nowak and McBride 1991). Consequently, relatively few trees were rated as excellent. Most of the dead and dying trees are in areas with minimal maintenance, naturalistic management, or in areas with more large trees that are not intensively managed (institutional land dominated by buildings). Dead and dying trees tend to be removed in the more intensively managed areas.

Species Composition

The most common species is the exotic and highly invasive buckthorn, accounting for 12.7 percent of the tree population. Seven of the 10 most common trees are native; three are genera of both native and exotic species. Four of the eight most common species are native pioneer species: green ash, boxelder, willow, cottonwood. These species have a propensity to colonize sites but have a shorter lifespan than more shade-tolerant species (Spurr and Barnes 1980; Burns and Honkala 1990). These species are common on all land uses but most common on vacant lands where they account for 47 percent of the population. Buckthorn is common on the three land uses that contain 95 percent of the trees (institutional lands dominated by vegetation, 1-3 family residential, and vacant lands). These land uses include many areas with relatively low maintenance (e.g., tree stands), which facilitates invasion by buckthorn. The most common ornamental species, exclusive of major pioneer species, planted on residential lands are silver maple, *Prunus* spp., blue spruce, crabapple, mulberry, Norway maple, arborvitae, honeylocust, American elm, and junipers.

The most common trees in Chicago are cottonwood and green/white ash, which make up 25 percent of the city's tree population. Green/white ash, both a pioneer and common ornamental tree, is common on most land uses in Chicago and accounts for 12 percent of all trees in the city. Cottonwood, which generally is not planted as an ornamental species, is the most common tree on vacant lands and institutional lands dominated by vegetation in Chicago. These land uses contain many low maintenance sites which facilitate invasion by cottonwood.

Species and Individual Tree Dominance

The most dominant species in total leaf area are silver maple, green/white ash, white oak, and American elm. These four species most likely have the greatest impact on the surrounding environment and constitute 34.8 percent of total leaf-surface area. Institutional lands dominated by vegetation are dominated by American elm, white oak, green/white ash, and red/black oak (39.8 percent of total leaf-surface area); 1-3 family residential areas are dominated by silver maple, green/white ash and white oak (31.7 percent); and vacant lands are dominated by the pioneer species of cottonwood, boxelder, willow, and poplar (other) (50.7 percent). Although buckthorn is the most common tree in the study area, it accounts for only 2.9 percent of total leaf-surface area due to its relatively small size.

The greatest average leaf-surface area on a per-tree basis occurs on white oak, swamp white oak, Norway spruce, silver maple, and Norway maple. Management activities should be directed toward preserving dominant individuals in a healthy condition so that their large environmental and social benefits, relative to smaller trees, are sustained (e.g., Schroeder and Cannon 1987; Nowak 1994a,b).

Diameter-growth rates of individual open-grown urban trees are relatively high (Nowak 1994b) and these growth rates are explained partially by the average LAI of individual trees in the study area (4.3), which is near the index level of maximum net growth. The overall urban tree LAI of 6.0 is at the low end of the normal range of LAI's exhibited for deciduous forests (Barbour et al. 1980). This relatively low index level is understandable considering the relative lack of lower level canopy (understory trees) in some urban areas that are common in deciduous forests. The urban forest understory of more intensively managed land uses often is occupied by grass or impervious surfaces.

Street Trees

Street trees in Chicago constitute 1 of every 10 trees overall and 1 of every 4 trees in 1-3 family residential areas. Chicago's street trees contribute 24 percent of the total city leaf-surface area, and 44 percent of total leaf area on 1-3 family residential lands. Street trees play a less important role in less urbanized areas, but can still contribute significantly to the street-corridor environment (Schroeder and Cannon 1987).

In suburban Cook County, street trees constitute 1 of every 37 trees (9.5 percent of total leaf-surface area) and 1 of every 10 trees on residential land. In the least urbanized sector, DuPage County, street trees account for 1 of every 77 trees (3.6 percent of total leaf-surface area) and 1 of every 26 trees on residential land. Thus, street trees become a more important component of the urban forest in more urbanized areas as artificial surfaces and land-use activities compete for tree space.

A high percentage of street trees in the Chicago area are greater than 46 cm d.b.h. (Chicago: 28.7 percent; suburban Cook County: 14.9 percent; DuPage County: 17.1 percent). There is a 4 to 6 times higher percentage of large street trees than non-street trees. Large trees are important to the urban environment, contributing significantly more air quality and carbon dioxide sequestration benefits than small trees (see Nowak 1994a,b: Chapters 5 and 6, this report).

Urban Ground Surfaces

Besides trees, a wide range of other urban surfaces interact with the surrounding environment and affect local gas and energy exchanges, visual quality, human stress, etc. The most abundant urban ground surfaces in the study area are maintained grass, tar, herbaceous plants (e.g., agriculture crops) and buildings. Impervious surfaces cover 60 percent of Chicago, 33 percent of suburban Cook County, and 25 percent of DuPage County. Tar generally is the most common ground-surface cover of commercial/industrial and transportation lands. Maintained grass often is the most abundant surface on residential and institutional lands. Converting nonessential impervious surfaces (e.g., abandoned parking lots) to more pervious surfaces (e.g., soil) could facilitate the formation of vegetation and reduce surface runoff. Understanding how various urban surfaces interact to affect the local environment and city inhabitants remains to be investigated.

Factors Influencing Current Vegetation Patterns

Vegetation within urban and urbanizing areas changes through time and space. Land use is one of the most significant factors affecting local vegetation patterns and distribution. In conjunction with its associated patterns of buildings and other artificial surfaces, land use influences the space available for trees and to some extent whether those spaces will be filled with trees and how they will be managed. Most of the nearly 51 million trees in Cook and DuPage Counties are on institutional lands dominated by vegetation, 1-3 family residences, and vacant land. This distribution pattern is similar to that for trees in Oakland, California (Nowak 1993a). These land uses generally are the most amenable to tree growth in urban areas and are likely where most of the trees exist in U.S. cities. Management plans should consider differences in tree distribution among land-use types to optimize tree configurations across the entire urban area. By understanding tree variations among land-use types, managers could focus planting efforts in areas typically lacking trees and direct species composition in more heavily-treed areas to meet specific management objectives and enhance the local environment.

In regions such as the Chicago area where trees are readily established through natural seeding, available planting space that is not filled with trees often has been actively managed to prohibit trees (e.g., mowing, use of herbicides, planting of herbs, selective tree removal). Such activities are necessary for land uses such as agriculture, airports, prairies, and sporting fields, but uses such as residential, commercial, and some transportation corridors could be used to increase tree cover if desired.

Tree cover can be increased through education and other promotional efforts that support tree planting and maintenance and/or encourage reducing management activities that prohibit trees and thereby allow trees to become established on the site naturally. Natural tree establishment can facilitate the development of invasive species so management activities should be directed toward altering species composition if certain invasive species are deemed undesirable.

The intensity of urban development also influences the amount of trees in a city, with tree density generally decreasing with urbanization. Average tree density in the Chicago area ranged from 68 trees/ha (28 trees/acre) in Chicago to 173 trees/ha

(70 trees/acre) in DuPage County. There are two primary reasons for the decrease in tree density with increased urbanization. First, in more heavily urbanized areas, more of the land is occupied by uses that preclude tree establishment (e.g., commercial/industrial, transportation).[5] Second, tree space tends to be more limited in highly urban areas (i.e., residential lots tend to be smaller; impervious surfaces occupy a higher proportion of the ground area).

Tree density on residential and commercial land in Chicago is comparable to those in Shorewood, Wisconsin, for the same land uses (Dorney et al. 1984). Tree density from other urban areas are 120 trees/ha (49 trees/acre) in Oakland, California (Nowak 1993a) and 373 and 40 trees/ha (151 and 16 trees/acre) for portions of South Lake Tahoe and Menlo Park, California, respectively (McBride and Jacobs 1986). By contrast, the average live tree density on timberland in Illinois is 1,186 trees/ha (480 trees/acre) (Raile and Leatherberry 1988).

Besides affecting management and various environmental functions, tree density affects visual quality of a landscape. Optimal foreground density for aesthetic quality in municipal parks has been estimated at approximately 125 trees/ha (51 trees/acre) (Schroeder and Green 1985). High tree densities and large trees are also preferred along streets (Schroeder and Cannon 1987).

Most of the differences in vegetation patterns within the study area are due to differences in land-use distribution, intensity of urbanization, and age of development. Chicago is the oldest, most urbanized area while DuPage County is the most suburban to rural area with newer residential developments and the highest proportion of agricultural areas.

Directing Future Urban Forest Structure in the Chicago Area

The future urban forest in the Chicago area, as indicated by the distribution of tree species less than 7 cm d.b.h., is likely to be dominated by green/white ash, boxelder, willow, cottonwood, black locust, and shagbark hickory. Other common species (buckthorn, *Prunus* spp., hawthorns, alders) in this smallest d.b.h. class generally do not reach a dominating size. American elm also is a common small tree, but sanitation programs and/or the planting of cultivars that resist Dutch elm disease must be continued or utilized if American elms are to maintain a dominant position in the Chicago area's urban forest.

This probable future forest will mean a shift from silver maple and white oak that codominate today toward more invasive pioneer species. While silver maple, white oak, and bur oak account for one-third of the trees greater than 46 cm d.b.h., they make up only 3.3 percent of the trees less than 7 cm d.b.h. However, planners and managers can alter or direct future species composition and structure (Nowak 1993c).

[5] Rural areas also can have land uses where low tree densities are typical (e.g., agriculture, vacant land in desert areas).

Education and management can influence the amount, type, and location of urban vegetation (e.g., tree planting in backyards and parking lots) and thereby direct future urban forest structure to a desirable outcome. Trees are not appropriate in all locations or land uses. However, where trees are desirable, planning and management can facilitate proper urban forest structure. The more space available for tree planting that is not inhibited by the existing land use, the more the natural environment and local planning and management can influence vegetation structure (e.g., vacant lands, parks).

Management plans should consider directing current urban forest structure toward a future structure that enhances healthy, functional leaf-surface area and optimizes species composition to maximize both social and environmental benefits of trees. Management plans should be developed to meet specific local needs, for example, enhancing the scenic beauty of a park or reducing air pollution in a certain area. Managing for one need or to maximize one benefit may reduce some other benefits derived from urban trees, so local and regional management priorities and plans must be developed. Besides preserving large trees, multilayer forest structures (stand conditions) should be sustained where appropriate, and healthy canopies should be maintained to maximize many tree benefits. Also, ample water should be supplied to trees to optimize benefits that are linked with transpiration (e.g., removal of gaseous pollutants and reduced air temperatures).

Implications for Research

The equations developed to predict the leaf-surface area of individual urban trees appear to yield reasonable estimates when applied within the bounds in which the regression equations were developed. However, more work is needed on developing shading coefficients and leaf-area predictions for individual species, particularly for large trees and coniferous species. Also needed is additional research on urban-forest structure and its link to various functions for other U.S. cities to help clarify and determine existing urban-forest patterns and processes. Finally, researchers need to investigate changes in urban forest structure and functions through time to better predict and understand the dynamics of these ecosystems, and to determine how urban surfaces interact in affecting the local environment and inhabitants.

Conclusion

Urban forest planning and management can direct urban forest structure toward a desired outcome. One of the first steps in properly directing urban forest structure is to understand if, and what, changes are necessary by analyzing the existing urban forest structure. By understanding forest structure and determining the relationships between structure and forest functions, various social and environmental benefits can also be quantified. The Chicago area urban forest contains 50.8 million trees, approximately 9 trees per resident. Most of the trees are small and predominantly found on institutional, residential and vacant lands.

The current pattern of urban vegetation has been formed through both present and past human and environmental factors. Education of both the public and private sectors can facilitate directing future urban forest structure toward desired results as dictated by urban forest management plans. However, the urban environment (e.g., land uses) presents many constraints on urban forest structure that managers and planners must consider.

Relatively short-lived pioneer species contribute significantly to the Chicago area urban forest and are most prevalent on land uses with minimal or naturalistic management (e.g., forest stand conditions). Street trees are also important elements of the urban forest, particularly in the City of Chicago. Trees are just one of many surfaces that interact to influence the urban environment; other prominent ground surfaces include tar and grass.

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Literature Cited

Airola, T. M.; Buchholz, K. 1982. **Forest community relationships of the Greenbrook Sanctuary, New Jersey.** Bulletin of the Torrey Botanical Club. 109: 205-218.

Bacon, C. G.; Zedaker, S. M. 1986. **Leaf area prediction equations for young southeastern hardwood stems.** Forest Science. 32(3): 818-821.

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

Box, E. O. 1981. **Foliar biomass: data base of the International Biological Program and other sources.** In: Bufalini, J. J.; Arnts, R. R. eds. Atmospheric biogenic hydrocarbons. Volume 1. Emissions. Ann Arbor, MI: Science Publishers:121-148.

Boyd, J. B. 1983. **Natural reproduction of exotic and indigenous trees in three urban environments.** Milwaukee, WI: University of Wisconsin. 97 p. M.S. thesis.

Buhyoff, G. J.; Gauthier, L. J.; Wellman, J. D. 1984. **Predicting scenic quality for urban forests using vegetation measurements.** Forest Science. 30(1): 71-82.

Burns, R. M.; Honkala, B. H. 1990. **Silvics of North America, Volume 2, hardwoods.** Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture. 877 p.

Cregg, B. M. 1992. **Leaf area estimation of mature foliage of *Juniperus*.** Forest Science. 38(1): 61-67.

Crow, T. R. 1988. **A guide to using regression equations for estimating tree biomass.** Northern Journal of Applied Forestry. 5: 15-22.

Dawson, J. O.; Khawaja, M. A. 1985. **Change in street-tree composition of two Urbana, Illinois neighborhoods after fifty years: 1932-1982.** Journal of Arboriculture. 11(11): 344-348.

Derrenbacher, W. E. 1969. **Plants and landscape: an analysis of ornamental planting in four Berkeley neighborhoods.** Berkeley, CA: University of California, Berkeley, CA. 231 p. M.S. thesis.

Dorney, J. R.; Guntenspergen, J. R.; Stearns, F. 1984. **Composition and structure of an urban woody plant community.** Urban Ecology. 8: 69-90.

Gacka-Grzesikiewicz, E. 1980. **Assimilation surface of urban green areas.** Ekologia Polska. 28(4): 493-523.

Hedman, C. W.; Binkley, D. 1988. **Canopy profiles of some Piedmont hardwood forests.** Canadian Journal of Forest Research. 18: 1090-1093.

Horie, Y.; Sidawi, S.; Ellefsen, R. 1991. **Inventory of leaf biomass and emission factors for vegetation in California's South Coast Air Basin.** Tech. Rep. III-C. El Monte, CA: South Coast Air Quality Management District. 149 p.

Impens R.; Delcarte, E. 1979. **Survey of urban trees in Brussels, Belgium.** Journal of Arboriculture. 5: 169-176.

Jim, C. Y. 1986. **Street trees in high-density urban Hong Kong.** Journal of Arboriculture. 12: 257-263.

Kramer, P. J.; Kozlowski, T. T. 1979. **Physiology of woody plants.** New York: Academic Press. 811 p.

McBride, J. R.; Froehlich, D. 1984. **Structure and condition of older stands in parks and open space areas of San Francisco, California.** Urban Ecology. 8: 165-178.

McBride, J. R.; Jacobs, D. F. 1986. **Presettlement forest structure as a factor in urban forest development.** Urban Ecology. 9: 245-266.

McLaughlin, S. B.; Madgwick, H. A. I. 1968. **The effects of position in crown on the morphology of needles of loblolly pine (*Pinus taeda* L.).** American Midland Naturalist. 80(2): 547-550.

McPherson, E. G. 1984. **Planting design for solar control.** In: McPherson, E.G. ed. Energy-conserving site design. Washington, DC: American Society of Landscape Architects: 141-164.

McPherson, E. G.; Nowak, D. J.; Sacamano, P. L.; Prichard, S. E.; Makra, E. M. 1993. **Chicago's evolving urban forest: initial report of the Chicago Urban Forest Climate Project.** Gen. Tech. Rep. NE-169. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 55 p.

McPherson, E. G.; Rowntree, R. A. 1989. **Using structural measures to compare twenty-two U.S. street tree populations.** Landscape Journal. 8(1): 13-23.

Miller, P. R.; Winer, A. M. 1984. **Composition and dominance in Los Angeles Basin urban vegetation.** Urban Ecology. 8: 29-54.

Monk, C. D.; Child, G. I.; Nicholson, S. A. 1970. **Biomass, litter and leaf-surface area estimates of an oak-hickory forest.** Oikos. 21: 138-141.

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. 1993a. **Compensatory value of an urban forest: an application of the tree-value formula.** Journal of Arboriculture. 19(3): 173-177.

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

Nowak, D. J. 1993c. **Historical vegetation change in Oakland and its implications for urban forest management.** Journal of Arboriculture. 19(5): 313-319.

Nowak, D. J. 1994a. **Air pollution removal by Chicago's urban forest.** In: McPherson, E. G., Nowak, D. J. and 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.

Nowak, D. J. 1994b. **Atmospheric carbon dioxide reduction by Chicago's urban forest.** In: McPherson, E. G., Nowak, D.J. and 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.

Nowak, D. J.; McBride, J. R. 1991. **Comparison of Monterey pine stress in urban and natural forests.** Journal of Environmental Management. 32: 383-395.

Peterson, D. L.; Arbaugh, M. J.; Lardner, M. A. 1988. **Leaf area of lodgepole pine and whitebark pine in a subalpine Sierra Nevada forest.** Canadian Journal of Forest Research. 19: 401-403.

Profous, G. V.; Rowntree, R. A. 1993. **The structure and management of the urban forest in Prague, Czechoslovakia. I. Growing space in metropolitan Prague.** Arboricultural Journal. 17: 1-13.

Profous, G. V.; Rowntree, R. A.; Loeb, R. E. 1988. **The urban forest landscape of Athens, Greece: aspects of structure, planning and management.** Arboriculture Journal. 12: 83-107.

Raile, G. K.; Leatherberry, E. C. 1988. **Illinois forest resource.** Resour. Bull. NC-106. St, Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 113 p.

Reich, P. B.; Walters, M. B.; Ellsworth, D. S. 1991. **Leaf age and season influence the relationships between leaf nitrogen, leaf mass per area and photosynthesis in maple and oak trees.** Plant, Cell and Environment. 14: 251-259.

Richards, N. A.; Mallette, J. R.; Simpson, R. J.; Macie, E. A. 1984. **Residential greenspace and vegetation in a mature city: Syracuse, New York.** Urban Ecology. 8: 99-125.

Richards, N. A.; Stevens, J. C. 1979. **Streetside space and street trees in Syracuse —1978.** Syracuse, NY: State University of New York, College of Environmental Science and Forestry. 66 p.

Rowntree, R. A. 1984. **Forest canopy cover and land use in four eastern United States cities.** Urban Ecology. 8: 55-67.

Sanders, R. A. 1984. **Some determinants of urban forest structure.** Urban Ecology. 8: 13-27.

Schmid, J. A. 1975. **Urban vegetation: a review and Chicago case study.** Res. Pap. (vol. 161). Chicago, IL: University of Chicago, Department of Geography. 266 p.

Schroeder, H. W.; Cannon, W. N. 1987. **Visual quality of residential streets: both street and yard trees make a difference.** Journal of Arboriculture. 13(10): 236-239.

Schroeder, H. W.; Green, T. L. 1985. **Public preference for tree density in municipal parks.** Journal of Arboriculture. 11(9): 272-277.

Shelton, M. G.; Switzer, G. L. 1984. **Variation in the surface area relationships of loblolly pine fascicles.** Forest Science. 30(2): 355-363.

Spurr, S. H.; Barnes, B. V. 1980. **Forest ecology.** New York: John Wiley and Sons. 687 p.

Stevens, J. C.; Richards, N. A. 1986. **Village and city street tree resources: a comparison of structure.** Arboricultural Journal. 10: 45-52.

Talarchek, G. M. 1985. **The New Orleans urban forest: structure and management.** New Orleans, LA: Xavier University of Louisiana, University Research Center. 104 p.

Vose, J. M.; Allen, H. L. 1988. **Leaf area, stemwood growth, and nutrition relationships in loblolly pine.** Forest Science. 34(3): 547-563.

Whitney, G. G.; Adams, S. D. 1980. **Man as a maker of new plant communities.** Journal of Applied Ecology. 17: 431-448.

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.

Chapter 3

Investigation of the Influence of Chicago's Urban Forests on Wind and Air Temperature Within Residential Neighborhoods

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Abstract

Ongoing research is examining the degree to which climate that surrounds people and houses in residential neighborhoods in the City of Chicago and adjacent communities is influenced by trees. The general research approach is to use windspeed, air temperature, and humidity at the nearest airport as reference conditions to compare differences in these climate variables between points in residential neighborhoods. Regression analysis is used to develop models to relate climate differences to measures of urban structure. The climate variables were measured for about 11 months at O'Hare International Airport, at two other reference locations, and in residential neighborhoods. The measurements in neighborhoods were made with four portable meteorological systems that were moved to sample 39 locations during the study period. Preliminary analyses indicate that it is possible to derive equations to predict the effect of buildings on windspeed separately from the effects of trees. The practical application of this is that, upon completion of the analysis, equations will be available to indicate the effect on wind within a neighborhood if the numbers or sizes of trees are changed. A goal of the study is to derive similar equations for tree effects on air temperature. Over three summertime days, temperatures in residential neighborhoods were higher on average than at the airport, though they were sometimes lower and sometimes higher than at the airport, depending largely of the net radiation balance. In the middle of a day with clear skies and bright sun, temperatures were slightly higher in a narrow space between two buildings than in a front yard near street trees. The relationships between climate and urban structure will apply best in the Chicago area, but extrapolation to other areas with a similar general climate and urban structure should be possible. These relationships are necessary for predicting effects of trees on energy use in buildings, human thermal comfort, and air quality.

Introduction

In this paper we describe ongoing research that is examining the degree to which climate at the height of people and houses in the Chicago area is influenced by trees. The general approach is to measure windspeed, air temperature, and humidity and then to develop equations to relate differences in these climate variables to measures of urban structure. By urban structure or morphology, we mean here the three-dimensional pattern of buildings, trees, and ground-surface characteristics (paved, grass, water, bare soil, etc.). The degree of success that we have in developing the relational equations will largely determine our ability to evaluate the effects of trees on climate within the urban area. The equations or models must be able to separate tree effects from building effects. Average windspeed and air temperature are the climate variables for primary consideration, though possible influences of tree distribution on humidity will also be examined.

Trees can have a major impact on the human environment in residential neighborhoods (Heisler 1986a; Oke 1989). For example, tree influences on wind (Heisler 1990a), air temperature and humidity (Grant 1991), and solar and long-wave radiation influence energy use in buildings (Heisler 1986a, 1990b; McPherson 1994; McPherson et al. 1988), human thermal comfort, air quality (Nowak 1994a), growth of smaller vegetation, and insect distribution (Heisler and Dix 1991). The influence of trees on solar radiation is directly related to geometrical factors that, although complex, have been studied sufficiently to provide at least approximate quantification of tree influences (e.g., Heisler 1986b, 1991). However, considering either a point in a residential neighborhood or the neighborhood generally, few tree effects on below-canopy air—its motion, temperature, humidity, and polluting constituents—can be estimated with sufficient accuracy for planning purposes. Below-canopy refers to the space below the general level of the tallest trees or buildings.

There have been few measurements of wind within residential neighborhoods (Heisler 1990a), and most available study reports, though containing valuable information, are for one season of the year or for a small number of sampling points (e.g., McGinn 1983). The general pattern of analysis in this study follows that used in a previous study in central Pennsylvania that showed a strong relationship between tree cover in the upwind direction and reductions in average windspeed in several neighborhoods that were typical of suburban developments (Heisler 1990a). Earlier studies with measurements in Dayton, Ohio, initially demonstrated the feasibility of developing prediction equations by statistical methods to relate windspeed at street level to building dimensions in the central business district (Grant et al. 1985; Heisler and Grant 1987).

Many studies have investigated the influence of urbanization on air temperature in both the above- and below-canopy space. Air temperatures have been related to land use, and clear distinctions in spatial and temporal patterns of air temperature have been observed between, for example, parks with many trees and surrounding building areas. The parks generally are cooler. However, such studies do not indicate the separate effects of buildings and trees. For example, given park land with 30-percent tree cover, it does not follow that a nearby neighborhood with streets and houses will have a similar temperature pattern if tree cover there also is 30 percent.

In discussions of tree effects on energy use, the potential of trees to save air conditioning costs through reductions in air temperature by evapotranspiration is often mentioned and incorporated in models (e.g., Huang et al. 1987). However, trees influence air temperature through other important aerodynamic and thermodynamic effects. For example, the trees throughout a neighborhood influence wind flow, which in turn influences exchange of the air below the general level of tree crowns with the air above. Some measurements (McGinn 1983) suggest that with moderate tree cover in a residential neighborhood, air temperatures may tend to be higher than with either more or less tree cover. This could be the result of the trees in the moderate-cover neighborhood reducing the air exchange while allowing most of the solar radiation to penetrate to ground level. In a forest with a complete canopy, there is little exchange of air between above- and below- canopy layers, but little solar radiation penetrates to heat the ground and below-canopy air. A complete forest may be approximated by the trees in a neighborhood with high tree cover, whereas with moderate tree cover, the trees cause significant reductions in below- to above-canopy air exchange but relatively small reductions in penetration of solar radiation to below-canopy species. Though solar radiation penetration may be greater in neighborhoods with low than with moderate tree cover, air exchange may be sufficient in the low tree density neighborhoods to keep them cooler at the height of people and buildings than in the neighborhoods with moderate cover.

Analogies can be made between the effects of the aggregate of trees in residential neighborhoods and traditional tree row windbreaks (Heisler and DeWalle 1988, McNaughton 1989). In the protected zone close behind windbreaks, air temperatures tend to be higher during the day, than upwind or farther downwind. At night, air temperatures in the near lee behind windbreaks may be relatively low because there are large losses of heat from the ground by long-wave radiation and relatively little mixing between the sheltered air and air flowing above the windbreak. Of course, in residential neighborhoods the situation is more complex because of interactive effects of trees and buildings on wind flow, heat storage, and radiation exchanges.

This study was carried out in conjunction with two other meteorological studies in the Chicago Urban Forest Climate Project. One study includes a description of the relationship between general weather patterns and air-flow fields over the city of Chicago (Grant 1993). That work is essential for interpreting meteorological observations in this study. The general area for meteorological data collection (Figure 1) was identical to that described in the study of local-scale energy and water exchange (Grimmond et al. 1994: Chapter 4, this report); data from the fixed meteorological measurement points at O'Hare Airport, the tall tower (ISPT3), and the Belmont Harbor light tower provide the reference conditions for this study. The land-use database described in Chapter 4 provides information for quantifying the urban structure in this study.

A general assumption is that climate variables at the airport site, which is in the middle of a large open area, are uninfluenced by trees and buildings. For purposes of developing the predictive models in this study, the differences that we are seeking to model generally are those between the hourly averages of windspeed and air temperature at points in residential neighborhoods and the reference point at O'Hare Airport. These differences form the dependent variables in the analysis. Descriptors of the structure of trees and buildings around the climate sample points in the residential neighborhoods form the independent variables. Some of the descriptors are derived from plat maps and aerial photographs and analyzed via a geographical information system (GIS); others are derived from analysis of hemispherical photographs taken from the climate sample points. An important objective of this study is to evaluate the efficiency with which descriptors can be developed by the different methods.

If the predictive model building is successful, the models will provide research tools to answer such questions as: What happens to wind and air temperature at specified kinds of sites or generally in a neighborhood configuration if we add a given number of trees of given sizes? The models will apply most directly to Chicago residential neighborhoods that have building and tree cover densities within the range of those included in this study. With this same constraint on range of cover densities, the models could be extrapolated to other cities with similar climates. The minimum input required to use the models would be some quantification of existing building and tree structure and general weather data for the period of interest. Weather data could be in the form of averages for each hour of a typical year. These data sets are available for over 200 cities in the United States (National Climate Center 1981).

Windspeed, wind direction, air temperature, and humidity were measured with 10 sets of sensors that operated almost continuously for nearly 11 months. The sensors were distributed among the three reference points and 39 below-canopy locations in residential neighborhoods (Figure 2). In this paper we describe the methods of data collection and the methods being used in the analysis of the entire data set. That analysis is not yet complete, but a partial analysis for a sample of the total meteorological and urban structural data is presented here to illustrate the methods.

METHODS

Meteorological Instrumentation

The meteorological sensors measured averages of windspeed, wind direction, air temperature, and humidity along with associated maximum and minimum values and standard deviations from July 16, 1992, to June 14, 1993. The wind, temperature, and humidity sensors were mounted perma-

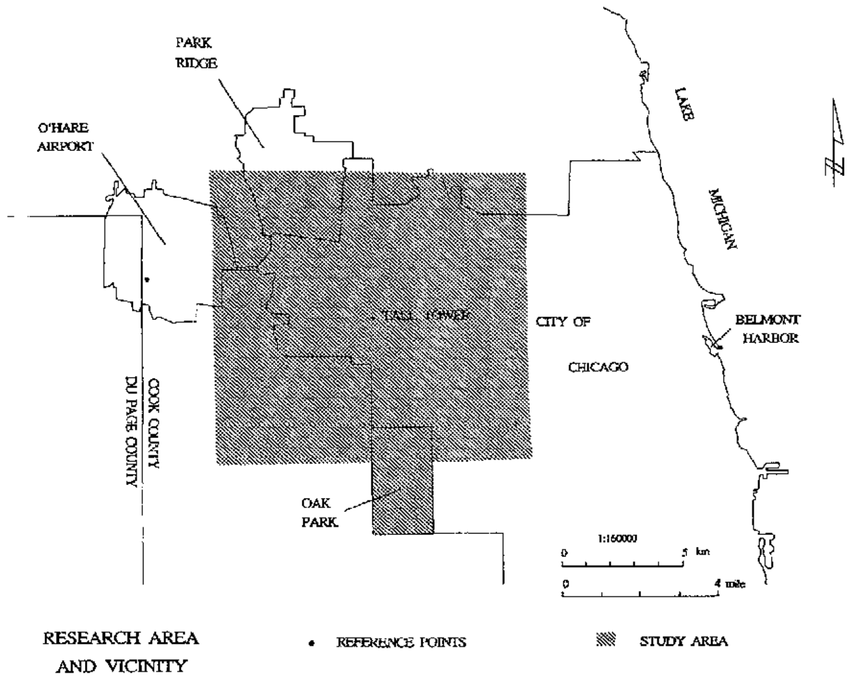


Figure 1.—Research area and meteorological reference points in and near Chicago. The tall tower is ISPT3 in Grimmond et al. (1994: Chapter 4,this report). The large portion of the shaded study area is bordered by Touhy Avenue on the north, Pulaski Road on the east, Chicago Avenue on the south, and Mannheim Road on the west.

nently at three reference locations: 1) within 8 feet (2.4 m) of the ground about 50 feet (15 m) from the National Weather Service instrument tower at O'Hare Airport; 2) at the 81-, 141-, and 228-foot (25-, 43-, and 69-m) levels on a radio tower about 6 miles (9.7 km) east of the airport location; and 3) on the shore of Lake Michigan at Belmont Harbor, about 15 miles (24 km) east of the airport (Figure 1). Specific instruments at the three reference sites are listed by brand name in Table 3, Chapter 4.

Below-canopy meteorological data were measured at the 39 sites (Figure 2) with five portable instrument packages mounted on TV antenna tripods (Figure 3) that were at a particular site for varying time periods. These measurements included air temperature and relative humidity at the 5-foot (1.5-m) height, and windspeed and direction at 7.8 feet (2.35 m).

Meteorological data were recorded on compact portable data loggers of a type that is widely used in environmental measurements. The loggers were programmed to provide instantaneous measurements every 5 seconds and, with one exception, average these over 15 minutes. For final analysis, the 15-minute averages will be combined into 1-hr averages of the meteorological data. There usually is a natural period in meteorological data near the surface of the earth such that averages over 30 minutes to 1 hour tend to represent the general trend of conditions, whereas averages over periods much shorter than 30 minutes include considerable random scatter associated with large-scale turbulent eddies (Panofsky and Dutton 1984). Because we had to substitute a data logger with a smaller memory for one that failed at O'Hare Airport, the averages there are over 1-hr periods for about 6 of the 11 months of data collection.

To acquire accurate temperature data, it is important to place the temperature sensor in a well-shielded and ventilated location to prevent errors from the influence of solar radiation on the temperature measurement. Although commercially produced shields are available, our experience is that none provides adequate shielding for the conditions we faced – some measurements in deep shade, some in full sun. With some temperature-measurement systems, errors frequently exceed 2°F (1°C). The requirement for battery operation for the portable units made design of the shield particularly crucial; the shields we used were designed specifically for this study (Grant and Heisler 1994). Each radiation shield held a small-bead thermistor inside a 1-inch-diameter inner tube and a combination temperature and humidity sensor that was protected only by a larger outer tube. A fan pulled air over both types of sensors. Tests of shielding efficiency suggest that the maximum radiation error for the small thermistor was about 0.18°F (0.10°C), whereas the maximum radiation error for the temperature sensor in the humidity unit was about 0.90°F (0.5°C).

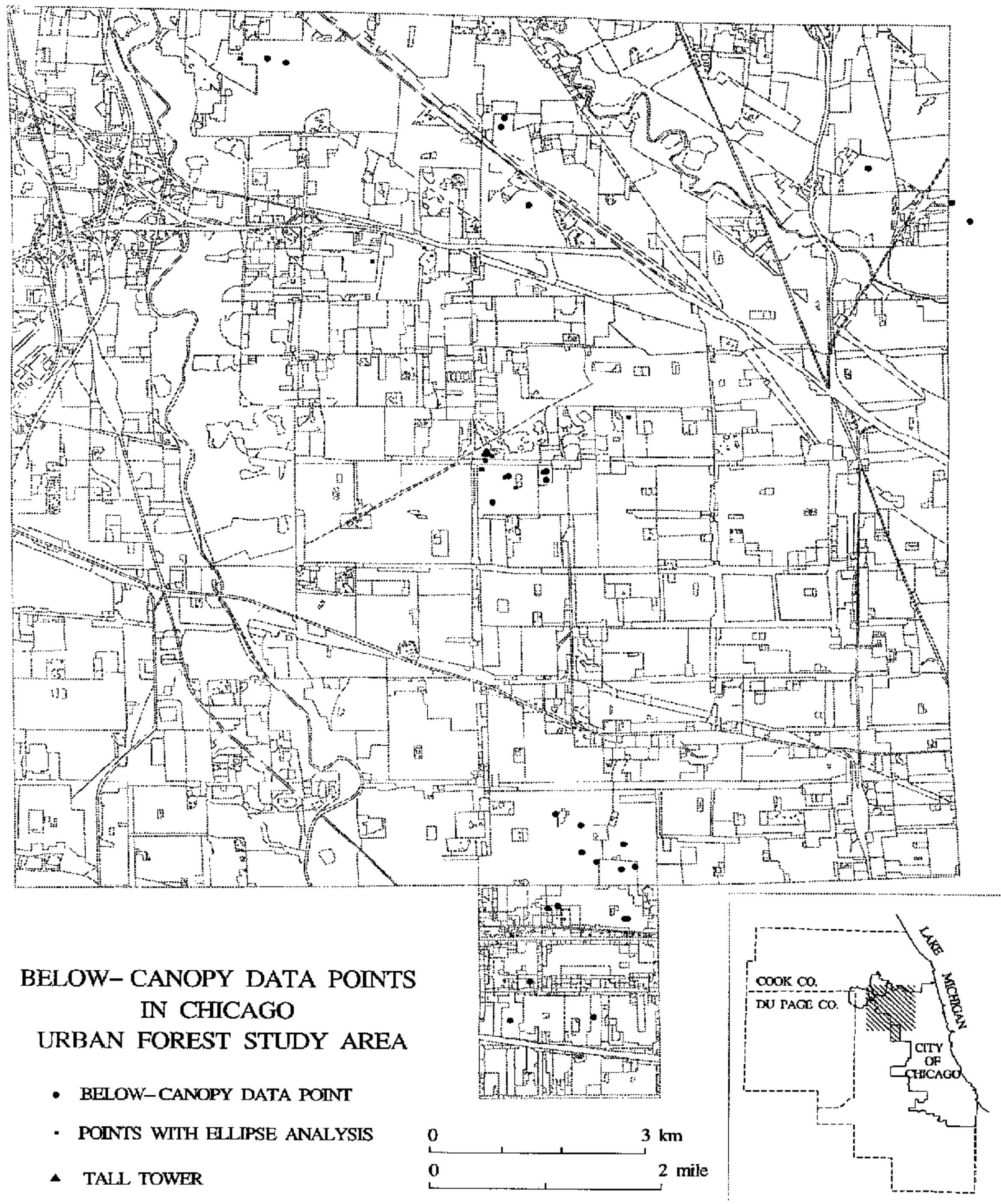


Figure 2.—Location of below-canopy meteorology sampling points and the pattern of land-use polygons in the land-use data base. Square symbols mark ten points used in ellipse spatial analysis.

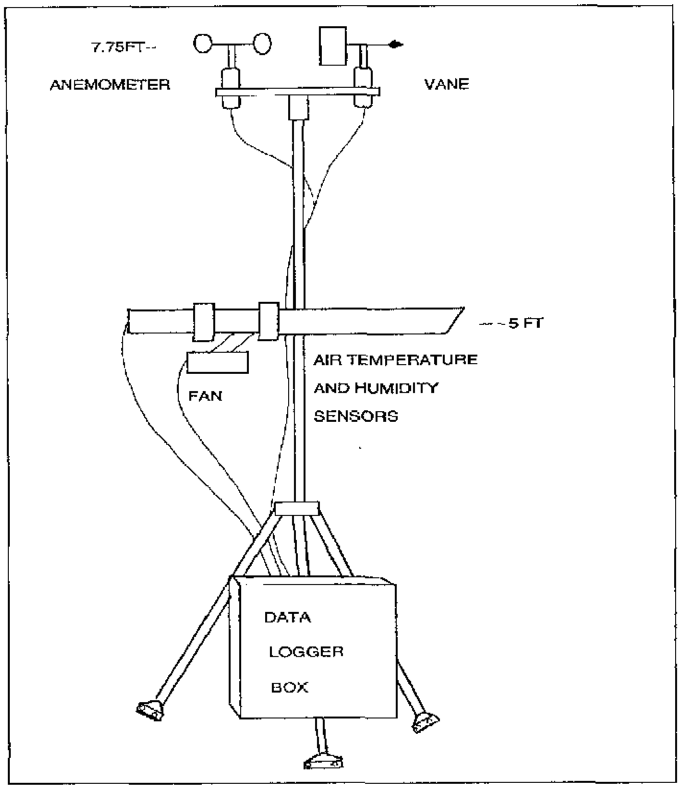


Figure 3.—Schematic of portable tripod and instruments for below-canopy measurements.

Each week, all sites in the network of meteorological instruments were visited for maintenance, to collect the data, and to move portable units scheduled for rotation. The below-canopy units generally performed well until mid-December 1992, when an ice storm apparently damaged some of the small-bead thermistors and caused some of the fans to fail. Fans on the below-canopy units and at the airport were changed, generally within several days of detected malfunctions.

Observation Site Selection

One of the five below-canopy units was maintained for the entire time in an area of tall grass near the ISPT3 tall tower (Figure 2). The other four units were rotated between sites in back yards, in front yards, in vacant lots, in narrow spaces between houses, and in an extensive woodlot, all between 3 and 9 miles (6 and 15 km) easterly from the airport, for 1 to 11 weeks (Table I). All except for the woodlot site (which is just off the east side of the GIS map) were in areas with 5 to 50 percent of the area covered by trees (Figure 4) and at least 10-percent coverage with trees, grass, and/or shrubs (Figure 5). A large proportion of points are located in Oak Park (Figures 1 and 2) partly because that community is developing a very complete tree inventory and GIS database of building structural features that will be made available for our analyses.

The sampling pattern and schedule had to be fairly flexible to accommodate homeowners' wishes. Location of the points depended partly on finding lawn space that was not heavily used for some purpose such as playing ball and where there was some degree of security. A goal was to sample each point in both summer and winter; however because of changes in ownership or homeowners' wishes, some points were sampled in only one season (Table I). Ideally, the rotation of instruments would have been done more frequently and each point would have been sampled several times during each season; however this was precluded by the limited availability of field personnel. More frequent rotation would have resulted in smaller differences between the sites in general weather conditions sampled. At some sites where the instruments provided particularly minimal inconvenience for the homeowner and also included morphologies that were in short supply elsewhere, we sampled for longer periods than at other sites.

If building and tree effects are to be separated in statistical models, it is necessary to sample over a wide range of both building and tree morphologies (particularly for areas covered by trees and buildings). Further, there must not be a high degree of correlation between the tree and building morphology. The number of points required to sample a sufficient range of building and tree morphologies depends in part on the variability of morphologies within the neighborhoods where measurements are made. To accommodate these requirements in so far as possible, we used aerial photographs and satellite images to visually explore the study area. We had some difficulty in finding a wide range of tree and building morphologies in the study area. Almost the entire area has older homes with relatively high building density and moderate tree cover. Tree cover tends to be inversely proportional to building density, and neighborhoods with either very low or very high cover are rare. We located the sample point in the woodlot to provide a sample of conditions at the upper limit of tree density. To the west of O'Hare International Airport there are many typical suburban neighborhoods with a wide range of building density and tree cover, but travel time and the lack of a tall tower reference prevented our sampling there.

Fortunately, the method of analysis, with the airport for a reference, greatly reduced the importance of uniform general weather conditions at each climate sampling point. Also, the range of structural conditions sampled varied substantially even at individual points, as the vegetation or buildings with greatest influence changed with wind direction. The Results Section has further discussion of the degree to which we succeeded in sampling in neighborhoods with differing morphologies.

For many of the points, a special effort was made to find lawn spaces between houses that were at least as wide as most of the houses so that meteorological conditions near the middle of the lawn would be representative of a possible house location. However, other points sampled a range of distances to nearest buildings, to dense conifer trees, to tall-crowned deciduous trees, and to hedges. Some points sampled narrow spaces between houses. In the prototype study by Heisler (1989 and 1990a), anemometers were located to sample the effects of the general aggregate of vegetation throughout the neighborhoods; dense tree rows and hedgerows were avoided. In this study we included the

Table 1. —Location of below-canopy meteorological instruments. Unit indicates which of the five below-canopy systems was used; and the "loc" column is the order of site placement, alphabetically, for that unit.

Site	Unit	Loc	Address	Total Days (Julian)	Started Date	Started Time	Finished Date	Finished Time	Leaves*
1	1	a	Irving Park Road and Harlem, Chicago	198-165	16 Jul 92	0900	14 Jun 93	1240	O,S,I,F
1	3	l	Irving Park Road and Harlem, Chicago	084-103	25 Mar 93	1147	13 Apr 93	1015	O
2	2	a	7915 Irving Park Rd., Chicago	199-206	17 Jul 92	1030	24 Jul 92	1000	I
3	3	a	3915 Neva, Chicago	199-206	17 Jul 92	1345	24 Jul 92	1025	I
3	3	k	3915 Neva, Chicago	068-082	9 Mar 93	1000	23 Mar 93	1245	O
4	4	a	3909 Neva, Chicago	199-206	17 Jul 92	1345	24 Jul 92	1045	I
5	5	a	3642 N. Nordica, Chicago	200-206	18 Jul 92	1030	24 Jul 92	1400	I
5	5	h	3642 N. Nordica, Chicago	033-047	2 Feb 93	1409	16 Feb 93	1131	O
6	2	b	3846 N. Sayre, Chicago 60634	206-212	24 Jul 92	1315	30 Jul 92	1600	I
6	3	h	3846 N. Sayre, Chicago 60634	026-040	26 Jan 93	1458	9 Feb 93	1319	O
7	3	b	3839 N. Nora, Chicago	206-212	24 Jul 92	1130	30 Jul 92	1730	I
7	3	j	3839 N. Nora, Chicago	054-068	23 Feb 93	1153	9 Mar 93	1639	O
8	4	b	6730 W. Byran, Chicago 60634	206-222	24 Jul 92	1230	30 Jul 92	900	I
8	3	i	6730 W. Byron, Chicago 60634	040-054	9 Feb 93	1435	23 Feb 93	1045	O
9	5	b	6727 W. Byron, Chicago 60634	214-217	27 Jul 92	1845	4 Aug 92	1917	I
10	2	c	7546 Bryn Mawr, Chicago	212-287	30 Jul 92	1630	13 Oct 92	1132	I
10	4	h	7546 Bryn Mawr, Chicago	012-033	12 Jan 93	1220	2 Feb 93	1446	O
11	3	c	6221 Knox, Chicago	212-252	30 Jul 92	1830	8 Sep 92	0835	I
11	5	f	6221 Knox, Chicago	330-357	25 Nov 92	1100	29 Dec 92	0930	O
12	4	c	6728 W. Byron, Chicago 60634	212-224	30 Jul 92	1617	11 Aug 92	1410	I
12	4	i	6728 W. Byron, Chicago 60634	033-047	2 Feb 93	1527	16 Feb 93	1103	O
13	5	c	4308 Moody, Chicago 60656	217-252	4 Aug 92	1648	8 Sep 92	1510	I
13	3	g	4308 Moody, Chicago 60656	364-026	29 Dec 92	1445	26 Jan 93	1407	O
14	4	d	Newland and Grace, Chicago	224-252	11 Aug 92	1515	15 Sep 92	1550	I
14	3	f	Newland and Grace, Chicago	329-364	24 Nov 92	1400	29 Dec 92	1345	O
15	3	d	5535 N. Linden Ave., Norwood Park	252-315	10 Sep 92	0915	10 Nov 92	1405	I,F
16	5	d	Pulaski Rd., Chicago	254-288	10 Sep 92	1200	17 Nov 92	1404	I
16	5	g	Pulaski Rd., Chicago	357-033	22 Dec 92	1505	2 Feb 93	0915	O
17	4	e	506 Western Ave., Park Ridge	259-315	15 Sep 92	1130	10 Nov 92	1239	I,F
18	2	d	505 Delphia, Park Ridge	287-321	13 Oct 92	1312	16 Nov 92	1235	F
19	4	f	6855 W. Thorndale	315-343	10 Nov 92	1330	8 Dec 92	0900	O
20	2	e	Pulaski Rd., Chicago	321-329	16 Nov 92	1515	24 Nov 92	0900	O
21	3	e	Pulaski Rd., Chicago	321-329	16 Nov 92	1200	24 Nov 92	0930	O
22	5	e	Pulaski Rd., Chicago	322-329	17 Nov 92	1500	24 Nov 92	0915	O
23	2	f	539 S. Chester Ave., Park Ridge	329-357	24 Nov 92	1500	22 Dec 93	1011	O
24	4	g	6460 Nordica, Chicago	343-012	8 Dec 92	0954	12 Jan 93	1134	O
25	2	g	7024 W. Devon Ave., Chicago, 60631	357-019	22 Dec 92	1104	19 Jan 93	1251	O
26	2	h	529 N. Harvey, Oak Park 60302	047-068	16 Feb 93	1349	9 Mar 93	1422	O
27	4	j	741 Fair Oaks Ave., Oak Park 60302	047-068	16 Feb 93	1300	9 Mar 93	1530	O
27	4	n	741 Fair Oaks Ave., Oak Park 60302	139-165	19 May 93	1134	14 Jun 93	0835	S,I
28	5	i	1133 N. Linden, Oak Park 60302	047-068	16 Feb 93	1230	9 Mar 93	1257	O
29	2	i	819 Mapleton, Oak Park 60302	068-082	9 Mar 93	1510	23 Mar 93	0925	O
29	2	l	819 Mapleton, Oak Park 60302	139-165	19 May 93	1245	14 Jun 93	0845	S,I
30	4	k	945 Fair Oaks Ave., Oak Park 60302	068-082	9 Mar 93	1621	23 Mar 93	1027	O
30	5	m	945 Fair Oaks Ave., Oak Park 60302	139-165	19 May 93	1210	14 Jun 93	0815	S,I
31	5	j	701 S. Elmwood, Oak Park 60302	068-082	9 Mar 93	1400	23 Mar 93	1145	O
32	2	j	233 N. Euclid, Oak Park 60302	082-084	23 Mar 93	1000	25 Mar 93	1310	O
33	4	l	213 S. Grove, Oak Park 60302	082-089	23 Mar 93	1130	30 Mar 93	1320	O
34	5	k	630 N. Lombard, Oak Park 60302	082-112	23 Mar 93	1215	22 Apr 93	1015	O,S
35	2	k	320 N. Euclid, Oak Park 60302	084-139	25 Mar 93	1325	19 May 93	1245	O,S
36	4	m	702 N. Elmwood, Oak Park 60302	089-139	30 Mar 93	1415	19 May 93	1130	O,S
37	3	m	725 S. Clinton	103-139	13 Apr 93	1145	19 May 93	0905	S
38	5	l1	175 N. Lombard, Oak Park 60302	112-117	22 Apr 93	1045	27 Apr 93	1430	S
39	5	l2	175 N. Lombard, Oak Park 60302	117-138	27 Apr 93	1430	18 May 93	1200	S

* I=in leaf, F=fall transition(Oct. 13- Nov.17, Days 287-322), O=out of leaf, S=spring transition(Apr. 13 to May 25, Days 73-115).

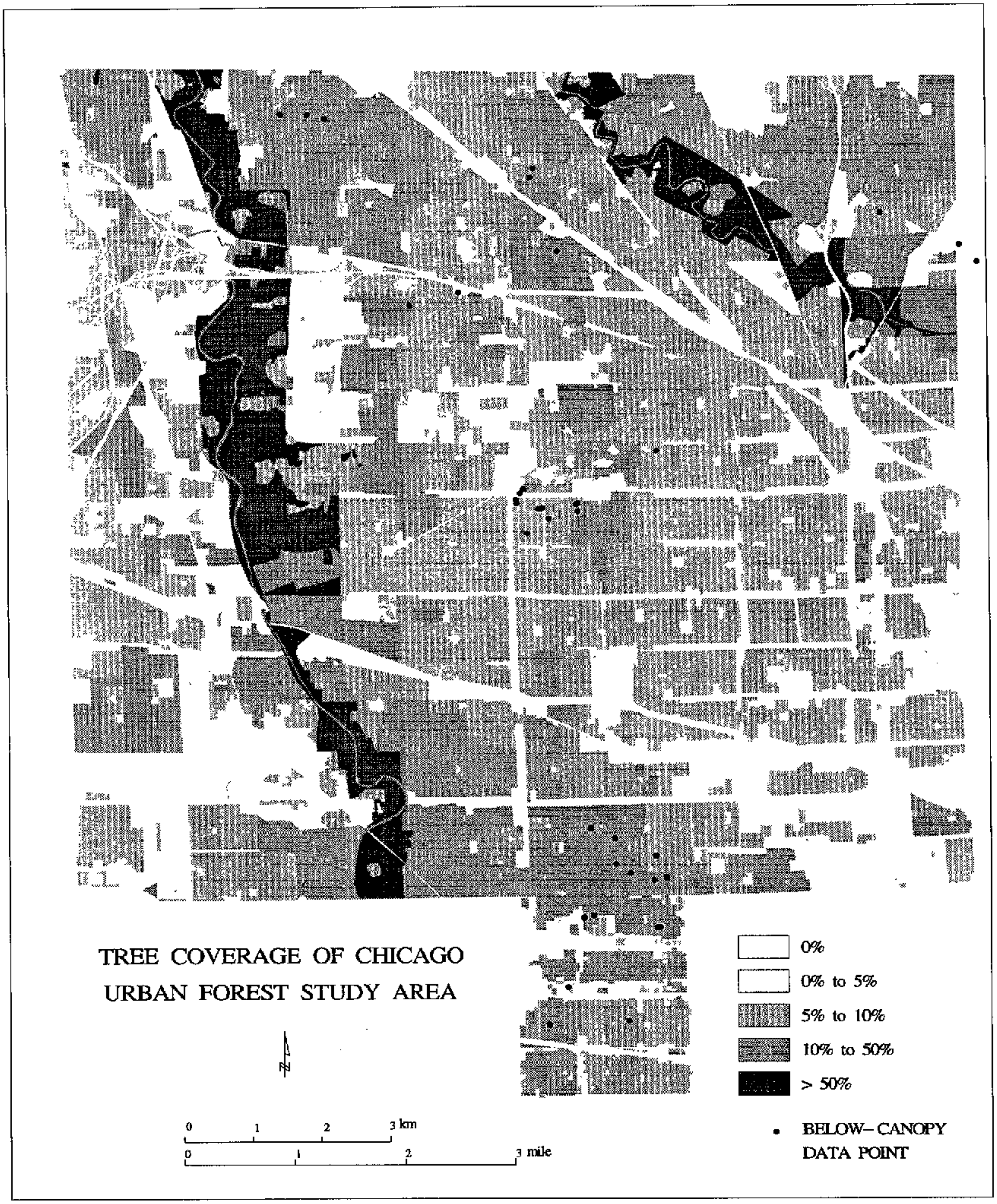


Figure 4.—Tree cover within study area and below-canopy points.

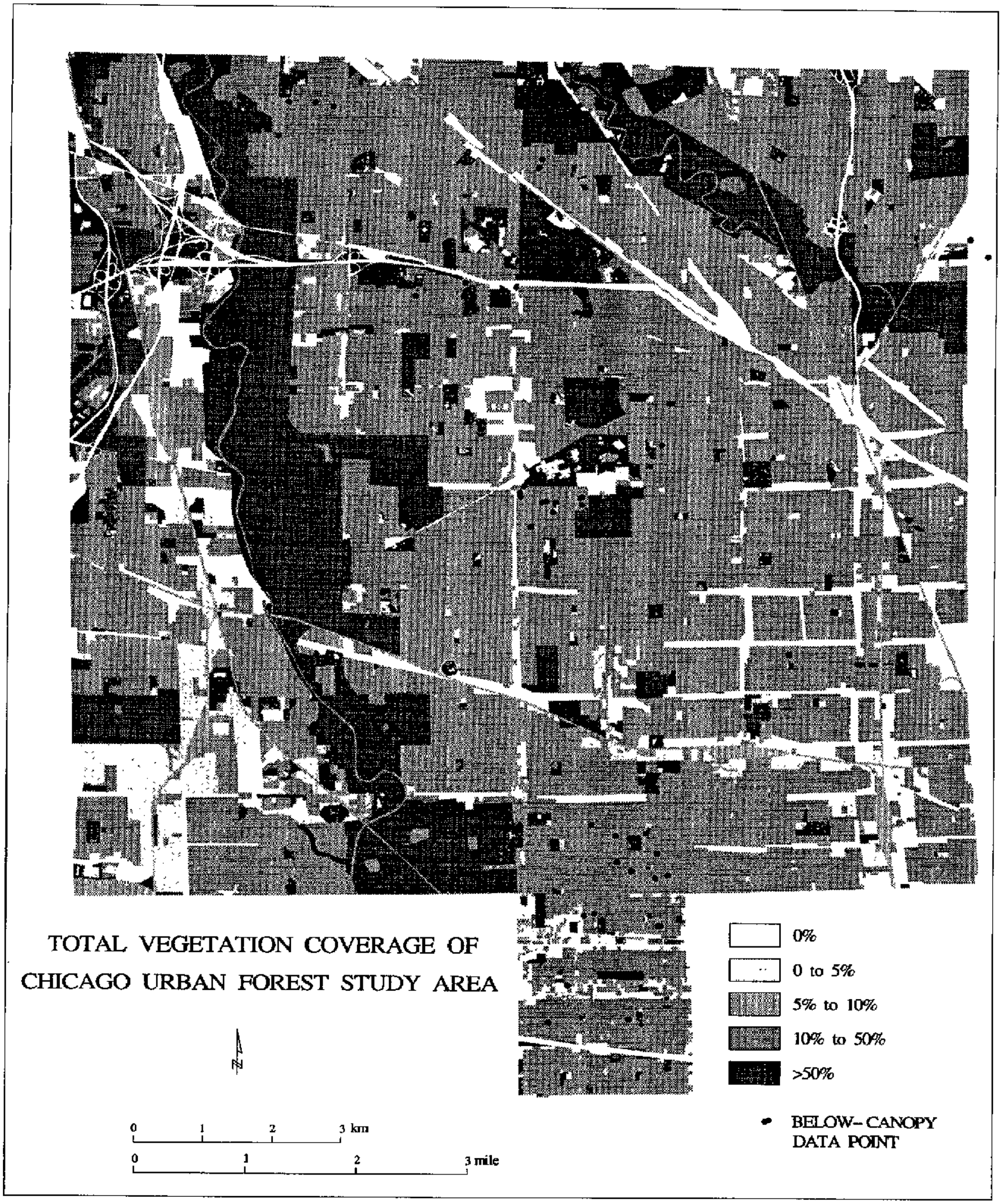


Figure 5.—Cover of all vegetation within study area and below-canopy points.

local effects of dense tree rows by locating some sampling points within one tree height of dense rows.

Reference Conditions

Although we make the assumption that the airport site is relatively uninfluenced by buildings and trees, we cannot assume that the general air flow over the airport site always is identical to the flow over the neighborhood sites, which are 3 to 9 miles (6 to 15 km) closer to the lake. Airport reference conditions will have to be adjusted to account for differences in wind, air temperature, and humidity between the boundary-layer air at the airport and over the below-canopy sites. The adjustments essentially will be an extrapolation from the airport conditions by first extrapolating vertically upward from the airport site, then across horizontally to above the residential neighborhoods, and then back down to the level of the below-canopy instruments at approximately 8 feet (2 m). The extrapolation must account for mesoscale variations, primarily the lake effect which prevails during part of the year (Grant 1993). The extrapolation will be derived for five classes of general (synoptic) weather conditions, as described in Grant (1993), so that for any hour of our observations, the lake effects can be estimated by knowing the general synoptic pattern. Vertical profiles of wind and air temperature derived from the three levels of measurement on the tall tower (ISPT3) along with the Belmont Harbor observations will facilitate the extrapolation. Indices of atmospheric thermal stability, which causes variations in the vertical profiles of wind and temperature, will aid in the extrapolations. The indices will be derived from our observations of net all-wave radiation (Grimmond and Cleugh 1994), which was measured at both the airport and ISPT3, and from the standard deviation of wind direction by a method of Slade (1968).

In the complete analysis, dependent variables will be formed as the differences between the values of windspeed and air temperature at the below-canopy sites and the extrapolated reference conditions. In the results presented here for tree and building effects on windspeed, the differences between the airport and below-canopy sites form the dependent variables, without extrapolation. This is a reasonable approach because results here are for essentially the same time period, and the below-canopy points are relatively close together.

Characterizing Urban Structure

Many characteristics of urban structure can be related to the meteorological differences that we measured. Looking from above in plan view, some possible characteristics are the areal coverage as a percentage or decimal fraction of buildings, trees, and impervious surfaces. Combined with these attributes, the average height of buildings and trees within land-use units adds the third dimension. These characteristics can be averaged over differently shaped and sized areas in the upwind direction in search of correlations with observed meteorological differences. Looking horizontally from below-canopy points, the heights of buildings and trees and the density of tree crowns in upwind directions, and to a smaller extent in downwind directions, also are related to microclimate, particularly windspeed.

In this study we are developing a set of independent variables to describe tree and building morphology, generally in the upwind direction from each below-canopy climate data point, to be entered into a data set with separate observations for each instrument-hour for each below-canopy point. The variables for describing the more distant morphology generally will be derived by GIS spatial analysis.

One source of data will be the surface database for the 8- by 8-mile (13- by 13-km) area used for hydroclimate analysis as described in Grimmond et al 1994: Chapter 4, this report. For each of the more than 2500 polygons shown in Figure 2, a set of attributes is assigned to indicate the percentage of area covered by buildings, trees, other vegetation or other surface characteristics (Table 6, Chapter 4). Because this database was developed for classes of land-use polygons, and some of the polygons have considerable variation in attributes within them, this database has limitations for developing descriptors of morphology for the near vicinity of particular points. The accuracy with which some of the attributes could be determined also was limited by the black-and-white aerial photos, which were available only for the leaf-off season for trees.

To provide land-use coverage for some of the sites near the edge or just off the original square area (Figure 2), we will digitize some additional areas on the northwest and northeast corners and around Oak Park. The sites included in the initial analysis reported here are near the center of the study area.

In our initial spatial-analysis to develop descriptors of morphology we used ARC/INFO GIS software, to average the attributes on an area-weighted basis across elliptically shaped areas in the upwind direction from each point. The ellipse shapes were cut from the coverage (cookie cutting) to determine the area of each land-use polygon within each ellipse as a proportion of ellipse area. The weighted average of an attribute within an ellipse was the sum over all land-uses in the ellipse of the attribute value for each polygon times proportional area. The attributes that have been used to date are: building cover; average building height; tree cover; total vegetation cover; and impervious, bare, and water-surface areas. The product of building cover times average building height forms an estimate of building volume (with dimensions feet^3 of building per foot^2 of land area), the building attribute that we expect to be most closely related to reductions in windspeed.

The spatial-analysis program averaged the attributes for ellipses centered on each 15 degrees for each of the below-canopy points. Thus, for each shape and attribute, there were 24 average values for each point. The average attributes were merged with the wind data by rounding wind direction over the residential area to the nearest 15-degree azimuth for which morphology averages were obtained in the spatial analysis. Wind direction at the ISPT3 tower is assumed to represent direction across the study area. The elliptical sample areas had lengths of 328, 984, 1640, and 3280 feet (100, 300, 500, and 1000 m), with widths equal to half the lengths, and with the downwind vertex over each below-canopy point. The spatial analysis for the ellipses has been completed for 10 of the 39 points. After the spatial analysis using ellipses was completed, average tree and shrub height was added as an attribute for each polygon, and this attribute will be used in any further analyses. The product of average tree and shrub height times tree and shrub cover fraction will provide an index of the volume of tree and shrub crowns.

Unlike the state of the technology related to above canopy source areas for vertical transfer of heat and vapor (Grimmond et al. 1994), there are few guidelines from previous experimentation that would aid in assigning appropriate shapes for averaging land-use structure that would relate to below-canopy microclimate. The elliptical averaging shapes were chosen for initial analysis partly because of their mathematical simplicity. Other shapes may better represent the land-use areas that influence wind and air temperature in the below-canopy space. The next step in analysis of the land-use database is to average attributes over sections of concentric circular bands at different distances from the below-canopy points. The band sections will be centered on mean wind direction and weighting will be applied according to angular distance from mean direction based on the standard deviation of wind direction on the tall tower during the sampling period. The band sections will be plus and minus 2 standard deviations, and weighting along the band, perpendicular to wind direction, will be based on area under a normal curve. Standard deviations on the tower are usually between 8 and 20 degrees. Hence, the band sections will range from about 30° to 80° wide as viewed from the below-canopy points. Five bands will be used: 0 to 100, 100 to 205, 205 to 410, 410 to 820, and 820 to 1640 feet from the point.

To provide more accurate descriptors of building morphology for areas near below-canopy points, another spatial GIS database of building footprints within 600 feet (180 m) of each below-canopy point (Figure 6) is being developed. The information sources are plat maps which are available for all Chicago locations and aerial photographs for other communities. A field survey and estimation from black-and-white stereo photos is providing approximate heights for each building. The building footprint database will provide average building density, height, and volume for differently shaped upwind areas, by a spatial analysis process similar to that applied to the larger land-use database. Ideally, color infrared aerial photographs for the trees-in-leaf season would have been available for development of a tree-cover database on the scale of the building footprint data, but no such current photos could be located.

The descriptors for building and tree morphology visible from the below-canopy points are being acquired from 180-degree hemispherical slide photos. These were taken at each point from a height of 3 feet (1 m) with the camera lens pointing directly overhead and with the top of the camera oriented toward north. The slides are projected onto polar grids from which technicians record, by 15-degree sector, average tree crown density and the maximum and minimum vertical angles from the horizon of the photo to the tops of visible buildings and trees. Tree crown density is estimated for upper and lower halves of the space between the horizon and the tallest tree within each sector. Separate photo sets were taken for the points where meteorological data were collected in both summer and winter. Changes in leaf phenology in the fall and spring transition periods (Table 1) were tracked with photos at a subset of the sample points.

Regression Analysis

Multiple regression models are being used to develop prediction equations to describe the influence of the vegetation and building morphology on the differences in airport to below-canopy wind and air temperature. Some of the morphological indicators are combined in physically meaningful ways prior to insertion in the model. For example, from the hemispherical photo data, distance to upwind buildings or trees relative to the building or tree height can be derived from the vertical angle from horizon to the top of the object. The product of normalized distances to upwind and downwind objects provides a descriptor that, if small, indicates that the point is between closely spaced obstacles and that wind tends not to penetrate downward into the canopy, but occurs mainly as skimming flow above the canopy (Oke 1987), resulting in large wind reductions below canopy.

The regression models are the usual general linear models with polynomial terms (Neter et al. 1985) or nonlinear models (Wilkinson 1990). The linear models are of the form

$$Y = B_0 + B_1X_1 + B_2X_2 + B_{12}X_1X_2 + B_{11}X_1X_1 + B_{22}X_2X_2 + ... + E \quad [1]$$

with E as the normally distributed error term with constant variance across all Y and X. In studying effects on windspeed, the dependent variable Y is, for example, a fractional reduction in windspeeds in the neighborhoods compared to the airport reference, and the X_i's are descriptors of either morphology or atmospheric conditions. In discussing wind reductions by trees, buildings, or other obstacles it is common practice to use a nondimensional normalized form rather than absolute windspeed (e.g., Heisler and DeWalle 1988, McNaughton 1989). Indices of atmospheric thermal stability calculated from vertical wind and temperature gradients, from net radiation (Grimmond and Cleugh 1994), or from windspeed and cloud cover (Turner's index, Panofsky and Dutton 1984) can be used to form descriptors of atmospheric conditions. The B_i's are regression coefficients. This is mathematically an additive effects model; each independent variable adds an effect, such as a fractional reduction in windspeed. The intercept B_0 will be near 0 if the X variables together account for most of the reductions in windspeed.

For studying effects of urban morphology on air temperature, the X_i's can include some of the same morphological characteristics as for windspeed in addition to others that are related to radiation exchanges, heat storage, moisture availability, and deficit of moisture in the air. Radiation exchange can be indexed by percent of unobscured sky above the below-canopy meteorological measurement point. In addition to building volume, heat storage may be significantly related to percentage of impervious cover from the land-use analysis. Impervious cover may also be related to moisture availability. Another index of moisture availability may be derived from the amount of precipitation over various lengths of time preceding the observation time. Moisture deficit is calculated as the difference between actual vapor pressure and vapor pressure if the air were saturated at the same temperature.

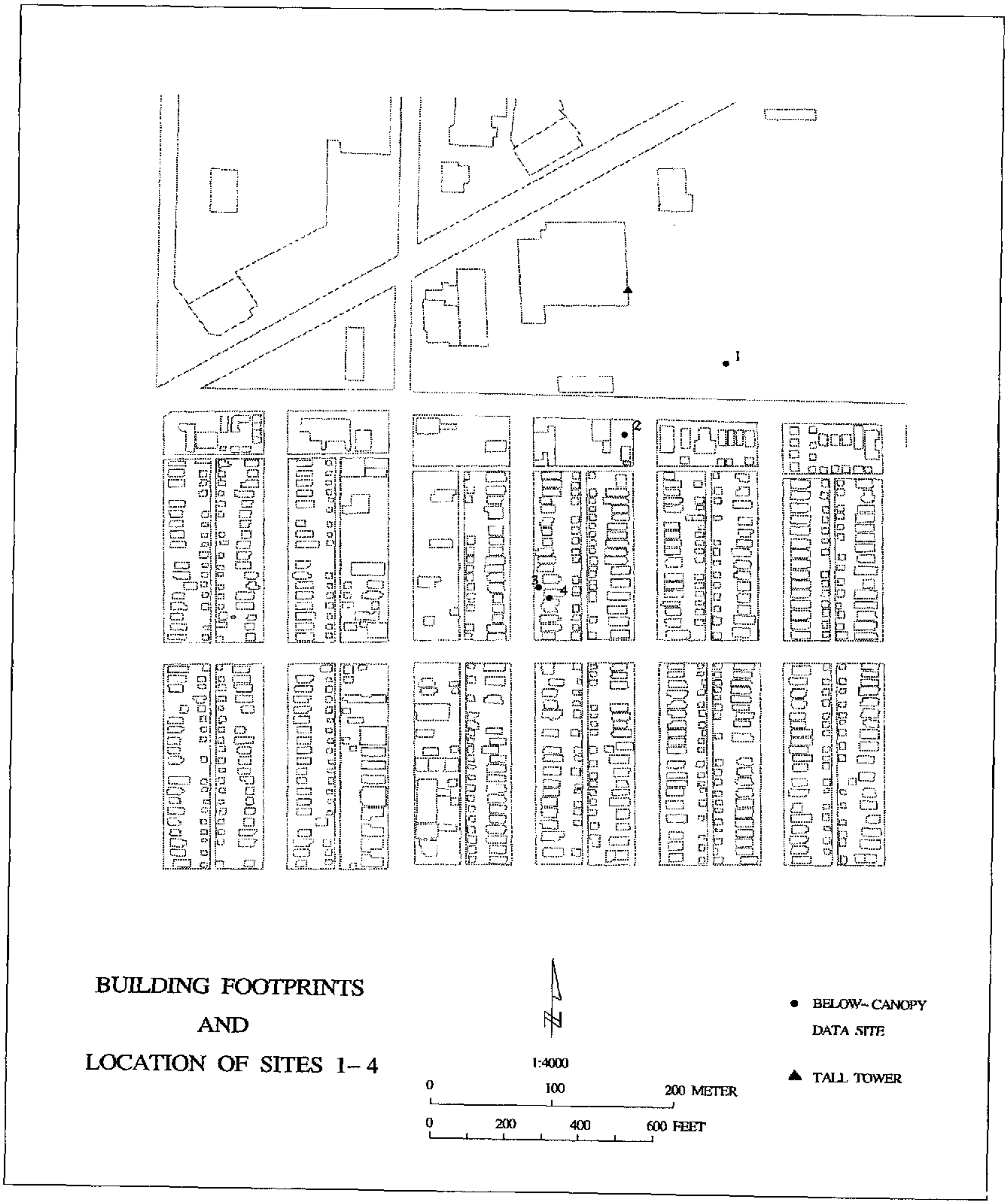


Figure 6.—Example of building footprints in GIS and location of four below-canopy points used in this analysis.

We might expect that the influence of morphology on microclimatic variables would be nonlinear. Nonlinear models can take various forms, such as

$$Y = B_0 \exp(B_1X_1 + B_2X_2 + \dots + B_nX_n) + E \qquad [2]$$

Here the Y would be, for example, a relative windspeed, that is, wind in the neighborhood divided by wind at the reference. Such models can be fit with standard nonlinear methods [e.g. SYSTAT (Wilkinson 1990)] depending on how many variables are included (interpretation of results becomes more difficult with each parameter that is added). Equation 2 is a multiplicative or exponential model, in that each independent variable has a multiplicative effect.

Results and Discussion

Land-Use Attributes

The study area has a complex pattern of land uses (Figure 4a, Chapter 4), including large areas in forest that are part of the Forest Preserve (areas with greater than 50 percent tree cover in Figure 4). Although overall tree cover is not high within Chicago (Nowak 1994a: Chapter 2, this report), the study area contains land-use categories with a wide range of tree cover (Figure 4). All vegetation combined typically covers 20 to 50 percent of the area in residential neighborhoods in which our below-canopy measurements were made (Figure 5).

One concern in interpreting the regression results is that some morphological descriptors that serve as independent variables are naturally correlated. Specifically, when building density is very high as in much of Chicago residential areas, tree cover generally also cannot be high. The relationship between building cover and tree cover is illustrated in the left side of Figure 7, which is derived from the land-use analysis with elliptic averaging shapes of different lengths and areas. The data for each scatter diagram are for 10 below-canopy points. Building cover ranged up to nearly 0.7 in some of the 328-foot (100-m) ellipses, and tree cover ranged up to about 0.4. The scatter of points shows a high degree of correlation between tree and building cover, particularly for the 328-foot ellipses. A small part of the reason for the close relation is an artifact of these data, because in development of the land-use database, only one type of coverage was allowed for any given sample point. Hence, where trees overhung buildings, the coverage category was trees rather than trees and buildings.

Steps can be taken to account for relationships between some independent variables in the regressions. The product of building-area coverage times height forms a building volume, which seems to be less well-correlated with tree cover (Figure 7, right column). Groups of below-canopy meteorological sites that have a wide range of morphological characteristics can be selected.

Initial Model Building

To illustrate the analysis that is being done to evaluate the effects of urban trees on wind, preliminary regression analyses were done for four sites, using a selection of the meteorological data collected within a 13-day period, July 21 (day 203) to August 2, 1992 (day 215). (The day of the year system is used because of ease of referring to dates in graphs.)

The sites

The locations of the sites, numbered 1 to 4, are plotted on a section of the GIS map of land-use in Figure 8. These four sites were all within 1000 feet (300 m) of each other and within about the same distance of the tall tower. Hence, these results serve to illustrate the range of microclimate within a short distance.

The hemispherical camera views (Figure 9) show the tree and building structure visible from each point. Site 1 was in a relatively open location in a large grassy field, but a natural stand of 25-foot (7.5-m) deciduous trees edges the north side of the field, about 75 feet (25 m) from the meteorological unit. Site 2 was in a vacant lot on the north edge of a residential development just 230 feet (70 m) south of site 1. Sites 3 and 4 were farther south within the development. Site 3 was in a small front yard along a street with many large street trees with crowns almost overhead; site 4 was in a narrow space between two houses.

General conditions

Windspeeds at O'Hare Airport ranged up to about 12 mph (5.5 m/s) between July 21 and July 24, days 203 through 206 (Figure 10). (Data for sites 2, 3, and 4 are available for these days only; site 1 also has data for days 212-215.) Windspeeds followed a diurnal pattern that is typical of locations within the atmospheric boundary layer—low speeds at night when the air becomes thermally stable because of radiational cooling near the ground. Figure 11 shows that day 203 had a smooth trace for both solar and net all-wave radiation, indicating a clear sky, resulting in high positive net radiation during the day and strong negative radiation at night compared to cloudy conditions on following nights). About 0.25 inch (3.8 mm) of rain fell on days 204 and 205 (Figure 8, Chapter 4).

Air temperatures

Air temperatures at below-canopy sites remained within 3.6°F (2°C) of the temperature at the same height at the airport (Figure 12a). Sites 2, 3, and 4, all in the residential neighborhood, were 0.5° to 0.7°F (0.28° to 0.39°C) warmer, on average, than the airport site. The general diurnal pattern, with temperatures in neighborhoods being warmer than the airport at night and cooler during the day is probably caused largely by different rates of heating and cooling in the neighborhoods compared to the airport. This pattern is fairly typical of the so-called urban heat island phenomenon (Oke 1987, 1989). For example, on day 203, which was cloud free, net radiation at night was strongly negative and open sites such as the airport cooled more quickly than the neighborhoods. This is more clearly seen in Figure 12b which shows that periods when sites 2, 3, and 4 were decidedly warmer than the airport (by up to 3.3°F or 1.8°C) are associated with negative net radiation. Neighborhood sites also tend to be warmer under periods of high positive net radiation resulting from high solar radiation. The fact that site 3 was close to trees and site 4 on the adjacent property was in a narrow space between two houses (Figure 9) appears to have resulted

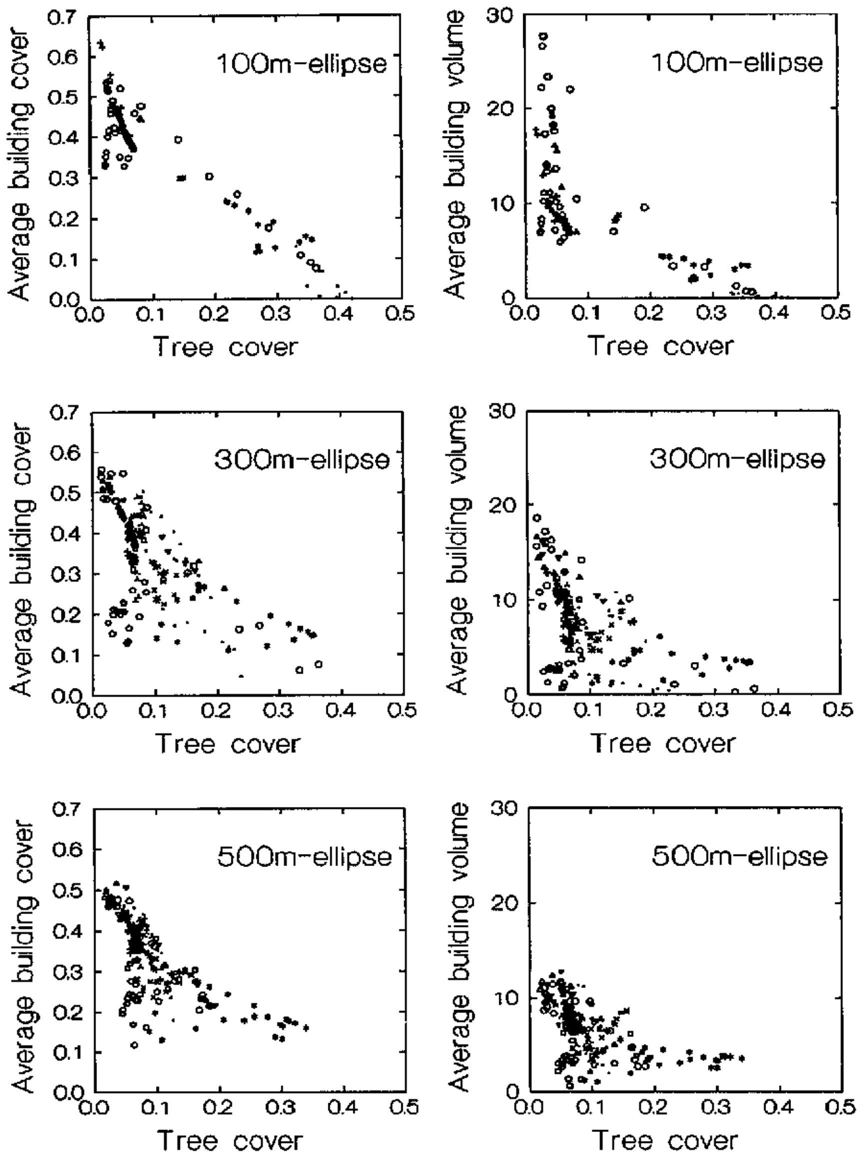


Figure 7.—Average building cover (fraction of area covered) and average building volume (cubic units of volume per squared units of area) versus tree cover (fraction of area covered) in elliptic sampling areas cut from the GIS database around ten of the below-canopy points. Different symbols show values for different points.

in site 3 being about 0.5°F (0.3°C) cooler at high values of net radiation (Figure 12b), even though the difference in overall average temperatures at the two sites was within the limits of instrumental error (0.18°F). Site 1 was cooler on average than the other below-canopy sites and had nearly the same mean temperature as the airport. The pattern of actual temperatures during days 203 through 206 (Figure 13) generally reflects the influence of the radiation balance, with a large diurnal swing accompanying the period of clear skies.

Effects of morphology on windspeed

Figure 10 shows that except for a few 15-minute observation periods with low windspeed at the airport, windspeeds at the below-canopy sites were lower than at the airport. However, there is considerable scatter in the 15-minute averages. A

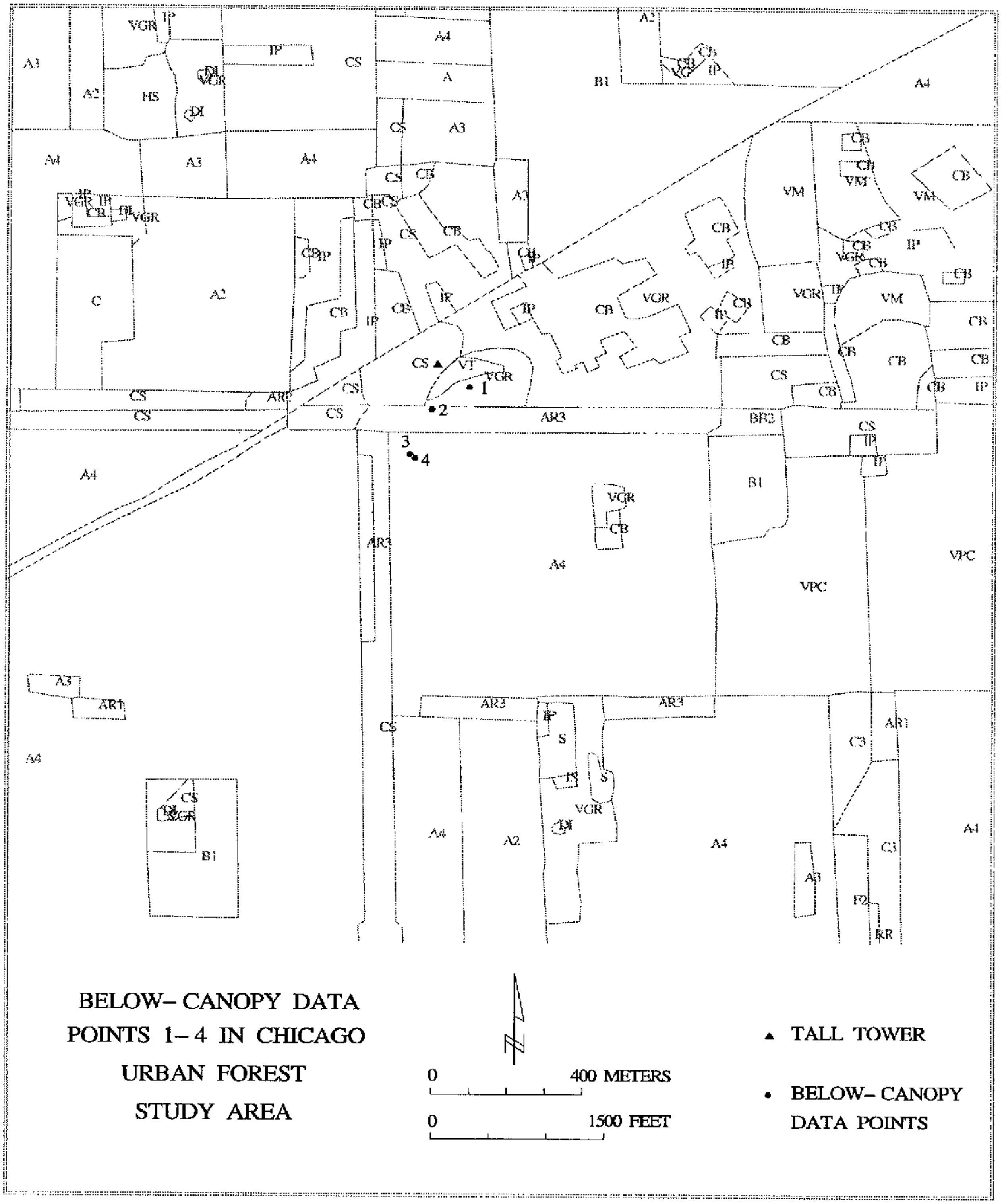
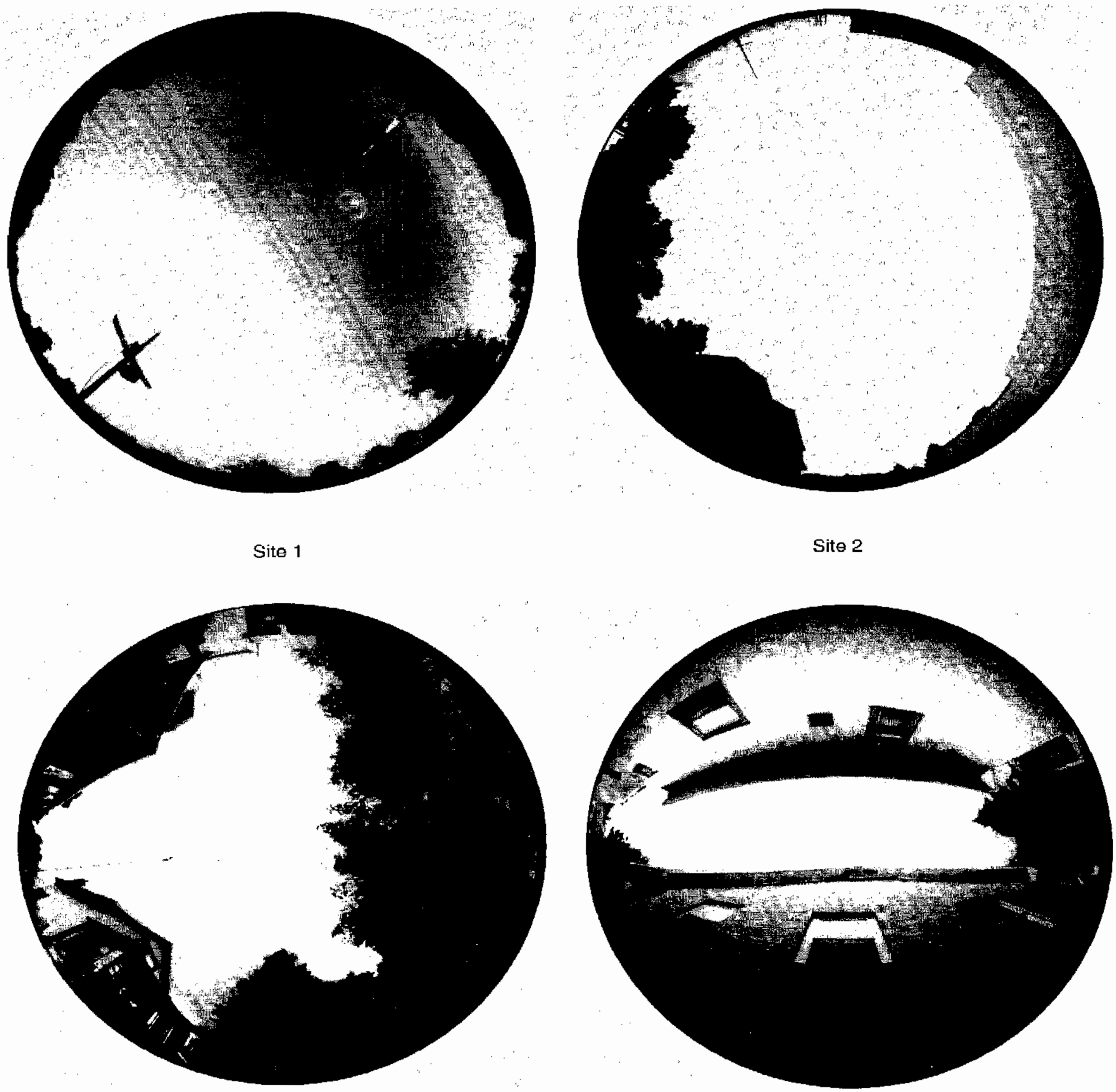


Figure 8.—Land-uses around the four below-canopy meteorology sampling points. See Table 5 in Chapter 4 for land- use categories. Near the points the classes are: VT, trees and shrubs; VGR, grass; A4, high density housing, yards small, mainly grass, few trees; CB, large commercial buildings, fewer than 6 stories; CS, small commercial buildings; AR3, apartments, highly mixed. (The tall tower is ISPT3 in Grimmond et al., Chapter 4, this report.).



Site 1

Site 2

Site 3

Site 4

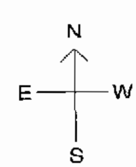


Figure 9.—Hemispherical photo views from horizon to zenith, from height of 3 feet at four sites.

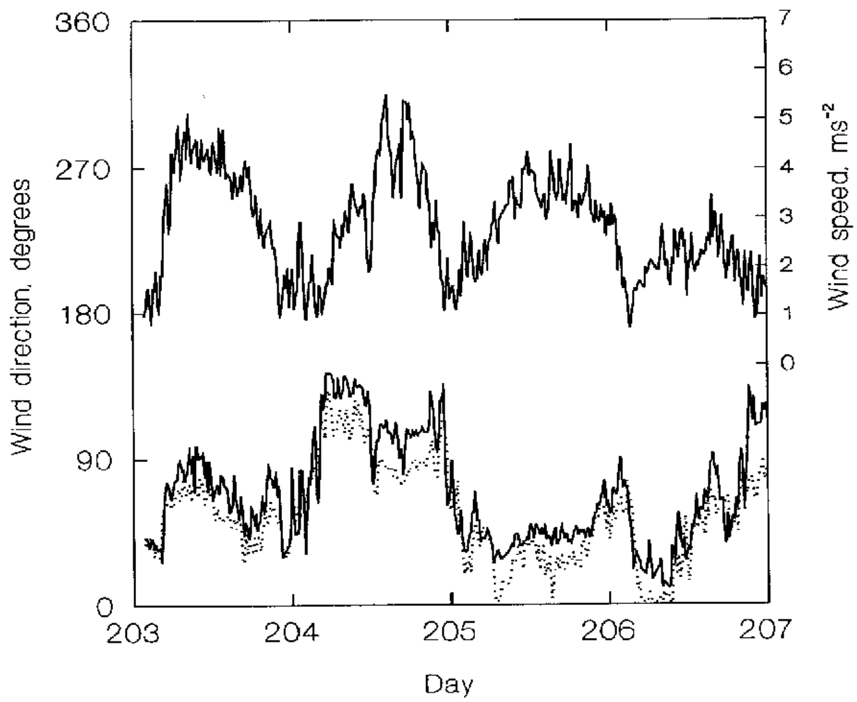


Figure 10.—Windspeed at O'Hare airport (upper curve) and wind direction at the airport (lower solid line) and on the lower level of the tall tower (dotted) on July 21 to July 24, 1992. The dates are shown at midnight starting the day.

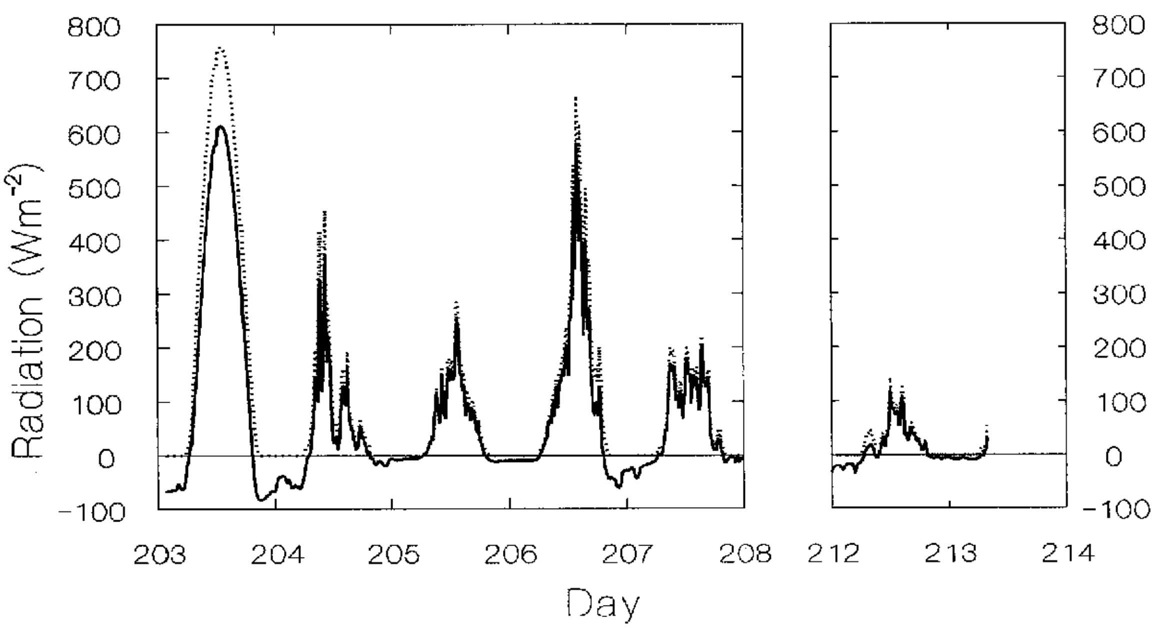
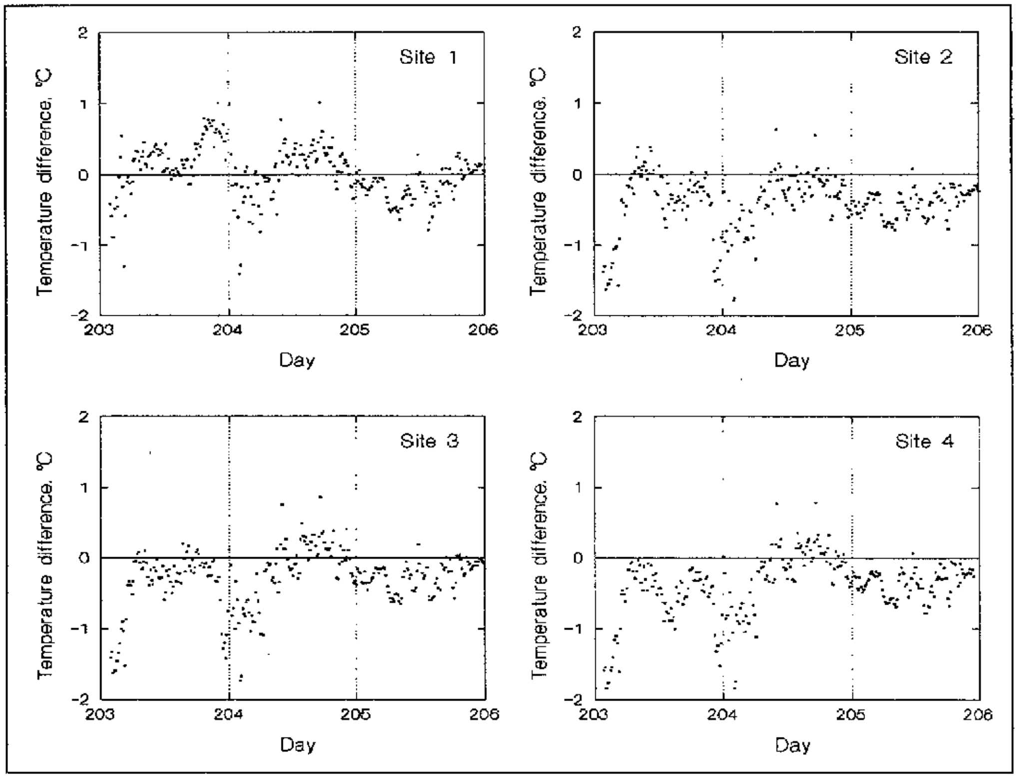


Figure 11.—Shortwave solar (dashed) and net all-wave radiation (solid) at the fixed tower.

12a



12b

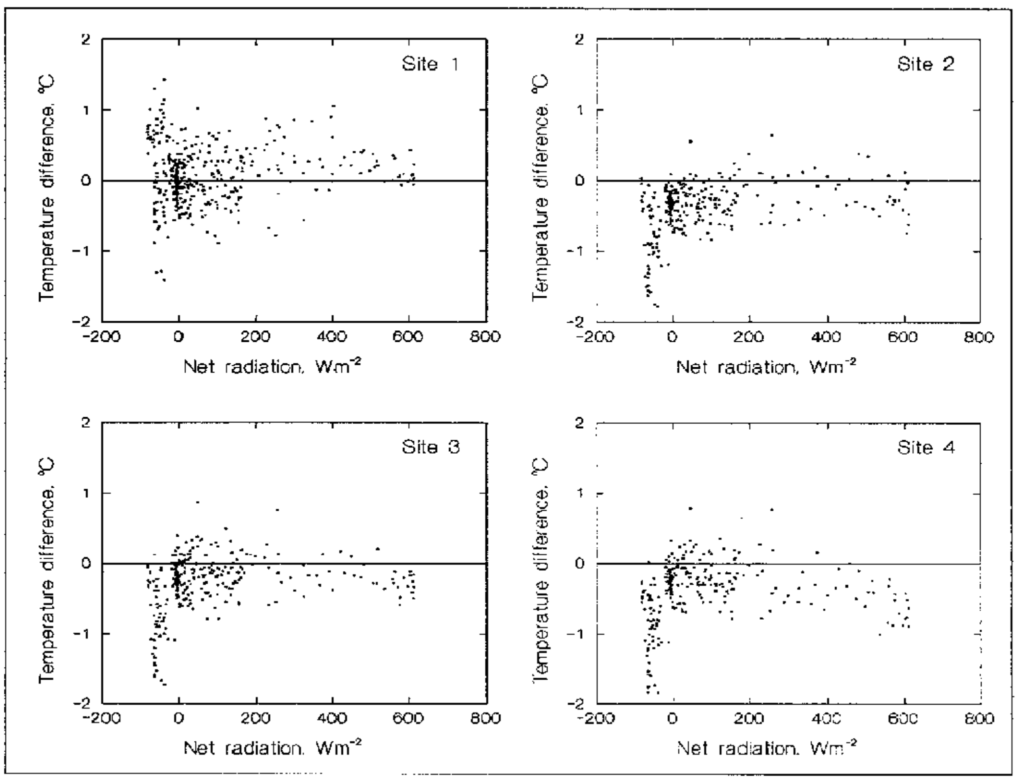


Figure 12.—Pattern of air-temperature differences (airport minus below-canopy) between O'Hare Airport and four below-canopy sites; a) time series, b) versus net radiation.

better sense of the pattern of windspeed differences is shown by plots of a normalized reduction in windspeed:

$$U_r = (U_{airport} - U_{site}) \ / \ U_{airport} . \qquad [3]$$

In Figure 14a, normalized reductions in windspeed are plotted for each site in a time series. The anemometers that we used had a threshold windspeed of 0.45 mph (0.2 m/s). Though the cups did not rotate until windspeed reached the threshold, the data loggers were programmed to indicate 0.45 mph (0.2 m/s) as a minimum speed, so that as wind reached the threshold speed and the cups began to rotate, the speed indicated was correct. However, the minimum recorded speed places a significant bias on the apparent reductions when wind is slow and anemometers at the below-canopy sites are stopped while the control at the airport is measuring a speed that is just slightly higher than the threshold. For airport speeds of 6.7 mph (3 m/s) or greater, the below-canopy anemometers generally indicated speeds above the threshold, and bias was negligible. Hence, data for airport speeds less than 6.7 mph were omitted from Figure 14a. From this point the discussion will pertain to the higher speed wind conditions.

With the higher reference windspeeds, the apparent effects of trees and buildings on windspeed vary less than at low relative windspeeds, and derivation of models to predict the effects of these obstacles is thus relatively more precise for the higher speeds. Also, influences of trees at higher windspeeds generally are of greatest importance for concerns such as energy use.

In Figure 14a we see a pattern of differences in windspeed reductions from site to site that is to some extent related to the amount of sky blockage in the hemispherical views (Figure 9). However, there is considerable within-site scatter, particularly at sites 1 and 2. Much of this scatter is explained by looking at wind reduction versus above-canopy wind direction (Figure 14b). For example, site 1 has large

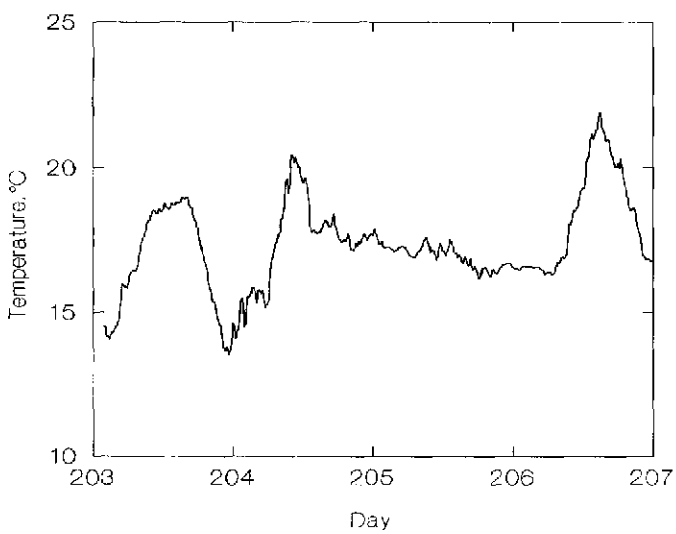


Figure 13.—Air temperatures at 5-foot height at O'Hare International Airport.

wind reductions when wind is from the north, apparently because wind is blocked by the tree row in that direction (Figure 9). The east is relatively free of obstacles and wind reductions are low in that direction (90 degree azimuth). At site 2, reductions are small at 45 degrees, evidently because wind comes relatively unabated through the opening between north and northeast. The very close buildings and street tree crowns account for large reductions at sites 3 and 4.

The descriptors obtained from the hemispherical photos and a nonlinear regression model provided an initial means of quantifying the relationship between morphology and reductions in windspeed. The photos were first analyzed in 15° sectors (see Methods). In the results reported here, we combined three sectors to describe average morphology in 45° sectors in the upwind and downwind directions (based on airport wind direction) for each 15-minute windspeed average for each below-canopy site. The most successful model included four independent variables. For buildings, we averaged the highest and lowest angles to the tops of buildings in the upwind direction (UBA) and in the downwind direction (DBA). For trees, similar descriptors were formed (UTA and DTA), but average angles were multiplied by fractional tree-crown density (0 to 1.00) estimated from the hemispherical photos. Thus a solid tree stand, with a visual density of nearly 1.00 as seen to the north of site 1 (Figure 9) would yield UTA and DTA values nearly equal to angular height. The street trees near site 3 have an overall visual density of less than 1.00, primarily because of the open space at the bottom and would yield UTA or DTA values of less than their angular height. Hence, trees often were weighted less than buildings of the same angular height.

The relationship between wind reductions and the morphology descriptors was explored by plots of wind reduction versus the descriptors or various combinations of descriptors. A combination of building and tree descriptors in the upwind and downwind directions that showed one of the closest relationships with wind reduction was BTUD; where

$$BTUD = \max(UBA,UTA) + (\max(DBA,DTA))/3, \quad [4]$$

"max" yields the larger of the two values in the following parentheses, and the divisor 3 is based on the trial assumption that downwind trees and buildings reduce windspeeds one-third as much as upwind buildings and trees. The scatter diagram of observations (Figure 15) suggested an exponential relationship with the general form of equation 3. The regression model

$$U_r = 1 - a^*BTUD + \exp(b^*BTUD), \qquad [5]$$

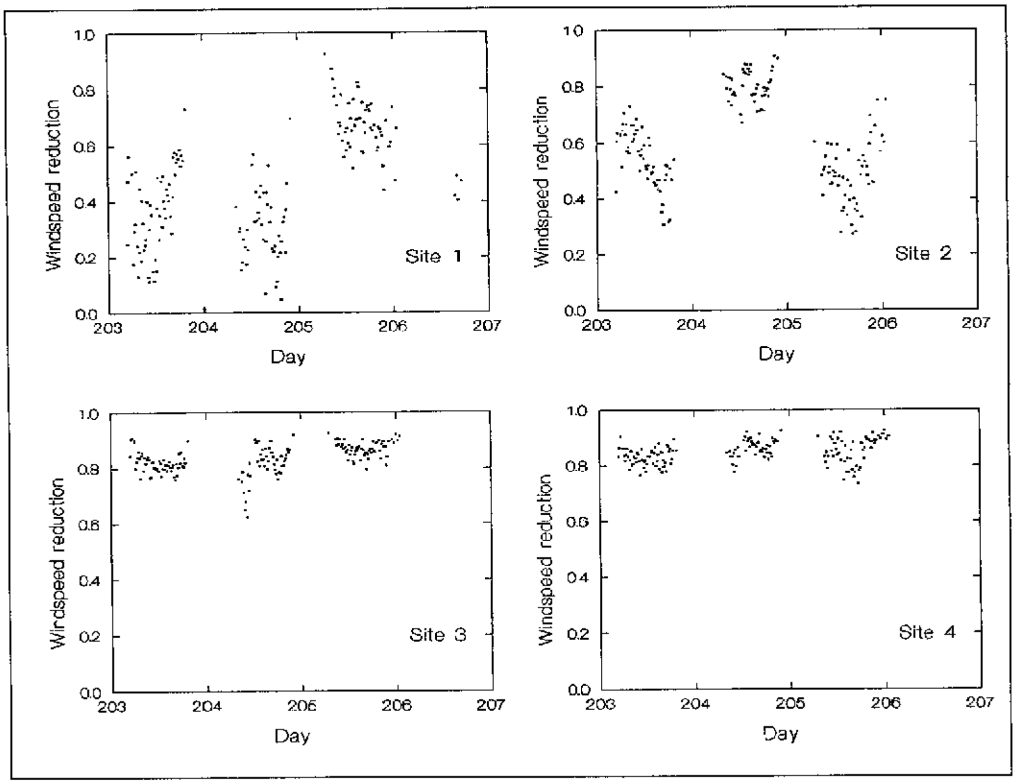
where a and b are parameters to be estimated, produced a good fit to the data (Figure 15) with a corrected correlation coefficient, R^2, of 0.78, indicating that about 78 percent of the wind reduction is explained by model [5]. Adding net radiation as an additional variable helped to explain additional variation and reduced residuals by about 0.1 at high positive values of net radiation.

With the four components of BTUD in the model separately, as

$$Ur = a + \exp(b^*UBA + c^*UTA + d^*DBA + e^*DTA), \quad [6]$$

where a, b, c, and d were coefficients to be estimated, R^2 increased to 0.80. The estimated coefficients were all

14a



14b

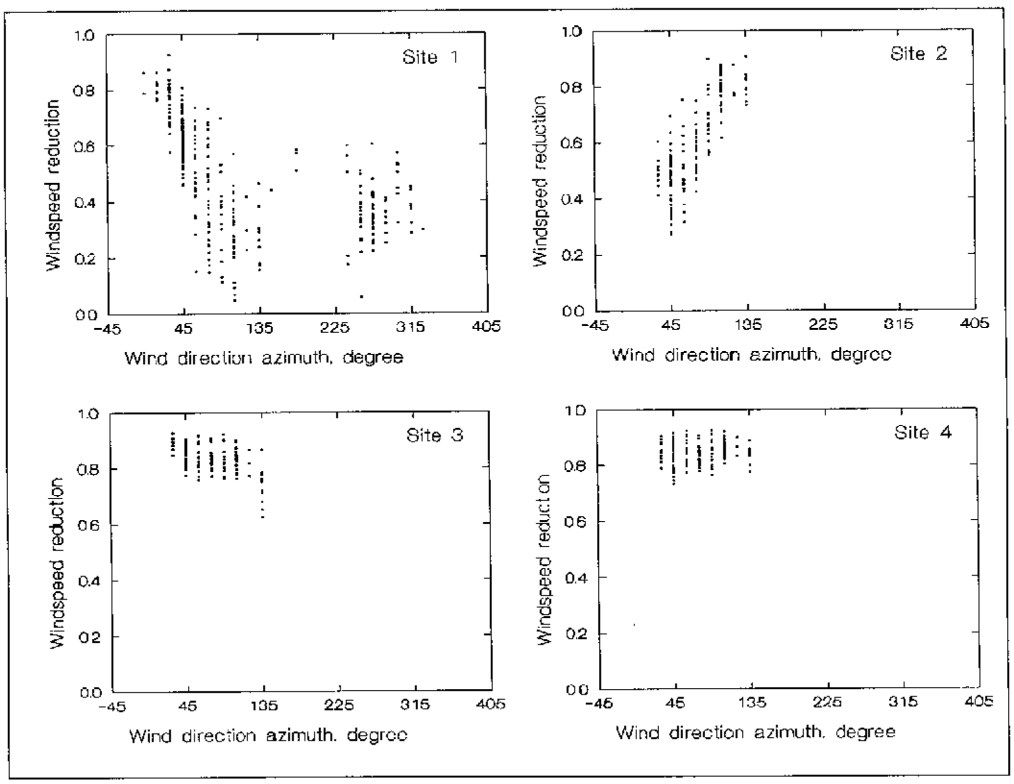


Figure 14.—Normalized reductions in windspeed at four below-canopy sites compared to the airport, with airport windspeeds greater than 6.7 mph (3 ms-1); a) as time series, b) versus wind direction.

significantly different from 0. (Because all variables were correlated over time, and because of the nature of nonlinear estimation, the test based on R^2 values is approximate.)

With the estimated coefficients, equation 6 becomes

$$U_r = 0.89 + \exp(-0.090{*}UBA - 0.073{*}UTA + 0.012{*}DBA - 0.019{*}DTA). \quad [7]$$

Equations of this type can be used to predict tree and building effects on windspeed, though care must be taken in interpretation. In the case of equation 7, the estimated coefficient d for downwind buildings DBA is positive, indicating smaller reductions with downwind buildings nearby. However, in this particular data, upwind and downwind building angles are positively correlated, and it is likely that one building-angle term tends to overestimate the building effect, while the other compensates for the overestimation. Inclusion of data from other sites combined with analysis of residuals (observed values minus estimates from the regression) will help in interpreting regression results.

Some of the residuals from the regressions are inflated partly by trees and structures obscured from view in Figure 9, partly by random turbulent eddies, partly perhaps because the assumption of no obstacle effect on wind at the airport is not completely met, and possibly in part by differences in thermal stratification in the atmosphere. The probability of this last effect being significant was reduced by our selection of higher speed winds for analysis. Future regressions will be based on hourly averaged data, which will reduce the effect of the random fluctuations. Descriptors of building and tree morphology from the GIS analysis will be included as independent variables to account for buildings and trees not visible in the hemispherical photos.

Conclusions and Application

Preliminary analysis of tree and building effects on windspeed and air temperature at points in one Chicago residential neighborhood over approximately one July week showed that windspeed was reduced by 83 to 85 percent on average compared to a location in the middle of O'Hare Airport, 6 miles to the west. Buildings occupied about 40 percent and tree crowns covered about 10 percent of the area within the neighborhood. In a long narrow open field adjacent to the residential area, windspeed was reduced an average of 46 percent, but reductions varied with distance to obstructions. When wind came to the field site from the direction of a 25- to 30-foot deciduous forest stand about 75 feet to the north, windspeeds were similar to those in the residential area.

Average air temperatures in the open field were essentially the same as the airport, but at times open field temperatures were from 2.5°F (1.4°C) greater to 2.3°F (1.3°C) less than at the airport in a pattern that reflected differences between the sites in rates of cooling and heating responses to the net radiation balance. Within the residential neighborhood, a

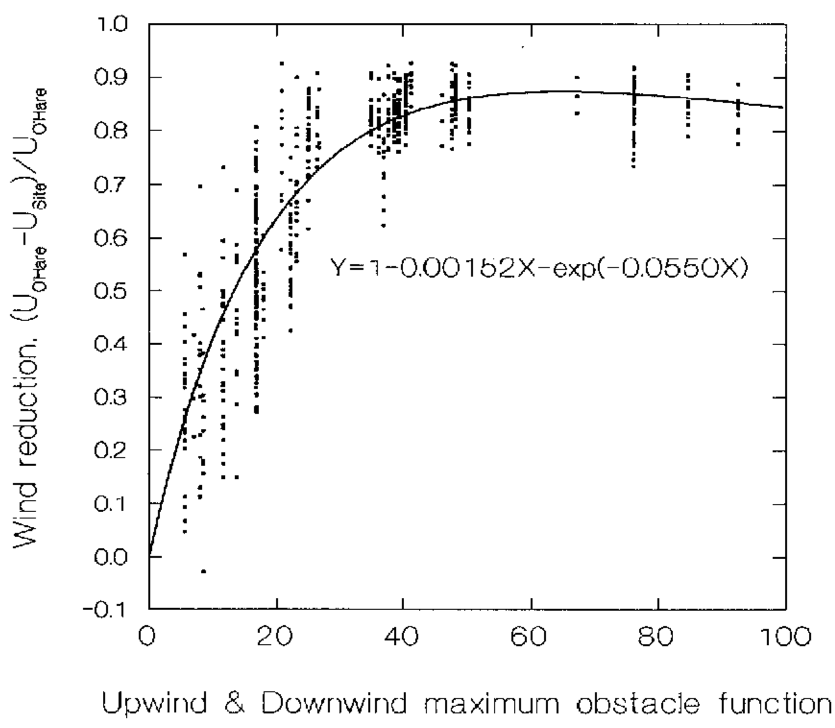


Figure 15.—Normalized wind reductions for all four sites versus a descriptor of upwind and downwind trees and buildings (BTUD) defined in the text. The curve is fit to the points by a nonlinear regression technique.

similar range and pattern of temperature differences from the airport were observed, but average temperatures were 0.5° to 0.7°F (0.28° to 0.39°C) higher in the neighborhood than in the open field.

One approach to developing information for planning tree management to save energy for heating and cooling is to simulate the effects of particular tree arrangements on energy use (Heisler 1991, McPherson 1994). This can be done by comprehensive, commercially available energy-analysis programs that include an hour-by-hour analysis of energy use in a building for an entire year. Input for these programs includes averaged or representative hourly weather data prepared specifically for energy analysis. However, the energy analysis programs do not include built-in procedures to estimate tree effects.

One method for including tree effects on wind, air temperature, and humidity in energy-use predictions, is to preprocess the representative weather data by algorithms that predict tree effects on these microclimatic variables. A primary goal of this study is to provide the algorithms to preprocess weather data. Although considerable analysis remains, the initial results reported here show considerable promise of success in predicting wind climate in residential neighborhoods. Most important, there is a strong likelihood that tree and building effects on windspeed can be reasonably well separated. The data from our airport reference site adjacent to a standard weather observing system, from which long-term weather data is archived, will enhance development of equations for preprocessing weather data for energy calculations. In further analysis, emphasis will be given to developing and using predictor variables that could be gathered without undue difficulty in extrapolating the methodology to other locations.

Different approaches to analysis of tree effects on temperature are possible using the 11 months of data. There are periods of 1 to 3 weeks in which the below-canopy sampling pattern remained stationary and when the sites were about the same distance from the lake. With data from such periods, temperature differences can be related to differences in tree and building cover directly, without extrapolation to the airport, thus reducing extrapolation errors. One reason for not using this method exclusively is that the range of morphological conditions sampled within each period generally will be smaller than when longer time periods and more sites are included. This method is similar to that used in an ongoing study in two neighborhoods in the Los Angeles area in which Simpson et al. (1994) used the below-canopy average temperature as a reference for comparing the neighborhoods.

The analysis has not yet proceeded to prediction equations for air temperature, and here the probability of success is less certain, at least in terms of separating tree and building effects. The differences in temperature will be relatively subtle and the physical causes of temperature difference between sites are far more complex than for wind. The comparisons of temperatures between neighborhoods as presented in the results indicate many of the considerations that must be included in model development.

Acknowledgments

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Literature Cited

Grant, R. H. 1991. **Evidence for vegetation effects on the daytime internal boundary layer over suburban areas.** In: Preprints of the 10th conference on biometeorology and aerobiology; 1991 September 10-13; Salt Lake City, UT. Boston, MA: American Meteorological Society: 75-78.

Grant, R. H. 1993. **Chicago, IL: Climate and meso-climate.** Unpublished final report on Cooperative Research Agreement 23-685. Available USDA Forest Service, 5 Moon Library, SUNY-CESF, Syracuse, NY 13210.

Grant, R. H.; Heisler, G. M. 1994. **A low-powered solar radiation shield for air temperature measurements.** In: Preprints of the 21st conference on agriculture and forest meteorology; 1994 March 7-10; San Diego, CA. Boston, MA: American Meteorological Society. (in press).

Grant, R. H.; Heisler, G. M.; Herrington, L. P.; Smith, D. 1985. **Urban winds: the influence of city morphology on pedestrian level winds.** Extended abstracts of 7th conference on biometeorology and aerobiology; 1985 May; Phoenix, AZ. Boston, MA: American Meteorological Society: 353-356.

Grimmond, C. S. B.; Cleugh, H. A. 1994. **A simple method to determine Obukhov lengths for suburban areas.** Journal of Applied Meteorology. 33: 435-440.

Grimmond, C. S. B.; Souch, C.; Grant,R.; Heisler,G. 1994. **Local scale energy and water exchanges in a chicago neighborhood.** In: McPherson, E. G., Nowak, D. J. and 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.

Heisler, G. M. 1986a. **Energy savings with trees.** Journal of Arboriculture. 12(5): 113-125.

Heisler, G. M. 1986b. **Effects of individual trees on the solar radiation climate of small buildings.** Urban Ecology. 9: 337-359.

Heisler, G. M. 1989. **Effects of trees on wind and solar radiation in residential neighborhoods.** Final Rep. on "Site Design and Microclimate Research" to Argonne National Laboratory, available from NTIS, #PB89-180202. USDA Forest Service, Northeastern Forest Experiment Station, University Park, PA. 164 pp.

Heisler, G. M. 1990a. **Mean wind speed below building height in residential neighborhoods with different tree densities.** ASHRAE Transactions. 96(1): 1389-1396.

Heisler, G. M. 1990b. **Tree plantings that save energy.** In: Rodbell, Philip D., Proceedings of 4th urban forestry conference; 1989 October 15-19; St. Louis, MO. Washington, DC: American Forestry Association: 58-62.

Heisler, G. M. 1991. **Computer simulation for optimizing windbreak placement to save energy for heating and cooling buildings.** In: Proceedings, 2nd international symposium on windbreaks and agroforestry; 1991 June 2-7; Ridgetown, ON. Ridgetown College: 100-104.

Heisler, G. M.; DeWalle, D. R. 1988. **Effects of windbreak structure on wind flow.** Agriculture, Ecosystems and Environment. 22/23: 41-69.

Heisler, G. M.; Dix, M. E. 1991. **Effects of windbreaks on local distribution of airborne insects.** In: Dix, Mary Ellen; Harrell, Mark O., eds. Insects of windbreaks and related plantings: Distribution, importance, and management. Conference proceedings; 1988 December 6; Louisville, KY Gen. Tech. Rep. RM-204. Fort Collins, CO; U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 5-12.

Heisler, G. M.; Grant, R. H. 1987. **Predicting pedestrian-level winds in cities.** In: Proceedings, 8th conference on biometeorology and aerobiology; 1987 September 14-18; W. Lafayette, IN. Boston, MA: American Meteorological Society: 356-359.

Huang, Y. J.; Akbari, H.; Taha, H.; Rosenfeld, A. H. 1987. **The potential of vegetation in reducing summer cooling loads in residential buildings.** Journal of Climate and Applied Meteorology. 26:1103-1116.

McGinn, C. E. 1983. **The microclimate and energy use in suburban tree canopies.** Davis, CA: University of California. 299 p. Ph.D. dissertation.

McNaughton, K. G. 1989. **Micrometeorology of shelter belts and forest edges.** Philosophical Transactions of the Royal Society of London, B. 324: 351-368.

McPherson, E. G. 1994. **Energy-saving potential of trees in Chicago.** In: McPherson, E. G., Nowak, D. J. and 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.

McPherson, E. G.; Heisler, G. M.; and Herrington, L. P. 1988. **Impacts of vegetation on residential heating and cooling.** Energy and Buildings 12(1988): 41-51.

Nowak, D. J. 1994a. **Urban forest structure: the state of Chicago's urban forest.** In: McPherson, E. G., Nowak, D. J. and 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.

Nowak, D. J. 1994b. **Air pollution removal by Chicago's urban forest.** In: McPherson, E. G., Nowak, D. J. and 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.

National Climatic Center. 1981. **Typical meteorological year user's manual.** Ashville, NC: National Climatic Center TD-9734. 25 p.

Neter, J.; Wasserman, W.; Kutner, M. H. 1985. **Applied linear statistical models.** Homewood, IL: Irwin. 1127 p.

Oke, T. R. 1987. **Boundary layer climates.** New York: Methuen. 435 p.

Oke, T. R. 1989. **The micrometeorology of the urban forest.** Philosophical Transactions of the Royal Society of London, B. 324: 335-349.

Panofsky, H.; Dutton, J. A. 1984. **Atmospheric turbulence: models and methods for engineering applications.** New York: John Wiley and Sons. 397 p.

Simpson, J. R.; Levitt, D. G.; Grimmond, C. S. B.; McPherson, E. G.; Rowntree, R. A. 1994. **Effects of vegetative cover on climate, local scale evaporation and air conditioning energy use in urban southern California.** In: Proceedings 11th conference on biometeorology and aerobiology; 1994 March 7-10; San Diego, CA. Boston, MA: American Meteorological Society. (in press).

Slade, D. H. ed. 1968. **Meteorology and atomic energy.** Oak Ridge, TN: U.S. Atomic Energy Commission, Division of Technical Information. 445 p.

Wilkinson, L. 1990. **SYSTAT: the system for statistics.** Evanston, IL: SYSTAT, Inc. 676 p.

Chapter 4

Local Scale Energy and Water Exchanges in a Chicago Neighborhood

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Abstract

Outlines the methods of measurement and analysis of "above-canopy" meteorological measurements undertaken to investigate the nature of surface controls on energy and water exchanges at the local scale. Observations were made over two periods: "intensive" (July 1992), and "extensive" (July 1992 through June 1993). During the intensive measurements, the vertical fluxes of sensible and latent heat were measured by eddy correlation methods at one above-canopy site. By combining these with measurements of net radiation and storage heat flux and detailed characterization of urban surface materials and morphology, a general understanding of energy exchanges of the urban surface at the local scale (100 to 1000 m) was obtained. Means of energy-balance values over the study period and their variability are presented and compared with results from other cities. Additional analyses to be conducted are described.

Introduction

Urban areas represent locations where a large and ever increasing proportion of the world's population lives, and where a disproportionate share of natural resources is used. Urbanization brings about significant changes in land-cover. The replacement of natural surface materials (the substitution of concrete, asphalt, trees, etc. for the natural vegetation) significantly alters the aerodynamic, radiative, thermal, and moisture properties of the surface. In turn the pre-urban balances of energy, mass, and momentum are altered. This leads to the modification of the atmosphere and the generation of an "urban climate" commonly characterized by enhanced temperatures, the "urban heat island" (Ackerman 1985, 1987), poorer air quality (Hanna 1971; Wadden et al. 1979; Sexton and Westberg 1980; Swinford 1980; Scheff et al. 1984), and other effects.

Increasing attention is being directed toward strategies that mitigate negative, inadvertent environmental effects of urbanization. For example, strategically planting trees or lightening building and pavement surfaces have been suggested as alternate ways to reduce the summertime urban heat island and thus reduce energy demand for cooling (Heisler 1974; Akbari et al. 1992). These strategies entail some alteration of the morphology or material properties of the urban surface, that have an effect through the alteration of surface energy and water exchanges. Relatively little research has been conducted to quantify these effects. Hence, we cannot make informed decisions about planning or directing urban morphological changes, as we do not know how such changes would affect the local environment and its inhabitants.

More fundamentally, our understanding of the biophysical processes involved in the generation of urban climates is limited. Direct observations of energy and mass exchanges in urban areas have been collected only in a restricted number of cities, with a small range of surface morphologies and climates (Oke 1988; Grimmond and Oke 1994). Thus, results of model simulations and predictions on the effects of changing the urban surface must be used with caution. To understand how urban morphology influences local climate (energy and water exchanges) it is necessary to undertake detailed investigations of local meteorology in conjunction with an understanding of urban surfaces. This paper reports on research conducted to study energy and water exchange processes in a neighborhood of Chicago. In addition to enhancing our understanding of biophysical processes, these data are to be used to evaluate physically based meteorological models, which, in turn, will be used to investigate the effects of proposed changes in urban morphology on the urban climate.

The surface-energy balance provides a framework with which to study energy and water exchanges at a range of spatial scales. It can be expressed:

$$Q^* + Q_F = Q_H + Q_E + \Delta Q_S + \Delta Q_A \ [\mathrm{W\ m^{-2}}]$$

where Q^* is the net all wave radiation (net available energy from solar and terrestrial radiation); Q_F is the anthropogenic heat flux (heat generated from fuel combustion); Q_H is the sensible heat flux (energy for heating the air); Q_E is the latent heat flux (energy for evapotranspiration); ΔQ_S is the net storage heat flux (energy for heating the urban fabric); and ΔQ_A is the net horizontal heat advection. Q_E, the term that links the energy and water balances, is the energy equivalent of evapotranspiration, a mass (water) term. If temperature is known, it is possible to convert between energy and mass (water) equivalents using the latent heat of vaporization. Thus, Q_E provides information about both energy and mass (water) exchanges. The surface energy balance concept, and the history of its application for an urbanized surface, was reviewed by Oke (1988).

Urban effects on climate are forced at a range of scales: from the urban canopy layer (UCL) where microclimates are determined by building/tree size and spacing, to the land-use scale, to the whole city. Table 1 (adapted from Oke 1984) illustrates this range of scales and associated atmospheric processes in urban areas. The Chicago study was conducted at three scales: micro (length scale 10^{-1}—10^{1} m), local (10^{2}—10^{3} m) and meso (10^{4} m) (Figure 1; Table 2).

We report on the local scale above-canopy studies (i.e., those representative of areas the size of city blocks to land-use zones) and outline the methodology used to select the study sites and collect meteorological data and information about the urban surface. The surface-energy balance provides the methodological framework (for measurement and modeling) for the local scale research. Using this framework, the partitioning of energy in Chicago is studied and compared with that in other cities, and research directions are described. The methodology and preliminary results from microscale "below-canopy" studies are presented in Heisler et al. 1994: Chapter 3, this report.

Methodology: Meteorology

To understand the nature of surface controls on energy and water exchanges, detailed measurements of local scale meteorology and surface conditions were conducted for one area within the City of Chicago.

Measurement Program

The meteorological measurements were conducted over two periods, referred to here as intensive (July 1992) and extensive (July 1992 through June 1993) (Table 2). The short-term intensive measurements were taken to collect direct observations of the energy and water fluxes from a representative neighborhood within Chicago. The extensive measurements were taken to provide data input for numerical modeling for all seasons; to aid in the development of relationships between routinely measured data at the National Weather Service (NWS) airport site and "urban" values representative of specific neighborhoods to allow NWS data to be extrapolated to urban sites; and to study relations between local scale and microscale conditions.

Table 1. —Framework for urban climate classification adapted from Oke (1984)

Turbulent Boundary Layers

	Layer	Flow characteristics	Dimensions[a]	Scale
I	Urban canopy layer (UCL)	Highly turbulent, controlled by roughness elements	Same as H[b] typically 10 m	Micro
	Roughness sub layer	Highly turbulent, wakes and plumes, transition zone	2D - 3D[b]	Micro
II	Urban boundary layer (UBL)	Turbulent, includes surface and mixed layers	Depends on surface fluxes of heat and momentum (typically 1 km day; 0.2 km night)	Local

Urban Morphology

Urban unit	Urban features	Urban climate phenomena	Dimensions[b] H	D	L	Scale
Building	Single building, tree or garden	Wake, plume, shadow	10 m	10 m	10 m	Micro
Canyon	Urban street and bordering buildings or trees	Canyon shelter, shade bioclimate	10 m	10 m	10 m	Micro
Block	City block, park, factory complex	Climates of parks, building clusters cumulus, mini-breezes		0.5 km	0.5 km	Micro
Land-use zones	Residential, commercial industrial	Local climates, winds, cloud modification		5 km	5 km	Local
City	Urban area	Heat island, urban circulation, urban effects in general		25 km	25 km	Meso

[a] Dimensions of boundary layers are depths of affected atmosphere; dimensions of urban units are those of urban structures or plan area
[b] H is building height; D is building spacing; L is building length.

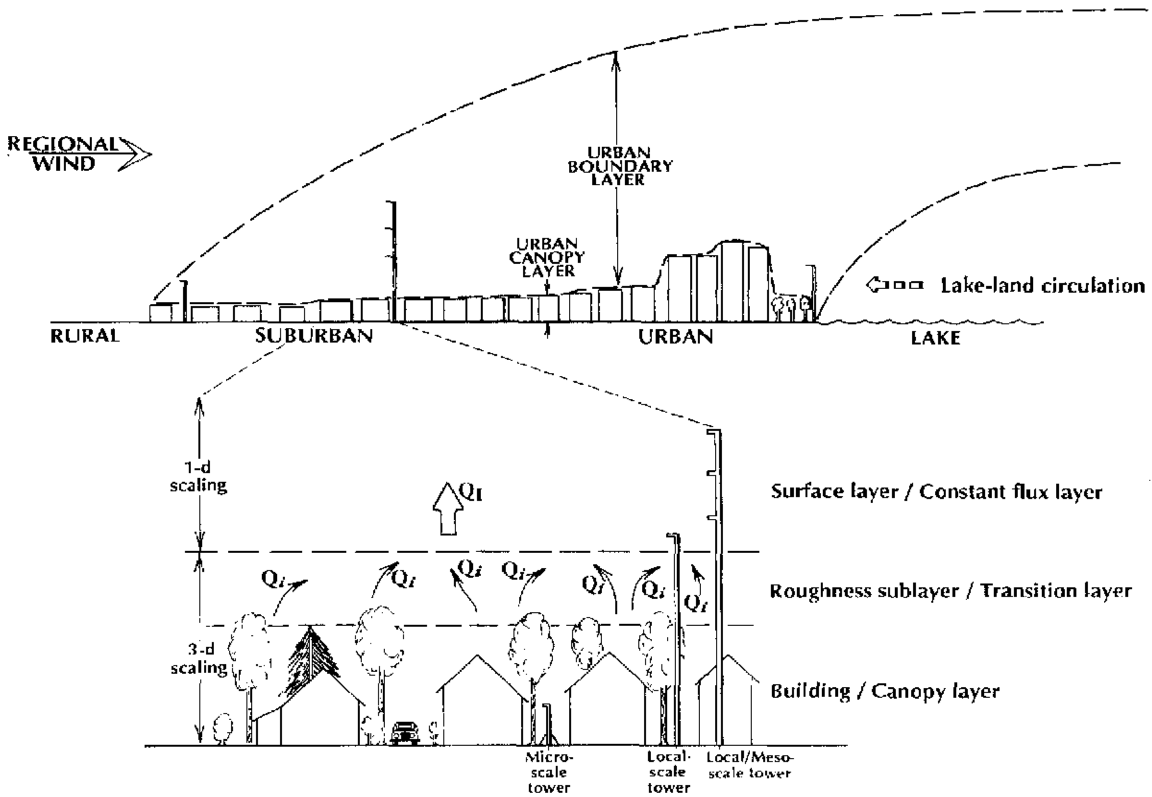


Figure 1.—Schematic representation of spatial scales and atmospheric processes in urban areas (adapted from Oke 1984; Oke et al.1989).

Table 2. —Scales of meteorological measurements in the Chicago Urban Forest Climate Project

Scale	Urban features	Urban climate phenomena	Tower sites	Measurement period
Regional 10^3 to 10^5 m	Cook and DuPage counties, Chicago Metropolitan area, Lake Michigan	Lake-land breeze	Belmont Harbor ISPT3 O'Hare airport	July 92 to June 93
Local 10^2 to 10^3 m	City-blocks, land-use zones, neighborhoods, community areas[a]	Above canopy local scale climates, constant flux layer, urban boundary layer	ISPT3 Pneumatic flux tower	July 92 to June 93 July 92
Micro 10^{-1} to 10^1 m	Individual properties buildings, gardens	Below canopy, shading, shelter	Below canopy 1 Below canopy 2-5	July 92 to June 93 < 1 month at a site rotated between sites

[a] Community area numbers referred to correspond to Figure 18 in McPherson et al. (1993): 9, 10-12[T], 13, 14, 15[T], 16, 17-19[T], 20-23, 25, 76[T], 87-91, 115[T]
[T] Community areas completely within 13 x 13 km study area (see Figure 3), remainder are partially in area.

Selection of Study Sites

Chicago is located along the southwest shore of Lake Michigan and occupies a plain which for the most part is only meters above the lake (Figure 2). The lake does not thermally modify the predominant synoptic-scale flow from the west, but it does generate a mesoscale breeze (lake-land breeze) as a result of differential heating between land and water. This effect decreases with distance due to the modification of airflow by the underlying urban surface. In this study it was essential to identify the effect of the lake on micro- and local scale climates from other controls. This required careful selection of study sites. Additional constraints on measurement locations were imposed by logistics, primarily by the location of pre-existing towers on which equipment could be mounted and where access was permitted.

The extensive meteorological measurements were conducted from three towers: City Parks Board tower at Belmont Harbor; Illinois State Police District 3 tower (ISPT3) near the intersection of Forest Preserve, Harlem, and Irving Park; and next to the NWS climate station at O'Hare International Airport (Figure 2). The intensive flux measurements were conducted on the grounds of the Read Mental Health Center, directly adjacent to ISPT3 (Figure 2). The sites are aligned along a transect east-west across the city, from the lake, past the intensive-flux site to the O'Hare station (Figure 2).

The area surrounding the ISPT3 and intensive-flux towers includes the neighborhoods of Harwood Heights and Norridge, Chicago. It has predominately two-storied densely packed houses and a large number of mature deciduous trees with many greenspaces (parks, cemeteries, etc.). In the immediate vicinity of the towers are large greenspaces (cemetery and grounds of the mental health facility) to the east, northeast, and west; a shopping mall and garages to the north and northwest; and houses to the south.

Meteorological Measurements

Intensive observations

The intensive observations consisted of direct measurements of sensible and latent heat flux, and net all-wave radiation (Table 3). The convective fluxes (Q_H and Q_E) were measured using eddy correlation techniques (Lenschow 1986; Oke 1987). All of the equipment was installed on a pneumatic tower that could be lowered when rainfall, high winds, and/or thunderstorms were anticipated. A Campbell Scientific Inc. (CSI) one-dimensional sonic anemometer and fine-wire thermocouple system (SAT: CA27) was used to measure vertical wind velocity and temperature; a CSI krypton hygrometer (KH20) was used to measure the absolute humidity. Fluctuations in the vertical wind velocity, air temperature and humidity were sampled at 5 Hz and the covariances determined over 15-minute periods. Flux corrections were made for oxygen absorption by the sensor and air density (Webb et al. 1980; Tanner and Greene 1989). Corrections were not made for frequency response and spatial resolution of the eddy correlation sensors, which probably would increase Q_E by 1 percent (M. Roth 1992 pers. commun.; Grimmond et al. 1993). All times have been corrected to Local Apparent Time.

Net all-wave radiation was measured at two levels (Table 3). It is not practical to measure ΔQ_S directly at urban/suburban sites due to the complexity of the materials and morphology of the urban surface (Oke and Cleugh 1987; Grimmond et al. 1991). Hence ΔQ_S is determined as a residual in the energy balance ($Q^*-(Q_H+Q_E)$) if Q_F and ΔQ_A are neglected. This approach has the inherent problem that all measurement errors of other energy balance fluxes are accumulated in the ΔQ_S term.

Q_F has not been determined for this site. Grimmond (1992) calculated the size of this flux for a suburban area of Vancouver, British Columbia, based on combustion from stationary and mobile sources and metabolic rates. The magnitude of this flux is dependent on the spatial pattern of the sources (Schmid et al. 1991). In residential areas, the most notable influences on Q_F are major roadway systems and significant non-residential stationary anthropogenic heat sources, for example, strip malls with energy-intensive users. Given the location of the local anthropogenic heat sources relative to the measurement sites, summertime air-conditioning, and the magnitude of Q_F calculated by various authors (Oke 1988), the peak diurnal values of Q_F at the study site probably were about 20 Wm^{-2} (4.5 percent of mean Q^* values).

Spatial differences in surface cover across the city result in differential heating and the lateral movement of energy (advection). The horizontal advection term (ΔQ_A) is difficult to determine. The observation site was located in an area that was extensively suburbanized, but, as discussed earlier, there are known regional scale circulations that are generated due to differential heating patterns between land and Lake Michigan (e.g. Hall 1954; Lyons 1972). The intensive flux-tower and ISPT3 site are less than 15 km from the lake (Figure 2), without intervening topographic barriers. Following an analysis in the Sunset neighborhood in Vancouver, where there is also a large water body which generates a sea-breeze circulation, Steyn (1985) concluded that advection could be neglected at the local scale when working under similar land-use conditions. For this report, ΔQ_A has been ignored, so the energy balance residual (ΔQ_S) should be interpreted accordingly. The influence of advection is the subject of further investigation.

Extensive observations

The instrumentation used in the extensive measurements, and the heights at which it was mounted, are listed in Table 3. A full description of ventilated temperature systems developed for the Chicago Urban Forest Climate Project is presented by Grant and Heisler (1994). All instruments used in the local scale study and the below-canopy study were inter-compared before and after the measurement campaigns (May 1992, July 1993). Appropriate corrections were made for inter-instrument differences.

Methodology: Surface Controls

Rationale

The active surface of any system is one of the most important determinants of climate because it is the primary site of

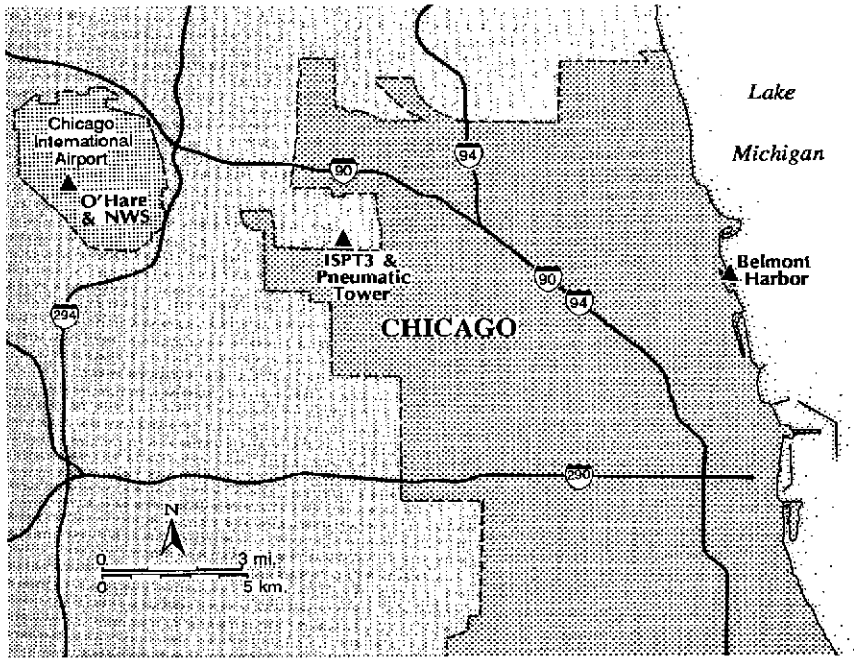


Figure 2.—Location of the local scale measurement sites across the city of Chicago (Chicago is identified with the darker shading).

Table 3. —Instrumentation used on pneumatic tower during intensive measurements and on fixed towers for extensive measurement period (July 1992 to June 1993)

Intensive Measurements

Variable	Instrumentation	Level installed (m)
Sensible heat flux (Q_H)	CSI sonic anemometer and fine wire thermocouple	18
Latent heat flux (Q_E)	CSI krypton hygrometer	18
Net all wave radiation (Q*)	Swissteco miniature net radiometer	18
Soil heat flux (Q_G)	REBS Soil heat flux plates	-0.08

Extensive Measurements

Variable	Instrumentation	Level installed (m)		
		Illinois StatePolice Tower ISPT3	Belmont Harbor	O'Hare
Air temperature	Vaisala HMP35C	24.6, 43.1, 69.5	17.1	1.5
	YSI thermistor 44020	24.6, 43.1, 69.5	17.1	1.5, 4.0
Relative humidity	Vaisala HMP35C	24.6, 43.1, 69.5	17.1	1.5
Wind speed	R.M. Young Wind Sentry	24.6, 43.1, 69.5	17.1	2.5
Wind direction	R.M. Young Wind sentry	24.6, 69.5	17.1	2.5
Net all-wave radiation	REBS Net radiometer	24.6		2.5
Solar radiation	Li-cor pyranometer	24.6		4.0
Precipitation	Texas Instruments rain gauge	3		
Surface moisture status	Weiss type wetness sensor	0		

transfer and transformation of energy, mass and momentum. Climatological and meteorological measurement and modeling studies require the surface datum to be defined and described to characterize the site where measurements have been conducted; provide input for numerical models; or ensure spatial consistency between measured and modeled data. In model evaluations, it is essential that surface parameters (the model domain) represent the same surface area for which the measurements were conducted (the measurements' source area) (Grimmond and Souch 1994). In this study the nature of surface controls on energy and water exchanges is of primary interest.

The source area for meteorological measurements is dependent on the physical process involved, the instrumentation used, and the meteorological conditions under which the measurements occurred. For radiant fluxes, the source area is fixed in time by the field of view of the instruments (i.e., by geometry). This source area can be determined using procedures outlined by Reifsnyder (1967) and Schmid et al. (1991). For turbulent fluxes, the source area is not fixed but varies through time as a sensitive function of sensor height, atmospheric stability, and surface roughness (in that order of importance). Numerical models, based on boundary-layer diffusion theory have been developed to determine the dimensions, weighting, and areal extent of the source area of turbulent measurements (e.g., Gash 1986; Schuepp et al., 1990; Leclerc and Thurtell 1990; Schmid and Oke 1990; Horst and Weil 1992).

In this study, a methodology to link a source area model for turbulent fluxes (based on Schmid and Oke 1990) to a surface database within a geographic information system (GIS) was developed (Grimmond and Souch 1994). This surface database in conjunction with the flux data will provide a basis for assessing the relationship between energy and water fluxes and vegetation (Demanes 1994).

Surface Database

Preliminary calculations based on the Schmid and Oke (1990) source area model for turbulent fluxes were used to identify the approximate dimensions of the source areas for the convective flux (Q_H and Q_E) measurements during the intensive study period. Based on these calculations a square approximately 13 km by 13 km, centered on the ISPT3 tower site, was delineated (Figure 3). A three-tier surface database was developed for this area, bounded by Touhy Avenue to the north, Chicago Avenue to the south, Mannheim Road to the west, and Pulaski Road to the east (Table 4). At the regional scale the spatial distribution of land use (Table 5) was mapped from aerial photographs. Given the focus of the

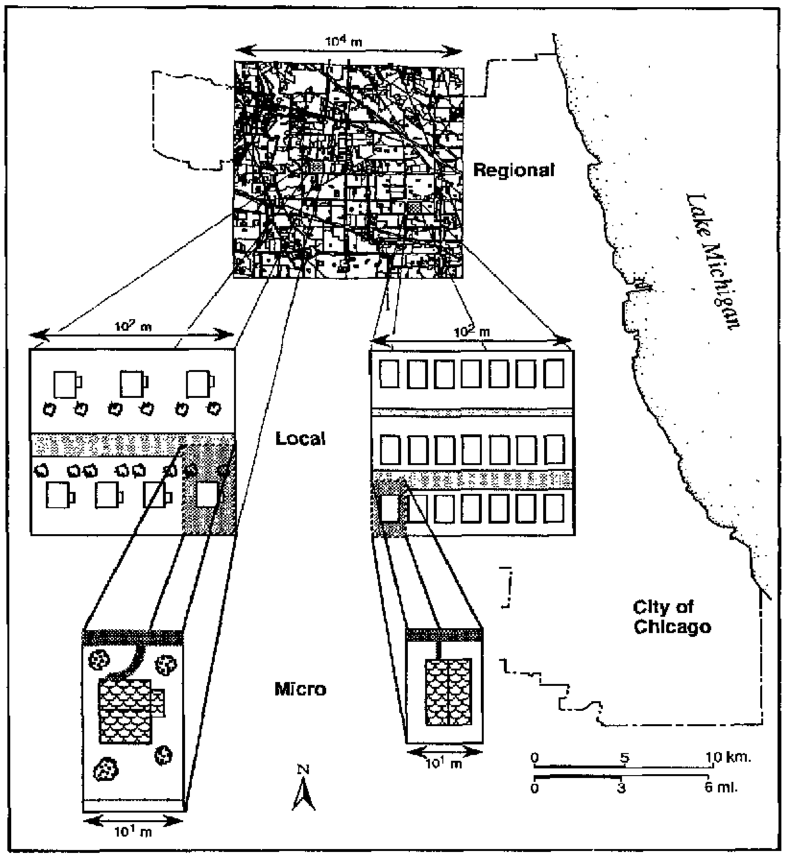


Figure 3.—Schematic representation of the structure of the surface database (adapted from Grimmond and Souch 1994).

study on the effects of vegetation on urban climate, the two primary criteria for identifying the land-use categories were building dimensions and density, and vegetation dimensions and density. The digitized, geo-corrected map contains more than 2500 polygons (Figure 4).

At the local scale (Figure 3), 200 m x 200 m grid squares were located randomly on a second set of more detailed (1:4800) aerial photos (Table 4). For each square the percent cover of building, grass, trees, pavement, and other variables (Table 6) was estimated. Based on replicates within each land-use category, means and standard deviations were calculated for building and vegetation densities and percent plan-area surface type (Table 6). These data were linked to the regional digital land-use map to allow the areal distribution of attributes to be illustrated.

At the microscale (Figure 3), field surveys were conducted to provide detailed information on surface cover at the scale of the individual lot in residential neighborhoods or 1/10 acre plot (0.04 ha) in non-residential areas. Weighted stratified random sampling was used to select sample plots within each land-use category to obtain detailed information on specific surface characteristics (Table 7). Data from 147 plots (87 residential, 60 nonresidential) were collected within the study region, 47 surveys conducted as part of the survey on urban forest structure (see Nowak 1994: Chapter 2, this report) and 100 supplementary sites. The additional surveys were conducted to ensure there were replicate surveys for each general land-use class. Field data stored in database files are linked to the regional scale land-use database to provide information on the attributes within land-use categories. These include building heights (of interest in the calculation of roughness length); surface materials (important for albedo, emissivity, drainage properties, storage heat flux modeling, etc.); and tree species and tree density (which aid in calculating leaf area index, important in evaporation modeling) (Grimmond and Souch 1994).

Figure 5 illustrates the spatial variability of vegetative cover and built impervious surfaces across the study region. Impervious surfaces are important in defining retention and detention storage capacities which are used in both runoff and evaporation modeling. Vegetative cover is important for defining surface resistances for evaporation and air quality modeling. When these figures are compared with the land-use map (Figure 4), differences in surface properties among the classes, which influence the energy and water exchanges become clear. For example, note the differences in surface cover within the residential A classes (A to A4) and how the city generally becomes more impervious toward the east.

Results

Representativeness of the Measurement Periods

Analysis of synoptic classifications during the study period show that the weather the Chicago area experienced was similar to that of the prior 10 years (Grant 1993). Cold fronts and warm sectors passed through the Chicago area 25 and 12 percent of the study period respectively; within 2 percent of the occurrence during the prior 3 years, and within the range of percent occurrence over the past 10 years. Chicago experienced fewer warm fronts during the study period than in the recent past, but experienced as many as have occurred in two of the last ten years. Polar high pressure was the dominant synoptic feature during the study period (35 percent of the time north, west or east of Chicago, and 11 percent of the time south of Chicago). The frequency of occurrence of the polar high located north, west, or east of Chicago equaled the occurrences in 3 of the past 10 years. The frequency of occurrence of the polar high south of Chicago exceeded the highest frequency of occurrence in the prior ten years. The presence of more frequent polar high pressure systems to the south of Chicago helps explain the relatively cold temperatures experienced during the study period (Table 8).

At O'Hare Airport a total of 95.8 mm of rain fell on 23 days during July 1992 (normal: 92.2 mm); longest period without rainfall was 2 days. Consequently, the surface was almost continuously wet throughout the study period (Figures 6 and 8). The range of general climatic conditions measured from the ISPT3 site in July 1992 (the intensive period) are presented in Figure 6.

Table 4. —Information source for surface database at each scale (See Figure 3 for scale dimensions)

Scale	Method	Area covered	Output
Regional	Land-use mapping on air photos Geonex Chicago Aerial Survey(CAS), Des Plaines Flown: March 2, 1992 scale: 1: 24000	13 km x 13 km square centered on ISPT3 Area bounded by Touhy Ave, Chicago Ave., Mannheim Rd. & Pulaski Rd.	Land-use categories (see Table 5)
Local	Detailed photo analysis Sidwell Company, West Chicago: Flown: Spring 1987 scale: 1: 4800 Geonex CAS: March 24, 1990,1:4800	Randomly located replicates within each land-use category	Attributes for each land-use (see Table 6)
Micro	Field surveys	147 randomly located points and immediate surrounding area within region	Surface details (see Table 7)

Table 5. —General land-use categories for Chicago

General Land-use	Categories and description
Residential (Single)	
A	High density housing, A1-A4 differentiated by shape of buildings and whether attached or not. Yards small, mainly grass, few trees.
B	Moderate density housing, small houses with trees
C	Moderate density housing, small houses, large yards. C1-C3 differentiated by size of houses. All have many trees/extensive landscaping
D	Large houses, small grass yards with some trees and shrubs
E	Large houses, large yards, yards landscaped with shrubs and trees
EA	Mixture of "A" and "E" type housing
F	Houses equally spaced, large grass yards, few trees, F1 and F2 differentiated on housing density
MH	Mobile homes
Apartments	
AA	5-6 stories, U-shaped, distinguished from AA2 based on arrangement of parking
AB	Square shaped buildings
AL	L-shaped buildings, 7 stories tall, no trees
AL1	Rectangular shape
AR1	Duplexes
AR2	Mixture of AR1 and A type houses
AR3	Highly mixed
BB	Low-level apartments (2 stories), rectangular shape. BB1, BB2 and BB3 distinguished on height and size
Commercial-Industrial	
CB	Large commercial buildings - < 6 stories
CC	Very tall commercial buildings - > 15 stories
CS	Small commercial buildings
I	Industrial - large low level buildings or many small buildings
Institutional	
HS	High school - large building, few trees, medium size parking lot
S	Elementary/ Junior High school - much smaller buildings than HS
U	University - large buildings, parking lot, vegetated grounds
Transportation	
MRI	Major roads e.g. interstates
RR	Railroad tracks or side/yards:
Vacant/Wild	
DI	Dirt
Vegetated	
VG	Golf course
VGR	100% grass
VM	50% grass/50% tree and shrub
VPC	Cemetery
VT	Trees and shrubs
Impervious Surfaces	
CN	Concrete
IP	Parking lot (impervious)
IS	Tennis court
Water	
WL/R	Lake/river

4a

4b

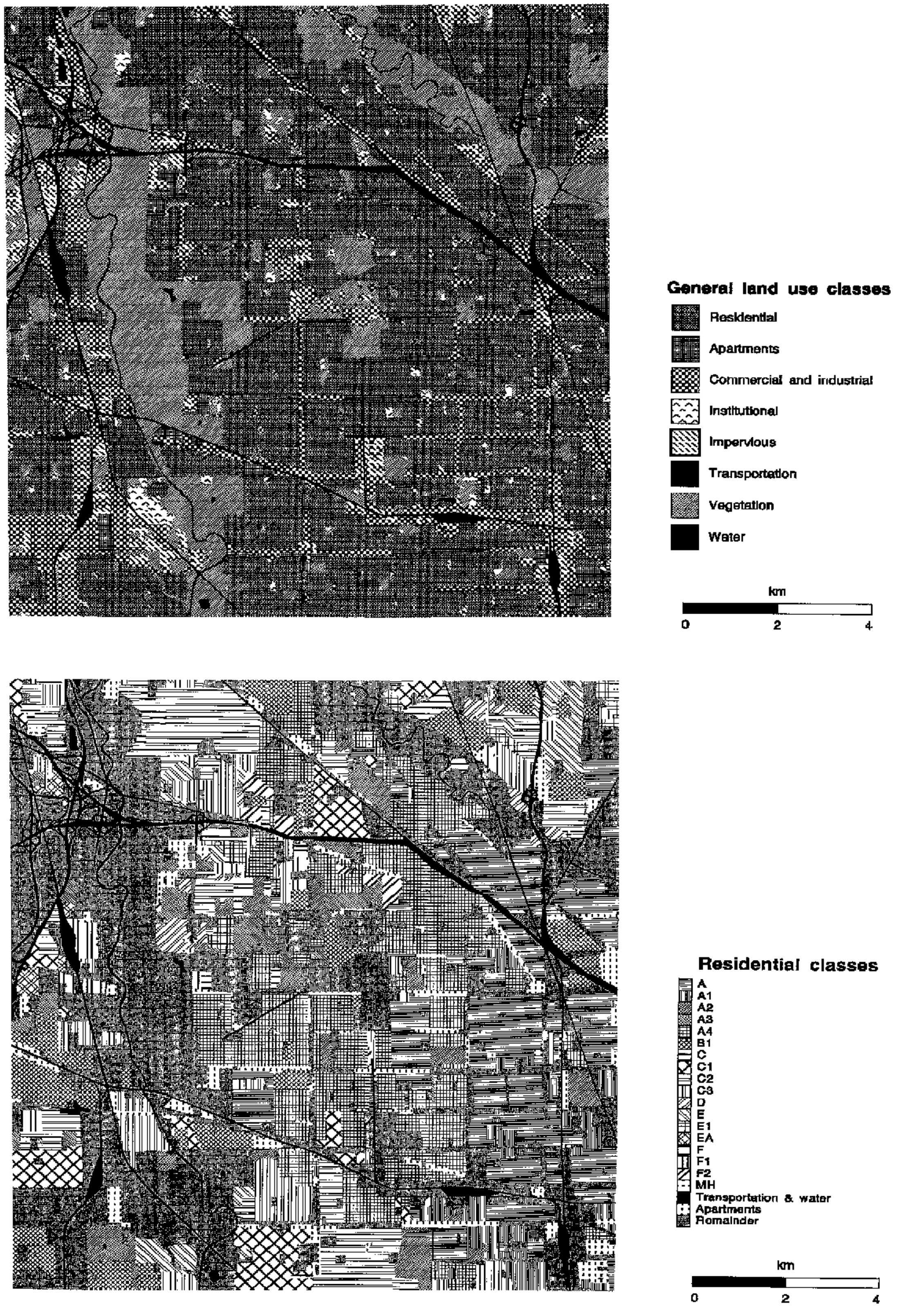


Figure 4.— a) General land-use classes across the study area b) Residential land-use classes (see Table 5 for descriptions)

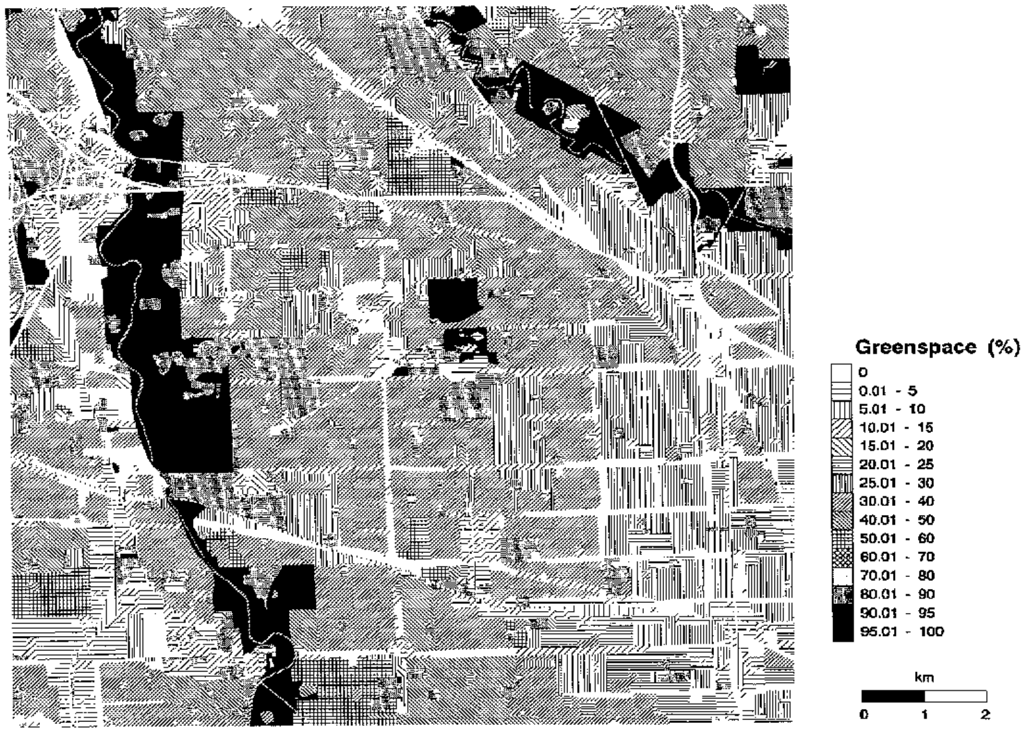
Table 6.— Attributes determined for each land-use category

Densities (number per area)
Buildings
Trees
Roads
Percent areal cover
Buildings
Garages
Grass
Trees/shrubs
Parking lot
Main road
Water
Dirt
Sand
Pavement (non parking lot)
Scruff

Table 7. —Information collected in the field survey

Non residential (0.1 acre, 0.04 ha plots)	
Landscape:	Managed/ unmanaged and condition
Land-use:	Residential, commercial etc. and % of plot covered
Ground cover:	% cover by: building, structures, cement, tar, wood, other impervious, soil, rock, duff/mulch, herbaceous/ivy, grass, wild grass, water, shrubs
Building attributes:	Type, length, width, material, azimuth from front door outward, age, height, number of floors, roof color, wall color, % wall glass, average distance to nearest building, height of nearest building
Structure shrub and trees:	Full listing of species and size of each tree and shrub, condition of tree, % beneath canopy of artificial surfaces, d.b.h, height, height to lower crown, crown width, crown shape, percent of crown volume occupied by leaves, tree condition.
Residential (variable size based on lot size; from mid-street to mid-alley or back of lot)	
Road:	Width of road, length of road in front of property, type, width of curb to sidewalk, % of strip covered by cement
Alley:	Width, length, surface type
Length:	Length of front part of lot, width of front part of lot, presence, type and height of any overhead obstructions
Irrigation:	% vegetation irrigated
Structure:	Length, width, height of structure, % plot occupied by structure, type of structure, material, structure of roof
Shrubs:	Species, length and height of shrub mass, % shrub volume occupied by leaves, density of leaf mass, number of stems in mass, average diameter of stems in mass
Trees:	Species, number of stems, d.b.h., tree height, bole height, crown width, crown shape, percent of crown volume occupied by leaves, crown density
Positions:	Sketch and photo of building and tree locations referenced to tree information

5a



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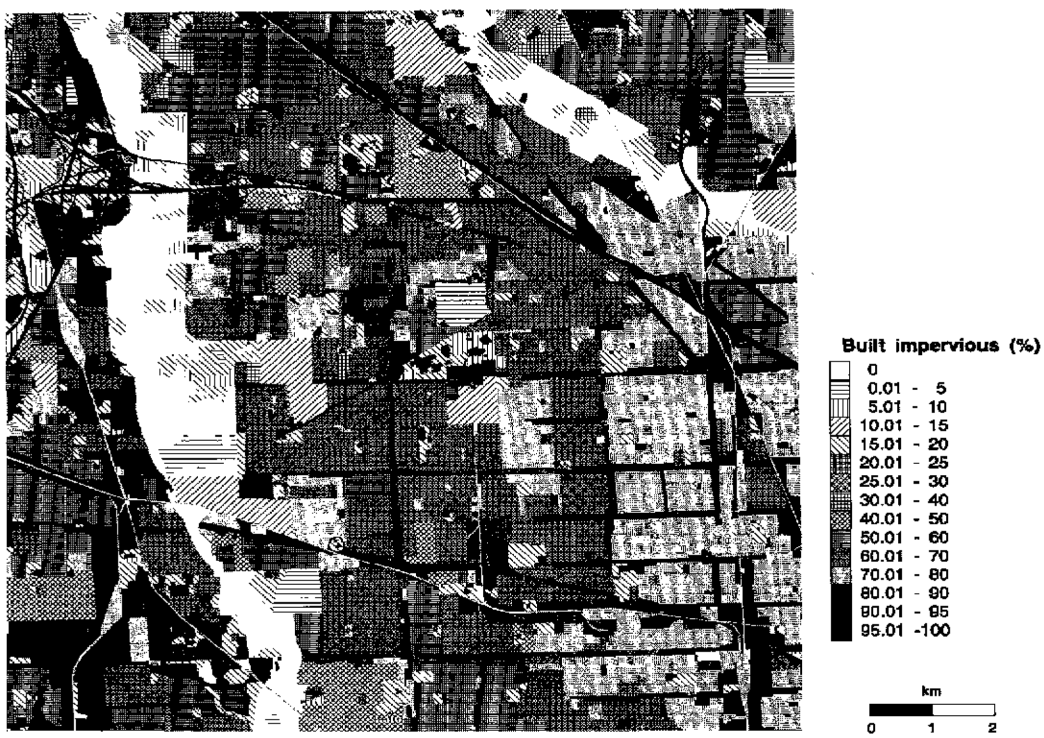


Figure 5.—Surface cover in the vicinity of the measurement site: a) Percent plan-area vegetation; b) Percent plan-area built impervious

Table 8.—Meteorological conditions during extensive study period (July 1992 to June 1993) and departures from Normal (1951-80). Source of data NOAA (National Climate Data Center, Local Climatological Data, Chicago O'Hare station). (SP study period; D departure from Normal).

Month	Temp (°C)		Precip (mm)	
	SP	D	SP	D
July	20.7	-2.1	95.8	3.6
August	19.4	-2.7	90.4	0.8
September	17.1	-1.1	109.5	24.4
October	10.2	-1.7	45.5	-12.4
November	3.5	-0.8	137.4	85.1
December	-1.9	0.5	63.3	9.9
January	-3.2	2.9	97.3	58.4
February	-4.2	-0.6	20.8	-13.7
March	1.2	-1.7	114.8	46.5
April	7.2	-2.0	116.1	23.6
May	8.8	0.4	46.5	-37.8
June	24.8	-1.2	253.0	157.0

The climatological conditions experienced during the extensive study period are summarized in Table 8 and Figure 7. Overall, the period was slightly cooler and wetter than normal.

Energy Balance Fluxes

During the intensive measurement period 127 hours of eddy correlation flux measurements were collected. Because the measurements were conducted during a period with a high frequency of rainfall, there are many breaks in the data (Figure 8). The mean value for each of the fluxes for each hour and their variability is shown in Figure 9. From Figures 8 and 9 it can be noted that clouds occurred throughout the day during the measurement period. The maximum output flux (i.e., removal of energy from the surface) was Q_E followed very closely by Q_H and ΔQ_S. The convective fluxes (Q_E and Q_H) peak at solar noon whereas ΔQ_S peaks about 1100 Local Apparent Time, with a marked hysteresis pattern (values higher in the morning and lower in the afternoon).

To allow direct comparisons of flux partitioning from day to day (i.e., to remove the effect of the available energy varying from day to day), each of the fluxes are normalized by net radiation to calculate ratios: $\chi(Q_H/Q^*)$, Υ (Q_E/Q^*) and Λ $(\Lambda Q_S/Q^*)$ (Figures 10 and 11). The ratio of the two convective fluxes, the Bowen ratio: $\beta = Q_H/Q_E$ (i.e., the amount of energy warming the air relative to that evaporating water), also is calculated. The mean daytime Bowen ratio for the observations, determined from the mean daytime fluxes, is 0.87. Thus, more energy is being removed from the surface by the latent heat flux than sensible heat flux (i.e., more energy during this period was going into drying the surface than into warming the air). The mean ratios of χ , Υ, and Λ are 0.32, 0.38, and 0.30 respectively for the daytime ($Q^*>0$) (32 percent of the energy going into heating the air, 38 percent into the evaporation of water, and 30 percent into heating the urban fabric), and 0.35, 0.49 and 0.16 for the day (24 hours) (35 percent heating the air, 49 percent evaporating water, and 16 percent heating the urban fabric). These results are biased to slightly higher Bowen ratios than the true average for the period as measurements were restricted to times when rainfall was neither occurring nor imminent (i.e., evaporation may have been more significant at the other times).

The variability of the fluxes from day to day can be seen by the ranges on Figure 10. It is notable that the data are remarkably consistent except for one day (Year/day; 92/210) when Bowen ratios were 3 to 5 (i.e., much greater Q_H than Q_E). This day was at the end of one of the slightly longer intervals between rainfall events (Figure 8). The high Bowen ratios were associated with a suppressed Q_E, while Q_H remained similar to that of previous days (Figure 8). Instead the energy went into storage heat flux (ΔQ_S) (heating the urban fabric). On the previous day (92/209), the largest Q_E fluxes in the measurement period were observed. By 92/210 there had been a significant reduction in availability of surface moisture (Figure 8: surface moisture sensors), so the surface was starting to exert a more significant control on energy partitioning. Throughout July 1992 in Chicago, it is probable that the influence of surface morphology on flux partitioning is not as evident as it may be at other times because of the frequency of rainfall events.

The Bowen ratio determined in this study, 0.87, is lower than the "typical" value of 1.0 suggested by Oke (1982) for suburban areas. It also is considerably lower than values observed in the summertime in Tucson, Sacramento, and Los Angeles (1.80, 1.40, and 1.38 respectively for daytime values) (Grimmond and Oke 1994). However, the value is not physically unrealistic given the conditions in Chicago in 1992. As was noted, flux measurements were restricted as to the time period for which they were conducted and the range of conditions experienced.

The χ ratio expresses how much energy is going into warming the air rather than drying the surface or warming the urban fabric. The χ ratio in Chicago behaves in a similar manner to that in other urban areas, showing an increase through the day ($Q^*>0$ time period) (Grimmond and Cleugh 1994). The mean daytime ratio (0.32) (daily value 0.35) is lower than the typical (0.39) values suggested by Oke (1982), and lower than those reported for Tucson, Sacramento and Los Angeles (0.46, 0.40 and 0.36) (Grimmond and Oke 1994). Given the prevailing meteorological conditions in Chicago during the study period, it is likely that more energy than usual was used to dry surfaces rather than warm the air or the urban fabric, i.e., the β and χ ratios are lower than would have been obtained under drier periods, and Υ is higher.

To obtain an idea of the variability of energy partitioning between seasons and years, it is useful to consider the data from Vancouver (Table 9). The Sunset neighborhood in Vancouver is one of the few urban sites where energy balance studies have been conducted over a number of years and thus under a range of synoptic conditions. There is considerable variability among seasons both within and across years (Table 9). However, it is important to note that

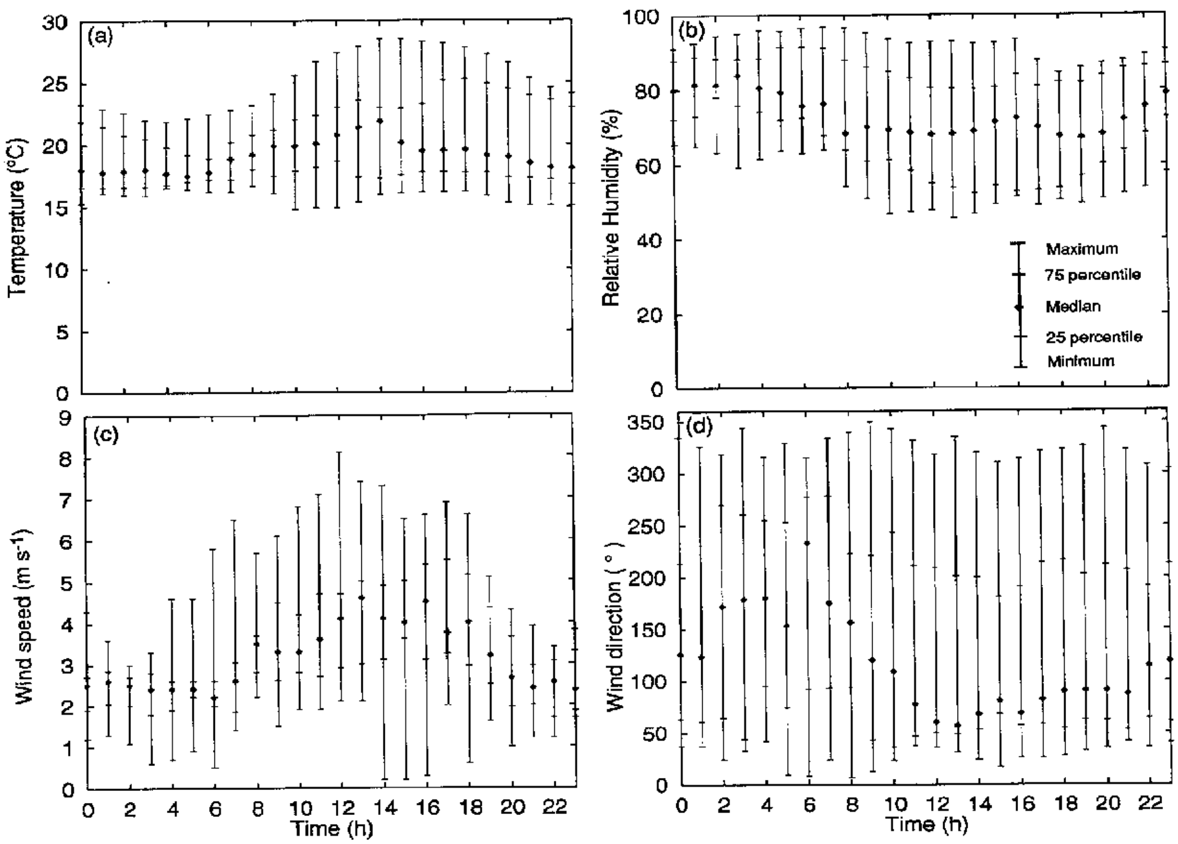


Figure 6.—Range of meteorological conditions during the intensive study period: Diurnal ranges of temperature, relative humidity, windspeed and wind direction. Minimum (0), 25, 50 (median), 75 and 100 (maximum) percentiles plotted. Median plotted as a diamond, and minimum, 25 and 75 percentile values, and maximum plotted as horizontal lines).

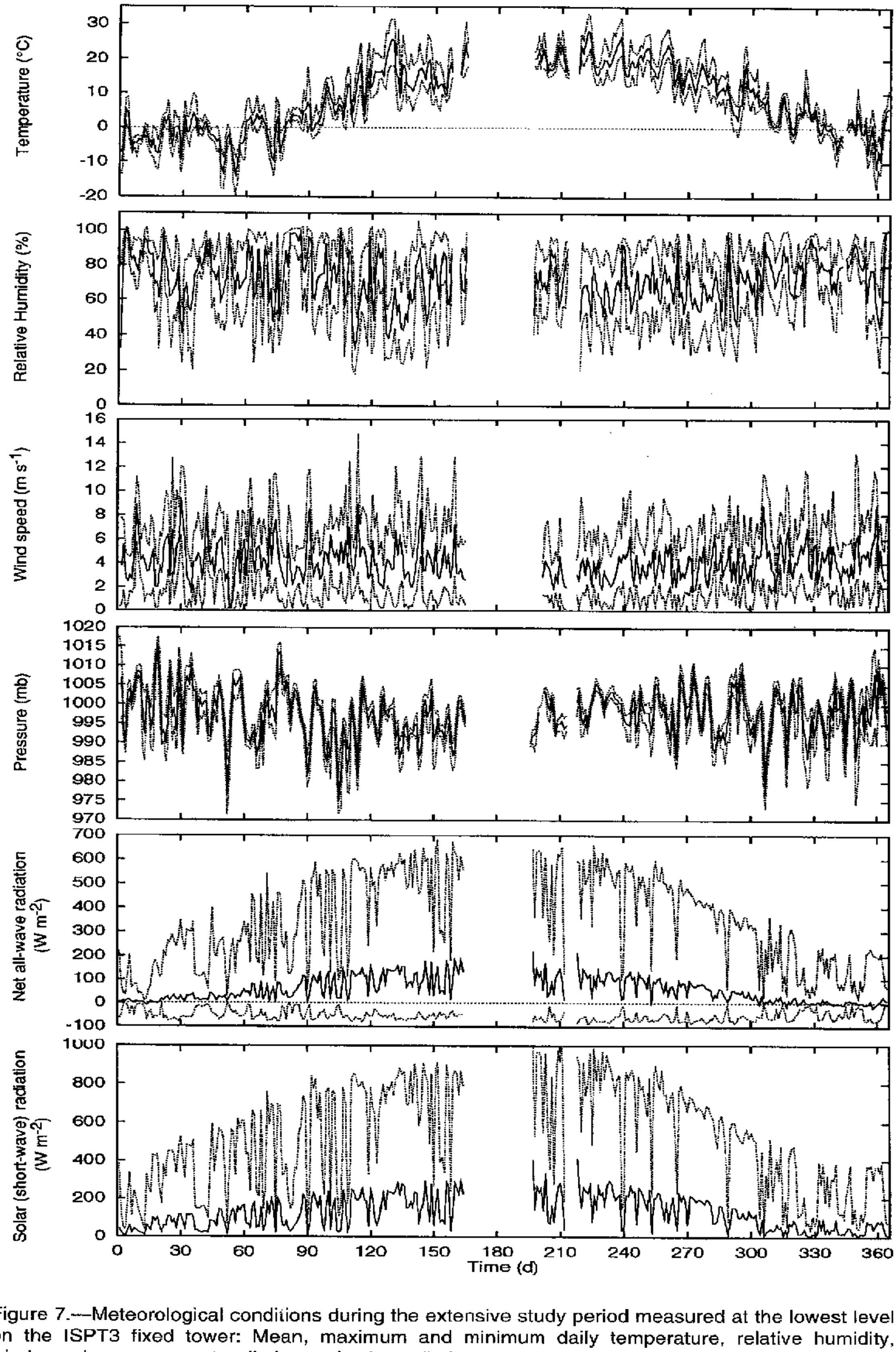


Figure 7.—Meteorological conditions during the extensive study period measured at the lowest level on the ISPT3 fixed tower: Mean, maximum and minimum daily temperature, relative humidity, windspeed, pressure, net radiation and solar radiation.

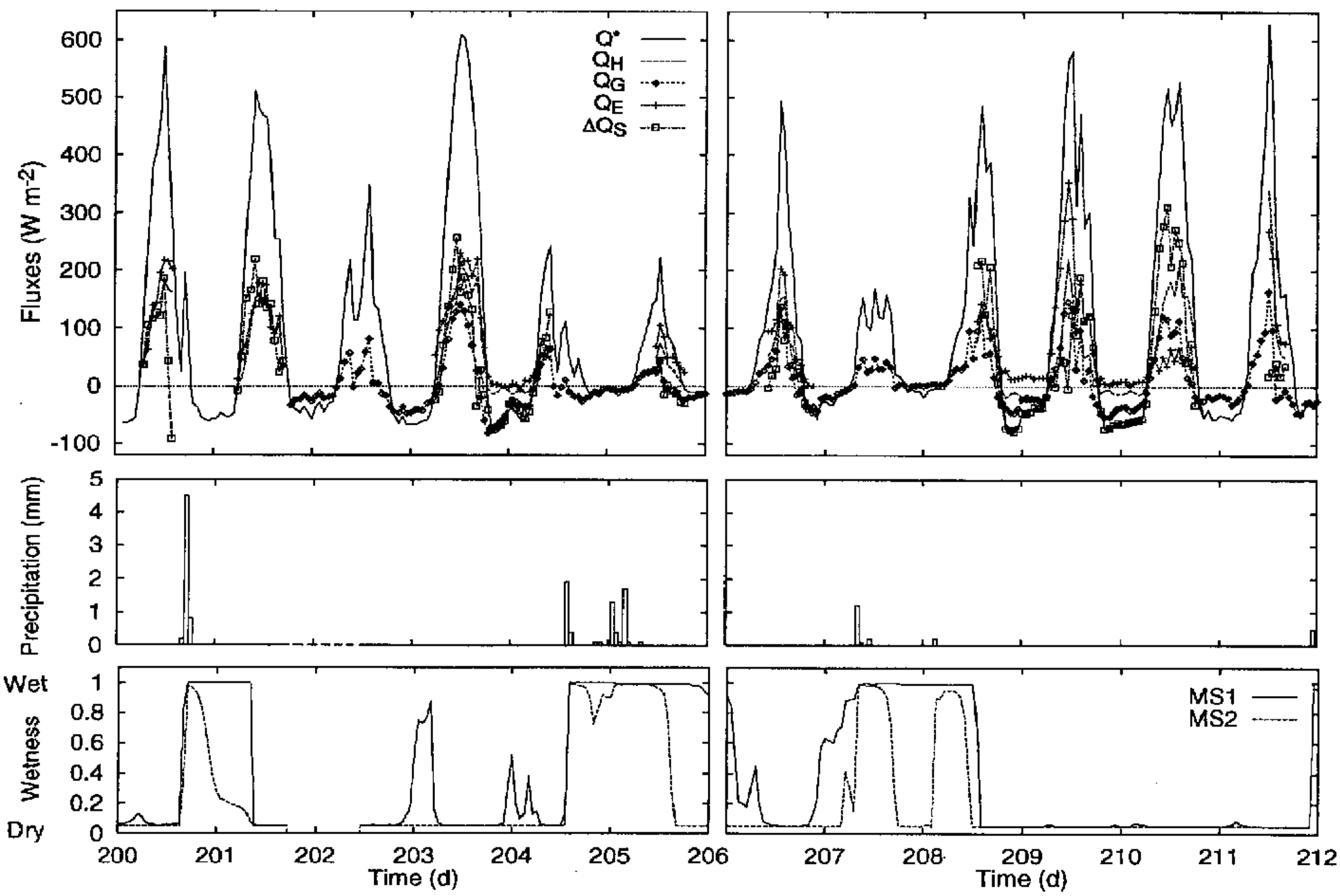


Figure 8.—Time series of energy balance fluxes, precipitation and surface moisture status. (Note the surface moisture status is a relative index, wet 1.0, dry 0.0, MS1 and MS2 are moisture sensors on vegetation and impervious surfaces respectively. These respond to both precipitation events and dew). (From July 20th (day 200) to July 31st (212), 1992). Note break in data 201-202.

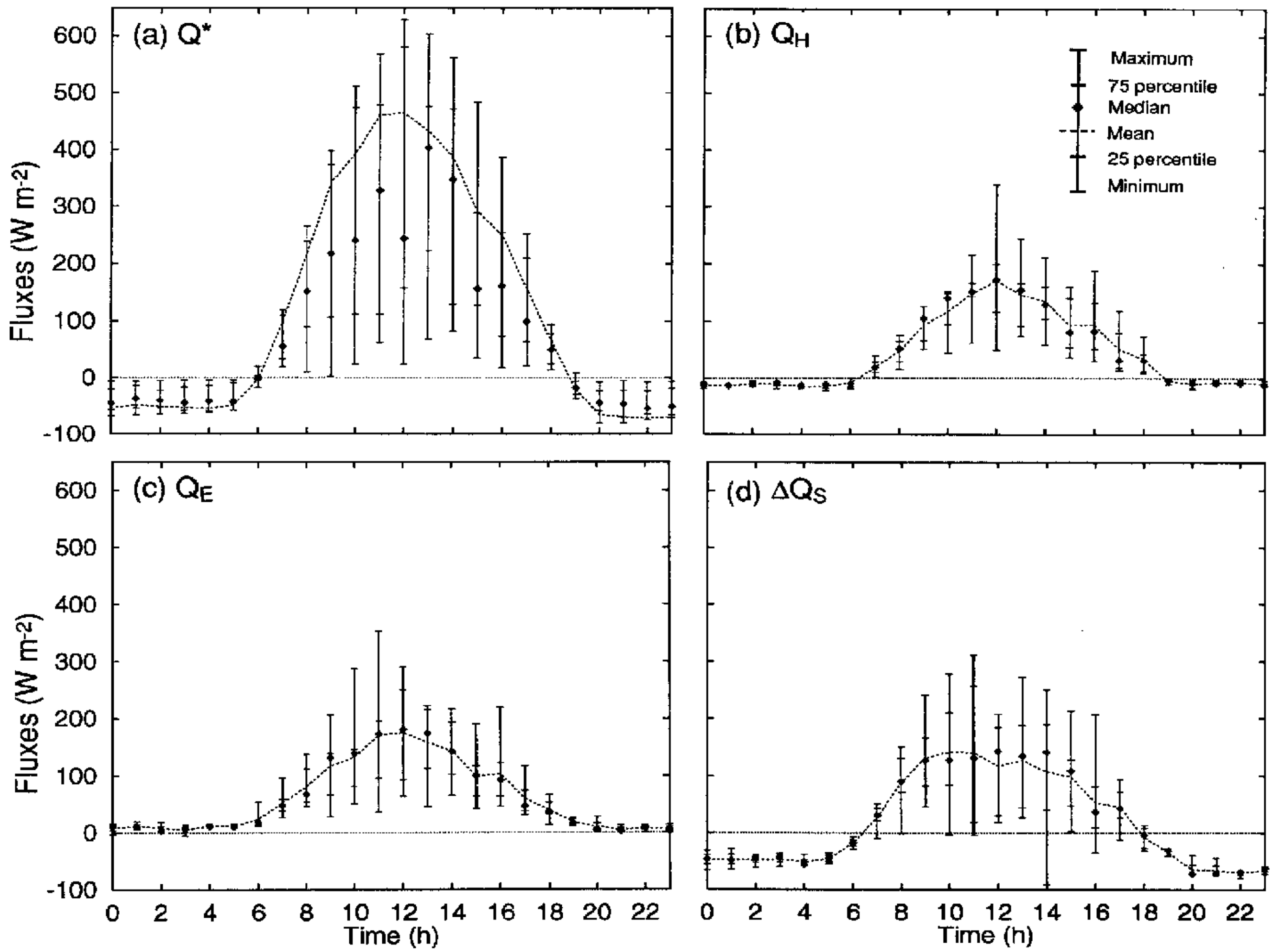


Figure 9.—Ensemble diurnal energy balance fluxes for Chicago, July 1992. Mean, 0 (minimum), 25, 50 (median), 75 and 100 (maximum) percentiles plotted. Mean values joined with a dashed line, median plotted as a diamond, and minimum, 25 and 75 percentile values, and maximum plotted as horizontal lines. The number of hours used in the analysis is indicated on Figure 11.

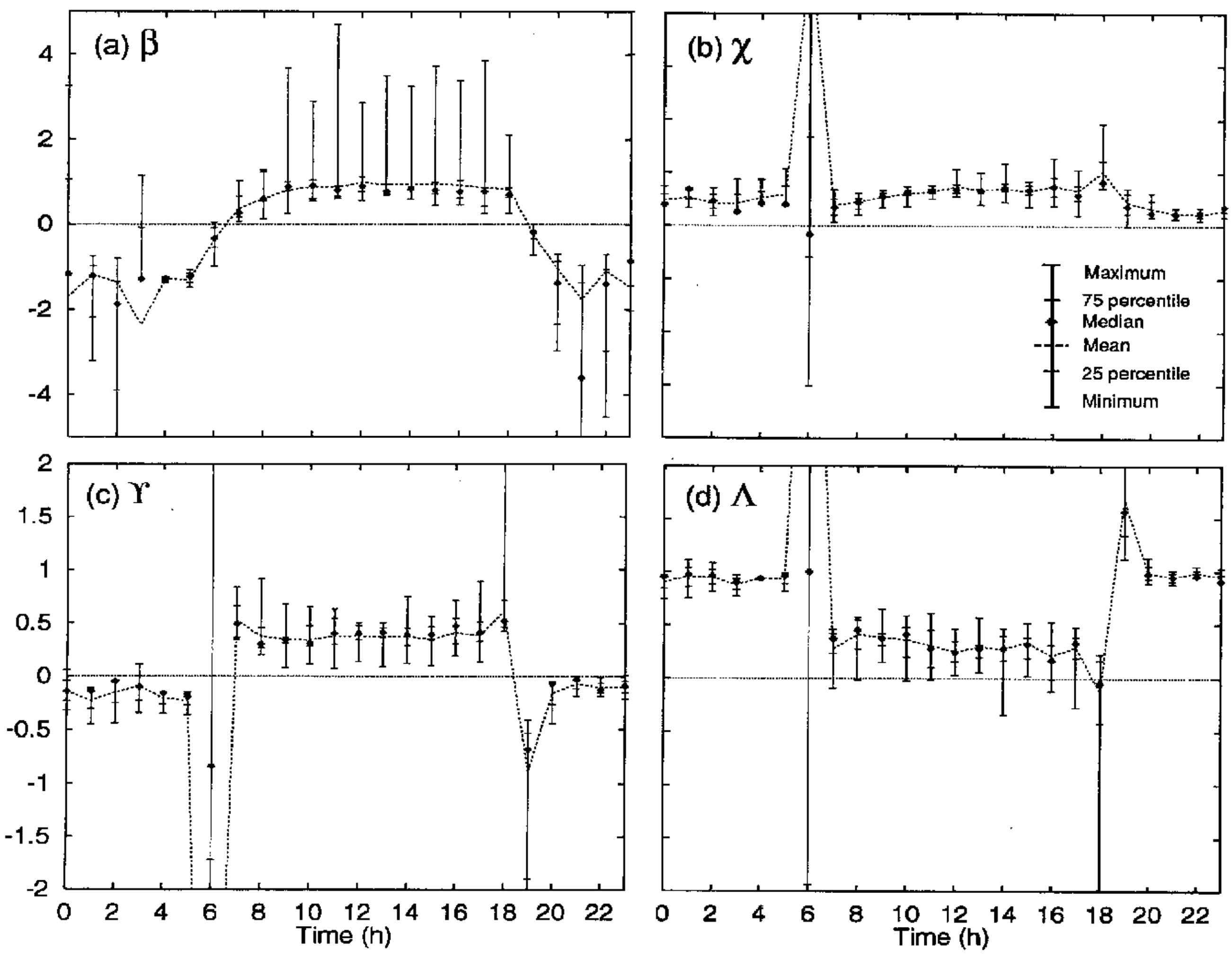
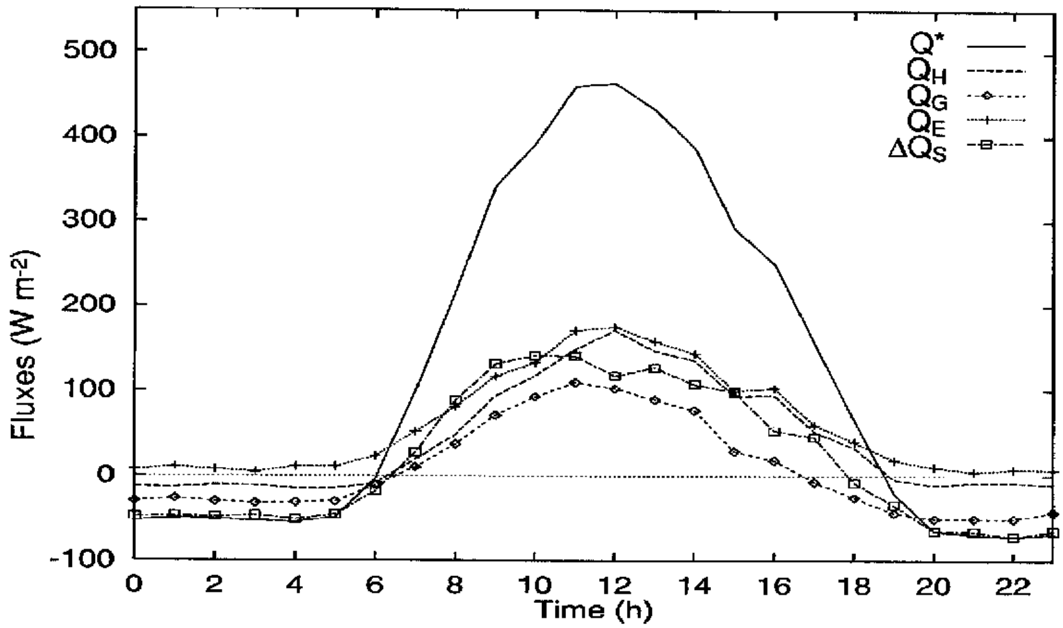


Figure 10.—Diurnal patterns and variability of β, χ, ϒ and Λ ratios (see text for explanation)



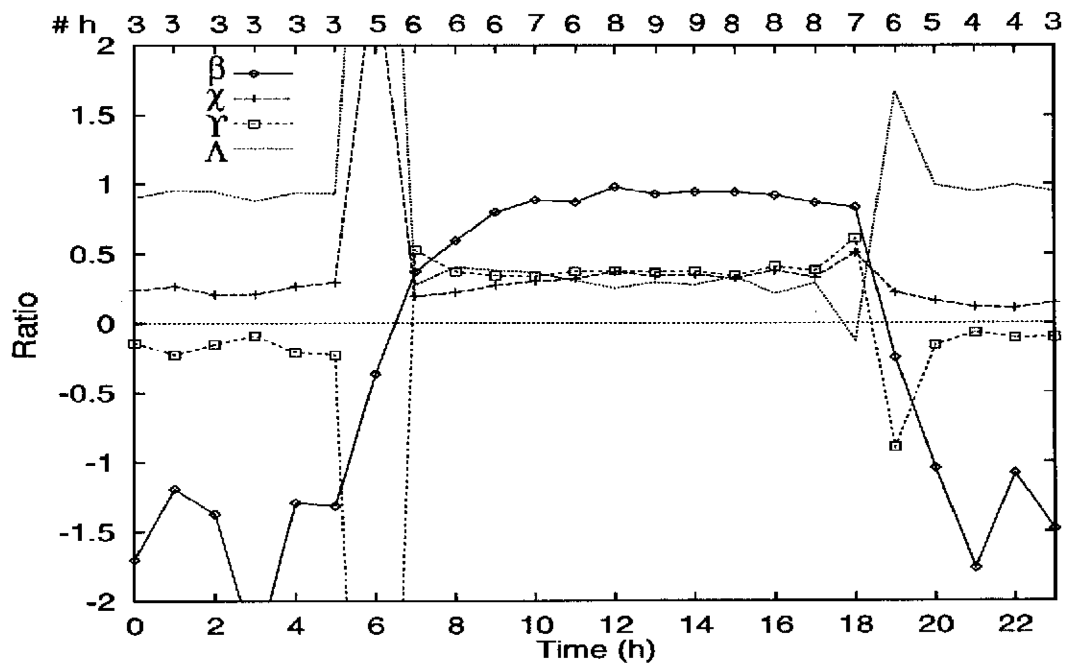


Figure 11.—Ensemble energy balance fluxes and ratios of flux partitioning for Chicago, July 1992. The numbers on the Figure #h indicate the number of hours used in analyses.

not all the results of the studies are directly comparable because of differences in instrumentation and methods between years (Table 9). As in this study, only Roth and Oke (in press) used eddy correlation techniques to measure directly both convective fluxes (Q_E and Q_H). Roth (1991) intercompared Bowen ratios determined from a Bowen ratio system β_B (*a reversing-temperature difference system*) and from eddy correlation techniques (β_{EC}). He concluded that the β_{EC} generally were lower in the daytime than the β_B. The data from Chicago fall within the range of observations for Vancouver.

Future Directions

An issue that needs further study is the representativeness of the observations reported here. This requires consideration of both the climatological and morphological conditions of the study period and site. There are obvious advantages to supplementing these data with further direct observations and data analysis to document the spatial and temporal variability of fluxes for this metropolitan area and to investigate further the role of advection.

Work is in progress to correlate fluxes (Q_E and Q_H) with tree-cover density (Demanes 1994), with the intention of investigating the influence of trees on flux partitioning, for example, the ratio Υ. The hypothesis is that greater Υ and smaller β ratios are associated with more heavily treed source areas; this would imply that energy is going into evaporation so that air below might be expected to be cooler. The GIS system will provide a basis for interpreting flux measurements in terms of the surface features influencing them and their spatial representativeness, and for objectively determining model input for surface parameters which are spatially consistent with the measured data used to evaluate numerical boundary layer models. These numerical models will be used to predict the effects of different tree-planting scenarios on local scale energy and water exchanges.

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Table 9.—Variability of ratios determined in the Sunset Neighborhood of Vancouver, British Columbia

		Daytime				Daily				Methods	
Reference	Period	β	χ	Υ	Λ	β	χ	Υ	Λ	ΔQ_S[1]	Conv[2]
Kalanda (1979)[3]	77/Aug 19 to Oct 3	1.03								A	a
Oke and McCaughey (1983)[4]	80/Jul to mid Aug	0.16	0.11	0.67	0.23	0.14	0.1	0.73	0.20	A	a
Cleugh & Oke (1986)	83/Jul 18 to Sep 22	1.28	0.44	0.34	0.22					A	b
Cleugh (1990)	86/Apr 5 to Oct 2	2.15	0.50	0.26	0.24					B	c
Grimmond (1992)[5]	87/Jan 21 to Feb 28	0.80	0.36	0.45	0.19	0.69	0.59	0.85	-0.44	B	c
	87/ Mar 1 to 31	1.29	0.42	0.32	0.26	1.19	0.53	0.45	0.02	B	c
	87/ Apr 1 to 30	0.87	0.35	0.40	0.25	0.85	0.42	0.49	0.09	B	c
	87/ May 1 to 31	1.26	0.40	0.33	0.29	1.36	0.48	0.36	0.16	B	c
	87/ Jun 1 to 28	1.40	0.42	0.30	0.29	1.47	0.50	0.34	0.17	B	c
Roth and Oke (1994)[6]	89/July	1.97								B	d

[1] ΔQ_S: A= Oke et al. 1981; B= Grimmond et al. 1991.

[2] Conv: Method of convective flux determination: a= Bowen ratio/energy balance—reversing temperature difference system; b= Q_H SAT and Q_E residual; c= Bowen ratio and SAT; d= KH20 and SAT eddy correlation systems.

[3] Mean of daytime β values (rather than determined from the mean of the fluxes for the period); median 0.77, range of daytime values 0.3 to 2.39.

[4] Very wet spring.

[5] Ratios are over Q^*+Q_F rather than Q^* only.

[6] Mean of daytime hourly mean β, median 1.85, range of mean hourly values during the daytime 1.25 to 3.0. Also determined using Bowen ratio methods; β was smaller using eddy correlation techniques.

Literature Cited

Ackerman, B. 1985. **Temporal march of the Chicago heat island.** Journal of Climate and Applied Meteorology. 24: 547-554.

Ackerman, B. 1987. **Climatology of Chicago area urban-rural difference in humidity.** Journal of Climate and Applied Meteorology. 26: 427-430.

Akbari, H.; Davis, S.; Dorsano, S.; Huang, J.; Winnett, S., eds. 1992. **Cooling our communities: a guidebook on tree planting and light-colored surfacing.** Washington DC: U.S. Environmental Protection Agency. 217 p.

Cleugh, H. A. 1990. **Development and evaluation of a suburban evaporation model: a study of surface and atmospheric controls on the suburban evaporation regime.** Vancouver, B.C., The University of British Columbia. 249 pp. Ph.D. dissertation.

Cleugh, H. A.; Oke, T. R. 1986. **Suburban-rural energy balance comparisons in summer for Vancouver, B.C.** Boundary Layer Meteorology. 36: 351-36

Demanes M. 1994. **Evaluation of surface controls on urban energy flux partitioning using GIS.** M.A. Thesis, Indiana University (in preparation).

Gash, J. H. C. 1986. **A note on estimating the effect of a limited fetch on micrometeorological evaporation measurements.** Boundary Layer Meteorology. 36: 409-414.

Grant R.H. 1993. **Chicago, IL: Climate and meso climate.** Unpublished report on file USDA Forest Service, Co-Operative Research Agreement #23-685, 17pp.

Grant, R. H.; Heisler, G. M. 1994. **Design of a low powered ventilated radiation shield.** In preprints of the 21st conference on agriculture and forest meteorology, San Diego, CA, March 1994.

Grimmond, C. S. B. 1992. **The suburban energy balance: methodological considerations and results for a mid-latitude west coast city under winter and spring conditions.** International Journal of Climatology. 12: 481-497.

Grimmond, C. S. B.; Cleugh, H. A.; Oke, T. R. 1991. **An objective hysteresis model for urban storage heat flux and its comparison with other schemes.** Atmospheric Environment. 25B: 311-326.

Grimmond C. S. B.; Oke, T. R; Cleugh, H. A.; . 1993. **The role of "rural" in comparisons of observed suburban—rural flux differences. Exchange processes at the land surface at a range of space and time scales.** International Association of Hydrological Sciences Publ. 212: 165-174.

Grimmond, C. S. B.; Cleugh, H. A. 1994. **A simple method to determine Obukhov lengths for suburban areas.** Journal of Applied Meteorology. 33: 435-440.

Grimmond C. S. B.; Oke T.R. 1994. **Comparison of heat fluxes from summertime observations in the suburbs of four N. American cities** (in preparation).

Grimmond, C. S. B.; Souch, C. 1994. **Surface description for urban climate studies: a GIS based methodology.** Geocarto International (in press).

Hall, C. D. 1954. **Forecasting the lake breeze and its effects on visibility at Chicago Midway airport.** Bulletin of the American Meteorological Society. 35: 105-111.

Hanna, S. R. 1971. **A simple method of calculating dispersion from urban area sources.** Air Pollution Control Association Journal. 21: 774-777.

Heisler G. M. 1974. **Trees and human comfort in urban areas.** Journal of Forestry. 72: 466-469.

Heisler, G. M.; Grimmond,S.; Grant, R.; Souch, C. 1994. **Investigation of the influence of Chicago's urban forest on wind and temperature within residential neighborhoods.** In: McPherson, E. G., Nowak, D. J. and 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.

Horst, T. W.; Weil, J. C. 1992. **Footprint estimation of scalar flux measurements in the atmospheric surface layer.** Boundary-Layer Meteorology. 59: 279-296.

Kalanda, B. D. 1979. **Suburban evapotranspiration estimates in Vancouver from energy balance measurements.** Vancouver, B.C. The University of British Columbia, 123 pp. M.Sc. thesis.

Leclerc, M.; Thurtell, G. W. 1990. **Footprint predictions of scalar fluxes using a Markovian analysis.** Boundary-Layer Meteorology. 52: 247-258.

Lenschow, D. H. 1986. **Probing the atmospheric boundary layer.** Boston, MA. American Meteorological Society. 269pp.

Lyons, W. A. 1972. **The climatology and prediction of the Chicago lake breeze.** Journal of Applied Meteorology. 11: 1259-1270.

McPherson, E. G.; Nowak, D. J.; Sacamano, P. L.; Prichard, S. E.; Makra, E. M. 1993. **Chicago's evolving urban forest: initial report of the Chicago Urban Forest Climate Project.** General Technical Report NE-169. Radnor, PA: US Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 55 pp.

Nowak, D. J. 1994. **Urban forest structure: the state of Chicago's urban forest.** In: McPherson, E. G., Nowak, D. J. and 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.

Oke, T. R. 1982. **The energetic basis of the urban heat island.** Quarterly Journal Royal Meteorological Society. 108: 1-24.

Oke, T. R. 1984. **Methods in urban climatology.** In: Kirchofer W.; Ohmura, A; Wanner. W. eds. Applied Climatology. Zurcher Schriften. 14. ETH, Zurich: 19-29.

Oke, T. R. 1987. **Boundary layer climates** (2nd ed.), London. Methuen, 435 pp.

Oke, T. R. 1988. **The urban energy balance.** Progress in Physical Geography. 12: 471-508.

Oke, T. R.; Cleugh, H. A. 1987. **Urban heat storage derived as energy budget residuals.** Boundary Layer Meteorology. 39: 233-245.

Oke, T. R.; McCaughey, J. H. 1983. **Suburban-rural energy balance comparisons for Vancouver, B.C.: an extreme case?** Boundary Layer Meteorology. 26: 337-354.

Oke T. R.; Cleugh, H. A.; Grimmond, C. S. B.; Schmid, H. P.; Roth, M. 1989. **Evaluation of spatially-averaged fluxes of heat, mass and momentum in the urban boundary layer.** Weather and Climate. 9: 14-21.

Oke, T. R; Kalanda, N.; Steyn, D. G. 1981. **Parameterisation of heat storage in urban areas.** Urban Ecology. 5: 45-54.

Reifsnyder, W.E. 1967. **Radiation geometry in the measurement and interpretation of radiation balance.** Agricultural Meteorology. 4: 255-265.

Roth, M. 1991. **Turbulent transfer characteristics over a suburban surface.** Vancouver, B.C. The University of British Columbia, 292 pp. Ph.D. dissertation.

Roth M.; T. R. Oke; 1994. **Comparison of modelled and "measured" heat storage in suburban terrain.** Contributions to Atmospheric Physics. (Beitrage zur Physik der Atmosphare). (submitted).

Scheff, P. A.; Wadden, R. A.; Alles, R. J. 1984. **Quantitative assessment of Chicago air pollution through analysis of covariance.** Atmospheric Environment. 18: 1044-1050.

Schmid, H. P.; Oke, T. R. 1990. **A model to estimate the source area contributing to turbulent exchange in the surface layer over patchy terrain.** Quarterly Journal Royal Meteorological Society. 116: 965-988.

Schmid, H. P.; Cleugh, H. A.; Grimmond, C. S. B.; Oke, T. R. 1991. **Spatial variability of energy fluxes in suburban terrain.** Boundary Layer Meteorology. 54: 249-276.

Schuepp, P. H.; Leclerc, M. Y.; McPherson, J. I; Desjardin, R. L. 1990. **Footprint prediction of scalar fluxes from analytical solutions of the diffusion equation.** Boundary Layer Meteorology. 50: 355-374.

Sexton, K.; Westberg, H. 1980. **Elevated ozone concentrations measured downwind of the Chicago-Gary urban complex.** Journal of the Air Pollution Control Association. 30: 911-914.

Swinford, R. 1980. **Vertical ozone profile in the lower troposphere over Chicago.** Journal of the Air Pollution Control Association. 30: 794-796.

Steyn, D. G. 1985. **An objective method to achieve closure of overdetermined surface energy budgets.** Boundary Layer Meteorology. 33: 303-311.

Tanner B.; Greene J.P. 1989. **Measurements of sensible heat and water vapor fluxes using eddy correlation techniques.** Unpublished report on file at U.S. Army Dugway Proving Grounds.

Wadden, R. A.; Ross, E. D. and Quon, J. E. 1979. **Characterization of ozone episodes in urban air.** Journal of the Environmental Engineering Division. 105: 621-628.

Webb E. K.; Pearman G. I; Leunig, R. 1980. **Correction of flux measurements for density effects due to heat and water vapour transfer.** Quarterly Journal of the Royal Meteorological Society. 106: 85-100.

Chapter 5

Air Pollution Removal by Chicago's Urban Forest

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Abstract

In 1991, trees in the City of Chicago (11 percent tree cover) removed an estimated 15 metric tons (t) (17 tons) of carbon monoxide (CO), 84 t (93 tons) of sulfur dioxide (SO_2), 89 t (98 tons) of nitrogen dioxide (NO_2), 191 t (210 tons) of ozone (O_3), and 212 t (234 tons) of particulate matter less than 10 microns (PM10). Across the study region of Cook and DuPage Counties, trees (in-leaf season) removed an average of 1.2 t/day (1.3 tons/day) of CO, 3.7 t/day (4.0 tons/day) of SO_2, 4.2 t/day (4.6 tons/day) of NO_2, 8.9 t/day (9.8 tons/day) of PM10 and 10.8 t/day (11.9 tons/day) of O_3. The value of pollution removal in 1991 was estimated at $1 million for trees in Chicago and $9.2 million for trees across the study area. Average hourly improvement (in-leaf season) in air quality due to all trees in the study area ranged from 0.002 percent for CO to 0.4 percent for PM10. Maximum hourly improvement was estimated at 1.3 percent for SO_2, though localized improvements in air quality can reach 5 to 10 percent or greater in areas of relatively high tree cover, particularly under stable atmospheric conditions during the daytime (in-leaf season). Large, healthy trees remove an estimated 60 to 70 times more pollution than small trees. This paper discusses the ways in which urban trees affect air quality, limitations to estimates of pollution removal by trees in the Chicago area, and management considerations for improving air quality with urban trees.

Introduction

Air pollution is a multibillion dollar problem that affects most major U.S. cities. Air pollution is a significant human health concern as it can cause coughing, headaches, lung, throat, and eye irritation, respiratory and heart disease, and cancer. It is estimated that about 60,000 people die annually in the United States from the effects of particulate pollution (Franchine 1991). In addition, air pollution damages vegetation and various anthropogenic materials. In some of the more heavily polluted areas of the world, observed material deterioration rates are 10 to 100 times faster than those in the preindustrial age (NAPAP 1991). Air pollution also reduces visibility. In the rural mountain/desert areas of the Southwest, the standard visual range is about 130 to 190 km. In rural areas south of the Great Lakes and east of the Mississippi River, the standard visual range is about 20 to 35 km. Aerosol data indicate that this difference is due to greater sulfate concentrations in the East (and the interaction of sulfates with the higher humidity of the East) (Trijonis et al. 1990). Air pollution also contributes to acidic deposition (Smith 1990).

Major air pollutants in urban areas are carbon monoxide (CO), predominantly from automobiles in urban areas; nitrogen oxides (NO_X), mainly from automobiles and stationary combustion sources; ozone (O_3), formed through chemical reactions involving the principal precursors of NO_x and volatile organic compounds; sulfur dioxide (SO_2), emissions mostly from stationary combustion sources and smelting of ores; and particulate matter.

Small particulate matter (PM10: particulate matter less than 10 μm) results from local soils, industrial processes, combustion products, and chemical reactions involving gaseous pollutants. Small particles can have significant health effects because particles less than 5 μm may escape the defense mechanisms of the upper respiratory tract and enter the lungs. Particles 0.5 to 5 μm may be deposited as deep as the bronchioles in the lung but usually are removed by cilia within a few hours. Particles less than 0.5 μm may reach and settle in the lung alveoli, remaining for weeks, months or years (Stoker and Seager 1976).

Air pollution is removed from the air primarily by three mechanisms: wet deposition, chemical reactions, and dry deposition. (Rasmussen et al. 1975; Fowler 1980). Wet deposition involves precipitation scavenging that includes "rainout" (transfer of pollutants to cloud droplets before they begin to fall) and "washout" (transfer of pollutants to falling rain/snowdrops) mechanisms. Gas phase reactions in the atmosphere can create aerosols that are removed by wet or dry deposition or produce oxidized products such as carbon dioxide (CO_2) and water vapor. Dry deposition is the mechanism by which gaseous and particulate pollutants are transported to and dry deposited on various surfaces, including trees.

Gaseous Pollutants

Dry deposition of gases to trees occurs predominantly through the leaf stomates, though some deposition occurs on the plant surface (Fowler 1985; Murphy and Sigmon 1990; Smith 1990). During daylight hours when plant leaves are transpiring water and taking up CO_2, other gases including pollutants are taken up into the leaf. Once inside the leaf, these gases diffuse into intercellular spaces and can be absorbed by water films on inner-leaf surfaces. Pollutant uptake by plants is highly variable as it is regulated by numerous plant, pollutant, and environmental forces (e.g., plant water deficit, light intensity, windspeed, gas solubility in water, leaf size and geometry) (Smith 1990). Once the gas reacts with the tree and is absorbed, it is removed from the atmosphere. However, plants also emit various compounds that can contribute to air pollution. The following sections outline plant-pollutant interactions for significant gaseous air pollutants in urban areas.

Carbon Monoxide

Carbon monoxide is harmful principally to animals due to its affinity for hemoglobin. When CO reacts with hemoglobin it reduces the ability of blood to transport oxygen (Ziegler 1973; Stoker and Seager 1976). It has been hypothesized that CO inhibits N_2-fixation in plants (Ziegler 1973). Most CO absorbed by plants is reduced and incorporated into serine, which is subsequently converted to sucrose (Bidwell and Fraser 1972).

Trees emit volatile organic compounds such as isoprene and monoterpenes into the atmosphere. These compounds are natural chemicals that make up essential oils, resins, and other plant products, and may be useful in attracting pollinators or repelling predators (Kramer and Kozlowski 1979). Complete oxidation of volatile organic compounds ultimately produces CO_2, but CO is an intermediate compound in this process. Oxidation of volatile organic compounds is an important component of the global CO budget (Tingey et al. 1991); CO also can be released from chlorophyll degradation (Smith 1990).

Nitrogen Dioxide

After nitrogen dioxide is absorbed through leaf stomates, it can react with water on the moist surfaces of the inner leaf to form nitrous (HNO_2) and nitric (HNO_3) acids. Pollutant interactions and altering of pH in the leaf can lead to altered plant metabolism (e.g., inhibition of CO_2 fixation, suppressed growth) (Ziegler 1973; Smith 1990). Visible leaf injury would be expected at concentrations around 1.6 to 2.6 ppm for 48 hours, 20 ppm for 1 hour, or a concentration of 1 ppm for as many as 100 hours (Natl. Acad. of Sci. 1977a). Concentrations that would induce foliage symptoms would be expected only in the vicinity of an excessive industrial source (Smith 1990). Trees generally are not considered as a source of atmospheric nitrogen oxides, though plants, particularly agricultural crops, are known to emit ammonia (NH_4). Emissions occur primarily under conditions of excess nitrogen (e.g., after fertilization) and during the reproductive growth phase (Schjoerring 1991); NH_4 in the atmosphere can be converted to NO_x.

Ozone

Ozone has low solubility in water but readily diffuses into stomatal cavities. The reactive nature of O_3 causes it to react rapidly on inner-leaf surfaces (Smith 1984). Eastern deciduous species are injured by exposures to O_3 at 0.20 to 0.30 ppm for 2 to 4 hours (Natl. Acad. of Sci. 1977b). The threshold for visible injury of eastern white pine is approximately 0.15 ppm for 5 hours (Costonis 1976). Sorption of O_3 by white birch seedlings shows a linear increase up to 0.8 ppm; for red maple seedlings the increase is up to 0.5 ppm (Townsend 1974). Severe O_3 levels in urban areas can exceed 0.3 ppm (Off. Technol. Assess. 1989). Injury effects can include altered photosynthesis, respiration, growth, and stomatal function (Shafer and Heagle 1989; Smith 1990).

Trees can contribute to O_3 formation by emitting volatile organic compounds (Brasseur and Chatfield 1991). Because these emissions are temperature dependent and trees generally lower air temperatures, it is believed that increased tree cover lowers overall volatile organic emissions and O_3 levels in urban areas, but additional research is needed (Cardelino and Chameides 1990). Volatile organic emissions of urban trees generally are less than 10 percent of total emissions in urban areas (Nowak 1991).

Sulfur Dioxide

Following absorption through leaf stomates, SO_2 is presumed to be dissolved in moisture films on inner-leaf cell walls. Eventually, sulfurous acid (H_2SO_3) and, following oxidation, sulfuric acid (H_2SO_4) are formed. Toxic effects of SO_2 may be due to its acidifying influence and/or the sulfite (SO_3^{2-}) and sulfate (SO_4^{2-}) ions that are toxic to a variety of biochemical processes (Smith 1990). Stomata may exhibit increases in either stomatal opening or stomatal closure when exposed to SO_2 (Smith 1984; Black 1985). Acute SO_2 injury to native vegetation does not occur below 0.70 ppm for 1 hour or 0.18 ppm for 8 hours (Linzon 1978). A concentration of 0.25 ppm for several hours may injure some species (Smith 1990).

Trees can make minor contributions to SO_2 concentration by emitting sulfur compounds such as hydrogen sulfide (H_2S) and SO_2 (Garsed 1985; Rennenberg 1991). H_2S, the predominant sulfur compound emitted, is oxidized in the atmosphere to SO_2. Higher rates of sulfur emissions from plants are observed in the presence of excess atmospheric or soil sulfur. However, sulfur compounds also can be emitted with a moderate sulfur supply (Rennenberg 1991).

Particulate Pollution

Particles can be dry deposited on plant surfaces through sedimentation under the influence of gravity or through impaction under the influence of wind. Particles hitting the tree may be retained on the surface, rebound off it, or be retained temporarily and subsequently removed (resuspended into air or transported to soil or other surface) (Smith 1990). The interception and retention of particles by plants is highly variable —smaller leaves and/or leaves with a rough surface are more efficient in collecting particles than larger and/or smoother leaves. Also, larger particles are deposited on leaves more rapidly than smaller particles (Smith 1984; Davidson and Wu 1990). Particle resuspension after 1 hour of initial retention varies from 91 percent for oak leaves to 10 percent for pines (Witherspoon and Taylor 1969).

Thus, vegetation generally is only a temporary retention site for atmospheric particles as particles can be resuspended to the atmosphere, be washed off by rain, or drop to the ground through leaf and twig fall. Trees can store various trace metals in their tissue, but the mechanisms and pathways of incorporation into trees needs to be clarified (Rolfe 1974; Baes and Ragsdale 1981; Baes and McLaughlin 1984). However, it is known that heavy metals can be absorbed directly through the cuticle (Ziegler 1973).

Trace metals can be toxic to plant leaves (Darley 1971; Smith 1990). The accumulation of particles on leaves also can reduce photosynthesis by reducing the amount of light reaching the leaf (Darley 1971; Ziegler 1973). Damage to plant leaves can occur from the deposition of acidic droplets (pH < 3.0) (Smith 1990). Acidic rain can be a source of the essential plant nutrients of sulfur and nitrogen, but also can reduce soil nutrient availability through leaching or toxic soil reactions (Shriner et al. 1990). Particles can also affect tree pest/disease populations (Darley 1971; Smith 1990). Trees can contribute to particle concentrations in urban areas by releasing pollen and emitting volatile organic and sulfur compounds that serve as precursors to particle formation (Smith 1990; Sharkey et al. 1991).

Effect of Urban Trees on Air Quality

Urban trees influence local air quality in various ways. First, trees can reduce or increase building energy use by shading buildings, altering air flows and lowering air temperatures through transpiration (e.g., Heisler 1986). In turn, this change in building energy use affects pollution emissions from power plants. By lowering air temperatures, trees also can affect O_3 photochemistry and O_3 precursor emission rates, thus influencing O_3 formation (Cardelino and Chameides 1990). Various tree configurations can alter wind profiles or create local inversions to trap pollutants such that the removal of local pollutants is enhanced (McCurdy 1978). As mentioned previously, trees emit volatile organic and other compounds that can contribute to pollution formation (Sharkey et al. 1991). Finally, trees can intercept atmospheric particles and absorb various gaseous pollutants.

There has been little research on the removal of atmospheric pollution by urban trees. Street trees in the St. Louis area have been estimated to remove approximately 3.1 kg/day (2.75 lb/acre/day) of particles for each hectare of land covered by street trees (DeSanto et al. 1976b). Other particle-removal estimates for individual trees are 1.5 to 4.4 kg/day for each hectare of land covered by trees (1.3 to 3.9 lb/acre/day); 1.5 to 4.7 kg/ha/day (1.3 to 4.2 lb/acre/day) for CO; 1.3 to 4.1 kg/ha/day (1.2 to 3.6 lb/acre/day) for nitrogen oxides; 22.7 to 74.4 kg/ha/day (20.2 to 66.3 lb/acre/day) for SO_2; and 34.7 to 111.5 kg/ha/day (30.9 to 99.5 lb/acre/day) for O_3 (DeSanto et al. 1976a).

Some of these estimates are higher than expected under typical urban conditions because average removal rates in $\mu g/m^2$ of leaf area/hr for vegetation were used. These rates are dependent on the pollutant concentrations used in the studies from which the average removal rate was derived. Often such concentrations in the literature are high so that plant responses to a pollutant can be studied under laboratory conditions. Thus, the removal rates are higher than would be expected under typical urban conditions. Other removal rates for SO_2 and NO_2 are given in Table 1.

The objective of this study was to estimate air pollution removal (dry deposition) of CO, NO_2, O_3, SO_2, and PM10 by trees in the Chicago region during 1991. The computations used to estimate pollution removal by urban trees should be considered a first-order approximation of a highly complex deposition system. Many factors influence dry-deposition removal rates, including aerodynamic roughness, atmospheric stability, pollutant concentration, solar radiation, temperature, turbulence, wind velocity, particle size, gaseous chemical activity and solubility, and vegetative surface characteristics (e.g., stomatal activity and resistances, leaf surface area) (Sehmel 1980).

Methods

Study Area

The study area (Figure 1 in Chapter 2) was fragmented into 117 community areas for detailed analyses of tree canopy cover (McPherson et al. 1993), pollution concentrations and total pollutant flux (Figure 1). Tree cover averages 11 percent in Chicago, 23 percent in suburban Cook County (i.e.,

Table 1.—Pollution-removal values (kg/ha/day) from the literature (divide removal rate by 1.12 to calculate lb/acre/day)

Pollutant	Removal rate	Site	Pollutant concentration (ppm)	Reference
SO_2	0.59	1,723 km^2 forest dominated area on Long Island, NY	0.015	Murphy et al. 1977
SO_2	0.20	Argonne National Laboratory, IL[a]	*	Wesely and Lesht 1988
SO_2	0.15	778 km^2 forest dominated area at Savannah River Plant, SC	0.008	Murphy et al. 1977
SO_2	0.04	Loblolly pine plantation at Savannah River Plant, SC	0.003	Lorenz and Murphy 1985
SO_2	0.03	Loblolly pine plantation in Alamance County, NC	**	Hicks et al. 1982
SO_2	0.03	Argonne National Laboratory, IL[a]	***	Wesely and Lesht 1988
NO_2	0.18	Salt Lake Valley, UT estimate[b]	0.02	Heggestad 1972
NO_2	0.04	Salt Lake Valley, UT estimate[b]	0.005	Heggestad 1972

[a] 50 percent white oak, 50 percent grass.
[b] 85 percent covered by vegetation.

* Peak modeled deposition in 1986 in-leaf season;
** Daytime peak removal extrapolated to entire day, therefore removal rate listed is an overestimate of the actual daily removal rate;
***Minimum modeled deposition in 1986 in-leaf season.

Cook County exclusive of Chicago), 19 percent in DuPage County, and 19 percent for the entire study area (McPherson et al. 1993).

Pollutant concentrations in Illinois in 1991 were typical of concentrations found in the mid-1980s through 1990; the exceptions were PM10 and nitrogen oxides, which were slightly below average (IEPA 1992). The average concentration of CO in the study area was 0.88 ppm. Peak hourly averages occurred in May (1.03 ppm) and minimum hourly concentrations occurred in June (0.65 ppm). The National Ambient Air Quality Standard (NAAQS) of 9 ppm (8-hr average) was not exceeded in the study area in 1991. Concentration levels cycled throughout the day (Figure 2).

Average hourly levels of NO_2 were highest in August (0.025 ppm) and lowest in November (0.019 ppm); the annual average in the study area was 0.021 ppm. Average levels of NO_2 varied through the day (Figure 3). During the in-leaf season, O_3 levels averaged 0.027 ppm; levels were highest in June (0.038 ppm) and lowest in October (0.013 ppm). Average hourly O_3 levels peaked at 2 p.m. (Figure 4). Levels of O_3 exceeded the NAAQS level of 0.12 ppm (1-hr average) on June 1, 18, 20, and 21 at four stations in Chicago and suburban Cook County (IEPA 1992).

The average concentration of SO_2 in the study area was 0.0084 ppm. Hourly averages were highest in January (0.011 ppm) and lowest in December (0.0062 ppm). Average hourly concentration peaked at 9 a.m. and 10 a.m. (Figure 5). The 24-hr average NAAQS level of 0.14 ppm was exceeded in the study area on October 16-17, November 14-15, and November 17-19 at one monitoring station in suburban Cook County (IEPA 1992).

The average level of PM10 in the study area was 34 $\mu g/m^3$. Levels were highest in July (45 $\mu g/m^3$) and lowest in December (27 $\mu g/m^3$). The 24-hr average NAAQS level of 150 $\mu g/m^3$ was exceeded on August 2 for one monitoring station in suburban Cook County (IEPA 1992). Regional air quality concentrations in 1991 probably were not high enough to induce visible damage to vegetation in the Chicago area.

Algorithms for Estimating Pollution Removal

To estimate pollutant flux to trees it is necessary to know the deposition velocity of each pollutant to trees and the local pollutant concentration (e.g., Hicks et al. 1987; Baldocchi 1988; Smith 1990). The deposition velocity may be thought of as the rate at which the surface "cleans" a pollutant from the air. If the deposition velocity of a pollutant is 1.0 cm/sec, then the surface is completely removing the pollutant from a layer of air 1.0 cm thick each second (Smith 1990). The pollutant flux (F) is calculated as the product of the deposition velocity (V_d) and the pollutant concentration (C) :

$$F\ (g/cm^2/sec) = V_d\ (cm/sec) \times C\ (g/cm^3) \qquad (1)$$

The pollutant flux is multiplied by the area of the surface (cm^2) over time periods for which the pollutant concentration is known around that surface (e.g., 1 hour: 3600 sec) to

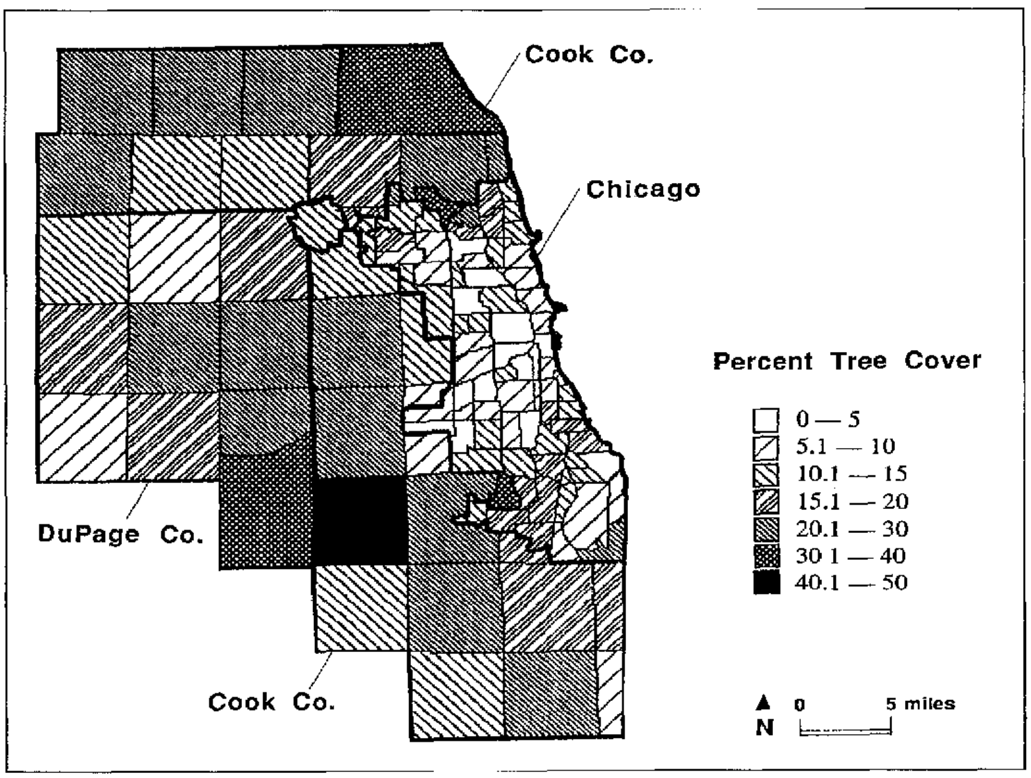

Figure 1. —Percent tree cover by community area.

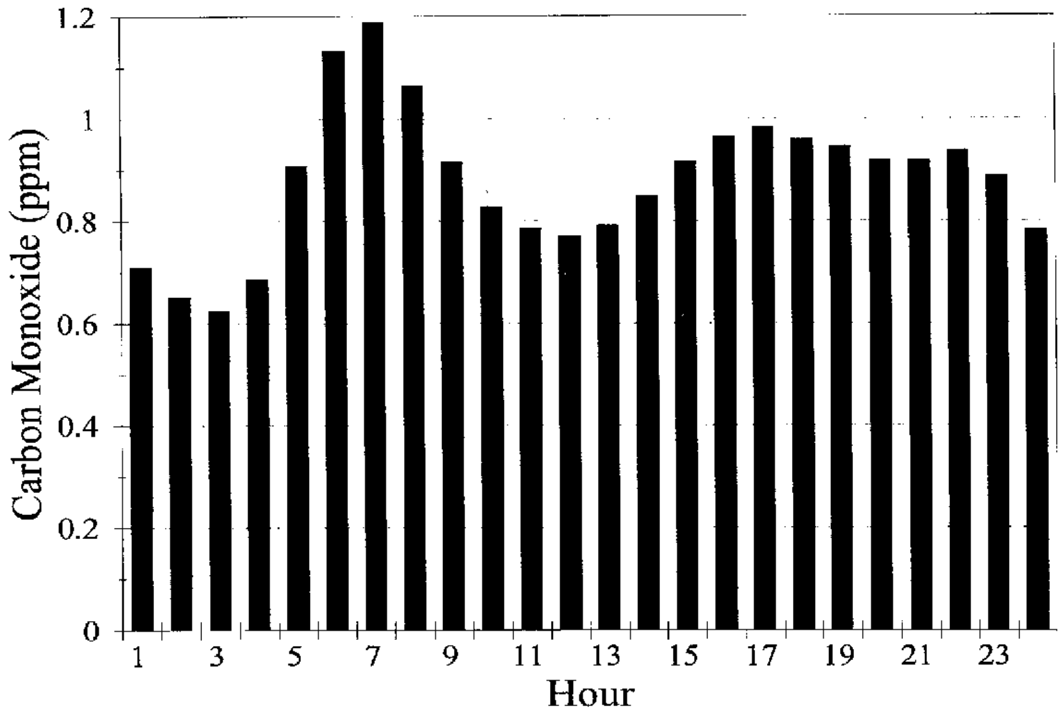

Figure 2. —Average hourly concentrations of CO calculated from seven IEPA monitoring sites in study area in 1991.

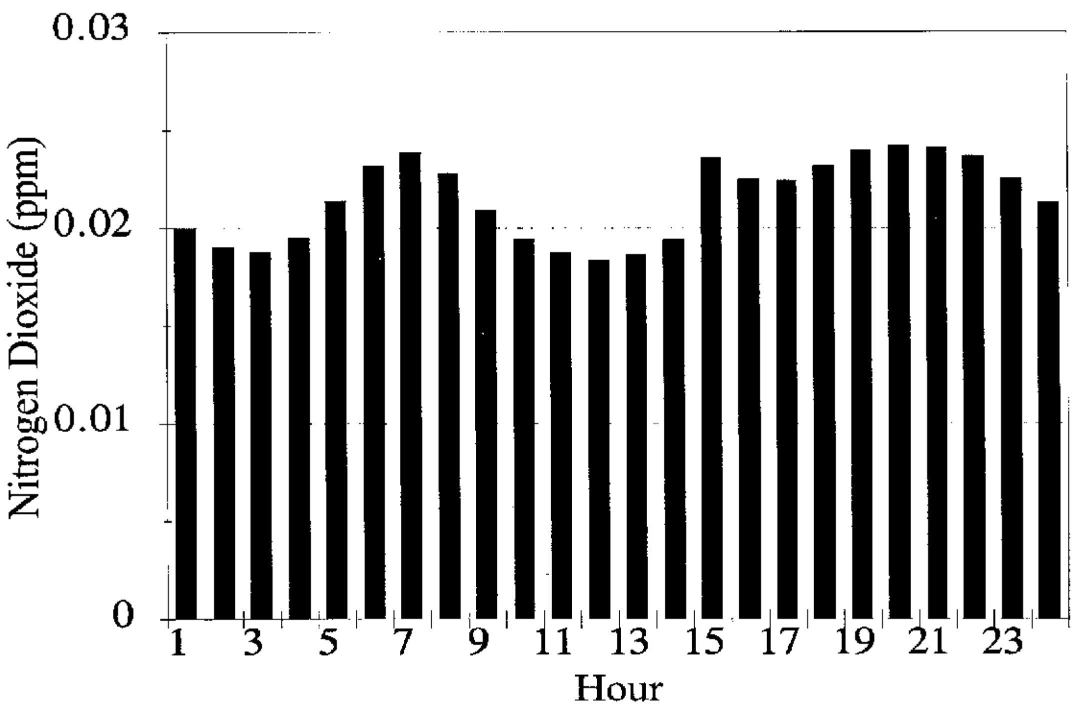

Figure 3. —Average hourly concentrations of NO_2 calculated from eight IEPA monitoring sites in study area in 1991.

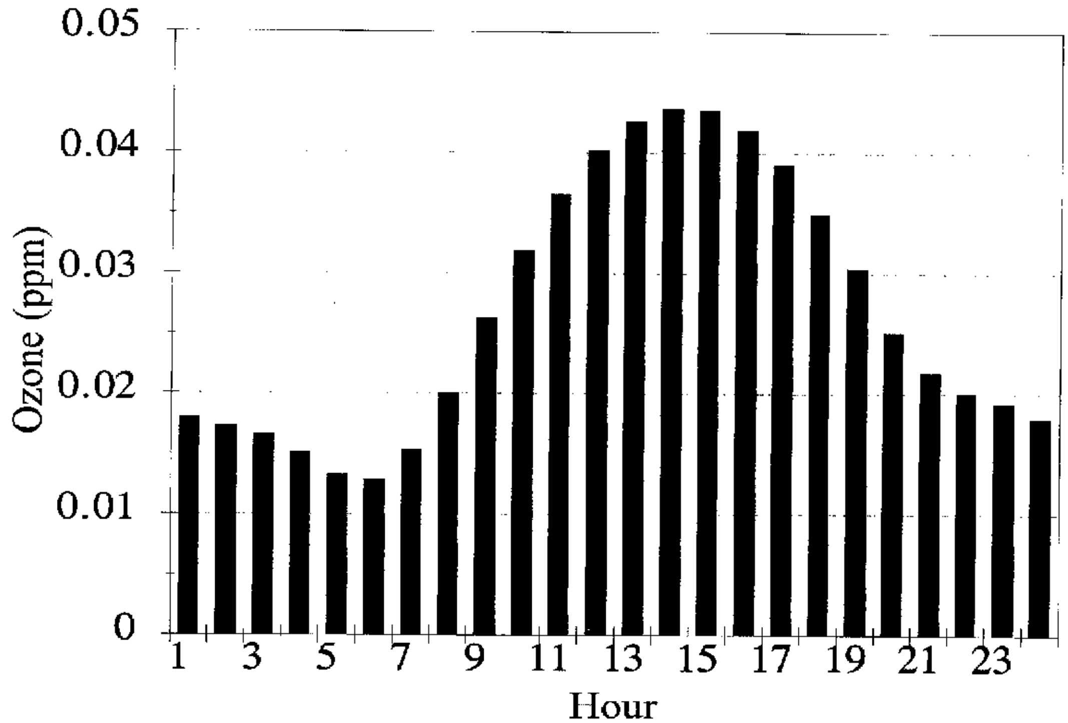

Figure 4. —Average hourly concentrations of O_3 calculated from 13 IEPA monitoring sites in study area during in-leaf season (May-October) of 1991.

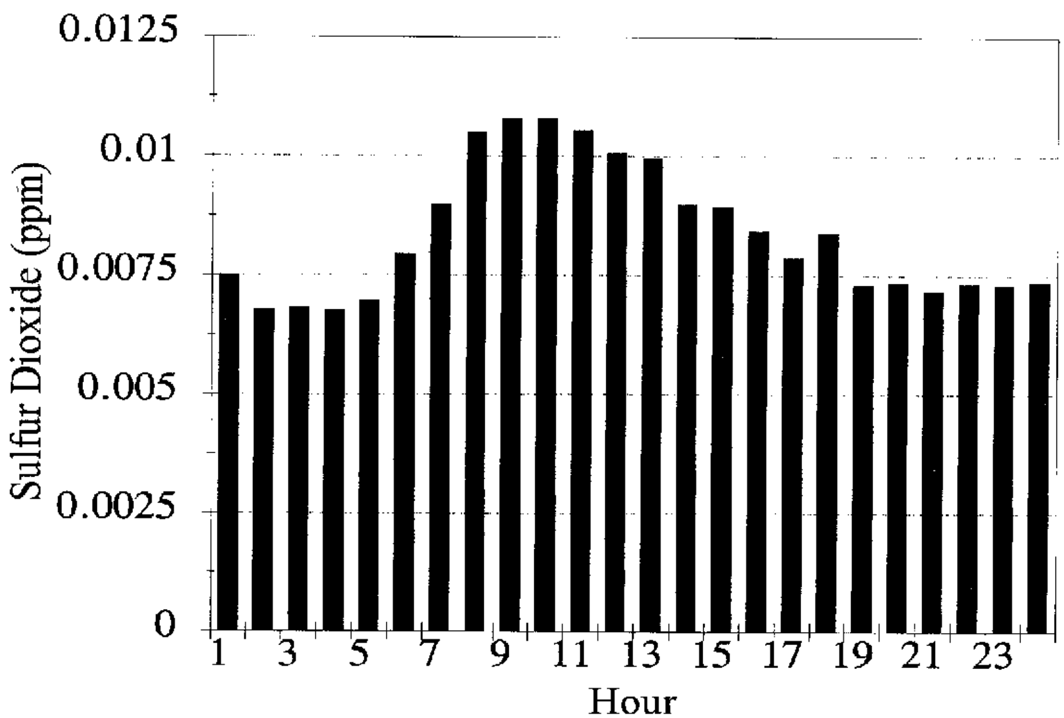

Figure 5. —Average hourly concentrations of SO_2 calculated from 10 IEPA monitoring sites in study area in 1991.

estimate total pollutant flux to the surface (e.g., g/hr). These hourly fluxes can be summed to estimate total daily, monthly, or yearly fluxes.

Deposition Velocities

The rate at which pollutants are transferred onto or into various surfaces is influenced by a series of resistances to pollutant transfer. 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 ($V_d = 1/(R_a + R_b + R_c)$). The aerodynamic resistance is associated with atmospheric turbulence, the quasi-laminar boundary-layer resistance is influenced by the diffusivity of the material being transferred, and the net canopy resistance is dominated by surface factors (Baldocchi et al. 1987). As the rate of turbulent mixing becomes high, pollutant transport to the surface is rapid as the resistance to transport through the boundary layer approaches zero and the resistance to deposition is limited by the surface resistance (Killus et al. 1984).

Aerodynamic and Quasi-laminar Boundary-Layer Resistances

Meteorological data from Chicago's O'Hare airport (3-hr averages) were used in estimating R_a and R_b. The aerodynamic and quasi-laminar boundary-layer resistances were estimated for the Chicago area with a method similar to that used in the Urban Airshed Model (Killus et al. 1984).

$$R_a = u(z)/u_*^2$$

where u(z) is the wind speed at height z (m/sec) and u_* is the frictional velocity (m/sec).

$$u_* = (k\, u(z-d))/[\ln((z-d)/z_o) - \psi_m((z-d)/L) + \psi_m(z_o/L)]$$

where k = von Karman's constant (0.40), d = displacement length (m), z_o = roughness length (m), ψ_m = stability function for momentum, and L = Monin-Obuhkov stability length (van Ulden and Holtslag 1985). L was estimated by classifying hourly local meteorological data into stability classes using Pasquill's (1961) stability classification scheme and then estimating 1/L as a function of Pasquill classes and z_o (Golder 1970). When L<0 (unstable):

$$\psi_m = 2\ln[(1+X)/2] + \ln[(1+X^2)/2] - 2\tan^{-1}(X) + \pi/2$$

(van Ulden and Holtslag 1985)

where $X = (1 - 28\, z/L)^{0.25}$ (Dyer and Bradley 1982). When L>0 (stable conditions):

$$\psi_m = -17\,(1 - \exp(-0.29(z-d)/L)$$

(van Ulden and Holtslag 1985).

The quasi-laminar boundary-layer resistance was estimated as:

$$R_b = B^{-1}u_*^{-1}$$

where $B^{-1} = 2.2u_*^{-1/3}$ (Killus et al. 1984).

R_a and R_b were calculated for every three hours throughout 1991 based on Chicago meteorological data. Each estimate of R_a and R_b was used to represent the corresponding 3-hr period of the day. These hourly values were combined to yield the average daily conditions for each month in 1991.

Canopy Resistance

The tree canopy resistances for each of the pollutants was estimated by averaging the R_c values derived from literature on individual trees and forests. R_c estimates were categorized by in-leaf season daytime, in-leaf season nighttime, and out-of-leaf season using a distribution of 90 percent deciduous and 10 percent coniferous leaf surface area (Nowak 1994: Chapter 2, this report) (Table 2). R_c estimates for particles and CO could not be found in the literature, so average deposition velocity minus average R_a and R_b for Chicago was substituted as the R_c for these pollutants. Fifty percent of the particles being deposited to trees were assumed to be resuspended from the trees to the atmosphere. Particle collection by deciduous trees in winter assumed a surface-area index for bark of 1.7 (m^2 of bark/m^2 of ground surface covered by tree crown) (Whittaker and Woodwell 1967). In-leaf daylight ranged from 11 hr/day in October to 15 hr/day in June. The in-leaf season for deciduous trees in the Chicago area was modeled as May 1 to October 31 based on local observation of foliation periods.

Hourly canopy resistances of trees were calculated for each hour in 1991 based on in-leaf vs. out-of-leaf season and day

Table 2.—Average canopy-resistance values (sec/cm) for trees in the Chicago area (90 percent deciduous; 10 percent coniferous leaf-surface area); values are estimates derived from the literature

Pollutant	In-leaf daytime	In-leaf nighttime	Out-of-leaf season
Carbon monoxide	500	500	10,000
Nitrogen dioxide	3.01	7.54	88.3
Ozone	1.74	17.2	---[a]
Particulate matter	0.78	0.78	2.39
Sulfur dioxide	1.87	9.54	58.2

[a] no pollutant concentrations collected during out-of-leaf season (November-April).

Sources: Bidwell and Fraser 1972; Roberts 1974; Fritschen and Edmonds 1976; Garland 1977; Garland and Branson 1977; Little 1977; McMahon and Denison 1979; Rogers et al. 1979; Sheih et al. 1979; Wesely and Hicks 1979; Galbally and Roy 1980; Sehmel 1980; Lindberg and Harriss 1981; Hicks et al. 1982; Hofken and Gravenhorst 1982; Granat and Johansson 1983; Gravenhorst et al. 1983; Greenhut 1983; Hofken et al. 1983; Lindberg and Lovett 1983; Wesely 1983; Wesely et al. 1983; Lindberg et al. 1984; Lovett and Lindberg 1984; Fowler 1985; Lorenz and Murphy 1985; Wesely et al. 1985; Voldner et al. 1986; Walcek et al. 1986; Dasch 1987; Dasch 1989; Shanley 1989; Wesely 1989; Davidson and Wu 1990; Murphy and Sigmon 1990.

vs. night. Tree-canopy resistance was combined with R_a and R_b to produce hourly estimates of deposition velocities to trees in the Chicago area. To limit deposition estimates to periods predominated by dry deposition, deposition velocities were set to zero during and immediately following periods of precipitation (1 hr).

Pollution Concentration

Hourly pollution concentrations (ppm) were obtained from the Illinois Environmental Protection Agency (IEPA) for CO (7 monitoring sites in study area), NO_2 (8 sites), O_3 (13 sites) and SO_2 (10 sites). Average daily concentrations of PM10 ($\mu g/m^3$) also were obtained from the IEPA (14 sites). No concentration data for O_3 were obtained for the out-of-leaf season (November-April).

Each of the 117 community areas were assigned the average hourly concentrations for each month from the closest monitoring station for each pollutant. The average hourly pollutant flux for each month of 1991 was calculated for each pollutant in each community area using equation (1). Hourly pollutant flux (g/m^2 of tree canopy coverage) for each community area was multiplied by the amount of tree canopy cover (m^2) in the community area to estimate total pollutant flux per hour for the average day in each month. These values were combined to yield estimates of daily, monthly, and yearly pollution flux to trees (for each pollutant) for Chicago, suburban Cook County, DuPage County, and the entire study area.[1]

Total pollutant flux also was calculated for the individual days that had the highest hourly reading of the year: CO (August 2), NO_2 (June 21), O_3 (June 18-21), SO_2 (October 16-17) and PM10 (July 17). Because of a lack of variance information on some of the averages used in the calculations, no error bounds could be computed for the removal estimates.

Boundary-Layer Height

The boundary layer is the atmospheric layer characterized by well-developed mixing (turbulence). The height of the boundary layer is not constant over time. By day, thermal mixing enables the boundary-layer height to extend to about 1 to 2 km. At night, mixing tends to be suppressed and the boundary-layer height can shrink to less than 100 m (Oke 1987). The height of the boundary layer is important because the deeper the boundary layer, the less the relative effect of trees on reducing overall concentrations of air pollutants given a well-mixed boundary layer.

To approximate boundary-layer heights in the study area, average mixing heights from the closest station to the study area (Peoria, IL) were used. Readings of average daily morning and afternoon mixing heights were extrapolated throughout the day to estimate the diurnal cycle of the boundary-layer height for each month (e.g., Holzworth 1972). The mixing heights used ranged from a low of 300 m in early morning (6 a.m.) to a high of 1,600 m for midafternoon (4 p.m.) in June. Average hourly mixing heights for each month were used in conjunction with data on pollution concentrations for each community area to calculate the amount of pollution within the mixing layer. This extrapolation from ground-layer concentration to total pollution within the boundary layer assumes a well-mixed boundary layer. The amount of pollution in the air was contrasted with the amount of pollution removed by trees to calculate the relative effect of trees in reducing local pollution concentrations:

$$E = R / R+A$$

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

Effect of Individual Trees

The ability of individual trees to remove pollutants was estimated for each diameter class using the formula:

$$I_x = R_t \times (LA_x/LA_t) / N_x$$

where I_x = pollution removal by individual trees in diameter class *x* (kg/tree); R_t = total pollution removed for all diameter classes (kg); LA_x = total leaf area in diameter class *x* (m^2); LA_t = total leaf area of all diameter classes (m^2); and N_x = number of trees in diameter class *x*. This formula yields an estimate of pollution removal by individual trees based on leaf-surface area (the major surface for pollutant removal) and a distribution of approximately 90 percent deciduous and 10 percent coniferous leaf-surface area (Nowak 1994: Chapter 2, this report).

Estimated Monetary Value of Pollution Removal

To estimate the monetary value of pollution removal by trees, current costs for emission control were used. The cost (dollars/metric ton) of preventing the emission of a similar amount of pollutant using these control strategies was multiplied by the metric tons of pollutant removed by trees to yield an indication of the pollution removal value of trees.[2] Dollar values (1990) per metric ton of pollutant removed were \$540/t (\$490/ton) for O_3, \$1,014/t (\$920/ton) for CO, \$1,441/t (\$1,307/ton) for PM10, \$1,801/t (\$1,634/ton) for SO_2 and \$4,863/t (\$4,412/ton) for NO_2 (California Energy Commission 1992).

Potential Future Effects of Tree Planting

To analyze the potential effects of future tree planting, available growing space (i.e., grass and soil area) was analyzed by land-use type throughout the study area. The future scenario assumed that none of the available space in agricultural or transportation (predominantly airport) would be planted with trees due to land-use limitations. Five percent of available

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1

$$F = \sum_{p=1}^{5} \sum_{m=1}^{12} \sum_{h=1}^{24} \sum_{ca=1}^{117} ((1/R_a+R_b+R_c) \times C)$$

where F = total annual pollution removal for five pollutants; p = pollutant species; m = month; h = hour; ca = community area (i.e., specific tree-cover data); R_a and R_b = aerodynamic and quasi-laminar boundary-layer resistances, respectively (calculated from local meteorological data for 3-hr periods); R_c = canopy resistance (varies by day, night, precipitation, and season); and C = average hourly pollutant concentration for each month (PM10 concentrations based on daily average).

2 The estimation of value is approximate as emission control strategies prevent the emission of pollution while trees remove pollution that already is in the atmosphere.

space was assumed to be planted and covered with trees on large commercial-industrial areas and institutional land dominated by vegetation (e.g., parks, forest preserves, cemeteries, golf courses). Ten percent of available space was assumed to be planted and covered with trees on institutional lands dominated by building (e.g., schools); 15 percent in residential areas, 20 percent in landscaped commercial complexes, and 25 percent on vacant lands and freeways.

Removal of pollutants by the additional trees was calculated based on average removal per acre of existing tree cover times the number of new acres of tree cover that result from the new plantings. This removal was subtracted from the amount of pollution in the atmosphere to calculate a new atmospheric concentration. Because the atmospheric concentration would be lower due to the additional trees, overall uptake per acre of trees also drops due to the lower concentrations. The new pollutant flux for all trees (original plus new trees) with a lower pollutant concentration was contrasted with the original flux rate to calculate the effect of the new tree plantings.

Results

In 1991, total estimated pollutant removal by trees in the study area was 5,575 t (6,145 tons) with PM10 and O_3 removed the most by trees (Table 3). Monthly removal rates varied, peaking in May for CO (41 t, 45 tons), in June for O_3 (498 t, 549 tons), in July for PM10 (348 t, 383 tons) and in August for NO_2 (152 t, 168 tons) and SO_2 (132 t, 145 tons). Minimum removal in the study area occurred in March for PM10 (30 t, 33 tons), in April for CO (1.6 t, 1.8 tons), in October for O_3 (117 t, 129 tons) (in-leaf season data only), in November for NO_2 (4.9 t, 5.4 tons) and in December for SO_2 (4.0 t, 4.4 tons) (Figure 6, Table 4). Monthly patterns of removal were similar in Chicago, suburban Cook, and DuPage Counties (Figures 7-9, Table 4).

Removal occurred mostly during the in-leaf season with daily in-leaf removal rates ranging from 1,155 kg/day (2,545 lb/day) for CO to 10,819 kg/day (23,850 lb/day) for O_3 (Table 5). Total removal per hectare of tree cover ranged from 3.4 kg/yr (3.1 lb/acre/yr) for CO to 30.7 kg/yr (27.4 lb/acre/yr) for O_3 (Table 5). Total removal per hectare of trees was 85.7 kg/yr (76.5 lb/acre/yr) for all five pollutants.

Maximum daily effects of pollution removal by trees in the study area was approximately 1.4 t (1.5 tons; 0.02 kg/ha of tree cover/day) for CO; 4.9 t (5.4 tons; 0.08 kg/ha of trees/day) for NO_2; 10.7 t (11.8 tons; 0.16 kg/ha of trees/day) for SO_2; 21.6 t (23.8 tons; 0.33 kg/ha of trees/day) for PM10; and 24.4 t (26.9 tons; 0.38 kg/ha of trees/day) for O_3. Peak-day effects (based on the day with highest hourly concentration) were lower than average-day effects for CO and NO_2 due to relatively low concentrations during nonpeak hours. Peak daily effects for these pollutants were based on peak average-day effects for a month (CO: September; NO_2: August).

The maximum hourly reduction in pollutant concentrations due to trees across the study area ranged from 0.007 percent for CO to 1.3 percent for SO_2 (Table 6). Average hourly reduction in concentrations during the in-leaf season ranged from 0.002 percent for CO to 0.4 percent for PM10. In large areas of 100-percent tree cover, reductions in concentrations due to trees likely reached 7 percent for sulfur dioxide (Table 6).

Under typical in-leaf daytime conditions in 1991, a hectare of urban tree cover would be expected to remove 0.0008 kg/hr (0.0007 lb/acre/hr) of CO, 0.0041 kg/hr (0.0037 lb/acre/hr) of SO_2, 0.0045 kg/hr (0.004 lb/acre/hr) of NO_2, 0.0056 kg/hr (0.005 lb/acre/hr) of PM10, and 0.0123 kg/hr (0.011 lb/acre/hr) of O_3. For concentrations at the NAAQS level, a hectare of tree cover would be expected to remove 0.007 kg/hr (0.006 lb/acre/hr) of CO (at 8-hr NAAQS); 0.067 kg/hr (0.06 lb/acre/hr) of SO_2 (at 24-hr NAAQS); 0.012 kg/hr (0.01 lb/acre/hr) of NO_2 (at annual NAAQS); 0.031 kg/hr (0.028 lb/acre/hr) of PM10 (at 24-hr NAAQS); and 0.046 kg/hr (0.041 lb/acre/hr) of O_3 (at 1-hr NAAQS). These removal rates should be considered high and of relatively short term.

Large individual trees have the greatest estimated pollution removal due to their relatively large leaf surface area. Trees larger than 76 cm (30 inches) in diameter at breast height (d.b.h. at 1.37 m or 4.5 ft) removed an estimated 1.4 kg (3.1 lb) of pollution in 1991; trees less than 8 cm (3 inches) in d.b.h. removed approximately 0.02 kg (0.05 lb) (Table 7).

The monetary value of pollution removal in 1991 was approximately \$1 million in Chicago (\$151/ha of tree cover/yr; \$61/acre of tree cover/yr); \$5.8 million in suburban Cook County (\$137/ha of trees/yr; \$55/acre of trees/yr); \$2.4 mil-

Table 3.—Total pollutant removal (t/yr) and removal per hectare of land (kg/ha/yr) in Chicago, suburban Cook County, DuPage County, and study area (multiply t by 1.102 to convert to tons; divide kg/ha by 1.12 to convert to lb/acre)

	Chicago		Cook County		DuPage County		Study area	
Pollutant	Total	per ha	Total	per ha	Total	per ha	Total	per ha
CO	15	0.3	147	0.8	61	0.7	223	0.7
SO_2	84	1.4	520	2.8	102	1.2	706	2.1
NO_2	89	1.5	470	2.5	248	2.9	806	2.4
PM10	212	3.5	1,179	6.3	449	5.2	1,840	5.5
O_3	191	3.1	1,328	7.1	481	5.6	2,000	6.0
Total	591	9.7	3,644	19.4	1,340	15.5	5,575	16.7

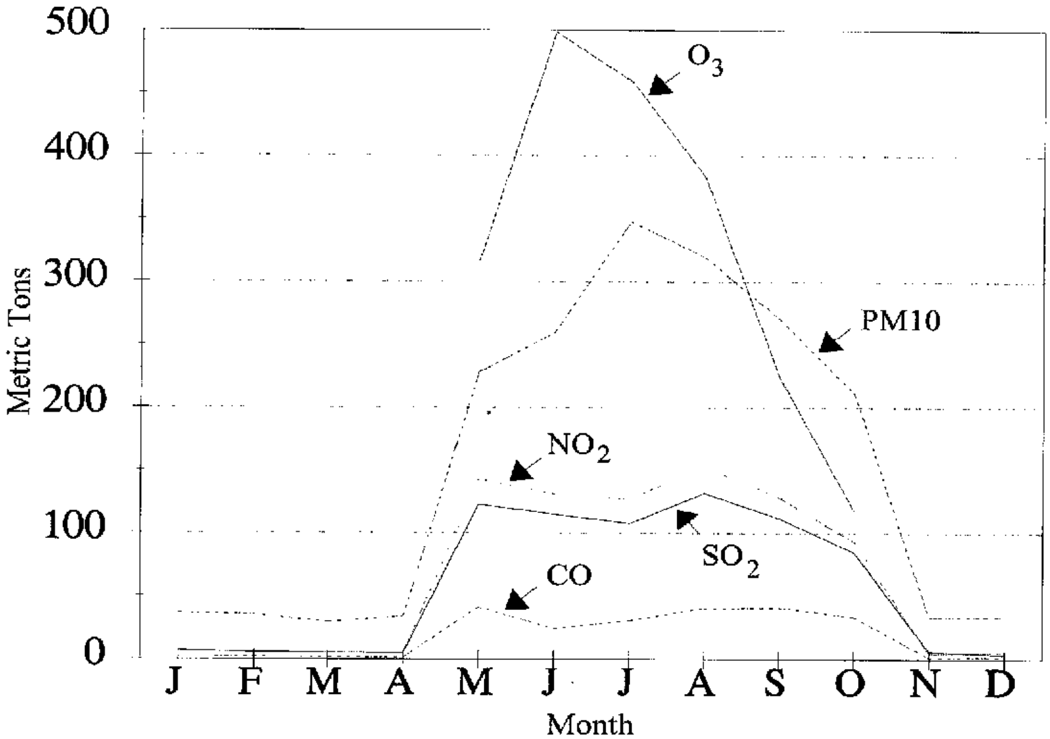

Figure 6. —Monthly estimates of pollution removal by trees in study area in 1991. Ozone removal estimates are for May-October only. Particulate removal assumes 50 percent resuspension back to the atmosphere.

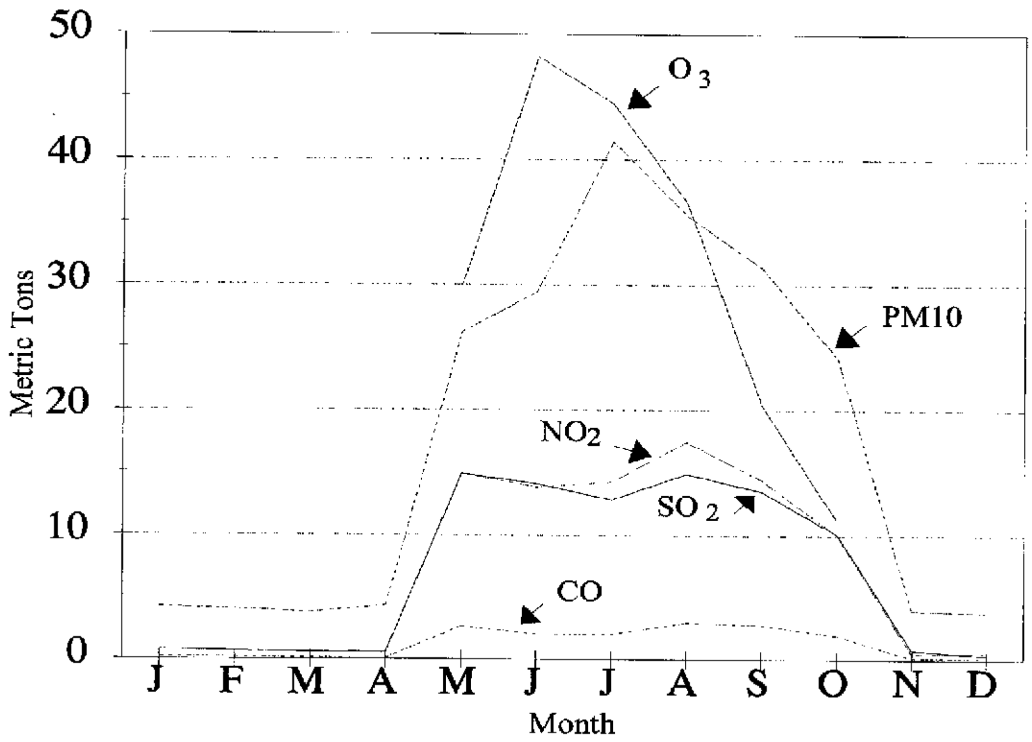

Figure 7. —Monthly estimates of pollution removal by trees in Chicago in 1991. Ozone removal estimates are for May-October only. Particulate removal assumes 50 percent resuspension back to the atmosphere.

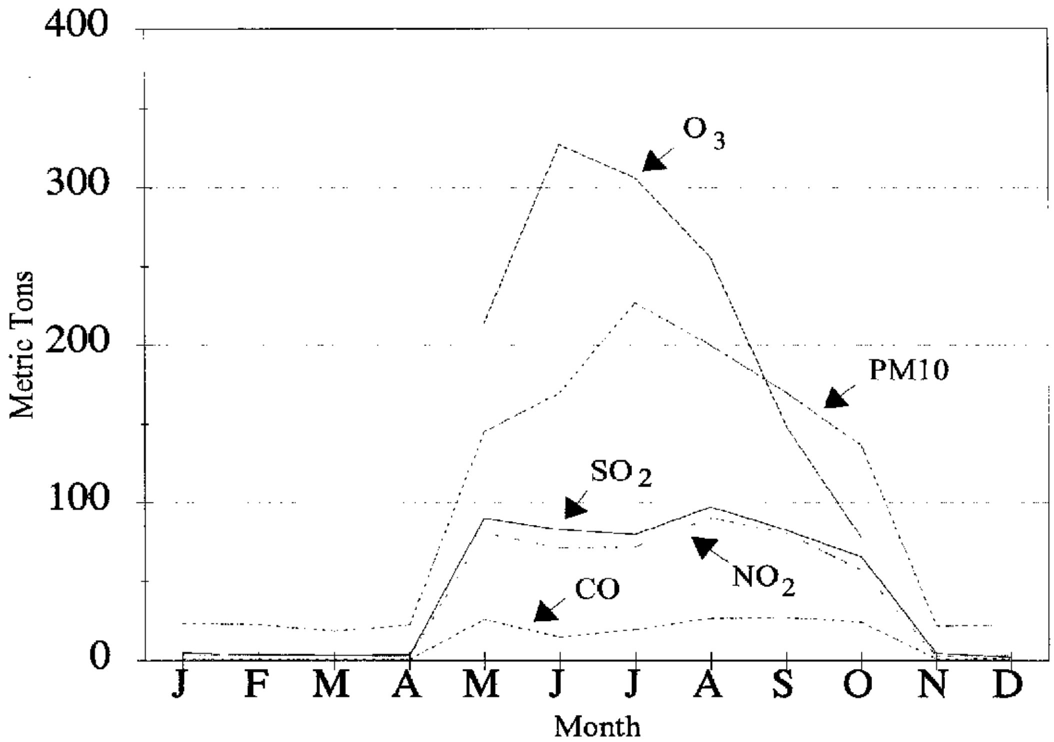

Figure 8.—Monthly estimates of pollution removal by trees in suburban Cook County in 1991. Ozone removal estimates are for May-October only. Particulate removal assumes 50 percent resuspension back to the atmosphere.

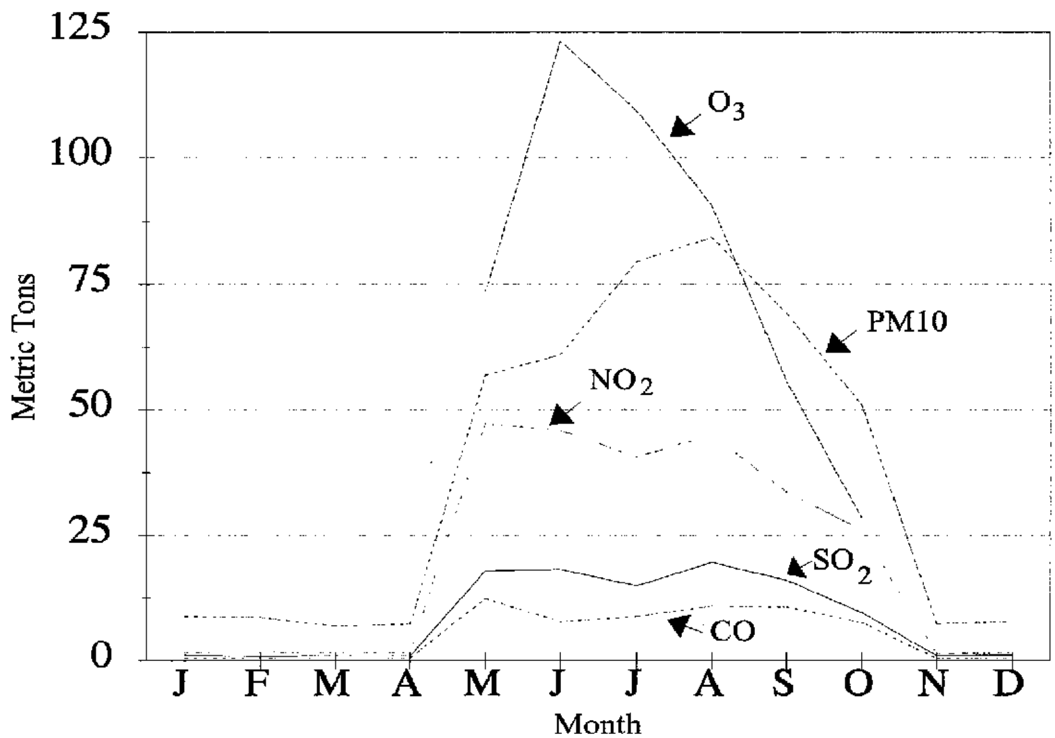

Figure 9.—Monthly estimates of pollution removal by trees in DuPage County in 1991. Ozone removal estimates are for May-October only. Particulate removal assumes 50 percent resuspension back into the atmosphere.

Table 4.—Total monthly removal rates (t/month) for pollutants by study area sector in 1991 (multiply t by 1.102 to convert to tons)

Month	CO	SO_2	NO_2	PM10	O_3
CHICAGO					
January	0.2	0.8	0.7	4.2	na
February	0.2	0.7	0.6	4.0	na
March	0.1	0.6	0.6	3.7	na
April	0.1	0.6	0.5	4.4	na
May	2.7	14.9	14.9	26.2	30.0
June	2.1	14.1	13.8	29.5	48.2
July	2.1	12.8	14.3	41.5	44.5
August	3.0	14.9	17.5	35.6	36.6
September	2.8	13.5	14.4	31.3	20.4
October	2.0	10.1	10.0	24.2	11.1
November	0.2	0.8	0.5	4.0	na
December	0.1	0.4	0.6	3.8	na
SUBURBAN COOK COUNTY					
January	1.1	5.0	3.8	23.7	na
February	1.2	4.0	3.4	22.7	na
March	1.2	3.6	3.3	18.9	na
April	1.0	3.6	3.0	22.6	na
May	26.1	89.7	80.3	144.7	213.2
June	15.0	82.3	71.0	169.3	327.0
July	19.8	79.7	71.8	226.7	305.9
August	26.9	97.1	90.0	199.7	255.6
September	27.4	82.3	80.2	170.0	148.7
October	24.3	65.3	57.1	136.3	77.8
November	1.4	4.8	3.0	22.1	na
December	1.5	2.6	3.4	22.4	na
DUPAGE COUNTY					
January	0.5	1.0	1.8	8.9	na
February	0.4	0.8	1.7	8.7	na
March	0.5	0.8	1.6	6.9	na
April	0.4	0.8	1.5	7.3	na
May	12.4	17.9	47.1	56.8	73.5
June	7.8	18.2	45.9	60.9	123.2
July	8.9	14.9	40.5	79.5	109.2
August	11.0	19.6	45.0	84.4	90.7
September	10.8	16.1	33.6	69.2	55.6
October	7.7	9.6	26.0	50.9	28.4
November	0.4	0.9	1.3	7.3	na
December	0.5	1.0	1.5	7.9	na
STUDY AREA					
January	1.7	6.8	6.3	36.7	na
February	1.8	5.4	5.7	35.4	na
March	1.8	5.0	5.5	29.5	na
April	1.6	5.1	5.1	34.2	na
May	41.2	122.5	142.3	227.7	316.7
June	24.9	114.7	130.7	259.7	498.4
July	30.7	107.5	126.6	347.7	459.6
August	40.8	131.6	152.5	319.6	382.8
September	41.0	111.9	128.2	270.5	224.7
October	33.9	85.0	93.2	211.3	117.2
November	1.9	6.5	4.9	33.4	na
December	2.1	4.0	5.5	34.2	na

na - not analyzed.

Table 5.—Average daily pollutant removal during in-leaf and out-of-leaf seasons (kg/day); total yearly removal per hectare of tree canopy cover (kg/ha/yr); and average daily pollutant removal during in-leaf and out-of-leaf seasons per hectare of tree canopy cover (kg/ha/day) in Chicago, suburban Cook County, DuPage County and entire study area (multiply kg by 2.204 to convert to pounds; divide kg/ha by 1.12 to convert to lb/acre)

Sector	Average daily removal		Removal per hectare of tree cover		
	In-leaf [a]	Out-of-leaf[b]	Total year	In-leaf [a]	Out-of-leaf[b]
CO					
Chicago	79	5	2.3	0.012	0.0007
Cook County	757	40	3.5	0.018	0.0009
DuPage County	318	15	3.8	0.020	0.0009
Study Area	1,155	60	3.4	0.018	0.0009
SO_2					
Chicago	437	21	12.6	0.065	0.0031
Cook County	2,697	131	12.3	0.064	0.0031
DuPage County	524	30	6.3	0.033	0.0019
Study Area	3,657	182	10.9	0.056	0.0028
NO_2					
Chicago	462	20	13.3	0.069	0.0030
Cook County	2,448	110	11.1	0.058	0.0026
DuPage County	1,294	52	15.4	0.081	0.0032
Study Area	4,205	182	12.4	0.065	0.0028
PM10					
Chicago	1,023	134	31.8	0.153	0.0201
Cook County	5,688	733	27.9	0.134	0.0173
DuPage County	2,183	260	27.9	0.136	0.0162
Study Area	8,894	1,127	28.3	0.137	0.0173
O_3					
Chicago	1,032	na	28.6	0.155	na
Cook County	7,185	na	31.4	0.170	na
DuPage County	2,602	na	29.9	0.162	na
Study Area	10,819	na	30.7	0.166	na

[a]May - October; kg/day
[b]November - April; kg/day

Table 6.—Estimated maximum and average in-leaf reduction in hourly pollution concentration (in percent) by trees in the Chicago area in 1991

Pollutant	Study area		100-percent forested area	
	Maximum	Average	Maximum	Average
CO	0.007	0.002	0.03	0.01
NO_2	0.8	0.2	4.2	1.1
SO_2	1.3	0.3	6.7	1.6
PM10[a]	0.5	0.4	2.5	2.1
O_3	1.0	0.3	5.2	1.6

[a] daily percent reduction

lion in DuPage County ($147/ha of trees/yr; $59/acre of trees/yr); and $9.2 million in the study area ($141/ha of trees/yr; $57/acre of trees/yr) (Table 8). The highest value was for NO_2 removal (43 percent of total monetary value), followed by PM10 (29 percent), SO_2 (14 percent), O_3 (12 percent) and CO (2 percent). Monetary values for individual trees in the study area ranged from $0.04/tree/yr for small trees to $2.31/tree/yr for large trees (Table 7) .

The proposed tree-planting scenario that would fill available grass and soil space on various land uses from 0 to 25 percent with trees would increase overall tree cover in the study area by 4.1 percent (from 19.4 to 23.5 percent tree cover). This additional cover likely would have removed an additional 1,180 t (1,300 tons) of pollution in 1991 (CO: 45 t, 50 tons; SO_2: 150 t, 165 tons; NO_2: 170 t, 185 tons; PM10: 390 t, 430 tons; O_3: 425 t, 470 tons) and reduced pollution concentrations by another 0.05 percent.

Discussion

The removal estimates in this paper are approximations based on computations that incorporate measured local urban tree canopy surface, local pollution concentrations, and local meteorology in diurnal and annual patterns. Average in-leaf pollution removal per hectare of tree cover per day for 1991 in the Chicago area was significantly less than estimated by DeSanto et al. (1976a) for all pollutants (from 11 to 32 times less for particles to 400 to 1,300 times less for SO_2). The estimates of DeSanto et al. are higher than those for the Chicago area because of high pollution concentrations in some of the studies used to determine removal rates and because diurnal leaf stomatal functions were disregarded. In-leaf daily removal of SO_2 per hectare of tree cover in the Chicago area was about half of that estimated by Murphy et al. (1977) and Lorenz and Murphy (1985) for equal pollutant concentration.

Results for the Chicago area improve on earlier estimates of pollution removal for urban trees. However, there remain many limitations to the Chicago results that have unknown bounds on the error of estimation. Thus, the results should be considered first-order approximations of pollution removal by urban trees. Additional research is needed to better determine various aspects of the calculations, and to test results under urban field conditions.

Factors Influencing Pollution Removal Estimates

Because tree-canopy resistances generally decrease from morning to midday and then increase until night (Grimmond and Oke 1991), the use of average in-leaf daytime R_c values likely overestimates pollution removal during the early morning and late evening, and underestimates removal during midday. Unfortunately, it is not known where the average R_c value from the literature falls within the diurnal resistance cycle. Research is needed to evaluate the diurnal cycle of tree canopy resistances to pollution deposition in urban areas.

The overall removal rate for trees is greater than reported in this study as results were limited to dry deposition. In periods after rain or during periods when dew collects on vegetation removal rates for urban trees increase as trees offer a large wet surface area upon which water-soluble pollutants can readily dissolve (e.g., SO_2, NO_2).

Estimates of particle removal also may be conservative as the model assumed 50 percent resuspension of deposited pollutants. This rate was estimated as a midvalue based on limited literature. Zinke (1967) estimated that retention of airborne materials ranged from 17 to 57 percent in pine stands and 82 to 86 percent in hardwood stands. For the Chicago area's urban forest, which is approximately 90 percent hardwoods, a resuspension rate of 20 percent would be reasonable given Zinke's estimates. However, due to the more open nature of urban forests relative to more natural forest stands, higher resuspension would be expected due to the increased probability of wind resuspension in tree canopies. Research is needed on the resuspension of particles in urban areas.

Average canopy-resistance values obtained from the literature probably are too high (leading to conservative deposition velocities) for SO_2 (average in-leaf daytime R_c = 1.9 sec/cm) and O_3 (average in-leaf daytime R_c = 1.7 sec/cm). Daytime tree-canopy resistances could be as low as 0.5 sec/cm for SO_2 and 0.4 sec/cm for O_3.[3] Average daytime in-leaf deposition velocities for forests and trees in the literature typically range from 0.2 to 2 cm/sec and average around 1.0 cm/sec for SO_2 (e.g., Garland 1977; McMahon and Denison 1979; Fowler and Cape 1983; Lovett and Lindberg 1984; Fowler 1985; Lorenz and Murphy 1985; Murphy and Sigmon 1990). Daytime deposition velocities for O_3 in the literature normally range from 0.3 to 1 cm/sec and average around 0.7 cm/sec (e.g., Greenhut 1983; Colbeck and Harrison 1985; Davidson and Wu 1990).

The deposition velocities used in this study were lower than averages in the literature (study SO_2 average in-leaf daytime V_d = 0.52 cm/sec; O_3 average in-leaf daytime V_d = 0.55 cm/sec) and are thought to be conservative (Wesely 1993, pers. commun.). Through the use of average R_c values, deposition velocities and pollution removal may be underestimated by a factor of 1.9 for SO_2 and a factor of 1.3 for O_3. Research is needed on improving R_c and V_d estimates for urban vegetation and other urban surfaces. The average deposition velocity of NO_2 was within the range of velocities in the literature.

The location of pollution monitors in the city can lead to an overestimation of pollution removal by urban trees. These monitors tend to be located in areas that are expected to have relatively high concentrations of pollution. Thus, extrapolations of these concentrations to larger areas may result in inflated concentration estimates. Detailed variations

[3] Based on minimum stomatal and mesophyll resistance of $r_s D_{H_2O}/D_x + r_{mx}$ where r_s is minimum stomatal resistance, D_{H_2O} is the molecular diffusivity of water vapor, D_x is the molecular diffusivity of gas x in air, and r_{mx} is mesophyll resistance of gas x (Wesely 1989). Minimum stomatal resistance was assumed to be 1.5 sec/cm (Baldocchi 1988). Leaf area index of urban forests was estimated to be 6 (see Nowak 1994: Chapter 2, this report).

Table 7.—Estimated removal rate per tree by d.b.h. class (kg/yr) and total annual dollar value per tree for removal of pollutants (see Table 8); particulate removal assumes 50 percent resuspension back to the atmosphere (multiply kg by 2.204 to convert to pounds)

D.b.h. class	CO	SO_2	NO_2	PM10	O_3[a]	Total	Dollars
0-7 cm	0.001	0.003	0.003	0.007	0.008	0.021	0.04
8-15 cm	0.003	0.008	0.009	0.021	0.023	0.064	0.10
16-30 cm	0.007	0.021	0.024	0.055	0.060	0.166	0.27
31-46 cm	0.017	0.054	0.062	0.141	0.153	0.428	0.70
47-61 cm	0.033	0.104	0.118	0.270	0.294	0.819	1.34
62-76 cm	0.043	0.136	0.155	0.355	0.385	1.074	1.76
77+ cm	0.056	0.178	0.204	0.465	0.505	1.409	2.31

[a] May-October only.

Table 8.—Total yearly monetary value (thousands of dollars) of pollutant removal and average daily monetary value (dollars) during in-leaf season for Chicago, suburban Cook County, DuPage County, and entire study area; estimated tons of pollutant removed by trees was multiplied by 1990 cost of preventing emission of similar amount of pollutant using current emission control strategies ($/t): CO = 1,014; SO_2 = 1,801; NO_2 = 4,863; PM10 = 1,441; O_3 = 540 (California Energy Commission 1992)

	Chicago		Cook County		DuPage County		Study area	
Pollutant	Total	Day	Total	Day	Total	Day	Total	Day
CO	16	80	149	770	62	320	227	1,170
SO_2	152	790	937	4,860	183	940	1,272	6,590
NO_2	431	2,250	2,287	11,910	1,204	6,290	3,922	20,450
PM10	306	1,470	1,699	8,190	646	3,140	2,651	12,800
O_3	103	560	717	3,880	260	1,410	1,080	5,850
Total	1,008	5,150	5,789	29,610	2,355	12,100	9,152	46,860

in pollution concentrations across a city need to be investigated more fully to better understand the limitations of extrapolating concentrations from limited monitoring points.

Boundary Layer

Current estimates of percent reduction in pollution concentrations in the Chicago area likely are conservative due to the effect of the breeze off Lake Michigan and the assumption of a well-mixed boundary layer. The lake breeze reduces mixing depths (Lyons and Olsson 1973), thus, increasing the relative effect of trees in reducing air pollution. The assumption of a well-mixed unstable atmosphere presumed little variation in pollution concentration with height (e.g., Colbeck and Harrison 1985). However, there are times, particularly at night, when there is limited mixing (van Dop et al. 1977; Colbeck and Harrison 1985). During these times of limited mixing, the effect of trees and other surfaces in removing pollutants is concentrated in the lower boundary layer, so trees have a greater relative effect on pollution reduction near the ground. This effect is of particular importance as this is the layer in which humans reside.

The depth of the boundary layer has an immense effect on the percent reduction in pollution concentration. Maximum tree effects occurred in early morning when stomates were assumed open and transpiring and the boundary-layer height still was relatively low. Research is needed on variations in stomatal resistances and boundary-layer heights in the Chicago region to improve the estimates of reductions in pollution concentration by Chicago's trees.

Emission Effects

Another factor that is not considered in estimates of pollution removal is that trees emit compounds that can increase local concentrations of pollution. These emissions offset some of the removal effects of trees. The relatively low removal of CO by trees likely is offset by their emission of volatile organic compounds, which can increase CO concentrations. It is possible that urban trees may be an overall source of CO; this sink/source relationship in urban areas needs further study. If trees are a source of CO, the source amount probably would be insignificant relative to automobile emissions.

Emissions of volatile organic compounds by trees can contribute to the formation of O_3 (Brasseur and Chatfield 1991). However, because these emissions are temperature dependent and trees generally lower air temperatures, it is believed that increased tree cover would lower overall volatile organic emissions and O_3 levels in urban areas (Cardelino and Chameides 1990).

Pollen emissions by trees can contribute significantly to local concentrations of total particles. However, tree pollen often is greater than 10 μm (Smith 1990) and likely contributes little to PM10 concentrations. Inhalation of noninfectious allergens can cause disease, the major response being allergic rhinitis, including seasonal hay fever and bronchial asthma (Smith 1978). Emissions of H_2S by trees generally occur in connection with moderate to high concentrations of sulfur in the atmosphere or soil. Thus, removal of SO_2 by trees under moderate to high SO_2 concentrations likely will be offset some by sulfur emissions by trees to the atmosphere.

Depending on their configuration around buildings, trees can increase or decrease building energy use. Trees generally conserve energy use in the summer but often increase use in the winter in colder climates (e.g., tree branches shade residences). This change in energy use alters pollutant emissions from local power plants. Thus, there are many interactive factors involving urban trees and air quality that remain to be investigated to more fully understand the impact of urban trees on air quality.

Model estimates of pollution removal by trees are specific to 1991 conditions in the Chicago area. Extrapolations to other years or other cities must consider specific pollution concentrations, tree configuration, and local meteorology.

Management Considerations

The majority of pollution removal by trees occurs under in-leaf daytime conditions as this is the time when leaf surfaces are actively transpiring and pollution concentrations can reach their maximum. The size of individual trees also affects total removal per tree. Large trees can remove 60 to 70 times more pollution a year than small trees. Thus, to maximize pollution removal by trees and other environmental benefits (e.g., reductions in air temperature), it is important to sustain healthy, functional (i.e., transpiring) trees, particularly large ones.

Future tree plantings can further enhance the air quality benefits of the urban forest and should be concentrated in polluted areas. When pollution concentrations become high, it is likely that stomates partially or fully close, reducing or eliminating most of the potential for pollution reduction of urban trees. However, tree response to pollutants varies by species and pollutant. Pollution-tolerant species (Kozlowski 1980) should be selected to enhance survival and subsequent air quality benefits.

Planting to reduce building energy use (McPherson 1994: Chapter 7, this report) also will improve air quality by reducing power plant emissions. Mass plantings can act as buffers from pollution sources (McCurdy 1978). Ample water should be supplied to enhance stomatal removal of pollution. Conifers should be planted to enhance particle removal, particularly in winter.

Monetary Value

Typical monetary values per tree are relatively small, ranging from \$0.04/yr for small trees to more than \$2/yr for large trees. These estimates are based on the cost of preventing the emission of a similar amount of pollutant with current control strategies. It is important to note that emission controls prevent pollution from entering the air while deposition to trees removes air pollutants already in the air. Using emission-control values likely overestimates the value generated by reducing pollutant concentrations after emission because once the pollutant is emitted, it can increase atmospheric concentrations and pollution effects around all surfaces, adversely affecting human health, materials, and visibility before being removed.

These estimates also do not fully incorporate the effects of trees on human health, materials, or visibility received through improvements in air quality. Other benefits and detriments not considered in this monetary valuation include possible lower concentrations of O_3 due to lower air temperatures, altered power plant emissions due to changes in building energy use, and changes in human perceptions of air quality. Perceptions can change through the production of pleasant odors, screening views from polluted air, and vegetation damage from pollution.

Research Issues

Continued research and field studies are needed to better evaluate and quantify aerodynamic and quasi-laminar boundary-layer resistances in urban areas. The R_a and R_b estimates in this study are minimal and in the range expected for forests (Fowler 1985). Considering that the stomatal influence on pollution removal is large, additional research is needed to investigate urban evapotranspiration (e.g., Grimmond and Oke 1991), particularly, urban tree transpiration, tree-canopy resistances to various pollutants, and the effect of pollutants on stomatal functioning (e.g., Baldocchi et al. 1987). Although advances are being made continually in these areas, particularly for forests and agricultural crops, field studies are needed to quantify pollution deposition in urban areas to begin to understand how various urban surfaces and combinations of surfaces influence pollution deposition and concentrations.

The study calculations are the first in a series to be developed to estimate pollution deposition in urban areas. Future calculations will incorporate all urban surfaces in a multi-layer model (e.g., Baldocchi 1988). Field measurements of urban tree stomatal resistance are planned to help improve these estimates. In addition, eddy-correlation estimates of pollutant deposition in urban areas are planned to test the removal estimates under summer field conditions.

Conclusion

Urban trees can improve air quality, removing approximately 590 metric tons (650 tons) of pollution in Chicago and 5,600 metric tons (6,100 tons) in Cook and DuPage Counties in 1991. These amounts relate to an average air quality improvement of approximately 0.3 percent, peaking at around one percent. These removal estimates are likely conservative, particularly for SO_2 and O_3. Further air quality improvement (reaching 5 to 10 percent or greater) can be obtained by increasing and sustaining healthy tree cover, particularly under stable atmospheric conditions. The majority of pollution removal by trees occurs during daylight in-leaf hours with the greatest overall removal effects for PM10 and O_3. Relatively minor removal was estimated for CO and urban trees may be an overall source of CO via tree volatile organic emissions. Research is needed to investigate the interactive relationships of pollution removal, trace-gas emissions, and air temperature and building energy use effects of urban trees on overall air quality.

Providing ample water to facilitate tree transpiration is critical to maximizing gaseous pollutant removal. Maximum percent reduction in pollution concentrations near the ground can be expected when trees are transpiring under stable atmospheric conditions and/or the boundary-layer height is relatively low. Trees offer both an active (via transpiration) and passive surface for gaseous and particulate pollutant removal, decreasing the amount of pollution inhaled by humans, deposited on anthropogenic material and available to decrease visibility. Trees should not be viewed as a substitute for emission controls, but rather as a supplement. Reduction of pollution emissions prevents possible pollution damage, reduction in ambient concentrations (e.g., via trees) only reduces the likelihood of possible damage. The effect of typical urban tree configurations on pollution emissions from both anthropogenic and biogenic sources remains to be investigated.

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Literature Cited

Baes, C. F.; Ragsdale, H. L. 1981. **Age-specific lead distribution in xylem rings of three tree genera in Atlanta, Georgia.** Environmental Pollution (Series B). 2: 21-35.

Baes, C. F.; McLaughlin, S. B. 1984. **Trace elements in tree rings: evidence of recent and historical air pollution.** Science. 224: 494-497.

Baldocchi, D. 1988. **A multi-layer model for estimating sulfur dioxide deposition to a deciduous oak forest canopy.** Atmospheric Environment. 22(5): 869-884.

Baldocchi, D. D.; Hicks, B. B.; Camara, P. 1987. **A canopy stomatal resistance model for gaseous deposition to vegetated surfaces.** Atmospheric Environment. 21(1): 91-101.

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

Black, V. J. 1985. **SO_2 effects on stomatal behavior.** In: Winner W. E., Mooney, H. A. and Goldstein R. A. eds. Sulfur dioxide and vegetation. Stanford, CA: Stanford University Press: 96-117.

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.

California Energy Commission. 1992. **1992 electricity report, air quality.** Sacramento, CA: California Energy Commission.

Cardelino, C. A.; Chameides, W. L. 1990. **Natural hydrocarbons, urbanization, and urban ozone.** Journal of Geophysical Research. 95(D9): 13,971-13,979.

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(11): 1807-1818.

Costonis, A. C. 1976. **Criteria for evaluating air pollution injury to forest trees.** Oslo, Norway: IUFRO Congress.

Darley, E. F. 1971. **Vegetation damage from air pollution.** In: Starkman, E. S. ed. Combustion-generated air pollution. New York: Plenum Press: 245-255.

Dasch, J. M. 1987. **Measurement of dry deposition to surfaces in deciduous and pine canopies.** Environmental Pollution. 44: 261-277.

Dasch, J. M. 1989. **Dry deposition of sulfur dioxide or nitric acid to oak, elm and pine leaves.** Environmental Pollution. 59: 1-16.

Davidson, C. I.; Wu, Y. L. 1990. **Dry deposition of particles and vapors.** In: Lindberg, S. E.; Page, A. L.; Norton, S. A. eds. Acid precipitation. Volume 3: sources, deposition, and canopy interactions. New York: Springer-Verlag: 103-216.

DeSanto, R. S.; Glaser, R. A.; McMillen, W. P.; MacGregor, K. A.; Miller, J. A. 1976a. **Open space as an air resource management measure. Volume II: design criteria.** EPA-450/3-76-028b. Research Triangle Park, NC: U.S. Environmental Protection Agency. 183 p.

DeSanto, R. S.; MacGregor, K. A.; McMillen, W. P.; Glaser, R. A. 1976b. **Open space as an air resource management measure. Volume III: demonstration plan (St. Louis, MO.).** EPA-450/3-76-028c. Research Triangle Park, NC: U.S. Environmental Protection Agency. 140 p.

Dyer, A. J.; Bradley, C. F. 1982. **An alternative analysis of flux gradient relationships.** Boundary-Layer Meteorology. 22: 3-19.

Fowler, D. 1980. **Removal of sulphur and nitrogen compounds from the atmosphere in rain and by dry deposition.** In: Drablos, D., Tollan, A. eds. Ecological impact of acid precipitation. Proceedings of an international conference, Sandefjord, Norway. Oslo, Norway: SNSF project: 22-32.

Fowler, D. 1985. **Deposition of SO_2 onto plant canopies.** In: Winner, W. E.; Mooney, H. A.; Goldstein, R. A. eds. Sulfur dioxide and vegetation. Stanford, CA: Stanford University Press: 389-402.

Fowler, D.; Cape, J. N. 1983. **Dry deposition of SO_2 onto a Scots pine forest.** In: Pruppacher, H. R.; Semonin, R. G.; Slinn, W. G. N. eds. Precipitation scavenging, dry deposition and resuspension. New York: Elsevier: 763-773.

Franchine, P. 1991. **Soot kills 60,000 a year in U.S., survey shows.** Chicago Sun-Times. June 30.

Fritschen, L. J.; Edmonds, R. 1976. **Dispersion of fluorescent particles into and within a Douglas fir forest.** In: Proceedings of symposium on atmosphere-surface exchange of particulate and gaseous pollutants. Richland, WA: 208-301.

Galbally, I. E.; Roy, C. R. 1980. **Destruction of ozone at the earth's surface.** Quarterly Journal of the Royal Meteorological Society. 106: 599-620.

Garland, J. A. 1977. **The dry deposition of sulphur dioxide to land and water surfaces.** Proceedings of the Royal Society of London Series A: Mathematical and Physical Sciences. 354: 245-268.

Garland, J. A.; Branson, J. R. 1977. **The deposition of sulphur dioxide to pine forest assessed by a radioactive tracer method.** Tellus. 29: 445-454.

Garsed, S. G. 1985. **SO_2 uptake and transport.** In: Winner W. E., Mooney, H. A. and Goldstein R. A. eds. Sulfur dioxide and vegetation. Stanford, CA: Stanford University Press: 75-95.

Golder, D. G. 1970. **A comparison of stability parameters.** University Park, PA: Pennsylvania State University. M.S. thesis.

Granat, L.; Johansson C. 1983. **Dry deposition of SO_2 and NO_x in winter.** Atmospheric Environment. 17(1): 191-192.

Gravenhorst, G.; Hofken, K. D.; Georgii, H. W. 1983. **Acidic input to a beech and spruce forest.** In: Beilke, S.; Elshout, A. J. eds. Acid deposition. Dordrecht, the Netherlands: Reidel: 155-171.

Greenhut, G. K. 1983. **Resistance of a pine forest to ozone uptake.** Boundary-Layer Meteorology. 27: 387-391.

Grimmond, C. S. B.; Oke, T. R. 1991. **An evapotranspiration-interception model for urban areas.** Water Resources Research. 27(7): 1739-1755.

Heggestad, H. E. 1972. **How plants fight "man-made" pollution.** The Science Teacher. (April): 21-24.

Heisler, G. M. 1986. **Energy savings with trees.** Journal of Arboriculture. 12(5): 113-125.

Hicks, B. B.; Baldocchi, D. D.; Meyers, T. P.; Hoskers, R. P.; Matt, D. R. 1987. **A preliminary multiple resistance routine for deriving dry deposition velocities from measured quantities.** Water, Air, and Soil Pollution. 36: 311-330.

Hicks, B. B.; Wesely, M. L.; Durham, J. L.; Brown, M. A. 1982. **Some direct measurements of atmospheric sulfur fluxes over a pine plantation.** Atmospheric Environment. 16(12): 2899-2903.

Hofken, K. D.; Gravenhorst, G. 1982. **Deposition of atmospheric aerosol particles to beech and spruce forest.** In: Georgii, H. W.; Pankrath, J. eds. Deposition of atmospheric pollutants. Dordrecht, the Netherlands: Reidel: 187-190.

Hofken, K. D.; Meixner, F. X.; Ehhalt, D. H. 1983. **Deposition of atmospheric trace constituents onto different natural surfaces.** In: Pruppacher, H. R.; Semonin, R .G., Slinn, W. G. N. eds. Precipitation scavenging, dry deposition and resuspension. New York: Elsevier: 825-835.

Holzworth, G. C. 1972. **Mixing heights, wind speeds, and potential for urban air pollution throughout the contiguous United States.** Research Triangle Park, NC: U.S. Environmental Protection Agency. 118 p.

Illinois Environmental Protection Agency. 1992. **1991 Illinois annual air quality report.** Springfield, IL: Illinois Environmental Protection Agency. 91 p.

Killus, J. P.; Meyer J. P.; Durran D. R.; Anderson, G. E.; Jerskey, T. N.; Reynolds S. D.; Ames, J. 1984. **Continued research in mesoscale air pollution simulation modeling. Volume V: refinements in numerical analysis, transport, chemistry, and pollutant removal.** EPA/600/3-84/095a. Research Triangle Park, NC: U.S. Environmental Protection Agency. 221 p.

Kozlowski, T. T. 1980. **Responses of shade trees to pollution.** Journal of Arboriculture. 6(2): 29-41.

Kramer, P. J.; Kozlowski, T. T. 1979. **Physiology of woody plants.** New York: Academic Press. 811 p.

Lindberg, S. E.; Harriss, R. C. 1981. **The role of atmospheric deposition in an eastern U.S. deciduous forest.** Water, Air, and Soil Pollution. 16: 13-31.

Lindberg, S. E.; Lovett, G. M. 1983. **Application of surrogate surface and leaf extraction methods to estimation of dry deposition to plant canopies.** In: Pruppacher, H. R.; Semonin, R. G., Slinn, W. G. N. eds. Precipitation scavenging, dry deposition and resuspension. New York: Elsevier: 837-848.

Lindberg, S. E.; Lovett, G. M.; Bondietti, E. A.; Davidson, C. I. 1984. **Recent field studies of dry deposition to surfaces in plant canopies.** In: 77th annual meeting of the Air Pollution Control Association; San Francisco, CA. Air Pollution Control Association: 2-15.

Linzon, S. N. 1978. **Effects of airborne sulfur pollutants on plants.** In: Nriagu, J. O. ed. Sulfur in the environment: Part II: ecological impacts. New York: John Wiley and Sons: 109-162.

Little, P. 1977. **Deposition of 2.75, 5.0 and 8.5 μm particles to plant and soil surfaces.** Environmental Pollution. 12: 293-305.

Lorenz, R.; Murphy, C. E. 1985. **The dry deposition of sulfur dioxide on a loblolly pine plantation.** Atmospheric Environment. 19(5): 797-802.

Lovett, G. M.; Lindberg, S. E. 1984. **Dry deposition and canopy exchange in a mixed oak forest as determined by analysis of throughfall.** Journal of Applied Ecology. 21: 1013-1027.

Lyons, W. A.; Olsson, L. E. 1973. **Detailed mesometeorological studies of air pollution dispersion in the Chicago lake breeze.** Monthly Weather Review. 101(5): 387-404.

McCurdy, T. 1978. **Open space as an air resource management strategy.** In: Hopkins, G. ed. Proceedings of the national urban forestry conference; Washington, DC. ESF Publ. 80-003. Syracuse, NY: State University of New York College of Environmental Science and Forestry: 306-319.

McMahon, T. A.; Denison, P. J. 1979. **Empirical atmospheric deposition parameters — a survey.** Atmospheric Environment. 13: 571-585.

McPherson, E. G. 1994. **Energy-saving potential of trees in Chicago.** In: McPherson, E. G., Nowak, D. J. and 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.

McPherson, E. G.; Nowak, D. J.; Sacamano, P. L.; Prichard, S. E.; Makra, E. M. 1993. **Chicago's evolving urban forest: initial report of the Chicago Urban Forest Climate Project.** Gen. Tech. Rep. NE-169. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 55 p.

Murphy, C. E.; Sigmon, J. T. 1990. **Dry deposition of sulfur and nitrogen oxide gases to forest vegetation.** In: Lindberg, S. E.; Page, A. L.; Norton, S. A. eds. Acid precipitation. Volume 3: sources, deposition, and canopy interactions. New York: Springer-Verlag: 217-240.

Murphy, C. E.; Sinclair, T. R.; Knoerr, K. R. 1977. **An assessment of the use of forests as sinks for the removal of atmospheric sulfur dioxide.** Journal of Environmental Quality. 6(4): 388-396.

National Academy of Sciences. 1977a. **Effects of nitrogen oxides on vegetation.** In: Nitrogen Oxides. Washington, DC: National Academy of Sciences:147-158.

National Academy of Sciences. 1977b. **Ozone and other photochemical oxidants.** Washington, DC: National Academy of Sciences. 789 p.

National Acid Precipitation Assessment Program. 1991. **1990 integrated assessment report.** Washington, DC: National Acid Precipitation Assessment Program. 520 p.

Nowak, D. J. 1991. **Urban forest structure and the functions of hydrocarbon emissions and carbon storage.** In: Proceedings of the fifth national urban forest conference; Los Angeles, CA. Washington, DC: American Forestry Association: 48-51.

Nowak, D. J. 1994. **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.

Office of Technology Assessment. 1989. **Catching our breath: next steps for reducing urban ozone.** OTA-O-412. Washington, DC: Office of Technology Assessment.

Oke, T. R. 1987. **Boundary layer climates.** New York: Methuen. 435 p.

Pasquill, F. 1961. **The estimation of dispersion of windborne material.** Meteorology. 90: 33-49.

Rasmussen K. H.; Taheri, M.; Kabel, R. L. 1975. **Global emissions and natural processes for removal of gaseous pollutants.** Water, Air, and Soil Pollution 4: 33-64.

Rennenberg, H. 1991. **The significance of higher plants in the emission of sulfur compounds from terrestrial ecosystems.** In: Sharkey, T. D., Holland, E. A., Mooney, H. A. eds. Trace gas emissions by plants. New York: Academic Press: 217-260.

Roberts, B. R. 1974. **Foliar sorption of atmospheric sulphur dioxide by woody plants.** Environmental Pollution. 7: 133-140.

Rogers, H. H.; Jeffries, H. E.; Witherspoon, A. M. 1979. **Measuring air pollutant uptake by plants: nitrogen dioxide.** Journal of Environmental Quality. 8(4): 551-557.

Rolfe, G. L. 1974. **Lead distribution in tree rings.** Forest Science. 20(3): 283-286.

Schjoerring, J. K. 1991. **Ammonia emission from the foliage of growing plants.** In: Sharkey, T. D., Holland, E. A., Mooney, H. A. eds. Trace gas emissions by plants. New York: Academic Press: 267-292.

Sehmel, G. A. 1980. **Particle and gas dry deposition: a review.** Atmospheric Environment. 14: 983-1011.

Shafer, S. R.; Heagle, A. S. 1989. **Growth responses of field-grown loblolly pine to chronic doses of ozone during multiple growing seasons.** Canadian Journal of Forestry. 19: 821-831.

Shanley, J. B. 1989. **Field measurements of dry deposition to spruce foliage and petri dishes in the black forest F.R.G.** Atmospheric Environment. 23(2): 403-414.

Sharkey, T. D.; Holland, E. A.; Mooney, H. A. eds. 1991. **Trace gas emissions by plants.** New York: Academic Press. 365 p.

Shieh, C. M.; Wesely, M. L.; Hicks, B. B. 1979. **Estimated dry deposition velocities of sulfur over the eastern United States and surrounding regions.** Atmospheric Environment. 13: 1361-1368.

Shiner, D. S.; Heck, W. W.; McLaughlin, S. B.; Johnson, D. W.; Irving, P. M.; Joslin, J. D.; Peterson, C. E. 1990. **Response of vegetation to atmospheric deposition and air pollution.** NAPAP SOS/T Report 18. Washington, DC: National Acid Precipitation Assessment Program. 206 p.

Smith, W. H. 1978. **Urban vegetation and air quality.** In: Hopkins, G. ed. Proceedings of the national urban forestry conference; Washington, DC. ESF Publ. 80-003. Syracuse, NY: State University of New York College of Environmental Science and Forestry: 284-305.

Smith, W. H. 1984. **Pollutant uptake by plants.** In: Treshow, M. ed. Air pollution and plant life. New York: John Wiley and Sons: 417-450.

Smith, W. H. 1990. **Air pollution and forests.** New York: Springer-Verlag. 618 p.

Stoker, H. S.; Seager, S. L. 1976. **Environmental chemistry: air and water pollution.** Glenview, IL: Scott, Foresman and Company. 231 p.

Tingey, D. T.; Turner, D. P.; Weber, J. A. 1991. **Factors controlling the emissions of monoterpenes and other volatile organic compounds.** In: Sharkey, T. D., Holland, E. A., Mooney, H. A. eds. Trace gas emissions by plants. New York: Academic Press: 93-119.

Townsend, A. M. 1974. **Sorption of ozone by nine shade tree species.** Journal of American Horticultural Science. 99(3): 206-208.

Trijonis, J. C.; Malm, W. C.; Pitchford, M.; White, W. H. 1990. **Visibility: existing and historical conditions — causes and effects.** NAPAP SOS/T Report 24. Washington, DC: National Acid Precipitation Assessment Program. 129 p.

van Dop, H.; Guicerit, R.; Lanting, R. W. 1977. **Some measurements of the vertical distribution of ozone in the atmospheric boundary layer.** Atmospheric Environment. 11: 65-71.

van Ulden, A. P; Holtslag, A. A. M. 1985. **Estimation of atmospheric boundary layer parameters for diffusion application.** Journal of Climate and Applied Meteorology. 24: 1196-1207.

Voldner, E. C.; Barrie, L. A.; Sirois, A. 1986. **A literature review of dry deposition of oxides of sulphur and nitrogen with emphasis on long-range transport modelling in North America.** Atmospheric Environment. 20(11): 2101-2123.

Walcek, C. J.; Brost, R. A.; Chang, J. S.; Wesely, M. L. 1986. **SO_2, sulfate and HNO_3 deposition velocities computed using regional land use and meteorological data.** Atmospheric Environment. 20(5): 949-964.

Wesely, M. L. 1983. **Turbulent transport of ozone to surfaces common in the eastern half of the United States.** In: Schwartz, S. E. ed. Trace atmospheric constituents: properties, transformations, and fates. New York: John Wiley & Sons: 345-370.

Wesely, M. L. 1989. **Parameterization for surface resistance to gaseous dry deposition in regional-scale numerical models.** Atmospheric Environment. 23(6): 1293-1304.

Wesely, M. L.; Cook, D. R.; Hart, R. L.; Hicks, B. B.; Durham, J. L.; Speer, R. E.; Stedman, D. H.; Tropp, R. J. 1983. **Eddy correlation measurements of the dry deposition of particulate sulfur and submicron particles.** In: Pruppacher, H. R.; Semonin, R. G., Slinn, W. G. N. eds. Precipitation scavenging, dry deposition and resuspension. New York: Elsevier: 943-952.

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(D1): 2131-2143.

Wesely, M. L.; Hicks, B. B. 1979. **Dry deposition and emission of small particles at the surface of the earth.** In: Proceedings of the fourth symposium on turbulence, diffusion and air quality; Reno, NV. Boston, MA: American Meteorological Society: 510-513.

Wesely, M. L.; Lesht, B. M. 1988. **Comparison of the RADM dry deposition module with site-specific routines for inferring dry deposition.** EPA/600/54-88/027. Research Triangle Park, NC: U.S. Environmental Protection Agency. 101 p.

Whittaker, R. H.; Woodwell, G. M. 1967. **Surface area relations of woody plants and forest communities.** American Journal of Botany. 54(8): 931-939.

Witherspoon, J. P.; Taylor, F. G. 1969. **Retention of a fallout simultant containing ^{134}Cs by pine and oak trees.** Health Physics. 17: 825-829.

Ziegler, I. 1973. **The effect of air-polluting gases on plant metabolism.** In: Environmental quality and safety. Volume 2. New York: Academic Press: 182-208.

Chapter 6
Atmospheric Carbon Dioxide Reduction by Chicago's Urban Forest

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Abstract

In terms of reducing atmospheric carbon dioxide (CO_2), trees in urban areas offer the double benefit of direct carbon storage and the avoidance of CO_2 production by fossil-fuel power plants through energy conservation from properly located trees. In the City of Chicago, trees store an estimated 855,000 metric tons (t) of carbon (942,000 tons), and trees throughout the study area of Cook and DuPage Counties store about 5.6 million t (6.1 million tons). Carbon storage by shrubs is approximately 4 percent of the amount stored by trees. Total carbon storage and annual sequestration are greatest on 1-3 family residential lands, institutional lands dominated by vegetation (e.g., parks, forest preserves) and vacant lands. Net carbon sequestration in the study area is estimated at 140,600 t (155,000 tons). Carbon storage by urban forests nationally likely is between 400 and 900 million t (440 to 990 millions tons).

Storage by individual trees is up to 1,000 times greater in large than in small trees, with sequestration rates up to 90 times greater for healthy large than healthy small trees. Estimated carbon emissions avoided annually due to energy conservation from existing trees throughout the study area is approximately 11,400 t (12,600 tons). Total carbon stored by trees in the study area, which took years to store, is equivalent to the amount of carbon emitted from the residential sector in the study area during a 5-month period. Net annual sequestration equals the amount of carbon emitted from transportation use in the study area in 1 week. The amount of carbon sequestered annually by one tree less than 8 cm (3 inches) in trunk diameter (d.b.h.) equals the amount emitted by one car driven 16 km (10 mi). Reasonable additional tree planting, in conjunction with efforts to sustain existing tree cover could increase carbon storage in the study area by another 1.2 million t (1.3 million tons), or the amount of carbon emitted by transportation use in the study area in less than 2 months. The advantages and limitations of urban trees in reducing atmospheric CO_2 are discussed.

Introduction

Increasing levels of atmospheric carbon dioxide (CO_2) and other "greenhouse" gases (e.g., methane, chlorofluorocarbons, nitrous oxide) are thought by many to be contributing to an increase in atmospheric temperatures by the trapping of certain wavelengths of heat in the atmosphere. Climate models indicate that the probable doubling of CO_2 within the next century would increase average global surface temperatures by 1.5° to 4.5°C (2.7° to 8.1°F) (U.S. National Research Council 1983). While no single gas is likely to have the direct impact on climate expected from CO_2, the sum of the radiative effects from other trace gases could effectively double the climatic impact of projected CO_2 increases (Wuebbles et al. 1989).

The observed increases in atmospheric concentrations of CO_2, methane (CH_4), chlorofluorocarbons (CFC's), and nitrous oxide (N_2O) during the 1980's, which resulted from human activities, contributed to the greenhouse effect by 56, 15, 24 and 5 percent, respectively (IPCC 1991). During this period, the contribution of different human activities to the change in the greenhouse effect is an estimated 46 percent from energy production and use; 24 percent from the production and use of CFC's and other halocarbons (e.g., from refrigerants, aerosol sprays); 18 percent from deforestation, biomass burning, and other changes in land use practices; 9 percent from agriculture (e.g., methane from rice cultivation and livestock and N_2O release from nitrogenous fertilizers); and 3 percent from other sources (e.g., methane from landfills) (IPCC 1991).

Urban Trees and Carbon Dioxide

Increased atmospheric CO_2 is attributable mostly to fossil fuel combustion (about 75 percent) and deforestation (Schneider 1989). Atmospheric carbon is estimated to be increasing by approximately 2.6 billion metric tons (t) (2.9 tons) annually (Sedjo 1989). By storing carbon through their growth process, trees act as a sink for atmospheric CO_2. Thus, increasing the number of trees can potentially slow the accumulation of atmospheric carbon (e.g., Moulton and Richards 1990).

In reducing atmospheric CO_2, trees in urban areas offer double benefits. First, they directly sequester and store atmospheric carbon. Second, when located properly, urban trees conserve energy, which results in lower CO_2 emissions from fossil-fuel power plants. Properly located trees shade residences in summer (reducing air-conditioning energy use), but also allow solar access and/or block winds in winter to reduce heating needs (Heisler 1986). Tree transpiration also reduces local air temperatures, which can affect local energy use. There has been little research on the amount of carbon that urban forests store, or on the effect of energy conservation by trees on the amount of carbon released to the atmosphere.

Biomass (dry weight) of trees in Shorewood, Wisconsin, a suburb of Milwaukee, has been estimated at 35.7 t per hectare (ha) of above-ground biomass (15.9 tons/acre) (Dorney et al. 1984). Biomass was calculated using a generalized formula from Whittaker et al. (1974). This biomass estimate converts to approximately 22.8 t/ha of carbon (10.2 tons/acre) (above and below ground). Shorewood's tree cover has been liberally estimated at 39 percent, with approximately 67 percent of the trees less than 15 cm (6 inches) in trunk diameter (d.b.h.) at 1.37 m (4.5 ft) (Dorney et al. 1984). Estimated carbon storage by trees in Oakland, California, (21 percent tree cover) is 145,800 t or 11.0 t/ha (160,700 tons or 4.9 tons/acre) (Nowak 1993).

Carbon storage by urban forests in the United States has been estimated at 350 to 750 million t (385 to 825 million tons) (Rowntree and Nowak 1991; Nowak 1993). It has been estimated that the establishment of 10 million urban trees annually over the next 10 years would sequester and offset the production of 363 million t (400 million tons) of carbon over the next 50 years, 77 million t (85 million tons) due to direct sequestration and 286 million t (315 million tons) due to avoided carbon emissions from power plants (Nowak 1993). This estimate assumes that the 100 million trees survive the 50-year period and were planted in optimal positions for energy conservation. Even so, this total is less than 1 percent of the amount of carbon emissions projected for the United States over the same 50-year period.

The purpose of this paper was to estimate total carbon storage, annual carbon sequestration, and carbon emissions avoided from power plants through energy conservation by trees in the Chicago area.

Methods

Ground Sampling of Trees

Data on 8,996 trees were collected on 652 randomly located plots throughout the study area (see Figure 1 in Chapter 2). 0.04-ha (0.1 acre) plots were used for all land uses except 1-3 family residential, where information on the entire residential lot was collected. Tree data collected included d.b.h., tree height, and species. Total shrub area was measured on each plot; on every tenth plot, diameters for individual shrubs were measured at 15 cm (6 inches) above groundline (see Nowak 1994: Chapter 2, this report).

Carbon and Tree Biomass

Biomass for each measured tree was calculated using allometric equations from the literature (Table 1). If no allometric equation could be found for an individual species, the genera average was substituted. If no genera equations were found, biomass was computed separately for each hardwood and conifer equation and the average result from the hardwood or conifer group was used.

To help determine whether allometric equations for forest-grown trees were applicable for urban trees, above-ground total fresh-weight biomass was collected for 30 street trees in Oak Park, Illinois. As the trees were removed, tree limbs were chipped and bagged and larger stems cut into logs. Logs and chips were weighed using a truck scale. Decay was evident in 10 trees but was not considered significant (Mike Stankovich, 1993, Village of Oak Park, pers. commun.). Measured trees ranged in d.b.h. from 20 to 99 cm (8 to 39 inches). Included were nine silver maple, eight American elm, four Norway maple, three ash, two pin oak, one elm, one linden, one tulip poplar and one sugar maple. Measured weight was matched against predicted weight using appropriate allometric equations. A pair-wise t-test was used to determine if significant differences existed between actual and predicted weights.

Measured biomass from street trees in Oak Park was significantly lower than that predicted from allometric equations from natural forest stands (alpha = 0.05). Biomass estimates of more open-grown trees were multiplied by a factor 0.8 to account for the discrepancy. No adjustment was made for trees found in more natural stand conditions (e.g., on vacant lands or in forest preserves).

Biomass equations differ in the portion of tree biomass that is calculated; whether fresh or oven-dry weight is estimated, and in the diameter ranges used to devise the equations (Table 1). Below-ground biomass of trees averages approximately 22 percent of total tree biomass (Bray 1963; Ovington 1965; Young and Carpenter 1967; Whittaker and Woodwell 1968; Andersson 1970; Woodwell and Botkin 1970; King and Schnell 1972; Whittaker and Marks 1975; Harriss et al. 1977; Hermann 1977; Husch et al. 1982; Raile and Jakes 1982; Czapowskyj et al. 1985; Harmon et al. 1990; Little and Shainsky 1992).

Average biomass per square meter of shrub cover was estimated for each land-use type by calculating the above-ground biomass (kg) using formulas in Smith and Brand (1983) and dividing the calculated biomass by individual shrub cover (m^2).

Below-ground biomass of small shrubs averaged approximately 61 percent of total shrub biomass (Whittaker 1962; Whittaker and Woodwell 1968; Woodwell and Botkin 1970). Many shrubs in the study area were larger than found in the literature, so a more conservative estimate of 40 percent of total biomass was used in converting above-ground shrub biomass to total shrub biomass. Equations that compute above-ground biomass were divided by 0.78 for trees and 0.6 for shrubs to convert to total biomass.

Equations that compute fresh-weight biomass were multiplied by species or genera specific conversion factors to yield dry-weight biomass. These conversion factors, derived from average moisture contents of species given in the literature, averaged 0.48 for conifers and 0.56 for hardwoods (U.S. Dept. Agric. 1955; Young and Carpenter 1967; King and Schnell 1972; Wartluft 1977; Stanek and State 1978; Wartluft 1978; Monteith 1979; Clark et al. 1980; Ker 1980; Phillips 1981; Husch et al. 1982; Schlaegel 1984a-d; Smith 1985).

For dead and dying trees, leaf biomass was removed from the estimate of total tree biomass using leaf biomass formulas derived as part of the Chicago Urban Forest Climate Project. Total biomass of dead trees was reduced by approximately 4 percent.

Table 1.—Attributes of biomass equations used to calculate tree biomass

Species	Tree part[a]	Weight[b]	D.b.h. range[c]	Reference
American beech	Above	Dry	3-56	Tritton and Hornbeck 1982
American beech	Above	Dry	3-66	Tritton and Hornbeck 1982
American beech	Above	Dry	5-51	Tritton and Hornbeck 1982
Aspen	Above	Dry	3-56	Tritton and Hornbeck 1982
Aspen	Total	Fresh	3-51	Wenger 1984
Balsam fir	Total	Dry	3-41	Stanek and State 1978
Balsam fir	Above	Dry	3-51	Tritton and Hornbeck 1982
Balsam fir	Total	Fresh	3-51	Wenger 1984
Black cherry	Above	Dry	5-51	Tritton and Hornbeck 1982
Black oak	Total	Dry	28-86	King and Schnell 1972
Chestnut oak	Above	Dry	5-51	Tritton and Hornbeck 1982
Douglas-fir	Total	Dry	3-122	Wenger 1984
Eastern hemlock	Total	Fresh	15-38	Stanek and State 1978
Eastern hemlock	Above	Dry	3-56	Tritton and Hornbeck 1982
Eastern hemlock	Above	Dry	3-51	Tritton and Hornbeck 1982
Eastern hemlock	Above	Dry	5-51	Tritton and Hornbeck 1982
Eastern hemlock	Total	Fresh	3-51	Wenger 1984
Eastern white-cedar	Above	Dry	3-30	Ker 1980
Green ash	Ab-lf	Dry	3-79	Schlaegel 1984a
Hickory	Total	Fresh	5-71	Wenger 1984
Hickory	Above	Dry	5-51	Tritton and Hornbeck 1982
Jack pine	Above	Dry	3-33	Stanek and State 1978
Jack pine	Total	Fresh	3-33	Wenger 1984
Lodgepole pine	Total	Dry	10-33	Stanek and State 1978
Longleaf pine	Total	Fresh	15-48	Wenger 1984
Norway spruce	Above	Dry	13-41	Jokela et al. 1986
Overcup oak	Ab-lf	Dry	3-86	Schlaegel 1984b
Paper birch	Total	Fresh	15-28	Stanek and State 1978
Paper birch	Above	Dry	3-51	Tritton and Hornbeck 1982
Pin cherry	Above	Dry	3-23	Tritton and Hornbeck 1982
Red maple	Above	Dry	3-56	Tritton and Hornbeck 1982
Red maple	Above	Dry	3-66	Tritton and Hornbeck 1982
Red maple	Above	Dry	5-51	Tritton and Hornbeck 1982
Red oak	Ab-lf	Dry	15-66	Clark et al. 1980
Red oak	Above	Dry	3-56	Tritton and Hornbeck 1982
Red oak	Above	Dry	5-51	Tritton and Hornbeck 1982
Red pine	Above	Dry	3-51	Tritton and Hornbeck 1982
Red pine	Total	Fresh	3-51	Wenger 1984
Red/white spruce	Total	Fresh	3-66	Wenger 1984
Scarlet oak	Ab-lf	Dry	13-51	Clark et al. 1980
Shortleaf pine	Total	Fresh	15-51	Wenger 1984
Slash pine	Total	Fresh	15-53	Wenger 1984
Spruce	Above	Dry	3-56	Tritton and Hornbeck 1982
Spruce	Above	Dry	3-66	Tritton and Hornbeck 1982
Sugarberry	Ab-lf	Dr	3-56	Schlaegel 1984c
Sugar maple	Above	Dry	3-56	Tritton and Hornbeck 1982
Sugar maple	Above	Dry	3-66	Tritton and Hornbeck 1982
Sugar maple	Total	Fresh	3-66	Wenger 1984
Sweetgum	Ab-lf	Dry	3-84	Schlaegel 1984d
Tulip-poplar	Ab-lf	Dry	15-71	Clark and Schroeder 1977

Table 1.—continued

Species	Tree part[a]	Weight[b]	D.b.h. range[c]	Reference
Tulip-poplar	Above	Dry	3-51	Tritton and Hornbeck 1982
Tulip-poplar	Above	Dry	5-51	Tritton and Hornbeck 1982
Western redcedar	Above	Dry	3-119	Stanek and State 1978
White ash	Above	Dry	5-51	Tritton and Hornbeck 1982
White oak	Above	Dry	5-51	Tritton and Hornbeck 1982
White pine	Above	Dry	3-56	Tritton and Hornbeck 1982
White pine	Above	Dry	3-66	Tritton and Hornbeck 1982
White pine	Total	Fresh	3-66	Wenger 1984
Yellow birch	Above	Dry	3-56	Tritton and Hornbeck 1982
Yellow birch	Above	Dry	3-66	Tritton and Hornbeck 1982
Yellow birch	Above	Dry	5-51	Tritton and Hornbeck 1982
Yellow birch	Total	Fresh	3-66	Wenger 1984

[a]Above = above-ground biomass; Ab-lf = above ground biomass excluding leaves; Total = total tree biomass (including roots).
[b]fresh or oven-dry weight.
[c]in cm

Total tree and shrub dry-weight biomass was converted to total stored carbon by multiplying by 0.5 (For. Prod. Lab. 1952; Millikin 1955; Ovington 1957; Reichle et al. 1973; Pingrey 1976; Ajtay et al. 1979; Chow and Rolfe 1989; Koch 1989). Total carbon storage by trees and shrubs was calculated by land-use type for each sector of the study area.

Because of a lack of information on errors in the basic formulas from which the projections were made and the various adjustment factors that were used, standard errors report sampling error rather than the error of estimation. Sampling errors underestimate the actual standard errors.

Urban Tree Growth and Carbon Sequestration

To estimate the amount of carbon sequestered annually by trees, urban tree-growth was estimated from measurements of radial growth increments. Sections cut at d.b.h. were obtained for 543 trees — 223 elms, 171 maples, 78 ash, 13 poplar, and 58 other (10 species) removed from Chicago, Oak Park, Glen Ellyn, and Bloomingdale during 1991-92. A radial line was marked across the section where average growth occurred (not compressed or elongated tree rings). To avoid measuring tree growth that might be affected by the condition of the removed trees (i.e., many trees were declining or dead), radial growth and tree cumulative radius to 0.05 cm (1/50 inch) were measured for each ring developed between 1965 and 1985. Average annual growth by diameter class was calculated for major genera. Average diameter growth from the appropriate genera and diameter class was added to the existing tree diameter (year *x*) to estimate tree diameter in year *x+1*. Average height growth was assumed to be 0.15 m/yr (0.48 ft/yr) (Fleming 1988). The difference in estimates of carbon storage between year *x* and year *x+1* is the amount of carbon sequestered annually.

Tree death will lead to the eventual release of stored carbon. This release is hastened when wood is burned or allowed to decay (e.g., not stored in durable wood products or landfills). To calculate the potential release of carbon due to tree death, estimates of annual mortality rates by diameter class were derived from a study of street-tree mortality (Nowak 1986). Annual mortality was estimated as 2.9 percent for trees 0 to 7 cm (0 to 3 inches) in diameter; 8 to 15 cm (3.1 to 6 inches) = 2.2 percent; 16 to 46 cm (6.1 to 18 inches) = 2.1 percent; 47 to 61 cm (18.1 to 24 inches) = 2.9 percent; 62 to 76 cm (24.1 to 30 inches) = 3.0 percent; and 77+ cm (30+ inches) = 5.4 percent. The amount of carbon sequestered due to tree growth was reduced by the amount lost due to tree mortality to estimate the net carbon sequestration rate.

Energy Conservation

Total distribution of residential natural gas in Chicago in 1992 was 4.16 billion m^3 (147 billion ft^3) (Peoples Energy Corp. 1993). In Dupage County, residential gas use in 1991 was 861 million m^3 (30.4 billion ft^3) (Northern Illinois Gas, 1992, pers. commun.). Cook County's estimated natural gas use, based on per capita consumption in Chicago and DuPage County, is 3.27 billion m^3 (115.6 billion ft^3). Natural gas consumption was converted to heating energy use by multiplying by 0.78 (Peoples Gas, 1992, pers. commun.); thousand m^3 of natural gas was converted to million Btu by multiplying by 36.55 (Energy Information Administration 1993). Total carbon emissions from natural gas were estimated based on the rate of 14.2 t (15.7 tons) of carbon per billion Btu for natural gas (Citizens Fund 1992). Total conservation of heating energy due to existing tree configurations (i.e., shading, wind modification) at 50 residences in Chicago has been estimated at 0.04 percent (Jo and Wilkin,1994). This value was used to estimate carbon emissions avoided due to the effects of existing trees on heating energy.

Total electrical energy generation by Commonwealth Edison in 1992 was 79.9 billion kWh with CO_2 emissions of 15.0 million t (16.5 million tons) (Commonwealth Edison, 1993, pers. commun.). Considering that 68 percent of Commonwealth Edison sales are in Cook and Dupage Counties (McPherson et al. 1993), 26.7 percent of sales are to residences (Commonwealth Edison, 1993, pers. commun.) and approximately 15 percent of residential energy use is for air conditioning (Greg McPherson, 1993, pers. commun.), it is estimated that air-conditioning energy use in the study area is 2.2 billion kWh. Commonwealth Edison's CO_2 emission rate is 0.051 t (0.056 tons) of carbon/MWh. Total conservation of air-conditioning energy use due to existing tree configurations at 50 residences in Chicago has been estimated at 8.4 percent (Jo and Wilkin 1994). This value was used to estimate carbon emissions avoided due to the effect of existing trees on air conditioning energy use.

Future Tree Planting

To analyze the potential effect of future tree plantings, available growing space (grass and soil area) was analyzed by land-use type throughout the study area. A reasonable tree-planting scenario assumes that none of the available space in agricultural or other transportation (predominantly airport) uses would be planted with trees due to land-use limitations. Five percent of available space could readily be planted and covered with trees on large commercial-industrial areas and institutional land dominated by vegetation such as parks, cemeteries, golf courses, and forest preserves. Ten percent of available space could be planted and covered with trees on institutional lands dominated by building such as schools, 15 percent in residential areas, 20 percent in landscaped commercial complexes, and 25 percent on vacant lands and along freeways.

Results

Total carbon storage by trees in the study area was about 5.6 million t or 85.7 t/ha of tree cover (6.1 million tons or 38.2 tons/acre). Trees in Chicago store 0.9 million t of carbon or 128.0 t/ha of tree cover (0.9 million tons or 57.1 tons/acre); suburban Cook County trees store 3.2 million t or 75.5 t/ha of tree cover (3.5 million tons or 33.7 tons/acre) and DuPage County trees store 1.5 million t or 95.0 t/ha of tree cover (1.7 million tons or 42.4 tons/acre) (Table 2). The most carbon stored by trees was on residential land and the least on agricultural lands. Total carbon stored by shrubs in the study area is estimated at 216,000 t (238,000 tons).

Tree carbon stored per ha in the study area averaged 16.7 t (7.4 tons/acre) and ranged from 14.1 t/ha (6.3 tons/acre) in Chicago to 17.7 t/ha (7.9 tons/acre) in DuPage County (Table 3). The highest carbon storage per ha was on institutional lands dominated by vegetation and least on agricultural lands (Table 3).

Average carbon storage by individual trees was 3 kg (7 lb) for a tree less than 8 cm (3 inches) d.b.h. to more than 3,100 kg (7,000 lb) for a tree greater than 76 cm (30 inches) d.b.h. (Figure 1, Table 4). Average carbon sequestration by individual trees ranged from 1.0 kg/yr (2.3 lb/yr) for a tree less than 8 cm d.b.h. to 93 kg /yr (204 lb/yr) for a tree greater than 76 cm d.b.h. (Figure 2, Table 4).

Table 2.—Total carbon stored (in thousands of metric tons) in Chicago, suburban Cook County, DuPage County, and entire study area (multiply thousands of metric tons by 1.102 to convert to thousands of tons)

	Chicago		Cook Co.		DuPage Co.		Study area	
Land use	Total	SE	Total	SE	Total	SE	Total	SE
Agriculture	0.0	0.0	0.0	0.0	2.9	2.6	2.9	2.6
Commercial[a]	0.2	0.2	8.9	5.1	8.6	4.9	17.7	7.1
Transportation[b]	40.5	25.5	0.0	0.0	19.7	19.7	60.2	32.2
Institutional (bldg.)[c]	28.7	25.9	0.0	0.0	42.1	31.6	70.7	40.9
Multiresidential[d]	100.9	87.8	24.0	11.6	7.0	1.7	131.9	88.5
Vacant	66.2	25.9	191.1	128.8	198.3	68.6	455.5	148.2
Institutional (veg.)[e]	198.2	46.1	1,308.4	192.6	310.6	66.4	1,817.2	208.9
Residential[f]	420.1	69.6	1,659.8	210.2	936.8	146.6	3,016.7	265.6
Total	854.8	129.1	3,192.2	313.1	1,525.9	178.9	5,572.9	383.0

SE = standard error (based on sampling error, not the error of estimation. Sampling errors underestimate the actual standard errors).
[a]Commercial/industrial.
[b]Airport, freeways, etc.
[c]Institutional lands dominated by buildings, e.g., schools, churches.
[d]Apartments with four or more units.
[e]Institutional lands dominated by vegetation, e.g., parks, cemeteries, forest preserves, golf courses.
[f]1-3 family residential buildings.

Table 3.—Carbon storage per hectare (metric tons) in Chicago, suburban Cook County, DuPage County, and entire study area (divide t/ha by 2.24 to convert to tons/acre)

	Chicago		Cook Co.		DuPage Co.		Study area	
Land use	Total	SE	Total	SE	Total	SE	Total	SE
Agriculture	0.0	0.0	0.0	0.0	0.2	0.2	0.1	0.1
Commercial	0.0	0.0	0.3	0.2	1.0	0.7	0.3	0.1
Transportation	7.2	4.5	0.0	0.0	9.0	9.0	3.5	1.9
Institutional (bldg.)	9.7	8.8	0.0	0.0	14.5	10.9	5.1	3.0
Multiresidential	17.3	15.0	5.7	2.8	3.2	0.8	10.8	7.3
Vacant	34.2	13.4	15.6	10.5	25.0	8.6	20.6	6.7
Institutional (veg.)	35.8	8.3	44.2	6.5	33.9	7.2	41.0	4.7
Residential	17.2	2.9	22.5	2.9	25.7	4.0	22.4	2.0
All uses	14.1	2.1	17.0	1.7	17.7	2.1	16.7	1.1

Table 4.—Average carbon stored (kg/tree) and sequestered (kg/tree/yr) in study area by d.b.h. class (multiply kg by 2.204 to convert to pounds)

	Carbon stored		Carbon sequestered	
D.b.h. class (cm)	Mean	SE	Mean	SE
0-7	3	0.05	1.0	0.02
8-15	24	0.3	4.4	0.05
16-30	105	1.4	9.4	0.1
31-46	399	6	19.1	0.3
47-61	962	19	34.6	0.8
62-76	1,808	51	55.3	1.8
77+	3,186	153	92.7	4.0

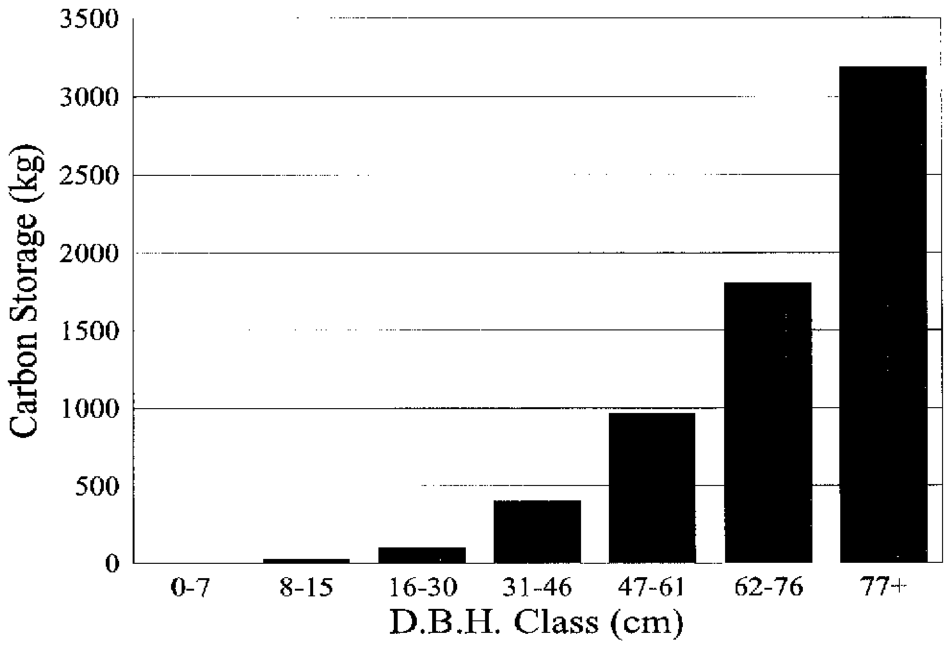

Figure 1. —Average carbon stored in individual urban trees by d.b.h. class (kg).

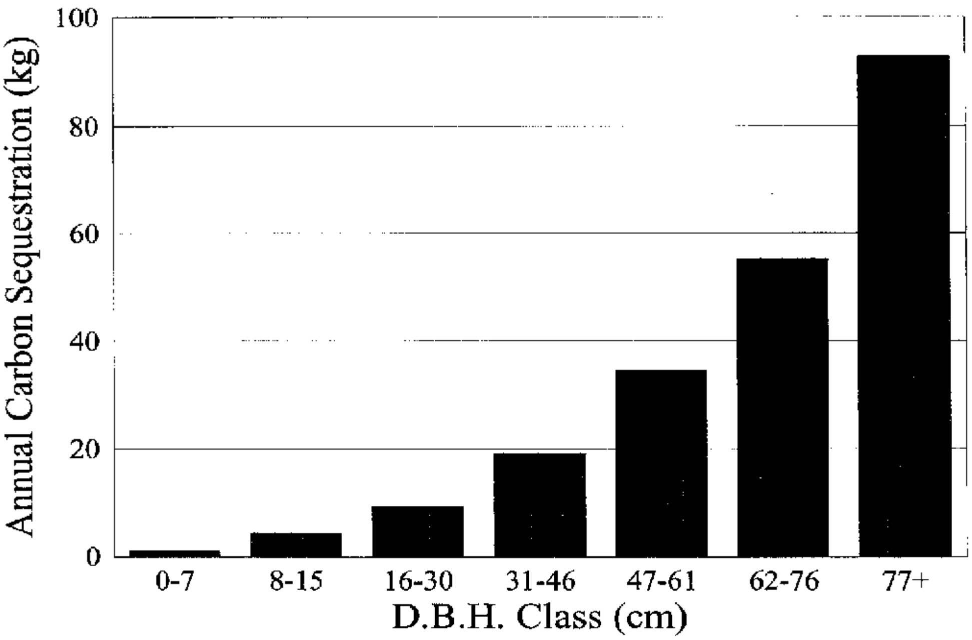

Figure 2. —Average annual carbon sequestration by individual urban trees by d.b.h. class (kg/year).

Average urban tree growth ranged from 0.78 to 1.02 cm/yr (0.31 to 0.40 inch/yr) (Table 5). Maximum total sequestration by trees in the study area (no tree mortality) is estimated at 315,800 t (348,000 tons) of carbon, ranging from 40,100 t (44,200 tons) in Chicago to 186,500 t (205,500 tons) in suburban Cook County (Table 6). Loss of carbon due to tree mortality in the study area (2.6 percent average annual mortality rate) is estimated at 175,200 t (193,000 tons) — 55 percent of the carbon sequestered — for a net sequestration rate of 140,600 t (155,000 tons) of carbon. This amounts to 0.4 t/ha of land and 2.2 t/ha of tree cover (0.2 ton/acre and 0.9 tons/acre). At an average mortality rate greater than 4.8 percent per year (assuming the same relative difference in mortality rates among the d.b.h. classes), more carbon would be lost due to tree mortality than would be sequestered by existing living trees.

Carbon emissions due to heating energy use in the study area total about 3.3 million t/yr (3.7 million tons/yr). Avoided carbon emissions due to savings in heating energy use from existing trees are estimated at 1,300 t/yr (1,500 tons/yr). Total carbon emissions due to air-conditioning use in the study area are approximately 109,900 t/yr (121,100 tons/yr). Avoided carbon emissions due to savings in air-conditioning use from existing trees are estimated at 10,100 t/yr (11,100 tons/yr).

If 0 to 25 percent of the available grass and soil space on various land uses were planted with trees, overall tree cover in the study area would increase from 19.4 to 23.5 percent. This planting assumes a tree-diameter structure comparable to what exists today and probably would take 40 to 80 years to become established. This tree establishment likely would store an additional 1.2 million t (1.3 million tons) of carbon. These trees also could reduce carbon emissions from power plants by lowering air temperatures through transpiration and by properly shading buildings and blocking winter winds.

Discussion

There are limitations to estimating carbon storage and sequestration by urban trees. Preliminary indications are that biomass equations derived from forest stands overestimate biomass from open-grown urban trees by a factor of 1.25. Open-grown trees typically are shorter but often have larger, more branchy crowns than forest-grown trees (Spurr and Barnes 1980). However, urban tree crowns often are pruned, which removes stored carbon. These differences in tree height and pruning likely contribute to the discrepancy between forest derived equations and measured biomass of urban trees. Pruning practices vary by location but street trees usually are well maintained; thus, the biomass equation adjustment factor (derived from street trees) likely is near maximum. Research is needed to further test the applicability of existing biomass equations to urban trees, and on how biomass-equation estimates vary by land-use type and associated maintenance practices.

D.b.h. ranges for biomass equations used in this study generally ranged from 3 to 66 cm (1 to 26 inches). The degree of error in predicting biomass outside of regression formula d.b.h. ranges is unknown, but visual inspection of biomass estimates for large trees (greater than 66 cm d.b.h.) indicates the estimates appear reasonable. Research is needed on root-shoot relationships of open-grown urban trees.

In U.S. forest ecosystems, 59 percent of the total carbon stored is in soils (Birdsey 1990). Estimates of carbon storage

Table 5.—Average tree-diameter growth rates (cm/yr), from a sample of street trees in the Chicago area, used for estimating carbon sequestration; dead and dying trees were given a growth rate of 0.0 cm/yr (divide cm by 2.54 to convert to inches)

	D.b.h. class (cm)						
Genera	0-7	8-15	16-30	31-46	47-61	62-76	77+
Ash	0.90	0.99	0.85	0.64	0.68	0.70	0.44
Elm	0.96	1.15	1.08	0.89	0.83	0.83	1.03
Maple	0.81	0.92	0.79	0.68	0.66	0.72	1.11
Other	0.80	1.10	0.87	0.73	0.73	0.71	0.42
Poplar	0.64	1.06	0.98	0.94	1.49	1.61	1.87
Average	0.85	1.02	0.90	0.79	0.78	0.84	0.95

Table 6.—Total carbon sequestered annually (in thousands of metric tons) in Chicago, suburban Cook County, DuPage County, and entire study area; estimates of sequestration are high because they do not account for tree mortality (multiply thousands of metric tons by 1.102 to convert to thousands of tons)

	Chicago		Cook Co.		DuPage Co.		Study area	
Land use	Total	SE	Total	SE	Total	SE	Total	SE
Agriculture	0.0	0.0	0.0	0.0	0.8	0.7	0.8	0.7
Commercial	0.1	0.1	2.2	1.4	0.7	0.3	2.9	1.4
Transportation	2.5	1.6	0.0	0.0	0.7	0.7	3.1	1.8
Institutional (bldg.)	1.2	1.0	0.0	0.0	1.7	1.1	3.0	1.5
Multiresidential	3.1	2.2	2.0	0.8	0.9	0.2	6.1	2.4
Vacant	4.4	1.6	13.5	5.9	21.3	6.6	39.2	9.0
Institutional (veg.)	10.7	2.2	94.4	12.4	17.9	3.4	123.0	12.8
Residential	18.2	2.7	74.4	8.0	45.1	6.3	137.7	10.5
Total	40.1	4.9	186.5	16.0	89.2	9.9	315.8	19.4

Table 7.—Average carbon stored (metric tons) per hectare of land in Oakland, CA, Chicago, suburban Cook County, and DuPage County; Oakland estimate is adjusted to meet same assumptions of biomass and carbon used in Chicago area estimates; land-use classes are combined to allow for equal comparison with Oakland estimates (Nowak 1993) (divide t/ha by 2.24 to convert to tons/acre)

Land use	Oakland	Chicago	Cook County	DuPage County
Commercial	0.6	0.0	0.3	1.0
Transportation	0.8	7.2	0.0	9.0
Residential[a]	10.4	17.2	21.6	24.4
Institutional/Wildland[b]	26.0	27.8	21.9	15.0
All uses	12.5	14.1	17.0	17.7

[a]Includes street trees that were categorized separately in Oakland.
[b]Wildlands, institutional and miscellaneous land uses, including agriculture.

for the Chicago area's urban forest include only carbon stored by trees and shrubs. Research is needed on carbon storage by soil, grass, and other components of the urban-forest ecosystem. Carbon storage by shrubs in the study area is approximately 4 percent of the amount stored by trees.

Estimates of carbon storage for the Chicago area differ from those for Oakland, California (Table 7). There are various factors that contribute to the differences observed among Oakland, Chicago, and Cook and DuPage Counties. One factor is the difference in land-use distribution among these areas. Oakland is relatively high in transportational land uses while Chicago is relatively high in commercial-industrial uses, and DuPage County is relatively high in agricultural use. As land-uses change, so does the amount of trees and associated tree biomass.

Land-use distribution affects overall tree density. Chicago had the lowest tree density with 68 trees/ha (28 trees/acre), followed by Oakland with 120 trees/ha (49 trees/acre), suburban Cook County with 169 trees/ha (68 trees/acre) and DuPage County with 173 trees/ha (70 trees/acre) (Table 3 , Chapter 2). The greater the tree density, the more biomass that is stored per ha given an equal diameter distribution.

Other factors that greatly influence carbon storage are tree species and diameter distribution. Tree species will differ in growth characteristics, so estimates of carbon storage can vary among trees of the same diameter. Chicago had relatively more large trees than other urban areas: 7.5 percent of Chicago's trees were larger than 46 cm (18 inches) d.b.h. compared with 4.5 percent for Oakland, 4 percent for DuPage County, and 3.5 percent for suburban Cook County. Cook and DuPage Counties had relatively more small trees with 78.7 and 76.7 percent of the trees less than 15 cm (6 inches) d.b.h. respectively. This compares with 63.5 percent in Chicago and 60.9 percent for Oakland (Table 9, Chapter 2).

Carbon stored per ha of tree cover was highest in Chicago at 128 t/ha (57 tons/acre), followed by DuPage County at 95.0 t/ha (42 tons/acre), suburban Cook County at 75.5 t/ha (34 tons/acre), and Oakland at 59.6 t/ha (27 tons/acre). Both tree density per ha of tree cover and tree-diameter distribution affect estimates of carbon storage per ha of tree cover. DuPage County had the highest density per ha of tree cover at 927 (375 trees/acre), followed by Cook County at 752 (304 trees/acre), Chicago at 619 (250 trees/acre), and Oakland at 571 (231 trees/acre). The estimate for Chicago may be too high due to the probability of a conservative estimate of tree cover from aerial photographs. The large amount and size of buildings in Chicago obscure small trees, so tree cover likely is underestimated and the amount of carbon stored per ha of tree cover probably is overestimated.

U.S. forest ecosystems store approximately 52.5 billion t (57.9 billion tons) of carbon, with 31 percent in live trees (Birdsey 1990). This estimate converts to 55 t of carbon/ha (24.5 tons/acre) of land in live trees in U.S. forests — 3 to 4 times greater than storage estimates for urban forests. This live-tree forest estimate of 55 t/ha is less than urban forest carbon storage estimates per ha with 100 percent tree cover because the former estimate is not based on 100 percent tree cover and the latter estimate includes dead trees (about 3 percent of total biomass). In the Chicago area, total carbon and residential carbon storage per ha appears to decrease with an increase in the density of urban development.

Carbon storage in urban forests nationally (28 percent tree cover) is estimated at 600 to 900 million t (660 to 990 million tons). This estimate falls at the upper end and beyond the estimated range (350 to 750 million t) of total carbon storage by U.S. urban forests (Nowak 1993).

Carbon Sequestration by Urban Trees

Total carbon stored by trees in the study area (5.6 million t), which took years to store, equals the amount of carbon emitted from the residential sector (including transportation use) in the study area during a 5-month period.[1] Net annual sequestration for all trees in the study area (140,600 t of carbon) equals the amount of carbon emitted from transportation use in the study area in one week.[2] The amount of carbon sequestered annually by one tree less than 8 cm d.b.h. is equivalent to the amount of carbon emitted by driving one car 16 km (10 mi). Annual sequestration by one tree greater than 77 cm d.b.h. is equivalent to driving one car approximately 1,460 km (900 mi).[3]

Carbon storage by individual trees is as much as 1,000 times greater in large than small trees, with sequestration rates as much as 90 times greater for healthy large than healthy small trees. Thus, to maximize carbon storage and sequestration from urban trees, it is necessary to ensure the survival and vigor of large trees and establish small ones.

The net sequestration rate is highly sensitive to mortality as tree death ultimately leads to the release of CO_2. An annual mortality rate of 2.6 percent was assumed in the estimate of net sequestration. This mortality rate is relatively low compared to that for newly planted street trees (Nowak et al. 1990). However, there is limited information on urban tree mortality, particularly for larger trees and nonstreet trees. If actual annual mortality of urban trees exceeds approximately 5 percent in the Chicago area (with no replacement plantings), it is likely that the urban forest will be a source of atmospheric CO_2. There will be a delay in the emission of CO_2 depending on the method of tree disposal (e.g., burning facilitates early emissions of CO_2). Trees removed today will contribute to CO_2 levels in the future, just as trees removed in the past are contributing to concentrations of CO_2 today. The cycle of carbon emissions due to urban tree removal needs further investigation.

[1] 2.24 t (2.47 tons) of carbon were emitted in 1991 from the residential sector (including transportation use) per capita in Illinois (Citizens Fund 1992). With 5.88 million people in the study area, an estimated 13.2 million t (14.5 million tons) of carbon are released annually from residences.

[2] 1.30 t (1.43 tons) of carbon were emitted on average in 1991 from all transportation uses per capita in Illinois (Citizens Fund 1992). With 5.88 million people in the study area, an estimated 7.6 million t (8.4 million tons) of carbon are released annually due to transportation use.

[3] 0.0636 kg of carbon emitted per vehicle km (0.226 lb/mi) (Citizens Fund 1992).

Average diameter growth of urban trees in this study ranged from 0.78 to 1.02 cm/yr (0.31 to 0.40 in/yr), within the range of average growth rates for street trees in New Jersey (0.58 to 1.09 cm/yr; 0.23 to 0.43 inch/yr) (Fleming 1988) but higher than those for trees in New York's Central Park (0.36 to 0.86 cm/yr; 0.14 to 0.34 inch/yr) (deVries 1987). The rates also are higher than those for forest trees in Illinois, which average 0.38 cm/yr (0.15 inch/yr) (Smith and Shifley 1984). Thus, the net sequestration rate is likely liberal as trees in more closed-canopy positions have slower growth rates than those in this study.

Energy Effects of Urban Trees

Estimated carbon emissions avoided annually due to energy conservation from existing trees throughout the study area total 11,400 t (12,600 tons). This amounts to about 8 percent of the net carbon sequestration rate. However, the heating energy conservation value (0.04 percent) likely is conservative as most of the sample buildings analyzed for energy use had a north-south orientation. Shading from trees on the south side of residences can increase winter heating use (Heisler 1986). If heating energy savings reached 3 percent (McPherson 1994: Chapter 7, this report), 113,600 t (125,200 tons) of carbon emissions would be avoided annually. More research is needed to evaluate the effect of existing tree configurations on residential energy use. Most studies to date have evaluated optimal tree configurations. A national average ratio of 4:1 carbon emissions avoided to carbon sequestered by urban trees has been estimated for optimal locations of urban trees (Nowak 1993). The actual ratio for existing urban tree configurations in the study area is probably much lower. Ratios can be higher in regions with little winter heating needs, but also can be negative in certain locations due to increased energy consumption from shading of homes in winter.

Avoided carbon emissions due to savings in air-conditioning energy use probably would be higher in other cities given the same energy savings as 83 percent of the study area's electricity is generated from nuclear sources.

Maximizing CO_2 Reduction with Urban Trees

There are two primary strategies for maximizing the effect of urban trees on atmospheric CO_2. The first is to sustain or enhance existing tree health to maximize sequestration while minimizing losses due to tree mortality. The net effect of existing trees is relatively minimal. However, due to the large amount of carbon stored in trees, existing trees could become a source of CO_2 through increased tree mortality in conjunction with minimal replanting to offset tree losses. A loss of urban trees without replacement is a net source of carbon to the atmosphere both directly and indirectly (loss of energy conservation around buildings).

The second strategy is to establish more properly chosen and located urban trees in available planting spaces. Planting trees to maximize building energy conservation will yield the greatest relative carbon benefit. A reasonable tree-planting program in conjunction with efforts to sustain existing tree cover could increase carbon storage in the study area by another 1.2 million t (1.3 million tons). This additional storage, which will take years to accrue, is the amount of carbon emitted through transportation use in the study area in less than 2 months. Future tree plantings must survive to ensure that they act as carbon sinks and not sources, that is, trees must live long enough to compensate for the CO_2 emitted due to planting and maintenance. Research is needed to analyze the carbon budget of urban trees.

Because trees are only a short term reservoir of carbon, future planting structures must be sustained to ensure that newly treed areas remain long-term carbon sinks. Although the benefit of carbon sequestering by trees will eventually be lost and the trees will need to be replanted, CO_2 emissions avoided by properly located urban trees are avoided forever.

Conclusion

Average carbon storage by trees in the Chicago area is between 14 and 18 t/ha (6 and 8 tons/acre), with more intensely urbanized areas having lower carbon storage. Estimates of carbon storage vary widely by land-use type and city depending on urban forest structure (e.g., species composition, tree density, diameter distribution). Estimates of carbon storage by urban forests nationally likely is between 400 and 900 million t (440 and 990 million tons). However, research is needed to refine this estimate and investigate urban forest characteristics and their influence on atmospheric CO_2. This research would include understanding variations in urban forests across the United States, carbon cycling and anthropogenic carbon emissions due to vegetation management, tree energy/carbon emission effects, and urban tree growth, mortality, and biomass. Although urban trees can help in reducing atmospheric CO_2, their effect is minimal relative to the magnitude of emissions in urban areas. The principal ways to decrease CO_2 emissions are increasing energy conservation and efficiency and converting to non-carbon or low-carbon fuels.

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Literature Cited

Ajtay, G. L.; Ketner, P.; Duvingneaud, P. 1979. **Terrestrial primary production and phytomass.** In: Bolin, B.; Degens, E. T.; Kempe, S; Ketner, P. eds. The global carbon cycle. SCOPE Rep. 13. New York: John Wiley and Sons: 129-181.

Andersson, F. 1970. **Ecological studies in Scanian Woodland and meadow area, southern Sweden.** II. Plant biomass, primary production and turnover of organic matter. Botaniska Notiser. 123: 8-51.

Birdsey, R. A. 1990. **Inventory of carbon storage and accumulation in U.S. forest ecosystems.** In: Burkhart, H. E.; Bonnor, G. M.; Lowe, J. J. eds. Research in forest inventory, monitoring, growth and yield. Proceedings, IUFRO world congress; Montreal, PQ. Publ. FWS-3-90. Blacksberg, VA: Virginia Polytechnic Institute and State University, School of Forestry: 24-31.

Bray, J. R. 1963. **Root production and estimation of net productivity.** Canadian Journal of Botany. 41: 65-71.

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.

Citizens Fund. 1992. **The heat is on: America's CO_2 polluters.** Res. Rep. Washington, DC: Citizens Fund. 60 p.

Clark, A.; Phillips, D. R.; Hitchcock, H. C. 1980. **Predicted weights and volumes of scarlet oak trees on the Tennessee Cumberland Plateau.** Res. Pap. SE-214. Ashville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 23 p.

Clark, A.; Schroeder, J. G. 1977. **Biomass of yellow-poplar in natural stands in western North Carolina.** Res. Pap. SE-165. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 15 p.

Czapowskyj, M. M.; Robinson, D. J.; Briggs, R. D.; White, E. H. 1985. **Component biomass equations for black spruce in Maine.** Res. Pap. NE-564. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 7 p.

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.

Dorney, J. R.; Guntenspergen, G. R.; Keough, J. R.; Stearns, F. 1984. **Composition and structure of an urban woody plant community.** Urban Ecology. 8: 69-90.

Energy Information Administration. 1993. **Household energy consumption and expenditures 1990.** Publ. DOE/EIA-0321(90)/S. Washington, DC: U.S. Department of Energy, Energy Information Administration.

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

Forest Products Laboratory. 1952. **Chemical analyses of wood.** Tech. Note 235. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 4 p.

Harmon, M. E.; Ferrell, W. K.; Franklin, J. F. 1990. **Effects of carbon storage of conversion of old-growth forests to young forests.** Science. 247: 699-702.

Harriss, W. F.; Kinerson, R. S.; Edwards, N. T. 1977. **Comparison of belowground biomass of natural deciduous forests and loblolly pine plantations.** In: Marshall, J. K. ed. The belowground ecosystem: a synthesis of plant-associated processes. Sci. Ser. No. 26. Fort Collins, CO: Colorado State University, Range Science Department: 29-37.

Heisler, G. M. 1986. **Energy savings with trees.** Journal of Arboriculture. 12(5): 113-125.

Hermann, R. K. 1977. **Growth and production of tree roots: a review.** In: Marshall, J. K. ed. The belowground ecosystem: a synthesis of plant-associated processes. Sci. Ser. No. 26. Fort Collins, CO: Colorado State University, Range Science Department: 7-27.

Husch, B.; Miller, C. I.; Beers, T. W. 1982. **Forest mensuration.** New York: John Wiley and Sons. 402 p.

Intergovernmental Panel on Climate Change. 1991. **Climate change. The IPCC response strategies.** Washington, DC: Island Press.

Jo, H.; Wilkin, D. 1994. **Landscape carbon budgets and planning guidelines for greenspace in urban residential lands.** Unpublished report on file at USDA Forest Service, Pacific Southwest Research Station, Davis, CA. 205 p.

Jokela, E. J.; VanGurp, K. P.; Briggs, R. D.; White, E. H. 1986. **Biomass estimation equations for Norway spruce in New York.** Canadian Journal of Forest Research. 16: 413-415.

Ker, M. F. 1980. **Tree biomass equations for seven species in southwestern New Brunswick.** Fredericton, NB: Canadian Forestry Service. 18 p.

King, W. W.; Schnell, R. L. 1972. **Biomass estimates of black oak tree components.** Tech. Note Bl. Norris, TN: Tennessee Valley Authority, Division of Forestry. 24 p.

Koch, P. 1989. **Estimates by species group and region in the USA of: I. Below-ground root weight as a percentage of ovendry complete-tree weight; and II. Carbon content of tree portions.** Unpublished report by Wood Science Laboratory, 942 Little Willow Creek Rd., Corvallis, MT 59828. 23 p.

Little, S. N.; Shainsky, L. J. 1992. **Distribution of biomass and nutrients in lodgepole pine/bitterbrush ecosystems in central Oregon.** Res. Pap. PNW-454. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 22 p.

McPherson, E. G.; Nowak, D. J.; Sacamano, P. L.; Prichard, S. E.; Makra, E. M. 1993. **Chicago's evolving urban forest: initial report of the Chicago Urban Forest Climate Project.** Gen. Tech. Rep. NE-169. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 55 p.

McPherson, E. G. 1994. **Energy-saving potential of trees in Chicago.** 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.

Millikin, D. E. 1955. **Determination of bark volumes and fuel properties.** Pulp and Paper Magazine of Canada. (December): 106-108.

Monteith, D. B. 1979. **Whole tree weight tables for New York.** AFRI Res. Rep. 40. Syracuse, NY: State University of New York, College of Environmental Science and Forestry, Applied Forestry Research Institute. 64 p.

Moulton, R. J; Richards, K. R. 1990. **Costs of sequestering carbon through tree planting and forest management in the United States.** Gen. Tech. Rep. WO-58. Washington, DC: U.S. Department of Agriculture, Forest Service. 46 p.

Nowak, D. J. 1986. **Silvics of an urban tree species: Norway maple (*Acer platanoides* L.).** Syracuse, NY: State University of New York, College of Environmental Science and Forestry. 148 p. M.S. thesis.

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

Nowak, D. J. 1994. **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.

Nowak, D. J.; McBride, J. R.; Beatty, R. A. 1990. **Newly planted street tree growth and mortality.** Journal of Arboriculture. 16(5): 124-129.

Ovington, J. D. 1957. **The volatile matter, organic carbon and nitrogen contents of tree species grown in close stands.** The New Physiologist. 56: 1-11.

Ovington, J. D. 1965. **Organic production, turnover and mineral cycling in woodlands.** Biological Review. 40: 295-336.

Peoples Energy Corporation. 1993. **Historical data book, fiscal year 1992.** Chicago, IL: Peoples Energy Corporation.

Phillips, D. R. 1981. **Predicted total-tree biomass of understory hardwoods.** Rep. Pap. SE-223. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 22 p.

Pingrey, D. W. 1976. **Forest products energy overview.** In: Energy and the wood products industry. Madison, WI: Forest Products Research Society: 1-14.

Raile, G. K; Jakes, P. J. 1982. **Tree biomass in the North Central region.** Res. Pap. NC-220. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 22 p.

Reichle, D. E.; Dinger, B. E.; Edwards, N. T.; Harris, W. F.; Sollins, P. 1973. **Carbon flow and storage in a forest ecosystem.** In: Woodwell, G. M.; Pecan, E. V. eds. Carbon and the biosphere. Proceedings of the 24th Brookhaven Symposium in Biology. 1973 May 16-18. U.S. Atomic Energy Commission: 345-365.

Rowntree, R. A.; Nowak, D.J. 1991. **Quantifying the role of urban forests in removing atmospheric carbon dioxide.** Journal of Arboriculture. 17(10): 269-275.

Schlaegel, B. E. 1984a. **Green ash volume and weight tables.** Res. Pap. SO-206. New Orleans, LA: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B. E. 1984b. **Overcup oak volume and weight tables.** Res. Pap. SO-207. New Orleans, LA: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 14 p.

Schlaegel, B. E. 1984c. **Sugarberry volume and weight tables.** Res. Pap. SO-205. New Orleans, LA: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 13 p.

Schlaegel, B. E. 1984d. **Sweetgum volume and weight tables.** Res. Pap. SO-204. New Orleans, LA: U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station. 14 p.

Schneider, S. H. 1989. **The changing climate.** Scientific American. 261(3): 70-79.

Sedjo, R. A. 1989. **Forests to offset the greenhouse effect.** Journal of Forestry 87: 12-15.

Smith, W. B. 1985. **Factors and equations to estimate forest biomass in the North Central region.** Res. Pap. NC-268. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 6 p.

Smith, W. B.; Brand, G. J. 1983. **Allometric biomass equations for 98 species of herbs, shrubs, and small trees.** Res. Note NC-299. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 8 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.

Spurr, S. H.; Barnes, B. V. 1980. **Forest ecology.** New York: John Wiley and Sons: 687 p.

Stanek, W.; State, D. 1978. **Equations predicting primary productivity (biomass) of trees, shrubs and lesser vegetation based on current literature.** Publ. BC-X-183. Victoria, BC: Canadian Forest Service. 58 p.

Tritton, L. M.; Hornbeck, J. W. 1982. **Biomass equations for major tree species of the Northeast.** Gen. Tech. Rep. NE-69. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 26 p.

U.S. Department of Agriculture. 1955. **Wood handbook.** Agric. Handbook 72. Washington DC: U.S. Department of Agriculture. 528 p.

U.S. National Research Council. 1983. **Changing climate: report of the Carbon Dioxide Assessment Committee.** Washington, DC: National Academy Press.

Wartluft, J. L. 1977. **Weights of small Appalachian hardwood trees and components.** Res. Pap. NE-366. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 4 p.

Wartluft, J. L. 1978. **Estimating top weights of hardwood sawtimber.** Res. Pap. NE-427. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 7 p.

Wenger, K. F. ed. 1984. **Forestry handbook.** New York: John Wiley and Sons: 1335 p.

Whittaker, R. H. 1962. **Net production relations of shrubs in the Great Smokey Mountains.** Ecology. 43(3): 357-377.

Whittaker, R. H.; Bormann, F. H.; Likens, G. E.; Siccama, T. G. 1974. **The Hubbard Brook ecosystem study: forest biomass and production.** Ecological Monographs. 44: 233-254.

Whittaker, R. H.; Marks, P. L. 1975. **Methods of assessing terrestrial productivity.** In: Lieth, H.; Whittaker, R. H. eds. Primary productivity of the biosphere. New York: Springer-Verlag: 55-118.

Whittaker, R. H.; Woodwell, G. M. 1968. **Dimension and production relations of trees and shrubs in the Brookhaven Forest, New York.** Journal of Ecology. 56(1): 1-25.

Woodwell, G. M.; Botkin, D. B. 1970. **Metabolism of terrestrial ecosystems by gas exchange techniques: the Brookhaven approach.** In: Reichle, D.E. ed. Analysis of temperate forest ecosystems. Ecological studies 1. New York: Springer-Verlag: 73-85.

Wuebbles, D. J.; Grant, K. E.; Connell, P. S.; Penner, J. E. 1989. **The role of atmospheric chemistry in climate change.** Journal of the Air Pollution Control Association. 39(1): 22-28.

Young, H. E.; Carpenter, P. M. 1967. **Weight, nutrient element and productivity studies of seedlings and saplings of eight tree species in natural ecosystems.** Orono, ME: University of Maine, Maine Agricultural Experiment Station. 39 p.