Using Structural Measures to Compare Twenty-Two U.S. Street Tree Populations

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Abstract: *Our purpose is to use certain ecological measures, commonly adopted for understanding natural forest populations, to describe and compare street tree populations. Street trees compose only about 10–20 percent of the urban forest, but there are sufficient data on these populations to apply certain measures of structure and to interpret the results. We used inventories from 22 U.S. cities to compare patterns in species and age composition, stocking levels, and growth forms. From this we infer trends toward greater species diversity and smaller tree size; these trends are discussed in terms of their implications for future costs and benefits. Our assessment is that certain measures can illuminate important attributes of the populations, especially when an array of populations is graphically compared.*

Can measures of ecological structure originally developed for natural tree populations be useful in urban forestry? While the urban forest is, in part, an artificial construction, many of its components operate as natural forests. Thus, measures devised for natural communities could be successfully used with, for example, residual islands of old-growth forest within the city.

The stiffest test of this idea would be to apply ecological indices to the most artificial component of the urban forest—street trees. It is here that the measures would be most likely to fail. The challenge lies in evaluating measures that help us to see patterns and trends in the structure of these populations. In the course of doing this, it may become clear that some measures add nothing and some need to be modified. To comprehend certain attributes of street tree populations, entirely new measures may need to be invented.

Purpose

The purpose of this study is to apply selected measures, or indices, of forest structure to 22 street tree populations in the United States. Our effort is based on the premise that urban forest knowledge is often best articulated at the level of the dendro-population. At that level, patterns and trends might be seen, especially when the indices are compared within a group of populations as we have done.

For this study we use the term "population" because street trees constitute an entity by sharing certain things in common. They are all planted. They live in similar environments, and are often managed in similar ways by the same institution (e.g., the municipality). At the same time, we recognize that in certain locales street trees may have more in common with adjacent yard trees than with other street trees.

While they account for only 10–20 percent of the total urban forest biomass (Rowntree and Stevens 1988), street trees influence the lives—for better or for worse—of many residents and are the subject of much concern. In the urban forest, they are the first to be

TABLE 1.
Descriptive Data on Cities and Street Trees Used in the Study

City Name	City Pop.	Pop. Density (pop/ha)	Potential Natural Vegetation[a]	USDA Hardiness Zone[b]	Street Length (km)	Total Street Trees	% Ave. Full Stocking	Trees Per Capita	Imp. Pat. Type[c]	Sps. Div. Index	Dia. Dist. Type[d]
Austin, Minnesota	26,000	16.7	Oak savanna	4	174	10,535	45.2	.41	CD	2.7	1
Bay City, Michigan	41,600	14.6	Oak-hickory forest	5	314	16,408	39.0	.39	SD	2.5	2
Bexley, Ohio	13,700	21.1	Beech-maple forest	6	68	5,458	60.3	.40	WD	2.4	3
Bratenahl, Ohio	1,485	5.7	Elm-ash forest	6	10	789	58.9	.53	WD	2.2	3
Edina, Minnesota	46,000	17.8	Oak savanna	4	418	10,682	19.1	.23	SD	2.7	2
Eugene, Oregon	106,000	10.2	Mosaic oak, cedar, fir	8	644	20,727	24.0	.19	WD	3.3	1
Great Neck Estates, N.Y.	3,000	14.6	Appalachian oak forest	7	21	1,577	56.3	.53	WD	3.0	3
Grosse Pointe Woods, Mich.	19,000	22.9	Oak-hickory forest	6	86	4,813	41.8	.25	CD	2.5	2
High Springs, Florida	2,600	1.7	Southern mixed forest	9	53	3,085	43.4	1.19	SD	2.4	3
Jessup, Iowa	2,350	6.0	Mosaic prairie, oak savanna	5	26	503	14.4	.21	CD	2.6	3
La Canada Flintridge, Calif.	21,150	9.6	Chapparal	9	129	12,154	70.5	.57	WD	3.4	1
Los Angeles Co., Calif. (Developed portion only)	1,022,260	1.0 (4.8)	California steppe	10	5690	200,085	26.2	.20	WD	3.9	1
Mount Horeb, Wisconsin	3,500	4.8	Great Lakes pine forest	5	29	871	22.4	.25	CD	2.5	2
Mount Vernon, Ohio	15,000	2.2	Appalachian oak forest	6	105	4,223	30.0	.28	SD	2.1	2
Murray City, Utah	29,000	11.8	Juniper-pinyon woods	5	177	4,594	19.4	.16	WD	2.7	1
Newark, New Jersey	329,250	52.6	Appalachian oak forest	7	628	30,565	36.3	.09	CD	2.3	3
Northwood, Iowa	2,190	2.1	Oak savanna	4	82	1,384	12.6	.63	CD	2.7	2
Perrysburg, Ohio	10,215	9.9	Elm-ash forest	6	80	5,863	54.7	.57	SD	3.0	1
Plymouth, Minnesota	37,050	4.0	Oak savanna	4	322	12,613	29.3	.34	WD	3.3	3
River Falls, Wisconsin	9,020	8.7	Maple-basswood forest	4	56	3,999	53.0	.44	WD	2.5	1
Syracuse, New York	170,100	38.0	Northern hardwoods	5	644	39,030	45.2	.23	CD	2.7	2
Vienna, Virginia	15,600	15.1	Oak-hickory-pine forest	7	37	2,129	42.9	.14	WD	2.9	1

a. Potential Natural Vegetation from Kuchler (1964).
b. United States Department of Agriculture (U.S.D.A.) Hardiness Zones (see Dirr 1877) are based on average annual minimum temperatures as shown below:

Zone No.	Temp. Range (C)
1	below −46
2	−46 to −40
3	−40 to −34
4	−34 to −29
5	−29 to −23
6	−23 to −18
7	−18 to −12
8	−12 to −7
9	−7 to −1
10	−1 to 4

c. Importance Patterns: SD = Strong dominance, CD = Codominance, WD = Weak dominance.
d. Diameter distribution types: 1 = Type 1, 2 = Type 2, 3 = Type 3.

inventoried and it is this fact that allows us to acquire sufficient data to perform our analyses. We cannot objectively assess the efficacy of each measure, but we render our summary opinion of how useful this exercise has been in the final section of the paper.

Species composition and age structure of street trees determine, to a great extent, the kinds and quantities of human benefits and costs generated by these populations. For example, the choice of species affects wildlife habitat suitability (McPherson and Nilon 1987). Large old trees contain more above-ground leafy and woody biomass than do small young trees. As a result, larger trees reduce the volume of stormwater runoff more effectively, remove more particles from the air, and provide more shade to buildings than do smaller trees (Rowntree 1986). In addition, larger trees along residential streets seem to provide more scenic beauty (Lien and Buhyoff 1986).

Also, species composition and age structure are indices of future maintenance costs. High species diversity is generally regarded as one of the bases for population stability (McBride and Jacobs 1976), and a reduced threat of large annual maintenance costs from monocultural disease and loss. Age structure indicates the relative need for replacement plantings and future removals (Richards 1983). Ecologists have developed the means for quantifying structure and change in natural plant communities (Watt 1947, Daubenmire 1966, Mueller-Dumbois and Ellenberg 1974, Whittaker 1974, Greig-Smith 1979, Woods 1979, Barbour et al. 1980, Krebs 1984). Some of these ecological measures have been applied to the urban forest. Tree species composition, cover, density, and age have been used to compare the implications of presettlement forest structure in two California communities (McBride and Jacobs 1986). Others have used importance values and species diversity indices to describe and compare the composition of yard and street tree populations (Sanders 1981, Dorney et al. 1984, Miller and Winer 1984, Rowntree 1984). Size and frequency data were used to assess age structure, productivity, and needs for replacement plantings (Lawson et al. 1972, Richards 1979, Richards and Stevens 1979, D'Ambrosio 1980, Dorney et al. 1984).

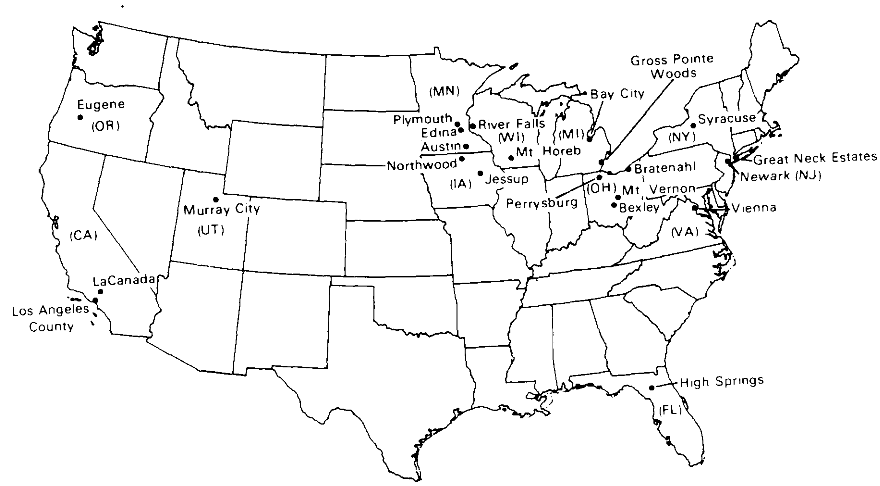


Figure 1. Locations of the 22 U.S. cities having street tree populations that were analyzed in this study.

Our study employs size and frequency data for trees from 22 U.S. street tree populations to describe the current and future attributes of each population. We chose street tree populations from different regions of the country and from cities with different development histories.

Certain patterns of structure are identified for the sample of 22 cities and some of the implications of these patterns for planning and management are briefly discussed. We display our indices in graphs so the reader may evaluate how well this information can be communicated visually.

Methods

Data Collection. Street tree inventory data listing the number of trees per diameter class in each species were obtained for 22 cities. Sixteen of the inventories were conducted by ACRT, Inc. (formerly Davey Environmental Services) under contract to the municipalities. Inventory data for the remaining cities were obtained from the foresters who supervised each inventory. Trained personnel collected the data used in this study. Eighteen of the inventories were conducted between 1982–1985, and all 22 were conducted since 1978. All inventories included 100 percent of the street tree populations except for a one-third sample in Vienna, Virginia. The number of individuals by species were listed for each of six diameter-at-breast-height (dbh) classes (< 15 cm, 15–30 cm, 30–45 cm, 45–60 cm, 60–80 cm, > 80 cm). Cities for which data were collected (Figure 1) represent a variety of demographic conditions and climatic regions (Table 1).

Species information was aggregated into genera or "other" categories (e.g., "other maple species"; "other species"). Lumping a number of low-frequency species into "other" categories results in an underestimation of species richness. The magnitude of the impact depends on the number of individuals in each aggregated category. We note where this number exceeds two percent of all trees.

Calculations. Importance values were calculated to quantify the relative degree to which a species dominates a street tree population. It considers both the number and size of all individuals as:

$$\text{Importance value of species X} = \text{Relative abundance} + \text{Relative dominance}$$

where:

$$\text{Relative abundance} = (\text{No. of individuals of species X} / \text{Total individuals of all species}) \times 100$$

$$\text{Relative dominance} = (\text{Basal area of species X} / \text{Total basal area of all species}) \times 100$$

The basal area for each species was calculated by summing the basal areas for trees in each dbh class. The total for each dbh class was determined by multiplying the number of trees in the dbh class by the basal area for a tree with a dbh in the middle of the size class (i.e., a dbh of 7.5 cm for the < 15 cm size class). Note that relative abundance and dominance were calculated without area measurements. Measures of abundance and dominance were originally devised for natural, nonlinear communities of spatially proximate individuals and incorporate calculations of area (Cottam and Curtis 1956). To make them applicable to linear street tree populations, we modified the original measures by leaving out the area calculation.

Results and Discussion

Stocking Level Index and Trees Per Capita. Wray and Prestemon (1983) define the full stocking of street tree populations as having a spacing between stems of approximately 15 meters. We used this figure and the total number of street trees and street length to calculate percentage of full stocking, or "stocking level." (The proportion of street length occupied by intersections and driveways was not included in the computation.) Thus, stocking level is simpy an index for comparing the density of trees along city streets and the concomitant opportunities for new planting.

In the 22 cities, stocking levels range from 12.6 percent to 70.5 percent with a mean value of 38.4 percent (Table 1). The mean number of trees per capita is 0.37, and the range extends from 0.09 to 1.19 trees per person (Table 1). Both stocking level and street trees per capita appear to be inversely related to city population (Table 2). A similar inverse relationship between total canopy cover and population was found by Halverson and Rowntree (1986).

TABLE 2.
Stocking Levels and Street Trees Per Capita

City Population	n	% Full Stocking	sd	Trees/ Capita	sd
> 100,000	4	33	9.8	0.18	.06
15,000–50,000	9	37	15.8	0.31	.14
< 15,000	9	42	19.7	0.53	.29
Means	22	38	16.4	0.37	.24

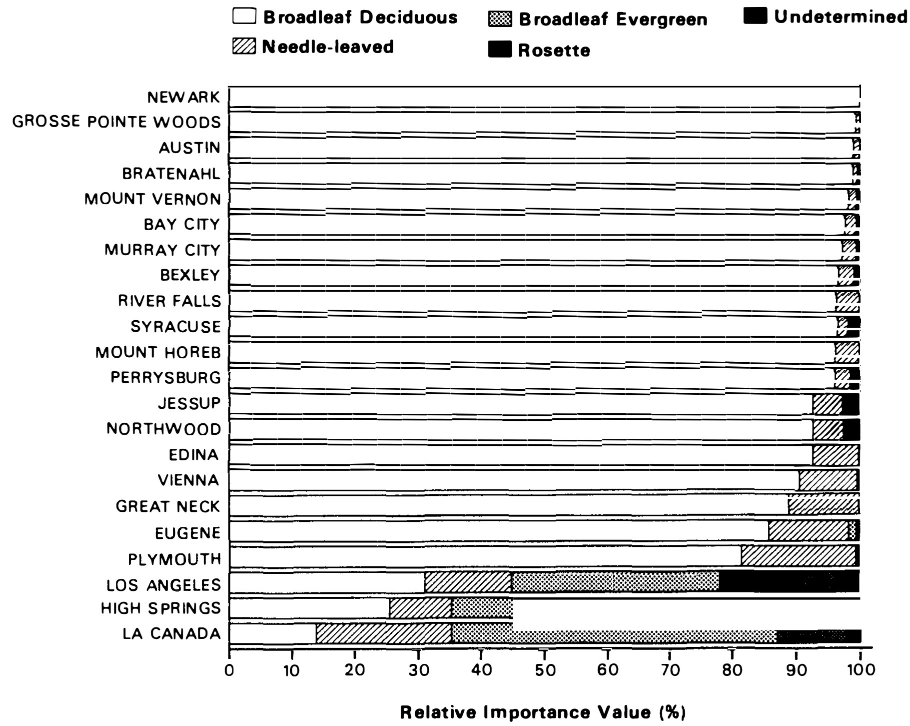


Figure 2. Percentage composition of street tree populations by growth form. Broadleaf deciduous forms dominate in temperate regions and broadleaf evergreens dominate in subtropical regions.

In densely populated areas there are more people per unit of street length and, some would say, a greater need for street trees than in areas of lower population density. Calculations of stocking level and street trees per capita may be useful in discussions of whether communities of higher population density should increase their stocking levels on the assumption that the importance of street trees in the lives of urban dwellers increases in proportion to population density.

Proportions of Different Growth Forms in the Populations. Growth forms make up the urban forest's scenic value and its capacity to ameliorate conditions such as poor air quality, excessive or insufficient amounts of solar radiation and air flow, and undesirable views and noise. For example, deciduous trees provide welcome solar radiation during winter months, but are less efficient than evergreens in capturing particulates and screening offensive sites from view during this time. Rosette trees, such as the palms, provide beautiful silhouettes against the skyline but contain relatively little biomass and may require annual pruning to remove dangerous fronds and fruits. In this study the percentage of total importance value represented by each growth form in each city was calculated and graphed (Figure 2).

Growth forms reflect environmental conditions. In temperate climates broadleaf deciduous street trees prevail. Deciduous trees are much less important in subtropical regions, where rosette trees and broadleaf evergreens predominate. Preference for year-round foliage may partially explain the predominance of broadleaf evergreens in regions where both deciduous and broadleaf evergreens grow. It is interesting to note that needle-leaved trees are relatively unimportant along streets regardless of climate zone.

Species Importance Values. The species composition of street tree communities reflects, first, human choices and, second, site factors (Whitney and Adams 1980, Dorney et al. 1984). Although a city may contain many street tree species, it is common that relatively few species exert a major influence by virtue of their size and numbers. After calculating importance values of the most common species, we found three patterns (Figure 3).

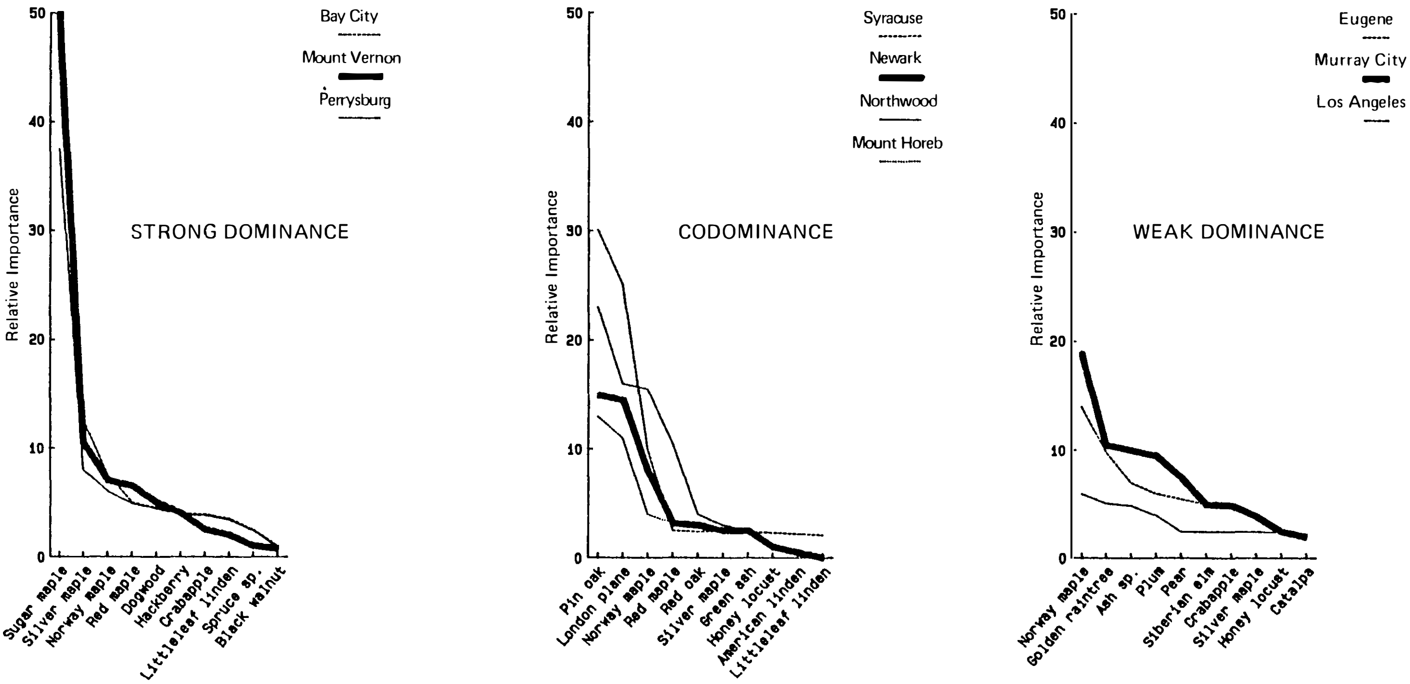


Figure 3. Three patterns of importance are shown as relative importance values for leading dominants in selected cities.

The first pattern, termed *strong dominance,* occurs when one species' relative importance is greater than 25 percent and there are no subdominants with importance values greater than 15 percent. Five cities contain street tree populations that exhibit this pattern (Table 1). Species of strong dominance are silver maple *(Acer saccharinum)* in Bay City, Michigan and Perrysburg, Ohio; sugar maple *(Acer saccharum)* in Mount Vernon, Ohio; American elm *(Ulmus americana)* in Edina, Minnesota; and laurel oak *(Quercus laurifolia)* in High Springs, Florida.

For each of these species their relative size is greater than their relative abundance. While this size indicates their ability to survive the tests of time, the dominant species are not always the most desirable species. For example, Fremont poplar *(Populus fremontii)* dominated the urban forest in Albuquerque, New Mexico until monocultural plantings of Siberian elm *(Ulmus pumila)* replaced it (Dykema 1985). Now the city is faced with enormous maintenance and removal costs because the elm is weak-wooded and subject to a variety of insect and disease problems. Street tree populations with one major dominant species will typically have lower maintenance costs due to the efficiency of repetitive work if the dominant species is a good one, but may still incur large costs in one year, or a few years, if decline, disease, or senescence in the dominant species requires large numbers of removals and replanting.

Newark, New Jersey best exemplifies what we found to be a second pattern of importance—*codominance.* Pin oak *(Quercus palustris)* and London plane *(Platanus acerifolia)* each have relative importance values greater than 10 percent, and their sum exceeds 25 percent. Codominance was the pattern for seven cities (noted in Table 1). All of the cities were in temperate climates and no two cities had the same two species as codominants. Species most commonly occurring as one of two codominants were silver maple (four cities), green ash *(Fraxinus pennsylvanica)* (three cities), Norway maple *(Acer platanoides)* (two cities), and American elm (two cities). Codominance provides greater population stability than strong dominance, and more closely resembles the condition in many natural forest ecosystems.

The third pattern of importance, *weak dominance,* exists when importance is relatively evenly dispersed among five to ten leading dominants. The importance curve is flattened (Figure 3). Using Murray City, Utah as an example, Norway maple is not strongly dominant because the relative importance value is less than 25 percent. The second most abundant species, flowering plum *(Prunus cerasifera),* is the fourth leading dominant due to the small sizes of these many young plantings. Ten cities (including all four western cities) were categorized as having a weak dominance pattern (Table 1). Of these ten populations, seven had species diversity indices (discussed below) that were among the ten highest for all cities. Weak dominance and high species diversity often go together. Certainly the weak dominance pattern tends to minimize risks of catastrophic losses of a single dominant species. However, planting a wide range of short-lived and/or poorly adapted species can result in shorter rotations and increased long term management costs (Richards 1983). In Murray City, five of the ten leading dominants are short-lived. This is in sharp contrast to Newark and Mount Vernon, where only one or two species are short-lived.

Species Diversity. Species diversity is derived from species richness (the number of species) and evenness (the distribution of trees among species). A greater number of species increases diversity, as does a more even distribution among species. For this study, the

Shannon-Wiener equation was used to measure diversity (Krebs 1984).

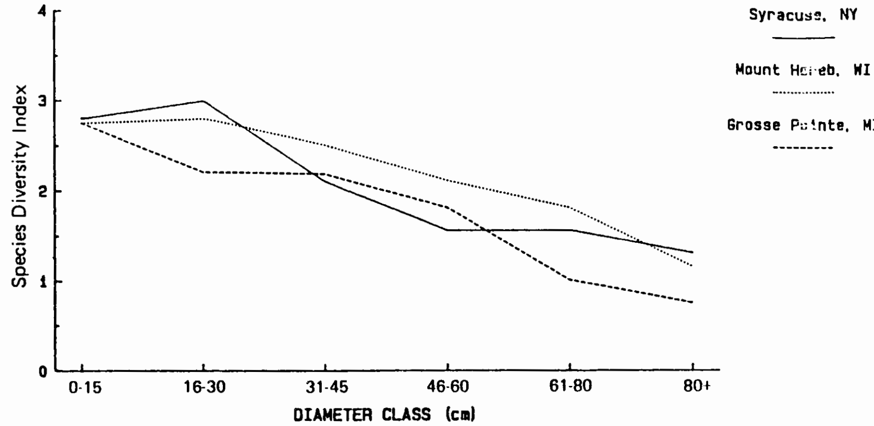
$$H = -\sum_{i=1}^{S}(\rho i)(\log_2 \rho i)$$

where

H = observed species diversity
S = number of species
ρi = proportion of the total sample belonging to the *i*th species

Species diversity indices were calculated for each street tree population (Table 1). Values ranged from 2.1 to 3.9. The mean species diversity index value for all cites was 2.7. Species diversity was greatest in cities with mild climates. Higher values in these cities is more a result of a larger number of species rather than evenness of distribution. The mean number of street tree species for all cities was 53, but Los Angeles and La Canada, California and Eugene, Oregon contained 77, 77, and 63 tree species, respectively. Relatively low diversity in the mild climate city of High Springs, Florida and high diversity in cold Plymouth, Minnesota are exceptions. The preponderance of old laurel oak, which is a well-adapted native species, accounts for the low diversity in High Springs. High species diversity in Plymouth may be related to the city's recent expansion and accelerated planting program. Ninety percent of the street trees are less than 35 cm dbh. Plymouth's inhabitants are aware of the recent high costs associated with the American elm monoculture and Dutch elm disease. Their policy of striving for high street tree diversity is likely to be in reaction to that catastrophe.

Species diversity was also calculated within each diameter class. We found that species diversity within a diameter class decreases as the diameter classes get larger. Two forms of this relationship occur in our sample of 22 cities (Figure 4). In the first form, which is typical of cities having suffered heavy losses from Dutch elm disease, species diversity drops sharply with increasing diameter. In these cities, relatively few species (generally maples) occur in the large diameter classes, and a diverse mixture of new plantings compose the smaller classes.



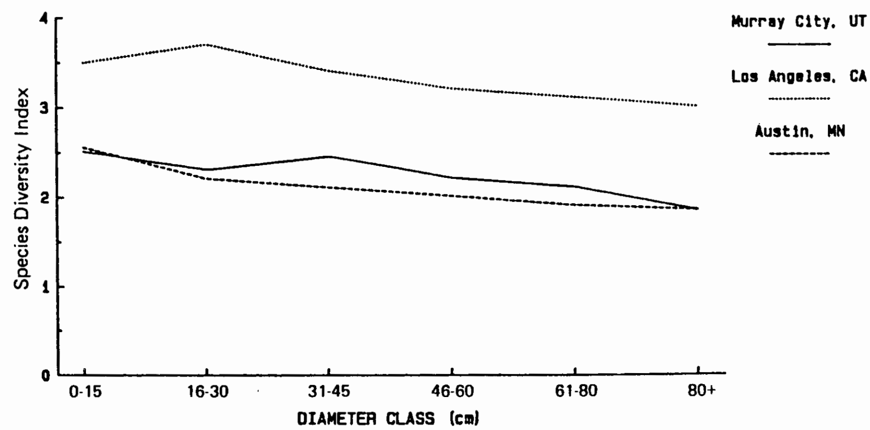


Figure 4. Two patterns of declining species diversity with increasing diameter class for six cities. The top plot describes a sharp drop in diversity and typifies populations decimated by Dutch elm disease. The bottom plot shows more gradual decline due to relatively higher species diversity in the larger size classes.

The second form of the relationship occurs in cities where American elms were not heavily planted and we find more diversity in the large diameter classes. Most western cities (e.g., Los Angeles, California and Murray City, Utah from our sample) have relatively few American elms. Larger numbers of American elms were planted in Austin, Minnesota (relative abundance of the species is 13 percent), but green ash, hackberry *(Celtis occidentalis)*, Siberian elm, and bur oak *(Quercus macrocarpa)* are also well represented in the large diameter classes; hence, the sharp decline in diversity does not occur as in the first form of this diversity/diameter class relationship.

Age Structure. The distribution of ages within a street tree population affects present and future tree care costs and the kinds of benefits available. An uneven-aged population may allow managers to allocate annual maintenance costs uniformly over many years. It may also assure a uniform distribution of canopy cover, with corresponding benefits, over time. The costs and benefits generated by street tree populations depend on the spatial scale of age structure and species patchiness (Sanders 1981). For example, neighborhood stands containing street trees of similar age and species can be more efficiently managed on an area basis than can block faces containing uneven-aged trees and mixed species (Kielbaso and Kennedy 1983). These advantages may be overshadowed by the disadvantages of having to bear large costs of removal in one or a few years. On a city-wide scale, street tree populations are generally uneven-aged. This fact can be attributed to two phenomena. One is the cyclical nature of tree planting, which commonly follows episodes of species-specific disease or decline (Richards 1983). Secondly, street tree planting is often coincident with land development. The result of these factors is that the all-aged urban forest is made up of a large number of even-aged "stands." These stands can be the size and shape of a residential housing tract, or a mile of linear replacement plantings, or any number of other sizes and shapes.

Because age was not measured from increment cores, we had to use the size of the stem as a surrogate. This was done with the understanding that the relationship between the tree's age and its diameter is not linear and, thus, the relationship can only be approximate.

We found three common patterns by visually grouping the diameter distribution curves (Figure 5). Type 1 characterizes young populations with over 40 percent of the total population in the smallest diameter class and 15–30 percent in the next smallest size class. This pattern coincides with Richards' (1983) recommendation of a good age distribution for population stability (40% < 20 cm, 30% at 20–40 cm, 20% at 40–60 cm, and 10% > 60 cm). The high proportion in the small diameter classes is needed to offset establishment-related mortality, and could be reduced if well-adapted species are selected for planting. Only eight of the twenty-two populations in this study exhibit the preferable Type 1 pattern (Table 1).

Only moderate rates of new and replacement planting identify populations with a Type 2 pattern, which has more individuals between 16–45 cm than in the 0–15 cm class. The number of trees in the smallest size class is substantially less for Type 2 than for Type 1 populations. We interpret this distribution pattern to mean that a large percentage of the Type 2 populations consist of trees planted 20 to 50 years ago, now in healthy maturity providing maximum benefits because of their size and condition, and requiring relatively little maintenance. As these trees continue to grow, maintenance and liability costs will increase. The Type 2 pattern describes the age structure for seven of the twenty-two street tree populations (Table 1).

The Type 3 pattern has a relatively even distribution of trees among all diameter classes (Figure 5). This pattern indicates that Type 3 populations have more trees with diameters greater than 45 cm than Type 1 or 2 populations. The Type 3 pattern suggests that many trees were planted before 1940 and are now mature or senescent. Benefits associated with large quantities of aboveground biomass for these trees may be partially negated by increased hazard liability and removal costs. Replacement planting programs should be high priorities in the seven cities with Type 3 age structures (Table 1).

Diameter/Age Diversity. Diameter/age curves can be constructed for individual species as well. In lieu of this, a single index may be preferable to characterize the distribution of trees of the same species among diameter/age classes—the Shannon-Wiener diversity function. High diversity indicates that the species is evenly distributed among diameter, or age, classes and will have continuity in the population over time. Diversity indices were plotted for the ten leading dominants in a typical city, Bay City, Michigan (Figure 6). Norway maple, green ash, and horsechestnut *(Aesculus hippocastanum)* have the lowest diversity indices. Diameter class diversity is low for horsechestnut and green ash because most horsechestnuts fall in the 45–60 cm size class, and most green ash were recently planted and are thus found in the < 15 cm diameter class (Figure 7). Silver maple exerts strong dominance in this population. The diameters of 72 percent of all silver maple exceed 60 cm. These large trees are evenly distributed among the three largest diameter classes, and the remaining trees are evenly distributed among the three smallest classes. The result is a high diversity index. How-

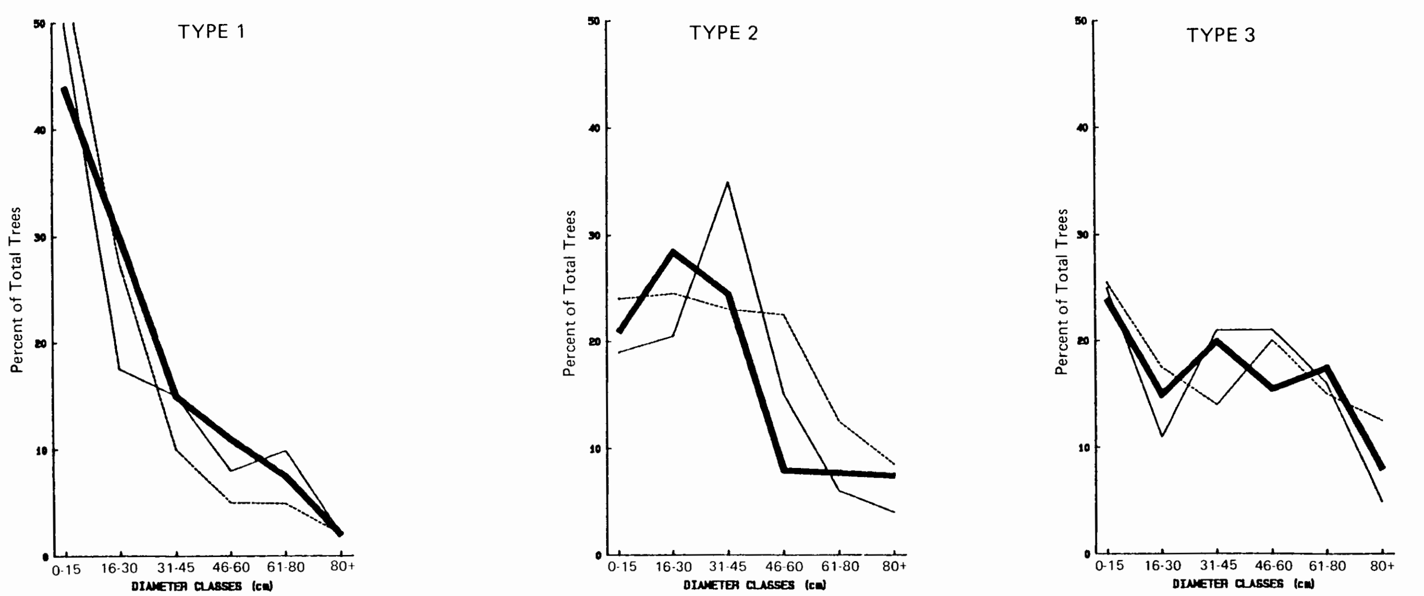


Figure 5. Three patterns of street tree age structure. Data from nine cities are plotted. The remainder of cities conform to these patterns.

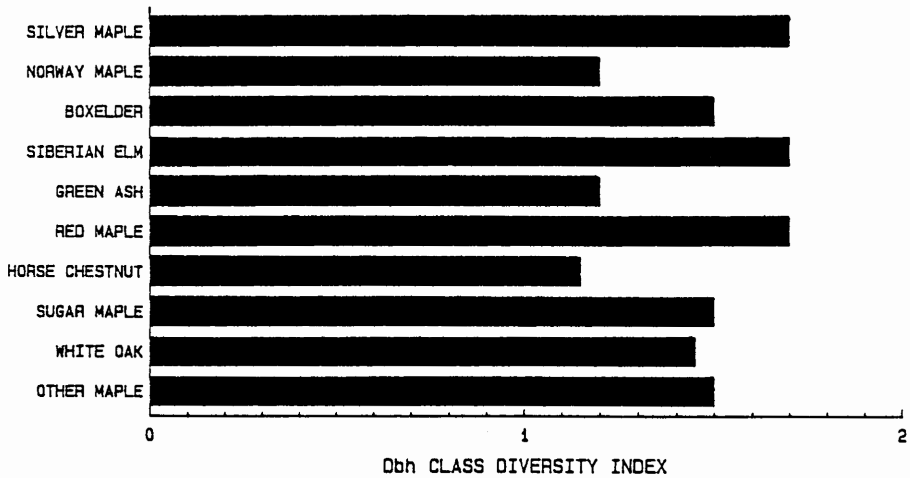


Figure 6. Diameter class diversity for ten leading dominant species listed in rank order for Bay City, Michigan. Indices reflect evenness of distribution among diameter classes.

ever, the predominance of silver maple will decline in the future when the large percentage of trees over 60 cm die. Red maple *(Acer rubrum)* has a similar index, but a different distribution. Most trees are either less than 15 cm dbh or greater than 45 cm dbh. The smaller numbers in the intermediate diameter classes are unexplained, but may be due to smaller numbers planted and/or higher than normal mortality during establishment or youth.

In Figure 7, the diameter/age curves are contrasted for four of the ten species from Figure 6. Species with the lowest diameter/age diversity indices exhibit peaked curves. Urban foresters may use either the index or these curves to plan the replacement, or phasing out, of particular species.

Likely Changes In Species Composition. Measures of age structure can also be used to infer changes in species composition over time. We compare the overall relative abundance of species with their relative abundance in the smallest (0–15 cm) diameter class (Figure 8). The left column lists species in descending order of relative abundance for the temperate climate cities in our sample. (In this column, trees of all size classes are included.) The right column lists only species of dbh less than 15 cm. Ascending lines, connecting the species, indicate that the species represents a larger proportion of all trees recently planted than it represents of all trees in the total population. The biggest gainers are ash (primarily green ash), dogwood *(Cornus sps.)*, crabapple *(Malus sps.)*, honeylocust *(Gleditsia triacanthos)* , linden *(Tilia sps.)* (primarily littleleaf linden), spruce *(Picea sps.)*, Norway maple, and flowering plum. Species most underrepresented in recent plantings are silver maple, American elm, sugar maple, London plane tree, pin oak, red maple, and pine *(Pinus sps.)*. Although these data are consistent with results from a survey of U.S. municipalities containing information on species being currently planted (Kielbaso and Kennedy 1983), there is an inherent bias in this approach. We assume no mortality (see Appendix A), but smaller, short-lived species are more likely to die sooner than larger, longer-lived species. Hence, what we interpret

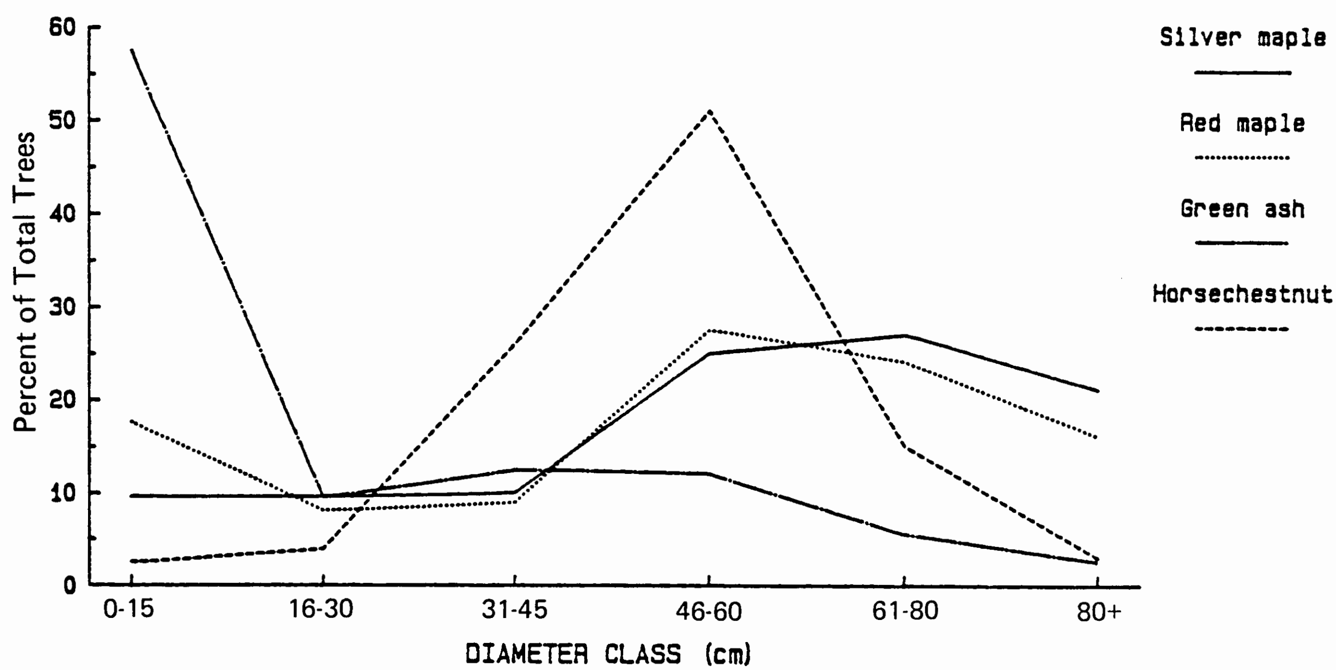


Figure 7. Diameter class distributions for four of the leading dominants shown in Figure 6. Distributions for silver and red maple exhibit greater evenness than for green ash or horsechestnut.

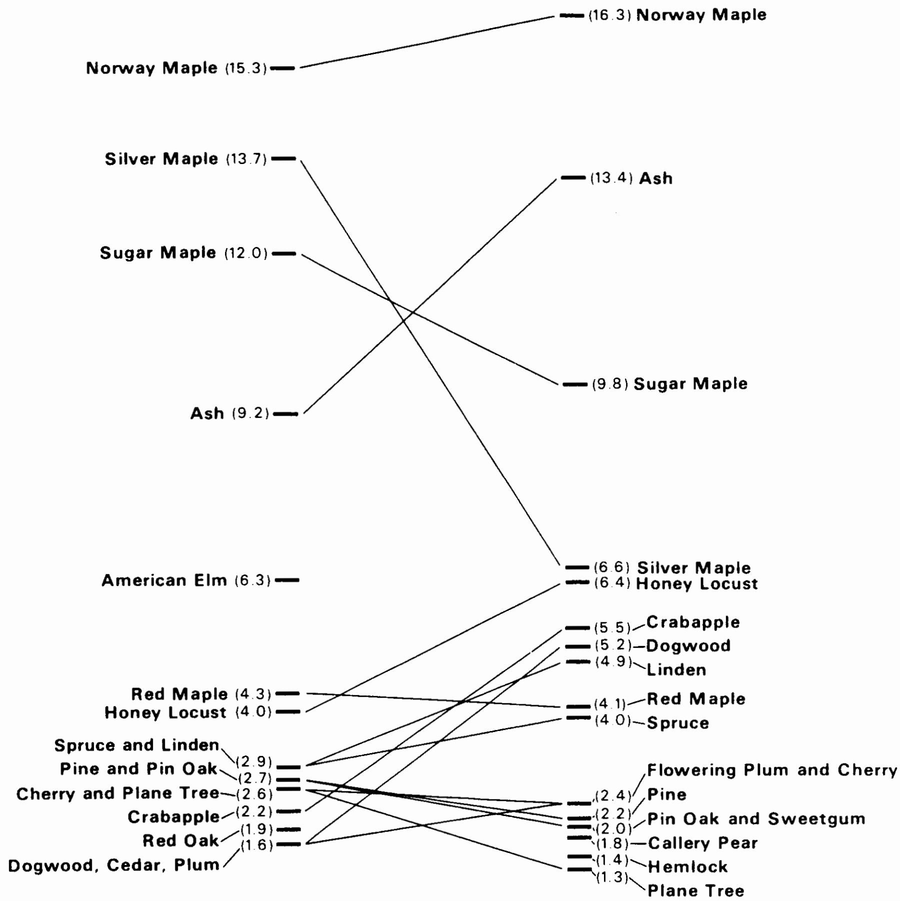


Figure 8. Changes in street tree species preferences. Numbers in parentheses following species in left column indicate mean relative abundance of street tree species in temperate climate cities. Numbers in the right column represent mean relative abundance of species occurring in the 0 to 15 cm diameter class. The degree of change in species preference is reflected by the connecting lines' degree of slope.

as high planting rates for dogwood and crabapple may be also a result of high mortality for older trees that have grown larger than 15 cm.

Likely Changes in Mature Height Composition. Data in the foregoing section suggest that fewer numbers of trees that grow into large-sized specimens are being planted. If this is true, the population of newly planted street trees should contain a lower percentage of species with large mature heights than does the total population. To confirm this, the present mature height composition was determined for each street tree population and compared with the mature height composition of newly planted trees. To determine present mature height composition, each species belonging to the ten most abundant species of each population was categorized as "large" (> 15 m), "medium" (9–15 m), or "small" (< 9 m) based on mature heights obtained from references (Dirr 1977, Williamson 1979). In this and following sections, "large," "medium," and "small" refer to mature heights. A procedure was developed to provide estimates of average annual planting rates for these three classes of trees (Appendix A). In the street tree populations of most of the cities we analyzed, large species accounted for 60–85 percent, medium for 10–25 percent, and small species for less than 20 percent of the total population.

In 82 percent of the cities, recent street tree plantings contain fewer large trees than does the entire population (Figure 9). Seventy-five percent of the cities show increased plantings of small trees. As an example, the city of Mount Vernon, Ohio, reflects the most drastic change in relative proportions of large and small trees. Large trees account for two-thirds of all street trees but less than 20 percent of the recent plantings. Seventy percent of the recent plantings are small trees. Our results also agree with trends towards more uniform distributions of basal areas and a greater variety of street tree species found after fifty years (1932–1982) for neighborhoods in Urbana, Illinois (Dawson and Khawaja 1985). The following reasons *may* explain the shift away from use of the large maples, oaks, and elms that were such popular choices in temperate climates about fifty years ago.

1. Episodic losses of monocultural elm plantings led to more diverse street tree plantings that often included a number of small species.

2. Urban foresters and landowners are now planting more small trees to reduce future pruning, removal, and liability costs.

3. Some cities are taking a less active role in financing and supporting street tree planting (Wray and Prestemon 1983), and citizens end up selecting street tree species. Many homeowners seem to want rapid-growing, low maintenance, ornamental varieties that are all small trees.

4. In northern latitudes of the United States, homeowners have come to appreciate the sunlight that became more available once the large elms were removed and have, subsequently, preferred small species.

Implications of Changes in Mature Height for Benefits and Costs of Street Trees. Although our calculations may agree with the survey findings of Kielbaso and Kennedy (1983), our interpretations differ. Kielbaso and Kennedy (1983, p. 437) conclude, "Unfortunately, urban tree managers have paid little attention to the negative consequences of large, single species plantings . . . ." We interpret the results of applying our measures as suggesting conscious efforts to increase species diversity and increase the number of small trees in street tree populations. The implications of these changes on benefits and costs have not been thoroughly addressed, however.

Richards (1979) has reported that long term replacement costs will increase if new plantings consist of short-lived species. Many of the small-sized species, such as flowering plum, mountain ash *(Sorbus aucuparia),* and crape myrtle *(Lagerstroemia indica),* have

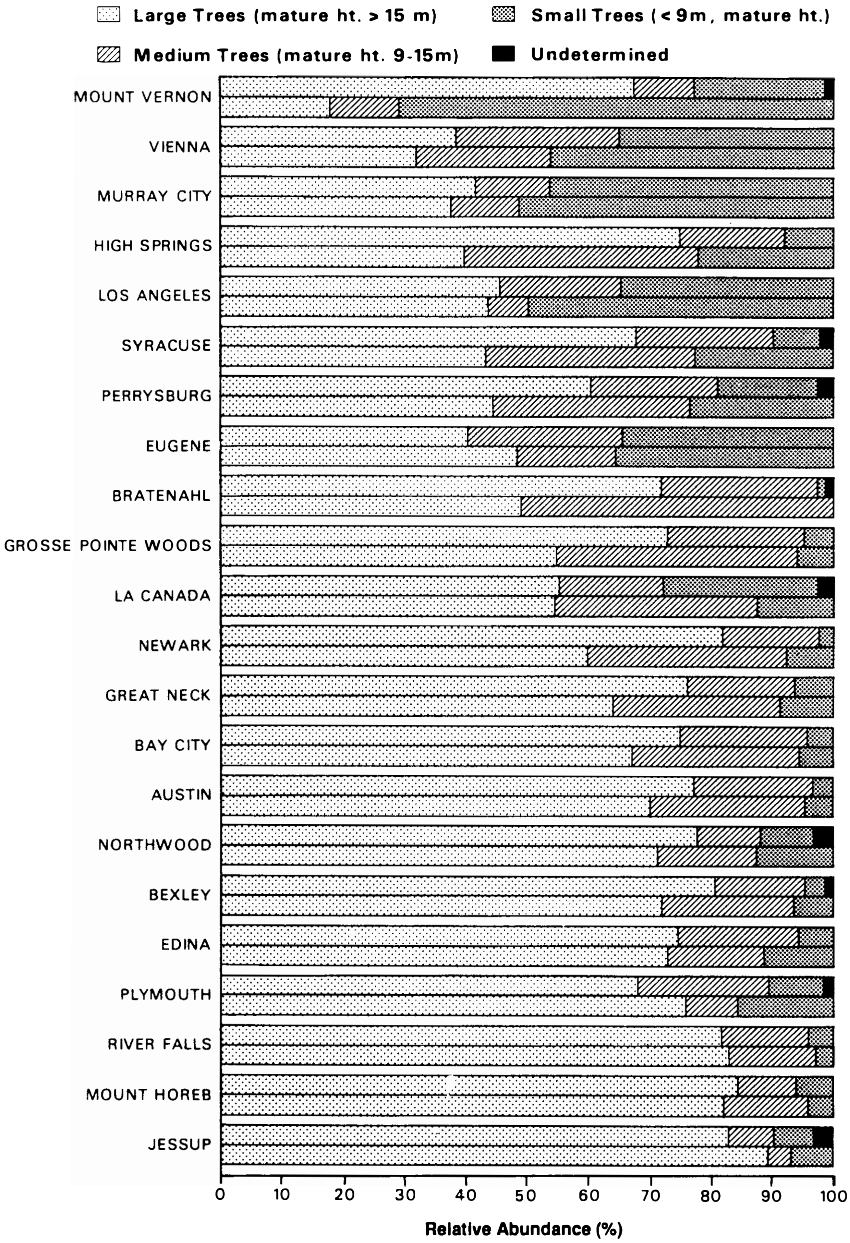


Figure 9. Changes in mature height composition of 22 U.S. street tree populations. The top bar indicates the percentage of all street trees in each mature height class, and the bottom bar shows the estimated percentage of trees recently planted (0–15 cm dbh) belonging to each mature height class. The trend towards decreased planting of larger-sized trees is evident.

relatively short life spans. Shorter rotations will result in more frequent replacement plantings to maintain current canopy cover. Although planting costs may increase, short term pruning and removal costs may decrease. Liability costs may also decrease with greater use of small trees. The shift to smaller street trees will save municipalities tree care dollars in the short run, but these savings may be offset by larger future planting costs. If tight budgets limit replacement plantings, tree canopy cover and stocking levels will drop dramatically.

The magnitude of many benefits increases as leaf surface area increases. Gacka-Grzesikiewicz (1980) reports that the functional value of trees is related to the outer surface area of the crown because interior layers are of little significance as regards photosynthesis, transpiration, and interception of atmospheric particulates. Small trees have disproportionately less outer surface area than large trees and thus, many street tree benefits will diminish as small street trees replace the large ones. Increased planting will be necessary to maintain existing canopy cover as smaller trees dominate more of the population. The shorter-lived small trees will intensify the need for more frequent street tree planting. Urban forestry programs now hard pressed to replace losses and maintain existing stocking levels will be forced to justify increased planting budgets as never before.

Concluding Discussion

Application of structural measures to a sample of 22 cities suggests that they can be useful tools for urban forest planning and management. Knowledge of existing age structure can be helpful in projecting whether future urban forest benefits are likely to diminish or increase. These data can then be used to evaluate present and future needs for planting, pruning, and removal. Measures of species diversity and importance can be analyzed to estimate the long-term performance of different species. Information on life span and performance can be used to develop better tree palettes for new plantings in different parts of the city.

Our analyses indicate that street tree populations are changing, with a trend towards increased planting of small-sized trees. Although this trend meets present concerns for increased diversity and reduced tree care costs, future planting costs may offset these savings if current amounts of canopy cover are to be sustained. If monies for replacement plantings are unavailable the result will be a dramatic reduction in canopy cover. It would be useful to determine the extent to which increased plantings of smaller trees will influence long term tree care costs, and to quantify how loss of canopy cover may affect environmental conditions in cities.

It is our assessment that the measures described in this study, together with their graphic representations, contribute to an understanding of the structure and dynamics of street tree populations in the urban forest. The indices reveal attributes useful to planning and management, but also to the growing body of knowledge nurturing the science and appreciation of urban vegetation, thereby helping us indeed to see the forest for the trees.

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APPENDIX A

The assumptions are:

1. Individuals of the same species are evenly distributed within the 0–15 cm dbh class.

2. Annual rates of planting and outgrowth (movement of individuals into the next diameter class) are constant. Mortality is not considered.

3. Even the slowest growing individuals will grow out of the 0–15 cm size class in 18 years.

4. Site conditions are typical of urban streetside space. Excellent and poor growing conditions offset each other.

The procedure incorporates species growth rate to account for the duration of time individuals remain in the size class. When one extrapolates back in time using inventory data, all individuals cannot be treated equally because the probability of being counted in a size class varies with growth rate. For example, one individual that requires ten years to grow out of the size class is equivalent to two individuals with five year outgrowth rates.

Reported growth rates of species located in streetside planting strips (Reisch et al. 1971) were analysed to define three rates of outgrowth: rapid (8 years), moderate (12 years), and slow (16 years). These rates correspond to annual ring widths of 0.64, 0.42, and 0.32 cm, respectively. References used to assign potential mature height classes were also used to assign growth rates to each of the ten most abundant species in every population (Dirr 1977, Williamson 1979).

The percentage that each species represents of the ten most abundant species was computed and the potential height and growth rate characteristics were identified. A 3 x 3 potential height/growth rate matrix was created. Columns correspond to potential height classes and rows are growth rates. The relative abundance of each species was placed in the appropriate matrix cell, and percentages were summed within each cell. The sums were divided by the growth rate to derive weighted values that account for both relative numbers and growth rate. These values were summed for large, medium, and small trees, and the average annual percentage of new plantings was obtained for each potential height class. There are some factors that were not considered. Growth rates vary with climate and soil as well as with species. The entire street tree population of each city was not analyzed. Establishment-related mortality as a function of species type and transplant size was not considered, nor was the increased probability of mortality in the 0–15 cm size class for smaller, short-lived species.