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Carbon Storage and Flux in Urban Residential Greenspace

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There is increasing concern about the predicted negative effects of the future doubling of carbon dioxide on the earth. This concern has evoked interest in the potential for urban greenspace to help reduce the levels of atmospheric carbon. This study quantifies greenspace-related carbon storage and annual carbon fluxes for urban residential landscapes. For detailed quantification, the scale of this study was limited to two residential blocks in northwest Chicago which had a significant difference in vegetation cover.

Differences between the two blocks in the size of greenspace area and vegetation cover resulted in considerable differences in total carbon storage and annual carbon uptake. Total carbon storage in greenspace was about 26·15 kg/m^2 of greenspace in study block 1, and 23·20 kg/m^2 of greenspace in block 2. Of the total, soil carbon accounted for approximately 78·7% in block 1 and 88·7% in block 2. Trees and shrubs in block 1 and block 2 accounted for 20·8% and 10·6%, respectively. The carbon storage in grass and other herbaceous plants was approximately 0·5–0·7% in both blocks. Total net annual carbon input to the study blocks by all the greenspace components was in the region of 0·49 kg/m^2 of greenspace in block 1 and 0·32 kg/m^2 of greenspace in block 2. The principal net carbon release from greenspaces of the two residential landscapes was from grass maintenance. Greenspace planning and management strategies were explored to minimize carbon release and maximize carbon uptake.

Keywords: climate change, carbon budget, greenspace planning, residential landscape, northwest Chicago.

1. Introduction

The greenhouse effect is one of the most serious concerns of our time. Carbon dioxide plays the major role in absorbing outgoing terrestrial radiation, contributing approximately half of the total greenhouse effect (Ciborowski, 1989; Rodhe, 1990). The atmospheric concentration of carbon dioxide has increased by approximately 25% in the last 100 to 150 years, and is currently rising by 4% per decade, because of fossil

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fuel combustion and deforestation (Ciborowski, 1989; Post *et al*., 1990). Existing studies (Manabe and Wetherald, 1987; Hansen *et al*., 1988; Washington and Meehl, 1989; Mitchell *et al*., 1990) predict that this continued trend in CO_2 emissions could result in a doubling of pre-industrial CO_2 concentrations and changes in the global climate within the next 50–100 years. If the existing projection is correct, these changes may pose a serious threat to global ecological and socio-economic systems (Emanuel *et al*., 1985; Gleick, 1987; Pastor and Post, 1988; Smith and Tirpack, 1988; Kempt, 1990; Melillo *et al*., 1990; Wilson, 1990; Schlesinger, 1991).

Rising concern about the greenhouse effect has provoked interest in the potential for urban greenspace to help reduce the level of atmospheric CO_2. Urban greenspace can reduce atmospheric carbon in two ways: (1) directly, through sequestration and (2) indirectly, through savings in the cooling and heating energy of buildings. Rowntree (1989) contended that urban forest design and management, that considers the objectives of energy conservation and carbon sequestering, can help mitigate the global CO_2 problem. The Minnesota Department of Natural Resources (1991) quantified carbon release by human activities and carbon sequestration by urban and rural forests in Minnesota, and developed a tree planting program to maximize carbon sequestration and energy conservation benefits. Nowak (1993) estimated diameter distribution and carbon storage of urban trees in Oakland, California, extrapolated this estimate to carbon storage by urban trees in the United States, and explored the effect of future tree plantings in urban areas on levels of atmospheric carbon.

While urban greenspace helps reduce atmospheric carbon, both directly, and indirectly, it also contributes to carbon emission through the consumption of energy for landscape management activities, such as mowing, pruning, irrigation, and fertilization. These can generate carbon either directly or indirectly (Pitt, 1984). Direct release occurs when, for example, gasoline is used to mow grass or electricity is used to pump water for irrigation. Indirect release occurs when material or equipment for maintenance is used that requires energy in its manufacture or installation.

Although several studies described energy consumption for landscape maintenance (Falk, 1976; Parker, 1982; Pitt, 1984), none considered carbon uptake to calculate the net effect. More comprehensive studies that quantify carbon flux and storage in urban greenspace (including grass, soils and trees) are required to improve our understanding of carbon cycling in urban landscapes and to provide detailed greenspace planning guidelines to reduce atmospheric carbon levels.

The first objective of this study was to quantify carbon uptake, storage and release by greenspace of residential neighborhoods in Chicago. The second objective was to develop effective planting and management strategies using the carbon flux data. The study did not consider indirect carbon reduction through energy savings by vegetation.

Greenspace has been defined here as any soil surface area capable of supporting vegetation (trees, shrubs, and grasses). Classification of trees and shrubs followed Dirr's (1977) system. Vines were included as shrubs in the study. Herbaceous plants were defined as all non-woody plants such as annual or perennial flowers and groundcover, with the exception of grasses.

2. Study area and methods

2.1. STUDY AREA

Two residential blocks located in northwest Chicago were selected as the study area. The criteria for selection of the study blocks were accessibility for data collection, similarity in building construction date, and difference in vegetation cover. Areal vegetative cover was relatively higher in study block 1 than in study block 2. Block 1 is enclosed by W. Catalpa Ave. and W. Rascher Ave., and N. Virginia Ave. and N. Francisco Ave. Block 2 is enclosed by W. Bryn Mawr Ave. and W. Gregory St., and N. California Ave. and N. Washtenaw Ave. Blocks 1 and 2 have 22 and 28 residential units, and are 1·86 and 1·61 ha in size, respectively. In block 2, one unusually large multi-family residential unit was excluded. Permission for access to survey individual lots was received from 16 residential units in block 1 (73% of total) and 17 residential units in block 2 (61% of total).

2.2. MEASUREMENT OF TOTAL CARBON STORAGE

2.2.1. *Trees and shrubs*

All the trees and shrubs in the sample of 33 residential lots were inventoried. For trees, the following dimensions were measured: diameter at breast height (dbh), using a tape or caliper, total height using an altimeter, and merchantable height (from a 0·3 m stump to the point where the central stem breaks into limbs for trees greater than or equal to 13 cm dbh). Total height, diameter at ground level and at 15 cm above ground were measured for shrubs.

Existing biomass equations generated from forest areas were used to estimate foliage, root and total biomass in dry weight, because there are no biomass equations from urban areas. Table 1 includes the sources of biomass equations used. A maximum of five equations were used for each individual plant, according to the diameter range for which the equations are applicable. If no biomass equation for a particular species existed, the average biomass estimate derived from formulas for the same genus or group (hardwood or conifer) was used. The dry-weight biomass of wood and foliage for each individual tree and shrub was converted to a carbon storage estimate by multiplying by 0·5 (USDA Forest Products Laboratory, 1952; Millikin, 1955; Ovington, 1956; Reichle *et al*., 1973; Pingrey, 1976; Ajtay *et al*., 1979; Chow and Rolfe, 1989).

Biomass of elm species, silver maple, Norway maple, ash species, and linden species was corrected using regression equations generated from fresh weight and dbh measurements from street trees removed by the Forestry Public Works Division in Oak Park, Illinois. The fresh weight was converted to dry weight based on data of tree moisture contents (Smith, 1985) and on a formula calculating dry weight from the use of the moisture contents (Phillips, 1981). Foliage biomass of common species in the study area (i.e. Norway and silver maple, elm species, green and white ash, and honeylocust) was corrected by regression equations generated from leaf dry weight data from urban trees measured as a part of the Chicago Urban Forest Climate Project (Nowak, 1994). The leaf dry weight was determined through random sampling of ten samples per tree with a 0·4 m^3 frame, from 54 healthy trees of Norway maple, American elm, green ash, honeylocust and hackberry.

For species without available root biomass equations, the average ratio of below ground biomass and above ground biomass was applied to estimate total biomass according to species and age. In a given species, the ratios of below ground biomass: above ground biomass decrease with age (Whittaker, 1962; Bray, 1963; Harris *et al*., 1973; Whittaker and Marks, 1975). Based on various literature (Whittaker, 1962; Bray, 1963; Ovington, 1965; Whittaker and Woodwell, 1968; Arts and Marks, 1971; Whittaker and Marks, 1975; Harris *et al*., 1977; Hermann, 1977), below ground biomass was

TABLE 1. Sources of biomass equations used to calculate biomass in dry weight of trees and shrubs

Species	Part	Diameter# range (CM)	Source
Arborvitae	TA, F	0·3–5·1*	Roussopoulos and Loomis (1979)
	R	2·5–10	Stanek and State (1978)
	TA, F	2·2–30·2	Ker (1980)
Ash	TA, F	0·9–27·9	Ker (1980)
	A	2·3–77·7	Schlaegel (1984)
	TA	5·1–50·8	Tritton and Hornbeck (1982)
	A	≥ 12·7	Hahn (1984), Phillips (1981) Smith (1985)
Buckthorn	TA, F	0·2–3·2*	Harrington *et al.* (1989)
	TA, F	0·5–1·8*	Grigal and Ohmann (1977)
Cherry	TA, F	0·7–2·9*	Harrington *et al.* (1989)
	TA, F	0·8–3·8*	Roussopoulos and Loomis (1979)
	TA	1–10	Tritton and Hornbeck (1982)
	F	2·5–7·5	Stanek and State (1978)
	TA	2·5–22·9	Tritton and Hornbeck (1982)
	TA	5·1–50·8	Tritton and Hornbeck (1982)
	R	***	Whittaker and Marks (1975)
Cottonwood	TA, F	0·5–3·3*	Roussopoulos and Loomis (1979)
	TA	2·5–55	Tritton and Hornbeck (1982)
	TA, F	4·5–33·0	Stanek and State (1978)
	TAB, R	14·5–24·5	Stanek and State (1978)
	R	***	Whittaker and Marks (1975)
Current	TA, F	0·4–1·4**	Smith and Brand (1983)
Dogwood	TA, F	0·3–1·8*	Connolly (1981)
	TA, F	0·3–3·6*	Roussopoulos and Loomis (1979)
	TA, F	0·7–3·3*	Harrington *et al.* (1989)
	A	2·5–12·4	Phillips (1981)
	F		Other hardwoods
Elm	A	<12·7	Hahn (1984), Phillips (1981), Smith (1985)
	A	≥ 12·7	Hahn (1984), Phillips (1981), Smith (1985)
	F		Other hardwoods
Fir	TA, F	0·5–3·3*	Roussopoulos and Loomis (1979)
	TAB	2·5–40·0	Stanek and State (1978)
Hawthorn	A	<12·7	Hahn (1984), Phillips (1981), Smith (1985)
	A	≥ 12·7	Hahn (1984), Phillips (1981), Smith (1985)
	F		Other hardwoods
Hemlock	TAB, F, R	0·0–7·5**	Stanek and State (1978)
	TA, F	1·5–33·8	Ker (1980)
	TA	2·5–50	Tritton and Hornbeck (1982)
	TAB	2·5–50·8	Wenger (1984)
	TAB, R	14·0–37·8	Stanek and State (1978)
Holly	TA, F	0·2–1·9**	Smith and Brand (1983)
Honeysuckle	TA, F	0·3–1·3*	Grigal and Ohmann (1977)
	TA, F	0·6–1·3*	Harrington *et al.* (1989)
Hornbeam	A	≥ 12·7	Hahn (1984), Phillips (1981), Smith (1985)
	F		Other hardwoods

TABLE 1. *(continued)*

Species	Part	Diameter# range (CM)	Source
Juniper	TA, F	0·8–2·9**	Smith and Brand (1983)
	A	<12·7	Hahn (1984), Phillips (1981), Smith (1985)
	A	≥12·7	Hahn (1984), Phillips (1981), Smith (1985)
Linden	A	<12·7	Hahn (1984), Phillips (1981), Smith (1985)
	A	≥12·7	Hahn (1984), Phillips (1981), Smith (1985)
	F		Other hardwoods
Maple	TA, F	0·3–4·3*	Roussopoulos and Loomis (1979)
	TA	≥10	Tritton and Hornbeck (1982)
	TA	1–30	Tritton and Hornbeck (1982)
	TA, F	1·1–40·5	Ker (1980)
	TA	2·5–66·0	Tritton and Hornbeck (1982)
	TAB	2·5–66·0	Wenger (1984)
	TA	5·1–50·8	Tritton and Hornbeck (1982)
	F	7–24	Tritton and Hornbeck (1982)
	R	***	Whittaker and Marks (1975)
Mockorange	TA, F	0·5–2·9**	Smith and Brand (1983)
Mountainash	TA, F	0·5–3·8*	Roussopoulos and Loomis (1979)
	A	<12·7	Hahn (1984), Phillips (1981), Smith (1985)
	F		Other hardwoods
Oak	TA, F	0·2–4·0**	Smith and Brand (1983)
Pine	TAB, F, R	2·1–5·2	Attiwill and Ovington (1968)
	F	2·9–31·7	Stanek and State (1978)
Raspberry	TA, F	0·3–1·4**	Smith and Brand (1983)
Rhododendron	TA, F	0·3–1·1	Telfer (1969)
Rose	TA, F	0·2–1·2**	Smith and Brand (1983)
Spirea	TA, F	0·1–1·3**	Smith and Brand (1983)
Spruce	TA, F	0·5–3·3*	Roussopoulos and Loomis (1979)
	TAB, F, R	1–3*	Czapowskj *et al.* (1985)
	TAB, F, R	1–15	Czapowskj *et al.* (1985)
	TAB, R	1·5–17·7	Ker and van Raalte (1981)
	TA, F	1·1–32·6	Harding and Grigal (1985)
	TA	2–32	Tritton and Hornbeck (1982)
	TA	2·5–66·0	Tritton and Hornbeck (1982)
	TAB	2·5–66·0	Wenger (1984)
	R	14·3–24·5	Stanek and State (1978)
	TA, F	≥12·7	Jokela *et al.* (1986)
Viburnum	TA, F	0·3–1·6*	Smith and Brand (1983), Telfer (1969)
	TA, F	0·3–3·1**	Smith and Brand (1983), Telfer (1969)
Willow	TA, F	0·3–3·0*	Connolly (1981)
	TA, F	0·5–3·0*	Roussopoulos and Loomis (1979)
	TA, F	0·8–3·8*	Ohmann *et al*,. (1976)
	A	<12·7	Hahn (1984), Phillips (1981), Smith (1985)
	F		Other hardwoods

TABLE 1. *(continued)*

Species	Part	Diameter# range (CM)	Source
Other hardwoods	F	<12	Tritton and Hornbeck (1982)
	A	2·5–12·4	Phillips (1981)
	TA, F	2·5–15·2	Tritton and Hornbeck (1982)
	TA	2·5–25·0	Monteith (1979)
	TA	2·5–25·4	Tritton and Hornbeck (1982)
	TA, F	>10	Tritton and Hornbeck (1982)
Other softwoods	A	<12·7	Hahn (1984), Phillips (1981), Smith (1985)
	TA	2·5–55·0	Monteith (1979)

TAB: Total above and below ground biomass. TA: Total above ground biomass. A: Above ground biomass except foliage. F: Foliage biomass. R: Root biomass. # Indicating dbh except figures with asterisk. * Diameter at 15 cm above ground. ** Diameter at ground level. *** From seedlings to trees.

assumed to be 100% of above ground biomass for multibranched, dense and low shrubs 1 m or less in height or 1 cm or less in diameter at ground level, 50% of above ground biomass for shrubs and small trees 3 m or less in height or 5 cm or less in diameter at ground level (or 2·5 cm or less in dbh), and 25% of above ground biomass for larger trees. The ratios of below:above ground biomass may not be consistent for different species and different growing sites. The use of ratios generated from forest areas could underestimate root biomass of plant species in the study area, because the ratios are higher for plants in a well-lit, open environment (Maggs, 1960; Whittaker, 1962). However, irrigation and fertilization in residential yards might decrease the ratios of below:above ground biomass. The ratios tend to be higher in species of drier soils (Bray, 1863). No studies concerned with root biomass in urban areas were found.

2.2.2. *Grass*

Total carbon in grass was determined using the maximum live above and below ground biomasses of grass found in bimonthly sampling (please see section 2.3.2 for detailed methods).

2.2.3. *Other herbaceous plants*

During August 1993, a sample of above ground herbaceous biomass was taken. The sample included 12 common species and a total of 32 plant individuals. Before harvesting, duplicate crown widths of the plants were individually measured, at 90° to each other using a 45 cm steel ruler in order to calculate plant cover. The number of samples in each species was less than desired due to limited permission from homeowners. The sampling was not random, because it had to be done without disturbing the appearance of gardens. However, plant individuals with relatively representative sizes in height and cover were sampled where possible.

The samples were individually bagged and oven-dried at 65°C for 24 h, and then weighed to an accuracy of 0·1 g. To estimate below ground biomass, the above ground dry weight was divided by 1·39 (Körner and Renhardt, 1987). Above and below ground dry weight biomass was converted to a carbon estimate by multiplying by 0·45 (Olson, 1970; Ajtay *et al*., 1979). Carbon per unit cover was applied for each species inventoried per residential unit. The carbon storage for species not sampled was determined by averaging that from plants with similar growth forms and sizes.

2.2.4. *Soil*

Total organic and inorganic carbon in the soils was estimated by averaging carbon storage measured in late April (beginning of the growing season) and early September (end of the growing season). One sample was randomly taken from 24 residences, 12 in each study block. Six composite samples were made in each block by mixing samples of two residential units through random selection. The sampling positions were determined from random numbers applied to a 1 m^2 grid laid over each property. The soils were cored to a depth of 60 cm using a split tube sampler of 5·1 cm in diameter, after live, above ground grass was removed. Sampling frequency and sample size were chosen as a compromise between the competing concerns for data reliability and the availability of money and labor.

Soil carbon in the samples was analyzed by TEI Analytical, Inc., Chicago, IL. Organic carbon was analyzed by a TOC Analyzer (Dohrman Model DC-80, Santa Clara, CA) and inorganic carbon, by the gravimetric method (Allison and Moodie, 1976). Standard samples (potassium acid phthalate ($KHC_8H_4O_4$) for analysis of organic carbon and sodium carbonate (Na_2CO_3) for analysis of inorganic carbon) were run four times to test the accuracy of analysis, before and after the carbon content in the soil samples was measured. The variation of the test relative to the standard samples averaged $\pm$1·3% for organic carbon and $\pm$0·4% for inorganic carbon.

2.3. Measurement of annual carbon uptake

2.3.1. *Trees and shrubs*

Estimates of annual carbon uptake by trees and shrubs were generated by calculating the annual change in biomass between the study year (1992) and the previous year. The previous year's dbh was calculated using the average radial growth rate for the last 5 years, measured through stem core sampling. This dbh was used to calculate the previous year's biomass through application of biomass equations. To estimate annual carbon uptake, the previous year's biomass was subtracted from the present year's biomass. If the biomass equations used merchantable height or total height as independent variables, the previous year's merchantable or total height was estimated by regression equations generated from dbh and heights taken from the study area. The amount of carbon in foliage was subtracted from the estimates generated.

The growth rate was estimated by measuring ring width from duplicate increment cores taken at 90° or 180° to each other. We attempted to take duplicate cores at 180°, but they were often taken at 90° because of the cross-sectional shape of trunks or to avoid obstacles (such as fences). After dbh was measured, the trunk at 1·3 m above the ground level was cored straight towards its center. The cores were put individually into plastic straws, and labelled with waterproof marker. Approximately 170 healthy trees and shrubs (except those that were small or unhealthy) were cored in 26 residential lots. Prior to processing, the cores were stored in a refrigerator (3–5°C) to prevent mould growth. The stem cores were glued into grooved wooden mounts with the end grains of the cores aligned vertically and with score lines by increment borer running along the edges of the mounts, so that individual ring boundaries were visible on the surfaces of the cores. The cores of trees with diffuse porous rings (for example, maples and crab apples) were imbedded into deeply grooved mounts. After the glue had dried, they were cut in half across their width so that the ring structure was clear when an incandescent light was shone onto the surface. The mounted cores were sanded by hand with a series of sandpaper grits (100, 220, and 400) to make the surface smooth. The annual ring widths of the prepared cores were measured using a Digital Positiometer at the USDA Forest Service laboratory (Durham, NH). The meter was composed of a sliding-stage micrometer interfaced with a microcomputer. Individual rings were measured to an accuracy of 0·01 mm by moving the samples on the sliding stage under a binocular microscope set up with a crosshair in one ocular lens. Measurements were recorded in the microcomputer by pressing a button when the crosshair was lined up on the ring boundaries of successive rings. Using an average growth rate from two increment cores per tree might not represent a growth rate at all depending on the directions of trunk for each tree. However, no more than two cores could be sampled from each tree, due to potential damage and limited permission.

Leaf fall will return carbon annually to the atmosphere through collection and decomposition. Therefore, all foliage carbon was subtracted from the deciduous species, and 25% of foliage carbon was subtracted for the evergreen species, assuming three-year leaf retention (Dirr, 1977; Rowntree and Nowak, 1991).

Annual carbon sequestration by those trees and shrubs that were not cored was estimated using age from regression equations and mean growth rate for the last 5 years. In generating the regression equations to estimate age, an iterative linear and non-linear approach was used to determine the most appropriate parameters for each species. The mean growth rate of some species was measured from pruned stems and used to estimate annual carbon uptake. Growth of small shrubs, for which extrapolation from core sample data is impossible, was estimated by applying an average annual diameter growth rate of 0·075 cm at ground level. This growth rate was obtained from a correction to the mean growth rate of shrubs in forest areas (0·05 cm) taken from existing data (Whittaker, 1962; Whittaker and Woodwell, 1968; Whittaker and Marks, 1975). The correction was determined through comparison of annual growth of several common species measured in the Chicago study area with that of the same species growing in forest areas (Harrington *et al.*, 1989).

2.3.2. *Grass*

The measurement of annual carbon accumulation for grass requires separate consideration of three parts: mown part, stubble, and live roots. Mown parts will not contribute to net carbon accumulation, but rather to carbon output (please see section 2.4). Annual carbon uptake of stubble and live roots was calculated using the following formula (Milner and Hughes, 1968; Falk, 1976 and 1980):

$$\text{Net carbon} = C\text{max}_{(s)}T_s + C\text{max}_{(r)}T_r$$

where: $C\text{max}_{(s)}$ = maximum carbon in live stubble, T_s = turnover rate of live stubble, $C\text{max}_{(r)}$ = maximum carbon in live roots, and T_r = turnover rate of live roots. Turnover rate (T) was calculated from the ratio of annual growth to total live stubble (or live root) mass (Dahlman and Kucera, 1965).

Sampling occurred during the first week of November and December 1992, and March, May, July and September 1993. During January and February, sampling was impossible due to snow cover and frozen soil. Only above ground material was sampled during the first week of December. Composite samples of three cores were taken from 17 residential units (30% of total residential units) using a split tube sampler of 5·1 cm diameter and 30·5 cm length. The samples were individually bagged and labelled. Prior to processing, the samples were stored in a refrigerator (-1–1°C) to prevent new growth and decomposition. Residential units and sampling positions were selected randomly as detailed previously in this paper.

The separation of the roots from soil was made by hand washing and with a sieve with 0·2 mm^2 mesh size. Live and dead roots were separated by testing for flotation, root color, and elasticity.

Dry weight was measured using an electronic balance, with an accuracy of 0·001 g. The samples were dried at 65°C for 24 h in an oven. The dry weight of grass roots was converted to ash-free dry weight by burning at 600°C for six h in a muffle furnace.

The dry weight of live stubble and the ash-free weight of live roots were converted to a carbon estimate by multiplying by 0·4269, since carbon content in grass averaged 42·69% of the dry weight. This carbon content was determined in the Soil, Water and Plant Analysis Laboratory at the University of Arizona from a total of 35 oven-dried grass samples (including both live and dead parts). The samples used in the carbon analysis were randomly taken from well mixed samples of live and dead, above and below ground grass stored during the grass and soil sampling from October 1992 to July 1993. The carbon content was determined by grinding the samples through a 40 mesh screen using a Wiley Mill, then subjecting them to high-temperature combustion (about 1000°C) in a Nitrogen Analyzer 1500. To test the accuracy of measurements, acetanilide standard with a known carbon content (71·09%) was run five times, and fenantrene additivato standard (93·32%) and tomato leaves standard (39·0%) were run once, respectively, prior to the carbon analysis of the samples. The variation of the test ranged within a maximum of $\pm 0{\cdot}23\%$, compared to the known carbon contents of the standards.

2.3.3. *Soil*

Annual soil–carbon accumulation was calculated by applying the formula used to determine annual carbon uptake by live grass. Variables in this application were dead above ground and below ground grass instead of live stubble and roots. The amount of annual decomposition was estimated through calculation of a decomposition constant (k), based on a method employed by Jenny *et al.* (1949) and Dahlman and Kucera (1965).

Soils were sampled from three different soil cover types: grass and trees, shrubs, and herbaceous plants. The method and processing of samples as well as the frequency and sample size were the same as that of grass. Nine samples of soils under shrubs, and six under herbaceous plants were taken without differentiation between the two blocks, because their areas were relatively small and accessibility for sampling was limited.

To identify carbon input to and carbon output from soil by all dead organic materials, the samples were washed and differentiated into live roots, dead materials, and other organic residue (small debris for which the identification of form was difficult). Dry weights of all the materials were converted to ash-free weight using the method mentioned previously in this paper. The ash-free weight of dead materials was again converted to a carbon estimate by multiplying by 0·4269 for grass, 0·45 for herbaceous plants and 0·5 for shrubs, trees and other organic residues (Kimura, 1963; Ajtay *et al.*, 1979).

Several limitations were found when obtaining deep soil samples. In addition to infrastructure (such as gas lines, water pipes, and drainage lines) in unknown below ground positions, it is often impossible to sample without using instruments of considerable size such as an hydraulic jack. This can disturb the grass and may be unacceptable in a residential area.

2.4. LANDSCAPE MANAGEMENT

Landscape management activities related to carbon release to the atmosphere were identified through interviews with homeowners. A total of 28 homeowners (or landscape managers) were interviewed. The questionnaire recorded data about the monthly frequency of mowing, the type of mower used, pruning time of trees and shrubs, the quantity of fuel consumed for mowing and pruning, and the treatment of raked, mown and pruned materials. It also included questions concerning the method, frequency and quantity of irrigation and fertilization, and the type and quantity of herbicides and pesticides applied.

The levels of mowing and pruning reported were corrected by values generated through actual measurement. To estimate carbon removal due to pruning, the pruning schedule of residents was identified at interview and through several additional letters sent in early spring and summer. Residents were asked to leave all pruned materials for collection. These were collected from 11 residential units in study block 1 and 10 residences in block 2. Fresh weight was weighed with scales then three to five random subsamples were placed into an aluminium container, 2400 cm^3 in size, and dried at 65°C for 48 h in a drying oven to obtain dry weight. The dry weight was converted to a carbon estimate by multiplying by 0·5. To estimate the amount of mowing, 1 m^2 quadrats were staked out randomly in the front or back yards of 26 residential units (50% of the total). Grass within the quadrats was clipped biweekly from October 1992 to September 1993 (except during winter when residents do not mow) and dry weight was measured then converted to carbon value, as detailed previously.

The approximate amount of water consumed through irrigation was calculated by measuring the water flow rate from a hose in a residential yard of the study area. Consumptive use was calculated using this value, and irrigation frequency and duration for each residence was recorded.

The amount of direct and indirect energy consumption by each type of landscape maintenance was converted to a Btu factor based on Pitt's (1984) study. Energy units were converted to carbon estimates by multiplying mBtu values by 6·1848.

3. Results and discussion

3.1. INTRODUCTORY INFORMATION

3.1.1. *Climate*

The climate of Chicago is a moist, mid-continental type with considerable seasonal variation in precipitation and temperature. Mean annual precipitation and temperature during 30 years from 1962–1991 were 898 mm and 9·4°C, respectively (NOAA, 1991). Precipitation is at a maximum in summer and at a minimum in winter. Drought and long rainy periods are rare in the city (Cutler, 1976).

3.1.2. *Soil*

The soil parent materials found in the areas around Chicago are primarily outwash (material deposited by glacial water), unsorted glacial till (material deposited by glacial ice), and fine-grained loess (silty wind deposit) (Fehrenbacher *et al*., 1967). It is known that clay and silt are primary components of soils in most of Chicago and the suburban areas. In the study area, clay was predominant lower than 20–30 cm below ground level.

3.1.3. *Vegetation*

In the study area, the total number of tree species was 34 for block 1 and 21 for block 2. The number of shrub species was 35 in block 1 and 24 in block 2. Main tree species found were maples (*Acer negundo, A. saccharinum*, *A. platanoides*), elms (*Ulmus americana, U. pumila*), mulberry (*Morus alba*), crabapple (*Malus* spp.), cherry (Prunus spp.) and spruces (*Picea pungens, P. abies, P. glauca*). Major shrub species were yew (*Taxus baccata*), honeysuckle (*Lonicera* spp.), privet (*Ligustrum* spp.), buckthorn (*Rhamnus cathartica, R. frangula*), dogwood (*Cornus* spp.), rose (*Rosa* spp.) and juniper (*Juniperus communis*). The number of tree and shrub individuals averaged about 119·4 per residential unit in block 1 and 25·5 in block 2.

Analysis of the dbh distribution of trees revealed that the tree population in the study area was quite young. Trees with dbh size of less than 30 cm accounted for 80% and 90% of all trees surveyed in block 1 and block 2, respectively. Dorney *et al*. (1984) and Nowak (1991) also found that the majority of urban trees had small diameters in Shorewood, Wisconsin and Oakland, California. For shrubs, distributions of diameter at 15 cm above ground level were similar between blocks, with about 80% of all shrubs in the 1·1–9·0 cm diameter size class.

Tree–shrub cover averaged about 41·6% per residential unit in block 1, and 13·1% in block 2, 3 times less than in block 1. McPherson *et al*. (1993) found that the average tree cover in residential areas of Chicago ranged from 7% for four-family (or more) residential lots to 15% for one- to three-family residential lots. Compared to their study, block 1 has a high level of tree cover, while block 2 is more characteristic of the city-wide average. Survey of tree planting potential revealed that tree cover can be increased by 77·9% of present tree cover in block 1, and 2·1 times present tree cover in block 2. The planting potential only included trees 3 m or more in mature crown diameter which can be grown without interfering with present above ground utility lines.

3.1.4. *Areal distribution of land cover types*

The total lot area ranged from approximately 583–2428 m^2, with an average of 978·9 m^2 in study block 1. Lot area ranged from about 308–1146 m^2, with mean 503·1 m^2 in block 2. The mean lot area was approximately twice as large in block 1 as in block 2. Percentages of land cover types per residence in block 1 averaged 37·7% for grass, 21·7% for building and garage, 18·7% for paving, 6·8% for soil (with herbaceous plants) and mulch and 15·1% for other pervious surfaces. In block 2, the percentages of land cover types averaged 36·4% for building and garage, 26·9% for grass, 25·9% for paving, 5·6% for soil and mulch and 5·2% for other pervious surfaces. While impervious surfaces in block 1 averaged about 40% of lot area, they averaged 62% of lot area in block 2. This reflects the higher building densities in block 2.

TABLE 2. Biomass equations generated from urban trees*

Species	Equations (kg)	R^2	N	dbh (cm)
Elm spp.	$1n\ TA = -1{\cdot}8945 + 2{\cdot}2822\ \ln dbh$	0·903	7	45·7–96·5
Silver maple	$TA = -3033 + 76{\cdot}02\ dbh$	0·929	9	53·3–99·1
Ash spp.	$F = (-20128 + 10287\ CR)/1000$	0·646	10	15·4–29·6
Honeylocust	$F = (907 + 55{\cdot}32\ V)/1000$	0·713	10	11·4–28·4
Norway maple	$F = (-37294 + 25581\ CR\text{-}2098\ dbh + 6483\ BH)/1000$	0·904	14	13·2–39·5
Hardwoods	$\ln TA = -3{\cdot}5618 + 2{\cdot}6645\ \ln dbh$	0·945	24	25·4–99·1
	$F = (-10375 + 8152\ CR\text{-}28690\ SC)/1000$	0·556	54	11·4–56·2

* TA: Total above ground biomass. F: Foliage biomass. CR: Crown radius (m). V: Crown volume (m^3). BH: Bole height (m). SC: Shading coefficient (McPherson, 1984). Hardwoods: Equation for TA was generated from elm species, silver maple, Norway maple, ash species, linden, and sugar maple. Equation for F was generated from Norway maple, American elm, green ash, honeylocust and hackberry.

3.2. TOTAL CARBON STORAGE AND ANNUAL CARBON UPTAKE

3.2.1. *Tree and shrubs*

Existing total carbon storage in trees and shrubs averaged approximately 3·42 kg/m^2 of lot area in study block 1, with 3249·5 kg per residential unit. Annual carbon uptake was, on average, approximately 0·22 kg/m^2, with 228 kg per residential unit. In block 2, average total carbon storage was 1·03 kg/m^2 and 513·5 kg per residence, and the mean annual carbon uptake was 0·07 kg/m^2 and 39·1 kg per residence. Shrubs accounted for about 10–11% of the total carbon storage and 21–25% of total annual carbon uptake. Foliage carbon comprised approximately 2–3% of the total carbon storage for trees and 10–12% for shrubs.

The amount of total carbon storage and annual carbon uptake per m^2 of lot area was three times greater in study block 1 than in block 2. Nowak (1993) found that carbon storage by trees alone in residential areas within Oakland, California was 0·98 kg/m^2. In this study, carbon storage by trees and shrubs in block 2 was slightly higher.

The carbon storage was estimated using biomass equations from forest areas. The use of biomass equations might underestimate biomass of trees and shrubs in the study area. An urban tree with the same dbh or height as a forest tree could have wider crown due to less competition, or more irrigation, and fertilization, and have a higher biomass. However, poor rooting conditions, air pollution, heat and severe pruning might lower biomass accumulation in an urban tree.

Above ground biomass (including foliage biomass) of elm species, silver maple, Norway maple, ash species and linden species, and foliage biomass of honeylocust were corrected using biomass equations generated from urban trees. The biomass equations from urban trees are included in Table 2. The application of corrected above ground biomass equations decreased total carbon of silver and Norway maples by approximately 35–45% (according to dbh ranges), compared to the equations for forest trees, while it increased total carbon of the other species by about 5–50%. The use of the corrected above ground biomass equations increased total carbon storage by 1·9% in study block 1, while it decreased the carbon storage by 1·8% in block 2. The application of corrected foliage biomass equations increased total leaf carbon by approximately 20%.

Annual carbon uptake was estimated based on the growth rate of trees determined by dbh measured from cored samples. Mean dbh growth rates were 1·09 cm per year ($N=118$) for hardwood trees (e.g. maple, elm, honeylocust, mulberry, crabapple, ash), and 0·51 cm per year ($N=17$) for softwood trees (spruce, Scots pine and Arborvitae). Dbh growth rates were 0·42 cm per year ($N=10$) for hardwood shrubs (buckthorn, dogwood and lilac), and 0·27 cm per year ($N=7$) for softwood shrubs (yew and juniper). Smith and Shifley (1984) found that the mean dbh growth rate of hardwood trees growing in forest areas of Indiana and Illinois was 0·4 cm per year. Hardwood trees in this study area are growing over twice as fast as are the trees in forest areas. Of the species sampled, cottonwood and mulberry had the greatest growth rates. Silver maple, ash species, honeylocust, and tree of heaven also showed relatively high growth rates. Fast-growing species may sequester more carbon per year.

3.2.2. *Grass*

Major species of grass planted in the study area were Kentucky bluegrass and fescue. The maximum carbon storage in grass stubble and roots occurred in July for both study blocks, while the minimum occurred in November for block 1 and in March for block 2. The changes in carbon storage of grass followed a typical pattern. It increased during the growing season from spring to summer and declined steadily from fall to winter. Other published studies (Dahlman and Kucera, 1965; Falk, 1976, 1980) also found that below ground biomass was at a maximum in the summer and a minimum in the early spring.

Maximum total carbon storage in grass was 186·07 g/m^2 (95% confidence interval of stubble: 71·43 ± 12·32, root: 114·64 ± 32·35) in block 1, and 246·62 g/m^2 (stubble: 91·02 ± 19·30, root: 155·60 ± 45·02) in block 2. Annual carbon uptake was 88·74 g/m^2 (stubble: 40·64, root: 48·1) in block 1, and 88·32 g/m^2 (stubble: 42·25 root: 46·07) in block 2. There was no difference (95% confidence level) in the total carbon storage in stubble and roots of grass between block 1 and block 2. For both blocks, the total carbon storage averaged 221·10 g/m^2 ($n=17$) (stubble: 82·18 ± 12·04, root: 138·92 ± 28·22), and annual carbon uptake was 87·90 g/m^2 (stubble: 40·81, root: 47·09). The turnover rate was approximately 2 years for stubble carbon and approximately 2·9 years for root carbon. In other words, about 50% of the total stubble carbon and 34% of the total root carbon would be replaced each year, even though certain segments might be exchanged more or less rapidly. The turnover rates were similar to those reported for a suburban lawn area in California (Falk, 1976). Falk (1976 and 1980) found that the range for lawn net primary production might be about 1000–1700 g/m^2 per year in California and Maryland lawns. Net annual primary production in temperate grasslands ranges from 100–1500 g/m^2 (Leith, 1975). In this study, when the carbon values were converted to dry weights, the annual net primary production of all the live and dead grass portions, including mown parts, was approximately 595·22 g/m^2 for block 1, and 500·68 g/m^2 for block 2, falling well under the range reported by Falk.

The analysis of carbon content in grass showed that an average content ($N=35$) was 42·69% of dry weight biomass. To identify carbon contents in different grass portions, the analysis was differentiated into live, dead, above- and below-ground materials. There was no significant difference between them, at a 95% confidence level.

3.2.3. *Other herbaceous plants*

Main herbaceous plant species in the study area were peony, lily, garden phlox, hosta, touch-me-not, and black-eyed Susan. Average carbon for the plants sampled ($N=32$)

was about $0{\cdot}18 \pm 0{\cdot}06$ kg/m^2 (95% confidence interval) of cover. Total carbon storage in the plants averaged 2·15 kg per residential unit in study block 1, with a minimum of 0 kg and a maximum of 7·29 kg. In block 2, the total carbon storage ranged from 0–3·28 kg, with a mean of 1·08 kg per residence. There was no difference (95% confidence level) in average carbon storage per residence between the two blocks. In both blocks, total carbon storage averaged $1{\cdot}60 \pm 0{\cdot}74$ kg per residential unit. Carbon storage in herbaceous plants was extremely small, compared to that in trees and shrubs.

3.2.4. *Soil*

Total carbon storage in soils (to the depth of 60 cm) of study block 1 averaged $4{\cdot}15 \pm 1{\cdot}81$ kg/m^2 (95% confidence interval) for inorganic carbon and $18{\cdot}48 \pm 2{\cdot}64$ kg/m^2 for organic carbon (a total of 22·63). In block 2, the inorganic carbon was $4{\cdot}44 \pm 1{\cdot}72$ kg/m^2 and the organic carbon was $14{\cdot}05 \pm 3{\cdot}76$ kg/m^2 (a total of 18·49). There was no significant difference (95% confidence level) in the total carbon storage between the two blocks. In both blocks, the soil carbon storage averaged $4{\cdot}30 \pm 1{\cdot}15$ kg/m^2 for the inorganic carbon and $16{\cdot}27 \pm 2{\cdot}30$ kg/m^2 ($n = 24$) for the organic carbon. Birdsey (1992) predicted that organic soil carbon storage ranges from 13–16 kg/m^2 for forest areas in North Central and Northeast U.S.A. Organic carbon storage in this urban study area was slightly greater than in those forest areas.

Total maximum carbon input to soils of grass areas by dead grass and tree materials (including other tiny organic residues for which a form could not be identified) occurred during early summer with $1{\cdot}40 \pm 0{\cdot}38$ kg/m^2 in block 1 and $1{\cdot}31 \pm 0{\cdot}22$ kg/m^2 in block 2 (no significant difference). In both blocks, the total maximum carbon input averaged $1{\cdot}36 \pm 0{\cdot}19$ kg/m^2 ($N = 17$) while the total minimum carbon input during the winter season was $0{\cdot}68 \pm 0{\cdot}18$ kg/m^2. Soil carbon input under shrub cover showed, in the maximum and minimum occurrence, the phenomenon similar to that in grass areas. Total maximum carbon input by shrub detritus, including mulches, occurred in July with $1{\cdot}36 \pm 0{\cdot}60$ kg/m^2 ($N = 9$) in both study blocks, and the minimum occurred in November with $0{\cdot}59 \pm 0{\cdot}19$ kg/m^2. In those soil areas with herbaceous plants, carbon input by dead materials, including organic fertilizers, reached a maximum in May with $1{\cdot}30 \pm 0{\cdot}59$ kg/m^2 ($N = 6$) and a minimum in September with $0{\cdot}99 \pm 0{\cdot}39$ kg/m^2. The occurrence of a maximum in May can be attributed largely to the input of organic debris (such as wood chips). Considering only the dead roots of herbaceous plants, the maximum carbon input occurred in November, when most plants are dying. The carbon input by all dead material was variable across different residential units, as indicated by the confidence interval. This might be due to the small number of samples, the uneven distribution of vegetation cover and differences in management input such as mowing, fertilization and litter collection.

Annual carbon input to soils of grass areas from dead material was 0·67 kg/m^2, and the amount of annual decomposition was 0·45 kg/m^2 (decomposition constant $k = 0{\cdot}33$) in both blocks. In soil areas covered with shrubs and herbaceous plants, the annual carbon input was 0·77 kg/m^2 and 0·30 kg/m^2, respectively, for both blocks. It is thought that the highest annual carbon input in shrub-covered soils was caused by input of mulches and incomplete litter removal. Decomposition from the soils was 0·49 kg/m^2 ($k = 0{\cdot}36$) for shrub areas and 0·24 kg/m^2 ($k = 0{\cdot}19$) for herbaceous plant areas. Low decomposition in soils with herbaceous plants might be due to the impact of mulches or organic fertilizers that decompose slowly. In this study, it may be more exact to express the annual decomposition as annual loss, because the term “decomposition” also includes losses due to wind, runoff, or leaching. For comparison, it has been estimated that the amount of carbon evolution from several temperate oak-dominated forest areas in the United States ranges from approximately 410–1000 g/m^2 per year (Edwards *et al.*, 1989). The amount of decomposition in the study area falls in the lower range of this estimate for forest areas.

3.3. CARBON RELEASE BY LANDSCAPE MANAGEMENT

There was no difference (95% confidence level) between the two blocks in the intensities of landscape maintenance concerned with mowing, pruning, irrigation and fertilization. The annual frequency of mowing averaged about 20 ± 2 (95% confidence interval) per residence in the study blocks. The power source for mowing was predominantly gasoline. Average annual gasoline consumption per residential unit was approximately $0{\cdot}072 \pm 0{\cdot}029$ l/m^2. Conversion of consumed gasoline to annual energy (Btu) and carbon generation yielded 2·36 kBtu/m^2 and 14·58 g/m^2 of carbon. Parker (1982) estimated that annual gasoline consumption for lawns amounted to 1·3 kBtu/m^2 in a Florida residential landscape. The Btu consumption by mowing in the study area appears to be nearly twice Parker's estimate. The amount of mowing was greatest in May for both blocks, and declined continuously until the end of the growing season. Total annual carbon output by mowing averaged about 113·2 g/m^2 in the study blocks. Indirect carbon release due to manufacture of a mower was 195·0 kg, assuming that a three horsepower mower was used. Multiplying the carbon by the number of residential units (one mower per residence) generated total 3704·1 kg carbon in block 1, and 4678·8 kg carbon in block 2 (excluding residential units using hand or electric mowers).

On average, homeowners in the study blocks pruned trees and shrubs once or twice each year. The pruning was done by hand- or electric pruners, usually in spring and summer. Annual mean carbon output by pruning was $0{\cdot}081 \pm 0{\cdot}037$ kg/m^2 of tree and shrub cover for both blocks. In block 1, the annual carbon output by pruning per residential unit was in the range of 2·6–112·7 kg, with an average of 31·2 kg. The carbon output in block 2 with lower vegetation cover ranged from 0–29·3 kg, with an average of 4·6 kg.

The annual frequency of irrigation averaged 28 ± 6 with a mean duration per event of about $1{\cdot}6 \pm 0{\cdot}7$. Sprinkler systems were mostly hand-moved or buried–automatic in both blocks. Multiplying the irrigation frequency by the duration for each residential unit generated an annual mean total irrigation duration of 41·9 h per residential yard. Based on the annual total duration and estimated water flow rate of about 1703 l/h, annual average water consumption by irrigation was 186·9 l/m^2 of grass area in block 1 and 473·9 l/m^2 in block 2. Residents indicated that most of the water was used to maintain grass. On the other hand, application of a water budget formula, using annual evapo-transpiration for Chicago (Bennett and Hazinski, 1992), resulted in an annual irrigation requirement of 816·6 l/m^2, which is approximately two to four times higher than the values calculated here. Part of this difference, if not all, would be due to the effects of natural sources of soil moisture (such as snowmelt and rainfall) on evapo-transpiration. Annual average water consumption, estimated using water flow rate, was converted to 0·22 kBtu/m^2 and 1·36 g/m^2 carbon in block 1, and 0·54 kBtu/m^2 and 3·33 g/m^2 carbon in block 2. The calculated Btu consumption is about two to four times greater than Pitt's (1984) estimate of 0·12 kBtu required for irrigation of turf in a temperate residential setting. Indirect carbon emission through both manufacture

TABLE 3. Annual carbon release by landscape management activities in study blocks*

Block	Mowing				Pruning		Irrigation		Fertilization	
	Gasoline		Mown grass							
	g/m²	Total	g/m²	Total	g/m²	Total	g/m²	Total	g/m²	Total
1	14·6	117·0	113·2	952·0	81·0	729·2	1·4	11·3	10·5	88·5
2	14·6	61·5	113·2	478·0	81·0	164·1	3·3	14·0	10·5	44·4

* The unit of total carbon release is kg. The unit area (m^2) of carbon release for pruning is cover of trees and shrubs. Total carbon release through mowing, irrigation, and fertilization was calculated multiplying carbon (g/m^2) of grass area by total grass area of each block (block 1: 8407·7 m^2, block 2: 4221·3 m^2). Calculation of carbon release by gasoline consumption in mowing excluded on the residential unit using a hand-operated mower in study block 1. Total carbon output by pruning was generated multiplying carbon (g) per m^2 of tree and shrub cover by the total cover (block 1: 9002·4 m^2, block 2: 2025·8 m^2). Pruning did not include direct carbon release through electricity use due to difficulty of quantification.

and installation of the sprinkler systems totalled 211·3 kg in block 1 (9·6 kg per residential unit), and 98·2 kg in block 2 (3·5 kg per residential unit).

Grass areas in the study blocks were, on average, fertilized approximately twice annually. Annual fertilizer input to grass was $36{\cdot}2 \pm 15{\cdot}7$ g/m^2 (the fertilizer input to other plants was negligible). Interview responses indicated that the nutrient ratio of fertilizers was 3:1:1 for N:P:K. Based on the fertilizer input and nutrient ratio, annual Btu consumption and carbon release were 1·70 $kBtu/m^2$ and 10·52 g/m^2 carbon in the study blocks. The Btu consumption by fertilization in the study area is slightly lower than Pitt's (1984) estimate of 1·85 $kBtu/m^2$ of lawn.

The interviews showed that use of herbicide, pesticide, and fungicide was negligible in the study area. Small amounts of herbicide were used infrequently with fertilizers.

Table 3 summarizes the total annual carbon release by landscape management activities in each study block. The figures do not include the indirect release of carbon required for manufacture of mowers and manufacture and installation of sprinkler systems due to the difficulty in converting total quantities to annual releases. Total annual carbon release from grass areas due to mowing, watering, and fertilization was about 1168·8 kg in block 1 and 597·9 kg in block 2 (0·14 kg/m^2 of grass area for both blocks). The greatest carbon release came from mowing (gasoline consumption and carbon output of mown grass).

Mulching with wood chips contributes to carbon input in residential landscapes. The mean area mulched was 6·4 m^2 per residential unit in each study block. Carbon input by mulching averaged 1·5 kg per residential unit. Extrapolation of the mean carbon input to all residences in the study area resulted in a total of 33·2 kg in block 1 and 42·3 kg in block 2.

3.4. ANNUAL CARBON BUDGETS

This section will discuss how much carbon flowed in to and out of greenspaces in the study blocks, and how much carbon was actually sequestered annually. Figures 1 and 2 summarize annual carbon inputs and outputs, and annual net carbon sequestration for each study block. The carbon budgets were calculated by extrapolating average carbon per residence or per unit area to the total number of residential units or total

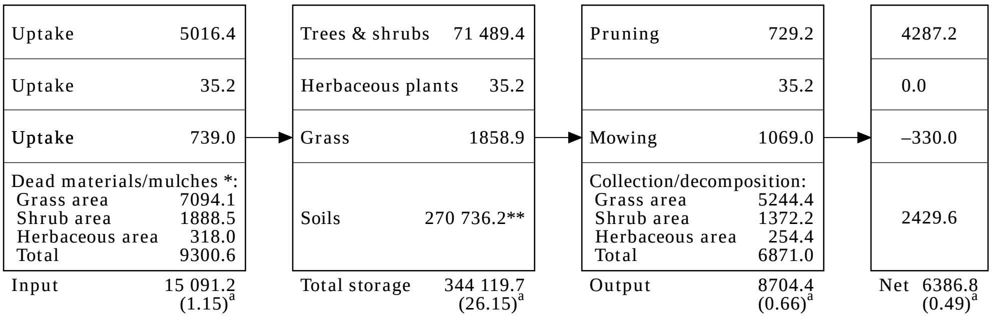

Figure 1. Annual carbon budget for greenspace of study block 1 (kg).
* Includes fallen leaves from trees and shrubs
** Soil carbon storage is a sum of inorganic and organic carbon
[a] The bold figures in parentheses indicate carbon/m^2 of greenspace (13 161·7 m^2)

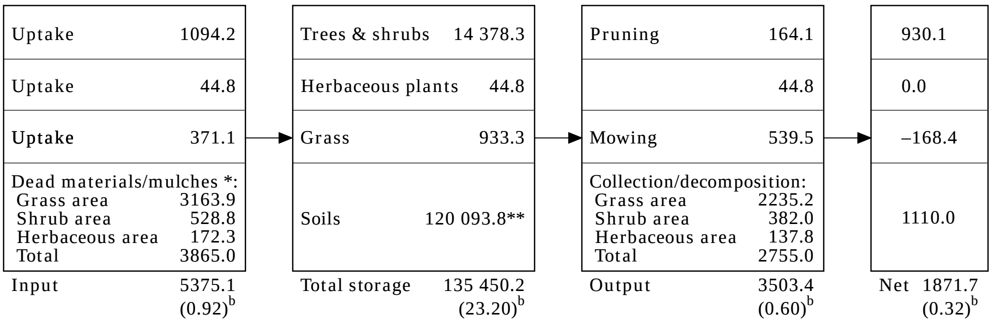

Figure 2. Annual carbon budget for greenspace of study block 2(kg).
* Includes fallen leaves from trees and shrubs
** Soil carbon is a sum of inorganic and organic carbon
[b] The bold figures in parentheses indicate carbon/m^2 of greenspace (5838·3 m^2)

areas of land cover types concerned, based on the data discussed in the previous sections. Figures 3 and 4 show, for each block, the annual fluxes and pool sizes of carbon for greenspace components.

Total carbon storage in study block 1 was approximately 26·15 kg/m^2 of greenspace (total: 344 120 kg), and 23·20 kg/m^2 (total: 135 450 kg) in block 2. Soil carbon (organic and inorganic in the top 60 cm accounted for 78·7% of all carbon stored in block 1 (270 736 kg) and 88·7% in block 2 (120 094 kg). Of the total soil carbon storage, inorganic soil carbon accounted for 20·9% (block 1: 56 595 kg, block 2: 25 105 kg). Trees and shrubs in block 1 and block 2 accounted for 20·8% (71 489 kg) and 10·6% (14 378 kg), respectively. The carbon storage in grass was in the region of 0·5–0·7% in both blocks. Forest ecosystems of the United States contain approximately 52 billion metric tons of carbon, 59% in soils (to a depth of 1 m), 31% in trees, 9% in litter above soil surfaces, and 1% in understory vegetation (Birdsey, 1990). The lower carbon storage in urban trees and shrubs increases the percentage carbon storage in urban soils, compared to forest soils.

In block 1, trees and shrubs took up approximately 0·56 kg of carbon per m^2 of cover per year (total 5016·4 kg, excluding leaf-fall carbon) from the atmosphere through

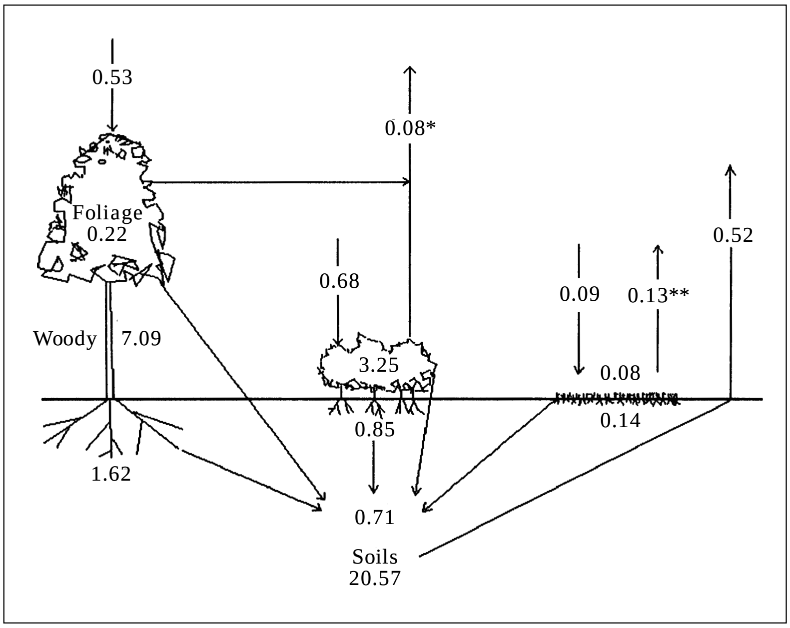

Figure 3. Annual fluxes and pool sizes of carbon (kg/m^2) for greenspace components (trees, shrubs, grass and soils) in study block1[a].

[a] The unit area (m^2) is areal cover for trees and shrubs, grass area for grass and pervious area for soils.

* The amount of carbon released by pruning

** The amount of carbon released by mowing

their growth. Subtracting carbon output (by pruning) of 0·08 kg per m^2 of cover (total: 729·2 kg) from the annual input resulted in net annual carbon uptake of about 0·48 kg m^2 (total 4287·2 kg). In block 2, annual direct carbon uptake of trees and shrubs was 0·54 kg of carbon per m^2 cover (total: 1094·2 kg), carbon output by pruning was 0·08 kg m^2 (total: 164·1 kg), and net annual carbon uptake was 0·46 kg m^2 (total: 930·1 kg).

Annual carbon uptake by herbaceous plants (differentiated from grass) totalled 35·2 kg in block 1 and 44·8 kg in block 2. Assuming that all the plants die, and are collected annually, there was no net carbon input by herbaceous plants in the study blocks. Even though some perennial herbaceous plants were growing in the study area, most of their above ground materials were replaced annually, and the contribution of remnant roots to net carbon would be very small, considering the area that was planted.

Annual carbon uptake by grass from the atmosphere was 0·09 kg/m^2 for each study block. Total annual carbon uptake was 739·0 kg in block 1 and 371·1 kg in block 2. However, mowing annually returned to the atmosphere 0·13 kg/m^2 carbon from both blocks. The total quantity of carbon released due to mowing amounted to 1069·0 kg per year in block 1 and 539·5 kg per year in block 2. Annual net carbon uptake by grass was estimated to be −0·04 kg/m^2 of grass area in both blocks. Total annual net carbon uptake by grass was −330·0 kg in block 1 and −168·4 kg in block 2. In these estimates, the carbon emission associated with the manufacture of fertilisers, mowers etc., was not included. Only on site carbon release was considered.

Annual carbon input by dead organic materials from grass, trees and shrubs (including mulches) was 0·71 kg/m^2 of pervious area (total: 9300·6 kg) in block 1 and

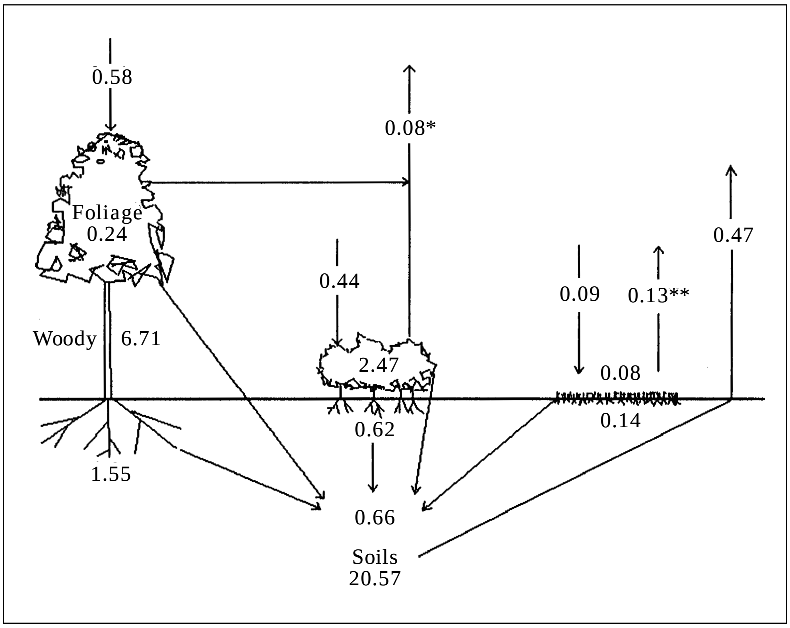

Figure 4. Annual fluxes and pool sizes of carbon (kg/m^2) for greenspace components (trees, shrubs, grass and soils) in study block 2[b].

[b] The unit area (m^2) is areal cover for trees and shrubs, grass area for grass, pervious area for soils

* The amount of carbon released by pruning

** The amount of carbon released by mowing

0·66 kg/m^2 (total 3865·0 kg) in block 2. The annual loss of carbon due to collection, runoff and decomposition was 0·52 kg/m^2 of pervious area (total: 6871·0 kg) in block 1 and 0·47 kg/m^2 in block 2 (total: 2755·0 kg). The difference between input and output, which is the annual net carbon input to soils from vegetation and management activities, was approximately 0·19 kg/m^2 in the study blocks. The annual net carbon input totalled 2429·6 kg in block 1 and 1110·0 kg in block 2. The carbon input and output by dead materials included the contribution from leaf fall from trees (block 1: 1460·9 kg, block 2: 335·7 kg) and shrubs (block 1: 468·7 kg, block 2: 125·1 kg). The contribution of leaf fall to net carbon was considered to be zero because most of the fallen leaves were collected and removed.

The total annual carbon input in block 1 was about 1·15 kg/m^2 of greenspace (total: 15 091 kg). The total annual carbon output in block 1 was 0·66 kg/m^2 (total 8704 kg). In block 2, the carbon input was approximately 0·92 kg/m^2 of greenspace (total: 5375 kg), and the carbon output was 0·60 kg/m^2 (total 3503 kg). The difference generated a net annual carbon value of 0·49 kg/m^2 (total 6387 kg) in block 1, and 0·32 kg/m^2 (total 1872 kg) in block 2.

Approximately 58–65% of the total carbon input was released annually back to the atmosphere due to landscape maintenance and decomposition. Trees and shrubs released annually, through pruning, 15% of carbon sequestered. Grass returned annually to the atmosphere 1·5 times the carbon sequestered due to mowing. The annual carbon loss from decomposition (including litter collection) accounted for approximately 79% of the total annual carbon output in both blocks.

Annual precipitation and mean annual temperature during the study period were

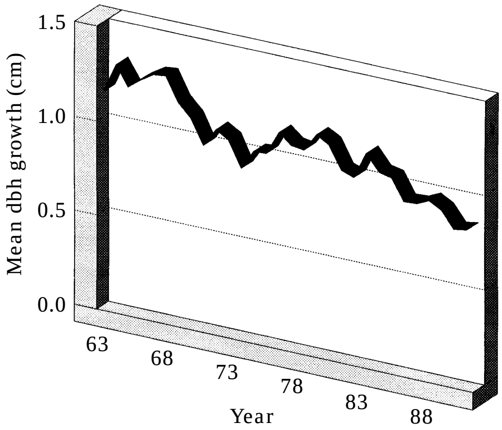

Figure 5. Changes in mean yearly dbh of hardwood trees from 1963 to 1992.

972 mm and 8·6°C, respectively (for a one-year period from June 1992 to May 1993) (NOAA, 1933). The mean temperature was lower by 0·8°C than that during the 30 years from 1962 to 1991. The annual precipitation during the study year was 74 mm higher than during the stated 30 year period. The lower temperature might have slowed the decomposition of soil organic matter during the study period (Kucera and Kirkham, 1971), assuming that the difference in air temperature is proportional to that in soil temperature. The increase in precipitation during the study period over the previous average might have resulted in an increase in carbon output from mowing.

The greenspaces in the study blocks were net sinks of carbon. However, total carbon storage and annual carbon uptake were estimated over a short period. The residential landscapes of the study area were largely composed of young growing trees and shrubs. The rate of carbon uptake by trees slows down as they age. Figure 5 shows the changes in mean yearly dbh growth of hardwood trees for 30 years from 1963 to 1992. The growth of the trees was relatively fast at a young age and declined gradually as the trees became older. All the carbon stored in trees and shrubs will ultimately be lost upon their death and removal. Existing trees and shrubs will therefore not be a long-term reservoir of carbon.

There was little new planting during the one-year study period. The only plantings observed were four small fruit trees in the back yard of one residential unit in block 2. The above carbon budgets did not consider carbon emission from automobiles driven to and from the garden centre.

3.5. GREENSPACE PLANNING AND MANAGEMENT GUIDELINES

The estimation of landscape carbon inputs and outputs for the study area indicated that soils and woody plants were carbon sinks, while grass was a net carbon source. Grass released annually, through mowing alone, 1·5 times the carbon sequestered. If carbon emission at power plants and factories by grass maintenance is included (e.g. electricity use for irrigation, manufacture of fertilizers, mowers and sprinkler systems), annual carbon release from grass would be much more significant. Therefore, less intensive grass management and reduction of lawn area are recommended to reduce carbon release. To decrease the number of mowers used, it might be desirable to share mowers among neighbors or to hire a mowing service for each residential block. Eliminating fertilization will minimize water requirements of grass during the dry season. All litterfall from vegetation can be utilized for composting to reduce the need for chemical fertilizers.

Soils were a large carbon pool with slow turnover rates. The turnover time of soil carbon in the study area was estimated to be about 41 years. Woody plants were more beneficial to annual net carbon uptake than herbaceous plants. Based on this short-term study, increasing pervious areas with trees and shrubs could result in increased carbon storage in urban greenspace. A survey of tree planting potential found that the total annual carbon uptake by trees could be increased by about 1·8 times the present sequestration in block 1, and 3·3 times the present sequestration in block 2. Reducing the present wide sidewalk and cement area in back yards could result in a larger pervious area and allow more trees to be planted. It may be better to plant fast-growing trees. These sequester more carbon annually than slow-growing species, such as conifers and shrubs.

Trees will sequester atmospheric carbon during their growing period. However, trees will mature and be removed at different times. After that time, they will act as a net carbon source due to decomposition or burning. Immediate replacement is needed to compensate for the carbon emitted from previously removed wood. Simply planting more trees may not be sufficient to decrease the level of atmospheric carbon. Circumspect management for longer productive lifespans should be accompanied to ensure trees remain as a long-term carbon reservoir. Planting of the right species in the right space is required to avoid a rapid release of carbon. Space for tree growth was often restricted by above ground utility lines and buildings in the study area. Planting large-growing trees in a small space could result in severe pruning and early removal.

4. Conclusion

The greenhouse effect can be interpreted to be a result of the combined, cumulative impacts of carbon emissions from every region of the world. Information on carbon fluxes from detailed local and regional studies may contribute to the moderation of climate change. Residential lands account for approximately half of all land area in cities. This research quantified carbon inputs, outputs and storage for greenspaces of two residential neighborhoods (blocks 1 and 2). There was little difference between the two blocks, in terms of carbon storage and annual carbon uptake per unit area of land cover type (e.g. grass) or vegetation cover. The differences in the size of greenspace area and vegetation cover resulted in greater carbon storage and annual carbon uptake in study block 1.

The reduction in carbon emissions from power plants, due to savings in cooling and heating energy by vegetation, was not considered in this research. The indirect carbon reduction by vegetation will be permanent (no release of carbon back to the atmosphere). The greenspace planning guidelines suggested in this study should be combined with tree planting strategies to maximize energy savings.

A major problem in conducting this study was to obtain permission from homeowners for data collection. The initial aim was to survey all the vegetation in each block and to interview all the homeowners (i.e. without sampling). However, permission for access to survey individual lots was received from only 73% of the residents in block 1, and 61% in block 2. The residents' active participation is important in studies of this kind. Another problem was that it was impossible to harvest trees and shrubs in the study area to measure biomass. Quantification of carbon storage and uptake by the trees and shrubs depended, for the most part, upon biomass data from forest areas, although the biomass equations used were generated for the same species as the urban trees in this study.

It was often difficult to compare the results of this study with those of other studies, because there are few that relate to biomass measurements of urban vegetation or carbon cycling in urban ecosystems. More studies, including biomass of urban trees and shrubs, their growth and mortality rates, and carbon contents of greenspace components, are required to deepen our understanding of carbon cycling in urban landscapes and to help reduce the levels of atmospheric carbon dioxide.

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